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EFFECTIVENESS OF SOIL AND WATER CONSERVATION TREATMENTS ON SOIL AND CATCHMENT RESTORATION

Doctoral Dissertation presented by

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Contents

List of abbreviations and symbols

Abbreviations

A	Mineral soil horizon rich in organic mater
AI	Agricultural micro-catchment with conservation measures
AMC	Antecedent moisture condition
ANOVA	Analysis of variance
ARES	Academie de Recherche et d'enseignement supèrieur
AT	Agricultural micro-catchment without conservation measures
CUD	Commission Universitaire pour le Développement
DF	Degraded micro-catchment with Eucalyptus forest
df	Degrees of freedom
DI	Degraded micro-catchment with conservation measures
DT	Degraded micro-catchment without conservation measures
DTM	Digital terrain model
ELI	Earth and Life Institute
EMAC	Empresa Pública Municipal de Aseo de Cuenca
ETAPA EP	Empresa pública municipal de telecomunicaciones, agua potable, alcantarillado y saneamiento
F value	Statistic value for ANOVA test
INEC	Instituto Nacional de Estadística y Censos
INECEL	Instituto Ecuatoriano de Electrificación
m a.s.l.	Meters above sea level
MC	Micro catchment
NGO	Non-governmental organization
O	Organic soil horizon with decaying plant tissue
OM	Organic Matter
PCA	Principal components analysis
pH	Potential of hydrogen
P_r	Probability
PROMAS	Programa para el manejo del agua y suelo
r	Coefficient of correlation
R^2	Determination coefficient
R^2_{adj}	Adjusted R^2
SCS	Soil Conservation Service
SD	Standard deviation

SENAGUA	Secretaría Nacional del Agua
ENESCYT	Secretaria Nacional de Ciencia, Tecnología e Innovación
SOC	Soil organic carbon
SWC	Soil and water conservation
UMACPA	Unidad de Manejo y Conservación de la Cuenca del Rio Paute
UNCCD	United Nations Conventions to Combat Desertification

Symbols

A	Catchment area (ha)
b_i	Bottom width of gully channel (m)
B_i	Top width of gully channel (m)
BD	Bulk density (g cm^{-3})
C	Carbon content (%)
C/N	Carbon - nitrogen ratio
CS_i	Cross area (m^2)
D	Depth equivalent discharge (mm)
h_i	Thickness of sediment accumulation (m)
I_{30max}	Maximum rainfall intensity in 30 minutes (mm h^{-1})
L	Length of the gully channel (m)
L_i	Length of sediment accumulated (m)
MD	Maximum of depth equivalent discharge (mm / 5 min)
m_{dry}	Mass of sample after drying in the oven (g)
m_{sat}	Mass of sample after it was saturated with water (g)
N	Nitrogen content (%)
P5	Total rainfall amount in the five days preceding the event (mm)
PR	Penetration resistance (g cm^{-2})
Q	Discharge (mm / 5 min)
R	Total rainfall depth (mm)
RC	Runoff coefficient (%)
R_{event}	Rainfall amount during one event (mm)

R_{tot}	Total daily rainfall (mm)
S	Mean slope gradient of the gully channel (%)
S_m	Sediment mass (t)
SP	Slope of the recession part of curve after the peak discharge (dimensionless)
SSY	Specific sediment yield (t ha^{-1} yr^{-1})
S_V	Sediment volume (m^3)
T	Time (yr)
TP	Time to peak (min)
V_b	Volume of the sample before drying (cm^3)
VC	Vegetation cover (%)
V_s	Volume of sediment (m^3)
θ_{sat}	Volumetric soil moisture content at saturation (%)
ρ_w	Density of water (g cm^{-3})

Resumen

La erosión del suelo es un componente importante de la degradación en las montañas Andinas. La erosión del suelo tiene como consecuencia la pérdida de la fertilidad del suelo, la disminución de la calidad de la tierra en las zonas erosionadas y la subsecuente disminución en la productividad agronómica con consecuencias negativas en la seguridad alimentaria y la calidad de vida. La degradación del suelo además tiene amplias implicaciones en la diversidad del paisaje y la calidad y cantidad de los servicios ambientales que son proporcionados o regulados por ecosistemas de montaña. En montanas tropicales andinas, el movimiento de masas y la erosión por agua son los procesos dominantes que trasportan el material del suelo pendiente abajo. La erosión hídrica es un proceso importante que lleva al desarrollo extensivo de redes de surcos y cárcavas y a la formación de “badlands”. Este es el caso de muchas áreas que han sido usadas para la producción agrícola. La rápida generación de escorrentía superficial resulta en la formación de cárcavas que posteriormente intensifican la degradación del suelo mediante la destrucción y pérdida de la capa superficial de suelo exponiendo el material parental por la acción del agua.

Existe un interés creciente en la conservación del suelo y en proyectos de restauración en los Andes Tropicales, desde una gran variedad de actores públicos y privados. El reto es desarrollar, esquemas de conservación de agua y suelo económicos y probados que puedan ser implementados

directamente en el campo por autoridades locales, administradores de tierras y agricultores.

En este estudio se mejoró la información actualmente disponible en la efectividad de esquemas de conservación de agua y suelo utilizando un enfoque basada en experimentos de campo. Los resultados están basados en 5 años de campaña de monitoreo que proporcionó información clave, con una resolución temporal muy alta, acerca de los cambios en las propiedades del suelo, erosión de suelo, flujos de agua y sedimentos en sitios experimentales. Los sitios piloto fueron seleccionados en los Andes tropicales y están localizados cerca de la ciudad de Cuenca (Ecuador). Los 5 sitios están situados en la microcuenca Loreto y tienen características biofísicas similares. Dos de estos sitios fueron mantenidos como referencia (en una zona agrícola y otra degrada) mientras que los tres restantes representan diferentes tratamientos de conservación/ restauración (en una zona agrícola, una zona degradada y una zona degradad forestada con *Eucalyptus*).

La premisa básica del enfoque usado consiste en que mucha de la degradación del suelo está localizada en zonas críticas de erosión que constituyen áreas relativamente pequeñas y que presentan tazas de erosión elevadas. Cuando estas zonas críticas se conectan e la red de drenaje, sus efectos se propagan devastando grandes áreas. Para maximizar la efectividad y minimizar costos de esquemas de conservación de agua y suelo, nuestro enfoque consiste en reducir la conectividad física en el paisaje. Por lo tanto nos enfocamos específicamente en zonas críticas de erosión y rutas de escorrentía y transferencia de sedimentos. Se dio especial atención a la recuperación y restauración de la vegetación natural mediante técnicas de bio- ingeniería que han probado ser eficientes y sostenibles en comparación con las clásicas obras civiles.

Los resultados muestran que la vegetación resulta ser una variable clave en el control de la variabilidad del carbono en el suelo. Los valores más altos de carbono se encontraron en suelos bajo bosques de eucalipto y pino. Estos bosques resultan ser eficaces como vegetación protectora para la restauración de suelos degradados.

Las obras de bioingeniería resultaron ser efectivas en la estabilización de cárcavas activas. Se estimó la producción de sedimento causada por cárcavas y la contribución de las técnicas de bio-ingeniería y vegetación en la reducción de la cantidad de sedimento exportado. La construcción de diques en los cauces de cárcavas incrementó la deposición y captura de sedimento. Estas obras resultaron tener una efectividad del 70 % en la reducción de la cantidad de sedimento exportado. Los bosques de *Eucalyptus* constituirían alternativas efectivas para la reducción de la producción de sedimento y aún más importantes para el control de erosión y restauración de suelo en zonas degradadas por erosión por cárcavas.

Los resultados muestran que las técnicas de bioingeniería son efectivas en la reducción de picos en los caudales hasta en un 73% para el caso de los diques y en un 56% debido al efecto del bosque de eucalipto.

Summary

Soil erosion is a serious threat to the long-term viability of mountain agriculture on steep slopes in the Tropical Andes, and its impact on agricultural production is expected to increase in the 21st century with potential impacts on the socio-economic development of rural areas. Soil erosion by water is one of the dominant soil degradation processes, and leads to the development of extensive rill and gully networks and formation of so-called badlands.

Vegetation restoration and regeneration is gaining momentum across the world, also in the Tropical Andes. This dissertation aims to enhance our understanding of the effectiveness of soil and water conservation treatments on soil chemical and physical properties, sediment production and transfer to the fluvial system. The analyses are based on a five-year monitoring campaign that provided key information on changes in soil properties, soil erosion, water and sediment fluxes at very high temporal resolution. Five micro-catchments were equipped as pilot sites: two pilot sites are untreated (control) sites; while conservation measures are implemented in the three remaining sites.

The experimental data show that bioengineering techniques are effective to reduce soil erosion and stabilize active gullies. Non-native eucalyptus trees enhance soil restoration in highly degraded sites, and play an important role in erosion control and sediment yield reduction. Vegetation restoration is shown to increase the soil carbon content, and enhances incipient soil development in highly degraded land. Restoration schemes have a measurable impact on the hydrological response of the micro-catchments, with a 50% reduction in peak flow.

Chapter 1 : Introduction

1.1 Trends in land degradation in the Andes

The tropical mountain regions of the Andes are amongst the most diverse ecosystems in the world (Mulligan et al., 2010). Natural resources, such as soils, minerals and water, are important for human development in the region (Balslev and Borchsenius, 2012). The Tropical Andes has been settled and cultivated for thousands of years, experiencing important environmental changes as consequence of human development – one of the main characteristics of the Anthropocene period (Chin et al., 2016). The Andes is currently home to millions of people, and constitutes the base for the development of several metropolis such as Bogotá, Quito, and La Paz.

The Andean mountain range is characterized by a large natural biological and ecological diversity, which is produced by strong and rapid alternations in topographical, meteorological, and ecological conditions over relatively short distances (Mulligan et al., 2010). They contain unique ecosystems consisting of a wide diversity of natural environments, which range from lowland rainforest to lower mountain rainforest, Andean cloud forest, grassland and shrub vegetation and páramo vegetation at the highest elevations. During the last few decades, the natural ecosystems of the Andean region have increasingly been disturbed by rapid demographic growth and socio-economic development (Hofstede et al., 1998; Vanacker et al., 2003a). Despite the fact that these environments are sources of water, energy and biological diversity that are essential to the survival of a large part of the Andean population, they are endangered by human pressure and natural ecological imbalances caused mainly by land use and climate change (Becker and Bugmann, 2001).

1.2 Soil degradation in the Andes

Soil erosion is an important component of soil degradation in the Andean mountains (Carretier et al. 2013; Harden 2006). On-site, soil erosion results in a loss of soil fertility, a decline of land quality on the eroded lands, and a subsequent reduction in agronomic productivity with negatives consequences on food security and quality of life. Soil erosion is a serious threat to the long-term viability of mountain agriculture on steep slopes in the Andes (Dercon et al., 2007), and its impact on agricultural production is expected to increase in the 21st century with potential impacts on the socio-economic development of the rural areas (Vanegas, 2015). Soil degradation has also wider implications on the diversity of the landscape, and the quality and quantity of environmental services that are provided or regulated by mountain ecosystems (such as water availability, sediment yield, hydrological response, carbon sequestration (Balthazar et al., 2015; Ponette-González et al., 2015).

In the tropical Andean mountains, mass movement and soil erosion by water (Figure 1) are the dominant processes that transport soil material downslope (Guns and Vanacker, 2013; Vanacker and Govers, 2007a). Water erosion is an important process of land degradation (Poesen et al., 2003), and leads to the development of extensive rill and gully networks and the formation of so-called badlands. This is particularly the case in areas that have been or are used for agricultural production (Figure 2). When the protective native vegetation cover is removed, the soil becomes rapidly exposed to the effects of erosive rainfall (Inbar and Llerena, 2000; Molina et al., 2007). The rapid generation of surface runoff results in the formation of gullies that further enhances soil degradation by destroying the soil layer and exposing the parent material by action of water (Harden and Scruggs, 2003). Gully erosion causes considerable soil losses and produces large volumes of sediment (Poesen et al., 2003). The off-site effects of soil erosion in tropical mountains are multifold: increasing costs for hydropower installations due to a reduced water storage capacity (Harden, 1993), pollution of water reservoirs and dams that are used to supply drinking water (Harden, 1993), and siltation of irrigation infrastructure.

Several studies (Harden, 1993; Harden and Mathews, 2002; Podwojewski et al., 2002; Molina et al., 2007) have indicated that soil erosion rates have changed significantly in the recent past increasing sediment transported by rivers. A recent study has shown that the average sediment yield in the northern Andes is about 14.85 t ha^{-1} yr^{-1} (Latrubesse and

Restrepo, 2014). However, large variation in sediment yield has been reported. Sediment yield can be increased up to 100 times if the vegetation cover in the upstream area is reduced (Vanacker et al., 2007b). Changes in land use and land management, modifications of agricultural practices, and climate change affect the magnitude of soil erosion processes (Vanacker et al., 2007b). Land use in the Tropical Andes has experienced remarkable changes during the last four decades, as a result of demographic, socio-economic and political reasons, overgrazing and poor land management (Vanacker and Govers, 2007; Molina et al., 2012; Farley and Kelly, 2004). Rural-urban migration has resulted in a depopulation of the impoverished rural areas, with important implications on land use and land management. In the Magdalena river basin, an increase of 33% in sediment load has been reported (from 550 t km^{-2} y^{-1} to 710 t km^{-2} y^{-1}) for the period 2000 – 2010: about 9 % of the sediment load can be attributed to deforestation (Restrepo et al., 2015).

The effects of land use change and land management on soil erosion processes are complex and dynamic (Molina et al., 2012). In the areas of agricultural expansion, the upward movement of the agricultural frontier results in deforestation of native forest. It is been shown that this increases the frequency of shallow mass movements and the intensity of water erosion (Guns and Vanacker, 2013; Vanacker et al., 2003b) At the same time, rural-urban migration is taking place in the rural areas that are located relatively close to urban centers (Vanegas, 2015). A decrease of the interest in farming activities leads to abandonment of agricultural parcels, regeneration of secondary forest, and afforestation with fast-growing commercial species, such as *Eucalyptus globulus* or *Pinus radiata* (Vanacker et al., 2003). Recent work has shown that vegetation restoration enhances soil infiltration and reduces runoff generation, factors that contribute to decrease erosion rates (Molina et al., 2007).

Figure 1: Photograph illustrating the magnitude of mass movements in the tropical Andes Mountains. After the mass movement, gully erosion actively contributes to the mobilization of loose material on the landslide scar (Photo by Marie Guns, 2013; Llavircay catchment, Southern Ecuadorian Andes).

In recent years, several international initiatives have been addressing sustainable land management and land degradation, such as the United Nations Conventions to Combat Desertification (UNCCD). The impact of global change on land degradation will depend on how processes like physical land degradation and soil erosion are affected by climate change and land use practices (Imeson, 2012). Runoff, as one of the main controlling factors of soil erosion by water, is expected to change in the future because global climate change is predicted to have an impact on rainfall patterns and rainfall erosivity (Huntington, 2006). Erosion and overland flow (runoff) are environmental problems that are likely to be magnified by unsustainable land use, climate variability and change (Kosmas et al., 1997; Yang et al., 2003; Imeson, 2012). However a lot of uncertainty exists about future changes in the torrentiality of rainfall events, particularly in tropical mountain regions (Buytaert et al., 2011; Vuille, 2013). It is therefore fundamental to investigate how soil erosion and surface runoff will be affected by human activities, land use and climate variability.

Figure 2: Photograph illustrating hotspots of water erosion (Photo by Pablo Borja, 2013).

1.3 Conservation and restoration of degraded land

Conservation and restoration of degraded land is increasing worldwide (De Groot et al., 2013). Assisted vegetation restoration and unassisted vegetation regeneration is gaining momentum across the world (Chazdon, 2008), and also in the Tropical Andes (Dehn and May 2008; Knoke et al. 2014). Restoration of the protective vegetation cover is an essential component of conservation and restoration projects. With an adequate vegetation cover, the soil surface is better protected against soil erosion and degradation,(Bochet et al., 1998) and the soil forming processes can help to recover the soil functionality (Moreno-de las Heras, 2009).

Recent studies have shown that the restoration of the vegetation cover on degraded hillslopes has a positive effect on erosion reduction (Aerts et al., 2006; Poesen et al., 1990). When augmenting the protective vegetation cover of the surface, the soil is less exposed to rain drop impact and surface runoff (Cammeraat, 2004). Decomposition of organic material leads to an increased input of organic C and N in the soil (De Baets et al., 2012), and the soil system can start to recover its functionality (Yüksek and Yüksek, 2011). Additionally, the vegetative cover reinforces the root matrix within the soil mantle such that the soil cohesion is enhanced (De Baets et al., 2007). The presence of vegetation on the ground reduces the generation of surface runoff, and augments the surface roughness reducing thereby the erosive capacity of surface flow (Govers et al., 1990).

In watershed or catchment restoration projects, assisted vegetation restoration and unassisted vegetation regeneration are frequently accompanied by stream and/or gully restoration. The latter involve stabilization of river banks and/or active gully channels and construction of check dams to control the water velocity and channel erosion (Boix-Fayos et al., 2008; Molina et al. 2009; Xiang-zhou et al., 2004). While vegetation restoration on the hillslopes aims to reduce soil erosion and surface runoff by restoring the soil functions on-site, stream and/or gully restoration aims to reduce transfer of surface runoff and sediments downstream. Integrated restoration projects involving restoration and regeneration of vegetation on the hill slopes and restoration of the stream and/or gully channels can be very effective to improve the hydrological regulation of the catchments (Conesa-García and García-Lorenzo, 2009).

1.4 Geotechnical and bioengineering techniques

Traditional conservation and restoration techniques are often based on so-called "hard" engineering techniques, such as the placement of gabions or rock dams in stream channels and/or gullies (Conesa-García and García-Lorenzo, 2009). Geotechnical engineering projects have been questioned for their sustainability in the long term: they involve high initial economic costs and require regular – and costly - maintenance (Morgan and Rickson, 1994) and are often not well integrated in the landscape. As such, geotechnical engineering projects are often considered as net-cost projects by national, regional or local governments, and require long-term financing, political will, labor, and institutional commitment (Chazdon, 2008; De Groot et al., 2013).

In recent years, low-cost geotechnical corrections that involve biotechnical and soil engineering treatments have been proposed as a viable alternative. Bioengineering techniques are vegetative-based treatments for soil erosion protection and slope stabilization (Morgan and Rickson, 1994). They are based on the use of living plant materials to construct soil bioengineering structures and perform some engineering function (Rey et al., 2005), and differ from biotechnical engineering that combines the use of plant living materials with inert structures (Morgan and Rickson, 1994). In small-scale restoration projects, the so-called 'soft' or bioengineering techniques can provide possibilities to complement, improve or in some cases even replace the traditional geotechnical engineering approaches (Rey, 2004; Rey, 2003). Biotechnical and soil engineering treatments are considered to be more cost-efficient than traditional geotechnical engineering approaches, with a better balance of implementation costs, resource use and results (Gray and Sotir, 1996).

Bioengineering techniques pursue technological, ecological, economic as well as design goals and seek to achieve these primarily by making use of living materials, i.e. seeds, plants, part of plants and plant communities, and employing them in near–natural constructions while exploiting the manifold abilities inherent in plants (Maestre and Cortina, 2004; Gray and

Sotir, 1996). Pioneering species such as those that are found on naturally disturbed land are preferentially used in bioengineering techniques. As Gray and Sotir (1996) indicated in their reference work on soil bioengineering techniques, “their ability to root from cuttings, their continued growth following burial and the ability to grow under harsh conditions serve to make these plants useful” for restoration.

Whatever the type of conservation of restoration activity, it remains essential to evaluate critically the benefits of conservation and restoration plans on the functioning of the restored ecosystem. This requires the setup of a monitoring program that collects both before and after the implementation of the restoration plan at reference and treatment sites critical data on the soil-vegetation system and the hydrological response. In addition, the monitoring period needs to be long enough to cover the potential restoration effects.

1.5 Examples of restoration projects

There are plenty of examples from gray and published literature documenting the implementation of bioengineering restoration projects. A multitude of projects mentions the importance of stakeholder participation to guarantee sustainability of conservation and restoration programmes (Alemayehu et al., 2009), and the need to account for socio-economic development of the region (Balana et al., 2012). Below, some examples are given to illustrate the potential of bioengineering projects for soil restoration.

In Ethiopia, integrated landscape restoration projects were implemented in the Tigray region with the aim to improve water harvesting and secure food production. Besides the widespread implementation of terrace agriculture, the construction of check dams in stream and gully channels and the establishment of "exclosures" are an important part of the soil and water conservation programme. The term "exclosures" refers to areas that were set aside and where agriculture and grazing became forbidden to allow natural regeneration of vegetation (Descheemaeker et al., 2006b). An evaluation of the effectiveness of the soil and water conservation techniques showed that the implementation of conservation measures reduced significantly runoff and soil loss in treated land compared to untreated land (Taye et al., 2015). Balana et al. (2012) evaluated the cost – benefit of the conservation measures, and concluded that the landscape restoration programme in Tigray generated a large positive net present value, with increases in crop productivity and food security for the farmers' communities in Ethiopia (Tesfaye et al. 2016).

In the Southern French Alps, bioengineering techniques were applied to control high sediment yields from marly catchments. The ecological engineering approach that was used by Rey and Burylo (2014) consisted in implanting vegetation barriers (made of *Salix purpurea* cuttings) perpendicularly to the flow to trap sediment in gully floors. Rey (2004) have shown that vegetation barriers are very effective in trapping eroded sediments, and reducing environmental degradation. Vegetation barriers with only 20% vegetation cover can be sufficient to trap all sediment eroded from a plot of about 500 m^2 and barriers with 33% vegetation cover to trap sediment from gullies draining a catchment area of < 1 ha (Rey, 2004a). Besides, an increase in biodiversity and ecosystem service measures was reported after restoration (Benayas et al., 2009).

Vegetation barrier also are used to trap sediment with tracers like Cd and Zn in polluted soils. In a flume experiment in Belgium retention efficiencies of 46 to 56 % were reported using different species (Lambrechts et al., 2014).

1.6 Thesis objectives and outline

There is an increasing interest in soil conservation and restoration projects in the Tropical Andes, from a wide variety of public and private actors. *This dissertation aims to enhance our understanding of the effectiveness of bioengineering solutions for soil erosion and runoff control by an in-depth analysis of original data from an experimental setup in Southern Ecuador.*

The *main objectives of the thesis* are to evaluate quantitatively (i) the effect of vegetation conservation and restoration on physical and chemical soil properties, (ii) the combined effects of vegetation establishment and gully restoration on sediment mobilization and transfer in badlands, and

(iii) the hydrological response of pilot catchments with different conservation and restoration treatments. The outcome of this research will contribute to establish robust and technically-proven soil and water conservation schemes that can directly be implemented in the field by local authorities, land managers and farmers.

The analyses are based on a five-year monitoring campaign that provided the key information on changes in soil properties, soil erosion, water and sediment fluxes at very high temporal resolution. Pilot sites are selected in the Tropical Andes, and are located close to the provincial town of Cuenca (Ecuador). Five micro-catchments are selected as the pilot sites, and are all located in the headwaters of the Loreto River, a tributary of the Jadan River (Figure 4). Two of the five pilot sites are untreated sites, and serve as control sites. In the three remaining sites, different soil and water conservation measures were implemented using bioengineering approaches.

The basic premise of the approach that will be used here is that much of the land degradation is localized in so-called erosion hot spots, relatively small areas that exhibit anomalously high erosion rates. When these erosion hotspots become connected to the stream network, their effects spread outwards to devastate larger areas. To maximize effectiveness and minimize costs of soil and water conservation schemes, our approach consists in reducing the physical connectivity in the landscape. Therefore, we specifically target our schemes at erosion hotspots and pathways of runoff and sediment transfer. Particular attention will be given to recovery and restoration of natural vegetation as bio-engineering techniques are proven to be particularly efficient, and more sustainable than hard civil engineering works.

This *thesis is divided in **five major parts***, according to its main objectives (Figure 3).

The **first Chapter** gives an overview of the **research context** - including a brief review on soil degradation in the Tropical Andes – and defines the research questions that are addressed in this PhD thesis. This chapter also contains a detailed description of the study area based on a literature review, and the experimental setup that was implemented in the pilot sites in the High Ecuadorian Andes. The chapter concludes with a presentation of the metadata that were collected in the scope of this PhD research project.

The **second Chapter** evaluates the **impact of soil bioengineering techniques on key soil properties**. In this part, we are particularly interested in the effect of unassisted vegetation regeneration and vegetation restoration on soil physical and chemical properties on short time scales (months to few years). A total of 64 sites were sampled in 2012, 2013 and 2014, and the soil surface samples were analyzed at the soil laboratory of the University of Cuenca (Ecuador) and the laboratory of ELIC (Belgium).

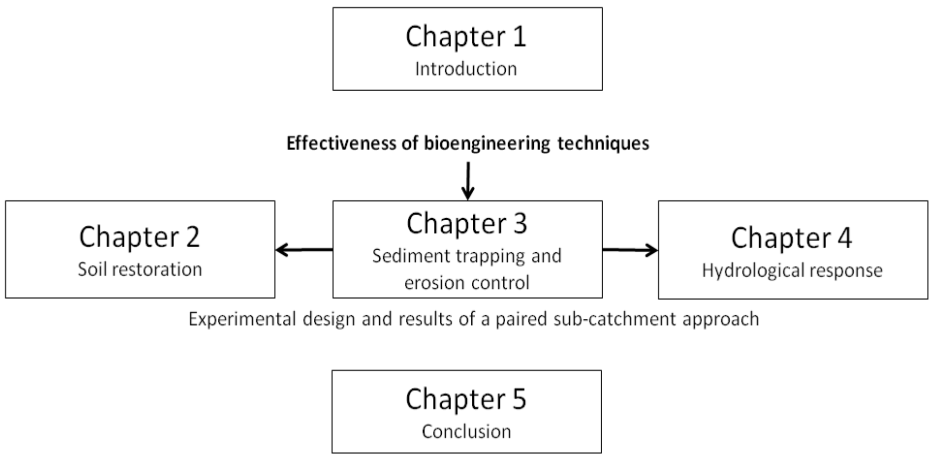

Figure 3: Outline of the thesis.

The **third Chapter** aims to evaluate the **effectiveness of bioengineering measures on sediment production, transfer and deposition** in the five pilot sites that were monitored during this PhD project. The five micro-catchments differ in vegetation cover and implementation of bioengineering techniques (construction of dams), which allows us to quantify the combined effects of vegetation and gully restoration (bioengineering techniques) on sediment production and transport in highly degraded lands.

The **fourth Chapter** investigates the **effectiveness of catchment restoration on the hydrological response of micro-catchments**. Rainfall-runoff mechanisms (generation of surface runoff in response to rainfall with different intensities) are analyzed at the level of 15 individual events, during a period of 41 months. Based on the runoff hydrographs, the event runoff dynamics were analyzed for the five micro-catchments having different conservation/restoration treatments. A statistical analyses of the peak discharge, the event-based runoff coefficient, the time to peak, and the slope of the recession limb allowed us to characterize the hydrological effects of catchment restoration.

Finally, the main findings are summarized in **Chapter 5**. This chapter ends with an outlook for future research.

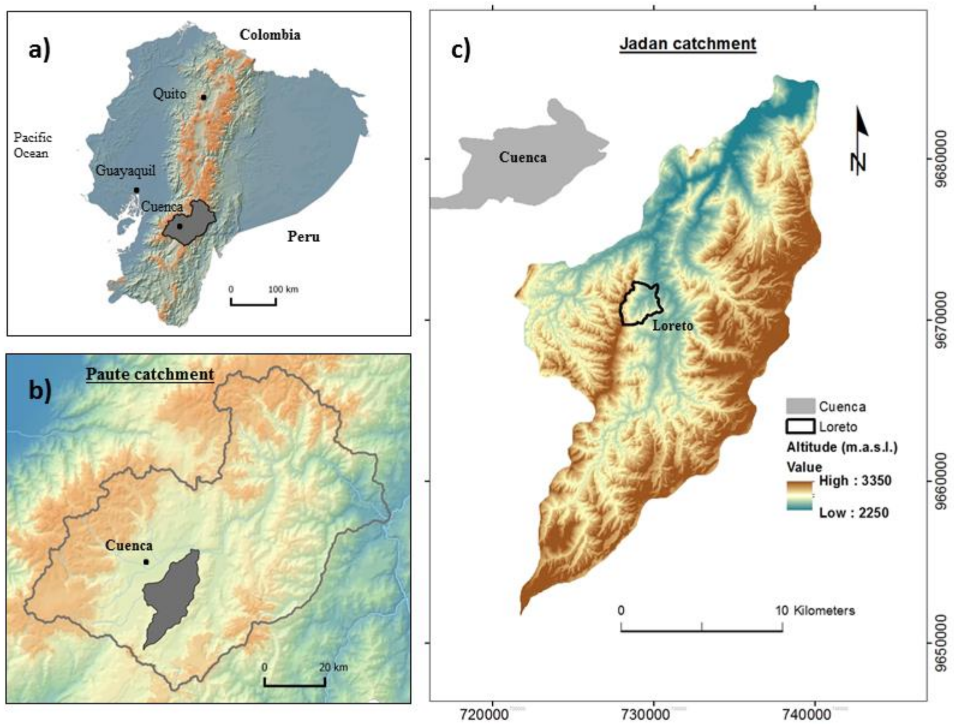

Figure 4: Location of the study sites. (a) Location of the Paute basin (gray polygon) in the Southern Ecuadorian Andes, (b) Location of the Jadan catchment (gray polygon) within the central part of the Paute basin, and (c) Situation of the Loreto catchment (black polygon) within the Jadan catchment.

1.7 Paute catchment in Southern Ecuador

1.7.1 Land use dynamics in Paute catchment

In the Paute catchment, the topography is very irregular and mountainous (Figure 4): the highest peaks are situated at 4200 m a.s.l. and the lowest parts at 1600 m a.s.l. The mean annual temperature is 20 °C, but shows large spatial and temporal variation. The annual mean precipitation is 2000 mm, but equally varies strongly as function of altitude and orientation.

