Effectiveness of bio-physical and soil amendment land management practices in reducing soil loss

(生物物理的手法及び土壌改良剤を用いた土地管理が土壌侵食の削減に及ぼす効果)

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The United Graduate School of Agricultural Sciences

Tottori University, Japan

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A dissertation submitted to The United Graduate School of Agricultural Sciences, Tottori University in partial fulfillment of the requirements for the Degree of Doctor of Philosophy

in Global Arid Land Science

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List of Abbreviations and Acronyms

A	Annual average soil loss
a.m.	Ante meridiem
AAS	Atomic Absorption Spectroscopy
ADSWE	Amhara Design and Supervision Works Enterprise
ALRC	Arid Land Research Center
B	Biochar
BD	Bulk Density
C	Control
C	Cover and management practices
CEC	Cation exchange capacity
CL1	Gently sloping cropland
CL2	Steeply sloping cropland
cm	Centimeter
ºC	Degree Celsius
º	Degree
%	Percent
θ	Degrees
dBD[a]	Dry bulk density
DBL	Degraded bush land
dS	DeciSeimens

E	exclosure
E	Storm kinetic energy
e	Unit kinetic energy
EC	Electric conductivity
EC	Electrical Conductivity
EI_{30}	Rainfall erosivity index
F	*Fanya juu*
G	Gypsum
g	Gram
ha	hectare
HEMC	High energy moisture characteristics
IR	Infiltration rate
J	Joule
K	Soil erodibility
Kg	Killo gram
L	dimensionless factor for slope length
L	Lime, litter
LM	Land management
m	A dimensionless slope length exponent
M	Particle size parameter
m	meter
m.a.s.l	meter above sea level
MJ	Mega joule

mm	millimeter
N	Number of years
n	number of storms
Np	Number of observations with a rock fragment
Nt	Total number of observations
OC	Organic Carbon
OM	Organic matter
P	Supporting practice factor
p	Permeability
P20	20 kg ha^{-1} Polyacrylamide
P40	40 kg ha^{-1} Polyacrylamide
P60	60 kg ha^{-1} Polyacrylamide
PAM	Polyacrylamide
pH	power of Hydrogen
R	Rainfall erosivity
R^2	Coefficient of Correlation
Rc	Surface rock fragment cover
RUSLE	Revised Universal Soil loss Equation
S	Slope steepness factor
SB	Soil bund
SBG	Soil bund with grass
SD	Standard Deviation
SI	Structural index

SL	Soil Loss
SL_c	Soil Loss from control
SLe	Estimated soil loss
SLM	Sustainable Land Management
SL_s	Soil loss from plots with structural measures
T	trench
t	tone
TRO	Time to runoff
UBN	Upper Blue Nile
yr	year

Chapter 1: General introduction

1.1 Background of the study

Soil is a natural resource composed of organic and inorganic materials on the surface of the earth that provides the medium for plant growth, and hence plays a vital role in human life. Soil degradation and nutrient depletion have become severe threats to reducing agricultural productivity, especially in developing countries like Ethiopia (Kebede and Yamoah, 2009). Unless additional proper measures are implemented to restore soil productivity, further degradation of soil may completely ruin its productive potential for human purposes (Hurni, 1993).

Soil erosion by water is a major cause of land degradation globally (Borrelli *et al*., 2017). Soil erosion entails the removal of top soil, which is rich in organic matter and nutrients (Mekuria et al., 2007). The multitude adverse on- and off-site consequences of soil erosion are reduction in soil fertility and crop production (Haregeweyn et al., 2008; Adgo et al., 2013), loss of vital ecosystem services (Lal, 2014), siltation of reservoirs (Haregeweyn et al., 2006; Vanmaercke et al., 2011), etc. In Ethiopia, these environmental and socioeconomic consequences are further aggravated by human intervention, including deforestation, overgrazing, and poor farming practices (Fenta et al., 2016; Haregeweyn et al., 2017; Hurni et al., 2015). The problem is also largely related to lack of suitable land management practices, application of proper policies to mitigate soil erosion and lack of raising awareness among farmers together with increased population pressure (Haregeweyn et al., 2017; Hurni et al., 2015; Nyssen et al., 2014).

Land management practices are applied to control the detachment and transportation of soil particles and to maintain soil fertility. Despite the long-term prevalence of land degradation and its severe consequences, the Ethiopian government became aware of the problem and initiated a soil and water conservation program in the 1970s (Haregeweyn *et al*., 2015), following devastating famines in 1973–1974 (Gebremichael et al., 2005). Since then, efforts have been made to introduce improved conservation practices, such as the construction of soil and stone bunds, short trenches, cut-off drains, check dams, hillside terraces, and *fanya juu* (a type of terracing system; Swahili for "throw uphill") (Sultan *et al*., 2018; Fenta *et al*., 2016). Although these management strategies were deemed to contribute to reverting soil erosion, in most cases the performances of the program was below expectation (Brevik and Hartemink, 2010). As a result, a new approach began with implementation of the land management (LM) program in 2008 (Schmidt & Tadesse, 2017), targeting wider geographic regions. The LM approach has promoted various practices, such as the construction of physical structures (bench terraces, check dams, etc.), planting of different shrub or tree species and establishing area exclosures; as well as combinations of structural and vegetative measures, and reduction of household livestock numbers (Tefera & Sterk, 2010). Rigorous researches have been also conducted to enhance the efficiency of LM practices to encourage efforts for wider adoption of LM practices and to improve land productivity through integrating plant species selected for their higher economic value with physical LM practices (Adimassu *et al*., 2012) and promising results have been obtained. Even though different bio-physical practices were used to reduce soil erosion by water in Ethiopia, land management practices that condition the soil/improve the soil physico-chemical properties and reduce soil loss were lacking in the previous times. To overcome this limitation, land management planning should consider the use of soil amendments as alternative management practices to reduce soil erosion through improving soil properties and creating favorable environment to plants/crops to better protect the soil under a changing climate.

Nowadays, the uses of soil amendments, such as anionic polyacrylamide (PAM), have become worldwide land management measure to reducing soil erosion, especially during the period between tillage and crop establishment in crop lands (Smets et al., 2008) when soil is bare and highly vulnerable to erosion. When applied to the soil, PAM enhances soil aggregate stability, reduces soil particles dispersion and surface crusting, and maintains high infiltration rates, and reducing runoff, soil and nutrient losses across a range of soil types (Busscher et al., 2006). Effectiveness of PAM in reducing runoff and soil loss is dependent on many factors including soils properties such as soil types, clay content and mineralogy, level of organic matter and electrolyte (e.g. Ca^{2+}) in the soil solution (Teo et al., 2006; Lee et al., 2010; Mamedov et al., 2010). Hence, applying PAM in combination with sources of electrolyte (Ca^{2+}), such as gypsum or lime enhances its effectiveness.

In addition, studies have also indicated that application of PAM in combination with organic soil amendments, such as biochar, was effective in reducing runoff and soil loss. For example, a study by Lee et al. (2015) and Abrol et al. (2016) have proved that application of biochar alone or in combination with PAM was effective to reducing runoff and soil loss. Biochar is a charcoal created by heating biomass at a temperature greater than 250˚c in a low oxygen environment (Antal Gronli, 2003), and the process is called pyrolysis. Biochar has become a soil conditioner and fertilizer due to its multiple benefits in increasing soil pH, and water and nutrient retention (Uzoma et al., 2011; Mangrich et al., 2015),) and soil microbial biomass (Lehmann et al., 2011). These effects of soil amendments generally improve vegetation or crop cover which in turn promotes infiltration and resistance to soil scouring through stabilizing soil structure with roots and intercepting rainfall and runoff (Li et al., 1992a, b; Pan and Shangguan, 2006), thereby playing an important role in soil and water conservation.

1.2 Statement of the problem and justification

Soil erosion by water is a major cause of land degradation globally (e.g. Borrelli et al., 2017) and in the Upper Blue Nile (UBN) basin of Ethiopia specifically (Gebrehiwot et al., 2014; Haregeweyn et al., 2015, 2017), where soil erosion by water causes soil loss rates of 37–246 t ha^{-1} yr^{-1} (Bewket & Teferi, 2009; Adimassu et al., 2014; Amare et al., 2014; Ebabu et al., 2019), and two million hectares of land have become severely degraded (Jagger & Pender, 2003). Different land management (LM) practices, such as bund, trenches and fanya juu have been constructed in Ethiopia to control soil erosion and their effectiveness in terms of reducing runoff and soil loss have already been evaluated in some areas of the Ethiopian highlands (e.g., Herweg & Ludi, 1999; Taye et al., 2013; Adimassu et al., 2014; Amare et al., 2014). Most Previous studies in the UBN basin or elsewhere in Ethiopia evaluated the effectiveness of LM practices focusing on specific watersheds that constitute a single land use and agro-ecological environment. But, effectiveness of LM practices, such as soil bunds with or without grass, fanya juu, and trenches combined with exclosures, have not yet been evaluated across land uses and agro-ecology in the UBN basin of Ethiopia.

In addition to bio-physical LM practices, the use organic and inorganic soil amendments have been used worldwide to protect soil erosion. Applications of amendments to the soil increases soil structural stability and porosity, improve infiltration rate and water holding capacity of the soil thereby reduce runoff and soil erosion (Mangrich et al., 2015). In addition to improving soil physico-chemical attributes, some soil amendments could also add important crop nutrients to the soil, thus creating favorable environment for crop growth and increase crop biomass and yield (Sojka et al., 2007).

Poly acrylamide (PAM) is an organic synthetic soil amendment used to reducing soil loss through improving soil structure especially when the vegetation cover cannot be established in crop lands (Smets et al., 2008). However, factors like rate of application affect its efficiency. Thus, determining the effective rate of PAM for a given soil type and rainfall pattern is essential before wide scale application. Effectiveness of PAM can further be enhanced by applying it together with other amendments, such as gypsum, lime and biochar. However, a comprehensive study about the effectiveness of PAM alone or integrated with gypsum, lime and biochar on reducing soil loss and related RUSLE factor values has not been conducted in Ethiopia. Furthermore, effectiveness of bio-physical practices combined with soil amendments on reducing soil loss has not been evaluated in Ethiopia or elsewhere.

1.3 Objectives of the study

The main objective of this study was, therefore, to evaluate the effectiveness of bio-physical and soil amendment LM practices in reducing soil loss by integrating laboratory and field studies. The specific objectives were (1) to evaluate the effectiveness of bio-physical LM practices (soil bund, *fanya juu*, soil bund with grass, trench with exclosure and different crop types) in reducing soil loss through determining C- and P-factors of RUSLE model; (2) to determine the effective PAM rate that best reduces runoff and soil loss under simulated consecutive rainfall storms; (3) to investigate effectiveness of PAM alone or integrated with other soil amendments (gypsum, lime and biochar) in reducing runoff, soil loss and RUSLE C-factor at field runoff plot condition using natural rainfall; and (4) to use RUSLE model and synthesize soil loss prediction scenarios for the best performing LM practices [SBG for bio-physical practices and P+L for soil amendments] and compare their separate and combined effectiveness in reducing soil loss.

1.4 Description of the study area

The study was conducted both in laboratory and field runoff plots. The laboratory experiment was conducted to determine the effective PAM rate using rainfall simulator located inside the central of Arid Land Research Center (ALRC) of Tottori University, Japan (Figure 1. 1).

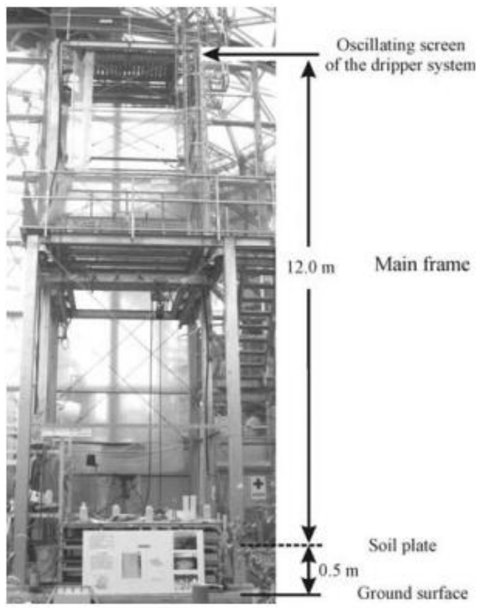

Figure 1. 1 Dripper-type rainfall simulator located inside the central of Arid Land Research Center dome, Tottori University, Japan (Abd Elbasit et al., 2010).

The field experiment was conducted using runoff plots established in three land use types (cropland, grazing land and degraded bush land use types) across three agro-ecologies in the UBN basin of Ethiopia. These sites were purposively selected to represent three different agro-ecologies in the UBN basin having different elevation, annual precipitation and land use types. The sites are Guder (highland), Aba Gerima (midland), and Dibatie (lowland). All the three study sites are located between 1487 and 2882 m above sea level within the tropical to humid tropical climatic zone (10°46′12″N–11°40′24″N, 36°15′51″E–37°29′49″E) (Figure 1. 2).

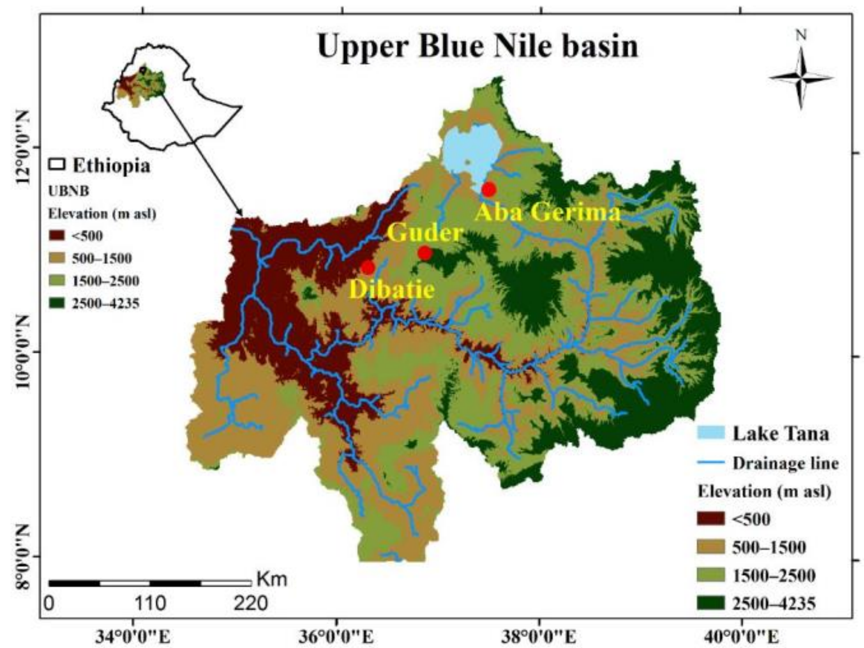

Figure 1. 2 Geographical location of the three study sites in the Upper Blue Nile basin, Ethiopia.

1.5 General methodological framework

The general methodological frame work of the thesis is shown in Figure 1.3. In this study effectiveness of bio-physical and soil amendments LM practices in reducing soil loss and related RUSLE model factors were evaluated. Chapter 2 describes about the effectiveness of the different bio-physical practices evaluated in cropland and non-crop land plots across three contrasting agro-ecologies in the Upper Blue Nile basin of Ethiopia through determining C- and P-factors of RUSLE, details are given in chapter 2. Chapter -3 deals with determination of the effective PAM rate for high intensity simulated consecutive rainfall storms that represent tropical climate condition using rainfall simulator at Arid Land Research Center (ALRC) of Tottori University, Japan, (chapter 3). This effective PAM rate integrated with other soil amendments was then applied to field runoff plots at Aba Gerima study site in the northwest Ethiopia, and its effectiveness in reducing runoff, soil loss and RUSLE's C-factor value was determined (Chapter 4). Then, soil loss scenarios for the best performing bio-physical and soil amendment LM practices were predicted using RUSLE model in field runoff plots of 30m length and 8% slope for the year of 2019 at Aba Gerima site. Finally, the separate and combined effectiveness of the practices in reducing soil loss was compared and conclusions and recommendations were drawn (chapter 5).

Methodological frame work

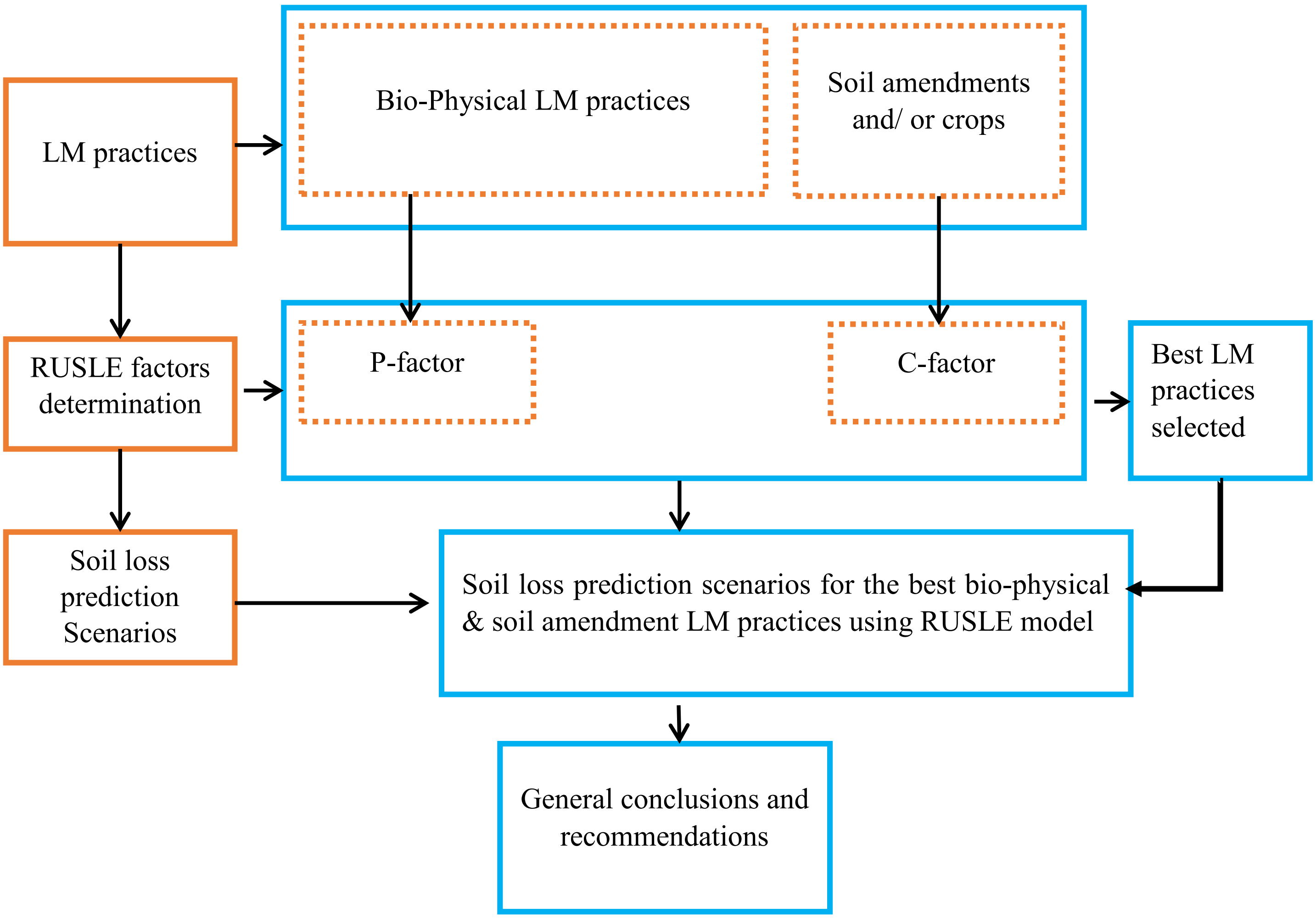

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Figure 1. 3 Methodological framework of the thesis

1.6 Organization of the thesis

The thesis is organized into five chapters (Figure 1.4). The first chapter (Chapter 1) is the general introductory section of the thesis and contains background, problem statement, objectives of the study and study area description. It explains the importance of soil, causes and consequences of soil erosion and degradation, management options to reduce soil erosion based on the available information from literature and field observation.
Chapter 2 deals with the evaluation of effectiveness of different bio-physical practices in reducing soil loss through determining the RUSLE model *C*-and *P*- factors for different land management practices across land uses and agro-ecologies in the UBN basin of Ethiopia.

Chapter 3 explains about laboratory experiment to determine effective PAM rate that best reduces runoff and soil loss under consecutive simulated rainfall storms for field application using runoff plots and natural rainfall.

Chapter 4 explains about investigation of effect of PAM integrated with other soil amendments on runoff, soil loss, selected soil attributes, crop biomass and RUSLE's C-factor.

Chapter 5 discusses about the separate and combined effects of the selected bio-physical and soil amendment practices, draws conclusions, implications and recommendations.

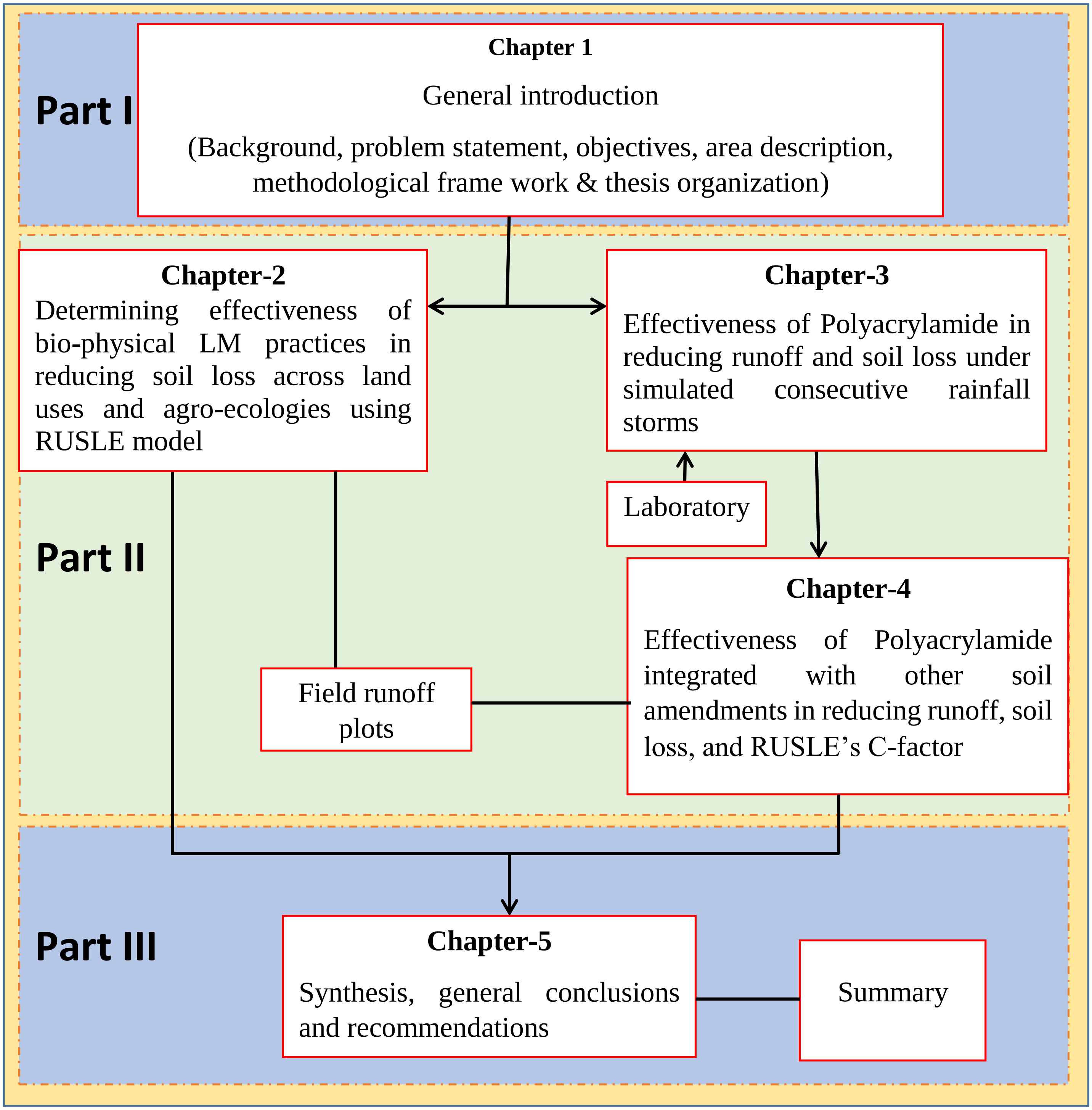

Figure 1. 4 Organization of the thesis

Chapter 2: Determining the effectiveness of bio-physical land management practices in reducing soil loss across land uses and agro-ecologies using RUSLE model:case studies from the Uper Blue Nile basin, Ethiopia

This chapter is published as:

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2. 1 Introduction

Soil erosion by water is a major cause of land degradation in the world in general (e.g. Borrelli et al., 2017) and in the Upper Blue Nile (UBN) basin of Ethiopia in particular (Gebrehiwot et al., 2014; Haregeweyn *et al*., 2015). In the UBN basin, soil erosion by water causes soil loss rates of 37–246 t ha^{-1} yr^{-1} (Bewket & Teferi, 2009; Adimassu *et al*., 2014; Amare *et al*., 2014; Ebabu *et al*., 2019) and two million hectares of land have become severely degraded (Jagger & Pender, 2003). Such high soil erosion rates and associated land degradation are attributable to factors such as increasing human and livestock populations (Hurni,1993; Sonneveld & Keyzer, 2003), and the resultant problems of overgrazing (Amsalu *et al*., 2007) and vegetation clearance from steep slopes and their subsequent cultivation (Hurni, 1993; Fenta *et al*., 2016; Berihun *et al*., 2019a,b). Furthermore, rainfall erosivity is strong in the highlands of Ethiopia, where large amounts of precipitation falling during high-intensity rainstorms exacerbate soil erosion (Nyssen *et al*., 2005; Meshesha *et al*., 2018; Fenta *et al*., 2017, 2020).

The effectiveness of LM practices in terms of reducing soil loss have already been evaluated in some areas of the Ethiopian highlands (e.g., Herweg & Ludi, 1999; Taye *et al*., 2013; Adimassu *et al*., 2014; Amare *et al*., 2014). Case studies conducted at various spatial scales (plot and watersheds) have shown that LM practices have reduced runoff and soil erosion (e.g., Haregeweyn *et al*., 2015; Admasu *et al*., 2017), thereby reversing the trend toward increased land degradation in Ethiopia (Amare *et al*., 2014). However, effectiveness varied greatly depending on the type of LM practice and the agro-ecology where it was implemented (Herweg & Ludi, 1999; Nyssen *et al*., 2009; Taye *et al*., 2013; Haregeweyn *et al*., 2017).

In this study, we adopted the Revised Universal Soil Loss Equation (RUSLE), with well-evaluated predictive power and low data requirements, to use for quantitative evaluation of effectiveness of different LM practices. The RUSLE model (A=KRLSCP) considers six factors that control the amount of erosion: soil erodibility (K), rainfall erosivity (R), slope length (L), slope steepness (S), cover and management practices (C), and support practices (P). From the six factors, *C*-and *P*-factors are used to quantify effect of different land management practices in reducing soil loss. The *C*-factor quantifies the effects of practices such as tillage and the use of cover crops and plant residues on soil loss, and its value varies seasonally as cover conditions change (Renard *et al*., 1991). The *C*-factor is usually estimated by comparing the weighted average ratio of soil loss in the experimental plot to that of a unit (control) plot, but it has been determined differently by different studies (Karpilo & Toy, 2004). The *P*-factor quantifies the effects of practices such as contour farming and the constructions of bunds, *fanya juu*, trenches and other physical LM structures on soil loss (Panagos *et al*., 2015); this factor is determined by calculating the ratio of soil loss due to runoff in a plot where a conservation practice has been implemented to soil loss from a unit plot (Renard *et al*., 1997). Factor *P* considers practices that reduce the erosion potential of runoff by modifying runoff concentration and velocity, drainage patterns, and the hydraulic force exerted by runoff on the soil surface (Renard *et al*., 1991). Previous studies in the UBN basin or elsewhere in Ethiopia determined *C*- and *P*-factor values focusing on specific watersheds that constitute a single land use and agro-ecological environment. For example, factor *C* values have been determined in land areas used for crops, including cereals, and grazing in different regions of Ethiopia (Hurni,1985; Nyssen *et al*., 2009; Taye *et al*., 2017); and factor *P* values have been determined for soil bund, soil bund with grass and *fanya juu* at different watersheds in the UBN basin (Amare *et al*., 2014; Herweg & Ludi, 1999), and for stone bunds with and without trenches in the dryland of northern Ethiopia (Nyssen *et al*., 2009; Gebremichael *et al*., 2005; Taye *et al*., 2017). However, values of these factors have not been determined for different land uses and LM structures across different agro-ecologies in the UBN basin of Ethiopia. Moreover, the UBN basin is characterized by a high spatial variability of rainfall (Rientjes *et al*., 2013; Fenta *et al*., 2014, 2018), which causes storm kinetic energy, and hence rainfall erosivity (factor *R*) to vary (Fenta *et al*., 2017) and greatly influences soil erosion rates (Haregeweyn *et al*., 2017). However, at present, no single storm kinetic energy determination model is recommended for calculating *R*-factor value for use in soil erosion modeling in the basin. Thus, selecting appropriate model for *R*-factor determination reduces uncertainties in the *C*-factor values.

