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Institutionen för skogens ekologi och skötsel


# Water infiltration in the Nyando River basin, Kenya


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Sveriges Lantbruksuniversitet — Utb program: Erasmus

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Handledare: Gert Nyberg och Ulrik Ilstedt, SLU, Inst för skogens ekologi och skötsel

Examinator: Anders Malmer, SLU, Inst för skogens ekologi och skötsel — Umeå 2009

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## Abstract

On-site and off-site effects of erosion in the Lake Victoria basin are some of the main contributors to the actual situation of poverty which is facing its population, one of the densest in the world. The Nyando River Basin (Western-Kenya) was identified as a regional erosion hotspot and one of the main sources of sediment and phosphorous into the Lake Victoria (ICRAF, 2001). In 2004, within the context of the Western Kenyan Integrated Ecosystem Management Project (WKIEM, ICRAF, 2003) measures to reduce erosion and increment agricultural productivity, through enhanced vegetation cover and improved management, were implemented in some plots located in East-Nyakach (lower Nyando River Basin).

This study investigated the effect of tree-planting and exclusion of grazing livestock in infiltrability, soil bulk density and erosion, as well as the existent relationships between these variables four years after rehabilitation started. A comparative study between WKIEM treated plots, where trees were planted, and adjacent controls was carried out in 9 paired-plots. Infiltrability was measured using double-ring infiltrometers in six sample points per plot, and by rainfall simulations at 20 and 60 mm/h intensities in two sample points. Steady-state infiltrability was afterwards estimated by means of curve-fitting to Philip's equation (Philip, 1957). Sediment generation rates (Mg $ha^{-1}$ $h^{-1}$) were calculated, for 20 and 60 mm/h rainfall intensities, by drying and weighing the amount of sediments present in the generated runoff after two hours of simulated rainfall. Four topsoil bulk density samples were collected in each plot and vegetation cover was assessed as percentage classes.

The results showed that the average steady-state infitrability was significantly higher, and the average sediment generation rate and bulk density significantly lower, in the treated plots than in the controls. The higher infiltrability rates in the treated plots as compared to the controls were significantly explained by the increment in vegetation cover and the reduction in soil bulk density. Vegetation cover also explained the variation in the amount of sediments generated, which were significantly lower under an enhanced cover. A negative relationship was found between the steady-state infiltrability and the sediment generation rate.

Tree planting practices constitute a feasible and functional management option to reduce erosion by decreasing surface runoff generation through enhanced soil infiltrability, and increasing vegetation cover.

**Keywords:** Lake Victoria basin, Nyando River basin, steady-state infiltrability, bulk density, erosion, gully, runoff, sediment generation rate, double-ring infiltrometer, rainfall simulator, trees, vegetation cover, agroforestry.

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# 1. Introduction

*1.1 The Lake Victoria and its basin: context*

The Lake Victoria Basin, which covers a surface area of 184.200 km$^2$ unevenly shared by Tanzania (44%), Kenya (22%), Uganda (16%), Rwanda (11%) and Burundi (7%) and comprises 11 major river basins and a large lakeshore area, supports one of the densest and poorest rural populations in the world, with densities up to 1,200 persons per square kilometre in parts of Kenya (Hoekstra and Corbett, 1995). National population growth rates, though declining due to the HIV/AIDS pandemic and other diseases, remain among the highest in the world and the population in the five riparian countries is expected to double over the next 25-35 years (UNPB, 2000). Over the last 40-50 years the lake and its basin have undergone enormous ecological changes, which are linked to a number of interrelated problems such as rapid population growth, poverty, land degradation and declining agricultural productivity and water quality (ICRAF, 2000). Sedimentation, nutrient runoff, urban and industrial point source pollution and biomass burning, have induced the rapid eutrophication of Lake Victoria over the latter part of the 20$^{th}$ century, resulting in invasion of water hyacinth, loss of endemic biodiversity and interrelated and compounded problems for the lake environment and the welfare of its people (ICRAF, 2000).

The highlands of western Kenya are home to 12 million people, or 40% of the country's population, but occupy only 15% of the land area. These lands have high agricultural potential, yet recent soil degradation has led to abject poverty in 30 to 50 % of the rural households (ICRAF, 2003). The Nyando River basin, which covers an area of 3,517 square kilometres of Western Kenya and contains some of the most severe problems of agricultural stagnation, environmental degradation and deepening poverty found anywhere in Kenya (ICRAF, 2004) has been identified as a regional erosion hotspot by ICRAF (2001). Approximately 46% of its 3,517 km$^2$ have experienced severe soil physical degradation, the most dramatic forms related to soil erosion, including the formation of gullies (Figure 1) and badlands in parts of the Kano plain and often severe sheet and rill erosion as well and landslides in the upper parts of the river basin, leading to low levels of soil fertility, as evidenced by low cation exchange capacity and soil organic matter contents, lack of mineralizeable nitrogen reserves and phosphorous deficiency. Forty eight percent of the remaining area which does not show signs of accelerated soil erosion has similar soil physical and chemical properties as the sites displaying visible signs of soil degradation. The most degraded parts of the landscape, both in terms of nutrient deficiencies and soil physical degradation, are areas used for open grazing and extraction of fuel wood. According to ICRAF (2000), the major causes of erosion, which are a result of high population pressure, include deforestation of headwaters and overuse of extensive areas of fragile lands on both hillslopes and plains. Furthermore, the Nyando River basin has been identified as one of the main contributors of sediment and phosphorous into the Lake Victoria based on analysis of Landsat TM satellite images. A huge sediment plume is projecting from the outlet of the river more than 20 km into the Winam Gulf (Figure 2). A sediment budget carried out by ICRAF (2004) indicates that the sediment source areas currently occupy more than 50% of the basin and the rates of soil loss in source areas have not been offset by sediment accretion in sink areas of the basin. This leads to an estimated export of $3.2 \cdot 10^6$ Mg $yr^{-1}$ of sediment to the Nyando River since 1963, and the consequent negative effects in water quality (Mbaria, 2006).

*Figure 1. Gully erosion in East-Nyakach (lower Nyando basin)*

The World Agroforestry Centre (ICRAF), Kenyan Agricultural Research Institute (KARI), the Climate Change Office of the Ministry of Envirnoment, Natural Resources and Wildlife, Kenyan Forestry Research Institute (KEFRI), Maseno University, local NGO´s and CBO´s are implementing agencies of Western Kenyan Integrated Ecosystem Management Project (WKIEM), which has fixed as a key development goal to reverse land degradation and promote income-generating activities for rural farmers in Western Kenya and contribute to improving rural water quality, within the context of an Integrated Ecosystem Management approach (ICRAF, 2003). Implementation of measures to reduce soil erosion and increase agricultural productivity started in 2004 in some plots in the Nyando River basin through enhancement of vegetation cover and improved management, e.g. agroforestry, grazing control and cover planting with trees, bushes and grass on eroded lands.

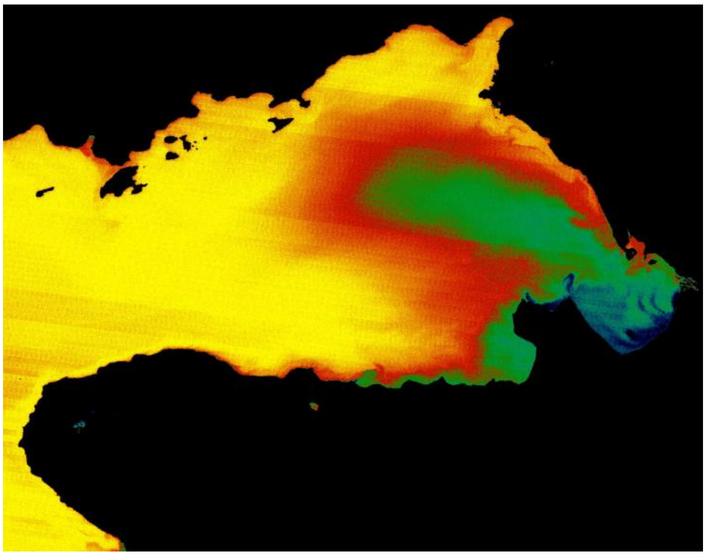

*Figure 2. Nyando sediment plume (~40 km$^2$) in Lake Victoria (based on Landsat ETM data Feb. 2000). Source: ICRAF, 2003.*

### *1.2 Soil water infiltration, land uses and erosion control*

Infiltration refers to the vertical entry of water into the soil through its surface. The rate of this process, relative to the rate of water supply, determines how much water will enter the root zone and how much, if any, will be distributed on the surface (Hillel, 2004). Thus, soil water infiltrability (Hillel, 2004), also known as soil’s infiltration capacity (Horton, 1940), is a key factor controlling not only the amount of water which can become available for terrestrial plants and micro-organisms or can recharge groundwater reservoirs, but also the amount of surface runoff which will be generated and its direct effects on soil erosion and stream flooding (Susswein et *al.*, 2001; Ilstedt et al., 2007, Lipiec et *al.*, 2006). The major adverse on-site effects of erosion are loss of soil available-water holding capacity, organic matter and plant nutrients, with consequent decline in crop yields (Young, 1989, Hashim et *al.*, 1998).