Land use is highly dynamic and changes rapidly in response to demographic growth, political decisions, social inequalities and short-term economic pressures (Vanacker et al., 2007b). From the 1960s onwards, there is a rapid expansion of the urban areas, intensification of the agricultural land use, fragmentation of the agricultural lands and strong upward movement of the agricultural frontier into the primary and secondary native forests and bushlands (Hofstede et al., 1998; Vanacker et al., 2003b; Wunder, 1996). More than half of the primary native forest is converted to agricultural land or replaced by secondary woody vegetation or more recently by exotic species (Wunder, 1996). Clearings and settlements have affected particularly the central part of the Paute catchment around the historic town of Cuenca (Figure 4). Remote parts of the catchment were largely left untouched and forested (Figure 5), but disturbance is now increasing rapidly as construction and utility roads around the Paute hydroelectric project have facilitated migration (Jokisch, 2002).

Figure 5: Panoramic view on the study area, located in the southern Ecuadorian Andes. The two main sources of sediment in the area are shown in the pictures below. Lower left: landslide activity in overgrazed pastures, and lower right: active gully erosion in badlands.

During the last decades, exotic tree plantations have been promoted as part of institutional and non-governmental projects to increase the wood production, conserve land and enhance carbon fixation. In the Paute catchment, the plantations of Eucalyptus and Pine trees are mainly concentrated in the central part of the basin. This change in forest cover dynamics has been documented for several regions in Ecuador, and consists in a deceleration of deforestation and increase of exotic tree plantations (Balthazar et al., 2015; Molina et al., 2012; Farley, 2007).

The landscape of the central part of the Paute catchment can be characterized as a patchwork of small agricultural fields, secondary forest, tree plantations, urban areas and infrastructure. Traditional and subsistence farming systems are dominant: they are typically mixed farms with cultivation of different crops on the same plot (association of corn-beans) and livestock farming with animals that are mainly fed on pasture and crop residues. Milk is the main product that constitutes a source of income for the farmers. Due to rapid demographic growth and severe land degradation, both the quality and quantity of land available is rapidly decreasing.

Access to land is highly unequal. Over half of the landowners have less than 1 ha of land, often located on steep slopes so that cultivation occurs on slope gradients up to 70% (INEC, 1991). These small farms or "minifundios" are further divided into even smaller landholdings. Overgrazing, and intensive cultivation of these poor soils has led to severe land degradation, low crop yields, and widespread poverty amongst the rural population. These harsh natural and socio-economic living conditions are an important driver of massive internal and external migration. As a result, the agricultural frontier is expanding and native forests are cleared for new agricultural land. Besides, rural people started to migrate to urban centers and foreign countries. The provinces of Azuay and Cañar in the southern Ecuadorian Andes are experiencing high levels of transnational migration. Although hard numbers are difficult to obtain, estimations are that over a few hundreds of thousands of inhabitants have left these two provinces in the last decades (Vanegas, 2015; Jokisch, 2002; Jokisch et al., 2010). Low agricultural production levels on degraded soils, and the lack of local livelihood alternatives are the main reasons for migration and rural decline (Vanegas, 2015; Jokisch et al., 2010).

1.7.2 Soil erosion in Paute catchment

The steep topography, erosive climate and highly erodible bedrock are among the principal natural causes for soil erosion in the Ecuadorian Andes (Vanacker et al., 2015). Human-induced land use has accelerated soil erosion, and soil losses on degraded land may exceed the rates of soil formation by as much as two orders of magnitude (Vanacker et al., 2007a). Sediment produced by erosion processes leads to high sedimentation rates in irrigation channels, reservoirs and dams, and hampers the production of surface drinking water. Drinking water companies face severe problems in the rainy season, and water intake for drinking water production has become impossible during flood events.

Since the early 1980s, the Ecuadorian authorities have considered the Paute drainage basin as the target area to combat soil erosion because of the presence of the largest national hydropower complex in the downstream part of the Paute catchment: the hydro-electric complex “Complejo Hidroelectrico Paute Integral” consisting of the Paute Mazar (170 MW), Paute Molino (1100MW), Paute Sopladora (in construction, 487 MW) and Paute Cardenillo (in plans, 596 MW) plants can generate 2353 MW, and currently foresees in about 40 to 50 % of the national electric energy demand (Corporacion Electrica del Ecuador. Unidad de Negocio, 2016). Two large reservoirs of 410 Hm^3 and 120 Hm^3 are feeding the hydropower complex, where respectively the Mazar and Daniel Palacios dam are obstructing the main channel of the Paute River. Sedimentation in the reservoirs represent a serious problem for the Paute hydroelectric project (Jerves, 2001). The costs of dredging sediment from the reservoirs (estimated at 3 $/$m^3$) as well as reparations of infrastructure damaged by flooding are considerable for the economy of the country (Jerves, 2001).

Because of the importance of the Paute-Mazar hydroelectric project, the interests in soil and water conservation extend largely beyond local stakeholders and authorities. The country is struggling to meet growing energy demands, particularly during prolonged droughts (Buytaert et al., 2006; Buytaert et al., 2011). Climate change may even further amplify this issue (Celleri et al., 2007; Iñiguez et al., 2015).

Erosion remediation schemes have been implemented by local authorities such as UMACPA[1], CGPaute[2] and various NGOs, and mainly focused at water erosion control on agricultural fields. The lack of direct involvement of local stakeholders and farmers' communities in the planning, implementation and maintenance of the remediation schemes is likely to affect their sustainability. Not only are these agricultural development programs and extension services struggling with low farmers' adoption levels, but also with low impacts on the overall quality and quantity of surface water as agricultural land is not a major source of surface runoff water and sediment production following recent investigations by Molina et al. (2007; 2008). It can now be stated with certainty that sediment production is localized in erosion hot spots, highly degraded abandoned areas and localized badlands, which represent a real financial loss for the landowners driving them to poverty.

There is a clear need for an integrated approach to soil and water conservation, that is sustainable and fits with local natural and socio-economic settings. Focusing soil and water conservation schemes at hotspots of erosion, instead of large extensive SWC schemes, allows maximizing effectiveness and minimizing costs and impacts for local communities. In scope of increasing water resource constraints and

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[1] Unidad de Manejo para la Cuenca del Rio Paute
[2] Consejo de Gestion de la Cuenca del Rio Paute

competition amongst public and private stakeholders in the Paute basin, the collection of ‘hard’ scientific data on the effectiveness of soil and water conservation schemes is critical to establish a baseline on which to negotiate compensation mechanisms for environmental services (Kuhlman et al., 2010).

1.7.3 Selection of the pilot sites

The PhD project is part of a five-year research for development project (2009-2014) on “*Strengthening the scientific and technological capacities to implement spatially integrated land and water management schemes adapted to local socio-economic, cultural and physical settings*” that was funded by the Commission Universitaire pour le Développement (CUD, now ARES). This project was a collaboration between the Universidad de Cuenca (Ecuador), Université Catholique de Louvain (Belgium), Université de Namur (Belgium), National Secretary of Water (SENAGUA) Municipal Sanitation Company of Cuenca (EMAC) and the Public Water Company of the Municipality of Cuenca (ETAPA EP).

This dissertation is based on field data that were collected in the Loreto catchment, one of the three catchments that were studied in the scope of the CUD project. The site was selected in agreement with local stakeholders and rural communities, and the selection is partly determined by the presence of a well-structured local ‘Water Board’, which facilitated active involvement of local communities in the research and extension activities and guaranteed the sustainability of the project. The local ‘Water Boards’ are supported by SENAGUA (former Comision de

Gestion de la Cuenca del Rio Paute, CGPaute) for different small projects and micro-credits, and they are legal organizations of local farmers (usually 20 to 30 members) that meet regularly to discuss local problems related to agricultural production, water (supply, irrigation) etc. In the CUD project, we worked directly with these local ‘Water Boards’ which allowed us to structure our participative approach.

The Loreto catchment (Figure 4) is located in the central part of the Paute catchment, close to the historical town of Cuenca. Figure 6 documents the changes in land cover in the catchment over the period 1963-2009. In the beginning of the 1960s, the forest cover in the catchment is very low: cropland and grassland are the dominant land cover, barren lands are widespread in the Eastern part of the catchment, secondary forests can be found at the watershed divide and small patches of Eucalyptus forest are present along the stream channels. Over time, there is an increase in the forest cover with an important extension of shrubs and scrubs as a result of natural regeneration, and strong expansion of Eucalyptus and Pine forests on the hillslopes. Overgrazing, and intensive cultivation of poor soils has led to severe land degradation, land abandonment and widespread poverty amongst the rural population. The rural villages of Jadan and Santa Ana are experiencing high levels of transnational migration.

Five experimental sites were selected and fully equipped in the Loreto catchment (Figure 7). The Empresa Municipal de Aseo de Cuenca (EMAC) facilitated access to the sites.

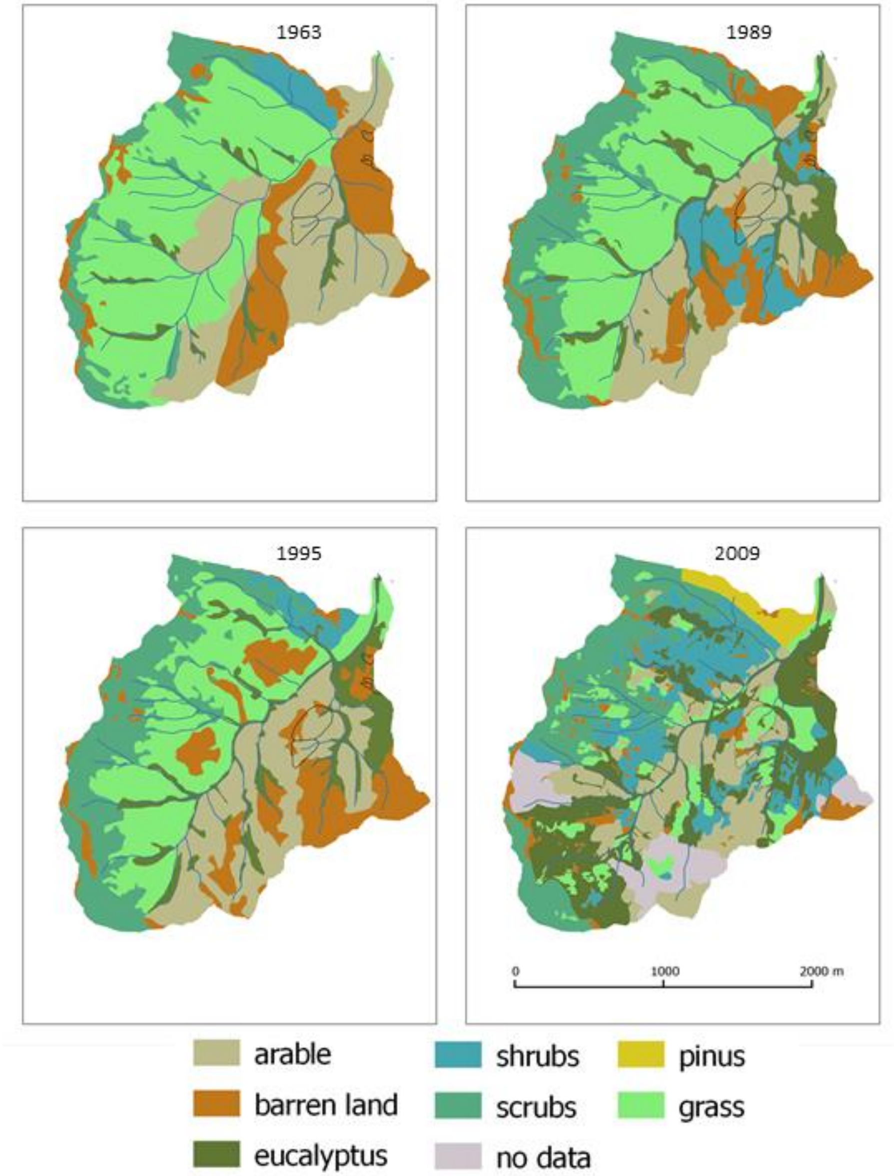

Figure 6: Land cover map for Loreto catchment for 1963, 1989, 1995 and 2009 based on the interpretation of aerial photographs.

1.7.4 Experimental setup

This research project involves the implementation, evaluation and extension of integrated soil and water conservation (SWC) schemes in five pilot sites. To maximize the effectiveness and minimize the costs and impacts for local communities, our approach accounted for recent findings with respect to water and sediment dynamics in Andean environments. Recent work has shown that the primary sources of sediment are not agricultural land: far more sediment is produced in badlands, on earth roads and footpaths (Molina et al., 2007; Vanacker et al., 2007). Therefore, we specifically targeted our soil and water conservation schemes at erosion hotspots and pathways of runoff and sediment transfer. Particular attention was given to use soil bioengineering techniques based on assisted vegetation restoration and unassisted vegetation regeneration, as an alternative to geotechnical engineering approaches that are traditionally implemented in the region.

The erosion remediation scheme was optimized by focusing on hillslopes and active gully systems. Gully systems play a pivotal role in the hydrological response of degraded catchments, as they concentrate the surface runoff generated at the hill slopes, and transport it to the river network. Given the fact that most erosion mitigation programs rarely incorporated any measures in gullied land, there is a large potential for improving the efficiency and effectiveness of current erosion remediation programs by small, but well-focused revegetation actions. As gullied land represents only a relatively small part of the total catchment area, restoration of active gullies by revegetating their gully bed can be a viable and cost-efficient means to combat the high sediment transfer to the fluvial system.

Within the Loreto catchment, five experimental sites (corresponding to hydrological units or micro-catchments) are selected with similar topographic, climatic, and geological setting. Two experimental sites are located in agricultural land, and three sites in highly degraded lands or so-called badlands. Soil and water conservation (SWC) measures and bioengineering techniques were applied in three of the five micro-catchments (DI, AI, DF), the so-called "treated" micro-catchments (Table 1). Two micro-catchments (DT, AT) are not treated, respectively in the agricultural and degraded land, and serve as 'reference' sites. In the text, we refer to the micro-catchments in agricultural land as "AT" (non-treated site) and "AI" (treated site), and to the ones in degraded land or badlands as "DT" (non-treated site), "DI" (treated site) and "DF" (treated site with reforestation) (Figure 7).

Table 1: Major characteristics of the experimental micro-catchments.

micro-catchment	Surface (ha)	Avg. Alt. (m)	Land use	Condition
			vegetation type	
DT	0.195	2698	Shrubs, barren	gullies
DI	0.415	2704	Shrubs, barren, *Eucalyptus*	gullies with wooden dams
DF	0.415	2657	*Eucalyptus*, Shrubs	gullies with *Eucalyptus*
AI	2.410	2706	Arable, grass, barren	arable land with wooden dams
AT	4.711	2684	Arable, grass, barren, *Eucalyptus*, shrubs	arable land

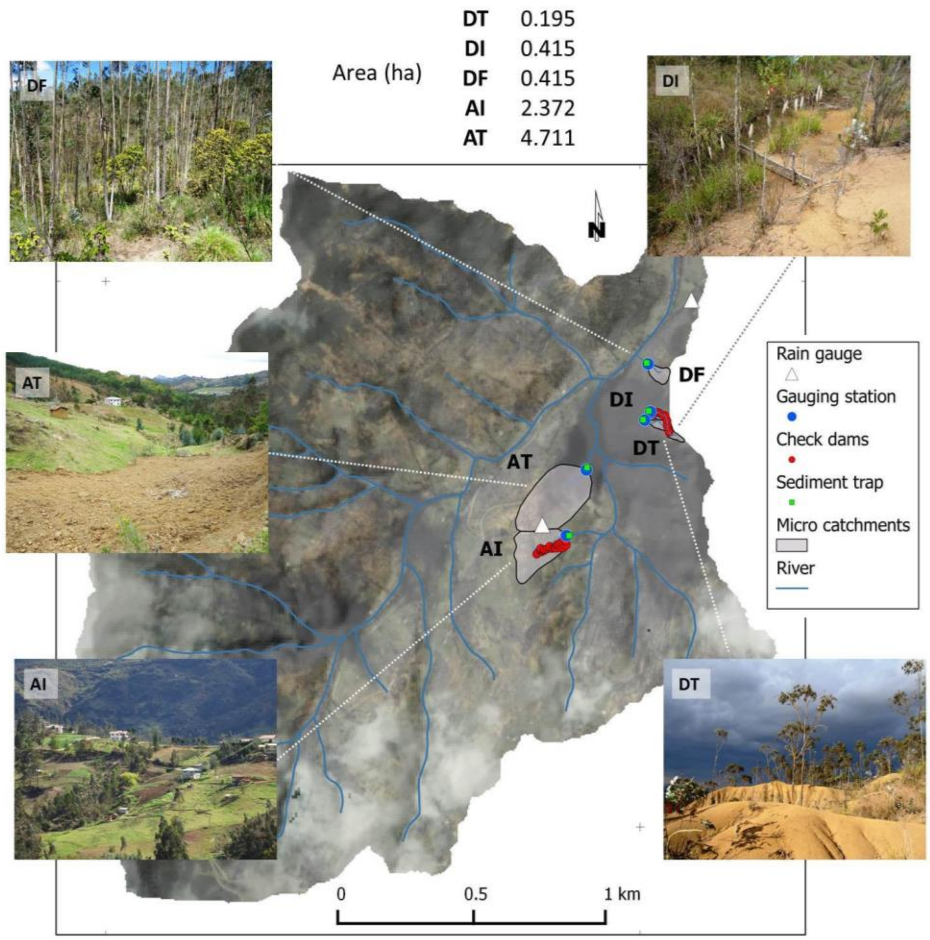

Figure 7: Experimental setup within the Loreto catchment. Three experimental sites are located in degraded land: DI – where SWC were implemented, DF – where Eucalyptus forest was planted in the 1960s, DT – reference site. Two experimental sites are located in agricultural land: AI – where SWC were implemented and AT – reference site.

Figure 7 shows the experimental design of the five micro-catchments. Aerial photographs were used to verify the land use history of the micro-catchments during the period 1963 till now (Figure 6). In the micro-catchments where the erosion measures were implemented, we selected areas where a combination of different SWC techniques could be evaluated including bioengineering techniques on the hill slopes and in the gully channels. It is important to note that the two micro-catchments in the agricultural land (AI, AT) and the three micro-catchments in the degraded land (DI, DF, and DT) had a very similar vegetation cover in the 1960s (Figure 6). The differences in soil properties and catchment response that are observed today are a result of the soil and water conservation schemes that were implemented since.

Details of soil and water conservation techniques

The *DT* and *AT* micro-catchments were not treated, and serve as reference sites in this study to evaluate the effectiveness of the erosion remediation program in respectively degraded or barren land (DT) and agricultural land (AT).

In the *DI* and *AI* micro-catchments, bioengineering techniques were implemented in the gully channels. They consist of vegetative barriers or so-called *dams* that are constructed perpendicular to the flow direction of the ephemeral streams. A total of 45 check dams were installed in 2011 and 2012 (Figure 8:). Check dams consisted of barriers of wooden poles, and barriers of recycled tires filled with soil and seedlings. In addition, assisted vegetation restoration and unassisted vegetation regeneration projects were implemented on the gully slopes in the mid-1980s and lasted till now. These interventions are mainly visible in DF, AI, AT and DI. The *Eucalyptus* forest in DI has not the same development than in DF. The forest is located at the extreme upper part of the micro-catchment, and corresponds to a recent plantation with low tree density.

The *DF* micro-catchment is covered with Eucalyptus forest. The Eucalyptus forest was planted in the 1980s (Figure 6), and has reached maturity. The Eucalyptus trees were planted at very low density, which permitted herbaceous and shrub undercover through natural regeneration of vegetation.

Figure 8: Construction of check dams in the active gully channels of the micro-catchments AI and DI (photo: Pablo Borja, 2012).

1.7.5 Monitoring program in experimental sites

The five experimental catchments were monitored in great detail during the 41 months in the period 2010-2014. Topographical and vegetation maps at 1/10.000 were constructed using GPS and topographic instruments.

Soil properties were studied by sampling 64 sites in total. The physical and chemical properties of top soil material (0 – 25 cm) were analyzed in the soil laboratory of the Universidad de Cuenca and the Earth and Life Institute (UCLouvain). The soil sampling involved agricultural lands (15 samples), forested lands (9 samples) and degraded lands (40 samples). The samples cover all 5 micro-catchments. The samples were analysed per category of water erosion.

Hydro-meteorological equipment was installed in 2011. Figure 9 gives an overview of the sensors that were installed and registered data on precipitation and water flow for the five micro-catchments over the time period January 2011- August 2014.

Two pluviometers were installed in the Loreto catchment. The two pluviometers (tipping bucket: P-Lo1 and P-Lo2 with resolution of 0.2 mm) recorded rainfall amounts and intensities. One pluviometer (P-Lo1) was installed nearby the three degraded micro-catchments (DT, DI and DF), and a second one (P-Lo2) nearby the agricultural micro-catchments (AI and AT). The two rain gauges are located at similar altitude and relatively close to each other, and there is a very good correlation between the precipitation data from both rain gauges. The P-Lo2 rain gauge shows major gaps in the data series in May – June 2011 and June – September 2014. Therefore, it was decided to construct one continuous time series for the Loreto catchment based on the time series of rainfall data from P-Lo1. The data from P-Lo2 was used to fill smaller data gaps of the P-Lo1 series and to verify the correct functioning of P-Lo1. The reconstructed time series has a gap of 4 days during the period 22/03/2014 - 26/03/2014 and 28 days during the period 17/07/2014 - 14/08/2014. The rainfall graph (mm/day) is shown in grey (Figure 9), and shows the large variation in precipitation between the years. The maximum 24h precipitation that was registered is 55.6 mm.

At the outlet of all five micro-catchments, a *San Dimas hydraulic weir* was installed. Water level recorders (*5 Schlumberger mini-divers and 2 baro-divers*) were installed to measure the water heights in the flume during the runoff events. The micro-catchments in the highly degraded land (DT, DI and DF) were equipped in spring 2011, and the measuring devices were installed in the hydraulic weirs in March 2011. Two larger data gaps are present, after vandalism and robbery of the equipment (December 2011, March 2012). The micro-catchments in the agricultural sites were equipped with a San Dimas flume in September 2011, and the pressure sensors started to record data from October 2011 till May 2014 in the two agricultural catchments. During 40 days (28/06/2012 to 7/08/2012) of the dry period all water level sensors were retired to permit maintenance of the hydraulic structures. Because of technical issues with the water height records in one of the micro-catchments, we made replicate measurements of the water heights during a period of 98 days to cross check the observations.

The divers recorded pressure with a resolution of 1 minute (5 minutes resolution during 24/08/2011 – 31/01/2013). Data on pressure levels from the mini-divers were corrected for barometric pressure and transformed to water height. The water heights were then converted into flow discharge using the water level-discharge relationships that were established for each one of the five San Dimas hydraulic weirs.

About 2 m downstream of the hydraulic weir, a sediment trap of 0.5 m^3 was constructed in the gully channel. As the velocity of the flow drops sharply when the runoff water enters the sediment trap, the majority of the sediment load is deposited in the trap. The volume of sediment that accumulated in the traps was measured on a regular basis (i.e. weekly during the rainy season) to have first estimates of the sediment export. However, we cannot exclude that during high intensity rainfall events, when the total runoff volume is much larger than the volume of water that can be stored in the trap, the reduction of the flow discharge is not sufficient to trap efficiently the fine fraction of the sediment load. Therefore, the sediment traps allow to have a first estimate of sediment export, that represent minimum sediment export rates during extreme events.

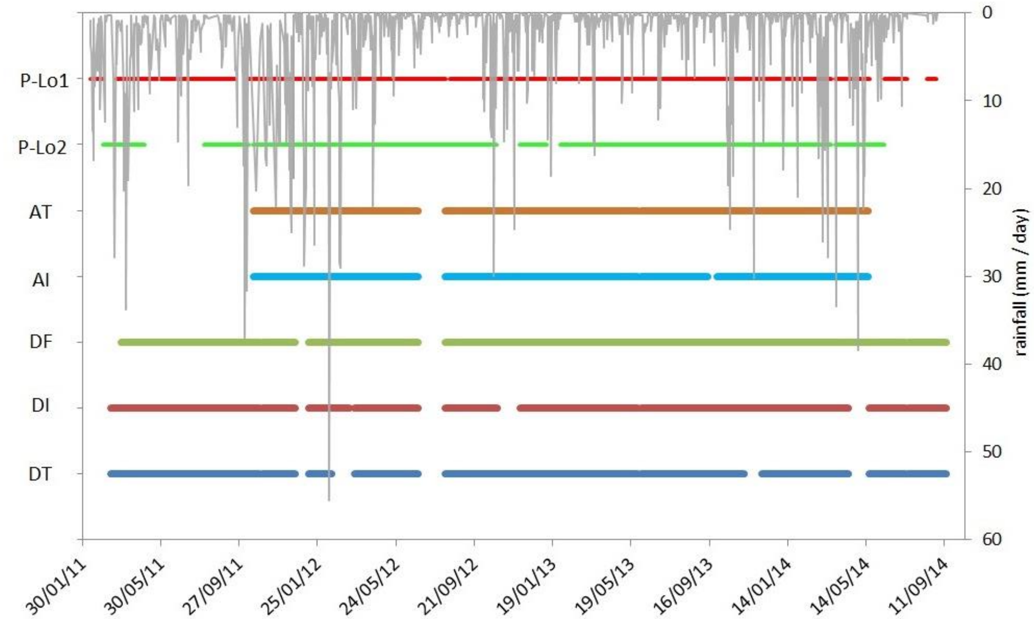

Figure 9: Two raingauges (P-Lo1 and P-Lo2), and five pressure sensors were installed in the experimental sites (AT, AI, DF, DI, DT). The pressure sensors are "divers" to estimate water level and convert it in water flow. The graph shows the time period during which each of the sensors has been collected information. Also, the daily rainfall (grey line) is plotted on the second Y-axis for reference.

Chapter 2 : The potential of soil bioengineering techniques to restore soil quality

Abstract

Badlands are characterized by intense water erosion, and active rill and gully processes can be observed. To combat erosion, soil and water conservation measures can be implemented that aim to reduce water and sediment connectivity in the gully channels and to enhance accumulation of eroded material behind check dams.

This chapter aims to evaluate the effect of soil bioengineering techniques on physical and chemical soil properties in highly degraded catchments. Other aspects that may affect soil dynamics, such as slope morphology, land cover and vegetation density, were equally taken into account. Soil samples were taken in sites with different vegetation cover, and physiographic position. The following physical and chemical parameters were measured: volumetric water content at saturation (θ_{sat}), bulk density (BD), penetration resistance (PR), pH, carbon (C) and nitrogen (N) content.

Based on a physical and chemical characterization of 64 samples, significant differences in soil quality were observed between sites. Vegetation cover (%) appears to exert a strong control on the C content in the mineral soils. The highest C values are found in soils of forest

plantations with *Eucalyptus* and *Pinus*. These plantations are located in areas that were previously affected by active gullying, which confirms that the establishment of a protective vegetation cover is an important factor in soil restoration. The saturation water content (θ_{sat}) does not vary significantly between different sites, or land use types, and mainly depends on the presence of macropores in these soils.

2.1 Introduction

Rill and gully erosion are the most widespread degradation processes that affect a wide range of environments and climates in the world (Jean Poesen et al., 2003). Tropical mountain regions are particularly vulnerable to water erosion, because of their shallow soils, steep topography and high intensity rainfall events (Vanacker et al., 2003; Harden, 2001). The direct consequence of water erosion is the removal of the upper part of the soil horizons that can result in exposure of parent material. Furthermore, soil loss reduces the agricultural productivity, and the eroded material leads to increased sediment yields in the river system, and rapid siltation of reservoirs for e.g. irrigation or hydropower plants (Nyssen et al., 2008; Verstraeten et al., 2006).

Various studies have shown that the vegetation cover is one of the most important factors in erosion control (Gyssels et al., 2005; Thornes, 1990). Maintaining land densely vegetated is an effective way to protect the ground surface for raindrop action allowing water infiltration and reducing surface runoff (Bruijnzeel, 2004). Vegetation cover prevents surface erosion and mobilization of eroded material downslope (Bellin et al., 2011; Molina et al., 2009; Rey et al., 2004; Sanders 1988; Stokes et al., 2008; Vanacker et al., 2003). The removal of the protective vegetation cover can accelerate water erosion rates up to 100 times (Vanacker et al., 2007).

Catchment restoration by bioengineering techniques has a large potential to reduce soil erosion and sediment yield (Rey and Burylo, 2014; Molina et al., 2008). An effective strategy to reduce sediment yield in badlands is the implementation of check dams for gully restoration. To stabilize the gully channel, a sequence of small dams are typically constructed in the gully floor as to reduce the flow of water and sediment (Rey, 2009; Rey et al., 2004). Small bioengineering techniques such as wooden dams, are a sustainable and economical alternative for large dam impoundments (Rey, 2009). For detailed information about gully restoration and the use of different bioengineering structures, we refer here to (Rey, 2009).

In addition to physical bioengineering structures, vegetation restoration in badlands can have positive effects for ecosystem restoration including a reduction of water erosion rates and sediment transport, and regulation of surface hydrology (Burylo et al., 2007; Chirino et al., 2006; Cohen and Rey, 2005; Erktan and Rey, 2013; Marden et al., 2014; Rey, 2003; Rey et al., 2005; Rey et al., 2004; Rey and Burylo, 2014; Zheng, 2006; Rey, 2009). (Rey, 2004; Rey, 2003) showed that the development of vegetation barriers in gully floors has more influence on sediment yield reduction than vegetation restoration on the hillslopes of the catchment. Results show that gully bed deposition in response to revegetation of the channel floor can represent more than 25 per cent of the volume of sediment generated within the catchment (Molina et al., 2009). Nevertheless, little is known about the role of gully bed vegetation compared to other factors such as soil properties, topography, and land use (Descheemaeker et al., 2006) The majority of studies on vegetation cover effects attribute soil loss reduction to the above-ground biomass only, but in reality this results from the combined effects of roots and canopy cover (Gyssels et al., 2005).

