

HYDROLOGICAL AND EROSION PROCESSES IN THE ETHIOPIAN HIGHLANDS

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# HYDROLOGICAL AND EROSION PROCESSES IN THE ETHIOPIAN HIGHLANDS

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## **Abstract**

With the objective to ameliorate the impact of recurrent drought and severe erosion, non-indigenous soil and water conservation structures are ubiquitous in the Ethiopian highlands. Nevertheless, erosion and drought remain critical problems in the sub-humid and humid highlands. The less than optimum performance of the non-indigenous practice is caused in part by not taking the varying landscape and climate conditions into account that determine the spatial pattern of runoff and erosion and thereby the performance of soil and water conservation practices.

This dissertation research was conducted with the objectives of understanding runoff generation processes and spatial and temporal runoff and erosion patterns at different (plot and watershed) scales. In addition, we investigated the potential of biochar, charcoal, and deep-rooted crops to improve soil hydraulic properties and greenhouse gas emissions.

Field and laboratory experiments were conducted in the 113 ha Anjeni watershed during the 2012 and 2013 rainy monsoon phases. Field infiltration tests were conducted and soil samples were analyzed for selected soil parameters. Soil column experiments were conducted to assess the effects of biochar and charcoal amendments on moisture retention and permeability of soils. Runoff and erosion rates were measured from 24 runoff-erosion plots and at watershed outlet. In

addition, gas samples were collected using static chambers and analyzed for nitrous oxide and methane emissions.

Results showed that while poor soil conditions (acidic, high clay and low organic matter content) are common, saturation excess runoff was the dominant runoff mechanism. Rainfall intensity would exceed median infiltration rate only 21% of the time. Soil degradation level and tillage significantly affected runoff. Sediment concentration and yield increased with greater plot length from 3 m to 15 m, but decreased at a plot length of 30 m. Sediment rating coefficients were also affected by plot size and with the progression of the rainy monsoon phase.

Wood charcoal improved permeability of soils near saturation (10 and 30 kPa). However, effect of charcoal on runoff and erosion at plot level was not significant. Significant differences were observed between seasonal nitrous oxide emissions, with greater fluxes observed during the end of the rainy monsoon phase.

Overall findings of this dissertation research imply that hillslope runoff and erosion rates are greatly variable both spatially and temporally. Taking the spatial pattern and mechanisms of runoff generation into account is of paramount importance for improving the performance of newly installed soil and water conservation practices.

## **BIOGRAPHICAL SKETCH**

Haimanote Kebede Bayabil was born and grew up in Gojjam Ethiopia. After completing high school at Debre Markos Comprehensive Senior Secondary School, he joined the then Alemaya University in 1999. During his four-year stay at university, he studied plant sciences and received his Bachelor degree in 2003. In September 2003, he was employed by Elfora Agro Industries P.L.C. where he served as Junior Agricultural Expert for eight months. Then in May 2004, he accepted a position at Finchaa Sugar Factory as a Plantation Section Manager, which requires managing a plantation section of 1300-hectare farm cultivated under irrigation and all the working staffs of the section. After working for three and half years, he left the organization to peruse his graduate studies and got his MPS degree from Cornell University in August 2009. For his masters research he modeled the rainfall-runoff relationships and assessed the impacts of soil and water conservation practices on soil physical and chemical properties at the rural Maybar watershed in the Northeastern Ethiopian Highlands. In October 2009, he joined the Sustainable Water Harvesting and Institutional Strengthening in Amhara (SWISHA), a project funded by the Canadian International Development Agency (CIDA), as an irrigated agriculture adviser. Soon after, in August 2010, Haimanote joined Cornell University to pursue his Ph.D. in the soil and water lab, in the department of Biological and Environmental Engineering.

*Dedicated to my beloved wife Erist Engidaw Wubet, our son Nahom Haimanote Kebede, and my  
late father Kebede Bayabil*

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I am thankful to the Amhara Regional Research Institute (ARARI) for allowing me to work at the Anjeni research site and providing long term hydrological and sediment data. I am indebted to my childhood best friend Ayenew Abebe. Thank you for everything you've done for me. I am also very grateful to my colleague and friend Tigist Y. Tebebu for her advice and support during hard times. The support and help I got from many colleagues and friends (Cathelijne Stoof, Seifu Adimassu, Abeyou Wale, Essayas Kaba, Christian Guzman, and John Recha) was invaluable.

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## CHAPTER 1: INTRODUCTION

The greatest challenges for the realization of the "*Green Revolution*" type led increase of agricultural productivity in Africa in general, and Ethiopia in particular, are the increasingly degraded soil conditions throughout the region that make soils less responsive to improved technologies. The land degradation trend is further exacerbated by increasing population pressures that requires more food. Thus, to meet the increasing food demand, all types of land including grazing and forest fields are extensively cultivated for crop production (Feoli et al., 2002a; Lu et al., 2007; Taddese, 2001). Soil erosion by water is severe (Bewket and Sterk, 2003; Demelash and Stahr, 2010; Temesgen et al., 2012) and droughts are common (Amsalu and Graaff, 2006; Biazin et al., 2011a; Hugo et al., 2002; Mouazen et al., 2007). Most areas in the Ethiopian highlands receive high amount of annual precipitation. However, rainfall distribution show high spatial and temporal variability (Bewket and Sterk, 2005a; Biazin et al., 2011b; McHugh et al., 2007). Thus, water scarcity is prevalent for longer periods of the year (Bewket and Sterk, 2005b; Biazin et al., 2011a), while at the same time most of the rainfall, during the rainy monsoon season, is lost as overland runoff causing erosion on the already degraded fields.

To curb the problem, extensive soil and water conservation structures have been constructed in most highland areas since the 1980s. However, these efforts have not resulted in decreasing sediment concentrations (Herweg and Ludi, 1999; Kato et al., 2011; Temesgen et al., 2012). In planning soil and water conservation practices (SWCP) the government followed a "one size fit" approach as SWCP were implemented throughout the highlands (Kato et al., 2011) regardless of soil, climate, and other ecological variations. Moreover, soil and water conservation structures

take significant area of the small farms that could be used for crop production otherwise. These have made construction and maintenance of soil and water conservation structures unaffordable by farmers and hence, non-sustainable. To overcome this limitation, soil and water management planning should aim towards developing integrated management practices that protect soil and water under a changing climate.

However, this can only be achieved through better understanding of soil physical and hydraulic properties as well as spatial and temporal runoff and erosion processes at field and watershed scales. Runoff and erosion problems in the Ethiopian highlands remain under-researched. Despite the magnitude of the problem, only few experimental watersheds, especially Soil Conservation Research Program (SCRIP) Watersheds, are available that serve as hydrological and monitoring sites. Existing trends of fitting discharge and sediment concentration data at a watershed outlet and applying the results to the landscape without field validation is likely faulty and misleading. Field studies, in other parts of the world, have shown that runoff and erosion in a landscape are governed by several factors (in addition to soil, land use, and slope) and extrapolation of results is not reliable (Bagarello and Ferro, 2004; Moreno-de las Heras et al., 2010; Parsons et al., 2006; Thomaz and Vestena, 2012). Therefore, there is an increasing need for detailed understanding of hillslope (plot) and catchment runoff and erosion processes as well as new and integrated approaches of soil management practices.

This dissertation focuses on field and laboratory research conducted for two rainy seasons (2012 and 2013) in the Anjeni Watershed, in the Ethiopian highlands. The main objectives of this research were to: (i). characterize dominant runoff processes, and assess potentials of biochar

and charcoal for soil and water management, (ii) investigate spatial and temporal runoff processes, and (iii) review the dominant hillslope erosion processes, investigate whether erosion rates are dependent on measurement scales, and assess seasonal effects of soil and water conservations structures on erosion rates.

In chapter 2 we investigate how soil properties can be improved. Because very little is known about the state of soil physical and hydraulic properties, our first specific aim was to study dominant runoff processes in the Ethiopian highlands by assessing soil properties in relation to storm characteristics. In addition, we assess how biochar and charcoal amendments affect soil-water retention characteristics of degraded soils in a context whether biochar and charcoal amendments from different biomass sources improve soil hydraulic properties, and thereby runoff and erosion.

In chapter 3 we investigate spatial and temporal runoff processes and its relationship with landscape position (with different soil degradation levels) and discharge at river outlet. We also assess the effects of crop type (barley with charcoal amendment and deep-rooted leguminous crop) on runoff responses.

Chapter 4 of this dissertation reviews sediment data observed during this dissertation research period (2012 and 2013) and long-term (196-1990) observations carried out as part of the ongoing hydrological and erosion monitoring activities in the Anjeni watershed. The paper uses daily runoff and erosion records from 40 plots of different sizes (3, 4.5, 9, 30, 180 m<sup>2</sup>) and the watershed outlet (113 ha).

In Chapter 5 greenhouse gas emission rates from plots amended with charcoal and under leguminous crop (Lupine) were assessed by collecting gas samples using static chambers. Seasonal and spatial nitrous oxide and methane concentrations were measured.

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## **CHAPTER 2: ASSESSING THE POTENTIAL OF BIOCHAR AND CHARCOAL TO IMPROVE SOIL HYDRAULIC PROPERTIES IN THE HUMID ETHIOPIAN HIGHLANDS: THE ANJENI WATERSHED<sup>1</sup>**

### **Abstract**

Biochar has shown promise for restoring soil hydraulic properties. However, biochar production could be expensive in the developing world, while charcoal is widely available and its production cost is relatively cheap. The objective of this study is therefore to investigate whether some of the charcoal made in developing countries can also be beneficial for improving soil hydraulic properties, and explore whether charcoal could potentially restore the degraded African soils. Laboratory and field experiments were conducted in the Anjeni watershed in the Ethiopian highlands, to measure soil physical properties including soil moisture retention and infiltration rates. Soils were dominantly clayey with pH in the acidic range, low organic carbon content, and steady infiltration rates ranging between 2 and 36 mm/h. Incorporation of woody feedstock (Acacia, Croton, and Eucalyptus) charcoals significantly decreased moisture retention at lower tensions (10 and 30 kPa), resulting in an increase in relative hydraulic conductivity coefficients at these tensions. While wood (oak) biochar decreased moisture retention at low tensions, corn biochar increased retention, but effects were only slight and not significant. Surprisingly, available water content was not significantly affected by any of the amendments.

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Overall findings suggest that wood charcoal amendments can improve soil hydraulic properties of degraded soils, thereby potentially reducing runoff and erosion.

*Keywords:* Soil physical properties, biochar, wood charcoal, soil water retention, soil and water management

## **2.1 Introduction**

Smallholder farm productivity in the Ethiopian highlands is constrained by land degradation due to accelerated soil erosion (Bewket and Sterk, 2003; Demelash and Stahr, 2010; Temesgen et al., 2012) and recurrent droughts (Amsalu and Graaff, 2006; Biazin et al., 2011; Hugo et al., 2002; Mouazen et al., 2007). To meet increasing food demand for growing populations, typically all types of land including grazing and forest fields are extensively cultivated for crop production (Feoli et al., 2002; Lu et al., 2007; Taddese, 2001). While annual precipitation is high in most African highland areas, its distribution is variable both in space and time (Bewket and Sterk, 2005; Biazin et al., 2011; McHugh et al., 2007). Water scarcity therefore prevails for 8-9 months every year (Bewket and Sterk, 2005; Biazin et al., 2011), while much rainfall is lost to runoff during the rainy monsoon season, causing erosion on the already degraded fields. To mitigate these negative impacts, soil and water conservation structures were built in most highland areas in Ethiopia. While these conservation efforts have considerably reduced surface runoff and soil erosion in some areas (Hurni et al., 2005; Nyssen et al., 2010), expectations were achieved only partially in most areas (Herweg and Ludi, 1999; Kato et al., 2011; Temesgen et al., 2012). The reason for this frequent lack of success may lie in that soil and water conservation practices often attempt to tackle symptoms of the problems (runoff and erosion) rather than their root causes

(such as poor soil permeability). Moreover, conservation efforts primarily use structural measures, regardless of apparent variations in edaphic, topographic, and hydrologic factors (Amsalu and Graaff, 2006; Kato et al., 2011; Shiferaw and Holden, 2000; Temesgen et al., 2012). These structural measures may, unless excess water is drained (Bayabil et al., 2010), cause field waterlogging and accelerated erosion when conservation structures on degraded soils are breached (Temesgen et al., 2012).

One of the ways to improve soil physical properties that has received increased attention recently is biochar, that is produced when biomass is thermally decomposed at a preset temperature with no or low supply of oxygen (Lehmann et al., 2011). Biochar amendments have been reported to improve soil bulk density, porosity, water retention, and hydraulic conductivity (Abel et al., 2013; Asai et al., 2009; Atkinson et al., 2010; Jeffery et al., 2011; Karhu et al., 2011; Laird et al., 2010). Several authors have also reported that biochar amended soils retained more nutrients (Dexter, 1991; Glaser et al., 2002; Joseph et al., 2007; Kookana et al., 2011; Major et al., 2010; McHenry, 2011; Oguntunde et al., 2004; Steiner et al., 2007; Verheijen et al., 2009). Despite the potential benefit of biochar amendment, lack of capital and poor infrastructure may prevent smallholder farmers to get access to pyrolysis kilns needed for biochar production. This poses considerable challenges on the use of biochar in rural Africa. Wood charcoal may be a good alternative as it is widely produced in most rural areas of Africa (Lehmann et al., 2006), using simple soil pits instead of high-tech kilns. Moreover, charcoal has been reported to have similar beneficial effects as biochar, as it can improve retention of both soil moisture (Glaser et al., 2002; Kameyama et al., 2010) and nutrients (Lehmann et al., 2006; Oguntunde et al., 2004; Steiner et al., 2007).

The objective of this study was to characterize soil hydrology and dominant runoff mechanisms in the Ethiopian highlands, and investigate whether biochar and wood charcoal can be used to improve soil hydraulic properties and potentially decrease surface runoff and erosion.

## **2.2 Materials and Methods**

### ***2.2.1 Site Description***

This study was conducted in the Anjeni watershed in northwest Ethiopia (Figure2-1). The watershed is one of the experimental watersheds established under the Soil Conservation Research Program (SCRIP) of the Ethiopian Ministry of Agriculture in collaboration with the Swiss Agency for Development and Cooperation (Hurni et al., 2005).

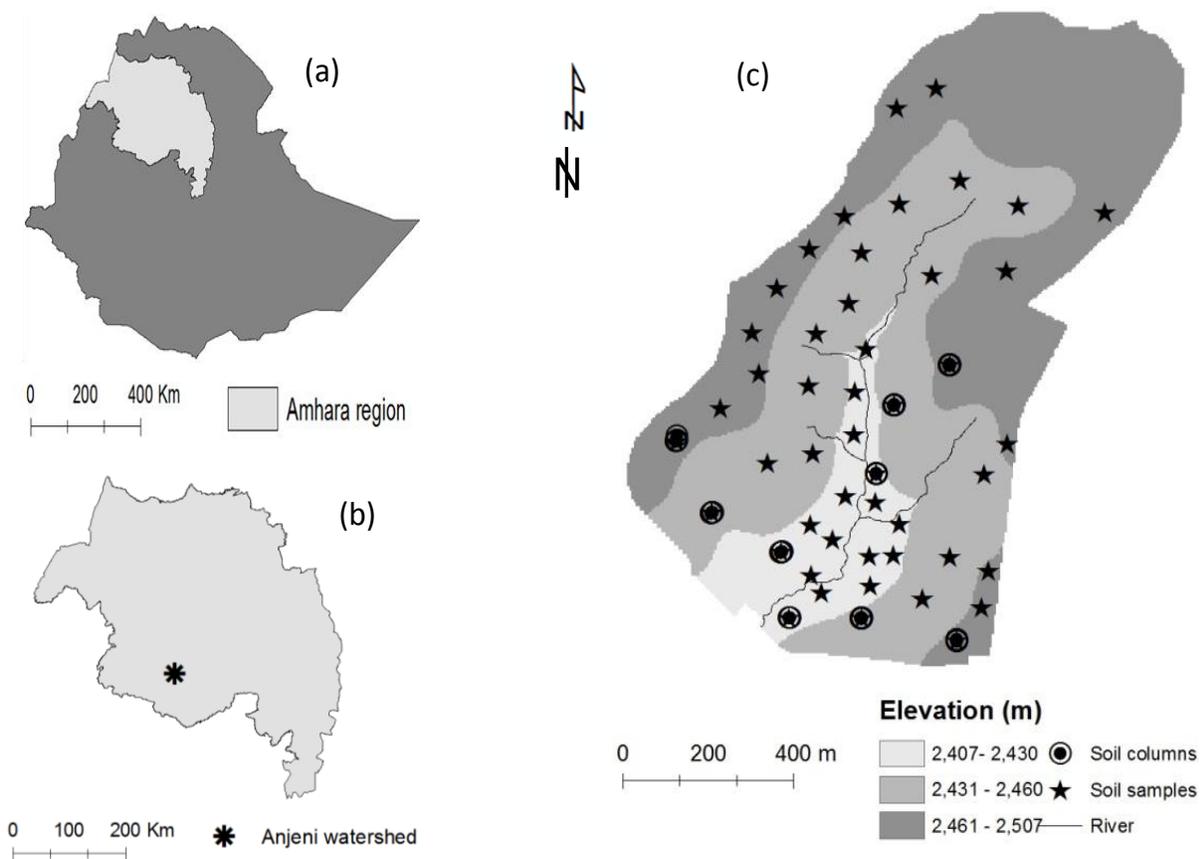


Figure 2-1. Map of Ethiopia (a) with the Amhara region (b) indicating the location of the Anjeni watershed (c). Sampling locations are indicated in (c).

Mean daily temperature in this region ranges from 9°C to 23°C, and mean annual rainfall is 1690 mm with a unimodal rainy season, which lasts from the middle of May to the middle of October. The Anjeni watershed drains a total catchment area of 113 ha, its gauging station is located at 10°40' N, 37°31' E (Tilahun et al., 2011). The watershed is oriented north-south and flanked on three sides by plateau ridges – elevation in the watershed ranges from 2407 to 2507 m (Herweg and Ludi, 1999). Finally, land use is mostly small scale agriculture, and soils have developed from basalt and volcanic ash, with Alisols, Nitisols, and Cambisols covering more than 80% of the area (Zeleeke, 2000). The deep Alisols cover the bottom part of the watershed; moderately

deep Nitisols cover the mid-transitional, gently sloping parts, and shallow Regosols and Leptosols cover the high, steepest areas. While the middle area of the watershed is covered by moderately deep Dystric Cambisols (Legesse, 2009; Zeleke, 2000).

### **2.2.2 Soil physical properties**

We assessed soil physical and basic chemical characteristics across the Anjeni watershed by measuring bulk density, soil moisture characteristics, soil texture, organic carbon content, pH, and infiltration rates. Moreover, runoff processes were determined by comparing infiltration rates with rainfall intensity computed using five-year rainfall records (1989-1993).

Since soils in the Ethiopian highlands vary with elevation (Amare et al., 2013), soil samples were taken and infiltration tests were performed at three elevation ranges ('low' 2407-2430 m, 'mid' 2431-2460 m, and 'high' 2461-2507 m a.s.l.), along a set of 16 downslope transects across the watershed. The sampling design yielded 48 sampling locations ('soil samples', Figure 2-1c). A distance of 125 m was maintained between transects, except when locations were inaccessible and samples were taken from adjacent locations that were accessible. In addition, transects in the northern part of the watershed lacked sampling locations in the low elevation range, hence more samples were collected from the lower elevation ranges of transects in the southern part of the watershed and to balance sample sizes between elevation ranges.

At each sampling location, we conducted *in situ* infiltration tests, extracted undisturbed soil samples (0-5 cm depth, using 91.2 cm<sup>3</sup> cores) to determine bulk density, and collected bulk soil samples (0-20 cm depth) for analyses of soil texture, organic carbon content (OC) and pH. Though organic carbon and pH are not soil physical parameters *per se*, they were measured

because of their effects on parameters and processes like aggregate stability, clay flocculation/dispersion, and thus their effect on soil physical properties.

Infiltration tests were done during the dry season, in March 2012, and to minimize water requirements, tests were conducted using a single ring infiltrometer (25 cm tall, 30 cm diameter). A wooden board was put on top of the infiltrometer and the infiltrometer was driven ~15 cm into the soil using a hammer. For each measurement, the drop in water level was measured at 5 min intervals using plastic rulers and a stopwatch. After each measurement, the ring was refilled with water to its initial level; and the test continued until the drop in water level was constant.

In addition, five-year (1989 to 1993) rainfall records were obtained from the Amhara Agricultural Research Institute (ARARI) that contained 8651 storm records from which we calculated storm duration, intensity (volume divided by duration), and frequency. The dominant runoff generation mechanism in the watershed (saturation vs. infiltration excess runoff) was subsequently identified using exceedance probabilities of storm intensity, by comparing five-year storm intensity values with the 25<sup>th</sup>, 50<sup>th</sup>, and 75<sup>th</sup> percentiles of measured infiltration rates.

### ***2.2.3 Effect of biochar and charcoal on soil water retention***

The effect of biochar and charcoal on soil water retention was assessed in the laboratory by incorporating biochar and charcoal into soils taken from the field.

For this, undisturbed soil columns (30 cm tall, 12 cm diameter) were extracted along three of the sixteen transects surveyed ('soil columns', Figure 2-1c). At each elevation range (low, mid, high) of the three transects, six replicate soil columns were extracted along the contour, yielding 54

soil columns in total. These columns were lined with cheesecloth and transported to the office station (in the watershed) and left to dry in the sun for 20 d before their dry weights were determined.

Since the effect of biochar and charcoal varies with feedstock source (Abel et al., 2013; Enders et al., 2012), we tested the effect of incorporation of two biochars (prepared from corn stover and oak) and three wood charcoals (*Eucalyptus camaladulensis*, *Acacia abyssinica*, and *Croton macrostachyus*) compared to a non-amended control. The two biochars (corn and oak) used in this study were previously used by Enders et al. (2012) as ‘corn 450°C’ and ‘oak 450°C’. They were produced by Best Energies Inc. (Cashton, WI, USA) by pyrolyzing pre-dried corn and oak feedstocks in the Daisy Reactor, a uniformly heated chamber at 450°C, for 80 to 90 min (Enders et al., 2012). All wood charcoals used were prepared in the Anjeni watershed following local farmers' practices. For this, trunks of each feedstock type (acacia, eucalyptus, and croton) with an approximate diameter of 20-30 cm were chopped into short logs (< 50 cm), placed inside separate pits (1 m deep, 1 m diameter) that had been excavated on open grounds, and were set on fire. To avoid complete combustion of biomass into ash, each pit was then covered by a layer of corn stubble, and backfilled with the excavated soil. The whole charring process took on average 3 to 5 d depending on the moisture status of both the feedstocks and the surrounding soils. After this, the charred biomass (charcoal) was extracted and manually crushed to obtain relatively uniform particle sizes (~2 mm diameter).

A fixed amount of biochar and charcoal (5 g/kg soil, or 0.5% by weight) was randomly added to columns in a randomized complete block design (Figure 2-1), by manually mixing the material

into the top 20 cm of soil. Because cultivation alone, even with no amendment, can also change soil properties, we also manually mixed the top 20 cm of the non-amended control columns.

To allow for aggregation of biochar and charcoal particles with the soil matrix, all columns including the control were put under wetting and drying cycles for 30 d, by leaving them outside in the sun without any shade with regular (every 7 d) supply of irrigation water. Subsequently, columns were taken inside the laboratory and put on a mesh, 50 cm above the ground, and they were irrigated until they became saturated. Afterwards, daily weights of the freely draining columns were measured for 6 d (with 24-h interval), until weights were constant. Finally, 54 bulk soil samples (~250 g) were taken by mixing the top (0-20 cm) of amended and control columns for laboratory moisture tests at different tensions.

#### ***2.2.4 Laboratory analyses***

Soil samples were transported to Adet Agricultural Research Center for laboratory analyses. Soil bulk density was determined after oven drying soil cores for 24 h at 105°C, and particle size distribution was determined using the Bouyoucos hydrometer procedure (Sahlemedihn and Taye, 2000). Organic carbon content was determined following the Walkley and Black method (Sahlemedihn and Taye, 2000), and soil pH was measured with the pH-water method using a 1:2.5 soil to water mixture (Sahlemedihn and Taye, 2000). Soil water retention measurements were conducted on 54 disturbed samples taken from biochar and charcoal treated and control columns. Moisture retention measurements were performed at five tensions (10, 30, 100, 500, and 1500 kPa) using a pressure plate apparatus.

In addition, in 2010, before conducting the column experiments, charcoal samples from different batches of *Eucalyptus* and *Acacia* biomass purchased from local markets near the Anjeni watershed were chemically analyzed at the Cornell University Soil and Water Lab. pH was determined with the pH-water method using a 1:2.5 charcoal to water mixture, and exchangeable base cation content ( $\text{Na}^+$ ,  $\text{K}^+$ ,  $\text{Ca}^{2+}$ ,  $\text{Mg}^{+2}$ ) contents determined using inductively coupled plasma (ICP) spectrometry. Because of limited supply, these analyses could unfortunately not be done for the Croton charcoal.

### 2.2.5 Analysis of effects on soil water retention

To allow for analysis of biochar and charcoal effects on soil water retention characteristics, we fitted the Van Genuchten (1980) soil moisture retention model (Eqs. 2-1 and 2-2) to the measured soil water retention data. First, unknown parameters of Eq. 2-1 were optimized, and results were used to calculate the relative degree of saturation (Eq. 2-2) and relative hydraulic conductivity ( $K_r$ ) or permeability coefficients (Eq. 2-3). Available water content was calculated as the moisture retention difference between 30 and 1500 kPa.

$$\theta(\psi) = \theta_r + (\theta_s - \theta_r) \left[ \frac{1}{1+(\alpha\psi)^n} \right]^m \quad (2-1)$$

$$S_e = \frac{\theta(\psi) - \theta_r}{\theta_s - \theta_r} \quad (2-2)$$

$$K_r = S_e^l \left[ 1 - \left( 1 - S_e^{\frac{1}{m}} \right)^m \right]^2 \quad (2-3)$$

where  $\theta_r$  and  $\theta_s$  are residual and saturated moisture contents, and  $\theta(\psi)$  and  $\psi$  represent the moisture and corresponding tension respectively.  $\alpha$  ( $\text{kPa}^{-1}$ ),  $n$ ,  $m$ , and  $l$  are dimensionless model fitting parameters, where  $\alpha$  is proportional to the inverse of the air entry value  $n$  and  $m$  are related to soil pore size distribution.  $S_e$  and  $K_r$  represent relative saturation and hydraulic conductivity of soils, respectively.  $l$  was assigned a value of 0.5, and  $m$  was assigned a value of one minus the inverse of  $n$  (i.e.,  $m = 1 - 1/n$ , provided  $n > 1$ ) to reduce the number of unknown parameters as proposed by Van Genuchten (1980).

### ***2.2.6 Statistical analyses***

Statistical data analysis and optimization of soil water retention curves to obtain Van Genuchten parameters was performed using R (R Development Core Team 2010). Since water retention data obtained from pressure plates and column drainage experiments violated assumptions of normality and equal variance, separate two-way ANOVA tests were run for observations from similar tensions or days. Treatment was used as a main factor, while elevation range was a block factor. For factors with significant Analysis of Variance (ANOVA) results, TukeyHSD mean comparison tests were performed to identify significant differences between groups.

## **2.3 Results**

The results of the soil properties and infiltration rate along the elevation gradient are presented first, followed by the effect of charcoal and biochar on soil physical properties.

### ***2.3.1 Soil physical properties***

Field and laboratory measurements summary results (Table 2-1) show that acidic to moderately acidic soils (pH <6), with high mean clay and silt contents (42 and 32%, respectively), and low in organic carbon (mean of 1.1%) were dominant in the study area. Soils were quite similar across elevation ranges, with only pH showing a significant trend (increase) with elevation (Table 2-1). Dry bulk density and sand content showed no apparent trend, while clay content slightly increased with elevation (39.7 to 43.3%) (Table 2-1).

Table 2-1. Mean values of soil properties at three elevations ranges (based on 16 replicate measurements per elevation range), with standard deviations given between parentheses. Values not sharing the same letter within the same column are statistically different

Elevation range	$f_s$	BD	pH	Clay	Silt	Sand	OC
	(mm/h)	(g/ cm <sup>3</sup> )	(-log[H+])	(%)			
Low	11.2 <sup>a</sup>	1.27 <sup>a</sup>	5.45 <sup>a</sup>	39.7 <sup>a</sup>	35.4 <sup>a</sup>	24.9 <sup>a</sup>	1.11 <sup>a</sup>
	(2.3)	(0.03)	(0.07)	(2.0)	(1.2)	(1.4)	(0.04)
Mid	11.0 <sup>a</sup>	1.28 <sup>a</sup>	5.67 <sup>ab</sup>	41.7 <sup>a</sup>	32.2 <sup>a</sup>	26.1 <sup>a</sup>	1.05 <sup>a</sup>
	(1.4)	(0.03)	(0.07)	(1.9)	(1.8)	(1.5)	(0.02)
High	8.5 <sup>a</sup>	1.25 <sup>a</sup>	5.95 <sup>b</sup>	43.3 <sup>a</sup>	35.4 <sup>a</sup>	21.3 <sup>a</sup>	1.10 <sup>a</sup>
	(2.8)	(0.02)	(0.09)	(1.7)	(1.48)	(1.6)	(0.02)

$f_s$ : steady infiltration rate, BD: bulk density, and OC: organic carbon content.

Correlations between soil parameters are presented in Table A1. As expected, clay content was strongly (negatively) correlated with the other two textural groups (sand and silt) with correlation coefficients (-0.62 and -0.60) respectively. Unexpectedly, bulk density (BD) was weakly positively correlated with steady state infiltration rate ( $f_s$ ), while pH showed a negative (albeit weak) correlation with clay and organic carbon (OC), with correlation coefficients of -0.18 and -0.12, respectively (Table A1).

### 2.3.2 Storm characteristics and infiltration capacity

Analysis of five-year (1989-1993) rainfall records showed that rainfall had a considerable seasonal variation, with four months (June through September) accounting for 76% of annual precipitation on average (Figure B1). Further analyses of 8651 storm records showed that short duration storms (<15 min, average intensity 6.3 mm/h) contributed for 68% of annual precipitation (Figure C1).

As steady infiltration rates did not significantly vary with elevation, 25<sup>th</sup>, 50<sup>th</sup> and 75<sup>th</sup> percentile infiltration rates were calculated from the data aggregated over all three-elevation ranges. The 25<sup>th</sup> percentile infiltration rate in the watershed was 4.6 mm/h, and the 50<sup>th</sup> and 75<sup>th</sup> percentile steady infiltration rates were 8.9 and 12.5 mm/h, respectively. Comparing five-year storm intensity records with these steady infiltration rates (Figure 2-2) showed that the probabilities for any storm intensity to match or exceed the 25<sup>th</sup>, 50<sup>th</sup>, and 75<sup>th</sup> percentile infiltration rates were 37, 23, and 16%, respectively.