Soil physical properties affecting infiltration are well known and have been reported in many papers. The hydraulic conductivity, and thus the infiltrability, depends on the soil structure as well as on its texture, being greater if the soil is highly porous, fractured or aggregated than if it is tightly compacted and dense (Hillel, 2004). The soil organic matter content is directly related to the percentage of aggregates and their stability as well as with bulk density and macroporosity (Mariscal et *al.*, 2007; Mapa et *al.*, 1995) due to the binding effects of organic products (Hillel, 2004, Grimaldi et al. 2003). Therefore, a decrease in soil organic matter implies a reduction in water infiltration as a consequence of soil structure and porosity loss, destruction of aggregates and pore obstruction by particles, or saturation by water (Brady and Ray, 2002; Hillel, 2004). Furthermore, stability and aggregation also have a direct influence on soil erosion since they affect soil erodibility (Lal and Greenland, 1979). Also related to soil organic matter and litter is the activity of soil macrofauna, that leads to a large macroporosity which in turn results in higher infiltration rates (Léonard et al., 2003, Savadogo et *al.*, 2007).

Land use and land management practices are directly and indirectly linked to soil physical properties and can also affect infiltrability by other means (e.g. terraces), determining therefore the potential top soil loss by erosion. Land use intensification results in a reduction in organic matter supply to soils, and a decrease in soil fertility and productivity, which in turn leads to soil degradation and increased erosion, with the consequent exposure of horizons with even lower infiltrability (Ilstedt et *al.*, 2007; Susswein et *al.*, 2001).

Beneficial effect of tree planting on soil infiltrability over a wide range of humidity levels in the tropics has been confirmed by Ilstedt et *al.*, (2007), supporting the idea that trees improve soil physical properties (Young, 1989, Grimaldi et *al.*, 2003). Mapa (1995) attributes the higher infiltration rates in reforested areas mainly to the high organic matter content in soils. Conversely, deforestation increases run-off due to decreased infiltration rates and lower soil water retention (Mapa, 1995; Susswein et *al.*, 2001), since soil structural degradation often occurs when forest or savannah vegetation is replaced by annual cropping systems (Grimaldi et *al.*, 2003).

Agroforestry practices encompass an entire spectrum of land use systems in which woody perennials are deliberately grown in association with herbaceous plants (crops, pastures) and/or livestock in some spatial or temporal arrangement (Young, 1989). This form of land use has two main objectives: productivity, involving a multitude of outputs, and sustainability, which implies the conservation, or even improvement of the environmental aspects of the system (Huxley, 1983).

Cultivation of trees can improve productivity trough biological nitrogen fixation, enhanced efficiency of nutrient use, reduction in soil erosion and improvement in soil physical conditions (Dadhawl and Solanki, 2003). According to Young (1989), the greatest potential of agroforestry in soil erosion control lies in its capacity to supply and maintain ground cover, which has experimentally been proved to reduce soil loss (Rose, 1994), trough litter incorporation from tree pruning. Mulching the soil surface with a layer of plant residue is an effective method of conserving water and soil because it reduces surface run-off, increases infiltration of water into the soil and retards soil erosion. In addition, mulching reduces the depletion of water within the root zone as a result of lower evaporation rates, and decreases crusting of the soil due to rainfall impact which in turn leads to reduced erosion because there is less particle detachment (Adekalu et *al.*, 2007, Omoro and Nair, 1993). Agroforestry can also reduce erosion due to the maintenance or increase of organic matter and its diversity, through continuous decomposition of litter and roots, in addition to the role that trees play in improving soil physical properties through their extensive root system and the enhancement of faunal activity (Young, 1989, Grimaldi et *al.*, 2003). McDonald (2002) concluded that rates of surface runoff and erosion were consistently and significantly lower in agroforestry than in agricultural treatment after forest clearance, which is probably linked to the fact that the transformation of original forest into various types of agroforests result in a smaller decrease in carbon stocks than the transformation of forests into croplands, pastures or degraded grasslands (Sanchez, 2000).

Apart from the implications organic matter has in soil erosion control, it is considered one of the most important components of soil fertility, especially in agriculture with low external inputs, where practices that enhance soil organic carbon are likely to enhance productivity as well (Schroth, 2003; Pierce and Lal, 1994). Subhrendu and Mercer (1996) provide empirical verification for the claim that agroforestry enhances soil quality and that this is positively correlated with net household income, which is, according to Garrity (2004), the key factor to make it attractive to farmers. In addition, through domestication tree crops could improve household welfare by providing traditional food and health products, boosting trade, generating income and diversifying farming systems, both biologically and economically, beyond the production of basic food crops (Sanchez and Leakey, 1997). Agroforestry appears to be promising to enable smallholders to increase on farm-food production and reduce erosion. Furthermore, since commercial fertilizers often cost two to six times as much in Africa as in

Europe or Asia, using fertilizer trees (N-fixing trees) is an economically attractive way to manage on-farm soil fertility (Garrity, 2004).

From the environmental point of view, agroforestry services that are of greatest relevance include watershed protection, biodiversity conservation and climate change mitigation and adaptation through increased carbon sequestration in soils and biomass (Garrity, 2004, Sanchez, 2000; Albretch and Kandji, 2003; Montagnini and Nair, 2004).

Maintaining an adequately high level of surface cover may be all that is necessary to provide an adequate level of protection against soil erosion, since in addition to the protection it offers against raindrops impact, a high level of cover also maintains high rates of infiltration, and therefore less runoff (Rose, 1994). In addition, afforestation of grasslands, shrublands and croplands has been found to increase evapotranspiration and thereby reduce stream flow (Jackson et *al.*, 2005).

There are numerous studies which report the negative effects of severe grazing on soil physical properties and thus on erosion and land degradation. The idea of increased soil bulk density, decrease in hydraulic conductivity and water infiltration and increase in surface runoff is commonly shared (Savadogo et *al.*, 2007; Greenwood and McKenzie, 2001; Willatt and Pullar, 1984). Cattle grazing results in mechanical pressure on the ground as animal trampling contribute to altered soil structure, soil compaction and reduction of soil porosity. Infiltration is also decreased due to reduced vegetation cover and amount of organic matter in the topsoil (Savadogo et *al.*). However, light and well managed grazing can improve fertility and physical properties of the soil through recycled manure, and trampling can stimulate seed germination and break-up hard soil crusts (Mohamed Saleem, 1998). According to Girma et *al*. (2007), the degree of soil physical and hydrological degradation depends on how livestock is managed in the grazing land.

The objective of this study is to answer the following hypothesis:

"*Water infiltrability into the soil will be higher on the treated WKIEM plots than in the adjacent areas, while water runoff and sediment generation will be reduced. The increase in infiltrability is inversely related to the amount of generated sediments and linked to an increase in vegetation cover and a decrease in soil bulk density.*

## 2. Material and methods

*2.1 Study area – The Nyando River basin*

The study was carried out in the lower area of the Nyando River basin (0º, 35º 10’E), near Kisumu city in Western Kenya, East Africa (Figure 3).

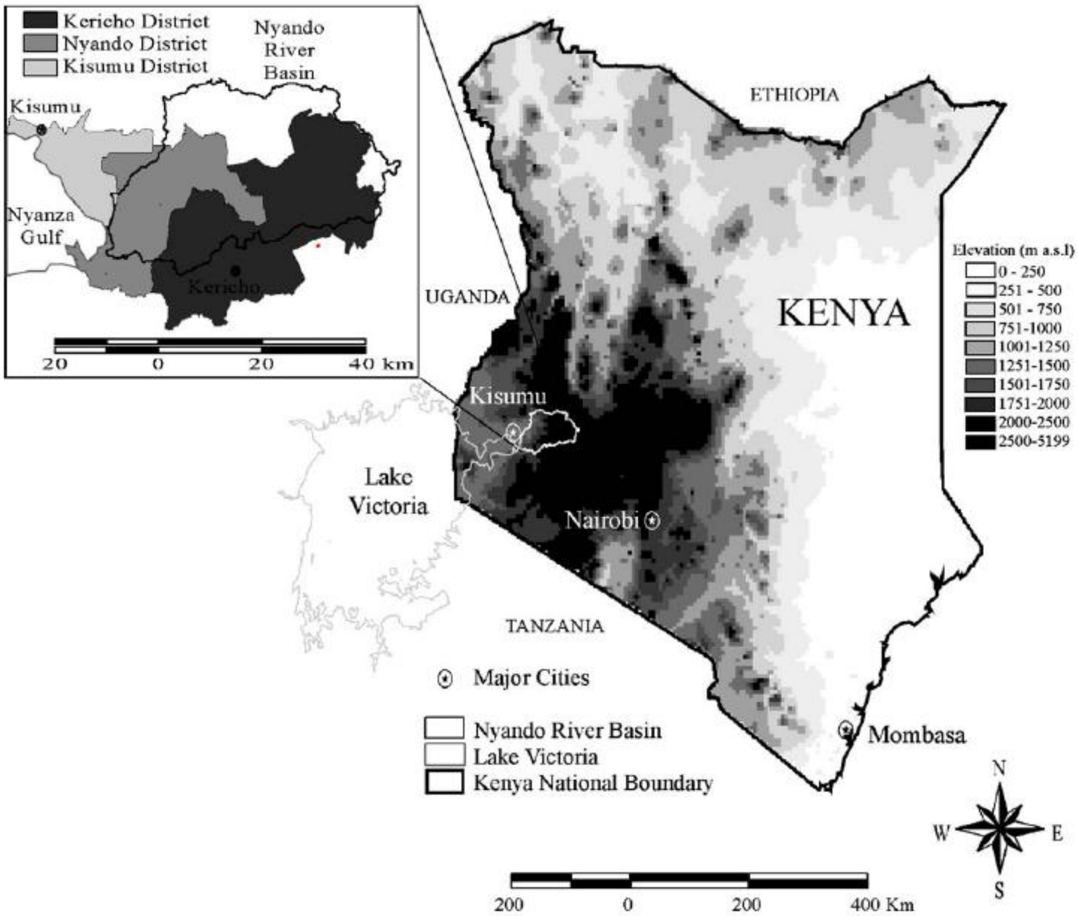


*Figure 3. Map of Kenya, Kisumu, Nyando and Kericho districts and Nyando River basin. Also shown is national elevation map, Lake Victoria and Kenya’s major urban centres. Source: Cohen et al., 2006*

2.1.1 Population and administrative boundaries

Administratively, the basin is divided into 17 Divisions in 6 Districts, Nyando, Nandi, Kericho, Nakuru, Baringo and Uasin Gishu. Nyando District is in Nyanza Province, while the rest falls into the Rift Valley Province (ICRAF, 2000).

