ADDIS ABABA UNIVERSITY
ETHIOPIAN INSTITUTE OF WATER RESOURCES (EIWR)

IMPACT OF SUSTAINABLE LAND MANAGEMENT (SLM) ON IMPROVING WATER SECURITY IN THE RAINFED BLUE NILE HIGHLANDS

BY

BERIHUN DUBALE MERSHA

DISSERTATION

SUBMITTED TO THE ETHIOPIAN INSTITUTE OF WATER RESOURCES (EIWR) IN FULFILLMENT OF THE REQUIREMENTS FOR THE ACQUISITION OF PhD DEGREE IN WATER RESOURCES ENGINEERING AND MANAGEMENT

ADDIS ABABA

MAY 2023

SUPERVISORS

Dr. Gete Zeleke

Water and Land Resource Center (WLRC), Addis Ababa University

Dr. Tena Alamirew

Water and Land Resource Center (WLRC), Addis Ababa University

Ethiopian Institute of Water Resources (EIWR), Addis Ababa University

Dr. Solomon Gebreyohannis

Water and Land Resource Center (WLRC), Addis Ababa University

Ethiopian Institute of Water Resources (EIWR), Addis Ababa University

Dr. Zeleke Agide

Ethiopian Institute of Water Resources (EIWR), Addis Ababa University

ADDIS ABABA UNIVERSITY
WATER RESOURCES ENGINEERING AND MANAGEMENT PROGRAM

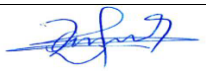
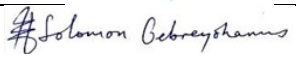
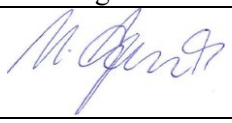
Impact of Sustainable Land Management (SLM) on Improving Water Security in the Rainfed Blue Nile Highlands

Berihun Dubale Mersha (ID No: GSR/0077/08)

Dissertation

Submitted to the Ethiopian Institute of Water Resources (EIWR) in Fulfillment of the Requirements for the Acquisition of PhD Degree in Water Resources Engineering and Management

Approved by Board of Examiners:

Main Advisor		Signature		date
Gete Zeleke (PhD)				25/10/2022
Co-Advisors		Signature		date
Tena Alamirew (PhD)				25/10/2022
Zeleke Agide (PhD)				24/10/2022
Solomon Gebreyohannis (PhD)				25/10/2022
Internal Examiners		Signature		date
Adane Abebe (PhD)				06/06/2023
Sirak Tekleab (PhD)				
External Examiner		Signature		date
Mekonen Ayana (PhD)				04/06/2023
Chair Person		Signature		date
Taye Alemayehu (PhD)				
EIWR Education Coordinator		Signature		date
Tekalegn Ayele (PhD)				
EIWR Director		Signature		date
Zeleke Agide (PhD)				

DEDICATION

To my mother

STATEMENT OF THE AUTHOR

I hereby declare that this dissertation represents my own work and is not submitted to any other institution elsewhere for the award of any degree, diploma, or certificate. All information (including diagrams and tables) or other information which is copied from, or based on, the work of others has its source clearly acknowledged in the text at the place where it appears.

TABLE OF CONTENTS

STATEMENT OF THE AUTHOR v

TABLE OF CONTENTS vi

SYMBOLS xv

ACRONYMS xvi

ACKNOWLEDGEMENTS xvii

ABSTRACT xviii

1. INTRODUCTION 1

1.1. Background and Justification 1

1.2. Problem Statement 3

1.3. Literature Review 5

1.3.1. Water security: a global perspective 5

1.3.2. Water security: a national perspective 5

1.3.3. Coping strategy: the need for a paradigm shift 10

1.3.4. Sustainable land management: a key strategy for improving water security in rainfed systems 13

1.4. Research Questions 18

1.5. Scope of the study 18

1.6. Research Objectives 19

1.6.1. General objective 19

1.6.2. Specific objectives 20

1.7. Structure of the Thesis 20

2. THE STUDY AREA 21

2.1. Geomorphology and soils 22

2.2. Hydroclimate 24

2.3. Farming system 25

3. DATA, MATERIALS, AND METHODS 29

3.1. Study Design 29

3.2. Data 31

3.3. Methods of Analysis 33

4. ASSESSING THE EFFECT OF SUSTAINABLE LAND MANAGEMENT ON IMPROVING WATER SECURITY IN THE BLUE NILE HIGHLANDS: A PAIRED-CATCHMENT APPROACH 35

4.1. Abstract 35

4.2. Introduction 35

4.3. Materials and Methods 38

4.3.1. Description of the study catchments 38

4.3.2. Study design 42

4.3.3. Data collection 42

4.3.4. Data analysis 44

4.4. Results and Discussion 52

4.4.1. Rainfall characteristics 52

4.4.2. Runoff characteristics 54

4.4.3. Soil moisture characteristics 54

4.4.4. Effect of SLM on infiltration-runoff dynamics 56

4.4.5. Effect of SLM on soil moisture dynamics 61

4.4.6. Implication of SLM for improving water security 64

4.5. Conclusions 65

5. SOIL MOISTURE SPATIO-TEMPORAL VARIABILITY UNDER TREATED AND UNTREATED CATCHMENT CONDITIONS IN A FRAGILE TROPICAL HIGHLAND ENVIRONMENT: IMPLICATION FOR WATER SECURITY 68

5.1. Abstract 68

5.2. Introduction 69

5.3. Materials and Methods 71

5.3.1. Study area description 71

5.3.2. Data collection 72

5.3.3. Data analysis 74

5.4. Results and Discussion 78

5.4.1. Soil moisture temporal variability 78

5.4.2. Soil moisture spatial variation 87

5.4.3. Implication for water security 91

5.5. Conclusions 93

6. INITIAL ABSTRACTION RATIO AND RUNOFF CURVE NUMBER COMPUTATIONS FROM DATA FOR DIFFERENT LAND USES AND MANAGEMENT CONDITIONS IN THE BLUE NILE HIGHLANDS, ETHIOPIA 95

6.1. Abstract 95

6.2. Introduction 96

6.3. Methods 99

6.3.1. Description of the study catchments 99

6.3.2. Data 100

6.3.3. Initial abstraction ratio and CN Computations from recorded P-Q data 102

6.3.4. Statistical analyses 104

6.4. Results and Discussion 105

6.4.1. Variations of λ and CN under different ARCs 105

6.4.2. Initial abstraction ratios and CNs computed from merged data 109

6.4.3. Evaluation of the adequacy of the CN method in the study catchments 110

6.4.4. Variations in hydrologic response under varying land uses and management conditions 112

6.4.5. Comparison of computed and NRCS-generated CNs 116

6.5. Conclusions 117

7. GENERAL CONCLUSIONS AND STUDY LIMITATIONS 119

7.1. General Conclusions 119

7.2. Study Limitations 128

8. REFERENCES 129

LIST OF TABLES

Table 1. Selected characteristics describing the experimental sites 48

Table 2. Characteristics of rainfall event received during the experimental period 53

Table 3. Fitting statistics of standard asymptotic curves for treatments under investigation 57

Table 4. Major topographic and soil characteristics of experimental sites 77

Table 5. Correlations between daily θ and climatic variables 78

Table 6. Minimum autocorrelation coefficients and associated time lags of daily θ for experimental sites in the control and treated catchments 79

Table 7. Multiple linear regression describing the relationships between soil moisture and climate variables 80

Table 8. Relationships between soil moisture monitored at different depths within the soil profile of each experimental site 89

Table 9. Cross correlations with zero time lag relating soil moisture time series of experimental sites within each study catchment 91

Table 10. Major biophysical characteristics describing the study catchments 99

Table 11. Major characteristics of experimental sites and data periods 101

Table 12. ARC thresholds in the study catchments 103

Table 13. Estimates of λ and CN from recorded P-Q data for ARC-1, ARC-2, and ARC-3 107

Table 14. Estimates of λ, S and CN from recorded P-Q data along with NRCS-generated CNs 111

LIST OF FIGURES

Figure 1. Location map of the study catchments 22

Figure 2. Soil cultivation using the animal-drawn *maresha* plow 26

Figure 3. Cropland affected by rill erosion at Aba Gerima control catchment 27

Figure 4. Open communal grazing at Aba Gerima control catchment 28

Figure 5. Terraces on cultivated land in Anjeni catchment 31

Figure 6. Experimental plot equipped with an iron sheet border and runoff collection tanks installed on an open grazing land (Anjeni) 32

Figure 7. The PR2/4 capacitance probe (a) and HH2 display unit (b) deployed for measuring soil moisture (Aba Gerima) 33

Figure 8. Location map of the Aba Gerima paired catchments 38

Figure 9. Landuse patterns of the Aba Gerima paired catchments 39

Figure 10. Distribution of structural SLM measures (soil bunds; contour trenches) in the treated Aba Gerima catchment 40

Figure 11. Some of the SLM practices implemented in the treated Aba Gerima catchment: (a) soil bund integrated with live hedges on cultivated land; (b) marginally degraded communal land protected from open grazing for recovery 41

Figure 12. Layout of a pair of runoff plots installed on cultivated land: (a) control runoff plot; (b) treated runoff plot 44

Figure 13. Rainfall pattern of the study area for the 2015 – 2018 period 53

Figure 14. Scatter plot of P–Q data pairs of treatments under investigation 54

Figure 15. Soil moisture regimes of treatments under investigation 56

Figure 16. Asymptotic CN plots of contrasting treatment pairs located on: (a) cultivated 5% slope; (b) cultivated 10% slope; (c) cultivated 15% slope; (d) degraded hillslope 60

Figure 17. Comparisons of changes in θ between contrasting treatment pairs relative to their respective initial θ recorded at the onset of the 2018 wet season: (a) cultivated land with 5% slope gradient; (b) cultivated land with 10% slope gradient; (c) degraded hillslope (17% slope gradient); (d) cultivated land with 15% slope gradient..... 62

Figure 18. Soil moisture dynamics of treatment pairs on cultivated land with 15% slope gradient: θC1, θC2, θC3 and θC4 represent θ at 10, 20, 30 and 40 cm depths, respectively, of the control site while θT1, θT2, θT3 and θT4 indicate θ at 10, 20, 30 and 40 cm depths of the treated site; FCC and FCT describe θ at field capacity of the control and treated sites, respectively 63

Figure 19. Rainfall pattern of the study area during the study period (2017–2018)..... 72

Figure 20. Temporal autocorrelation of soil moisture time series for individual experimental sites: (a) control catchment; and (b) treated catchment 81

Figure 21. Soil moisture temporal patterns for experimental sites in the treated and control catchments 83

Figure 22. Relationship between temporal mean of θ and elevation at different soil profile depths..... 85

Figure 23. Soil moisture regimes of TC3 and CC4 at 10, 20, 30, and 40 cm soil profile depths versus soil moisture content at field capacity (FC)..... 86

Figure 24. Relationships between temporal mean of θ and CV: (a) control catchment and (b) treated catchment 87

Figure 25. Relationships between θ and soil depth under treated and untreated catchment conditions: (a) CC1; (b) CC2; (c) CC3; (d) CC4; (e) TC1; (f) TC2; (g) TC3; and (h) TC4..... 90

Figure 26. Rainfall pattern of the study catchments..... 100

Figure 27. Scatter plots of observed P-Q data for six runoff plots and their best-fit curves describing CN-1, CN-2 and CN-3 108

Figure 28. Values of NSE describing the performance of the CN method in the study area..... 112

Figure 29. Observed variations in initial abstraction ratio (λ) across experimental sites 113

Figure 30. Variability of CN across experimental sites 115

SYMBOLS

Symbol	Description	Dimension
BD	Bulk Density	$M\ L^{-3}$
CN	Curve number	
CN_∞	Asymptotic curve number	
CN_o	Enveloping curve number	
F	Cumulative infiltration	L
FC	Field capacity	$L^3\ L^{-3}$
I_a	Initial abstraction	L
k	Asymptotic curve number parameter	L^{-1}
kWh	Kilowatt hour	ML^2T^{-2}
NMSE	Normalized mean square error	
NSE	Nash and Sutcliffe efficiency coefficient	
OC	Organic carbon content	
$\bar{O}_o$	Mean of observed runoff	L
P	Rainfall	L
P_{min}	Minimum precipitation at which CN_∞ is attained	L
Q	Runoff	L
Q_{ei}	The i^{th} computed runoff	L
Q_{oi}	The i^{th} observed runoff	L
RH	Relative humidity	
S	Storage index	L
T	Air temperature	Q
$t\ ha^{-1}$	Tons per hectare	$M\ L^{-2}$
WP	Wilting point	$L^3\ L^{-3}$
θ	Volumetric soil moisture content	$L^3\ L^{-3}$
λ	Initial abstraction ratio	
σ	Standard error of parameter estimate	

ACRONYMS

AGNPS	Agricultural Nonpoint Source
ARC	Antecedent Runoff Condition
ENSO	El Nino Southern Oscillation
EPIC	Erosion productivity Impact Calculator
ET	Evapotranspiration
FAO	Food and Agriculture Organization of t United Nations
GDP	Gross Domestic Product
HSG	Hydrologic soil group
ITCZ	Inter-Tropical Convergence Zone
NRCS	Natural Resources Conservation Service
PET	Potential Evapotranspiration
SCRP	Soil Conservation Research Program
SCS	Soil Conservation Service
SLM	Sustainable Land Management
SST	Sea Surface Temperature
SWAT	Soil and Water Assessment Tool
US	United States
USDA	United States Department of Agriculture

ACKNOWLEDGEMENTS

Let me start my acknowledgements by thanking God the Almighty for His blessings and protection. Without His grace, it is hard for me to imagine where I would be today.

My deepest gratitude goes to Dr. Gete Zeleke, my advisor, for shaping my career ever since I got the privilege of being his advisee while I was doing my MSc research a decade ago. The success of the present work is due to his insightful guidance and direct involvement in the design and setup of my field experimentation. I also gratefully acknowledge my co-advisors Dr. Tena Alamirew, Dr. Solomon GebereYohannes, and Dr. Zeleke Agide without whose support the successful completion of this work wouldn't have been possible. Dr. Tena, I wholeheartedly thank you for all the kindness and support I have been enjoying until this day ever since I knew you professing me at Haramaya University. My deepest appreciation goes to Dr. Solomon for his quick feedbacks and for the time he generously invested in reviewing the preliminary versions of my work. I sincerely thank Dr. Zeleke for his encouragement and support always accompanied by a kind treatment.

I sincerely acknowledge FCDO for funding my PhD research through WLRC and the REACH program.

My warmest appreciation goes to Dr. Ermias Teferi, Dr. Demamu Mesfin, and Dr. Tibebu Kassawmar whose insights, encouragement, and support were vital for the successful completion of this work. I am also indebted to Mastewal Mengist and Melese Bililgn whose assistances were instrumental during field experimentation and data collection. I thank Dr. Dejene Sahilu, Menelik Getaneh, Mamo Kasegn, Gebiyaw Sitotaw, Nigist Getahun, Habte Honelign, Misganaw Muchie, and Beqalu Dawit for being there whenever I needed their support.

Finally, deserving my appreciation are my wife Tamralech Getahun for her patience and care and my baby boys Nathan and Ezra for their refreshing smiles.

Berihun Dubale Mersha

May 2023

Impact of Sustainable Land Management (SLM) on Improving Water Security in the Rainfed Blue Nile Highlands

Berihun Dubale Mersha

Addis Ababa University, 2023

ABSTRACT

Securing water to meet ever growing human needs is a major global challenge. Particularly, feeding a rapidly expanding world population of 10 billion by 2050 presents a dominant water security challenge, with a projected water requirement of 12600 km^3 yr^{-1}. To meet this demand, management and planning of water resources dominated by blue water whose availability is limited needs to be widened to include green water. Green water, which originates from naturally infiltrated rain, sustains 80% of the croplands globally. In sub-Saharan Africa, over 95% of the croplands are rainfed. While rainfall in vast regions of the world is sufficient to support agricultural production without large-scale irrigation, grain yields in rainfed systems are disproportionately low due to management-induced green water scarcity resulting from nonproductive rainwater losses of up to 70 – 85%. Investments in sustainable land management (SLM) initiatives in rainfed systems have led to reductions in nonproductive rainwater losses and increases in grain yields by 5 – 10 times. Large-scale implementation of SLM initiatives in Ethiopia dates back to the early 1970s, which primarily involved the construction of terraces on agricultural lands. Such basic practices of SLM have contributed significantly to controlling soil erosion on agricultural land; however, they produced little impact on improving water security and agricultural productivity. This led to the emergence of a new approach of SLM, where every land use type within a catchment is treated for improving its productivity and ecological functionality. The main goal of the present thesis is to assess the impact of SLM on improving water security in the Blue Nile Highlands, where rainfed agriculture is the dominant economic activity. A paired-catchment approach provided an opportunity for using a combination of historical and newly recorded data for analysis. Runoff and causal rainfall data recorded from 26 experimental plots across

three Blue Nile headwater catchments were used for an event-based water-budget analysis. Soil moisture data collected intensively at multiple sites and soil profile depths were used to assess the impact of SLM on soil water availability and soil moisture spatio-temporal variability. Relative to traditionally managed sites, sites under SLM practices presented computed runoff curve numbers reduced by -13.9 to -21.6 units, revealing substantial increases in infiltration and water storage potentials of the agricultural landscapes in the study area. An increasing trend in the initial abstraction ratio was also observed across sites where the SLM interventions have been adequate, indicating a lag effect in runoff response. Sustainable land management practices in the study area reduced soil moisture temporal variability and spatial heterogeneity, resulting in increased soil moisture storage by 16 – 800%. The emerging SLM approach in the study area that leaves no land use type untreated within a catchment reveals great promises for improving rainwater infiltration and soil water storage potentials in the study area. Apart from securing water for sustaining rainfed production, SLM interventions have minimal effect on overall catchment runoff and, hence, on downstream water users and ecosystems. We conclude that a major effort needs to be exerted on mainstreaming water security into SLM initiatives to draw adequate attention among local land users, experts, and policy makers about the role of SLM for improving water security in the rainfed Blue Nile Highlands.

Key terms: *Green Water, Hydrologic Functionality, Infiltration-Runoff Dynamics, Management-Induced Water Scarcity, Rainfed Systems, Soil Water Storage, Sustainable Land Management, Water Security.*

1. INTRODUCTION

1.1. Background and Justification

Water is a fundamental natural resource playing a central role in sustaining lives, livelihoods, economies, and ecosystems (Rockström et al., 2004). Water provides key productive opportunities in a spectrum of economic sectors, including agriculture, energy, and industry. On the other hand, it can also be a threat to human well-being, the economy, and the natural environment through droughts, floods, and erosion. Hence, human societies have always been under a continuing struggle to harness the productive potential of water and limit its destructive impact (Grey & Sadoff, 2007). This led to the emergence of the concept of water security, which has gained increased importance in framing the interactions between human societies and water (Gain et al., 2016).

Water security is a concept that emerged in the 1990s with a more or less equivalent meaning with food security and energy security (Cook & Bakker, 2012; Gerlak & Mukhtarov, 2015). Food security and energy security generally mean reliable access to sufficient supplies of food or energy to meet basic needs of individuals or societies (Grey & Sadoff, 2007). However, water security has since evolved to include issues ranging from sustainable use and development of water resources to mitigation of water-related hazards and protection of water functions and services for humans and the environment (van Beek & Arriens, 2014).

Water is Earth's ubiquitous element, covering nearly two-thirds of its surface. However, less than 3% is freshwater, of which only 1% is available as liquid water on the Earth's surface while the rest is locked in the ice-caps and glaciers or in deep aquifers (Dingman, 2015). Hence, the estimated ceiling of the realistically accessible water upon which all social and economic activities on Earth depend is 12,500 km^3 yr^{-1} (Rockström, 2001).

Owing to rapid increases in global population and economic development, the increases in demand for water to sustain water-dependent economic sectors, such as agriculture, industry, and municipalities, are unprecedented (Hoekstra & Mekonnen, 2012; Wheater, 2015). As a result, global water withdrawal between 1900 and 2010, for instance, increased by nearly eight times

from about 500 km^3 yr^{-1} to 4000 km^3 yr^{-1} (Wada et al., 2016). While municipal and industrial water uses constitute, respectively, 10% and 20% of the global water withdrawal, 70% is used to sustain agriculture in irrigation (Hejazi et al., 2013), making agriculture the world's largest water consuming economic sector by far. With world population estimated to reach about 10 billion by mid-century, there will likely be a 70% increase in global food demand and up to a 100% increase in developing countries (Rodrigues et al., 2014; Wheater & Gober, 2015). Further, Hanasaki et al. (2013) projected that approximately 50% of the world's population will be living under severe water stress by 2071 – 2100 stemming from increases in water-related human needs associated with the rapidly growing global population.

Apart from increases in demand for water stemming from rapid expansion of global population and economic development, the world is also experiencing human-induced changes in the availability of water (Hanasaki et al., 2013). As a consequence, water resources are under increasingly severe pressure from climate change due to changes in rainfall patterns, soil moisture, humidity, river flows, and intensity and frequency of floods and droughts (Kumar, 2018). Gosling and Arnell (2013) projected that climate change will cause water scarcity to some 0.5 – 3.1 billion people by 2050. In addition, urbanization, deforestation, and agricultural development and intensification represent another aspect of human influences on the terrestrial water cycle (Wheater, 2015). Hence, with an estimated forest loss of 5.2 million ha annually, the hydrological impacts of deforestation have also been the subject of much hydrological research with profound outcomes (FAO, 2012).

While much has been reported about the hydrological effects of climate change and deforestation, the implications of changes associated with agricultural land management are much less well understood (Wheater & Gober, 2015). Given cultivated land accounts for 1.5 billion ha globally (FAO, 2012), it has been hypothesized that changes in land management associated with agricultural land would substantially change the quantity and quality of surface water and groundwater resources (McNally et al., 2019; Wheater, 2015). While deforestation and intensification of agricultural production have the potential to enhance flood risk, changes in land management due to implementation of appropriate agricultural management practices equally have the potential to mitigate such impacts (Wheater & Gober, 2015). However, many of these impacts are poorly understood as well as poorly modeled due to lack of skill and/or relevant data.

As a consequence, the hydrological effects of agricultural intensification remain quite uncertain even in densely-gauged environments (McIntyre et al., 2014). In sparsely-gauged environments, effects of modern changes in land management, such as zero tillage, have had little attention from a hydrological perspective due to lack of relevant data to understand their impact on the local water balance (Wheater, 2015).

Given effects of land management are complex, they depend very much on local context (Wheater, 2015). This is particularly true in Ethiopia, where the climate is diverse associated with topographic diversity, ranging from a temperate climate in the plateaus to a hot tropical climate in the lowlands (Zaitchik et al., 2012). Ethiopia's rich geographic diversity consists of rugged mountains, flat-topped plateaus, deep gorges, and river valleys, with the highest altitude (4620 m above sea level) at Ras Dejen and the lowest (120 m below sea level) at Afar Depression (Eshete et al., 2020).

Ethiopia's hydrology is significantly influenced by its topography. The country has abundant water resources, with a mean total surface water flow of roughly 124.4 billion m^3 yr^{-1} (Melesse et al., 2013). However, they are unevenly distributed between areas, with the western region receiving relatively high rainfall while the central and eastern parts of the country face water shortages. Given all these contrasts in water resources availability across the country's territory, it is not clear how sustainable land management (SLM) practices affect water availability and storage in Ethiopia, where subsistence rainfed farming, which is the dominant economic activity in the country, has been greatly undermined by management-induced water scarcity.

1.2. Problem Statement

Agriculture is the greatest consumer of freshwater than any other economic sector, with about 70% of the total blue water withdrawal globally being used in irrigation (Rockström et al., 2010). Irrigated agriculture has contributed greatly to increasing agricultural water security in vast regions of the world; however, current water withdrawal for irrigation is causing stress in many of the world's major river basins (Falkenmark, 2013). Hence, irrigated agriculture already has a limited potential for further expansion (Cooper et al., 2008; Rost et al., 2009), rendering rainfed agriculture to remain a major player in food production for a growing world population.

The great untapped potential in rainfed agriculture to increase grain yields is constrained by disproportionately high rainwater losses through nonproductive pathways (Biazin et al., 2012; Rockström et al., 2010). Though rainfall is generally erratic in vast regions of the world, it is often not hydrologically limiting to rainfed crop production. Hence, rainfed agriculture in these regions suffers greatly not from the scarcity of rainfall itself but from management-induced water scarcity, leading to low grain yields (Biazin et al., 2012).

Crop failures commonly blamed on drought might be overcome by appropriate management interventions, with potential increases in grain yields by up to 5 – 10 times (Rockström et al., 2004). Globally, a host of management options reveals great promises for bridging management-induced water scarcity and sustaining rainfed agricultural productivity in various regions of the world (Bossio et al., 2010). Hence, SLM practices provide a key role in improving green water security and agricultural productivity in rainfed systems around the world (Rockström et al., 2010). In effect, such practices have gained increasing importance in rainfed systems, with success stories of optimized rainwater use (Abera et al., 2020; Rockström et al., 2009).

Large-scale implementation of soil and water conservation measures in Ethiopia dates back to the early 1970s, following the drought-induced famines that caused huge devastations and losses of lives in the country (Abera et al., 2020). Since then, huge investments have been made with the support of extensive community participation to reverse land degradation, mainly in the degradation-prone Ethiopian Highlands where more than 90% of the land under cultivation in the country is contained. While simulation modelling and field experimentation revealed that SLM practices have played a substantial role in protecting soil erosion onsite (SCRP, 2000) and hence reducing sediment loading in streams (Betrie et al., 2011), the potential of such a strategy in improving water security is poorly investigated in the fragile Ethiopian Highlands, where human settlement and rainfed farming have a long history (Gumma et al., 2021). Such an understanding is critical for mainstreaming water security into current and future SLM initiatives across the nation, where subsistence rainfed agriculture remains to be the major economic activity providing food and livelihoods for the vast majority of the population.

1.3. Literature Review

1.3.1. Water security: a global perspective

Access to adequate water to meet the various needs of a rapidly growing world population is a major global challenge (Mishra et al., 2021). Investments in water development and management are deemed an urgent priority for ensuring food and energy security and for eradicating poverty around the world (Damkjaer & Taylor, 2017). Currently, four billion people worldwide face severe water scarcity for at least one month of the year while half a billion of them live under conditions of severe water scarcity all year round (Mekonnen & Hoekstra, 2016). Hence, this challenge paved the way for water security to be recognized as one of the sustainable development goals (SDGs) (Gain et al., 2016).

The level of investment in water infrastructure and institutions determines the water security status of countries globally. In wealthy regions of the world, where a comparably easy hydrology is maintained by low rainfall spatio-temporal variability and perennial river flows, water security has been achieved predominantly through investments in substantial water infrastructure, such as multipurpose dams, inter-basin transfer schemes, dense hydrometeorological networks, and strong institutions to manage them (Grey et al., 2013). By contrast, it remains a pressing challenge for many of the world's poorest countries to achieve water security, where the capacity to provide the exceptionally heavy investments in water institutions and infrastructure to manage their difficult hydrologic legacy is limited (Grey & Sadoff, 2007). The hydrologic legacy of these countries is characterized by markedly seasonal rainfall, with high inter-annual and intra-seasonal variability. This is particularly the case in sub-Saharan African countries, where the transition to a water-secure future is under a serious threat from the limited capacity to cover the comparatively high cost of investments in infrastructure and institutions (Grey et al., 2013). In these countries, only a small fraction of the infrastructure and institutional capacity are available to develop and manage their scarce water resources.

1.3.2. Water security: a national perspective

Water security concerns are diverse, varying from country to country depending on a range of inherent and anthropogenic factors (Grey & Sadoff, 2007; Rockström et al., 2010). Inherent

factors refer to the various forms of the natural hydrologic legacy that a society inherits, ranging from the absolute level of water availability to its temporal variability and spatial distribution (Cook & Bakker, 2012). On the other hand, anthropogenic factors include global environmental changes arising from human activities perturbing the terrestrial water cycle, such as land degradation and climate change (Rost et al., 2008). In Ethiopia, both inherent and anthropogenic factors are responsible for undermining the effort to build the minimum platform of water infrastructure and management capacity needed to achieve water security.

In Ethiopia, only about 10% (4676 MW) of the country's hydropower potential (45000 MW) has been developed so far (Hailu, 2022), which corresponds to one of the lowest per capita electric power consumptions in the world, at 133 kWh yr^{-1} (Boke et al., 2022). Besides, while over 6 million ha of land is irrigable in Ethiopia (Worqlul et al., 2017), only about 3.02 million ha has been developed (Gebul, 2021), with even lower figures for the Blue Nile Basin, where only about 79800 ha is currently under irrigation relative to a 738184 ha potential of the basin (Yimere & Assefa, 2022). Overall, the greatest proportion (97%) of stream flows in Ethiopia is lost to downstream countries (Melesse et al., 2013), pointing to the need for large investments in water resources infrastructure and management to harness the country's water resources potential and promote water security.

1.3.2.1. Hydrologic variability

Ethiopia is cited to have a relatively generous endowment of water resources, with a mean total runoff volume of about 124.4 billion m^3 yr^{-1} (Melesse et al., 2013) and an estimated groundwater potential of 40 billion m^3 (Mengistu et al., 2019). However, the country's hydrologic legacy is characterized by extreme spatial and temporal variability. Hence, rainfall is highly variable seasonally, annually, and across the country's territory (Melesse et al., 2013).

The seasonal distribution of rainfall over Ethiopia is influenced by a range of weather systems, such as Subtropical Jet (STJ), Intertropical Convergence Zone (ITCZ), Red Sea Convergence Zone (RSCZ), Tropical Easterly Jet (TEJ), and Somali Jet, leading to the occurrence of both unimodal and bimodal rainfall patterns in different parts of the country (Conway, 2000; Melesse et al., 2013; Seleshi & Zanke, 2004). As a result, rainfall is unimodal in the southwestern and northwestern parts of the country, with more extended rains in the southwest (February –

November) than in the northwest (June – September). On the other hand, rainfall is bimodal in central, eastern, southern, and southeastern Ethiopia comprising main and shorter rainy seasons. In central and eastern Ethiopia, the main rains occur during June – September while the shorter rains are received during March – May. In southern and southeastern Ethiopia, the main rains occur during February – May and the shorter rains during October – November.

Further, Ethiopia's topography is diverse, with elevation ranging from 120 m below mean sea level at the Afar depression to over 4600 m above mean sea level at Ras Dejen (Eshete et al., 2020; Melesse et al., 2013; Seleshi & Zanke, 2004). In effect, rainfall in the country is spatially extremely variable, with mean annual rainfall varying from 141 mm in the eastern and northeastern lowlands to 2275 mm in the southwestern highlands (Conway, 2000; Melesse et al., 2013). Overall, 69.83% of the country's surface water resources are concentrated in the western region where rainfall is ample (Melesse et al., 2013). Yet, rainfall over most parts of Ethiopia exhibits marked interannual variability associated with changes in sea surface temperature and El-Nino Southern Oscillation (ENSO) episodes in the Atlantic and Indian oceans (Asfaw et al., 2018; Block & Rajagopalan, 2007; Seleshi & Zanke, 2004; Wagesho et al., 2013), with irregularities in the timing and amount of rainfall. As a result, El-Nino years are characterized by below average June – September rainfall.

The heavy impacts of seasonal and interannual rainfall variability on the energy, agriculture, and water-supply sectors in Ethiopia cost the country over one-third of its average annual GDP (Verhoeven, 2013). Particularly, Ethiopia's agriculture, which is predominantly rainfed, is a water-vulnerable sector despite its dominant role in the country's economy, sustaining 85% of the country's population (Eshete et al., 2020; Tabari et al., 2015) and contributing 24% of its GDP growth (Planning and Development Commission, 2021). In Ethiopia, dry spells occur almost every year and drought events once or twice every decade, leading to crop failures and famines, such as those experienced in 1984, 1974, and 1973 (Conway & Schipper, 2011).

1.3.2.2. Climate change

The environment globally is heavily impacted by anthropogenic activities mainly linked to the burning of fossil fuels, leading to climate change (Rockström, 2003). Increases in average temperature and changes in precipitation patterns and in the frequency and intensity of extreme

events associated with climate change are, in turn, expected to have impacts on water resources and agriculture (Shukla et al., 2022). Projections indicate that increases in the frequency of drought events arising from climate change will lead to a progressive decline in overall rainfall, with strong implications on agriculture, especially in already drought-prone environments (Rockström, 2003). Globally, food production will likely experience a 9% reduction by the mid of the century as a result of climate change (Rost et al., 2009). In sub-Saharan Africa, climate change is expected to increase water scarcity, causing declines in regional soil moisture upon which agricultural production depends (Cooper et al., 2008). In countries of the Sahel and Horn of Africa, climate change is projected to cause declines in blue water resources by up to 25 – 100% between 2020 and 2040 (Faramarzi et al., 2013).

The impact of climate change is already evident in Ethiopia (Asfaw et al., 2018; Faramarzi et al., 2013; Tabari et al., 2015; Wagesho et al., 2013; Worku et al., 2019). Examining the changes in rainfall during 1901 – 2014 for northcentral Ethiopia, Asfaw et al. (2018) found a decreasing trend for the June – September and annual rainfall and an increasing trend for the frequency of drought events. Faramarzi et al. (2013) projected that precipitation over Ethiopia could decline by up to 25% during 2020 – 2040. Investigating long-term rainfall trends for 16 stations within the upper Blue Nile Basin, Tabari et al. (2015) identified a decreasing tendency in annual and seasonal rainfall for over 80% of the stations. Again, Wagesho et al. (2013) indicated decreasing wet season and annual rainfall for the northern, northwestern, and western parts of the country based on a precipitation trend analysis involving five decades (1951 – 2000). Besides, Worku et al. (2019) found observed increases in the extremes of rainfall for most sites within the upper Blue Nile Basin.

Due to reductions in rainfall amount and increases in the frequency of drought events related to climate change, water stress is expected to increase in Ethiopia, posing a major threat to the country's water security (Ayalew et al., 2022; Gelete et al., 2020; Hailemariam, 1999; Heyi et al., 2022; McCartney et al., 2013; Taye et al., 2018; Worqlul et al., 2018). For instance, stream flows will decrease in the future in the Blue Nile Basin (Gelete et al., 2020), with a likely 19% reduction in wet season flow (Worqlul et al., 2018) and a possible 37% reduction in hydropower production potential (McCartney et al., 2013) by the end of the century. Besides, mean annual streamflow is projected to decrease markedly in the headwaters of the Blue Nile Basin (Ayalew

et al., 2022). Also, analysis of the hydrologic impact of climate change in the Awash Basin demonstrated that there would be decreases in runoff (Hailemariam, 1999) and increases in water stress (Taye et al., 2018) over the basin stemming from a drier and warmer climate in the future. Especially, significant reductions in mean annual and wet season flows are anticipated to take place in the upper Awash Basin (Heyi et al., 2022). Overall, there appears to be a strong need to implement climate change adaptation and mitigation strategies in Ethiopia to improve the country's water security status.

1.3.2.3. Land degradation

Agriculture forges a stronger association with water security than any other economic sector. Accounting for about 70% the overall freshwater withdrawals globally, not only is agriculture a major freshwater consumer, but also impacts water security through land degradation and associated changes in the terrestrial water balance and water availability (McNally et al., 2019). In many developing countries, agricultural practices are responsible for the degradation of land, water, and forest resources (Bai et al., 2008). Expansion of agricultural land at the expense of forest land during the past several decades has been the major cause of land degradation (Rockström et al., 2010), leading to reduced global evapotranspiration and increased stream discharge (Rost et al., 2008).

In Ethiopia, human settlement and rainfed agriculture have a long history, particularly in the Highlands, where the climate is favorable (Gumma et al., 2021; Hurni et al., 2010; Wassie, 2020). Hosting over 90% of the country's population on roughly 50% of the country's land area (Hurni et al., 2010), the Ethiopian Highlands are under immense pressure, leading to deforestation and encroachment of rainfed farming onto steep slopes in an effort to increase food production to meet the needs of the fast-growing population (Hurni et al., 2005; Nyssen et al., 2004). Deforestation in Ethiopia has been taking place for centuries in association with agricultural development in the country (Hurni et al., 2010). Over the years, cropped areas have expanded substantially in Ethiopia while forests, grasslands, and water bodies are shrinking (Wassie, 2020). At an average soil loss rate of 40 t ha^{-1} yr^{-1} (SCRP, 2000), the country has long suffered from the impacts of severe soil degradation, including disturbances in local rainfall

partitioning and, hence, reductions in water productivity, availability, and storage, due to reduced soil depth, soil organic matter content, and water-holding capacity (Hurni et al., 2010).

Depletion of soil organic matter content associated with chronic soil degradation leads to the collapse of soil structure and soil surface crusting, adversely affecting soil porosity and infiltration and water-holding capacity (Bossio et al., 2010). This in turn leads to increased non-productively rainwater losses through direct soil evaporation, surface runoff, and deep drainage, and subsequently, to agricultural water scarcity (Biazin et al., 2012; Verhoeven, 2013). Non-productive rainwater losses arising from poor farming practices are estimated to be in the order of 70 – 85% of the rainfall received in rainfed agricultural systems (Rost et al., 2009). Hence, in rainfed systems affected by land degradation, only a limited fraction (15 – 30%) of the rainfall is available for crop transpiration (Hsiao et al., 2007; Rockström et al., 2010).

In the Ethiopian Highlands, where the conversion of forest land into agricultural land has long been a common practice to increase agricultural production, surface runoff is reported to have increased by a factor of 5 – 30 times (Hurni et al., 2005). Hence, though rainfall received in the Ethiopian Highlands is hydrologically sufficient (Hurni et al., 2010; Nyssen et al., 2004) to sustain rainfed farming in the region without large-scale irrigation, limited availability of green water in the root zone due to reduced water infiltration and retention capacity of the degraded soils has led to low level of productivity, with base yields remaining in the range 0.5 – 1.2 t ha^{-1} (Biazin et al., 2012; Rockström et al., 2009). In effect, about 10% of the country's population are chronically food insecure, being unable to meet their food needs even in average rainfall years (Conway & Schipper, 2011).

1.3.3. Coping strategy: the need for a paradigm shift

The overall available freshwater resource in the terrestrial hydrologic cycle almost comes from precipitation over land, which is partitioned into green water (infiltrated rain in the unsaturated zone) and blue water (runoff and deep aquifer recharge) (Hoekstra, 2019; Rockstrom et al., 2009). However, water resources planning and management has been traditionally dominated by blue water, which forms only about 11% of the total annual terrestrial precipitation (Schuol et al., 2008). Appropriation of blue water for agriculture, energy, industry, and domestic use at a rate

higher than the growth rate of world population has caused unprecedented perturbation of the terrestrial water cycle (Rost et al., 2008, 2009). Compounded by the effect of climate change, this has led to a situation where water has become increasingly scarce (Liu et al., 2022), with blue water shortage to affect 59% of the world population by 2050 (Rockstrom et al., 2009).

Given the agriculture sector is world's largest water user, consuming over two-thirds of the total freshwater withdrawal globally (Liu et al., 2022; McNally et al., 2019), access to sufficient supply of water to produce food for a rapidly expanding world population remains to be a dominant water security challenge. World population is expected to reach 10 billion before leveling off by 2050 (de Fraiture & Wichelns, 2010). Cumulative water demand to feed a world population of 10 billion by the mid of the century will reach 12600 km^3 yr^{-1} (Rockström et al., 2010). Relative to a current agricultural consumptive water use estimate of 8000 km^3 yr^{-1} (Rost et al., 2009), there will be a water deficit of 4600 km^3 yr^{-1} to secure food for a world population of 10 billion by 2050. This requires at least the doubling of agricultural yields in economies dominated by smallholder rainfed farming (Rockström et al., 2009), which corresponds to a nearly five-fold increase in agricultural water demand in sub-Saharan Africa (Rockström et al., 2010).

With about 20% of the croplands already equipped for irrigation globally, irrigation development has driven improved agricultural water security and agricultural production in large regions of the world (Falkenmark, 2013). However, freshwater and arable land are approaching their limits to increase food production through additional blue water withdrawal and horizontal expansion of irrigated agriculture (Rockström et al., 2010; Rost et al., 2009). By increasing water withdrawals for irrigation expansion together with the improvement of water use efficiencies in existing irrigation schemes, irrigated agriculture can bridge only about 1000 km^3 yr^{-1} of the water deficit required to meet the global food security challenge by the mid of the century (Rockström et al., 2007). Hence, there remains a water deficit of 3600 km^3 yr^{-1} exceeding the potential of irrigated agriculture to meet the food needs of the world population by 2050. This strategy can only achieve a 9% increase in cropped area and 13% increase in agricultural water withdrawal (Rockstrom et al., 2009; Rost et al., 2009). In Ethiopia, though irrigation development has received increased attention with a substantial rise in the area equipped with irrigation facilities, inadequate water management is a major challenge in irrigation schemes, limiting the country's

prospect to alleviate poverty in the face of a rapidly growing population and intensifying extreme climate conditions (Bekele et al., 2021).

Until the recent past, studies of freshwater availability have ignored the green water resource, which has a critical role in sustaining ecosystem goods and services upon which humans depend (Liu et al., 2022; Rockstrom et al., 2009; Rockström et al., 2010). Green water (soil moisture in the root zone originating from naturally infiltrated rain) is a fundamental component of the terrestrial hydrologic cycle sustaining all terrestrial ecosystems, including biodiversity, forests, grasslands, wetlands, and rainfed crops, representing a consumptive water use of 49000 – 56500 $km^3\ yr^{-1}$ (Rost et al., 2008). The role of green water in meeting water needs for food production and human welfare is quite abundant (Falkenmark, 2013; Rockström et al., 2010). Hence, food production in many of the world's countries with difficult hydrologic legacy is operated by green water (Falkenmark, 2013). Globally, about 60% of the food production is rainfed (Cooper et al., 2008; Faramarzi et al., 2013). In sub-Saharan Africa, more than 95% of the croplands are rainfed, accounting for over 90% of the staple food production in the region (Cooper et al., 2008). In Ethiopia, quite a similar share (90 – 95%) of the total annual food production is rainfed (Worqlul et al., 2017).