The main purpose of this study was, therefore, to evaluate effectiveness of different LM practices through determining the *C*- and *P*- factor values of RUSLE for different land uses and LM practices by using measured data from experimental runoff plots in different agro-ecologies in the UBN basin of Ethiopia. The specific objectives of the study were to: (i) evaluate candidate models for determining storm kinetic energy and rainfall erosivity (R) factor; and to select the model most applicable to our study sites; (ii) determine support practices (*P*) factor values for different LM practices, including soil bunds, *fanya juu*, soil bunds with grass, and trenches with exclosure; and (iii) determine cover and management (*C*) factor values for croplands used to grow different crop types, grasslands, and degraded bushlands in three agro-ecologies in the UBN basin.

2.2 Materials and methods

2.2.1 Study area description

Experimental runoff plots for this study were established at three study sites in three agro-ecologies in the UBN basin of Ethiopia (Figure 2.1).

These sites were purposively selected to represent three different agro-ecologies in the UBN basin having different elevation, annual precipitation and land use types.The sites are Guder (highland), Aba Gerima (midland), and Dibatie (lowland). All the three study sites are located between 1487 and 2882 m above sea level within the tropical to humid tropical climatic zone (10°46′12″N–11°40′24″N, 36°15′51″E–37°29′49″E) (Figure 2.1).

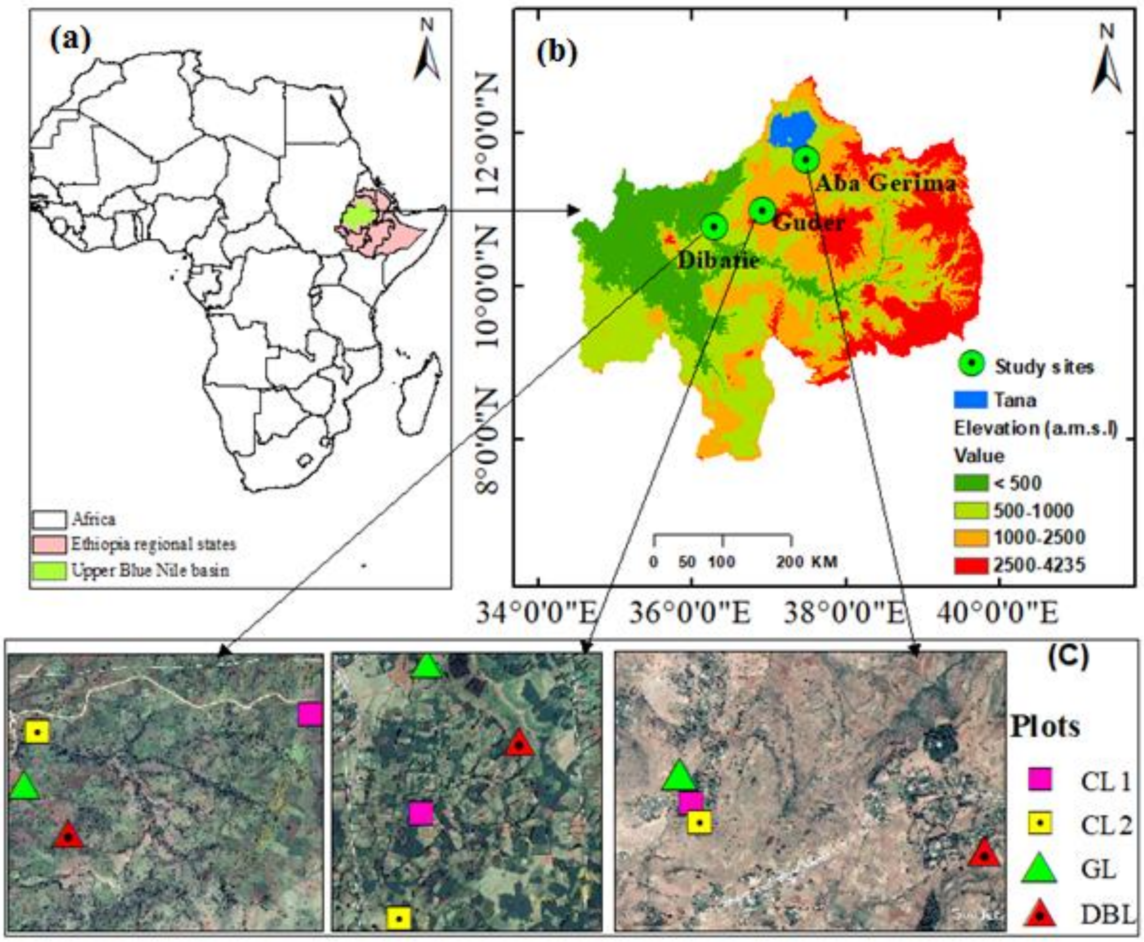

Figure 2. 1 Geographical location of the study sites: (a) location of Ethiopia in Africa, and Upper Blue Nile basin in Ethiopia; (b) study site locations in the Upper Blue Nile basin; (C) Locations of the runoff plots and their land use types; (CL1 = gently sloping cropland, CL2 = steeply sloping cropland, GL = steeply sloping grassland, DBL = steeply sloping degraded bush land).

Table 2. 1 Description of characteristics of the study sites

Study Site	Guder (Highland)	Aba Gerima (Midland)	Dibatie(Lowland)
Mean elevation (m a.s.l.)	2728	1998	1490
Mean annual rainfall (mm)	2495	1343	1022
Mean daily temp.(°C)	15–24	17–31	18–29
Soil types	Acrisols, Luvisols, Leptosols, Vertisols	Luvisols, Leptosols	Luvisols , Vertisols, Leptosols
Agro-ecology zone [a]	Moist subtropical (*Wet Dega*)	Humid subtropical (*Moist Weyna Dega*)	Tropical hot humid (*Moist Kolla*)
Dominant crops [b]	Barley, teff, wheat, potatoes	Teff, finger millet, wheat, maize	Chili pepper, teff, Finger millet, maize, groundnut
Rainfall threshold (mm) for crop land	6	11	10
Rainfall threshold (mm) for non-crop land	5	9	9

[a] Moist Kolla = 500 to 1500 m elevation and 900 to 1400 mm annual rainfall; Moist Weyna Dega = 1500 to 2300 m elevation and 900 to 1400 mm annual rainfall; Wet Dega = 2300 to 3200 m elevation and ≥1400 mm annual rainfall (Hurni *et al*., 2016; Nigussie *et al*., 2016; Sultan *et al*.,2018).

[b] Crops: Barley (*Hordeum vulgare*); Teff (*Eragrostis tef*); wheat (*Triticum aestivum*); potato (*Solanum tuberosum*); finger millet (*Eleusine coracana*); maize (*Zea mays*); and groundnut (*Arachis hypogaea*); (Sultan *et al*., 2018; Ebabu *et al*., 2019).

The four dominant soil types of the study sites are: (1) Acrisols, soils with the subsurface accumulation of low-activity clays (i.e., highly weathered), low cation exchange capacity, and low base saturation; (2) Luvisols, very deep and well-drained soils that form on gentle slopes; (3) Leptosols, thin, degraded soils on steep slopes; and (4) Vertisols, soils with a high content of clay minerals that shrink and swell during dry and wet seasons. All four types occur in Guder, whereas only three (Luvisols, Vertisols, and Leptosols) are present in Dibatie and two (Luvisols and Leptosols) in Aba Gerima (Ebabu *et al*., 2019).

The major land use types are croplands, grasslands, and degraded bushlands (Sultan *et al*., 2017, 2018). At all sites, there are a larger proportion of croplands than non-croplands, and the farming system is mixed crop–livestock, characterized by rain fed and continuous cropping. Location of study plots, mean annual rainfall, dominant soil types and other characteristics of the study sites are described in Figure 2.1 and Table 2.1.

2.2.2 Methods

(i) Experimental plots set-up

A total of 42 experimental runoff plots, each 6 m by 30 m, were established in three land use types: cropland, grassland, and degraded bushland. All plots were bounded by inserting a 35-cm-wide metallic sheet to a depth of 15 cm into the soil; thus, the sheet extended 20 cm above ground level to prevent the inflow of runoff into the plot and outflow from the plot. At each site, four cropland-1 plots (CL1; 5% slope), four cropland-2 (CL2; 15% slope) plots, three grasslands (15% slope) plots, and three degraded bushlands (35% slope) plots were established (Table 2.2).

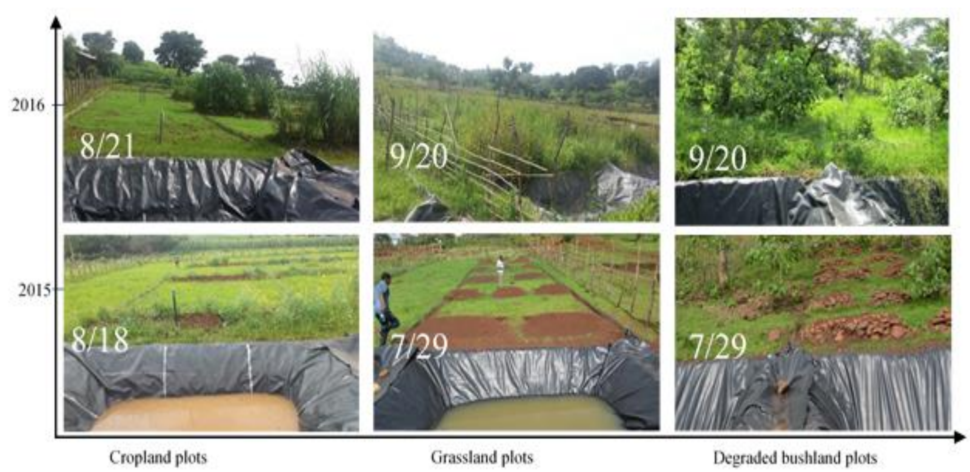

Figure 2. 2 Evolution of vegetation cover during the two monitoring years (2015 & 2016) in the study plots under different land management practices in cropland (left), grassland (middle) and degraded bushland (right).

Cropland LM treatments were control, *fanya juu*, soil bund, and soil bund reinforced with grass. The soil bunds were planted with Desho grass (*Pennisetum pedicellatum*) at Guder, with elephant grass (*Pennisetum purpureum*) at Aba Gerima, and with Vetiver grass (*Vetiveria zizanioides*) at Dibatie. Grassland and degraded bushland LM practices were control (i.e., free grazing was allowed), exclosure (E), and trench with exclosure (T+E). In grassland and degraded bushland free grazing was allowed for control plots, however, control plots in croplands were fenced. The temporal evolutions of plots in the two study years are shown in Figure 2.2.

Runoff collector trenches, trapezoidal in cross section (5 m long, 3 m wide, 1.5 m deep), were constructed at the lower end of each plot and lined with a plastic geo-membrane to prevent loss of water by infiltration. Ebabu *et al*. (2019) has described the plot set-up in detail and Sultan *et al*. (2018) has described the structural measures. The croplands were tilled manually with a hoe only once, at planting time, to disturb the soil as little as possible during the experiment. Crops sown in each site were selected by considering local practice, climatic conditions, and crop rotation as a soil management practice; at each site, all agronomic activities were similar to those practiced by the local farmers. In the first year (2015), barley (Figure 2.3a) was grown at Guder, finger millet (Figure 2.3b) at Aba Gerima, and chili peppers (Figure 2.3c) at Dibatie whereas in the second year (2016), teff was sown in all cropland plots (Figure 2.3d).

During the two study years, all cropland plots were fenced, including the control plots, to protect them from animal and human interference, and crop residues were left in the plots. Note that the cropland plot practices differed from the practices of the local farmers in some respects. However, our experimental results may provide additional information to support government efforts to reduce erosion and improve the productivity of the land, in addition that provided by studies using unfenced cropland runoff plots in other parts of Ethiopia.

Table 2. 2 Characteristics of the runoff plots and LM practices used in the three land use types at the three study sites

Study Sites	Land use	Slope (%)	LM practices	Soil texture class	dBD[a] (g/cm^{-3})	Total no. of LM practices per plot
Guder	CL1	5	C,SB,F,SBG	Clay loam	1.18	3
	CL2	15	C,SB,F,SBG	Clay loam	1.3	4
	GL	15	C,E,T+E	Loam	1.14	15
	DBL	35	C,E,T+E	Silty loam	1.11	20
Aba Gerima	CL1	5	C,SB,F,SBG	Clay	1.27	3
	CL2	15	C,SB,F,SBG	(Sandy) Loam	1.22	4
	GL	15	C,E,T+E	Clay loam	1.39	15
	DBL	35	C,E,T+E	Clay loam	1.43	20
Dibatie	CL1	5	C,SB,F,SBG	Heavy clay	1.11	3
	CL2	15	C,SB,F,SBG	Heavy clay	1.14	4
	GL	15	C,E,T+E	Heavy clay	1.24	15
	DBL	35	C,E,T+E	Clay loam	1.14	20

Note: land uses: C = control plot, CL1 = gently (5%) sloping cropland, CL2 = steeply (15%) sloping cropland, GL = grassland (15% slope), DBL = degraded bush land (35% slope), SB = soil bund, F = *fanya juu*, SBG = soil bund with grass, E = Exclosure, T+E = trench with exclosure.

dBD[a] = Dry bulk density

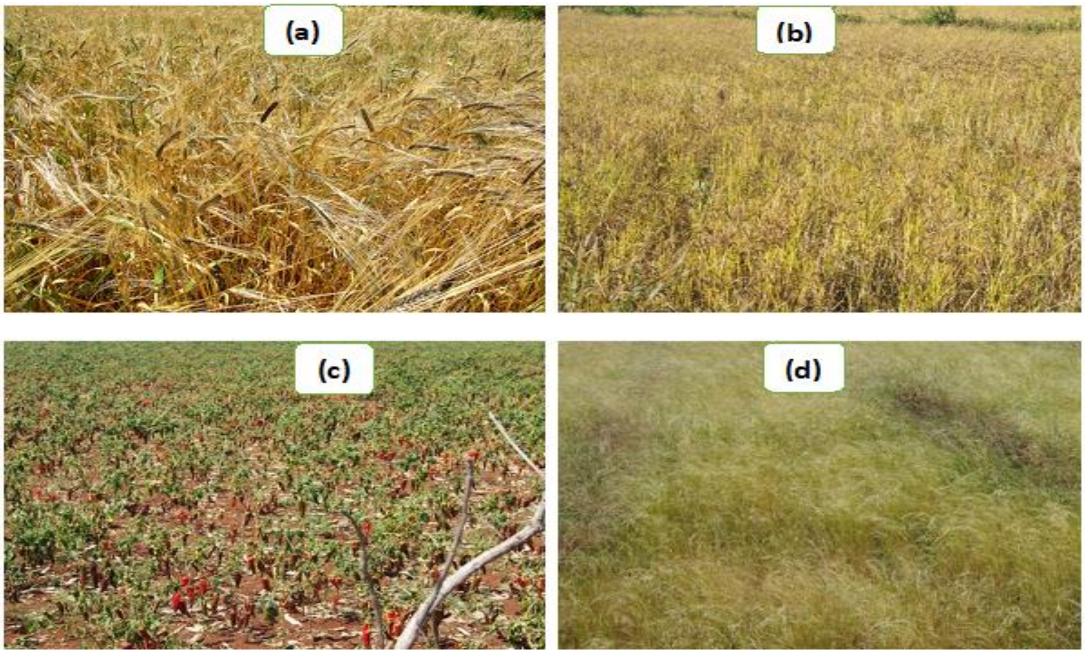

Figure 2. 3 Cover and management practices for C-factor determination for cropland plots in the three study sites: (a) Barley at Guder, (b) Finger millet at Aba Gerima, (c) Chili pepper at Dibatie, and (d) Tef at Dibatie. All plots were hoed once during sowing time with minimum pulverization of the soil as part of conservation tillage to increase infiltration, reduce runoff and erosion.

(ii) Collection of rainfall, runoff, and soil loss data

Rainfall data were collected during the two study years from the three sites by an automatic tipping-bucket (HOBO UA-003-64) rain gauge installed at each site. Runoff and soil loss data for rainfall events that generated runoff were collected on a daily basis, mainly during the rainy season. Relationship between the water level from total runoff in the collector trench (water depth) and total runoff volume was developed (Ebabu *et al*., 2019), and then this relationship was used to calculate the total daily runoff volume from the measured runoff water depth in the collector trench for each plot. Finally, the actual runoff volume was determined by subtracting the rainfall volume from the total runoff volume in the trench. During runoff data collection, the runoff water in the collector trench was vigorously agitated with floor brushes until the sediment that had settled was suspended in the water; then a 1-L depth-integrated runoff sample was collected for determination of the sediment concentration. Following these measurements, the plastic geo-membrane in each collector trench was emptied manually, carefully cleaned, and checked for any damage. The collected 1-L runoff samples were transported to the laboratory, filtered through Whatman 42 filter paper, and oven-dried at 105°C for 24 h; the sediment concentration (g L^{-1}) was then determined by weighing the sediment + filter together on a digital balance and subtracting the weight of the filter. Daily soil loss (t ha^{-1}) was calculated as the product of the sediment concentration and the actual daily runoff volume, and seasonal soil loss was calculated by summing the daily soil loss data.

(iii) Revised Universal Soil Loss Equation factors

The equation for RUSLE is given as:

$$A = RKLSCP \tag{1}$$

where

A is the long-term average annual soil loss rate (t ha^{-1} yr^{-1}), R is the rainfall erosivity factor (MJ mm ha^{-1} h^{-1} yr^{-1}), K is soil erodibility (t h MJ^{-1} mm^{-1}), L is a slope length factor (dimensionless), S is a slope steepness factor (dimensionless), C is a cover and management practices factor (dimensionless), and P is a supporting practice factor (dimensionless).

Rainfall erosivity (*R*) model selection: R is one of the most important input factors of RUSLE. To determine this factor, rainfall kinetic energy (E) must be known. Although it is difficult to measure E directly and sophisticated instruments are required, E can be computed from rainfall intensity data. Wischmeier and Smith (1978) developed the rainfall erosivity index (EI_{30}), which they defined as the product of E and the maximum continuous 30-min intensity (I_{30}) during a storm event. Following Wischmeier and Smith (1978), many models have been also developed to determine E from rainfall intensities for areas where automatic recording rain gauges are available. In this study, we compared seven candidate models for determining rainfall kinetic energy/R-factor that have been developed or tested in Ethiopia: (1) Hurni1985 (Hurni, 1985), which was developed for the highlands of Ethiopia; (2) WS1978 (Wischmeier and Smith, 1978), which Krauer (1988) used to construct an iso-erodent map of Ethiopia; (3) Renard1997 (Renard *et al*., 1997), which is widely used to determine R-factor of the RUSLE; (4) Nyssen2005 (Nyssen *et al*., 2005), which was developed in the northern highlands of Ethiopia; (5) Meshesha2014 (Meshesha *et al*., 2014), which was developed in the Rift Valley of Ethiopia; (6) Meshesha2018 (Meshesha *et al*., 2018), developed in the northwestern highlands of Ethiopia; (7); and Van Dijk2002 (Van Dijk *et al*., 2002), which was developed using global data from research results in different parts of the world. All these models, except Hurni1985, use rainfall intensity data to calculate rainfall kinetic energy.

To select the best model for determination of R, first R values were obtained from the rainfall kinetic energy calculated by each model as described below, and then averaged over the two study years. The average annual soil loss (A) was estimated from steeply sloping control cropland plots at the three study sites (i.e., P = 1) by multiplying the averaged R value obtained by each model with the other five RUSLE factors (K, L, S, C, P), using K, L, and S values calculated by using empirical equations described below, and the value of C used to evaluate the models was 0.25 for croplands in Ethiopia as estimated in Hurni (1985) for Ethiopian arable lands. Then A estimated by each model was compared with the observed soil loss averaged over the two study years in the respective control plots and calculated the percentage bias (%bias) of each model. Finally, the model that resulted in the lowest %bias was selected as the best model for determining *R*-factor at all sites.

Data from the selected rainfall events from June to September in 2015 and 2016, collected at 10 min intervals by automatic tipping bucket rain gauge, from the three sites were used to calculate the unit kinetic energy (*e*), the storm kinetic energy (*E*), and the rainfall erosivity factor (*R*) as follows:

1. $\mathrm{R} = 5.5 * \mathrm{Ps} - 47$ (Hurni, 1985), where Ps is seasonal rainfall in millimeters
2. $e_r = 0.119 + 0.0873 * \log(i_r)$ (Wischmeier and Smith, 1978) for

   $i_r \leq 76$ mm h^{-1}, and $e_r = 0.283$ for $i_r > 76$ mm h^{-1}
3. $e_r = 0.29[1 - 0.72\exp(-0.05\, i_r)]$ (Renard *et al*., 1997)

   $i_r \leq 76$ mm h^{-1}, and $e_r = 0.283$ for $i_r > 76$ mm h^{-1}
4. $e_r = 36.65(1 - (0.6/i_r))$ for $i_r > 0.6$ mm h^{-1} (Nyssen *et al*., 2005)
5. $e_r = 7.56 * \ln(i_r) + 9.98$ (Meshesha *et al*., 2014)
6. $e_r = 6.8 * \ln(i_r) + 5.96$ (Meshesha *et al*., 2018)
7. $e_r = 28.3[1 - 0.52\exp(-0.042 i_r)]$, (Van Dijk *et al*., 2002)

Here, e_r is the unit rainfall kinetic energy for the r^{th} time interval (J m^{-2} mm^{-1}), and i_r is the rainfall intensity in the r^{th} time interval (mm h^{-1}) during a storm event, and storm kinetic energy (*E*) (MJ m^{-2}) is calculated as follows:

$$E = \sum_{i=1}(e_r * p_r), \qquad (2)$$

where *E* is the total storm kinetic energy for storm *i*, and p_r is the rainfall amount in the r^{th} time interval (mm) in a storm event;

Then rainfall erosivity *R* (MJ mm ha^{-1} h^{-1} $season^{-1}$) is calculated as follows:

$$R = \frac{\sum_{j=1}^{n}(EI_{30})_i}{N} \quad (3)$$

where $(EI_{30})_i$ is the erosivity index for storm *i*, n is the number of storms in the season or year, I_{30} is the maximum 30-min rainfall intensity, and *N* is number of years if more than one-year data is used.

Soil erodibility (*K*). Soil erodibility (*K*) was determined from soil texture, soil organic matter content, soil structure class, and permeability by the method of Wischmeier and Smith (1978) and by applying correction factor for surface rock fragment cover in degraded bushland at Aba Gerima as suggested by Nyssen *et al*. (2009):

$$K = [\, 2.1 * (10^{-4})(12 - OM)M^{1.14} + 3.25(s - 2) + 2.5(p - 3)] * \frac{0.1317}{100} * e^{-0.04*(Rc-10)} \quad (4)$$

where *K* is soil erodibility (t h MJ^{-1} mm^{-1}), *OM* is organic matter (%), *M* is a particle size parameter calculated as [silt (%) + very fine sand (%)] × [100 –clay (%)]; *s* is the soil structure class, which ranges between 1 (very fine granular) and 4 (blocky, platy, or massive), with a default value of 2; and *p* is the permeability class, which ranges between 1 (rapid) and 6 (very slow), with a default value of 3; *Rc* is surface rock fragment cover, measured in the field by the point-count method (Nyssen et al., 2009).

where *Rc* was calculated as: *RC* (%) = Np/Nt* 100, where: Np = number of observations with a rock fragment; Nt = total number of observations. The effect of surface rock fragment cover was accounted for degraded bushland at Aba Gerima, but it was not taken into account in all other land uses as it was negligible.

Slope length and steepness (*L* and *S*). *L* is a dimensionless factor for slope length determined by the equation of Renard *et al*., (1997):

$$L = (\lambda/22.13)^m \quad (5)$$

Where λ is the length of the horizontal projection of the slope and *m* is a dimensionless slope length exponent. For soils that are moderately susceptible to both rill and inter-rill erosion, values for *m* were estimated from Foster *et al*., (1977) and McCool *et al*. (1989) to be 0.5, 0.65, and 0.72 for gently (5% gradient), steeply (15%), and very steeply (35%) sloping plots, respectively.

The equation proposed by McCool *et al*., (1987) for estimating slope steepness factor(*S*) in RUSLE under predicts the factor for very steep slopes (Nearing, 1997). Hence, we used the empirical equation developed by Nearing (1997) that better fits to all slope ranges, considering our data is from all slope categories, to determine the dimensionless slope steepness factor *S*:

$$S = -1.5 + 17/(1 + e^{(2.3-6.1\,sin\theta)}) \quad \textit{(6)}$$

where θ(degrees) is the slope gradient angle.

Support practices (*P*) factor. *P*, the support practices factor, reflects the effects of LM practices on reducing soil loss and runoff. *P* can be determined as the product of a sub factor due to contouring (P_C) and LM structure factor (P_{Str}) (Renard *et al*., 1997). P_{Str} is computed as the product of sub factors for each individual structure type (e.g., bunds, *fanya juu*, and trenching). In this study, tillage was performed only once at sowing time with a hoe and a defined pattern related to the slope contours was not produced; and hoeing was considered as a minimum tillage and incorporated in *C*-factor as a management practice. Therefore, *P*-factor was calculated as a reflection of LM structures only as follows:

$$P = SLs/SLc, \quad (7)$$

where SL_s is the measured seasonal soil loss on a plot where LM structures have been constructed and SL_c is the seasonal soil loss from the control plots without LM structures (t ha^{-1} season^{-1}).

Cover and management practices (*C*) factor. The C-factor (dimensionless) accounts for how land cover (for non-croplands) and crops and crop management (for croplands) affect soil erosion as compared to bare fallow areas (Wischmeier & Smith, 1978). *C*-factor value is estimated as the ratio of annual soil loss from the control plot divided by the product of the calculated RUSLE factors. *C*-factor calculated in this way mainly takes into account the effect of cropping and management practices (tillage practices, planting of cover crops, and incorporation of plant residues) on erosion (Renard *et al*., 1997):

$$C = SL_c / RKLSP, \tag{8}$$

where SL_c = seasonal soil loss from the control plot (t ha^{-1} season^{-1}).

2.3 Results

2.3.1 Rainfall erosivity (R) factor

The result of average *R*-factor estimates (Table 2.3) showed wide variation among the seven storm kinetic energy models; specifically, *R* value from Hurni1985 was quite higher than *R* values from other models in the three agro-ecologies, which in turn caused significant differences in soil loss prediction.

Table 2. 3 Comparison of measured seasonal soil loss from control plot (SL_c) and estimated soil loss (SLe) from the three steeply sloping control cropland plots (CL2-C) using R-factor from different-storm kinetic energy models in RUSLE.

Site	Model	Land use	*R*	*K*	*L*	*S*	*C*	*P*	SL_e	SL_c	%Bias
Guder	Hurni1985	Cl2-C	7343	0.02	1.18	1.88	0.25	1	76.3	10.05	659
	WS1978	Cl2-C	1175	0.02	1.18	1.88	0.25	1	13.7	10.05	21
	Renard1997	Cl2-C	1072	0.02	1.18	1.88	0.25	1	11.1	10.05	11
	Nyssen2005	Cl2-C	1765	0.02	1.18	1.88	0.25	1	18.3	10.05	82
	Meshesha2014	Cl2-C	1644	0.02	1.18	1.88	0.25	1	17.1	10.05	70
	Meshesha2018	Cl2-C	1325	0.02	1.18	1.88	0.25	1	13.8	10.05	37
	Van Dijk2002	Cl2-C	1121	0.02	1.18	1.88	0.25	1	11.6	10.05	16
Aba Gerima	Hurni1985	Cl2-C	3714	0.02	1.18	1.88	0.25	1	44.6	2.34	1808
	WS1978	Cl2-C	1317	0.02	1.18	1.88	0.25	1	15.8	2.34	576
	Renard1997	Cl2-C	1272	0.02	1.18	1.88	0.25	1	15.3	2.34	553
	Nyssen2005	Cl2-C	1829	0.02	1.18	1.88	0.25	1	22.0	2.34	839
	Meshesha2014	Cl2-C	1920	0.02	1.18	1.88	0.25	1	23.1	2.34	886
	Meshesha2018	Cl2-C	1572	0.02	1.18	1.88	0.25	1	18.9	2.34	708
	Van Dijk2002	Cl2-C	1275	0.02	1.18	1.88	0.25	1	15.3	2.34	555
Dibatie	Hurni1985	Cl2-C	3072	0.003	1.18	1.88	0.25	1	5.6	2.00	182
	WS1978	Cl2-C	1165	0.003	1.18	1.88	0.25	1	2.1	2.00	7
	Renard1997	Cl2-C	1118	0.003	1.18	1.88	0.25	1	2.0	2.00	3
	Nyssen2005	Cl2-C	1592	0.003	1.18	1.88	0.25	1	2.9	2.00	46
	Meshesha2014	Cl2-C	1708	0.003	1.18	1.88	0.25	1	3.1	2.00	57
	Meshesha2018	Cl2-C	1401	0.003	1.18	1.88	0.25	1	2.6	2.00	29
	Van Dijk2002	Cl2-C	1123	0.003	1.18	1.88	0.25	1	2.1	2.00	3

Note: *R* is rainfall erosivity factor (MJ mm ha^{-1} h^{-1} yr^{-1}), *K* = soil erodibility (t h MJ^{-1} mm^{-1}), *L* = slope lenth, *S* = slope steepness, *P* = Support practice factor, and *C* = cover & management factors; SL_c = meausred seasonal soil loss from control plot (t $ha^{-1}season^{-1}$) and SLe = estimated soil loss from control plots (t ha^{-1} yr^{-1}).

Generally, soil loss predicted by RUSLE, using *R* value from models, was higher than the observed soil loss. Among the seven candidate models, the %biases in soil loss predicted based on Renard1997 were the lowest followed by Van Dijk2002 with small variation between the two models (Table 2.3). As a result, the Renard1997 model was selected and used to determine *R* values in RUSLE (Table 2.3).