The Nyando basin supports a population of approximately 746,000 inhabitants, with the districts of Kericho (42%), Nyando (35%) and Nandi (19%) accounting for 96% of the total. The average population density is 214 persons per $km^2$, with some areas of the basin exceeding 1,200 persons per $km^2$. Rapid population growth, ~3.2% $yr^{-1}$over the last 50 years, has lead to cultivation of marginal lands on steep slopes, shorter agricultural fallow periods, decreased natural vegetation and an increased number of livestock (ICRAF, 2003; Verchot et *al.*, 2007).

### 2.1.2 Hydrology and Geomorphology

The river Nyando drains into the eastern portion of the Winam (or Nyanza) Gulf of Lake Victoria. Its associated drainage basin is 3,590 $km^2$ in size and the total drainage network length is approximately 2,175 km. The river has three main tributaries: Ainabngetuny, Kipchorian and Awach (Onyango et *al.*, 2005). The general drainage pattern of this area is controlled by the level of Lake Victoria in the west, which currently stands at approximately 1,138 meters a.s.l., and the peak of the Tinderet Volcano at 2,930 m a.s.l in the north-eastern portion of the basin. The basin is divided into lowlands (<1,500m) and highlands (>1,500), separated by the steep Nandi and Mau escarpments. At the base of the scarps numerous streams cut deeply through poorly-sorted beds of coarse gravel, sands and sandy clays in the Kano plain. Steep hydrological gradients occur in the basin with long slopes in excess of 20º inclination (ICRAF, 2003).

### 2.1.3 Climatology

The lowland climate is sub-humid tropical (~1,100 mm rain/year), with a seasonally bimodal rainfall distribution characteristic of African equatorial latitudes located near the inter-tropical convergence zone (ICRAF, 2003). The first rainy season is the most pronounced and is experienced throughout the basin from March to May (Verchot et *al.*, 2007). The second rainy season differs slightly depending on the location, but usually occurs in September/October (Onyango et *al.*, 2005). Highlands climate is humid tropical (~1,700 mm rain/year) and generally shows an attenuated bimodal rainfall distribution supplemented by convectional rainfall from Lake Victoria (ICRAF, 2003).

Temperatures remain relatively stable throughout the year, although average annual temperatures change spatially depending on the altitude. In the lowlands, average annual temperature is 22.2 ºC while in the highlands it is 16.9 ºC (Verchot et *al.*, 2007). The annual monthly maximum temperatures in the lowlands range from 29 to 31 ºC, while the annual monthly minimum temperatures range from 12 to 16 ºC (Onyango et *al.*, 2005). In the highlands, the annual monthly maximum temperatures range from 19 to 27 ºC, while the annual monthly minimum temperatures range from 5 to 12 ºC (Onyango et *al.*, 2005).

### 2.1.4 Native vegetation and land uses

In the highlands, native vegetation communities, now rare, consist of evergreen broadleaf forest, where the most important tree species are *Croton megalocarpus, Diospyrus abyssinica, Funtumia latifolia, Olea welwetchii, Dombeya spp* and *Dovyalis abyssinica.* In the lowlands, a main distinction can be made between the native vegetation from the plains and the one from the mid-slopes; in the first case, grasslands (*Eragrostis spp., Cynodon dactylon* and sedges) with scattered acacia trees like *Acacia seyal, Acacia drepanalobium* and *Acacia hockii* dominate. Native communities in the mid-slopes include perennial grasslands (*Themeda triandra, Hypairhenia hirta, Panicum spp., Eragrostis spp.,* among others) interspersed with evergreen and semi-deciduous bushlands (*Dodonea angustifolia, Carissa edulis, Rhus natalensis, Rhus vulgaris* and *Euclea divinorum* mainly). In the inland valleys and at the river mouth, *Cyperus spp.* wetlands and riparian vegetation (*Ficus natalensis*, and thickets of *Dombeya spp.,* among others) are the main native communities.

The basin includes areas of cultivated arable land used for agricultural production and areas of agro-pastoral use for both crop cultivation and rising of livestock. Within the cultivated areas a mixed cropping system is used, where cereals (*Zea mais,* maize; *Sorghum bicolor,* sorghum; and *Panicum milaiceum,* millet), pulses (beans; *Vigna radiata,* green gram) and root (*Manihot esculenta,* cassava; *Maranta arundinacea,* arrowroot; *Ipomoea batatas,* sweet potato) are grown (Verchot et *al.*, 2007).

Dominant agricultural land-uses in lowland areas are maize (*Zea mais*), sorghum (*Sorghum bicolor*), sugarcane (*Saccharum officinarum*), irrigated rice (*Oryza sativa)* and communal pasture (ICRAF, 2003). The plain is flood-prone and several rice irrigation schemes have been set up. Annual flooding near the delta leaves rich alluvial deposits that are cultivated and yield good harvests (Onyango et *al.*, 2005). In the highlands, tea (*Camellia sinensis),* maize (*Zea mais),* sugarcane (*Saccharum officinarum*), commercial woodlots and restricted grazing are the dominant land uses (ICRAF, 2003).

During the last 40 years, the region has experienced dramatic land use changes, as the land was converted from small-scale farming to intense smallholder cultivation. The impacts of these changes on the environment are seen in emerging environmental problems such as deforestation and landslides (Onyango et al., 2005). The majority of the watershed is more or less continuously cropped. The few exceptions are two remaining forest areas, Tinderet and Mau forests, that are currently being heavily deforested, and the steep sloping escarpments- originally Government trust land- that are quickly being devegetated due to charcoal burning and illegal farming (ICRAF, 2000).

### 2.1.5 The soils

Soils in the Kano plain are derived from Holocene alluvial deposits. Luvisol, Vertisol (locally known as Black Cotton soils), Planosol, Cambisol and Solonetz types (FAO-UNESCO, 1988) are common, frequently in saline or sodic phases (Andriesse and van der Pouw, 1985), with deep profiles and moderate to low fertility (Cohen et *al.*, 2006).

Highland soils are derived from a wide variety of parent materials including phonolites, quartzites, nephelinites, granitoid gneisses and intrusives such as dolorites, monzonites and granidiorites, which are representative of a large part of the Kenyan portion of the Victoria Lake Basin. Predominant soil types (FAO-UNESCO, 1988) include Ferrasols, Nitisols, Cambisols and Acricsols (Andriesse and van der Pouw, 1985), and are structurally stable (Cohen et *al.*, 2006).

## *2.2 Experimental design and general outline*

A comparative study was carried out between plots where rehabilitation measures were implemented within the WKIEM project context (treated plots, n=9) and the adjacent control areas (untreated plots, n=9) (Figure 4). In each plot the studied variables were steady state infiltrability (mm/h), bulk density ($g/cm^3$), vegetation cover (%), sediment generation rates (Mg $ha^{-1}$ $h^{-1}$) and soil textural class. The filed work started in April 2009, two weeks after the first rains of the season, and went on until the end of May.

*Figure 4. Treated plot (left) and its control, the adjacent area where trees are not present (right)*

Within the treated plots a distinction was made between woodlots, where native and exotic species were grown to provide wood to the families in plots close to the houses, and rehabilitation sites, which were initially much degraded plots where the vegetation cover had been enhanced and grazing and fuel collection restricted. From a total of nine treated plots, five consisted on woodlots and four were rehabilitation sites. Two of the rehabilitation plots were situated in the upper part of a gully and the other two in the middle part of its slope. The woodolots were all located in the plain where the gully was inserted. This extended 9 km along Asawo River and was 50 m deep at its deepest point.

Infiltrability was measured using the double-ring infiltrometer method (Bouwer, 1986) in six sample points per plot and a small rainfall simulator in two sample points, working with a rainfall intensity of 40 and 60 mm/h respectively. Rainfall simulations were run for two hours and the runoff was collected in buckets in order to quantify the amount of sediments generated for both rainfall intensities during this time interval. Hydrochloric acid was added to the runoff solution to precipitate the sediments in the water, which was removed using a syringe. The sediments were afterwards dried in the oven at 70 ºC and weighted. Sediment generation rate (Mg $ha^{-1}$ $h^{-1}$) was calculated for both rainfall intensities of 40 and 60 mm/h.

Surface soil bulk density and water content was determined from four undisturbed samples per plot. The samples were collected using a cylindrical sampler 5 cm long and 5 cm in diameter, and weighted in the field. Afterwards they were dried in the oven at 70 ºC and weighted again.

Topsoil texture class (USDA, 2009) was assessed in the field by feel in one sample point per plot.

Vegetation cover was assessed in each plot as percentage class, describing the vegetation type (short grass, long grass, bushes/shrubs and trees) and the main trees and shrub species present in the plot.

The sample points were randomly chosen and followed a geometrical pattern, when possible, where distances changed depending on the size of the plot (figure 5).