The effect of catchment restoration on soil properties has received less attention. Its effect on physical and chemical soil properties like soil moisture (Chen et al., 2010; Wang et al., 2013; Yang et al., 2015), soil organic carbon (Deng et al., 2013; Lü et al., 2012; Ruiz-Colmenero et al., 2013; Wang et al., 2011) or other soil properties (Fattet et al., 2011; Li et al., 2012; Vallauri et al., 2002; Yüksek and Yüksek, 2011) is less understood. (Ruiz-Colmenero et al., 2013) showed that vegetation restoration has significant short-term effects on soil quality, where an increment in soil organic carbon can be measured after only 2 years. Similarly, reforestation of gullied lands can induce soil profile development with organic matter accumulation in the upper soil horizons and contribute to increase aggregate stability (Dlapa et al., 2012).

An interesting study about the effect of vegetation restoration on soil development, erosion and sediment transport is carried out in an artificial catchment called Chicken Creek (Gerwin et al., 2011; Gerwin et al., 2009). In this catchment, the process of ecosystem restoration is continuously monitored by measuring changes in soil physical and chemical properties, in relation to changes in vegetation type, structure and density. The first results suggest that the soil properties, hydrological behavior and vegetation succession are near to point zero in its initial phase of establishment (Gerwin et al., 2011). Soil sampling is planned to be done every 5 years.

In highly degraded environments or active badland systems, the intense water erosion removes and transports the material that is produced by mechanical and chemical weathering of bedrock. In these weathering-limited environments, soil restoration is only possible when the surface is effectively protected against the action of raindrop impact and runoff water. In these conditions, the slow process of soil formation consist in mechanical breakdown of rocks and rock fragments by biological, chemical, and mechanical agents and chemical weathering of primary minerals (Huggett, 2007).

The aim of this study is to gain new insights in the impact of soil bioengineering techniques including vegetation restoration and unassisted vegetation regeneration on soil restoration in active badlands. We are particularly interested in the potential of soil bioengineering techniques to restore soil physical and chemical properties on relatively short time-scales (months to few years). The setup of this study differs from previous work where the effectiveness of ecological restoration was often evaluated by comparing the pre- and post-intervention situation. Here, we intend to collect information of non-intervention (reference sites where no intervention was done) to avoid systematic bias that is often associated with controlled and idealized experimental designs. The dataset consists of a samples that are randomly selected in treated areas – i.e. areas where soil conservation measures were applied – and non-treated areas – i.e. reference sites where no soil conservation measures were implemented.

2.2 Materials and Methods

2.2.1 Study site

The southern Ecuadorian Andes has been selected for this study. All sites are located in the Loreto catchment that is situated at 25 km distance from the provincial town of Cuenca (Figure 10). In the Loreto catchment, the altitude ranges between 2620 and 3100 m a.s.l, the mean temperature is between 15ºC and 17ºC (Dercon et al., 1998) and a mean annual rainfall of 810 mm is measured at the station of Cochapamba-Quingeo (Molina et al., 2009). The underlying bedrock is of Middle to Late Eocene age, and consists essentially of sedimentary rocks including poorly sorted conglomerates, sandstones and red and purple siltstones (Hungerbühler et al., 2002). The major soil types in the agricultural lands are Vertic Luvisols, Vertic Cambisols, and Eutric Cambisols: their soil texture is loamy with dominance of silt. On the slopes, Dystric Leptosols and

Dystric Regosols are found : they are characterized by an almost total absence of organic material (ISSS Working Group RB, 1988).

The land cover in the Loreto catchment is characterized by agricultural land, matorral, pine forest, Eucalyptus forest and shrubs. The farming system is traditional subsistence agriculture. On the croplands, the soil is prepared for planting by animal tillage. Intercropping of maize-beans-peas is common, and bird manure is used as fertilizers. Due to the low productivity, croplands are converted to pastures of natural grass (*Pennisetum*) after some years of cultivation. Grassland productivity is very low due to soil compaction by trampling of animals. Rill and gully erosion have affected the lower part of the Loreto catchment, where active badlands are observed. The active gullies are developed on poorly consolidated and highly weathered argillites and argillaceous sandstone/siltstone (Molina et al., 2009).

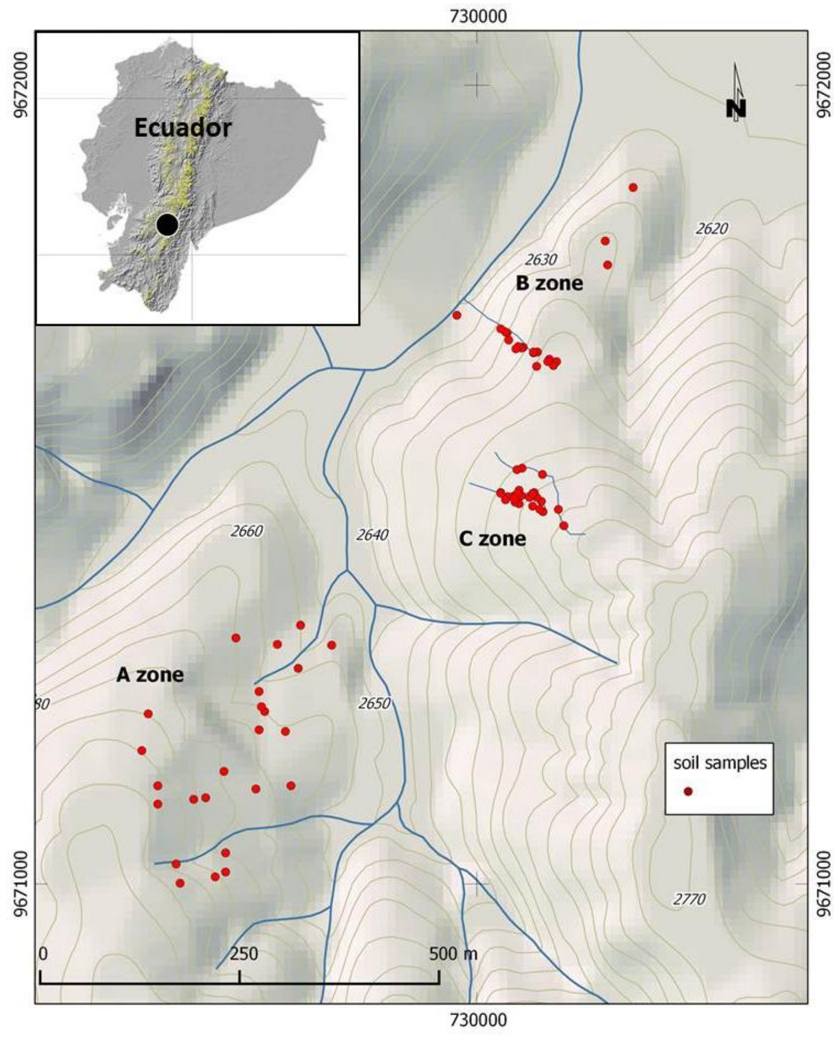

Figure 10: Location of the 64 soil sampling sites (red dots) in the Loreto catchment, southern Ecuadorian Andes. The blue lines represent the drainage network, and the contour lines are drawn in green.

To organize the soil sampling and analysis (Figure 10), an erosion map was established for the Loreto catchment. Three main erosion classes (A, B, and C) were identified based on the type and degree of water erosion (Figure 11 and Table 2).

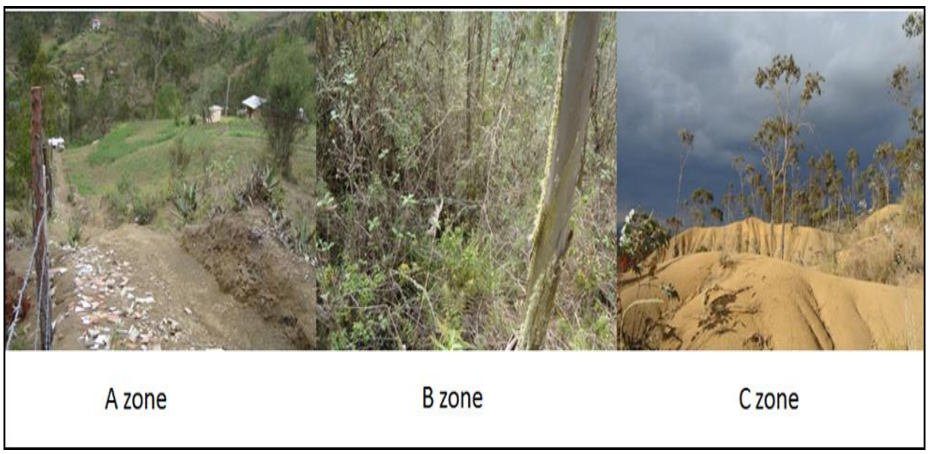

Figure 11: Typology of water erosion processes within the study area, with areas with dominance of (A) rill and inter-rill erosion, (B) gully erosion that is partially stabilized by vegetation and (C) active gully erosion.

Figure 12 illustrates in a schematic way the distribution of the areas that are classified as being affected by erosion type A, B, and C. Sites that are affected by erosion type A show signs of intense rill- and inter rill erosion. This erosion class is dominant in the catchment. Areas affected by type A erosion are used for agriculture (croplands and pasture). Sites affected by type B erosion correspond to stabilized gully areas, and are now largely covered by low-density plantation forest with herbaceous and shrub undercover. Sites that are classified as type C erosion are the most active erosion sources. They can be characterized as highly dissected badlands with active gully systems, and very low vegetation cover.

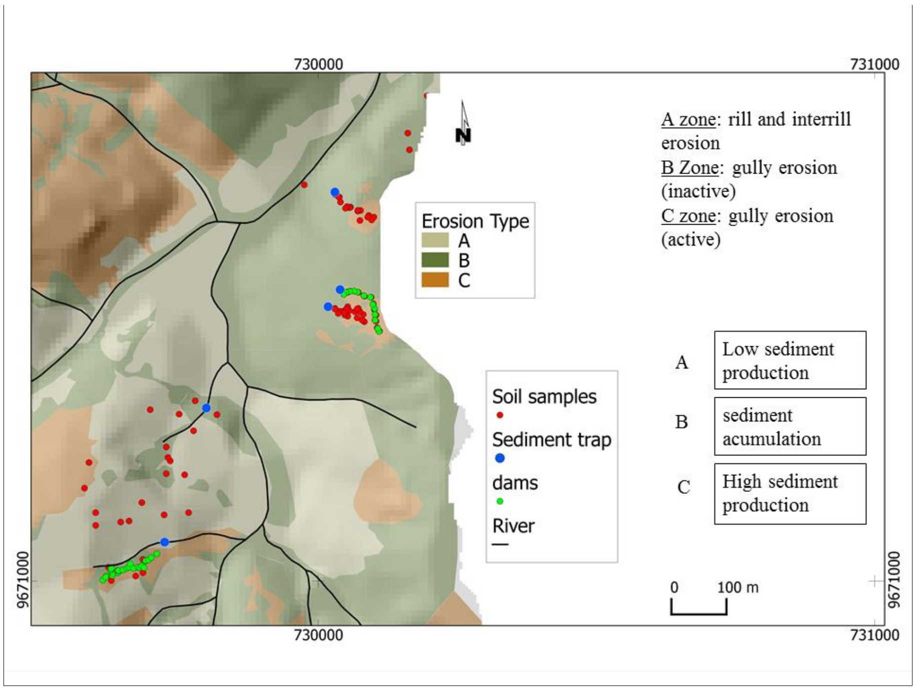

Figure 12: Erosion map that was established for the Loreto catchment, with indication of areas that are affected by erosion type A (rill and inter-rill erosion), B (gully erosion that is partially stabilized by vegetation) and C (active gully erosion).

(A) The area that is classified as ‘A’, is mainly affected by rill and inter-rill erosion. While gullies are present, they are not the dominant source of sediment in this zone. The zone is mainly characterized by crops (*Zea mays*, *Pisum sativum*, *Phaseolus vulgaris*, *Vicia phaba* and vegetables) and grasslands (*Penisetum clandestinum*, *Medicago sativa*). The soils are shallow with low content of organic matter and relatively poor fertility. The incorporation of chemical fertilizers (N, P, K) and natural fertilizers such as bird manure is common. The slope gradients are moderate. Cambisols and Luvisols are commonly found in the agricultural plots, and Regosols and Leptosols are found on the steep slopes. The size of the agricultural parcels is typically less than 1 ha. Land use is very dynamic, with agricultural land producing crops for self-consumption. The local agricultural system rotates arable lands with pasture. Mechanized farming was only recently introduced. Nowadays, sloping plots are levelled with tractors to prepare the land for cultivation. The depth of plowing is typically 25 to 30 cm.

Erosion on agricultural plots is evident only during the first stages of crop development when the soil is not yet entirely covered with vegetation. Once the plants develop in an inter-cropping system of *Zea mays* with *Phaseolus vulgaris,* the plants protect the soil efficiently. Abandoned lands covered with natural pastures (*Pennisetum clandestinum)* are well protected against action of water and constitute an important source of organic matter for soil. Within the agricultural zone, foot paths and unpaved roads are likely the main sources of sediment.

(B) The area that is classified as 'B' is mainly affected by gully erosion. Highly dissected gully topography is visible, but most of the gullies are stabilized by vegetation. Aerial photographs of the 1960s show that the area was heavily affected by gully erosion in the past that resulted in the loss of the fertile soil. The area is now characterized by the presence of poorly developed soils that are in their first stages of development. Under the forest cover, the formation of a litter layer is noticeable, and a few centimeters of soil material can be observed. The main soil types are Leptosols.

Vegetation consists of shrubs, *Pinus patula* and *Eucalyptus globulus* in the forested lands, and a lot of small colonizing plant and shrub species. According to the aerial photographs available for the zone, the plantation of eucalyptus trees started after 1963. In 1989, the presence of an *Eucalyptus* forest is evident from the airphotos. Forestation with *Pinus* is more recent (after 1995) and is visible only on the satellite images of 2009. Eucalyptus forest was established without any forest management. Trees were planted randomly, and the mean distance between tree trunks is about 4 to 5 m. In more degraded zones, the growth of trees is very irregular and not all plants have survived. The density of *Pinus* trees is typically higher (distance between tree trunks of about 2.5 to 3 m), and they were planted in the gully floor and on the hillslopes in row patterns. Growth of new trees from seeds or sproutings is visible in the *Eucalyptus* and *Pinus* forests. Due to the lush understory vegetation that developed in the *Eucalyptus* and *Pinus* forest, sediment production is limited and deposition of eroded soil is strongly enhanced.

(C) The area that is classified as ‘C’ is characterized by the presence of active gully systems. The topography is highly dissected, and typical for active badland areas. Some pedestals with remnants of soil material are present, but most of the soil is eroded as consequence of severe erosion processes. In most of the C zone, the weathered bedrock is outcropping. The exposed rock material, mudstone and claystone, is highly erodible and slakes when it is in contact with water. Vegetation is very scarce and consists of small vegetation patches of herbs and shrubs in the gully channels.

Table 2: Main characteristics of the sites where the erosion type A, B and C are observed.

	TYPE A	**TYPE B**	**TYPE C**
Erosion process	Rill- and interill erosion	Gully erosion (inactive)	Gully erosion (active gullies)
Degree of erosion	Intermediate	Intermediate	Very high
Soil type	Luvisols Cambisols Leptosols	Leptosols	Regosols
Soil state	loose, recovery	recovery, formation	no soil (erosion)
Slope gradient	low - moderate - high	moderate - high	high
Lithology	Poorly consolidated sedimentary rock, very friable and erodible		
Vegetation type	crops, pastures, bare lands, agriculture	shrubs, *Eucalyptus*, *Pinus*, restoration	bare lands, shrubs, restoration
Vegetation cover	low high	medium high	very low

2.2.2 Bioengineering techniques for soil and water conservation

Soil and water conservation schemes were first implemented in the area in the 1980s by UMACPA (Unidad de Manejo y Conservacion de la Cuenca del Rio Paute). In the Loreto catchment, this mainly concerned plantations with exotic rapid-growing species in the regions that were most affected by gully erosion such as the 'B' or 'C' zone. The selected species for forestation was *Eucalyptus globulus*.

In the scope of the CUD-PIC project 'Strengthening the scientific and technological capacities to implement spatially integrated land and water management schemes adapted to local socio-economic and physical settings', a new scheme of soil and water conservation works was implemented in 2010. In the A and C zone where active gully systems are present, a sequence of wooden dams was constructed in the main gully channels using local wood (*Eucalyptus*) and recycled tires. Wooden dams are typically small: they vary in width between 0.9 and 3.2 m, and in height between 0.35 and 0.8 m. The distance between two dams ranges between 1 and 22 m. Their construction started in July 2010.

The physical structures (such as the wooden dams) are expected to enhance the development of vegetation in the gully channels, through coevolution of plant communities and soil development in sediment bodies. Once established, these vegetation islands further colonize degraded sites by spreading in the gully floor and on the steep gully walls.

2.2.3 Soil sampling

To get a better understanding of the spatial variation in soil properties, a total of 64 sites were selected for soil sampling (Figure 10). A stratified procedure was used, were samples were stratified according to vegetation cover, geomorphic position and slope gradient (see Table 3 for more details). Because of practical and logistical reasons (access to sites), the number of samples per class is not strictly the same. Table 4 gives an overview of the number of samples that were taken per land cover type and per erosion class. As the variability in soil properties is higher in the active badlands, most samples were taken at mid-slope and bottom-slope position in the zone with erosion type C (active gully erosion). Of a total of 64 soil samples (64 sites), 40 samples were taken in barren land, 9 in forest, 7 in cropland and 8 in grassland. The samples in the croplands were taken at the end of growing season.

Soil sampling was done in 2012, 2013 and 2014. The depth of sampling was restricted to the top soil (first 25 cm) because soils in the zone are very shallow. A total of 55 samples were taken at depths between 5 - 15 cm (below organic horizon), and 9 samples at depths between 20 - 25 cm. The depth of sampling varies according to the thickness of the topsoil layer, tillage, and presence of roots and stones. Soil samplings was done according to the guide for soil description (FAO, 1990).

Table 3: Sampling campaign was optimized due to stratified sampling. Four variables were taken into account: land cover (LC), vegetation cover (VC), geomorphic position, and slope gradient (S).

Variable	Data type	Categories
Land cover	categorical	crop, barren land, forest, pasture
Vegetation cover	numerical, reclassified into 5 classes	0 - 5, 5 - 10, 10 - 20, 25 - 40, >40 %
Vegetation type	categorical	corn, eucalyptus, herbs, kikuyo grass (*Pennisetum clandestinum*), *Pinus*, recolonized, shrubs
Geomorphic position	categorical	bottom, slope, top
Slope	numerical, reclassified into 4 classes	0 - 5, 5 - 15, 15 - 45, >45 %

Table 4: Number of soil samples per land cover category (cropland, grassland, barren land and forest) and per erosion type with A : area affected by rill and interrill erosion, B : area with stabilized gully systems, and C : active gully erosion.

Zone	cropland	grassland	barren land	forest
A	7	8		
B			10	9
C			30	
TOTAL	7	8	40	9

In all sampling sites, detailed information on location was registered with GPS. The slope gradient of the site was measured with a clinometer. The land cover was derived from 1/10.000 maps of land cover that were established based on mapping GPS in the field (Table 3). The surface vegetation cover was assessed visually using the procedure described in (Molina et al., 2007). Vertical photographs of the soil surface were taken at about 1.5 m height. All photographs were focused on a wooden frame with grid cells of 0.10 x 0.10 m that was placed on the surface in order to have a good reference scale for the analysis. The ImageJ software (Schneider et al., 2012) was used to estimate the fractional vegetation cover, that is a function of the fraction of the pixels covered by vegetation.

From the 64 sites, 6 were selected for a sampling at different depths. The aim was to analyze the variation of soil chemical and physical properties as a function of soil depth. The soil pits were dug manually till the refusal layer: 2 pits are located in cropland (1 in a fallow corn field, 1 in a recently plowed field), 3 in grassland and 1 in plantation forest (*Eucalyptus* forest). The selected sites in cropland are sites more or less stables in time and recent converted lands were not considered for sampling. For each soil pit, samples were taken at different depths in the profile (resp. 5, 15, 20, 35, 45 cm). Note that we were not able to dig a soil pit in barren land. These areas are highly affected by water erosion, and the maximum depth of the regolith is 10 cm. As such, no depth information is available for zone C (area affected by active gully erosion).

Figure 13: Undisturbed soil samples taken in Loreto catchment. The samples illustrate the variability in organic matter content of the topsoil material.

Figure 14: Analysis of soil physical properties at the laboratory of Soil Sciences at the Faculty of Agricultural Engineering, Universidad de Cuenca.

2.2.4 Soil analysis

Undisturbed soil core samples were collected in rings of 100 cm^3 (3 replicates) for analysis of bulk density (BD) and saturated soil water content by volume (θ_{sat}). To determine the saturated water content, 100 cm^3 of soil material was weighted before and after saturation (saturated on saturation mat for 5 days) and finally oven dried (105 ºC for 24 h, Figure 13 and Figure 14):

$$\theta_{sat} = \frac{m_{sat} - m_{dry}}{\rho_{w \times V_b}} \qquad \text{(Eq. 1)}$$

Where m_{wet} and m_{dry} are the masses of the sample respectively before and after drying in the oven; ρ_w is the density of water; and V_b is the volume of the sample before drying the sample. The actual soil water content is determined in a similar way by subtracting the dry weight of the sample from the actual weight, and dividing this by the density of water and the volume of the sample.

Bulk samples were collected (1 kg) to determine the pH, and the content in N and C. The samples were air-dried and crushed to pass through a 2 mm sieve before analysis. Penetration resistance (PR) was registered in the field with a pocket penetrometer and it was averaged from 5 individual measurements. The pH was determined with an electrode in a 1:1 soil –water suspension. Soil carbon content (%) and N (%) was determined in the soil laboratory at ELIC using a vario MAX CN Macro (Elementar Analysensysteme GmbH, Germany).

2.2.5 Statistical analyses

The analysis of the data was carried out with R 2.15.1 (R Development Core Team, 2012). At first time, normality tests were applied for verification of data distribution. The correlation matrix was evaluated to check relationships between variables, using the Pearson r coefficient and rho Spearman coefficient for variables with non-parametric distributions.

To analyze potential differences in soil chemical and physical properties between the three different erosion zones, a first exploratory analysis was made using principal components analysis (PCA) and cluster techniques. For PCA, all variables were normalized and the cluster technique was based on a hierarchical clustering approach. ANOVA (parametric) and Kruskal – Wallis (non-parametric) tests were used to test for statistical differences between the three zones A, B, C with different erosion type and intensity, and are based in the p-values for rejection of null hypothesis considering the zone as a factor.

2.3 Results

A summary with the major outcomes of the analysis of variance can be seen in Table 5. According to the methods detailed above, the dataset was split into three groups that represent the areas with different type and intensity of erosion processes (A, B and C, see Figure 12 and Table 2). Our results of the statistical tests (ANOVA and Kruskal - Wallis tests for non-normal variables) clearly show that the soil chemical and physical properties are significantly different between the three groups (Table 5). Below, we provide a discussion of the results for each one of the variables that was included in the analysis, and continue with an analysis of the interrelationships between variables based on PCA analysis.

Table 5: Variation in soil chemical and physical properties between the areas affected by erosion type A, B, and C. The table shows the results from the Analysis of Variance (ANOVA) and Kruskal - Wallis test. The erosion type (A, B, C) was used here as factor. The number of soil samples that were used in the analysis is 64.

	ANOVA		Mean and S.D.		
	F	Pr	A (15)	B (19)	C (30)
BD (g cm^{-3})	0.779	0.4633	1.34 ± 0.1	1.28 ± 0.1	1.33±0.2
θ_{sat} (%)	7.345	0.0014	51.6 ± 3.9	47.7 ± 4.6	45.7±5.4
PR (g cm^{-2})	2.086	0.1329	2.5 ± 1.1	2 ± 1	1.8±1.1
pH	1.954	0.1504	6.1 ± 0.5	5.7 ± 0.8	5.5±1
	Kruskal-Wallis		Mean and SD		
	Chi-squared	Pr	A (15)	B (19)	C (30)
Depth (cm)	11.414	0.0033	14.3 ± 8.4	13.2 ± 5.1	8.7±2.3
S (%)	11.761	0.0028	26.1 ± 14.1	16.9 ± 13.2	13.5±13.8
Vegetation cover (%)	39.118	0.0000	64.4 ± 31.8	68.8 ± 24.2	13.9±13.5
N (%)	21.991	0.0000	0.2 ± 0.08	0.2 ± 0.06	0.1±0.01
C (%)	16.049	0.0003	1.5 ± 1.1	1.8 ± 1.8	0.5±0.2

BD = bulk density, PR = penetration resistance, θ_{sat} = volumetric water content at saturation, S = slope gradient, n = number of observations

2.3.1 Soil moisture

For the soil moisture content at saturation (θ_{sat}), there is a significant difference in θ_{sat} between the three zones (Figure 16). The highest values of saturated soil moisture content are observed in the A zone where the agricultural land is concentrated with a mean value of θ_{sat} = 51.5 %. Croplands have higher values of soil moisture content at saturation (53 %) compared to grasslands (50.3 %). Interestingly, samples taken in the stabilized badlands (Zone B) have slightly higher θ_{sat} values (47.6 %) compared to the soils in active badlands (Zone C where mean θ_{sat} = 45.7 %).

2.3.2 Bulk density

Our results show no significant differences in bulk density (BD). The mean bulk density of all samples is 1.32 ± 0.1 g cm^{-3}. The bulk density in the A and B zones is very similar, with slightly lower values in the B zone mainly in sites where there is an evident development of incipient soil like in the *Eucalyptus* forest. The C zone has a mean value of BD of 1.33 g cm^{-3}, which is very similar to soils in the A and B zone.

Our data suggest that there are small differences in the bulk density between croplands (mean BD = 1.23 g cm^{-3}) and grasslands (mean BD = 1.44 g cm^{-3}). Croplands are frequently plowed by animal tillage and fertilized with bird manure. The structure of the soil in croplands is better developed with presence of large macro pores. Permanent grasslands suffer from soil compaction, which is here illustrated by the higher BD values.

We observe a positive weak relationship between BD and slope gradient (R^2 = 0.20). In zones with low slopes, the conditions are more favorable for plant development. Root growth and input of soil organic matter can improve the soil structure and reduce the bulk density.

For 6 soil profiles, the depth-variation of bulk density is plotted in Figure 15. With exception of the profile sampled in recently plowed land, the bulk densities do not show major differences as a function of land cover, and show very little variation with depth. A slight tendency is visible, where the bulk density of the croplands is lower than the grassland, and *Ecualyptus* forest.

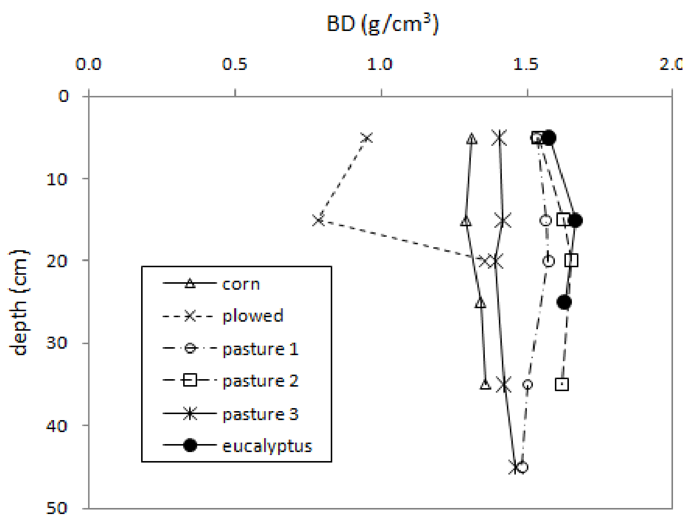

Figure 15: Variation in BD with depth, for six representative soil profiles.

2.3.3 Organic matter

Our results indicate that the carbon content of the topsoil is significantly different between the three zones. Topsoil material in the zone with active gullies (C-type erosion) has significantly lower carbon content (0.5 ± 0.2 %). The highest concentrations of carbon content in the topsoil are observed in zone A and B. It is not surprising to find higher C content in the zone A where agricultural plots are concentrated (Figure 16). In this zone, the soil still conserves some of its original properties and can eventually achieve 25 cm depth. Pastures present slightly higher values of C compared to cropland (respectively 1.87 vs. 0.97 %). The pastures that now cover the majority of land in the A zone protect the soil effectively against soil erosion and constitute a good input source of organic matter. As a part of the agricultural practices, farmers often incorporate manure to the soil. This manure is produced at the farm by the farm animals (in little quantities) or is bought as commercial fertilizer (bird manure).

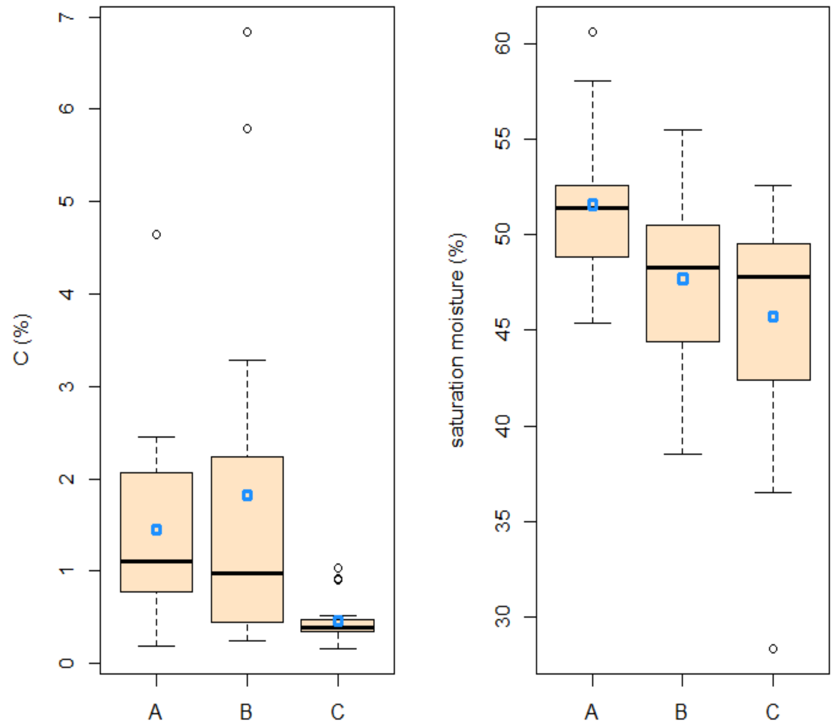

Figure 16: Boxplots showing the variation in C content (%) of the topsoil, and the variation in soil moisture content at saturation (θ_{sat}). The mean values are plotted in blue rectangles. The samples are classified according to the type and degree of erosion of the plots where they were sampled, and represent: A, Rill and inter-rill erosion, B, Passive gully systems; C, active gully systems.