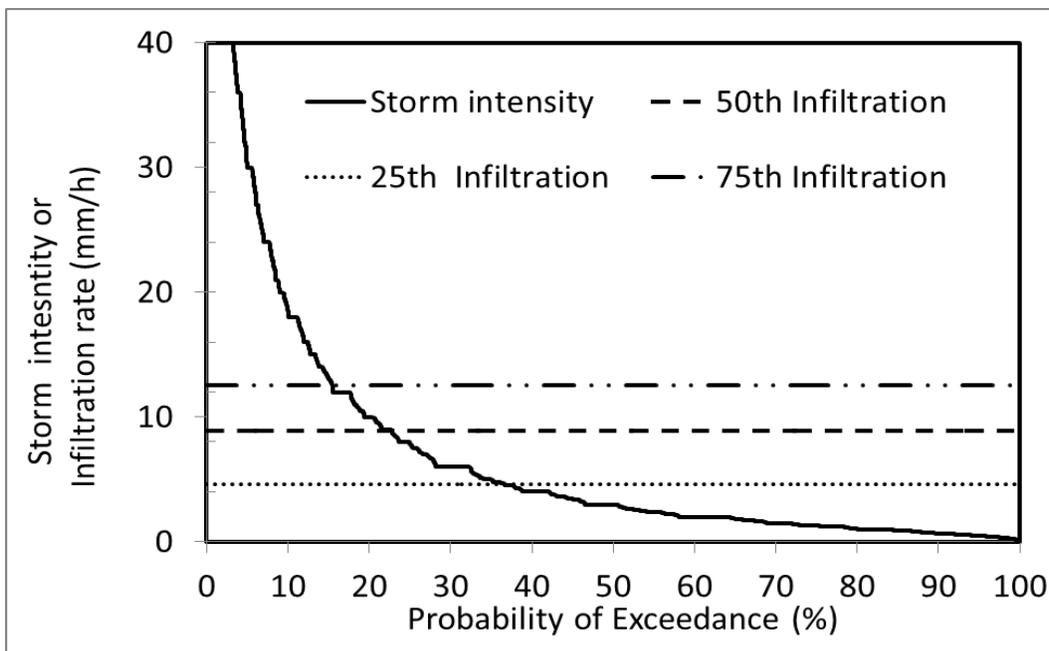


Figure 2-2. Exceedance probability of rainfall intensity compared with 25<sup>th</sup>, 50<sup>th</sup>, and 75<sup>th</sup> percentile infiltration rates.

Though some of the highest average infiltration rates were found at the lower elevations (Table 2-1) and the risk of infiltration excess runoff may therefore be limited, overland flow may still occur at these locations. This is because these soils have gentle slopes and may saturate due to interflow from the steeper uplands, thereby producing saturation excess overland flow. At the higher elevations where infiltration rates were lowest, improvement of infiltration capacity can

increase infiltration rates and thereby decrease the risk of infiltration excess overland flow during the most intense storms.

### 2.3.3 Effects of biochar and charcoal on soil water retention

Analysis of soil water retention data (Figure 2-3) indicated that all biochar and charcoal amendments except corn biochar decreased soil water retention at most tensions considered. However, these effects were only significant at 10 and 30 kPa (Figure 2-3). At 10 kPa, water retention of soils amended with the three charcoals (acacia, croton, and eucalyptus) was significantly lower than for biochar (corn and oak) amended and control soils. At 30 kPa, the lower water retention of charcoal amended soil was only significant for croton (Figure 2-3).

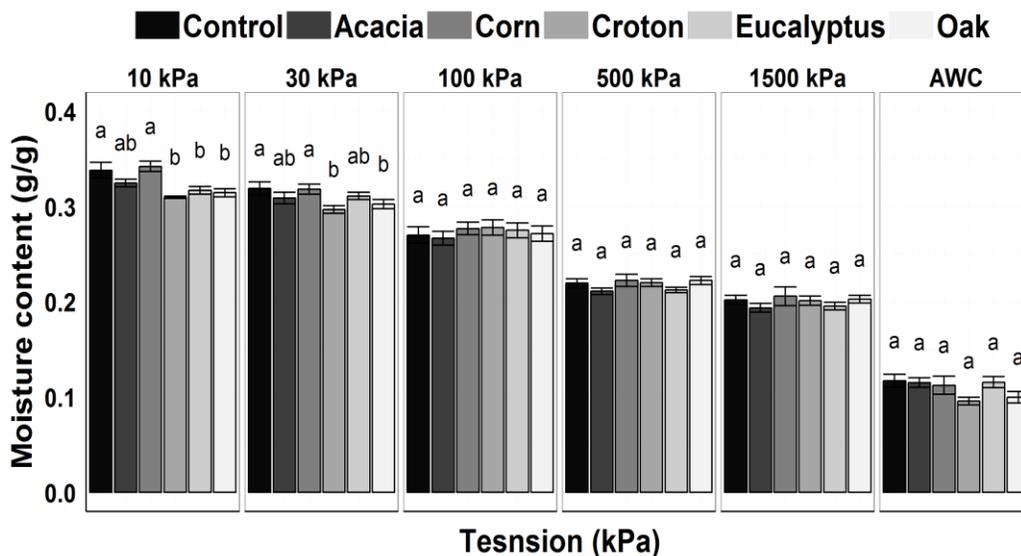


Figure 2-3. Treatment effect on moisture retention at different tensions. Different letters at each tension indicate significant difference at  $p < 0.05$ . Acacia, croton, and eucalyptus are wood charcoals, and corn and oak are biochars.

Surprisingly, available water content was neither affected by charcoal, nor by biochar (Figure 2-3). Available water content was also not affected by elevation (Figure 2-4), though elevation did significantly affected soil water retention at lower (10 and 30 kPa) and higher (1500 kPa)

tensions (Figure 2-4). TukeyHSD mean comparison results indicated that at these tensions, soils at low elevations retained significantly more water than soils at high elevations.

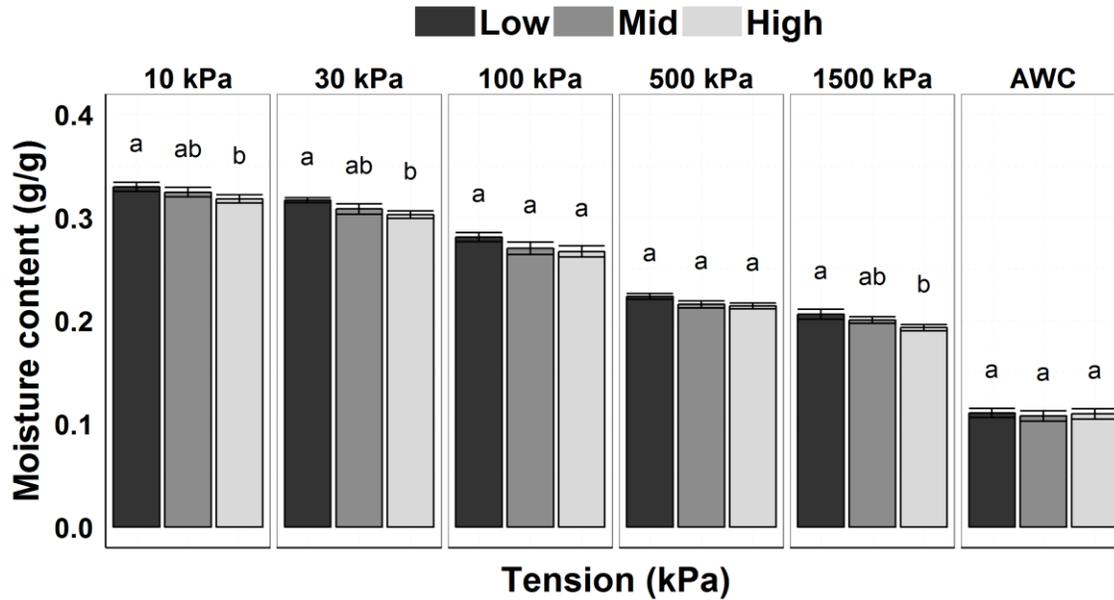


Figure 2-4. Effect of elevation (low-mid-high) on moisture retention at different tensions. Different letters at similar tension indicate significant difference at  $p < 0.05$ .

Results from column weight measurements corresponded with the soil water retention data obtained from pressure plates (Figure 2-5). Biochar from oak feedstock and all wood charcoals decreased water retention during most observation days; and treatment effects were significant for the first two days (Figure 2-5).

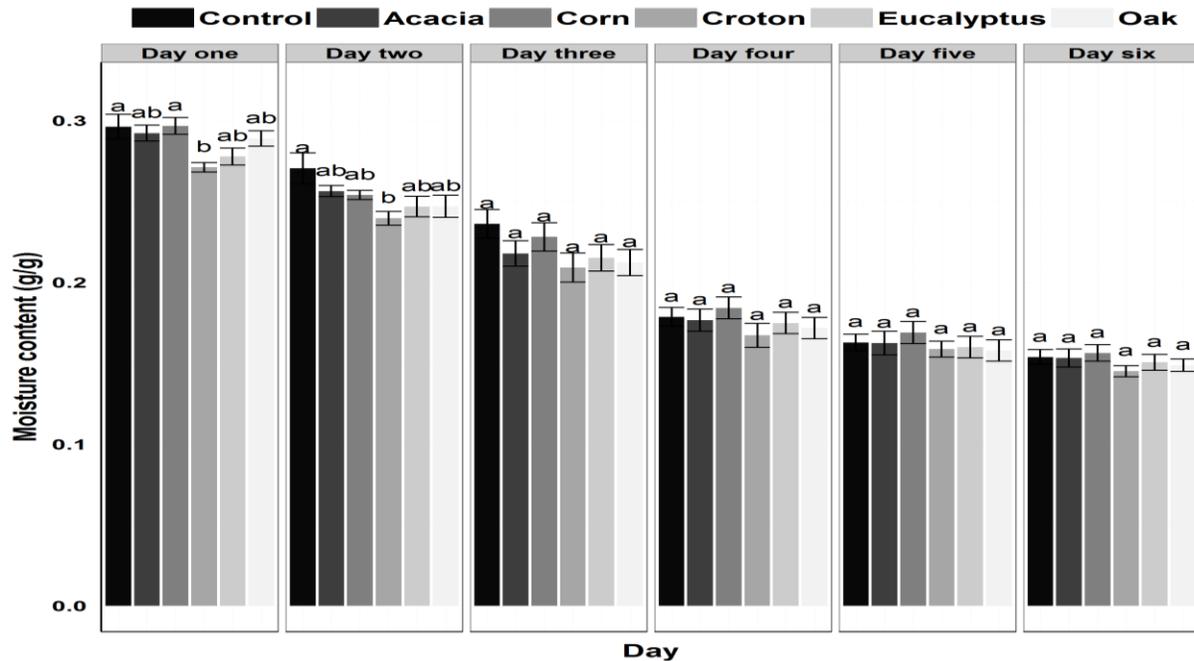


Figure 2-5. Summary of treatment effects on soil moisture retention by day. Values are averages of replications (n = 9). Bars with different letters (with in the same day) indicate significant difference ( $p < 0.05$ ). Acacia, croton, and eucalyptus are wood charcoals, and corn and oak are biochars.

TukeyHSD mean comparison results indicated that amended soils retained significantly less water than the non-amended control after one day of free drainage (croton and eucalyptus charcoal; oak biochar), and after two days of free drainage (croton charcoal only). There was no significant effect of elevation on water retention in these free drainage column experiments, for any of the observation days.

Both the pressure plate data and the column weight experiments corroborate that wood charcoal amendments were effective in reducing soil moisture retention near saturation, without affecting available water content, while reduction from oak biochar was not significant.

### 2.3.4 Effects of biochar and charcoal on soil hydraulic properties

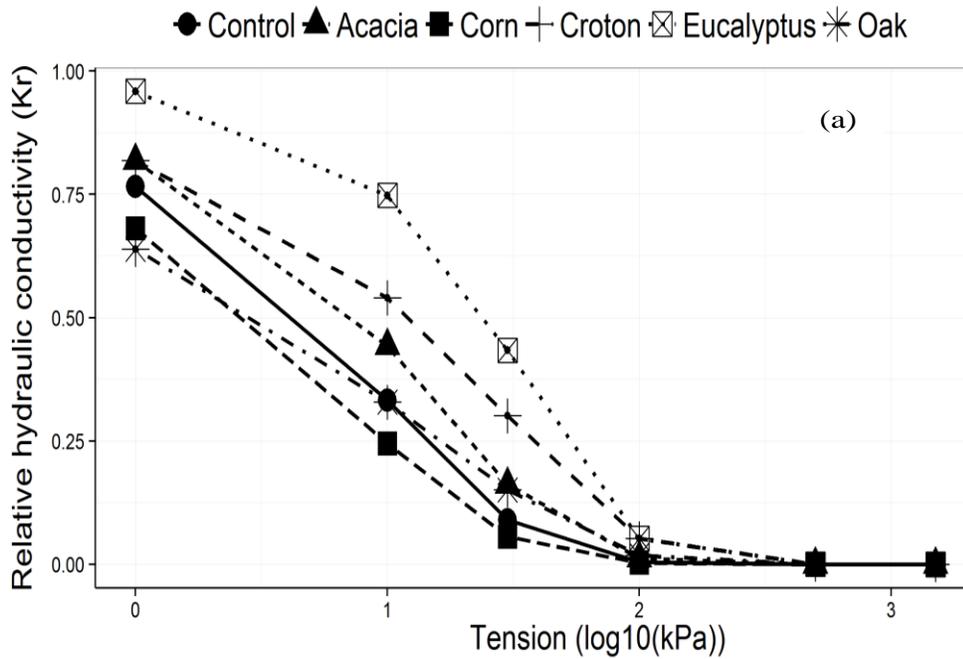
The Van Genuchten (1980) model fitted the observed data well, with  $R^2$  between 0.89 and 0.94 and RMSE coefficients between 0.01 and 0.02 (Table 2-2). As expected, the model under-predicted residual moisture content ( $\theta_r$ ) for all treatments compared with observed values at 1500 kPa (Figure 2-3). On average, fitted  $\alpha$ -values (inverse of air entry pressure) ranged from 0.01 to 0.03  $\text{kPa}^{-1}$  and n-values from 1.50 to 1.96.

Table 2-2: Summary of the Van Genuchten model fitting parameters and goodness of fit for charcoal and biochar treated and control soils. Results are based on combined data from all three elevation ranges together (n=3).

Treatment	$\theta_r$	$\theta_s$	n	$\alpha$	$R^2$	RMSE
		(g/g)	(-)	( $\text{kPa}^{-1}$ )		
Control	0.18 <sup>a</sup>	0.34 <sup>a</sup>	1.59 <sup>a</sup>	0.03 <sup>ab</sup>	0.90	0.02
Biochar						
Corn	0.17 <sup>a</sup>	0.35 <sup>a</sup>	1.50 <sup>a</sup>	0.03 <sup>ac</sup>	0.89	0.02
Oak	0.16 <sup>a</sup>	0.32 <sup>bc</sup>	1.50 <sup>a</sup>	0.02 <sup>bc</sup>	0.91	0.01
Wood charcoal						
Acacia	0.17 <sup>a</sup>	0.33 <sup>ab</sup>	1.65 <sup>a</sup>	0.02 <sup>ab</sup>	0.94	0.01
Croton	0.17 <sup>a</sup>	0.31 <sup>c</sup>	1.65 <sup>a</sup>	0.01 <sup>b</sup>	0.91	0.01
Eucalyptus	0.18 <sup>a</sup>	0.32 <sup>c</sup>	1.96 <sup>a</sup>	0.01 <sup>b</sup>	0.94	0.01

Interestingly, average n-values of all charcoal amendments (acacia, croton, and eucalyptus) exceeded those of the control treatment, while n-values of the biochars (corn and oak feedstocks) were smaller than the control (Table 2-2), indicating that the capillary rise was less for charcoal treatments and therefore consistent with the results in Figure 2-3.

The values in Table 2-2 allow us to look at the effects of biochar and charcoal on relative hydraulic conductivity ( $K_r$ ) of soils as a function of tension and soil moisture content, by calculating relative hydraulic conductivity rates using Eq. 2-3. This is shown in Figure 2-6, which illustrates the distinct differences between relative hydraulic conductivity rates at low tensions (<100 kPa, Figure 2-6a) and high moisture contents (approximately > 0.28 g/g, Figure 2-6b). In these tension and moisture content ranges, all charcoals (acacia, croton, and eucalyptus) had relatively greater relative hydraulic conductivity ( $K_r$ ) coefficients, while both biochars (corn and oak) had lower  $K_r$  coefficients compared with the control.



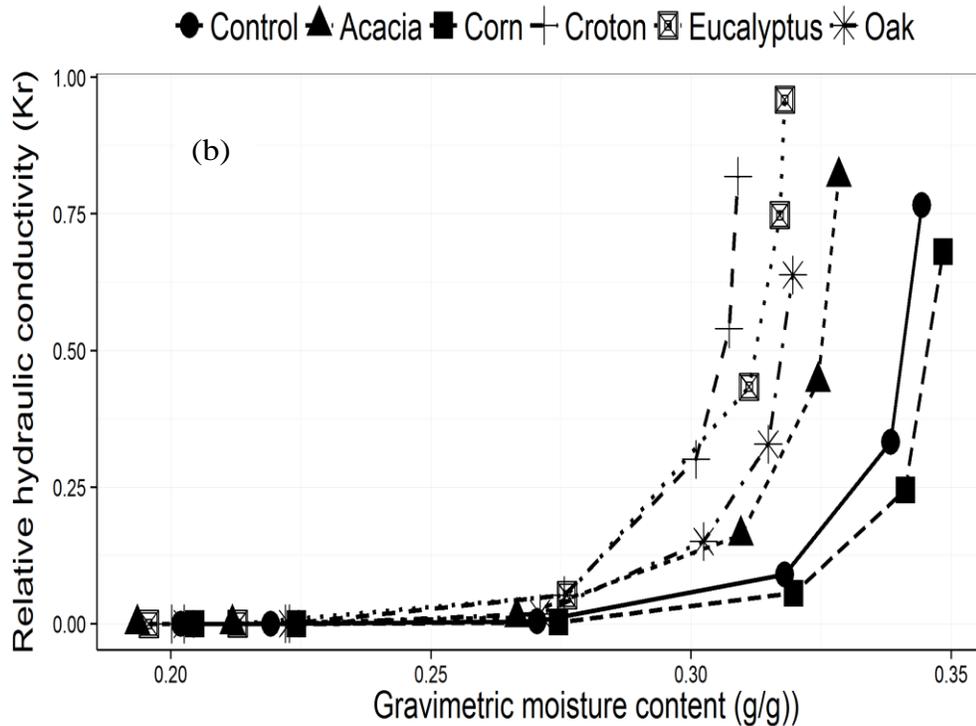


Figure 2-6. Relative hydraulic conductivity curves as a function of tension (a) and moisture content (b). Acacia, croton, and eucalyptus are wood charcoals, and corn and oak are biochars.

## 2.4 Discussion

### 2.4.1 Soil physical properties

Following USDA classification (USDA, 1999), the soils in the Anjeni watershed can be classified as clay loam (low elevations) to clay soils (mid to high elevations). Interestingly however, most studied parameters except pH were not significantly affected by elevation. These findings are in agreement with those of Adgo et al. (2013) and Assefa (2007) who found similar results for the Anjeni Watershed (Table 2-1). Likewise, these results (Table 2-1) concur with several authors who concluded that soils in the Ethiopian highlands area acidic (Chibsa and Ta, 2009; Demelash and Stahr, 2010; Feoli et al., 2002) and that its soil organic carbon pool is depleted (Hailu et al., 2012; Taddese, 2001; Zeleke et al., 2004). Soil acidity in the region is

partly due to continuous weathering processes and leaching of base cations (Amare et al., 2013; Hodnett and Tomasella, 2002), while depletion of soil organic carbon is further exacerbated by scarcity of farm inputs (including organic biomass) among other factors (Abegaz and Van Keulen, 2009; Feoli et al., 2002; Taddese, 2001). Organic carbon serves as a bridge (binding material) between primary soil particles (Bronick and Lal, 2005), and it is commonly accepted that both acidic pH (Dexter, 1988) and depletion of organic carbon (Bronick and Lal, 2005; Dexter et al., 2008; Hati et al., 2007; Lal, 2004; Reeves, 1997; Reynolds et al., 2007; Watts and Dexter, 1997) can enhance clay dispersion. A study by Dexter (1988) suggested that low pH results in net negative surface charges on clay particles that subsequently induce clay dispersion due to increased inter particle repulsion. Clay dispersion causes soil structural deterioration by blocking larger (hydraulically active) pores, causing a reduction in soil permeability (Chen et al., 1983; Daoud and Robert, 1992). Combined impacts of low organic carbon contents and low pH in these clayey soils therefore suggest high vulnerability to deteriorated soil physical condition (e.g. poor structural aggregation and stability), poor permeability (Watts and Dexter, 1997), and subsequent initiation of overland flow from open fields and waterlogged conditions on poorly drained fields (Temesgen et al., 2012) unless soil permeability is improved through appropriate management (Bayabil et al., 2010).

#### ***2.4.2 Infiltration capacity and storm intensity***

The soils of Anjeni have developed from the basaltic Trapp series of Tertiary volcanic eruptions and is similar to most parts of central Ethiopia, with major soils: Alisols (41.5ha) and Nitisols (23.8ha) around 60% of the watershed area (SCRIP, 2000; Zeleke, 2000), which could suggest good infiltration.

In contrast, however, construction of shallow ditches (10 - 15 cm deep) by local farmers (Figure D1) supports the abovementioned view that prevalence of deteriorated physical conditions and poor permeability of soils in the Anjeni watershed. Moreover, compared with reports from similar watersheds, infiltration rates in Anjeni (Table 2-1, Figure 2-2) were relatively lower: Engda (2009) and Derib (2005) for instance reported steady infiltration rates of 24-870 and 19-600 mm/h for the Andit Tid and Maybar watersheds, respectively. Like the Anjeni watershed, these watersheds are also located in the highlands, though Andit Tid and Maybar are situated at higher elevation (3040-3548 m and 2530-2858 m, respectively (Herweg and Ludi, 1999) vs. 2407-2507 for Anjeni) with steeper gradients.

Comparison of storm intensities and steady soil infiltration rates, as shown in Figure 2-2, suggests that for the far majority of rainstorms, infiltration capacity considerably exceeds storm intensity. This indicates that saturation excess runoff, rather than infiltration excess runoff, is the root cause of observed overland flow in the Anjeni watershed. This is supported by a study by Tilahun et al. (2011) who analyzed long term rainfall and discharge data at the watershed outlet and reported that saturation excess runoff (mainly from saturated areas) was the dominant runoff mechanism.

### ***2.4.3 Changes in soil hydraulic properties due to biochar and charcoal***

The observed reduction in soil water retention at low tensions (near saturation) due to woody biochar and charcoal amendments (Figure 2-3) is in agreement with previous study (Tryon, 1948), that observed significant reduction in water retention of clayey soils after incorporation of charcoal. These findings are also in line with the observed increase in both in relative hydraulic conductivity ( $K_r$ ) at similar tensions (Figure 2-6) and the Van Genuchten model parameter  $n$  for

most soils amended with woody biochar and charcoal (Table 2-2). These *n*-values suggest steeper slopes of the soil water retention curve, which results in a significant reduction in soil moisture content for small changes in tension (Hodnett and Tomasella, 2002).

In contrast to charcoal, corn biochar (prepared from corn stover) did not decrease but rather increase soil water retention or had no effects (Figure 2-3). In other studies mainly for sandy soils, organic amendments including biochar enhanced soil water retention (Abel et al., 2013; Bauer and Black, 1992; Feoli et al., 2002; Glaser et al., 2002; Hollis et al., 1977; Rawls et al., 2003) as well as available water content of medium textured soils (Emami and Astarai, 2012; Karhu et al., 2011). Differences in impacts of biochar and charcoal on soil hydraulic properties could be due to variations in physico-chemical properties of feedstock sources (Enders et al., 2012; Verheijen et al., 2009). Physico-chemical properties of organic amendments may affect soil hydraulic properties in different ways. Direct substitution of clay particles by relatively larger biochar or charcoal particles might improve soil permeability by inducing tensile stresses around clay matrixes causing the formation of macropores or cracks as suggested by Dexter (1988) or just due to simple rearrangement of soil particles without altering total porosity of soil (Nimmo, 1997). For clayey soils, a small increase in macroporosity can significantly affect water flow near saturation (Eusufzai and Fujii, 2012; Sharma and Bhushan, 2001), whereas at higher tensions soil water retention is mainly affected by clay particles (texture), and thus organic amendments have diminished impacts (Saxton and Rawls, 2006). In line with this, Tryon (1948) reported coarse charcoal particles to be more effective in reducing moisture retention of clayey soils than fine charcoal particles. This would explain why the (coarser) charcoal significantly reduced water retention in the wet range of the water retention characteristic, while (finer)

biochar only slightly caused a reduction in this range (Figure 2-3). Finally, biochar and charcoal amendments could also alter structural aggregation and stability of soils. Biochar and charcoal particles can bond with soil mineral surfaces through carboxylic and phenolic functional groups thereby contributing soil aggregate and structural stability (Soenne et al., 2014).

Another potential explanation for the fact that biochar and charcoal had different effects may lie in the interaction between biochar/charcoal and clay, and the mechanisms by which biochar and charcoal could alter the chemistry of clay particles. Several studies reported that substituting monovalent cations ( $\text{Na}^+$  and  $\text{K}^+$ ) on exchange sites of clay particles by divalent cations with high charge density (such as  $\text{Ca}^{2+}$  and  $\text{Mg}^{2+}$ ) enhanced clay flocculation, while the reverse processes induces clay dispersion (Dexter, 1988; Emami and Astarai, 2012; Marchuk and Rengasamy, 2010). Clay dispersion often leads to clogging of macropores (Dexter, 1988; So and Aylmore, 1993), whereas flocculation of clay particles enhances macropores size and network (Rao and Mathew, 1995). Another study by Chen et al. (1983) reported the major mechanism for hydraulic conductivity reduction to be the dispersion of the ‘fine soft fraction’ (mostly clay aggregates) and its rearrangement *in situ* to form a dense network of particles and smaller pores, and not the extensive migration of clay and the subsequent formation of an impermeable layer.

Low hydraulic conductivity ( $K_r$ ) coefficients for corn biochar, at low tensions, were in accordance with higher sodium adsorption ratios 2, 3, and 8 times higher and potassium adsorption ratios 7, 49, and 62 times higher than oak biochar, and acacia and eucalyptus charcoal amendments, respectively (Table E1). High sodium adsorption ratio (SAR) (Dexter, 1988; Emami and Astarai, 2012) and high potassium adsorption ratio (PAR) (Chen et al., 1983;

Marchuk and Rengasamy, 2010) induced clay dispersion, but with varying magnitude depending on clay mineralogy (So and Aylmore, 1993). This suggests that, in addition to soil physical properties (texture), clay mineralogy, as well as elemental constituents of amendments could significantly affect impacts of pyrolyzed organic amendments.

## **2.5 Conclusion**

In the Anjeni watershed, half of the catchment area generates infiltration excess runoff 23% of the time (Figure 2-2). On these areas, management practices should focus on improving soil infiltration rates. Wood charcoal and biochar incorporation reduced soil moisture retention at lower tensions (<100 kPa) by increasing relative hydraulic conductivity ( $K_r$ ) at these tensions. This was likely because of improved pore networks caused by binding clay particles that otherwise plug the major pathways for drainage. Therefore, we conclude that woody charcoal (acacia, croton, and eucalyptus) and biochar (oak) incorporation can improve soil physical properties (such as hydraulic conductivity) of degraded soils, which in turn could potentially reduce runoff, erosion, and field waterlogging. Results furthermore suggest that wood charcoal amendment may even be more effective than biochar, as biochar amendments (corn and oak) considered did not result in a significant improvement in these soil hydraulic parameters, for the soils considered here. Since none of the amendments significantly changed available water capacity, this study finally indicates that amendment with wood charcoals can improve soil drainage while having no effect on plant available water.

Overall findings of this study imply that decades of soil and water management planning approach needs to be adjusted. Future soil and water management practices need to target causes

of runoff and erosion in relation to the dominant rainfall characteristics and the state of soil physical properties in a landscape. This study indicates that wood charcoal can be a viable low-cost alternative for improving soil physical properties, for instance in places like rural Africa where high-tech biochar is not available or too costly. However, a word of caution is needed here as all biomass serves multiple purposes in daily livelihoods of smallholder farmers. Future studies therefore need to include socio-economic factors to verify feasibility of biochar and charcoal use as soil amendments.

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## 2.8 Appendix

### Appendix A. Correlation between soil properties

**Table A 1.** Linear correlation between observed soil properties

	$f_s$ (mm/h)	BD (g/cm <sup>3</sup> )	pH (-log [H <sup>+</sup> ])	Clay	Silt	Sand (%)	OC
$f_s$	1.00	-0.25	-0.24	-0.13	0.03	0.13	0.12
BD		1.00	-0.01	0.20	-0.13	-0.11	-0.12
pH			1.00	-0.18	0.22	-0.01	-0.12
Clay				1.00	-0.60	-0.62	-0.02
Silt					1.00	-0.26	0.05
Sand						1.00	-0.03
OC							1.00

$f_s$ : steady infiltration rate, BD: bulk density, and OC: organic carbon content.

**Appendix B. Mean monthly rainfall and discharge, and storm characteristics**

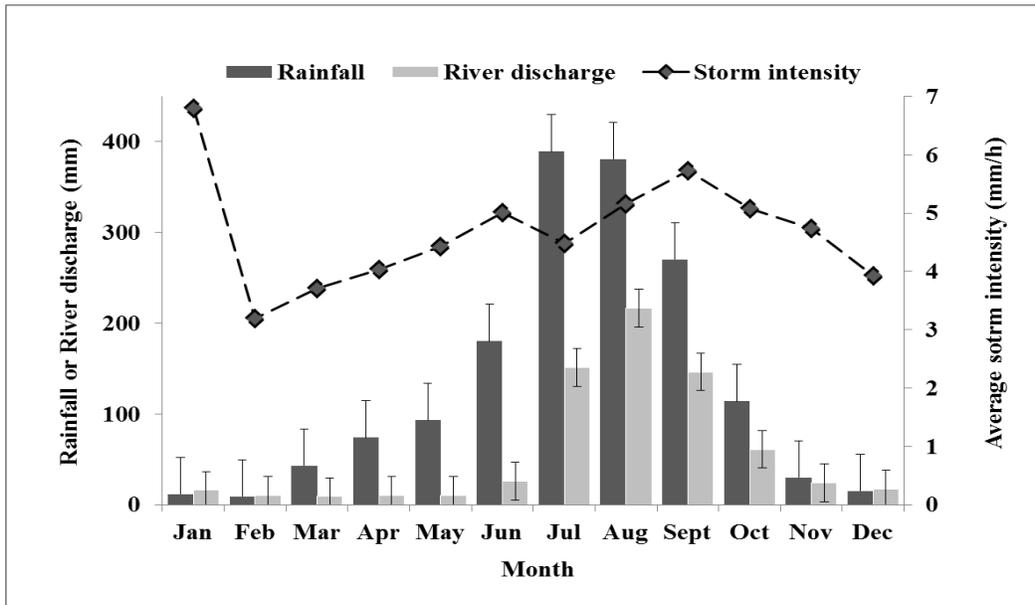


Figure B1. Monthly rainfall, discharge, and average storm intensity in the Anjeni watershed based on five year (1989 to 1993) data analysis results.