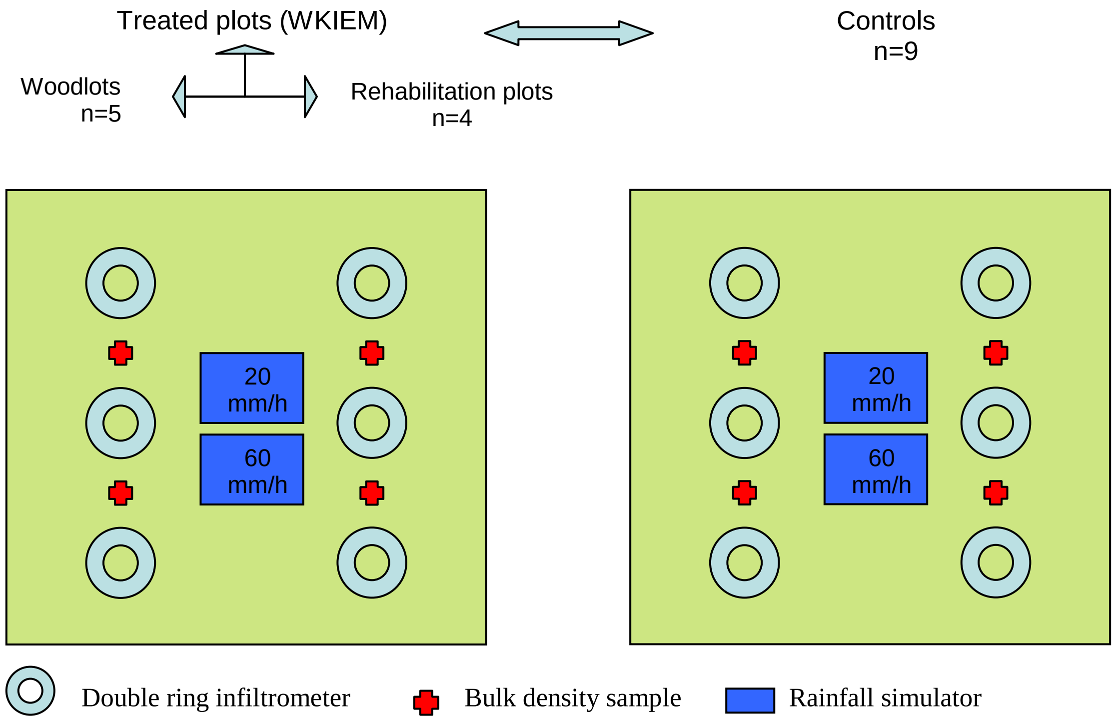


*Figure 5. Experimental design overview and sample points distribution within the plot.*

## *2.3 Infiltrability*

### 2.3.1 Theoretical basis

The infiltration rate is defined as the volume flux of water flowing into the profile per unit of soil surface area. For the special condition in which the rainfall rate exceeds the ability of the soil to absorb water, infiltration proceeds at a maximal rate, which Horton (1940) called the soil's infiltration capacity, for which later Hillel (1971) suggested the term infiltrability. Therefore, infiltrability can be measured using soil controlled infiltration tests, such as cylinder infiltrometer and sprinkling infiltrometer (rainfall simulations).

In general, soil infiltrability is relatively high in the early stages of infiltration, particularly when the soil is initially quite dry, but it tends to decrease and eventually to approach asymptotically a constant rate known as steady-state infiltrability or final infiltration capacity (Figure 6).

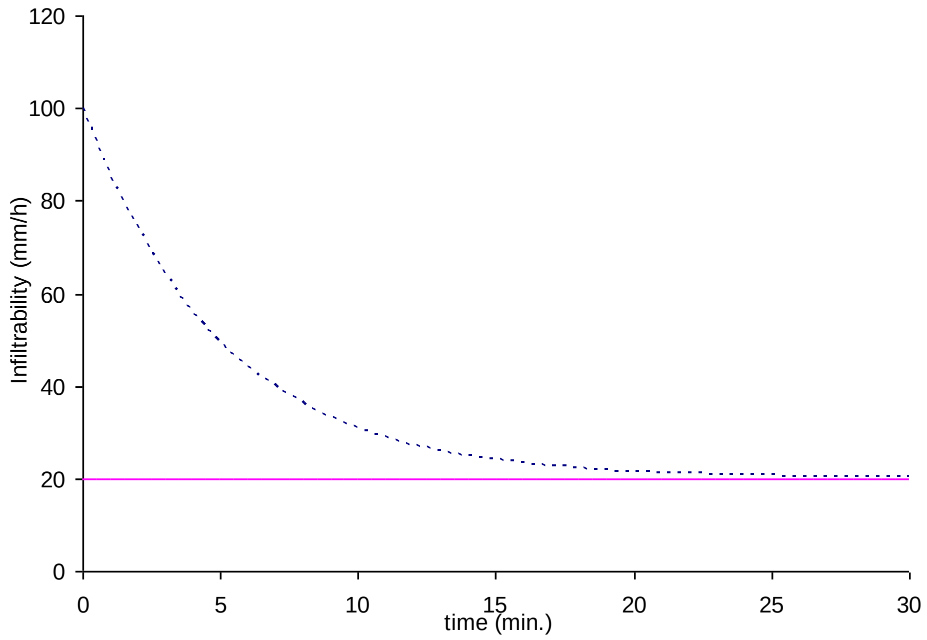


*Figure 6. Theoretical infiltrability curve (dashed line) and steady-state infiltrability constant (continuous line)*

Downward infiltration into an initially unsaturated soil is generally due to two main gradients or moving forces: the gravitational head (δz/δz=1/1) and the pressure head (δh/δz) - or matric suction gradient when it is negative, i.e. unsaturated soil-. This is reflected in Darcy's law:

$$q = -k(h)\frac{\partial H}{\partial z}$$

$$q = -k(h)\frac{\partial (h+z)}{\partial z} = -k(h)(\frac{\partial h}{\partial z} + 1)$$

where:

- q (mm/h) is the flux density, in our case the infiltration rate
- k(h) (mm/h) is the hydraulic conductivity of the wetted zone for a given pression head, h
- δH/δz (m/m) is the hydraulic head gradient
- δz/δz (m/m) is the gravitational head gradient
- δh/δz (m/m) is the pressure head gradient

The decline in infiltrability with time results mainly from the gradually decrease in the pressure head gradient (δh/δz), which occurs inevitably as infiltration goes on. As the infiltration proceeds and the depth of the wetting front increases, the effect of the pressure head gradient decreases and eventually becomes negligible. From this moment, the gravitational gradient, which equals one, is the main moving force that acts and it follows that the infiltration rate tends to approach the hydraulic conductivity as a limiting value. In a uniform profile under prolonged pounding, the water content of the wetted zone should, theoretically, approach saturation. Hence, the infiltration rate in such conditions will tend asymptotically to the saturated hydraulic conductivity, which will represent the steady-state infiltrability (Hillel, 2004).

For rainfall simulations of a uniform intensity greater than the soil's infiltrability the theoretical infiltrability continuous function (Figure 6) becomes step-like as shown in Figure 7. Initially, the rate of infiltration equals the intensity, but as infiltration proceeds, the soil infiltrability decreases and, as soon as the rainfall intensity exceeds the soil's infitlrability, the latter becomes the actual infiltration rate. The rainfall excess starts to pond on the surface and may generate run-off. The elapsed time between the start of the simulation and the moment at which the break in the infiltration curve occurs is called the time to ponding. From the time to ponding, the infiltration rate follows the infiltrability curve (Cammeraat, 2005).

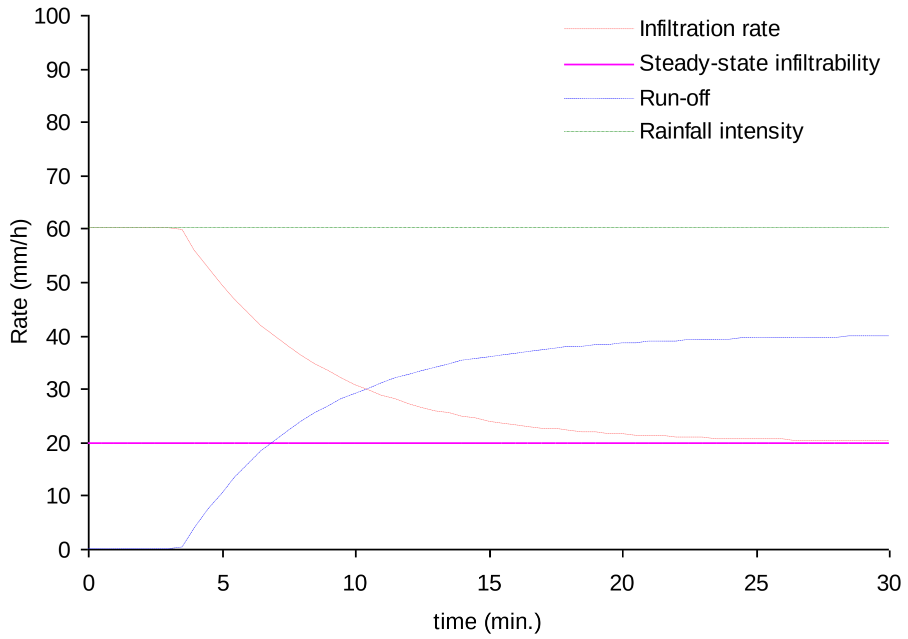


*Figure 7. Rates of infiltration and run-off versus time for infiltrability measurements with rainfall simulators working at a constant rainfall intensity.*

2.3.2 Field measurement

Infiltrability was measured in the field using the double-ring infiltrometer method (Bouwer, 1986) and rainfall simulations.