The growing realization that blue water resources are not the only freshwater involved in societal development has led to a paradigm shift, where the focus of water resources planning and management has evolved from a blue to a blue-green approach (de Fraiture & Wichelns, 2010; Liu et al., 2022; Rockstrom et al., 2009; Rockström et al., 2010). Hence, it is critical to take into account the green water resource in the analysis of water availability and requirement for understanding the overall water resources potential of a country or region (de Fraiture & Wichelns, 2010; Liu et al., 2022; Rockstrom et al., 2009). For instance, Ethiopia is heading toward chronic blue water shortage (Rockstrom et al., 2009). However, the country's overall supply of water is adequate to be self-sufficient in food production if its green water resource is adequately managed. Hence, the key to the country's water secure future is co-managing its green and blue water resources to meet its agricultural, societal, and environmental needs. Globally, through the co-management of blue and green water resources to improve water productivity in both irrigated and rainfed systems, the water deficit required to meet global food needs by the mid of the century can be reduced by about 80% (de Fraiture & Wichelns, 2010).

1.3.4. Sustainable land management: a key strategy for improving water security in rainfed systems

Agriculture bears the challenge of feeding a rapidly growing world population. However, food production requires large quantities of water (de Fraiture & Wichelns, 2010). For instance, global food demand is projected to increase by 35% by 2030 (McNally et al., 2019). This corresponds to a 40% increase in water demand, presenting a major global water security challenge. The consumptive use of water in agriculture is dominated by green water, particularly in Europe, Africa, and South America, where food productions are almost exclusively dependent on green water (Rockstrom et al., 2009). Given the potential of irrigated agriculture in meeting increasing global food needs is approaching its limit, rainfed agriculture will continue to have a major role in providing food and livelihoods for a significant number of the world's growing population. Especially, this applies to regions with difficult hydrologic legacy, such as sub-Saharan Africa, where stream flows are small and occur only during the rainy season (Rockström et al., 2010).

Under better management conditions, Earth's existing water and land resources are sufficient to meet increasing global food needs over the next 50 years (de Fraiture & Wichelns, 2010). However, in rainfed systems, where food production is operated by infiltrated rain in the soil (green water), food production often suffers from management-induced green water scarcity resulting from large nonproductive rainwater losses (Falkenmark, 2013; Rockström et al., 2009; Rost et al., 2008). Hence, though rainfall in rainfed systems is predominantly enough to achieve substantial increases in grain yields without large-scale irrigation, up to 70 – 85% of the rainfall is lost nonproductively due to lack of adequate management interventions (Rost et al., 2009). This, in turn, has led to low yield levels, particularly in smallholder rainfed systems, where average grain yield is 1.5 t ha^{-1}, compared with 5 – 6 t ha^{-1} in commercial rainfed agriculture and 3.1 t ha^{-1} in irrigated agriculture (Rockström et al., 2010). In sub-Saharan Africa, only 10 – 30% of the seasonal rainfall is used for productive green-water flow (transpiration) (Rockström et al., 2010) while up to 50% is lost as nonproductive evaporation, 30% as surface runoff, and 25% as deep percolation (Hsiao et al., 2007). A water budget analysis carried out for Ethiopia indicates rainwater losses of up to 72– 81% through ET and 11.3 – 24.1% through runoff, leaving only 2.3 – 4.4% of the total annual precipitation as green water storage in the soil (Schuol et al., 2008).

In sub-Saharan Africa, where food production is almost entirely rainfed, about 60% of the croplands have been affected by water erosion (Rockström, 2003). In Ethiopia, nearly 85% the highlands, where over 90% of the country's agricultural land is contained, have been affected by water erosion (Gumma et al., 2021). In the long term, unmanaged water erosion can reduce soil organic matter content, soil aggregate stability, soil porosity, and rooting depth, subsequently reducing infiltration capacity and soil water availability (Bossio et al., 2010). This presents a major challenge to increasing and sustaining agricultural production in the country (Adimassu et al., 2014; Etsay et al., 2019; Ewunetu et al., 2021; Hurni et al., 2010).

In Ethiopia, due to limited availability of land for agricultural expansion, intensification of agriculture on the land already under cultivation has gained a major importance, particularly in the highlands, to increase food production for the country's growing population (Alemu, 2016). However, this has driven accelerated degradation of water and land resources in the country (Ewunetu et al., 2021; Hurni et al., 2010; Schmidt & Tadesse, 2019). Given agricultural intensification must materialize without threatening environmental sustainability (Davis et al., 2016), large scale adoption of technologies that promote sustainable land management (SLM) has been implemented in Ethiopia since the 1970s (Ewunetu et al., 2021). Sustainable land management is defined as a knowledge-based approach that integrates land, water, biodiversity, and environmental management to meet rising food and fiber needs of a growing world population while sustaining ecosystem services and livelihoods (World Bank, 2006). By SLM we mean that land is managed to not only meet the needs of the present generation, but also the needs of future generations. Overall, SLM promotes the use of land resources, including soil, water, animals, and plants, for the production of goods to meet changing human needs while ensuring the long-term productive potential of these resources and ensuring their environmental functions (Alemu, 2016). Hence, SLM is gaining importance as a key strategy to sustain economic growth in Ethiopia, which, in turn, has promoted massive investments in SLM across many areas in the country to restore and enhance agricultural productivity (Ewunetu et al., 2021; Schmidt & Tadesse, 2019).

Although traditionally neglected, investing in SLM harbors a great untapped potential to improve on-farm water productivity in rainfed agriculture, even in semiarid regions (Bossio et al., 2010; de Fraiture & Wichelns, 2010). In rainfed agricultural systems, SLM can substantially reduce

nonproductive rainwater losses and thereby increase grain yields by a factor of 5 – 10 times (Rockström et al., 2004). At the global scale, the difference between total green water flows and productive green water flows (3500 km^3 yr^{-1}) constitutes a sizable potential water resource to make rainfed agriculture more productive (Rockstrom et al., 2009). If this huge potential water resource in rainfed systems is exploited to achieve a 72% increase in cereal yields globally and a more than double increase in sub-Saharan Africa, all the food required to feed the global population by 2050 can be met (de Fraiture et al., 2009). Particularly, reducing nonproductive soil evaporation in rainfed systems provides an enormous opportunity to harness the green water resource in situ to improve yields without substantially affecting blue water (runoff) flows for water users and ecosystems downstream (Rockström et al., 2010). In sub-Saharan Africa, where the yield gaps in rainfed agriculture are large and the capacity to invest in irrigated agriculture is limited (de Fraiture & Wichelns, 2010), investing in SLM options offers a large and relatively cheaper option for upgrading rainfed agriculture to meet increasing food needs.

Conventionally, engineering structures, such as terraces and soil bunds, have often been used to manage runoff (blue water) on agricultural land to control onsite soil erosion (Alemu, 2016; Ewunetu et al., 2021; Rockström et al., 2009, 2010). In Ethiopia, Anjeni catchment represents the conventional approach to SLM, where terraces have been installed in the catchment since 1986 to control water erosion exclusively on croplands (Figure 5). However, the success rates of such interventions are limited. In regions where water is a key limiting factor for rainfed crop production, such as eastern Africa, such basic SLM practices applied to control soil erosion exclusively on agricultural land have barely contributed to increased agricultural productivity (Abera et al., 2020; Adimassu et al., 2014; Gebreegziabher et al., 2009). Under conventional SLM practices, yield levels in rainfed systems remain low, at 0.5 – 1 t ha^{-1} (Rockström et al., 2009; SCRP, 2000). Hence, this accentuates the need for a new SLM approach geared towards managing rainwater and its partitioning instead of just runoff after it is formed (Rockström et al., 2009).

The fact that conventional SLM interventions in Ethiopia have contributed little to improved agricultural production and productivity has eventually promoted the need to shift to a new SLM strategy, where several technologies are integrated at the catchment scale to promote SLM (Ewunetu et al., 2021). For instance, the host of SLM practices put in place in Aba Gerima since

2012 represents the new SLM approach implemented in Ethiopia (Figure 11), where terraces are integrated with fodder species on cultivated land to not only control soil erosion but also make productive use of these structures and in the meantime ensure the stability of the structures (Figure 11a). In addition, grasslands have been protected from open communal grazing and enriched with vegetative species in Aba |Gerima to promote fast recovery of these degraded sites (Figure 11b).

Given soil degradation has causes and impacts beyond the onsite scale, so is the target of the new SLM approach to ensure that all land use types within a catchment are actively managed, including croplands, forest lands, grasslands, wastelands, and home gardens (Bossio et al., 2010; Rockström et al., 2010). This offers a large opportunity to not only address soil degradation problems specific to each land use type, but also to enhance the multi-functionality of agricultural landscapes. Hence, the emerging SLM approach promotes the need to maintain high spatial configuration of different land uses within a catchment, which is critical for harnessing potential benefits of SLM, ranging from improved availability of green and blue water resources to increased productivity and income.

A wide spectrum of SLM options are available for managing rainwater in rainfed systems around the globe to improve water storage in the root zone and, hence, increase grain yields (de Fraiture & Wichelns, 2010; Rockström et al., 2010). Enhanced rainwater infiltration and soil moisture availability in rainfed systems promoted by SLM interventions can boost water use efficiency by up to 25 – 40% and water productivity by 70 – 100% (Bossio et al., 2010). Experiences from sub-Saharan Africa reveal that shifting farming practices from conventional ploughing to conservation tillage has improved water productivity by 10 – 40% and increased grain yield by 20 – 120% (Rockström et al., 2010).

In Ethiopia, impacts of SLM interventions under varying environmental settings across the country have been documented in a considerable body of literature (Abera et al., 2020; Adimassu et al., 2014; Akale et al., 2017, 2019; Descheemaeker et al., 2006, 2008; Gebreegziabher et al., 2009; Gumma et al., 2021; Haregeweyn et al., 2012; Mekuria et al., 2015; Melaku et al., 2018; Nyssen et al., 2010; Rockström et al., 2009; Schmidt & Zemadim, 2015). For instance, a broad range of SLM interventions (e.g., soil and stone bunds, *Fanya Juu* terraces, vegetative strips,

conservation tillage, etc.) installed across many agricultural landscapes in the country are reported to have caused substantial reductions in surface runoff (Abera et al., 2020; Adimassu et al., 2014; Akale et al., 2017, 2019; Descheemaeker et al., 2006, 2008; Gebreegziabher et al., 2009; Haregeweyn et al., 2012; Melaku et al., 2018; Nyssen et al., 2010; Schmidt & Zemadim, 2015) and soil loss (Abera et al., 2020; Adimassu et al., 2014; Akale et al., 2017; Gebreegziabher et al., 2009; Gumma et al., 2021; Haregeweyn et al., 2012; Melaku et al., 2018; Schmidt & Zemadim, 2015) and increases in soil organic carbon (Abera et al., 2020; Adimassu et al., 2014; Mekuria et al., 2015). However, not all of these SLM interventions have been effective in enhancing agricultural productivity. More specifically, soil bunds and *Fanya Juu* terraces installed on agricultural lands as standalone practices revealed little positive gains in agricultural productivity (Abera et al., 2020; Adimassu et al., 2014; Rockström et al., 2009). By contrast, increased grain and biomass yields have been recorded in areas where engineering structures have been integrated with vegetative measures (Abera et al., 2020; Gumma et al., 2021; Haregeweyn et al., 2012; Mekuria et al., 2015; Rockström et al., 2009). Other impacts commonly reported in the literature include increased base flow (Akale et al., 2019) and groundwater recharge (Akale et al., 2019; Nyssen et al., 2010) and enhanced vegetation cover intensity (Descheemaeker et al., 2006, 2008; Gumma et al., 2021; Haregeweyn et al., 2012; Mekuria et al., 2015).

Over the past several decades, SLM interventions have expanded substantially across diverse geographies within Ethiopia. However, impacts studies in the past have been largely concentrated in the Ethiopian Highlands, where major investments in SLM have been made (Abera et al., 2020; Akale et al., 2017, 2019; Descheemaeker et al., 2006, 2008; Ewunetu et al., 2021; Gumma et al., 2021; Haregeweyn et al., 2012; Mekuria et al., 2015; Nyssen et al., 2010). Overall, though the geographical distribution of SLM interventions across the country covers over 24 agro-ecologies, impact studies so far are limited to only around nine agro-ecologies (Abera et al., 2020). Hence, this suggests past studies lack adequate geographical representation to clearly understand the cumulative impacts of SLM interventions in the country.

Besides, impact studies in the past in relation to hydrologic responses of SLM interventions have exclusively been focused on the blue water component of the terrestrial water cycle (Adimassu et al., 2014; Akale et al., 2017, 2019; Descheemaeker et al., 2006, 2008; Gebreegziabher et al.,

2009; Haregeweyn et al., 2012; Melaku et al., 2018; Nyssen et al., 2010). As a result, the green water component has been overlooked despite its dominant role in sustaining the country's agriculture, livelihoods, and economy. In fact, it is not adequately understood how SLM interventions have affected both green water storage and flow across the diverse ecological settings of the country. Hence, more impact studies are required in the future to bridge these and other research gaps. Especially, given food production in Ethiopia needs to keep pace with its fast-growing population in the face of a changing and highly variable climate, impact studies related to green water flow as well as storage are critical for planning and implementing adequate mitigation and adaptation strategies to enhance the country's rainfed agricultural productivity.

1.4. Research Questions

The present investigation addresses the following key research questions:

- Can SLM interventions modify hydrologic response in the Blue Nile Highlands, where traditional agricultural practices prevail?
- Can SLM help improve agricultural water security in an intensively cultivated rainfed system in the Blue Nile Highlands?
- How effective have SLM practices been in the long term in improving water availability and storage in the Blue Nile Highlands?

1.5. Scope of the study

The Blue Nile Basin has been described as the ‘‘great unknown” due to limited research and data availability. Since recently, however, the basin is increasingly attracting more research attention, with research interventions dominated by the application of hydrologic simulation models, such as SWAT, whose application has been limited by the absence of long-term field data for calibration and verification. Collection of long-term hydrometeorological data in the Blue Nile Basin has been limited to a small number of sites irrespective of the vastness of the basin and the topographic diversity prevailing within the basin. As a result, hydrologic simulation modeling studies in the Blue Nile Basin often end up in unjustified mass balance results.

Again, available historical data in the basin are dominated by runoff, sediment loading, and meteorological data, with some critical components of the hydrologic balance, such as soil moisture, often ignored. Hence, we only have a very limited body of recorded data to reliably apply simulation modeling to understand the hydrologic impact of land management. In the present study, we combined historical hydrometeorological data with newly recorded ones, including some new variables to complement the historical data and partly fill some critical data gap, such as soil moisture data, to understand the impact of sustainable land management (SLM) on hydrologic response and green water security in the Blue Nile Highlands using the case of three rainfed agricultural headwater catchments, which have been under varying land management conditions. Empirical techniques were applied in the present study, mainly focusing on investigating: (1) infiltration-runoff dynamics; (2) soil moisture dynamics; and (3) soil moisture spatial and temporal variability as influenced by catchment management, including the evolution of SLM interventions over the years.

1.6. Research Objectives

1.6.1. General objective

SLM practices implemented in Ethiopia for the past several decades focused on managing (or controlling) runoff (blue water), which is a major erosion agent washing away fertile topsoil from cultivated fields in the Ethiopian Highlands. Since recently, however, the focus of SLM practices has shifted from exclusively managing runoff as an erosion agent to managing rainfall to reduce nonproductive rainwater losses. This implies a paradigm shift, where SLM practices are increasingly gaining importance as a water management strategy for upgrading agricultural water security and productivity in the subsistence rainfed production systems dominating the Ethiopian Highlands. With this in mind, the present study was motivated by the need to mainstream water security into SLM initiatives in Ethiopia now and in the future for meeting the challenge of feeding the country's expanding population who depend largely on subsistence rainfed farming for their livelihoods and food needs. Hence, the major objective of the present study is to understand the impact of SLM on green water availability and storage in the Blue Nile Highlands, which have increasingly been under intensive cultivation to meet the food demand of the growing population.

1.6.2. Specific objectives

The present investigation seeks to address the following research objectives:

- To assess the impact of SLM on infiltration-runoff and soil moisture dynamics in intensively cultivated agricultural catchments in the Blue Nile Highlands;
- To investigate the impact of SLM on soil moisture spatial and temporal variability in intensively cultivated agricultural catchments in the fragile Blue Nile Highlands; and
- To examine the evolution in SLM interventions over the years and associated impacts on hydrologic response in the Blue Nile Highlands.

1.7. Structure of the Thesis

The present thesis is organized into seven main chapters. Chapter 1 presents a general introduction about the area of investigation addressed in the present thesis and highlights the research questions and objectives. Chapter 2 describes the study area. Chapter 3 details the data and methods. Chapter 4 examines the impact of SLM on infiltration-runoff and soil moisture dynamics. Chapter 5 determines soil moisture spatio-temporal variability as influenced by catchment management. Chapter 6 investigates the evolution of SLM practices in the Blue Nile Highlands over the years and associated impacts on hydrologic response. Finally, general conclusions are drawn in Chapter 7.

2. THE STUDY AREA

The present study involved three headwater catchments in the Blue Nile Basin located in the Anjeni and Aba Gerima areas, where two of the study catchments have received SLM treatments while one has been managed traditionally. Figure 1 presents the location map of these catchments. The treated catchments include Anjeni catchment (113.4 ha) located ~103 km south of Lake Tana and another catchment (900 ha) located ~6 km southeast of Lake Tana in Aba Gerima area. The third catchment (358.9 ha) with a history of traditional management was installed adjacent to the treated Aba Gerima catchment to provide a control counterpart for both treated catchments.

The treated Aba Gerima catchment has been under SLM interventions since 2012 while Anjeni catchment received SLM treatments in 1986 (Hurni et al., 2005). Anjeni catchment was established in 1984 by the Soil Conservation Research Program (SCRP) as one of the seven benchmark sites installed across the various agro-ecological zones of the country to conduct research to support national efforts to curb land degradation (Hurni et al., 2005; SCRP, 2000). On the other hand, the control catchment, which was installed in 2012 adjacent to the treated Aba Gerima catchment, has been under traditional farming practices since time immemorial.

The selection of two study catchments under improved management conditions stems from the need to account for the evolution in SLM approaches over the years. In Anjeni catchment, SLM treatments were introduced 36 years ago (1986), which exclusively targeted farmlands to control soil erosion using different forms of terraces. By contrast, the approach adopted in the treated Aba Gerima catchment 10 years ago (2012) focused on treating every land use type (grazing land, cropping land, home garden, etc.) within the catchment to enhance landscape productivity and restore natural hydrologic functionality. The control catchment, on the other hand, represents the traditional farming practices that persisted for centuries in the study area, presenting a contrast to both treated catchments.

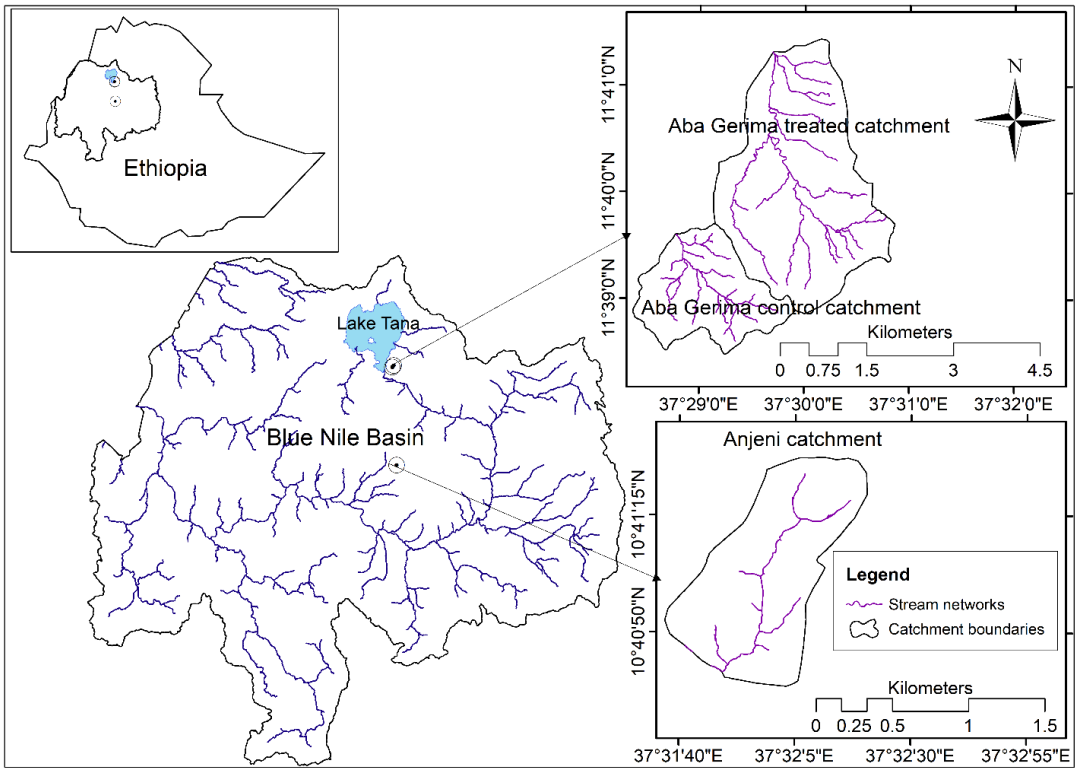


Figure 1. Location map of the study catchments.

2.1. Geomorphology and soils

Originating at 1780 m above mean sea level from Lake Tana, and joined by several tributaries along its path, the Blue Nile River forms the Blue Nile Basin (Elshamy et al., 2009; Haregeweyn et al., 2017), which comprises a large portion (~67%) of the Ethiopian Highlands (Conway, 1997). With a size of approximately 185000 km^2 (Betrie et al., 2011; Elshamy et al., 2009), the Blue Nile Basin makes up about 17% of the country's land area (Conway, 2000; Tabari et al., 2015) and 8% of the entire Nile Basin (Gebrehiwot et al., 2011). However, with a discharge of 48.5 km^3 yr^{-1} (Conway, 2000), the Blue Nile River is a major tributary of the River Nile, contributing ~60% of its annual flows (Elshamy et al., 2009).

Geologically, the basin is composed mainly of three different formations, including volcanic, sedimentary, and Precambrian basement rocks (Conway, 2000; Gani et al., 2009). While deep valleys in the southern part of the basin are underlain by sedimentary formations, comprising about 11% of the basin, a major part of the basin (52%), including the north, central, and eastern

parts, is composed of volcanic rocks (Gebrehiwot et al., 2011). The lower part of the basin is covered by crystalline basement rocks exposed over 32% of the basin's catchment area (Haregeweyn et al., 2017).

The basin's morphology is an outcome of various tectonic episodes and geomorphic processes that took place in the remote past, including volcanic eruptions and major structural faults (Gani et al., 2009). Swelling and folding processes together with volcanic lava flows led to the formation of extremely diverse topography within the basin, with elevation ranging from 500 m above mean sea level at Ethiopian-Sudanese border to 4620 m at Mount Ras Dejen (Conway, 1997). About 67% of the basin's area lies above 1500 m above mean sea level, making up a highland environment whose topography is dominated by plateaus and steep terrains (Gebrehiwot et al., 2011; Haregeweyn et al., 2017). The remaining part of the basin is a lowland environment in the west composed of gentler topography (Gani et al., 2009).

Given the rocks in the Blue Nile Basin are predominantly volcanic ashes and reworked materials resulting from Tertiary and Quaternary volcanic eruptions, the dominant soil texture in the basin is described as clay (Gebrehiwot et al., 2011). The dominant soil types are Alisols (21%), Leptosols (21%), Nitosols (17%), Vertisols (15%), and Cambisols (9%) (Betrie et al., 2011; Haregeweyn et al., 2017). Alisols, Leptosols, and Cambisols dominate the high relief areas and the northern part of the basin whereas Nitosols cover the western Blue Nile Highlands that are underlain by deeply weathered bedrocks. On the other hand, the foot slopes of the basin are dominated by Vertisols (Zaitchik et al., 2012).

Under undisturbed conditions, soils in the Blue Nile Basin tend to be deep, with natural depths exceeding 50 cm over most parts of the basin (Zaitchik et al., 2012). However, the soils are highly susceptible to erosion, especially on lands cultivated traditionally using the animal-drawn maresha plow, where erosion rate exceeds soil regeneration rate by a factor of 4 – 10 times (Rockström et al., 2009). High rates of soil erosion in the basin have resulted in reduced soil water availability and soil nutrient levels, subsequently leading to reduced agricultural productivity (Rockström et al., 2009; Zaitchik et al., 2012).

2.2. Hydroclimate

The climate in the Blue Nile Basin is described as tropical highland monsoon with a single rainy season spanning from June to September (Nawaz et al., 2010; Zaitchik et al., 2012). The rainy season starts upon the northward movement of the inter-tropical convergence zone (ITCZ), bringing moist air masses from the Indian and Atlantic oceans, which are responsible for about 70% of the annual rainfall in the basin (Betrie et al., 2011; Conway, 2000). Beginning in October, the ITCZ moves southward, resulting in a long dry season extending from October to May (Conway, 2000). The seasonality of rainfall is a key determinant of the seasonal distribution of runoff in the Blue Nile River, where 80% of its annual discharge occurs during July – October (Betrie et al., 2011; Nawaz et al., 2010). While the smaller streams within the Blue Nile Basin exhibit rapid runoff responses, with little baseflow, or often drying out, in the dry season, the peaks of runoff at the basin's outlet usually occur in August, exhibiting a lag effect by one month from the peaks of rainfall (Conway, 1997).

Based on rainfall data recorded at 11 raingauge stations over the period from 1900 to 1998, annual rainfall for the Blue Nile Basin varied from 1148 to 1757 mm (Conway, 2000). Such high inter-annual variability of precipitation is driven by the effects of the El Nino Southern Oscillation (ENSO) and, perhaps, Indian Ocean Sea Surface Temperature (SST) events (Zaitchik et al., 2012).

Precipitation during the June – September rainy season plays a pivotal role in rainfed agricultural production in the Blue Nile Highlands, sustaining 85 – 95% of the annual crops grown in the basin (Block & Rajagopalan, 2007). However, the Blue Nile Highlands experience high erosion rates during the rainy season stemming from poor farming practices. High erosion rates in the Blue Nile Highlands have in the long term led to reduced agricultural productivity onsite and excessive sedimentation of reservoirs downstream (Zaitchik et al., 2012). As a result, despite the presence of Lake Tana, with a considerable reservoir capacity (3000 km^2 surface area) subject to high evaporation losses, runoff ratio of the Blue Nile Basin in the long term remains at around 20% (Conway, 2000; Nyssen et al., 2010; Takele et al., 2021). Such a high runoff ratio suggests that runoff response is quick in the basin promoted by land degradation arising from poor farming practices (Conway, 2000; Elshamy et al., 2009; Nyssen et al., 2010).

Apart from ITCZ, elevation appears to be another determinant factor influencing spatial variability of climate in the Blue Nile Basin. Climate contrasts in the basin are high between the lowlands and the highlands, with temperature falling by 5.8 ˚C for every 1000 m rise in elevation (Conway, 2000). Hence, this has led to a temperate climate in the highlands and a tropical climate in the lowlands (Conway, 1997). Besides, the diurnal range of temperature exceeds the seasonal range in the basin, with temperature exhibiting little fluctuation throughout the year (Conway, 2000). Average annual temperature in the basin is 20.8 ˚C (Nawaz et al., 2010). In addition, topography influences rainfall spatial variability, presenting a climate ranging from semiarid to humid, with annual rainfall correspondingly varying from 800 mm in the lowlands to 2200 mm in the southwest (Betrie et al., 2011; Haregeweyn et al., 2017; Zaitchik et al., 2012).

Total annual potential evapotranspiration (PET) of the basin has been estimated at around 1300 mm and actual evapotranspiration (AET) at 900 mm (Elshamy et al., 2009), with AET being equal to PET only during the wet season when there is enough moisture in the soil.

2.3. Farming system

Agriculture in Ethiopia is highly traditional dominated by a crop-livestock mixed system (Rockström et al., 2009; Zaitchik et al., 2012). Particularly, the cropping systems throughout the Ethiopian Highlands use the ancient Ethiopian *maresha* plough for seedbed preparation (Figure 2). While this simple animal-drawn implement is well suited for breaking through hard dry tropical soils with vertic nature (Zaitchik et al., 2012), it involves operations that often lead to high erosion rates. Soil cultivations using the traditional *maresha* implement are carried out at a constant depth of 12 – 15 cm, leading to the development of plough pans (Figure 3), potentially reducing the soil's infiltration and water retention potentials. Besides, the frequency of cultivation required to grow most grain crops in the area is high, affecting soil physical and hydraulic properties. Taking *tef*, for instance, a weed sensitive staple grain crop in the study area with fine grains, farmers often cultivate the soil up to five times and subsequently practice animal trampling in order to secure an even and weed free seedbed (Rockström et al., 2009). Given these operations are carried out over an extended period until the beginning of August when the most erosive storm events are received, the process from seedbed preparation to planting of *tef* leaves little ground cover to protect the soil against the most erosive storms. This

coupled with the traditional practice of open grazing on communal lands (Figure 4) has in the long term led to a high level of soil degradation in the study area.



Figure 2. Soil cultivation using the animal-drawn *maresha* plow. This traditional implement can only plow to a constant depth of 10 – 15 cm, which leads to the development of plow pans. Besides, soil cultivation using the *maresha* plow is carried out up to 5- 6 times prior to planting, which represents a high level of soil disturbance. The overall implication is reduced rainwater infiltration and soil water storage, leading to green water scarcity (photo by Berihun D. Mersha, 18 May 2017).

The cumulative effect of soil degradation in the study area has led to reduced soil depth and organic matter content, with adverse impacts on infiltration and water holding capacity, and hence, on water productivity, availability and storage. Depletion of soil organic matter content is responsible for the collapse of soil structure, affecting total soil porosity and infiltration and water holding capacity (Bossio et al., 2010). Such changes in soil structure and porosity in turn increase the risks of water scarcity resulting from increased rainwater losses through non-productive pathways (Biazin et al., 2012; Verhoeven, 2013). Non-productive rainwater losses through soil evaporation, surface runoff, and deep drainage resulting from poor farming practices could be in the order of 70 – 85% of rainfall, leading to reduction in the amount of green water

(soil moisture) available for productive transpiration (Rost et al., 2009). Observations in the Ethiopian Highlands reveal that nonproductive rainwater losses in the form of surface runoff from cultivated land are 5 – 30 times higher than losses from forest land (Hurni et al., 2005). Soil degradation in Ethiopia contributes greatly to low level of land productivity, which is in the range 0.5 – 1.1 t ha^{-1} (Rockström et al., 2009).



Figure 3. Cropland affected by rill erosion in Aba Gerima control catchment. The rill exposes a shallow soil layer (about 10 cm) with a plough pan underneath. The field is planted with *tef*, which is at its early growth stage. Clearly, rainwater infiltration and soil water storage would be limited by the plough pan and reduced soil volume, respectively, at this site. The overall impact is reduced soil moisture to meet the requirement of the *tef* crop (photo by Berihun D. Mersha, 28 August 2017).



Figure 4. Open communal grazing in Aba Gerima control catchment. Open communal grazing is a traditional practice in the study area, where every farm animal is left to graze freely on communal land. The outcome is reduced ground cover arising from overgrazing and diminished soil porosity due to trampling (compaction) by livestock, leading to a disturbed natural hydrologic functionality, with increased runoff response and reduced rainwater infiltration (photo by Berihun D. Mersha, 7 July 2017).

3. DATA, MATERIALS, AND METHODS

3.1. Study Design

Globally, impact studies of anthropogenic activities on catchment hydrology have widely involved either long-term before- and after-intervention monitoring of a single catchment or a paired-catchment approach involving two or more catchments (Minella et al., 2008; Van Loon et al., 2019). The long-term time trend approach involving a single catchment has often been limited by its inability to separate treatment effects from climate effects (Neary, 2016; Stednick, 1996; Whitehead & Robinson, 1993). By contrast, the paired-catchment approach allows for climatic variability to be accounted for (Folton et al., 2015; Van Loon et al., 2019; Yurtseven et al., 2018). In addition, the paired-catchment approach requires relatively shorter time periods ranging from a few months to years for the analysis of treatment effects (Neary, 2016; Van Loon et al., 2019).

Conventionally, the paired-catchment approach provides an experimental investigation of the effects of intentional treatments on catchment hydrology (Van Loon et al., 2019). However, most human activities taking place at the catchment scale are not applicable as intentional treatments. Besides, lack of monitoring before the start of human activities under investigation often limits the availability of pre-treatment data. Under such circumstances, the paired-catchment approach can be applied based on existing observational data with satisfactory results given proper care is taken during the selection of study catchments (Van Loon et al., 2019).

In the present study, a paired-catchment approach was applied involving three agricultural headwater catchments in the Blue Nile Basin, where two of the study catchments have received SLM options (treated catchments) while the third catchment has been under traditional farming practices (a control catchment). The treated catchments include Anjeni catchment, which has received SLM measures since 1986, and the Aba Gerima treated catchment, which has been under SLM since 2012. The two treated catchments were chosen to represent the evolution of SLM practices in the study area over the years, where the target of SLM interventions has markedly shifted from controlling soil erosion on agricultural land (Anjeni catchment) to managing every land use type within a catchment to enhance productivity and restore natural

hydrologic functionality (treated Aba Gerima catchment). Hence, the SLM measures implemented in Anjeni catchment were predominantly structural (soil bunds and *Fanya Juu* terraces) to control soil erosion on cultivated land through runoff management while the treated Aba Gerima catchment involved the treatment of every land use type (including both agricultural and nonagricultural) within the catchment to improve productivity and restore natural hydrologic functionality. Thus, soil bunds, which have been installed as a standalone measure to control soil erosion on cultivated land in Anjeni catchment (Figure 5), have been planted with fodder species in the treated Aba Gerima catchment (Figure 11a) to stabilize the structures and in the meantime produce biomass for animal feed. In the treated Aba Gerima catchment, the traditional practice of post-harvest grazing has also been abandoned on cultivated land to facilitate the buildup of soil cover and organic matter content, which is critical for improving soil hydrology. Besides, open communal grazing has been entirely eliminated to allow fast recovery of marginally degraded sites (Figure 11b). Further, home gardens have been introduced in the treated Aba Gerima catchment as a cash-generating scheme for the local land users.



Figure 5. Terraces on cultivated land in Anjeni catchment. These structures have been in place since 1986, providing substantial protection to the soils from water erosion; however, they contribute little to improving the soils' physical properties and, hence, are less efficient in providing a water management role (photo by Berihun D. Mersha, 17 December, 2009).

3.2. Data

Anjeni catchment was established in 1984 to investigate the impact of SLM measures on controlling soil erosion on cultivated land. The catchment was instrumented, with weather and streamflow data being monitored since 1984, including during a two-year period (1984 – 1986) prior to the implementation of soil and water conservation measures (SCRP, 2000). As a result, a considerable amount of hydrometeorological data has been generated for the catchment for the last 38 years now. Apart from the generation of weather and hydrosedimentological data at the catchment scale, monitoring of runoff and soil loss data had been carried out intensively at the plot scale in Anjeni catchment for an extended period between 1984 and 1993. On the other hand, the remaining two study catchments were established adjacent to each other in 2012 in the Aba Gerima locality, where one has been maintained under traditional management (control catchment) while the other has received SLM treatments since 2012 (treated catchment). These

two study catchments have also been instrumented to monitor climate data since 2014 and runoff and soil moisture data during the 2017 – 2018 period.

The present analyses were carried out based on a considerable amount of runoff and soil moisture data generated at the plots scale. The experimental plots were equipped with facilities required for monitoring these data, such as corrugated iron sheet borders and runoff collection tanks to collect runoff data (Figure 6) and access tubes along with a capacitance type PR2/4 profile probe and a HH2 display unit to monitor soil moisture data (Figure 7).



Figure 6. Experimental plot equipped with an iron sheet border and runoff collection tanks installed on an open grazing land (Anjeni).



Figure 7. The PR2/4 capacitance probe (a) and HH2 display unit (b) deployed for measuring soil moisture (Aba Gerima).

3.3. Methods of Analysis

Conventionally, paired-catchment studies have been used in the analysis of the impacts of anthropogenic activities on blue water flows (annual, monthly, low, and base flows) and water quality (Brown et al., 2005). Subsequently, field-scale overland flow data have also been involved in paired-catchment studies (Almeida et al., 2016; Dung et al., 2012). More recently, paired-catchment studies have expanded to include new variables and areas of investigation (Van Loon et al., 2019).

In the present study, a paired-catchment approach has been employed to investigate the impact of SLM practices on improving green water security, which is critical for sustaining ecosystem

goods and services upon which humans depend, including rainfed agriculture. Runoff and soil moisture data along with climate data were used to determine whether implementation of SLM has contributed to improving green water security in the Blue Nile Highlands, where rainfed agricultural production is highly constrained by management-induced water scarcity. An event-based water budget analysis together with a range of statistical data analysis techniques were applied to evaluate the impact of SLM on improving green water availability and storage in the study area.

4. ASSESSING THE EFFECT OF SUSTAINABLE LAND MANAGEMENT ON IMPROVING WATER SECURITY IN THE BLUE NILE HIGHLANDS: A PAIRED-CATCHMENT APPROACH[†]

Berihun D. Mersha[1, 2,*], Gete Zeleke[2], Tena Alamirew[1, 2], Zeleke A. Dejen [1], Solomon G. Gebrehiwot[1, 2]

4.1. Abstract

The Blue Nile Highlands, Ethiopia, has been experiencing serious land degradation, menacing water security and then human well-being. However, sustainable land management (SLM) may be the way to curb land degradation and improve water security. Therefore, in order to assess benefits after a 5-year catchment restoration effort, we conducted a paired-catchment study to investigate runoff and soil moisture dynamics. First and second catchments were used as control and treated, respectively. After comparing observations gathered from four sites within each of the study catchments, we found that implementing SLM reduced runoff curve numbers by -13.9 to -21. 6 units and increased soil moisture storage by 15.6 to 800%, then promoting rapid recovery of the hydrologic functionality of the natural landscapes. We conclude that SLM initiatives can greatly improve water security for the rainfed agriculture prevailing in the drought-prone Blue Nile Highlands.

4.2. Introduction

Water resources are under stress worldwide. The problem is posing a major threat when it comes to the African continent and more specifically to the eastern African region (Vörösmarty et al., 2005). Situated in the eastern African region, Ethiopia is seriously affected by water stress,

---

[†] Chapter 4 has been published in the "Environmental Monitoring and Assessment" Journal: Environ Monit Assess (2022), 194(3), 197. https://doi.org/10.1007/s10661-022-09837-5.

1 Ethiopian Institute of Water Resources (EIWR), Addis Ababa University

2 Water and Land Resource Center (WLRC), Addis Ababa University

[*] Corresponding author. Address: Water and Land Resource Center (WLRC), Addis Ababa University, Addis Ababa, Ethiopia. Mobile: +251-47 23 05 60; P.O.Box: 3880. E-mail address: berihun2010@gmail.com.

resulting from land degradation and climate variability and change (Conway & Schipper, 2011; Simane et al., 2016; Tabari et al., 2015). About 25% of the country's land area is degraded as a result of deforestation, overgrazing and inadequate agricultural practices (Kirui et al., 2021). Water erosion and soil moisture depletion are the main forms of land degradation affecting freshwater availability in Ethiopia (Betrie et al., 2011; Hurni et al., 2005). Climate variability and change further compound the problem by increasing the irregularities in the onset and cessation of rainfall and, hence, increasing the frequencies of agricultural drought (Asfaw et al., 2018; Conway & Schipper, 2011; Dile et al., 2013). Historically, the country suffered recurrent droughts that led to famine and loss of life (Conway & Schipper, 2011; Wagesho et al., 2013).

Sustaining over 90% of the country's population on around 50% of the country's land area, the Ethiopian Highlands present a fragile ecosystem susceptible to degradation and climate variability and change (Asfaw et al., 2018; Hurni et al., 2005). With about 70% of its area lying in the Ethiopian Highlands, the Blue Nile Basin has been seriously affected by land degradation, subsequently witnessing marked changes in its hydrology (Bewket & Sterk, 2005; Nyssen et al., 2010; Rientjes et al., 2011). The basin receives major significance in Ethiopia and beyond due to its greatest water yield relative to the other basins in Ethiopia, with a contribution of about 60% of the flow of the Nile (Conway, 2000). Owing to deforestation and subsequent agricultural expansion, streamflows in the Blue Nile Basin have experienced significant decreases in annual and low flows and increases in high flows, particularly in its headwater catchments (Bewket & Sterk, 2005; Rientjes et al., 2011). Furthermore, in spite of a substantial surface storage capacity (25.6 billion m^3) provided by Lake Tana (source of the Blue Nile River), the Blue Nile Basin has the highest runoff coefficient relative to other basins of similar extent in Ethiopia (Nyssen et al., 2010), highlighting the adverse impact of land degradation on the hydrology of the basin.

The impact of climate change on water resources in the Blue Nile Basin has been revealed by observed reductions in wet season (June – September) rainfall on which streamflows and rainfed agriculture are heavily dependent (Tabari et al., 2015). Climate projections suggest that apparent decreases in the basin's precipitation could result in likely reductions in streamflows (Dile et al., 2013; Setegn et al., 2011). In addition, potential increases in temperature and PET are expected to increase the water stress in the Blue Nile Basin (Elshamy et al., 2009).

In order to improve the water security status in Ethiopia, there is a need to put in place appropriate mitigation and adaptation strategies for combating land degradation and climate variability and change. Historically, the drought-induced famines of 1973 and 1974 promoted the massive implementation of sustainable land management (SLM) initiatives in Ethiopia, which are still widely implemented across many regions of the country, such as the Blue Nile Highlands, through community labor (Abera et al., 2020). Over the years, SLM practices in Ethiopia have evolved from field-scale structural measures (e.g., soil and stone bunds) to an integrated catchment-scale approach (Akale et al., 2019; Haregeweyn et al., 2012). In the past, the impacts of SLM practices have been assessed under different ecological settings using different methods and approaches. For instance, Gumma et al. (2021) assessed land use dynamics, sediment deposition, biomass productivity and vegetative cover intensity across treated and untreated catchments, employing satellite imagery and ground data. Haregeweyn et al. (2012) mapped land use and land cover dynamics and assessed associated changes in runoff and soil loss rates. Jan Nyssen et al. (2010) examined catchment response to SLM interventions using catchment-scale and plot-scale runoff data coupled with water table depth measurements. Akale et al. (2019) used simulation modeling to understand the impact of SLM interventions on surface runoff and groundwater flow.