### Appendix C. Mean monthly rainfall and discharge, and storm characteristics

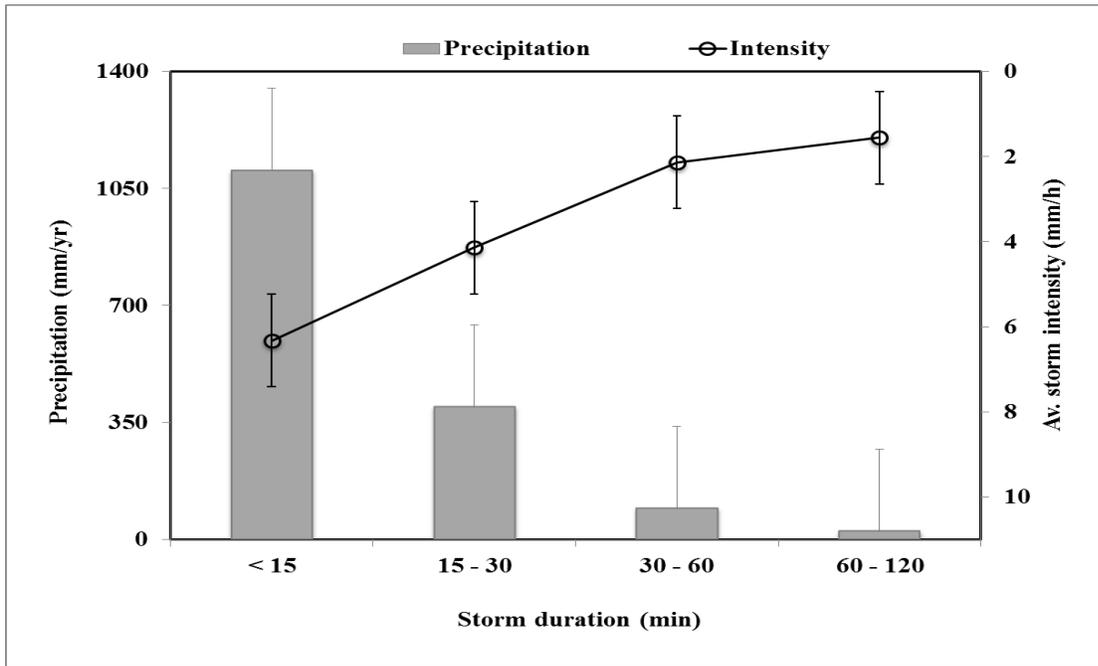


Figure C1. Changes in storm characteristics with duration ranges: contribution to annual precipitation and average storm intensity based on five year (1989 to 1993) observations.

## Appendix D. Waterlogging issues and farmers practices

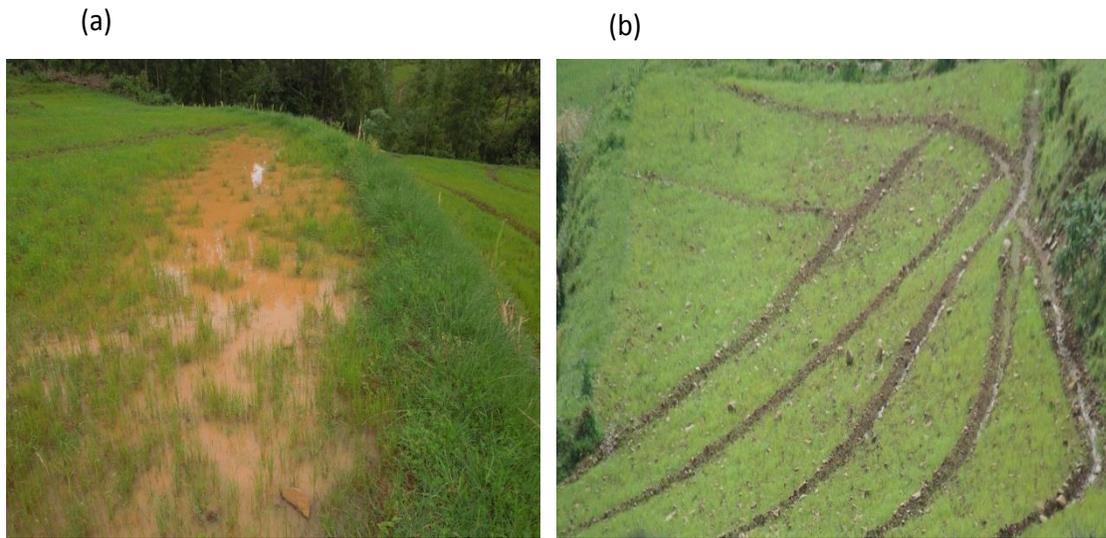


Figure D1. Field waterlogging (a), and networks of shallow drainage ditches (b).

## Appendix E. Exchangeable cations concentrations for biochars and selected charcoals

Table E1 Exchangeable bases, adsorption ratio of sodium (SAR) and potassium (PAR) for biochar and selected charcoal treatments. Data for corn and oak were averages of 400 and 500°C readings of respective feedstock (adapted from Endres et al., (2012)).

Treatment	*pH	Ca <sup>2+</sup>	Mg <sup>2+</sup>	K <sup>+</sup>	Na <sup>+</sup>	SAR	PAR
		(mmol <sub>C</sub> /kg)				(mmol <sub>C</sub> /kg) <sup>0.5</sup>	
Corn	9.4	194.7	161.2	427.25	4.45	1.05	101.28
Oak	7.5	12.6	0.30	11.1	0.50	0.62	13.82
Acacia	8.2	0.08	0.10	0.15	0.01	0.13	1.64
Eucalyptus	8.7	0.07	0.01	0.13	0.03	0.40	2.07

# CHAPTER 3: SPATIAL AND TEMPORAL RUNOFF PROCESSES IN THE DEGRADED ETHIOPIAN HIGHLANDS: THE ANJENI WATERSHED<sup>1</sup>

## Abstract

As runoff mechanisms in the Ethiopian highlands are not well understood, performance of many soil and water conservation measures is inadequate because of ineffective placement outside the major runoff source areas. To improve understanding of the runoff generating mechanisms in these highlands, we monitored runoff volumes from 24 runoff plots constructed in the 113 ha Anjeni watershed, where historic data of rainfall and stream discharge were available. In addition, we assessed the effectiveness of charcoal and crop rooting depth in reducing runoff, in which we compared the effect of lupine (a deep-rooted crop) to that of barley. Daily rainfall, surface runoff, and root zone moisture contents were measured during the monsoon seasons of 2012 and 2013 (with all plots being tilled in 2012, but only barley plots in 2013). In addition, long-term surface runoff (from four plots) and outlet discharge data from the research site (1989-1993) was analyzed and compared with our observations. Results showed that the degree of soil degradation and soil disturbance (tillage) were significant factors affecting plot runoff responses. As expected, runoff was greater from more degraded soils, while tilled plots had greater soil storage and thus less runoff. Overall, barley plots produced significantly less runoff than lupine plots.

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Specifically, considerable difference was observed for smaller rainfall events (ca. < 20 mm) in 2013, when lupine plots (non-tilled) resulted in greater runoff than barley plots (tilled). This suggests that plot rainfall-runoff relationships are greatly affected by root-zone storage, which is directly affected by soil degradation and tillage practices.

*Keywords:* rainfall, runoff, topography, crop cover, soil and water management

### **3.1 Introduction**

Soil and water conservation practices are ubiquitous in the Ethiopian highlands, and necessary to counteract the loss the top fertile soil from farmlands. However, surprisingly, most non-traditional soil and water conservation practices are ineffective because their placement is often not sufficiently aligned with where the runoff occurs. Planning effective soil and water management measures require knowledge of where runoff hotspots are located in a landscape. While most areas in the Ethiopian highlands receive a high amount of annual precipitation, its distribution is variable both spatially and temporally (Biazin et al., 2011; Bitew et al., 2009; McHugh et al., 2007). Location of the hotspots depends on whether runoff is generated by infiltration or saturation excess. There is no agreement on the causes of runoff and erosion in the Ethiopian Highlands. Previous studies highlight land use and topography as the critical factors in governing runoff processes (Bayabil et al., 2010; Bewket and Sterk, 2005; Taddese, 2001; Tilahun et al., 2013), mainly based on analysis of changes in the hydrograph at the outlet. Land use is important because it controls stream flow volumes. For example, several studies reported land use change from natural vegetation to agricultural lands to increase discharge during the

rainy monsoon phase and reduce base flows during the dry phase (Bewket and Sterk, 2005; Feoli et al., 2002; Taddese, 2001; Zeleke, 2000).

Topography is important because of its control on water routing and thus where the runoff goes after it has been generated. A field study by Bayabil et al. (2010) found that in the Maybar watershed, with highly conductive soils, topography was the most important factor for runoff initiation by channeling water through the hillsides as interflow and saturating the lower lying fields that became hotspot areas for runoff. Likewise, in Debra Mawi watershed in the northern Ethiopian highlands, both degraded hillslope soils and the saturated lower lying fields contributed most of the surface runoff (Tilahun et al., 2013).

Many of the previous watershed studies did not consider the role of hardpans on runoff. A hardpan is a restrictive layer that impedes downward flow of water and growth of plant roots (Biazin et al., 2011; Tebebu et al., 2013; Temesgen et al., 2009). Hardpans are ubiquitous in the Ethiopian Highlands and have in many cases formed after deforestation of the primary forests. Once the trees are removed, the soil loses its organic matter and becomes more acidic. As discussed by (Bayabil et al., 2015), clay particles in these acidic soils can disperse easily and therefore be picked up in the infiltrating water and cause plugging of the original macropores. Tebebu et al. (2013) found that hardpan formation in the Anjeni watershed was greater on intensively cultivated agricultural fields compared with forest land. Temesgen et al. (2009) observed peak penetration resistance of soils at 20 cm depth. In other countries as well, soil degradation after cutting down the forests resulted in decreased infiltration rates (Hanson et al., 2004; Mendoza and Steenhuis, 2002; Nyberg et al., 2012; Shougrakpam et al., 2010). Installation of physical soil and water conservation measures (eg. *fanya juu* terraces in the Anjeni watershed) on fields with hardpan will cause waterlogged conditions and runoff will not be reduced unless

soil infiltration is improved (Bayabil et al., 2010; Temesgen et al., 2012). This can be done by restoring the macropore network through the hardpan (either physically or biologically), and by improving soil organic matter contents to limit further clay dispersion. As traditional tillage practices using oxen pulled plow only loosen the top 10-15 cm of the soil (Biazin et al., 2011; Temesgen et al., 2012), these methods are inadequate for disrupting the more deeply located hardpans.

A solution may be planting deep-rooted crops that penetrate the hardpan and thereby increase hardpan conductivity (Angers and Caron, 1998; Cresswell and Kirkegaard, 1995; Lesturgez et al., 2004; Meek et al., 1992). Another solution, improving soil acidity and organic carbon pool through the addition of biochar or charcoal, which is known to improve soil physical and hydraulic properties (Abel et al., 2013; Asai et al., 2009; Bayabil et al., 2015; Glaser et al., 2002; Kameyama et al., 2010; Karhu et al., 2011; Laird et al., 2010; Spokas, 2010). Although biochar and charcoal amendments can both be effective in improving soil water relationships, (Bayabil et al., 2015) argued that charcoal to be a more viable solution for rural Africa because it is widely produced in most rural areas of Africa (Lehmann et al., 2006) and therefore more accessible to smallholder farmers than biochar.

Since effective soil and water management requires accurate understanding of runoff generating mechanisms, the objective of this study was, therefore, to assess the drivers of runoff generation in the Ethiopian Highlands. We characterized effects of soil degradation status and landscape position, and investigated the effects crop rooting depth (barley with and without charcoal, and deep-rooted lupine crop) on spatial and temporal rainfall-runoff relationships. Understanding rainfall-runoff processes and identifying runoff source areas in the landscape will aid future

efforts towards planning effective soil and water management practices that allow improved use of green (rain) water to boost smallholder farm productivities.

## **3.2 Materials and Methods**

### ***3.2.1 Study site***

This study was carried out in the Anjeni watershed, situated in the northwestern part of Ethiopia (Figure 3-1). Among other reasons, this watershed was selected because of the availability of historic discharge records at the outlet and from runoff plots inside the watershed. The watershed has a drainage area of 113 ha and is one of the experimental watersheds established under the Soil Conservation Research Program (SCRP) of the Ministry of Agriculture of Ethiopia in collaboration with the Swiss Agency for Development and Cooperation (SDC) (Hurni et al., 2005). Its gauging station is located at 10°40' N, 37°31' E. The watershed has a unimodal rainy season that lasts from mid-May to mid-October, with a mean annual rainfall of 1690 mm yr<sup>-1</sup>. The topography of Anjeni is typical of Tertiary volcanic landscapes; it has been deeply incised by streams, resulting in the current diversity of landforms (SCRP, 2000) with elevation between 2407 and 2507 m (Herweg and Ludi, 1999). The soils of Anjeni have been developed from the basaltic Trapp series of Tertiary volcanic eruptions and is similar to most parts of central Ethiopia with major soils Alisols (41.5 ha), Nitisols (23.8 ha), Cambisols (18.9 ha), and Regosols (10 ha) covering more than 80% of the watershed (Figure A1 in Appendix material A) (SCRP, 2000; Zeleke, 2000). The deep Alisols cover the bottom part of the watershed; moderately deep Nitisols cover the mid-transitional, gently sloping parts of the watershed, while the shallow Regosols and Leptosols cover the high, steepest part of the watershed (Zeleke, 2000). Fields are intensively cultivated for crop production and large proportion of the watershed is degraded

(SCRIP, 2000). In 1986, graded fanya-juu structures were installed resulting in terraces across the landscape (SCRIP, 2000).

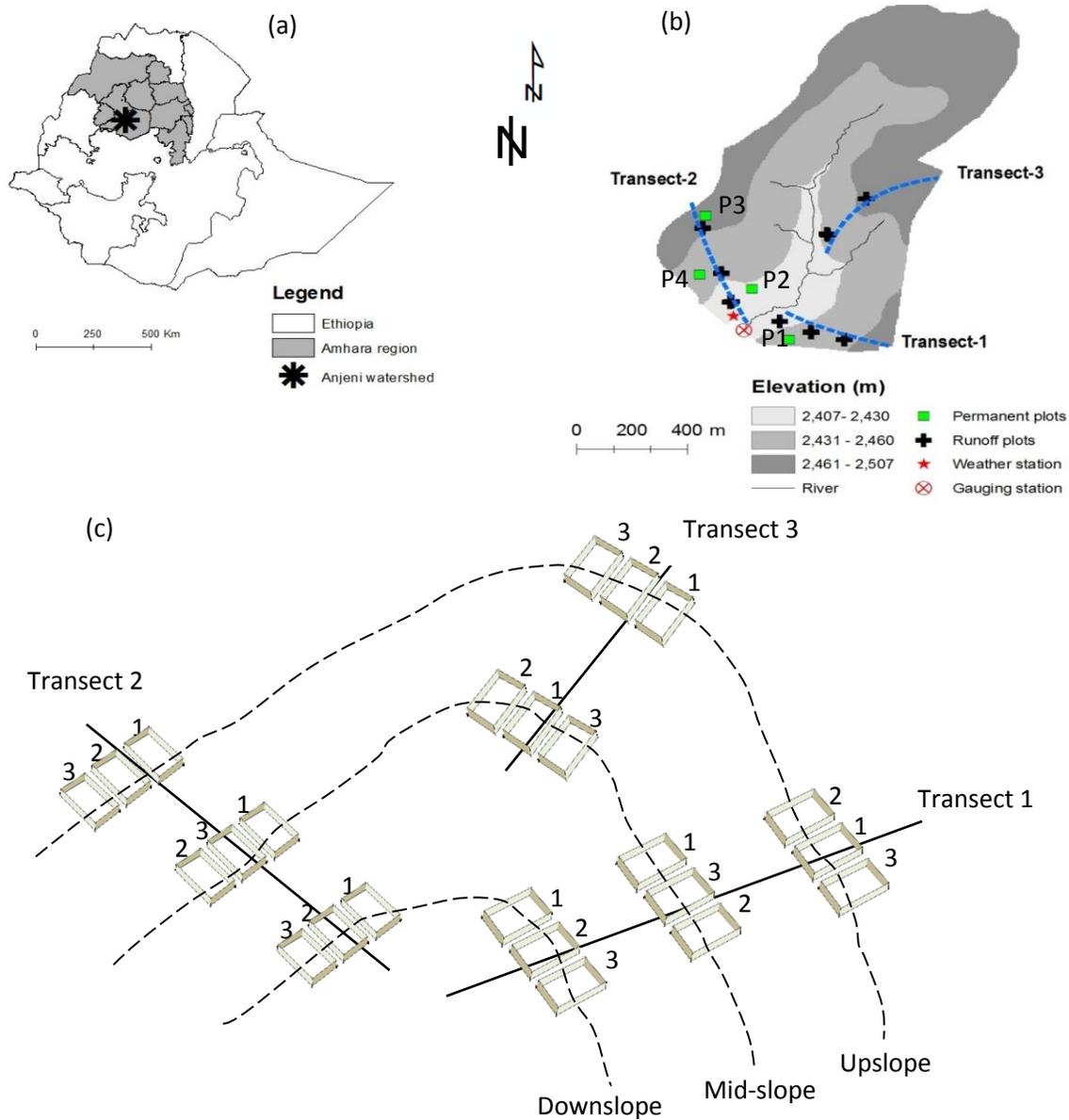


Figure 3-1. Location of the Anjeni watershed in the Amhara region in Ethiopia (a), with the location of downslope transects and runoff plots indicated in (b) and (c – not to scale). Dashed lines in (c) are elevation contours. Plots labels represent treatment types: 1 = barley without amendment (Control), 2= barley with charcoal, and 3= Lupine.

### 3.2.2 *Experimental setup*

Runoff generating mechanisms were studied using 24 runoff plots installed across the watershed, accounting for spatial variability in soil degradation status and slope position (Figure 3-1). Effects of charcoal amendment and crop rooting depth were assessed for each transect location. The 24 plots were positioned in groups of three along three transects perpendicular to the slope (Figure 3-2). Soil degradation varied between transects: Transects 1 and 2 are located in the southeast and southwest part of the watershed (Figure 3-1b), and have deep soils while Transect 3, located between Transects 1 and 2, is characterized by shallow and degraded soils. Transects 1 and 3 are steep (with slopes ca. 14.5 and 15.6 %, respectively), while Transect 2 has moderate slope (ca. 11.8%). Effects of landscape position were assessed by placing plots at different slope positions: at downslope, mid-slope, and upslope positions along Transects 1 and 2; and at the two upper positions along Transect 3 (Figure 3-1c).



Figure 3-2. Groups of three runoff plots setup at downslope position along Transect 2. Water storage tanks are positioned below the plots, on the downslope side of the terrace edge. Dark brown lines above runoff plots are traditional conservation practices (drainage ditches) constructed by farmers to channel out excess water from fields.

At the start of the 2012 growing season (June), all plots were plowed and two plots were seeded with barley. Effects of charcoal amendment were assessed by amending one of the barley plots with charcoal during plowing, the non-amended barley plot serving as a control treatment. Effects of crop rooting depth were assessed by seeding the third plot at each transect location with the deep-rooted lupine (*Lupinus albus* L.) crop, with again the non-amended barley plot serving as a control treatment. Barley and lupine crops were assigned randomly to plots; and the same crop was maintained on each plot for two years (2012 and 2013). These crops were chosen as they are widely grown throughout the Ethiopian highlands. Farmers grow lupine as intercrop with cereals (eg. barley and wheat) or as sole crop on marginal lands without additional farm inputs.

### ***3.2.3 Agronomic practices on plots***

Barley, one of the predominantly grown crops in the watershed (SCRIP, 2000), was grown following local farmers' cultural practices and thus barley plots were tilled in both 2012 and 2013. While, lupine seedbeds are typically not tilled, tillage was done in 2012 as plots were originally designated to be sown with alfalfa, another deep rooted crop. When alfalfa proved to be unsuccessful, lupine was sown on the tilled soil. The year after, in 2013, only barley plots were tilled and seeded, while lupine seeds were sown on untilled plots, which is a more common practice in the area. Also in line with farmer practices, all barley plots were fertilized with 100 kg/ha Di-Ammonium Phosphate (DAP; 46% Nitrogen, 23% Phosphorous, and 21% Potassium) during seeding, and 100 kg/ha of Urea (100% Nitrogen) one month after sowing. While lupine plots were not fertilized.

Moreover, on charcoal-amended barley plots, charcoal was applied at a fixed rate of 12 ton/ha during tillage in 2012 and 2013. Charcoal (prepared from *Eucalyptus camaladulensis* biomass in a way similar to that described by Bayabil 2015) was manually crushed to obtain relatively uniform particle size (ca. 2 mm diameter) and then manually incorporated on the top 0.2 m of the soil.

### **3.2.4 Plot installation and data collection**

While crop and charcoal treatments were applied to 9 m<sup>2</sup> (3 m wide, 3 m long) areas, runoff was only measured from a central area of 4.5 m<sup>2</sup> (1.5 m wide, 3 m long), to avoid trampling and disturbing of the soil inside the runoff plots while taking auxiliary measurements such as soil moisture contents. For this, runoff plot boundaries were installed 0.75 m inside the seeded area from both sides. As illustrated in Figure 3-2 above, all runoff plots were constructed at the level bottom ends of terraces. The plot boundaries consisted of 50 cm high metal sheets of which 25 cm were belowground and 25 cm were aboveground, and the lower plot boundaries were reinforced with concrete. A 2-inch PVC pipe carried surface runoff into a primary collection tank (ca. 76 L volume). When the primary tanks were full, excess water flowed through divisor slots directing one-tenth (10%) of the excess flow into secondary tanks (ca. 76 L volume). The tanks were made from barrels cut in half and were covered on the top to minimize evaporation and prevent rainfall entry.

All runoff plots were monitored manually for runoff volumes on a daily basis during the monsoon season (from June 29 to October 4 in 2012 and from June 25 to October 8 in 2013). When runoff occurred, the depth of water in the two tanks was measured and then the water was drained out through valves fitted at the bottom of the tanks. Daily rainfall totals were measured

using a manual rain gauge installed at the weather station (see Figure 3-1b 'Weather station'). In addition, during the 2013 growing period, soil moisture contents,  $\theta$  ( $\text{g g}^{-1}$ ), were measured gravimetrically by taking bulk soil samples from the top 20 cm depth at 10-day intervals. To prevent disturbance, samples were taken inside the seeded area but just outside each runoff plots.

### ***3.2.5 Long-term plot runoff and river discharge data***

In addition to runoff data from the 24 newly installed plots, we obtained long-term data from the Amhara Regional Agricultural Research Institute (ARARI). The data consist of runoff from four long-term 30 m<sup>2</sup>-plots (2 m wide, 15 m long) (Figure 3-1b, 'Permanent plots') and discharge at the outlet of the watershed (Figure 3-1b, 'Gauging station'). To place our newly installed plot-scale runoff observations into a broader and longer-term context, we compared our data with historic plot-scale runoff data available in the watershed for the years 1989 through to 1993. These data were measured on 2 m wide by 15 m long plots with slopes of 12, 16, 22, and 28%. The 16% sloped plot was on grassland, while the other three plots were cultivated with food crops (eg. barley and wheat) (SCRIP, 2000). Discharge was measured continuously since 1984 (two years before the installation of the '*fanya juu*' conservation structures) as part of the ongoing hydrological and erosion monitoring activities (SCRIP, 2000), and we used discharge data for the 2012 and 2013 monsoon seasons to compare our plot-scale observations with watershed-scale patterns. Rainfall data obtained from the watershed (Figure 3-1b, 'Weather station') was available for the same period.

### ***3.2.6 Data quality control and aggregation***

We checked all daily data to make sure that peaks of daily rainfall and runoff coincided, both visually and by calculating the daily runoff coefficients (Rcoef; the quotient of the daily runoff

depth and precipitation). Plot-scale rainfall-runoff data (Figure B1-B3 in Appendix material B) showed that there were 214 events (spread over 11 and 32 days in 2012 and 2013, respectively) out of 5232 events total (4.1 %) where daily runoff was greater than corresponding rainfall amount recorded on the same day (i.e.  $R_{coef} > 1$ ). In some cases, large rainfall events were visible that did not produce runoff on the same day, but for which peak runoff appeared on the following day (see for example the black arrows in Figure B1-B3 in Appendix material B). In other cases, there was more runoff than rainfall without delays (see spikes of blue, green, and red lines in Figure B1-B3 in Appendix material B). Runoff in excess of rainfall can be caused by rainfall and runoff measurement periods that do not coincide. Here, rainfall was measured at 8 am each day. The first of the 24 runoff plots was also measured at 8 am but emptying the barrels and scooping out the sediment is time consuming, causing the last plot to be emptied around noon. Other potential causes for runoff exceeding rainfall are high spatial variation in rainfall not picked up by our single rain gauge, and interflow from outside the plot entering the plot during large rainstorms.

To reduce the impact of delayed peak runoff, we, therefore, decided to aggregate rainfall and runoff data over a 3-day period, resolving most of the high runoff coefficients. Yet 47 events (2.6% of total) observed from the 24 plots and recorded from 11 observation days spread over the two-year study period were left with  $R_{coef} > 1$  (Figure C1 in Appendix material C). Further data aggregation, even on a weekly interval, did not solve these high runoff events. One of the options to deal with such outlier data points would be excluding observations from data analysis. However, to avoid bias between treatments and spatial locations, all observations from those 11 days need to be discarded for all (24) plots, which would result in discarding 264 observations.

Losing this many observations (14.9% from 1777 total observations) would considerably reduce the power of our analysis.

Thus, to achieve a balance between the number of runoff events remaining for analysis and the objective to analyze large runoff events, such high runoff events ( $R_{coef} > 1$ ), after data aggregation on 3-day intervals, were therefore assigned a maximum value that equals the 3-day rainfall amount - resulting in a runoff coefficient of 1. As such, adjusted and 3-day aggregate runoff data were used for all statistical data analyses in this paper.

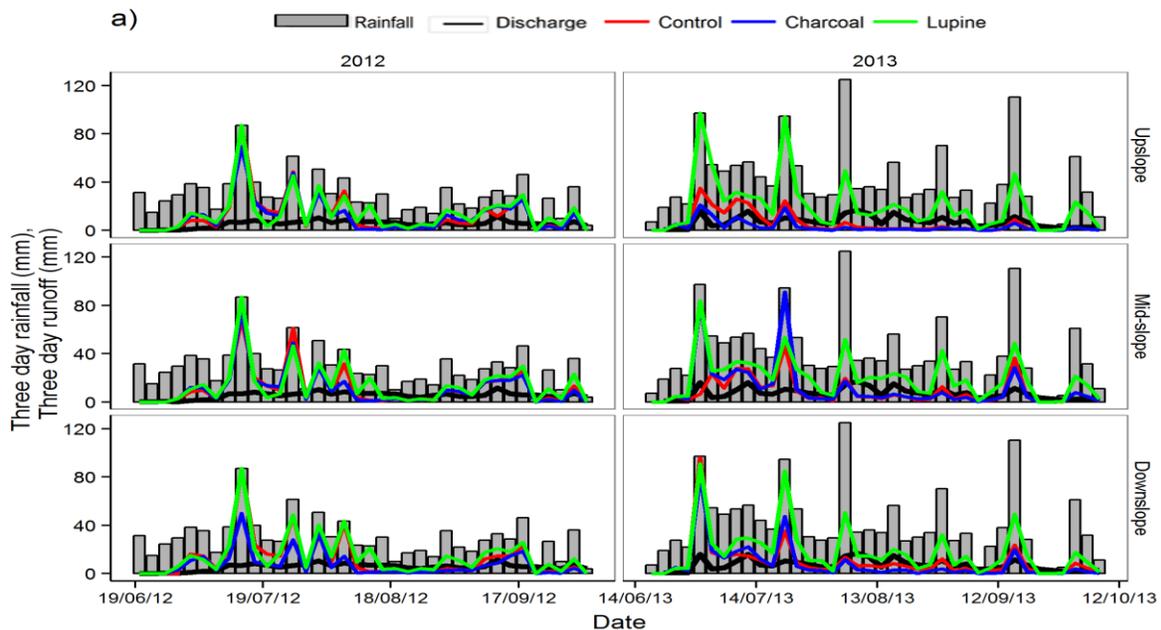
### ***3.2.7 Statistical analysis***

Data analysis aimed at detecting spatial and temporal trends in rainfall-runoff relationships during the two-year study period. Statistical data analysis was performed using R (R Development Core Team 2010). To determine the impact of charcoal amendment and deep-rooted lupine as well as spatial location with different soil degradation levels (transects) and slope position, a linear mixed effect model was fitted using the *nlme* package in R. In this model, crop type, slope position, and transect were used as fixed factors, and individual plots as random factors. For fixed factors with significant effects, post hoc mean comparison tests were performed using the *lsmeans* package in R to identify group pairs with significant difference.

### 3.3 Results and Discussion

#### 3.3.1 Plot-scale rainfall-runoff response and effect of charcoal amendment and deep-rooted lupine

The adjusted runoffs during the monsoon seasons of 2012 and 2013 for all eight groups of plots along three transect are shown in Figure 3-3. In 2013, runoff response from lupine plots was considerably greater than barley plots; while in 2012, runoff tended to be more or less similar for all treatments. In addition, a summary of observed rainfall and original (non-adjusted) runoff data recorded from all 24 plots is presented in Table D1 in Appendix material D. Average monthly rainfall in 2012 was similar to the 5-year average (based on 1989-1993 observations; Figure E1 in Appendix material E), while in 2013 it exceeded the 5-year average.





As discussed in the Methods section, runoff exceeding rainfall (i.e.  $R_{coef} > 1$ ), as shown in Figure B1-B3 in Appendix material B and Figure C1 in Appendix material C, is not expected and worrisome. We therefore checked historic long-term data (1989-1993) from four permanent plots measured by the well-trained technicians at the experimental station, and found the same “problem” that in many cases there was more runoff than rainfall (Figure 3-4a).

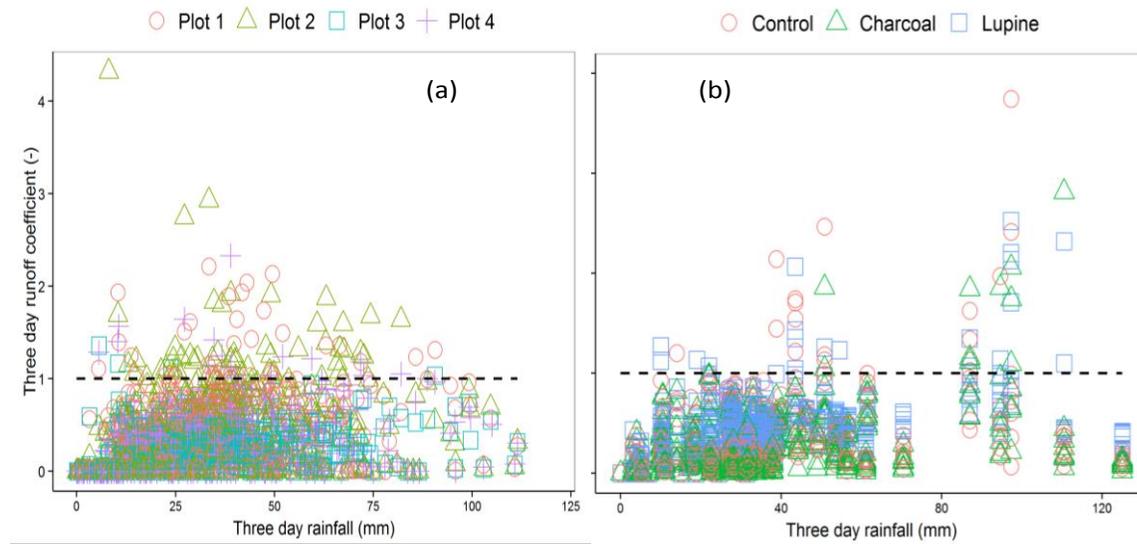


Figure 3-4. Runoff coefficients computed from observations from long-term monitoring plots (a) and plots in 2012 and 2013 (b). Black dash horizontal line represents  $R_{coef} = 1$

This indicates that our daily observations with  $R_{coef} > 1$  (Figure 3-4b) are real and not caused by measurement errors. This phenomena of runoff exceeding rainfall has not been reported often for temperate climates, and it is therefore likely that rainfall in monsoon climates is more variable over short distances than rains in temperate climates. Studies found that rainfall in the Ethiopian highlands significantly varies in space (Bewket and Conway, 2007; Bitew et al., 2009). Bitew et al. (2009) observed up to 424% coefficient of variation of daily rainfall between rain gauges over 1km distance. These authors further noted that in areas with complex topography (like the Anjeni

watershed), extrapolation of point rainfall observations to larger scales are prone to be inaccurate.