In the double-ring infiltrometer test (Figure 8.a), two metal cylinders, with diameters 20 and 30 centimetres respectively, were driven a proper distance into the soil in order to avoid any water leakage, the smaller placed concentrically inside the larger one. Water was poured into the inner ring carefully to avoid disturbing the soil surface, and also in the area between the inner and the outer ring, maintaining a head of 3-4 cm during the measurements. Water level in the inner ring was measured every 3 minutes (Figure 8.b) during 17.30 minutes long intervals per hour in each plot, having 5 infiltrability measurements per ring every hour, until a constant infiltration rate was reached or after four hours since the measurements started.

The idea behind the double-ring infiltrometer is to let the outer space between the two rings to “absorb” all the edge and divergence effects due to non-vertical flow, so that the infiltration from the inner ring would be a true measure of the vertical infiltration rate of the soil (Bouwer, 1986). The simplicity and low cost of the method are its main advantages (Teixeira et *al.*, 2003).

*a)* *b)*

*Figure 8. a) Double-ring infiltrometer. b) Water level measurement in the inner ring.*

Rainfall simulations were done using a small drip-plate type (drop-former type) rainfall simulator (Figure 9.a), based on the model from Pla Sentís (1981). The drip-plate was 30x20 $cm^2$ and was set up 2 m above the surface level, where a runoff collector with the same dimensions was installed in such a way that the projected area of rainfall overlapped the plot from which the run-off was collected (Figure 9.b). The collector was made of four metallic sheets bound together which were inserted some centimetres into the soil to prevent any leakage. In one of the walls, a pipe set at the surface level allowed the run-off evacuation and collection. Runoff was collected every five minutes during two hours.

Rainfall intensities of 20±5 mm/h and 60±5 mm/h were used in each plot, i.e. two rainfall simulations per plot. At the beginning of the experiment water was collected in a tray during five minutes to determine the initial intensity. This was repeated two more times at the end of the measurements, after two hours. The infiltration rate was afterwards deduced by substracting the runoff rate from the average intensity. In those cases where no runoff was generated during the two hours, the steady-state infiltrability was set to 60 mm/h.

Rainfall simulations are more apt to simulate natural processes, such as runoff generation and sealing, than the double-ring infiltrometer tests (Cammeraat, 2005; Teixeira et *al.*, 2003 ).

*a)* *b)*

*Figure 9. a) Rainfall simulator. b) Runoff collecting plot*

2.3.3 Data analysis

The steady-state infiltrability and the sorptivity constants were estimated by means of curve-fitting to Philip's equation (Philip, 1957) by minimizing the squared residuals using the Excel's solver tool.

According to Philip's equation:

$$I = st^{1/2} + i_c t$$

which, deriving (δt) leads to:

$$i = \frac{s}{2t^{1/2}} + i_c$$

where:

- I (mm) is the cumulative infiltration at time t
- i (mm/h) is the infiltration rate at time t
- s (mm/h$^{½}$) is the term defined as sorptivity
- $i_c$ (mm/h) is the term defined as transmissivity or steady-state infiltrability

The main assumption from this equation is that an infinitely deep, uniform soil of constant wetness ($\theta_i$) is assumed at time 0 to be submerged under a thin layer of water that instantaneously increases soil moisture at the surface to a new value near saturation ($\theta_o$), that is thereafter maintained constant (Hillel, 2004). This equation is applicable to head-controlled tests and to rainfall simulations once ponding has occurred (Cammeraat, 2005).

The sorptivity constant depends both on the initial volumetric water content of the uninfiltrated soil ($\theta_i$), and the saturated wetness ($\theta_o$) (Hillel, 2004). For low initial soil moisture contents there is a high sorptivity due to the greater matric suction gradients. This is reflected by high infiltration rates at the first stages under ponding and thus a steeper slope when plotting infiltrability versus time.

The soil's transmissivity is the infiltration rate at the steady-state, or steady-state infiltrability, which can be assumed to be equal to the saturated hydraulic conductivity.

*2.4 Statistical analysis*

A descriptive statistical analysis was done for each variable using the data obtained from the "means" procedure in SAS (SAS Instiute, 1998). The mean value, minimum, maximum, standard deviation, lower quartile, mediane and upper quartile were calculated for treated (n=9) and control plots (n=9). Due to the different treatments that existed within the treated plots, the descriptive statistical values were also calculated for woodlots (n=5) and rehabilitation sites (n=4) separately, and their respective controls (n=5 and n=4).

A unilateral hypothesis means contrast was done for all the studied variables in order to find if there were any significant differences (the significance level was set to 95%) in the mean value of a given variable between treated and control plots. A two-sample Student´s t-test in SAS (SAS Instiute, 1998) was used to solve the hypothesis means contrast. Furthermore, a unilateral hypothesis medians contrast was posed using the Wilcoxon two-sample test in SAS (SAS Instiute, 1998), procedure “Npar1way”. Both contrasts were also done separately for woodlots, rehabilitation sites and their respective controls.

Linear regression models were fitted to different sets of data relating pairs of studied variables (steady-state infiltrability, bulk densinty, vegetation cover, sediment generation rate) and afterwards an analysis of variance (ANOVA) was done using the “Reg” procedure in SAS (SAS Instiute, 1998) to find out if the entered dependant variable was significantly explained by the independent one or not.

## 3. Results and discussion

### *3.1 Vegetation cover*

The average total vegetation cover at the treated and control plots was 135% and 55% respectively. In the control plots this cover was mainly constituted by grass, although some sparse bushes and shrubs (*Acacia seyal, Acacia drepanalobium* and *Acacia gerradii*) appeared as well in a few cases. A difference could be made between the control plots linked to the woodlots and the ones linked to the rehabilitation sites according to its vegetation cover. While in the woodlots controls the total vegetation cover was 69% in average, in the rehabilitation sites controls this was reduced to approximately 37% due to the greater degradation of these sites. It should be observed that the grass cover was not constant all year around and when it was estimated, during the rainy season, grass appeared in places that were bare during the major part of the year. The main present grass species were *Cynodon dactylon* and *Sedge* spp.

In the treated plots the total vegetation cover was higher in part due to the presence of trees and more shrubs. In contrast to the control plots, in the treated plots the vegetation cover was continuous in the site due to a more or less regular plantation pattern of the trees and a regular grass cover. The rehabilitation sites were covered mainly by grass and shrubs but no yet trees, the most important shrubs species being *Acacia polyacantha, Acacia senegal, Acacia* seyal and *Acacia tortilis,* all of them native. In the woodlots the tree-shrub cover was much more diversified in species because one of their purposes was to provide a large range of products to the families, and introduced species appeared as well together with the native; some of the most common species were *Grevillea robusta, Leucaena leucocephala, Markhemia lutea, Cassuarina equistifolia, Terminalia brownii, Croton megalocarpus, Senna siamea, Albizia seman, Harissonia abyssinica, Lantana camara, Azadirachta indica, Balanites aegyptiaca, Acacia polyacantha, Acacia seyal, Acacia gerrardii, Acacia nilotica, Acacia perciciflora, Acacia drepanalobium, Acacia tortilis* and *Acacia senegal.* In general, it could be observed that native species performed much better (surveillance, height, diameter) than the introduced ones despite the perceptions of local people on the contrary prior to planting.

### *3.2 Texture and bulk density*

The most common top soil texture classes in the studied plots were clay, sandy-clay-loam and silty-clay. Sandy-clay and silty-clay-loam top soil texture classes were also found in a few plots. The top soil texture class was the same or very similar between paired treated-control plots and thus differences in this variable are assumed to have negligible effect on infiltration in the present study.

The average dry bulk density in the treated plots was 1.16 g/cm$^3$ and was significantly lower (P=0.004) than the one obtained in the control plots, which was 1.34 g/cm$^3$ (Table 1, Figure 10), reflecting the higher degree of topsoil compaction in the control plots as compared to the treated ones. This difference was significant between the two different treatments (rehabilitation sites and woodlots) and their respective controls as well (P=0.042 and P=0.038, Table 1). The lower bulk density values obtained in the treated plots are probably due to the increment in soil organic matter (SOM) that usually follows the presence of trees (Scrotch, 2003; Yimer et *al.*, 2008; Nyamadzawo, 2007 and Celik, 2004) and the consequent soil structure improvement due to the SOM binding effect (Mapa, 1995; Mariscal, 2007 and Scrotch, 2003.) and to the increase in root and faunal activity.