These studies documented that SLM interventions are effective in curbing land degradation and restoring the ecological functionality of agricultural landscapes. However, while the ecological benefits of SLM interventions are well documented, their role as a water management strategy is inadequately investigated. The Ethiopian Highlands are famously diverse in topography and associated local climate contrasts (Berhane et al., 2014; Haregeweyn et al., 2017). Therefore, there is a need for more research interventions to better understand the water management role of SLM interventions in these diverse ecosystems.

An empirical approach was adopted in the present study to examine the dynamics in water balance resulting from the implementation of SLM practices. The relevance of using such an empirical approach lies in the fact that the application of simulation modeling in the study area is limited by the paucity of long-term data (Akale et al., 2019; Conway, 2000; Dile et al., 2018). The overall objective of this study is to understand the implication of implementing SLM for improving water security in the fragile tropical Blue Nile Highlands. The specific objectives are

to: (1) quantify the impact of SLM on infiltration-runoff dynamics; and (2) evaluate the influence of SLM on soil moisture dynamics.

4.3. Materials and Methods

4.3.1. Description of the study catchments

A paired-catchment approach was adopted in the present study, where two agricultural catchments were selected in the headwaters of the Blue Nile basin, Ethiopia, ~6 km southeast of Lake Tana, source of the Blue Nile River (Figure 8). While making up only ~8% of the area of the entire Nile basin (Gebrehiwot et al., 2011), the Blue Nile basin contributes the largest flow to River Nile (~60%; 48.5 billion m^3 $year^{-1}$) draining a large portion of the Ethiopian Highlands, which make up ~ 67% of the basin's area (Betrie et al., 2011; Conway, 2000; Hurni et al., 2005; Schmidt & Zemadim, 2015).

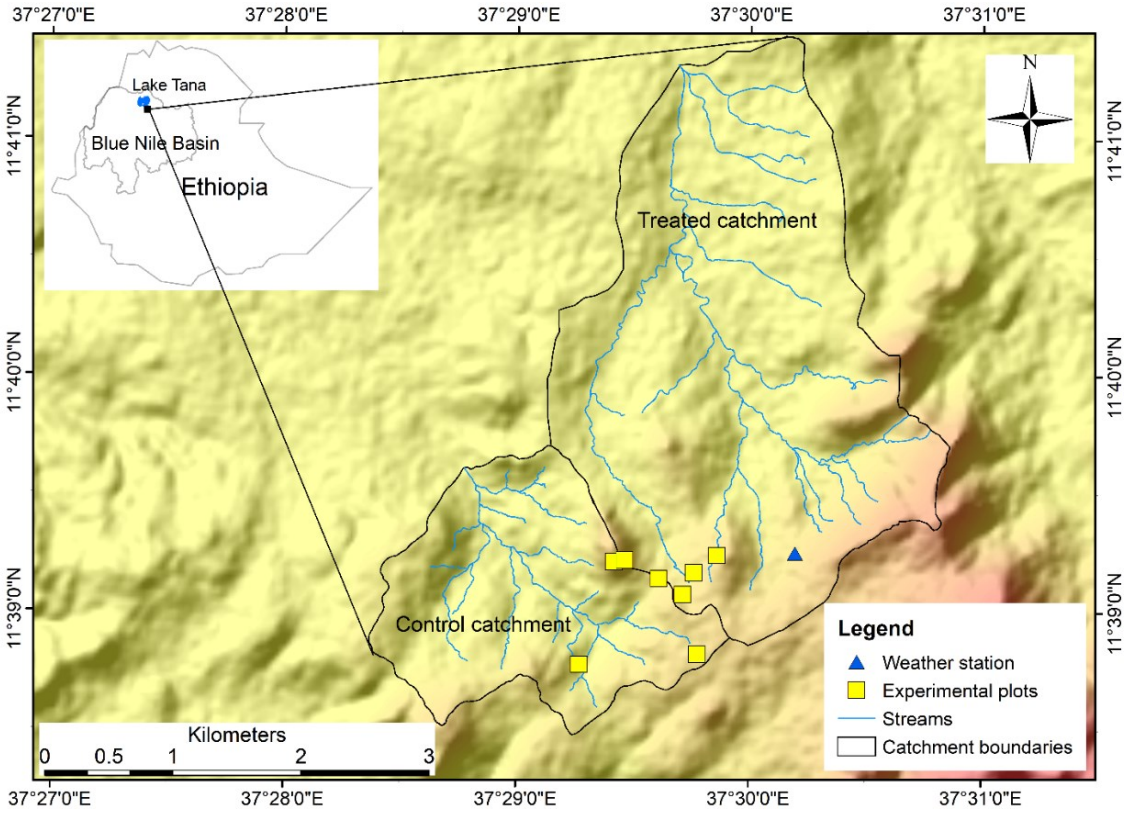


Figure 8. Location map of the Aba Gerima paired catchments.

The Aba Gerima paired catchments share many similar characteristics, including soils, land uses, drainage direction, climate and topography. The landscapes of both catchments are dominated by rolling topography with degraded and shallow soils due to chronic water erosion. The major soils include Regosols (>27%), Cambisols (>13%), and Leptosols (>7%). The largest parts of both catchments are under subsistence rainfed crop cultivation (finger millet, maize, *tef,* wheat), covering ~69.5% in the control catchment and ~66% in the treated catchment. The second largest land use in both study catchments is grazing land, comprising ~17.4% of the control catchment and ~18.7% of the treated catchment. Both catchments drain northwest to Lake Tana, source of the Blue Nile River. While sharing many similar biophysical characteristics, the study catchments have been managed differently, where the first catchment (358.9 ha) has been managed traditionally (control catchment) since time immemorial while the second (900 ha) has received improved catchment management options since 2012 (treated catchment). Figure 9 presents the landuse patterns of the study catchments.

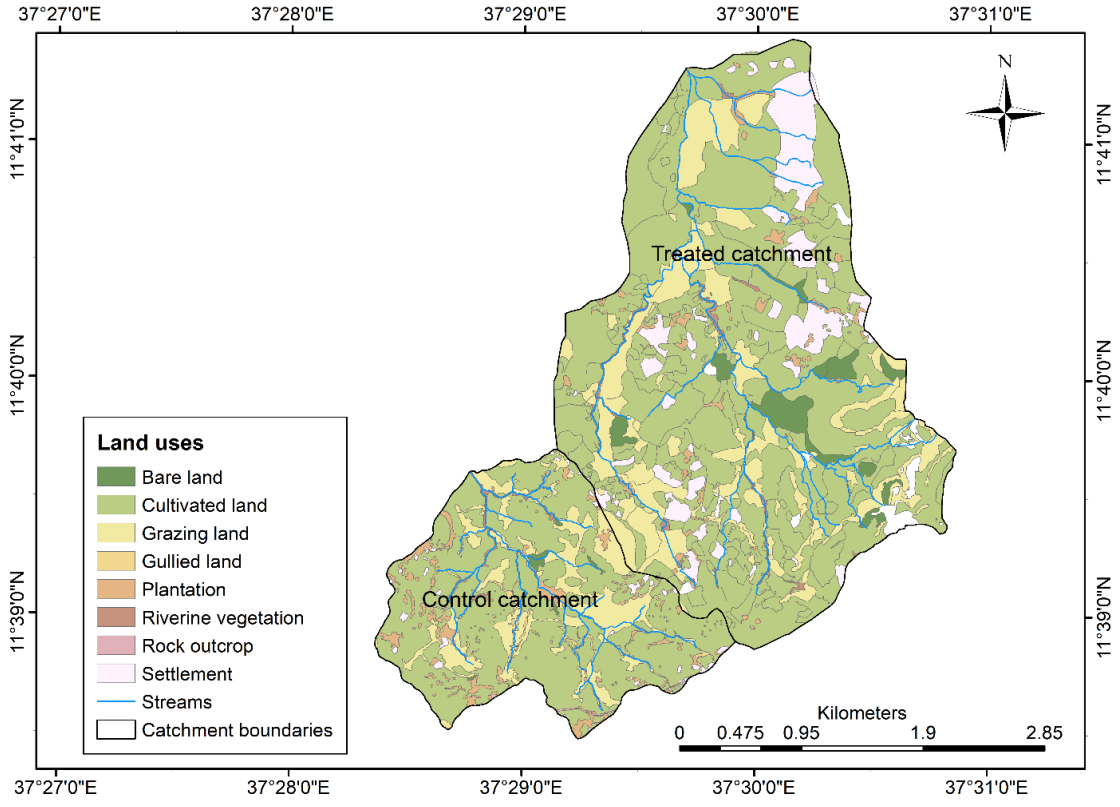


Figure 9. Landuse patterns of the Aba Gerima paired catchments.

The climate of the study area is characterized by a highly seasonal unimodal rainfall pattern modulated by the Inter Tropical Convergence Zone (ITCZ) (Conway, 2000; Nyssen et al., 2005). The rainy season, which extends from June – September, begins upon the northward migration of ITCZ, contributing about 80% of the total annual rainfall. The southward movement of ITCZ results in the commencement of the long dry season (October – May), during which about 20% of the annual rainfall is received. At a mean of about 20 ˚C, temperature exhibits little seasonal variation in the study area.

The Aba Gerima paired catchments are under contrasting land management conditions, where one of the catchments (900 ha) has been under SLM since 2012 (treated Aba Gerima catchment) while the other (358.9 ha) has been managed traditionally since time immemorial (control Aba Gerima catchment). The distribution of the structural SLM measures (soil bunds; contour trenches) implemented in the treated catchment has been mapped in Figure 10.

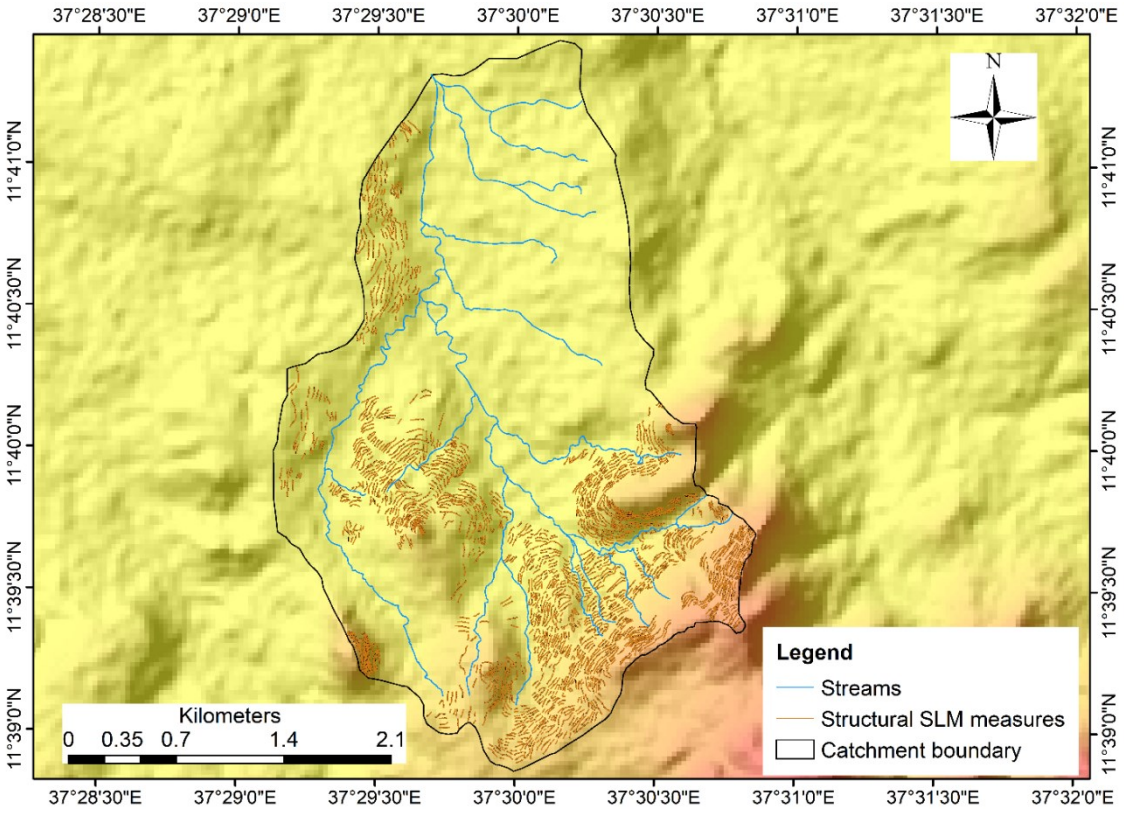


Figure 10. Distribution of structural SLM measures (soil bunds; contour trenches) in the treated Aba Gerima catchment.

Figure 11 illustrates some of the SLM practices put in place in the treated Aba Gerima catchment. Soil bunds integrated with live hedges have been installed on cultivated land. While protecting the structures, the live hedges serve to supply feed for livestock. The bunds are connected to natural channels via a network of waterways to safely dispose excess runoff into the natural channels. The traditional practice of post-harvest grazing has been abandoned on cultivated land to protect the live hedges. On marginally degraded areas, elimination of open grazing has been implemented along with contour trenching. The elimination of open grazing has led to the emergence of a new practice where local land users feed their animals by cutting the biomass instead of allowing them to graze freely. In addition, sustaining the spatial configuration of different land uses at the catchment scale has received importance in the new SLM approach, introducing home gardens as a major component of the land management system in the study area, with a cash generating role.

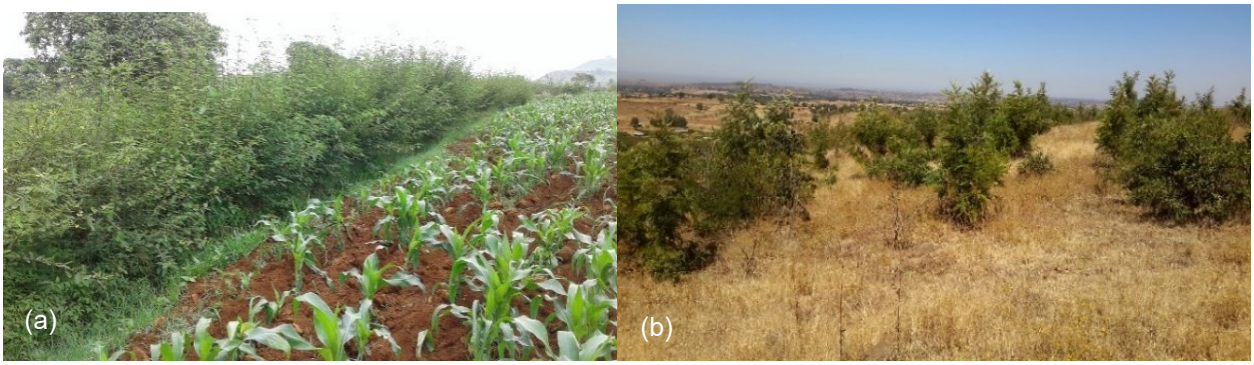


Figure 11. Some of the SLM practices implemented in the treated Aba Gerima catchment: (a) soil bund integrated with live hedges on cultivated land; (b) marginally degraded communal land protected from open grazing for recovery.

The traditionally managed (control) Aba Gerima catchment lacks any of the SLM practices installed in the treated Aba Gerima catchment, except some spillover effects, such as the abandonment of open grazing on some patches of marginally degraded hillslopes due to complete loss of productivity. Traditionally, mixed farming is practiced in the study area, where the local land users rely on subsistence grain production and farm animals for their livelihood. In the traditional farming practice, the land is intensively cultivated without fallow for grain

production and animals are left to graze in the open. As a result, the soil surface is devoid of protective cover, due to the removal of crop residues and overgrazing.

4.3.2. Study design

To understand the impact of SLM on water balance, one could conduct long-term monitoring during pre- and post-intervention years. However, this approach is deemed costly and time-consuming. In this particular study, a relatively simpler approach was adopted, where a catchment under SLM interventions was selected along with an adjacent control catchment to simultaneously monitor processes during post-intervention years so that space can replace time (Hajabbasi et al., 1997).

Within each of the study catchments, four runoff plots were installed, three of them on cultivated land with 5, 10, and 15% slope gradients and one plot on a marginally degraded hillslope (17% slope gradient) with a use history of open communal grazing. Biophysical characteristics of the experimental sites are summarized in Table 1. Care was taken to install all the runoff plots reasonably close together (Figure 8) in order to minimize variations in weather and other site factors that could affect the experiment. Each runoff plot was isolated by iron sheet borders, with roofed concrete structures constructed at their downstream ends to collect runoff. In addition, each experimental plot was equipped with two access tubes for soil moisture data monitoring.

Each runoff plot in the treated catchment was paired with a counterpart in the control catchment. Figure 12 illustrates the layout of a pair of treated and control runoff plots installed on cultivated land. Sizes of all runoff plots was 15m length by 6m width (90 m^2); but, sizes of the runoff collection structures were made to vary according to runoff potentials of the experimental sites, with the smallest structure having a size of 2 m^3 and the largest 3.75 m^3. The runoff collection structures provided sufficient capacity for all runoff events throughout the experimental period.

4.3.3. Data collection

The present study involved the collection of mainly rainfall, runoff, and soil moisture data. Rainfall data was collected since 2014 using an automatic raingauge station located close to the

experimental sites (Figure 8). The automatic raingauge is a HOBO RG3-M type (USA), which records rainfall events at 10 minutes resolution and 0.2 mm precision.

The runoff plots were used to generate runoff data after every causal rainfall event during the 2017 and 2018 rainy seasons. Depth of the runoff collected in the concrete structures installed at the downstream ends of the runoff plot was multiplied by area of bottom of the structure to determine a runoff volume (m^3), which was, then, converted to a normalized direct runoff depth (mm) by dividing the runoff volume by area of the runoff plot.

Soil moisture data was collected at two points per runoff plot using the PR2/4 capacitance type soil profile probe, which is especially suited for use on mineral soils (Delta-T Devices Ltd, 2016). Prior to the deployment of the PR2/4 profile probe, sensor evaluation was made against thermogravimetric soil moisture measurements taken at multiple soil profile depths within selected sites in the study catchments. Sensor evaluation yielded a highly significant association ($p < 0.001$) between sensor readings and thermogravimetric soil moisture measurements (coefficient of determination, $R^2 = 0.72$; mean bias error, MBE = 0.097 $m^3\ m^{-3}$). Then, the PR2/4 profile probe was used along with a portable HH2 display unit to take readings of volumetric soil moisture content on a daily basis at 10, 20, 30 and 40 cm soil profile depths.

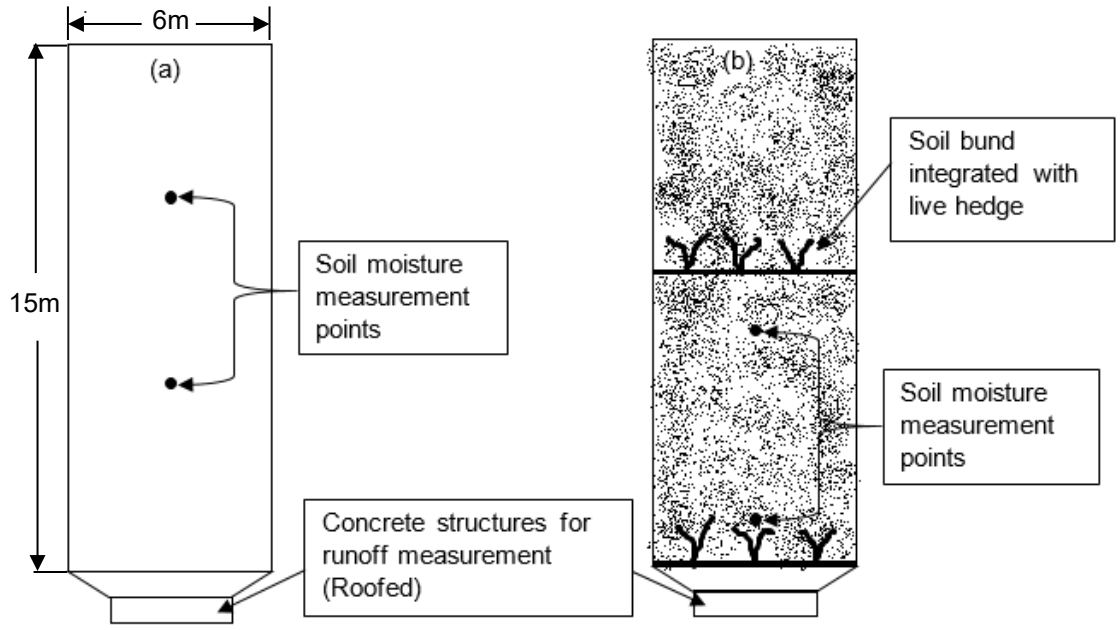


Figure 12. Layout of a pair of runoff plots installed on cultivated land: (a) control runoff plot; (b) treated runoff plot.

4.3.4. Data analysis

4.3.4.1. Infiltration-runoff dynamics

An event-based water budget analysis as described in Equation (4.1) was used to assess the impact of SLM practices on infiltration-runoff dynamics in the study area. This approach is deemed appropriate for events of sufficiently short duration where other components of the water budget, such as evapotranspiration, are negligible (Tedela et al., 2012; USDA-NRCS, 2004; Yuan et al., 2001).

$$Q = P - I_a - F \tag{4.1}$$

where,

Q = direct event runoff depth (mm),

P = causal event rainfall depth (mm),

I_a = initial abstraction (rainfall amount recorded before runoff begins, mm), and

F = cumulative infiltration exceeding initial abstraction (mm).

Combined with two fundamental hypotheses, Equation (4.1) yields the expression described in Equation (4.4), which is famously known as the curve number (CN) method developed by the Natural Resources Conservation Service (NRCS) of the United States Department of Agriculture (USDA) (Tedela et al., 2012; USDA-NRCS, 2004). The two hypotheses are: (1) the ratio of direct runoff to potential maximum runoff is equal to the ratio of infiltration to storage index (Equation (4.2)); and (2) the initial abstraction is some fraction of the storage index (Equation (4.3)).

$$Q/(P - I_a) = F/S \tag{4.2}$$

$$I_a = \lambda S \tag{4.3}$$

where,

λ = initial abstraction ratio (dimensionless),

S = storage index (mm).

$$Q = (P - \lambda S)^2/[P + (1 - \lambda)s] \tag{4.4}$$

The standard CN method assigns a value of 0.2 to λ, which reduces Equation (4.4) to Equation (4.5) (USDA-NRCS, 2004):

$$Q = (P - 0.2S)^2/(P + 0.8S) \quad \text{If P} > 0.2\text{S} \tag{4.5a}$$

$$Q = 0 \text{ If P} \leq 0.2\text{S} \tag{4.5b}$$

In the presence of observed precipitation and runoff data, S could be determined for any P-Q data pairs using Equation (4.6), which is obtained by solving Equation (4.5a) for S, where $0 < Q < P$. Then, S is transformed into a dimensionless CN parameter using Equation (4.7).

$$S = 5[P + 2Q - (4Q^2 + 5PQ)^{0.5}] \quad (4.6)$$

$$CN = 25400/(254 + S) \quad (4.7)$$

Theoretically, the value of CN varies from 0 (indicating zero runoff) to 100 (indicating zero infiltration) as S varies from ∞ to 0 (Hawkins, 1993). Standard handbook values of CN are officially maintained by NRCS, reflecting different combinations of soil hydrologic groups and land management conditions. However, CN values derived from observed P-Q data pairs are more reliable than the handbook values for local conditions (Elhakeem & Papanicolaou, 2009; Hawkins, 1993; Oliveira et al., 2016; Tedela et al., 2012).

When CNs are calculated from observed P-Q data pairs, a strong secondary relationship often emerges between CN and P, with CN decreasing asymptotically to a constant value with increasing P (Bonta & Shipitalo, 2013; Bonta, 2013; Hawkins, 1993; Oliveira et al., 2016; Tedela et al., 2012). Hence, the P-CN data pairs should be fitted to Equation (4.8):

$$CN(P) = CN_\infty + (100 - CN_\infty)\exp(-kP) \quad (4.8)$$

where,

CN_∞ = asymptotic CN (dimensionless) representing the runoff potential of large rainfall events; and

k = fitting parameter that quantifies the rate at which CN approaches CN_∞ (mm^{-1}).

The asymptotic curve number (CN_∞) is a function of the limiting interaction of vegetation, land management, and surface and subsurface soil and percolation characteristics for large P (Bonta & Shipitalo, 2013; Bonta, 2013).

Typically, the asymptotic curve of P-CN data pairs cannot plot below an enveloping value of zero runoff (CN_o), which is calculated using Equation (4.9) for observed P-CN data pairs, where P = 0.2S for Q = 0 (Bonta & Shipitalo, 2013; Bonta, 2013; Hawkins, 1993).

$$CN_o = 25400/[254 + (P/0.2)] \quad (4.9)$$

Since Equation (4.8) decays exponentially, the minimum precipitation (P_{min}) above which CN_∞ is applicable can be computed at the point where 99% of CN_∞ is reached using Equation (4.10).

$$P_{min} = -ln(0.01)/k \tag{4.10}$$

In the present study, observed CNs were determined for individual treatments using the frequency-matching approach (Hawkins, 1993), where the observed P-Q event pairs were sorted independently in descending order and merged to calculate S and CN using Equations (4.6) and (4.7), respectively. Then, Equation (4.8) was used to make an asymptotic curve fitting for the P-CN data pairs of individual treatments.

Table 1. Selected characteristics describing the experimental sites.

Location	Geographic coordinates		Land use	Slope (%)	Soil depth (cm)	Soil characteristics						
	Latitude (˚N)	Longitude (˚E)				Sand fraction (%)	Silt fraction (%)	Clay fraction (%)	Organic carbon (%)	Dry bulk density (kg m^{-3})	K_{sat} of limiting layer (mm h^{-1})	Hydrologic soil group
Treated catchment	11.6513	37.4961	Cultivated land	5	0-10	34	38	28	1.44	1164	4.91	C
					10-20	26	40	34	1.6	1249		
					20-30	26	30	44	1.37	1261		
					30-40	16	22	62	0.9	1138		
	11.6540	37.4977	Cultivated land	10	0-10	26	42	32	2.15	1223	5.44	C
					10-20	14	30	56	1.44	1057		
					20-30	26	34	40	1.44	929		
					30-40	36	26	38	0.74	994		
	11.6512	37.4953	Cultivated land	15	0-10	36	38	26	1.83	1178	1.86	D
					10-20	24	26	48	0.70	1235		
					20-30	34	26	40	0.27	1253		
					30-40	28	26	46	0.39	1192		
	11.6536	37.4911	Degraded hillslope	17	0-10	36	40	24	2.81	1262	0.02	D
					10-20	32	40	28	2.61	1086		
					20-30	28	38	34	1.83	1134		
					30-40	22	42	36	1.29	969		
Control catchment	11.6523	37.4936	Cultivated land	5	0-10	14	24	62	0.94	1338	4.35	C
					10-20	24	20	56	1.05	1064		
					20-30	34	18	48	0.59	1074		
					30-40	38	20	42	1.52	914		
	11.6470	37.4963	Cultivated land	10	0-10	24	26	50	1.68	1019	2.78	D
					10-20	30	26	44	1.87	1036		
					20-30	42	26	32	0.98	1117		
					30-40	46	26	28	0.90	1023		
	11.6463	37.4879	Cultivated land	15	0-10	60	20	20	0.74	1064	2.33	D
					10-20	44	38	18	2.69	1184		
					20-30	62	22	16	0.43	1082		
					30-40	64	22	14	0.47	1237		
	11.6535	37.4903	Degraded hillslope	17	0-10	58	32	10	2.89	1020	2.00	D
					10-20	60	20	20	2.59	1032		
					20-30	30	40	30	2.46	934		
					30-40	18	36	46	1.76	888		

4.3.4.2. Soil moisture dynamics

Soil moisture operates the subsistence rainfed agriculture in the study area. Hence, assessing soil moisture dynamics is critical for understanding the impact of SLM on improving water availability. In the present study, daily readings of volumetric soil moisture content (θ), which were taken at two measurement points and four monitoring depths per experimental site, were used to derive different values of θ as indicated in Equations (4.11) to (4.14). These values were the bases for further analyses to understand the impact of SLM on soil moisture dynamics.

For each measurement occasion, mean volumetric soil moisture content at a monitoring depth within an experimental site was determined as:

$$\bar{\theta}_{i,t} = \frac{1}{n}\sum_{j=1}^{n} \theta_{i,j,t} \quad (4.11)$$

where,

i = monitoring depth, j = measurement point, t = measurement occasion and

n = total number of measurement points per experimental site.

In addition, the mean volumetric soil moisture content of a site at each measurement occasion was calculated as:

$$\bar{\theta}_{s,t} = \frac{1}{nd}\sum_{j=1}^{n}\sum_{i=1}^{d} \theta_{i,j,t} \quad (4.12)$$

where,

i = monitoring depth, j = measurement point,

t = measurement occasion,

n = number of measurement points, and

d = number of measurement depths.

Further, the temporal mean of soil moisture content of a monitoring depth within an experimental site was calculated as:

$$\bar{\theta}_i = \frac{1}{nT}\sum_{j=1}^{n}\sum_{t=1}^{T}\theta_{i,j,t} \quad (4.13)$$

where,

i = monitoring depth,

j = measurement point,

t = measurement occasion,

n = number of measurement points, and

T = number of measurement occasions.

Again, the temporal mean of soil moisture content of an experimental site was calculated as:

$$\bar{\theta}_s = \frac{1}{ndT}\sum_{j=1}^{n}\sum_{i=1}^{d}\sum_{t=1}^{T}\theta_{i,j,t} \quad (4.14)$$

where,

i = monitoring depth,

j = measurement point,

t = measurement occasion,

n = number of measurement points,

d = number of measurement depths, and

T = number of measurement occasions.

4.3.4.3. Statistical methods

In the present study, comparisons between contrasting treatment pairs were made by comparing their respective best-fit parameters (CN_∞ and k) of the asymptotic P-CN curves using a t test (Motulsky & Christopoulos, 2003). In this test, the null hypothesis is that both data sets have the same parameter, where one common model can describe the data from both treatments. On the other hand, the alternative hypothesis is that the parameter values are distinct, and therefore entirely separate response curves should be fitted to each data set. The differences between CN_∞ and k parameters of contrasting treatment pairs are considered significant if the t test yields $p \leq 0.05$. Otherwise, the two parameter values are considered the same. A significant difference in CN_∞ of contrasting treatment pairs highlights a real shift in infiltration potential of a site promoted by the change in land management (Bonta & Shipitalo, 2013; Bonta, 2013).

The t ratios for CN_∞ and k parameters were calculated using Equations (4.15) and (4.16), respectively.

$$t = \frac{|CN_{\infty c} - CN_{\infty t}|}{\sqrt{\sigma_c^2 + \sigma_t^2}} \tag{4.15}$$

where,

$CN_{\infty c}$ = asymptotic curve number of the control treatment,

$CN_{\infty t}$ = asymptotic curve number of the SLM treatment,

σ_c = standard error of estimate of the asymptotic curve number of the control treatment, and

σ_t = standard error of estimate of the asymptotic curve number of the SLM treatment.

$$t = \frac{|k_c - k_t|}{\sqrt{\sigma_c^2 + \sigma_t^2}} \quad (4.16)$$

where,

k_c = fitting parameter of the control treatment and

k_t = fitting parameter of the SLM treatment,

σ_c = standard error of estimate of the fitting parameter for the control treatment, and

σ_t = standard error of estimate of the fitting parameter for the SLM treatment.

Contrasting treatment pairs were also compared for soil moisture dynamics using the nonparametric Wilcoxon signed rank test. The use of a parametric test was deemed inappropriate due to the fact that the distribution of the soil moisture data sets did not fit the normal distribution.

4.4. Results and Discussion

4.4.1. Rainfall characteristics

Rainfall exhibited high seasonality in the study area, with the peaks occurring in July – August. The rainfall pattern for the years 2015 – 2018 is illustrated in Figure 13. During this period, high interannual variability in rainfall was also observed. The lowest annual amount (895.8 mm) was received in 2015 and the highest (1581.6 mm) in 2018, which varied by a margin of 685.8 mm (43.36%) relative to the highest value. Such a characteristic high intra- and inter-annual variability in rainfall is typical of the Ethiopian Highlands, with a significant impact on water resources and the subsistence rainfed agriculture in the region (Asfaw et al., 2018; Conway, 2000).

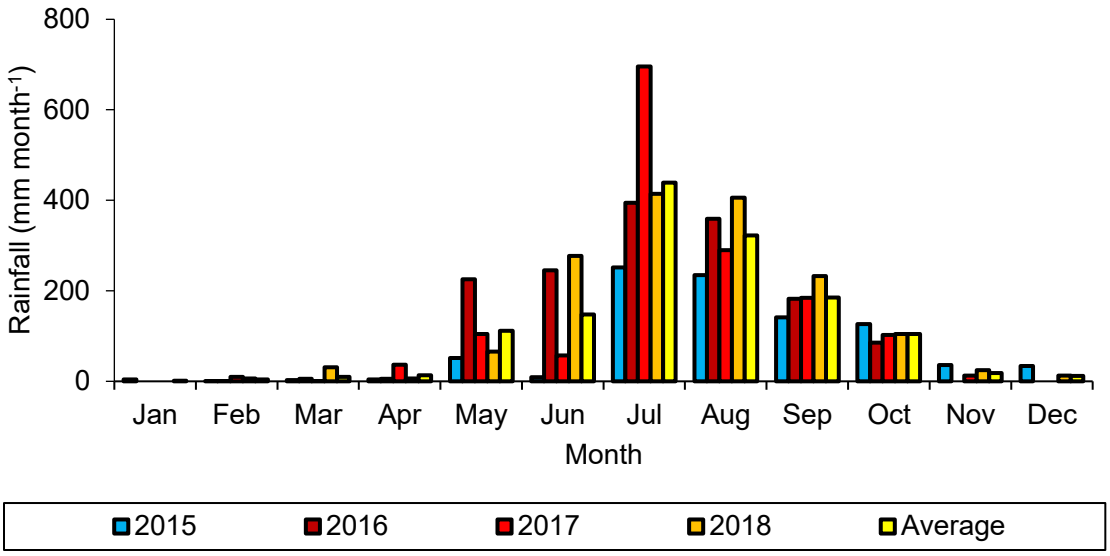


Figure 13. Rainfall pattern of the study area for the 2015 – 2018 period.

On the other hand, the amounts received during the experimental period (2017 – 2018) varied only by a relatively smaller margin (86.2 mm; 5.45% relative to the higher value). Characteristics of rainfall events received during the experimental period are summarized in Table 2.

Table 2. Characteristics of rainfall event received during the experimental period.

			Event depth (mm)		Event duration (min)		Event intensity (mm h^{-1})	
Year	Season	n	Mean	Standard deviation	Mean	Standard deviation	Mean	Standard deviation
2017	Dry	64	3.4	4.9	55.9	72.3	3.1	2.6
	Wet	105	9.3	16.3	98.1	108	4.7	4.4
2018	Dry	41	5.2	6.7	66.4	64.5	4.5	5.1
	Wet	108	10.1	11.8	111.9	96.6	5.3	4.8

4.4.2. Runoff characteristics

The entire event-based P-Q data pairs of all treatments under investigation are plotted in Figure 14. The data indicate that runoff potential is higher in the control than in the treated catchment. As a result, runoff volumes and number of events were lower under the treated than the control runoff plots. From field observations, we noted that runoff response to rainfall events was rapid in the study area with little indication of saturation excess runoff. This is consistent with what has been reported in Descheemaeker et al. (2006) for the Tigray Highlands, northern Ethiopia.

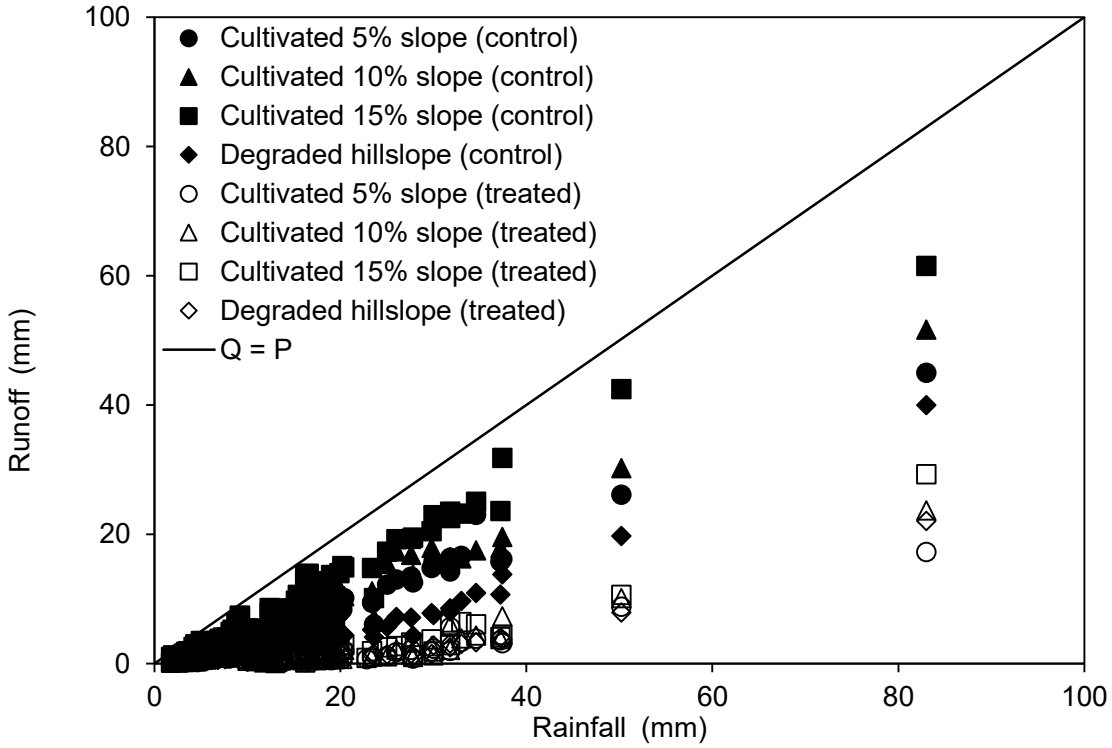


Figure 14. Scatter plot of P–Q data pairs of treatments under investigation.

4.4.3. Soil moisture characteristics

The soil moisture (θ) regimes of all experimental sites are illustrated in Figure 15. The data indicate that seasonal variation in rainfall is the main factor controlling the seasonal fluctuation of θ in the study catchments. Thus, the two peaks in Figure 15 indicate the two wet seasons that occurred at the beginning and end of the data period during which

the level of θ attained its maximum. The trough, on the other hand, represents the dry season that occurred between the two wet seasons during which θ experiences depletion. In Figure 15, it can be clearly noted that θ increased dramatically upon the start of the wet season; however, a lag effect in θ response was apparent for all treatments, where rainfall peaks occurred in July – August while θ peaks occurred in September – October. Such a lag effect in θ response is attributable to the fact that cultural activities are at their peak during the July – August period, leaving the soil surface devoid of protective cover in favor of high overland flow and evaporation losses. This, in turn, reduces the amount of rainwater infiltration and soil water storage.

Figure 15 demonstrates that the driest θ regimes are associated with the runoff plots on the marginally degraded hillslopes in both the control and treated catchments. These sites had a long history of use for open communal grazing, with the natural hydrologic functionality of the sites severely affected due to livestock trampling (soil surface compaction) and overgrazing (poor ground cover). However, the treated site presented a substantially higher mean daily θ (0.205 $m^3 m^{-3}$) than its control counterpart (0.104 $m^3 m^{-3}$), indicating recovery of the site promoted by SLM intervention.

The moistest θ regimes belonged to the experimental sites on cultivated land with 15% (mean daily θ = 0.369 $m^3 m^{-3}$; control catchment) and 10% (mean daily θ = 0.279 $m^3 m^{-3}$; treated catchment) slope gradients. Figure 15 illustrates that these two sites had little variation in θ regime during the two wet seasons of the data periods; however, the sites presented different θ regimes during the dry season. Despite a steeper terrain (15% slope gradient) and lack of best practices (no SLM practices) the site in the control catchment had a lower θ depletion rate (0.170 mm day^{-1}) than the site in the treated catchment (0.347 mm day^{-1}). The fact that the site in the control catchment lies at a lower elevation (1965 m above mean sea level) than the one in the treated catchment (2013 m) implies a likely occurrence of a shallow groundwater table at the site situated in the control catchment, where continuous replenishment of θ from the groundwater table is possible by capillary rise, resulting in a wetter and more stable θ regime.

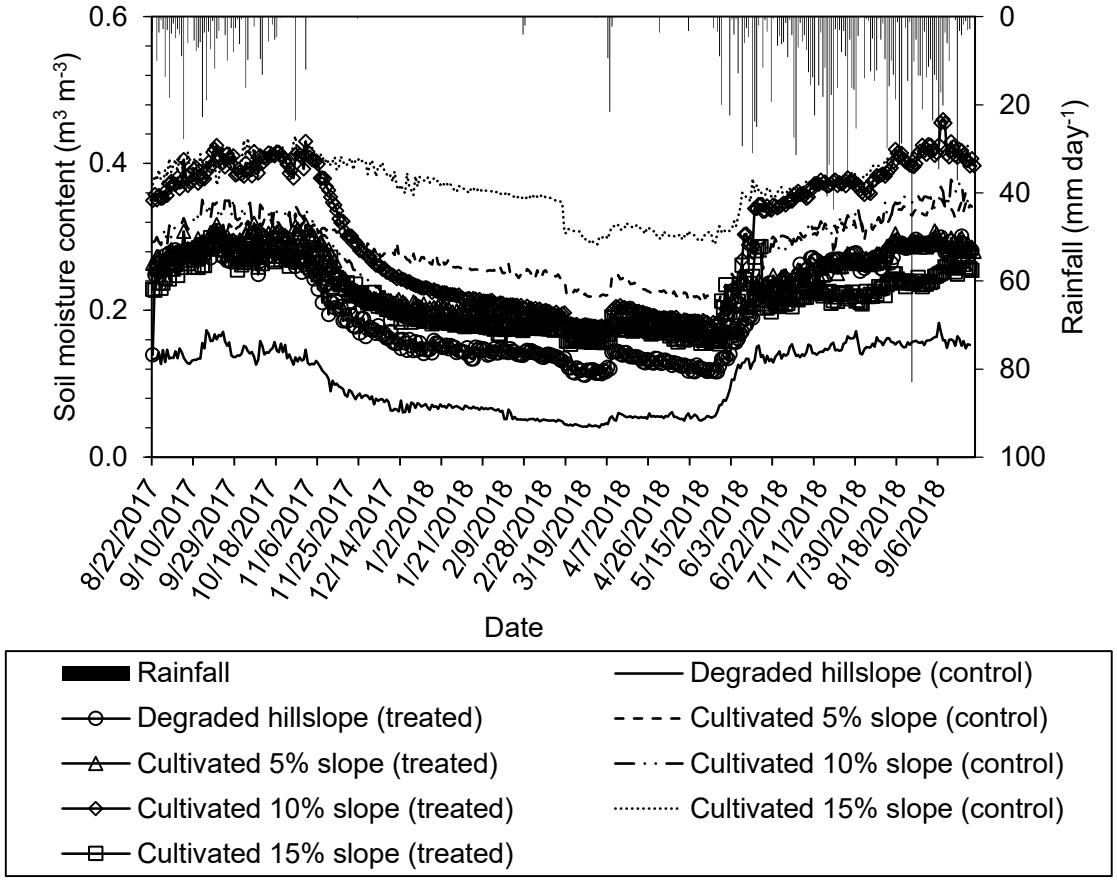


Figure 15. Soil moisture regimes of treatments under investigation.