2.3.2 Support practices (P) factor values for different LM practices

The factor *P* values determined for each LM practices under different land uses and agro-ecologies in the two study seasons are presented in Table (2.4). Generally, *P* value ranged from 0.06 to 0.53 in cropland plots and from 0.03 to 0.42 in non-cropland plots (Table 2.4). In the cropland plots at the three sites, the *P* values range from 0.15 to 0.53 for soil bund, from 0.18 to 0.5 for *fanya juu*, and 0.06 to 0.44 for soil bund with grass while the values ranged from 0.03 to 0.42 for non-cropland plots (Table 2.4).

Most LM practices were associated with reduced *P* values in the second year at all sites (Tables 4 & 5). The lowest values were observed in the second year for soil bund with grass in cropland plots at all sites, and the highest was observed for soil bund in the CL1 plot in the first year at Aba Gerima. The amount of reduction in *P* (temporal evolution between 2015 and 2016) depended on both the LM practice type and agro-ecology (the study site), details are presented in Table (2.4). The average *P* value for each LM practice also differed among the agro-ecologies: ranging from 0.3 in Dibatie to 0.38 in Aba Gerima for soil bund, from 0.28 in Dibatie to 0.43 in Aba Gerima for fanya juu, and from 0.18 in Dibatie to 0.32 in Aba Gerima for soil bund with grass (Table 2.5).

Table 2. 4 Calculated support practices (P) factor values for different LM practices in the three land use types at each study site.

Land use		Slope (%)	Year	Guder SL	Guder P	Aba Gerima SL	Aba Gerima P	Dibatie SL	Dibatie P
CL1	C	5	2015	0.57	1.00	5.39	1.00	1.25	1.00
	SB	5	2015	0.25	0.44	2.87	0.53	0.38	0.30
	F	5	2015	0.19	0.33	2.61	0.48	0.47	0.38
	SBG	5	2015	0.12	0.21	2.36	0.44	0.39	0.31
CL2	C	15	2015	6.73	1.00	4.30	1.00	3.88	1.00
	SB	15	2015	3.35	0.50	1.60	0.37	1.68	0.43
	F	15	2015	3.09	0.46	2.17	0.50	0.89	0.23
	SBG	15	2015	2.53	0.38	1.35	0.31	0.81	0.21
GL	C	15	2015	2.85	1.00	39.67	1.00	2.26	1.00
	E	15	2015	1.79	1.00	33.10	1.00	1.17	1.00
	T+E	15	2015	1.04	0.36	16.70	0.42	0.64	0.28
DBL	C	35	2015	1.55	1.00	22.82	1.00	2.49	1.00
	E	35	2015	0.58	1.00	8.46	1.00	1.05	1.00
	T+E	35	2015	0.27	0.17	1.94	0.09	0.28	0.11
CL1	C	5	2016	0.39	1.00	1.54	1.00	0.98	1.00
	SB	5	2016	0.06	0.15	0.48	0.31	0.15	0.15
	F	5	2016	0.10	0.26	0.49	0.32	0.18	0.18
	SBG	5	2016	0.08	0.21	0.41	0.27	0.12	0.12
CL2	C	15	2016	13.37	1.00	0.38	1.00	1.73	1.00
	SB	15	2016	2.06	0.15	0.12	0.32	0.52	0.30
	F	15	2016	2.83	0.21	0.16	0.42	0.55	0.32
	SBG	15	2016	1.12	0.08	0.10	0.26	0.11	0.06
GL	C	15	2016	1.57	1.00	17.67	1.00	2.04	1.00
	E	15	2016	1.09	1.00	0.80	1.00	1.03	1.00
	T+E	15	2016	0.46	0.29	1.29	0.07	0.41	0.20
DBL	C	35	2016	0.95	1.00	24.70	1.00	2.10	1.00
	E	35	2016	1.13	1.00	4.83	1.00	0.15	1.00
	T+E	35	2016	0.08	0.08	0.84	0.03	0.08	0.04

Note: P = Support practice factor, SL = measured seasonal soil loss rates from experimental plots (t ha^{-1}); C = control plot, CL1 = gently (5%) sloping cropland, CL2 = steeply (15%) sloping cropland, GL = grassland (15% slope); and practices: DBL = degraded bush land (35% slope), SB = soil bund, F = *fanya juu*, SBG = soil bund with grass, E = Exclosure, T+E = trench with exclosure.

Generally, across agro-ecologies, average *P* values for all LM practices in cropland plots decreased in the order Aba Gerima (mid-land) > Guder (highland) > Dibatie (lowland) in cropland plots (Table 2.5). LM practices used in non-cropland plots, represented by grassland and degraded bushland in this study, were exclosure and trench + exclosure. *P* values calculated for trench + exclosure for each year varied from 0.07 to 0.42 in grassland plots and from 0.03 to 0.17 in degraded bushland plots (Table 2.4). The two-year average *P* values for non-cropland plots from the three sites ranged from 0.15 to 0.23 by agro-ecology, decreasing in the order Guder > Dibatie > Aba Gerima (Table 2.5).

2.3.3 Cover and management practice (C) factor for different land uses

The *C*-factor values calculated for different land use types using equation 8 are presented in Table (2.6). For the cropland plots, the highest *C* value (0.64) was observed at Dibatie for chili peppers in the first year (2015) and the lowest value (0.004) for teff at Aba Gerima in 2016 (Table 2.6). Similarly, for grassland, both the highest and lowest *C* values (0.49 and 0.01, respectively) were observed at Aba Gerima; and for degraded bushland, the lowest value (0.001) was observed at Dibatie and the highest value (0.20) at Aba Gerima (Table 2.6).

The two-year average *C* values determined for cropland and non-cropland plots from the three sites were 0.24 and 0.1, respectively (Table 2.7). Furthermore, in the non-cropland plots, the average *C* values were determined as 0.19 and 0.07 for control, and 0.11 and 0.02 for exclosed grassland and degraded bushland plots, respectively (Table 2.7). Across agro-ecologies, the average *C* values increased in the order of Guder (0.12) < Aba Gerima (0.13) < Dibatie (0.46) for cropland plots, and Guder (0.02) < Dibatie (0.08) < Aba Gerima (0.2) for the non-cropland plots.

Table 2. 5 Average P-factor values for soil bund (SB), fanya juu (F), and soil bund with grass (SBG) in the cropland; and trench with exclosure (T+E) for non-cropland plots in the two study seasons at the three study sites

Land uses	LM practices	Guder			Aba Gerima			Dibatie			Overall mean
		2015	2016	Mean	2015	2016	Mean	2015	2016	Mean	
Cropland	SB	0.47	0.15	0.31	0.45	0.31	0.38	0.37	0.23	0.3	0.33
	F	0.4	0.23	0.32	0.49	0.37	0.43	0.3	0.25	0.28	0.34
	SBG	0.29	0.14	0.22	0.38	0.26	0.32	0.26	0.09	0.18	0.24
	Mean	0.39	0.17	0.28	0.44	0.31	0.38	0.31	0.19	0.25	0.30
Non -cropland	T+E	0.27	0.19	0.23	0.25	0.05	0.15	0.2	0.12	0.16	0.18

Table 2. 6 Cover and management practice (C) factor values calculated using measured seasonal soil loss from control plot (SLc) in gentle and steep control croplands (CL1-C & CL2-C), control/freely grazed and exclosed grassland (GL-C & GL-E), and degraded bush land (DBL-C & DBL-E) plots at the three sites during the two study years (2015 and 2016) and other RUSLE factor values. P-factor = 1 for control

Site	Land use code	Cover type	Slope (%)	Year	*R*	*K*	*L*	*S*	SLc	*C*
Guder	CL1-C	Barley	5	2015	1038	0.02	1.13	0.53	0.5	0.04
	CL2-C	Barley	15	2015	1038	0.02	1.18	1.88	6.7	0.16
	GL-C	Grass	15	2015	1038	0.02	1.18	1.88	2.8	0.06
	GL-E	Grass	15	2015	1038	0.02	1.18	1.88	1.7	0.04
	DBL-C	Grass+ shrubs	35	2015	1038	0.03	1.18	5.80	1.5	0.00
	DBL-E	Grass+ shrubs	35	2015	1038	0.03	1.18	5.80	0.5	0.00
	CL1-C	Teff	5	2016	1106	0.02	1.13	0.53	0.3	0.03
	CL2-C	Teff	15	2016	1106	0.02	1.18	1.88	13.	0.29
	GL-C	Grass	15	2016	1106	0.02	1.18	1.88	1.5	0.03
	GL-E	Grass	15	2016	1106	0.02	1.18	1.88	1.0	0.02
	DBL-C	Grass+ shrubs	35	2016	1106	0.03	1.18	5.80	0.9	0.00
	DBL-E	Grass+ shrubs	35	2016	1106	0.03	1.18	5.80	1.1	0.00
Aba Gerima	CL1-C	Finger millet	5	2015	1295	0.02	1.13	0.53	5.3	0.32
	CL2-C	Finger millet	15	2015	1295	0.03	1.18	1.88	4.3	0.05
	GL-C	Grass	15	2015	1295	0.03	1.18	1.88	39.	0.49
	GL-E	Grass	15	2015	1295	0.03	1.18	1.88	33.	0.40
	DBL-C	Grass+ shrubs	35	2015	1295	0.01	1.18	5.80	22.	0.18
	DBL-E	Grass+ shrubs	35	2015	1295	0.01	1.18	5.80	8.4	0.07
	CL1-C	Teff	5	2016	1249	0.02	1.13	0.53	1.5	0.09
	CL2-C	Teff	15	2016	1249	0.03	1.18	1.88	0.3	0.00
	GL-C	Grass	15	2016	1249	0.03	1.18	1.88	17.	0.22
	GL-E	Grass	15	2016	1249	0.03	1.18	1.88	0.8	0.01
	DBL-C	Grass+ shrubs	35	2016	1249	0.01	1.18	5.80	24.	0.20
	DBL-E	Grass+ shrubs	35	2016	1249	0.01	1.18	5.80	4.8	0.04
Dibatie	CL1-C	Chili pepper	5	2015	1181	0.00	1.13	0.53	1.2	0.64
	CL2-C	Chili pepper	15	2015	1181	0.00	1.18	1.88	3.8	0.45
	GL-C	Grass	15	2015	1181	0.00	1.18	1.88	2.2	0.18
	GL-E	Grass	15	2015	1181	0.00	1.18	1.88	1.1	0.10
	DBL-C	Grass +	35	2015	1181	0.02	1.18	5.80	2.4	0.02
	DBL-E	Grass +	35	2015	1181	0.02	1.18	5.80	1.0	0.01
	CL1-C	Teff	5	2016	1055	0.00	1.13	0.53	0.9	0.56
	CL2-C	Teff	15	2016	1055	0.00	1.18	1.88	1.7	0.2
	GL-C	Grass	15	2016	1055	0.00	1.18	1.88	2.0	0.19
	GL-E	Grass	15	2016	1055	0.00	1.18	1.88	1.0	0.09
	DBL-C	Grass +	35	2016	1055	0.02	1.18	5.80	2.1	0.02
	DBL-E	Grass +	35	2016	1055	0.02	1.18	5.80	0.1	0.00

Table 2. 7 Cover and management practice (C) factor values obtained by this study and from previous studies in Ethiopia and other countries

Cover type/ land use	C values by this study				C-factor values from previous studies				
	No of events	Range	Mean	Median	Taye et al. (2017) (Ethiopia)	Nyssen et al. (2009) (Ethiopia)	Hurni (1985) (Ethiopia)	Nill (1993) (Cameroon)	Panos et al. (2015) (Europe)
Teff	190	0.04 - 0.56	0.20	0.16		0.07	0.25		
Finger millet	56	0.05 - 0.32	0.18	0.18					
Barley	98	0.04 - 0.16	0.10	0.10	0.12	0.21	0.15		0.21
Chili pepper	43	0.45 - 0.64	0.54	0.54					
Cropland	387	0.02 - 0.64	0.24	0.19	0.03 - 0.35	0.14	0.10 - 0.25		
GL-free	387	0.03 - 0.49	0.19	0.18	0.23 - 0.82	0.42	0.05	0.4	
GL-exclosed	387	0.01 - 0.4	0.11	0.06					
DBL-free	387	0.004 - 0.18	0.07	0.02					
DBL-exclosed	387	0.001 - 0.07	0.02	0.01					
Forest						0.004	0.001		

Note: GL = grassland, DBL = degraded bushland

2.4 Discussion

2.4.1 Support practices (P) factor

P-factor values in RUSLE indicate the effectiveness of physical LM practices. A lower value for an LM practice means higher effectiveness and vice versa. In general, lower *P* values were observed in the second year at all sites (Table 2.4). Many factors might contribute to higher *P* values in the first year after their construction. For example, effects from previous traditional land use activities (conventional tillage and intensive grazing) that degraded the topsoil (Mwendera & Saleem, 1997) and the fresh disturbance of soils during construction of the LM practices together with the absence of protective vegetative cover on the bunds and trenches at the beginning of rainy season following construction can lead to higher erosion rates that increased *P*-factor values in the first year. But, since the plots were fenced to prevent animals from entering and damaging the LM practices, and to allow grasses and other weeds to grow and stabilize the structures, their effectiveness was increased (lower *P*-factor values) in the second year (Tables 2.4 & 2.5). The result from this study, however, contradicts with the result of Taye *et al*. (2015), who reported that effectiveness of stone bunds and trenches in Tigray region of Ethiopia declines over time. This could be attributed to lack of fencing of plots during the dry seasons in their experimental period that increases grazing pressure and erosion rate (Nyssen *et al*., 2009), but decreases structures' effectiveness (higher *P* values) over time; because sediment at the ground surface become loosened by the hooves of the grazing animals when the grass cover was insufficient before the onset of the rainy seasons (Herweg & Ludi, 1999; Stavi *et al*., 2016).

The average *P*-factor value of 0.34 from our cropland plots (Table 2.5) was greater than the *P* value of 0.19 reported by Herweg & Ludi (1999) for *fanya juu* at Anjeni watersheds. However, the average *P*-factor value of 0.33 for soil bund from our plots (Table 2.5) was lower than the values (0.65) reported by Amare *et al*. (2014) from Debre Mewi experimental watershed, and (0.38) reported by Herweg & Ludi (1999) from Anjeni watersheds, both in the UBN basin. The average *P* value (0.24) from our plots (Table 2.5) was also lower than the value (0.61) reported by Amare *et al*. (2014) for soil bund with grass. Generally, the lower *P* value in our cropland plots than previous studies was attributed mainly to differences in the management practices (tillage and fencing), climatic and biophysical conditions between the study areas.

Comparing the averaged *P* values among the different LM practices in the crop land plots, soil bund with grass had the lowest *P* value (0.24) while *fanya juu* had the highest (0.34) (Table 2.5). This suggests that integrating structural and vegetative measures to control soil erosion from cultivated lands is more effective than the use of physical structures only, especially in areas with steep slopes, where relatively higher soil loss can be expected. This result is consistent with the findings of Amare *et al*. (2014), who reported that soil bund with grass is efficient means of controlling soil loss in Debre Mewi experimental watershed of the UBN basin. In addition to reducing soil loss, the grass provides many economic benefits as a source of forage to farmers and allows effective use of the land occupied by the soil bund. For the non-cropland plots, the average *P* value (0.1) for exclosure with trench in our plots is lower than the range of values (0.22 to 0.33) reported by Taye *et al*. (2017) for trenches in the semi-arid Tigray region. The lower average *P* value in our plots could be related to the effect of exclosure that prevented the trench from damage by animal trampling, thereby increasing its effectiveness (lower *P*-factor value) to reducing soil loss.

Generally, average *P* value from cropland plots (0.30) was higher than those of non-cropland plots (0.18) (Table 2.5), indicating that the LM practices on non-cropland plots were more effective in controlling soil loss than those on croplands. Similarly, Taye *et al*. (2017) reported that *P*-factor values are generally higher in cropland than in rangeland plots. The difference in *P* values between these two land use types is attributable to variations in soil loss; seasonal soil losses from non-cropland plots are generally higher than those from cropland plots, mainly because smaller rainfall events that generate runoff and erosion in non-cropland plots might not generate runoff and soil loss in cropland plots (Taye *et al*., 2017). In the non-cropland control plots, small but frequent rainfall events caused a relatively larger proportion of the soil loss, but that soil loss was effectively controlled by LM practices that reduced *P*-factor values. In contrast, from cropland plots, tillage obstructs runoff from small-intensity events and soil loss occurs mainly during larger rainfall events (Taye *et al*., 2013). As a result, the small events occupy some of the storage capacity of the structure, which fills up sooner and limits the effectiveness of LM practices and also causes the sediment-loaded runoff to overflow or seep through partially permeable bunds and *fanya juu* (Taye *et al*., 2015), which decreases the efficiency of LM practices (higher *P* values) in cropland plots. Moreover, trenches are more effective in reducing runoff and soil loss than soil bund and *fanya juu*, which are partly permeable to sediment-loaded runoff (Taye et al., 2015).The variations in *P* values in the two land use types could also be explained by differences in climate and other biophysical conditions that result in spatial and temporal variability in soil erosion (Ebabu *et al*., 2019).

Looking the *P*-factor values across agro-ecologies (Table 2.5), the higher average *P*-factor value (0.38) from the cropland plots at Aba Gerima, as compared to Guder (0.28) and Dibatie (0.25), could be due to the high rainfall intensity at the beginning of the rainy seasons (Ebabau *et al*., 2019) that resulted in higher rainfall erosivity (*R*-factor), which could in turn increase soil loss rate and *P*-factor values at Aba Gerima (Table 2.5). However, the lower average *P*-factor value at Dibatie could be attributed to the high rainfall threshold required to generate runoff (10 mm), as compared to Guder (6 mm), that increased infiltration, reduced runoff (Sultan *et al*., 2018) and soil loss (lower *P* value) (Table 2.5). In contrast, the higher average *P* value (0.23) in the non-cropland plots (Table 2.5) at Guder could be explained by the low rainfall threshold (6 mm), as compared to Aba Gerima (9 mm) and Dibatie (9 mm), that caused a longer period smaller rainfall events to generate runoff and fill up structures sooner (Sultan *et al*., 2018), thus decreasing their effectiveness (higher *P* values) from non-crop land plots. Conversely, the smaller average *P*-factor values from non-crop land plots at Aba Gerima (0.15) and, Dibatie (0.16) (Table 2.5) could also be explained by the high rainfall threshold (9 mm) in the two sites that increased infiltration but decreased runoff and soil loss (lower *P* values).

2.4.2 Cover and management practices (C) factor

Another of our objectives was to determine *C* values for different land uses and crops. The average *C* value for cropland (0.24) in our study (Table 2.7), is in the range of values reported by other studies (e.g., Hurni, 1985; Taye *et al*., 2017) but higher than that reported by Nyssen *et al*., (2009), for cereal-based cropland systems in Ethiopia (Table 2.7). Similarly, average *C* value for teff (0.20) (Table 2.7, Figure 2.3d) from our plot was higher than the value (0.07) reported by Hurni (1985) but lower than the value (0.25) reported by Nyssen *et al*. (2009); *C*-factor value (0.10) for barley (Table 2.7, Figure 2.3a) from our plot was also lower than those reported in Ethiopia (Taye *et al*., 2017; Nyssen *et al*., 2009; Hurni, 1985) and Europe (Panagos et al., 2015). Generally, lower *C* values from our cropland plots could be related to the relatively low soil loss in our plots due to fencing and conservation tillage whereby crop residues are incorporated into the soil when cropland plots were plowed in the second year, which strengthened the top soil layer, increased resistance to erosion, and decreased soil loss (Nyssen *et al*., 2009). In non-cropland plots, the average *C* value (0.15) in grassland was much higher than that in degraded bushland (0.05) (Table 2.7), mainly due to high vegetative cover and relatively less grazing pressure that reduced soil loss and *C* values in the degraded bushland plots. The average *C* values for freely grazed plots were also higher than those exclosed ones in both grassland and degraded bushland plots (Tables 2.6 & 2.7), implying that exclosure of non-cropland plots is more effective in restoring of the natural vegetation and then reducing soil erosion. Descheemaeker *et al*. (2006) also reported that exclosures have a high sediment-trapping efficiency, ranging from 70% to 99%. Amdihun *et al*. (2014), who studied in the UBN basin recommended exclosure of steeply sloping areas and their conversion to forest and grasslands to reduce erosion. Positive effects of exclosure observed in the non-cropland plots included regeneration of natural vegetation (Mekuria *et al*., 2007) and reduction of runoff and soil loss (Descheemaeker *et al*., 2006; Haregeweyn *et al*., 2012; Girmay *et al*., 2009).

Generally, higher average *C* value (0.24) was observed in cropland plots, compared to the non-cropland plots (0.1), mainly due to differences in cover and management practices. The variation in the cover conditions in the two land uses could be further explained, such that exclosure in non-cropland plots provided better protection to the soil throughout the year thereby reducing soil loss and *C*-factor values, compared with cropland plots (Mekuria *et al*., 2007; Maetens, 2013). However, this result is contrary to the findings of Taye *et al*. (2017), who obtained larger *C* values for grasslands than croplands in Tigray region: the variation in the two studies could be explained by the climatic difference between the humid tropical highlands where our study was conducted and the semi-arid Tigray region. The relatively high rainfall in our study area leads to better grass and vegetation cover in non-croplands for longer span of the year, which reduces soil erosion and *C* values. In contrast, in Tigray region where annual rainfall is relatively small, the vegetation cover is less and grazing pressure is high (Nyssen *et al*., 2009), all of which cause soil erosion and *C* values to be relatively higher in grasslands (Taye *et al*., 2013).

The average *C* values, both in cropland and non-cropland plots, also varied across agro-ecologies. The lower average *C* values (0.12, 0.02) from cropland and non-cropland plots, respectively, at Guder, as compared to Aba Gerima (0.13, 0.2) and Dibatie (0.46, 0.08) (Table 2.6), could be caused by the longer rainy period that maintained good moisture and cover condition for most seasons of the year and reduced *C*-factor values. On the other hand, the higher average *C* value (0.46) obtained at Dibatie cropland plots (Table 2.6), is attributable to the row crop chili peppers that provided negligible cover in the early stage during which high erosion is expected because they are planted in widely spaced rows, compared to broadcasted cereals (Figure 2.3c). The higher average *C* value (0.2) at Aba Gerima for non-cropland plots could also be related to the previous intensive grazing practices that significantly increased soil loss in the first year.

2.5 Conclusions

This study was conducted to determine values of *C*- and *P*- factors of RUSLE for different cover conditions and land management practices across agro-ecologies in the upper Blue Nile basin. The results show that soil bund with grass in cropland and exclosure with trench in the non-cropland plots had the smallest average *P* values of all other practices, implying that these practices were more effective in reducing soil loss. In addition, a lower average *C* value (0.1) was observed in non-cropland plots, compared to the cropland plots (0.24). The lower average *C*-factor value observed from the non-cropland suggests that exclosure in the non-cropland plots provided better cover conditions to the soil than crops in cropland plots. However, the effect of agro-ecology on *C*- and *P*-factor values was not consistent, mainly due to variations in climate, biophysical conditions and management practices across study sites. In addition to *C*- and *P*- factors, Renard1997 was selected as the best model to determine *R*-factor in the basin. Generally, determining *C*- and *P*- factor values, together with the recommended *R*-factor model, will help minimize uncertainties in future soil loss prediction studies in the basin and elsewhere with similar environments. It should be noted that since *C*-factor was determined using RUSLE factor values, the potential errors introduced in calculated *R*, *K* and LS factors may affect *C*-factor values.

Chapter 3: Effectiveness of Polyacrylamide in reducing runoff and soil loss under simulated consecutive rainfall storms

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3.1 Introduction

Soil erosion by water is the most threatening global problem causing adverse on-and off-site consequences, such as the depletion of soil fertility (Haregeweyn et al., 2008; Lal, 1995), siltation of downstream reservoirs (Haregeweyn et al., 2006; Vanmaercke et al., 2011), loss of vital ecosystem services, and associated economic costs (Kirui et al., 2014). As a result, soil erosion has become a major threat to food security particularly for developing countries where livelihoods predominantly depend on agriculture (Lal, 1995; Fenta et al., 2016, 2017, Haregeweyn et al., 2017). Hence, soil and water conservation are essential for sustaining food production and preserving the environment (Graber et al., 2006).

The major causes of soil erosion are the physical disintegration and dispersion of surface soil aggregates by the impact energy of raindrops (Mamedov et al., 2000; Yu et al., 2003) and the physico-chemical dispersion and migration of soil clays with the infiltrating water into the soil, thereby clogging the conducting pores (Teo et al., 2006; Mamedov and Levy et al., 2019), all leading to crust formation, decreased infiltration, and increased runoff and soil loss (Mamedov and Levy et al., 2019). Soil erosion can be prevented by using physical and biological measures, or through conventional management practices such as mulching, growing cover crops, etc. An alternative practice is the modification of soil properties through the application of chemical amendments to the soil, such as polyacrylamide (PAM) (Flanagan et al., 2002) to decrease aggregate disintegration.

PAM is a water-soluble, organic anionic polymer having a long molecule of identical atom chains held together by covalent bonds (Seybold, 1994) that form bridges with the soil particles through cations in soil solution (Cochrane et al., 2005). PAM has proved to be superior to other polymers in controlling erosion (Peterson et al., 2002) and is also used to improve soil physical properties

(Sojka et al., 2007). PAM remains effective in reducing soil loss by limiting the physical disintegration of aggregates caused due to water drop impact (Sepaskhah and Bazrafshan, 2005) by adsorption to the soil aggregates and increasing cohesion among soil particles (Sojka et al., 2007), thus increasing the resistance of aggregates to the direct impact of raindrops or dragging force by runoff. Factors like soil characteristics, water quality, and PAM properties such as charge density and molecular weight play important roles in the adsorption of PAM (Cochrane et al., 2005). PAM can stabilize an existing soil structure by preserving pervious pore structure during the surface seals formation (Mamedov et al., 2007), but unable to remediate a poor soil structure (Lentz and Sojka, 2007). PAM is infinitely soluble in water but dissolves very slowly (Kumar and Saha, 2011) [29]. It is more readily adsorbed by the water of a higher electrolyte concentration than by water with lower electrolyte concentration (Shainberg, 1992). Nevertheless, the addition of dissolved PAM may have some negative effects, such as enhancing water viscosity at the beginning, which in turn could lead to a decrease in infiltration rate (IR) and increased runoff, although it may decrease soil erosion (Sojka et al., 2007; Ajwa and Trout, 2006). Furthermore, the high application rates required for effective erosion control and the large volume of water required for effective dissolution are the two major obstacles constraining the usage of PAM in agriculture (Petersen et al., 2007).

Although anionic PAM is the most effective polymeric soil amendment to control erosion (Tumsavas and Kara et al, 2007), its effectiveness can be enhanced by introducing a source of electrolyte that can create a cation bridge and help the polymer to adsorb to the soil (Peterson et al.,2002). The introduction of electrolyte (such as Ca^{2+}) at the soil surface reduces chemical dispersion and migration of clay particles by strengthening the bonds between primary soil particles, thus reducing seal formation (Kumar and Saha, 2011; Mamedov et al., 2007).

Electrolytes are typically introduced in the form of gypsum and lime. The increase in the electrolyte concentration in soil replaces exchangeable Sodium (Na^+) ions from the exchange complex with dissolved Calcium (Ca^{2+}) ions. In addition, it decreases clay dispersion and surface sealing (Mamedov et al., 2007), enhances soil structure, increases infiltration rate (IR), and decreases runoff and sediment loss (Lepore et al, 2009). Furthermore, the effectiveness of PAM application depends primarily on the soil type and percentage of clay in the soil (Mamedov et al., 2017).

Many studies have been conducted to determine the effective rates of anionic PAM that can reduce runoff and soil loss through soil structure stabilization (Lado and Inbar, 2016; Abrol, et al., 2013). Most of these studies have shown that application rates of 10–20 kg ha^{-1} were effective in stabilizing the surface structure and decreasing runoff and soil erosion (Abrol, et al., 2013). In the earlier times, Gabriels et al. (1973) found that applying 38 kg ha^{-1} of anionic PAM to soil surface resulted in increased IR and reduced runoff while other researchers (Cochrane et al, 2005; Smith et al., 1999; Shainberg et al., 1990) suggested the use of 20 kg ha^{-1} PAM as an effective and economical application rate. However, these effective PAM rates in most of the studies were determined using a maximum of three simulated rainfall storms (< 250 mm rainfall), which may represent dry land regions.

For example, Lado et al. (2015) evaluated the effectiveness of granular PAM at rates of 0, 25, 50 and 100 kg ha^{-1} to reduce post-fire erosion in a Calcic Regosol affected by different fire conditions using three consecutive rainfall storms of 80 mm depth each with an intensity of 47 mm h^{-1}. They found that the application of 50 kg ha^{-1} granular PAM increased runoff during the first storm due to increased viscosity of runoff. However, this rate was more effective in reducing soil loss during the three storms in un burnt and moderately burnt soils, with a total reduction of 42% and 34%, respectively. In addition, Abrol et al. (2013) also evaluated the effect of the application rate of granular PAM (0, 5, 10 and 20 kg ha^{-1}) on IR as a function of cumulative rainfall in silt loam soil using 2-h simulated rainstorm at rainfall intensity of 37 mm h^{-1}. That study showed that the application rate of 10 kg ha^{-1} was more effective in increasing IR and reducing erosion.