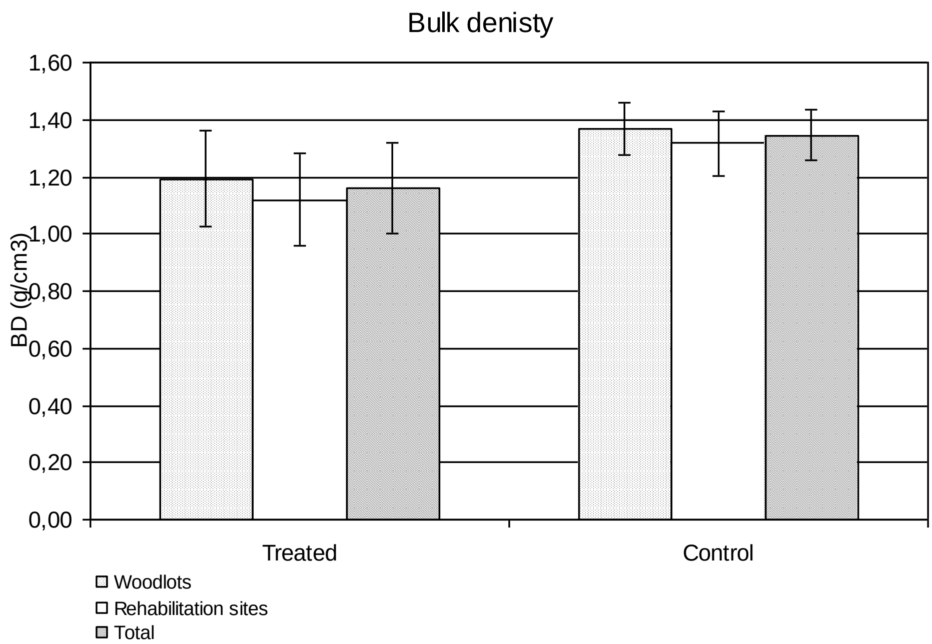


*Figure 10. Average dry bulk density (g/cm$^3$) for treated (woodlots, rehabilitation sites and total) and control plots in the study area (East-Nyakach, lower Nyando). The error bars indicate the SD.*

### *3.3 Steady-state infiltrability*

Average steady-state infiltrability was significantly higher in the treated plots than in the controls for both used methodologies (Table 1, Figure 11); the average steady-state infiltrability obtained with the double-ring infiltrometers and the 60 mm/h and 20 mm/h rainfall simulations was, for the control plots, 6.55 (±4.94) mm/h, 1.97 (±5.91) mm/h and 3.15 (±4.09) mm/h, while in the treated plots these values were increased to 29.35 (±19.46) mm/h, 28.45 (±25.50) mm/h and 27.75 (±24.41) mm/h respectively, and were significantly higher (P=0.004; P=0.007 and P=0.008). The P-values for the difference of the average steady-state infiltrability values of the woodlots and their controls were all <0.1 (P=0.053; P=0.068 and P=0.054), indicating a higher infiltrability in the woodlots. In the case of the rehabilitation sites, the average steady-state infiltrability values were significantly higher than in their controls for both the double-ring infiltrometer and the 60 mm/h rainfall simulations methodologies but not for the 20 mm/h simulations (P=0.016; P=0.049 and P=0.07).

The fact that the average steady-state infiltrability was lower in the rehabilitation sites than in the woodlots, but that the difference from the controls was more pronounced in the first case (Table 1, Figure 12) is probably due to a worse soil degradation state in both the controls linked to the rehabilitation sites and the original soils in these rehabilitated plots, which probably has lead to a greater response to the tree-planting treatment.

Furthermore, the median steady-state infiltrability obtained with the double-ring infiltrometers and the rainfall simulations at 20mm/h and 60 mm/h, which was 5.73, 0 and 0.53 mm/h respectively for the control plots and 24.41, 26.52 and 11.26 mm/h for the treated plots, was significantly higher in the treated plots than in the control plots for the three methodologies used (P<0.001, Table 2). From the total set of individual ring data, the median steady-state infiltrability appeared to be higher for the treated plots than for the controls as well (Figure 13). As shown in Figure 13, none of the rings in the control plots reached a steady-state infiltrability over 40 mm/h while in the treated plots about a 1/3 of the measurements were higher, and values up to 130 mm/h were registered. Moreover, the trend of the cumulative frequency diagram in Figure 13 shows that the steady-state infiltrability corresponding to any given cumulative percentage was always higher in the treated plots than in the controls.

The latter results are consistent with the observations reported by other authors in the sense that land uses including trees improve the steady-state infiltrability of soils in comparison to land uses without trees, both in the tropics (Nyamadzawo et *al.*, 2007; Yimer et *al.*, 2008; Ilstedt et *al.*, 2007; Malmer et *al.*, 2009; Mbagwu, 1997; Mapa, 1995;) and other latitudes (Celik, 2005). In spite of the observed significant differences in the average steady-state infiltrability between treated and control plots it is worthy of notice that in both cases the soils would be susceptible to generate runoff, and therefore vulnerable to erosion, given any rainfall episode that exceeded an intensity of 30 mm/h, which is not rare in the area (Rowntree, 2006).

However, it is important to remember that the trees were planted four years ago and that within this short period of time relative great improvements have occurred concerning the infiltration rates of water into the soil. Therefore, we should expect the soil infiltrability to increase in the future in the treated plots and lead to a greater reduction in runoff generation. Also, the provability of the surface runoff generated to infiltrate in the soil again at spots with higher infiltrability is much higher in the treated plots.

The average coefficient of variation (CV) for the double-ring infitrometer steady-state infiltrability measurements was 73.77% in the treated plots and 132.84% in the control plots (Table 1), indicating the large spatial variability in soil hydraulic properties in accordance to the literature (Teixeira et *al.*, 2003; Williams & Bonell, 1988; Bouwer, 1986, Van de Genachte et *al.*, 1996). Mbagwu (1997) reports a CV of 39.3% in Philip’s predicted quasi-steady infiltrability measured using double-ring infiltrometers in highly permeable tropical savannah soils from Nigeria. Van de Genachte et *al.* (1996) obtained CVs of 81% and 33% in steady-state infiltrability predicted by Philip’s equation and measured using single ring infiltrometer for Ferralsols and Arenosols respectively from a tropical rain forest in Guyana.

This spatial variability could be reduced by increasing the diameter of the infiltration rings or by increasing the number of samples per plot (Williams & Bonell, 1988; Bouwer, 1986); the first option can be a good idea in those cases where the plots to be studied are easily accessible and one should not carry the rings by himself, but this is not usually the case in many field situations. Moreover, larger even surfaces would be needed and it would be more difficult to introduce the rings into the soil. The second option involves more measurements and therefore more time. A compromise decision should be made in order to reduce the existent variability in a feasible way. In our case, however, the fact that we got significant differences in spite of large variation indicates that the number of samples per plot was enough.

The significant (P=0.005, Table 1) increment in relative variability between treated and control plots can be due to a combination of two factors: the existence, in the control plots, of a greater number of rings where zero steady-state infiltrability values where obtained, and the presence of a more irregular vegetation cover in the control plots and thus more spatial variability in these soil physical properties related to the vegetation. The absolute variability, however, was larger in the treated plots than in the controls since in the first case the average steady-state infiltrability was higher.

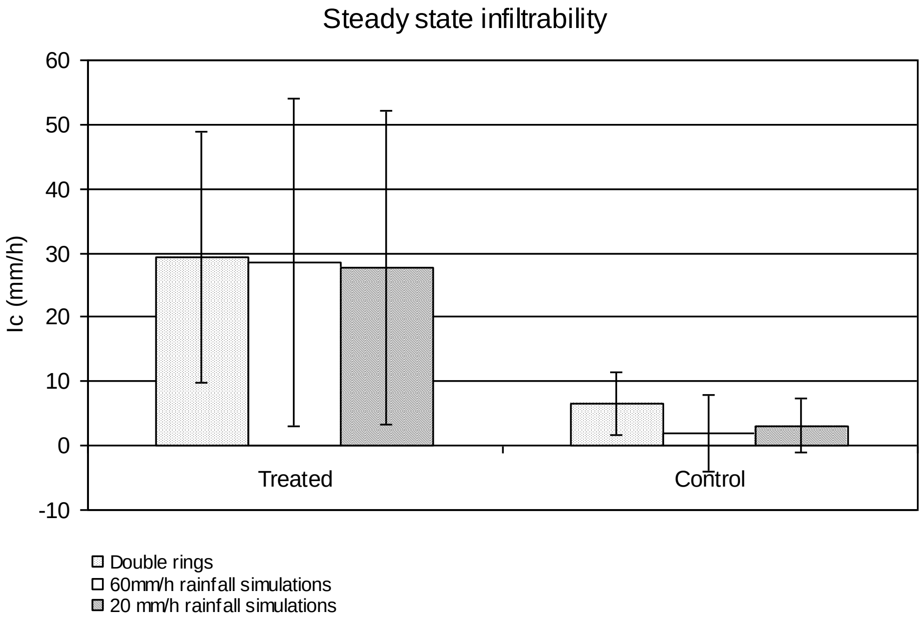


*Figure 11. Average steady-state infiltrability (mm/h) obtained with the double-ring infiltrometers and 60 mm/h and 20 mm/h rainfall simulations for both treated and control plots in the study area (East-Nyakach, lower Nyando). The error bars indicate the SD.*