4.4.4. Effect of SLM on infiltration-runoff dynamics

The two-parameter asymptotic curves fitted to the P-CN data pairs of all treatments were found to decrease asymptotically approaching a constant value of CN_{∞} as P becomes large. Such curves, which decrease asymptotically to produce a P-independent CN_{∞} as P becomes large, are referred to as standard asymptotic curves (Bonta & Shipitalo, 2013; Bonta, 2013; Hawkins, 1993; Oliveira et al., 2016; Tedela et al., 2012). The standard asymptotic curves fitted to the P-CN data of all contrasting treatment pairs in this investigation are presented in Figure 16. The curves of the treated sites plot below the curves of their control counterparts, clearly indicating the recovery in infiltration potential of the treated catchment promoted by SLM interventions.

The statistics describing the fitted asymptotic curves indicate that the standard asymptotic curve fitting procedure adequately described the P-CN relationships for all treatments (Table 3). Runoff plots in the treated catchment presented high P_{min} and low CN_∞ and k parameter values. In contrast, low P_{min} and high CN_∞ and k parameter values were associated with runoff plots in the control catchment. The highest P_{min} (114.2 mm) and lowest CN_∞ (67.9) and k (0.04 mm^{-1}) values were determined for the marginally degraded site in the treated catchment which is currently closed for livestock and humans. Minimum surface disturbance coupled with regeneration of natural vegetation at the site promoted recovery of the infiltration potential of the site (substantially reduced CN_∞). On the other hand, the control runoff plot on cultivated land with 15% slope gradient produced the lowest P_{min} (31.3 mm) and highest CN_∞ (96.6) and k (0.147 mm^{-1}) values, suggesting low infiltration potential due to steep terrain and high surface disturbance. Further, Table 3 reveals that the asymptotic CNs computed from observed local data have better represented actual site variations than the standard tabular values published in NRCS tables. Hence, unlike the tabular CNs, the asymptotic CNs presented different values for sites that are under similar management conditions but varying slope gradients.

Table 3. Fitting statistics of standard asymptotic curves for treatments under investigation.

			Parameter estimate		Standard error				
Catchment	Treatment	Slope (%)	CN_∞	k (mm^{-1})	CN_∞	k (mm^{-1})	P_{min} (mm)	R^2	Tabular CN
Treated	Cultivated land	5	68.6	0.044	0.872	0.002	104.6	0.94	77
		10	73.0	0.055	0.752	0.003	83.5	0.92	77
		15	75.1	0.055	0.396	0.002	84.1	0.97	77
	Degraded hillslope	17	67.9	0.040	0.616	0.001	114.2	0.98	65
Control	Cultivated land	5	90.2	0.066	0.360	0.004	70.0	0.88	85
		10	92.1	0.066	0.335	0.005	70.3	0.83	85
		15	96.6	0.147	0.120	0.013	31.3	0.66	85
	Degraded hillslope	17	81.8	0.043	0.295	0.001	107.1	0.99	71

Statistical comparisons made between the CN_∞ of all contrasting treatment pairs using the t test yielded significant differences at $p < 0.0001$; however, the k parameters of only two treatment pairs (cultivated land with 5 and 15% slope gradients) were significantly different at $p < 0.0001$ while the other treatment pairs produced k parameters that only varied marginally ($p = 0.035$; degraded hillslopes) and did not vary at all ($p = 0.99$; plots on cultivated land with 10% slope gradient). Since k is only a fitting parameter in the standard asymptotic curve fitting procedure (Hawkins, 1993), more importance was placed on the CN_∞ parameter to describe the infiltration-runoff dynamics resulting from the implementation of SLM in the study area.

The data indicate that all experimental sites in the treated catchment had lower CN_∞ values than their control counterparts. CN_∞ decreased by 13.9 units (from 81.8 for the control plot to 67.9 for the treated plot) on marginally degraded hillslopes; 21.6 units (from 90.2 to 68.6) on cultivated land with 5% slope gradient; 19.2 units (from 92.1 to 73.0) on cultivated land with 10% slope gradient; and 21.5 units (from 96.6 to 75.1) on cultivated land with 15% slope gradient. Similar reductions in CN were reported in the literature promoted by improved land management (Bonta & Shipitalo, 2013; Bonta, 2013; Dabney et al., 2012; Feyereisen et al., 2008). For instance, Feyereisen et al. (2008) found a reduction in CN under a strip tillage practice (CN = 71) than under conventional tillage (CN = 82). Dabney et al. (2012) reported reductions in CN from 78 (under a conventional corn system) to 70 (hedge only system) and further to 61 (hedge-berm system). Bonta (2013) noted a decrease in CN induced by the change in management from 29 years of pasture (77.0) to 3.7 years of meadow (59.6). Again, Bonta and Shipitalo (2013) demonstrated that a shift in management from a conventional to a no-till corn system reduced CN from 90.6 to 66.3.

The observed decreases in CN at sites in the treated catchment clearly indicate that implementation of SLM has contributed to the recovery of infiltration potential in the treated catchment. Owing to the structural measures (soil bunds, contour trenches, etc.), an increased fraction of the rainfall is captured in situ to readily infiltrate into the soil. In addition, the elimination of livestock trampling (soil compaction) in the treated catchment as a result of the abandonment of open grazing should help upgrade the water intake

capacity of the soils due to improved structure and porosity. In general, the data reveal that, while being susceptible to land degradation, the fragile Blue Nile Highlands also are highly responsive to management interventions, with the natural hydrologic functionality of the agricultural landscapes restored quite rapidly.

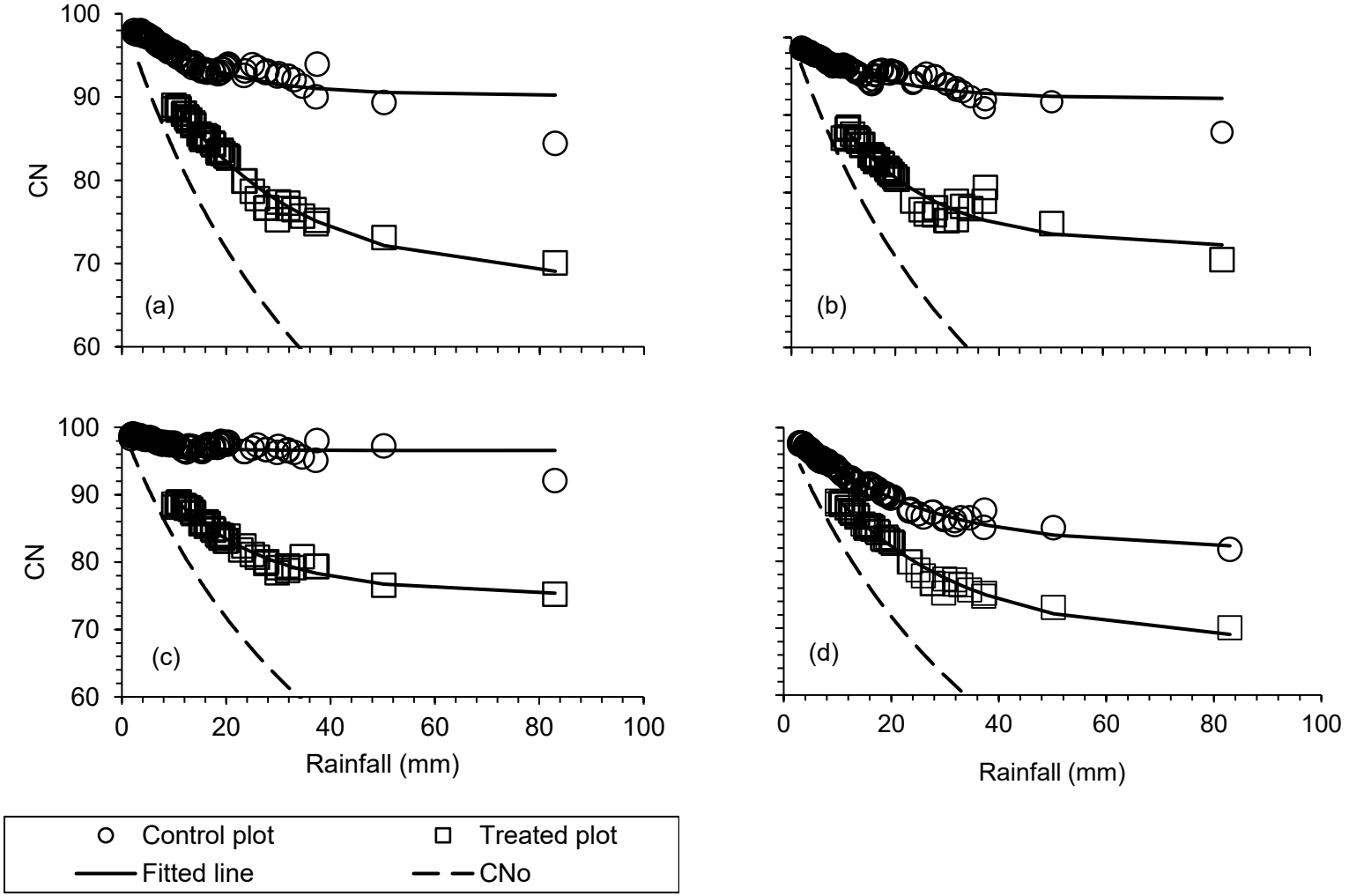


Figure 16. Asymptotic CN plots of contrasting treatment pairs located on: (a) cultivated 5% slope; (b) cultivated 10% slope; (c) cultivated 15% slope; (d) degraded hillslope.

4.4.5. Effect of SLM on soil moisture dynamics

The changes in soil moisture content (θ) relative to a reference value recorded at the onset of the 2018 wet season are presented in Figure 17, describing soil water recharge. Noticeably, θ increased with the increase in cumulative rainfall at all experimental sites, indicating the buildup of soil moisture as the wet season progresses. However, the experimental sites in the treated catchment exhibited higher soil water recharge than their control counterparts except the one on cultivated land with 15% slope gradient, which presented little increase in soil water recharge relative to its control counterpart (Figure 17d). Apparently, the control plot maintained exceptionally wet soil moisture condition (Figure 18), where θ at 40 cm depth exceeded field capacity (FC) throughout the experimental period. Site characteristics indicate that θ at FC of the treated (0.38 $m^3 m^{-3}$) and control (0.37 $m^3 m^{-3}$) sites were quite similar, suggesting that both sites have similar soil water retention characteristics. The exceptionally wet soil moisture condition at the control site, thus, implies that the site is receiving continuous groundwater flow via capillary rise.

Further, some control sites presented higher θ than their treated counterparts at some soil profile depths, such as cultivated land with 5% slope gradient (at 40 cm depth) and marginally degraded hillslopes (at 10 cm depth). This may be attributable to inherent soil properties, such as soil porosity, that favor increased soil water retention. It was found that total soil porosities of the control sites were higher than the porosities of their treated counterparts (e.g., 0.655 versus 0.571 for cultivated land with 5% slope gradient; and 0.615 versus 0.524 for marginally degraded hillslopes). This implies that some inherent soil physical properties have not been affected by the SLM interventions, probably due to the fact that the period is too short for such effects to be apparent, given the SLM interventions have been in place only since 2012.

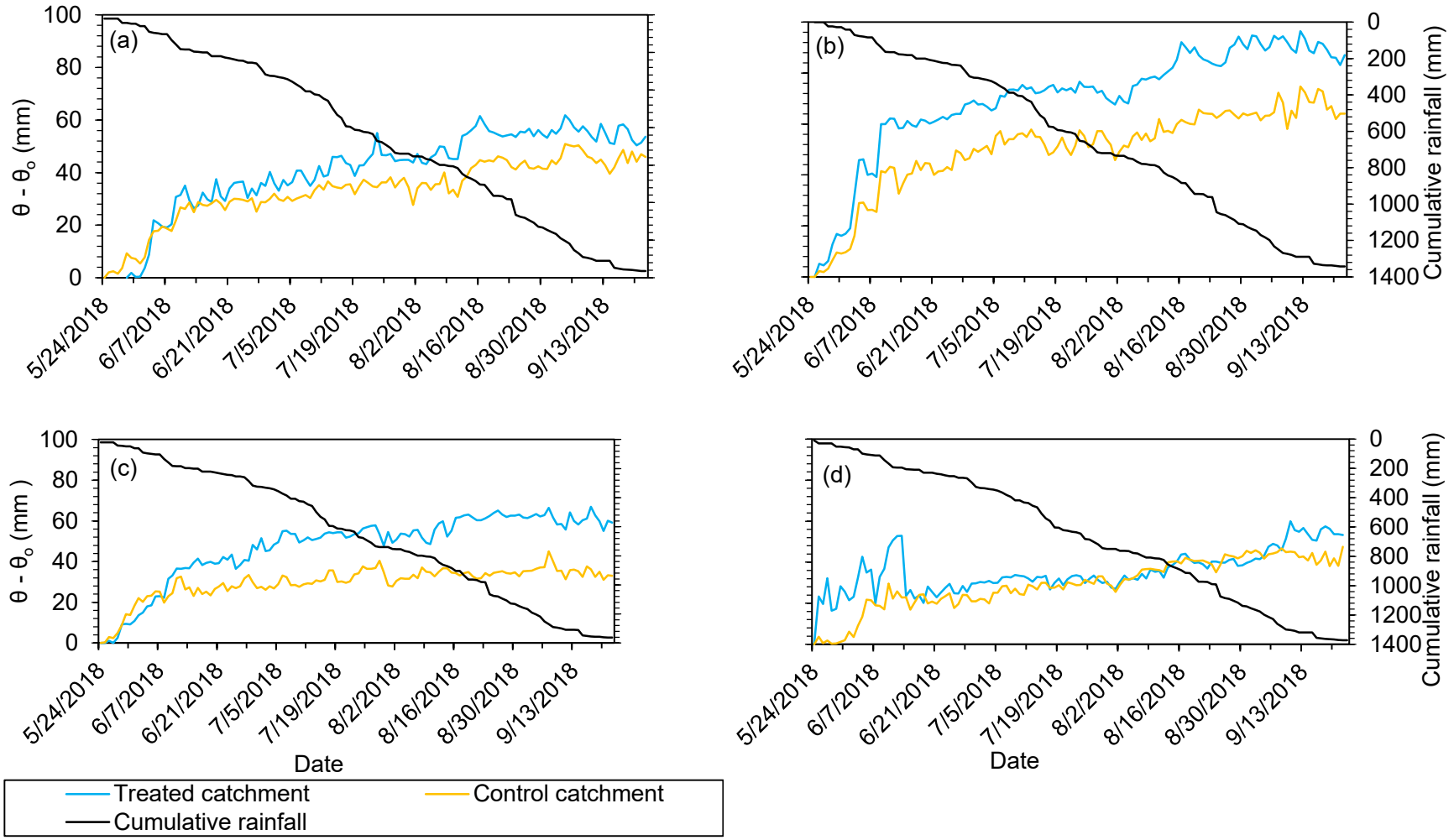


Figure 17. Comparisons of changes in θ between contrasting treatment pairs relative to their respective initial θ recorded at the onset of the 2018 wet season: (a) cultivated land with 5% slope gradient; (b) cultivated land with 10% slope gradient; (c) degraded hillslope (17% slope gradient); (d) cultivated land with 15% slope gradient.

SLM promoted increased θ in the treated catchment due to improved soil water recharge and retention, resulting from the recovery of the hydrologic functionality of the natural landscapes. Such changes in the hydrologic functionality of the agricultural landscapes in the treated catchment are caused by the in-situ management of overland flow through structural SLM measures (soil bunds, contour trenches, etc.), which contributed to increased soil water recharge. Besides, the abandonment of open grazing in the treated catchment promoted the accumulation of litter and soil organic matter on the soil surface, leading to reduced soil evaporation and increased soil moisture retention.

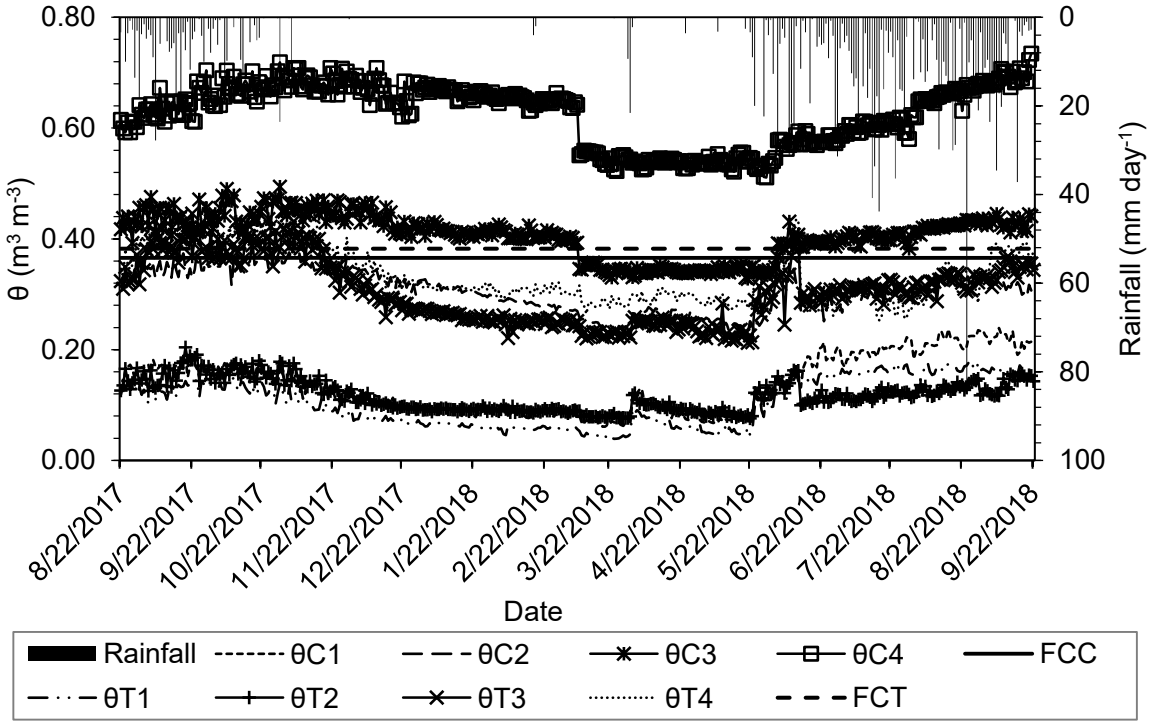


Figure 18. Soil moisture dynamics of treatment pairs on cultivated land with 15% slope gradient: θC1, θC2, θC3 and θC4 represent θ at 10, 20, 30 and 40 cm depths, respectively, of the control site while θT1, θT2, θT3 and θT4 indicate θ at 10, 20, 30 and 40 cm depths of the treated site; FCC and FCT describe θ at field capacity of the control and treated sites, respectively.

4.4.6. Implication of SLM for improving water security

The availability of freshwater resources in the Ethiopian Highlands has been undermined by perturbation of the water balance by land degradation and climate variability and change (Conway, 2000; Hurni et al., 2005, 2010; Seleshi & Zanke, 2004). As a result, water stress is affecting the vast majority of Ethiopians whose livelihoods are dependent on subsistence rainfed agriculture. The productivity of Ethiopia's rainfed agriculture, which accounts for 47% of the country's GDP, is highly affected by hydrologic variability and reduced freshwater availability (Dile et al., 2013; Verhoeven, 2013). Therefore, only a small fraction of the rainfall is available as soil moisture for consumptive use by plants due to erratic rainfall and nonproductive freshwater losses through soil evaporation, surface runoff, and deep percolation, leading to plant water stress and yield losses.

Mitigating the deleterious impacts of land degradation and climate variability and change on freshwater availability is critical for improving water security in the Ethiopian Highlands, where 90% of the country's population live, the majority of whom are subsistence farmers. There is a great untapped potential for improving the productivity of the subsistence rainfed agricultural system in the Ethiopian Highlands through appropriate adaptation strategies that can improve freshwater availability. This investigation reveals that SLM interventions have led to rapid restoration of the hydrologic functionality of the natural landscapes in the fragile tropical Blue Nile Highlands, with a great potential to bridge freshwater deficits constraining the productivity of the subsistence rainfed agriculture. Even though the implementation of SLM in the Ethiopian Highlands has been to primarily control rampant soil erosion, these practices can also provide a crucial role in managing freshwater resources.

The current practices of SLM in the Blue Nile Highlands involve a variety of structural and nonstructural elements integrated at the catchment scale, providing different roles in managing water resources in situ. For instance, the structural measures (soil bunds, contour trenches, etc.) significantly improve infiltration and water storage potential of the agricultural landscapes in the study area. On the other hand, the nonstructural measures,

such as the practice of area closure in communal grazing lands and the abandonment of post-harvest grazing on cultivated land, help improve the water retention capacity of the soils and reduce nonproductive evaporative water losses. In addition, the current practices of SLM attempts to sustain diverse landuse mosaics at the catchment scale, including protected areas (gullied lands and communal grazing lands), cultivated land and home gardens, with the aim of harnessing potential uses, services and values from a catchment (Haregeweyn et al., 2012). For instance, the development and expansion of home gardens in the study area targets the production of cash-generating crops.

Further, recovery of the storage potentials of the agricultural landscapes promoted by SLM would contribute to increased groundwater recharge (Akale et al., 2019), leading to increased water yields in streams, springs and wells for meeting water needs for domestic use and for the development of home gardens, which are important sources of cash for the local farmers. Overall, the implementation of SLM in the fragile tropical Blue Nile Highlands reveals great promises for improving freshwater availability and water security. Through the implementation of SLM practices as in-situ water management strategies, enough supply of freshwater could be harnessed to double agricultural productivity in subsistence rainfed systems (Rockström et al., 2007, 2010).

Water stress, which is already an important water security problem today, is likely to get worse in the future as climate change intensifies (Lebel et al., 2015). Apart from mitigating current water stresses, SLM practices will remain effective as in-situ water management strategies in the face of a changing climate, bridging future water stresses in many sub-Saharan African countries including Ethiopia (Lebel et al., 2015; Rockström et al., 2010). Therefore, mainstreaming water security into SLM initiatives in the Ethiopian Highlands would greatly improve the livelihood of millions today and in the future.

4.5. Conclusions

Implementation of SLM in Ethiopia started in the mid-1970s with the intention to control the rampant soil erosion in the highlands. However, SLM practices have since evolved from field-scale structural measures to an integrated catchment-scale approach,

increasingly gaining importance as water management strategies. This experimental study was conducted in the fragile tropical Blue Nile Highlands, Ethiopia, to understand the impact of SLM on improving freshwater availability and water security.

The results reveal that while the Blue Nile Highlands are ecologically fragile, they are also highly responsive to management interventions. Hence, significant reductions in CN (ranging from -13.9 to -21. 6 units) and increases in θ (ranging from 15.6 to 800%) were observed, promoted by SLM interventions. SLM interventions restored the natural hydrological functionality of the agricultural landscapes, leading to recovery of infiltration and soil water retention potentials. This will particularly benefit the subsistence rainfed production system in the study area by boosting soil moisture availability in the root zone, which has been highly depleted due to degraded soils and frequent dry spells.

Besides, due to increased infiltration promoted by SLM, there would be improved blue water supplies in streams, wells and springs to meet water demand for domestic uses as well as for irrigation and expansion of home gardens. In effect, home gardens, which are important sources of cash in the study area, are expanding and the construction of new hand-dug wells are on the rise in the treated catchment (personal observation). Overall, the implementation of SLM in the fragile tropical Blue Nile Highlands reveals great promises for mitigating the impacts of land degradation and adapting to climatic variability and change, thereby contributing to improved freshwater availability and hence water security. We conclude that mainstreaming water security into current and future SLM initiatives is critical in the Blue Nile Highlands and in other similar tropical highland ecosystems in the developing world for a water secure future and to build resilience against future water-related risks and uncertainties.

In the present study, runoff plots have been used to conduct event-scale water budget analysis to understand the role of SLM practices as in-situ water management strategy. Hence, the study has limitations in addressing components of the water balance that gain importance at the catchment scale, such as evapotranspiration and groundwater flow.

Such components of the water balance should receive the attention of future catchment-scale simulation and empirical studies.

5. SOIL MOISTURE SPATIO-TEMPORAL VARIABILITY UNDER TREATED AND UNTREATED CATCHMENT CONDITIONS IN A FRAGILE TROPICAL HIGHLAND ENVIRONMENT: IMPLICATION FOR WATER SECURITY[†]

Berihun D. Mersha[1, 2,*], Gete Zeleke[2], Tena Alamirew[1, 2], Zeleke A. Dejen [1], Solomon G. Gebrehiwot[1, 2]

5.1. Abstract

This paper reports soil moisture spatio-temporal variability as influenced by catchment management in the Blue Nile Highlands, Ethiopia. The study was motivated by the need to support the mainstreaming of water security into improved catchment management initiatives. The specific objectives are to: (1) determine major environmental factors influencing soil moisture spatio-temporal variability; (2) assess soil moisture spatio-temporal variability under traditional and improved catchment management conditions; and (3) understand the implication of improving catchment management for improving water security. Soil moisture and weather data were monitored intensively between 2017 and 2018 in a pair of agricultural catchments receiving traditional and improved management. A range of factors was found to contribute to soil moisture spatio-temporal variability in the study catchments, including, rainfall, air temperature, relative humidity, antecedent soil moisture content, elevation, measurement depth, groundwater table, and catchment management. Soil moisture daily variation was strongly influenced by air temperature, rainfall, relative humidity, and antecedent soil moisture content, whereas soil moisture seasonal fluctuation was controlled by rainfall seasonal pattern. Of all experimental sites, the highest elevation (2032 m) presented the driest soil moisture

[†] Chapter 5 has been published in the Arabian Journal of Geosciences: Arabian Journal of Geosciences (2022) 15(8), 1–16. https://doi.org/10.1007/s12517-022-10016-6.

1 Ethiopian Institute of Water Resources (EIWR), Addis Ababa University

2 Water and Land Resource Center (WLRC), Addis Ababa University

[*] Corresponding author. Address: Water and Land Resource Center (WLRC), Addis Ababa University, Addis Ababa, Ethiopia. Mobile: +251-47 23 05 60; P.O. Box: 3880. E-mail address: berihun2010@gmail.com.

regime (0.104 m^3 m^{-3}) while the lowest elevation (1965 m) displayed the moistest soil moisture regime (0.369 m^3 m^{-3}). The results demonstrated that improved catchment management has modified soil moisture response in the treated catchment, reducing soil moisture spatial heterogeneity and temporal variability. We conclude that improved catchment management can play a crucial role for improving agricultural water security in rainfed systems by improving soil moisture availability and storage.

5.2. Introduction

Understanding soil moisture spatio-temporal variability is critical for water resources management (Niu et al., 2015). Soil moisture (θ) forms a fundamental component of the hydrologic cycle due to its major control on many hydrologic processes, such as rainfall infiltration, surface runoff, groundwater recharge, and evapotranspiration (Falkenmark, 2013; Hébrard et al., 2006; Korres et al., 2015; Pellet & Hauck, 2017; Shen et al., 2014). Soil moisture operates the production of food and biomass in rainfed agricultural systems, which represent around 60 – 70% of current crop production worldwide (Rockström et al., 2010; Rost et al., 2009). Soil moisture spatio-temporal variability is influenced by a range of site and anthropogenic factors, among which catchment management is a major one (Cantón et al., 2004; Chen et al., 2010; Fu et al., 2003; Korres et al., 2015). Catchment management influences θ spatio-temporal variability through its impact on infiltration, soil water retention, surface runoff, and evapotranspiration (Hébrard et al., 2006; Korres et al., 2015; Niu et al., 2015; Perkins et al., 2014). Irrational management of agricultural catchments may impact on θ availability and storage, affecting local rainfall partitioning through increased runoff and decreased and uneven rainfall infiltration (Niu et al., 2015).

Agriculture in the Blue Nile Highlands is predominantly rainfed, comprising 85 – 95% of the basin's current crop production (Block & Rajagopalan, 2007). Hydrologically, the Blue Nile Highlands receive sufficient amount of rainfall to significantly increase yields without large-scale irrigation. However, rainfall in the basin is inherently uneven and highly seasonal, with over 70% of the annual precipitation occurring during a short rainy season between June and September (Conway, 2000). In addition, precipitation in the

Blue Nile Highlands often exhibits great intra-seasonal variability, directly impacting on the performance of the subsistence rainfed agriculture due to irregularities at the beginning and end of the rainy season (Berhane et al., 2014).

Rainfall seasonality and variability exert a major control on the flow regimes of streams in the Blue Nile Highlands, causing over 80% of the annual discharge to be concentrated within the period between June and October (Conway, 1997). In effect, many of the streams go empty during the long dry season, limiting the potential for irrigation development in the Blue Nile Highlands. Hence, meeting the challenge of feeding the basin's growing population through appropriate water management strategies is an issue of agricultural water security in the Blue Nile Highlands.

The rainfed agriculture in the Blue Nile Highlands is highly constrained by management-induced water scarcity stemming from irational catchment management practices. Hence, land degradation resulting from irrational catchment management has in the long term led to reduced soil depth, water-holding capacity, and soil organic carbon content, affecting local rainfall partitioning as well as water productivity, availability and storage (Bewket & Sterk, 2005; Descheemaeker et al., 2006; Hurni et al., 2005; Nyssen et al., 2009, 2010; Rientjes et al., 2011). Poor farming practices might cause rainwater losses of up to 70 – 85% non-productively through soil evaporation, surface runoff, and deep drainage, reducing the amount of infiltrated rain held in the soil as soil moisture (Rost et al., 2009). Observations in the Ethiopian Highlands demonstrated a dramatic increase in rainwater losses through surface runoff (5 – 30 times) promoted by conversion of forestland into cultivated land (Hurni et al., 2005).

On the other hand, improving the management of agricultural catchments can influence water flow pathways and thereby hydrologic continuity, resulting in improved rainfall infiltration and soil moisture retention (Hébrard et al., 2006; Mersha et al., 2022). Improved land management practices can modify soil physical and hydraulic properties that can influence soil moisture retention and redistribution (Montenegro et al., 2019; Perkins et al., 2014). Since the early 1970s, catchment management initiatives have been undertaken extensively in the Ethiopian Highlands, mainly at the scale of small

catchments, to reverse land degradation and restore ecosystem functionality in the agricultural landscapes (Descheemaeker et al., 2006, 2009; Hurni et al., 2005; Nyssen et al., 2010). Hurni et al. (2005) noted that improved catchment management reduced sediment loss in the Blue Nile Highlands. Descheemaeker et al. (2006) reported high in-situ sediment deposition in the northern Ethiopian Highlands promoted by vegetation restoration. Descheemaeker et al. (2009) indicated increased infiltration and transpiration due to improved catchment management in the northern Ethiopian Highlands. Nyssen et al. (2010) observed reductions in runoff and a rise in groundwater table in the northern Ethiopian Highlands promoted by improved catchment management. While a considerable number of studies in the past have documented different impacts of catchment management in the fragile tropical Ethiopian Highlands, little is reported about the effect of catchment management on θ spatio-temporal variability, which is critical for improving the productivity of the subsistence rainfed agriculture in the region.

The present paper reports an empirical study conducted in the Blue Nile Highlands, northwestern Ethiopia, based on a paired-catchment approach to understand soil moisture spatio-temporal variability under improved and traditional management conditions. The study was motivated by the need to provide support for mainstreaming water security into current as well as future catchment management initiatives. The specific objectives are to: (1) determine major environmental and anthropogenic factors controlling θ spatio-temporal variability in the fragile tropical Blue Nile Highlands; (2) assess soil moisture spatio-temporal variability under traditional and improved catchment management conditions; and (3) understand the implication of improving catchment management for improving water security in a fragile tropical highland ecosystem.

5.3. Materials and Methods

5.3.1. Study area description

Detailed descriptions of the study catchments are presented in Section 4.3.1. Figure 19 presents the rainfall patterns of the study area during the data period (2017 – 2018) along with minimum and maximum air temperatures. Annual rainfall received in 2017 (1495.4 mm) and 2018 (1581.6 mm) varied only slightly, with the amount received in 2018

exceeding that received in 2017 by a small margin (5.8%). However, the data indicate high rainfall variability between the 2017 and 2018 wet seasons. For instance, rainfall received during the month of June was higher in 2018 (277.4 mm) than the amount received in 2017 (57.2 mm) by a margin of 384.97%. On the other hand, rainfall received during the month of July in 2017 (695.6 mm) exceeded the amount received in 2018 (416.6 mm) by 66.97%. In addition, long-term records indicate the occurrence of high inter-annual rainfall variability. Based on records during 2015 – 2018, annual rainfall ranged from 895.8 mm (2015) to 1581.6 mm (2018), with the rainfall received in 2018 exceeding that received in 2015 by 76.56% (685.8 mm). However, unlike rainfall, temperature in the study area exhibits little temporal variability, with a mean of 20 ˚C.

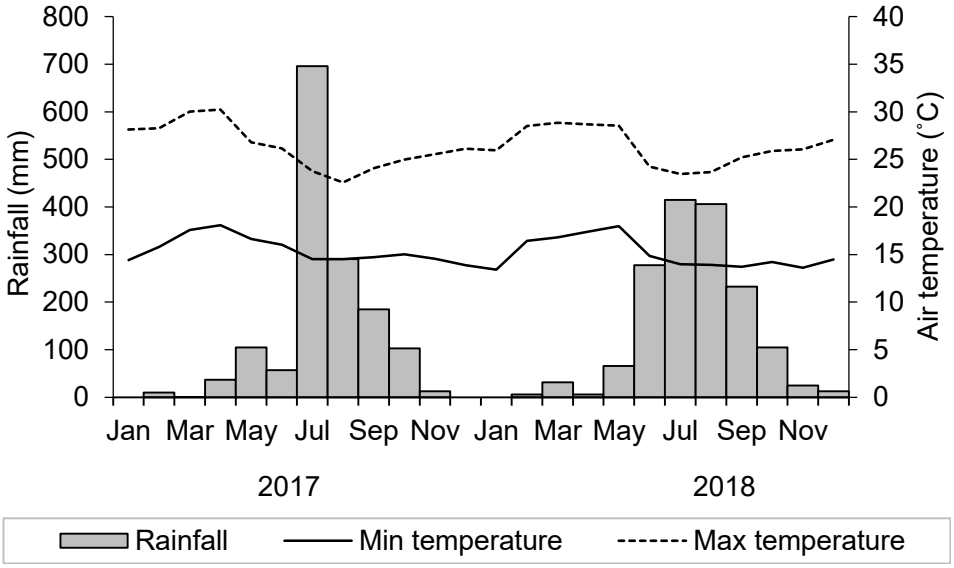


Figure 19. Rainfall pattern of the study area during the study period (2017–2018).

5.3.2. Data collection

A paired-catchment approach was adopted to collect soil moisture and weather data in two adjacently located agricultural catchments under traditional and improved management conditions. Soil moisture data were collected intensively along with weather data during two wet seasons and one dry season between 2017 and 2018. Within each study catchment, four parcels of 15 m length by 6 m width were set on varying land uses, land management, and topography, where three of them were located on cultivated land

with 5, 10, and 15% slope gradients and one on a marginally degraded grazing land with a long history of open communal grazing (17% slope gradient). Care was taken to locate all experimental sites reasonably close together in the upper parts of both study catchments in order to achieve minimum variations in site factors other than catchment management that could influence θ response, such as topography and soil characteristics. The experimental sites and their respective characteristics are summarized in Table 4.

A suite of best land management practices have been introduced in the treated catchment since 2012. Cultivated areas have received a soil bund-live hedge system along with a network of ditches for safe disposal of overland flow, which otherwise would cause erosion damage to the soils. Besides, the traditional practice of post-harvest grazing has been abandoned on cultivated land to promote the buildup of stubble mulch and eliminate livestock trampling. Open communal grazing has been entirely abandoned on grazing land to allow recovery of marginally degraded areas. In addition, the grazing lands have received contour trenches to promote fast recovery. Following the elimination of open communal grazing, a new practice has evolved in the treated catchment where the local land users feed their animals by cutting grass and biomass instead of leaving them to graze freely.

In the traditionally managed (control) catchment, any of the best land management practices implemented in the treated catchment is lacking, except some spillover effects, such as the protection of some pockets of marginally degraded grazing sites that have entirely lost their productivity. Thus, no stubble cover is maintained on the soil surface of cultivated fields in the control catchment, partly due to removal for use as fuelwood and partly due to overgrazing.

Two access tubes were installed per experimental site to monitor θ at four soil profile depths (10, 20, 30, and 40 cm) using the capacitance type PR2/4 profile probe (Dalta-T devices Ltd, UK) and a hand-held HH2 display unit. Readings of θ were taken on a daily basis at regular intervals during early morning hours (6:00 – 8:00 am). The PR2/4 probe is especially suited for monitoring θ in mineral soils, providing a high measurement accuracy (±6%) under its factory calibration (Delta-T Devices Ltd, 2016). In the present

study, prior to the deployment of the PR2/4 profile probe for monitoring θ, sensor evaluation was made against thermogravimetric θ measurements taken at multiple sites and soil profile depths within the study catchments. The evaluation presented a strong association between sensor readings and thermogravimetric measurements, with a coefficient of determination (R^2) of 0.72 and mean bias error (MBE) of 0.097 m^3 m^{-3} ($p < 0.001$).

Weather data, such as rainfall (P), air temperature (T), and relative humidity (RH), were generated at 5 min resolution at an automatic weather station close to the θ monitoring sites (Figure 1). Further, soil samples were collected at each θ monitoring depth within each experimental site to determine major soil characteristics, such as soil texture, organic carbon content (OC), dry bulk density (BD), wilting point (WP), and field capacity (FC).

5.3.3. Data analysis

In the present study, daily readings of θ (m^3 m^{-3}) taken at two measurement points and four monitoring depths within each experimental site were used to derive different values of θ as described in Equations (5.1) to (5.4). These values were then subjected to a variety of statistical data analysis techniques, such as correlation, autocorrelation, cross correlation, and regression, using the IBM SPSS® statistical package (version 24) to investigate θ response to varying site and anthropogenic factors. The influences of climate variables on soil moisture response were examined using correlation and regression techniques. Autocorrelation analysis was applied to the time series of θ to determine a time scale characteristic for θ response. Cross correlation and stepwise regression analyses were used to examine θ responses to different site and anthropogenic factors. Further, coefficients of variation (CVs) were computed for the time series of θ of individual monitoring depths within each experimental site to examine the level of soil moisture seasonal variability under treated and untreated catchment conditions.

Mean θ of a monitoring depth within an experimental site for each measurement occasion was determined as:

$$\bar{\theta}_{i,t} = \frac{1}{n}\sum_{j=1}^{n} \theta_{i,j,t} \quad (5.1)$$

where,

i = monitoring depth,

j = measurement point,

t = measurement occasion, and

n = total number of measurement points per experimental site.

Mean θ of an experimental site for each measurement occasion was calculated as:

$$\bar{\theta}_{s,t} = \frac{1}{nd}\sum_{j=1}^{n}\sum_{i=1}^{d} \theta_{i,j,t} \quad (5.2)$$

where,

i = monitoring depth,

j = measurement point,

t = measurement occasion,

n = total number of measurement points,

d = total number of measurement depths.

The temporal mean of θ for a monitoring depth within an experimental site was computed as:

$$\bar{\theta}_{i} = \frac{1}{nT}\sum_{j=1}^{n}\sum_{t=1}^{T} \theta_{i,j,t} \quad (5.3)$$

where,

i = the monitoring depth,

j = the measurement point,

t = the measurement occasion,

n = the total numbers of measurement points, and

T = the total numbers of measurement occasions.

The temporal mean of θ for an experimental site was obtained as:

$$\bar{\theta}_s = \frac{1}{ndT}\sum_{j=1}^{n}\sum_{i=1}^{d}\sum_{t=1}^{T}\theta_{i,j,t} \quad (5.4)$$

where,

i = monitoring depth,

j = measurement point,

t = measurement occasion,

n = total number of measurement points,

d = total number of measurement depths, and

T = total number of measurement occasions.

Table 4. Major topographic and soil characteristics of experimental sites.