Figure 16 shows that the C content in the topsoil shows considerable spread for a given zone. When analyzing the spread in the data, we observe that most of the variation in C content is explained by the erosion type and intensity and the vegetation cover of the sampling site. The latter is illustrated in Figure 17, where the C content of the topsoil is plotted against the fractional vegetation cover of the sampling site. There is a clear and positive relationship between the C content (Y-axis) and the vegetation cover of the site. Clearly, the highest C content is observed for sites with very high vegetative cover (above 75 %) in areas that are not affected by intense active gully erosion. As expected, the zone with active gullies (Zone C) has the lowest values of C content, as the vegetation cover is generally very low.

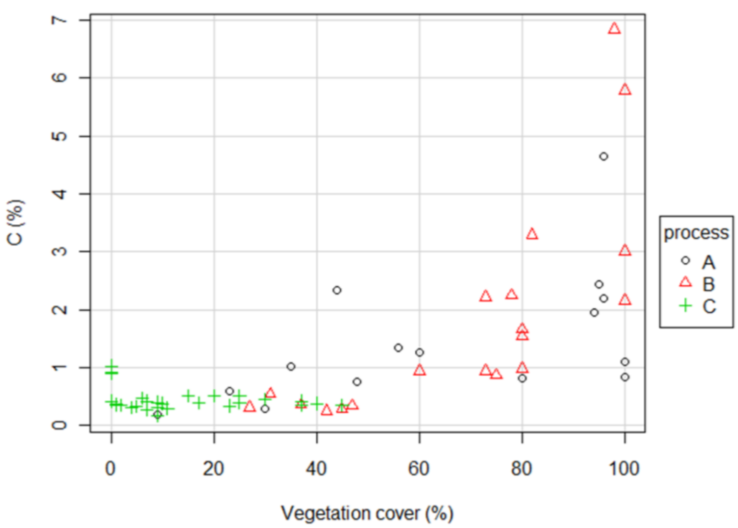

Figure 17: Scatterplot of carbon content (%) of topsoil material as a function of the vegetation cover (%) of the surface. The samples are colored according to the type and degree of erosion in the plots where they were taken.

For 6 soil profiles sampled in cropland, grassland and forest, we have analyzed the variation in C content (%) with depth. The major differences are observed in the first 10 cm, and are probably related to differences in accumulation rates of organic material. Below 15 cm, there is only very little variation of C content with depth for a given profile. Figure 18 nicely illustrates the differences in C content as a function of land use and soil management. The corn field has the lowest C values (C < 0.40 %), and the carbon content varies very little with depth. Higher C-values are observed for grasslands and *Eucalyptus* forest, with a clear increase of the C content in the upper 10 cm of the soil. The effect of soil management is clearly visible for the croplands: the corn field has very low C values, in contrast to the recently plowed soil where higher values are found in the topsoil probably due to the organic fertilizers added.

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2.3.4 Other variables

Penetration resistance (PR) is a measure of soil compaction and porosity. This variable shows a positive relationship with BD ($R^2 = 0.29$). Soils in the zone C have the lowest value of penetration resistance, while the values in the zones A and B are systematically higher.

The N content of the topsoil samples shows similar patterns as the C content, with higher values for soils of Zone A. With respect to the pH, the values for this variable are between moderate acid and slight acid and there are no significant differences between Zones A, B and C.

2.3.5 Interaction between vegetation cover, physical and chemical soil properties

Figure 19 shows the association between the soil moisture content at saturation θ_{sat} (%), bulk density BD (g cm^{-3}), Carbon content (%) and vegetation cover VC (%), and the correlation coefficients are given in Table 6. A strong correlation is observed between the vegetation cover and the C content of the topsoil. An increase of the organic matter content in the topsoil can enhance soil physical properties like soil structure, porosity and bulk density, and saturated soil water content θ_{sat}. However, this is not observed in our dataset as we do not observe a clear association between C content and θ_{sat}. Given that the C content is generally very low in the studied soils, the water content of the soils likely depends more on physical soil properties. A good correlation is found between the saturated volumetric soil water content and bulk density (r = - 0.52) and low correlation between θ_{sat} and C (r = 0.06).

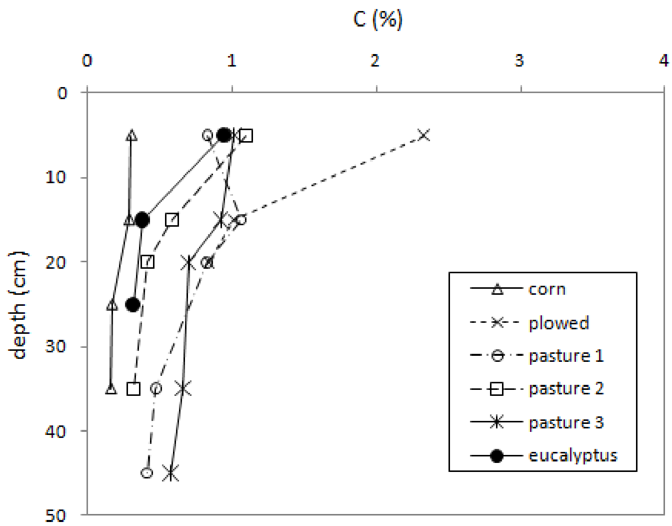

Figure 18: Variation in C content with depth, for six representative soil profiles.

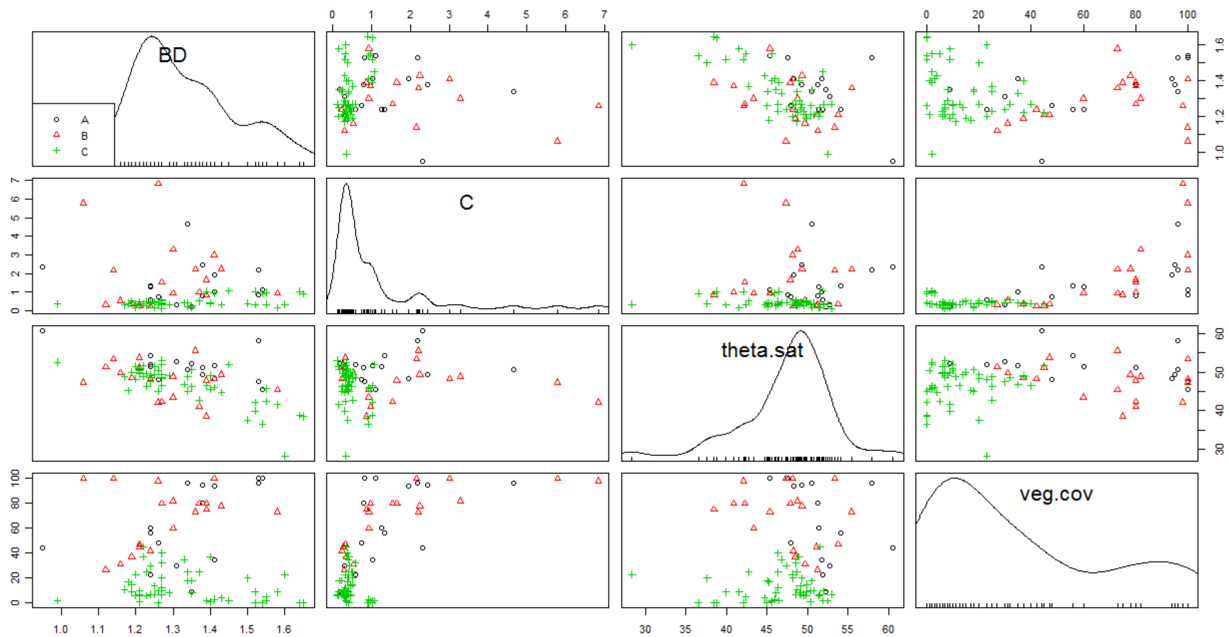

Figure 19: Relation between $\theta_{sa\,(\%)}$ BD, and C content (%) and vegetation cover of the sampling site. A total of 64 samples were analyzed. Samples are coloured according to the type of erosion (A: black circle, B: red triangle, C: green cross).

Table 6: Spearman correlation matrix (n = 64).

	Depth	BD	θ_{sat}	Slope	Vegetation cover	PR	N	C	CN
BD	-0.18								
θ_{sat}	0.22	-0.52							
Slope	-0.14	0.44	-0.15						
Vegetation cover	0.35	0.02	0.19	0.24					
PR	-0.002	0.62	-0.15	0.44	0.22				
N	-0.03	0.30	0.06	0.63	0.63	0.55			
C	0.01	0.20	-0.001	0.60	0.61	0.52	0.85		
CN	0.03	0.17	-0.03	0.56	0.58	0.51	0.76	0.98	
pH	0.11	0.08	0.12	0.27	-0.08	-0.09	-0.04	0.05	0.09

Table 7 presents the results of the principal components analysis (PCA) for the variables summarized in Table 6. The first two components explain respectively 29 % and 27 % of the total variance. Table 7 shows the loadings of the variables on the three first principal components. The first component is related to the organic part of soil (C content and vegetation cover), while the second principal component is more related with the structural part of soil (bulk density and soil moisture content at saturation. Figure 20 shows the correlations between the variables and the two first components, which together account for 56 % of the total variance of the dataset.

Table 7: Eigenvectors matrix (loadings) with the correlation of the variables (resumed in Table 6) with first three principal components.

	PC1	PC2	PC3
C	**-0.445**	0.383	-0.302
depth	-0.077	0.314	**0.570**
BD	-0.450	**-0.482**	0.176
θ_{sat}	0.221	**0.552**	0.174
VC	**-0.445**	0.455	-0.066
pH	-0.047	-0.052	**0.715**
PR	**-0.586**	-0.095	0.078

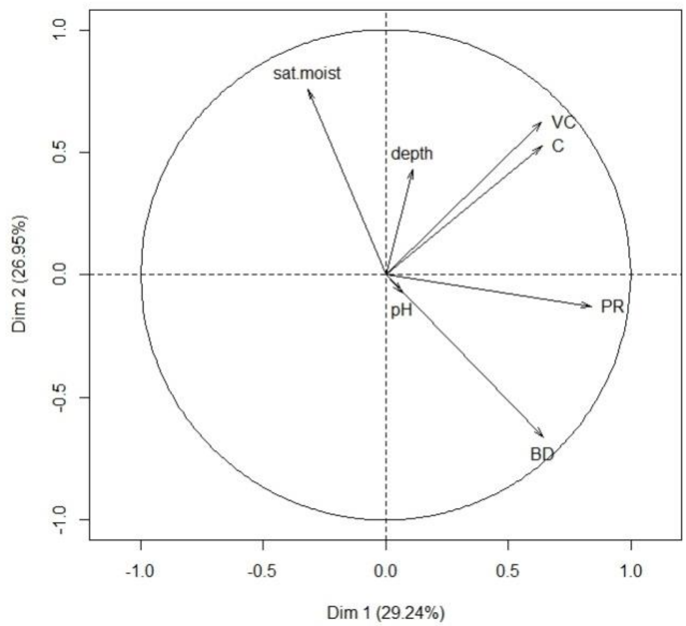

Figure 20: Orientation of quantitative variables in the vector space formed by the first two principal components (Dim1 and Dim2). The first component (Dim1) accounts for 29 % of the total variance of the dataset, and the second component (Dim2) for 27 %.

The results of the PCA show a pattern of soil chemical and physical characteristics that is closely linked with the type and intensity of water erosion processes (Figure 20). In the first vector space, there is a separation along the first component between the areas with active gully erosion (Zone C) on the one side, and the area with rill and inter-rill erosion (Zone A) and stabilized gullies (Zone B) on the other side (Figure 21).

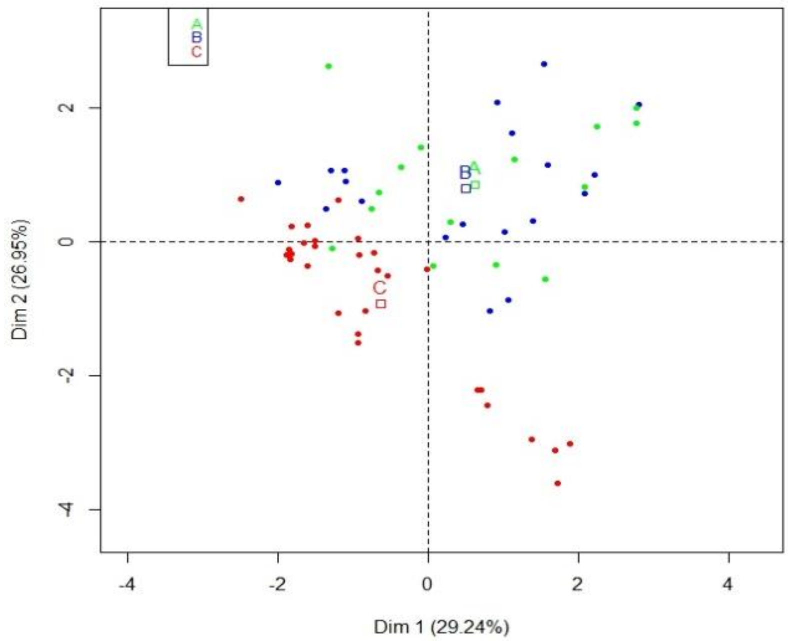

Figure 21: Distribution (factor scores) of soil samples in the space formed by the first two principal components (Dim1 and Dim2). The symbols are colored according to the dominant erosion type and its intensity in the sampling plot.

Figure 20 and Figure 21 show that the variation in the physical and chemical soil properties of soils in areas affected by erosion type A and B is mainly related to C content, and vegetation cover. For the remaining soils in the C zone, the variation is strongly related to physical soil properties such as bulk density and saturated volumetric water content as vegetation cover and organic matter input are very low for these soils.

The fact that samples taken in zone A and B are clustering (Figure 21) is maybe not so surprising. In both zones, the vegetation cover is often well developed, and the carbon and nitrogen content in the topsoil is relatively superior. The soil depth is typically higher in these zones, either because of the presence of truncated soils or incipient soil development after restoration, however is deeper in zone A. The correlation with soil depth is less pronounced, probably because soils are generally very shallow with a soil depth of max. 35 cm. In most cases, the soil is only a few centimeters thick. Depth appears more relevant the third component along with pH and its influence is more or less the same for al samples.

2.3.6 Spatial variability

Soil bioengineering techniques that include vegetation restoration and unassisted vegetation regeneration are based on the use of living materials for soil restoration. Restoration actions have an effect on the spatial distribution of organic matter (Xin et al., 2016), through changes in the vegetation cover. As organic matter is an important component of soils, it is important to understand how this variable evolves on relatively short time-scales (months to few years).

In catchments strongly influenced by anthropogenic actions, one of the factors that affect the spatial distribution of soil organic matter is land use and vegetation cover. This is also observed in our dataset where vegetation cover is strongly related to the soil C content (r = 0.61, Table 6).

Our observations show that the spatial variability in soil C content in the badland area (C-Zone) is very low. In this highly dissected topography, soils are very shallow and have low C and N contents. Physical factors like local slope gradient and topography condition the evolution of Regosols and Leptosols. In the agricultural lands, the soil chemical and physical properties are more dependent on land use and land management (use of fertilizers, plowing). The soil profile is more developed and it is common to find well developed soil profiles (Figure 18).

2.4 Discussion

The results of this study show on the one hand the relationship of soil organic matter with visible changes in soil properties in incipient soils and on the other hand the importance of vegetation in reducing soil erosion and supplying organic matter.

2.4.1 The role of vegetation in restoration

Vegetation affects the stabilization of gullies in several ways. Dense vegetation cover in the study site is found in forests of *Pinus* and *Eucalyptus* (Zone B) and grasslands (Zone A). The dense vegetation cover in forest plantations or artificial pastures protects the soil against the effects of water erosion. The roots of plants act like a network that improve the mechanic resistance of the soil and provide organic matter that accelerates the soil forming processes. The values of resistance to penetration measured in zones A and B are also related with the presence of roots from surrounded plants that were abundant in many sampling points. As can be seen in Figure 17, the vegetation cover also raises the C content in the incipient soils, improving the soil structure.

In regions that are highly susceptible to water erosion because of their biophysical settings (lithology, climate and topography), it is fundamental that tensioning factors (vegetation removal) are reduced to facilitate ecosystem restoration. This can be observed in Zone B, where water erosion is strongly reduced in spots that were restored by forestation. In Zone C where all this conditioning factors still continue to act, the implementation of bioengineering works can provide an efficient solution for sediment retention.

The evolution of land cover suggest that zone B and C were very similar in vegetation cover before the 1970s (Figure 6). Both zones were characterized by a very low vegetation cover, and were highly degraded with presence of active erosion features (rills and gullies). Whereas the restoration works in zone C only started in 2010, restoration by forest plantations in zone B started in the 1980s.

The comparison of the physical and chemical soil properties of the two areas provide us with insights in the changes in soil properties after restoration. During the 4-year measuring period, there was no clear evidence of incipient soil formation in areas near to wooden dams, because no significant differences were found in the soil C content in comparison with the mineral soil sampled in the gullies without restoration measurements. However, the sediment deposited behind the wooden dams provides a favorable environment for establishment of pioneer species. The pioneers stabilize the sediment deposits and can colonize surrounding areas, including the gully floor.

For the formation of a litter layer and posterior evolution of an *O* and *A* horizon, it is necessary that a good vegetation cover exists. In the sediment accumulations in the gully floor, the presence of roots from plants and pioneers can become an input source of organic matter in the future. Thus, a correct implementation of bioengineering works in the gully floor enhances sediment deposition in the gully channels, and constitutes a favorable environment for plant development and colonization.

2.4.2 Soil restoration and its influencing factors

To restore the active badlands (C zone), it is fundamental that the erosion rates diminish so that incipient soil formation and development can start. Soil restoration is a slow and complex process. The formation of a protective soil surface by e.g. vegetation (pioneer plants, and rapidly growing tree species that allow development of understory) diminishes the impact of rain drops, enhances soil infiltration and reduces surface runoff.

With the addition of organic matter by decomposition of vegetal tissues begins the process of soil formation. Degraded areas may have a significant potential for C sequestration after abandonment because erosion zones have a higher potential to increase their SOC (De Baets et al., 2013). The results in this study show topsoil C values of 3.4 % for forested areas in zone B, in comparison with 0.48 % for topsoil in eroded land of zone C. These would be the maximum potential that could be achieved by the ecosystem, because the increase in SOC is most evident in the first 10 to 50 years following abandonment according to (De Baets et al., 2013). The forest plantations in zone C were established in the 1980s and are mature *Eucalyptus* forests. During the first years of ecosystem restoration, one can expect a very rapid and dynamic development (Gerwin et al., 2011). After this period of time, the rate of increase of C in the topsoil is expected to approach steady state.

The superior C content in topsoil of the restored land (B zone) is due to the formation of a thin *A* horizon and also an *O* horizon in the forest soils (Figure 22). Soil C content is considered to be an important indicator for ecological restoration (Ruiz-Jaen and Mitchell Aide, 2005). Soils under *Eucalyptus* forests had amongst the highest C/N ratios that were measured in this study (C/N = 11.58), suggesting slow decomposition of plant litter and a net accumulation of organic C. In comparison soils under Pinus have C/N ratio = 17.35, crops of corn have C/N = 6.1 and grass have values of C/N = 8, being the lowest values for herbs vegetation in degraded lands with C/N = 3.6. However, these values are not as high as they are typically in natural and mature forests, where the organic matter can interact with other elements and microorganisms. In a semiarid climate in northern Arizona in restoration patches with Pinus, C/N ratios of 17 were found and 4 for grass patches (Abella et al., 2013).

Figure 22: Incipient soil development in the restored catchment (Zone B). This picture is taken from a plot inside the *Pinus* forest (Photo taken by Pablo Borja, Loreto catchment, 2013).

The process of soil restoration under these conditions can be very slow. In the case of the restored catchment, it has taken approximately 30 years (based on the observations from the aerial pictures that show the time when the *Eucalyptus* plantation started). The fact that the pH does not show significant differences between the three zones may be due to the changes at this stage of soil development that are mostly noticeable in the physical soil properties. The vegetation in this phase is acting mainly at the level of soil structure increasing the total volume of macro pores, which is reflected in differences in bulk density and water retention capacity. Our observations suggest that the pH is lower for the C zone (pH = 5.5) in comparison with B zones (pH = 5.7), but the results are not statistically significant. In the case of samples from *Eucalyptus* forest the pH mean value is also 5.5 This finding is generally consistent with the concept that a lower pH is indicative for higher chemical and biological activity in the topsoil, as soil forming processes cause a decrease in pH with respective increase in SOC content (Ruiz Sinoga et al., 2012). In soils in Uruguay Farley et al. (2008) found that *Eucaliptus* caused soil acidification for the registering values of pH = 5 for a 0 -10 cm depth.

2.5 Conclusions

The establishment of vegetation exerts a significant role in the process of gully restoration. Vegetation physically protects the soil against soil erosion agents, and enhances retention of eroded material. One additional benefit of vegetation is the direct supply of organic matter to the topsoil that contributes to soil formation processes. Our results show that vegetation cover is the most important factor explaining the observed variation in C content of top soils.

The implementation of bioengineering techniques is an effective initiative not only for reducing the sediment export but also for restoring the soil and vegetation. The increment of carbon content in zones under restoration is 3.6 times compared with zones were gullies are still active. In the badlands, an appropriate plan of reforestation is an effective alternative for long term restoration. The soil analyses indicate that soil physical and chemical properties have improved after restoration of this strongly degraded environment. In the zone where exotic rapidly growing tree species were implemented, the changes in soil properties like C content in the topsoil have favored a first stage of soil development and an *O* horizon is visible.

Chapter 3 : Bioengineering techniques reducing sediment mobilization in active gully systems in the Andean mountains

Abstract

Gully erosion is an important process of land degradation in mountainous regions, and is known to be one of the major sediment sources in degraded catchments. Recent studies have suggested that bioengineering can be effective for ecosystem restoration. However, few quantitative studies exist on the effect of bioengineering techniques on sediment production and transport in the tropical Andean mountains.

In this study, sediment mobilization and transport was studied in great detail in five micro-catchments with different soil and water conservation treatments. The bioengineering techniques that were used for soil and water conservation involve soil bioengineering on the hillslopes and biotechnical structures in active gully channels. To characterize the routing of sediment transport within the micro-catchments, we measured erosion and sediment deposition areas in the gully channels. Sediment yield was estimated from measurements of sediment accumulation in sediment traps that were constructed at the outlet of the micro-catchments.

Results point out that bioengineering techniques are effective to stabilize active gullies. The construction of wooden barriers (or so-called dams) in active gully channels enhances sediment deposition in the channels. The deposition of sediments behind dams is strongly dependent on previous rainfall events, as well as gully channel slope, and its vegetation cover. Flow barriers are shown to be very effective in stabilizing active gully systems in badlands through significant reduction (more than 70%) of the amount of sediment exported from the micro-catchments. Also, the restoration of vegetation on the slopes is shown to be very effective in controlling soil erosion, sediment production and mobilization. The experimental data suggest that there exists a threshold value of rainfall intensity (I_{30}max) of about 23 mm, above which all sections of the gully system are actively contributing water and sediment to the river network.

3.1 Introduction

Gully erosion is an important process of land degradation, and is known to be one of the major sediment sources in a wide range of environments across the world (Poesen et al., 2003). Tropical mountain areas are particularly vulnerable for soil erosion due to their irregular and steep topography (Harden, 2001; Vanacker et al., 2003a). In the Ecuadorian Andes, the formation of gullies dates back to at least the 1950s and 1960s. Soil erosion poses important limitations to the development of agriculture, as elevated soil loss has a negative effect on crop productivity (Pimentel et al., 1995). Transport of eroded soil particles in the river system also affects water quality in downstream areas threatening drinking water production and hydropower generation.

Quantitative information on sediment production and transport in highly degraded areas is still scarce. An important issue in erosion research is the spatial extent of the study area, as erosion and deposition processes are scale-dependent. The majority of erosion research in the Ecuadorian Andes focused on rill and inter-rill erosion based on plot scale experiments, and cannot be directly extrapolated to larger spatial scales. Few quantitative studies exist on the importance of gully erosion to the overall sediment budget, and the dynamics of soil erosion and sediment deposition in gully systems (Molina et al., 2009). Harden (1993) cite data from non-published technical reports with erosion rates up to 95 t ha^{-1} yr^{-1} for the Jadan catchment, which are the highest erosion values reported in the region. Harden (2001) measured sediment detachment rates of 3.7 g mm^{-1} (the maximum observed) in abandoned lands in southern Andean Ecuador. In the northern part of the country, erosion rates up to 836 t ha^{-1} yr^{-1} are measured with a portable rainfall simulator (Harden, 1988). In a recent study in the Southern Andes , the RUSLE model was used to estimate soil losses and maximum rates of 936 t ha^{-1} yr^{-1} for croplands and pastures and 1.5 – 40 t ha^{-1} yr^{-1} for sites under natural forest vegetation have been estimated (Ochoa-Cueva et al., 2015).

(Molina et al., 2008) pointed out that small hydrological units such as micro-catchments (10^{-2} – 10^{2} km^{2}) are interesting environments to study erosion processes in mountain areas, given the large spatial heterogeneity in climate, topography, soils and vegetation cover that exists over short distances. Analyzing erosion at the spatial scale of hill slopes and/or micro-catchments allows one to limit the existing variability in climate, lithology and topography to a minimum, and isolate the effects of e.g. vegetation cover or land use.

Vegetation is often cited as one of the most important factors controlling erosion processes. According to (Vanacker et al., 2007), vegetation cover exerts an important control over present day erosion rates at the catchment scale. An increase in vegetation cover can help to protect the ground surface for raindrop action allowing infiltration and reducing surface runoff (Bruijnzeel, 2004), regulate surface hydrology (Thornes, 1990) and reduce rates of water erosion and sediment transport (Bellin et al. 2011; Stokes et al., 2008; Molina et al., 2009; Rey et al., 2004; Vanacker et al., 2007). Modeling results show that an increase in vegetation cover results in a reduction of surface runoff (Collins et al., 2004; Istanbulluoglu et al., 2004), water erosion and sediment transport rates (Istanbulluoglu and Bras 2005). A dense and short vegetation cover is an effective way to control water erosion (Sanders, 1988).

In degraded lands, bioengineering techniques that incorporate vegetation restoration can be effective in reducing water runoff (Xu et al., 2008; Chirino et al., 2006; Descheemaeker et al., 2006), and soil erosion and sediment yields (Burylo et al., 2007; Chirino et al., 2006; Cohen and Rey, 2005; Erktan and Rey, 2013; Marden et al., 2014; Rey, 2009; Rey, 2004; Rey, 2003; Rey et al., 2005; Rey and Burylo, 2014; Zheng, 2006).

The position and distribution of vegetation patches is crucial to understand sediment mobilization and transport within degraded catchments (Rey, 2003). Rey (2004) and Rey and Burylo (2014) demonstrated that vegetation barriers in gully floors are very effective to trap sediment, and can reduce sediment transfer. (Molina et al., 2009) showed that the presence of a dense vegetation cover in gully floors can result in significantly lower sediment transfer: the deposition of eroded sediments in the gully bed can represent more than 25 per cent of the volume of the sediment generated by erosion processes within the catchment (Molina et al., 2009).

In the dry valley of Minjiang River, SW China, three species of plants were tested for their effectiveness to reduce water runoff and soil loss, finding that they had positive effects on soil porosity and soil water content, and a high effectiveness in controlling soil loss. However only one of these species had clear effects on runoff (Xu et al., 2008). In a semi-arid region of Spain, a study with *Aleppo sp* pine found that that runoff coefficients of vegetated plots were less than 1% of the precipitation; whereas runoff in bare lands was nearly 4%. Soil losses in vegetation plots averaged 0.04 t ha^{-1} $year^{-1}$ and increased 40-fold in open-land plots (Chirino et al., 2006).

Little is known about the combined effects of soil bioengineering for gully restoration on sediment production and transport in highly degraded lands. In this paper, we will quantify the efficiency of soil and water conservation treatments on sediment mobilization in active gully systems, and sediment export to the river network. The soil and water conservation project consists of vegetation restoration with plantations of *Eucalyptus* and *Pinus*, and gully stabilization with implementation of a set of flow barriers or so-called dams in the gully floor. In addition, a time series of erosion and sedimentation data will be analyzed to determine the critical rainfall thresholds for gully erosion.

3.2 Study area

The study area is located 23 km south of the city of Cuenca located in the southern Ecuadorian Andes (Figure 23). The area is characterized by the presence of active gullies, and elevated soil erosion rates. Recent work by (Molina et al., 2008) in southern Ecuadorian Andes has reported catchment-wide erosion rates varying between 0.26 and 151 t ha^{-1} yr^{-1}, largely as a result of variations in vegetation cover, lithology and topography. The land cover pattern is very fragmented, with a mix of cropland, rangeland, shrubs and bushes, and patches of forests of *Eucalyptus* and *Pinus*. The rainfall regime is bimodal with two rainy seasons, registering between 600 and 1000 mm of yearly rainfall (Celleri et al., 2007). In the study area, an average rainfall of 913 mm yr^{-1} was measured during the period February 2011 to August 2014.

Five experimental micro-catchments were selected in the Loreto catchment, a tributary of the Jadan River, for this study. Catchments were selected in the lower part of the Loreto catchment where soil erosion is evident. Agricultural land is concentrated on the most suitable (less eroded) soils that have different degrees of soil erosion dependent on their slope gradient and vegetation cover. Two micro-catchments (DT, AT) are not treated, respectively in the agricultural and degraded land, and serve as ‘reference’ sites. In the text, we refer to the micro-catchments in agricultural land as “AT” (non-treated site) and “AI” (treated site), and to the ones in degraded land or badlands as “DT” (non-treated site), “DI” (treated site) and “DF” (treated site with reforestation) (Figure 7).