In both studies, the higher PAM rates, 100 kg ha^{-1} (Lado et al., 2015) and 20 kg ha^{-1} (Abrol et al. (2013), were less effective, as compared to lower and medium rates. This could be due to the amount of rainfall applied in these storms was less to completely dissolve PAM of higher rates, as higher rates require a large volume of water for effective dissolution (Petersen et al., 2007). This suggests the importance of applying sufficient rainfall such that all the applied PAM gets fully dissolved to effectively act on soil aggregate stabilization and then monitoring the effect of PAM application rates on IR, runoff, and soil loss. PAM effectiveness is, therefore, believed to be strongly influenced by the rainfall pattern that a certain soil is subjected to and requires study under consecutive rainfall storms of high intensity. Hence, treating a test soil with different PAM rates, exposing it to consecutive rainfall storms of high intensity, which may represent humid and sub-humid regions, and determining the effective PAM rate that produces better results in terms of reducing runoff and soil loss is the best way forward. Moreover, applying PAM mixed with some source of electrolytes (e.g., gypsum and lime) at the effective rate and exposing it to consecutive rainfall storms will help to understand whether the mixture of PAM and electrolytes can further enhance the PAM effectiveness for a given test soil under consecutive storms.

The purpose of this study was, therefore, to evaluate the effectiveness of PAM rates (0, 20, 40, 60 kg ha^{-1}) in increasing IR and reducing runoff and soil loss when applied to a test soil (acidic Oxisol) and subjected to consecutive rainfall storms. The two main objectives were:

(i) To determine the effective granular PAM rate that increases IR and reduces runoff and soil loss under consecutive rainfall storms and

(ii) To verify whether the effectiveness of the selected PAM rate can further be enhanced by applying it mixed with gypsum (4 t ha^{-1}) or lime (2 t ha^{-1}) as a source of electrolyte.

3.2 Materials and Methods

3.2.1 Experimental setup and materials

This experiment was conducted at Arid Land Research Center (ALRC) of Tottori University during January–March 2019. The test soil used for this experiment was acidic clay red soil (similar to Oxisols by US Taxonomy or Acrisols and Luvisols by FAO classification), one of the widely abundant soils in Japan. Small runoff boxes of dimensions 50-cm length, 30-cm width, and 5-cm depth were filled with soil (Figure 3.1). The boxes have two outlets for collecting surface runoff and percolating (infiltration) water. The amendments used were environmentally friendly, non-toxic anionic PAM (Superfloc A-110, granular powder, 10–12% hydrolysis, and 12×10^6 Mg / mole), gypsum and lime.

The soil was air-dried, gently crushed, and sieved through a 5 mm sieve. Firstly, we packed gravel of 1–2 mm diameter up to a depth of 2 cm in the small runoff boxes to allow the percolation of infiltrated water. The sieved soil was then packed up to a depth of 3 cm in each box over the 2 cm layer of gravel, compacted to a bulk density of 1.2 g cm^{-3} using a wooden log, and finally, amendments were applied. PAM was applied onto the soil surface at a rate of 0 (C), 20 kg ha^{-1} (P20), 40 kg ha^{-1} (P40), and 60 kg ha^{-1} (P60) in granular form. The entire PAM rates were then exposed to the six consecutive storms, runoff and soil loss were measured, and effective PAM rate was selected. Later, gypsum (G) at a rate of 4 t ha^{-1}, and lime (L) at a rate of 2 t ha^{-1} were applied alone or in mixture with the effective PAM rate selected above. Each amendment was applied (uniformly distributed) over the soil surface by hand. For the combined treatments of PAM mixed with gypsum or lime, first PAM alone was applied followed by the application of gypsum or lime.

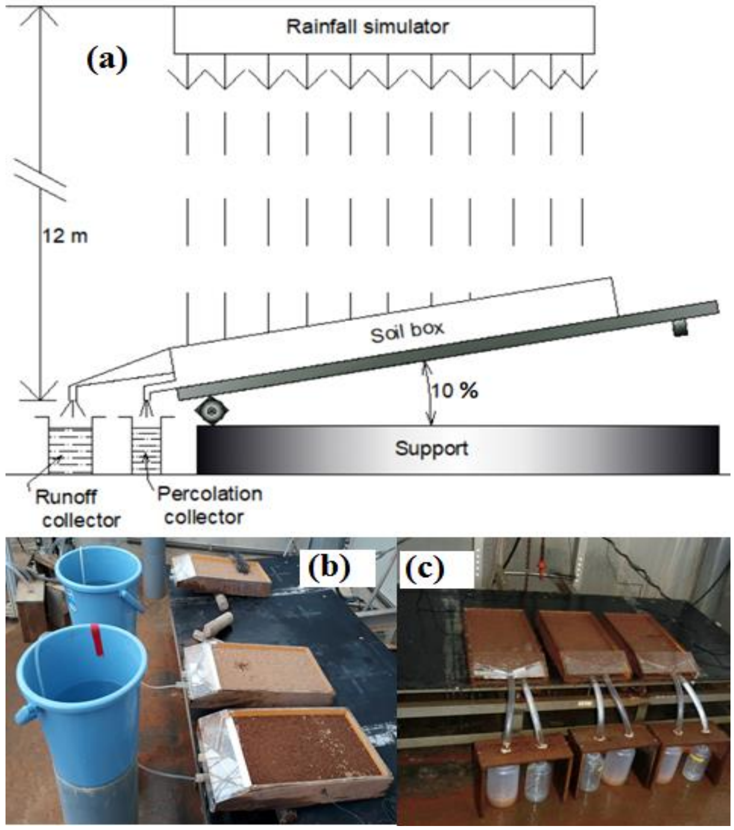

Figure 3. 1 Experimental set-up showing (a) rainfall simulator, experimental soil box and runoff and percolating water collector, (b) different treatments being saturated from below, and (c) runoff and percolating water collection during simulation.

3.2.2 Rainfall simulation procedure

A drip-type rainfall simulator facility with raindrop fall-height of 12 m, raindrop diameter of 3 mm, and rain kinetic energy of 29 J m^{-2} mm^{-1} at the ALRC, Tottori University, Japan was used in this study. The simulation was conducted in two phases: the first phase using four PAM rates (0, 20, 40, and 60 kg ha^{-1}) and the second phase using gypsum, lime, and effective PAM rate mixed with gypsum or lime. Out of the four rates, the effective PAM rate was selected based on the simulation results from the first phase.

In both phases, after all amendments were applied, the boxes were placed over the simulator tray in a horizontal position and saturated from below for 10 to 15 minutes with deionized water in order to facilitate the immediate measurement of infiltrating water during the first simulation. After saturation, the boxes were air-dried for 24 h prior to the start of the simulation. The simulator tray was set to a slope of 10%, and three soil-packed runoff boxes, with treatments assigned randomly, were positioned side-by-side at a time on the sloped platform to allow simultaneous testing of different treatments. Boxes were then exposed to six consecutive rainstorms using tap water (EC= 0.07 dS m^{-1}). Rainfall intensity was 70 mm h^{-1} lasting for 1-h rainfall duration, and time interval between two consecutive rainstorms was two days (48 h). Although the rainfall intensity of 70 mm h^{-1} lasting for 1-h may not be common under natural conditions, globally large-magnitude events are assumed to be a dominant cause of soil erosion (Zhanli, 1987). Initially, runoff boxes were covered with plastic covers, and after the rain intensity was stabilized, we removed the plastic covers and started recording the time taken to initiate runoff (TRO), the infiltrated water that come to the box outlet as sub subsurface flow, and runoff (Figure 3.1).

During each storm, runoff and infiltrating water were collected at every 10-min interval, throughout the 1-h storm duration, using temperature resistant graduated plastic bottles placed underneath the outlet at the bottom of the box. At the end of the simulation, the 10-min volume of runoff and infiltrated water was measured using a graduated cylinder. The runoff volumes were oven-dried at 105°C using the temperature resistant plastic bottles, and the weights of sediments in the runoff were determined. Then, the total soil loss of a treatment was found by summing up the amount of sediments obtained from the oven dried 10-min runoff volume in the six consecutive storms. Splash from the runoff boxes, which could be directly related to the rain intensity or erosivity (Abd Elbasit et al., 2010), was not measured. The cumulative runoff and cumulative soil loss were subjected to statistical analysis using IBM SPSS Statistics version 22 software. The simulation datasets (cumulative runoff and soil loss) were tested for normality and found to be significantly different from the normal distribution; hence parametric statistical tests could not be used for this study. Therefore, the differences among the median cumulative runoff and soil loss were subjected to analysis of variance using the Kruskal–Wallis non-parametric test (Kruskal and Wallis, 1952) at a significance level of 0.95 ($\alpha = 0.05$).

3.2.3. Soil physico-chemical properties and aggregate stability determination

Original soil properties and soil properties following the final rainfall storm were determined. Soil samples were collected from each treatment, dried, and used to determine the soil properties. Soil texture was determined for untreated soil samples using the hydrometer method (Gee and Bauder, 1986). The air-dried soil samples were used for the determination of electrical conductivity at 1:5 ratio, soil pH at 1:2.5 ratio, soil organic carbon with C/N coder apparatus, and exchangeable cation using Atomic Absorption Spectroscopy (AAS) after extraction with ammonium acetate buffered at pH 7 (Table 3.1). The soil is characterized by clay texture (44.4, 13.4, and 42.2% for clay, silt, and sand, respectively) and with 3.6% organic matter content.

Aggregate (structure) stability with three replications (taken from upper 5 mm of the soil in each box) was determined using the modified high energy moisture characteristics (HEMC) method (Levy and Mamedov, 2002). In this method, soil aggregates are wetted rapidly in a controlled manner, and a moisture content curve, at a matric potential range of 0–50 cm, corresponding to drainable pores of > 60 µm, with small steps of 1–2 cm, was generated using a hanging water column.

Table 3. 1 Effect of treatments on soil properties. Different treatments are control (C), PAM of 20 kg ha^{-1} (P20), 40 kg ha^{-1} (P40), 60 kg ha^{-1} (P60), PAM of 40 kg ha^{-1} + Gypsum (4t ha^{-1}) (P+G), Gypsum (4t ha^{-1}) (G), lime (2t ha^{-1}) (L), and PAM of 40 kg ha^{-1} + Lime (2t ha^{-1}) (P+L).

Treatments	pH	EC ($dS\ m^{-1}$)	Exchangeable cations (cmol(+) kg^{-1})			
			Ca^{2+}	Mg^{2+}	K^{+}	Na^{+}
Untreated soil	4.84	0.05	10.83	4.40	0.71	0.41
C	4.86	0.05	9.52	4.20	0.62	0.35
P20	4.86	0.05	9.80	4.33	0.69	0.36
P40	5.03	0.05	9.88	4.29	0.59	0.39
P60	5.09	0.05	9.49	4.96	0.64	0.39
P40+G	5.11	0.08	11.03	6.00	0.57	0.18
G	4.85	0.12	10.93	4.50	0.56	0.19
L	6.75	0.37	13.52	4.80	0.67	0.24
P40+L	5.14	0.09	11.79	4.10	0.69	0.28

An index of aggregate stability or structural index (SI) was determined from differences among the water retention curves (differences in pore size distribution) of the treatments by using their specific water capacity curves. The volume of drainable pores and modal suction (matric potential at the peak of the specific water capacity curve corresponds to the most frequent pore size) were determined and SI was defined as the ratio of the volume of drainable pores to modal suction, and used to characterize soil aggregate and structure stability; the higher the value of SI, the higher the stability of samples (Levy and Mamedov, 2002).

3.3 Results

3. 3.1 Effect of treatments on soil properties

The effect of soil amendments on soil properties is presented in Table 3.1. Generally, the lime substantially increased the soil pH and EC more than other treatments. However, this effect of lime was decreased when combined with PAM. Gypsum did not increase pH but slightly increased EC. This difference could probably be attributed to leaching of gypsum by the continued rainfall due to its high solubility.

3.3.2 Time to runoff

Time to runoff (TRO) for the different treatments is presented in Figure 3.2. During storm 1, the shortest TRO i.e., 67% reduction was observed for higher PAM rates (P40 and P60), followed by 33% reduction for P20 and PAM associated treatments (i.e., P+G and P+L) while 33% increment was observed for gypsum treatment, compared with the control (Figure 3.2). For storms 2 to 6, the runoff was initiated immediately after the start of rainfall.

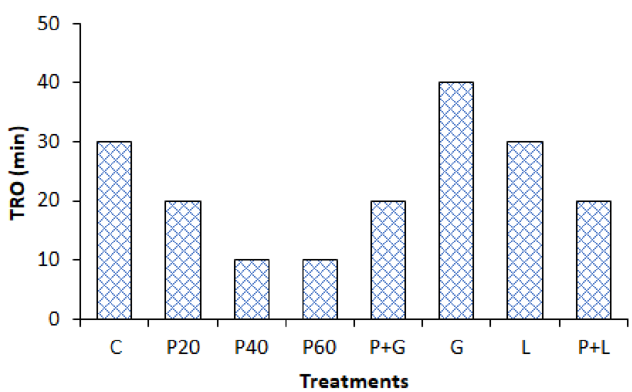

Figure 3. 2 Effect of treatments on time to runoff (TRO) for control (C), PAM of 20 kg ha^{-f} (P20), 40 kg ha$^{-($ (P40), 60 kg ha$^{-($ (P60), PAM of 40 kg ha$^{-($ + Gypsum (4 t ha^{-+}) (P+G), Gypsum (4 t ha^{-}) (G), lime (2 t ha^{-}) (L), and PAM of 40 kg ha^{-} + Lime (2 t ha^{-+}) (P+L).

3.3.3. Effect of PAM rates on infiltration rate (IR), runoff and soil loss

IR, runoff, and soil loss measured from the fine texture Oxisol treated with PAM rates of 0, 20, 40, and 60 kg ha^{-1} and subjected to 6 consecutive rainstorms separated by drying periods are presented in Figure 3.3. At the beginning of storm 1, compared with the control, IR for all PAM rates reduced by 4, 24, and 46% for P20, P40, and P60, respectively. However, as the rainfall continued from storms 2 to 6, IR for all PAM rates increased by 61–147%, 63–268%, and 20–338% for P20, P40, and P60, respectively. Furthermore, IR for higher PAM rates slightly decreased during storm 2, compared with IR during storm 1. However, IR sharply increased during storms 3 and 4 but decreased again during storm 5 and 6 for the higher rates. IR values from P20 continuously decreased throughout the consecutive storms, however, the final infiltration rate (FIR) during storm 6 increased by 61, 182, and 229% for P20, P40, and P60, respectively, compared with the control (Figure 3.3).

Following the reduction in IR at different PAM rates, when compared with the control, cumulative runoff increased during storm 1 by 5, 28, and 54% for P20, P40, and P60 rates, respectively. However, cumulative runoff decreased for all the rates later in the consecutive storms depending on the changes in IR (Figure 3.4a). During storms 2 to 6, cumulative runoff reduced by 7–27%, 15–29%, and 5–40% for P20, P40, and P60, respectively. The cumulative runoff increased in the order P60 > P40 > P20 for storms 1 and 2 and in the order C > P20 > P60 > P40 for storms 4 to 6. Nevertheless, runoff from control was the lowest during storm 1, but increased sharply during storm 2, and was the highest of all treatments throughout the consecutive storms 2 to 6 (Figure 3.4a). The total runoff from all the six consecutive storms decreased in the order of C > P20 > P60 > P40 (Table 3.2). The Kruskal–Wallis test ($P \leq 0.05$) indicates that all PAM rates reduced storm runoff significantly, compared to the control, but with no significant difference among them (Figure 3.5a).

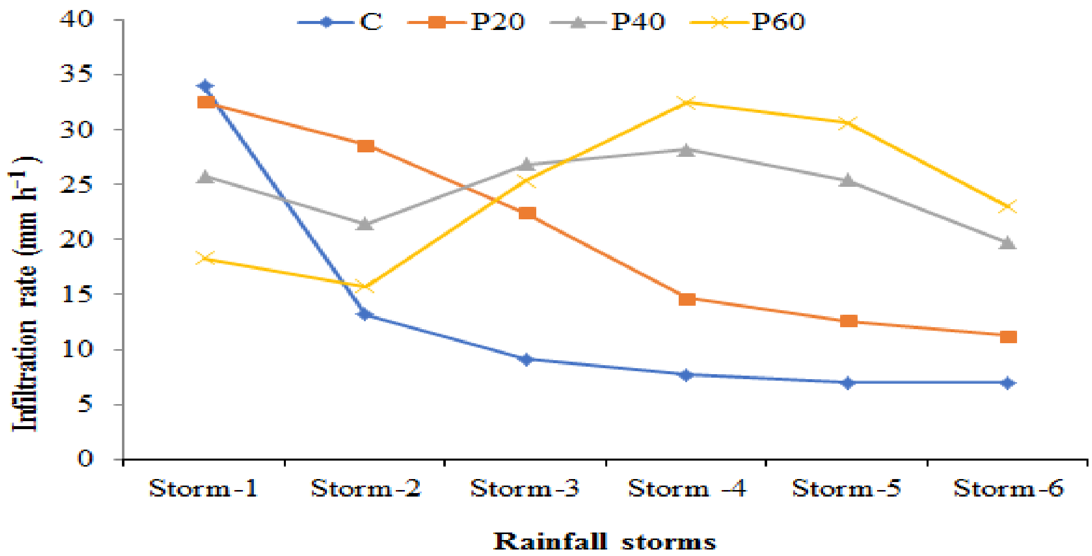

Figure 3. 3 Effect of PAM rates on infiltration rate (mm h^{-1}). Each of the six storms had an average rainfall intensity of 70 mm h^{-1} for 1-h duration and 48-h drying period. The treatments were control (C), PAM of 20 kg ha^{-1} (P20), 40 kg ha^{-1} (P40), and 60 kg ha^{-1} (P60).

On the other hand, compared with the control, all PAM rates reduced cumulative soil loss ranging from 12–59%, 35–78%, and 41–90% for P20, P40, and P60 rates, respectively during storms 1 to 6 (Figure 3.4b). For all the PAM rates, the maximum percentage reduction in soil loss was observed during the first and minimum was observed during the last storm. After six storms, the total soil loss decreased in the order C > P20 > P40 > P60 (Table 3.2). Despite the fact that all PAM rates reduced storm soil loss, P40 and P60 reduced storm soil loss significantly (using Kruskal–Wallis test, $P \leq 0.05$), compared with the control, but with no significant difference between P40 and P60 treatments (Figure 3.5b).

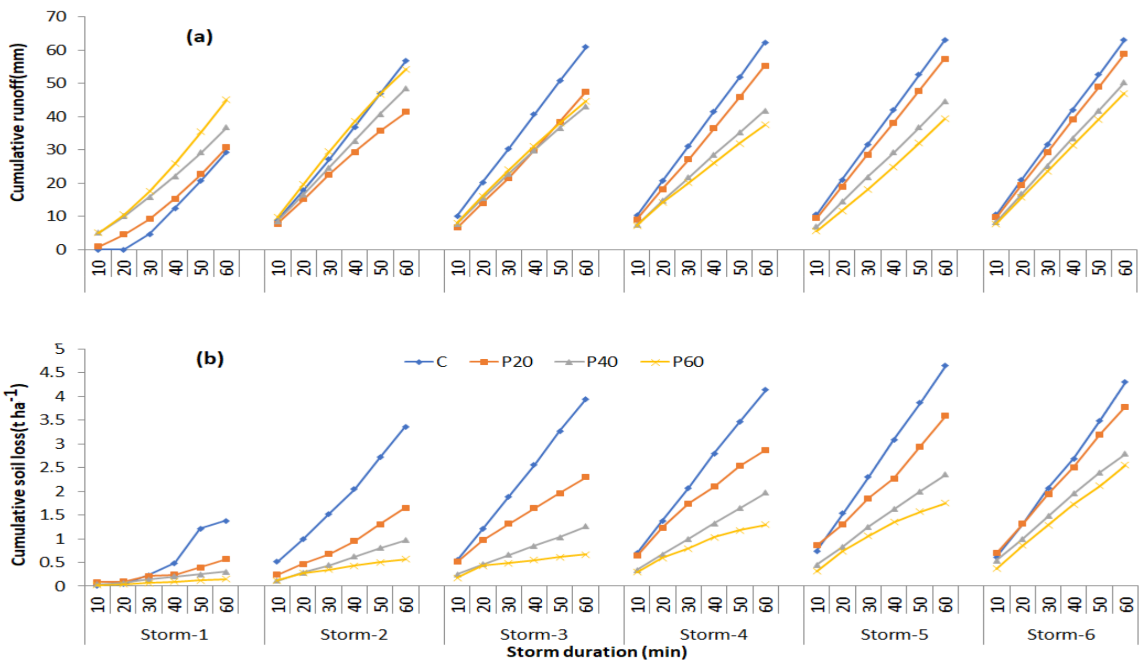

Figure 3. 4 Effect of PAM rates on (a) cumulative runoff (mm) and (b) cumulative soil loss (t ha^{-1}). Each of the 6 storms had an average rainfall intensity of 70 mm h^{-1} for 1-h duration and 48-h drying period interval. Data were collected at 10-min intervals in the 1-h rainfall duration. The treatments were control (C), PAM of 20 kg ha^{-1} (P20), 40 kg ha^{-1} (P40), and 60 kg ha^{-1} (P60).

Table 3. 2 Total runoff, soil loss and percent reduction (%), compared to control, from soils treated with different PAM rates and exposed to six consecutive rainfall storms of 1-h duration and 70 mm h^{-1} intensity. The treatments were control (C), PAM of 20 kg ha^{-1} (P20), 40 kg ha^{-1} (P40), and 60 kg ha^{-1} (P60).

Treatments	Runoff		Soil loss	
	Total runoff (mm)	Reduction (%)	Total soil loss (t ha^{-1})	Reduction (%)
C	336		21.52	
P20	292	13	14.91	31
P40	265	21	9.19	57
P60	267	20	6.93	68

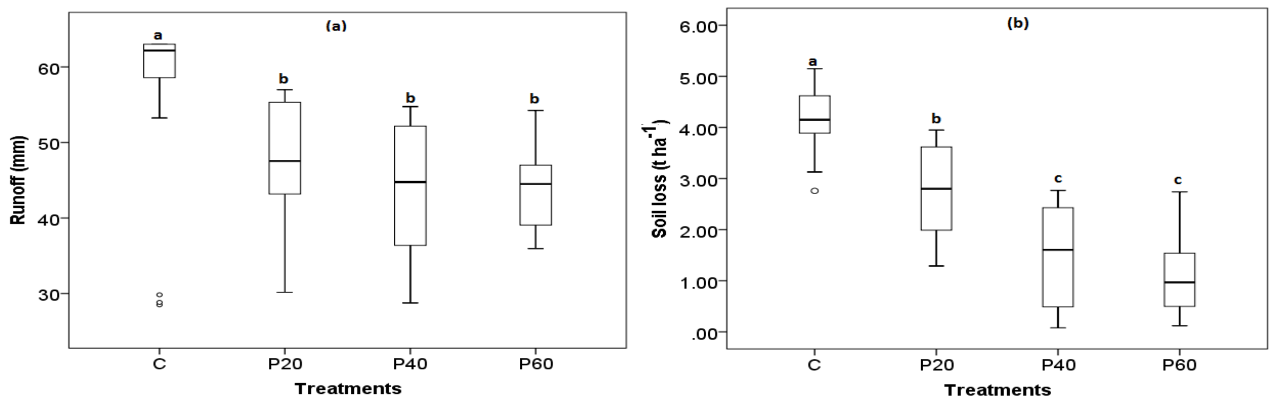

Figure 3. 5 Box-plot diagram of storm runoff (a) and soil loss (b) from six consecutive rainfall storms of 70mm h^{-1} using Kruskal-Wallis test ($P \leq 0.05$). The plots show the median, the 25th and 75th percentile scores (boxes) and the range outliers (dots). Different letters following treatments indicate statistically significant differences among treatments. The treatments were control (C), PAM of 20 kg ha^{-1} (P20), 40 kg ha^{-1} (P40), and 60 kg ha^{-1} (P60).

3.3.4. Effect of PAM, gypsum, and lime on infiltration rate (IR), runoff and soil loss

As P40 was determined as the effective rate in the first phase of the experiment, it was applied mixed with gypsum or lime at the same rate and subjected to the same number of storms as PAM treatments. During storm 1, the highest IR was observed for gypsum treatment (41 mm h^{-1}) i.e., an increase of 21% compared with the control while the lowest was observed for P40 (26 mm h^{-1}) with reduction of 24%. During storms 2 to 6, as the rain progressed, IR continued to decrease for all the treatments, except P40 in the order P+L > P+G > G or L > C (Figure 3.6).

However, IR for P40 treatment sharply increased during storm 3 and reached its maximum (28 mm h^{-1}, 268% increment) during storm 4, however, it continuously dropped during storms 5 and 6. Thus, for storms 4 to 6, IR decreased in the order P40 > P+L > P+G > G or L > C, with respective final IR increment of 182, 100, 63, 9, and 4% (Figure 3.6). Due to the reduced IR during storm 1, runoff from P40 increased by 28% during storm 1 but decreased up to 33% in the subsequent storms. For other treatments, reduction in cumulative runoff ranged between 8–37% for P+L, 7–28% for P+G, 1–24% for gypsum, and 0–11% for lime (Figure 3.7a).

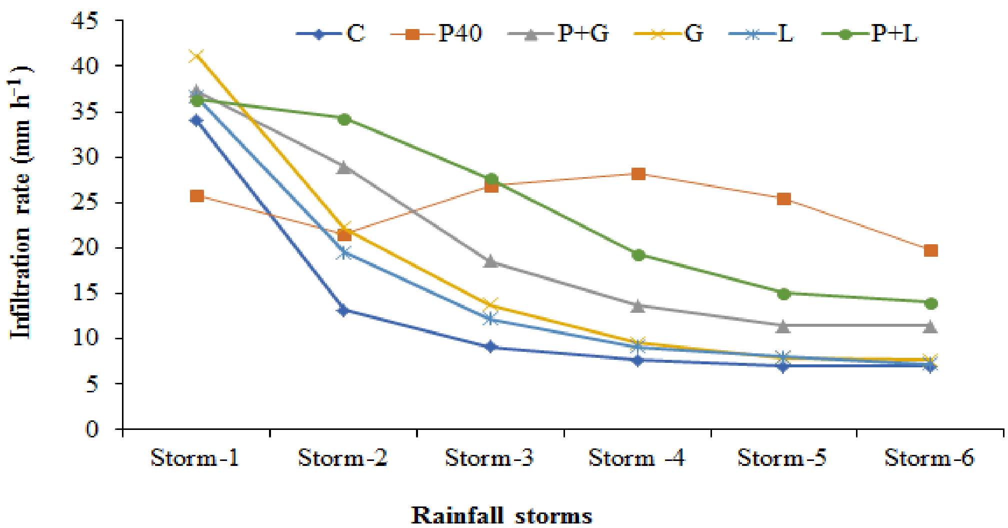

Figure 3. 6 Effect of treatments on infiltration rate (mm h^{-1}). Each of the six storms had an average rainfall intensity of 70 mm h^{-1} for 1-h duration and 48 h drying period interval. Treatments were control (C), 40 kg ha^{-1} (P40), PAM of 40 kg ha^{-1} + Gypsum (4t ha^{-1}) (P+G), Gypsum (4t ha^{-1}) (G), lime (2t ha^{-1}) (L), and PAM of 40 kg ha^{-1} + Lime (2t ha^{-1}) (P+L).

On the other hand, the amount of cumulative soil loss consistently increased in the consecutive storms in the order of P40 < P+L < P+G < G < L < C, with percentage reduction ranging between 35–78% for P40, 20–56% for P+L, 11–48% for P+G, 8–44% for gypsum, and 2–26% for lime, compared with the control (Figure 3.7b). At the end of six consecutive rainfall storms, both total runoff and soil loss increased in the order P40 < P+L < P+G < G < L < C (Table 3.3). The boxplot statistical analysis using Kruskal-Wallis test ($P \leq 0.05$) revealed that P40 and P+L treatments reduced median storm runoff and soil loss significantly, as compared to the control (Figure 3.8)

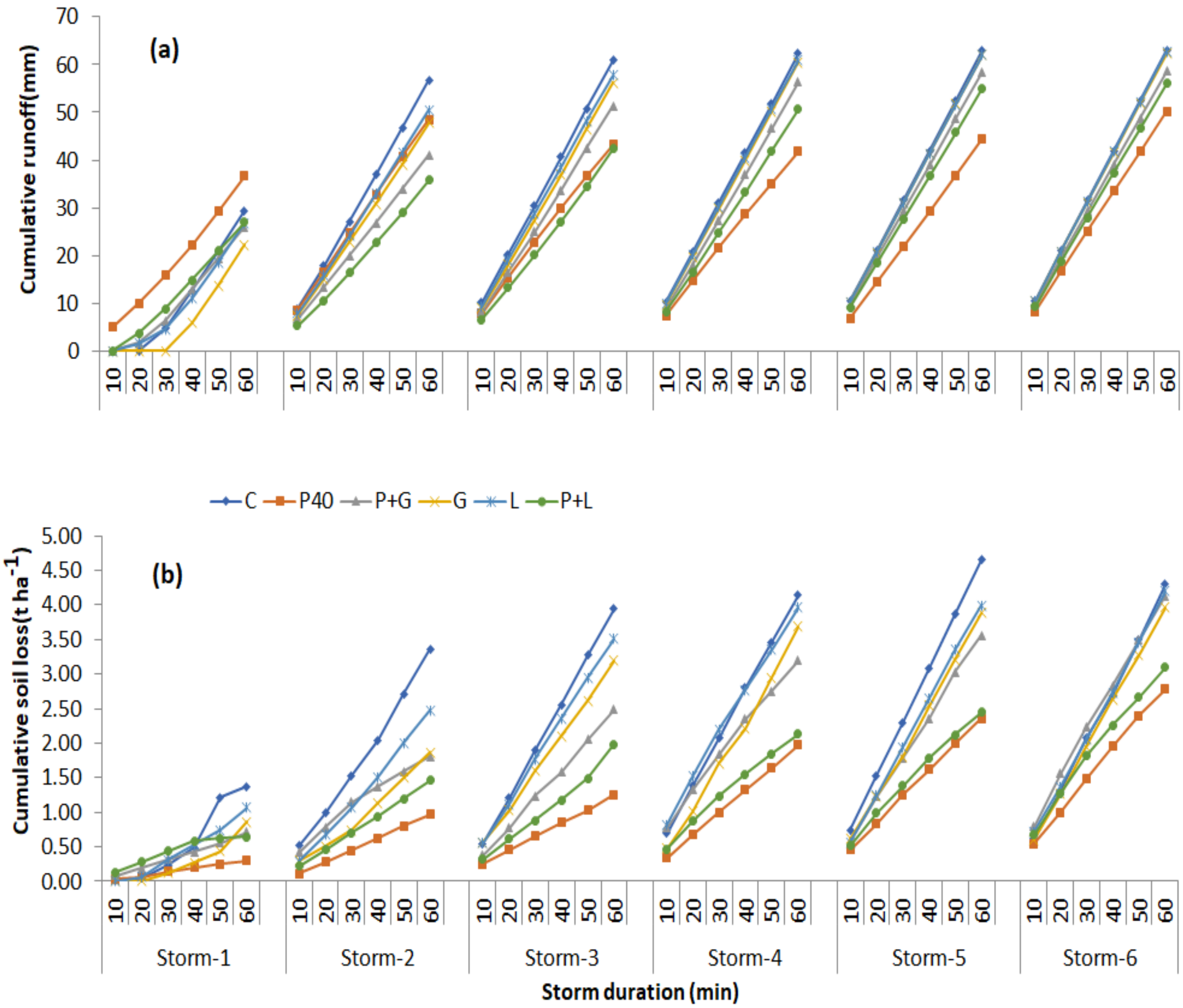

Figure 3. 7 Effect of treatments on (a) cumulative runoff (mm) and (b) cumulative soil loss (t ha^{-1}). Each of the six storms had an average rainfall intensity of 70 mm h^{-1} for 1-h duration and 48 h drying period interval. Treatments were control (C), 40 kg ha^{-1} (P40), PAM of 40 kg ha^{-1} + Gypsum (4t ha^{-1}) (P+G), Gypsum (4t ha^{-1}) (G), lime (2t ha^{-1}) (L), and PAM of 40 kg ha^{-1} + Lime (2t ha^{-1}) (P+L).