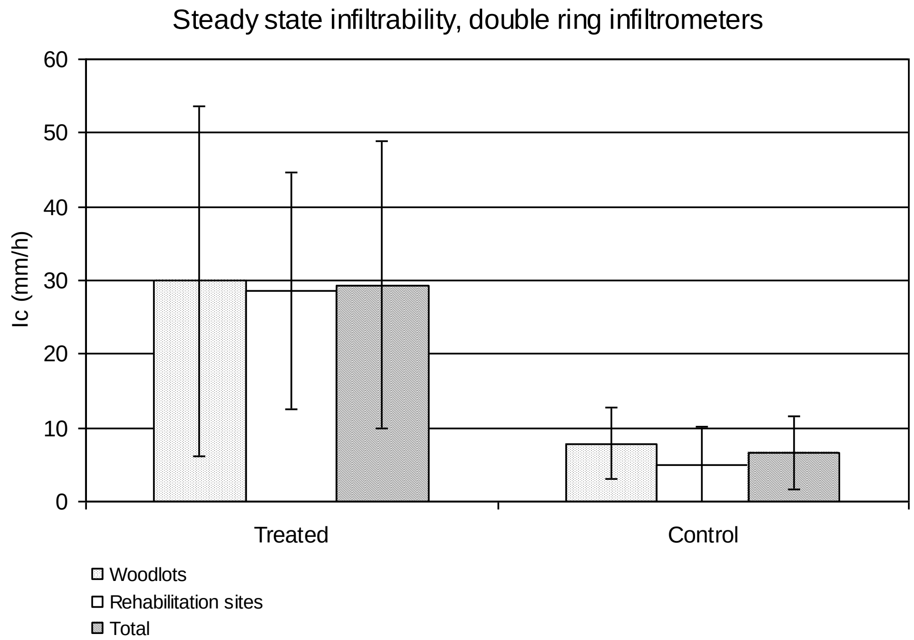


*Figure 12. Average steady-state infiltrability (mm/h) obtained with the double-ring infiltrometers for treated (woodlots, rehabilitation sites and total) and control plots in the study area (East-Nyakach, lower Nyando). The error bars indicate the SD.*

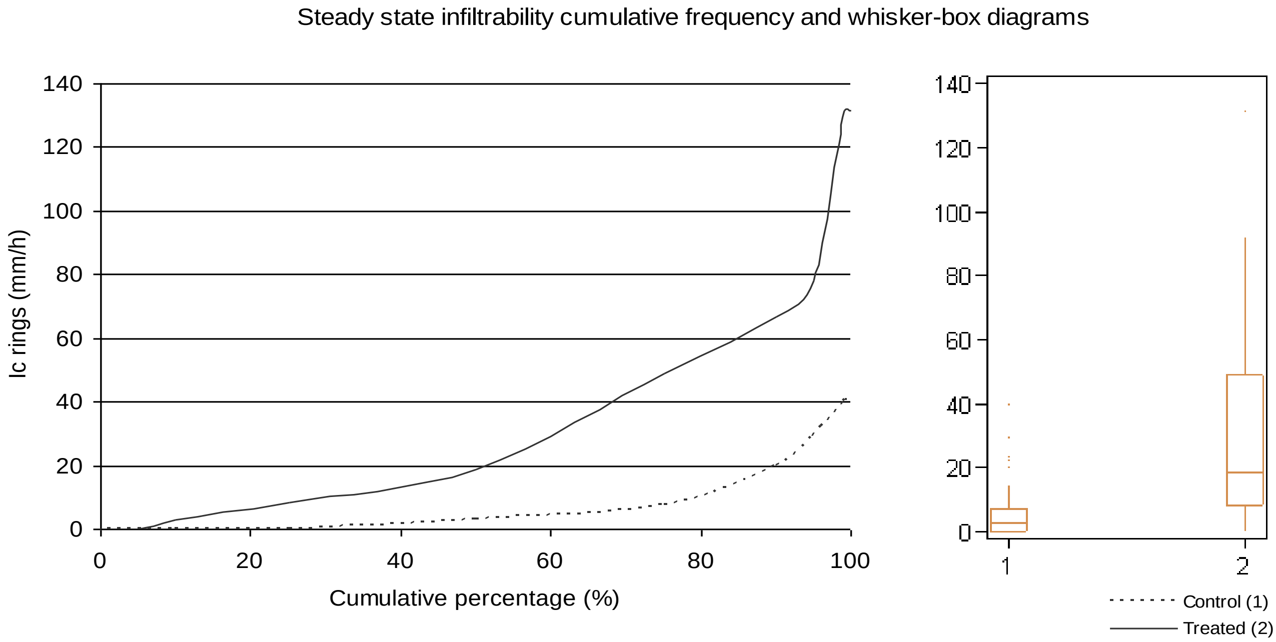


*Figure 13. Cumulative frequency diagram for the whole set of individual steady-state infiltrability values obtained with the double-ring infiltrometers in the control and treated plots (left) and whisker-box diagram for the same set of data (right).*

*3.4 Sediments*

The average sediment generation rates under rainfall simulations of 60 mm/h and 20 mm/h intensities were significantly higher (P=0.011 and P=0.02) in the control plots, with average values of 2.97 Mg $ha^{-1}$ $h^{-1}$ and 0.56 Mg $ha^{-1}$ $h^{-1}$ respectively, than in the treated ones, where these rates were reduced to 0.20 Mg $ha^{-1}$ $h^{-1}$ and 0.01 Mg $ha^{-1}$ $h^{-1}$ (Table 1).

In the particular case of rehabilitation sites and woodlots, similar average sediment generation rates were obtained for 60 mm/h intensity rainfall simulations (0.15 and 0.24 Mg $ha^{-1}$ $h^{-1}$), while in their respective controls the average value was higher in these linked to the rehabilitation sites (4.06 versus 0.98 Mg $ha^{-1}$ $h^{-1}$). However, the differences in this variable were significant between the woodlots and their controls (P=0.008) but they were not in the case of the rehabilitation sites (P=0.082) because of the high variability in their controls (Table 1, Figure 14). This variability can probably be due to the differences in the soil degradation state between the control plots located in the mid-slope gully, where higher amounts of sediments were collected, and the ones in the top-gully position. The soil degradation state could be one of the factors also explaining the higher sediment generation rates in the rehabilitation sites controls compared to the ones from the woodlots.

We estimated the amount of sediments that would be generated in the whole basin after a rain episode of half an hour with a constant intensity of 60 mm/h in two hypothetical scenarios, one where 10% of the land in the basin was rehabilitated with tree planting (as in the case of our treated plots) and 90% of it was not, and a second case where there was not any rehabilitated piece of land. In the first scenario 483,394 Mg of sediments would be generated and in the second 533,115 Mg, giving a difference of 49,722 Mg. If we assume that 60% of these sediments are redistributed in sink areas within the same basin, then the amount of sediments reaching Lake Victoria would be reduced to 19,889 Mg.

These figures are far from reality for many reasons. The results obtained in a portion of the lower part of the basin cannot be extrapolated to the whole catchment because of the evident differences in soil types, slopes, land uses, etc. It is also not realistic to think about a rainfall episode which has a constant intensity and covers the whole basin. Furthermore, the average sediment generation rates were obtained by a simple methodology that makes them valid for comparative purposes. However, such rates are probably lower than those at the slope scale due to two factors, the reduced sampling area and therefore the lack of a slope length factor that would increase the runoff water energy, and the design of the sampling-collecting plot, which could trap part of the generated sediments. Nevertheless, these results underline the magnitude of the sediments generated at a regional scale and highlight the great importance of proper land management practices, and particularly tree planting, in reducing the amount of sediments exported to the lake as well as the loss of fertile soil, with its linked consequences. From this point of view, it is necessary to persuade other communities in the basin to enrol in similar projects aiming to rehabilitate and protect their land and promote and support these initiatives.

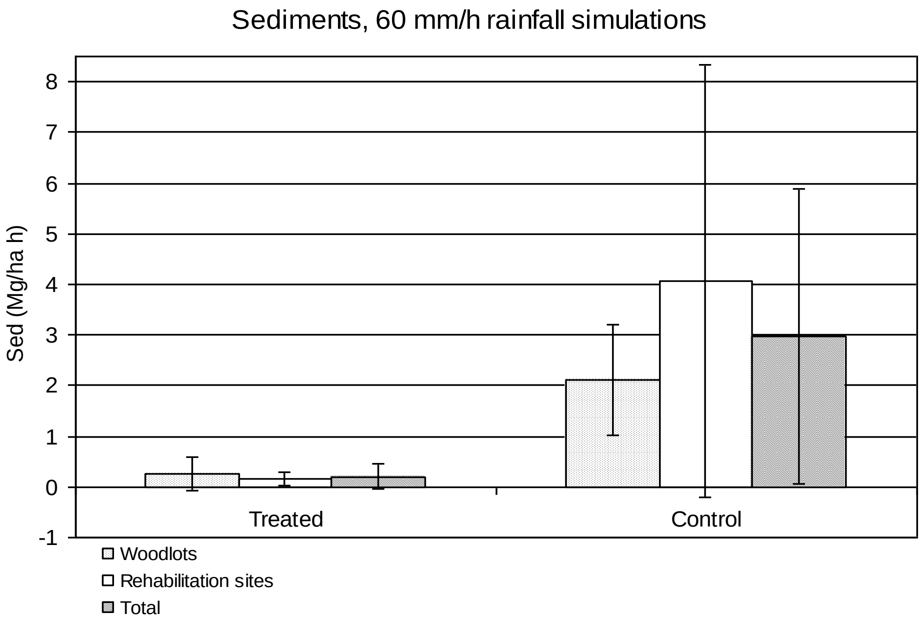


*Figure 14. Average sediment generation rate under 60 mm/h rainfall simulations for both treated and control plots in the study area (East-Nyakach, lower Nyando). The error bars indicate the SD.*

Table 1 : Mean values and standard errors in parentheses, divided into treated (W: woodlots, RS: rehabilitation sites) and control plots, for the average steady-state infiltrability obtained with the double-ring infiltrometers (Ic_rings, mm/h), the steady-state infiltrability obtained from rainfall simulations with an intensity of 60 and 20 mm/h (Ic_60 and Ic_20, mm/h), the average bulk density (BD, g/cm$^3$), the sediment generation rates obtained from the rainfall simulations at 60 and 20 mm/h intensity (Sed_60 and Sed_20, Mg ha$^{-1}$ h$^{-1}$) and the coefficient of variance for the steady-state infiltrability values obtained with the double-ring infiltrometers (CV, %). The p-values obtained from the two-sample t-test for the mean difference contrast between control and treated plots are presented as well.