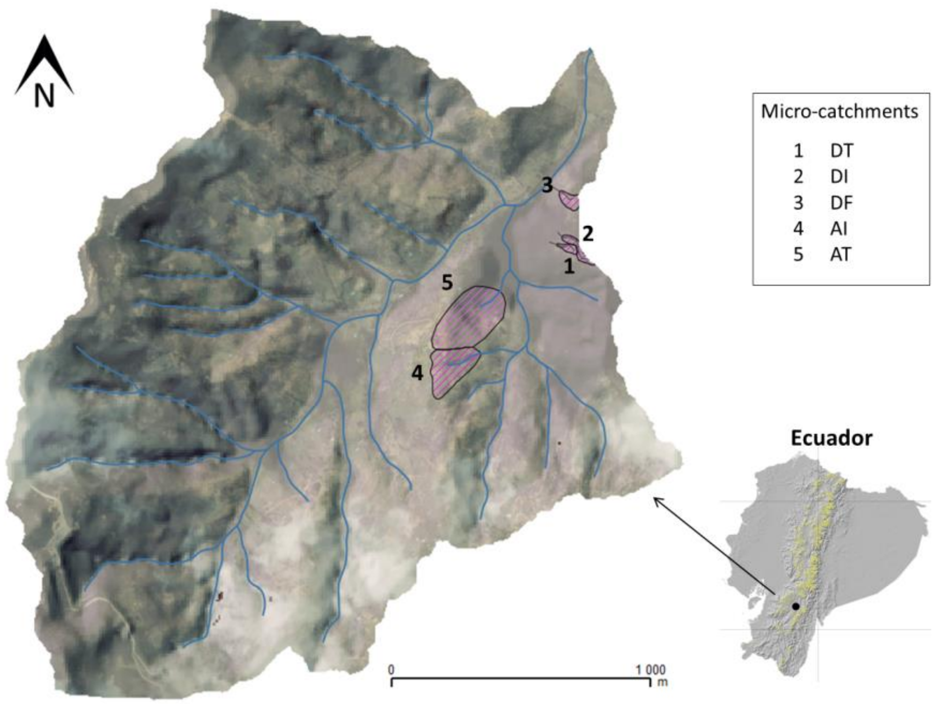

Figure 23: Location of the study area in the Southern Ecuadorian Andes. The five micro-catchments are part of the Loreto catchment. Two micro-catchments are located in the agricultural area: (AI) with gully restoration and (AT) without any restoration. In the highly degraded land, three micro-catchments were selected: (DI) catchment with gully restoration, (DT) without gully restoration and (DF) vegetation restoration with *Eucalyptus* plantations. Topographic information is derived from a 10 m DTM, and a satellite image (Digital Globe Quickbird, 2008) is given as background.

3.3 Materials and Methods

3.3.1 Characterisation of the micro-catchments

Digital topographical maps (1:10 000) were used for the delimitation of the micro-catchments. The dense network of gullies was mapped in the field using a handheld GPS with a horizontal accuracy of 5 m. Based on the digital topography, maps of local hillslope gradient and orientation were made in ArcGIS.

A vegetation map (1:10000) was made based on field observations of vegetation type and density. Polygons with homogeneous vegetation cover were identified in the field, and were mapped using a handheld GPS. The survey started in 2011, and consisted in several campaigns during 2012 and 2014. The following vegetation classes were identified: grasses, Eucalyptus forest, scrubs, shrubs, arable land and barren land. The ground vegetation cover (%) was determined based on point measurements of the surface vegetation cover and includes all types of living materials. A wooden square frame (1m x 1m) was located on the ground (Figure 24), and a vertical photograph was taken to estimate the percent vegetation cover at ground level (0 – 0.1m). The estimation of image fraction occupied by vegetation pixels was done using ImageJ software (Schneider et al., 2012).

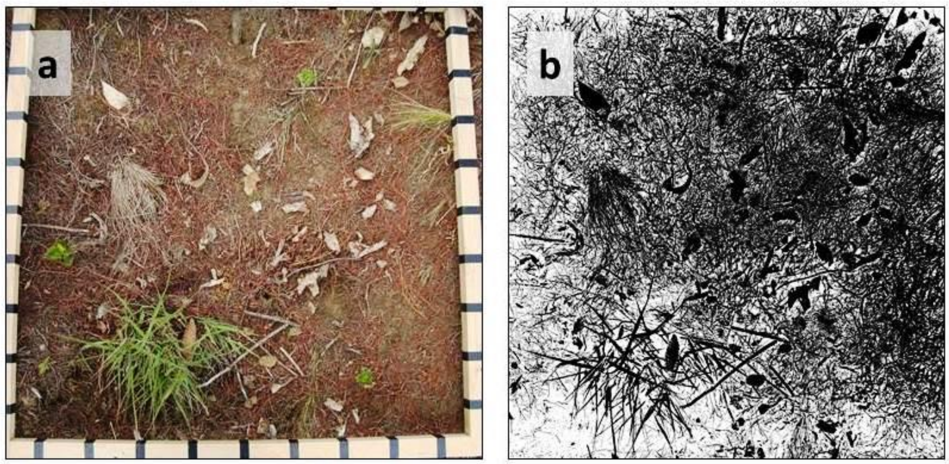

Figure 24: Estimation of surface vegetation cover (%) based on analysis of digital images of 1 x 1 m vegetation plots. a) Original image. b) Processed image to extract automatically the pixels covered by living plant materials.

3.3.2 Soil and water conservation

Erosion mitigation in the wider area was largely an initiative taken by the former Instituto Nacional de Electrificación (INECEL) and the Unidad de Manejo de la Cuenca del Rio Paute (UMACPA) in the 1980s and 1990s to restore highly degraded catchments and reduce sediment transport in the river system. The restoration programs of UMACPA consisted in the construction of gabion dams in small tributaries to adjust the gradient of the gully channels, reduce the erosive power of the river and reduced the sediment export to the Paute River. It also included vegetation restoration through tree plantations of *Pinus* and *Eucalyptus* on degraded land.

In the scope of the CUD project on “Strengthening the scientific and technological capacities to implement spatially integrated land and watershed management schemes adapted to local socio-economic and physical settings”, a gully restoration project was implemented in the five micro-catchments. This project is complementary with the vegetation restoration project of UMACPA that was initiated in the area in the 1980s and 1990s.

Wooden dams or small flow barriers were constructed with poles of wood having a 15 cm diameter. After hammering two to three strong wooden poles in the base of the gully floor, wooden stakes were piled up behind the line of poles. The structures are typically 1.5 m wide and 1 m high, and were constructed in the floor of active gully channels. The dams were made of wooden poles of *Eucalyptus*, and in some of them recycled tires were used (Figure 25). The poles and stakes are fixed by means of metal wire and nails so that the structure is firm and stable. The construction of dams started in July 2010. The dams vary in width (0.9 to 3.2 m) and height (0.35 to 0.8 m). The distance between consecutive dams is 1 to 22 m. A total of 47 dams were constructed: 12 dams and 35 dams in the DI and AI micro-catchment respectively. Their location is not random, and they are often located downstream of a confluence of two or more gully channels in order to keep the number of dams at a minimum.

During the duration of the study, seven of a total of 47 dams were damaged and two were completely destroyed. Destruction was due to vandalism and theft of wood. In the agricultural micro-catchment (AI), the majority of damaged dams were repaired. In the micro-catchment in degraded land (DI), two dams were completely destroyed and were not repaired. In the agricultural lands, the dams were implemented in private fields which facilitated the maintenance of the structures. In the degraded land (DI micro-catchment), the land is owned by a public company (EMAC), and local inhabitants frequent the area for wood extraction and grazing.

a) barren land b) arable land

c) forested land d) wooden dams for sediment control

Figure 25: Land cover types of the study area with (a) barren land, (b) arable land, (c) forested land with *Eucalyptus.* (d) Picture showing the implementation of small structures for sediment control.

3.3.3 Monitoring of sediment transfer and export

During two rainy seasons, the sediment accumulated behind the dams was measured and the data cover the period 18/07/2011 till 30/05/2012. Our method for calculating sediment deposition in the gully floor is based on (Bellin et al., 2011; Sougnez et al., 2011). The sediment height was measured regularly behind the newly constructed dams to estimate the volume S_V (m³) of sediment retained based on the measurement of the geometry of the section using the following equations:

$$S_V = \sum_{i=1}^{n} \left(\frac{CS_i + CS_{i+1}}{2}\right) . L_i \tag{Eq. 2}$$

where CS_i is cross area (m^2) of ith section, and L_i (m) is the length of sediment cumulated between two cross sections (Figure 26). The cross section CS_i is determined as follow:

$$CS_i = \frac{(B_i + b_i)}{2} . h_i \tag{Eq. 3}$$

With h_i is the thickness of the sediment deposit (m), and B_i and b_i are respectively the top and bottom width of the sediment accumulation area (m).

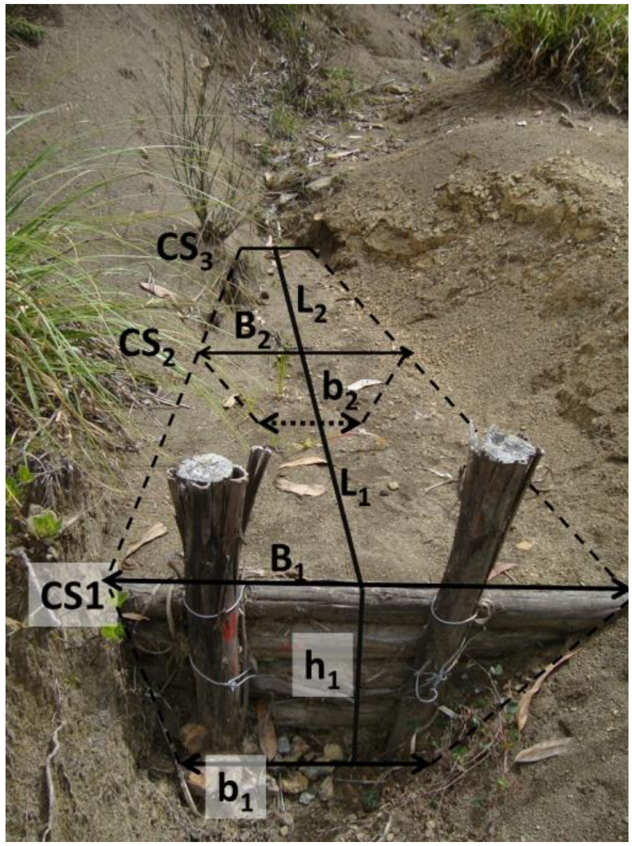

Figure 26: Schematic representation of a wooden dam, with indication of the measurements that were realized in the field for determination of sediment volumes. We refer to Equation 2 and 3 for more details on the calculation of the sediment volumes.

At the outlet of every micro-catchment (DI, DT, DF and AI), a sediment trap was constructed to estimate the volume of sediment that is exported from the catchment. All traps consist in a pit of 1 x 1 x 0.5 m that was dug in the main gully channel. The base of the trap was covered with concrete and the walls were reinforced with wooden poles till a height of 0.5 m. The material accumulated in the traps was measured at weekly basis to quantity the amount of sediment that is exported out of the micro-catchments (Figure 27). Occasionally when the trap was completely filled with sediment, sedimentation occurred in the gully channel downstream of the sediment trap. On these occasions, the total volume of sediment deposited was determined by summing the volume of sediment accumulated in the trap and the sediment deposited in the channel based on GPS measurements in the field.

Figure 27: An example of a sediment trap that was constructed at the outlet of the micro-catchment to estimate the sediment export.

In our results, the sediment accumulation in a given structure, S_V, is expressed in m³ of sediment accumulation. Values of S_V were divided by the maximum storage volume of the structure to obtain the relative infilling S_{VE} ($m^3\ m^{-3}$) which allows comparison in filling rates between dams of different size. The volume of sediment that accumulates in the sediment traps is expressed in m³ per surface area of its drainage basin (ha).

In order to convert the sediment volumes into sediment mass, the bulk density was calculated from 104 sediment samples taken in metallic cylinders of 100 cm³. The average bulk density (BD) of the sediment was determined at 1.43 ± 0.12 g cm^{-3}. To get a first estimate of sediment fluxes (t $ha^{-1}\ yr^{-1}$) in the micro-catchments, the sediment masses (t) were divided by the time period of observations (yr) and the contributing drainage area (ha).

Preliminary data from grain size analyses show that the sediment that is accumulated behind the wooden dams is slightly coarser than the topsoil material. Based on 5 samples of sediment, we estimated that the sand, silt and clay fraction in sediments is respectively 16.9, 53.2 and 29.9 %. Ten samples from topsoil material of the eroding slopes have a sand, silt and clay fraction of resp. 22.6, 55.0 and 22.4 %. This suggests that the dams are efficient in trapping the eroded material, as there are only slight differences in grain size between the sources of erosion and the deposition sites. Our data are not corrected for the trapping efficiency of the structures, and sediment fluxes should be interpreted accordingly. The global efficiency of the structures was evaluated by comparing the sediment accumulated in the dams and the sediment trapped at the outlet of the micro-catchments.

3.3.4 Monitoring of rainfall amounts

Two tipping bucket rain gauges (Davis Rain Collector) were installed to monitor rainfall intensity during the period February 2011 till August 2014. The rain gauges are situated in the Loreto catchment, in proximity of barren lands and arable lands. The resolution of the rain gauges is 0.2 mm and both gauges were installed at a height of 1.8 m. The rain gauge “P-Lo1” (barren land) was used for estimating rainfall in the major part of the study area, while “P-Lo2” (agricultural land) was used only to complete data gaps during short periods when P-Lo1 was damaged. The two rain gauges are located at similar altitude and at less than 1 km distance from each other.

One continuous time-series was constructed based on the data from P-Lo1, and the few data gaps (Table 8) where filled with the original data from P-Lo2. There is a gap in the rainfall series of 4 days during the period 22/03/2014 - 26/03/2014 and 28 days for the period 17/07/2014 - 14/08/2014 (Figure 9).

Table 8: Periods of time without data in rainfall.

Rain gauge	Missing data
P-Lo1	4/03/2011 - 22/03/2011; 22/09/2011 - 11/10/2011; 10/08/2012 - 14/08/2012; 22/03/2014 - 26/03/2014; 20/05/2014 - 10/06/2014; 17/07/2014 - 14/08/2014
P-Lo2	7/05/2011 - 4/08/2011; 13/10/2011 - 18/10-2011; 27/10/2012 - 29/11/2012; 11/01/2013 - 30/01/2013; 21/03/2014 - 26/03/2014

3.3.5 Data Analysis

The sediment yield data was normalized for the total area of each micro-catchment. Rainfall data was cumulated to a daily resolution and periods without data were filled with information of the secondary rain gauge (Lo2). Finally, the I_{30max} (mm) was calculated for each period of sediment monitoring. Regression analyses were used to explore relationships between evaluated variables.

3.4 Results

3.4.1 Vegetation cover of the five micro catchments

The two degraded catchments (DI and DT) have rather similar vegetation cover, with eroded, gullied land covering 59% and 75% of the area of catchment DI and DT respectively (Figure 28). The DI catchment is slightly larger than the DT catchment, and has *Eucalyptus* plantations with understory of shrubs in the upper part of the catchment. Given that the low density Eucalyptus forest only covers 6 % of the total catchment area (in the headwaters) and has a poor ground vegetation cover (21.4 %), it has minimal effect on sediment production and retention. The DF micro-catchment is mainly covered by *Eucalyptus* forest that has an understory of bushes and shrubs (61% of its surface area), although barren land is present in the upper part of the catchment (39% of its surface). In the agricultural part of the catchment, pastures cover 64 % of the surface area of the micro-catchment in AT and 19 % of the surface area in AI. Cropland is restricted to a small area in AT covering only 18% but near to the half of total area in AI micro catchment.

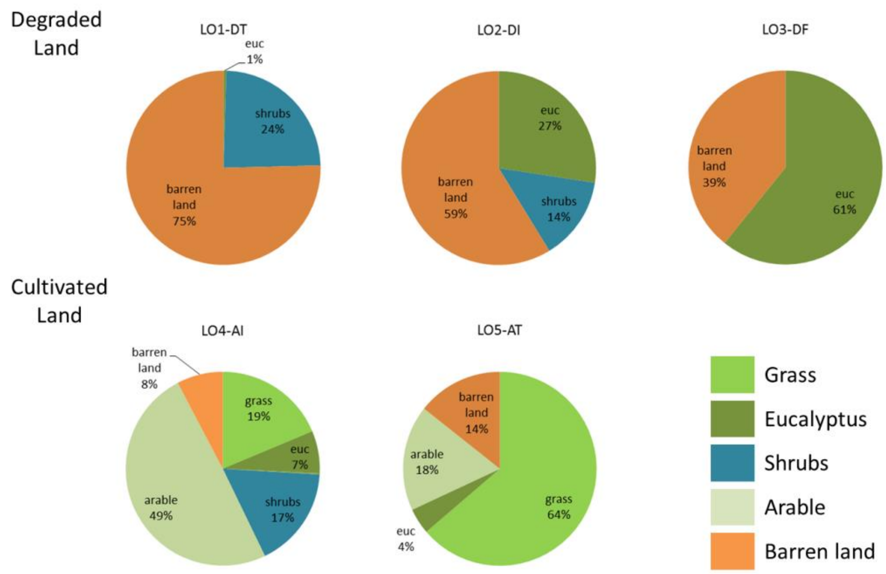

Figure 28: Vegetation cover for the five micro-catchments in degraded and cultivated land.

3.4.2 Rainfall characteristics

Rainfall data cover four years (Jan 2011 till Aug 2014). Rainfall amounts registered in the two gauges are very similar at daily time step. Small differences exist at hourly time step, for storm events. Figure 29 shows that the summer months of July, August and September are typically dry, and that the wet season covers the period from October to May. Only 8 % of all rainfall events have an intensity that is higher than 10 mm day $^{-1}$ and the events are mostly individual rain events bound by a period of no rainfall. The maximum rainfall intensity registered was 80.8 mm h^{-1}, and occurred during 30 minutes. This event has a recurrence time of 38 years.

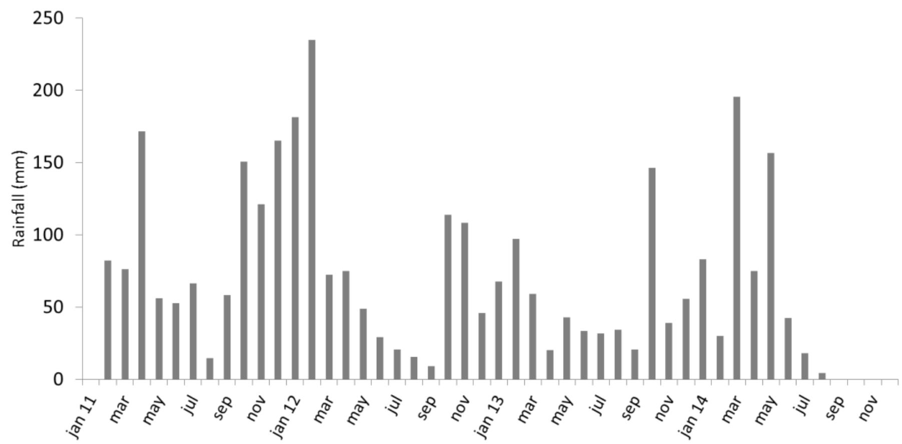

Figure 29: Monthly rainfall (mm) in Loreto catchment (Jan 2011-Aug 2014).

3.4.3 Sediment transfer in gully system

The accumulation of sediment behind the check dams is strongly associated with rainfall-runoff events. The results show that in all dams the sediment accumulation was very low until 05/10/2011 (280 days after construction), when the rainy season started and a significant increment in sediment accumulation is observed (Figure 30). Dam 11 located in the DI micro-catchment is the first one to fill (12/10/2011), almost 285 days after its construction. Dam 1, 7 and 9 are filled a few days later in 01/11/2011 (305 days after construction). Figure 30 nicely illustrates that the filling history of the dams can be very different.

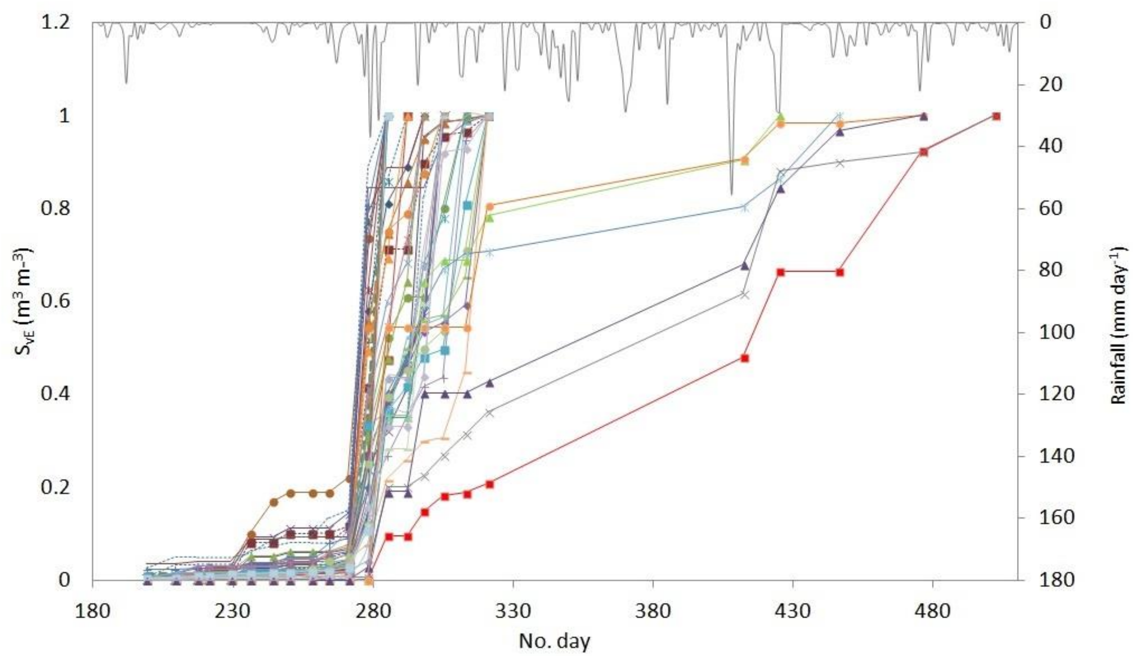

Figure 30: Cumulated sediment volume (m^3 m^{-3}) in the micro catchment DI and AI. The amount of sediment (m^3) is divided by the total capacity of the check dam (m^3) as to compare the filling history of the dams. The time correspond to number of days since the construction of the dams.

Figure 30 plots the sediment accumulation in dams in response to rainfall for the DI and AI micro-catchments. The surface hydrology is controlling the sediment deposition behind the dams, as the deposition of sediment occurs during flow events. After very high rainfall intensity events, low sediment accumulation rates were observed in the dams. This suggests that very strong surface runoff can remove sediment that has been deposited before.

To understand the spatial variation in sediment accumulation in the gully channel, we analyze the relation between the rainfall and sediment accumulation as a function of the spatial location of the dam. Figure 31 and Figure 32 plot the correlation coefficient between the rainfall (I_{30max}) and sediment measured in every check dam (S_V) as a function of the position of the dam in the gully system, for respectively the DI and AI micro-catchments. The latter is represented by the distance between the outlet of the catchment and the dam, with dams located close to the outlet of the catchment having low numbers and vice versa.

In the DI micro-catchment, the upper part of the catchment seems to be more responsive than the lower part. The same tendency can be observed for I_{30max}, where more distant dams show a better correlation between the amount of sediment accumulation and the I_{30max} of the preceding rainfall event (Figure 31). While the sediment accumulation in the most upstream dams can clearly be linked to the local hydrology of the most upstream part of the drainage basin, this is less the case for the downstream dams. Along the gully channel, the implementation of dams alters the local hydrology through e.g. re-infiltration of runoff water in sedimentation areas. Besides, the morphology of the gullies is often complex in its downstream part where runoff can be the result of concentrated runoff from several slopes due to the effect of flow convergence (Verstraeten et al., 2006).

In the AI micro-catchment (Figure 32), this tendency is not clear. One possible reason is the fact that the surface hydrology of the catchment is highly altered by construction works, roads and irrigation infrastructure in the micro- catchment AI. Infrastructure can alter the flow connectivity of the gully networks. In addition, the AI micro-catchment has a larger number of dams in comparison to the DI micro-catchment, and some dams are located in artificial flow paths (footpaths).

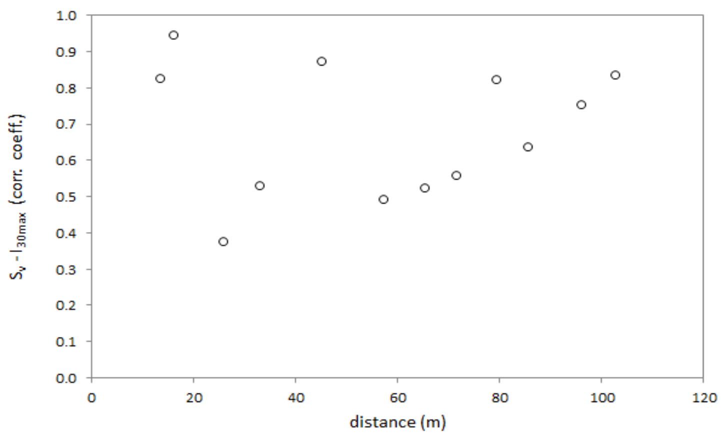

Figure 31: Pearson coefficient of correlation (r) between the amounts of sediment accumulated behind each check dam (S_V) and the I_{30max} of the rainfall (axe Y) related with the position of check dam in the DI micro catchment (distance between sediment trap and every check dam).

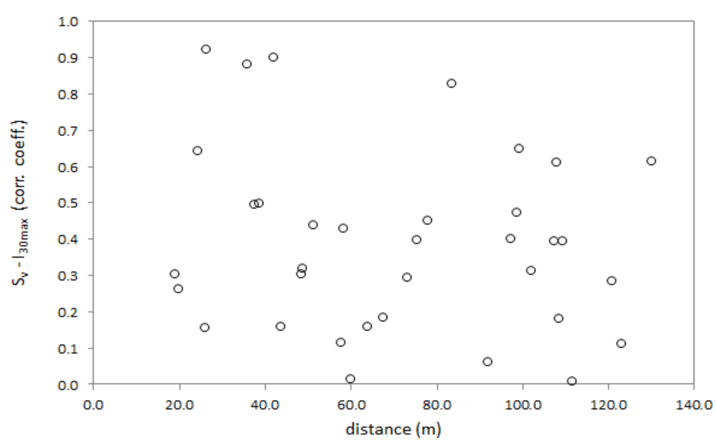

Figure 32: Pearson coefficient of correlation (r) between the amounts of sediment accumulated behind each dam (S_V) and the I_{30max} of the rainfall (axe Y) related with the position of check dam in the AI micro catchment (distance between sediment trap and every check dam).

3.4.4 Effectiveness of dams to trap sediment

The sediment traps at the outlet of the micro-catchments give us a good idea of how effective the dams retain sediment within the gully channel. By comparing the micro-catchments with/without dams (AT vs. AI, DT vs. DI micro-catchments), it becomes possible to quantify the amount of sediment that can be retained by the small structures in the gully channels.

We observe a positive and strong correlation (R^2 = 0.92) between the sediment accumulation data in the sediment traps in the treated (DI) and non-treated (DT) micro-catchment (Figure 33). This shows that the two micro-catchments are responding to the same trigger, i.e. high intensity rainfall events, and that the sediment accumulation data are consistent. Figure 33 also shows that the amount of sediment transfer is very different between the micro-catchments. The sediment accumulation rates in the trap in the DT catchment is – on average – 3.5 times higher than the sediment accumulation rate in the DI catchment, illustrating the efficiency of the soil and water conservation measures in the DI to reduce sediment export.

Figure 34 shows the correlation between the sediment filling in the flow barriers and the sediment accumulation in the sediment trap for the DI micro-catchment. There is a positive and strong correlation between the amount of sediment trapped in the channel and the sediment exported from the catchment. This highlights that the data on sediment mobilisation and export are based on robust measurements, and suggests that the sediment mobilisation within the channel and export at the outlet of the micro-catchment is largely responding to a common external trigger - high intensity precipitation events. The correlation between sediment filling of flow barriers and accumulation in the traps is highest for the dams that are located close to the sediment trap (Figure 34).

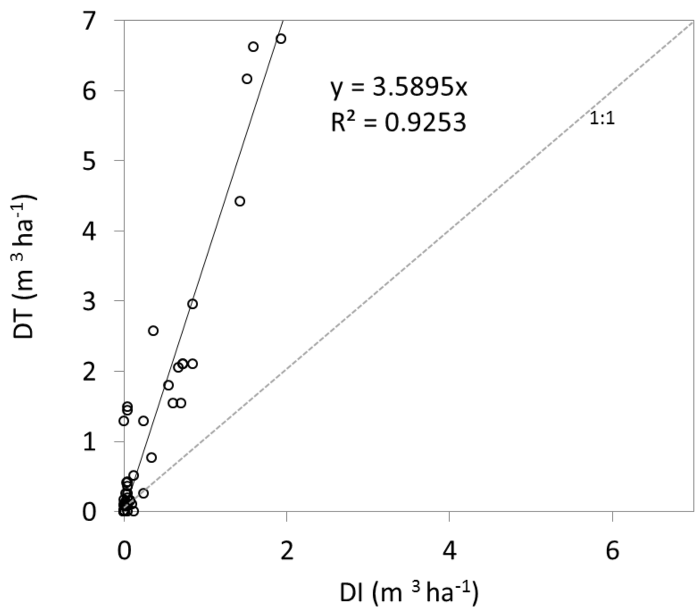

Figure 33: Scatter plot between sediment accumulated in sediment trap DT (m^3 ha^{-1}) and sediment accumulated in DI (m^3 ha^{-1}) for the total period of monitoring.