Catchment	Land use	‡Experimental site	Slope (%)	Elevation (m)	Soil depth (cm)	Clay fraction (%)	Silt fraction (%)	Sand fraction (%)	‡OC (%)	‡BD (g cm^{-3})	‡θ_{FC} (m^3 m^{-3})	‡θ_{WP} (m^3 m^{-3})
Control	Degraded hillslope	CC1	17	2032	0-10	10	32	58	2.89	1.02	0.382	0.186
					10-20	20	20	60	2.59	1.03	0.384	0.273
					20-30	30	40	30	2.46	0.93	0.274	0.173
					30-40	46	36	18	1.76	0.89	0.334	0.207
	Cultivated land	CC2	5	1978	0-10	62	24	14	0.94	1.34	0.403	0.241
					10-20	56	20	24	1.05	1.06	0.323	0.195
					20-30	48	18	34	0.59	1.07	0.328	0.208
					30-40	42	20	38	1.52	0.91	0.324	0.207
		CC3	10	2013	0-10	50	26	24	1.68	1.02	0.300	0.183
					10-20	44	26	30	1.87	1.04	0.326	0.213
					20-30	32	26	42	0.98	1.12	0.415	0.253
					30-40	28	26	46	0.90	1.02	0.387	0.223
		CC4	15	1965	0-10	20	20	60	0.74	1.06	0.354	0.222
					10-20	18	38	44	2.69	1.18	0.341	0.203
					20-30	16	22	62	0.43	1.08	0.365	0.230
					30-40	14	22	64	0.47	1.24	0.402	0.240
Treated	Degraded hillslope	TC1	17	2025	0-10	24	40	36	2.81	1.26	0.476	0.233
					10-20	28	40	32	2.61	1.09	0.356	0.204
					20-30	34	38	28	1.83	1.13	0.376	0.233
					30-40	36	42	22	1.29	0.97	0.317	0.201
	Cultivated land	TC2	5	2002	0-10	28	38	34	2.48	1.16	0.335	0.197
					10-20	34	40	26	2.76	1.25	0.381	0.246
					20-30	44	30	26	2.36	1.26	0.416	0.266
					30-40	62	22	16	1.55	1.14	0.403	0.253
		TC3	10	2013	0-10	32	42	26	2.15	1.22	0.336	0.217
					10-20	56	30	14	1.44	1.06	0.317	0.189
					20-30	40	34	26	1.44	0.93	0.286	0.167
					30-40	38	26	36	0.74	0.99	0.325	0.210
		TC4	15	2001	0-10	26	38	36	1.83	1.18	0.317	0.198
					10-20	48	26	24	0.70	1.24	0.400	0.280
					20-30	40	26	34	0.27	1.25	0.415	0.269
					30-40	46	26	28	0.39	1.19	0.397	0.261

‡CC1 – CC4: Experimental sites 1 – 4 of the control catchment; TC1 – TC4: Experimental sites 1 – 4 of the treated catchment; OC: organic carbon content; BD: bulk density; θ_{FC}: soil moisture content at field capacity; θ_{WP}: soil moisture content at wilting point.

5.4. Results and Discussion

5.4.1. Soil moisture temporal variability

5.4.1.1. Soil moisture daily variation

At the daily time scale, θ was controlled by a range of climatic variables, including rainfall (P), air temperature (T), relative humidity (RH), and antecedent soil moisture content (θ_A), showing strong correlations ($p < 0.001$) with daily θ both in the treated and control catchments (Table 5). In the treated catchment, daily θ presented the highest correlation (r) with θ_A ($0.959 < r < 0.996$) followed by T ($-0.874 < r < -0.807$), RH ($0.734 < r < 0.798$), and P ($0.368 < r < 0.501$). The associations between daily θ and climate variables in the control catchment exhibited a similar trend with the treated catchment, exhibiting the highest r for θ_A ($0.978 < r < 0.992$) followed by T ($-0.913 < r < -0.684$), RH ($0.575 < r < 0.868$), and P ($0.371 < r < 0.447$). The ranges of r values were similar between the treated and control catchments, indicating that climate variables exerted similar controls on daily θ in both study catchments. All the climate variables except T presented positive correlations with θ. The negative correlation between θ and T indicates an inverse relationship between T and θ, where an increased T causes θ depletion due to increased evaporation and transpiration losses.

Table 5. Correlations between daily θ and climatic variables.

		Correlation coefficients (r)			
Catchment	Experimental site	θ_A	P	T	RH
Control catchment	CC1	0.990^{**}	0.371^{**}	-0.913^{**}	0.868^{**}
	CC2	0.990^{**}	0.447^{**}	-0.843^{**}	0.720^{**}
	CC3	0.992^{**}	0.440^{**}	-0.834^{**}	0.703^{**}
	CC4	0.978^{**}	0.375^{**}	-0.684^{**}	0.575^{**}
Treated catchment	TC1	0.995^{**}	0.368^{**}	-0.874^{**}	0.798^{**}
	TC2	0.991^{**}	0.463^{**}	-0.807^{**}	0.734^{**}
	TC3	0.996^{**}	0.501^{**}	-0.857^{**}	0.795^{**}
	TC4	0.959^{**}	0.473^{**}	-0.821^{**}	0.774^{**}

**Correlation is significant at the 0.01 level. θ_A, antecedent soil moisture content ($m^3\ m^{-3}$). P, rainfall (mm). T, air temperature (˚C). RH, relative humidity (-).

Strong associations between θ and climate variables were also reported by other researchers (Venkatesh et al., 2011), suggesting that multiple linear regression models could be developed to estimate θ at deeper soil layers from surface θ acquired via satellite remote sensing. Hence, multiple linear regression models were developed for different soil profile depths, yielding strong associations between climate variables and daily θ ($p < 0.001$). Table 6 presents the results of the regression analysis relating θ with climate variables.

The time series of daily θ were subjected to an autocorrelation analysis for increasing time lags in days in order to determine a time scale characteristic for θ response. Figure 20 presents the plots of temporal autocorrelations of daily θ versus time lag (days) for experimental sites in the treated and control catchments. The temporal autocorrelation revealed that the treated and control catchments appeared to attain their minimum values (troughs) at varying time lags. The experimental sites in the treated catchment presented consistently larger time lags than those in the control catchment (Table 6), suggesting a reduced θ response to daily climatic variation and, hence, a relatively more stable θ regime in the treated than in the control catchment (De Lannoy et al., 2006).

Table 6. Minimum autocorrelation coefficients and associated time lags of daily θ for experimental sites in the control and treated catchments.

Catchment	Experimental site	Minimum value of autocorrelation	Time lag (days)
Control catchment	CC1	-0.526**	176
	CC2	-0.467**	162
	CC3	-0.542**	175
	CC4	-0.520**	166
Treated catchment	TC1	-0.498**	186
	TC2	-0.490**	186
	TC3	-0.532**	177
	TC4	-0.475**	176

**Autocorrelation is significant at the 0.01 level. CC1 – CC4: experimental sites in the control catchment; TC1 – TC4: experimental sites in the treated catchment.

Table 7. Multiple linear regression describing the relationships between soil moisture and climate variables.

Catchment	Experimental site	Dependent variable	Intercept	Regression coefficients of descriptor variables					R^2
				θ_A	θ_U	P	T	RH	
Control catchment	CC1	θ_1	0.046	0.901	-	$-3.46*10^{-5}$	-0.002	0.017	0.990
		θ_2	-0.015	0.544	0.236	1.49*10-5	0.001	0.012	0.967
		θ_3	-0.006	0.529	0.296	-1.08*10-5	0.00	0.011	0.905
		θ_4	0.018	0.291	0.392	$-3.58*10^{-5}$	-0.001	-0.013	0.901
	CC2	θ_1	0.051	0.882	-	$-5.34*10^{-5}$	-0.002	0.014	0.981
		θ_2	0.013	0.789	0.169	$5.22*10^{-5}$	0.00	-0.009	0.982
		θ_3	0.089	0.611	0.192	$-7.03*10^{-5}$	0.00	0.009	0.912
		θ_4	0.043	0.868	0.134	0.00	-0.001	-0.006	0.918
	CC3	θ_1	0.016	0.955	-	$-3.74*10^{-5}$	-0.001	0.010	0.989
		θ_2	0.017	0.885	0.054	$-2.98*10^{-5}$	-0.001	0.013	0.991
		θ_3	0.026	0.681	0.328	0.00	0.001	0.002	0.968
		θ_4	0.081	0.436	0.392	$-4.33*10^{-5}$	0.001	-0.007	0.810
	CC4	θ_1	0.077	0.854	-	$8.85*10^{-5}$	-0.003	0.003	0.969
		θ_2	0.024	0.785	0.095	$5.05*10^{-5}$	0.001	0.009	0.880
		θ_3	0.028	0.735	0.283	$-3.52*10^{-5}$	0.00	0.003	0.946
		θ_4	-0.023	0.809	0.386	$-4.87*10^{-6}$	$-3.97*10^{-5}$	-0.016	0.968
Treated catchment	TC1	θ_1	0.046	0.908	-	$-2.02*10^{-5}$	-0.002	0.007	0.975
		θ_2	-0.04	0.671	0.346	$-4.50*10^{-5}$	0.002	0.041	0.986
		θ_3	0.049	0.837	0.105	$3.62*10^{-5}$	-0.001	-0.007	0.973
		θ_4	-0.054	0.262	0.651	0.00	0.003	0.012	0.936
	TC2	θ_1	0.059	0.858	-	$-8.45*10^{-6}$	-0.002	0.001	0.962
		θ_2	0.026	0.622	0.414	$-2.61*10^{-5}$	0.00	0.001	0.976
		θ_3	0.029	0.601	0.437	$4.70*10^{-6}$	0.001	-0.004	0.960
		θ_4	-0.045	0.756	0.329	$-2.18*10^{-5}$	0.001	-0.004	0.965
	TC3	θ_1	0.054	0.912	-	$6.03*10^{-5}$	-0.002	0.012	0.989
		θ_2	-0.005	0.703	0.254	$5.34*10^{-5}$	0.001	0.005	0.992
		θ_3	-0.04	0.682	0.400	$5.66*10^{-5}$	0.002	0.008	0.992
		θ_4	0.03	0.681	0.360	0.00	0.00	-0.014	0.976
	TC4	θ_1	0.024	0.858	-	$8.38*10^{-5}$	-0.001	0.016	0.964
		θ_2	-0.022	0.721	0.135	$1.63*10^{-5}$	0.002	0.012	0.811
		θ_3	0.125	0.287	1.066	$-6.29*10^{-5}$	-0.002	0.012	0.884
		θ_4	-0.049	0.619	0.491	$6.18*10^{-5}$	0.002	-0.029	0.796

θ_A: antecedent soil moisture content; θ_U: soil moisture content of immediate upper layer; P: rainfall; T: air temperature; RH: relative humidity. CC1 – CC4: Experimental sites 1 – 4 in the control catchment; TC1 – TC4: Experimental sites 1 – 4 in the treated catchment. θ_1 – θ_4: soil moisture content measurements at 10, 20, 30, and 40 cm soil profile depths.

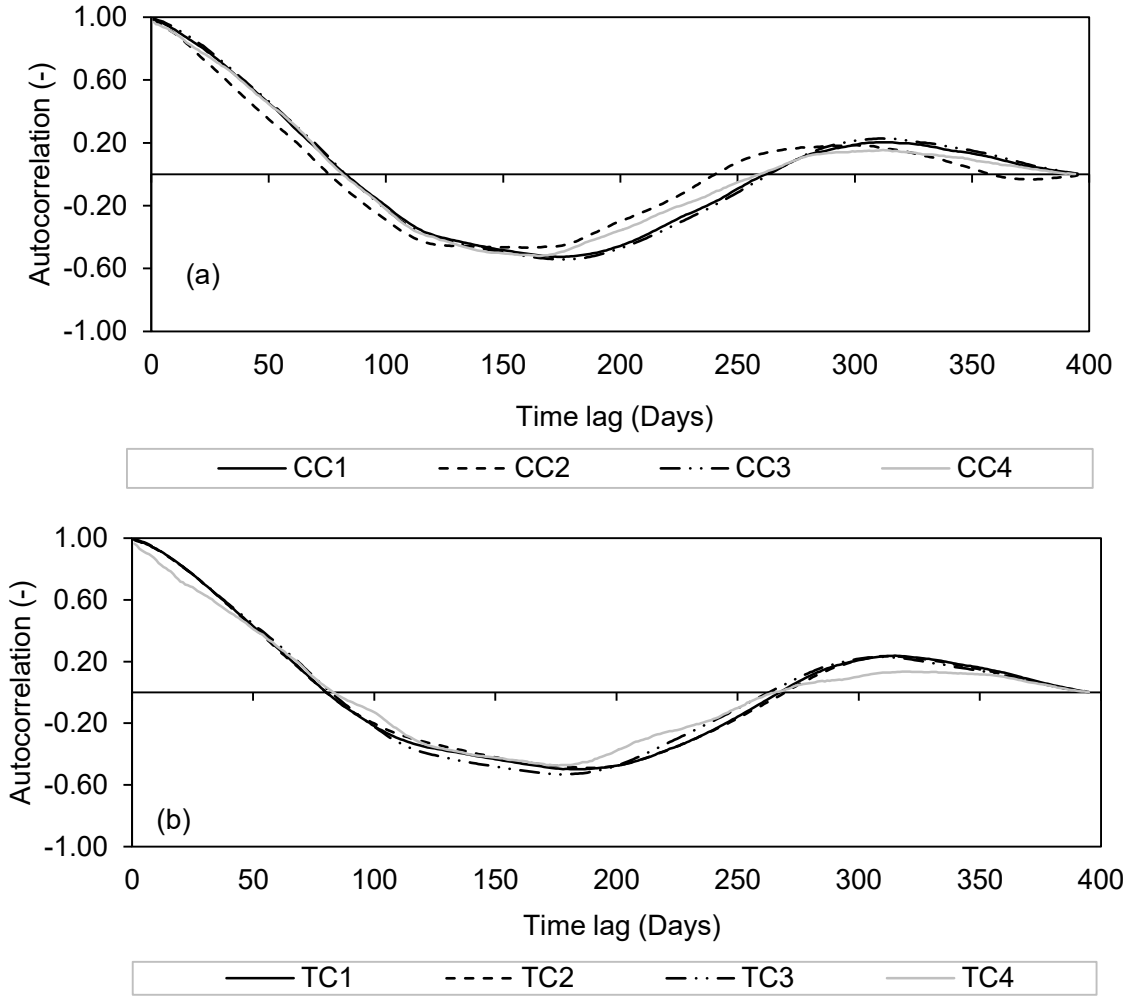


Figure 20. Temporal autocorrelation of soil moisture time series for individual experimental sites: (a) control catchment; and (b) treated catchment.

5.4.1.2. Soil moisture seasonal patterns

Figure 21 presents a plot of the time series of daily θ across all experimental sites in the treated and control catchments along with daily rainfall. The θ regimes in both study catchments demonstrate that the seasonal pattern of rainfall had a direct control on the seasonal fluctuation of θ. The two peaks observed in the θ regimes of all the experimental sites are associated with the two wet seasons that occurred during the data period while the single trough corresponds to the single dry season of the data period, generally resulting in a wet-dry-wet θ sequence. Soil moisture exhibited dramatic depletion upon the start of the dry sequence and then remained relatively stable until a couple of rainfall

events received on 30 and 31 March 2018 resulted in a small peak within the dry sequence. The rainfall received on 30 March 2018 (9.4 mm) barely caused any θ response; however, a larger rainfall event (21.6 mm) received on 31 March triggered θ response across all experimental sites. However, the experimental sites presented different time lags for attaining the small peaks of θ triggered by those rainfall events. Hence, soil moisture reached its peak 4 days later (4 April) at CC2 and CC4 and 5 days later (5 April) at CC1 and CC3 of the control catchment. By contrast, θ reached its peak faster in the treated catchment, for instance one day later at TC4 (1 April) and three days later at TC2 (3 April). The reduced lags observed in the treated catchment clearly indicate increased soil water recharge promoted by improved catchment management.

Soil moisture content increased dramatically across all experimental sites upon the start of the wet season. However, a lag effect in θ response was apparent, with the peaks of rainfall occurring in July – August while the peaks of θ occurred in September – October. Such a lag effect in θ response could be explained by the fact that the July – August period represents the early stage of the growing season in the study area during which cultural activities are at their peaks, resulting in high soil surface disturbance and poor ground cover. Hence, overland flow and soil evaporation are high during the July – August period, limiting the amount of rainwater infiltration and soil water storage.

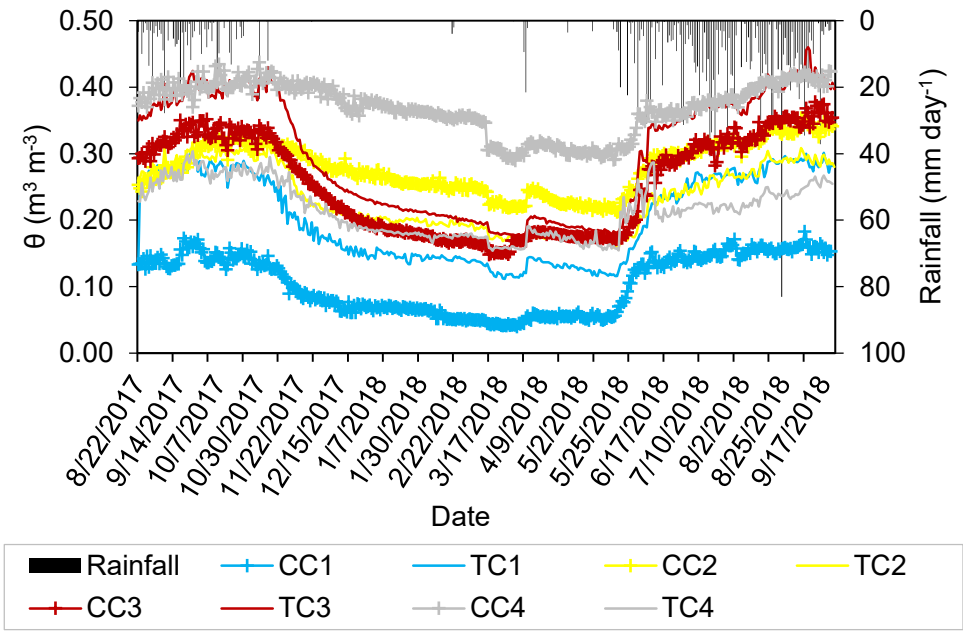


Figure 21. Soil moisture temporal patterns for experimental sites in the treated and control catchments.

The driest θ regime was observes at CC1 of the control catchment (a marginally degraded grazing land), exhibiting the lowest values of daily θ (mean θ = 0.104 m^3 m^{-3}) throughout the data period. The lowest soil water retention capacity observed at CC1 is associated with the level of soil degradation experienced at the site due to a long history of open communal grazing. The soils at the site have lost their water intake and storage capacities due to prolonged surface disturbances, including trampling (surface compaction) and overgrazing by livestock. In addition, CC1 lies on the highest elevation (2032 m above mean sea level) than any of the experimental sites across the study catchments (Table 4). A stepwise regression analysis applied to relate elevation to θ revealed a strong association between elevation and temporal mean of θ, with the influence of elevation on θ gaining importance at deeper soil layers, such as 30 cm depth (R^2 = 0.54; p = 0.038) and 40 cm depth (R^2 = 0.75; p = 0.006) (Figure 22). This suggests that elevation has partly contributed to reduced θ at CC1.

The moistest θ regimes were observed at CC4 (mean θ = 0.369 m^3 m^{-3}) and TC3 (mean θ = 0.300 m^3 m^{-3}). The mean of θ during the 2017 wet sequence was 0.401 m^3 m^{-3} at CC4

and 0.391 m^3 m^{-3} at TC3. During the 2018 wet sequence, θ decreased slightly to a mean of 0.385 m^3 m^{-3} (-3.99% reduction) at CC4 and 0.375 m^3 m^{-3} (-4.09%) at TC3. The higher mean of θ experienced during the 2017 wet sequence relative to the 2018 wet sequence is attributable to the observed intra-seasonal rainfall variability. Total wet season rainfall in 2018 (1330.4 mm) was slightly higher than that received in 2017 (1227.8 mm) by 8.36%. However, rainfall received during the month of July in 2017 (695.6 mm) was higher than that received in 2018 (416.6 mm) by a much greater margin (66.97%), explaining the observed increases in θ in the 2017 wet sequence.

These two moistest sites presented roughly similar levels of θ during the two wet sequences of the data period (Figure 20). However, these sites demonstrated clear deviations in their θ patterns during the dry sequence, where TC3 experienced a higher θ depletion (0.347 mm day^{-1}; mean θ = 0.229 m^3 m^{-3}) than CC4 (0.170 mm day^{-1}; mean θ = 0.349 m^3 m^{-3}). Besides, the time series of θ presented a higher coefficient of variation (CV) at TC3 (CV = 0.30) than at CC4 (CV = 0.11), indicating higher θ temporal variability at TC3 than at CC4. These two sites belonged to the same land use (cultivated land) but were different in topographic positions and land management conditions, where CC4 was located at 1965 m elevation in the control catchment on a 15% slope gradient while TC3 was situated at 2013 m elevation in the treated catchment on a 10% slope gradient. It appears that CC4 is the lowest lying site than any of the experimental sites across the study catchments. Hence, the moistest θ regime displayed at CC4 despite a steeper terrain (15% slope gradient) and untreated site condition (control site) is attributable to low elevation, obviously subjecting the site to the influence of the groundwater table.

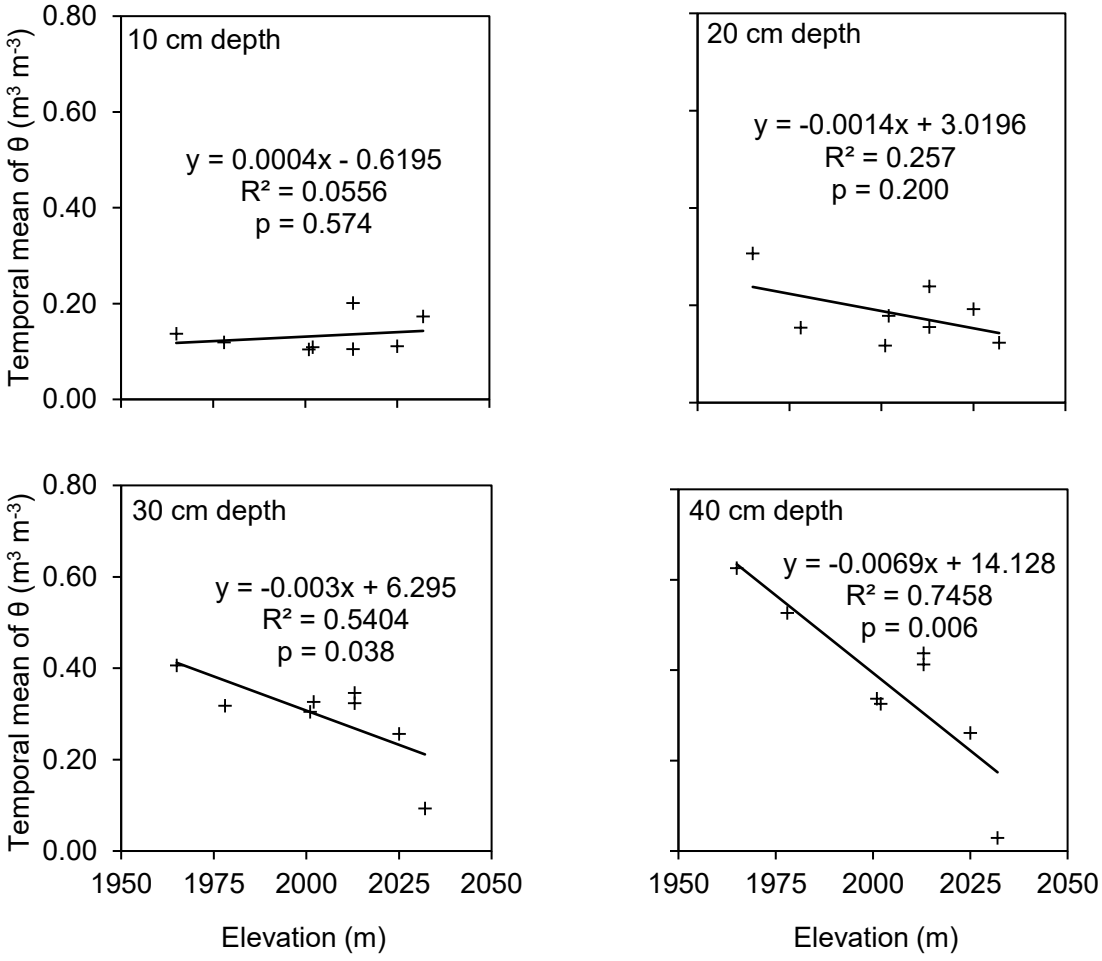


Figure 22. Relationship between temporal mean of θ and elevation at different soil profile depths.

Figure 23 compares the θ regimes of these two moistest sites. Clearly, the θ regime of CC4 at 40 cm depth consistently exceeded FC throughout the data period despite high rainfall seasonal fluctuation experienced during the data period. This suggests the occurrence of a continuous flow of water from a shallow groundwater table to the soil profile via capillary rise to keep the soil consistently wet at this particular site. The influence of groundwater table on θ of CC4 was also reflected by the observed lag in θ response to rainfall seasonal fluctuation. During the data period, the greatest θ depletion rate at TC3 and the majority of the experimental sites occurred upon the start of the dry sequence (in November 2017), leading to a dramatic loss of θ. However, the highest θ

depletion rate at CC4 took place four months later (in March 2018), clearly indicating that rainfall seasonal fluctuation has little control on the seasonal variation of θ at CC4. Hence, the delayed occurrence of the highest θ depletion in March at CC4 should likely be associated with the fall of the groundwater table at this particular site.

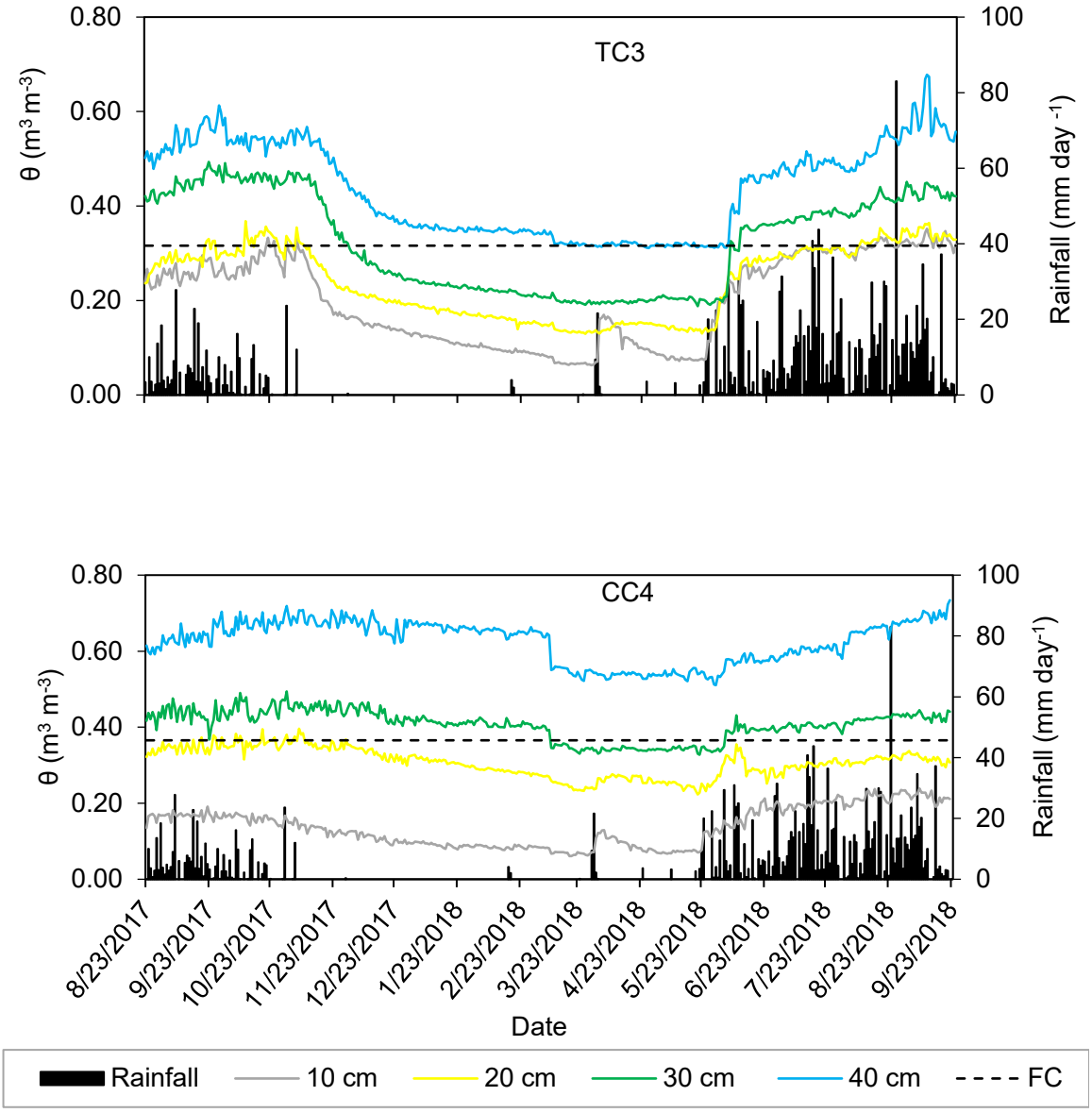


Figure 23. Soil moisture regimes of TC3 and CC4 at 10, 20, 30, and 40 cm soil profile depths versus soil moisture content at field capacity (FC).

Further, a stepwise regression analysis presented a characteristic inverse relationship between the temporal mean and CV of θ, with negative regression slopes (Figure 24). Apparently, reduced regression slope and R^2 values were found for the treated than the control catchment, indicating a reduced θ response to rainfall seasonal fluctuation. This implies that θ is temporally more stable in the treated than in the control catchment, due to improved rainfall infiltration and soil moisture retention and redistribution promoted by improved catchment management.

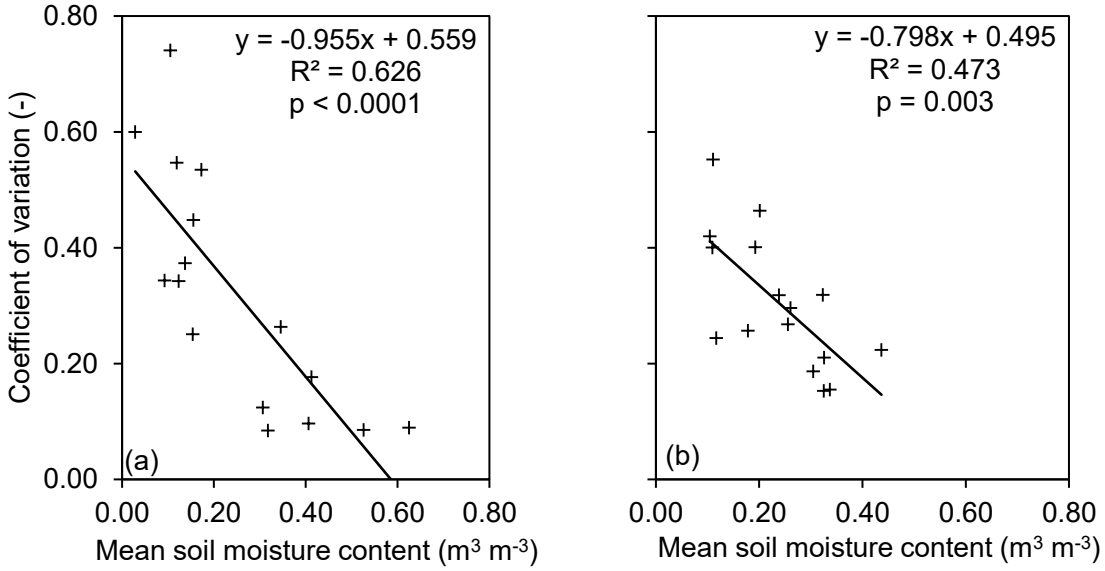


Figure 24. Relationships between temporal mean of θ and CV: (a) control catchment and (b) treated catchment.

5.4.2. Soil moisture spatial variation

A cross correlation analysis was applied to examine the time lag relationships between any two monitoring depths within an experimental site (Table 8). The values in parentheses indicate the time lags that yielded the highest correlation coefficients. Time lags in days ranged from 0 to 62 for the control catchment and 0 to 18 for the treated catchment. Adjacent monitoring depths presented the highest correlation coefficients at zero time lag, indicating faster soil water exchange between these layers. On the other hand, the correlation coefficients associated with nonadjacent θ measurement depths

within an experimental site often yielded time lags different from zero, indicating a slower soil water exchange between those layers. In general, the highest time lags are associated with the control catchment, obviously suggesting a relatively uneven θ profile distribution. By contrast, the reduced time lags observed in the treated catchment indicate a relatively faster soil water exchange and, hence, more uniform θ profile distribution, explained by the positive impact of improved catchment management (increased infiltration and soil moisture redistribution).

Further, the results of a stepwise regression applied to understand the relationship between θ and soil depth is presented in Figure 25. The data revealed an increasing θ with increasing soil depth across all experimental sites except CC1, where θ decreased with increasing soil depth. The data presented consistently lower regression slopes and R^2 values for experimental sites in the treated catchment than in the control. This implies a relatively more uniform θ distribution (reduced θ profile variability) within the treated catchment, due to increased rainwater infiltration and soil moisture redistribution promoted by improved catchment management. An exceptionally decreasing trend of θ with increasing soil depth at CC1 should be associated with site characteristics, given this site is located at the highest elevation in the control catchment on a marginally degraded grazing land.

Table 8. Relationships between soil moisture monitored at different depths within the soil profile of each experimental site.

Catchment	Experimental site	Cross correlation coefficients					
		10 cm vs 20 cm	10 cm vs 30 cm	10 cm vs 40 cm	20 cm vs 30 cm	20 cm vs 40 cm	30 cm vs 40 cm
Control	CC1	0.957(0)	0.800(4)	0.753(2)	0.869(0)	0.822(0)	0.859(0)
catchment	CC2	0.915(0)	0.682(12)	0.555(22)	0.746(0)	0.512(0)	0.650(0)
	CC3	0.937(0)	0.727(5)	0.673(2)	0.869(0)	0.799(0)	0.938(0)
	CC4	0.529(17)	0.574(33)	0.613(62)	0.887(0)	0.799(12)	0.879(0)
Treated	TC1	0.969(0)	0.816(8)	0.629(7)	0.879(8)	0.714(6)	0.920(0)
catchment	TC2	0.971(0)	0.800(9)	0.866(18)	0.804(7)	0.883(7)	0.892(0)
	TC3	0.971(0)	0.911(3)	0.913(6)	0.952(0)	0.949(3)	0.975(0)
	TC4	0.738(0)	0.682(0)	0.385(0)	0.830(0)	0.735(0)	0.864(0)

Values in parentheses are time lags in days.

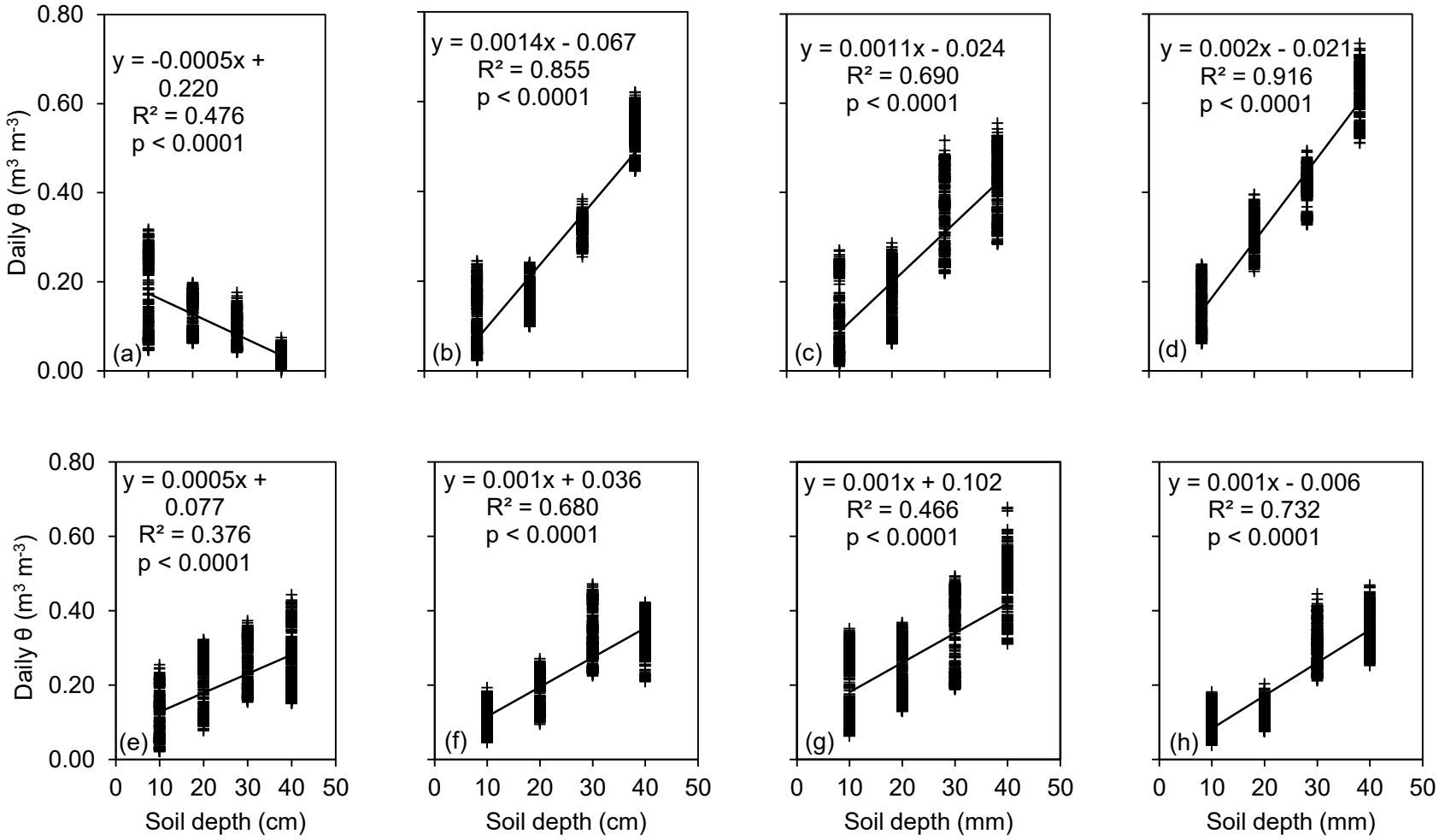


Figure 25. Relationships between θ and soil depth under treated and untreated catchment conditions: (a) CC1; (b) CC2; (c) CC3; (d) CC4; (e) TC1; (f) TC2; (g) TC3; and (h) TC4.

The degrees of association between the time series of θ across experimental sites within each study catchment were examined using a cross correlation analysis with a zero time lag. The analysis yielded quite strong associations between θ of experimental sites within both study catchments (Table 9). However, the correlation coefficients tended to be consistently higher within the treated catchment, indicating a spatially more uniform θ response across the experimental sites within the treated catchment promoted by improved catchment management. On the other hand, correlation coefficients within the control catchment were relatively lower, indicating a spatially more varied θ response. In addition, θ measurements within the control catchment yielded higher CV (0.44) than those within the treated catchment (0.18), indicating a spatially more heterogeneous θ distribution in the control catchment. This implies that θ response in the control catchment is controlled by spatially varied hydrologic processes, such as surface runoff and rainfall infiltration (De Lannoy et al., 2006; Hébrard et al., 2006; Niu et al., 2015).

Table 9. Cross correlations with zero time lag relating soil moisture time series of experimental sites within each study catchment.

	Control catchment					Treated catchment			
	CC1	CC2	CC3	CC4		TC1	TC2	TC3	TC4
CC1	1				TC1	1			
CC2	0.863**	1			TC2	0.957**	1		
CC3	0.958**	0.903**	1		TC3	0.982**	0.971**	1	
CC4	0.727**	0.872**	0.805**	1	TC4	0.870**	0.906**	0.900**	1

**Correlation is significant at the 0.01 level.

5.4.3. Implication for water security

Agriculture forms the basis of Ethiopia's economy, supporting about 47% of the GDP, 90% of the exports and 85% of the employment (Asfaw et al., 2018; Conway & Schipper, 2011; Dile et al., 2013; Simane et al., 2016; Verhoeven, 2013). Infiltrated rain held in the soil operates agricultural production in Ethiopia. However, extreme hydrologic variability and chronic land degradation have jeopardized the productivity of the subsistence rainfed agriculture in Ethiopia due to their deleterious impact on soil moisture availability and storage (Asfaw et al., 2018; Berhane et al., 2014).

Inherently, rainfall is highly erratic in Ethiopia, causing meteorological droughts to strike once or twice every decade and dry spells almost every rainy season (Rockström et al., 2004; Verhoeven, 2013). Historically, there were instances where meteorological drought events lead to famine and loss of life in Ethiopia, such as in 1973, 1974, and 1984 (Conway & Schipper, 2011; Tabari et al., 2015). So sensitivity is Ethiopia's agriculture to rainfall variability that the economic cost is disproportionately high, at over one-third of the country's average annual growth potential (Asfaw et al., 2018; Conway & Schipper, 2011; Dile et al., 2013; Tabari et al., 2015; Verhoeven, 2013).

Owing to favorable temperature, human settlement and agriculture have a long history in Ethiopia (Gumma et al., 2021), particularly in the Highlands, where over 90% of the country's population live. The Ethiopian Highlands receive sufficient rainfall to significantly increase crop yields without large-scale irrigation. However, the productivity of the rainfed agriculture in the Ethiopian Highlands is constrained by management-induced water scarcity. Chronic land degradation has reduced the depth, water holding capacity, and organic carbon content of the soils in the Ethiopian Highlands, affecting local rainfall partitioning and, hence, soil moisture availability and storage (Betrie et al., 2011; Bossio et al., 2010; Kirui et al., 2021; Zaitchik et al., 2012). Poor farming practices are responsible for nonproductive rainwater losses of up to 70 – 80% through soil evaporation, surface runoff, and deep drainage (Rost et al., 2009). Observations in the Ethiopian Highlands reveal dramatic increases in rainwater losses (5 – 30 times) through surface runoff triggered by the clearing of forests for agricultural expansion (Hurni et al., 2005).

Since the early 1990s, soil moisture has gained increased attention as a fundamental component of the hydrologic cycle (Falkenmark, 2013). In the Ethiopian Highlands where the dominant form of freshwater is the rainwater held in the soil (soil moisture), upgrading rainfed agriculture is an issue of agricultural water security. The suite of catchment management options investigated in the present study reveal great promises for bridging soil water stress and sustaining the subsistence rainfed agriculture in the Blue Nile Highlands, which accounts for 85 – 95% of the current annual food production of the basin (Block & Rajagopalan, 2007).

Through improved catchment management, a temporally more stable and a spatially more uniform soil moisture pattern has been achieved in the Blue Nile Highlands, implying increased soil moisture availability and storage. Improved catchment management options in the Blue Nile Highlands have helped improve green water storage in the rooting zone by up to 16 – 800 % compared to traditional practices (Mersha et al., 2022). Improving the management of agricultural catchments will greatly benefit the subsistence rainfed agriculture in the study area today and in the future, by securing water not only for current agricultural production but also for building resilience for coping with future water security risks and uncertainties (Lebel et al., 2015). Apart from benefiting the local population in the Blue Nile Highlands, improved catchment management barely affects overall catchment runoff to downstream areas, creating a large opportunity for improving yield without affecting downstream water users and ecosystems (Hurni et al., 2005; Rockström et al., 2010).