Table 3. 3 Total runoff, soil loss and percent reduction (%), compared to control, from soils treated with different soil amendments and exposed to six consecutive rainfall storms of 1hr duration and 70 mm h^{-1} intensity. Treatments were control (C), 40 kg ha^{-1} (P40), PAM of 40 kg ha^{-1} + Gypsum (4t ha^{-1}) (P+G), Gypsum (4t ha^{-1}) (G), lime (2t ha^{-1}) (L), and PAM of 40 kg ha^{-1} + Lime (2t ha^{-1}) (P+L).

Treatments	Runoff		Soil loss	
	Total runoff (mm)	Reduction (%)	Total soil loss (t ha^{-1})	Reduction (%)
C	336		21.52	
P40	265	21	9.19	57
P+G	291	13	16.16	25
G	313	7	17.96	17
L	322	4	19.08	11
P+L	267	20	11.53	46

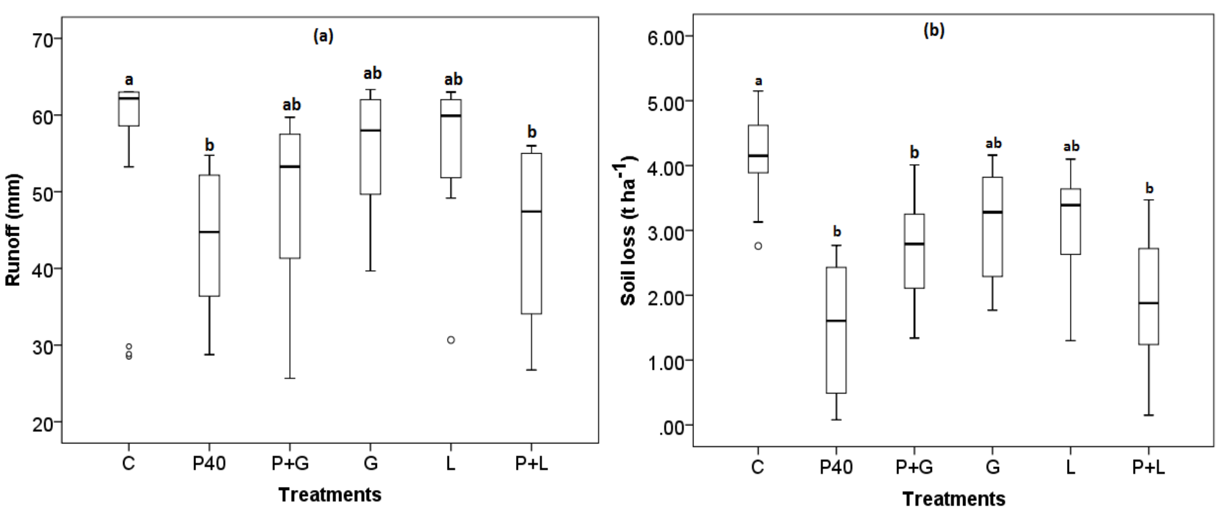

Figure 3. 8 Box-plot diagram of storm runoff (a) and soil loss (b) from six consecutive rainfall storms of 70 mm h^{-1} using Kruskal-Wallis test ($P \leq 0.05$). The plots show the median, the 25th and 75th percentile scores (boxes) and the range outliers (dots). Different letters following treatments indicate statistically significant differences among treatments. Treatments were control (C), 40 kg ha^{-1} (P40), PAM of 40 kg ha^{-1} + Gypsum (4t ha^{-1}) (P+G), Gypsum (4t ha^{-1}) (G), lime (2t ha^{-1}) (L), and PAM of 40 kg ha^{-1} + Lime (2t ha^{-1}) (P+L).

3.3.5. Effect of aggregate and structure stability on runoff and soil loss

Aggregate and structure stability, as described by SI, is an index that indicates the status of the soil aggregates in each treatment at the end of the experiment. After 6 consecutive rainfall storms, SI in P60 increased by 26% while it increased by 21% both in P40 and P+L compared with the control. The SI for all other treatments increased by less than 17% (Figure 3.9). In addition, both the total runoff and total soil loss showed a strong negative linear relationship with SI ($R^2 > 0.9$) (Figure 3.9).

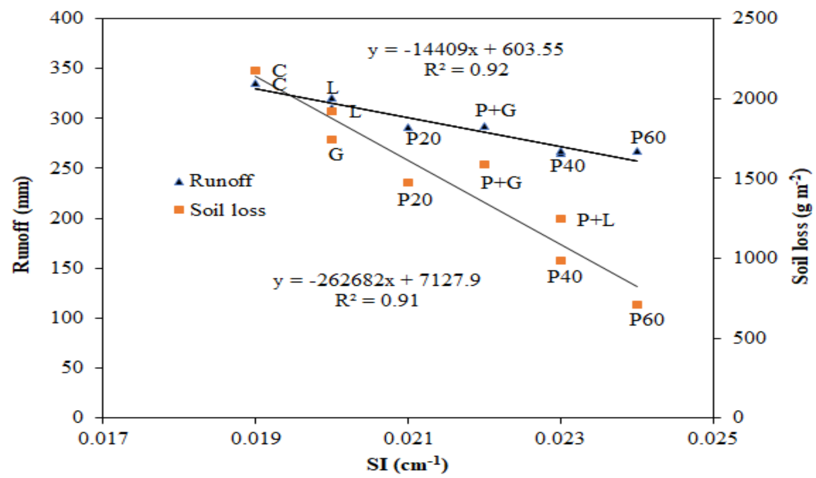

Figure 3. 9 Relationship between structural index (SI) at the end of simulation with runoff (triangle) and soil loss (square).

3.4. Discussion

3.4.1. Effect of PAM rates on TRO, IR, runoff, and soil loss

The shorter TRO from PAM treatments, when compared with the control, was because of the high viscosity caused by the dissolution of PAM granules that led to a decrease in the IR and an increase in runoff (Sojka et al., 2007; Ajwa and Trout, 2006). Our result is consistent with the report by Lee et al. (2010) who measured TRO for a silt loam soil amended with three PAM rates (0, 20, and 40 kg ha^{-1}), at a slope of 10, 20, and 40%. They found that the application of PAM (P20 and P40) decreased TRO for the 10% slope by an average of 5%.

During storm 1, IR for all the PAM rates reduced, compared with the control (Figure 3.3). Nevertheless, IR from control was significantly reduced and became the lowest through storms 2 to 6. This result is in agreement with observations by Inbar et al. (2015), who used different PAM rates and consecutive rainstorms to mitigate post-fire soil erosion. They reported that the IR for control was higher during the first storm compared with the PAM treated setup. However, the effect of PAM treatments on IR and runoff in this study was not consistent. The application of granular PAM at higher rates (P40 and P60) decreased IR and increased runoff during storms 1 and 2, compared with the control. However, IR for these rates substantially increased during storms 3 and 4 while IR from lower rate (P20) decreased continuously through storms 1 to 6. The mechanism responsible for decreasing IR and increasing runoff for the PAM treatments during the initial storm events was not the soil surface seal formation, as is the case with the control, but the large viscosity in the soil solution produced by the dissolution of PAM granules Inbar et al. (2014). On the other hand, the reduction in IR from control during storms 2 to 6 was due to the breakdown of aggregates by raindrop impact and subsequent seal formation (Ben-Hur, 2008).

On PAM dissolution during a rainfall event, the dissolved molecules are partially sorbed on the soil clay particles and improve aggregate stability, thereby increasing their resistance to detachment. Moreover, the non-sorbed segments of the molecules extend into the pores and drag the infiltrating water (Abrol, et al, 2013), decreasing the hydraulic conductivity of the soil and thus the infiltration rate (Yu et al., 2003). However, the effect of viscosity disappeared after drying cycles, when the formation of bonds between PAM and soil particles was favored (Lado et al., 2015) and the positive effect of PAM applied at higher rates were enhanced after two raining and drying cycles that reversed the situation and sharply increased IR and decreased runoff later during storms 3 and 4. The decrease in viscosity of the percolating solution together with the stabilization of aggregates through wetting and drying periods granules Inbar et al. (2014) led to an increase in the IR and a decrease in runoff during storms 3 and 4 (Sojka et al, 2007; Ajwa and Trout, 2006). The IR, however, continuously declined during storms 5 and 6, mainly due to washing out of PAM by runoff in the consecutive storms Inbar et al. (2014). Furthermore, there was an increased reduction in IR with an increasing PAM rate at the beginning of the simulation; however, this reduction was diminished with increasing PAM rate at the latter stage of the simulation (Figure 3.3). This implies that higher PAM rates could either be applied as a split application or each of the first two rainfall storms could be applied in two or more applications with fair drying period intervals to minimize the excess runoff at the beginning.

Despite the higher runoff volume during storms 1 and 2 as a result of high viscosity, higher PAM rates were effective in reducing soil loss throughout the consecutive storms. Similarly, Abrol, et al. (2013) reported that soil erosion decreases with an increase in viscosity of runoff in spite of the increase in runoff volume. Likewise, Inbar et al. (2014) observed that increased viscosity of runoff reduces flow velocity and shear or drag forces that can detach soil particles. Furthermore, unlike the cumulative runoff and IR, cumulative soil losses for PAM treatments consistently decreased with increasing PAM rate but increased with time during the consecutive storm events. The reduction in soil loss with increasing PAM rate in this study is consistent with the results of Lado et al. (2015) who reported that the relative viscosity of runoff increases with increasing PAM rate, thus decreasing runoff erosivity and hence soil loss. The consistent reduction in soil loss with increasing rates of PAM could also be attributed to the increased positive effect of PAM in preserving the soil aggregate structure (Tümsavas and Kara, 2011) by increasing the soil erodibility resistance even during the last storm, which is in accordance with the reports by other researchers (Sepaskhah and Bazrafshan-Jahromi, 2006; Sepaskhah and Mahdi-Hosseinabadi, 2008).

Similarly, the increase in soil loss for all PAM rates in the subsequent storms can be explained by the loss of effectiveness of PAM with time (e.g. dissolution, washing, and leaching), causing a decrease in IR and an increase in runoff, thus leading to increased soil loss during the last storms. The results from our study agree with the results of (Li et al., 2011) who reported that the cumulative soil loss increases with increasing rainfall duration for each PAM rate because PAM effectiveness diminishes with time (Lee et al., 2011; Lado et al., 2015). Furthermore, the lower PAM rate losses its effectiveness faster towards the end of the simulation when compared with the higher rates which could be due to the washout of PAM by runoff in the consecutive storms (Inbar et al., 2014; Sepaskhah and Mahdi-Hosseinabadi, 2008). This leads to increased detachment and washed−in of the finer clay particles that plug the soil pores more at lower PAM rate (P20) because of the unavailability of sufficient PAM to protect the soil (Cochrane et al., 2005).

At the end of the simulation, it was observed that all the PAM rates were more effective at reducing total soil loss than total runoff, due to increased runoff observed at the beginning. This is in agreement with the observation made in a study by Yu et al. (2003), who reported that the PAM application has less influence on runoff than soil loss. Furthermore, P40 and P60 reduced the median cumulative runoff and soil loss significantly, compared with the control, however, there was no significant difference in reduction when compared with each other, implying that P40 was the most suitable application rate (cost-effective) in reducing both runoff and soil loss from the given test soil under consecutive storms (Table 3.2). Nevertheless, higher final IR (Figure 3.3) and increased aggregate stability (Figure 3.9) after the last storm indicated that the PAM rates, especially the higher rates (P40 and P60) were still effective to reduce runoff and erosion for some more storms. The following figure (Figure 3.10) shows crusting of soil surfaces after six consecutive storms.

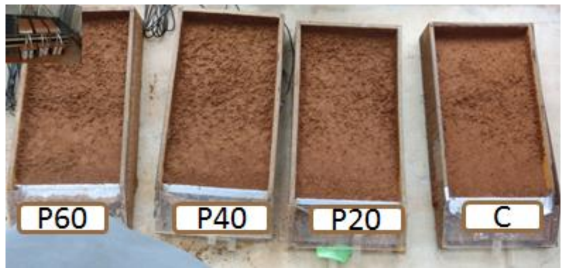

Figure 3. 10 Crusts formed on the soil surface after six consecutive storms. Treatments were control (C), PAM of 20 kg ha^{-1} (P20), 40 kg ha^{-1} (P40), and 60 kg ha^{-1} (P60).

3.4.2. Effect of gypsum, lime and their mixture with PAM on TRO, IR, runoff, and soil loss

The reduction in TRO for PAM mixed with gypsum or lime treatments, as compared with the control, was attributed to the increased viscosity from the dissolution of PAM granules that reduced IR and increased runoff. On the other hand, the long TRO from gypsum could be related to the high solubility of gypsum that interacts with the soil faster. Our result agrees with the finding of (Cochrane et al., 2005), who compared the effects of phosphogypsum (5 t ha^{-1}) and PAM (20 kg ha^{-1}) on TRO, runoff, and erosion using a highly weathered Brazilian Alfisol. They reported that compared with control, the application of gypsum was effective in delaying runoff by increasing the TRO by 75%, whereas PAM and P+G decreased TRO by 16% and 85%, respectively. They attributed the longer TRO from gypsum treatment, which led to increased IR and delayed runoff, to changes in soil surface chemistry and the shorter TRO in PAM treatment to increased viscosity of the runoff.

The IR, runoff, and soil loss from gypsum, lime, and PAM mixed with gypsum or lime amendments varied during the consecutive rainfall storms. In the beginning, IR for P40 treatment was lower than that for other treatments and the control (Figure 3.6), due to increased soil solution viscosity, as discussed in section 4.1. However, the effect of viscosity on the P40 solution started disappearing during storm 3, when there is enough rainfall for the complete dissolution of the PAM and drying period for irreversible bonding or sorption to the soil aggregates (Mamedovet al., 2009). This improved soil aggregates and structural stability, increased IR and decreased cumulative runoff from P40 treatment substantially during storms 4 to 6, compared with the control (Figure 3.6, 3.7a). Despite the high runoff during storms 1 and 2 from P40, cumulative soil loss from P40 was the lowest of all the treatments throughout the consecutive storms (Figure 3.7b), because the increased viscosity reduced the erosivity of the runoff at the beginning Inbar et al. (2014). But when the viscosity decreased latter in the consecutive storms, PAM becomes adsorbed to and binds soil particles through cation bridging, thus, increasing aggregate stability and cohesion strength between the soil particles and decreasing soil erosion (Roa-Espinosa, 2000). Thus, depending on the treatments and storm period, amendments affected surface soil aggregates and structural stability differently (e.g., resistant to disintegration and macro porosity).

Both gypsum and lime treatments increased IR, compared with the control (Figure 3.6). The higher IR from gypsum at the beginning could be due to the high solubility of gypsum that it is likely to dissolve, release Ca^{2+} cations and interact with the soil at a faster rate than lime (Bennett, and Cattle, 2010). On the other hand, the lower IR from lime implied the occurrence of higher dispersion from the lime treatment that sealed soil pores and decreased IR in the subsequent storms. As a result, runoff and soil loss from lime was higher than that from gypsum, and this difference in the two treatments could be explained by the varying effects of the two amendments on soil properties, mainly soil pH (Table 3.1).

Upon dissolution, lime dissociates into calcium and carbonate ions. The carbonate combines with hydrogen ions from the soil to form carbon dioxide and water. The removal of hydrogen ions from the soil in the form of water due to lime addition leads to an increase in negative charges in the soil while raising soil pH; hence the repulsive forces dominate between the soil particles and lead to dispersion (lime (Bennett, and Cattle, 2010; Tama and El Swaify, 1978) that seal soil pores and decrease IR. Our result is in agreement with the findings of de Castro (1988) who studied the effect of gypsum and lime on clay dispersion and infiltration in Oxisols. They found that incorporating lime to the soil increased clay dispersion and reduced IR significantly than incorporating gypsum. Furthermore, the removal of hydrogen ions following the addition of lime to the soil in our study increased the pH of the Oxisols from 4.84 to 6.75 (Table 3.1). The increment in the soil pH of the Oxisols from lime addition might have also increased soil dispersion leading to a decrease in IR compared with the gypsum treatment that had no effect on the soil pH (Table 3.1). Similarly, Roloff (1987) demonstrated the rise in pH values of Oxidic soils leads to clay dispersion that seals pores in the soil surface and decreases IR. In addition, de Castro (1988) has also highlighted a negative correlation between pH values and infiltration rate in acidic Oxisols i.e., the IR was found to decrease significantly with increasing lime application rates.

In the case of P40 mixed with gypsum or lime treatments, increased cumulative runoff and cumulative soil loss were observed, compared with P40 alone treatments (Figure 3.7a, b). This can be explained by the increase in cations in the soil solution with the addition of gypsum or lime mixed with P40 promoted coiling of the dissolved PAM molecules and reduce the interaction of the charged functional groups with soil particles (Lado et al., 2015). This reduces PAM effect on aggregate stabilization, increase surface seal formation, runoff and soil loss.

On the other hand, the presence of a higher concentration of electrolytes in P + G treatment, (4 t ha^{-1} of gypsum) compared with that in P + L (2 t ha^{-1} of lime), shortened the polymer chains and prevented the polymers from stretching and long-range bridging between the soil particles. This makes P + G less efficient in binding soil particles that are far apart and maintain soil aggregates, thus leading to increased soil erosion (Yu et al., 2003). Hence, P + L was more effective than P + G in reducing both runoff and soil loss whereas both are less effective compared with P40 treatment. Furthermore, P + G and P + L increased IR and decreased cumulative runoff and cumulative soil loss, compared with treatments using gypsum or lime alone (Figure 3.7a, b). This implies that gypsum or lime was more effective for Oxisols when applied in mixture with PAM than when applied alone for Oxisols. Furthermore, the high clay content in the Oxisols could also be another factor that maintained the effectiveness of PAM alone treatment in the Oxisols (Tumsavas, Z.; Kara, 2011).

The result of our study partially agrees with the study by Lepore et al. (2009). They reported that soil loss reduction in silt loam by P + G (58%) was lower than P + L (67%) while gypsum and lime reduced soil loss only by 18% and 30%, respectively. Our result was in agreement with that study in terms of highlighting that gypsum and lime were more effective when applied in mixture with PAM than when applied alone. Also, it showed that P + L was more effective than P + G in reducing soil loss. However, contrary to that study, our result showed that gypsum was less effective than lime when applied individually. The difference in the results between the two studies could be attributed to differences in soil properties (soil type, clay mineralogy), amendment rates applied (Teo et al., 2006) and their effect on aggregates and structural stability associated with treatments, rainfall duration, and wetting-drying condition in the two experiments. Yu et al. (2003) also observed that the use of PAM mixed with gypsum increases the final infiltration rate and reduces runoff and wash erosion than gypsum alone application in loamy sand and clay soils. Furthermore, Yu et al. (2003) reported that application of 20 kg ha^{-1} PAM is the most effective in reducing soil losses for the silt loam and sandy clay soils compared with gypsum (2 t ha^{-1} and 4 t ha^{-1}) or 20 kg ha^{-1} PAM mixed with gypsum (2 t ha^{-1} and 4 t ha^{-1}), in spite of low IR and the resultant high runoff.

The SI of treatments after six consecutive storms also indicated that P60 increased SI by 26% while both P40 and P + L treatments increased SI by 21%, however, SI for other treatments remains lower than 17% (Figure 3.9), supporting the results of the effect of different treatments on runoff and soil loss. For all the eight treatments (control, PAM, G, L and their combinations) there was a negative linear relationship between soil SI and soil loss or runoff with coefficient of determination ($R^2 > 0.9$) (Figure 3.9), showing that (i) both the runoff and soil loss were significantly affected by treatments and (ii) the steeper slope for soil loss versus SI implies that soil loss was more sensitive to SI than runoff. Therefore, such an approach could be used for the evaluation of the effect of amendments on soil status to resist erosion under consecutive rainfall storms.

3.5. Conclusions

In this study, we tested the effectiveness of different PAM rates in reducing runoff and soil loss under consecutive rainfall storms and selected the effective PAM rate for fine-textured Oxisols. The application of PAM at a rate of 20 kg ha^{-1} (i.e., P20) was more effective in reducing runoff during the first two storms and higher rates (i.e., P40 and P60) were more effective towards the end of the consecutive storms. However, the effectiveness of PAM in reducing soil loss increased with increasing PAM rate but diminished with time over the entire duration of the simulation. Furthermore, the application of PAM leads to a better reduction in soil loss compared to runoff. Our results also revealed that reductions in runoff and soil loss by P40 and P60 were not statistically different. Hence, by taking into consideration the price and application cost, P40 was selected as the most appropriate application rate for the given test soil. The selected PAM rate was further tested by applying it mixed with sources of electrolytes (gypsum and lime), but the concurrent application of P40 with gypsum or lime was less effective in reducing runoff and soil loss than P40 alone treatments. However, as only one application rate of gypsum and lime was considered, the result from this study may not be conclusive and further tests using multiple rates of gypsum or lime mixed with P40 and wetting-drying experimental conditions are recommended. Furthermore, the split application of P40 and evaluation of its effectiveness in reducing runoff against its gross application may provide additional knowledge about the effective PAM application rate.

Chapter 4: Effectiveness of Polyacrylamide integrated with other soil amendments in reducing runoff, soil loss, and RUSLE's *C*-factor: case Study from Northwest Ethiopia

4.1. Introduction

Soil erosion is a major global environmental problem with on-site and off-site consequences that include removal of fertile topsoil and degradation of land productivity (Wickama et al., 2014), river siltation, and water pollution (Haregeweyn et al., 2006, 2008). Soil erosion and land degradation have become severe threats to food security for developing countries like Ethiopia, where the livelihood of most of the population depends predominantly on agriculture (Haregeweyn et al., 2012,2015). Although the Ethiopian government has made large and successful efforts in the last 30 years to conserve soil and water resources, soil degradation remains a severe problem (Adimassu et al., 2014).

In recent years, the use of synthetic polymers, such as Polyacrylamide (PAM), has become an emerging alternative to conservation practices to reduce soil erosion. Anionic PAM is an environment-friendly soil conditioner applied mainly to improve soil aggregate stability, increase infiltration rates, and reduce runoff and soil loss (Sojka et al., 2007). It is adsorbed mostly on the external surfaces of soil aggregates (Mamedov et al., 2009) and binds soil particles that are far apart (Ben-Hur and Keren, 1997). This increases the stability of soil aggregates and their resistance to splash erosion by raindrop impact and detachment by runoff (Yu., 2003), thus enhances infiltration rates while reducing runoff and soil loss (Kebede et al., 2020). The effectiveness of PAM, however, depends on soil properties such as clay content and mineralogy, soil type, level of organic matter and electrolytes (e.g. Ca^{2+}) in the soil solution, and application methods (Sojka et al., 2007). Hence, applying PAM in mixture with sources of calcium electrolyte (Ca^{2+}), such as gypsum or lime, enhances its effectiveness in reducing runoff and soil loss, consequently improving soil aggregation (Lee et al., 2010). In addition, liming increases soil pH, decreases soil acidity, and benefits plant growth in acidic environments (Gabriel et al., 2018).

Studies have indicated that PAM alone or mixed with gypsum or lime reduces runoff and soil loss to differing degrees. For example, a field study by Kumar & Saha, (2011) showed that PAM or gypsum alone reduces soil loss by 7% and 64%, respectively, whereas PAM combined with gypsum reduces soil loss by 67%. Conversely, a recent study by Kebede et al., (2020) indicated that application of PAM alone reduces runoff and soil loss more than PAM mixed with gypsum or lime in Oxisols in a humid region. Similarly, Yu et al., (2003) reported that application of PAM combined with gypsum was less effective than PAM alone in reducing soil loss from silty loam and clay soils in semi-arid regions. PAM has also been shown to contribute to water conservation in soils by buffering the root zone against water loss in dryland regions with frequent drought periods (Letey et al., 1992).

PAM combined with organic soil amendments, such as biochar, has also been tested as a measure to improve soil quality and reduce soil erosion. Biochar improves soil aggregate stability and water holding capacity (Ravindran et al., 2019), as well as nutrient retention and soil fertility (Lee, et al., 2015; Saffari et al., 2020). Application of biochar, alone or combined with PAM, has been shown to reduce both runoff and soil loss (Saffari et al., 2020; Abrol et al., 2016). A filed study by (Li et al., 2017b) indicated that biochar reduces annual runoff by 19–28% and soil loss by 11% (Li et al., 2017a). This finding implies that biochar may play a vital role in improving infiltration, but there is also risk of increasing soil loss under tillage condition, particularly in sloping areas. Hence, the authors of that study suggested that biochar may need to be applied in combination with other amendments such as PAM (Li et al., 2017a).

A related factor is that cultivation of teff (*Eragrostis tef*) together with soil amendments may also affect the numerical value of the cover and management (C)-factor in the Revised Universal Soil Loss Equation (RUSLE). The *C*-factor quantifies the effects of canopy cover, surface cover, and surface roughness as well as management practices related to plant residues, consolidation changes, and soil disturbance on erosion (Karpilo, Jr. & Toy, 2004). If *C*-factor values can be reliably tied to amendment practices, that would improve the predictions of the RUSLE model for plots treated with soil amendments.

Reviews of the literatures show that application of PAM, alone or mixed with other amendments, are effective in reducing runoff and soil loss from croplands. However, the studies have involved different environmental and management conditions, raising the need for additional investigations in actual field conditions. Moreover, a comprehensive study has not been conducted in natural field conditions at Ethiopian localities in the tropical highland climate zone, such as the Upper Blue Nile basin in northwestern Ethiopia. Such an evaluation is essential before PAM can be adopted as a soil conditioner on a large scale. The main objective of this study was, therefore, to determine the effectiveness of PAM, alone or integrated with other amendments (gypsum, lime or biochar), in reducing runoff and soil loss. We also sought to evaluate the effect of these amendments on soil physico-chemical properties, teff biomass yield, and the *C*-factor of RUSLE.

4.2. Materials and methods

4.2.1. Study area

The study took place over two growing seasons (2018 and 2019) in plots located at 11°39′08″N, 37°29′40″E at an elevation of 2000 m in the Aba Gerima watershed in the Upper Blue Nile basin of Ethiopia (Figure 4.1). The study area is characterized by a sub-humid tropical climate with a mean annual rainfall of 1343 mm, of which 80% occurs between June and September, and mean monthly temperatures ranging between 17 and 23 °C. According to the FAO classification system.

Luvisol is the dominant soil type in the study area, which is suitable for growing teff, finger millet (*Eleusine coracana*), and maize (*Zea mays*) as major cereal crops (Ebabu et al., 2018).