The effect of the dam construction on the sediment transfer in the gully systems can also be observed in the Figure 35. Here, the accumulation of sediment in the traps at the outlet of the DT, DI and DF micro-catchments are plotted as a function of time. The graph shows that sediment export is strongly associated with the rainfall patterns. The accumulation rates in the sediment traps are highest during and shortly after high-intensity rainfall events. Interestingly, the sediment export in the micro-catchments DI and DF where soil and water conservation measures are installed is systematically lower than the sediment export in the non-treated micro-catchment (DT). In the forested micro-catchment, only 6 rainfall event triggered sediment mobilisation in the gully channel and sediment accumulation in the trap. Results from AT are not given in the graph as only two events led to (very low amounts of) sediment accumulation in the traps (Figure 35).

From Figure 35, it is also clear that the SWC measures have a long-term impact on sediment mobilisation and export in the degraded catchments. Even when the flow barriers are entirely filled with sediment, the sediment export in the treated micro-catchment (DI) stays consistently low. Our data do not show an increase in sediment export after filling of the flow barriers, indicating that they have a long-lasting effect on the sediment mobilisation within the gully channels. To analyse the effect of the filling of the flow barriers on the sediment export, we analysed the relationship between the sediment trapped in DI and DT (m3 ha) before and after the fillings of the flow barriers. The R^2 values of the regression analysis are 0.97 and 0.91 for the period before and after the dams filling respectively, suggesting that the flow barriers have a lasting impact on sediment retention after filling.

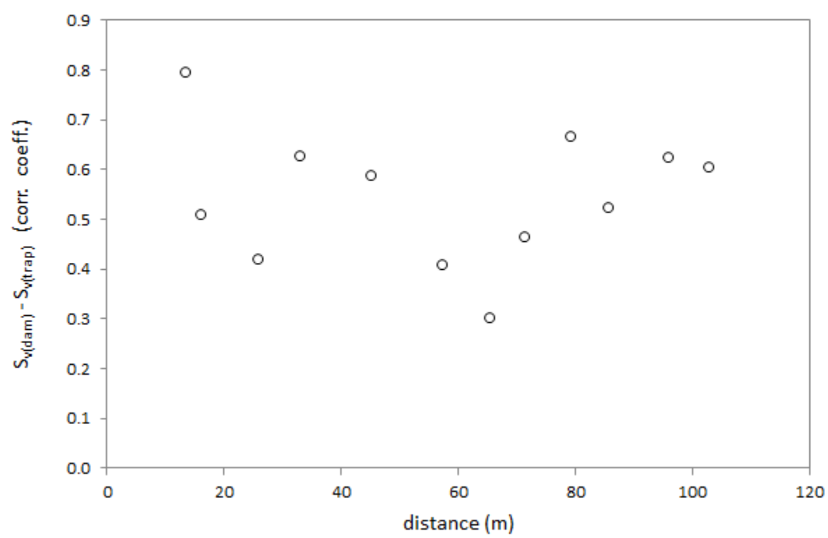

Figure 34: Pearson coefficient of correlation (r) between the amounts of sediment filling of flow barriers and the amount of sediment (S_v) accumulated in the sediment trap at the outlet (Y-axis), plotted as a function of the spatial location of the flow barrier or so-called dam (X-axis) in the DI micro catchment.

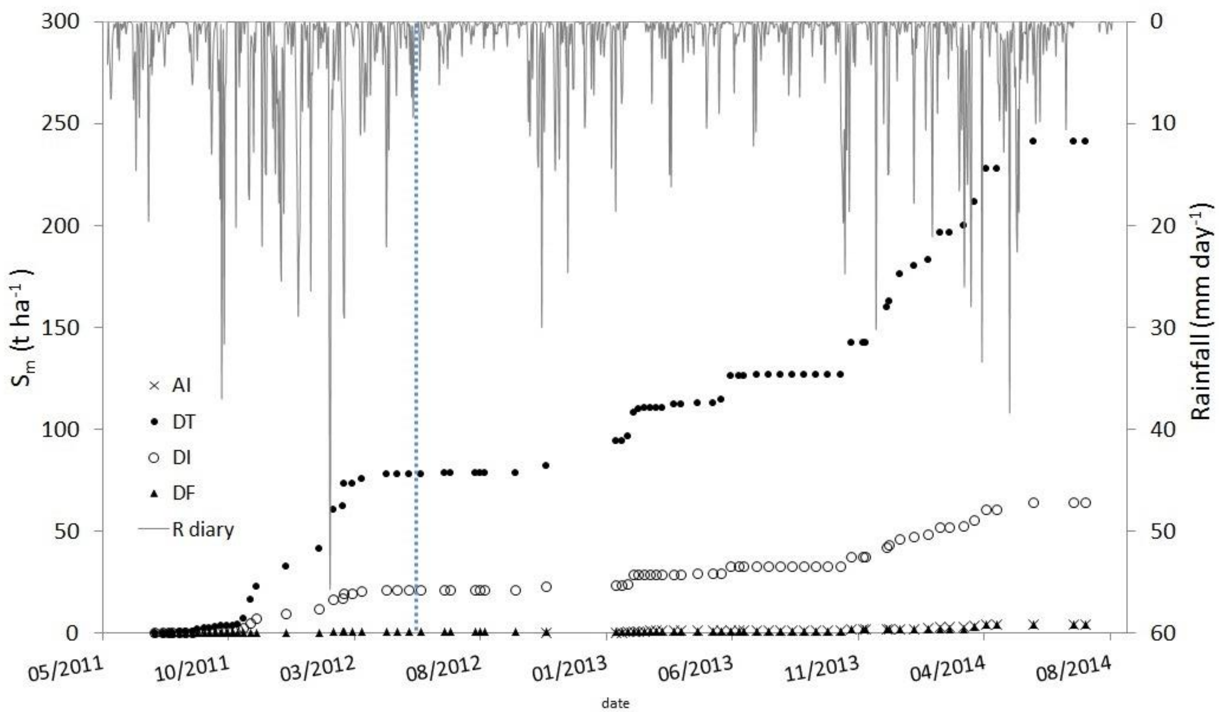

Figure 35: Sediment retained in traps (cumulative) at the outlet of the treated micro-catchments DI (black circles) and DF (black triangles) and the non-treated micro-catchment DT (black dot). The vertical blue dotted line gives the period when all dams in the DI micro-catchments are filled with sediment.

3.4.5 Sediment budgets of the micro-catchments

Based on the data from the sediment filling in flow barriers or so-called dams and the data from the sediment accumulation in the sediment traps, it is possible to establish a first-order estimate of the sediment budget of the micro-catchments. The data on flow barrier filling allow us to quantify sediment deposition within the gully channels, and the sediment accumulation data from the traps to quantify sediment export from the micro-catchments. Given that the data are not corrected for trapping efficiency, the estimates of sediment export are minimum estimates. By comparing the sediment budgets of the five micro-catchments with different soil and water conservation treatments, it is possible to determine the efficiency of the treatments on sediment mobilization and export.

The sediment trap data show that the sediment export highly depends on the sediment production and sediment retention in the micro-catchments. Based on the sediment accumulation rates in the traps, a first estimate of the specific sediment yield can be made for the five micro-catchments (Table 9). The sediment export is highest in the degraded land, with the highest values of 89.1 t ha^{-1} yr^{-1} for the degraded reference site (DT) where no SWC measures were implemented, and significantly lower values of 24.34 t ha^{-1} yr^{-1} for the treated micro-catchment, DI. This roughly corresponds to a surface lowering of resp. 6.22 and 1.7 to mm yr^{-1} for degraded catchments affected by gully erosion (Table 9).

The micro-catchments that have a dominant land cover of grasslands and croplands, AT and AI, have much lower rates of sediment export of respectively 0.23 and 2.49 t ha^{-1} yr^{-1}, corresponding to erosion rates of approximately 0.02 and 0.17 mm yr^{-1} (Table 9). For the AT micro-catchment, sediment accumulation registered in the trap was very low during the period of observations. The value of 0.23 t ha^{-1} yr^{-1} corresponds to a 2 unique measurements during heavy rainfall events, when sediment was deposited in the gully channel.

Table 9: Sediment budget for the five micro-catchments. The 1st column gives the sediment export based on sediment accumulation rates in traps. The 2nd column gives the sediment trapped in the gully floor based on data of flow barrier filling. The total amount of sediment produced in the catchment is given in the 3rd column, and is the sum of the flow barriers filling and sediment trap accumulation. This value is also expressed as the catchment-wide erosion rate (mm yr^{-1}). The values between brackets do not account for the sediment deposition in the barriers. Note that these values correspond to minimum estimates, as no correction is made for the trap efficiency of the structures.

	Traps (t ha^{-1} yr^{-1})	Dams (t ha^{-1} yr^{-1})	Total (t ha^{-1} yr^{-1})	Erosion rate (mm yr^{-1})
DT	89.1		89.1	6.22
DI	24.34	81.5	105.9	5.69 (1.7)
DF	1.18		1.18	0.08
AI	2.49	15.7	18.2	1.1 (0.17)
AT	0.23		0.23	0.02

The degraded micro-catchment that was restored with forest plantations (DF) has a very low sediment export of 1.18 t ha^{-1} yr^{-1}, corresponding to a catchment-wide erosion rate of 0.08 mm/yr. In the micro-catchment with forest plantations (DF), export of sediment was only registered during six events. This shows the effectiveness of *Eucalyptus sp.* in reducing sediment mobilization and export.

The data on sediment export and erosion rates are within the range of earlier measurements by (Molina et al., 2007), who reported specific sediment yield values of 0.26 to 151 t ha^{-1} yr^{-1} for small catchments in the Paute river basin. The latter values are established based on field measurements of reservoir infillings for catchments of 1 to 250 km^2, and integrate over a period of 10 to 40 years. For comparison, the average specific sediment yield for the Paute river basin is estimated at 6.50 t ha^{-1} yr^{-1} (Jerves, 2001). The large variation in specific sediment yield is related to differences in vegetation cover, land use and lithology (Molina et al., 2009).

If we assume that our measurements are a good and reliable estimate of sediment mobilization within the catchment, we can use them to assess the effectiveness of the structures on sediment retention. In the DI micro-catchment, the dominant erosion processes is gully erosion. From the total amount of sediments that is mobilized within the micro-catchment (105.9 t ha^{-1} yr^{-1}), about 77% of sediment is trapped in the dams (Figure 36). In the AI micro-catchment, the sediment export that is derived from the sediment trap accumulation rates is only 2.49 t ha^{-1} yr^{-1}.

The flow barriers that were implemented in the agricultural micro-catchments are very effective sediment traps, and are able to trap about 86% of the sediment that is mobilized within the AI micro-catchment.

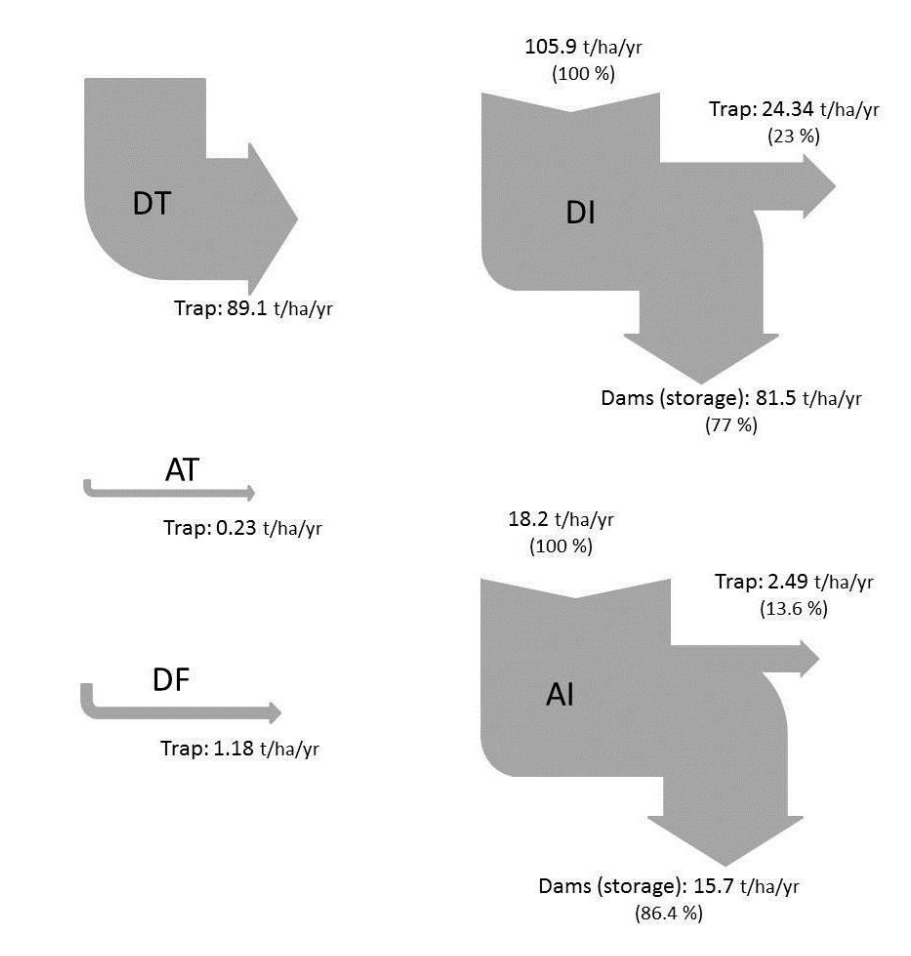

Figure 36: Simplified sediment budget for water erosion for the five micro-catchment.

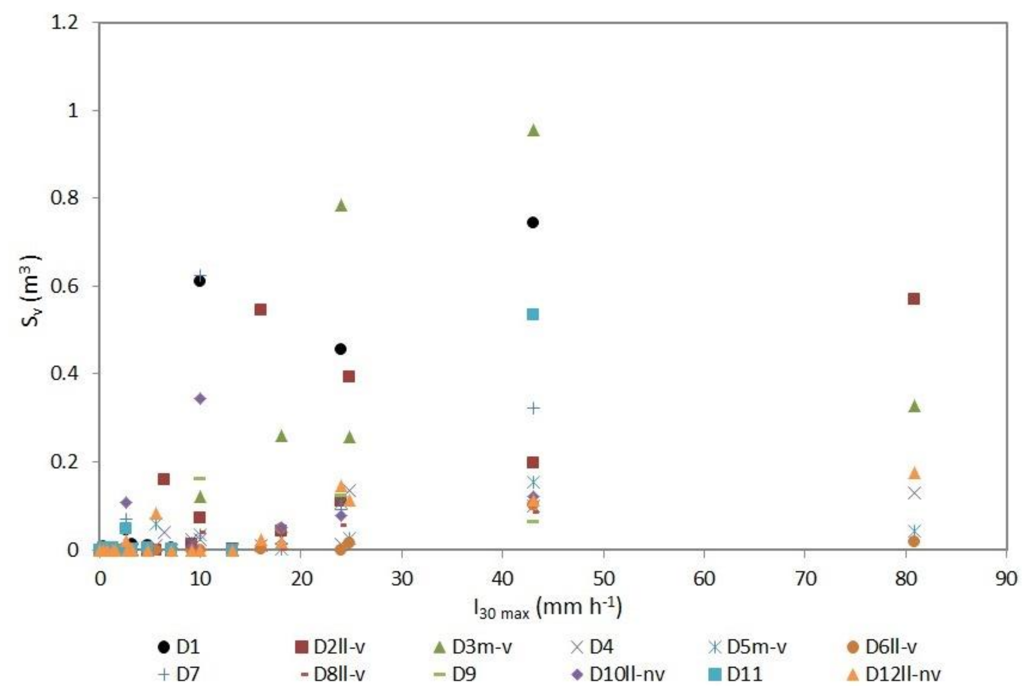

Figure 37: Relationship between the I_{30max} (mm h^{-1}) of the rainfall event and the corresponding amount of sediment infilling in flow barriers (m^3). All data are derived from the DI catchment.

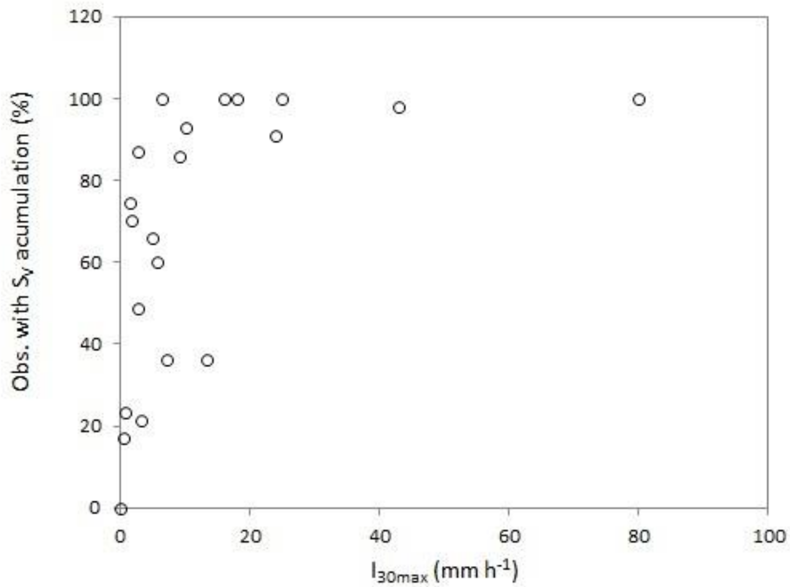

Figure 38: Percentage of the flow barriers where sediment accumulation was measured, as a function of the rainfall intensity. A value of 100 % means that for a determined I_{30max}, all dams register sediment accumulation.

3.5 Discussion

3.5.1 Sediment connectivity during rainfall events

Our data show that the largest amount of sediment is mobilized in the highly degraded micro-catchments, DT and DI. When comparing the standardized data of sediment filling of flow barriers in the treated micro-catchments, it is evident that the dam structures are very efficient in trapping eroded material and reducing the sediment export from the degraded catchments (Figure 30 and Table 9). Dams located in the lower part of the micro-catchments are first filled with sediment (Dams: 7, 8, 9, 10 and 11), except dam 12. This is also what one would expect, as the total surface area that contributes water and sediment to the downstream part of the gully system is higher compared to upstream sections. However, there are other factors that can influence the response of the gullies and the time of filling such as the density of gully channels, the distribution of the vegetation in the micro-catchments, and the runoff characteristics.

The rainfall intensity (and runoff volume) is controlling the amount of sediment that is trapped within the gully system. In badlands, the infiltration capacity of the soils is very low, and a rainfall intensity of 25 mm h^{-1} can exceed the infiltration capacity of the barren land and generate Hortonian overland flow (Molina et al., 2007). Most of the rainfall events have a low intensity, and do not result in runoff and sediment mobilization in the gully system. Only 8 % of all rainfall events has exceeded 10 mm day^{-1} during the monitoring period (see Chapter 4). This is also observed in Figure 37 where rainfall lower than I_{30max} of 10 mm h^{-1} do not lead to significant deposition in the gully floor, probably because the sediment production and transport is limited during low intensity events.

Figure 38 suggests that the rainfall intensity during the event needs to reach 16 mm h^{-1} or higher to provoke flow and mobilization of sediment in the gully channel. The filling of flow barriers happens during rainfall events of medium intensity (10 mm h^{-1} < I_{30max} < 20 mm h^{-1}). A rainfall event with I_{30max} of 43 mm h^{-1} was responsible for most of the sediment deposition in the gully system of the DI micro-catchment (Figure 37). This event was followed by an event with an I_{30max} of 24 mm h^{-1} that also provoked sediment deposition in all dams.

During very high intensity storm events (I_{30max} > 50 mm h^{-1}), Hortonian overland flow in the badlands becomes important and mobilizes large volumes of water and sediment. The maximum I_{30max} registered in the study area was 80 mm h^{-1}, that corresponds to a daily precipitation of 55.6 mm. This event did not lead to the highest amounts of sediment filling in the dams, probably because the retention capacity of the dams was already strongly reduced as a result of deposition during previous events. This assumption could be confirmed by the fact that the amount of sediment accumulated in the three sediment traps at the outlet of catchments was the largest of all registers.

The temporal variation in sediment mobilization and export rates is likely to be related to the precedent rainfall and antecedent soil moisture content. During the rainy season, the continuation of individual rainfall events can increment the soil water content considerably, and reduce the infiltration capacity of the soil (Molina et al., 2009). Under these conditions, overland flow can occur during rainfall events of lower intensities.

The sediment yield in forested catchment is very low and could be compared with the natural benchmark values obtained from cosmogenic nuclides that provide values of 1.5 – 1 t ha^{-1} yr^{-1} (Vanacker et al., 2007b). In the plantation forests, there exists a well-established ground vegetation of herbs and shrubs that protects entirely the soil surface. Although the forest covers only 61 % of the catchment area, it has a drastic impact on the reduction of sediment export. A key element is that the forest is situated in the lower part of the catchment, so all sediment generated in the upper part is retained here.

After all check dams were filled, no major change in the rate of sediment export was observed for the micro catchments DI. The correlation between the sediment measured in the traps at the outlet of the DI and DT micro-catchments is only slightly lower for the period after filling of dams (R^2 = 0.91) compared to the period during when the dams were trapping sediment (R^2 = 0.97). Although all dams are filled with sediment, sediment trapping in the gully channels is still ongoing. The sediment deposits trigger the growth of pioneer plants in the gully channel. The dense root systems of the pioneer species contribute to further stabilize the gully channels, and an upstream extension of the sediment accumulation area is often observed. Besides, the dams lower the slope of the gully channel, and reduce the energy of the runoff. The long-term effectiveness of the structures will largely depends on the maintenance of the physical structures and the vegetation cover.

In the case of AI and AT, it was not possible to establish a quantitative evaluation because in these agricultural catchments there are constant alterations and changes i.e. addition of new roads, cleaning of channels, changes in the limits of croplands. Also, it was not possible to construct sediments traps until October 2012.

3.6 Conclusions

Our data show that soil and water conservation treatments such as vegetation restoration and/or construction of flow barriers in gully systems are very effective in reducing sediment transport in gully systems. The flow barriers that were built in active gully systems are effective in trapping sediment and can significantly reduce the sediment export to the fluvial systems: in the degraded and agricultural micro-catchment, about 78% and 87 % of the eroded material is trapped in the flow barriers and not exported to the fluvial network. Besides, forestation of active gully systems with rapidly growing exotic species such as Eucalyptus has a very positive effect on the stabilization and restoration of the badlands, and effectively reduces the sediment export. Rainfall intensity controls runoff volumes and sediment mobilization: during high intensity rainfall events, the conservation works seem to be less efficient in trapping sediment, although a detailed analysis is necessary to make conclusive statements.

Chapter 4 : Impact of soil and water conservation treatments on the hydrological response of degraded catchments

Abstract

Reforestation programs with *Eucalyptus* and *Pinus* have been implemented in many regions of Andean mountains with different purposes such as protection and restoration. Some studies report a reduction in water yield as consequence of forestation. The positive effect of vegetation restoration and bioengineering techniques on sediment retention and erosion reduction in active gully channels has been quantified, but less is known about the potential impact of catchment restoration schemes on surface hydrology. Particularly, the effect of restoration on the rainfall-runoff behavior of small ephemeral channels in highly degraded environments is understudied.

In this chapter, we will study the impact of catchment restoration schemes - that involve bioengineering works in the channels and vegetation restoration on the hill slopes – on the hydrological response of ephemeral channels. The analysis is based on a five-year intensive monitoring campaign of 5 micro-catchments that are located in the central part of the Jadan river basin in southern Ecuador. Two micro-catchments are located in the agricultural zone, and three micro-catchments in the highly degraded zone where active gullies are present. In each zone, there is one reference and one or two micro-catchments where conservation works

were implemented. Rainfall and surface runoff were measured in all catchments using pluviometers and pressure sensors installed in hydraulic weirs. The results suggest that the response time of the restored catchments (DI) is bigger when compared to the reference catchments (DT) and more similar to agricultural lands. However, there is a large difference in the hydrographs between the reference and restored catchments in the highly degraded sites, with a more pronounced peak flow in the reference catchments (DI). Our data show that the peak flows could be reduced by resp. 73% and 56%, as a result of the implementation of restoration techniques in active gully channels and vegetation restoration by plantation forest. This study offers the basic insights in the potential changes in hydrological response of degraded catchments after implementation of restoration schemes.

4.1 Introduction

Land use changes might alter the hydrological regulation in the tropical Andes. During the last decades, there have been notable changes in land use in the Inter-Andean basins. While deforestation (conversion of native forests to agricultural land) continues to be an important process at the agricultural frontiers, there is a reversing trend of vegetation regeneration and active restoration close to urban centers. The use of vegetation for slope stabilization or erosion control is denominated bioengineering (Morgan and Rickson, 1994) and any part of vegetation (live or death) has been used for these purposes. In degraded lands, bioengineering techniques can be effective in reducing water runoff as suggested by e.g. (Chirino et al., 2006; Descheemaeker et al., 2006; Xu et al., 2008), and reducing soil erosion and sediment yields (Burylo et al., 2007; Chirino et al., 2006; Cohen and Rey, 2005; Erktan and Rey, 2013; Marden et al., 2014; Rey, 2009; Rey, 2004; Rey, 2003; Rey et al., 2005b; Rey and Burylo, 2014; Zheng, 2006). The latter has also been documented for the southern Ecuadorian Andes by e.g. Vanacker et al. (2003) who showed that vegetation restoration effectively reduced soil erosion rates(Molina et al., 2012; Vanacker et al., 2007b). Experiments of rainfall runoff mechanisms provided the evidence that runoff generation in degraded Andean ecosystems is mainly controlled by land use and land management (Molina et al., 2007). The potential impact of land use change on the rainfall-runoff response of hydrological units is less investigated, with exception of the work on annual water yield by (Molina et al., 2015; Molina et al., 2012).

In Ecuador, as in many Andean tropical regions, intensive reforestation and afforestation programs have been implemented during the last four decades. Exotic species such as *Eucalyptus* and *Pinus* have been widely used for forest plantations (Vanacker et al., 2003). *Eucalyptus globulus* was introduced in Ecuador in the XIX century and is one of the most popular exotic species in the Inter-Andean basins (2000 – 2900 m a.s.l.) because of its rapid grow, adaptability and suitability for timber production and fuelwood. The commercial use of eucalypt and pine trees in forests plantations for the production of pulp and wood has been questioned because of potentially large impacts on soil quality and surface hydrology.

Earlier work by (Farley et al., 2008; Farley and Kelly, 2004) has shown that pine plantations at high altitudes (2800 – 3800 m a.s.l.) can have negative impacts on chemical and physical soil properties, more specifically on the soil water retention and soil C content. High-density plantations of e.g. pine can alter the hydrological regulation of streams through increase in the base flow, storm flow and sediment concentrations after harvesting and burning (Waterloo et al., 2007). (Buytaert et al., 2007) also showed that exotic forest plantations in high elevation páramo ecosystems affect the water yield and increase evapotranspiration, as exotic forest species present a high water consumption during the first stages of their life cycle. Similar results were obtained by Bruijnzeel (2004), Farley et al. (2005) and Molina et al. (2015).

Figure 39: Difference in environmental impact between forest plantations in high-altitude native grasslands (upper panel) and in highly degraded environments (lower panel).

However, the vast majority of researchers have concentrated their efforts on quantifying the impacts of forest plantations on soil quality and water regulation in high-altitude native grasslands or so-called páramo ecosystems, and their conclusions representatively demonstrate the consequences of converting native tundra ecosystems to commercial forest plantations. Transposing the results that were obtained in páramo ecosystems to the tropical Andean region is not viable, given the complex and heterogeneous land use pattern that currently exists in the Inter-Andean basins Figure 39.

Forest plantations have been an integral part of vegetation restoration and catchment restoration schemes in the period 1980s-2000s. Areas that were affected by very severe soil degradation and where the fertile soil mantle was practically stripped by soil erosion have been restored by active restoration schemes. Eucalyptus trees have been widely used in restoration programs, as very few tree species can adapt in these extreme conditions. Field data have shown that the establishment of Eucalyptus forests in badlands areas can be effective in erosion control, as the dead leaves and branches increase the surface cover and retard the generation of surface runoff (Molina et al., 2009). Controlled field experiments in gully channels demonstrated that vegetation restoration in active gully channels reduces the hydrological connectivity and the transfer of runoff water downstream (Molina et al., 2009). Factors as forest density, composition and age, and the altitude, climate, and soil type of the forest stand can have an effect on the effectiveness of forest plantations.

The objective of this research is to investigate the potential impact of catchment restoration schemes on the hydrological response of ephemeral streams. To do so, we will use hydro-meteorological data from five micro-catchments in the Ecuadorian Andes where different restoration schemes were implemented. The restoration schemes include vegetation restoration through reforestation with exotic species, and gully restoration with live check dams. In ephemeral streams, few extreme events in a year are responsible of a major part of erosion (Coppus and Imeson, 2002). The relationship between rainfall and runoff processes can change significantly during the course of a storm. For that reason, it is important to go beyond the analyses of annual water yields and look into changes in the hydrological response of the micro-catchments at level of individual rainfall events (event time scale). From an analysis of the hydrographs of individual storm events, we will look into differences in the response, duration, and flashiness of the runoff events between the five catchments with different treatments.

4.2 Materials and Methods

4.2.1 The study area

The Loreto catchment (a tributary of the Jadan River) is located in the south of Ecuador and belongs to Paute river basin Figure 40. This belongs to a mountainous region in the Andes with an altitudinal range of 2610 to 3100 m a.s.l. The principal river Loreto presents a low flow during most of the year. Many of its tributaries are ephemeral, and their discharge increases rapidly during the rainy period with a rapid response to the rainstorms. The bimodal rainfall regime presents a wet period in April and December and the dry period from June to September, with a mean annual precipitation of 800 to 900 mm yr^{-1}.