### 3.3.2 Plot runoff and outlet discharge

All plots on degraded soils along Transect 3 produced greater 3-day runoff than plots along the other two transects with relatively deeper soils (Figure 3-5). While we expected that slope position affect runoff, results from the linear mixed effect model showed that plot-scale runoff responses between slope positions were not significant. Because of this, 2012 and 2013 runoff responses of barley (control and charcoal amended) and deep-rooted lupine were grouped by transect and then compared. Statistical test results showed that, for all transects, lupine plots produced significantly more than both the control and charcoal-amended barley plots.

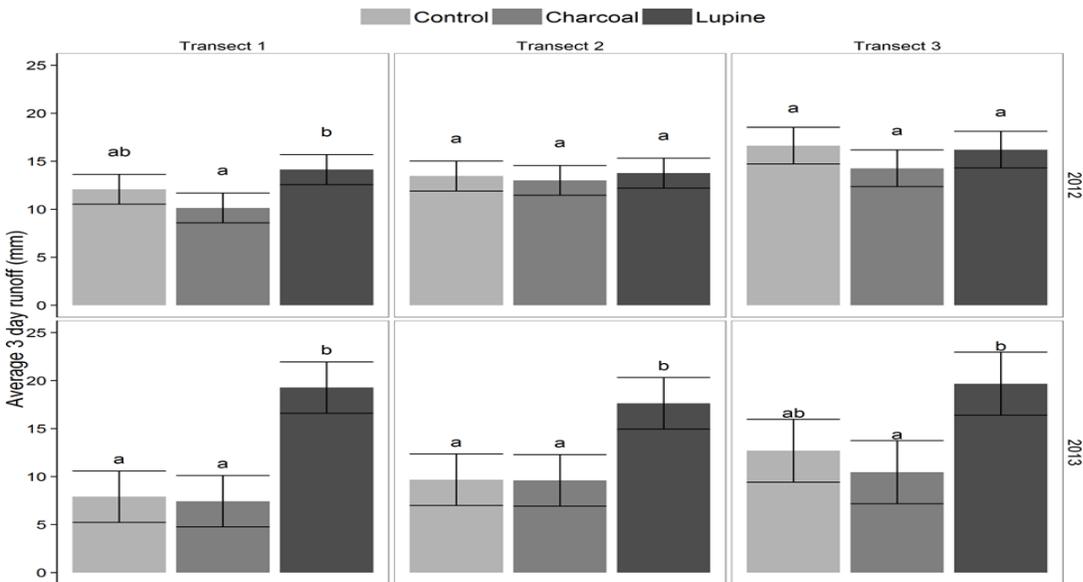


Figure 3-5. Effect of charcoal amendment and deep-rooted lupine crop on plot-scale runoff (3-day total) for each transect and year. Treatments not sharing the same letter within an individual transects for a given year are significantly different at  $p < 0.05$ .

Charcoal amendment, on the other hand, caused no significant difference effects (Figure 3-5). The cumulative runoff for the lupine plots followed the cumulative runoff for the outlet more than the barley plots, particularly in 2013 (Figure 3-6).

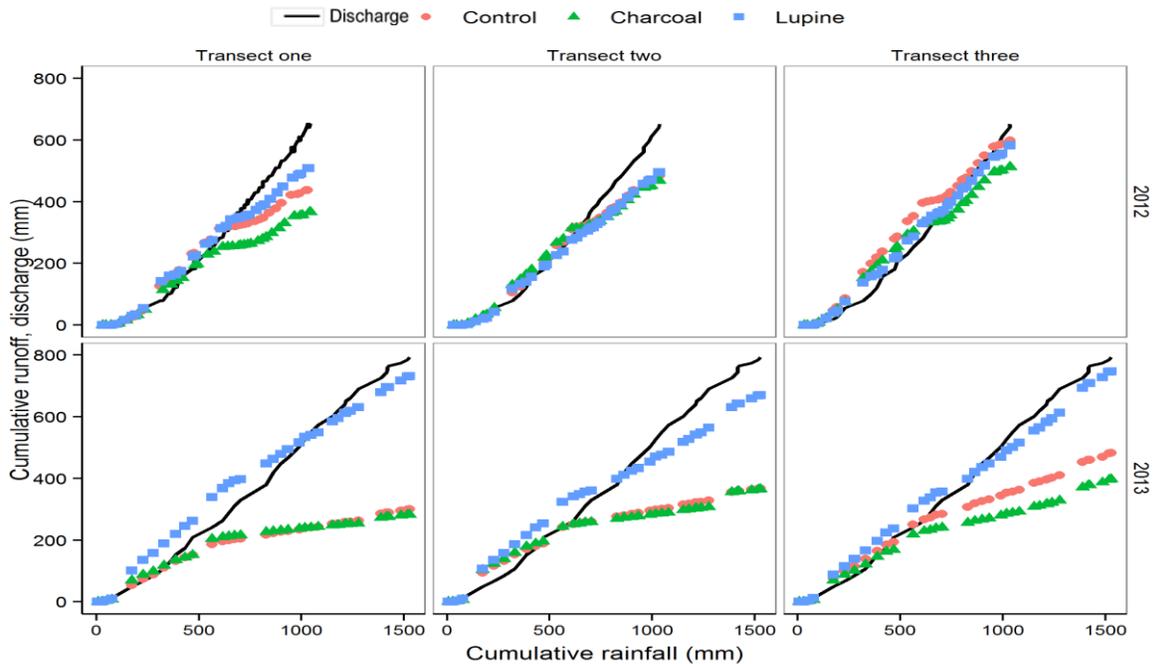


Figure 3-6. Cumulative rainfall vs. runoff from three transects and at the watershed outlet, for 2012 and 2013.

Comparison of plot-scale cumulative runoff and cumulative river discharge observed at the watershed outlet with cumulative rainfall indicated that approximately 100 mm of cumulative rainfall was needed before runoff was initiated from all plots. In general, during the start of the monsoon season (ca. 500 mm cumulative rainfall in Figure 3-6), plot-scale runoff response generally exceeded watershed-scale discharge response. Nevertheless, as the rainy season progresses, starting from the middle of August and at approximately 500 mm cumulative rainfall, watershed-scale discharge starts to exceed plot-scale runoff depths. The difference between plot

runoff and outlet discharge at early season of the monsoon indicates that detention storage at a watershed scale; while the difference during later the monsoon season represents base flow at the watershed outlet. This is consistent with previous observations by Tilahun et al (2013 a, b); Bayabil (2010) where initially the runoff from the hillsides infiltrate on the lower slope position and then later in the season these bottom lands start to contribute both subsurface flow and surface runoff.

A considerable difference in the runoff response of barley and lupine plots was observed in 2013 compared with 2012. In 2012, runoff tended to be more or less similar for all treatments, whereas in 2013 runoff from barley and lupine plots began to deviate after approximately 250 mm cumulative rainfall (Figure 3-6). In agreement with this, a closer look at the plots (Figure 3-3) clearly shows that for most of the high rainfall amounts, there is little difference in runoff response between the barley and lupine plots. Only for smaller rain events (ca. < 20 mm) and during the start of the 2013 rainy season (around July 1), runoff from lupine plots exceeded that of barley plots. It is interesting that this is the case for all three transects in 2013, but does not occur in 2012. The only management difference between these two years is that lupine was tilled in 2012 but not in 2013. This implies that tillage resulted in relatively greater soil water storage for lupine plots in 2012 than in 2013, and that the difference in rainfall-runoff response between treatments in 2013 may be ascribed to the fact that barley plots were tilled and lupine plots were not. Soil water storage predicted using the SCS-CN equation (Steenhuis et al., 1995) confirmed smaller storage for lupine than for barley (Figure 3-7). This would mean that there is very little infiltration in the lupine plots other than abstracted water to compensate evapotranspiration loss by the lupine.

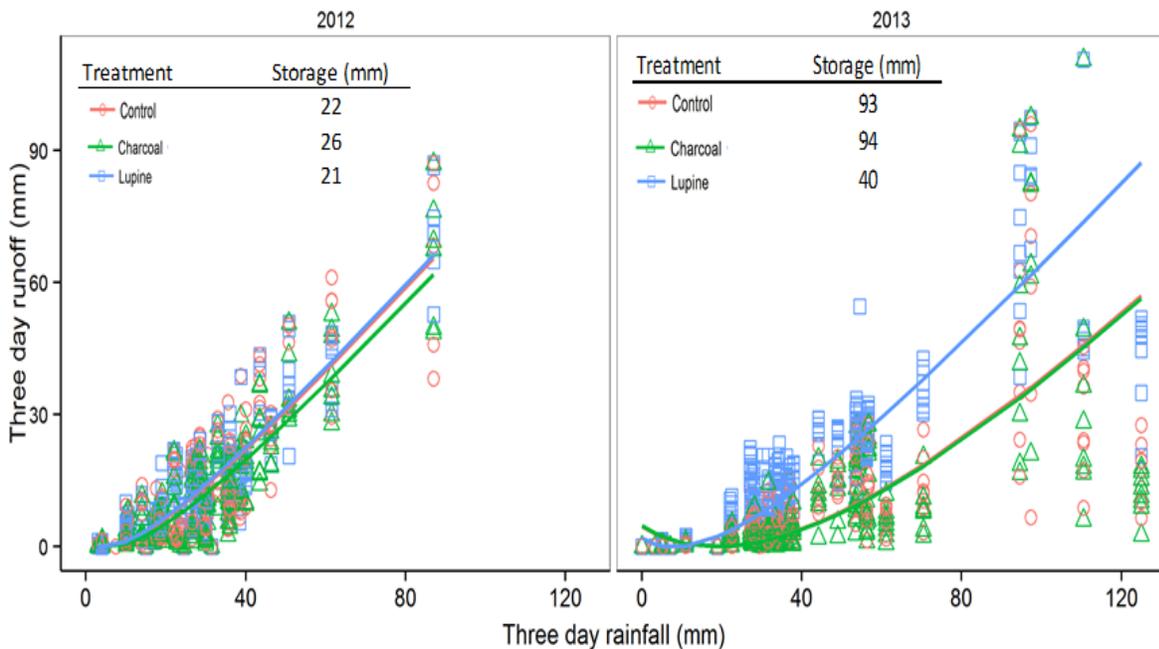


Figure 3-7. Effect of charcoal amendment and deep-rooted lupine on 3-day soil water storage: estimated by fitting three day runoff to three day rainfall using SCS-CN equation by Steenhuis et al. (1995).

Our findings indicated that both soil degradation status (here visible as differences in soil depth) and disturbance (tillage) were important factors affecting rainfall-runoff relationships in a landscape. In addition to tillage activities, inherent differences in plant root morphology (e.g. length and density) between the barley and lupine could likely be another factor. Most of the root masses of barley are located at shallow depths in the upper part of the soil profile (Lugg et al. (1988) and thereby take water from the top soil, whereas lupine roots grow deeper than barley and extract water from deeper depths French and Buirchell (2005)). These differences in root water uptake are somewhat visible in the slightly greater, albeit not significant, root depth moisture readings observed (measured from the top 20 cm) for lupine plots beginning in August in 2013 (not shown).

It is important to note that the fact that lupine did not decrease runoff during this study period does not imply it would not reduce runoff in the long-term. When the roots of lupine decompose, it is likely that biopores and channels would be created as reported by Meek et al. (1992) and Lesturgez et al. (Lesturgez et al., 2004) that have greater vertical and lateral continuity due to an improved network of macropores (Yunusa and Newton, 2003), which thereby would result in reduced surface runoff and associated erosion.

### **3.4 Conclusions and implications**

Our findings have several important implications for future efforts towards modeling rainfall-runoff relationships and planning effective soil and water management practices that allow better use of green water (rainfall) for smallholder agriculture systems in the Ethiopian Highlands. First, results support previous findings that plot or hillslope scale rainfall-runoff relationships are mostly different from watershed outlets. This implies physical models that represent variable field conditions (eg. soil and crop) into consideration are needed; analysis of hydrographs at the watershed outlet does not fully depict field conditions. Second, plot-scale rainfall-runoff relationships are greatly affected by root-zone soil storage capacity, which our data shows is directly affected by soil degradation status and tillage practices that in turn affect soil storage (Figures 3-5 and 3-7). Finally, tillage practices (eg. plowing) and root morphology (e.g. root length and density) of crops significantly affect storage of soils and therefore affect runoff responses.

In the near term, decreased soil water storage for lupine implies smaller rainfall threshold for runoff initiation. In the long term however, lupine may have the potential to actually reduce runoff by improving infiltration rates through the creation of biopores once its large taproot

decomposes. The long-term impact of lupine growth on runoff processes therefore requires further investigation. Understanding the drivers of hardpan formation and permeability is essential for the development of management approaches that can effectively tackle hardpan occurrence and its hydrologic impacts, in order to ultimately reverse the land degradation trend.

### **3.5 Acknowledgements**

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### 3.7 Appendix

#### Appendix A: Soil Map of the Anjeni Watershed

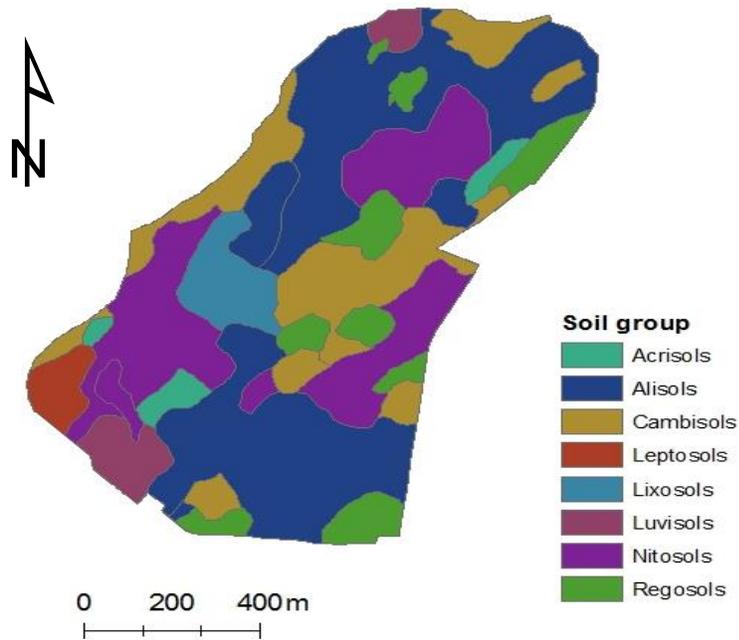


Figure A1. Soil map of the Anjeni watershed. Source (Zeleeke, 2000)

## Appendix B: Daily Plot Runoff Response

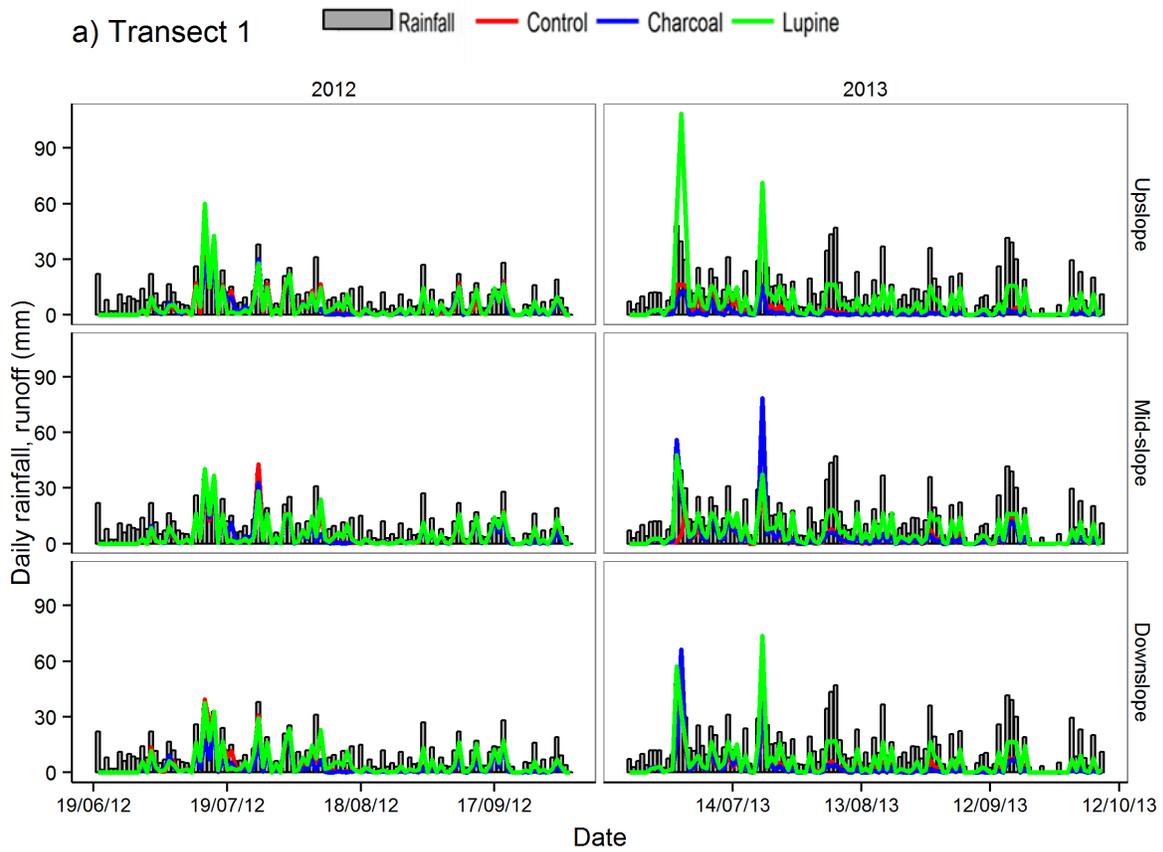


Figure B1. Daily rainfall versus runoff from plots along Transect 1

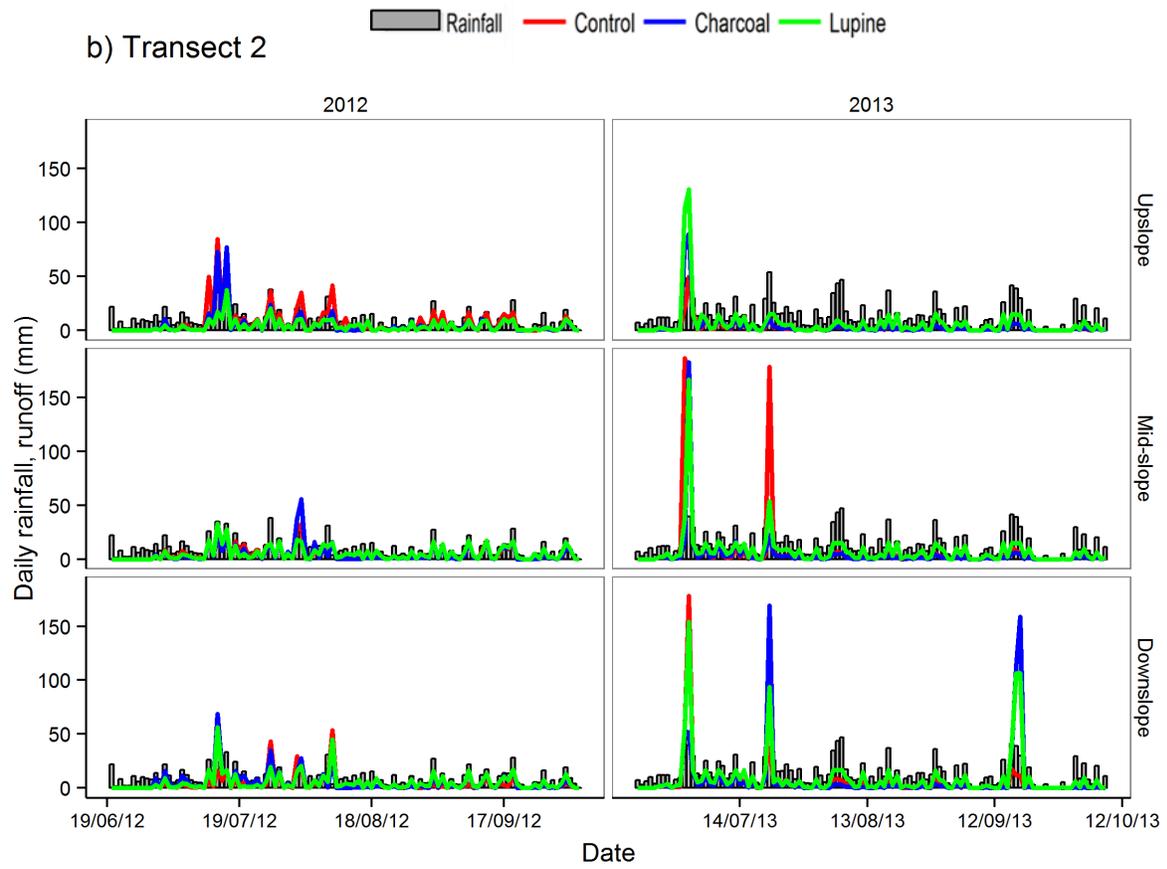


Figure B2. Daily rainfall versus runoff from plots along Transect 2

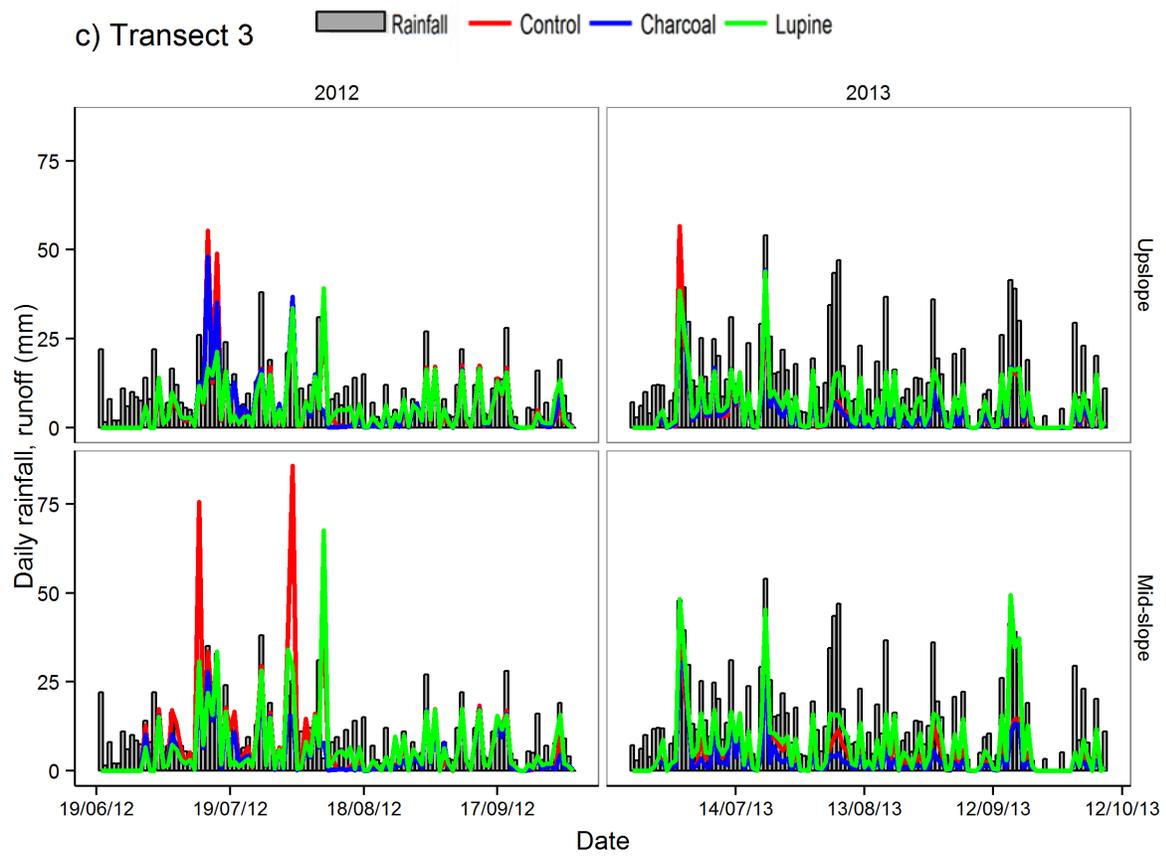
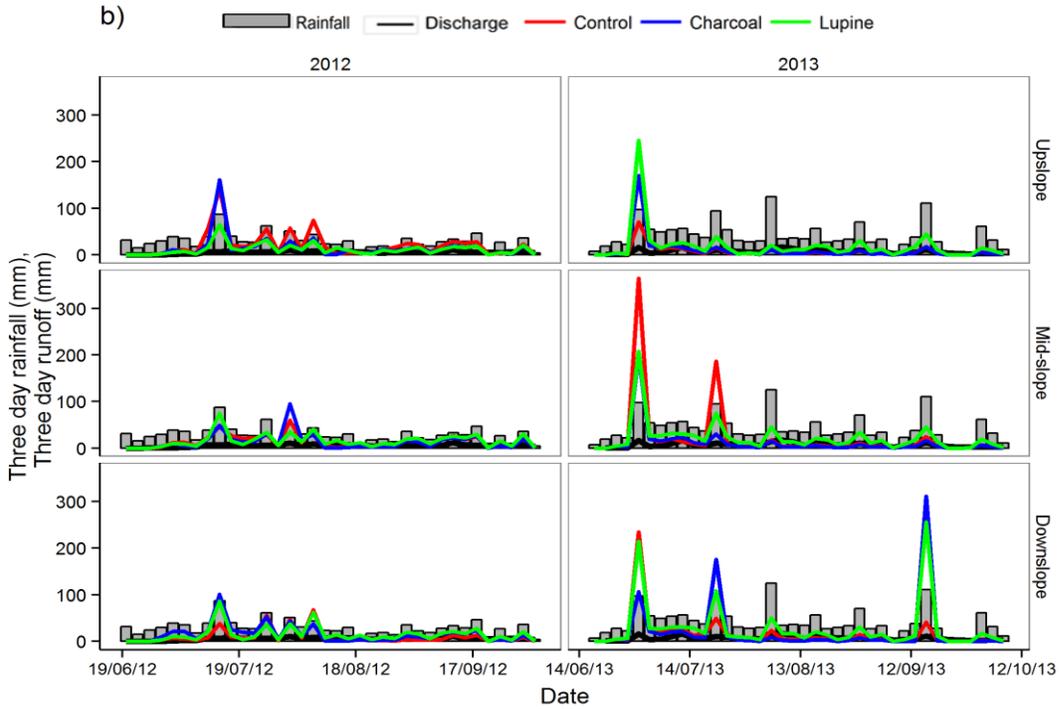
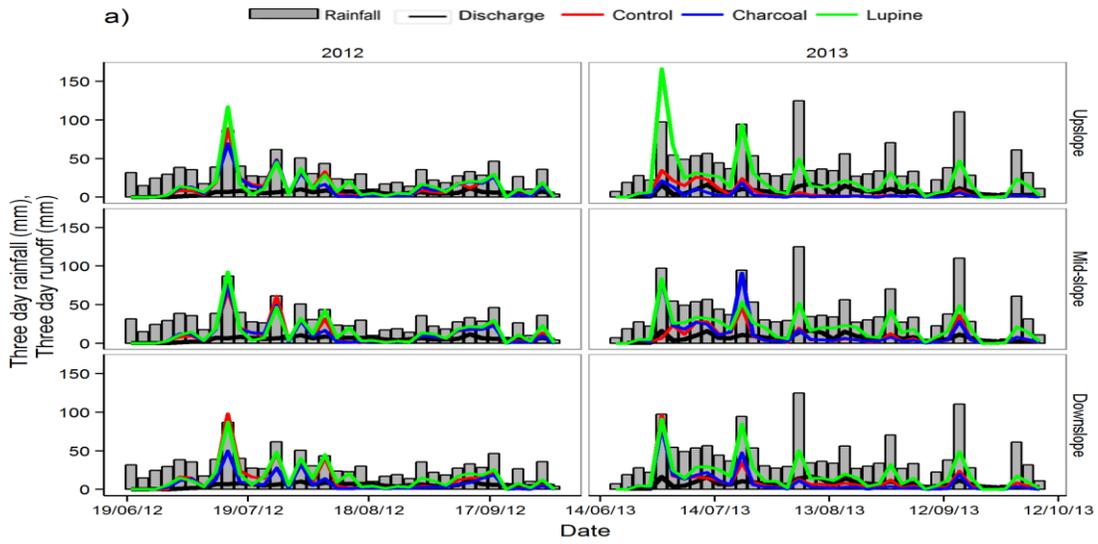


Figure B3. Daily rainfall versus runoff from plots along Transect 3

Su

## Appendix C: Aggregated Plot Runoff Response



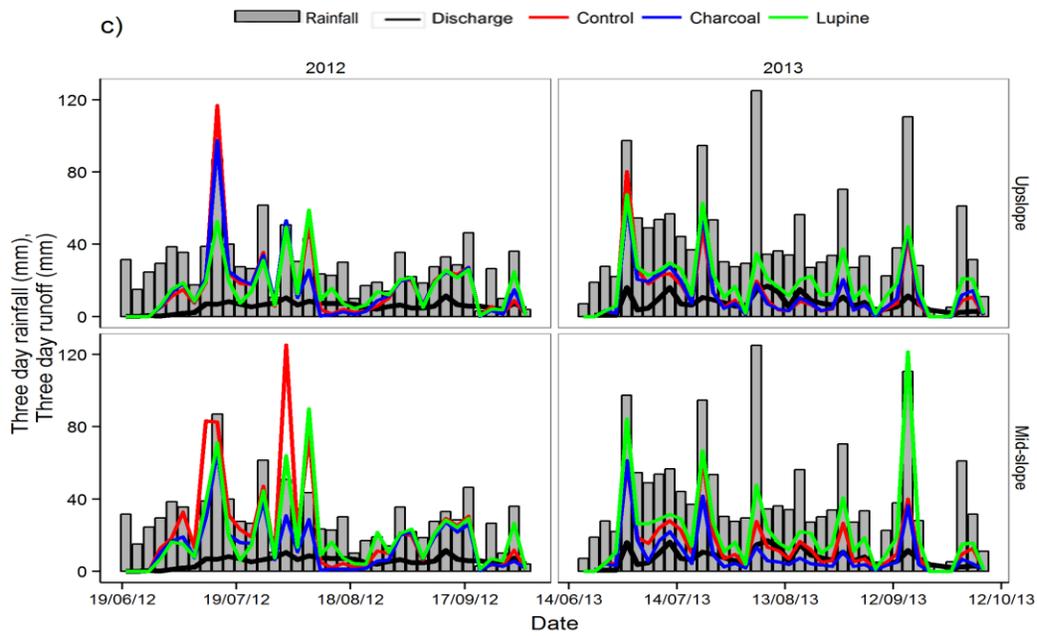


Figure C1. Three day rainfall and unadjusted three day runoff (aggregated on 3-day intervals) for individual plots from different slope positions along Transect 1 (a), Transect 2 (b), and Transect 3 (c)

## Appendix D: Summary of Annual Plot Runoff Response

Table D1. Summary of total runoff during the year (Total runoff), average 3-day runoff in mm (Mean runoff) and Standard error (SE) in mm averaged of the plots in the transect for 2012 and 2013. The annual precipitation in 2012 and 2013 was 1036 and 1528 mm, respectively.