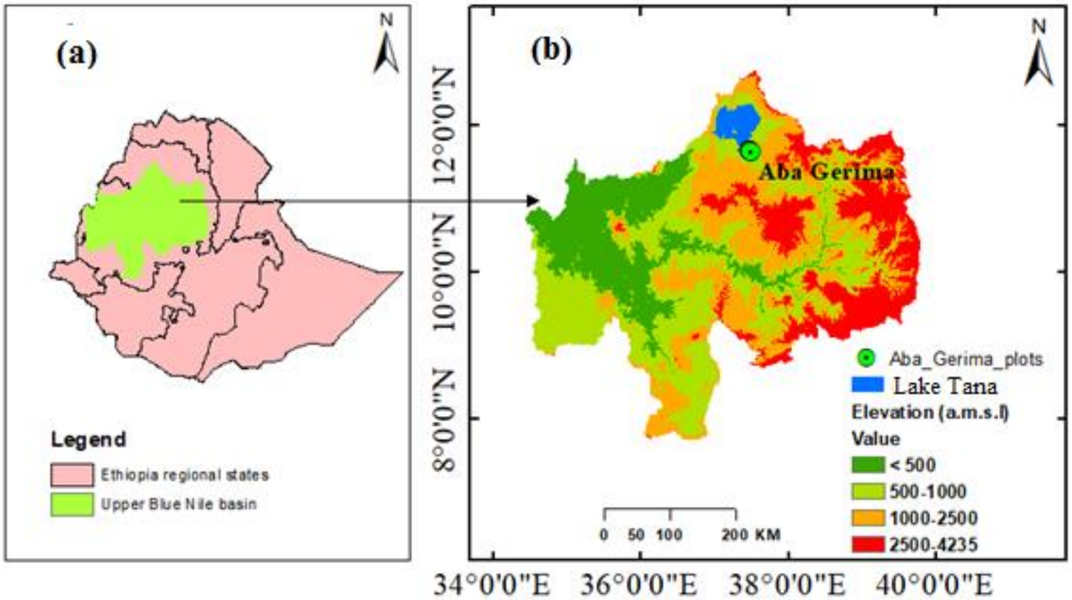

Figure 4. 1 Map of the study area. (a) Location of the Upper Blue Nile basin in Ethiopia, (b) Location of the study plots in the Upper Blue Nile basin.

4.2.2. Experimental design

Experimental plots were established on cropland measuring 4 m by 1.3 m at a slope gradient of 8% (Figure 4.2). Runoff collector trenches with trapezoidal cross sections (2.5 m long and 1 m wide on the top, 1.5 m long and 0.5 m wide on the bottom, and 0.6 m deep) were dug below each plot and lined with a geo-membrane plastic sheet to prevent water loss by infiltration (Figure 4.2). The perimeter of each plot was lined with a barrier of sheet metal 35 cm high driven 15 cm into the ground to bar runoff from outside the plot. Plots were fenced to prevent damage by grazing animals and to allow crop residues and soil organic matter to build up.

Four soil amendment types were applied at different rates: PAM (40 kg ha^{-1}, applied in split or divided in two applications), gypsum (5 t ha^{-1}), lime (4 t ha^{-1}) and biochar (8 t ha^{-1}). The biochar was obtained from commercial sellers of charcoal, produced from *Acacia decurrens* in the Guder watershed following local practices. The charcoal was crushed and passed through a 4 mm sieve to obtain relatively uniform particle sizes <4 mm across and to increase its surface area for better contact with the soil. Gypsum and lime were bought from local distributers. PAM (Superfloc A-110, granular powder, 10–12% hydrolysis, and 12 Mg mole^{-1} molecular weight) was obtained from the Kemira Oyj Company, Japan.

The experiment followed a randomized complete block design using eight treatments and three replications. All plots, including control, were planted with teff. The treatments were control (no soil amendment but planted with teff), PAM, gypsum (G), lime (L), biochar (B), PAM plus gypsum (P+G), PAM plus lime (P+L), and PAM plus biochar (P+B). In addition, one bare plot (with no teff and no amendments) was established for *C*-factor determination. Plots were tilled by manual hoeing three times each year: during the dry season (May), at the beginning of the rainy season (June), and in late July followed by teff sowing. In the 2018 season lime, gypsum, and biochar were applied in June following the second tillage and thoroughly mixed in the soil, and 20kg ha^{-1} of PAM was applied immediately afterward while the remaining 20kg ha^{-1} of PAM was applied in the third tillage during teff planting (Figure 4.2). In the 2019 season PAM alone was applied in split during the second tillage at the beginning of June and in late July after the third tillage, such that a total of 40 kg ha^{-1} was applied over the course of the experiment (Figure 4.2).

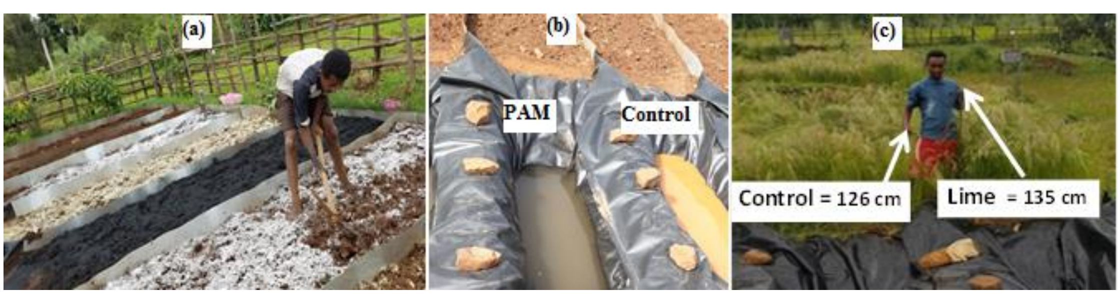

Figure 4. 2 Field runoff plots in the experimental site at different stages: (a) tillage and amendment application, (b) runoff data collection, and (c) teff crop during the grain filling stage**.**

4.2.3. Data collection and analysis

Daily rainfall was measured and runoff and sediment data were collected on a daily basis for all rainfall events that generated runoff during the rainy seasons of the study period: July to October in 2018 and June to September in 2019. Rainfall data were collected using an automatic tipping-bucket rain gauge (HOBO UA-003-64) installed at the experimental site. There were 133 days rainfall events during the study period (57 in 2018 and 76 in 2019), in which 79 larger rainfall events (40 events in 2018 and 39 events in 2019) generated runoff. Runoff and sediment concentration data were collected daily at 7 a.m. whenever rainfall events generated runoff. The runoff water in the collector trench was first vigorously agitated with floor brushes until the settled sediment was suspended in the water, then 0.5 L of the runoff water was collected for determination of the sediment concentration. The trenches were then emptied manually using plastic buckets of known volume to determine the runoff volume. The geo-membrane plastic sheets lining the trenches were then carefully cleaned and checked for any damage.

In the laboratory, the runoff samples were filtered through Whatman 42 filter paper, which was then oven-dried at 105 °C for 24 hr. The sediment concentration (g L^{-1}) was determined by weighing the dried sediment and filter paper together on a digital balance and subtracting the weight of the filter paper. The actual runoff volume generated from the plots was determined by subtracting the volume of rainfall falling directly into the trench from the total runoff volume. Daily soil loss was calculated as the product of the sediment concentration and the daily runoff volume, and seasonal soil loss was calculated by summing the daily soil loss records. The two year seasonal runoff and soil loss data were subjected to analysis of variance using IBM SPSS Statistics version 22 software and the Kruskal–Wallis non-parametric test, at a significance level of 0.95 ($\alpha = 0.05$). The *C*-factor, which quantifies the effect of amendments on soil loss reduction, was estimated as the ratio of the soil loss from the control or treated plot to the soil loss from the bare plot (Renard et al., 1997).

Soil moisture data were collected on a daily basis from June to November of 2019, using automatic moisture sensors, and aggregated to monthly averages. For this purpose, eight sensors connected to two moisture data loggers (EM43539 and EM43540) were installed at a depth of 20 cm to monitor change is soil moisture as influenced by treatments.

Soil samples from each plot were collected before and after the experiment. The first samples were composited and analyzed as baseline data. Soil texture was determined for the baseline soil data using the hydrometer method (Gee and Bauder, 1986) at Amhara Design and Supervision Works Enterprise (ADSWE), Bahir Dar, Ethiopia. All soil samples were transported to the Arid Land Research Center of Tottori University, Japan, where they were air-dried and used for determining soil pH (at 1:2.5 ratio), electrical conductivity (at 1:5 ratio), soil organic carbon by a C/N coder apparatus, and exchangeable cations by Atomic Absorption Spectroscopy (AAS) after extraction with ammonium acetate buffered at pH 7 (Table 3.1). The soil had a clay loam texture (32% sand, 33% silt, and 35% clay) with an organic matter (OM) content of 2.55%. To determine the aboveground biomass of the teff crop, the plants were harvested from three 0.5 by 0.5 m quadrants in each plot, sun dried, and weighed for total biomass with an adjustment to average moisture content (Table 3.2).

Soil samples for aggregate stability measurement were randomly taken from the surface layer of plots (0–5 cm) at the end of vegetation growth of the two study seasons (November). Soil aggregate stability, expressed in terms of the structural index (SI), was determined using the sensitive modified high energy moisture characteristics (HEMC) method (Levy and Mamedov, 2002). In this method, soil aggregates are wetted rapidly in a controlled manner and a water retention curve is generated using a hanging water column. SI is then determined from the differences among the water retention curves of the treatments by using their specific water capacity curves. SI, the ratio of the volume of drainable pores to modal suction, is used to characterize soil aggregate and structure stability. The higher the value of SI, the more stable the soil sample (Levy & Mamedov, 2002).

4.3. Results

4.3.1. Effects of treatments on soil attributes and crop biomass

Soil chemical attributes measured before and after two years of treatments are presented in Table 4.1. The results indicate that treatments involving lime and biochar (L, B, P+L, and P+B) appreciably increased soil pH, the highest being from plots treated with P+L followed by that with lime. Treatments involving biochar (B and P+B) substantially increased soil organic carbon and thus OM content. Treatments generally increased the remaining soil properties (electrical conductivity, exchangeable cations) to different degrees (Table 4.1).

Soil moisture was monitored daily for six months (from June to November 2019) and monthly averages were used for analysis (Figure 4.3a). All of the treated plots had mean soil moisture contents that were higher (by 8–59%) than the control plots (Figure 4.3b). The largest increments in average moisture content were from P+B and PAM treatments (59 and 49%, respectively) and the lowest was from gypsum treatment (8%). Soil moisture in all plots increased between June and October and dropped sharply in November (Figure 4.3a).

Table 4. 1 Chemical properties of soil samples collected before treatment (baseline) and after two years of experimental treatments. All treatments included sowing, weeding, and harvesting of teff. Treatments for plots other than the control plot included the indicated soil amendments. Measured soil parameters were: Soil pH, electrical conductivity (EC), exchangeable cations (Exch.) and organic matter (OM).

Treatment	pH	EC	Exch. Ca	Exch. Mg	Exch. Na	Exch. K	OM
	(H_2O)	(dS m^{-1})	(cmol(+) kg^{-1})				(%)
Baseline	5.34	0.06	9.64	2.4	1.29	0.32	2.55
Control	5.35	0.06	9.29	4.79	1.88	0.56	2.6
PAM (P)	5.38	0.06	9.59	5.18	1.82	0.68	2.79
Gypsum (G)	5.28	0.11	10.1	5.21	1.87	0.7	2.61
Lime (L)	5.91	0.08	10.72	4.76	1.88	0.51	2.82
Biochar (B)	5.89	0.06	10.45	5.03	1.91	0.64	3.16
P+G	5.34	0.09	10.82	4.89	1.93	0.69	2.73
P+L	6.05	0.08	11.23	4.92	1.92	0.74	2.86
P+B	5.73	0.05	10.58	5.19	1.85	0.62	3.22

All of the treatments resulted in higher SI values compared to the control (Figure 4.3b). The largest increment in SI resulted from PAM treatment (132%) and the smallest resulted from gypsum and biochar treatments (both 24%).

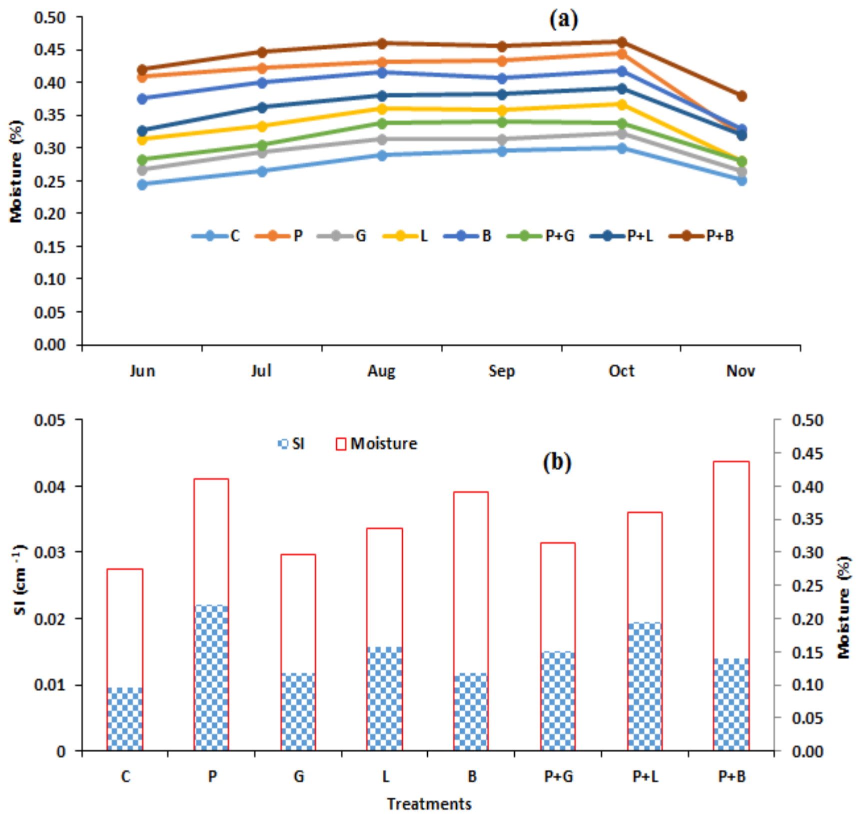

Figure 4. 3 (a) Monthly soil moisture and (b) average soil moisture content (red open bars) and soil structural index (SI, blue solid bars) under the experimental treatments. Treatments for plots other than the control plot (C) included soil amendment with PAM (P), gypsum (G), lime (L), biochar (B), PAM plus gypsum (P+G), PAM plus lime (P+L), and PAM plus biochar (P+B).

The total biomass measured in the experimental plots clearly varied with treatment types and study seasons (Table 4.2). Generally, the greatest increase in average biomass yield was in plots with P+B (33%) and P+L (28%) treatments and the smallest was in plots with P+G (5%) and gypsum (8%) treatments. The biomass yield was higher in the second season than in the first for all plots treated with soil amendments.

Table 4. 2 Biomass yield and relative increases over the control case from the different treatments during the two study seasons and the study period.

Treatment	2018		2019		Average	Increment (%)
	Biomass (t ha^{-1})	Increment (%)	Biomass (t ha^{-1})	Increment (%)		
Control	5.8		5.7		5.8	
PAM (P)	6.2	7	6.6	14	6.4	11
Gypsum (G)	6.3	9	6.2	8	6.2	8
Lime (L)	7.2	25	6.9	21	7.1	23
Biochar (B)	6.9	19	7.3	26	7.1	23
P+G	5.9	2	6.2	8	6.1	5
P+L	7.3	26	7.5	30	7.4	28
P+B	7.4	27	8.0	39	7.7	33

4.3.2. Effect of treatments on runoff, soil loss, and C-factor

The magnitudes of monthly runoff and soil loss clearly varied among treatments and between study seasons. Monthly runoff was higher in August in the first season (2018) and higher in June in the second season (2019) (Figure 4.4). However, monthly soil loss was generally higher at the beginning and decreased with time in both years for most treatments.

Seasonal runoff ranged from 293 to 403 mm in 2018 and from 378 to 623 mm in 2019. The treatments reduced runoff by 12–27% in 2018 and 15–39% in 2019 with respect to the control plots (Table 4.3). The greatest reductions in seasonal runoff resulted from treatment with PAM

(27% in 2018 and 32% in 2019) and P+B (27% in 2018 and 39% in 2019), and the smallest resulted from treatment with gypsum (12% in 2018 and 15% in 2019). The statistical test indicated that PAM and P+B only significantly reduced the seasonal runoff, as compared to the control (Figure 4. 5a).

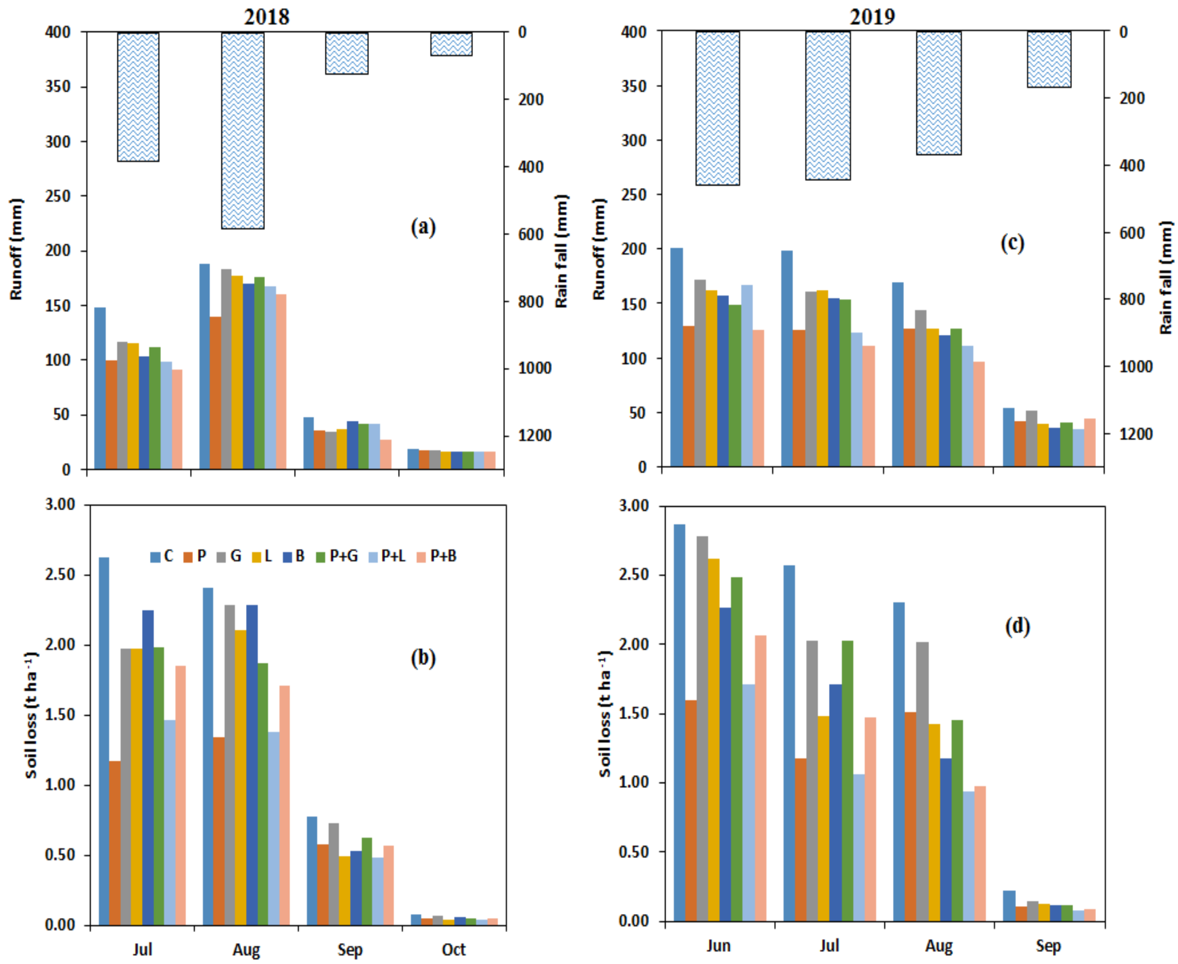

Figure 4. 4 Histogram plots showing the monthly relationships of (a) runoff (lower bars) to rainfall (upper bars) and (b) soil loss to treatments in 2018, (c) runoff to rainfall and (d) soil loss to treatments in 2019.

Table 4. 3 Seasonal runoff and soil loss, and their relative reduction, as compared to the control, for the two study periods. Teff crop was grown to all plots including the control and other management practices, such as wedding, remain same for all plots. (N = 79 events).

	2018				2019			
Treatment	Runoff (mm)	Reduction (%)	Soil loss (t ha^{-1})	Reduction (%)	Runoff (mm)	Reduction (%)	Soil loss (t ha^{-1})	Reduction (%)
Control	403		5.89		623		7.95	
PAM (P)	293	27	3.13	47	424	32	4.42	44
Gypsum (G)	354	12	5.06	14	529	15	6.94	13
Lime (L)	348	13	4.61	22	490	21	5.57	30
Biochar (B)	334	17	5.13	13	470	25	5.2	35
P+G	345	14	4.53	23	470	25	6.03	24
P+L	326	19	3.37	43	436	30	3.74	53
P+B	296	27	4.17	29	378	39	4.52	43

Seasonal soil loss from the different treatments ranged from 3.13 to 5.89 t ha^{-1} in 2018 (13–47% reduction from control) and from 3.74 to 7.95 t ha^{-1} in 2019 (13–53% reduction). The greatest reduction resulted from PAM (47%) and P+L (43%) in 2018 and from P+L (53%) and PAM (44%) in 2019, whereas the smallest reduction resulted from biochar (13%) in 2018 and gypsum (13%) in 2019 (Table 4.3). The seasonal soil losses from P+L, PAM and P+B were statistically significant, as compared to the control (Figure 4. 5b). The average SI and crop biomass had negative linear relationships with both median runoff and soil loss, but except for the relationship between SI and runoff (R2 = 0.76), these relationships were weak (Figure 4.6).

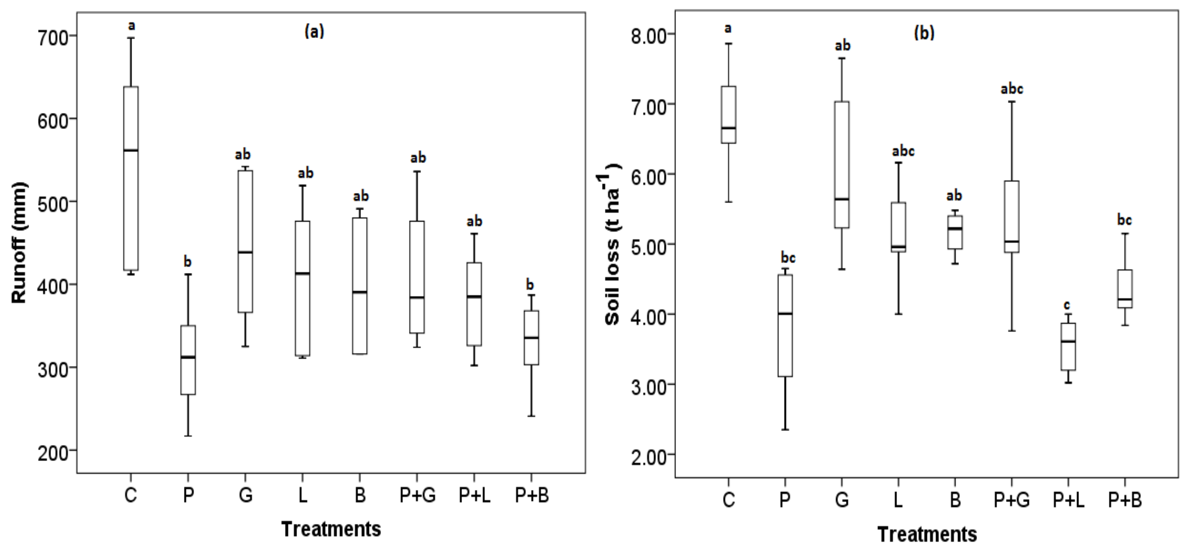

Figure 4. 5 Box-plot diagram of seasonal runoff (a) and soil loss (b). The Kruskal-Wallis test ($\alpha = 0.05$) was used when the data distribution was skewed. The different letters indicate statistically significant differences in seasonal runoff and soil loss among treatments. Treatments were control (C), PAM (P), gypsum (G), lime (L), biochar (B), PAM + gypsum (P+G), PAM + lime (P+L), and PAM + biochar (P+B). Teff crop was planted in all plots and other management practices, such as weeding, remain the same across plots. (N = 79)

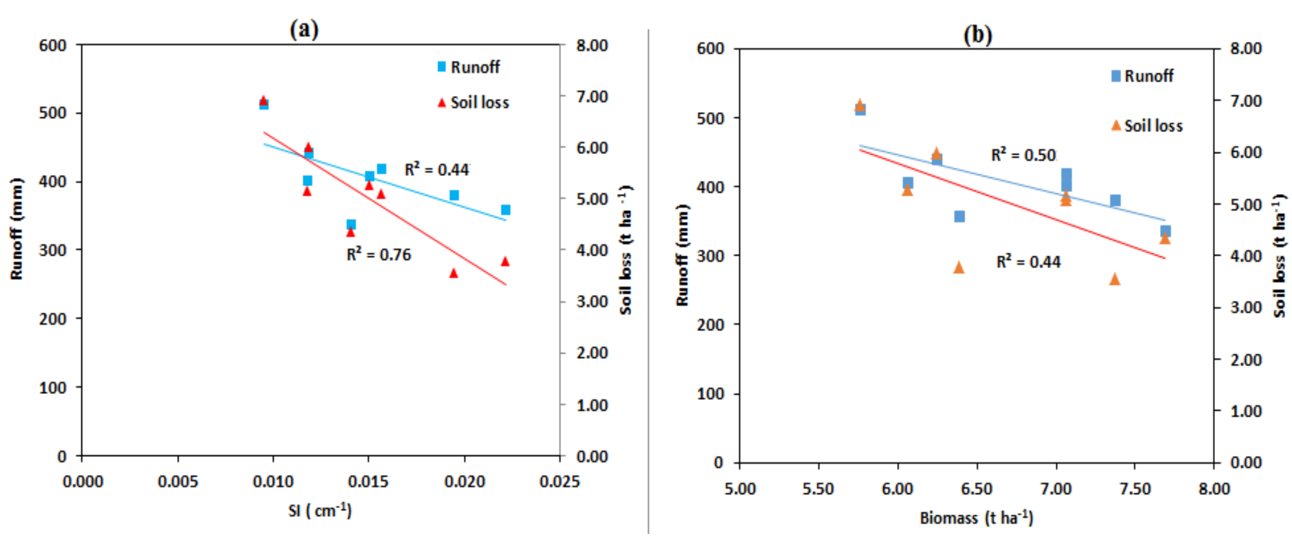

Figure 4. 6 Plots of median runoff (squares) and soil loss (triangles) against (a) average SI and (b) average biomass, including regression lines with their coefficients of determination (R^2).

The soil treatments reduced *C*-factor values by different amounts (Table 4.4). The two-season average *C* values for the different treatments ranged from 0.12 to 0.23, compared to the value of 1 for the bare plot. The lowest value (0.12) was from the P+L treatments and the highest (0.23) was from the control plot.

Table 4. 4 *C*-factor values for the different treatments.

	C-factor value		
Treatment	2018	2019	Mean
Control	0.22	0.24	0.23
PAM (P)	0.12	0.13	0.13
Gypsum (G)	0.19	0.21	0.20
Lime (L)	0.17	0.17	0.17
Biochar (B)	0.19	0.16	0.17
P+G	0.17	0.18	0.17
P+L	0.13	0.11	0.12
P+B	0.16	0.13	0.15

4.4 Discussion

4.4.1. Effect of treatments on soil attributes and plant biomass

The experimental treatments modified soil attributes, notably pH, OM, water retention (moisture content), and SI, and also increased plant biomass (Tables 4.1 and 4.2). The improved soil pH and OM were associated with a positive response to the amendments (mainly P+B and P+L), which reduces exchangeable aluminum (Al) and increases exchangeable Ca. This response in turn improves soil microbial activity, the stabilization and accumulation of soil organic carbon (Haynes & Naidu, 1998), and uptake of available nutrients by the roots of teff, all of which favor crop growth and increased biomass (Doan et al., 2015; Anikwe et al., 2016; Gabriel et al., 2018). The humus fraction of soil OM contributes to these benefits through pH buffering and increased water holding capacity (Haynes & Naidu, 1998).

The gypsum treatments (G and P+G) yielded lower increments in biomass ($\leq$ 8%), probably owing to the reduction in pH (Table 4.1). In acidic soils, the low pH increases the solubility of cations (Al^{3+} and Mn^{2+}) that can be toxic to plants (Anderson et al., 2013) and could reduce plant biomass. Our result is supported by Gabriel et al. (2018), who reported that a higher rate of gypsum application decreased the shoot dry weight of plants in acidic soil and attributed this result to higher electrical conductivity in the soil solution lowering nutrient uptake.

The treatments also improved soil structure (higher SI) and water retention (moisture content) compared to the control. PAM treatment contributes to SI by a cementing effect as the long-chain PAM molecules are adsorbed on to soil particles (mostly on their exterior surfaces) (Mamedov et al., 2007), which enhances soil aggregate and structure stability more than organic or inorganic amendments (Karami, et al., 2012; Sojka et al., 2007) (Figure 4.3). The biochar, P+B, PAM, and P+L treatments produced consistently higher soil moisture content compared to the control. PAM and lime are known to be effective in stabilizing soil aggregate structure (Mamedov et al., 2010; Paradelo et al., 2015) and increasing infiltration rate (Xiong et al., 2018), and biochar particles offer large surface areas for adsorbing water (Lee et al., 2015) that increases soil porosity and infiltration (Abrol et al., 2016). The smaller benefit in SI and soil moisture content from gypsum treatments (G and P+G) could be linked to the removal of Al, which binds soil particles in acidic soils, probably due to the higher rate of gypsum; this change would cause local dispersion, and thus reduce infiltration and available water retention, and also promote leaching of cations under intense rainfall (e.g., Kebede et al., 2020; Roth and Pavan, 1991; Ernani, et al, 2006). Our results are compatible with those of Nishimura et al. (2005) who reported that application of gypsum to acidic soil enhances dispersion and thus increases runoff and soil loss. The higher soil moisture from treated plots for the months June to October could also be due to the high rainfall, and the sharp drop in November could be due to teff harvesting and the cessation of rainfall that dries the soil.