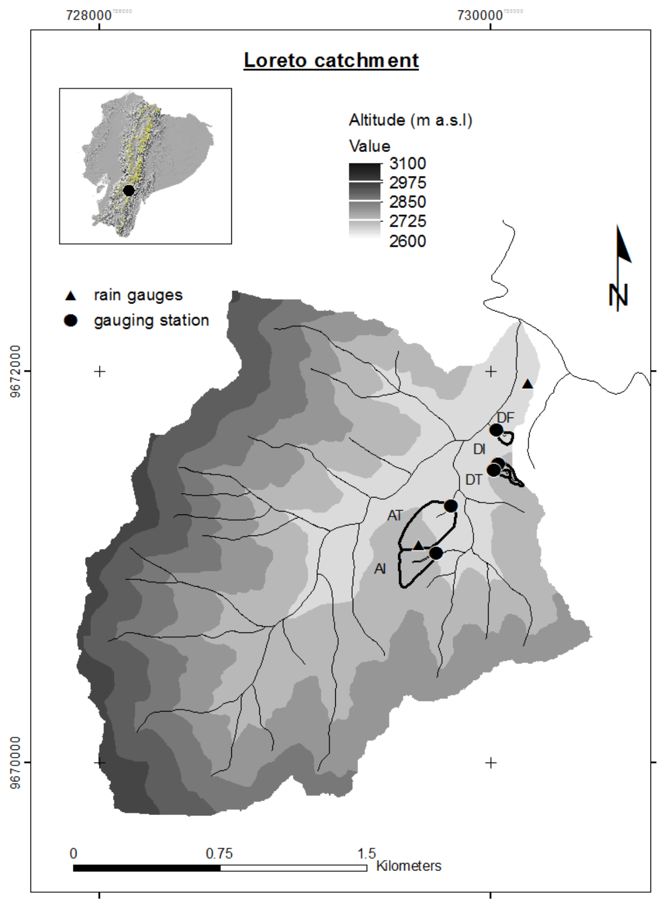

Figure 40: Map with the study zone including experimental micro catchments and the location of the monitoring stations in Loreto catchment.

The geology of the micro-catchments is characterized by sedimentary deposits (Middle to Late Eocene) of poorly consolidated rocks and poorly sorted conglomerates and sandstones and red and purple siltstones (Hungerbühler et al., 2002). There several rocky outcrops in the area as a result of intense water erosion. The sedimentary rocks are highly weathered and very friable. The morphology is steep, and flat areas only occur in the flood plains and bottom of river valleys. Soils are shallow, rich in clay minerals, and very low in organic matter (see Chapter 2). They have a low water holding capacity, and can be classified as Leptosols and Regosols in the bad lands areas. Cambisols, Luvisols and Vertisols (ISSS Working Group RB, 1988) occur in the zones that are less affected by water erosion, with more flat topography.

The native forest has practically disappeared, and some relicts of secondary forest can be observed in very small patches near to ephemeral channels and rivers. The predominant vegetation types are shrubs and herbs as a result of natural regeneration of vegetation after abandonment. Plantations of *Eucalyptus* and *Pinus* are present since the late 1960s. Agricultural land is restricted to gentle slopes, and grasslands and plots with corn can be observed.

4.2.2 Experimental design

The five experimental micro-catchments (Figure 40) are located in Jadan basin that belongs to the Paute river basin. The latter is of great importance for the region and the whole country in terms of water resources. In degraded land, three micro-catchments were selected: (DI) catchment with gully restoration, (DT) with active gullies and (DF) vegetation restoration with *Eucalyptus* plantations. Two micro-catchments were selected in the agricultural area: (AI) with gully restoration and (AT) without any restoration. The main differences between the 5 micro-catchments (Table 10) are land use and vegetation cover, and there are no major differences in climate, topography and geology (see Chapter 3).

4.2.3 Bioengineering techniques and forest characteristics

Three of the five micro-catchments were treated with catchment restoration measures, and the remaining two catchments were kept as reference sites. This includes vegetation restoration and regeneration on the slopes and restoration of the active gully channels. In the highly degraded site, one micro-catchment (DI) was restored by vegetation regeneration and construction of small check dams in the main gully channel. A similar treatment was given to one agricultural catchment (AI). The dams were made of wooden poles of *Eucalyptus*, and some of them are made of recycled tires and wooden poles. The construction of dams started in July 2010.

Table 10: Main characteristics of the five micro-catchments, including the surface area (ha), the length of the gully channel, L (m), and the mean slope gradient of the gully channel, S (m/m).

micro-catchment	Surface (ha)	Land cover (vegetation type)	Condition	L (m)	S (m/m)	Soil
T	0.195	Shrubs, barren	gullies	84	0.27	Regosols, Leptosol
DI	0.415	Shrubs, barren	gullies with wooden dams	179	0.20	Regosols, Leptosol
DF	0.414	Eucalyptus, Shrubs	gullies with Eucalyptus	94.7	0.42	Regosols, Leptosol
AI	2.410	Arable, grass, barren	arable land with wooden dams	209	0.19	Cambisols, Leptosol, Luvisols
AT	4.711	Arable, grass, barren	arable land	340	0.21	Cambisols, Leptosol, Luvisols, Cambisols

In addition, in the highly degraded site, one micro-catchment (DF) that was restored by forest plantations was selected. The plantation of Eucalyptus trees was implemented after 1963, and trees were planted at very low density. Nowadays, about 50 years after the first plantings; there is a dense understory of herbs and shrubs. About 60% of the surface area of the DF micro-catchment is covered by *Eucalyptus* forest, and the remaining 40% is covered with shrubs.

4.2.4 Monitoring and processing

Two tipping bucket rain gauges (Davis Rain Collector) were installed to monitor rainfall amount and intensity. The resolution of the rain gauges is 0.2 mm and both gauges were installed at a height of 1.8 m. The rain gauge “P-Lo1” was located near to the outlet of Loreto micro catchment where the landscape is characterized by the presence of gullies. The second rain gauge “P-Lo2” was located between the AI and AT micro-catchments in the agricultural land (Figure 40).The distance between the two rain gauges is 990 m. The rain gauge P-Lo1 presents very few data gaps, and was used to reconstruct a continuous time series of rainfall data for the area. For the analysis of individual events the data from more nearest was used specially with high rainfall intensities. The gaps in the rainfall record of P-Lo1 were filled based on the observations of "P-Lo2" using data without any correction. The periods with gaps in P-Lo1 were: 4/03/2011 - 22/03/2011 (18 days); 22/09/2011 - 11/10/2011 (19 days); 10/08/2012 - 14/08/2012 (4 days); 22/03/2014 - 26/03/2014 (4 days); 20/05/2014 - 10/06/2014 (21 days); 17/07/2014 - 14/08/2014 (28days) (Figure 9).

The runoff was measured at the outlet of the five micro-catchments. A hydraulic weir (metallic San Dimas flume) was constructed Figure 41, and installed in the five sites. In cylindrical tubes connected with the main structure (Figure 42), one or two divers pressure sensors (Schlumberger) were installed. Additionally, 3 baro-divers were used for atmospheric compensation, and were located in three sites (barren land, arable land

and forested land). Water pressure recordings were made at 1 (february 2011 – march 2012) and 5 (april 2012 to june 2014) minutes resolution during a period of 41 months.

Figure 41: Flume in AI micro-catchment: a) during and b) after an event.

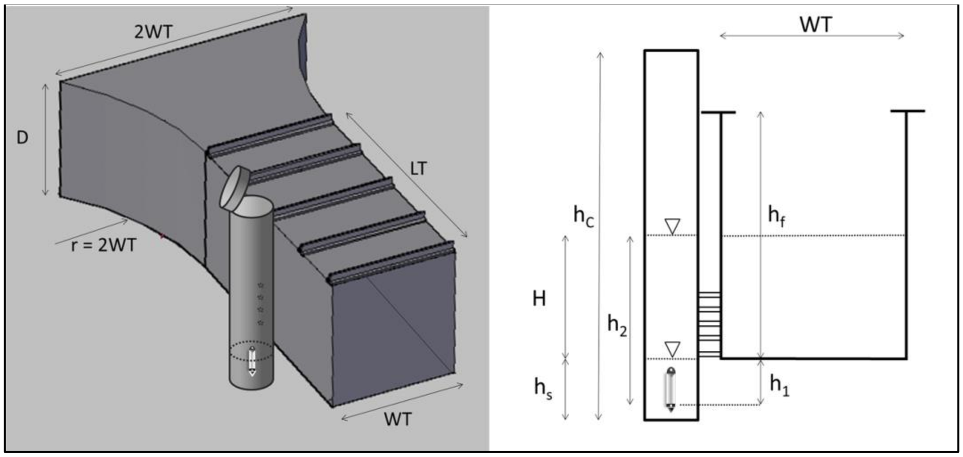

Figure 42: San Dimas flume cross scheme with external cylinder for diver sensor.

The San Dimas equation was used to convert the water heights to discharge values: (Eq. 2), (Eq. 5).

$$Q = 6.35\,.W^{1.04}.H^{(1.5-n)} \qquad \text{(Eq. 4)}$$

$$n = 0.179.W^{0.32} \qquad \text{(Eq. 5)}$$

where, Q is discharge ($ft^3\ s^{-1}$), W is the flume's width (ft), H is the water height (in). The unit discharge was obtained by dividing the values by the catchment area (ha). In the text, tables and figures, the discharge is expressed in mm $5min^{-1}$ in order to have data on the hydrological response that are independent of the catchment area and that allows us to compare the hydrological response of the five catchments directly.

Because of temperature fluctuations during the day (evaporation) and external factors (such as sediment accumulation within the cylinder), there is a considerable amount of noise on the measurements of the water heights by the pressure sensors. To facilitate the data analysis, we extracted from the time series of pressure heights the segments that correspond to well-identified rainfall events. The maximum intensity of the rainfall during 30 minutes, I_{30max} (mm h^{-1}), was determined for each event.

The rainfall series were aggregated to 5 minutes and also the resolution of water level series was transformed from 1 minute to 5 minutes in order to reduce the noise in the signal. Given that the catchments are drained by ephemeral streams, we can assume that the streams only have flow during and shortly after an intense rainfall event. As such, for the interpretation of the pressure levels of the divers, we can safely assume that the reference water level in the flume is zero at the start of a rainfall event. Shortly after the event, the water level will eventually return to this reference value, 0.

4.2.5 Data processing and analysis

The micro catchments in the study zone are drained by ephemeral streams. For that reason, it is not possible to obtain a continuous data set for stream flow being necessary to select specific rainfall events that produce measurable water level in the gauging stations. For the analysis, 15 rainfall events were selected based on the total daily rainfall (R_{tot}). A previous analysis showed that events with R_{tot} higher than 10 mm day^{-1} produced adequate information. However 5 events with R_{tot} less than 10 mm were included in the analysis to characterize also the events with low intensity considering that only 8 % of total events during 3.5 years were higher than 10 mm day^{-1}. The availability of water level data also limited the number of events (malfunctions in sensors and vandalism).

In order to analyze the effect of the treatment on the hydrological response of the micro-catchments, we derived five hydrological indicators that are indicative of the catchment's response to rainfall.

We have calculated the runoff coefficient, RC (%), as follows:

$$RC = \frac{D}{R_{event}} . 100 \qquad \text{(Eq. 6)}$$

where D is the total discharge (mm) and R_{event} (mm) is the total rainfall of the event.

Furthermore, based on the hydrographs we calculated the maximum discharge, MD (mm 5min^{-1}), the time to peak, TP (min), the slope of the recession part of curve after the peak discharge, SP (dimensionless), and the maximum rainfall intensity during 30 minutes I_{30max} (mm h^{-1}).

A principal component analysis (PCA) was used to explore the dataset, and the direction and strength of the relationships between the variables. An analysis of variance (ANOVA) was used to assess the significance of the treatment (conservation measures) on the hydrological response of the micro-catchments.

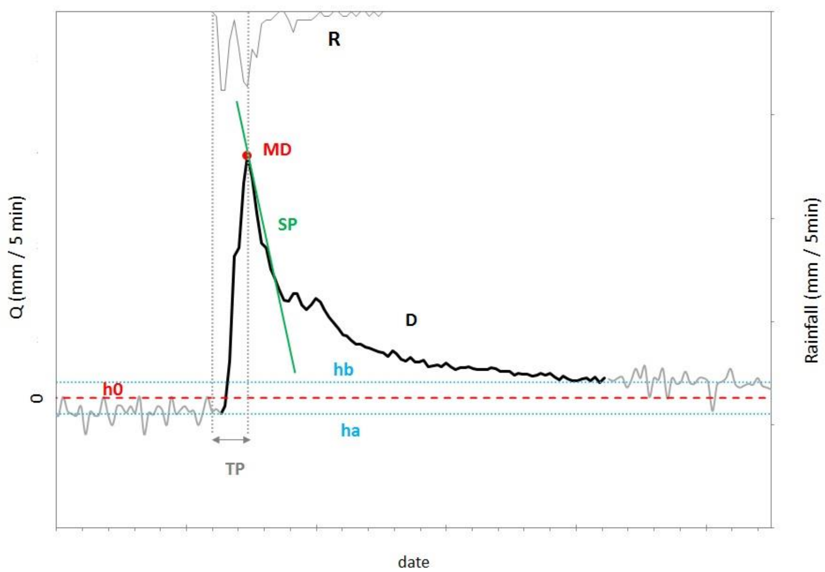

Figure 43: Scheme of hydrograph and the main considered variables. ha and hb are the height before and after the rainfall event and ho is the reference level.

4.3 Results

4.3.1 Characterization of rainfall events generating runoff

The Figure 44 shows the measured rainfall data at a daily resolution covering a period of 3.5 years, based on the raingauge Lo1. The frequency analysis (Figure 45) shows that few events (8 % of all rainfall events) have an intensity that is higher than 10 mm day $^{-1}$. Figure 44 also shows that the rainfall events are mostly individual rain events bound by a period of no rainfall. It is rare to observe various rainfall events during one day (Table 11). Most of the rainfall events have a low intensity, and do not result in runoff in the small ephemeral streams. The maximum rainfall intensity that was registered is 80.8 mm.h^{-1}, and occurred during 30 minutes. This event has a recurrence time of 38 years, based on the 8 years meteorological data of PROMAS at the University of Cuenca (PROMAS, 2016). For the Cuenca meteorological station (located at 23 km distance from Loreto), Harden (1993) reported only 3 to 4 events per year with an intensity of > 27 mm.h^{-1}.

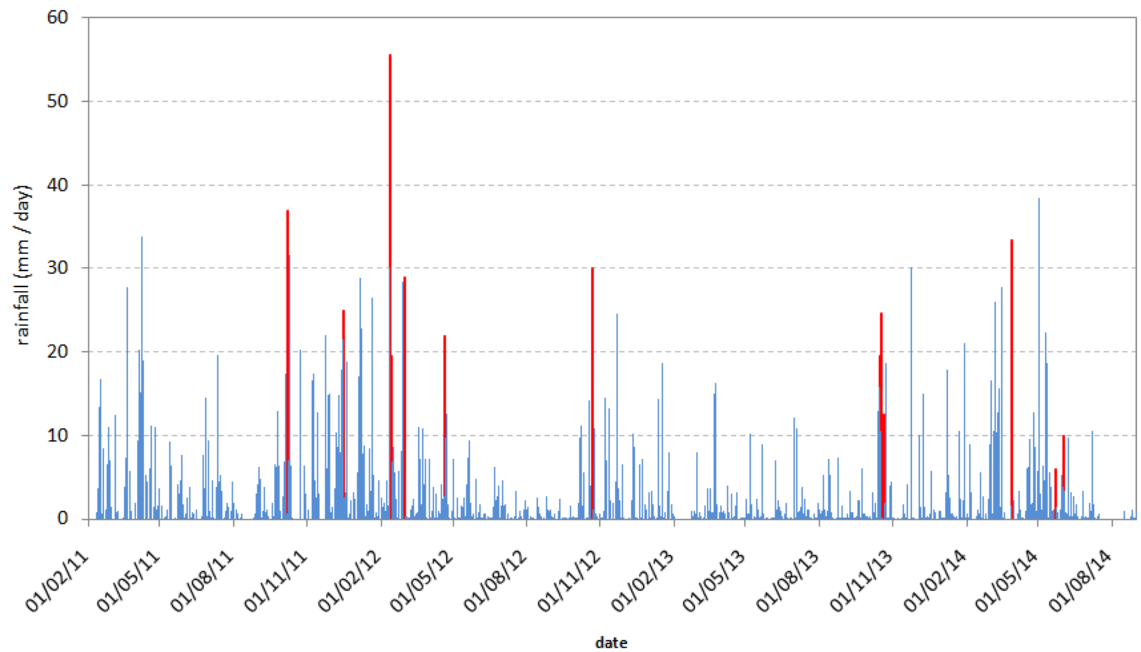

Figure 44: Daily rainfall for Loreto catchment with selected events in red.

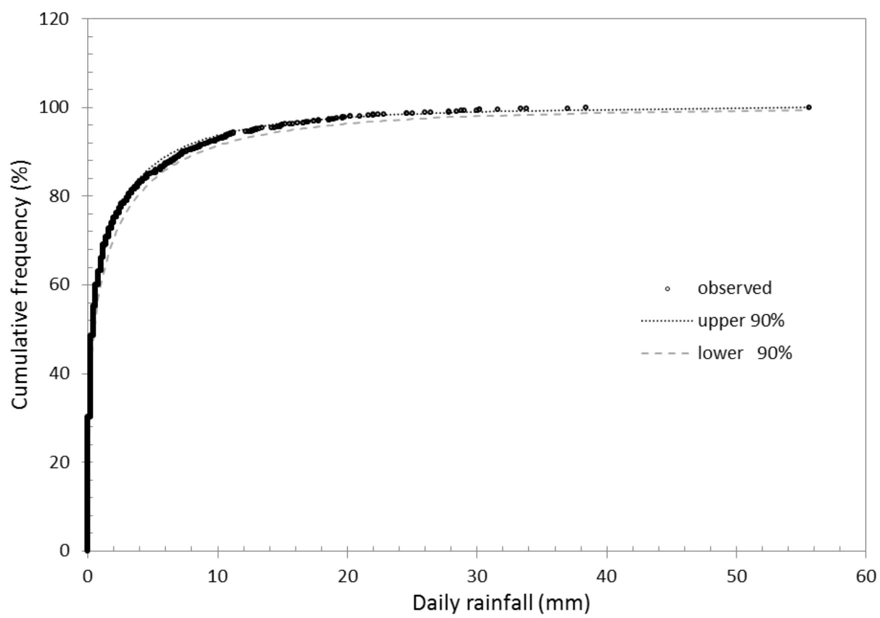

Figure 45: Plot of cumulative frequency of daily precipitation data.

Fifteen rainfall events were selected (Table 11). The I_{30max} of these events ranges between 4 mm/h and 80.8 mm/h. As such, they allow us to evaluate the effect of rainfall intensity on the hydrological response of the micro-catchments. The antecedent moisture condition, AMC, was determined based on the total rainfall amount in the five days preceding the event (P5) using the SCS method (Soil Conservation Service (SCS), 1985): AMC I: P5 < 12.5 mm (dry), AMC II: 12.5 < P5 < 28 mm (average) and AMC III: P5 > 28 mm (high). I_{30max} has a good correlation with maximum discharge with an R^2 of 0.36.

Table 11: Characteristics of the fifteen rainfall events in the Loreto catchment that were selected for the hydrological analyses.

	I_{30max}	R_{tot}	R_{event}	duration	P5	AMC
	mm h^{-1}	mm	mm	hrs	mm	
06/10/11	29.6	37.2	37.2	7.67	31.6	III
16/12/11	11.20	25.00	24.80	7.00	61.8	III
12/02/12	80.80	55.60	55.60	1.08	37.2	III
13/02/12	32.40	19.60	19.60	1.50	92.6	III
29/02/12	24.80	29.00	28.60	3.00	51.6	III
19/04/12	16	31.8	28.6	8.67	10.4	I
21/10/12	24.80	30.00	30.00	2.75	32.8	III
15/10/13	19.2	19.6	15.4	1.67	38.8	III
17/10/13	25.20	24.60	24.40	2.58	68.8	III
19/10/13	9.20	12.16	12.40	1.67	70.8	III
29/03/14	62.40	33.40	33.40	0.75	2.6	I
22/05/14	4.40	6.00	6.00	1.58	4.4	I
31/05/14	4.00	13.80	6.20	2.42	6.8	I
01/06/14	4.00	10.00	4.60	2.25	10.6	I
02/06/14	4.00	3.40	1.00	0.58	20.2	II

I_{30max} (mm h^{-1}); Rtot: total daily rainfall (mm); R_{event}: Rainfall amount during one event (mm), Duration: duration of the event (hours); P5: total 5-day antecedent rainfall (mm); AMC: antecedent moisture condition class following SCS (1985).

The rainfall distribution within the Loreto catchment is relatively uniform, as the two rain gauges Lo1 and Lo2 are recording similar amounts of precipitations. The mass curve of the daily precipitation data for Lo1 and Lo2 has a correlation coefficient, R^2, of 0.9. For individual events, the correlation coefficient between the data from Lo1 and Lo2 is 0.84 (non cumulative). Figure 46 shows that there is a small deviation in the rainfall amounts during high magnitude events. For this reason, the data from the individual rain gauges were used for the calculations, Lo1 for the three micro-catchments in degraded land (DI, DT, DF) and Lo2 for the two micro-catchments in agricultural land (AI, AT).

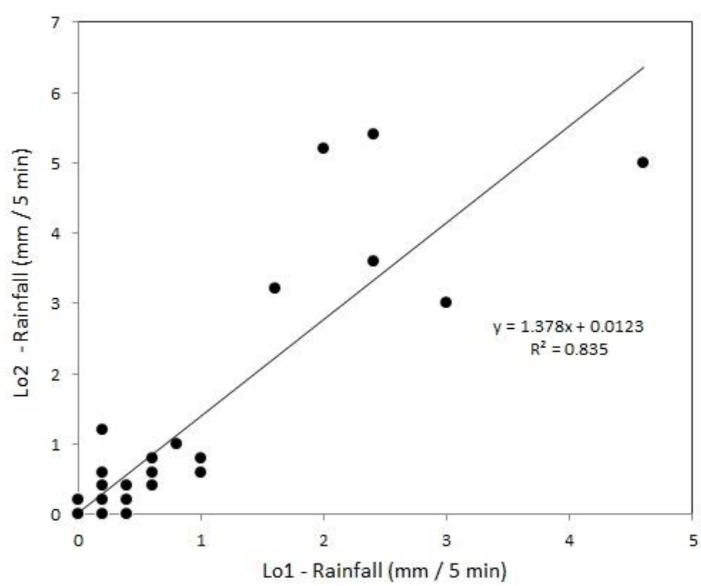

Figure 46: Scatter plot of rainfall data from two rain gauges (Lo1 and Lo2) used in the study (rainfall event of 17/10/2013).

4.3.2 Characterization of runoff events

Table 12 summarizes the main characteristics of the fifteen rainfall-runoff events for the five micro-catchments. The majority of the events occurred during the rainy season, with exception of the event of 1/06/2014. The channels in the five micro-catchments are ephemeral, and only have measurable stream flow after intense rainfall events.

In the degraded land (micro-catchments DT, DI and DF), the stream flow quickly drops to zero after a rainfall event. The degraded zone is characterized by the presence of active gully systems, topography is highly dissected. Vegetation is scarce and consists of small vegetation patches of herbs and shrubs in the gully channels. Due to these characteristics the soil has a poor infiltration capacity and surface runoff is high. Restored badlands (DF) have slightly higher values of soil moisture content compared to the soils in active badlands showing a better water holding capacity (see Chapter 2). DF micro catchment has the lowest runoff coefficient values.

In the agricultural land (micro-catchments AT and AI), there is a longer recession stage and sometimes the channels have low flow during several days after the rainfall event. The soils from this area have the highest values of soil moisture content (Chapter 2). The agricultural land has soils with higher organic matter and greater depth compared with soils from degraded lands. The zone is mainly characterized by crops and grasslands. Abandoned lands covered with natural pastures are well protected against action of water. Soils in agricultural land can have high infiltrations rates as showed by Molina et al. (2007) who reported infiltration capacities of above 29 mm h^{-1} for croplands and forests in the same region.

In general, the highest runoff peaks are observed for degraded lands with active processes of gully erosion. The condition of antecedent soil moisture seems to be less important. During few events (e.g. 16/12/2011 and 13/02/2012), it is possible to see the effect of soil moisture on runoff volume (Table 12): when the soil is wet, and has accumulated water during the days previous to the event, the runoff is slightly higher.

The relation runoff – rainfall for the study area is illustrated in Figure 47 where for all events exist a linear relationship with a negative y intercept (R^2 = 0.72). The slope represents the fraction of the rainfall in excess of the x intercept that becomes runoff. Normally this difference represent the amount of rainfall that is lost before runoff is generated and correspond to evapotranspiration and storage in the catchment, however the storage fraction in degraded catchments would be very small (not the case in forested and well vegetated catchments). Analyzing this relationship for different micro catchments we obtain R^2 values of 0.80, 0.85, 0.82, 0.66 and 0.46 for DI, DT, DF, AI and AT respectively.

Table 12: Main hydrological characteristics of the fifteen runoff events for the five micro-catchments (DT, DI, DF, AI and AT), with RC: runoff coefficient, MC: micro catchment; MD: maximum discharge, TP: time to peak, SP: slope of the recession part of the hydrograph after the peak discharge, D: discharge

MC	RC (%)	MD (mm 5min^{-1})	TP (min)	SP (/)	D (mm)	date
DT	97.214	2.7870	40.00	-0.06005	36.22	
DI	57.324	2.0699	30.00	-0.02341	21.32	06/10/11
DF	67.603	1.4959	45.00	-0.01616	25.17	
DT	49.449	5.7017	20.00	-35.85437	12.28	
DI	16.574	0.6439	25.00	-0.55042	4.11	
DF	17.587	0.3215	15.00	-0.22207	4.36	16/12/11
AI	38.285	0.4756	155.00	-0.28600	9.49	
AT	36.371	0.4264	155.00	-0.18208	9.02	
DT	69.205	14.5054	45.00	-32.64910	38.54	
DI	13.903	0.6178	50.00	-1.27171	7.73	
DF	70.880	9.2975	40.00	-8.21303	39.44	12/02/12
AI	63.129	6.3818	40.00	-3.55614	29.67	
AT	44.747	3.1438	45.00	-3.57949	21.03	
DT	75.495	7.0094	35.00	-21.79138	14.82	
DI	44.938	0.4474	50.00	-1.44308	8.81	
DF	42.683	0.6942	10.00	-1.47328	8.37	13/02/12
AI	30.947	0.9528	35.00	-1.71410	3.84	
AT	32.793	0.7131	40.00	-0.59354	4.07	
DF	32.408	0.4871	20.00	-0.00744	9.28	
AI	75.364	2.9309	35.00	-0.08335	21.55	29/02/12
AT	65.593	1.7164	35.00	-0.02826	18.76	
DT	63.839	0.1643	40.00	-0.00390	18.29	
DI	71.074	0.1848	60.00	-0.00150	20.33	
DF	37.393	0.1312	60.00	-0.00105	10.70	19/04/12
AI	45.928	0.2017	40.00	-0.00271	10.01	
AT	32.622	0.1909	45.00	-0.00156	7.11	

DT	43.505	3.6334	20.00	-0.19184	13.07	21/10/12
DI	55.256	0.9313	40.00	-0.01617	16.58	
DF	24.949	1.0389	30.00	-0.04265	7.49	
AI	9.644	0.1831	30.00	-0.00339	2.89	
AT	8.543	0.1706	55.00	-0.00275	2.56	
DT	66.770	0.8614	15.00	-0.13382	10.30	15/10/13
DI	38.010	0.4214	45.00	-0.00844	5.85	
DF	23.050	0.2021	30.00	-0.00550	3.27	
AI	51.186	1.1060	20.00	-0.02762	6.64	
AT	57.144	0.5929	30.00	-0.01573	8.24	
DT	70.510	1.7904	40.00	-1.42912	17.20	17/10/13
DI	50.035	1.3651	85.00	-1.20302	12.21	
DF	16.950	0.5540	50.00	-0.68240	4.14	
AI	36.920	1.8392	35.00	-1.02680	12.85	
AT	47.642	1.2811	40.00	-0.57961	16.58	
DT	29.623	0.6655	70.00	-1.75062	3.67	19/10/13
DI	14.715	0.2821	140.00	-0.99648	1.82	
DF	0.540	0.0105	130.00	-0.01873	0.07	
AI	9.690	0.1168	125.00	-0.17628	1.20	
AT	12.307	0.1201	125.00	-0.06371	1.53	
DT	53.594	1.5835	20.00	-1.91795	17.93	29/03/14
DI	36.312	3.4672	20.00	-3.68757	12.13	
DF	49.475	3.8001	25.00	-2.80254	16.54	
AI	65.566	4.5175	20.00	-3.67043	20.31	
AT	64.313	2.2011	25.00	-1.69560	10.63	
DT	34.739	0.1327	95.00	-0.06006	2.08	22/05/14
DI	14.621	0.0544	95.00	-0.03390	0.88	
DF	24.337	0.0559	50.00	-0.01266	1.46	
DT	4.122	0.0512	5.00	-0.00021	0.26	31/05/14
DI	0.245	0.0025	10.00	-0.00016	0.02	
DF	0.069	0.0010	10.00	-0.00010	0.00	
DT	35.329	0.0451	30.00	-0.00082	1.63	01/06/14
DI	23.144	0.0477	95.00	-0.00044	1.06	

DF	0.352	0.0023	45.00	-0.00005	0.02	
DT	79.679	0.0600	55.00	-0.00085	0.80	02/06/14
DI	23.824	0.0202	45.00	-0.00030	0.24	
DF	0.258	0.0007	45.00	-0.00004	0.003	

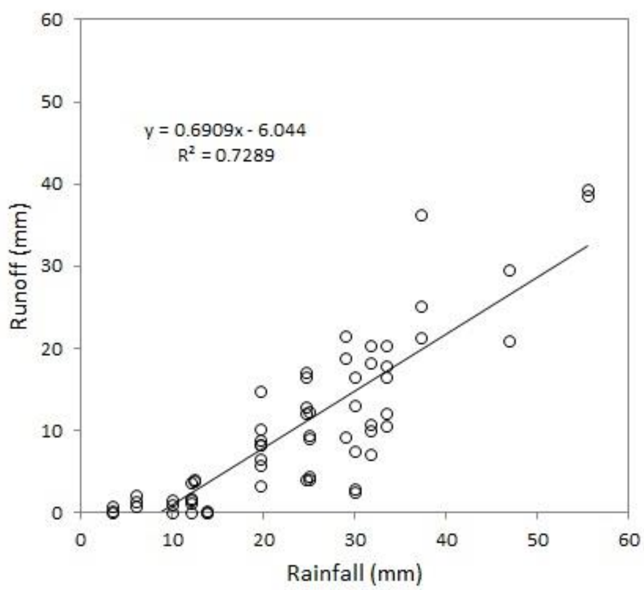

Figure 47: Scaterplot between the Runoff and Rainfall.