Plot	Rainfall/Runoff (mm)	2012			2013		
		Transect			Transect		
		1	2	3	1	2	3
Control	Total runoff	438	487	603	300	367	482
	Mean runoff	12	14	17	10	17	15
	SE	2.8	2.5	3	1.7	2.7	2.5
Charcoal	Total runoff	365	467	512	282	365	397
	Mean runoff	10	13	14	13	18	14
	SE	2.2	2.5	2.6	2.1	3	2.3
Lupine	Total runoff	509	500	588	732	670	747
	Mean runoff	14	14	16	20	21	19
	SE	2.8	2.4	2.5	3.3	3.4	3.1
Rainfall	Total rainfall	1036			1528		
	Mean rainfall	29			40		
	SE	0.5			0.8		

\*SE: Standard error of the mean

## Appendix E: Monthly Rainfall Distribution during the Monsoon Season

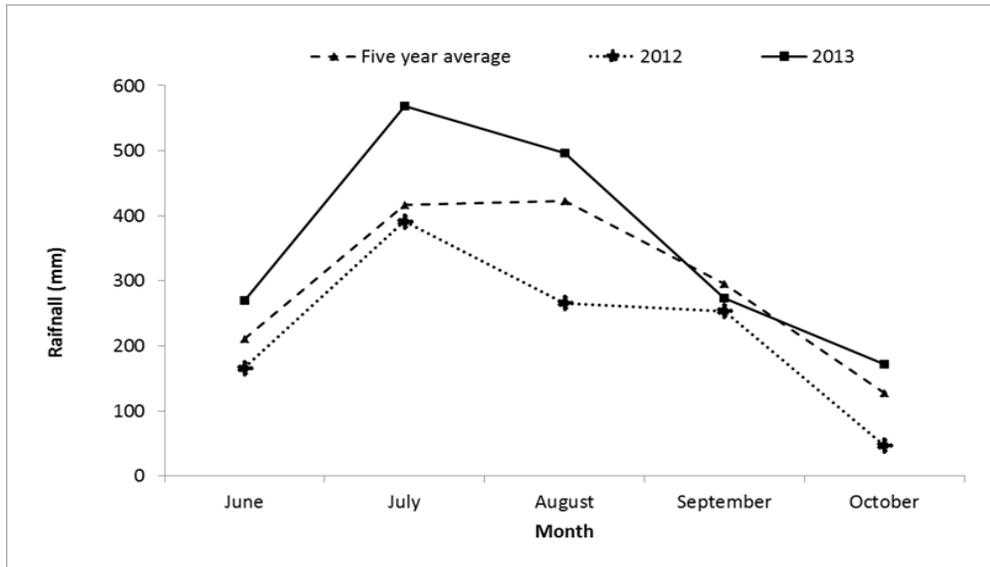


Figure E1. Monthly rainfall distribution during the monsoon period of 2012, 2013 and 5-year average

## CHAPTER 4: EROSION PROCESSES IN THE UPPER BLUE NILE BASIN: THE ANJENI WATERSHED<sup>1</sup>

### Abstract

Long-term erosion monitoring data in the Ethiopian highlands are only available from the Soil Conservation Research Program (SCRIP) Watersheds including the Anjeni Watershed. The 113 ha Anjeni watershed was established in 1984 and *fanya juu* terraces were installed in 1986. Runoff and erosion data are available from three different plot sizes and at the watershed outlet. The objective of this study was to investigate how erosion processes and sediment rating parameters vary with plot sizes and the progression of the rainy monsoon phase. We analyzed runoff and sediment loss data from 40 plots and the watershed outlet. The dataset included erosion data from 24 newly constructed plots (3m length) during the monsoon phase of 2012 and 2013 and 16 long-term plots (with 12, 16, 22, and 24% slopes and 3, 15 and 30 m lengths) and the watershed outlet during the period between 1986 to 1990. Sediment concentration ( $C$ ) was fitted to runoff ( $Q$ ) using a power equation ( $C = aQ^b$ ). Sediment concentration and yield increased with increasing plot length; from 3 m to 15 m, but sediment yield decreased as plot length increased to 30 m. The coefficients ( $a$  and  $b$ ) were affected by plot size and the progression of the rainy monsoon phase. As plot size increases, the  $a$  value increased, while the  $b$  value decreased.

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<sup>1</sup>This paper is submitted for publication as Bayabil, H.K., et al. (2015). Erosion Processes in the Upper Blue Nile Basin: The Anjeni Watershed. Earth Surface Processes and Landforms (submitted).

Overall findings suggest that as plot size gets larger (eg. 30m length plots and watershed) and during early periods of the monsoon season, erosion processes appear to be at transport limiting conditions, while as plot size gets smaller (eg. 3m length micro plots) and towards the end of the monsoon season that sediment source would be limiting.

*Keywords:* plot erosion, sediment yield, Blue Nile basin, soil and water conservation

## **4.1 Introduction**

Erosion has affected the livelihood of people for centuries (Dregne, 1990). In the humid Ethiopian highlands where most of the fields are intensively cultivated for crop production with an increasing population, erosion is especially severe (Bewket and Sterk, 2003; Hurni, 1988; Steenhuis et al., 2009; Temesgen et al., 2012) and reservoirs downstream are being filled up (Bewket and Sterk, 2003; Steenhuis et al., 2009; Temesgen et al., 2012). This has resulted in the loss of the most productive top soils shortly after forestlands were converted to agriculture (Hurni, 1988).

To decrease sedimentation, soil and water conservation structures have been installed throughout most of the highlands (Herweg and Ludi, 1999; Hurni, 1988; Nyssen et al., 2008). Effectiveness of soil and water conservation structures in reducing runoff and erosion depends on the annual rainfall. In arid and semi-arid monsoonal areas, several studies (Descheemaeker et al., 2006; Herweg and Ludi, 1999; Nyssen et al., 2007), found that conservation structures were effective in reducing runoff and erosion. However, in humid areas, performance of the same conservation structures were always not satisfactory (Kato et al., 2011; Mitiku et al., 2006; Temesgen et al., 2012).

Previous erosion studies in the Ethiopian highlands employed plots (Descheemaeker et al., 2006; Nyssen et al., 2008), rill measurements (Bewket and Sterk, 2003; Zegeye et al., 2010), and used sediment concentration at watershed outlets (Guzman et al., 2013; Setegn et al., 2010; Steenhuis et al., 2009; Tilahun et al., 2014, 2013). Sediment rating curves obtained by fitting sediment load to discharge are widely reported as viable options to understanding erosion processes (Asselman, 2000; Gao, 2008; Guzman et al., 2013; Syvitski et al., 2000). Guzman et al. (2013) found that discharge and sediment concentration were poorly correlated when fitted on daily basis. However, aggregating the data on a 14-day interval he found a unique relationship between discharge and sediment concentration with the progression of the rainy monsoon phase.

Hurni (1988) estimated an overall average of  $42 \text{ t ha}^{-1}$  erosion rate for cultivated fields in the Ethiopian highlands in the nineteen eighties. However, erosion rates vary greatly across different agro-ecologies. Soil loss rates are greater from areas with rainfall over  $1000 \text{ mm y}^{-1}$  and with a sub-humid or humid climate than in the arid and semi-arid areas (Herweg and Ludi, 1999). In the arid and semi-arid region of Tigray in the Northern Ethiopian highlands, specific sediment yield from reservoir studies varied between  $5 \text{ t ha}^{-1} \text{ y}^{-1}$  to  $18 \text{ t ha}^{-1} \text{ y}^{-1}$  (Haregeweyn et al., 2005), while observed sediment loss from plots was on the average  $15 \text{ t ha}^{-1}$  (Nyssen et al., 2008). Rill erosion ranged between 13 to  $61 \text{ t ha}^{-1}$  in the sub-humid Chemoga watershed (Bewket and Sterk, 2003) and 8 to  $32 \text{ t ha}^{-1}$  for the Debre Mawi watershed (Zegeye et al., 2010). In the same Debre Mawi watershed, where active gullyng is taking place, Tilahun (2012) and Tebebu et al. (2010) reported erosion rates ranging from  $31 \text{ t ha}^{-1} \text{ y}^{-1}$  to  $530 \text{ t ha}^{-1} \text{ y}^{-1}$  at sub watershed level depending on the severity of the gullyng. Average sediment yield at the outlet of the Anjeni watershed was  $25 \text{ t ha}^{-1} \text{ y}^{-1}$  (Guzman et al., 2013; SCRIP, 2000; Setegn et al., 2010).

Studies in temperate regions in other countries have shown that as plot length increases, sediment yield initially also increased, but then subsequently decreased (Bagarello and Ferro, 2004; Moreno-de las Heras et al., 2010; Parsons et al., 2006; Thomaz and Vestena, 2012). Parsons et al. (2006), for example, observed a maximum sediment load from plots of 7 m length in Southern Arizona. Increasing plot length above that decreased runoff and sediment load (Bagarello and Ferro, 2004; Parsons et al., 2006). Similarly in Ethiopia Guzman et al. (2013); SCRCP, (2000); Setegn et al., (2010) reported less loss per unit area at the outlet than at the plot scale. However, systematic investigations do not exist in Ethiopia on the effect of scale with the progression of the monsoon (rainy) phase on erosion processes. Therefore, the objective of this study was to investigate how sediment transport varies with scale and the progression of the monsoon season by comparing erosion processes of plots of various lengths along the hillslope and at the watershed outlet. In addition, we investigated the effect of soil and water conservation structures on soil erosion rates.

The Anjeni experimental watershed was chosen for this study. Discharge and soil loss data have been collected since the 1980's from both the watershed outlet and runoff plots with different sizes and slopes; and with and without conservation practices (Hurni et al., 2005). The data is available from the Soil Conservation Research Program (SCRCP). Sediment data at the watershed outlet was analyzed by several researchers (Guzman et al., 2013; Setegn et al., 2010), but ignoring hillslope runoff and erosion processes. Sediment losses at the outlet were simulated by Setegn et al. (2010), on a monthly basis, who predicted monthly sediment yield at the outlet using SWAT model and relatively small Hydrological response units (HRU's). Zeleke (1999) tested the Water Erosion Prediction Project (WEPP) model to simulate erosion losses from agricultural plots but not at the outlet. Herweg and Ludi (1999) compared erosion rates from

plots with different conservation structures. Our study provides more detailed information about scale and seasonal trends on erosion rates.

## **4.2 Materials and Methods**

### ***4.2.1 The Anjeni watershed***

Anjeni watershed is located in the northwestern part of Ethiopia (Figure 4-1a). The watershed is one of the six experimental watersheds established, throughout the Ethiopian highlands, under the Soil Conservation Research Program (SCRIP) of the Ministry of Agriculture of Ethiopia in collaboration with the Swiss Agency for Development and Cooperation (SDC) (Hurni et al., 2005). The watershed's total catchment area is 113 ha with elevation between 2407 and 2507 m (Herweg and Ludi, 1999) and its gauging station is located at 10°40' N, 37°31' E (Tilahun et al., 2011).

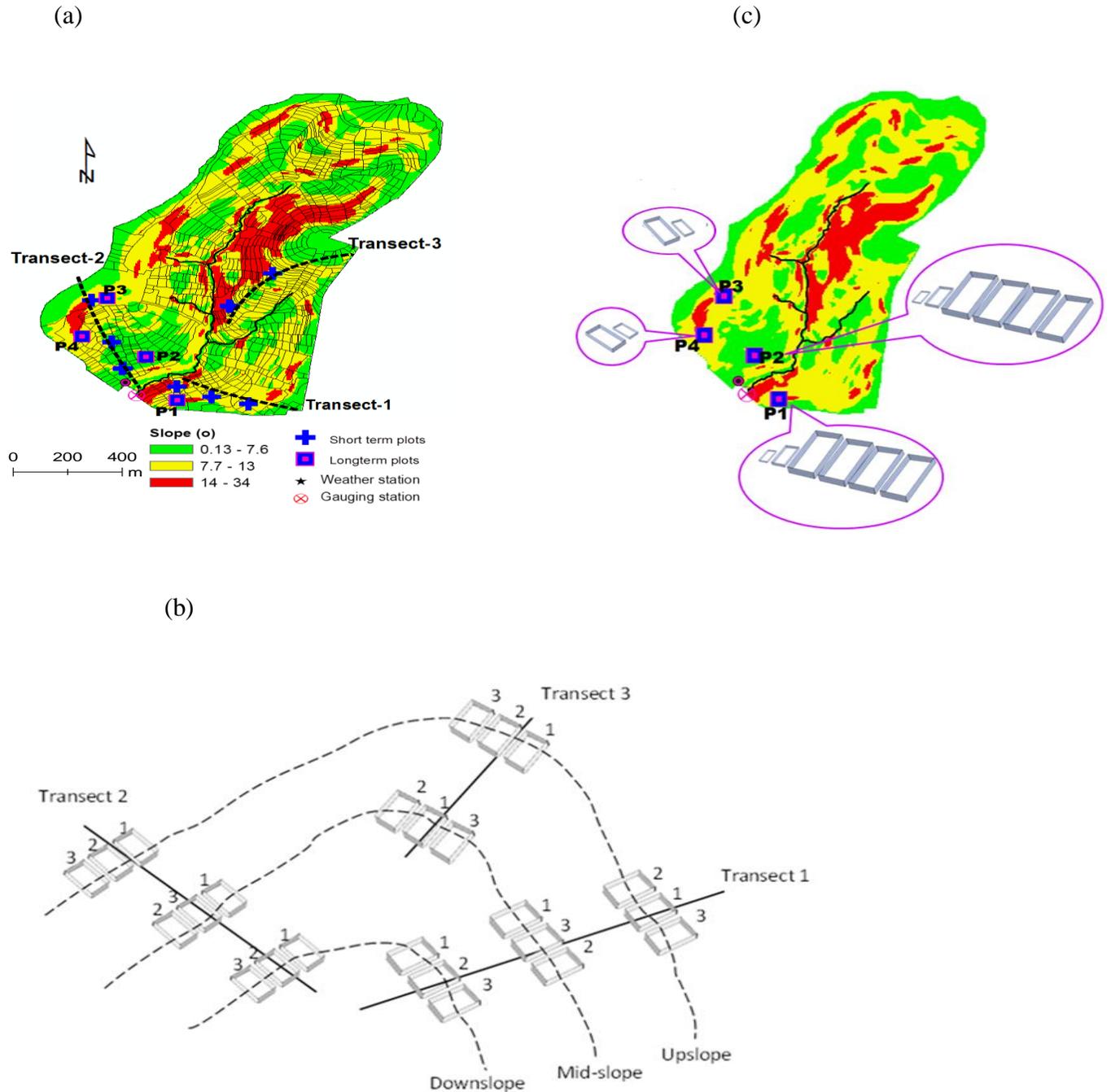


Figure 4-1. Slope map of the Anjeni watershed (a). Location of erosion plots constructed in 2012-2013, and long term plots (b). Long term plot locations P1 and P2 represent groups of plots (1 micro, 1 test, and 4 experimental plots at 28 and 12%, respectively), and P3 and P4 represent groups of plots (1 micro and 1 test plots at 16 and 22%, respectively) (c). Details about land use and topographic attributes of plots are presented in Table 1. Dashed lines are elevation contours. Plots labels represent treatment types: 1 = barley without amendment (Control), 2= barley with charcoal (Charcoal), and 3= Lupine grown as sole crop (b and c-not to scale). Figure 4-1c is adapted from (Bayabil et al., 2015).

Table 4-1. Spatial and ecological attributes of long term runoff-erosion plots (adapted from SCRIP, 2000)

Group	Plot type	Slope (%)	Length (m)	Width (m)	Land use	*SWC structure
P1	Micro plot (MP5)	28	3	1	Cultivated	No
	Test plot (TP1)	28	15	2	Cultivated	No
	Experimental plot (EP1)	28	30	6	Cultivated	No
	Experimental plot (EP2)	28	30	6	Cultivated	Graded bund
	Experimental plot (EP3)	28	30	6	Cultivated	Graded <i>fanya juu</i>
	Experimental plot (EP4)	28	30	6	Cultivated	Grass strip
P2	Micro plot (MP6)	12	3	1	Cultivated	No
	Test plot (TP2)	12	15	2	Cultivated	No
	Experimental plot (EP5)	12	30	6	Cultivated	Graded <i>fanya juu</i>
	Experimental plot (EP6)	12	30	6	Cultivated	Graded bund
	Experimental plot (EP7)	12	30	6	Cultivated	No
	Experimental plot (EP8)	12	30	6	Cultivated	Grass strip
P3	Micro plot (MP7)	16	3	1	Grassland	No
	Test plot (TP3)	16	15	2	Grassland	No
P4	Micro plot (MP8)	22	3	1	Cultivated	No
	Test plot (TP4)	22	15	2	Cultivated	No

\*SWC: Soil and Water Conservation

The soils of Anjeni are developed from the basalt and volcanic ash, with major soils Alisols, Nitisols, and Cambisols covering more than 80% of the watershed (Zelege, 2000). The deep Alisols cover the bottom part of the watershed; moderately deep Nitisols cover the mid-transitional, gently sloping parts of the watershed, while the shallow Regosols and Leptosols cover the high and steepest part of the watershed (Zelege, 2000).

#### **4.2.2 Data preparation and analysis**

In this study, we analyzed two sets of sediment data from plots and at the watershed outlet. The first data set consists of discharge and sediment concentration data from 24 plots ('short term plots', Figure 4-1a) with 3m length and 1.5m width collected during the monsoon seasons of 2012 and 2013. The second data set was collected by the Soil Conservation Research Program (SCRIP, Hurni et al., 2005) and contained discharge and sediment concentration data from the period from 1986 to 1990 for sixteen plots of different sizes ('long term plots', Figure 4-1a) and at the watershed outlet. These plots are located in the watershed in four groups P1, P2, P3, and P4 as shown in Figure 4-1c each with a distinct slope ranging from 28% (P1) to 12% (P2). The slope of P3 was 16% and for P4 22% (Table 4-1). There were three types of plots: four micro plots (3 m length and 1m width), four test plots (15 m length and 2 m width), and eight experimental plots (30 m length and 6 m width) (Table 4-1). There was one micro plot (3x1m) and one test plot (15x2m) at each of the four groups (Figure 4-1c, Table 4-1). Groups P1 and P2 in addition each contained four experimental plots (30x6m) for testing the effectiveness of soil and water conservation practices (Figure 4-1c, Table 4-1) These practices consisted of a control, graded bund, graded *fanya juu* and grass strip (Table 4-1). Except the third group of plots (P3, at 16 % slope), which was constructed on grassland, the rest of the groups of plots (P1, P2, and P4) were constructed on cultivated fields.

From February to April 1986, *fanya juu* (throwing soil up slope from a ditch to form a bund along the contour) structures were installed throughout the watershed. These practices were effective and maintained during this period. During 2012 and 2013, terraces were fully formed behind the soil dug from the furrows. Detailed information about these long term monitoring plots and the watershed can be found in the Soil Conservation Research Program report series (SCRIP, 2000).

The watershed has unimodal rainfall pattern (Bayabil et al., 2015; SCRIP, 2000) and most of the runoff and erosion takes place during the monsoon (rainy) phase (June to September) (Herweg and Ludi, 1999; SCRIP, 2000). We therefore analyzed data from the rainy monsoon phase (June to September) only. This then allows us comparison of our observations during 2012 & 2013 with the long term results. Only data from experimental plots without conservation structure were used for comparison of erosion rates from micro plots, test plots, and experimental plots. We analyzed also the effect of soil and water conservation practices on erosion processes on the experimental plots. Non-parametric analysis of variance (Kruskal-Wallis) tests were performed in R (R Development Core Team 2010) to test the significance of plot size on runoff and erosion rates.

### ***4.2.3 Runoff and sediment measurement***

All plots were monitored manually (under natural rainfall conditions) for runoff water and sediment on a daily basis. After runoff collected from a plot, water in the primary collection tanks was measured taking three depth measurements and then water was drained through valves fitted at the bottom of the tanks. Runoff depth from each plot was converted to volume based on size of the collection tanks. Stage discharge measurements at the watershed outlet were

conducted in two ways: using float-actuated recorder and manual recording. Long term stage data were converted to discharge depth (mm) using the discharge rating curve for the watershed (Bosshart, 1997).

Sediment measurement for the plots included both sediment load settled at the bottom of the collection tanks of plots and sediment suspended in the runoff water. When there is settled sediment at the bottom of the tanks, 500 g sediment sample was scooped, and total sediment weight was recorded. The 500 g sample was oven dried to determine net dry soil weight. Suspended sediment was determined by taking 1-L sample of runoff water and subsequently filtering, using filter papers (Figure 4-2) and oven drying the samples. Total suspended sediment was then calculated by multiplying the net dry soil weight in 1-L sample by the total runoff volume during that day. Finally, the sediment concentration (SC) was calculated by dividing the sum of settled sediment and total suspended sediment by total runoff volume.



Figure 4-2. Runoff water samples from plots, river outlet, and subsequent filtering of suspended sediment using filter papers.

At the watershed outlet, sediment concentrations were measured by taking 1-liter samples 10 and 30 min intervals depending on the change in the color of the water. Total suspended sediment load was then calculated by multiplying suspended sediment concentration (in 1-liter water sample) by the total runoff volume at the watershed outlet (Figure 4-2). For consistency with plot measurement, sediment concentration at the watershed outlet was calculated by dividing total sediment load by amount of direct runoff assuming that the subsurface water was sediment free. The direct runoff component was calculated using EcoHydRology package in R (R Development Core Team 2010) that uses recursive digital filter base flow separation method (Eq. 4-1) by (Nathan and McMahon, 1990):

$$f_i = \alpha f_{i-1} + \frac{(1+\alpha)}{2} (y_i - y_{i-1}) \quad (4-1)$$

where  $f_i$  is the filtered quick response at the  $i^{\text{th}}$  sampling instant,  $y_i$  is the original stream flow,  $\alpha$  and the filter parameter with value 0.925, which is the recommended value (Nathan and McMahon, 1990). Filtered baseflow is thus defined as  $y_i - f_i$ .

#### ***4.2.4 Relating sediment concentration with discharge***

In order to better understand the factors governing hillslope and catchment erosion processes, sediment rating curves were determined for each plot. A power regression model (Eq. 4-2), that relates sediment concentration to discharge similar to the one suggested by (Crawford, 1991; Gao, 2008; Syvitski et al., 2000; Walling, 1977) was fitted by nonlinear least square technique.

$$C = aQ^b \quad (4-2)$$

where,  $C$  is sediment concentration ( $\text{g l}^{-1}$ ),  $Q$  is the runoff ( $\text{mm d}^{-1}$ ) and  $a$  and  $b$  are parameters fitted for sediment concentration and are determined by regression analysis using observed data (Gao, 2008; Walling, 1977).

One of the limitations of rating curves is that they do not account for variations in sediment concentration at similar discharges with the progression of the rainy season (Steenhuis et al., 2009; Tilahun et al., 2014, 2013). To resolve this issue, power relationship was fitted using subset data for each month from each plot and watershed outlet.

## **4.3 Results**

### ***4.3.1 Flow characteristics at watershed outlet***

Based on five-year data (1986-1990), baseflow accounts on average 51% of total flow during the rainy monsoon phase (June - September) in the Anjeni watershed (Figure A1 in appendix A). This diminished the dilution effect of baseflow with the progression of the rainy phase of the monsoon that would result in decreased sediment concentrations in the river. Therefore, we calculated all sediment concentration at the outlet as the amount of sediment lost per day divided by the direct runoff during that day.

### ***4.3.2 Sediment concentration***

As shown in Figure 4-3a, sediment concentration are greater at the beginning of the rainy monsoon phase (ca. June /July) from all cultivated plots with the exception of the micro plots (3 m length) and watershed outlet. Sediment concentration from micro plots and all grassland plots regardless of size did not have the characteristic elevated sediment concentration and the concentrations were consistently smaller than at the outlet at the beginning of the rainy monsoon

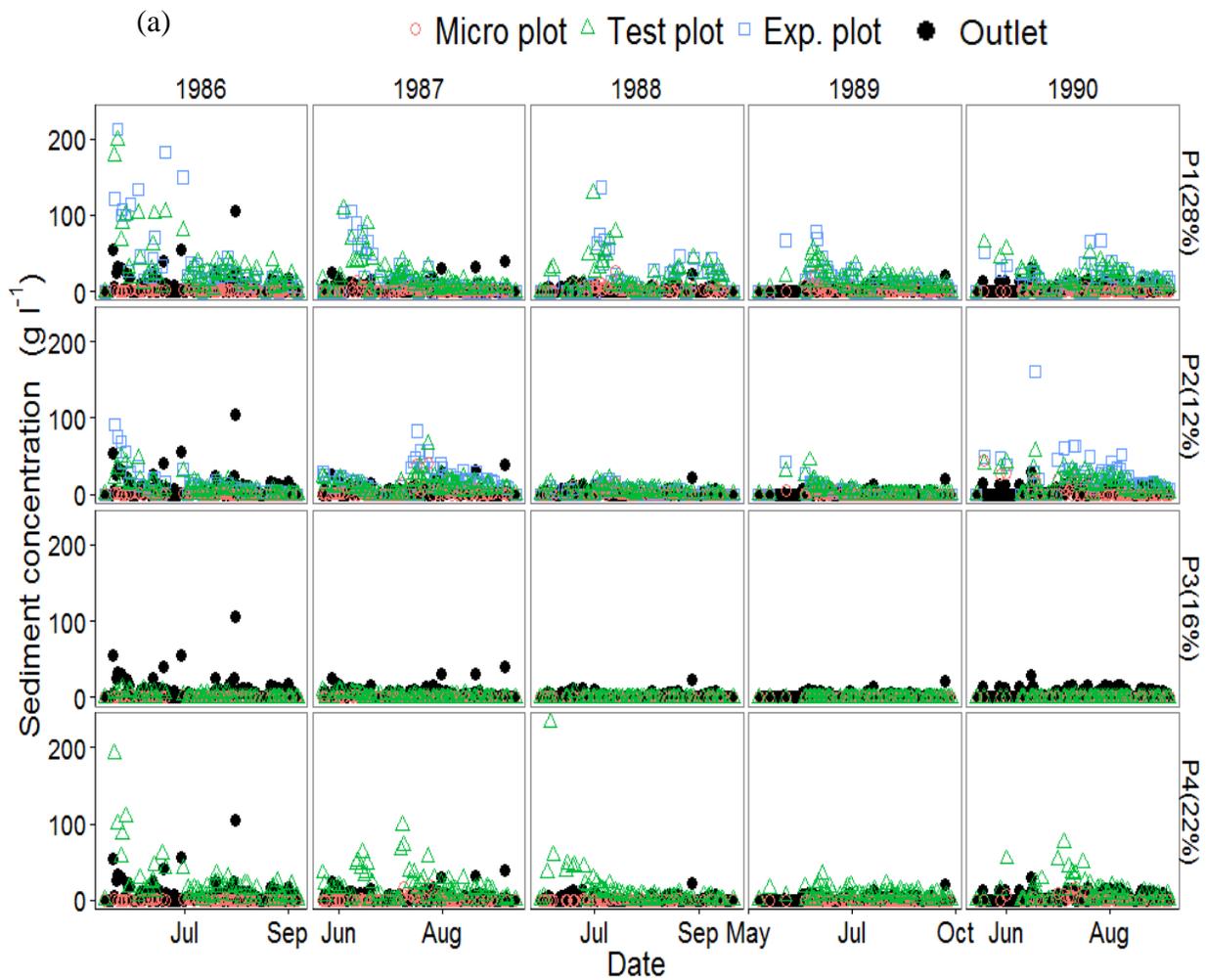
phase (Table 4-2, Figure 4-3). The data for the long term plots in Figure 4-3a are replotted as a function of discharge in Figure B1a in appendix B. The sediment concentration of the plots under grass (Group P3) are independent of the discharge.

Table 4-2. Runoff and erosion summary (based on 4 months: June-September) results from different plots without soil and water conservation structure and watershed outlet (based on five year data: 1986-1990), with standard error of the mean (calculated using year as replication, n=5) given between parentheses. Values not sharing the same letter within the same groups of plots (P1-P4) are significantly different ( $p < 0.05$ ).

Group	Plot type	Slope (%)	Length (m)	Width (m)	Direct runoff (mm y <sup>-1</sup> )	<sup>a</sup> SC (g l <sup>-1</sup> )	<sup>b</sup> SSY (t ha <sup>-1</sup> y <sup>-1</sup> )
P1	Micro plot (MP5)	28	3	1	820.4 (44.4)a	1.6 (0.2)a	26 (7.6)a
	Test plot (TP1)	28	15	2	702.4 (60.1)b	20.4 (1.3)b	160.8 (15.9)b
	Experimental plot (EP1)	28	30	6	442.3 (47.3)c	19.7 (1.6)b	104.2 (14.3)c
	Watershed outlet		113 ha		291 (20.7)d	3.7 (1)c	15.2 (4.1)a
P2	Micro plot (MP6)	12	3	1	681.9 (66.3)a	2.4 (0.3)a	37 (13.9)a
	Test plot (TP2)	12	15	2	715.4 (97.9)a	9 (0.6)b	108.6 (39)b
	Experimental plot (EP7)	12	30	6	435.5 (56.5)c	12.3 (1)c	82.3 (31.8)b
	Watershed outlet		113 ha		291 (20.7)d	3.7 (1)d	15.2 (4.1)c
P3	Micro plot (MP7)	16	3	1	430.1 (80.5)a	0.1 (0)a	1.1 (0.7)a
	Test plot (TP3)	16	15	2	468.4 (47.8)a	0.8 (0.1)b	4.5 (2.5)b
	Watershed outlet		113 ha		291 (20.7)c	3.7 (1)c	15.2 (4.1)c
P4	Micro plot (MP8)	22	3	1	506.1 (59.7)a	0.9 (0.1)a	12.8 (5.4)a
	Test plot (TP4)	22	15	2	484.1 (57.5)a	16 (1.7)b	109.7 (24.3)b
	Watershed outlet		113 ha		291 (20.7)b	3.7 (1)c	15.2 (4.1)a

<sup>a</sup> Sediment concentration calculated based on direct runoff; <sup>b</sup> Specific Sediment Yield.

The cultivated test and experimental plots (Groups P1, P2 and P4, Figure B1a in appendix B) have the greatest concentration for low flows that occur during the beginning of the rainy phase (Figure 4-3a). At the middle and end of the rainy phase (July and August) after the watershed has wetted up discharge increases and the sediment concentration increases with increasing discharge but not to the same level as in the beginning of rainy phase (Figures 4-3a and B1a in appendix B).