5.5. Conclusions

In the present study, an empirical paired-catchment approach was adopted to understand spatio-temporal soil moisture (θ) variability under a treated versus traditionally managed agricultural catchment in the fragile Blue Nile Highlands. The study reveals that catchment management, topography, and climate jointly influence θ spatio-temporal variation in the study catchments. The temporal variation of θ was primarily influenced by climate, topography, and catchment management. The daily variation of θ was controlled by antecedent soil moisture content (θ_A), rainfall (P), air temperature (T), and relative humidity (RH), with θ_A exerting the major control followed by T, RH, and P. On the other hand, the seasonal fluctuation of θ was controlled by the rainfall seasonal pattern, with θ regimes peaking during the wet season and falling during the dry season. The driest θ regime (mean θ = 0.104 m^3 m^{-3}) was determined for the highest measurement elevation (2032 m above mean sea level) while the moistest (mean θ = 0.369 m^3 m^{-3}) was obtained for the lowest elevation (1965 m) of all experimental sites under investigation. The present investigation revealed that the two study catchments differ markedly in θ response, with reduced θ daily and seasonal variations and increased temporal stability observed in the treated catchment.

Soil moisture spatial variability in the study catchments was influenced by depth of θ measurement, topography, and catchment management. Soil moisture generally presented an increasing trend with increasing soil depth across both study catchments. The treated catchment presented reduced soil moisture profile variability, due to improved rainwater infiltration and θ redistribution. Though elevation masked the effect of catchment management at some particular experimental sites, causing a control experimental site to display the moistest θ regime (lowest elevation) and a treated site the driest θ regime (highest elevation), the treated catchment generally displayed reduced θ spatial heterogeneity (CV = 0.18) than the control (CV = 0.44), indicating relatively more uniform θ response in the treated catchment than in the control.

In conclusion, improved catchment management options revealed great promises for improving agricultural water security in the Blue Nile Highlands by reducing soil moisture spatio-temporal variability and, hence, improving soil moisture availability and storage. The evidences generated in the present study are critical for supporting current and future efforts to mainstream water security into improved catchment management interventions in the Blue Nile Highlands. Given the diverse topography and associated climate contrasts prevailing in the Blue Nile basin, we recommend similar studies at a larger spatial scale in the future.

6. INITIAL ABSTRACTION RATIO AND RUNOFF CURVE NUMBER COMPUTATIONS FROM DATA FOR DIFFERENT LAND USES AND MANAGEMENT CONDITIONS IN THE BLUE NILE HIGHLANDS, ETHIOPIA[†]

Berihun D. Mersha[1, 2,*], Gete Zeleke[2], Tena Alamirew[1, 2], Zeleke A. Dejen [1], Solomon G. Gebrehiwot[1, 2]

6.1. Abstract

The curve number (CN) method is widely used around the world, independently or as part of hydrologic simulation models, for estimation of direct runoff from storm events. The adequacy of the CN method depends on the accuracy of its initial abstraction ratio (λ) and CN parameters, which exhibit considerable variations across different regions worldwide. However, values of λ and CN have not been adequately generated for Ethiopia based on local data. In the present analysis, we applied an iterative least squares procedure to determine λ and CN for different land uses and management conditions using rainfall-runoff (P-Q) data recorded from 26 runoff plots across 3 agricultural headwater catchments in the Blue Nile Basin, Ethiopia. The data revealed that neither antecedent runoff condition (ARC) nor hydrologic soil group (HSG) influenced variations in λ and CN in the study catchments. Variabilities of λ and CN across the study catchments were greatly influenced by land management practices, with sites under better management presenting increased λ and reduced CN. While recorded P-Q data resulted in computed λ much lower than the standard value (λ = 0.2), implying quick runoff response, CN estimates appeared to be predominantly higher than values published by the Natural Resources Conservation Service (NRCS).

[†] Parts of Chapter 6 have been presented on the 2nd African Graduate Students Conference, AGSC II, 27 – 28 May 2022, and on the 1st Abbay International Water conference, AIWC, 16 – 17 September 2022, held at Bahir Dar University, Bahir Dar, Ethiopia.

1 Ethiopian Institute of Water Resources (EIWR), Addis Ababa University

2 Water and Land Resource Center (WLRC), Addis Ababa University

[*] Corresponding author. Address: Water and Land Resource Center (WLRC), Addis Ababa University, Addis Ababa, Ethiopia. Mobile: +251-47 23 05 60; P.O. Box: 3880. E-mail address: berihun2010@gmail.com.

We conclude that λ and CN computed from data are critical for improving output uncertainties associated with CN-based hydrologic simulation in the study area.

6.2. Introduction

Hydrologic modeling is an integral part of water resources planning and management, playing a crucial role for estimating key hydrologic variables, among which direct runoff is a major one. Direct runoff is routinely needed in engineering design, estimating soil erosion and sediment yield, and analyzing long-term changes in water resources availability (Hawkins, 1993; Krajewski et al., 2020). The increased availability of computers has led to the development and use of complex models for simulating water balance and predicting direct runoff (Descheemaeker et al., 2008; Ponce & Hawkins, 1996). However, the wider applicability of such models is limited by intensive input data and calibration requirements (Abon et al., 2011; Shi et al., 2009; Soulis & Valiantzas, 2012). Particularly, in poorly gauged regions, conceptual models with few input data requirements and clearly stated assumptions can provide a good alternative, sharing the simplicity of empirical models and the wider applicability of physically-based simulation models (Ponce & Hawkins, 1996; Shi et al., 2009; Soulis & Valiantzas, 2012).

The curve number (CN) method is a conceptual model that is widely used around the world, independently or as part of hydrologic simulation models, for the estimation of direct runoff from storm events (Caletka et al., 2020; Krajewski et al., 2020). The popularity of the CN method stems from its simplicity, versatility, low input data requirement, and applicability in ungauged catchments (Hawkins et al., 2009; Hawkins, 1993; Jain et al., 2006; Michel et al., 2005; Oliveira et al., 2016; Ponce & Hawkins, 1996). The CN method was developed in the 1950s by the Natural Resources Conservation Service (NRCS) of the United States Department of Agriculture (USDA), formerly known as the Soil Conservation Service (SCS), to design flood control projects on agricultural catchments (Dile et al., 2016; Ponce & Hawkins, 1996; Soulis & Valiantzas, 2012; Tedela et al., 2012). Originally, the method was developed based on annual maximum rainfall-runoff (P-Q) data collected during 1928 – 1954 from 199 catchments across the US with mainly a single soil group and cover complex, which ranged in size from 0.0971 to 18600 ha (Ponce & Hawkins, 1996; Stewart et al., 2012; Tedela et al., 2012; Woodward et al., 2003). In

subsequent applications, the CN method involved the use of P-Q data recorded at the plot scale (Descheemaeker et al., 2008; Elhakeem & Papanicolaou, 2009; Feyereisen et al., 2008; Huang et al., 2007; Oliveira et al., 2016).

For over six decades since its inception, the CN method has been widely adopted in various regions, land uses, and climates (Huang et al., 2007; Soulis & Valiantzas, 2012; Tedela et al., 2012). Besides, the method has evolved from an event-based infiltration loss model that does not account for long-term processes (e.g., evapotranspiration) to being an integral part of several complex hydrologic and water quality computer models (e.g., EPIC, AGNPS, and SWAT) for simulating continuous processes (Arnold et al., 1998; Sharpley & Williams J R, 1990; Young et al., 1994).

The adequacy of the CN method depends on the accuracy of its initial abstraction ratio (λ) and CN parameters, which exhibit considerable variations across different regions worldwide. Though the CN method was developed based on a tremendous amount of empirical data, the data were collected entirely within the US (Ponce & Hawkins, 1996; Tedela et al., 2012). As a result, the validity and applicability of the method in different geographic and climatic regions out of the US is not fully understood (Dile et al., 2016; Elhakeem & Papanicolaou, 2009; Jain et al., 2006; Ponce & Hawkins, 1996; Shi et al., 2009; Zelelew, 2017). For instance, though λ takes a fixed value of 0.2 in the original CN method, it has often been cited to vary across different geographical locations worldwide, suggesting the method's responsiveness to variations in geologic and climatic settings (Baltas et al., 2007; Descheemaeker et al., 2008; Hawkins, 1993; Mishra et al., 2006; Ponce & Hawkins, 1996). Given the fact that the CN method was developed for a temperate environment with low rainfall intensity and seasonality and porous soils, it is expected to perform differently in a tropical highland ecosystem, where rainfall is highly seasonal and the soils are degraded with low infiltration and water retention capacities (Descheemaeker et al., 2008; Dile et al., 2016).

Contributing about 60% of the annual Nile's flow, the Blue Nile Basin has gained the focus of increased research attention to address its water resources problems arising from environmental degradation and climate variability and change (Ahmed et al., 2019; Betrie et al., 2011; Conway, 1997, 2000; Haregeweyn et al., 2017; Hurni et al., 2005). In the past, several CN-based hydrologic simulation studies have been

conducted in the Blue Nile Basin to support ecosystem restoration and water resources planning and management efforts (Betrie et al., 2011; Dile et al., 2016; Setegn et al., 2010, 2011; Takele et al., 2021). Setegn et al. (2010) evaluated the applicability of SWAT for sediment yield simulation in a gauged headwater catchment in the Blue Nile Basin. Setegn et al. (2011) used SWAT to investigate water resources responses to climate change in the Lake Tana sub-basin within the Blue Nile Basin. Betrie et al. (2011) applied SWAT to simulate sediment yield under different management scenarios for the entire Blue Nile Basin. Dile et al. (2016) investigated the capability of SWAT for estimating direct runoff in three agricultural catchments in the Blue Nile Highlands. Takele et al. (2021) employed SWAT to simulate the water balance of the entire Blue Nile basin. However, these studies were predominantly conducted based on model calibration involving the manipulation of model input parameters, often resulting in unrealistic input parameter values and unjustified hydrological mass balance results (Berhanu et al., 2013; Dile et al., 2016; Van Griensven et al., 2012). Particularly, such CN-based hydrologic simulation studies conducted in the Blue Nile Basin are cited to have resulted in underestimation of direct runoff (Descheemaeker et al., 2008; Dile et al., 2016; Zelelew, 2017), which suggests that CN parameter values published by NRCS are inadequate to represent local conditions (Feyereisen et al., 2008; Van Griensven et al., 2012).

Estimation of direct runoff by the CN method is much more sensitive to errors in CN parameter values than similar levels of error in storm rainfall (Hawkins, 1993). Hence, model output uncertainties resulting from CN-based simulation studies are often associated with unrealistic CN estimates (Arnold et al., 1998; Betrie et al., 2011; Feyereisen et al., 2008; Setegn et al., 2010, 2011). While CN estimates from recorded P-Q data are cited to considerably reduce model output uncertainty and provide policy-relevant simulation results (Descheemaeker et al., 2008; Feyereisen et al., 2008; Hawkins, 1993; Zelelew, 2017), such studies to determine local CN from recorded data are rare in Ethiopia. The overall objective of this paper is to apply computed λ and CN parameters for assessing hydrologic responses under different land uses and management practices. The specific objectives are: (1) to estimate λ and CN from recorded P-Q data for different land uses and management practices in the Blue Nile Highlands; (2) to determine the applicability of the CN method in the Blue

Nile Highlands; and (3) to evaluate computed CNs against NRCS-generated CNs published in standard tables.

6.3. Methods

6.3.1. Description of the study catchments

The present analysis was based on the wealth of P-Q data collected during an extended period between 1984 and 2018 across three agricultural headwater catchments in the Blue Nile Basin, including two adjacently located catchments in Aba Gerima area and a third catchment in Anjeni (Figure 1). A description of major site characteristics has been summarized in Table 10. Detailed descriptions of the study catchments can also be found in Section 2. The study catchments are predominantly agricultural, with crop cultivation being carried out without fallows due to shortage of land. The crops grown in the study catchments require fine seedbed preparation and provide inadequate ground cover during the period of erosive rainstorms (July – August), greatly contributing to increased runoff and soil loss rates. As a result, the soils across the study catchments are highly degraded, with low pH ($pH < 6.4$) and organic carbon content ($OC < 4\%$).

The study catchments are characterized by a sub-humid climate with a highly seasonal (unimodal) rainfall pattern, where most rainstorms occur between May and October. Mean monthly rainfall exceeds 100 mm during May – October (Figure 26), contributing 96% of the annual rainfall in Aba Gerima and 89% in Anjeni.

Table 10. Major biophysical characteristics describing the study catchments.

		Major land uses (%)			
Catchment	Elevation (m)	Cropland	Grassland	Plantation	Settlement
Aba Gerima (treated)	1875 – 2116	66.6	17.9	1.9	9
Aba Gerima (control)	1896 – 2028	69.5	19.3	6.4	1.5
Anjeni	2407 – 2507	67.8	10.9	13.4	0.9

6.3.2. Data

Rainfall-runoff daily data generated from a total of 26 runoff plots of varying sizes, slopes, land uses, and management condition were used to compute λ and CN for 26 individual sites in the present analysis. Detailed descriptions of the experimental sites are presented in Table 11. The P-Q data used for the present analysis were recorded during two rainy seasons (2017 – 2018) in Aba Gerima and between 1984 and 1993 in Anjeni catchment, yielding a total of 3564 data pairs. Depending on site characteristics and length of data period, the number of P-Q event pairs generated for each runoff plot ranged from 39 (treated cropland with a 5% slope gradient, AGT2) to 228 (untreated cropland with 22% slope gradient, AJT4). The runoff plots include those installed on cropland (77%) and grassland (23%), where 42% of the sites were treated and 58% untreated. Sizes of the runoff plots ranged from 3 to 180 m^2 and slope gradients from 5 to 28%.

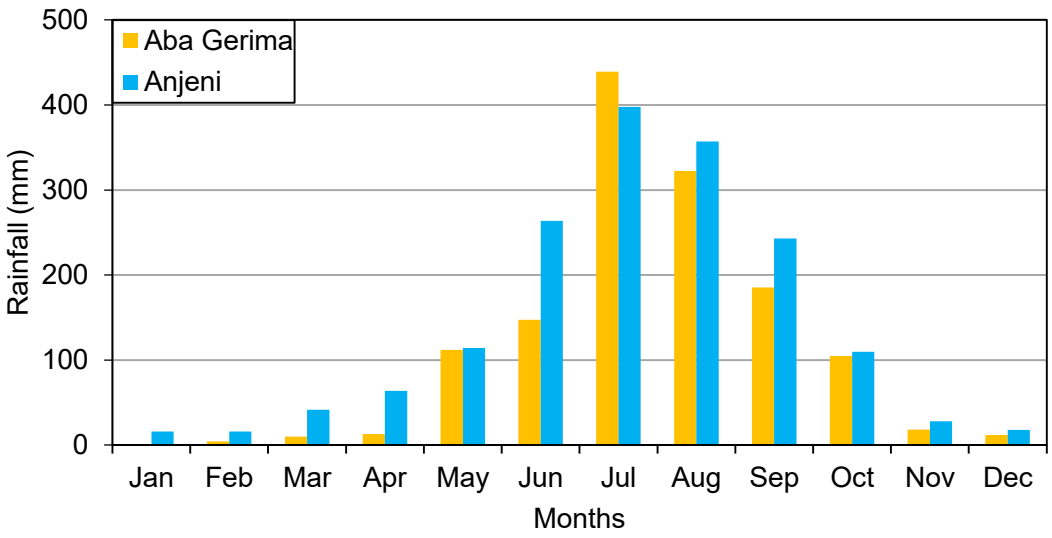


Figure 26. Rainfall pattern of the study catchments.

Two different approaches were adopted in the process of implementing improved land management practices in the study catchments. These include an old approach adopted in 1986 in Anjeni focused on controlling overland flow and soil erosion exclusively on agricultural land while a new approach was adopted in Aba Gerima in 2012, where every parcel of land within the catchment has been targeted for improving productivity and ecological functionality.

Table 11. Major characteristics of experimental sites and data periods.

Catchment	Plot code	Plot size (m^2)	Slope (%)	Top layer soil texture	Hydrologic soil group	Land use	Management practice	No. of P-Q data pairs	Data period
Aba Gerima (control)	AGC1	90	17	Sandy loam	D	Grassland	Protected	93	2017 – 2018
	AGC2	90	5	Clay	C	Cropland	None	102	2017 – 2018
	AGC3	90	10	Clay	D	Cropland	None	106	2017 – 2018
	AGC4	90	15	Sandy clay loam	D	Cropland	None	110	2017 – 2018
Aba Gerima (treated)	AGT1	90	17	Loam	D	Grassland	Protected + trenches	49	2017 – 2018
	AGT2	90	5	Clay loam	C	Cropland	Soil bunds + hedges	39	2017 – 2018
	AGT3	90	10	Clay loam	C	Cropland	Soil bunds + hedges	46	2017 – 2018
	AGT4	90	15	Loam	D	Cropland	Soil bunds + hedges	48	2017 – 2018
Anjeni	AJ11	3	28	Clay	C	Cropland	None	190	1984 – 1991
	AJ12	3	12	Clay	C	Cropland	None	193	1984 – 1991
	AJ13	3	16	Clay	C	Grassland	None	192	1984 – 1991
	AJ14	3	22	Clay	C	Cropland	None	196	1984 – 1991
	AJ21	30	28	Clay	C	Cropland	None	225	1984 – 1993
	AJ22	30	12	Clay	C	Cropland	None	195	1984 – 1993
	AJ23	30	16	Clay	C	Grassland	None	207	1984 – 1993
	AJ24	30	22	Clay	C	Cropland	None	228	1984 – 1993
	AJ31	180	28	Clay	C	Cropland	None	156	1986 – 1991
	AJ32	180	28	Clay	C	Cropland	*Fanya Juu* terraces	142	1986 – 1991
	AJ33	180	28	Clay	C	Cropland	Soil bunds	137	1986 – 1991
	AJ34	180	28	Clay	C	Cropland	Grass strips	97	1986 – 1991
	AJ35	180	12	Clay	C	Cropland	None	137	1986 – 1991
	AJ36	180	12	Clay	C	Cropland	*Fanya Juu* terraces	140	1986 – 1991
	AJ37	180	12	Clay	C	Cropland	Soil bunds	137	1986 – 1991
	AJ38	180	12	Clay	C	Cropland	Grass strips	135	1986 – 1991
	AJ39	180	16	Clay	C	Grassland	None	134	1986 – 1991
	AJ310	180	16	Clay	C	Bare land	None	130	1986 – 1991

6.3.3. Initial abstraction ratio and CN Computations from recorded P-Q data

The CN method is a conceptual rainfall-runoff model developed based on an event-based water budget equation (Equation (6.1)) and two fundamental hypotheses (Equations (6.2) and (6.3)) (Huang et al., 2007; Oliveira et al., 2016; Ponce & Hawkins, 1996).

$$Q = P - I_a - F \tag{6.1}$$

$$\frac{Q}{P-I_a} = \frac{F}{S} \tag{6.2}$$

$$I_a = \lambda S \tag{6.3}$$

where,

P = total rainfall (mm),

I_a = initial abstraction (mm),

F = cumulative infiltration (mm),

Q = direct runoff (mm),

S = potential retention after runoff begins (mm), and

λ = initial abstraction ratio (dimensionless).

The combination of Equations (6.1), (6.2), and (6.3) yields the expression:

$$Q = \frac{(P-\lambda S)^2}{P+(1-\lambda)S} \text{ for } P > \lambda S \tag{6.4a}$$

$$Q = 0 \text{ for } P \leq \lambda S \tag{6.4b}$$

In the presence of observed P-Q data, Equation (6.4a) can be solved for λ and S parameters using an iterative least squares curve fitting procedure involving a sequence of parameter approximations until curve fitting error becomes a minimum (Descheemaeker et al., 2008; Woodward et al., 2003). Once λ and S were determined

iteratively from recorded data, computed S can be transformed into a dimensionless CN parameter using Equation (6.5), which varies in the range $100 \geq CN \geq 0$ as S correspondingly varies in the range $0 \leq S \leq \infty$ (Tedela et al., 2012).

$$CN = \frac{25400}{254+S} \tag{6.5}$$

In the present analysis, the iterative least squares procedure was applied to the partial duration series of P-Q data recorded for individual runoff plots. The use of a partial duration series was recommended by Stewart et al. (2012) who noted that there existed no difference between partial duration series and annual maximum series in CN computation and that selecting a partial duration series instead of annual maximum series provides a sufficiently large dataset for analysis. Also, Tedela et al. (2012) reported CNs computed from partial duration series.

Traditionally, CN computations are made based on the five-day total antecedent rainfall (P_5) describing three antecedent runoff conditions (ARC), including ARC-1 (low runoff potential), ARC-2 (average runoff potential), and ARC-3 (high runoff potential) (Ponce & Hawkins, 1996). In the present analysis, P-Q data pairs obtained from each runoff plot were split into three subsets based on P_5 as described in Table 12 to represent ARC-1, ARC-2, and ARC-3 and then fitted separately to Equation (6.4a) using the iterative least squares procedure to determine respective values of λ and S. Then, Equation (6.5) was used to transform computed S values of ARC-1, ARC-2, and ARC-3 into corresponding values of CN-1 (low runoff potential), CN-2 (average runoff potential), and CN-3 (high runoff potential), respectively. Since the rainfall pattern in the study area is unimodal allowing the collection of P-Q data only during a single rainy season, definition of ARC in the present analysis was based on rainfall limits set for the growing season. Thus, the dormant season criteria set for defining ARC were not applicable for the study sites.

Table 12. ARC thresholds in the study catchments.

Antecedent runoff condition	Description	Rainfall limits
ARC-1	Low runoff potential	$P_5 < 35.6$ mm
ARC-2	Average condition	35.6 mm $< P_5 <$ 53.3 mm
ARC-3	High runoff potential	$P_5 > 53.3$ mm

6.3.4. Statistical analyses

The nonparametric Friedman test was used to examine the differences among the λ and CN parameter values computed for ARC-1, ARC-2, and ARC-3. Besides, the adequacy of the CN method for estimating runoff in the study catchments was evaluated using the normalized mean square error (NMSE) (Equation (6.6)) and Nash and Sutcliffe efficiency coefficient (NSE) (Equation (6.7)).

$$NMSE = \frac{\sum_{i=1}^{n}(Q_{oi}-Q_{ei})^2}{\sum_{i=1}^{n}(Q_{oi}-\overline{Q_o})^2} \tag{6.6}$$

where,

Q_{oi} = i^{th} observed runoff,

Q_{ei} = i^{th} computed runoff, and

$\overline{Q}_o$ = mean of observed runoff.

The minimum value of NMSE is 0, which indicates that computed and observed runoff match perfectly (Feyereisen et al., 2008). An NMSE value of 1.0 suggests that the average of observed values is as good a predictor of runoff as the computed values. NMSE values greater than 1.0 indicate poor agreement between computed and observed runoff, suggesting that the average of observed values is a better predictor of runoff than the computed values.

$$NSE = 1 - \left[\frac{\sum_{i=1}^{n}(Q_{ei}-Q_{oi})^2}{\sum_{i=1}^{n}(Q_{oi}-\overline{Q_o})^2}\right] \tag{6.7}$$

The values of NSE range from -∞ to 1 (Legates & McCabe, 1999). An NSE value of 1 indicates a perfect match between observed and computed runoff. While an NSE value in the range $0 < NSE < 1$ theoretically indicates an acceptable model performance, an NSE value of 0 or less suggests large differences between computed and observed runoff, with the mean of observed values being a better predictor of runoff than the computed values.

6.4. Results and Discussion

6.4.1. Variations of λ and CN under different ARCs

6.4.1.1. Initial abstraction ratios

Table 13 illustrates values of λ and CN computed for individual experimental sites for ARC-1, ARC-2, and ARC-3. Computed values of λ varied in the ranges 0.0 – 0.087 for ARC-1 and 0.0 – 0.076 for ARC-2. However, all experimental sites across the study catchments yielded the same value of λ= 0.0 for ARC-3, where high runoff potential is assumed. Besides, while 84.6% of the experimental sites under investigation presented the same value of λ = 0.0 for ARC-1, ARC-2, and ARC-3, including all the runoff plots in Anjeni catchment and in the control Aba Gerima catchment, values of λ exceeding 0.0 were computed for ARC-1 and ARC-2 of the experimental sites in the treated Aba Gerima catchment. Overall, all values of λ computed for ARC-1, ARC-2, and ARC-3 in the present analysis were found to be much lower than the customary value of λ = 0.2.

6.4.1.2. Runoff curve numbers

Table 13 indicates that CNs computed from the P-Q data of individual experimental plots varied in the ranges 44.40 – 97.58 (CN-1), 46.50 – 97.63 (CN-2), and 37.79 – 97.00 (CN-3). Figure 27 provides scatter plots of P-Q event pairs for six experimental sites along with their best-fit curves representing CN-1, CN-2 and CN-3. The data reveal that the results in the present analysis are not consistent with the basic assumption of the CN method, which matches CN-1 with the lowest runoff potential and CN-3 with the highest (CN-1 < CN-2 < CN-3). In Figure 27, while only AJ24 matches CN-3 with the highest runoff potential and CN-1 with the lowest, the remaining five experimental sites fail to meet this basic assumption of the CN method, with CN-2 presenting either the greatest (AJ12, AJ14, and AJ22) or the lowest (AJ32 and AGC3) runoff potential. Overall, the data imply that the variability of CN in the study catchments is not related to the 5-day total antecedent rainfall (P_5). Descheemaeker et al. (2008) also found similar trends for the semi-arid northern Ethiopian highlands, Tigray, that deviate from the basic assumption of the CN method, with CN-1 < CN-3 < CN-2 and CN-2 < CN-3 < CN-1. It was also noted by

Stewart et al. (2012), Huang et al. (2007), and Dile et al. (2016) that there is no apparent relationship between ARC and CN. This could be explained by the fact that the limits of P_5 originally set to define ARC were based on data obtained from unspecified locations where scale effect and the effect of regional differences had not been accounted for (Ponce & Hawkins, 1996).

Table 13. Estimates of λ and CN from recorded P-Q data for ARC-1, ARC-2, and ARC-3.

Experimental plot	‡Initial abstraction ratio (λ)			‡Potential maximum retention (S)			Runoff curve number (CN)		
	ARC-1	ARC-2	ARC-3	ARC-1	ARC-2	ARC-3	CN-1	CN-2	CN-3
AJ11	0.000 (0.378)	0.000 (0.455)	0.000 (0.280)	6.309 (4.261)	6.173 (5.027)	7.869 (4.163)	97.576	97.627	96.995
AJ12	0.000 (0.299)	0.000 (0.293)	0.000 (0.159)	11.658 (7.966)	8.115 (4.810)	11.087 (3.905)	95.612	96.904	95.818
AJ13	0.000 (0.125)	0.000 (0.141)	0.000 (0.076)	43.000 (27.704)	22.321 (9.884)	29.643 (8.307)	85.522	91.922	89.549
AJ14	0.000 (0.194)	0.000 (0.168)	0.000 (0.122)	21.194 (12.887)	14.003 (5.888)	17.033 (5.684)	92.298	94.775	93.716
AJ21	0.000 (0.265)	0.000 (0.178)	0.000 (0.136)	11.274 (6.800)	12.101 (5.241)	14.128 (4.778)	95.750	95.452	94.731
AJ22	0.000 (0.189)	0.000 (0.214)	0.000 (0.110)	26.721 (17.482)	11.687 (6.005)	20.085 (6.775)	90.481	95.601	92.672
AJ23	0.000 (0.124)	0.000 (0.103)	0.000 (0.078)	32.807 (17.962)	23.589 (7.935)	25.376 (6.571)	88.561	91.502	90.917
AJ24	0.000 (0.142)	0.000 (0.109)	0.000 (0.094)	32.297 (17.910)	23.878 (8.868)	20.016 (5.450)	88.719	91.407	92.695
AJ31	0.000 (0.212)	0.000 (0.106)	0.000 (0.072)	33.452 (25.287)	39.536 (20.924)	30.200 (8.094)	88.362	86.531	89.374
AJ32	0.000 (0.104)	0.000 (0.072)	0.000 (0.051)	73.986 (57.358)	90.805 (60.111)	56.985 (16.374)	77.442	73.665	81.676
AJ33	0.000 (0.083)	0.000 (0.068)	0.000 (0.052)	158.803 (133.163)	93.549 (59.615)	58.294 (17.097)	61.531	73.083	81.334
AJ34	0.000 (0.194)	0.000 (0.089)	0.000 (0.047)	65.373 (83.183)	89.956 (68.591)	76.383 (26.006)	79.531	73.847	76.881
AJ35	0.000 (0.095)	0.000 (0.080)	0.000 (0.038)	119.625 (125.759)	63.083 (34.379)	89.673 (27.011)	67.983	80.105	73.907
AJ36	0.000 (0.107)	0.000 (0.096)	0.000 (0.036)	111.933 (125.379)	58.989 (36.820)	88.118 (24.806)	69.412	81.153	74.243
AJ37	0.000 (0.182)	0.000 (0.132)	0.000 (0.076)	45.736 (43.402)	25.682 (12.283)	44.825 (16.113)	84.741	90.817	85.000
AJ38	0.000 (0.163)	0.000 (0.162)	0.000 (0.060)	53.257 (50.831)	41.559 (33.623)	48.922 (14.714)	82.667	85.939	83.850
AJ39	0.000 (0.287)	0.000 (0.163)	0.000 (0.143)	15.746 (12.151)	18.345 (8.735)	15.004 (5.438)	94.163	93.264	94.423
AJ310	0.000 (0.227)	0.000 (0.163)	0.000 (0.105)	22.576 (16.889)	22.372 (12.580)	18.136 (5.317)	91.837	91.905	93.336
AGC1	0.000 (0.019)	0.000 (0.012)	0.000 (0.020)	84.338 (16.234)	85.611 (4.813)	92.671(13.431)	75.073	74.791	73.268
AGC2	0.000 (0.029)	0.000 (0.049)	0.000 (0.033)	40.090 (6.679)	54.815 (9.648)	38.133 (4.882)	86.368	82.250	86.947
AGC3	0.000 (0.029)	0.000 (0.070)	0.000 (0.070)	33.867 (4.970)	37.328 (7.880)	16.972 (6.211)	88.235	87.187	90.243
AGC4	0.000 (0.071)	0.000 (0.117)	0.000 (0.111)	17.245 (4.198)	18.035 (4.889)	11.004 (2.593)	93.642	93.370	95.848
AGT1	0.000 (0.008)	0.043 (0.009)	0.000 (0.008)	318.074 (81.814)	181.236 (9.215)	410.290 (85.885)	44.400	58.359	38.236
AGT2	0.070 (0.071)	0.004 (0.015)	0.000 (0.009)	113.382 (58.577)	292.220 (44.091)	418.077 (91.089)	69.138	46.501	37.793
AGT3	0.087 (0.066)	0.057 (0.021)	0.000 (0.017)	98.037 (40.448)	154.140 (15.866)	354.334 (134.756)	72.152	62.234	41.753
AGT4	0.032 (0.035)	0.076 (0.025)	0.000 (0.018)	135.431(50.959)	115.497 (11.418)	280.151 (87.304)	65.223	68.742	47.552

‡Values in parentheses are standard errors of parameter estimates.

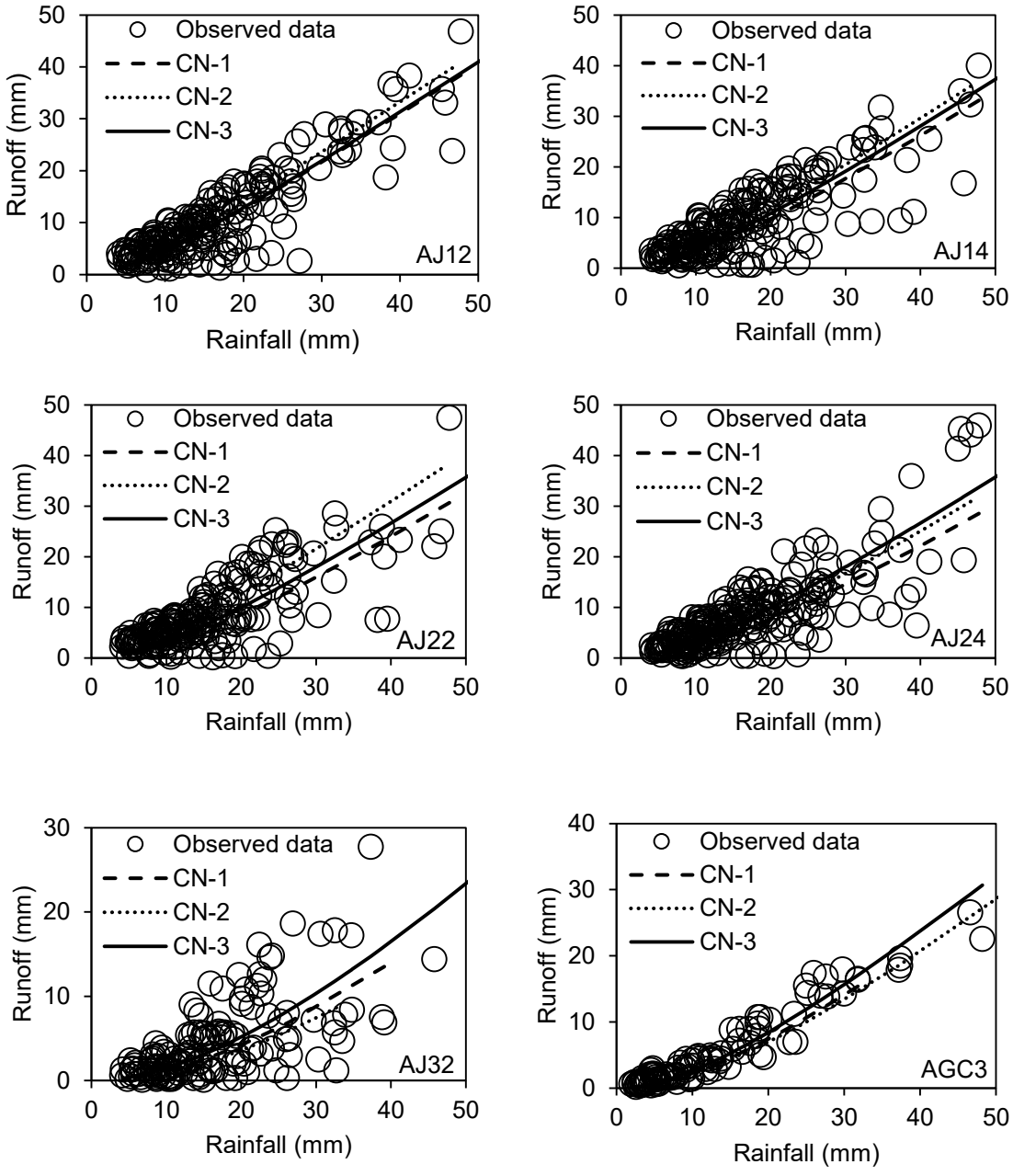


Figure 27. Scatter plots of observed P-Q data for six runoff plots and their best-fit curves describing CN-1, CN-2 and CN-3.

6.4.2. Initial abstraction ratios and CNs computed from merged data

Apart from the results of the iterative least squares curve fitting presented in Figure 27, the nonparametric Friedman test revealed that none of the λ and CN parameters computed for all experimental sites exhibited statistically significant variations associated with ARC ($p > 0.05$), implying that the variability of λ and CN in the study catchments is not influenced by P_5. The results are in agreement with those reported by Stewart et al. (2012) who stated that the P_5-based criteria to describe ARC are not relevant for CN analysis. Descheemaeker et al. (2008) and Dile et al. (2016) also noted that variations in P_5 had no effect on CN-based runoff estimation. Thus, the iterative least squares procedure was applied to fit the entire P-Q data of individual experimental sites to the basic CN Equation without splitting them based on the P_5 criteria.

6.4.2.1. Initial abstraction ratios

Table 14 presents the λ and CN parameter estimates obtained from the merged data of individual experimental sites. Overall, the merged data resulted in computed values of λ much lower than 0.2, which obviously varied in the range 0.0 – 0.030. The data presented the same value of $\lambda = 0.0$ for 84.6% of the experimental sites under investigation. Thus, only 15.4% of the computed values of λ were found to be exceeding 0.0. Several other studies have also reported values of λ lower than 0.2 (Baltas et al., 2007; Krajewski et al., 2020; Shi et al., 2009). Baltas et al. (2007) determined λ values ranging from 0.014 to 0.054. Krajewski et al.(2020) found values in the range 0.002 – 0.188. The values computed by Shi et al. (2009) varied between 0.010 and 0.154.

6.4.2.2. Runoff curve numbers

Curve number computations from the merged P-Q data resulted in values ranging from 44.64 to 97.25 (Table 14). The lowest CN corresponds to a treated cultivated land with a 5% slope gradient (AGT2) while the greatest value corresponds to an untreated cultivated land with a 28% slope gradient (AJ11). The results reveal that the range of CN computed for the study catchments is similar with the ranges reported in the literature (Descheemaeker et al., 2008; Tedela et al., 2012). Tedela et al. (2012) computed CN

values in the range 40 – 84 for P-Q data recorded from 10 forested catchments across the US, stressing that variations in CN were attributable to differences in hydrologic soil group (HSG). In addition, CN values reported by Descheemaeker et al. (2008) for semi-arid northern Ethiopian Highlands, Tigray, varied between 29 and 97, with the lowest value found for a church forest and the highest for a communal grazing land.

6.4.3. Evaluation of the adequacy of the CN method in the study catchments

The iterative least squares fitting of merged P-Q data to the basic CN equation resulted in R^2 values in the range 0.179 – 0.962 (Table 14). Further, values of the normalized mean square error (NMSE) and the Nash and Sutcliffe efficiency coefficient (NSE) computed for the experimental sites were found to be in the ranges 0.038 – 0.821 and 0.179 – 0.962, respectively. Figure 28 presents the NSE values computed for the experimental sites under study, describing the performance of the CN method for the study area. Apart from the values of R^2, the ranges of both NMSE and NSE computed for the experimental sites suggest the adequacy of the CN method for estimating direct runoff in the study catchments.

Table 14. Estimates of λ, S and CN from recorded P-Q data along with NRCS-generated CNs.

Experimental plot	Number of P-Q events	‡λ	‡S	R^2	Computed CN	NRCS-generated CN
AJ11	190	0.000 (0.210)	7.185 (2.740)	0.773	97.246	85
AJ12	193	0.000 (0.130)	10.539 (2.890)	0.783	96.016	85
AJ13	192	0.000 (0.060)	28.764 (6.330)	0.539	89.828	89
AJ14	196	0.000 (0.090)	16.879 (4.110)	0.640	93.769	85
AJ21	225	0.000 (0.100)	13.143 (3.180)	0.736	95.080	85
AJ22	195	0.000 (0.090)	18.544 (4.820)	0.533	93.196	85
AJ23	207	0.000 (0.060)	25.566 (4.850)	0.701	90.855	89
AJ24	228	0.000 (0.070)	22.422 (4.610)	0.644	91.889	85
AJ31	156	0.000 (0.060)	32.159 (7.370)	0.555	88.762	85
AJ32	142	0.000 (0.040)	63.396 (15.310)	0.385	80.026	79
AJ33	137	0.000 (0.040)	74.034 (18.700)	0.179	77.431	79
AJ34	97	0.000 (0.040)	77.096 (23.750)	0.274	76.715	79
AJ35	137	0.000 (0.030)	86.689 (22.090)	0.268	74.555	74
AJ36	140	0.000 (0.030)	83.562 (21.440)	0.218	75.245	79
AJ37	137	0.000 (0.060)	40.134 (11.400)	0.187	86.355	82
AJ38	135	0.000 (0.050)	47.913 (13.290)	0.337	84.130	82
AJ39	134	0.000 (0.100)	15.810 (4.270)	0.760	94.140	86
AJ310	130	0.000 (0.080)	19.418 (4.650)	0.727	92.898	94
AGC1	93	0.000 (0.010)	88.282 (5.060)	0.962	74.208	71
AGC2	102	0.000 (0.020)	43.996 (3.680)	0.947	85.236	85
AGC3	106	0.000 (0.030)	31.678 (3.370)	0.935	88.911	85
AGC4	110	0.000 (0.060)	14.136 (1.990)	0.956	94.728	85
AGT1	49	0.020 (0.010)	227.416 (18.360)	0.940	52.761	65
AGT2	39	0.003 (0.010)	314.998 (31.620)	0.920	44.640	77
AGT3	46	0.027 (0.010)	194.702 (22.570)	0.900	56.608	77
AGT4	48	0.030 (0.020)	155.1150 (16.910)	0.915	62.085	77

‡Values in parentheses are standard errors of parameter estimates.

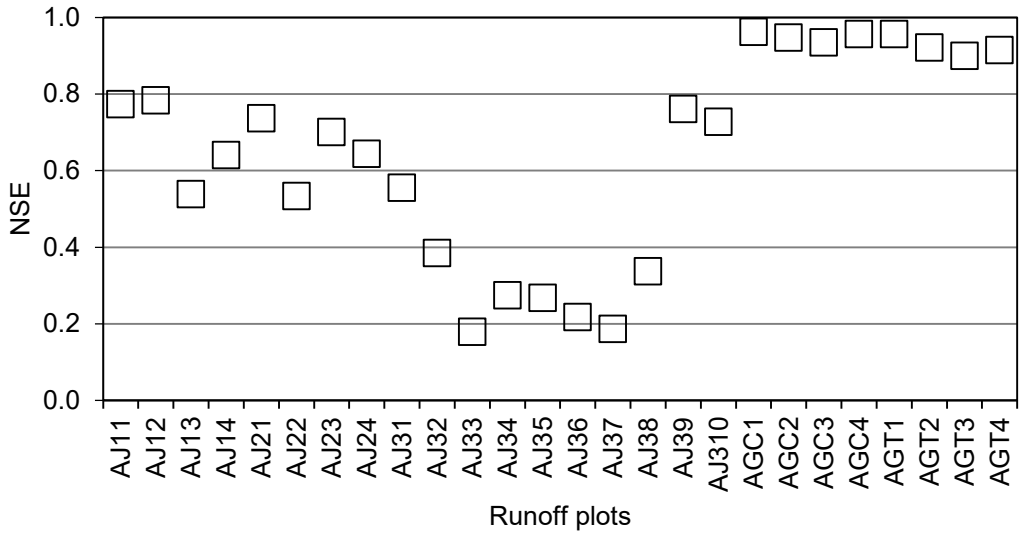


Figure 28. Values of NSE describing the performance of the CN method in the study area.