4.4.2. Effect of treatments on runoff, soil loss, and C-factor values

The variability in monthly runoff and soil loss with different treatments may reflect variation in rainfall across months and the resulting changes in vegetation cover over the course of the growing season (Figure 4.4). The decreasing trend in runoff and soil loss towards the end of the season can be explained by the combination of decreasing rainfall and increased canopy cover (Ebabu et al., 2019).

Runoff and soil loss were slightly higher in 2019, could be due to the relatively higher rainfall in this season (Figure 4.4). However, most of the treatments were more effective in reducing runoff and soil loss than in 2018. The improved performance in 2019 may be attributed to factors such as the improved soil structure resulting from increased soil organic carbon (Table 4.1) and the increased microbial activity resulting from the amendments and crop residue (Li et al., 2017; Paradelo et al., 2015; Grandy, et al., 2002). Fencing and tillage in 2019 may also have facilitated incorporate amendments and crop residues into the root zone, thus increasing soil organic carbon and hence the OM content (Girmay et al., 2009) as well as the effectiveness of amendments (Caesar-Ton et al., 2008). Soil OM acts as a cementing agent and favors good aggregation and water infiltration (Lado, Paz, & Ben-Hur, 2004), preventing soil degradation by raising the soil's ability to retain water and nutrients (Li et al., 2017).

Treatments were more effective in reducing soil loss than reducing runoff, which may be related to plot size as well as the effects of crop cover associated with amendment applications. In small plots, the short slope length reduces flow accumulation and runoff erosivity, thus resulting in lower erosion (Bagarello et al., 2018; Erlangung, 2016). Changes in crop cover resulting from soil treatments may also reduce soil loss as vegetation dissipates the kinetic energy of raindrops (Ruiz-Colmenero et al., 2013), increases soil aggregation in the root zone, reduces soil crusting, and reduces splash erosion by raindrop impact (Ma et al., 2015). This finding is consistent with other studies showing that amendments (PAM, gypsum, and lime) reduce soil loss to a greater extent than they reduce runoff (kebede et al., 2020). Treatments reduced both seasonal runoff and soil loss compared to the control, but no statistically significant difference was noted among most treatments (Figure 4.5). The greatest reduction in runoff was from PAM and P+B treatments (27 and 39%), and the greatest reduction in soil loss was from PAM and P+L treatments (43 and 53%) (Tables 4.3). This implies that application of PAM alone or mixed with other amendments was generally effective in reducing both runoff and soil loss.

The efficacy of PAM in reducing runoff and soil loss can be explained by its ready adsorption onto soil aggregates (Mamedov et al., 2009) and its binding effect on soil particles that are far apart, thereby increasing the soil's aggregate stability (Ben-Hur & Keren, 1997), shear strength (Abu-Zreig et al., 2007), and structural stability (Mamedov et al., 2010). Moreover, PAM improves cohesion strength between soil particles and their resistance to splash erosion by raindrop impact and detachment by runoff (Yu et al., 2003). The efficacy of PAM may also be affected by soil type, clay mineralogy, and the type and concentration of ions in the soil solution or added by accompanying amendments (Yu et al., 2003; Mamedov, et al., 2010).

The effectiveness of P+B treatment in reducing runoff and soil loss (Tables 3 and 4) may reflect a combination of the high water holding capacity of PAM molecules and the large surface area of biochar particles for adsorbing water (Lee et al.,2015; van Zwieten et al., 2010). Biochar aids the rearrangement of soil particles by inducing tensile stresses around clay matrixes and causing the formation of cracks or macropores (Dexter, 1988), thus improving soil aggregate stability and water retention (Sharma and Bhushan, 2001; Saffari, et al., 2020). Biochar incorporated in the soil surface also helps reduce soil loss by absorbing raindrop energy and minimizing soil particle detachment and surface crusting (Lee et al., 2008).

Biochar alone yielded a smaller reduction than P+B in runoff and soil loss. The reason may be that larger biochar particles are preferentially removed by runoff due to weaker coherence with soil particles and shorter incubation periods, especially at the beginning of the rainy season after tillage and biochar application (Li et al., 2017). High sediment concentrations, mixed with biochar particles, were recorded in the plots treated with biochar at the beginning of the 2018 rainy season before crop establishment, when the soil was loose and had low strength (Li et al., 2017). In addition, disruption of aggregate by the effects of tillage (Li et al., 2017) and the absence of vegetation may be responsible for the increased soil loss (Ruiz-Colmenero et al., 2013; Ma, et al., 2015) early in the rainy seasons. Sadeghi et al., (2017) and Li et al., (2017) have reported their observation of some biochar particles in the runoff from biochar treatments. Biochar application increases moisture content, soil organic carbon, pH and (Glaser et al., 2002), but the addition of PAM may further improve the biochar–soil particles coherence and reduces runoff and soil loss from P+B treatment by increasing soil strength (Li et al., 2017) and countering the rainfall-related loss of biochar (Li et al., 2017). Generally, the improved biomass or crop cover from P + B treatment could intercept rainfall and runoff while crop root stabilizes soil structure and promotes infiltration and resistance to soil scouring (Li et al., 1992a, b). Our result is consistent with the study by Lee et al. (2015), who reported that applications of biochar plus PAM were more effective in reducing soil loss than biochar alone.

The advantage of P+L over lime treatment alone in reducing runoff and soil loss could be attributed to the increased concentration of Ca^{2+} in the soil solution (Table 4.1), which enhances cation bridging and PAM sorption to soil particles with negative charges (Mamedov et al., 2007; Nishimura et al., 2005). This in turn decreases the tendency of soil aggregate disintegration and surface seal formation by increasing the electrolyte concentration, and replacing exchangeable Na with Ca (Lee et al., 2010). The resulting effects on soil aggregation and infiltration rate thus reduced runoff and soil loss (Tables 4.3). In addition, the improved soil attributes from P+L treatment, especially soil pH due to liming, decrease exchangeable Al, which favors water and nutrient uptake, microbial activity, and growth of crops (Gabriel et al., 2018; Paradelo et al., 2015), leading to higher biomass (Table 4.2) and reduced soil erosion (Smets et al., 2008). Also, the lower effectiveness of lime treatment alone compared with P+L, given the limited solubility and mobility of lime, may reflect the time required for lime to effectively dissolve and interact with the soil (Haynes and Naidu, 1998; Anikwe et al.2016). Determining the effective rates of both PAM and lime is essential for optimizing the efficiency of P+L treatment.

Lime and gypsum differ in the physico-chemical mechanisms by which they interact with soil. The relatively lower effectiveness of P+G in acidic Luvisols could be related to shrinkage of the PAM structure along with decreased viscosity of PAM in the presence of excess electrolyte, probably from higher rate, resulting from the relatively rapid dissolution of gypsum (Ernani et al., 2006).

High electrolyte concentrations inhibit stretching of the PAM molecular chains and reduce its effectiveness in bridging soil particles, whereas modest concentrations reduce the viscosity of PAM, which ensures uniform spreading on the soil surface (Tang et al., 2006) and enhances flocculation (Yu & Lei, 2003). The poor efficacy of treatment with gypsum alone, compared to other treatments, could be due to its higher application rate. Treating acidic soils with medium to high rates of gypsum, which has high solubility (~2 g L^{-1}) and mobility, could remove Al compounds (which tend to bind soil particles) in the soil solution by forming the ion pair $AlSO_4$ (Ernani et al., 2006), which would enhance dispersion and thus increase runoff and soil loss (Nishimura et al., 2005). Our results are consistent with previous studies comparing lime and gypsum treatments (Yu & Lei, 2003; Kebede et al., 2020; Lepore et al., 2009). Kebede et al. (2020) found that P+G treatment was less effective than PAM alone or P+L in reducing soil loss, and Lepore et al. (2009) reported that soil loss in silt loam was reduced by 67% with P+L treatment and 58% by P+G treatment, but by only 30% with lime and 18% with gypsum treatment.

Both SI and biomass exhibited a negative relationship with runoff and soil loss, indicating their sensitivity to runoff and soil loss. The differences in slopes of their trend lines in Figure 4.6 suggest that soil loss was more sensitive to both SI and biomass than runoff, although the weakness of these correlations may also reflect wide variations in the effectiveness of amendments among treatments and between study seasons. Our results also show that lower *C*-factor values are a good indicator of the effectiveness of treatments in reducing soil erosion, and vice versa (Table 4.4). The *C*-factor values indicate that PAM and P+L treatments were most effective (0.12) whereas gypsum (0.20) was only slightly better than control (0.23) in reducing soil loss in the study plots in our experimental conditions.

4.5 Conclusions

This study demonstrated that application of PAM alone and combined with other soil amendments on croplands is an important practice for conserving soil and water resources and improving crop production in the Ethiopian highlands. Amendments generally improved soil pH, OM, and SI, resulting in increased infiltration rates and reduced runoff and soil loss. Compared to the untreated control plot, the reductions in seasonal runoff and soil loss were substantial, as great as 39% reduction in runoff from P+B treatment and 53% reduction in soil loss from P+L treatments both in 2019, and also were associated with high biomass. Soil moisture was most effectively maintained during the growing season by the P+B and PAM treatments, owing to their positive influence on soil macro-pores and hence infiltration rate. Treatments were generally more effective in the second year (2019), perhaps indicating a long-term effect of treatment combined with the benefits of fencing and incorporation of crop residues. This two-year study suggests that P+B was effective in increasing soil moisture (water retention) and reducing runoff. However, P+L was the most effective treatment in reducing soil loss and C-factor values, and increasing soil pH, compared to all other treatments. Since amendments have effects lasting longer than the term of our experiments, studying their effects over extended periods could help to better understand their effectiveness over time. Furthermore, the availability and costs of amendments determine the overall viability of amendments as alternative LM practices, hence, additional cost-benefit analysis study is needed to complement this study.

Chapter 5: Synthesis, general conclusions and recommendations

5.1 Synthesis and general conclusions

This study evaluated effectiveness of bio-physical and soil amendment land management practices by integrating laboratory and field runoff plots studies. The bio-physical practices were tested for four land management practices in three land use types across three agro-ecologies: Guder (highland), Aba Gerima (midland), and Dibatie (lowland) study sites. Effectiveness of the LM practices was determined on the basis of *C*- and *P*-factors of RUSLE through selecting appropriate rainfall erosivity (R)-factor model. The results revealed that generally effectiveness of LM practices vary between study seasons and across land uses, type of LM practices and agro-ecologies. Effectiveness of the practices was increased in the second season, compared with the first season. This is mainly attributed to previous traditional land use activities (conventional tillage and intensive grazing) that degraded the topsoil and the fresh disturbance of soils during construction of the LM practices together with the absence of protective vegetative cover on the bunds and trenches at the beginning of rainy season following construction that lead to a higher erosion rate in the first year. Among the different bio-physical practices, the highest effectiveness, the smallest P-factor value, was observed for SBG in the crop land plots and trench with enclosure in the non-crop land plots, compared to other LM practices. This implies that SBG and trench with exclosure are relatively effective management options for cropland and non-cropland uses, respectively. In addition, the result indicated the importance of integrating structural and vegetative measures to control soil erosion from cultivated lands than the use of physical structures only, especially in areas with steep slopes, where relatively higher soil loss can be expected. The lower *P*-factor values for trench with exclosure as compared to SBG implies that trench with exclosure was more effective than SBG, and this could be due to trenches are less permeable to sediment-loaded runoff than bunds and *fanya juu*. By agro-ecology, LM practices were more effective at Dibatie for croplands and Aba Gerima for non-cropland plots, which could be related to variations in the rainfall pattern and other bio-physical conditions among the study sites. The C-factor values were generally lower in non-crop lands than crop lands, mainly due to exclosure in non-crop land plots maintained better vegetation/ grass cover for most seasons of the year.

On the other hand, the effective PAM rate was determined using consecutive simulated rainfall storms in laboratory. The P20 was found to be effective in reducing runoff in the beginning while P40 and P60 were more effective in reducing runoff and soil loss starting from the third storm through the end of the consecutive storms, but with no statistically significant difference between P40 and P60. Hence, P40 was selected as the most suitable rate for the given test soil and rainfall pattern. The variation in effectiveness of PAM rates in reducing runoff with storm duration could indicate that the effective rates shall be selected based on the climatic region (rain fall pattern) in that lower rates for the short rains or higher rates for elongated rains. The mixed application of P40 with gypsum or lime increased infiltration rate (IR) in the first two storms through improving soil solution viscosity. However, effectiveness of the mixtures had diminished by various degrees as rain progressed, as compared to P40 alone, which could be attributed to the rate and properties of G and L. Moreover, combined application of PAM with lime improved soil pH, which could offer good option to both fairly reduce soil erosion and improve land productivity especially in acidic soils, which requires further field verification.

Field application of the selected effective PAM rate alone or integrated with other soil amendments indicated that PAM was effective in reducing runoff and soil loss but less effective in increasing crop biomass. However, PAM integrated with lime was more effective in reducing soil loss, improving soil attributes and increasing crop biomass. Applying PAM mixed with lime (P+L) significantly reduced soil loss, compared to the control while substantially improved soil pH, SI, soil moisture and OM, and teff biomass, as compared to the control, implying that P+L can be potentially best alternative land management practice. The study further predicted soil loss scenarios for the best performing bio-physical (SBG) and soil amendment (P+L) practices on cropland (Table 5.1). The result indicated that teff crop cove alone reduced soil loss by 77%, teff crop land treated with SBG and P+L reduced soil loss by 88% and 93% while the combination of SBG with P+L reduced soil loss by 96%. This implies that the combined application of SBG and P+L was much more effective in reducing soil loss than their separate application (Table 5.1).

Table 5. 1 The predicted soil loss scenarios using RUSLE model in cropland plot of 30m long and 8% slope crop land plot planted with teff in the year 2019. Treatments were PAM + lime (P+L) and soil bund with grass (SBG) and other management practices, such as weeding, remain the same.

Treatments	R	K	L	S	C	P	Soil loss (t ha^{-1} $year^{-1}$)	% reduction
Control	1272	0.03	1.19	1.73	1	1	78.56	
Teff	1272	0.03	1.19	1.73	0.23	1	18.07	77
Teff+(P+L)	1272	0.03	1.19	1.73	0.12	1	9.43	88
Teff+SBG	1272	0.03	1.19	1.73	0.23	0.32	5.78	93
Teff+(P+L)+SBG	1272	0.03	1.19	1.73	0.12	0.32	3.02	96

Note: *R* is the rainfall erosivity factor (MJ mm ha^{-1} h^{-1} yr^{-1}), *K* is soil erodibility (t h MJ^{-1} mm^{-1}), *L* is a slope length factor (dimensionless), *S* is a slope steepness factor (dimensionless), *C* is a cover and management practices factor (dimensionless), and *P* is a supporting practice factor (dimensionless).

5.2 Recommendations for future studies

Although the study revealed that PAM mixed with lime was effective in reducing soil loss and increasing soil pH, repeating the test in different agro-ecologies with different soil and climatic condition may help to better know about its performance in Ethiopia. Furthermore, since incubation periods for lime and biochar may not be same, continuing the experiment for extended period and studying associated physico-chemical changes will further clarify the effect of treatments overtime. In addition to reducing runoff and soil loss, soil amendments also affect the soil nutrient balance in croplands. Hence, study about effect of soil amendments on soil nutrient balance is another important topic to be considered in future studies. Despite SBG was more efficient bio-physical LM practice in reducing soil loss, the effect of P+L in improving soil properties and hence crop biomass yield could compensate its relatively low efficiency in reducing soil loss. Hence, further study to compare the overall costs and benefits of these practices from the view point of ecosystem services could help to better understand the efficiency and viability of the practices.

References

Abd Elbasit, M. A., Yasuda, H., Salmi, A., & Anyoji, H. (2010). Characterization of rainfall generated by dripper-type rainfall simulator using piezoelectric transducers and its impact on splash soil erosion. Earth Surface Processes and Landforms, 35(4), 466-475.

Abrol, V., Shainberg, I., Lado, M., & Ben-Hur, M. (2013). Efficacy of dry granular anionic polyacrylamide (PAM) on infiltration, runoff and erosion. European journal of soil science, 64(5), 699-705.

Abrol, V., Ben-Hur, M., Verheijen, F. G. A., Keizer, J. J., Martins, M. A. S., Tenaw, H., … Graber, E. R. (2016). Biochar effects on soil water infiltration and erosion under seal formation conditions: rainfall simulation experiment. Journal of Soils and Sediments, 16(12), 2709–2719.

Abu-Zreig, M., Al-Sharif, M., & Amayreh, J. (2007). Erosion control of arid land in Jordan with two anionic polyacrylamides. Arid land research and management, 21(4), 315-328.

Adgo, E., Teshome, A., Mati, B., 2013. Impacts of long-term soil and water conservation on agricultural productivity: the case of Anjenie watershed, Ethiopia. Agric. Water Manage. 117, 55–61

Adimassu, Z.,Mekonnen, K., Yirga, C., Kessler, A., 2014. Effect of soil bunds on runoff, soil and nutrient losses, and crop yield in the central highlands of Ethiopia. Land Degrad. Dev. 25 (6), 554–564

Amare, T., Zegeye, A. D., Yitaferu, B., Steenhuis, T. S., Hurni, H., & Zeleke, G. (2014). Combined effect of soil bund with biological soil and water conservation measures in the northwestern

Ethiopian highlands. Ecohydrology and Hydrobiology, 14(3), 192–199.

Amdihun, A., Gebremariam, E., Rebelo, L., & Zeleke, G. (2014). Suitability and scenario modeling to support soil and water conservation interventions in the Blue Nile Basin , Ethiopia, 1–13.

Amsalu, A., Stroosnijder, L., & de Graaff, J. (2007). Long-term dynamics in land resource use and the driving forces in the Beressa watershed, highlands of Ethiopia. Journal of Environmental management, 83(4), 448-459.

Anderson, N. P., Hart, J. M., Sullivan, D. M., Christensen, N. W., Horneck, D. A., & Pirelli, G. J. (2013). Applying Lime to Raise Soil pH for Crop Production (Western Oregon). Oregon State University Extension Publication, (EM 9057), 1–21.

Anikwe, M. A. N., Eze, J. C., & Ibudialo, A. N. (2016). Influence of lime and gypsum application on soil properties and yield of cassava (Manihot esculenta Crantz.) in a degraded Ultisol in Agbani, Enugu Southeastern Nigeria. Soil and Tillage Research, 158, 32-38.

Bagarello, V., Ferro, V., Keesstra, S., Comino, J. R., Pulido, M., & Cerdà, A. (2018). Testing simple scaling in soil erosion processes at plot scale. Catena, 167, 171-180.

Ben-Hur, M., & Keren, R. (1997). Polymer Effects on Water Infiltration and Soil Aggregation. Soil Science Society of America Journal, 61(2), 565.

Ben-Hur, M. (2008). Seal formation effects on soil infiltration and runoff in arid and semiarid regions under rainfall and sprinkler irrigation conditions. In Climatic Changes and Water Resources in the Middle East and North Africa (pp. 429-452). Springer, Berlin, Heidelberg.

Berihun, M.L., Tsunekawa, A., Haregeweyn, N., Meshesha, D.T., Adgo, E., Tsubo, M., Masunaga, T., Fenta, A.A., Sultan, D. and Yibeltal, M. (2019a). Exploring land use/land cover changes, drivers and their implications in contrasting agro-ecological environments of Ethiopia. Land Use Policy, 87, 104052.

Bewket, W., & Teferi, E. (2009). Assessment of soil erosion hazard and prioritization for treatment at the watershed level: case study in the Chemoga watershed, Blue Nile basin, Ethiopia. Land Degradation & Development, 20(6), 609-622.

Borrelli, P., Robinson, D.A., Fleischer, L.R., Lugato, E., Ballabio, C., Alewell, C., Meusburger, K., Modugno, S., Schütt, B., Ferro, V. and Bagarello, V. (2017). An assessment of the global impact of 21st century land use change on soil erosion. Nature Communications, 8(1), 1–13.

Brevik, E. C., & Hartemink, A. E. (2010). Early soil knowledge and the birth and development of soil science. Catena, 83(1), 23–33.

Caesar-TonThat, T. C., Busscher, W. J., Novak, J. M., Gaskin, J. F., & Kim, Y. (2008). Effects of polyacrylamide and organic matter on microbes associated to soil aggregation of Norfolk loamy sand. Applied Soil Ecology, 40(2), 240–249.

Descheemaeker, K., Nyssen, J., Rossi, J., Poesen, J., Haile, M., Raes, D., Muys, B., Moeyersons, J. and Deckers, S. (2006). Sediment deposition and pedogenesis in exclosures in the Tigray Highlands, Ethiopia. Geoderma, 132(3-4), 291-314.

De Vente, J., Poesen, J., Verstraeten, G., Govers, G., Vanmaercke, M., Van Rompaey, A., Arabkhedri, M. and Boix-Fayos, C. (2013). Predicting soil erosion and sediment yield at regional scales: where do we stand?. Earth-Science Reviews, 127, 16-29.

Dexter, A. R. (1988). Advances in characterization of soil structure. Soil and Tillage Research, 11(3–4), 199–238.

Doan, T. T., Henry-des-Tureaux, T., Rumpel, C., Janeau, J. L., & Jouquet, P. (2015). Impact of compost, vermicompost and biochar on soil fertility, maize yield and soil erosion in Northern Vietnam: a three year mesocosm experiment. Science of the Total Environment, 514, 147-154.

Ebabu, K., Tsunekawa, A., Haregeweyn, N., Adgo, E., Meshesha, D.T., Aklog, D., Masunaga, T., Tsubo, M., Sultan, D., Fenta, A.A. and Yibeltal, M. (2018). Analyzing the variability of sediment yield: A case study from paired watersheds in the Upper Blue Nile basin, Ethiopia. Geomorphology, 303, 446–455.

Ebabu, K., Tsunekawa, A., Haregeweyn, N., Adgo, E., & Tsegaye, D. (2019). Science of the Total Environment Effects of land use and sustainable land management practices on runoff and soil loss in the Upper Blue Nile basin , Ethiopia. Science of the Total Environment, 648(August 2018), 1462–1475.

Erlangung, Z., Eltner, D. G. A., Ries, J. B., & James, P. D. M. R.(2016). Photogrammetric techniques for across-scale soil erosion assessment, thesis.

Ernani, P. R., Miquelluti, D. J., Fontoura, S. M. V., Kaminski, J., & Almeida, J. A. (2006). Downward movement of soil cations in highly weathered soils caused by addition of gypsum. Communications in Soil Science and Plant Analysis, 37(3–4), 571–586.

Fenta, A. A., Rientjes, T., Haile, A. T., & Reggiani, P. (2014). Satellite rainfall products and their reliability in the Blue Nile Basin. In Nile River Basin (pp. 51-67). Springer, Cham.

Fenta, A. A., Yasuda, H., Shimizu, K., Haregeweyn, N., & Negussie, A. (2016). Dynamics of Soil

Erosion as Influenced by Watershed Management Practices: A Case Study of the Agula Watershed in the Semi-Arid Highlands of Northern Ethiopia. Environmental Management, 58(5), 889–905.

Fenta, A.A., Yasuda, H., Shimizu, K., Haregeweyn, N., Kawai, T., Sultan, D., Ebabu, K. and Belay, A.S. (2017). Spatial distribution and temporal trends of rainfall and erosivity in the Eastern Africa region. Hydrological Processes, 31(25), 4555–4567.

Fenta, A.A., Yasuda, H., Shimizu, K., Ibaraki, Y., Haregeweyn, N., Kawai, T., Belay, A.S., Sultan, D. and Ebabu, K. (2018). Evaluation of satellite rainfall estimates over the Lake Tana basin at the source region of the Blue Nile River. Atmospheric Research, 212, 43-53.

Fenta, A.A., Tsunekawa, A., Haregeweyn, N., Poesen, J., Tsubo, M., Borrelli, P., Panagos, P., Vanmaercke, M., Broeckx, J., Yasuda, H. and Kawai, T. (2020). Land susceptibility to water and wind erosion risks in the East Africa region. Science of The Total Environment, 703, 135016.

Flanagan, D. C., Ascough, J. C., Nearing, M. A., & Laflen, J. M. (2001). The water erosion prediction project (WEPP) model. In Landscape erosion and evolution modeling (pp. 145-199). Springer, Boston, MA.

Foster, G. R., Meyer, L. D., & Onstad, C. A. (1977). A runoff erosivity factor and variable slope length exponents for soil loss estimates. Transactions of the ASAE, 20(4), 683-0687.

Gabriel, C. A., Cassol, P. C., Simonete, M. A., Moro, L., Pfleger, P., & Mumbach, G. L. (2018a). Lime and gypsum applications on soil chemical attributes and initial growth of eucalyptus. Floresta, 48(4), 573–582.

Gabriel, C., Cassol, P. C., Simonete, M. A., & Moro, L. (2018b). Lime and gypsum applications on soil chemical attributes and initial growth of eucalyptus LIME AND GYPSUM APPLICATIONS ON SOIL CHEMICAL, (October).

Gebrehiwot, S. G., Bewket, W., Gärdenäs, A. I., & Bishop, K. (2014). Forest cover change over four decades in the Blue Nile Basin, Ethiopia: Comparison of three watersheds. Regional Environmental Change, 14(1), 253–266.

Gebremichael, D., Nyssen, J., Poesen, J., Deckers, J., Haile, M., Govers, G., & Moeyersons, J. (2005). Effectiveness of stone bunds in controlling soil erosion on cropland in the Tigray Highlands, northern Ethiopia. Soil Use and Management, 21(3), 287–297.

Girmay, G., Singh, B. R., Nyssen, J., & Borrosen, T. (2009). Runoff and sediment-associated nutrient losses under different land uses in Tigray, Northern Ethiopia. Journal of Hydrology, 376(1–2), 70–80.

Glaser, B., Lehmann, J., & Zech, W. (2002). Ameliorating physical and chemical properties of highly weathered soils in the tropics with charcoal–a review. Biology and fertility of soils, 35(4), 219-230.

Graber, E. R., Fine, P., & Levy, G. J. (2006). Soil stabilization in semiarid and arid land agriculture. Journal of Materials in Civil Engineering, 18(2), 190-205.

Grandy, A. S., Porter, G. A., & Erich, M. S. (2002). Organic amendment and rotation crop effects on the recovery of soil organic matter and aggregation in potato cropping systems. Soil Science Society of America Journal, 66(4), 1311-1319.

Haregeweyn, N., Poesen, J., Nyssen, J., De Wit, J., Haile, M., Govers, G., & Deckers, S. (2006).

Reservoirs in Tigray (Northern Ethiopia): characteristics and sediment deposition problems. Land Degradation & Development, 17(2), 211–230.

Haregeweyn, N., Poesen, J., Nyssen, J., Govers, G., Verstraeten, G., de Vente, J., Deckers, J., Moeyersons, J. and Haile, M. (2008). Sediment yield variability in Northern Ethiopia: A quantitative analysis of its controlling factors. Catena, 75(1), 65–76.

Haregeweyn, N., Fikadu, G., Tsunekawa, A., Tsubo, M., & Meshesha, D. T. (2012). The dynamics of urban expansion and its impacts on land use/land cover change and small-scale farmers living near the urban fringe: A case study of Bahir Dar, Ethiopia. Landscape and urban planning, 106(2), 149-157.

Haregeweyn, N., Tsunekawa, A., Nyssen, J., Poesen, J., Tsubo, M., Tsegaye Meshesha, D., Schütt, B., Adgo, E. and Tegegne, F. (2015). Soil erosion and conservation in Ethiopia: A review. Progress in Physical Geography, 39(6), 750–774.

Haregeweyn, N., Tsunekawa, A., Poesen, J., Tsubo, M., Meshesha, D.T., Fenta, A.A., Nyssen, J. and 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.

Haynes, R. J., & Naidu, R. (1998). Influence of lime, fertilizer and manure applications on soil organic matter content and soil physical conditions: A review. Nutrient Cycling in Agroecosystems, 51(2), 123–137.

Herweg, K., & Ludi, E. (1999). The performance of selected soil and water conservation measures—case studies from Ethiopia and Eritrea. Catena, 36(1–2), 99–114.

Hurni H. (1993). Land degradation, famine, and land resource scenarios in Ethiopia. In World soil erosion and conservation, Pimentel D (ed.), Cambridge Studies in Applied Ecology and Resource Management. Cambridge University Press: Cambridge; 27–61.

Hurni, H. (1985). Erosion – Productivity – Conservation Systems in Ethio- pia. In I. Pla Sentis (Ed.), Internat. Conf. on Soil Conservation (4th ed). (pp. 654–674). Maracay, Venezuela.

Hurni, K., Zeleke, G., Kassie, M., Tegegne, B., Kassawmar, T., Teferi, E., Moges, A., Tadesse, D., Ahmed, M., Degu, Y. and Kebebew, Z. (2015). Economics of Land Degradation (ELD) Ethiopia Case Study: Soil degradation and sustainable land management in the rainfed agricultural areas of Ethiopia: An assessment of the economic implications.

Jagger, P., & Pender, J. (2003). The role of trees for sustainable management of less-favored lands: the case of eucalyptus in Ethiopia. Forest Policy and Economics, 5(1), 83-95.

Karami, A., Homaee, M., Afzalinia, S., Ruhipour, H., & Basirat, S. (2012). Organic resource management: impacts on soil aggregate stability and other soil physico-chemical properties. Agriculture, Ecosystems & Environment, 148, 22-28.

Karpilo, Jr., R. D., & Toy, T. J. (2004). Rusle C-Factors for Slope Protection Applications. Journal American Society of Mining and Reclamation, 2004(1), 995–1013.

Kebede, B., Tsunekawa, A., Haregeweyn, N., Mamedov, A.I., Tsubo, M., Fenta, A.A., Meshesha, D.T., Masunaga, T., Adgo, E., Abebe, G. and Berihun, M.L. (2020). Effectiveness of Polyacrylamide in Reducing Runoff and Soil Loss under Consecutive Rainfall Storms. Sustainability, 12(4), 1597.