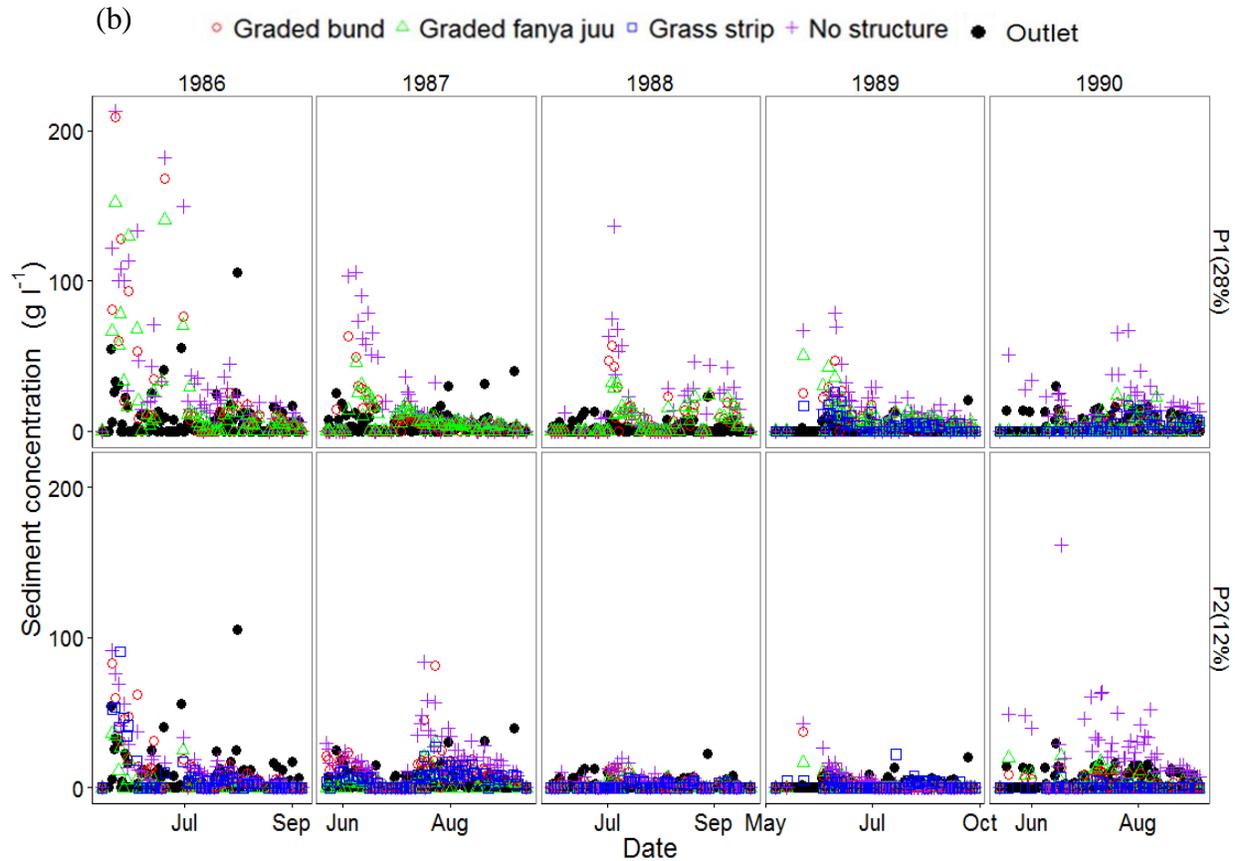


Figure 4-3. Long term (1986-1990) daily sediment concentration from plots of different sizes without conservation structures and watershed outlet (a) and experimental plots with conservation structures and watershed outlet (b)

Unlike the 3 m long term (sloping) micro plots in the five years (1986-1990), the sediment concentration in 2012 and 2013 of the 3-m plots at the end of the relatively flatter terraces showed distinct elevated concentrations for the low flows in the beginning of the rainy phase (Figure B1b in appendix B). The sediment concentration at the outlet of the watershed has the same pattern as the cultivated plots with length of 15 and 30 m (Figures 4-3a and B1c in appendix B).

### ***4.3.3 Specific sediment yield***

Specific sediment yield ranged between 1 t ha<sup>-1</sup> y<sup>-1</sup> for micro plot (MP7) under grassland to 161 t ha<sup>-1</sup> y<sup>-1</sup> from cultivated fields (Table 4-2). Runoff and erosion yields were significantly affected by measurement scale. Runoff increased as plot sizes become smaller (Table 4-2). Sediment yield was significantly smaller from all plots under grassland regardless of plot size including the whole watershed. Greatest sediment yield was observed from test plots (15 m length) followed by experimental plots (30 m length). In general, sediment yield per unit area for cultivated plots without conservation measures, increased significantly as plot size increases from 3 m (micro plots) to 15 m (test plots), and then (unlike sediment concentration) decreased with further increase in plot length to 30 m and for the watershed outlet (Table 4-2).

Plotting annual cumulative annual sediment yield against cumulative annual runoff from different plot sizes and for the watershed outlet clearly showed that plots with cultivated land produced the greatest sediment yield at all plot scales, while sediment yield from grassland was the smallest (Figure 4-4).

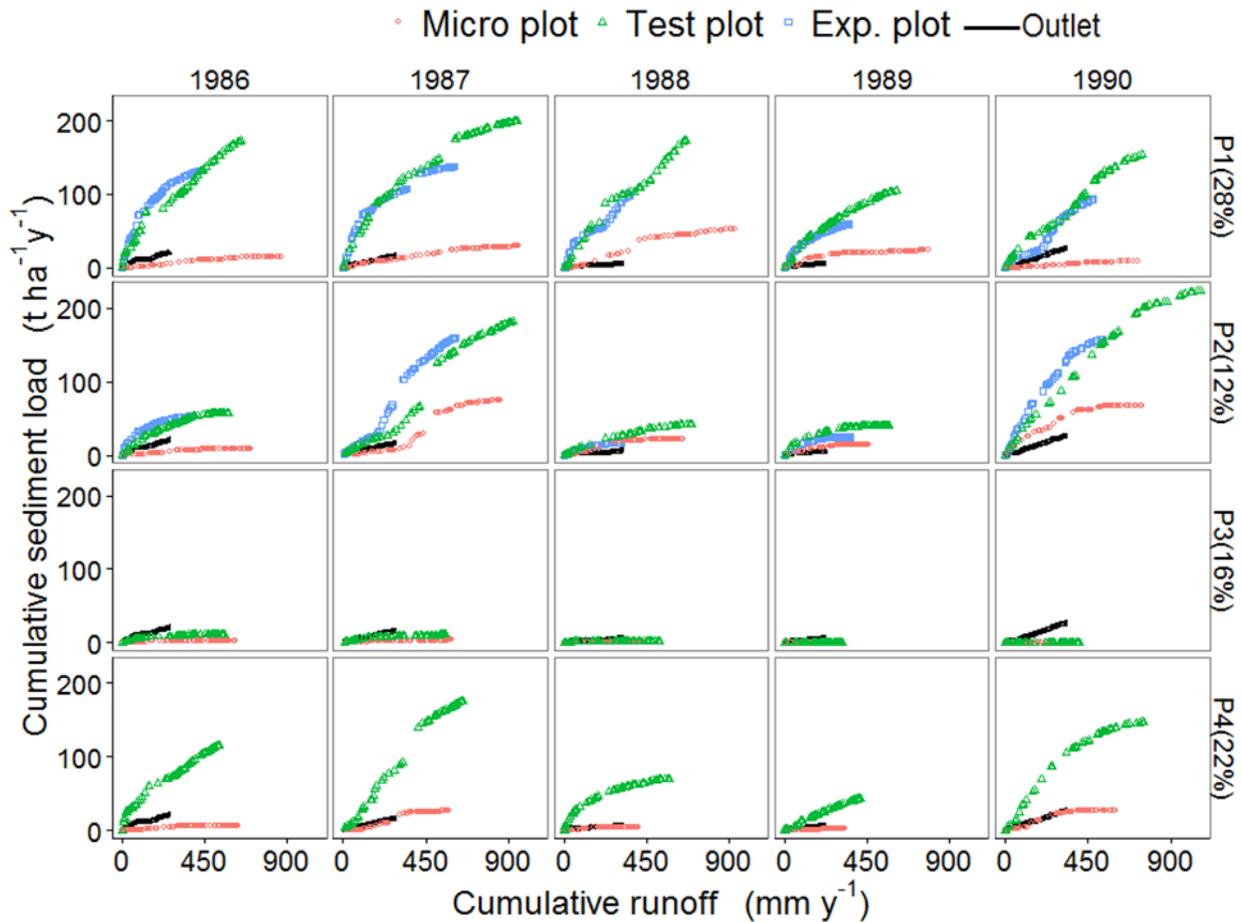


Figure 4-4. Cumulative runoff vs. cumulative sediment load from micro plots (3 m length, 1 m width), test plots (15 length, 2 m width), and experimental plots (30 m length, 6 m width). Rows represent group of plots (P1-P4) at different slope position, while columns show results for a given year. All plots are cultivated, except plots at P3 (16 %), where all plots are grassland.

In addition, it can be seen in Figure 4-4 that cumulative sediment yield from 30 m long cultivated experimental plots followed the same trend as the cumulative sediment yield from the 15 m long cultivated test plots (Figure 4-4), and the difference in sediment yield appears mainly because cumulative runoff was significantly smaller from experimental (EP1 and EP7) plots without conservation structures (Table 4-2).

Our observations, in 2012 and 2013, were in the same order of magnitude with long term sediment concentration and yield results from the 3 m long cultivated micro plots measured in 1986-1990 (Table 4-3). Greater sediment concentration and yield was observed from lupine plots with the greatest runoff amounts, while sediment concentration and yield from control (barley without amendment) and charcoal (barley with charcoal amendment) plots were comparable but lower than the lupine plots.

Table 4-3. Summary of runoff and erosion results from 24 plots (based on two-year observations: 2012-2013), with standard error of the mean given between parentheses. Values not sharing the same letter are significantly different ( $p < 0.05$ )

Treatment	Direct runoff (mm y <sup>-1</sup> )	<sup>a</sup> SC (g l <sup>-1</sup> )	<sup>b</sup> SSY (t ha <sup>-1</sup> y <sup>-1</sup> )
Control	407.9 (37.9)a	2.4 (0.5)a	25.2 (5.1)a
Charcoal	391.4 (27.4)a	2.5 (0.6)a	24.2 (4.4)a
Lupine	617.4 (29.7)b	3.4 (0.7)a	37.8 (5.4)b

<sup>a</sup> Sediment concentration calculated based on direct runoff; <sup>b</sup> Specific Sediment Yield. Control: is barley without any amendment; charcoal: is barley + charcoal; and Lupine: is lupine without any amendment.

#### 4.3.4 Effect of soil and water conservation structures

Summary results of sediment yield from plots with different conservation structures indicated that soil and water conservation structures were effective in reducing runoff and sediment loss (Figure 4-5, Table 4-4). Overall, effect of conservation measures was greater on fields at steep slope (28%) than at relatively gentler slope (12%). Despite observed reduction in sediment concentration and yield from plots with conservation structures, however, observed erosion rates from most of these plots (with conservation measures) were still greater than sediment yield at the watershed outlet with the exception of the *fanya juu* plots at the 12% slope and the grass strip

plots (Figure 4-5, Table 4-4). In agreement with results from test plots (15 m length), sediment concentration from experimental plots with conservation structures were consistently greater during the beginning of the rainy monsoon phase (June and July) than during later towards the end of the rainy phase (August and September) (Figure 4-3b).

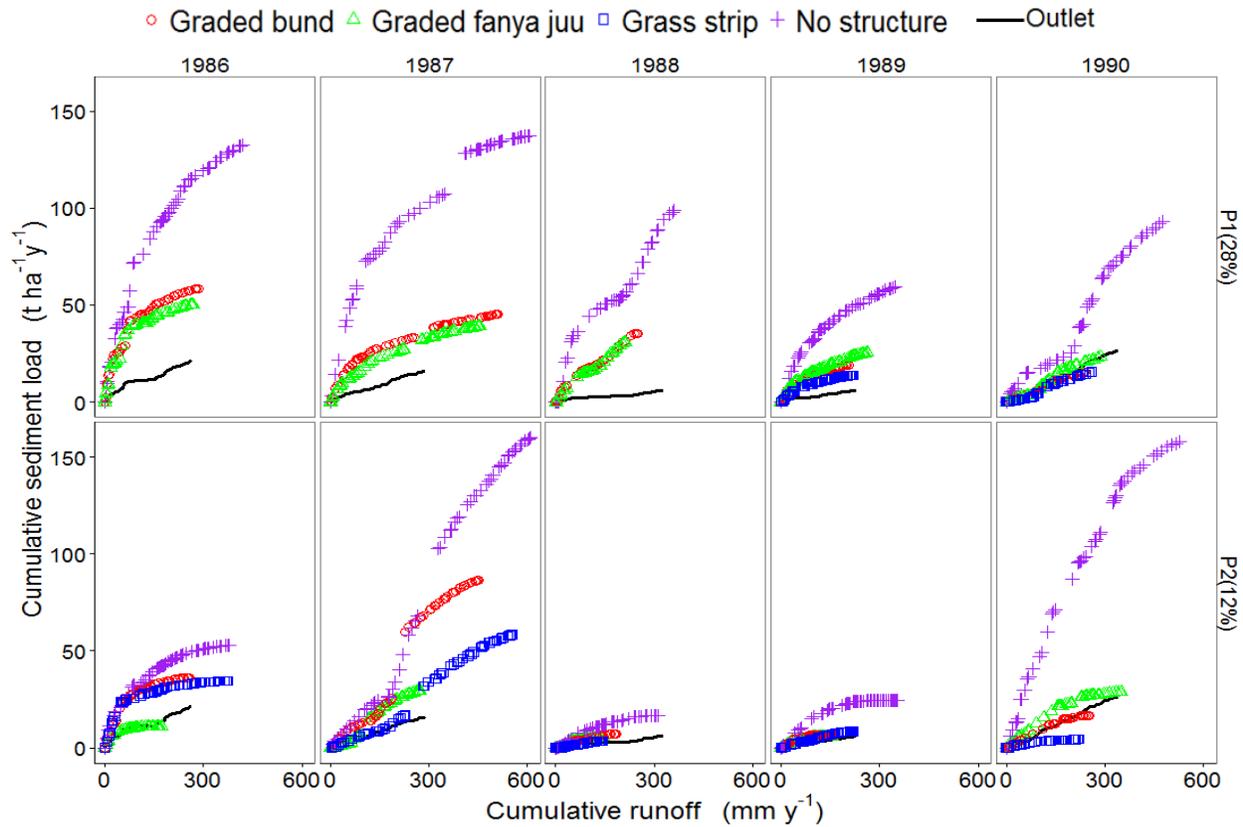


Figure 4-5. Annual cumulative runoff vs. annual cumulative sediment yield from experimental plots (30 m length, 6 m width) with different soil and water conservation structures at two slope positions (28 % and 12%).

Even though only 2 years of data were available from experimental plots with grass strips at 28% slope, grass strips effectively and significantly reduced both runoff and soil loss compared with other conservation measures and non-conserved (control) plots (Table 4-4). At relatively flatter slope (12%) graded *fanya juu* structures resulted in the greatest reduction of runoff and soil loss compared with other structures, and results were significant compared with non-conserved control plots (Table 4-4).

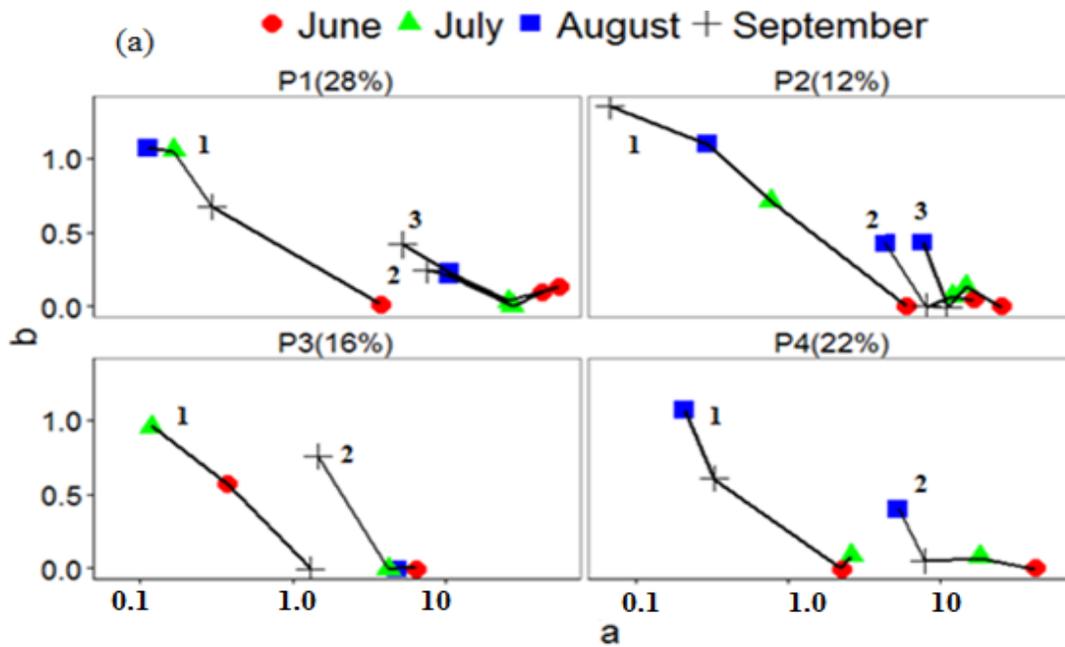
Table 4-4. Runoff, and erosion summary (based on 4 months: June-September) results from plots with different soil and water conservation structure and watershed outlet (based on five-year data: 1986-1990). Standard error of the mean is given between parentheses. Values not sharing the same letter within the same groups of plots (P1-P2) are significantly different ( $p < 0.05$ ).

Group	Plot type	Slope (%)	Length (m)	Width (m)	Direct runoff (mm y <sup>-1</sup> )	<sup>a</sup> SC (g l <sup>-1</sup> )	<sup>b</sup> SSY (t ha <sup>-1</sup> y <sup>-1</sup> )
P1	No structure (EP1)	28	30	6	442.3 (47.3)a	19.7 (1.6)a	104.2 (14.3)a
	Graded bund (EP2)	28	30	6	297.4 (54.5)b	8.7 (1.1)b	34.7 (8.1)b
	Graded <i>fanya juu</i> (EP3)	28	30	6	297.8 (41.4)b	8.7 (0.9)b	33.7 (4.9)b
	Grass strip (EP4)	28	30	6	238.9 (20)b	3.4 (0.4)c	14.8 (1)c
	Watershed outlet			113 ha	291 (20.7)b	3.7 (1)c	15.2 (4.1)c
P2	No structure (EP7)	12	30	6	435.5 (56.5)b	12.3 (1)c	82.3 (31.8)b
	Graded bund (EP6)	12	30	6	262.7 (50.6)a	5.6 (0.6)b	30.6 (14.9)a
	Graded <i>fanya juu</i> (EP5)	12	30	6	220.1 (39.8)a	3 (0.3)a	16.2 (5.4)a
	Grass strip (EP8)	12	30	6	356.7 (119.5)ab	5.3 (0.7)bd	32.1 (15.9)a
	Watershed outlet			113 ha	291 (20.7)a	3.7 (1)ad	15.2 (4.1)a

<sup>a</sup> Sediment concentration calculated based on direct runoff; <sup>b</sup> Specific Sediment Yield.

### 4.3.5 Relating sediment concentration with discharge

Figures 4-6 and C1 in appendix C show how sediment rating coefficients ( $a$  and  $b$ ) for plots (with different sizes) and watershed outlet vary with the progression of the monsoon phase. Although the coefficient of determination ( $R^2$ ) values were small (not shown), it is illustrative and helpful in explaining the differences in scale to relate sediment concentration with discharge as shown in Figure D1 in appendix D. Summary of the rating coefficients are also presented in Tables E1-E2 in appendix E.



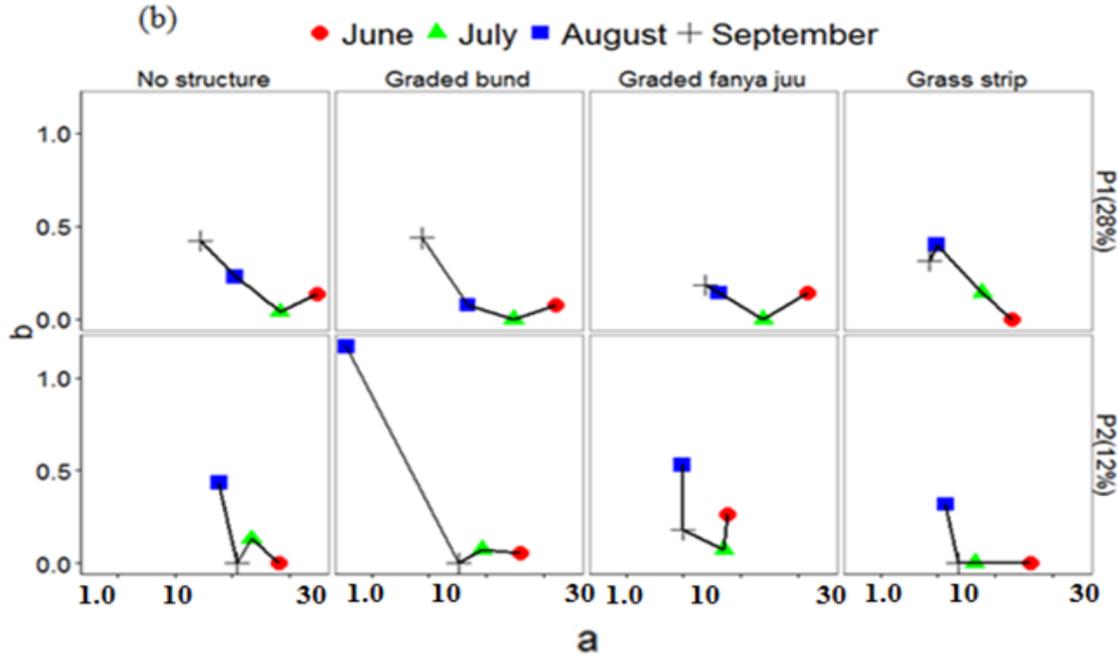


Figure 4-6. Seasonal variations of sediment rating coefficients for groups of long-term (1986-1990) plots without soil and water conservation structures (a), and experimental plots with soil and water conservation structures (b). In Figure 6a numbers 1, 2, and 3 represent different plot sizes of 1, 15, and 30m lengths, respectively.

Coefficient *a* showed an increase with increase in plot size (without conservation structure), micro plots (3 m length) had the smallest value of *a* followed by the watershed outlet. On the other hand, value of *b* decreased with increase in plot size but not significantly.

Soil and water conservation structures resulted in significantly smaller coefficients of *a* compared with control experimental plots (without conservation measure). Rating coefficients (*a* and *b*) at the watershed outlet were mostly with the same order of magnitude with that of experimental plots with conservation structures at 12% slope position (Table E2 in appendix E).

Figures 4-6a and b show how sediment-rating coefficients of cultivated plots vary with the progression of the monsoon season. During June/July at the beginning of the rainy phase, the *a*

values are general greater and the  $b$  values smaller than during August/September towards the end of the rainy phase when the  $a$  values are smaller and  $b$  values greater independent of either plot size or conservation structures (Figures 4-6a and b). However, at the watershed outlet, this trend did not hold. Only June has the greatest  $a$  and smallest  $b$  values (Figure C1 in appendix C) consistent with plot scale results. Rating coefficients  $a$  and  $b$  at the watershed outlet were mostly comparable with the *fanya juu* plots as expected because the watershed was treated with *fanya juu* terraces (see also Tables 4-4 and Figures 4-6b and C1 in appendix C).

## 4.4 Discussion

### 4.4.1 Effect of measurement scale on erosion processes

Overall, sediment concentration and sediment yield were consistently smaller from plots under grassland regardless of measurement scale (Figures 4-3a and 4, Table 4-2). In agreement with this, several studies (Cerdan et al., 2010; Hurni et al., 2005; Le Bissonnais et al., 1998; Ngatunga et al., 1984) found smaller erosion rates from fields under permanent covers (e.g., grassland). In other parts of the world (such as China) Guo et al. (2015) observed erosion rates from grassland between the ranges of 1.6 to 4.0 t ha<sup>-1</sup> y<sup>-1</sup>, while rates from farmland under conventional tillage were between 7.7 to 49 t ha<sup>-1</sup> y<sup>-1</sup>. Likewise, Wilcox (1994) noted that runoff and erosion rates were substantially greater from disturbed plots. Level of soil disturbance not only affect erosion rates but also runoff amount (Moreno-de las Heras et al., 2010). Nyssen et al. (2009b) argued that sediment fluxes are greatly increased by tillage practices. The above findings are consistent with findings of Asselman (1999), Bewket and Sterk, (2003), Nyssen et al. (2004) and Vanmaercke et al. (2010) in that the amount of soil loss depends on the availability of sediment for transport by surface runoff. Tillage practices in the Ethiopian highlands loosens up

the soil, so that it easily can be picked up by the overland flow and thereby increases the soil losses greatly.

We found that when plot length increased from the 3-m long micro plots to the 15-m test plots, sediment concentration from cultivated fields increased significantly by factors of 4 to 18 and sediment yields from 3 to 9 times. Runoff depth was unaffected (Figure 4-4, Table 4-2). Further increase in plot length to 30 m, sediment concentration did not show a trend while the sediment yield decreased due to infiltration of water and decreasing the direct runoff (Figure 4-4, Table 4-2). At the 113 ha watershed outlet, the direct runoff and soil loss was approximately the same as the 30 m runoff plots at the 12% slope with the graded *fanya juu* structures (Group P2 in Table 4-4). This is not unexpected that the losses of the 12% 30 m long plots are comparable with the watershed. The watershed was fully covered with newly installed “*fanya-juu*” structures and has an average watershed slope of approximately 12%. In addition, there is relatively little flat land around the river. While runoff from the *fanya-juus* plots at 28% slope was comparable to direct runoff at the outlet, sediment losses were significantly greater at this slope compared with sediment loss for *fanya-juus* plots at 12% slope and at the outlet indicating that the transport capacity for runoff with slope decreases and sediment will drop out. Previous studies found too that soil loss depends on measurement scale and slope. Le Bissonais (1998) and Moreno-de las Heras et al., (2010) observed that 10 and 15 m long plots, respectively have a greater sediment yield than 1 m plots. Castro et al., (1999) also noted a marked decrease of sheet and rill erosion and soil loss from smaller (1 m<sup>2</sup>) plots to bigger (77 m<sup>2</sup>) and catchment scales.

The initial increase in erosion rates with increase in plot length suggest that at the beginning the transport capacity at the 3 m plots was not reached because rills were not formed for this length of plots as we observed in the 2012 and 2013 experiments (Table 4-3). In the 15 m long plots

rills were formed in the beginning of the rainy phase and thus there was sufficient sediment available to reach the transport capacity resulting in great values of coefficient  $a$  in the sediment rating curve in Eq. 4-2 (Table E1 in appendix E). The observed reduction in sediment yield with further increase in plot length (to 30m) is caused by re-infiltration of runoff water after a greater but short rainfall intensity temporarily exceeding the infiltration capacity of the soil (Van de Giesen et al., 2000). This in turn decreases the amount of sediment that the runoff water can carry when at sediment transport capacity.

#### ***4.4.2 Performance of soil and water conservation structures***

All conservation measures reduced direct runoff, sediment concentration and sediments yield at the 12 and 28% slopes. The effects were greater on fields at steep slope (28%) than at relatively gentler slope (12%) (Figure 4-5 and Table 4-4). This was in agreement with findings by several authors mostly from arid regions in the Ethiopian Highlands (Haregeweyn et al., 2008; Herweg and Ludi, 1999; Nyssen et al., 2009a). These authors indicated that reduction in sediment loss were mainly due to decreased runoff because of soil and water conservation structures. Our experiments show that in addition sediment concentration decreases as well.

#### ***4.4.3 Implication of sediment rating coefficients***

Sediment rating curve fitting coefficients from this study were comparable with reports by Vanmaercke et al. (2010) in the northern Ethiopian highlands where they fitted sediment concentration to discharge for ten sub-catchments with areas ranging between 121 to 4592 km<sup>2</sup> within the Geba (5,133 km<sup>2</sup>) catchment in the Blue Nile basin. The authors found that sediment rating coefficients ( $a$  and  $b$ ) varied between 0.0 to 7.37 and 0.27 to 1.4, respectively with coefficient of determination ( $R^2$ ) between the ranges 0.06 to 0.80.

Our results clearly showed existence of relationship between sediment rating coefficients ( $a$  and  $b$ ) with plot size and the progression of the monsoon season (Figure 4-6a). Greater values of coefficient  $a$  were observed for larger plots (15 and 30 m plots), while coefficient  $b$  decreased as plot size increased. This is in agreement with Asselman (2000) who suggested that sediment rating coefficient  $b$  could be affected by measurement scales. Vanmaercke et al. (2010) observed temporal differences in sediment rating coefficients, values of coefficient  $a$  were greater at the beginning of the rainy period and decreased with the progression of the rainy season. Syvitski et al. (2000) also observed that rating coefficient  $a$  is inversely proportional to discharge. Vanmaercke et al. (2010) attributed the change in rating coefficient  $a$  to changes in sediment supply during the rainy season. This was in agreement with our observations of decreasing trend of coefficient  $a$  as the monsoon (rainy) phase progresses and the rill network reached an equilibrium state in which the runoff water could be carried off through the existing network without the need to further widen the rills.

Sediment rating coefficient  $a$  indicates availability of sediment, where smaller values of  $a$  indicating erosion processes being at source limit (Wang et al., 2008). While coefficient  $b$  indicates the power of runoff water to transport sediment, where smaller  $b$  values suggesting erosion processes being at transport limit (Hassan, 2013; Wang et al., 2008). These suggest that smaller plots with 3m length (micro plots) sediment source and not the transport would be limiting, while as plot size increases to 15 m (test plots) sediment transport would be at peak. However, with further increase in plot length to 30 m (experimental plots) and since the watershed remained at the 12% slope watershed outlet erosion processes would likely be at transport limit, especially during the beginning of the monsoon (rainy) phase.

## 4.5 Conclusion

In the (humid) Anjeni watershed, observed results from 2012 and 2013 monsoon season as well as long-term (1986-1990) plot and watershed outlet data confirmed that erosion processes are significantly affected by measurement scales and land use. Sediment rating coefficients were also affected by measurement scales and with the progression of the monsoon phase. Greater erosion rates were observed from agricultural lands, compared with grassland. Sediment concentrations were peak at the beginning of the rainy monsoon phase (June - July). This highlights the significance of tillage practices and level of soil disturbance in making sediment available for transport.

Initially, greater sediment concentration and yield were observed with increases in plot length from 3 m (micro plot) to 15 m (test plot). Similarly, sediment rating coefficient  $a$  showed an increase with increase in plot length, while coefficient  $b$  decreased with increase in plot length. Soil and water conservation measures were effective in reducing runoff and erosion rates, which was corroborated by significantly smaller values of sediment rating coefficient  $a$  compared with control plots without conservation structures. As the watershed was treated with *fanya juu* terraces, erosion rates and sediment rating coefficients ( $a$  and  $b$ ) at the watershed outlet were within the same order of magnitude with the *fanya juu* plots at 12% slope.