6.4.4. Variations in hydrologic response under varying land uses and management conditions

The effect of severe soil degradation has greatly modified soil properties in the study catchments, subsequently reducing the variation in soil hydrologic properties. Hence, the results revealed little association between HSG and λ and CN for the study catchments. The values of λ computed in the present analysis for the study catchments were much lower than the standard value of λ = 0.2, describing poor infiltration capacity and, hence, quick runoff response (Baltas et al., 2007; Woodward et al., 2003). Losses of soil structure and porosity resulting from depletion of soil organic matter and high level of surface disturbance are responsible for such a poor hydrologic condition observed in the study catchments. Similarly, Descheemaeker et al. (2008) found no association between HSG and CN in semi-arid northern Ethiopian Highlands affected by quite a similar level of soil degradation.

On the other hand, the present analysis reveals strong associations between land management and λ and CN, with land management influencing the potential of a site to retain rainwater during storm events. Figure 29 presents the computed values of λ for the experimental sites under study. The effect of land management was reflected in the observed values of λ, with all inadequately treated sites presenting λ = 0.0 while

sites under adequate management interventions resulted in λ exceeding 0.0. Hence, values of λ exceeding 0.0 were found for all experimental sites in the treated Aba Gerima catchment (AGT1, AGT2, AGT3, and AGT4), where a new integrated SLM approach has been implemented. The new approach of implementing SLM in this particular catchment has promoted recovery of natural hydrologic functionality, leading to reduced runoff response, which is evidenced by increases in λ. The emerging SLM approach introduced in the treated Aba Gerima catchment focuses on integrating multiple practices, such as engineering structures (*Fanya Juu* terraces; soil bunds), vegetative measures (planting of fodder species on engineering structures; area closure), and elimination of post-harvest grazing on cultivated land, allowing the retention of more surface cover and the buildup of soil organic matter, which is critical for improving soil water uptake and retention.

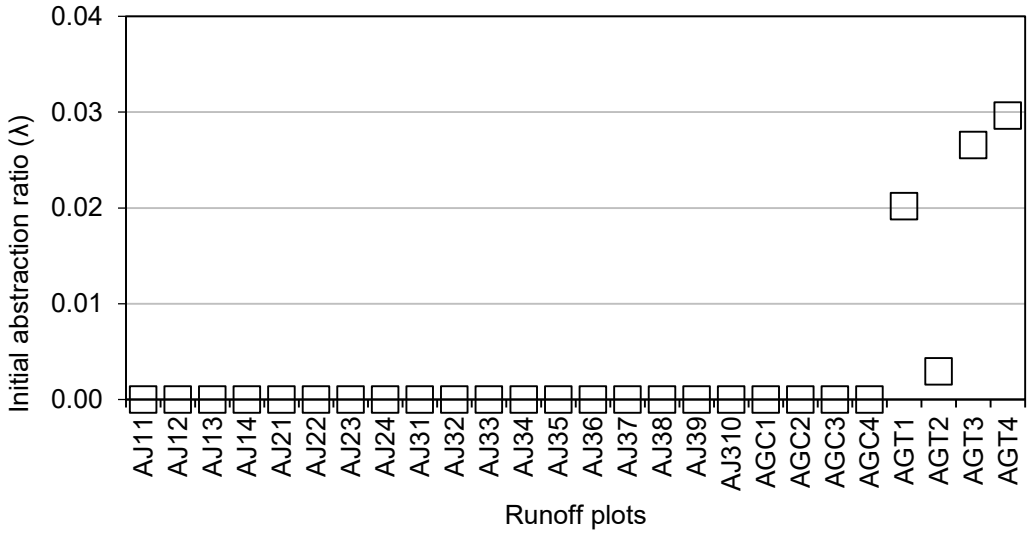


Figure 29. Observed variations in initial abstraction ratio (λ) across experimental sites.

In Anjeni catchment, no variation in computed λ was observed between treated and control sites, indicating that the SLM options dominated by structural measures installed exclusively on cultivated land exerted little influence on hydrologic response and, hence, did not adequately contribute to water resources management. Long-term records of the runoff coefficient for Anjeni catchment also exhibited little reduction over the years (Hurni et al., 2005), presenting further evidence that SLM treatments implemented in this particular catchment contributed little to improving water

resources management. The use of engineering structures alone in Anjeni catchment without allowing for the retention of crop residues provided little role for improving surface cover and the building up soil organic matter content, which is critical for enhancing soil water infiltration and retention. This suggests that there is a need to focus on practices that improve soil hydrologic and hydraulic properties that promote rainwater infiltration and retention.

The hydrologic effect of land management was also reflected by observed variations in CN across the study catchments (Figure 30), where all the treated experimental sites across Anjeni (AJ32, AJ33, AJ34, AJ36, AJ37, and AJ38) and Aba Gerima (AGT1, AGT2, AGT3, and AGT4) demonstrated substantial reductions in CN. However, observed reductions in CN were greater in the treated Aba Gerima catchment than in Anjeni catchment. For instance, AJ32 presented 17.3 and 15.1 units of reduction in CN compared with untreated sites (AJ11 and AJ21, respectively) lying on similar soil (clay), land use (cropland), and slope gradient (28%). Similarly, AJ36 (a cropland with 12% slope gradient) displayed redactions in CN of 20.8 and 18 units relative to AJ12 and AJ22 (sites belonging to similar soil, land use, and slope gradient), respectively. On the other hand, CNs computed for AGT2 (a cropland with a 5% slope gradient) and AGT4 (a cultivated site on a 15% slope gradient) were 40.6 and 32.6 units lower than their control counterparts (AGC2 and AGC4, respectively), revealing a greater impact of SLM on improving water availability and storage in the treated Aba Gerima catchment.

Further, of all the runoff plots installed in Anjeni catchment, four (AJ13, AJ23, AJ39, and AJ310) were set up on grazing land under the traditional practice of open communal grazing, which displayed the same value of $\lambda = 0.0$ and CNs in the range 89.8 – 94.1. For AGC1 and AGT1 (Aba Gerima), on the other hand, where open communal grazing has been entirely abandoned, the P-Q data resulted in computed λ values of 0.0 and 0.02, respectively. In addition, relative to those four sites in Anjeni catchment under open communal grazing, reductions in CN were substantially high at both AGC1 (CN = 74.2) and AGT1 (CN = 52.8). Though both AGC1 and AGT1 are sites that have been protected from open communal grazing since 2012, an additional treatment of contour trenching that AGT1 has received led to a relatively faster site

recovery, which is reflected by a relatively higher λ and lower CN computed for AGT1.

Under untreated conditions, both cultivated and grazing lands across the study catchment resulted in quite similar ranges of λ and CN, suggesting that there is little variation in hydrologic response between cultivated and grazing lands under untreated conditions in the study catchments. On the other hand, treated sites in Anjeni and Aba Gerima displayed substantial variations in λ and CN. While all treated sites in Anjeni catchment yielded $\lambda = 0.0$ and $75.25 < CN < 86.36$, values of λ and CN computed for sites in the treated Aba Gerima catchment varied in the ranges 0.003 – 0.030 and 44.64 – 62.09, respectively, revealing a considerable variation in hydrologic response between these two treated catchments stemming from the difference in approach while implementing catchment management schemes. In Anjeni, where SLM measures were implemented 36 years ago, the primary goal was to control overland flow and soil erosion on agricultural land. By contrast, in the treated Aba Gerima catchment, where a spectrum of SLM options have been put in place since 2012, every parcel of land within the catchment has been targeted for enhancing productivity and ecological functionality. Hence, observed increases in λ and greater reductions in CN displayed by experimental sites in the treated Aba Gerima catchment reveal great promises for improving water availability and storage through properly designed SLM interventions.

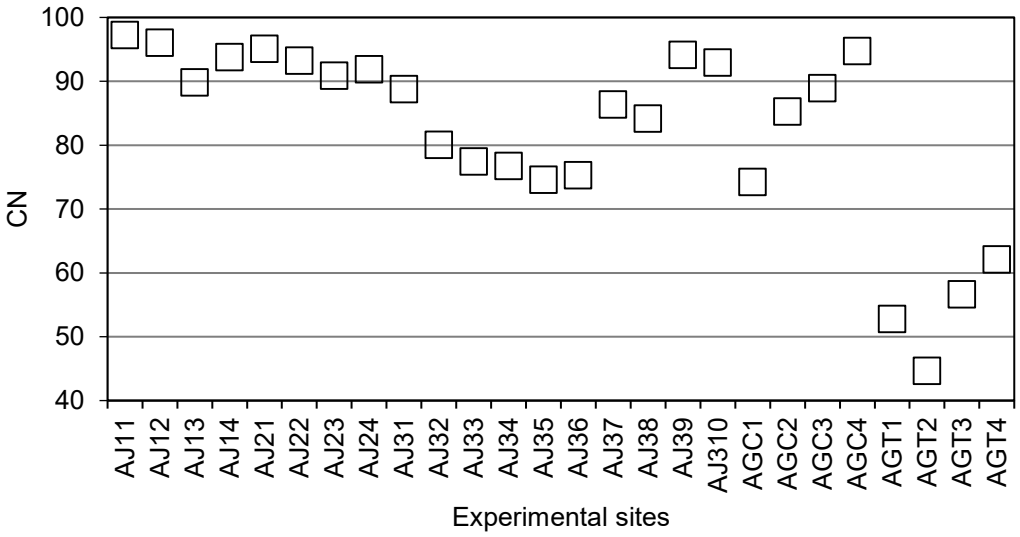


Figure 30. Variability of CN across experimental sites.

The impact of SLM on improving soil water uptake and retention capacity in agricultural landscapes has often been reported in the literature (Bonta & Shipitalo, 2013; Bonta, 2013; Dabney et al., 2012; Descheemaeker et al., 2008; Elhakeem & Papanicolaou, 2009; Feyereisen et al., 2008; Oliveira et al., 2016). Bonta & Shipitalo (2013) reported a reduction in CN (from 90.6 to 74.6) promoted by a change in management from conventional to mulch tillage. Dabney et al. (2012) found considerable variations in CN among cultivated plots under different treatments, including no treatment (CN = 78), plant hedges (CN = 70), and plant hedges plus berms (CN = 61). Elhakeem & Papanicolaou (2009) indicated that higher levels of residue cover can result in reduced CN. Further, Feyereisen et al. (2008) obtained reduced CN resulting from a shift in management from conventional to strip tillage.

Again, Bonta (2013), Descheemaeker et al. (2008), and Oliveira et al. (2016) noted that CNs tend to be low under undisturbed surface conditions. Thus, Descheemaeker et al. (2008) obtained CNs in the range 90 – 97 for grasslands under open communal grazing in semi-arid northern Ethiopia Highlands, Tigray, which decreased to 49 – 90 and 35 – 57 after closure for 3 – 5 and 20 years, respectively. Descheemaeker et al. (2008) computed CN = 29 for a church forest, which describes an undisturbed condition. For an undisturbed condition, Oliveira et al. (2016) also reported CN = 49, which increased to as high as 79.3 for a bare soil. Further, Bonta (2013) noted that CN decreased from 77.0 to 65.4 upon a change in management from a 29-year of pasture to a 3.7-year of meadow.

6.4.5. Comparison of computed and NRCS-generated CNs

In the present study, the experimental sites under investigation resulted in computed CNs predominantly higher than values published in standard NRCS tables (69%). Observed differences between computed and tabular CNs ranged from 0.24 to 12.25 units. Stewart et al. (2012), Feyereisen et al. (2008), and Huang et al. (2007) also reported that computed CNs are higher than tabular values. In addition, hydrologic simulation studies in the Blue Nile Basin based on CNs derived from NRCS standard tables often resulted in underestimation of direct runoff (Descheemaeker et al., 2008; Dile et al., 2016; Zelelew, 2017). This implies that tabular CNs published by NRCS are inadequate to represent local conditions in the study area (Feyereisen et al., 2008; Van Griensven et al., 2012).

Though CNs computed for untreated sites predominantly exceeded tabulated CNs in the study catchments, CNs lower than tabular values were also found for treated sites whose runoff potentials have been considerably reduced. Similarly, Oliveira et al. (2016) and Bonta & Shipitalo (2013) found computed CNs lower than tabular values for sites characterized by low runoff potential. Overall, given the fact that runoff estimation is much more sensitive to errors in CN estimates than causal rainfall (Hawkins, 1993), the observed differences between computed and published CNs in the study catchments suggest a call for local CNs computed from recorded P-Q data for reducing output uncertainties associated with CN-based hydrologic simulations in the study area.

6.5. Conclusions

A study was conducted to determine initial abstraction ratios (λ) and runoff curve numbers (CN) for varying land uses and management conditions in the Blue Nile Highlands, Ethiopia. The analysis was based on 3564 rainfall-runoff (P-Q) event pairs measured from 26 runoff plots across three agricultural headwater catchments. Computations of λ and CN were carried out for each of the 26 runoff plot using an iterative least squares curve fitting procedure.

The present analysis revealed that variations in λ and CN in the study area are related neither to antecedent runoff conditions (ARC) nor to hydrologic soil group (HSG). Instead, differences in land management exerted the greatest influence on observed variations in λ and CN across the study catchments. The hydrologic effect of land management practices was strongly evidenced by the observed variability in λ, where all inadequately managed experimental sites displayed $\lambda = 0.0$ while those sites installed in the treated Aba Gerima catchment, where a new integrated SLM approach has been implemented, presented λ exceeding 0.0. Overall, values of λ computed for the experimental sites were found to be much lower than the standard value of $\lambda = 0.2$ determined by NRCS, indicating quick runoff response in the study area.

Runoff CNs computed for all experimental sites across the study catchments varied in the range 44.64 – 97.25. The hydrologic effect of land management practices in the study catchments was clearly demonstrated by the observed variability in CN, where the lowest computed values of CN were associated with sites under improved

catchment management conditions and the greatest values with untreated sites. Most experimental sites presented CN values higher than values published in NRCS standard tables (69%).

Evaluation of the adequacy of the CN method for the study area presented normalized mean square error (NMSE) and Nash and Sutcliffe efficiency coefficient (NSE) values in the ranges 0.038 – 0.821 and 0.179 – 0.962, respectively. The limits of NMSE and NSE determined in the present analysis suggest that the CN method is adequate for estimating direct runoff in the study area. We conclude that the underestimation of direct runoff often reported by previous CN-based simulation studies in the Blue Nile basin could be rectified via model parameterization based on locally recorded P-Q data.

7. GENERAL CONCLUSIONS AND STUDY LIMITATIONS

7.1. General Conclusions

Water is a fundamental natural resource, providing key productive opportunities in a spectrum of economic sectors, including agriculture, industry, and energy. Owing to rapid increases in global population and economic development, water demand for sustaining water-dependent economic activities has increased unprecedentedly. Globally, water withdrawal has increased by nearly 8 times between 1900 and 2010.

Of all water-dependent economic activities, agriculture forges the strongest association with water security, accounting for about 70% of the total freshwater withdrawal globally. Given agriculture is by far the greatest consumer of freshwater than any other economic sector, there will be a 70% increase in global food demand, and up to a 100% increase in developing countries, to feed a projected world population of 10 billion by the mid of the century. Hence, access to sufficient supply of water to meet projected increases in global food demand presents a dominant water security challenge.

The potential of irrigated agriculture is limited in meeting the food needs of a rapidly increasing world population. Agricultural water withdrawal represents only about 20% of the total consumptive water use in agriculture while the remaining 80% is green water use in rainfed systems. Besides, irrigated agriculture would only supply about 16% of the water deficit required to meet the global food security challenge by the mid of the century, even with its maximum potential harnessed through increased water withdrawals and improved water use efficiencies. Hence, securing the food needs of the rapidly increasing world population will continue to rely on green water use in rainfed systems.

About 60% of world's food production is rainfed, which accounts for over 90% of the staple food production in sub-Saharan Africa. However, while green water plays a central role in sustaining agricultural production in rainfed systems, it received little attention in studies of freshwater availability in the past. Since recently, there is a growing realization that blue water is not the only freshwater resource involved in societal development, leading to a paradigm shift in water resources planning and

management to balance between blue and green water needs for agriculture, nature, and society.

Apart from increases in water demand associated with increasing food needs due to rapidly expanding world population, agriculture impacts water security through changes in the terrestrial water cycle stemming from agricultural development and intensification. While rainfall in rainfed systems is predominantly enough to significantly increase grain yields without large-scale irrigation, up to 70 – 85% of the rainfall is lost non-productively through soil evaporation, deep percolation, and surface runoff. Such large non-productive rainwater losses often lead to disproportionately low grain yields particularly in smallholder rainfed systems, with an average productivity of 1.5 t ha^{-1} compared with 5 – 6 t ha^{-1} in commercial rainfed farming. In sub-Saharan Africa, where food production almost entirely depends on green water availability in the root zone, only 10 – 30% of the seasonal rainfall is used in productive green water flow (transpiration), with up to 50% lost as non-productive soil evaporation, 30% as surface runoff, and 25% as deep percolation. In Ethiopia, non-productive rainwater losses are extremely high, with 72 – 81% lost through evapotranspiration and 11.3 – 21.4% through runoff, leaving only 2.3 – 4.4% of the seasonal rainfall as green water storage in the soil.

Investing in sustainable land management (SLM) interventions can greatly improve on-farm water productivity in rainfed agricultural systems by reducing non-productive rainwater losses, with a potential to increase grain yields by up to 5 – 10 times. Experiences in sub-Saharan Africa show increases in grain yields by 20 – 120% promoted by shifts in farming practices from conventional plowing to conservation tillage. In Ethiopia, superimposing conservation farming on already terraced fields resulted in the doubling of the grain yield of *tef*.

Large-scale implementation of sustainable land management (SLM) practices in Ethiopia dates back to the early 1970s primarily to curb soil degradation in the drought-prone Ethiopian Highlands, where more than 90% of the country's cultivated land is contained. Until the recent past, the focus of SLM practices in Ethiopia had been to control onsite soil erosion exclusively on croplands, predominantly through runoff (blue water) management using soil bunds and other forms of terraces. While field experimentation and simulation modelling revealed that such basic SLM

practices have played a substantial role in controlling onsite soil erosion, they have barely contributed to improved green water security and agricultural productivity in the long term.

On the other hand, success stories around the globe revealed that, when properly designed, SLM practices can effectively bridge management-induced water scarcity and improve agricultural productivity in rainfed systems. By enhancing rainwater infiltration and soil moisture availability and storage, SLM options can help boost water use efficiency by up to 25 – 40% and water productivity by 70 – 100% in rainfed systems. In sub-Saharan Africa, improvements in water productivity by 20 – 120% have been reported as a result of shifting farming practices from conventional plowing to conservation tillage.

Yet, while much has been reported about changes in water availability associated with climate change and deforestation, the implications of changes associated with agricultural land management are much less understood as well as inadequately modeled especially in sparsely-gauged environments, such as the Ethiopian Highlands, due to lack of skill and/or relevant data. Given effects of land management are complex, they depend very much on local context, which particularly applies to Ethiopia, where topographic diversity and associated climate contrasts are high. As a result, the potential of SLM practices for improving green water security and agricultural productivity is poorly understood in the subsistence rainfed agricultural system prevailing in the Ethiopian Highlands.

The major objective of the present thesis has been to assess the impact of SLM practices on improving water security in the Blue Nile Highlands, where early human settlement and subsistence rainfed farming have in the long term led to serious soil degradation and associated green water scarcity. The present study was motivated by the need to provide support for the mainstreaming of water security into current SLM initiatives widely implemented across the fragile Blue Nile Highlands. Given the reliability of simulation modeling in the study area is limited by the paucity of long-term data, a paired-catchment approach based partly on historical data and partly on data recorded during the study period provided an effective tool for analysis.

Impact studies of anthropogenic activities on catchment hydrology have often involved either long-term before- and after-intervention monitoring of a single catchment or a paired-catchment approach involving two or more catchments. The paired-catchment approach requires relatively shorter time periods ranging from a few months to years for the analysis of treatment effects than the long-term time trend approach involving a single catchment. Apart from providing an experimental investigation of the effects of intentional treatments on catchment hydrology, the paired-catchment approach can be applied based on existing observational data with satisfactory results.

To meet the study objectives, three agricultural catchments were selected in the headwaters of the Blue Nile Basin, where two of the study catchments were under SLM interventions while the third was managed traditionally. The selection of two study catchments under SLM interventions stems from the need to account for the evolution of SLM practices over the years. On the other hand, having been under traditional farming practices since time immemorial, the third catchment presented a control counterpart for the two treated catchments.

The treated catchments include Anjeni catchment, which has been under SLM interventions since 1986 (36 years ago), and a second catchment in Aba Gerima area, which has been treated since 2012 (10 years ago). In Anjeni catchment, mainly structural measures, such as soil bunds and *Fanya Juu* terraces, were put in place to control onsite soil erosion exclusively on cultivated land. By contrast, the SLM options introduced in the Aba Gerima area include quite diverse practices targeting every parcel of land at the catchment scale, including cultivated land, gullied land, grazing land, etc., to enhance landscape productivity and ecological functionality.

The data used in the present analysis include runoff, soil moisture, and climate data recorded between 1984 and 2018 across multiple sites within the study catchments. Impacts of SLM practices in the study catchments over the past several decades have been examined based on the dynamics in various components of the terrestrial water cycle. Using runoff and causal rainfall data recorded over such an extended period, local values of the initial abstraction ratio (λ) and runoff curve number (CN) parameters have been generated for varying land uses and management conditions across the study catchments, providing a reliable tool for assessing the impact of SLM

practices on hydrologic response and associated infiltration-runoff dynamics. Volumetric soil moisture content monitored intensively at multiple soil profile depths across multiple sites were used to assess soil water recharge and soil moisture spatial and temporal variability under improved and traditionally managed catchment conditions.

The impact of SLM interventions on infiltration-runoff dynamics was examined using a standard asymptotic curve fitting procedure describing the relationship between rainfall events (P) and associated curve numbers (CN). Curve numbers computed from observed rainfall-runoff data presented a strong relationship with P, with CN decreasing asymptotically to a constant P-independent value as P becomes large, which is termed as the asymptotic curve number (CN_∞). Hence, in the treated Aba Gerima catchment, where an emerging integrated SLM approach has been implemented, all experimental sites exhibited substantial reductions in CN_∞ (-13.9 to -21.6 units) relative to their control counterparts that are under the traditional farming system, indicating recovery of infiltration capacity and soil water retention potential of the agricultural landscapes of the treated catchment.

The impact of SLM interventions on the recovery of natural landscape hydrologic functionality in the study area has also been demonstrated by the change in volumetric soil moisture content (θ) as the wet season progresses. Hence, though θ increased with an increase in cumulative rainfall across both the treated and control sites, indicating the buildup of soil moisture as the wet season progresses, changes in θ were predominantly higher in the treated Aba Gerima catchment, demonstrating higher soil water recharge promoted by SLM interventions This was, in turn, reflected in observed increases in volumetric soil moisture content in the treated catchment, which were in the range 16 – 800%, relative to the untreated catchment.

Further, correlation, autocorrelation, cross correlation, and regression techniques were applied to the time series of volumetric soil moisture content (θ) to investigate θ response to SLM interventions. The results revealed that soil moisture temporal variability and spatial heterogeneity have been substantially reduced in the treated Aba Gerima catchment relative to its control counterpart. Correlation analysis presented strong associations ($p < 0.001$) between daily θ and a range of climate

variables, such as rainfall (P), air temperature (T), relative humidity (RH), and antecedent soil moisture content (θ_A) for all monitoring sites both in the treated and control catchments. However, a temporal autocorrelation analysis applied to the time series of daily θ for increasing time lags revealed consistently larger time lags for the experimental sites in the treated Aba Gerima catchment than for those in the control Aba Gerima catchment. This, therefore, suggests that SLM interventions caused reduced θ response to daily climatic variation, hence, leading to a temporally more stable θ regime in the treated than in the control catchment. In addition, a stepwise regression analysis presented a characteristic inverse relationship between the temporal mean of θ and CV, with regression slopes and R^2 reduced for the treated than the control catchment. This implies a reduced θ response to rainfall seasonal fluctuation in the treated than in the control catchment. A temporally more stable θ regime observed in the treated catchment regardless of daily climate variability and rainfall seasonality indicates improved rainfall infiltration and soil moisture retention and redistribution promoted by improved catchment management.

Cross correlation and stepwise regression techniques were applied to investigate the impact of SLM interventions on soil moisture spatial heterogeneity. A cross correlation analysis applied to examine the time lag relationships between any two soil moisture monitoring depths within an experimental site presented generally higher time lags for the control than the treated catchment. Hence, while the highest time lags associated with the control catchment in Aba Gerima suggest a relatively uneven θ profile distribution, the reduced time lags observed in the treated catchment indicate a relatively faster soil water exchange and, hence, more uniform θ profile distribution, promoted by improved catchment management. Besides, a stepwise regression applied to understand the relationship between θ and soil depth presented consistently lower regression slopes and R^2 values for soil moisture monitoring sites in the treated catchment than in the control, implying a relatively reduced θ profile variability in the treated catchment.

Also, a cross correlation analysis at a time lag of zero applied to understand the degrees of association between the time series of θ across experimental sites within a catchment yielded consistently higher correlation coefficients for the treated than the control catchment, indicating that θ response is spatially more uniform in the treated

catchment than in the control. The relatively lower correlation coefficients displayed by the control catchment indicate a spatially more varied θ response across sites, due to the effect of spatially varying hydrologic processes, such as surface runoff and rainfall infiltration.

Making use of the wealth of rainfall-runoff data generated in the study catchments over an extended time period spanning from 1984 to 2018, local values of the initial abstraction ratio (λ) and the runoff curve number (CN) parameters have been generated for the study catchments to evaluate the dynamics in land management practices and associated changes in hydrologic response. Though λ traditionally takes a fixed value of 0.2 in CN-based simulation modeling, it has often been cited to vary depending on geographic location. In addition, despite the fact that estimation of direct runoff by the CN method is much more sensitive to errors in CN parameter estimates than similar levels of error in storm rainfall, standardized tabular CNs are used in CN-based simulation modeling across various regions of the world. Hence, unrealistic estimates of λ and CN are often the causes of model output uncertainties in CN-based simulation studies. Therefore, to considerably reduce model output uncertainty and provide policy-relevant simulation results, estimation of λ and CN from locally recorded rainfall-runoff data are often recommended. However, such studies to determine λ and CN from local rainfall-runoff data are lacking in Ethiopia. In the present analysis, an attempt has been made to adapt the CN method to the Blue Nile Highlands using a considerable amount of rainfall-runoff data recorded over the years. To do this, an iterative least squares procedure was applied to fit the rainfall-runoff data of individual experimental sites to the basic CN Equation. The iterative least squares fitting of rainfall-runoff data to the basic CN equation resulted in Nash and Sutcliffe efficiency coefficients (NSE) in the range 0.179 – 0.962, suggesting the adequacy of the CN method for estimating direct runoff in the study catchments.

The data resulted in computed values of λ much lower than the customary value of 0.2, which varied in the range 0.0 – 0.030, and CNs predominantly higher than values published in standard tables, with observed differences between computed and tabular CNs ranging from 0.24 to 12.25 units. Much lower values of λ as well as higher CNs than the standard tabular CNs observed in the study catchments describe poor infiltration capacity and, hence, quick runoff response, stemming from soil hydrologic

properties modified by chronic soil degradation prevailing in the study catchments. However, the present analysis reveals that λ and CN parameters have strong associations with land management, indicating the influence of land management on hydrologic response in the study catchments.

The effect of land management on hydrologic response was reflected by observed variations in λ across the experimental sites, where all inadequately treated sites presented λ = 0.0 while λ exceeded 0.0 for sites under adequate management interventions. In effect, values of λ exceeding 0.0 were found for all experimental sites in the treated Aba Gerima catchment, where a new integrated SLM approach has been implemented, indicating a lag effect in runoff response. The hydrologic effect of land management was also reflected by observed variations in CN across the study catchments, where both experimental sites under basic SLM treatments (Anjeni catchment) and integrated SLM interventions (the treated Aba Gerima catchment) demonstrated substantial reductions in CN relative to the control catchment under traditional farming practices. Yet, the observed reductions in CN were greater in the treated Aba Gerima catchment than in Anjeni catchment, indicating the influence of the evolution in SLM approach on hydrologic response, with the emerging integrated SLM approach presenting a more effective water management role.

Overall, SLM practices in the Blue Nile Highlands have evolved substantially over the years from practices that focused on controlling soil erosion on agricultural land to an approach that targets every land use within a catchment for enhancing landscape productivity and restoring natural hydrologic functionality. Our results showed that SLM practices dominated by the construction of soil bunds and *Fanya Juu* terraces in Anjeni catchment to exclusively protect cultivated land from soil erosion exhibited little improvement over the traditional farming practices prevailing in the study area in terms of enhancing green water availability and storage. On the other hand, in Aba Gerima, where the emerging SLM approach targeting every land use type at the catchment scale has been implemented since 10 years ago, rainwater infiltration and soil water storage have been substantially improved relative to the traditional farming practices prevailing in the control catchment as well as to the basic SLM practices introduced 36 years ago in Anjeni catchment to control soil erosion on agricultural land.

The results reveal that while the Blue Nile Highlands are ecologically fragile, they are also highly responsive to management interventions. SLM interventions in the Blue Nile Highlands have evolved into practices that possess great promises for restoring natural hydrologic functionality in the agricultural landscapes of the study area, laying the basis for improved agricultural water security and increased productivity. Given the primary target of SLM practices is to reduce nonproductive rainwater losses and enhance green water availability and storage, we conclude that SLM practices provide a large opportunity for upgrading agricultural productivity in the study area without significantly affecting overall catchment runoff for downstream water users and ecosystems.

In studies in the past, soil moisture data have not received adequate attention to assess the impact of SLM interventions in the Ethiopian Highlands. The present study has demonstrated the potential of soil moisture data in revealing the impact of SLM practices on improving water availability and storage in the Ethiopian Highlands. Analysis of water resources systems is never complete without accounting for the green water resource, which is the critical freshwater resource in rainfed systems. Hence, this gap needs to be filled through soil moisture data monitoring as part of the hydrometeorological data monitored in gauged catchments. This would remain critical for improving our understanding of the impact of anthropogenic activities on water resources availability.

Since recently, CN-based hydrologic simulation models, such as the Soil and Water Assessment Tool (SWAT), have been increasingly used in the Blue Nile Basin to support ecosystem restoration and water resources planning and management efforts. However, these studies were predominantly conducted based on model calibration involving the manipulation of model input parameters, often resulting in unrealistic input parameter values and unjustified hydrological mass balance results. In the present analysis, an attempt has been made to adapt the CN method to the Blue Nile Highlands by generating local values of λ and CN from observed rainfall-runoff data for varying land uses and management conditions. By doing so, the present analysis has contributed a substantial part to an improved application of CN-based simulation modeling in the Blue Nile Highlands. In the future, there is a need to rely on local values of λ and CN than the customary NRCS-generated values in applying CN-based

simulation models in the Blue Nile Highlands to reduce model output uncertainty and come up with policy-relevant simulation results.

7.2. Study Limitations

The present study has contributed significantly to understanding the impact of SLM practices on improving water security in the Blue Nile Highlands. However, the analyses were predominantly focused on event-based components of the terrestrial water cycle, such as infiltration and runoff dynamics and soil moisture dynamic and spatio-temporal variability. Hence, components of the terrestrial water cycle that gain importance in the long term, such as evapotranspiration and groundwater flow, have not been addressed in the present study, mainly because the present study was superimposed on catchments where long-term catchment-scale monitoring is lacking both before and after SLM interventions. Hence, we suggest that these components of the terrestrial water cycle should receive the attention of future research for a more complete understanding of the impact of SLM practices on improving water security in the study area and beyond.

Simulation modeling appears to be a more feasible approach for understanding the impact of SLM on evapotranspiration and groundwater flow components of the terrestrial water cycle. However, most hydrologic simulation studies in the Blue Nile Highlands have often resulted in uncertain results stemming from model input parameters that poorly represent local conditions. Hence, though the present study has not addressed these important components of the terrestrial water cycle, it has laid the basis for an improved and more reliable hydrologic simulation modeling in the future in the study area by generating data-driven values of most critical input parameters of CN-based models, namely the initial abstraction ratio (λ) and the runoff curve number (CN) for different land uses and management conditions.

8. REFERENCES

Abera, W., Tamene, L., Tibebe, D., Adimassu, Z., Kassa, H., Hailu, H., Mekonnen, K., Desta, G., Sommer, R., & Verchot, L. (2020). Characterizing and evaluating the impacts of national land restoration initiatives on ecosystem services in Ethiopia. *Land Degradation and Development*, *31*(1), 37–52. https://doi.org/10.1002/ldr.3424

Abon, C. C., David, C. P. C., & Pellejera, N. E. B. (2011). Reconstructing the Tropical Storm Ketsana flood event in Marikina River, Philippines. *Hydrology and Earth System Sciences*, *15*(4), 1283–1289. https://doi.org/10.5194/hess-15-1283-2011

Adimassu, Z., Mekonnen, K., Yirga, C., & Kessler, A. (2014). Effects of soil bunds on runoff , soil and nutrient losses , and crop yield in the Central Highlands of Ethiopia. *Land Degrad. Develop.*, *25*, 554–564. https://doi.org/10.1002/ldr.2182

Ahmed, Y., Al-Faraj, F., Scholz, M., & Soliman, A. (2019). Assessment of Upstream Human Intervention Coupled with Climate Change Impact for a Transboundary River Flow Regime: Nile River Basin. *Water Resources Management*, *33*(7), 2485–2500. https://doi.org/10.1007/s11269-019-02256-1

Akale, A. T., Dagnew, D. C., Belete, M. A., Tilahun, S. A., Mekuria, W., & Steenhuis, T. S. (2017). Impact of soil depth and topography on the effectiveness of conservation practices on discharge and soil loss in the Ethiopian highlands. *Land*, *6*(4), 1–17. https://doi.org/10.3390/land6040078

Akale, A. T., Dagnew, D. C., Moges, M. A., Tilahun, S. A., & Steenhuis, T. S. (2019). The effect of landscape interventions on groundwater flow and surface runoff in a watershed in the upper reaches of the Blue Nile. *Water (Switzerland)*, *11*(10). https://doi.org/10.3390/w11102188

Alemu, M. M. (2016). Sustainable Land Management. *Journal of Environmental Protection*, *07*(04), 502–506. https://doi.org/10.4236/jep.2016.74045

Almeida, A. C., Smethurst, P. J., Siggins, A., Cavalcante, R. B. L., & Borges, N.

(2016). Quantifying the effects of Eucalyptus plantations and management on water resources at plot and catchment scales. *Hydrological Processes*, *30*(25), 4687–4703. https://doi.org/10.1002/hyp.10992

Arnold, J. G., Srinisvan, R., Muttiah, R. ., & Williams, J. R. (1998). Large area hydrologic modeling and assessment, Part I: Model development. *Journal of the American Water Resources Association*, *34*(1), 73–89.

Asfaw, A., Simane, B., Hassen, A., & Bantider, A. (2018). Variability and time series trend analysis of rainfall and temperature in northcentral Ethiopia: A case study in Woleka sub-basin. *Weather and Climate Extremes*, *19*, 29–41. https://doi.org/10.1016/j.wace.2017.12.002

Ayalew, D. W., Asefa, T., Moges, M. A., & Leyew, S. M. (2022). Evaluating the potential impact of climate change on the hydrology of Ribb catchment , and Sileshie Mes fi n Leyew. *Journal of Water and Climate Change*, *13*(1), 190–205. https://doi.org/10.2166/wcc.2021.049

Bai, Z. G., Dent, D. L., Olsson, L., & Schaepman, M. E. (2008). Proxy global assessment of land degradation. In *Soil Use and Management* (Vol. 24, Issue 3, pp. 223–234). https://doi.org/10.1111/j.1475-2743.2008.00169.x

Baltas, E. A., Dervos, N. A., & Mimikou, M. A. (2007). Hydrology and Earth System Sciences Technical Note: Determination of the SCS initial abstraction ratio in an experimental watershed in Greece. In *Hydrol. Earth Syst. Sci* (Vol. 11). www.hydrol-earth-syst-sci.net/11/1825/2007/

Bekele, R. D., Mirzabaev, A., & Mekonnen, D. (2021). Adoption of multiple sustainable land management practices among irrigator rural farm households of Ethiopia. *Land Degradation and Development*, *32*(17), 5052–5068. https://doi.org/10.1002/ldr.4091

Berhane, F., Zaitchik, B., & Dezfuli, A. (2014). Subseasonal analysis of precipitation variability in the Blue Nile River Basin. *Journal of Climate*, *27*(1), 325–344. https://doi.org/10.1175/JCLI-D-13-00094.1

Berhanu, B., Melesse, A. M., & Seleshi, Y. (2013). GIS-based hydrological zones and soil geo-database of Ethiopia. *Catena*, *104*, 21–31. https://doi.org/10.1016/j.catena.2012.12.007

Betrie, G. D., Mohamed, Y. A., Van Griensven, A., & Srinivasan, R. (2011). Sediment management modelling in the Blue Nile Basin using SWAT model. *Hydrology and Earth System Sciences*, *15*(3), 807–818. https://doi.org/10.5194/hess-15-807-2011

Bewket, W., & Sterk, G. (2005). Dynamics in land cover and its effect on stream flow in the Chemoga watershed, Blue Nile basin, Ethiopia. *Hydrological Processes*, *19*(2), 445–458. https://doi.org/10.1002/hyp.5542

Biazin, B., Sterk, G., Temesgen, M., Abdulkedir, A., & Stroosnijder, L. (2012). Rainwater harvesting and management in rainfed agricultural systems in sub-Saharan Africa - A review. *Physics and Chemistry of the Earth*, *47–48*, 139–151. https://doi.org/10.1016/j.pce.2011.08.015

Block, P., & Rajagopalan, B. (2007). Interannual variability and ensemble forecast of upper Blue Nile basin Kiremt season precipitation. *Journal of Hydrometeorology*, *8*(3), 327–343. https://doi.org/10.1175/JHM580.1

Boke, M. T., Moges, S. A., & Dejen, Z. A. (2022). Optimizing renewable-based energy supply options for power generation in Ethiopia. *PLoS ONE*, *17*(1 January 2022). https://doi.org/10.1371/journal.pone.0262595

Bonta, J. V., & Shipitalo, M. J. (2013). Curve numbers for long-term no-till corn and agricultural practices with high watershed infltration. *Journal of Soil and Water Conservation*, *68*(6), 487–500. https://doi.org/10.2489/jswc.68.6.487

Bonta, J. V. (2013). Managing landscape disturbances to increase watershed infiltration. *Transactions of the ASABE*, *56*(4), 1349–1359. www.unh.edu/unhsc

Bossio, D., Geheb, K., & Critchley, W. (2010). Managing water by managing land: Addressing land degradation to improve water productivity and rural livelihoods. *Agricultural Water Management*, *97*(4), 536–542.

https://doi.org/10.1016/j.agwat.2008.12.001

Brown, A. E., Zhang, L., McMahon, T. A., Western, A. W., & Vertessy, R. A. (2005). A review of paired catchment studies for determining changes in water yield resulting from alterations in vegetation. *Journal of Hydrology*, *310*(1–4), 28–61. https://doi.org/10.1016/j.jhydrol.2004.12.010

Caletka, M., Michalková, M. Š., Karásek, P., & Fučík, P. (2020). Improvement of SCS-CN initial abstraction coefficient in the Czech Republic: A study of five catchments. *Water (Switzerland)*, *12*(7). https://doi.org/10.3390/w12071964

Cantón, Y., Solé-Benet, A., & Domingo, F. (2004). Temporal and spatial patterns of soil moisture in semiarid badlands of SE Spain. *Journal of Hydrology*, *285*(1–4), 199–214. https://doi.org/10.1016/j.jhydrol.2003.08.018

Chen, H., Zhang, W., Wang, K., & Fu, W. (2010). Soil moisture dynamics under different land uses on karst hillslope in northwest Guangxi, China. *Environmental Earth Sciences*, *61*(6), 1105–1111. https://doi.org/10.1007/s12665-009-0428-3

Conway. (1997). Un modèle de bilan pour le bassin du haut Nil Bleu en Ethiopie. *Hydrological Sciences Journal*, *42*(2), 265–286. https://doi.org/10.1080/02626669709492024

Conway, D. (2000). The climate and hydrology of the Upper Blue Nile River. *The Geographical Journal*, *166*(1), 49–62.

Conway, D., & Schipper, E. L. F. (2011). Adaptation to climate change in Africa: Challenges and opportunities identified from Ethiopia. *Global Environmental Change*, *21*(1), 227–237. https://doi.org/10.1016/j.gloenvcha.2010.07.013

Cook, C., & Bakker, K. (2012). Water security: Debating an emerging paradigm. *Global Environmental Change*, *22*(1), 94–102. https://doi.org/10.1016/j.gloenvcha.2011.10.011

Cooper, P. J. M., Dimes, J., Rao, K. P. C., Shapiro, B., Shiferaw, B., & Twomlow, S.