Table 12 indicates that the hydrological response is very fast in all five micro-catchments. Figure 48 and Figure 49 show the hydrographs of the five micro-catchments for two selected events during the rainy season in the autumn of 2013. The event of 17/10/2013 corresponds with a high-intensity event with a I_{30max} of 25.20 mm h^{-1}, while the 19/10/2013 represents an event of intermediate intensity with an I_{30max} of 9.20 mm h^{-1}.

The figures show that the lag time between the start of the rainfall and the start of the runoff is generally short. This is particularly the case for the micro-catchment DT, that registers the highest values of discharge and the highest peak flows (Figure 48, Figure 49). This flashy runoff response of the DT catchment is also illustrated by the rapid fall of the discharge after the peak flow. Compared to the reference catchment DT, the restored catchment DF (plantation forest) has much lower values of the unit discharge. During certain runoff events, there is no measurable flow at the outlet of DF.

The two micro-catchments in the agricultural land (AT and AI) have a very similar hydrological response, although the total discharge is higher in the restored micro-catchment AI compared to the reference catchment AT. The slope of the hydrograph recession curve is low, compared to the DI and DT micro-catchments.

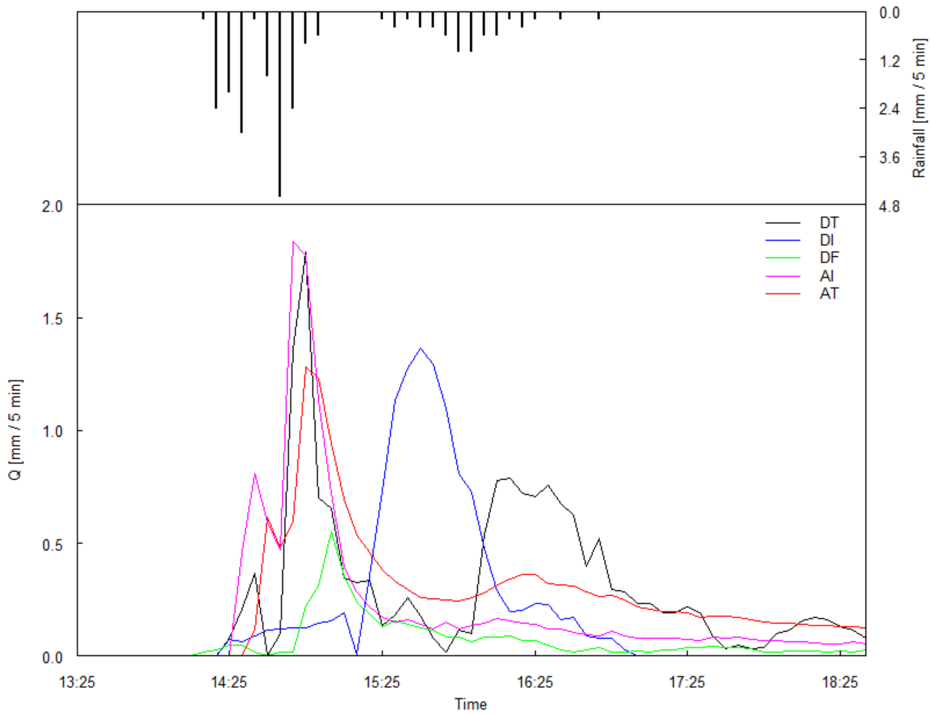

Figure 48: Records of the rainfall (mm/5 min) and runoff (mm/5 min) for the five micro-catchments during the 17-10-2013 event.

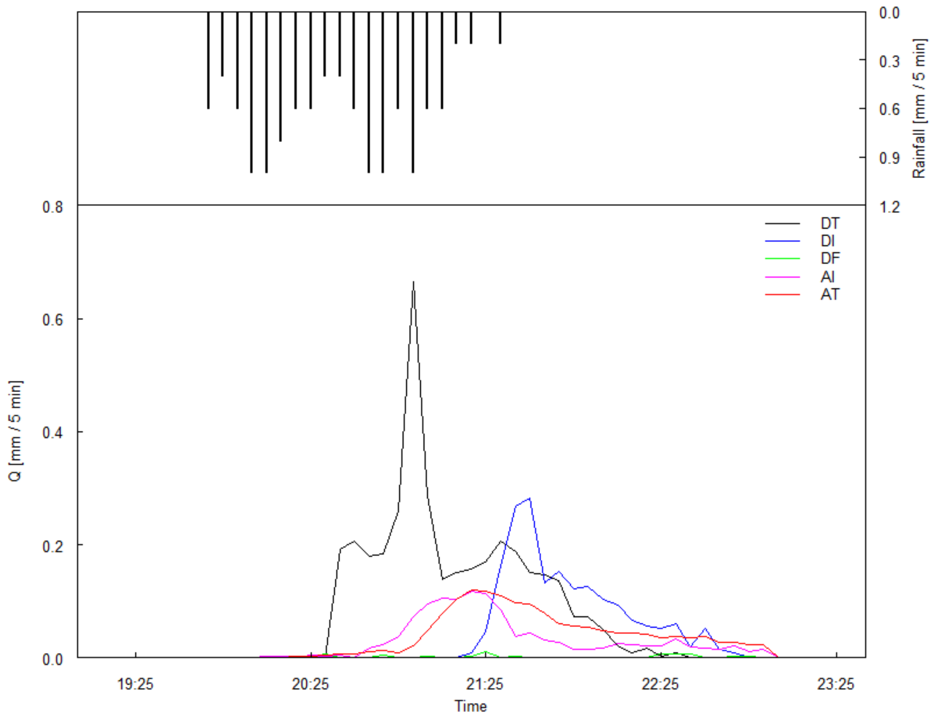

Figure 49: Records of the rainfall (mm/5 min) and runoff (mm/5 min) for the five micro-catchments during the 19-10-2013 event.

4.3.3 Analysis of factors explaining rainfall-runoff behavior of micro-catchments

The analysis of the principal components reveals the structure in the dataset. When including the five hydrological indicators (RC, TP, SP, MD, and D), rainfall characteristics (duration of rainfall and R_{I30max}) and the total 5-day antecedent rainfall (P5), the first two components of the PCA can explain 77% of the observed variation in the data (Figure 50). The first component is related to the discharge and the variables that are directly related to the generation of surface runoff. It is positively related to the maximum discharge, MD, the runoff coefficient, RC, the rainfall intensity as measured by I_{30}, and negatively related to the slope of the recession curve (SP). The SP depends directly of MD because at greater values of this variable the slope of the recession part of the hydrograph will be more intense in magnitude but negative. R_{tot} was not included in the analysis of PCA because is correlated with discharge (D), runoff (RC) and I_{30max}. The second component is related to the shape of the hydrograph and variables such as the time to the peak discharge, TP, and the preceding rainfall amount, P5, have high loadings on the 2nd component. The duration of the rainfall and the total discharge, D, have relatively little explanatory power.

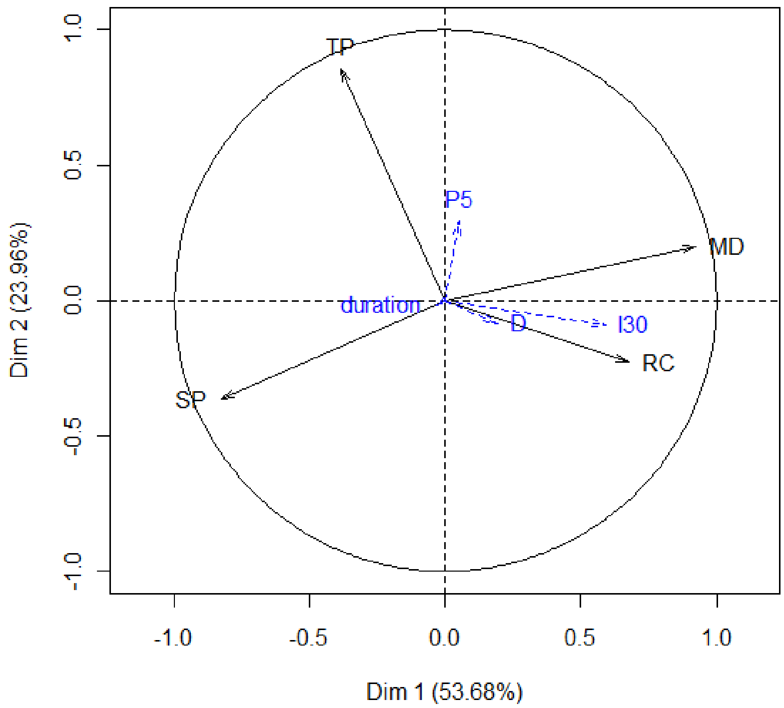

Figure 50: Outcomes of the principal component analysis. The first two components explain 77% of the observed variation in the dataset. The variables that are marked in blue are not considered in the analysis of variance, ANOVA.

Based on the data from the 15 runoff events, we can observe a positive relationship between the runoff coefficient, RC, and the maximum unit discharge, D (Figure 51). The reference site in the degraded land (DT) has consistently the highest values of the runoff coefficient and the maximum discharge.

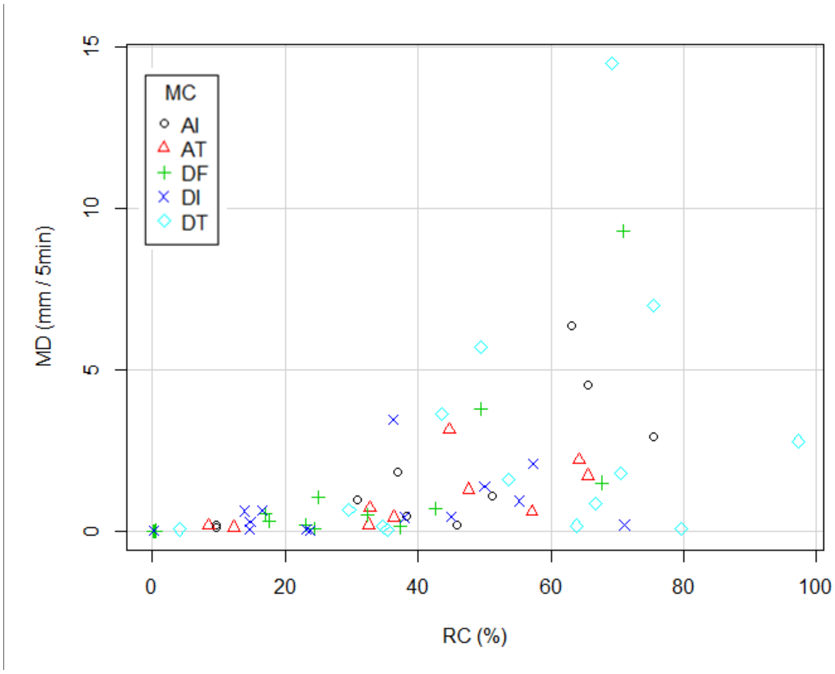

Figure 51: Scatterplot between the maximum discharge (mm / 5 min) and the runoff coefficient (%) for the five micro-catchments, based on data from 15 individual runoff events

The runoff coefficient, RC, varies between 0.07 and 97 % with an average of 39 %. The maximum value occurred during a rainfall event with an I_{30max} of 29.6 mm h^{-1}. The RC has a positive correlation with R_{tot} (R^2 = 0.20) and its relation is better than with I_{30max} (R^2 = 0.14). Considering individual micro catchments, DF has the highest correlation for R_{tot} and RC (R^2 = 0.71). The lowest value is for AI (R^2 = 0.09).

The total 5-day antecedent rainfall (P5) appears to be of minor importance for the runoff generation in the micro-catchments as its inclusion in the PCA analysis only diminishes the explained variance. In the degraded land, the soils are highly eroded and parent material is often outcropping. Saturation of the surface is almost never reached. Even after a prolonged period of intense precipitation (with a total of 52 mm during the 5 preceding days and 107.2 mm for 8 days), the actual soil moisture content was only 21.2% (54.6 % of the total capacity of water retention during: 12/10/2011). The same tendency was found for 12 different dates with samples taken in 16 sites. This suggests that the dominant mechanism for runoff generation is infiltration excess overland flow, which is consistent with earlier work based on rainfall simulations by Molina et al. (2007).

When analyzing the effect of treatment (catchment restoration) on the hydrological response of the micro-catchments, we observe that there is a significant difference in the runoff coefficients of the degraded lands. The reference catchment has significantly higher runoff coefficients than the restored catchments. The ANOVA shows that there are significant differences in the runoff coefficient at 95% probability level ($P = 0.017$; $df = 4$; F value = 3.248). The type of treatment (forest plantation vs. restoration of active gully channels) apparently has no major effect, as the two restored catchments have similar runoff coefficients (Figure 52). Interestingly, we do not see any significant difference in the runoff coefficient between the reference and restored catchment in the agricultural land, as the two micro-catchment have intermediate runoff coefficients.

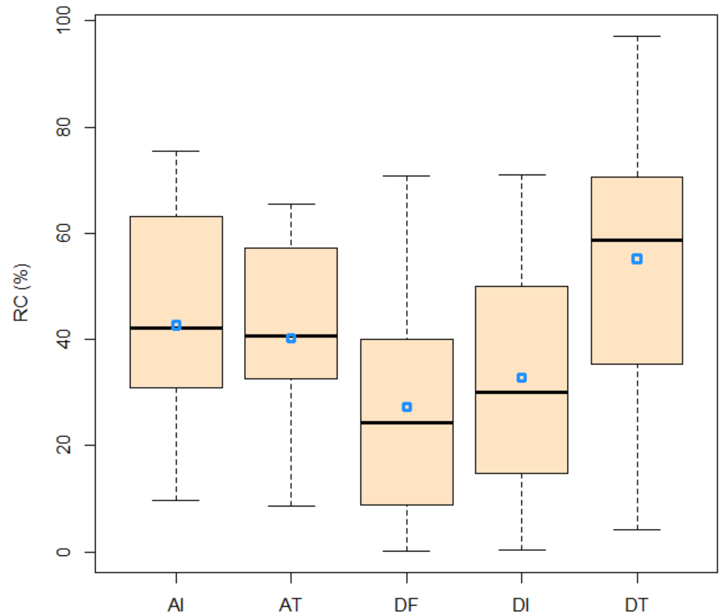

Figure 52: Boxplot of runoff coefficients of the five micro-catchments, showing the effect of treatment on runoff behavior. Blue squares correspond to mean value that were calculated based on 15 individual rainfall-runoff events.

When analysing the recession curve of the peak hydrographs, we observe significant differences in the slope of the recession curve between the reference and the restored micro-catchment in the degraded land (Figure 53). The reference catchment (DT) clearly has a flashier hydrological response, with a rapid decrease of the hydrograph after the peak of the discharge. Also, there is a large variability in the shape of the hydrograph for the DT micro-catchment. No differences are observed for the catchments in the agricultural land. The ANOVA statistical analysis confirms that there are statistical differences at 95 % of probability, comparing means of SP (P = 0.051; df = 4; F value = 2.514. No significant dioferences were found for TP and MD variables.

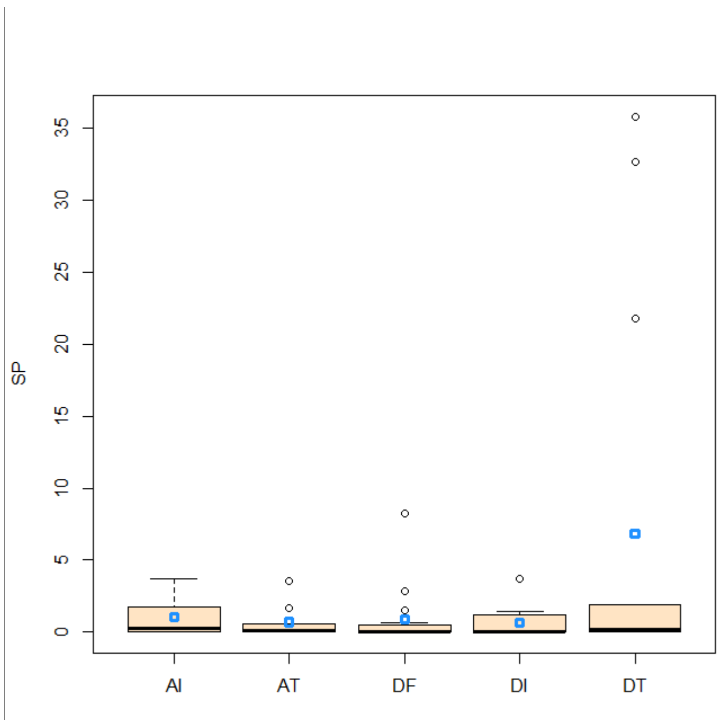

Figure 53: Boxplot with slope after the peak discharge for the micro catchments of the study (absolute values).

4.4 Discussion

The runoff events were selected based on the maximum rainfall intensity in 30 minutes (I_{30max}), which appears to be a better indicator of runoff generation than the daily rainfall amount, especially for peak flows that are of special interest by their erosive characteristics. I_{30max} have a correlation with MD with a $R^2 = 0.36$ (in comparison R_{tot} have a correlation with MD with a $R^2 = 0.18$). The duration of an average rain storm in the Loreto catchment is around 2.5 hours, and storm events are single, individual events that are followed by a period of no rainfall. The knowledge of the rainfall intensity in the study sites is very important to understand the runoff mechanisms.

Our dataset suggests that infiltration excess is the major runoff generation mechanism, as the hydrological response of the micro-catchments is strongly associated with the runoff coefficient, maximum discharge and rainfall intensity. The fact that the cumulative rainfall in the 5 days preceding the rainfall events (P5) is not clearly associated with the magnitude of the surface runoff sugest that the soil water storage is limited and/or has no major influence on the hydrological response of the degraded catchments. Soil saturation would be rare and more probable in areas with a good vegetation cover. In the lithology of the zone, more than half of the clay minerals predominantly consist of 2:1 dioctahedral clays that are prone to swelling. The very low organic matter content and poor vegetation cover lead to very low infiltration coefficient in bare land. The presence of large amounts of swelling clays in all outcropping lithology (Molina et al., 2007) leads to rapid sealing of the soil surface that prevents further infiltration of rain water.

The soil infiltration capacity and the water holding capacity of the thin soils is very low, which limits the buffering effect that soils can have on the hydrological response. However, these conditions could be improved with an increase in vegetation cover. Molina et al. (2009) found that cumulative infiltration is higher for gullies with more than 50% vegetation cover under dry hydrological conditions.

There is a clear (and significant) difference in the hydrological response between the restored and non-restored catchments in the degraded lands. The poor vegetation cover in the non-restored catchment DT gives way to the rapid generation of infiltration excess overland flow that concentrates in the gully channels. This is directly reflected in the SSY for this micro catchment (Chapter 3) where the amount of sediment retained in trap is larger. The hydrographs for DT exemplify the flashy response of the catchment (Figure 48, Figure 49). In contrast, the restored micro-catchments DI and DF have both a lower runoff coefficient and a slower return of the recession curve to zero discharge. The construction of dams in the active gully channel of the DI catchment strongly reduces the hydrological connectivity of the channels, as also suggested by controlled flow experiments by (Molina et al., 2009). The check dams retard the flow of runoff water, and promote the re-infiltration of runoff water in the gully channels. This results in lower peak flows (MD), and delays the flood peak time at the outlet of the catchments. Our data confirm that restoration of active gullies is key in slowing down and reducing the runoff in degraded catchments. Additionally, the sediment deposited in the check dams promotes the colonization of pioneer species that contribute to capture sediment, reduce the water velocity and give stability to the material accumulated.

The micro-catchment that was restored with plantation forest, DF, has the lowest values of unit discharge and runoff coefficient. This was also verified during the 41-months monitoring campaign, as only very few high-magnitude events gave rise to runoff generation in the DF catchment. The dense *Eucalyptus* plantation forest is able to reduce the generation of excess infiltration runoff, as a result of enhanced infiltration in the soils that are protected by a vegetative and dead leaf cover (Vanacker et al., 2003). The establishment of *Eucalyptus* forest has altered the water balance of the degraded areas, and is expected to have increased the rates of consumption and transpiration (Bruijnzeel, 2004). There is a total reduction in sediment export in the DF micro-catchment, with the exception of a limited number of extreme rainfall events as data show in Chapter 3.

Catchment restoration has a measurable impact on the hydrological response of small catchments during normal storm events –i.e. events with a recurrence interval of 1 to 2 years. The effectiveness of restoration measures to reduce and retard peak flows during extreme events is less obvious. During our observation period, one extreme event having a 37 years recurrence interval was registered. Our data suggest that the regulation capacity of conservation measures is much lower during such extreme events, as we observe smaller differences in the runoff coefficients between restored and non-restored catchments during the event of 12/02/2012. This also happens with the ability to capture sediments as data show in Chapter 3 where after events with high intensity some check dams showed a low sediment accumulation.

4.5 Conclusions

Catchment restoration is very effective to reduce and retard peak flows in degraded catchments. The effectiveness of the restoration schemes is highest during normal rain storms, and our data suggest that this decreases for extreme events. Living check dams in active gully channels decrease the hydrological connectivity, promote the re-infiltration of surface runoff and slow down the velocity of the overland flow. Infiltration excess overland flow is the main runoff generation mechanism in degraded catchments. By augmenting the infiltration capacity of the surface (either through soil restoration or living mulches), the amount of surface runoff is reduced. Our data show that the peak flows could be reduced by resp. 73% and 56%, as a result of the implementation of restoration techniques in active gully channels and vegetation restoration by plantation forest. The latter is not only beneficial for hydrological regulation of torrential streams, but also contributes to soil restoration as shown in Chapter 3.

Chapter 5 : General conclusions and recommendations

Tropical mountain ecosystems host several mega-cities in the Andes, and provide goods and services for settlements, agriculture and industries in the InterAndean valleys. The rising pressure on natural resources has caused environmental degradation and loss of environmental services. This has urged private and public stakeholders to develop and implement environmental management plans, including conservation and restoration programs. Given the local context, these initiatives have to be cost effective and technically applicable for land managers, smallholders and agro-industry.

Land degradation is often concentrated in erosion "hot spots", relatively small areas that exhibit anomalously high erosion rates. When these erosion hotspots become connected to the stream network, the off-site effects of soil erosion can have large implications for a wider area. Soil and water conservation schemes can maximize their effectiveness and minimize costs when they are applied in the erosion hotspots and when the pathways of runoff and sediment transfer from the hot spots to the river systems are disconnected. Bioengineering techniques have proven to be particularly efficient, and more sustainable than hard civil engineering works. Vegetation restoration and regeneration is a key factor for erosion control and sediment retention.

5.1 Erosion processes in InterAndean basins

The thesis focuses at the effectiveness of soil and water conservation techniques on catchment restoration. Its main objective was to enhance our understanding of the effect of soil conservation and restoration schemes on (i) erosion control in active gully systems, (ii) l physical and chemical soil properties and soil formation, and (iii) the hydrological response of ephemeral streams. An exhaustive monitoring campaign provided the baseline information on soil, water and nutrient fluxes of small hydrological units, and their response to land use and land management.

Bioengineering techniques (wooden dams) can contribute to reduce the sediment export of active gully systems by more than 70%, as a result of gully stabilization. Plantations of *Eucalyptus* trees stabilize the gully channels, thereby reducing the erosion rates to levels near to natural erosion. Through the increase in soil vegetation cover (plants and dead leaves), there is an additional input of organic matter resulting in the formation of an *O* and *A* horizons and favoring the establishment of other plant species.

Our results confirm that agricultural lands are not the major source of sediment in these catchments. Agricultural plots are relatively well covered by vegetation, as intercropping (e.g. corn with beans) is commonly practiced. Arable fields are interspersed with rangelands (*Pennisetum clandestinum*) that act as buffering zones trapping sediment.

Bad lands and roads are the main sources of sediment, and are true hotspots of sediment production within the catchments

Catchments can be divided into three zones that are characterized by different types of soil erosion. Their identification is useful to delimit critical zones and understand the interactions between erosion processes, soils and vegetation cover. The highly degraded zone is characterized by active gully erosion. Restored lands have stabilized gully systems, and soil restoration occurs. Rill and inter-rill erosion is the dominant process in agricultural lands. Not all rainfall events result in high rates of sediment production and transfer. In agricultural zones and forest plantations, a limited number of rainfall-runoff events (< 3/yr) have sufficient energy to mobilize and transport sediment, and the erosion rates are very low (resp. 1.2, 18.2 and 0.2 t ha^{-1} yr^{-1} for the forested and two agricultural catchments).

There is a clear effect of catchment restoration on the hydrological response of ephemeral channels. In catchments that were restored with forest plantations, there is a reduction of the peak flows and improvement of the regulating capacity of the catchment. Gully channel restoration with wooden flow barriers has similar effects on the surface hydrology.

5.2 Impact of soil and water conservation treatments on sediment mobilisation

The establishment of vegetation plays a significant role in the process of gully restoration (Chapter 2). Vegetation physically protects the soil against soil erosion agents. Vegetation also acts as a barrier in flow paths and reduces the water velocity, which enhances deposition of eroded material. An additional benefit of vegetation is the supply of organic matter to the topsoil contributing to soil formation processes, by means of decomposition of organic material that accumulates at the soil surface and is later incorporated into the soil matrix by edaphic processes. Our results show that vegetation cover is the most important factor explaining the observed variation in C content of top soils.

The implementation of bioengineering techniques is effective to reduce sediment mobilization and transport. The trapped material behind the flow barriers or so-called dams offer good soil physical conditions for plant colonization and accumulation of organic matter. In a short period of time, the sediment trapped in depositional zones shows better soil physical properties for vegetation establishment than surrounding eroded areas, because of a lower bulk density and higher soil water content. These well-vegetated patches upstream the flow barriers favour soil formation, and further enhance gully stabilization. The pioneer plants that colonize the depositional zones in the gully channel have dense root systems that provides cohesion.

In the badlands, an appropriate forestation scheme is an effective means of long term soil restoration. The soil analyses indicate that soil physical and chemical properties have improved after the plantation of *Eucalyptus* trees in this strongly degraded environment. In the zone where rapidly growing exotic tree species were implemented, the changes in soil properties like C content in the topsoil have favoured a first stage of soil formation with development of an *O* horizon.

In Chapter 3, our data show that biotechnical measures such as wooden flow barriers (or so-called dams) are very effective in reducing sediment transport in gully systems. The dams built with wood and tires have an efficiency of more than 70% and are effective in stabilizing active gullies in badlands through significant reduction (of about 77 and 86%) of the amount of sediment exported from the catchment. The comparison of the erosion rates between the five micro-catchments shows clearly that the main sources of sediment are the active gully systems. Agricultural land is not the major source of sediment in these environments. Besides, forestation of active gully systems with rapidly growing exotic species such as *Eucalyptus* has a very positive effect on the stabilization and restoration of the badlands, and effectively reduces the sediment export. Rainfall intensity controls runoff volumes and sediment deposition during events of high rainfall intensity. During extreme rainfall events, the conservation works seem to be less efficient in trapping sediment, although a detailed analysis is necessary to make conclusive statements.

5.3 Impact of soil and water conservation treatments on hydrological response

In Chapter 4, we analyze the effect of vegetation restoration and bioengineering techniques, like the construction of flow barriers in gully channels, on the hydrological response of ephemeral channels. Our data show a clear (and significant) difference in the hydrological response between the treated and non-treated micro-catchments for the degraded lands. In contrast to the non-treated catchment DT, the micro-catchments DI (w/ gully restoration) and DF (w/ vegetation restoration) have both a lower runoff coefficient and a slower return of the recession curve to zero discharge. The soil and water conservation treatments are effective in slowing down and reducing the runoff in degraded catchments. Catchments where agricultural land use results in a patchwork of arable lands and pastures (AI and AT), are less affected by gully erosion. In these catchments, bioengineering techniques are less effective, as the rates of soil erosion and sediment mobilization are lower.

Infiltration excess overland flow is the main runoff generation mechanism in degraded catchments. By augmenting the infiltration capacity of the surface (either through soil restoration or living mulches), the amount of surface runoff is reduced. Our data show that the peak flow can be reduced by resp. 73% and 56%, as a result of implementation of restoration techniques in active gully channels and vegetation restoration by plantation forest. Catchment restoration is very effective in decreasing the flashiness of the flow, which is currently causing major damage to civil infrastructure.

5.4 Further research and recommendations

A next step is the calibration of a geomorphological model, that is able to reproduce the basic mechanisms of runoff generation, soil erosion and sediment transfer and deposition. Such a physically-based model would allow one to assess the potential impact of conservation and restoration schemes on water and sediment fluxes, for different climate scenarios. Our data show that a distributed model is necessary to represent correctly the patchy distribution of the vegetation, and the effects on infiltration, runoff generation and soil loss.

Our results were collected at the scale of small hydrological units. In our micro-catchments, water erosion is the dominant erosion mechanism and active gullies are the main sources of sediment. Our conclusions are relevant for similar geographical settings. However, direct extrapolation of our results to larger hydrological units is not possible without a thorough validation of the ongoing erosion mechanisms. This is particularly the case of larger hydrological units, where mass wasting becomes an important geomorphological process.

This study shows the high efficiency of conservation and restoration techniques. It also highlights the potential of soft restoration programs that combine vegetation restoration on the hill slopes with gully restoration with live check dams. Forest plantations with *Eucalyptus* can play an important role in restoration programs in highly degraded lands. Because of the controversy that currently exists on the effectiveness of exotic plantations, it is necessary to do more research on the potential of plantation forests to be catalyst in the restoration of soil and water resources in the humid tropics. Topics that need further attention are how the density of trees, the soil-water balance, and the vegetation richness in the understory vegetation are controlling the effectiveness of the restoration scheme. In addition, it could be interesting to explore the possibilities with other species like *Pinus* that is widely used in afforestation and reforestation projects.

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