Overall findings confirmed that as plot size gets smaller (eg. 3 m length) sediment transport processes are at source limit and with increase in plot length (15 m) sediment source increased with the formation of rill and sediment transport was at its peak. Further increase in plot length to 30 m (experimental plots) and at watershed scale exhibit sediment transport processes at transport limit. While this study focuses in highlighting how erosion processes are affected by

measurement scales, land use, and management practices, further studies are needed to identify optimum distances at which installation of soil and water conservation structures would be effective.

## **4.6 Acknowledgements**

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## 4.8 Appendix

### Appendix A

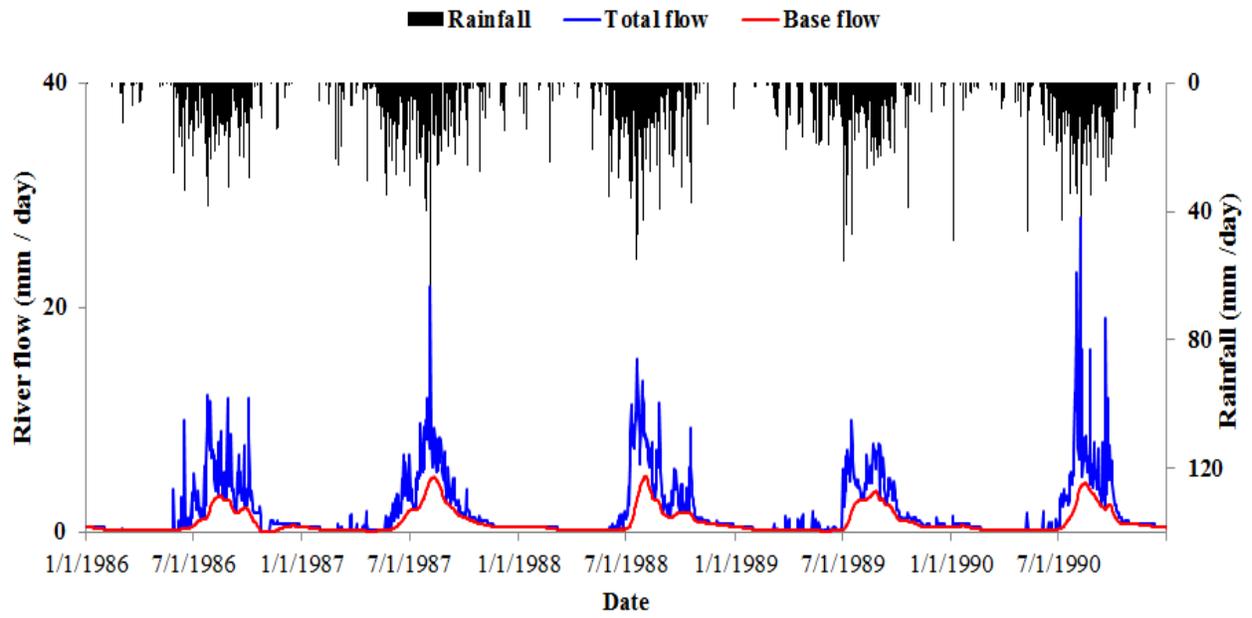


Figure A1. Flow characteristics at the Anjeni watershed outlet (based on data from 1986-1990).

## Appendix B

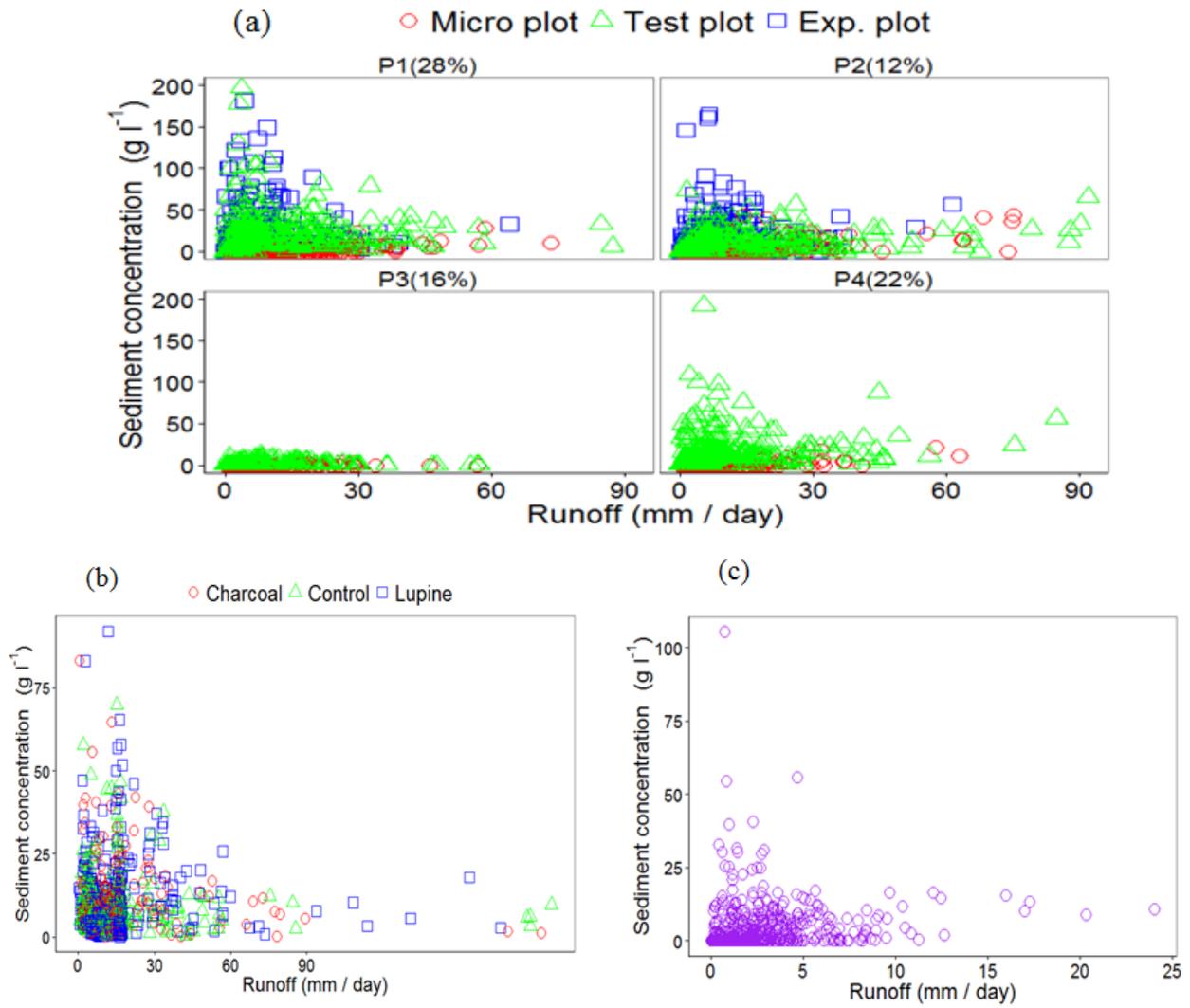


Figure B1. Event runoff vs. sediment concentration from groups (P1-P4) of long term plots without conservation structures at different slope positions and of different sizes (micro plot: 3 m length, 1 m width; test plot: 15 m length, 2 m width; Experimental plot: 30 m length, 6 m width) (a), observations in 2012 and 2013 from plots (3 m length, 1.5 m width) (b), and watershed outlet (113 ha) (c).

Appendix C

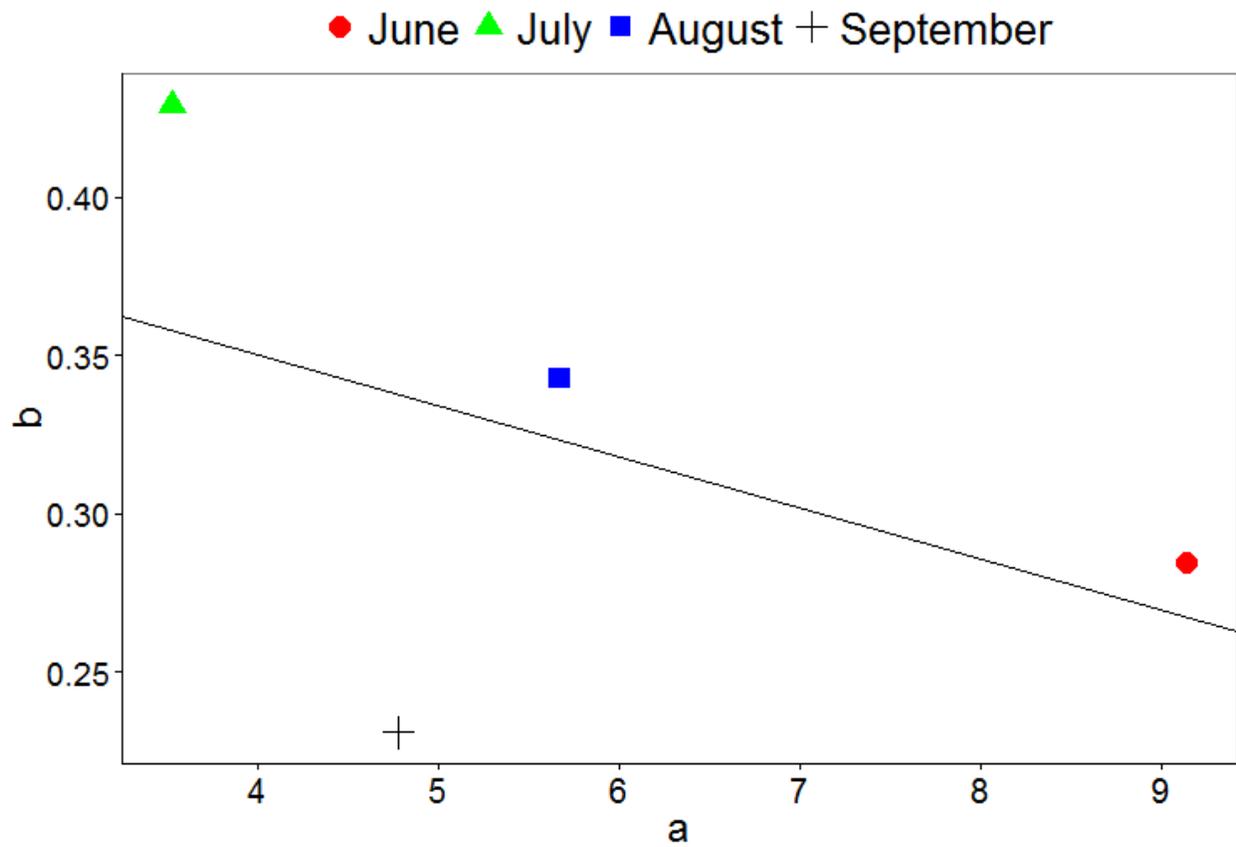
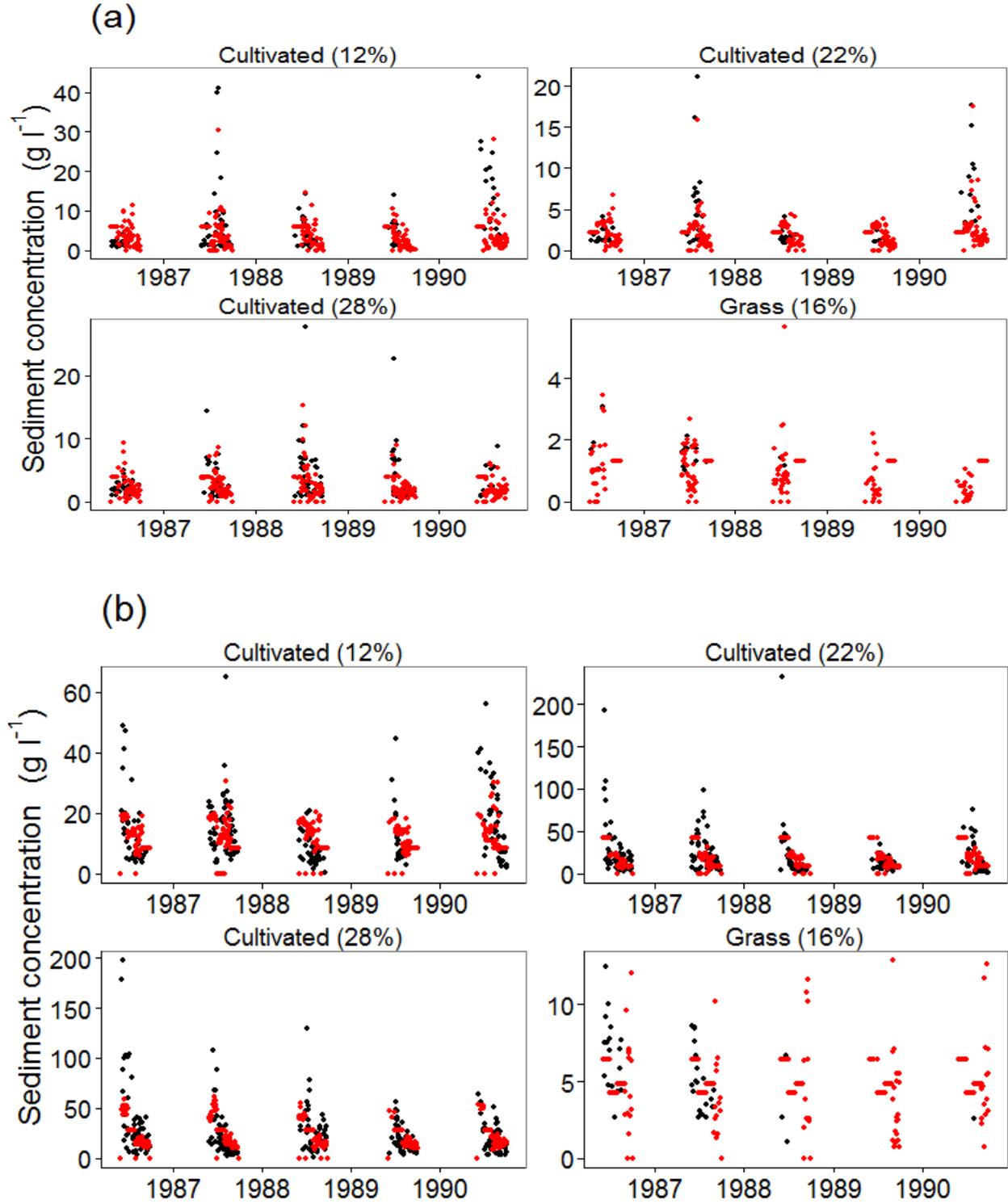


Figure C1. Seasonal variations of sediment rating coefficients at watershed outlet.

Appendix D



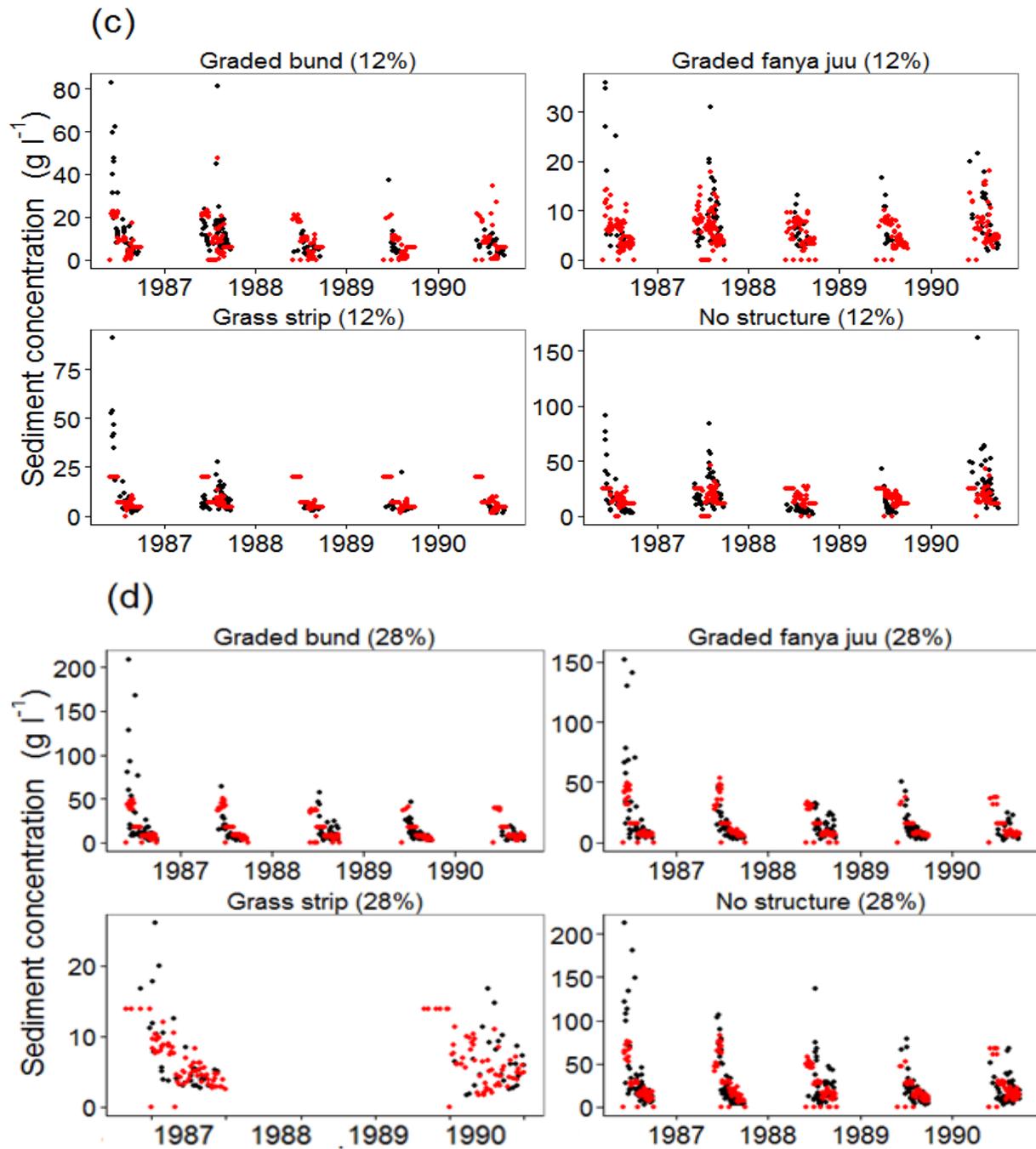


Figure D1. Observed and sediment concentration ( $\text{g l}^{-1}$ ) (black circles) and simulated sediment concentration using power regression model (red circles) at different land uses and plots scales: 3 m long micro plots (a), 15 m long test plots (b), 30 m long experimental plots with different soil and water structures at 12% slope (c) and 28 % slope (d)

## Appendix E

Table E1. Summary of sediment rating curve-fitting parameters for plots (without soil and water conservation structures) and watershed outlet. Standard error of the mean is given between parentheses. Values not sharing the same letter within the same groups of plots (P1-P4) are significantly different ( $p < 0.05$ ).

Group	Plot type	Slope (%)	Length (m)	Width (m)	a (-)	b (-)
P1	Micro plot (MP5)	28	3	1	1.1 (0.9)a	0.7 (0.2)a
	Test plot (TP1)	28	15	2	22.2 (8.2)b	0.1 (0.1)a
	Experimental plot (EP1)*	28	30	6	24.3 (11.2)b	0.2 (0.1)a
	Watershed outlet			113 ha	6.6 (1.2)c	0.3 (0.1)a
P2	Micro plot (MP6)	12	3	1	1.8 (1.4)a	0.5 (0.3)a
	Test plot (TP2)	12	15	2	10.3 (2.6)bc	0.1 (0.1)b
	Experimental plot (EP7)*	12	30	6	14.7 (3.8)b	0.1 (0.1)b
	Watershed outlet			113 ha	6.6 (1.2)c	0.3(0.1)ab
P3	Micro plot (MP7)	16	3	1	0.6 (0.4)a	0.5 (0.3)a
	Test plot (TP3)	16	15	2	4.2 (1)ab	0.2 (0.2)a
	Watershed outlet			113 ha	6.6 (1.2)b	0.3 (0.1)a
P4	Micro plot (MP8)	22	3	1	1.3 (0.6)a	0.4 (0.2)a
	Test plot (TP4)	22	15	2	18.4 (8.4)b	0.1 (0.1)a
	Watershed outlet			113 ha	6.6 (1.2)c	0.3 (0.1)a

\*Experimental plots without soil and water conservation structure.

Table E2. Summary of sediment rating curve-fitting parameters for plots (with soil and water conservation structures) and watershed outlet. Standard error of the mean are given between parentheses. Values not sharing the same letter within the same groups of plots (P1-P2) are significantly different ( $p < 0.05$ ).

Group	Plot type	Slope (%)	Length (m)	Width (m)	a (-)	B (-)
P1	No structure (EP1)	28	30	6	24.3 (11.2)a	0.2 (0.1)b
	Graded bund (EP2)	28	30	6	16.9 (8.5)ab	0.1 (0.1)a
	Graded <i>fanya juu</i> (EP3)	28	30	6	16.2 (7.6)ab	0.1 (0)a
	Grass strip (EP4)	28	30	6	6.8 (2.6)b	0.2 (0.1)a
	Watershed outlet		113 ha		6.6 (1.2)b	0.3 (0.1)a
P2	No structure (EP7)	12	30	6	14.7 (3.8)b	0.1 (0.1)a
	Graded bund (EP6)	12	30	6	8.8 (4.1)ab	0.3 (0.3)a
	Graded <i>fanya juu</i> (EP5)	12	30	6	5.3 (1.2)a	0.3 (0.1)a
	Grass strip (EP8)	12	30	6	8.8 (3.8)ab	0.1 (0.1)a
	Watershed outlet		113 ha		6.6 (1.2)a	0.3 (0.1)a

# CHAPTER 5: NITROUS OXIDE AND METHANE EMISSIONS FROM AGRICULTURAL SOILS IN TROPICAL REGIONS: THE ANJENI WATERSHED

## Abstract

While agricultural practices are widely reported to greatly contribute to anthropogenic greenhouse gas (GHG) emissions, there are only limited measurements available for emission levels in Africa. We measured nitrous oxide (N<sub>2</sub>O-N) and methane (CH<sub>4</sub>) emission rates from agricultural fields in the sub-humid Anjeni watershed in the Ethiopian highlands before, during, and after the rainy monsoon phase. Nitrous oxide emission rates varied from -275 to 522 μg m<sup>-2</sup> h<sup>-1</sup> and methane emissions ranged from -206 to 264 μg m<sup>-2</sup> h<sup>-1</sup> with overall means of 51 and 5 μg m<sup>-2</sup> h<sup>-1</sup> for N<sub>2</sub>O-N and CH<sub>4</sub>, respectively. Generally, greater fluxes of nitrous oxide were observed towards the end of the rainy monsoon phase. Nitrous oxide emissions were regularly greater from charcoal and lupine plots, 38 and 150% greater than control plots, respectively. While in contrast, methane emissions were mostly negative from lupine and charcoal treated plots, 73 and 164% lower than control plots, respectively.

*Keywords:* Greenhouse gas emission, nitrous oxide, methane, charcoal

## 5.1 Introduction

While existing literature shows that burning of fossil fuel takes the lion's share of world's Greenhouse Gas (GHG) ( $\text{CH}_4$ ,  $\text{CO}_2$ , and  $\text{N}_2\text{O}$ ) emissions, agriculture contributes for 52 to 84% of world anthropogenic methane and nitrous oxide emissions (Smith et al., 2008). Africa accounts for only 3.7 % global emissions (Canadell et al., 2009). The major source of anthropogenic greenhouse gas emission in Africa is land use change, clearing of forestlands for food production (Bellarby et al., 2014; Bombelli et al., 2009). With increasing need for agricultural land for food production, due to population pressure, deforestation is severe in Sub-Saharan Africa (Feoli et al., 2002; Taddese, 2001). While the great portions of former forested land are presently converted to agricultural lands, the contribution of agricultural lands to greenhouse gas emission is not well documented (Bellarby et al., 2014).

Agricultural soils can act both as sources and sinks of greenhouse gases (Chapuis-Lardy et al., 2009; Smith et al., 2008). Tropical forests contribute significant proportion of global nitrous oxide emission (Keller et al., 1986). Though, greenhouse gas emissions are expected to show greater spatial and temporal variations (Andersson, 2003; H'enault et al., 2012; Luo et al., 2013; Molodovskaya et al., 2012), data availability is much scarce in the Sub-Saharan region including Ethiopia, and thus, emissions are not quantified and spatial and seasonal variations in greenhouse gas emission from agricultural fields are unknown. Such information is critical, in order to seek effective climate mitigation strategies.

Studies in many parts of the world have shown that biochar effectively reduced greenhouse gas emissions. Several authors (Kammann et al., 2012; Singla and Inubushi, 2014; Wang et al., 2012; Zhang et al., 2010) observed significant reduction in nitrous oxide emissions due to biochar addition to soils. In contrast, however, effect of biochar on methane emission has shown contrasting results. Karhu et al. (2011); Feng et al. (2012) observed reduced methane emissions from biochar amended soils, while (Singla and Inubushi, 2014; Wang et al., 2012; Zhang et al., 2010) measured increased methane emissions from biochar treated soils.

The aim of this study was twofold: (1) to quantify seasonal and spatial soil emissions of nitrous oxide and methane from smallholder agricultural fields by conducting field measurements, and (2) to assess the effects of charcoal amendment and deep-rooted leguminous (lupine) crop on nitrous oxide and methane emission rates.

## **5.2 Methodology**

### ***5.2.1 Study site description***

The Anjeni watershed is located in the Ethiopian Highlands at 10°40' N, 37°31'E (Figure 5-1). The watershed total catchment area is 113 ha, with elevation between 2407 and 2507 m (Herweg and Ludi, 1999). Agriculture is the main land use system in the watershed, and most fields have been cultivated for longer than fifty years (Zelege 2000). The watershed has a unimodal rainfall distribution with average precipitation of 1610 mm y<sup>-1</sup>. Minimum and maximum air and soil (at 5 cm depth) temperature ranges between 9 - 23 and 8 - 26 °C, respectively (Figure 5-2). The soils of Anjeni have developed from the basalt and volcanic ash with major soils Alisols, Nitisols, and

Cambisols covering more than 80% of the watershed (Zeleeke, 2000). The deep Alisols cover the bottom part of the watershed; moderately deep Nitisols cover the mid-transitional, gently sloping parts of the watershed, while the shallow Regosols and Leptosols cover the high, steepest part of the watershed (Zeleeke, 2000).

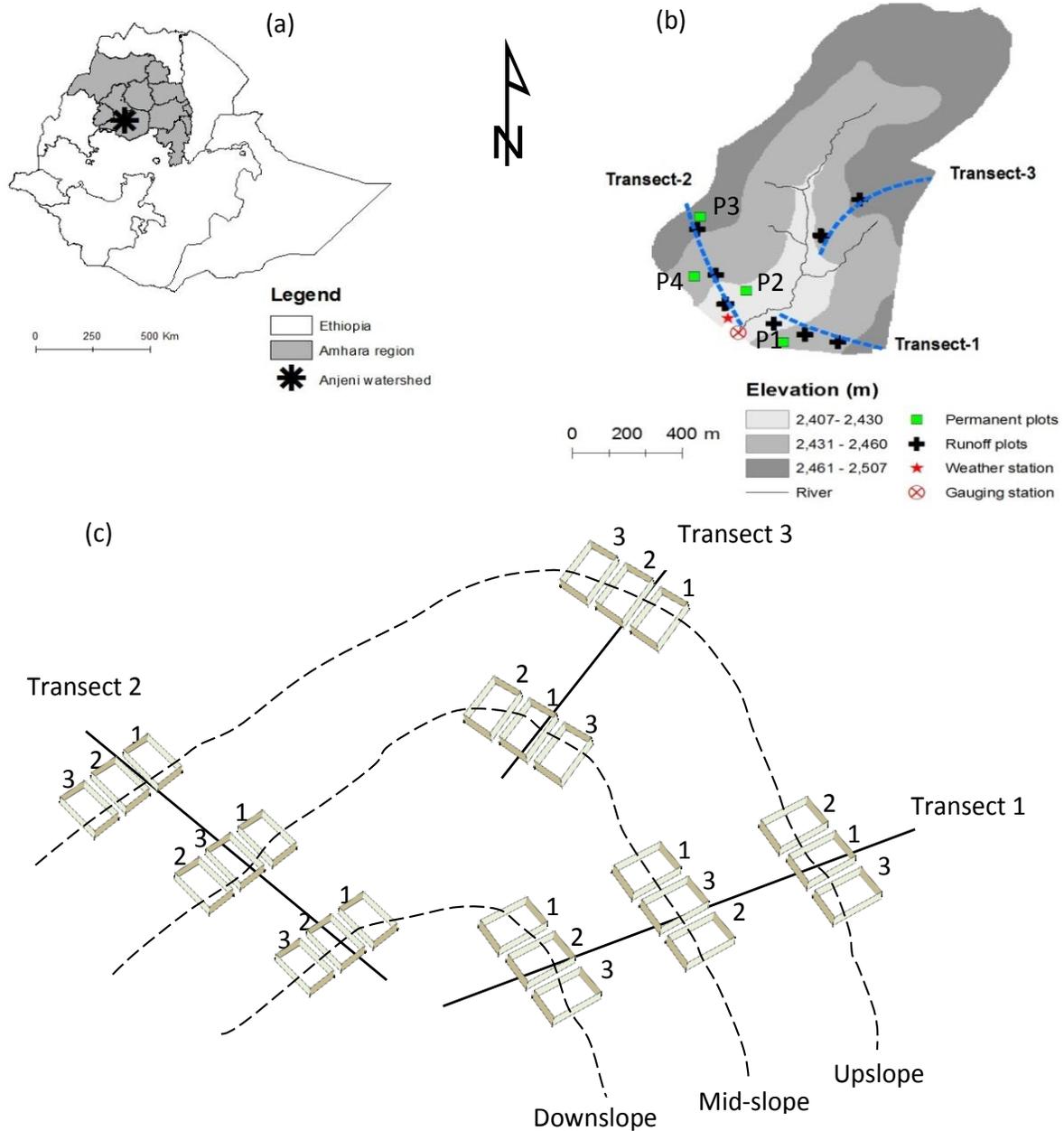


Figure 5-1. Location of the Anjeni watershed in the Amhara region in Ethiopia (a), with the location of downslope transects and runoff plots indicated in (b) and (c – not to scale). Dashed lines in (c) are elevation contours. Plots labels represent treatment types: 1 = barley without amendment (Control), 2= barley with charcoal, and 3= Lupine. Figure is adopted from Bayabil et al. (2015b).