(2008). Coping better with current climatic variability in the rain-fed farming systems of sub-Saharan Africa: An essential first step in adapting to future climate change? *Agriculture, Ecosystems and Environment*, *126*(1–2), 24–35. https://doi.org/10.1016/j.agee.2008.01.007

Dabney, S. M., Wilson, G. V., McGregor, K. C., & Vieira, D. A. N. (2012). Runoff through and upslope of contour switchgrass hedges. *Soil Science Society of America Journal*, *76*(1), 210–219. https://doi.org/10.2136/sssaj2011.0019

Damkjaer, S., & Taylor, R. (2017). The measurement of water scarcity: Defining a meaningful indicator. In *Ambio* (Vol. 46, Issue 5, pp. 513–531). Springer Netherlands. https://doi.org/10.1007/s13280-017-0912-z

Davis, K. F., Gephart, J. A., Emery, K. A., Leach, A. M., Galloway, J. N., & Odorico, P. D. (2016). Meeting future food demand with current agricultural resources. *Global Environmental Change*, *39*, 125–132. https://doi.org/10.1016/j.gloenvcha.2016.05.004

de Fraiture, C., Karlberg, L., & Rockström, J. (2009). Can Rainfed Agriculture Feed the World ? An Assessment of Potentials and Risk. In S.P. Wani et al. (Ed.), *Rainfed Agriculture: Unlocking the Potential* (pp. 124–132). CAB International.

de Fraiture, C., & Wichelns, D. (2010). Satisfying future water demands for agriculture. *Agricultural Water Management*, *97*, 502–511. https://doi.org/10.1016/j.agwat.2009.08.008

De Lannoy, G. J. M., Verhoest, N. E. C., Houser, P. R., Gish, T. J., & Van Meirvenne, M. (2006). Spatial and temporal characteristics of soil moisture in an intensively monitored agricultural field (OPE3). *Journal of Hydrology*, *331*(3–4), 719–730. https://doi.org/10.1016/j.jhydrol.2006.06.016

Delta-T Devices Ltd. (2016). *User Manual for the SDI-12 Profile Probe Type SDI PR2/4 and SDI PR2/6*. www.delta-t.co.uk

Descheemaeker, K., Nyssen, J., Poesen, J., Raes, D., Haile, M., Muys, B., & Deckers, S. (2006). Runoff on slopes with restoring vegetation: A case study from the

Tigray highlands, Ethiopia. *Journal of Hydrology*, *331*(1–2), 219–241. https://doi.org/10.1016/j.jhydrol.2006.05.015

Descheemaeker, K., Poesen, J., Borselli, L., Nyssen, J., Raes, D., Haile, M., Muys, B., & Deckers, J. (2008). Runoff curve numbers for steep hillslopes with natural vegetation in semi-arid tropical highlands, northern Ethiopia. *Hydrological Processes*, *22*(20), 4097–4105. https://doi.org/10.1002/hyp.7011

Descheemaeker, K., Raes, D., Nyssen, J., Poesen, J., Haile, M., & Deckers, J. (2009). Changes in water flows and water productivity upon vegetation regeneration on degraded hillslopes in northern Ethiopia: A water balance modelling exercise. *Rangeland Journal*, *31*(2), 237–249. https://doi.org/10.1071/RJ09010

Dile, Y. T., Berndtsson, R., & Setegn, S. G. (2013). Hydrological Response to Climate Change for Gilgel Abay River, in the Lake Tana Basin - Upper Blue Nile Basin of Ethiopia. *PLoS ONE*, *8*(10). https://doi.org/10.1371/journal.pone.0079296

Dile, Y. T., Karlberg, L., Srinivasan, R., & Rockström, J. (2016). Investigation of the Curve Number Method For Surface Runoff Estimation In Tropical Regions. *Journal of the American Water Resources Association*, *52*(5), 1155–1169. https://doi.org/10.1111/1752-1688.12446

Dile, Y. T., Tekleab, S., Kaba, E. A., Gebrehiwot, S. G., Worqlul, A. W., Bayabil, H. K., Yimam, Y. T., Tilahun, S. A., Daggupati, P., Karlberg, L., & Srinivasan, R. (2018). Advances in water resources research in the Upper Blue Nile basin and the way forward: A review. *Journal of Hydrology*, *560*, 407–423. https://doi.org/10.1016/j.jhydrol.2018.03.042

Dingman, S. L. (2015). *PHYSICAL HYDROLOGY* (Third Edit). Waveland Press, Inc.

Dung, B. X., Gomi, T., Miyata, S., Sidle, R. C., Kosugi, K., & Onda, Y. (2012). Runoff responses to forest thinning at plot and catchment scales in a headwater catchment draining Japanese cypress forest. *Journal of Hydrology*, *444–445*, 51–62. https://doi.org/10.1016/j.jhydrol.2012.03.040

Elhakeem, M., & Papanicolaou, A. N. (2009). Estimation of the runoff curve number via direct rainfall simulator measurements in the state of Iowa, USA. *Water Resources Management*, *23*(12), 2455–2473. https://doi.org/10.1007/s11269-008-9390-1

Elshamy, M. E., Seierstad, I. A., & Sorteberg, A. (2009). Hydrology and Earth System Sciences Impacts of climate change on Blue Nile flows using bias-corrected GCM scenarios. *Hydrol. Earth Syst. Sci*, *13*, 551–565. www.hydrol-earth-syst-sci.net/13/551/2009/

Eshete, D. G., Sinshaw, B. G., & Legese, K. G. (2020). Critical review on improving irrigation water use efficiency: Advances, challenges, and opportunities in the Ethiopia context. *Water-Energy Nexus*, *3*, 143–154. https://doi.org/10.1016/j.wen.2020.09.001

Etsay, H., Negash, T., & Aregay, M. (2019). Factors that influence the implementation of sustainable land management practices by rural households in Tigrai region, Ethiopia. *Ecological Processes*, *8*(1). https://doi.org/10.1186/s13717-019-0166-8

Ewunetu, A., Simane, B., Teferi, E., & Zaitchik, B. F. (2021). Relationships and the determinants of sustainable land management technologies in north gojjam sub-basin, upper blue nile, Ethiopia. *Sustainability (Switzerland)*, *13*(11). https://doi.org/10.3390/su13116365

Falkenmark, M. (2013). Growing water scarcity in agriculture: Future challenge to global water security. *Philosophical Transactions of the Royal Society A: Mathematical, Physical and Engineering Sciences*, *371*(2002). https://doi.org/10.1098/rsta.2012.0410

FAO. (2012). *State of the World ' s Forests* (Tenth Edit). FAO.

Faramarzi, M., Abbaspour, K. C., Ashraf Vaghefi, S., Farzaneh, M. R., Zehnder, A. J. B., Srinivasan, R., & Yang, H. (2013). Modeling impacts of climate change on freshwater availability in Africa. *Journal of Hydrology*, *480*, 85–101. https://doi.org/10.1016/j.jhydrol.2012.12.016

Feyereisen, G. W., Strickland, T. C., Bosch, D. D., Truman, C. C., Sheridan, J. M., & Potter, T. L. (2008). Curve number estimates for conventional and conservation tillages in the southeastern Coastal Plain. *Journal of Soil and Water Conservation*, *63*(3), 120–128.

Folton, N., Andréassian, V., & Duperray, R. (2015). Mise en perspective de la méthode des bassins appariés et de la modélisation pluie–débit sur l'impact hydrologique d'un incendie de forêt. *Hydrological Sciences Journal*, *60*(7–8), 1213–1224. https://doi.org/10.1080/02626667.2015.1035274

Fu, B., Wang, J., Chen, L., & Qiu, Y. (2003). The effects of land use on soil moisture variation in the Danangou catchment of the Loess Plateau, China. *Catena*, *54*(1–2), 197–213. https://doi.org/10.1016/S0341-8162(03)00065-1

Gain, A. K., Giupponi, C., & Wada, Y. (2016). Measuring global water security towards sustainable development goals. *Environmental Research Letters*, *11*(12). https://doi.org/10.1088/1748-9326/11/12/124015

Gani, N. D., Abdelsalam, M. G., Gera, S., & Gani, M. R. (2009). Stratigraphic and structural evolution of the Blue Nile Basin, Northwestern Ethiopian Plateau. *Geological Journal*, *44*(1), 30–56. https://doi.org/10.1002/gj.1127

Gebreegziabher, T., Nyssen, J., Govaerts, B., Getnet, F., Behailu, M., Haile, M., & Deckers, J. (2009). Contour furrows for in situ soil and water conservation , Tigray , Northern Ethiopia. *Soil & Tillage Research*, *103*, 257–264. https://doi.org/10.1016/j.still.2008.05.021

Gebrehiwot, S. G., Ilstedt, U., Gärdenas, A. I., & Bishop, K. (2011). Hydrological characterization of watersheds in the Blue Nile Basin, Ethiopia. *Hydrology and Earth System Sciences*, *15*(1), 11–20. https://doi.org/10.5194/hess-15-11-2011

Gebul, M. A. (2021). Trend, status, and challenges of irrigation development in Ethiopia—A review. In *Sustainability (Switzerland)* (Vol. 13, Issue 10). MDPI AG. https://doi.org/10.3390/su13105646

Gelete, G., Gokcekus, H., & Gichamo, T. (2020). Impact of climate change on the

hydrology of Blue Nile basin , Ethiopia : a review Gebre Gelete , Huseyin Gokcekus and Tagesse Gichamo. *Journal of Water and Climate Change*, *11*(4), 1539–1550. https://doi.org/10.2166/wcc.2019.014

Gerlak, A. K., & Mukhtarov, F. (2015). 'Ways of knowing' water: integrated water resources management and water security as complementary discourses. *International Environmental Agreements: Politics, Law and Economics*, *15*(3), 257–272. https://doi.org/10.1007/s10784-015-9278-5

Gosling, S., & Arnell, N. (2013). A global assessment of the impact of climate change on water scarcity. *Climatic Change*, *May 2014*. https://doi.org/10.1007/s10584-013-0853-x

Grey, D., & Sadoff, C. W. (2007). Sink or Swim? Water security for growth and development. *Water Policy*, *9*(6), 545–571. https://doi.org/10.2166/wp.2007.021

Grey, Garrick, D., Blackmore, D., Kelman, J., Muller, M., & Sadoff, C. (2013). Water security in one blue planet: Twenty-first century policy challenges for science. *Philosophical Transactions of the Royal Society A: Mathematical, Physical and Engineering Sciences*, *371*(2002). https://doi.org/10.1098/rsta.2012.0406

Gumma, M. K., Desta, G., Amede, T., Panjala, P., Smith, A. P., Kassawmar, T., Tummala, K., Zeleke, G., & Whitbread, A. M. (2021). Assessing the impacts of watershed interventions using ground data and remote sensing: a case study in Ethiopia. *International Journal of Environmental Science and Technology*. https://doi.org/10.1007/s13762-021-03192-7

Hailemariam, K. (1999). Impact of climate change on the water resources of Awash River Basin , Ethiopia. *Clim Res*, *12*, 91–96.

Hailu, A. D. (2022). Ethiopia hydropower development and Nile basin hydro politics. *AIMS Energy*, *10*(1), 87–101. https://doi.org/10.3934/ENERGY.2022006

Hajabbasi, M. A., Jalalian, A., & Karimzadeh, H. R. (1997). Deforestation effects on soil physical and chemical properties, lordegan, iran. *Plant and Soil*, *190*(2), 301–308. https://doi.org/10.1023/A:1004243702208

Hanasaki, N., Fujimori, S., Yamamoto, T., Yoshikawa, S., Masaki, Y., Hijioka, Y., Kainuma, M., Kanamori, Y., Masui, T., Takahashi, K., & Kanae, S. (2013). A global water scarcity assessment under Shared Socio-economic Pathways - Part 1: Water use. *Hydrology and Earth System Sciences*, *17*(7), 2375–2391. https://doi.org/10.5194/hess-17-2375-2013

Haregeweyn, N., Berhe, A., Tsunekawa, A., Tsubo, M., & Meshesha, D. T. (2012). Integrated watershed management as an effective approach to curb land degradation: A case study of the enabered watershed in northern Ethiopia. *Environmental Management*, *50*(6), 1219–1233. https://doi.org/10.1007/s00267-012-9952-0

Haregeweyn, N., Tsunekawa, A., Poesen, J., Tsubo, M., Meshesha, D. T., Fenta, A. A., Nyssen, J., & Adgo, E. (2017). Comprehensive assessment of soil erosion risk for better land use planning in river basins: Case study of the Upper Blue Nile River. *Science of the Total Environment*, *574*, 95–108. https://doi.org/10.1016/j.scitotenv.2016.09.019

Hawkins, R. (1993). Asymptotic determination of runoff curve numbers from data. *Journal of Irrigation and Drainage Engineering*, *119*, 334–345.

Hawkins, Ward, T. J., Woodward, Donald, E., & Van Mullem, Joseph, A. (2009). *Curve number hydrology : state of the practice*. American Society of Civil Engineers.

Hébrard, O., Voltz, M., Andrieux, P., & Moussa, R. (2006). Spatio-temporal distribution of soil surface moisture in a heterogeneously farmed Mediterranean catchment. *Journal of Hydrology*, *329*(1–2), 110–121. https://doi.org/10.1016/j.jhydrol.2006.02.012

Hejazi, M., Edmonds, J., Chaturvedi, V., Davies, E., & Eom, J. (2013). Scenarios of global municipal water-use demand projections over the 21st century. *Hydrological Sciences Journal*, *58*(3), 519–538. https://doi.org/10.1080/02626667.2013.772301

Heyi, E. A., Dinka, M. O., & Momo, G. (2022). Assessing the impact of climate

change on water resources of upper Awash River sub-basin , Ethiopia. *Journal of Water and Land Development*, *52*, 232–244. https://doi.org/10.24425/jwld.2022.140394

Hoekstra, A. Y. (2019). Green-blue water accounting in a soil water balance. *Advances in Water Resources*, *129*, 112–117. https://doi.org/10.1016/j.advwatres.2019.05.012

Hoekstra, A. Y., & Mekonnen, M. M. (2012). The water footprint of humanity. *PNAS*, *109*(9), 3232–3237. https://doi.org/10.1073/pnas.1109936109/-/DCSupplemental

Hsiao, T. C., Steduto, P., & Fereres, E. (2007). A systematic and quantitative approach to improve water use efficiency in agriculture. *Irrigation Science*, *25*(3), 209–231. https://doi.org/10.1007/s00271-007-0063-2

Huang, M., Gallichand, J., Dong, C., Wang, Z., & Shao, M. (2007). Use of soil moisture data and curve number method for estimating runoff in the Loess Plateau of China. *Hydrological Processes*, *21*(11), 1471–1481. https://doi.org/10.1002/hyp.6312

Hurni, H., Abate, S., Bantider, A., Debele, B., Ludi, E., Portner, B., Yitafer, B., & Zeleke, G. (2010). Land degradation and sustainable land management in the Highlands of Ethiopia. In H. Hurni & U. Wiesmann (Eds.), *Global Change and Sustainable Development: A Synthesis of Regional Experiences from Research Partnerships* (pp. 187–207). https://doi.org/10.13140/2.1.3976.5449

Hurni, H., Tato, K., & Zeleke, G. (2005). The implications of changes in population, land use, and land management for surface runoff in the Upper Nile Basin area of Ethiopia. *Mountain Research and Development*, *25*(2), 147–154. https://doi.org/10.1659/0276-4741(2005)025[0147:tiocip]2.0.co;2

Jain, M. K., Mishra, S. K., Babu, P. S., & Venugopal, K. (2006). On the Ia - S relation of the SCS-CN method. *Nordic Hydrology*, *37*(3), 261–275. https://doi.org/10.2166/nh.2006.011

Kirui, O. K., Mirzabaev, A., & von Braun, J. (2021). Assessment of land degradation

‘on the ground’ and from ‘above.’ *SN Applied Sciences*, *3*(3). https://doi.org/10.1007/s42452-021-04314-z

Korres, W., Reichenau, T. G., Fiener, P., Koyama, C. N., Bogena, H. R., Cornelissen, T., Baatz, R., Herbst, M., Diekkrüger, B., Vereecken, H., & Schneider, K. (2015). Spatio-temporal soil moisture patterns - A meta-analysis using plot to catchment scale data. *Journal of Hydrology*, *520*, 326–341. https://doi.org/10.1016/j.jhydrol.2014.11.042

Krajewski, A., Sikorska-Senoner, A. E., Hejduk, A., & Hejduk, L. (2020). Variability of the initial abstraction ratio in an urban and an agroforested catchment. *Water (Switzerland)*, *12*(2). https://doi.org/10.3390/w12020415

Kumar, C. P. (2018). Water Security-Challenges and Needs. *I N T E R N A T I O N A L E D U C A T I O N A L S C I E N T I F I C R E S E A R C H J O U R N A L*. https://www.researchgate.net/publication/322569213

Lebel, S., Fleskens, L., Forster, P. M., Jackson, L. S., & Lorenz, S. (2015). Evaluation of In Situ Rainwater Harvesting as an Adaptation Strategy to Climate Change for Maize Production in Rainfed Africa. *Water Resources Management*, *29*(13), 4803–4816. https://doi.org/10.1007/s11269-015-1091-y

Legates, D. R., & McCabe, G. J. (1999). Evaluating the use of “goodness-of-fit” measures in hydrologic and hydroclimatic model validation. *Water Resources Research*, *35*(1), 233–241. https://doi.org/10.1029/1998WR900018

Liu, X., Liu, W., Tang, Q., Liu, B., Wada, Y., & Yang, H. (2022). Global Agricultural Water Scarcity Assessment Incorporating Blue and Green Water Availability Under Future Climate Change Earth ’ s Future. *Earth’s Future*, 1–16. https://doi.org/10.1029/2021EF002567

McCartney, M. P., Demissie, S. S., & Girma, M. M. (2013). *Implications of climate change on existing and planned Water Resource Development in the Upper Blue Nile. Paper presented at Africa 2013:Water Storage and Hydropower Development for Africa Conference, Addis Ababa, Ethiopia, 16-18 April 2013. 8p.*

McIntyre, N., Ballard, C., Bruen, M., Bulygina, N., Buytaert, W., Cluckie, I., Dunn, S., Ehret, U., Ewen, J., Gelfan, A., Hess, T., Hughes, D., Jackson, B., Kjeldsen, T. R., Merz, R., Park, J. S., O'Connell, E., O'Donnell, G., Oudin, L., … Wheater, H. (2014). Modelling the hydrological impacts of rural land use change. *Hydrology Research*, *45*(6), 737–754. https://doi.org/10.2166/nh.2013.145

McNally, A., McCartney, S., Ruane, A. C., Mladenova, I. E., Whitcraft, A. K., Becker-Reshef, I., Bolten, J. D., Peters-Lidard, C. D., Rosenzweig, C., & Uz, S. S. (2019). Hydrologic and agricultural Earth observations and modeling for the water-food nexus. In *Frontiers in Environmental Science* (Vol. 7, Issue MAR). Frontiers Media S.A. https://doi.org/10.3389/fenvs.2019.00023

Mekonnen, M. M., & Hoekstra, A. Y. (2016). Sustainability: Four billion people facing severe water scarcity. *Science Advances*, *2*(2). https://doi.org/10.1126/sciadv.1500323

Mekuria, W., Langan, S., Johnston, R., Belay, B., Amare, D., Gashaw, T., Desta, G., Noble, A., & Wale, A. (2015). Restoring aboveground carbon and biodiversity: a case study from the Nile basin, Ethiopia. *Forest Science and Technology*, *11*(2), 86–96. https://doi.org/10.1080/21580103.2014.966862

Melaku, N. D., Renschler, C. S., Flagler, J., Bayu, W., & Klik, A. (2018). Integrated impact assessment of soil and water conservation structures on runoff and sediment yield through measurements and modeling in the Northern Ethiopian highlands. *Catena*, *169*, 140–150. https://doi.org/10.1016/j.catena.2018.05.035

Melesse, A. M., Abtew, W., & Setegn, S. G. (2013). Nile River Basin: Ecohydrological challenges, climate change and hydropolitics. In A. M. Melesse, W. Abtew, & S. G. Setegn (Eds.), *Nile River Basin: Ecohydrological Challenges, Climate Change and Hydropolitics* (Issue February, pp. 1–718). Springer International Publishing. https://doi.org/10.1007/978-3-319-02720-3

Mengistu, H. A., Demlie, M. B., & Abiye, T. A. (2019). Groundwater resource potential and status of groundwater resource development in Ethiopia.

Hydrogeology Journal, *27*, 1051–1065. https://doi.org/10.1007/s10040-019-01928-x

Mersha, B. D., Zeleke, G., Alamirew, T., Dejen, Z. A., & Gebrehiwot, S. G. (2022). Assessing the effect of sustainable land management on improving water security in the Blue Nile Highlands: a paired catchment approach. *Environmental Monitoring and Assessment*, *194*(3), 197. https://doi.org/10.1007/s10661-022-09837-5

Michel, C., Andréassian, V., & Perrin, C. (2005). Soil Conservation Service Curve Number method: How to mend a wrong soil moisture accounting procedure? *Water Resources Research*, *41*(2), 1–6. https://doi.org/10.1029/2004WR003191

Minella, J. P. G., Walling, D. E., & Merten, G. H. (2008). Combining sediment source tracing techniques with traditional monitoring to assess the impact of improved land management on catchment sediment yields. *Journal of Hydrology*, *348*(3–4), 546–563. https://doi.org/10.1016/j.jhydrol.2007.10.026

Mishra, Kumar, P., Saraswat, C., Chakraborty, S., & Gautam, A. (2021). Water security in a changing environment: Concept, challenges and solutions. *Water (Switzerland)*, *13*(4). https://doi.org/10.3390/w13040490

Mishra, S. K., Sahu, R. K., Eldho, T. I., & Jain, M. K. (2006). An improved Ia-S relation incorporating antecedent moisture in SCS-CN methodology. *Water Resources Management*, *20*(5), 643–660. https://doi.org/10.1007/s11269-005-9000-4

Montenegro, A. A. A., Lopes, I., De Carvalho, A. A., De Lima, J. L. M. P., De Souza, T. E. M. S., Araújo, H. L., Lins, F. A. C., Almeida, T. A. B., & Montenegro, H. G. L. A. (2019). Spatio Temporal Soil Moisture Dynamics and Runoff under Different Soil Cover Conditions in a Semiarid Representative Basin in Brazil. *Advances in Geosciences*, *48*, 19–30. https://doi.org/10.5194/adgeo-48-19-2019

Motulsky, H., & Christopoulos, A. (2003). Fitting curves with GraphPad Prism. In *Fitting Models to Biological Data Using Linear and Nonlinear Regression*.

Nawaz, R., Bellerby, T., Sayed, M., & Elshamy, M. (2010). Blue Nile Runoff Sensitivity to Climate Change. In *The Open Hydrology Journal* (Vol. 4).

Neary, D. G. (2016). Long-term forest paired catchment studies: What do they tell us that landscape-level monitoring does not? *Forests*, *7*(8), 1–15. https://doi.org/10.3390/f7080164

Niu, C. Y., Musa, A., & Liu, Y. (2015). Analysis of soil moisture condition under different land uses in the arid region of Horqin sandy land, northern China. *Solid Earth*, *6*(4), 1157–1167. https://doi.org/10.5194/se-6-1157-2015

Nyssen, J., Clymans, W., Descheemaeker, K., Poesen, J., Vandecasteele, I., Vanmaercke, M., Zenebe, A., Van Camp, M., Haile, M., Haregeweyn, N., Moeyersons, J., Martens, K., Gebreyohannes, T., Deckers, J., & Walraevens, K. (2010). Impact of soil and water conservation measures on catchment hydrological response-a case in north Ethiopia. *Hydrological Processes*, *24*(13), 1880–1895. https://doi.org/10.1002/hyp.7628

Nyssen, J., Poesen, J., & Deckers, J. (2009). Land degradation and soil and water conservation in tropical highlands. *Soil and Tillage Research*, *103*(2), 197–202. https://doi.org/10.1016/j.still.2008.08.002

Nyssen, J., Poesen, J., Moeyersons, J., Deckers, J., Haile, M., & Lang, A. (2004). Human impact on the environment in the Ethiopian and Eritrean highlands - A state of the art. *Earth-Science Reviews*, *64*(3–4), 273–320. https://doi.org/10.1016/S0012-8252(03)00078-3

Nyssen, J., Vandenreyken, H., Poesen, J., Moeyersons, J., Deckers, J., Haile, M., Salles, C., & Govers, G. (2005). Rainfall erosivity and variability in the Northern Ethiopian Highlands. *Journal of Hydrology*, *311*(1–4), 172–187. https://doi.org/10.1016/j.jhydrol.2004.12.016

Oliveira, P. T. S., Nearing, M. A., Hawkins, R. H., Stone, J. J., Rodrigues, D. B. B., Panachuki, E., & Wendland, E. (2016). Curve number estimation from Brazilian Cerrado rainfall and runoff data. *Journal of Soil and Water Conservation*, *71*(5), 420–429. https://doi.org/10.2489/jswc.71.5.420

Pellet, C., & Hauck, C. (2017). Monitoring soil moisture from middle to high elevation in Switzerland: Set-up and first results from the SOMOMOUNT network. *Hydrology and Earth System Sciences*, *21*(6), 3199–3220. https://doi.org/10.5194/hess-21-3199-2017

Perkins, K. S., Nimmo, J. R., Medeiros, A. C., Szutu, D. J., & von Allmen, E. (2014). Assessing effects of native forest restoration on soil moisture dynamics and potential aquifer recharge, Auwahi, Maui. *Ecohydrology*, *7*(5), 1437–1451. https://doi.org/10.1002/eco.1469

Planning and Development Commission, F. D. R. of E. (2021). *Ten Years Development Plan (2021 - 2030): A Pathway to Prosperity*.

Ponce, V. M., & Hawkins, R. H. (1996). Runoff Curve Number: Has It Reached Maturity? *Journal of Hydrologic Engineering*, *1*(1), 11–19. https://doi.org/10.1061/(asce)1084-0699(1996)1:1(11)

Rientjes, T. H. M., Haile, A. T., Kebede, E., Mannaerts, C. M. M., Habib, E., & Steenhuis, T. S. (2011). Changes in land cover, rainfall and stream flow in Upper Gilgel Abbay catchment, Blue Nile basin - Ethiopia. *Hydrology and Earth System Sciences*, *15*(6), 1979–1989. https://doi.org/10.5194/hess-15-1979-2011

Rockström. (2003). Water for food and nature in drought-prone tropics: Vapour shift in rain-fed agriculture. In *Philosophical Transactions of the Royal Society B: Biological Sciences* (Vol. 358, Issue 1440, pp. 1997–2009). Royal Society. https://doi.org/10.1098/rstb.2003.1400

Rockström, J. (2001). Green water security for the food makers of tomorrow: windows of opportunity in drought-prone savannahs. *Water Sci and Technol*, *43*(4), 71–78. https://doi.org/https://doi.org/10.2166/wst.2001.0181

Rockstrom, J., Falkenmark, M., Karlberg, L., Hoff, H., Rost, S., & Gerten, D. (2009). Future water availability for global food production : The potential of green water for increasing resilience to global change. *WATER RESOURCES RESEARCH*, *45*, 1–16. https://doi.org/10.1029/2007WR006767

Rockström, J., Folke, C., Gordon, L., Hatibu, N., Jewitt, G., Penning de Vries, F., Rwehumbiza, F., Sally, H., Savenije, H., & Schulze, R. (2004). A watershed approach to upgrade rainfed agriculture in water scarce regions through Water System Innovations: An integrated research initiative on water for food and rural livelihoods in balance with ecosystem functions. *Physics and Chemistry of the Earth*, *29*(15-18 SPEC.ISS.), 1109–1118. https://doi.org/10.1016/j.pce.2004.09.016

Rockström, J., Karlberg, L., Wani, S. P., Barron, J., Hatibu, N., Oweis, T., Bruggeman, A., Farahani, J., & Qiang, Z. (2010). Managing water in rainfed agriculture-The need for a paradigm shift. *Agricultural Water Management*, *97*(4), 543–550. https://doi.org/10.1016/j.agwat.2009.09.009

Rockström, J., Lannerstad, M., & Falkenmark, M. (2007). Assessing the water challenge of a new green revolution in developing countries. *Proceedings of the National Academy of Sciences of the United States of America*, *104*(15), 6253–6260. https://doi.org/10.1073/pnas.0605739104

Rockström, Kaumbutho, P., Mwalley, J., Nzabi, A. W., Temesgen, M., Mawenya, L., Barron, J., Mutua, J., & Damgaard-Larsen, S. (2009). Conservation farming strategies in East and Southern Africa: Yields and rain water productivity from on-farm action research. *Soil and Tillage Research*, *103*, 23–32. https://doi.org/10.1016/j.still.2008.09.013

Rodrigues, D. B. B., Gupta, H. V., & Mendiondo, E. M. (2014). A blue/green water-based accounting framework for assessment of water security. *Water Resources Research*, *50*(9), 7187–7205. https://doi.org/10.1002/2013WR014274

Rost, S., Gerten, D., Bondeau, A., Lucht, W., Rohwer, J., & Schaphoff, S. (2008). Agricultural green and blue water consumption and its influence on the global water system. *Water Resources Research*, *44*(9). https://doi.org/10.1029/2007WR006331

Rost, S., Gerten, D., Hoff, H., Lucht, W., Falkenmark, M., & Rockström, J. (2009). Global potential to increase crop production through water management in

rainfed agriculture. *Environmental Research Letters*, *4*(4). https://doi.org/10.1088/1748-9326/4/4/044002

Schmidt, E., & Tadesse, F. (2019). The impact of sustainable land management on household crop production in the Blue Nile Basin , Ethiopia. *Land Degrad Dev.*, *30*(January 2018), 777–787. https://doi.org/10.1002/ldr.3266

Schmidt, E., & Zemadim, B. (2015). Expanding sustainable land management in Ethiopia: Scenarios for improved agricultural water management in the Blue Nile. *Agricultural Water Management*, *158*, 166–178. https://doi.org/10.1016/j.agwat.2015.05.001

Schuol, J., Abbaspour, K. C., Yang, H., Srinivasan, R., & Zehnder, A. J. B. (2008). Modeling blue and green water availability in Africa. *Water Resources Research*, *44*(7). https://doi.org/10.1029/2007WR006609

SCRP. (2000). *Area of Anjeni, Gojam, Ethiopia: Long-term Monitoring of the Agricultural Environment 1984-1994 Soil Erosion and Conservation Database 2000 Centre for Development and Environment in Association with The Ministry of Agriculture, Ethiopia*. http://www.cde.unibe.ch

Seleshi, Y., & Zanke, U. (2004). Recent changes in rainfall and rainy days in Ethiopia. *International Journal of Climatology*, *24*(8), 973–983. https://doi.org/10.1002/joc.1052

Setegn, S. G., Dargahi, B., Srinivasan, R., & Melesse, A. M. (2010). Modeling of sediment yield from anjeni-gauged watershed, Ethiopia using swat model1. *Journal of the American Water Resources Association*, *46*(3), 514–526. https://doi.org/10.1111/j.1752-1688.2010.00431.x

Setegn, S. G., Rayner, D., Melesse, A. M., Dargahi, B., & Srinivasan, R. (2011). Impact of climate change on the hydroclimatology of Lake Tana Basin, Ethiopia. *Water Resources Research*, *47*(4). https://doi.org/10.1029/2010WR009248

Sharpley, A. N., & Williams J R. (1990). *EPIC—Erosion/productivity impact calculator: 1. model documentation*.

Shen, Q., Gao, G., Fu, B., & Lü, Y. (2014). Soil water content variations and hydrological relations of the cropland-treebelt-desert land use pattern in an oasis-desert ecotone of the Heihe River Basin, China. *Catena*, *123*, 52–61. https://doi.org/10.1016/j.catena.2014.07.002

Shi, Z. H., Chen, L. D., Fang, N. F., Qin, D. F., & Cai, C. F. (2009). Research on the SCS-CN initial abstraction ratio using rainfall-runoff event analysis in the Three Gorges Area, China. *Catena*, *77*(1), 1–7. https://doi.org/10.1016/j.catena.2008.11.006

Shukla, P. R., Skea, J., Reisinger, A., Slade, R., Fradera, R., Pathak, M., Al, A., Malek, K., Renée Van Diemen, B., Hasija, A., Lisboa, G., Luz, S., Malley, J., Mccollum, D., & Some, S. (2022). *Climate Change 2022 Mitigation of Climate Change Working Group III Contribution to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change Summary for Policymakers Edited by*. www.ipcc.ch

Simane, B., Zaitchik, B. F., & Foltz, J. D. (2016). Agroecosystem specific climate vulnerability analysis: application of the livelihood vulnerability index to a tropical highland region. *Mitigation and Adaptation Strategies for Global Change*, *21*(1), 39–65. https://doi.org/10.1007/s11027-014-9568-1

Soulis, K. X., & Valiantzas, J. D. (2012). SCS-CN parameter determination using rainfall-runoff data in heterogeneous watersheds-the two-CN system approach. *Hydrology and Earth System Sciences*, *16*(3), 1001–1015. https://doi.org/10.5194/hess-16-1001-2012

Stednick, J. D. (1996). Monitoring the effects of timber water yield harvest on annual. In *Journal Journal of Hydrology* (Vol. 176).

Stewart, D., Canfield, E., & Hawkins, R. (2012). Curve Number Determination Methods and Uncertainty in Hydrologic Soil Groups from Semiarid Watershed Data. *Journal of Hydrologic Engineering*, *17*(11), 1180–1187. https://doi.org/10.1061/(asce)he.1943-5584.0000452

Tabari, H., Taye, M. T., & Willems, P. (2015). Statistical assessment of precipitation

trends in the upper Blue Nile River basin. *Stochastic Environmental Research and Risk Assessment*, *29*(7), 1751–1761. https://doi.org/10.1007/s00477-015-1046-0

Takele, G. S., Gebre, G. S., Gebremariam, A. G., & Engida, A. N. (2021). Hydrological modeling in the Upper Blue Nile basin using soil and water analysis tool (SWAT). *Modeling Earth Systems and Environment*. https://doi.org/10.1007/s40808-021-01085-9

Taye, M. T., Dyer, E., Hirpa, F. A., & Charles, K. (2018). Climate Change Impact on Water Resources in the Awash Basin, Ethiopia. *Water (Switzerland)*, *10*, 1–16. https://doi.org/10.3390/w10111560

Tedela, N. H., Mccutcheon, S. C., Asce, M., Todd, ;, Rasmussen, C., Hawkins, R. H., Asce, F., Swank, W. T., Campbell, J. L., Mary, ;, Adams, B., Jackson, ; C Rhett, & Tollner, E. W. (2012). Runoff Curve Numbers for 10 Small Forested Watersheds in the Mountains of the Eastern United States. *Journal of Hydrologic Engineering*. https://doi.org/10.1061/(ASCE)HE

USDA-NRCS. (2004). *Estimation of Direct Runoff from Storm Rainfall. Chapter 10, Part 630, Hydrology. National Engineering Handbook, Title 210*. Washington, DC: USDA.

van Beek, E., & Arriens, W. L. (2014). *Water Security: Putting the Concept into Practice*. www.gwptoolbox.org

Van Griensven, A., Ndomba, P., Yalew, S., & Kilonzo, F. (2012). Critical review of SWAT applications in the upper Nile basin countries. In *Hydrology and Earth System Sciences* (Vol. 16, Issue 9, pp. 3371–3381). https://doi.org/10.5194/hess-16-3371-2012

Van Loon, A. F., Rangecroft, S., Coxon, G., Naranjo, J. A. B., Van Ogtrop, F., & Van Lanen, H. A. J. (2019). Using paired catchments to quantify the human influence on hydrological droughts. *Hydrology and Earth System Sciences*, *23*(3), 1725–1739. https://doi.org/10.5194/hess-23-1725-2019

Venkatesh, B., Lakshman, N., Purandara, B. K., & Reddy, V. B. (2011). Analysis of observed soil moisture patterns under different land covers in Western Ghats, India. *Journal of Hydrology*, *397*(3–4), 281–294. https://doi.org/10.1016/j.jhydrol.2010.12.006

Verhoeven, H. (2013). The politics of African energy development: Ethiopia's hydro-agricultural state-building strategy and clashing paradigms of water security. *Philosophical Transactions of the Royal Society A: Mathematical, Physical and Engineering Sciences*, *371*(2002). https://doi.org/10.1098/rsta.2012.0411

Vörösmarty, C. J., Douglas, E. M., Green, P. A., & Revenga, C. (2005). Geospatial indicators of emerging water stress: An application to Africa. *Ambio*, *34*(3), 230–236. https://doi.org/10.1579/0044-7447-34.3.230

Wada, Y., Flörke, M., Hanasaki, N., Eisner, S., Fischer, G., Tramberend, S., Satoh, Y., Van Vliet, M. T. H., Yillia, P., Ringler, C., Burek, P., & Wiberg, D. (2016). Modeling global water use for the 21st century: The Water Futures and Solutions (WFaS) initiative and its approaches. *Geoscientific Model Development*, *9*(1), 175–222. https://doi.org/10.5194/gmd-9-175-2016

Wagesho, N., Goel, N. K., & Jain, M. K. (2013). Temporal and spatial variability of annual and seasonal rainfall over Ethiopia. *Hydrological Sciences Journal*, *58*(2), 354–373. https://doi.org/10.1080/02626667.2012.754543

Wassie, S. B. (2020). Natural resource degradation tendencies in Ethiopia: a review. *Environmental Systems Research*, *9*(1), 1–29. https://doi.org/10.1186/s40068-020-00194-1

Wheater, H. S. (2015). Water Security - Science and management challenges. *IAHS-AISH Proceedings and Reports*, *366*, 23–30. https://doi.org/10.5194/piahs-366-23-2015

Wheater, H. S., & Gober, P. (2015). Water security and the science agenda. *Water Resources Research*, *51*(7), 5406–5424. https://doi.org/10.1002/2015WR016892

Whitehead, P. G., & Robinson, M. (1993). Experimental basin studies-an

international and historical perspective of forest impacts. *Journal of Hydrology*, *145*(3–4), 217–230. https://doi.org/10.1016/0022-1694(93)90055-E

Woodward, D. E., Hawkins, R. H., Jiang, R., Hjelmfelt, A. T., Van Mullem, J. A., & Quan, Q. D. (2003). Runoff curve number method: Examination of the initial abstraction ratio. *World Water and Environmental Resources Congress*, 691–700. https://doi.org/10.1061/40685(2003)308

Worku, G., Teferi, E., Bantider, A., & Dile, Y. T. (2019). Observed changes in extremes of daily rainfall and temperature in Jemma Sub-Basin , Upper Blue Nile Basin , Ethiopia. *Theoretical and Applied Climatology*, *135*, 839–854. https://doi.org/10.1007/s00704-018-2412-x

World Bank. (2006). *Sustainable Land Management CHALLENGES, OPPORTUNITIES, AND TRADE-OFFS*.

Worqlul, A. W., Dile, Y. T., Ayana, E. K., Jeong, J., Adem, A. A., & Gerik, T. (2018). Impact of Climate Change on Streamflow Hydrology in Headwater Catchments of the Upper Blue Nile. *Water (Switzerland)*, *10*(120), 1–18. https://doi.org/10.3390/w10020120

Worqlul, A. W., Jeong, J., Dile, Y. T., Osorio, J., Schmitter, P., Gerik, T., Srinivasan, R., & Clark, N. (2017). Assessing potential land suitable for surface irrigation using groundwater in Ethiopia. *Applied Geography*, *85*, 1–13. https://doi.org/10.1016/j.apgeog.2017.05.010

Yimere, A., & Assefa, E. (2022). Current and Future Irrigation Water Requirement and Potential in the Abbay River Basin , Ethiopia. *Air, Soil and Water Research*, *15*, 1–15. https://doi.org/10.1177/11786221221097929

Young, R. A., Onstad, C. A., Bosch, D. D., & Anderson, W. P. (1994). *Agricultural non-point source pollution model. Version 4.03, AGNPS User's Guide.*.

Yuan, Y., Mitchell, J. K., Hirschi, M. C., & Cooke, R. A. C. (2001). Modified SCS curve number method for predicting subsurface drainage flow. *Transactions of the American Society of Agricultural Engineers*, *44*(6), 1673–1682.

https://doi.org/10.13031/2013.7028

Yurtseven, I., Serengil, Y., Gökbulak, F., Şengönül, K., Ozhan, S., Kılıç, U., Uygur, B., & Ozçelik, M. S. (2018). Results of a paired catchment analysis of forest thinning in Turkey in relation to forest management options. *Science of the Total Environment*, *618*, 785–792. https://doi.org/10.1016/j.scitotenv.2017.08.190

Zaitchik, B. F., Simane, B., Habib, S., Anderson, M. C., Ozdogan, M., & Foltz, J. D. (2012). Building climate resilience in the Blue Nile/Abay Highlands: A role for earth system sciences. *International Journal of Environmental Research and Public Health*, *9*(2), 435–461. https://doi.org/10.3390/ijerph9020435

Zelelew, D. G. (2017). Spatial mapping and testing the applicability of the curve number method for ungauged catchments in Northern Ethiopia. *International Soil and Water Conservation Research*, *5*(4), 293–301. https://doi.org/10.1016/j.iswcr.2017.06.003

LIST OF PUBLICATIONS

Peer-reviewed Articles

Mersha, B. D., Zeleke, G., Alamirew, T., Dejen, Z. A., & Gebrehiwot, S. G. (2022). Assessing the effect of sustainable land management on improving water security in the Blue Nile Highlands: a paired catchment approach. *Environmental Monitoring and Assessment*, *194*(3), 197. https://doi.org/10.1007/s10661-022-09837-5

Mersha, B. D., Zeleke, G., Alamirew, T., Dejen, Z. A., & Gebrehiwot, S. G. (2022). Soil moisture spatio-temporal variability under treated and untreated catchment conditions in a fragile tropical highland environment: implication for water security. *Arabian Journal of Geosciences*, *15*(8), 1–16. https://doi.org/10.1007/s12517-022-10016-6

Fikadu, T., Teferi, E., Dubale, B., Gusha, B., Mantel, S. K., Tanner, J., Palmer, C. G. T., Woldu, Z., Alamirew, T., & Zeleke, G. (2022). Implications of Watershed Management Practices on Water Availability Using Hydrus-1D Model in the Aba Gerima Watershed, Upper Blue Nile Basin, Ethiopia. Water 14, 3095. https://doi.org/10.3390/w14193095

Conference Papers

Mersha, B. D., Zeleke, G., Alamirew, T., Dejen, Z. A., & Gebrehiwot, S. G. (2022). Initial Abstraction Ratio and Runoff Curve Number Computations from Data for Different Land Uses and Management Conditions in the Blue Nile Highlands, Ethiopia. 2nd African Graduate Students Conference (AGSCII), 27 – 28 May 2022, Bahir Dar University, Bahir Dar, Ethiopia

Mersha, B. D., Zeleke, G., Alamirew, T., Dejen, Z. A., & Gebrehiwot, S. G. (2022). Observed Variability in the Initial Abstraction Ratio under Different Land-use and Management Conditions in the Blue Nile Highlands. 1st Abbay International Water Conference, 16 – 17 September 2022, Bahir Dar University, Bahir Dar, Ethiopia

In Preparation

Mersha, B. D., Zeleke, G., Alamirew, T., Dejen, Z. A., & Gebrehiwot, S. G. (2022). Initial Abstraction Ratio and Runoff Curve Number Computations from Data for Different Land Uses and Management Conditions in the Blue Nile Highlands, Ethiopia.

Mersha, B. D., Zeleke, G., Alamirew, T., Dejen, Z. A., & Gebrehiwot, S. G. (2022). Observed Variability in the Initial Abstraction Ratio under Different Land-use and Management Conditions in the Blue Nile Highlands.