Kebede, F., & Yamoah, C. (2009). Soil fertility status and numass fertilizer recommendation of typic hapluusterts in the northern highlands of Ethiopia. World Applied Sciences Journal, 6(11), 1473-1480.

Kirui, O. K., & Mirzabaev, A. (2014). Economics of land degradation in Eastern Africa (No. 128). ZEF working paper series.

Krauer, J. (1988). Rainfall, erosivity and isoerodent map of Ethiopia. Soil Conservation Research Project, Research Report 15, University of Berne, Switzerland. 132 pp

Kumar, A., & Saha, A. (2011). Effect of polyacrylamide and gypsum on surface runoff, sediment yield and nutrient losses from steep slopes. Agricultural water management, 98(6), 999-1004.

Kuok, K. K. K., Mah, D. Y. S., & Chiu, P. C. (2013). Evaluation of C and P Factors in Universal Soil Loss Equation on Trapping Sediment: Case Study of Santubong River. Journal of Water Resource and Protection, 5(12), 1149–1154.

Lado, M., Paz, A., & Ben-Hur, M. (2004). Organic Matter and Aggregate Size Interactions in Infiltration, Seal Formation, and Soil Loss. Soil Science Society of America Journal, 68(3), 935–942.

Lado, M., Inbar, A., Sternberg, M., & Ben-Hur, M. (2016). Effectiveness of granular polyacrylamide to reduce soil erosion during consecutive rainstorms in a calcic regosol exposed to different fire conditions. Land Degradation & Development, 27(5), 1453-1462.

Lal, R. (1995). Erosion-Crop Productivity Relationships for Soils of Africa. Soil Science Society of America Journal, 59(3), 661–667.

Lal R. (2014). Soil conservation and ecosystem services. Int. Soil Water Conservation Res. 2(3):36-47.

Lee, S. S., Gantzer, C. J., Thompson, A. L., & Anderson, S. H. (2010). Polyacrylamide and gypsum amendments for erosion and runoff control on two soil series. Journal of Soil and Water Conservation, 65(4), 233–242.

Lee, Sang Soo, Gantzer, C. J., Thompson, A. L., Anderson, S. H., & Ketcham, R. A. (2008). Using High-Resolution Computed Tomography Analysis To Characterize Soil-Surface Seals. Soil Science Society of America Journal, 72(5), 1478–1485.

Lee, Sang Soo, Shah, H. S., Awad, Y. M., Kumar, S., & Ok, Y. S. (2015). Synergy effects of biochar and polyacrylamide on plants growth and soil erosion control. Environmental Earth Sciences, 74(3), 2463–2473.

Lepore, B. J., Thompson, A. M., & Petersen, A. (2009). Impact of polyacrylamide delivery method with lime or gypsum for soil and nutrient stabilization. Journal of soil and water conservation, 64(3), 223-231.

Letey, J., Clark, P., & Amrhein, C. (1992). A greenhouse experiment finds water-sorbing polymers do not conserve water. California Agriculture, 46(3), 9-10.

Levy, G. J., & Mamedov, A. I. (2002). High-Energy-Moisture-Characteristic Aggregate Stability as a Predictor for Seal Formation. Soil Science Society of America Journal, 66(5), 1603–1609.

Li, Y., X. Q. Xu, and X. M. Zhu. (1992a). Preliminary study on mechanism of plant roots to increase the anti-scouribility on the Loess Plateau. Sci. China Series B- Chemistry. 35(9): 1085-1092.

Li, Y., X. Q. Xu, and X. M. Zhu. (1992b). Effectiveness of plant roots on increase the soil permeability on the Loess Plateau. Chinese Sci. Bull. 37(20): 1735-1738.

Li, Z. guo, Gu, C. ming, Zhang, R. hua, Ibrahim, M., Zhang, G. shi, Wang, L., … Liu, Y. (2017). The benefic effect induced by biochar on soil erosion and nutrient loss of slopping land under natural rainfall conditions in central China. Agricultural Water Management, 185, 145–150.

Ludi, E. (2004). Economic analysis of soil conservation: case studies from the highlands of Amhara region, Ethiopia.

Ma, B., Liu, Y., Liu, X., Ma, F., Wu, F., & Li, Z. (2015). Soil splash detachment and its spatial distribution under corn and soybean cover. Catena, 127, 142–151.

Mekuria, W., Veldkamp, E., Haile, M., Nyssen, J., Muys, B., & Gebrehiwot, K. (2007). Effectiveness of exclosures to restore degraded soils as a result of overgrazing in Tigray, Ethiopia. Journal of arid environments, 69(2), 270-284.

Maetens W (2013) Effectiveness of land use and soil conservation techniques in reducing runoff and soil loss in Europe and the Mediterranean. Dissertation, KU Leuven, Groep Wetenschap & Technologie, Arenberg Doctoral School.

Mamedov, A. I., Beckmann, S., Huang, C., & Levy, G. J. (2007). Aggregate stability as affected by polyacrylamide molecular weight, soil texture, and water quality. Soil Science Society of America Journal, 71(6), 1909-1918.

Mamedov, A. I., Shainberg, I., Wagner, L. E., Warrington, D. N., & Levy, G. J. (2009). Infiltration and erosion in soils treated with dry PAM, of two molecular weights, and phosphogypsum. Australian Journal of Soil Research, 47(8), 788–795.

Mamedov, A. I., Wagner, L. E., Huang, C., Norton, L. D., & Levy, G. J. (2010). Polyacrylamide

Effects on Aggregate and Structure Stability of Soils with Different Clay Mineralogy. Soil Science Society of America Journal, 74(5), 1720–1732.

McCool, D. K., Brown, L. C., Foster, G. R., Mutchler, C. K., & Meyer, L. D. (1987). Revised slope steepness factor for the Universal Soil Loss Equation. Transactions of the ASAE, 30(5), 1387-1396.

McCool, D. K., Foster, G. R., Mutchler, C. K., & Meyer, L. D. (1989). Revised slope length factor for the Universal Soil Loss Equation. Transactions of the ASAE, 32(5), 1571-1576.

Mekuria, W., Veldkamp, E., Haile, M., Nyssen, J., Muys, B., & Gebrehiwot, K. (2007).Meshesha, D. T., Tsunekawa, A., Tsubo, M., Haregeweyn, N., & Adgo, E. (2014). Drop size distribution and kinetic energy load of rainfall events in the highlands of the Central Rift Valley, Ethiopia. Hydrological Sciences Journal, 59(12), 2203–2215.

Meshesha, D. T., Tsunekawa, A., Tsubo, M., Haregeweyn, N., & Adgo, E. (2014). Drop size distribution and kinetic energy load of rainfall events in the highlands of the Central Rift Valley, Ethiopia. Hydrological Sciences Journal, 59(12), 2203–2215.

Meshesha, D.T., Atsushi Tsunekawa, and Nigussie Haregeweyn. 2018. "Influence of Raindrop Size on Rainfall Intensity, Kinetic Energy, and Erosivity in a Sub-Humid Tropical Area: A Case Study in the Northern Highlands of Ethiopia." Theoretical and Applied Climatology: 1–11.

Mwendera, E. J., & Saleem, M. M. (1997). Infiltration rates, surface runoff, and soil loss as influenced by grazing pressure in the Ethiopian highlands. Soil use and management, 13(1), 29-35.

Nearing, M. A. (1997). A single, continuous function for slope steepness influence on soil loss. Soil Science Society of America Journal, 61(3), 917-919.

Nishimura, T., Yamamoto, T., Suzuki, S., & Kato, M. (2005). Effect of gypsum and polyacrylamide application on erodibility of an acid Kunigami mahji soil. Soil Science and Plant Nutrition, 51(5), 641–644.

Nyssen, J., Haile, M., Poesen, J., Deckers, J., & Moeyersons, J. (2001). Removal of rock fragments and its effect on soil loss and crop yield, Tigray, Ethiopia. Soil Use and Management, 17(3), 179-187.

Nyssen, J., Vandenreyken, H., Poesen, J., Moeyersons, J., Deckers, J., Haile, M., … Govers, G. (2005). Rainfall erosivity and variability in the northern Ethiopian highlands. Journal of Hydrology, 311, 172–187.

Nyssen, J., Poesen, J., Haile, M., Moeyersons, J., Deckers, J., & Hurni, H. (2009). Effects of land use and land cover on sheet and rill erosion rates in the Tigray highlands, Ethiopia. Zeitschrift Für Geomorphologie, 53(2), 171–197.

Nyssen, J., Frankl, A., Haile, M., Hurni, H., Descheemaeker, K., Crummey, D., Ritler, A., Portner, B., Nievergelt, B., Moeyersons, J. and Munro, N. (2014). Environmental conditions and human drivers for changes to north Ethiopian mountain landscapes over 145 years. Science of the total environment, 485, 164-179.

Panagos, P., Borrelli, P., Meusburger, K., Alewell, C., Lugato, E., & Montanarella, L. (2015). Estimating the soil erosion cover-management factor at the European scale. Land Use Policy, 48, 38–50.

Panagos, P., Borrelli, P., Meusburger, K., Yu, B., Klik, A., Lim, K. J., … Ballabio, C. (2017). Global rainfall erosivity assessment based on high-temporal resolution rainfall records. Scientific Reports, 7(1), 1–12.

Paradelo, R., Virto, I., & Chenu, C. (2015). Net effect of liming on soil organic carbon stocks: a review. Agriculture, Ecosystems & Environment, 202, 98-107.

Ravindran, B., Nguyen, D. D., Chaudhary, D. K., Chang, S. W., Kim, J., Lee, S. R., … Lee, J. J. (2019). Influence of biochar on physico-chemical and microbial community during swine manure composting process. Journal of Environmental Management, 232, 592–599.

Renard, K. G., Foster, G. R., Weesies, G. a., & Porter, J. P. (1991). RUSLE: Revised universal soil loss equation. Journal of Soil and Water Conservation, 46(1), 30–33.

Renard, K., Foster, G., Weesies, G., McCool, D., & Yoder, D. (1997). Predicting soil erosion by water: a guide to conservation planning with the Revised Universal Soil Loss Equation (RUSLE). Agricultural Handbook No. 703.

Rientjes, T., Haile, A. T., & Fenta, A. A. (2013). Diurnal rainfall variability over the Upper Blue Nile Basin: A remote sensing based approach. International journal of applied earth observation and geoinformation, 21, 311-325.

Roth, C. H., & Pavan, M. A. (1991). Effects of lime and gypsum on clay dispersion and infiltration in samples of a Brazilian Oxisol. Geoderma, 48(3-4), 351-361.

Ruiz-Colmenero, M., Bienes, R., Eldridge, D. J., & Marques, M. J. (2013). Vegetation cover reduces erosion and enhances soil organic carbon in a vineyard in the central Spain. Catena, 104, 153–160.

Sadeghi, S. H., Hazbavi, Z., & Harchegani, M. K. (2016). Controllability of runoff and soil loss from small plots treated by vinasse-produced biochar. Science of the Total Environment, 541, 483–490.

Saffari, N., Hajabbasi, M. A., Shirani, H., Mosaddeghi, M. R., & Mamedov, A. I. (2020). Biochar type and pyrolysis temperature effects on soil quality indicators and structural stability. Journal of Environmental Management, 261, 110190.

Sharma, P. K., & Bhushan, L. (2001). Physical characterization of a soil amended with organic residues in a rice - Wheat cropping system using a single value soil physical index. Soil and Tillage Research, 60(3–4), 143–152.

Smets, T., Poesen, J., & Bochet, E. (2008). Impact of plot length on the effectiveness of different soil-surface covers in reducing runoff and soil loss by water. Progress in Physical Geography, 32(6), 654-677.

Sojka, R. E., Bjorneberg, D. L., Entry, J. A., Lentz, R. D., & Orts, W. J. (2007, January 1). Polyacrylamide in Agriculture and Environmental Land Management. Advances in Agronomy. Academic Press.

Sonneveld, B. G. J. S., & Keyzer, M. A. (2003). Land under pressure: soil conservation concerns and opportunities for Ethiopia. Land degradation & development, 14(1), 5-23.

Stavi, I., Barkai, D., Knoll, Y.M., Zaady, E., 2016. Livestock grazing impact on soil wettability and erosion risk in post-fire agricultural lands. Sci. Total Environ. 573, 1203–1208.

Sultan D, Tsunekawa A, Haregeweyn N, et al. 2017. Analyzing the runoff response to soil and water conservation measures in a tropical humid Ethiopian highland. Physical Geography, 38(5): 423–447

Sultan, D., Tsunekawa, A., Haregeweyn, N., Adgo, E., Tsubo, M., Meshesha, D. T., ... & Ebabu, K. (2018). Efficiency of soil and water conservation practices in different agro-ecological environments in the Upper Blue Nile Basin of Ethiopia. Journal of Arid Land, 10(2), 249-263.

Sultan, D., Tsunekawa, A., Haregeweyn, N., Adgo, E., Tsubo, M., Meshesha, D.T., Masunaga, T., Aklog, D., Fenta, A.A. and Ebabu, K. (2018). Impact of Soil and Water Conservation Interventions on Watershed Runoff Response in a Tropical Humid Highland of Ethiopia. Environmental Management, 61(5), 860–874.

Tang, Z., Lei, T., Yu, J., Shainberg, I., Mamedov, A. I., Ben-Hur, M., & Levy, G. J. (2006). Runoff and Interrill Erosion in Sodic Soils Treated with Dry PAM and Phosphogypsum. Soil Science Society of America Journal, 70(2), 679.

Taye, G., Poesen, J., Wesemael, B. Van, Vanmaercke, M., Teka, D., Deckers, J., … Haregeweyn, N. (2013). Effects of land use, slope gradient, and soil and water conservation structures on runoff and soil loss in semi-arid Northern Ethiopia. Physical Geography, 34(3), 236–259.

Taye, G., Poesen, J., Vanmaercke, M., van Wesemael, B., Martens, L., Teka, D., … Hallet, V. (2015). Evolution of the effectiveness of stone bunds and trenches in reducing runoff and soil loss in the semi-arid Ethiopian highlands. Zeitschrift Für Geomorphologie, 59(4), 477–493.

Taye, G., Poesen, J., Vanmaercke, M., Van Wesemael, B., Tesfaye, S., Nyssen, J., … Haregeweyn, N. (2017). Determing RUSLE P- and C-Factors for stone bunds and trenches in rangeland and cropland, northern Ethiopia. Land Degradation & Development, Draft(January), 1–13.

Van Dijk, A. I. J. M., Bruijnzeel, L. A., & Rosewell, C. J. (2002). Rainfall intensity–kinetic energy relationships: a critical literature appraisal. Journal of Hydrology, 261(1-4), 1-23.

Van Zwieten, L., Kimber, S., Morris, S., Chan, K. Y., Downie, A., Rust, J., … Cowie, A. (2010). Effects of biochar from slow pyrolysis of papermill waste on agronomic performance and soil fertility. Plant and Soil, 327(1), 235–246.

Vanmaercke, M., Poesen, J., Verstraeten, G., de Vente, J., & Ocakoglu, F. (2011). Sediment yield in Europe: Spatial patterns and scale dependency. Geomorphology, 130(3–4), 142–161.

Wischmeier, W.H., Smith, D.D., 1978. Predicting rainfall erosion losses. A Guide to conservation planning. United States Department of Agriculture, Agricultural Research Service (USDA-ARS) Handbook No. 537. United States Government Printing Office, Washington, DC.

Xiong, B., Loss, R. D., Shields, D., Pawlik, T., Hochreiter, R., Zydney, A. L., & Kumar, M. (2018). Polyacrylamide degradation and its implications in environmental systems. Npj Clean Water, 1(1).

Yu, J., Lei, T. W., Shainberg, I., Mamedov, A. I., & Levy, G. J. (2003). Infiltration and erosion in soils treated with dry PAM and gypsum. Soil Science Society of America Journal, 67(2), 630-636.

Summary

Soil erosion by water is a major cause of land degradation globally and in the Upper Blue Nile (UBN) basin of Ethiopia specifically. Soil erosion has become severe threats to food security for developing countries like Ethiopia, where the livelihood of most of the population depends predominantly on agriculture. The multitude adverse on- and off-site consequences of soil erosion are reduction in soil fertility and crop production, loss of vital ecosystem services, siltation of reservoirs, etc. In Ethiopia, these environmental and socioeconomic consequences are further aggravated by human intervention, including deforestation, overgrazing, poor farming practices and lack of suitable land management (LM) practices.

So far, Ethiopia has made many efforts to control soil erosion and its consequences using different bio-physical LM practices and promising results have been obtained. However, effectiveness of these practices was less studied across land uses and agro-ecologies. Furthermore, the practices focused on reducing soil erosion and less attention was given to alternative management practices dealing with conditioning the soil, such as PAM that improve soil properties. Earlier studies indicate that application of PAM integrated with other soil amendments, such as gypsum, lime and biochar, could further improve effectiveness of PAM in reducing soil erosion through improving soil properties. But PAM technology as a soil conditioner has not been tested for soils in the tropical highland humid environments, such as northwest Ethiopia.

The main aim of this study was, therefore, to contribute for the development of alternative LM practices against soil erosion through testing the separate and combined effectiveness of bio-physical and soil amendment practices by integrating laboratory and field studies. The study was conducted under laboratory using a rainfall simulator as well as field conditions in the UBN basin.

The specific objectives were (1) to determine the effectiveness of bio-physical practices [soil bund (SB), fanya juu (F), soil bund with grass (SBG), trench with exclosure (T+E) and different crop types] on the bases of the C- and P-factors of the Revised Universal Soil Loss Equation (RUSLE); and (2) through first determining the effective PAM rate under laboratory condition, to investigate its effectiveness when applied alone or integrated with other soil amendments (gypsum, lime and biochar) in reducing runoff, soil loss, and RUSLE's C-factor under field condition. These objectives cover chapters 2–4 of this thesis, which comprises a total of five Chapters, including the introduction; and the general synthesis, conclusions and recommendations, as summarized below:

Chapter 1 explains the introductory section of the study that contains background, problem statement, objectives, description of the study area, general methodological framework and overall organization of the thesis.

Chapter 2 evaluates effectiveness of various bio-physical LM practices (SB, F, SBG and T+E) implemented to tackle soil erosion in the UBN basin, Ethiopia, through adopting the RUSLE model and determining support practice (P) and cover and management (C) factors for different LM practices in three agro-ecologies: Guder (highland), Aba Gerima (midland), and Dibatie (lowland). Two seasons daily soil loss data were collected from 42 runoff plots. The result showed that P-factor values ranged from 0.15 to 0.53 for SB, 0.18 to 0.5 for F, and 0.06 to 0.44 for SBG in cropland, the lowest being for SBG; and 0.03 to 0.42 for T+E in non-cropland plots. The average P values also varied with agro-ecology in the order Aba Gerima > Guder > Dibatie for cropland and Guder > Dibatie > Aba Gerima for non-cropland plots, which could be attributed to climatic and other bio-physical variations among study sites. The SBG was found the most effective bio-physical practice across all the three studied sites. The C-factor values varied from 0.004 to 0.64

in cropland and from 0.001 to 0.49 in non-cropland plots implying that the management practices were more effective in non-crop lands than croplands due to better cover condition in most seasons of the year than tilled croplands.

Chapter 3 determines the effective PAM rate that best reduces runoff and soil loss from Oxisols, one of the dominant soils in humid tropics/Ethiopia. Different PAM rates of 0(C), 20 kg ha^{-1} (P20), 40 kg ha^{-1} (P40), and 60 kg ha^{-1} (P60) were applied onto soil surface and run for six consecutive simulated rainfall storms of 70 mm h^{-1} intensity for 1-hr duration to determine the effective PAM rate. The P20 was found to be more effective in reducing runoff in the beginning while P40 and P60 were more effective in reducing both runoff and soil loss starting from the third storm through the end of the consecutive storms, but with no statistically significant difference between P40 and P60. Hence, P40 was selected as the most suitable rate for the given test soil and rainfall pattern.

Chapter 4 evaluates the potential of the selected PAM rate (i.e. P40) to reducing soil erosion in field runoff plots condition at Aba Gerima site in northwest Ethiopia. We assessed the effectiveness of PAM alone or integrated with gypsum, lime, or biochar in reducing soil loss. We collected daily runoff and sediment loss data from plots planted with teff during the 2018 and 2019 rainy seasons and investigated associated changes in soil properties and crop growth parameters. Treatments reduced seasonal runoff by 12–39% and soil loss by 13–53%. The highest reduction in soil loss was observed from PAM combined with lime (P+L) treatment. Integrating PAM with other amendments improved soil moisture content, pH, organic matter and crop biomass yield. The effects of these treatments were also reflected in improving C-factor of the RUSLE model, contributing for erosion modeling in this region and beyond. Unlike PAM, biochar, and lime amendments take time to be effective after application, hence, continuing the field experiment and studying associated physicochemical mechanisms for extended periods will better elucidate their effectiveness over time.

Chapter 5 provides the general synthesis, concussions and recommendations of the whole thesis based on the key findings obtained from Chapter 2–4. Soil loss prediction scenarios considering best performing practices out of the tested bio-physical and soil amendment practices (i.e. SBG from bio-physical and/or P+L from soil amendment), were estimated using RUSLE model to evaluate the separate and combined effectiveness of these best alternative LM practices in reducing soil loss on cropland (teff) runoff plot of 3m by 30m. The results showed that P+L and SBG separately reduced soil loss by 48 and 68%, respectively, while their combination reduced by 83%. Although SBG was more efficient in reducing soil erosion, as compared to P+L treatment, the effect of P+L in improving soil properties and hence crop and biomass yield could compensate its relatively low efficiency in reducing soil loss. Hence, further study to compare the overall costs and benefits of these practices from the view point of ecosystem services could help to better evaluate the efficiency of the practices.

学位論文概要

水による土壌侵食は、世界的に、そして特にエチオピアの青ナイル川上流域（UBN）における土地劣化の主要な原因である。土壌侵食は、人口のほとんどの生計を主に農業に依存しているエチオピアのような発展途上国の食糧安全保障に対する深刻な脅威となっている。土壌侵食の多数の有害なオンサイトおよびオフサイトの影響は、土壌の肥沃度の低下、作物生産、重要な生態系サービスの喪失、貯水池における沈泥などである。エチオピアでは、これらの環境的および社会経済的影響は、森林破壊、過放牧、貧しい農業慣行、適切な土地管理慣行の欠如などの人為的要因によってさらに悪化している。

これまでのところ、エチオピアは土壌侵食を制御するために多くの努力を払っており、さまざまな生物物理学的土地管理手法を使用してその結果がもたらされ、有望な結果が得られている。しかし、これらの慣行の有効性は、異なる土地利用と農業生態系全体ではあまり研究されていない。さらに、土壌の侵食を減らすことに重点を置いた慣行が、土壌の特性を改善するポリアクリルアミド（polyacrylamide: PAM）などの土壌の状態調整を扱う代替の管理慣行に与えられた。以前の研究は、石膏、石灰、バイオ炭などの他の土壌改良剤と統合されたPAMの適用が、土壌特性の改善を通じて土壌侵食を低減するPAMの効果をさらに改善できることを示している。しかし、土壌改良剤としてのPAM技術は、エチオピア北西部などの熱帯高地の湿気の多い環境の土壌では検証されていない。

そこで、本研究ではその主たる目的を、実験室とフィールドの研究を統合することにより、生物物理学と土壌改良の個別および組み合わせた効果を検証することにより、土壌侵食に対する代替の土地管理技術の開発に貢献することとした。本研究は、青ナイル川上流域のフィールド条件を使用して、降雨シミュレーターと実験室で行った。具体的な目的は、①改良型普遍的土壌損失方程式（RUSLE）のC係数（作物係数）およびP係数（保全係数）に基づく生物物理学的手法（ソイルバンド（SB）、ファニャジュ（F）、植栽されたソイルバンド（SBG）、禁牧されたトレンチ（T ＋ E）およびさまざまな作物タイプ）の効果の評価、②実験室条件下で有効なPAM施用量を最初に決定し、これを単独で適用した場合、または他の土壌改良剤（石膏、石灰、バイオ炭）と統合した場合の流出、土壌損失、および現場条件下でのRUSLEのC係数の削減における有効性の解析である。これらの目的は、この論文の2〜4章を構成する。そして、一般的な合成、結論、および推奨事項を含めて、以下に要約される。

第1章では、背景、問題の説明、目的、研究領域の説明、一般的な方法論のフレームワーク、および論文の全体的な構成を含む、研究の概要について説明する。

第2章では、3つの農業生態系、すなわちGuder（高地）、Aba　Gerima（中地）、およびDibatie（低地）において、異なる土地管理慣行の要因に対してRUSLEモデルを採用し、保全係数（P）と作物係数（C係数）を決定することにより、エチオピアの青ナイル川上流域の土壌侵食に取り組むために導入されたさまざまな生物物理学的土地管理（SB、F、SBGおよびT + E）の有効性を評価した。42の流出区から 2つの季節の毎日の土壌損失データを収集した。その結果、P係数の値は、農地のSBで0.15〜0.53、Fで0.18〜

0.5、SBGで0.06〜0.44の範囲で、最低はSBGであった。非作付区画のT ＋ Eでは0.03〜0.42、平均P値は、農地の場合はAba Gerima> Guder> Dibatie、非農地の場合はGuder> Dibatie> Aba Gerimaの順に変化した。これは、調査サイト間の気候やその他の生物物理学的変動に起因する可能性がある。 SBGは、調査された3つのサイトすべてで最も効果的な生物物理学的手法であることが示された。 C係数の値は、耕作地では0.004から0.64で、非耕作地では0.001から0.49で変化した。これは、年間のほとんどの季節の耕作条件が耕作地よりも良好であるため、管理慣行が耕作地よりも非耕作地でより効果的であることを意味している。

　第3章では、エチオピアの湿潤熱帯地方における主要な土壌の1つであるオキシソル(Oxisols)　からの流出と土壌損失を最も効果的に減らす効果的なPAM施用量を決定した。 0 kg/ha （C）、20 kg/ha（P20）、40 kg/ha（P40）、および60 kg/ha（P60）の異なるPAM施用量を土壌表面に適用し、降雨シミュレーターを用いて6つの連続した人工的な降雨を与えた。有効なPAM施用量を決定するために、70　mm/hの強度で1時間の降雨を与えた。 P40とP60は、3回目の降雨から始まり、連続した降雨の終わりまで、流出と土壌損失の両方を低減するのに効果的であったが、P40とP60の間に統計的有意差は認められなかった。　したがって、与えられた供試土壌と降雨パターンに最も適したPAM施用量としてP40が選択された。

第4章では、エチオピア北西部のAba Gerimaサイトでのフィールド表面流出プロット条件で土壌侵食を低減するための選択したPAM施用量（すなわちP40）の可能性を評価した。土壌損失の低減におけるPAMの単独または石膏、石灰、またはバイオ炭との統合

の有効性を評価した。　2018年と2019年の雨季にテフを植えた区画から、毎日の流出と土砂損失のデータを収集し、土壌特性と作物成長パラメータの関連する変化を調査した。処理により、季節的な流出が12〜39%減少し、土壌損失が13〜53%減少した。土壌損失の最高の減少は、石灰処理と組み合わせたPAM（P + L）から観察された。 PAMを他の処理と統合すると、土壌の含水率、pH、有機物、作物のバイオマス収量が向上した。これらの処理の効果は、RUSLEモデルのC係数の改善にも反映され、以降の侵食モデルに用いられた。　PAMとは異なり、バイオ炭と石灰の処理は、適用後の効果が出るまでに時間がかかるため、フィールド実験を継続し、関連する物理化学的メカニズムを長期間研究することで、時間の経過に伴うそれらの効果をよりよく解明できる。

第5章では、第2章から第4章で得られた主要な調査結果に基づいて、論文全体の一般的な結論、考察、および推奨事項を示した。　RUSLEモデルを使用して、検証された生物物理的手法および土壌改良剤の施用（すなわち生物物理学的改良を目的としたSBGおよび/または土壌改良を目的としたP ＋ L）のベストプラクティスを考慮した土壌損失予測シナリオを推定し、これらの最良の代替土地管理策は、3m x 30mのテフ耕作地流出プロットでの土壌損失を削減した。すなわちP ＋ LとSBGがそれぞれ土壌損失をそれぞれ48%と68%削減する一方で、それらの組み合わせは83%削減することが示された。P ＋ L処理と比較して、SBGは土壌侵食を減らすのにより効率的だったが、土壌特性の改善におけるP ＋ Lの効果、これによる作物およびバイオマス収量は、土壌損失の低減におけるその比較的低い効率を補うことができる。したがって生態系サービスの観点からこれらの実践の全体的なコストと利点を比較するためのさらなる研究によって、保全策導入の効果をよりよく評価することができるであろう。

List of Publications

1. Kebede, B., Tsunekawa, A., Haregeweyn, N., Adgo, E., Ebabu, K., Meshesha, D.T., Tsubo, M., Masunaga, T. and Fenta, A.A. (2020). Determining *C*- and *P*-factors of RUSLE for different land uses and management practices across agro-ecologies: case studies from the Upper Blue Nile basin, Ethiopia. Physical Geography (online, DOI: 10.1080/02723646.2020.1762831, this article covers Chapter 2 in the thesis).

2. Kebede, B., Tsunekawa, A., Haregeweyn, N., Mamedov, A.I., Tsubo, M., Fenta, A.A., Meshesha, D.T., Masunaga, T., Adgo, E., Abebe, G. and Berihun, M.L. (2020). Effectiveness of Polyacrylamide in Reducing Runoff and Soil Loss under Consecutive Rainfall Storms. Sustainability, 12 (4), 1597 (online, DOI: 10.3390/su12041597, this article covers Chapter 3 in the thesis).