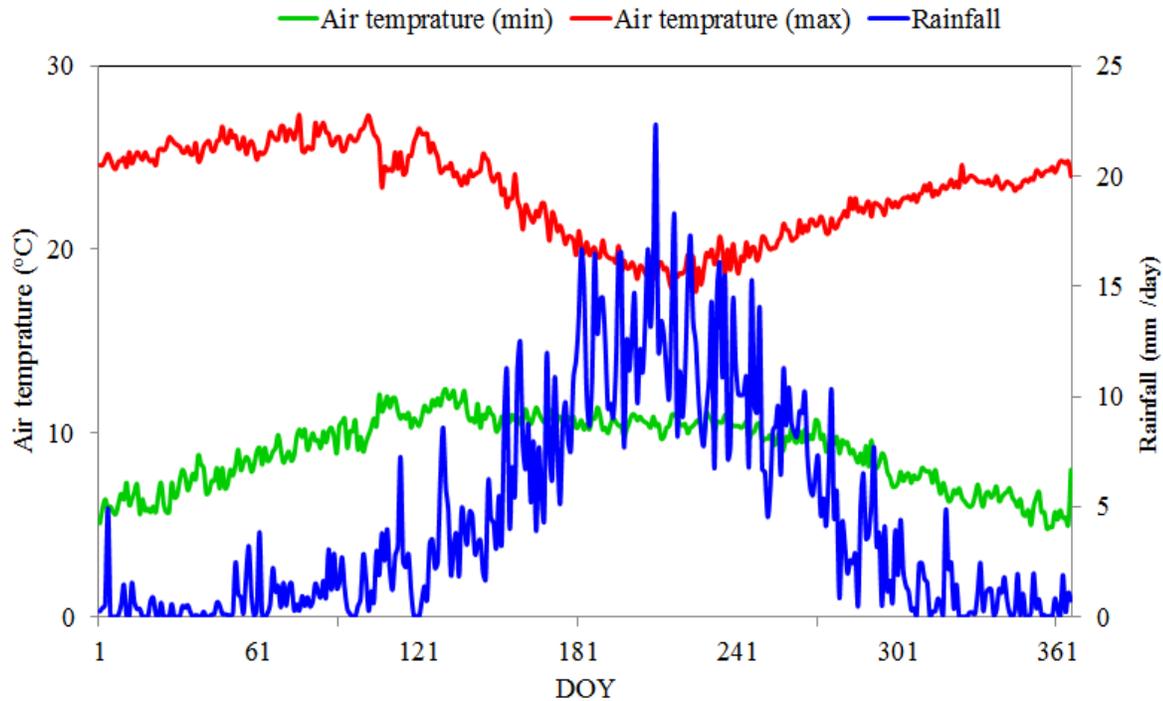


Figure 5-2. Long term average daily air temperature (minimum and maximum) and daily rainfall for the Anjeni watershed (based on observations from 1986 to 1996).

### 5.2.2 Experimental Setup

This study was conducted in parallel with another study Bayabil et al. (2015b) that used 24 runoff plots that were laid out in eight replicates (Figure 5-1c). Each replicate consisted of three treatments (barley grown on non-amended and charcoal amended soils and a leguminous crop, lupine, grown on non-amended soil). Static chambers were installed adjacent to 24 runoff plots (15 cm far from plot boundaries) on fields with the same treatments as the runoff plots. Detailed information about the plots setup and treatments can be found in Bayabil et al. (Bayabil et al., 2015a, 2015b). Twenty-four chambers (one chamber per plot) were installed immediately after

seeding of the plots. We followed the same construction procedure for the static chambers as by Mason (2014) and Molodovskaya et al., (2011). Plastic buckets (19-L volume) were cut in half, and the top part was carefully installed wide end down (5 cm below ground) using handheld hoe. A removable second plastic bucket (19-L volume) fitted with sampling and vent ports and rubber bottle septum (easily penetrable by sampling syringes) was used to as a top cover during extraction of gas samples. To ensure airtight seal between chamber installed in the soil and top cover, rubber band was put around the outside part of the static chamber, on which the cover was put at the top. Similar to the one used by (Molodovskaya et al., 2011) a second septum with aluminum pipe (5 cm length) was used to maintain air pressure equilibrium inside the chambers.

### ***5.2.3 Agronomic practices on plots***

Charcoal amendments and deep-rooted leguminous crop (Lupine) were applied to plots in both 2012 and 2013, while gas samples were collected only in 2013. While lupine seedbeds are typically not tilled, tillage was done in 2012 as plots were originally designated to be sown with alfalfa seeds, another deep rooted leguminous crop (Bayabil et al., 2015b). When alfalfa proved to be unsuccessful, lupine was sown on the tilled soil. In 2013, only the barley plots were tilled (three times between the end of May to middle of June) and seeded. Lupine seeds were sown on untilled plots, which is a more common practice in the area. Also in line with farmer practices, all barley plots were fertilized with 100 kg/ha Di-Ammonium Phosphate (DAP; 46% Nitrogen, 23% Phosphorous, and 21% Potassium) during seeding (middle of June), and 100 kg/ha of Urea (100% Nitrogen) one month after sowing. While lupine plots were not fertilized.

Charcoal was applied on charcoal-amended barley plots, at a fixed rate of 12 ton/ha during tillage in 2012 and 2013. Charcoal was prepared from *Eucalyptus camaladulensis* and manually crushed to obtain relatively uniform particle size (2 mm diameter) and then manually incorporated on the top 20 cm of the soil (more details are given in Bayabil et al. 2015a).



Figure 5-3. Static chamber (collar) installed in the field adjacent to runoff plots (left) and static chambers with covers on the top ready for gas sampling (right).

#### ***5.2.4 Gas sampling and flux calculation***

Four field sampling campaigns were conducted in 2013 before, during, and at the end of the rainy monsoon phase that lasts from June to September. The rest of the year has very little rain and the soils dry out. Two sampling campaigns were conducted towards the end of the dry season on May 10 and 14, 2013; the third campaign was conducted towards the end of the rainy monsoon phase on September 5, 2013; and the last (fourth) sampling campaign was conducted

after the end of the monsoon season on October 15, 2013 when the soils are drying out. During the field campaigns soil temperature was measured at 5 cm depth) using Taylor TruTemp® Instant Read Digital Thermometer.

During each sampling campaign, samples were collected from all 24 chambers the same day and sampling lasted usually between 10:00 AM to 4:00 PM. Four samples were extracted from each chamber at 10 minutes interval (0, 10, 20, and 30 minutes). A 15 ml volume syringe was used to extract gas samples from the static chambers. Samples were then put into 10 ml volume pre-evacuated and air tight sealed glass vials. Gas samples were then shipped to the USA for laboratory analysis and were analyzed in the soil and water lab, at Cornell University. Concentrations of N<sub>2</sub>O and CH<sub>4</sub> in the samples were determined using a Gas Chromatograph (Agilent Technologies 6890N) that has a 95% confidence interval detection limit of + / - 10.2 µg N<sub>2</sub>O-N m<sup>-2</sup> hr<sup>-1</sup> (Mason, 2014). Finally, N<sub>2</sub>O and CH<sub>4</sub> fluxes from each chamber from a given day were computed by calculating the slope of a linear regression model by fitting concentrations vs. sampling time and using the equation (Eq. 5-1) below:

$$F_g = \left( \frac{\Delta C}{\Delta t} \right) \left( \frac{V_c}{A_c} \right) \left( \frac{M_G}{V_m} \right) \quad (5-1)$$

where  $F_g$  nitrous oxide or methane in µg m<sup>-2</sup> h<sup>-1</sup>),  $\frac{\Delta C}{\Delta t}$  is slope of linear regression model (by fitting gas concentration vs. sampling time),  $V_c$  volume of chamber in m<sup>3</sup>,  $A_c$  area of chamber at the base in m<sup>2</sup>,  $M_G$  molecular mass of gas (nitrous oxide or methane in g mol<sup>-1</sup>), and  $V_m$  is molar volume of gas (in m<sup>3</sup> mol<sup>-1</sup>) at chamber pressure and temperature.

Finally, non-parametric analysis of variance (Kruskal-Wallis) tests were performed in R (R Development Core Team 2010) to test the significance of treatments and elevation range on nitrous oxide and methane emissions.

## 5.3 Results

### 5.3.1 Nitrous oxide ( $N_2O-N$ ) emission

There was a large variation in measured nitrous oxide ( $N_2O-N$ ) emissions, which ranged from -275 to 522.3  $\mu\text{g m}^{-2}\text{h}^{-1}$  (Figure 5-4) and with overall mean of 51.3  $\mu\text{g m}^{-2}\text{h}^{-1}$  (Table 5-1). Nitrous oxide emissions during the dry periods (both before and after the monsoon rainy season) were smaller than emissions during the rainy phase when the soils were near saturation (Table 5-1, Figure 5-5).

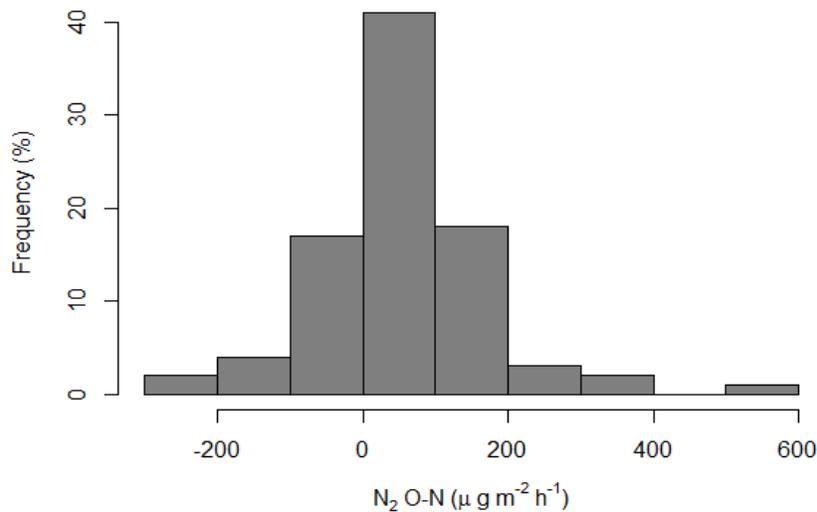


Figure 5-4. Distribution of observed nitrous oxide emissions from 24 static chambers and four sampling campaigns in 2013.

Table 5-1. Summary of methane and nitrous oxide emissions, with standard error of means given between parentheses. Results are based on combined data from all three elevation ranges along three transects.

Sampling date	Control	Charcoal	Lupine	Control	Charcoal	Lupine
	N <sub>2</sub> O-N ( $\mu\text{g m}^{-2} \text{h}^{-1}$ )			CH <sub>4</sub> ( $\mu\text{g m}^{-2} \text{h}^{-1}$ )		
10-May-2013	60 (93)	34 (92)	43 (119)	26(35)	13(33)	58(71)
14-May-2013	12 (104)	1 (147)	40 (120)	20(54)	-9(98)	35(53)
5-Sep-2013	54 (59)	122 (167)	176 (51)	41(123)	-15(101)	-42(63)
15-Oct-2013	-3 (74)	7(156)	41 (125)	1(57)	-43(70)	-11(89)
Mean*	32 (15) <sup>a</sup>	44 (26) <sup>ab</sup>	80 (22) <sup>b</sup>	22 (14) <sup>a</sup>	-14 (15) <sup>b</sup>	6 (15) <sup>ab</sup>

\*statistical tests were performed on combined data from all four sampling campaigns.

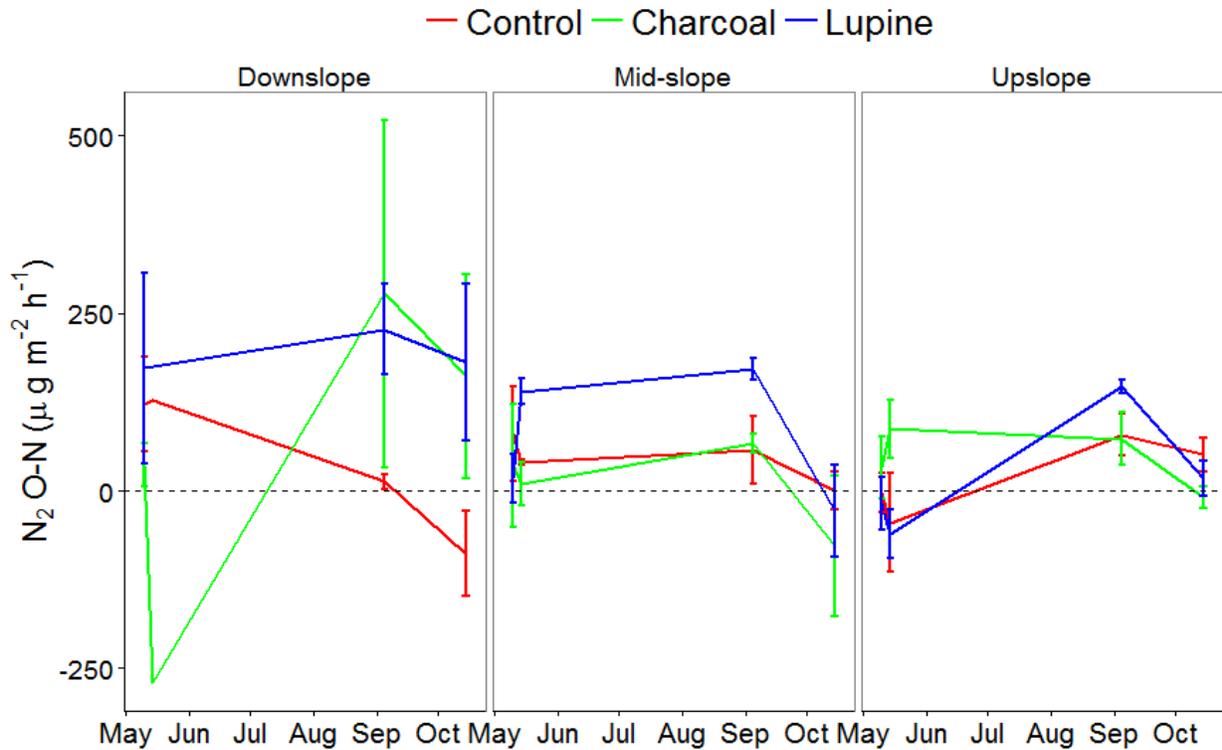


Figure 5-5. Seasonal and spatial (along the elevation gradient) emissions of nitrous oxide from plots with different treatments. Vertical lines represent standard errors of the means for each treatment during each sampling campaign.

Overall, average nitrous oxide emissions from Lupine plots were greater followed by emissions from charcoal plots (Table 5-1). Nitrous oxide emissions were slightly greater from plots at downslope position compared to upslope and mid-slope plots, especially during and after the end of the rainy monsoon phase, but differences were not significant (Figure A1a in appendix A). Correlation between nitrous oxide emissions and soil temperature (at 5 cm depth) was poor for all treatments (Figure B1a in appendix B).

### ***5.3.2 Methane (CH<sub>4</sub>) emissions***

Similar to nitrous oxide emissions, methane (CH<sub>4</sub>) emissions also showed large variation between the ranges -206 to 264  $\mu\text{g m}^{-2}\text{h}^{-1}$  (Figure 5-6). Methane emissions from charcoal plots were mostly negative, with overall average emission of -14  $\mu\text{g m}^{-2}\text{h}^{-1}$ . While average emissions of the control plots were consistently positive during all sampling campaigns (Figure 5-7, Table 5-1). The difference in overall mean methane emissions between the control and the charcoal plots was statistically significant ( $p < 0.1$ ) (Table 5-1). Methane emission from lupine plots was intermediate between the charcoal and control plots. There was no consistent trend in methane emissions along the slope catena (Figure A1b in appendix A). Correlation between methane emissions and soil temperature (at 5 cm depth) was also poor (Figure B1b in appendix B).

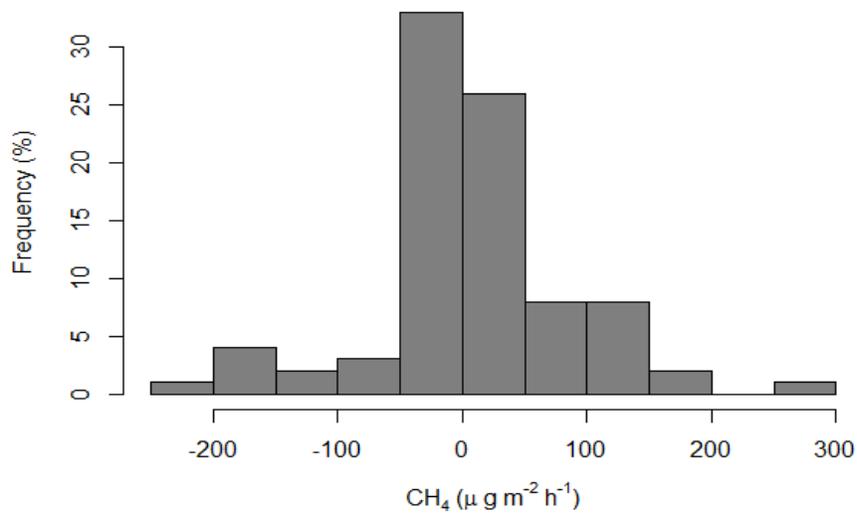


Figure 5-6. Distribution of observed methane emissions from 24 static chambers and four sampling campaigns in 2013.

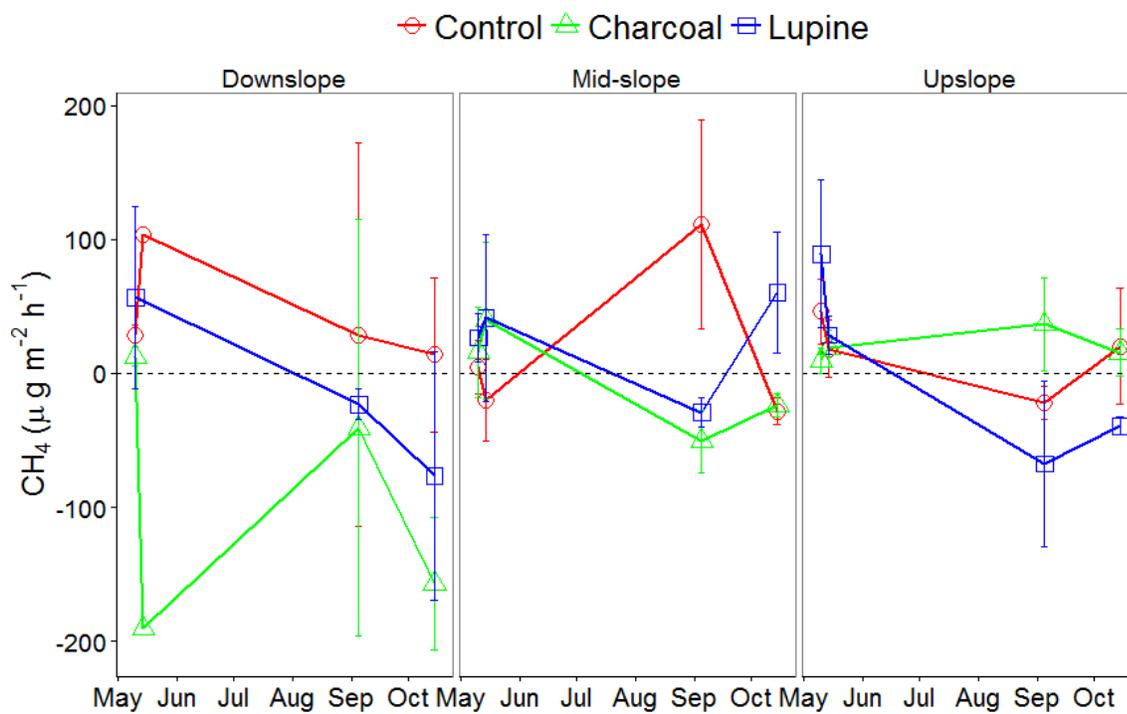


Figure 5-7. Seasonal and spatial (along the elevation gradient) methane emissions from plots with different treatments. Vertical lines represent standard errors of the means for each treatment during each sampling campaign.

## 5.4 Discussion

Our observations of negative emissions of nitrous oxide are common and were also observed by (Luo et al., 2013) across three different ecosystem types in Germany, Australia and China; Molodovskaya et al., 2012 in upstate New York State, USA; Zhang et al., 2010 in China. Overall, average nitrous oxide emissions from Lupine plots were greater followed by emissions from charcoal plots. Thus, charcoal did not reduce nitrous oxide emissions compared to non-amended control plots contrary to the observations by Zhang et al. (2010); Kammann et al. (2012); Wang et al. (2012).

Nitrous oxide fluxes measured in our study were almost an order of magnitude greater than results observed by (Andersson, 2003) who reported a maximum mean of  $5.5 \mu\text{g m}^{-2} \text{h}^{-1}$  nitrous oxide emission from savanna fields in the arid region of Gambella, in Ethiopia. However, the trend in nitrous oxide fluxes in Gambella was similar with our observations where increased fluxes were observed at the end of the rainy season when soils were wet and the pores filled with water. Similarly, outside Ethiopia (Andersson, 2003; Luo et al., 2013; Molodovskaya et al., 2012; (Luo et al., 2013; Sainju et al., 2012; Zhang et al., 2010, 2015) found greater fluxes when the soil was wet but Chapuis-Lardy et al., (2009) in Madagascar did not.

While there was no previous report about methane emission in Ethiopia, studies in other parts of the world show that negative fluxes are common (Zhang et al., 2010). Negative emissions of methane ( $\text{CH}_4$ ) from charcoal amended plots (Figure 5-4, Table 5-1) were in agreement with

other studies that found biochar promotes uptake of methane resulting in negative fluxes (Karhu et al., 2011). In contrast, however, increased methane emissions were also observed from biochar amended soils (Karhu et al., 2011; Wang et al., 2012; Zhang et al., 2012, 2010). Zhang et al. (2012) argued that while biochar could have immediate effect on nitrous oxide emissions, its effect on methane emissions could take longer periods.

Negative fluxes of both nitrous oxide (N<sub>2</sub>O-N) and methane (CH<sub>4</sub>) confirmed that soils could act both as sinks and sources of greenhouse gas emissions depending on the environmental conditions (Luo et al., 2013; Yu et al., 2013). While both nitrous oxide and methane emissions were expected to increase with increase in soil temperature and moisture status (Gogoi and Baruah, 2012; Sainju et al., 2012; Yu et al., 2013; Zhang et al., 2010), we found that soil temperature was poorly correlated with both nitrous oxide and methane emissions (Figure C1 in appendix C). This was in agreement with Luo et al. (2013) who also found poor correlation with emission rates and soil temperature.

This demonstrates that other factors, generally low availability of nitrogen and organic matter pools (Bayabil et al., 2015a) in accordance to the relatively open nutrient cycling system with a net negative nutrient balance in the region controls Greenhouse Gas (GHG) emission dynamics (Hénault et al., 2012).

Nitrous oxide emissions are regulated by nitrification rates, which in turn is controlled by soil nitrogen status (organic and organic nitrogen sources) (Gogoi and Baruah, 2012; Hénault et al., 2012; Zhang et al., 2010). In the Ethiopian highlands with degraded soils (Bayabil et al., 2015a),

waterlogged (low redox potentials) conditions that favor methane emission are less likely. Sainju et al. (2012) observed that lack of nitrogen fertilization increased methane uptake. This suggests that observed negative methane fluxes (methane uptake) (Figure 5-6, Table 5-1) from this study could be due to degraded soil conditions with low nitrogen inputs.

## **5.5 Conclusion**

Our observations of nitrous oxide ( $\text{N}_2\text{O-N}$ ) and methane ( $\text{CH}_4$ ) emissions in the sub-humid Anjeni watershed provide insights about seasonal and spatial Greenhouse Gas (GHG) emission rates from cultivated fields with charcoal amendment and leguminous lupine crop. We observed both positive and negative fluxes of nitrous oxide and methane emissions. This confirms that soils act as both sources and sinks of greenhouse gases. In addition, seasonal differences in nitrous oxide emissions were observed, with greater fluxes observed towards the end of the rainy monsoon phase. There was no consistent trend of nitrous oxide and methane emissions along the slope catena (Figure A1 in appendix A). Nitrous oxide emissions were mostly greater from lupine plots, while methane emissions were mostly negative from lupine and charcoal treated plots. While this study provides quantified nitrous oxide and methane fluxes, observed fluxes showed greater variability and thus, studies with longer duration (across multiple seasons and years) are needed to verify the results.

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## 5.8 Appendix

### Appendix A

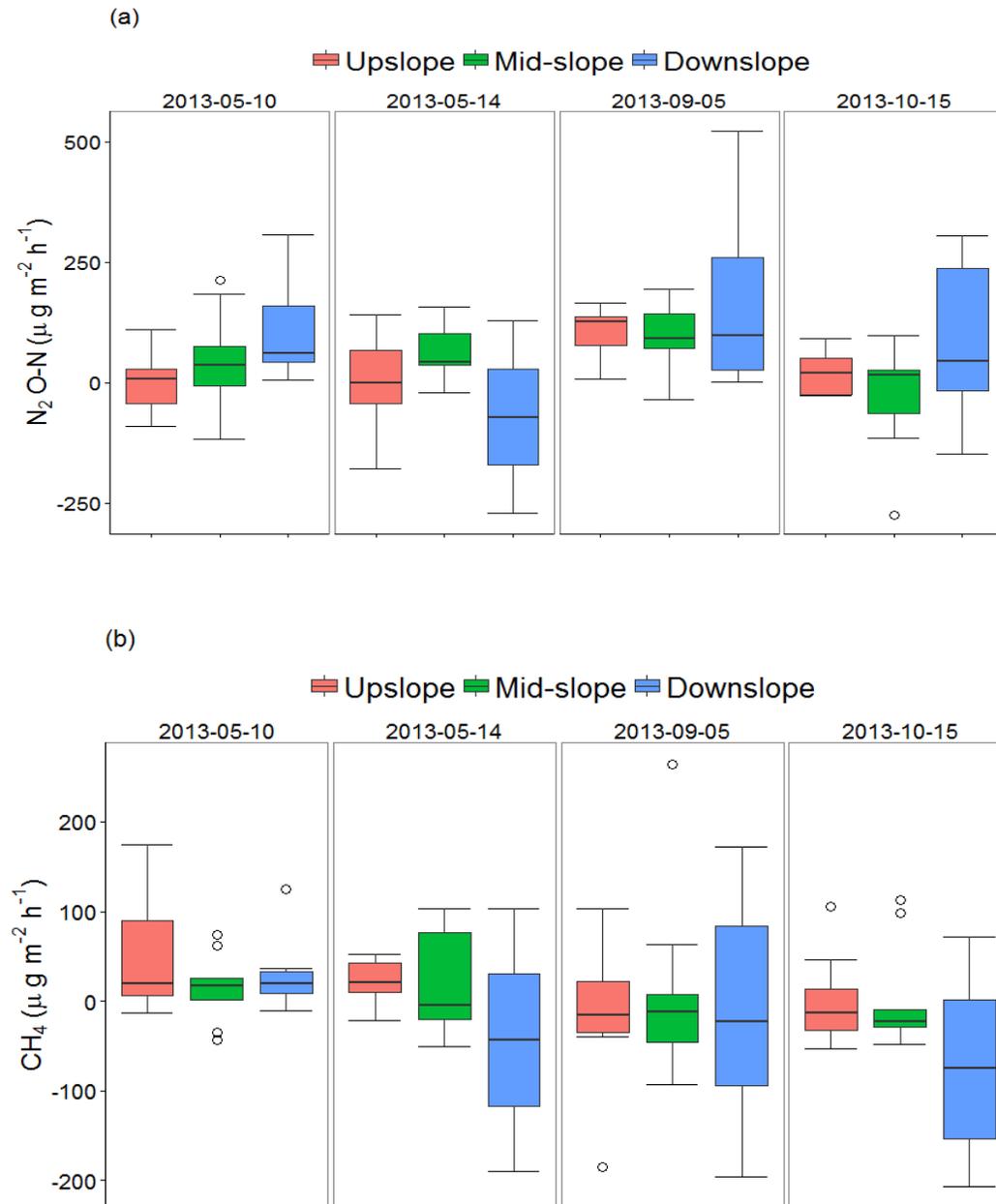


Figure A1. Greenhouse gas emissions along the slope catena (a) nitrous oxide and (b) methane

Appendix B

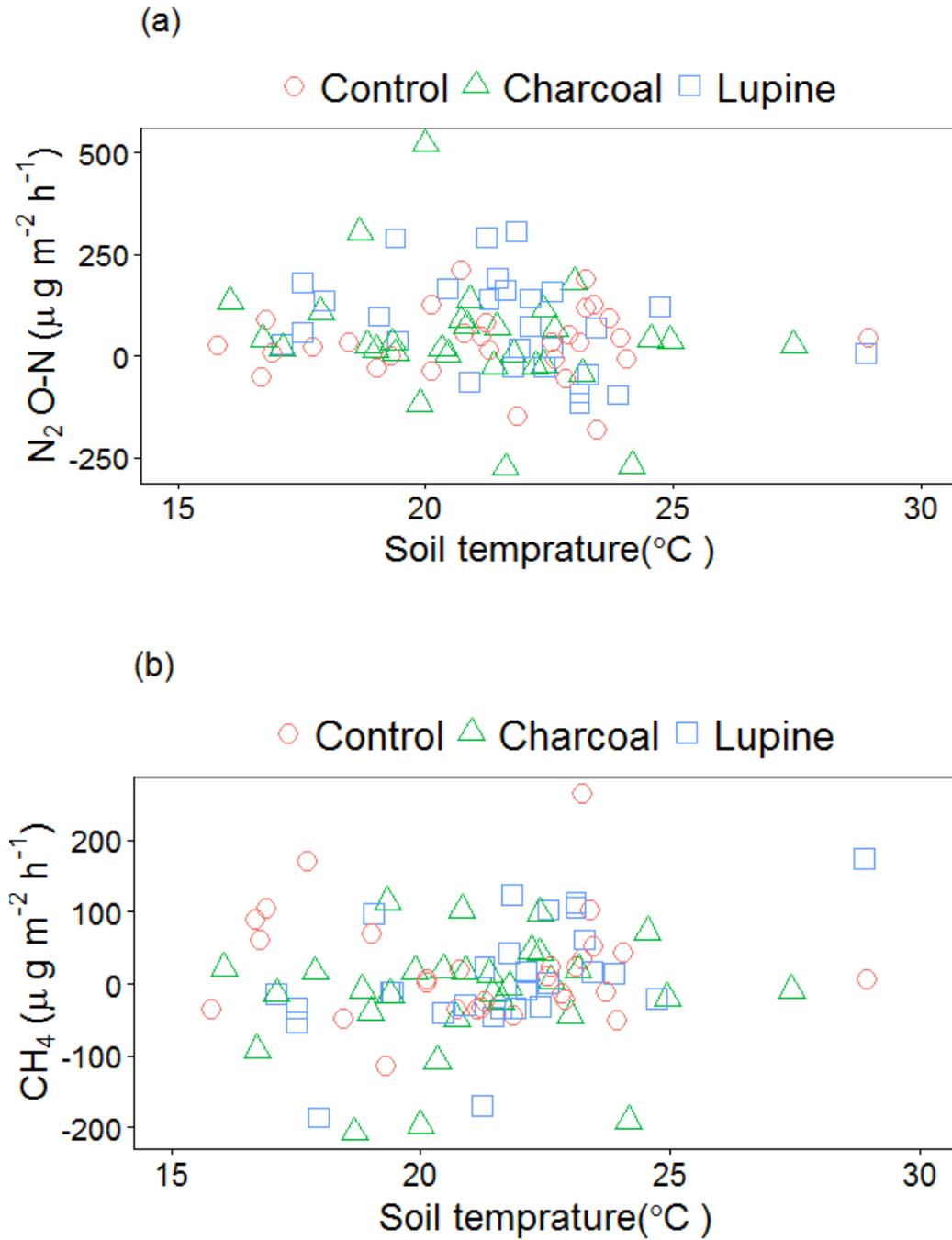


Figure B1. Correlation between soil temperature (at 5 cm depth) and nitrous oxide (a) and methane emissions (b).