| | TREATED | | | CONTROL | | | p-value | | |
|---|---|---|---|---|---|---|---|---|---|
| | **Total n=9** | **W n=5** | **RS n=4** | **Total n=9** | **W n=5** | **RS n=4** | **Total n=9** | **W n=5** | **RS n=4** |
| **Ic_rings** | 29.35 (19.46) | 29.91 (23.71) | 28.64 (16.08) | 6.55 (4.94) | 7.83 (4.80) | 4.94 (5.29) | 0.004** | 0.053$^{ns}$ | 0.016* |
| **Ic_60** | 28.45 (25.50) | 28.23 (29.43) | 28.74 (24.07) | 1.97 (5.91) | 3.54 (7.92) | 0 (0) | 0.007** | 0.068$^{ns}$ | 0.049* |
| **Ic_20** | 27.75 (24.41) | 29.60 (27.76) | 25.45 (23.44) | 3.15 (4.09) | 3.93 (4.45) | 2.17 (3.99) | 0.008** | 0.054$^{ns}$ | 0.070$^{ns}$ |
| **BD** | 1.16 (0.16) | 1.20 (0.17) | 1.12 (0.16) | 1.34 (0.09) | 1.37 (0.09) | 1.32 (0.09) | 0.004** | 0.038* | 0.042* |
| **Sed_60** | 0.20 (0.25) | 0.24 (0.33) | 0.15 (0.13) | 2.97 (2.92) | 2.11 (1.10) | 4.06 (4.27) | 0.011* | 0.008** | 0.082$^{ns}$ |
| **Sed_20** | 0.01 (0.01) | 0.01 (0.01) | 0.01 (0.01) | 0.56 (0.67) | 0.22 (0.22) | 0.98 (0.84) | 0.020* | 0.051$^{ns}$ | 0.052$^{ns}$ |
| **CV_rings** | 73.77 (20.32) | 67.83 (25.13) | 80.16 (12.11) | 132.84 (50.90) | 99.42 (29.47) | 174.62 (39.51) | 0.005** | 0.106$^{ns}$ | 0.004** |

ns: Not significantly different ($\alpha \leq 0.05$)
*: Significantly different at 5%
**: Significantly different at 1%

Table 2: Median values and interquartil range (Q3-Q1) in parentheses, divided into treated (W: woodlots, RS: rehabilitation sites) and control plots, for the average steady-state infiltrability obtained with the double-ring infiltrometers (Ic_rings, mm/h), the steady-state infiltrability obtained from rainfall simulations with an intensity of 60 and 20 mm/h (Ic_60 and Ic_20, mm/h), the average bulk density (BD, g/cm$^3$), and the sediment generation rates obtained from the rainfall simulations at 60 and 20 mm/h intensity (Sed_60 and Sed_20, Mg ha$^{-1}$ h$^{-1}$). The p-values obtained from the Wilcoxon two-sample test for the median difference contrast between control and treated plots are presented as well.

| | TREATED | | | CONTROL | | | p-value | | |
|---|---|---|---|---|---|---|---|---|---|
| | **Total n=9** | **W n=5** | **RS n=4** | **Total n=9** | **W n=5** | **RS n=4** | **Total n=9** | **W n=5** | **RS n=4** |
| **Ic_rings** | 24.41 (16.91) | 27.97 (18.26) | 22.98 (19.37) | 5.73 (9.10) | 10.36 (8.60) | 3.58 (7.96) | <0.001** | 0.028* | 0.014* |
| **Ic_60** | 26.52 (53.0) | 14.14 (53.0) | 26.83 (29.66) | 0 (0) | 0 (0) | 0 (0) | <0.001** | 0.064$^{ns}$ | 0.014* |
| **Ic_20** | 11.26 (50.13) | 9.87 (50.33) | 15.63 (29.11) | 0.53 (8.15) | 2.05 (7.95) | 0.27 (4.34) | <0.001** | 0.036* | 0.029* |
| **BD** | 1.14 (0.19) | 1.20 (0.08) | 1.06 (0.21) | 1.39 (0.12) | 1.39 (0.02) | 1.30 (0.12) | 0.008** | 0.088$^{ns}$ | 0.1$^{ns}$ |
| **Sed_60** | 0.16 (0.26) | 0.16 (0.26) | 0.16 (0.18) | 1.50 (2.1) | 1.50 (1.99) | 3.09 (6.52) | <0.001** | 0.004** | 0.014* |
| **Sed_20** | 0.02 (0.02) | 0.02 (0.02) | 0.01 (0.02) | 0.36 (0.64) | 0.12 (0.31) | 0.91 (1.21) | <0.001** | 0.004** | 0.014* |

ns: Not significantly different ($\alpha \leq 0.05$)
*: Significantly different at 5%
**: Significantly different at 1%

*3.4 Relationship between steady-state infiltrability and vegetation cover*

A significant (P=0.0002) positive correlation was found between total vegetation cover and the average steady-state infiltrability measured using the double-ring infiltrometer (Table 3, Figure 15), suggesting the importance of those land management practices, such as agroforestry, that enhance vegetation cover and reduce water runoff and topsoil loss by erosion. Gifford & Hawkins (1978), Rose (1994) and Adekalu (2007) also observed that increases in ground cover often resulted in increases in soil infiltration rate. Moreover, Ilstedt et *al.* (2007) noticed that greater improvements in soil physical properties, such as infiltrability, were observed under afforestation as compared to agroforestry, indicating that the various potential benefits of planted trees and shrubs (e.g. litterfall, deep roots, shade, etc.) (Young, 1989) were likely to be more effective with increased cover. On the other hand, a land use change that involves a reduction in vegetation cover would result in a degradation of the soil physical properties, leaving them more susceptible to erosion, as concluded by Celik (2005).

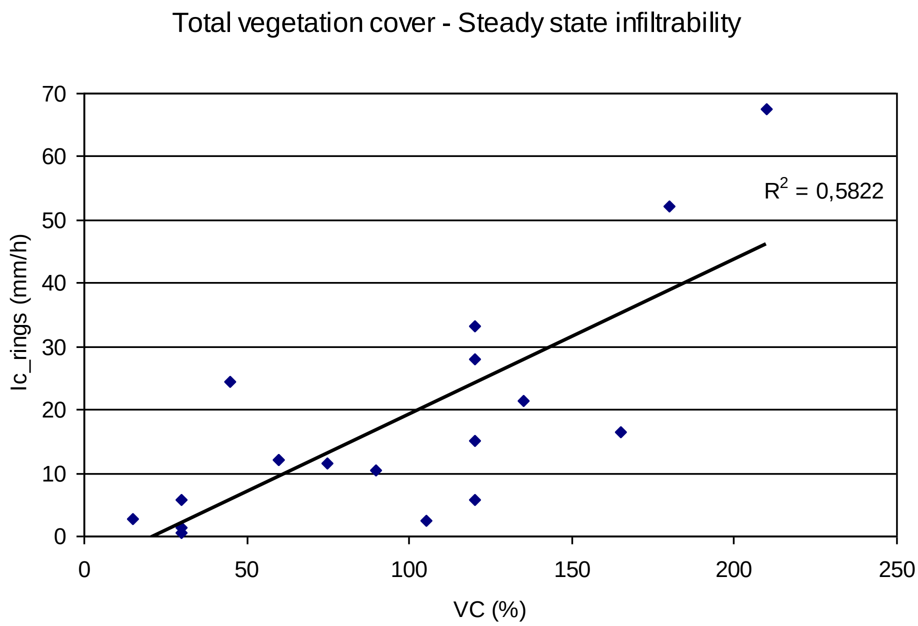


*Figure 15. Regression trend between total vegetation cover (%) and average steady-state infiltrability measured with the double-ring infiltrometer (mm/h).*

*3.5 Relationship between steady-state infiltrability and bulk density*

Changes in steady-state infiltrability could be significantly (P=0.0081, Table 3) explained by the bulk density variable. As shown in Figure 16 and Table 3, an increase in bulk density resulted in a reduction of the steady-state infiltrability. This negative correlation is consistent with the literature (Yimer et *al.*, 2008; Mbagwu, 1997; Willat and Pullar, 1984) and agrees with theory, which states that changes in bulk density affect the size, amount and distribution of pores and thereby the rate of water infiltration (Abdul-Megid et *al.*, 1987; Grimaldi et *al.*, 2003; Teixeira et *al.*, 2003). Higher bulk density implies relatively more compaction, a diminution in the total quantity of pores, and an increase in micropores at the expense of macropores, therefore slowing down infiltration (Lipiec et *al.*, 2006; Mbagwu, 1997; Léonard et *al.*, 2003; Mapa, 1995).

All those management practices aiming to reduce the soil dry bulk density, or at least stop its increase, will indirectly contribute to improve the resistance of soil to erosion trough an enhancement of water infiltration. From this point of view, the actual rehabilitation practices taking place in the study area are playing an important role, as mentioned previously, but an attempt should be done in order to diminish even more soil compaction. Grazing restriction could be a good way to reach this purpose due to the reduction in soil compaction (Greenwood and McKenzie, 2001; Willatt and Pullar, 1984; Savadogo, 2007). Even though grazing restriction was one of the main aims of the initial ICRAF project (ICRAF, 2003), this has never been attained in reality as has been observed during the field work. Cut and carry practices would probably constitute the best option to both avoid soil compaction, maintain a short grass cover and guarantee the supply of grass to cattle.

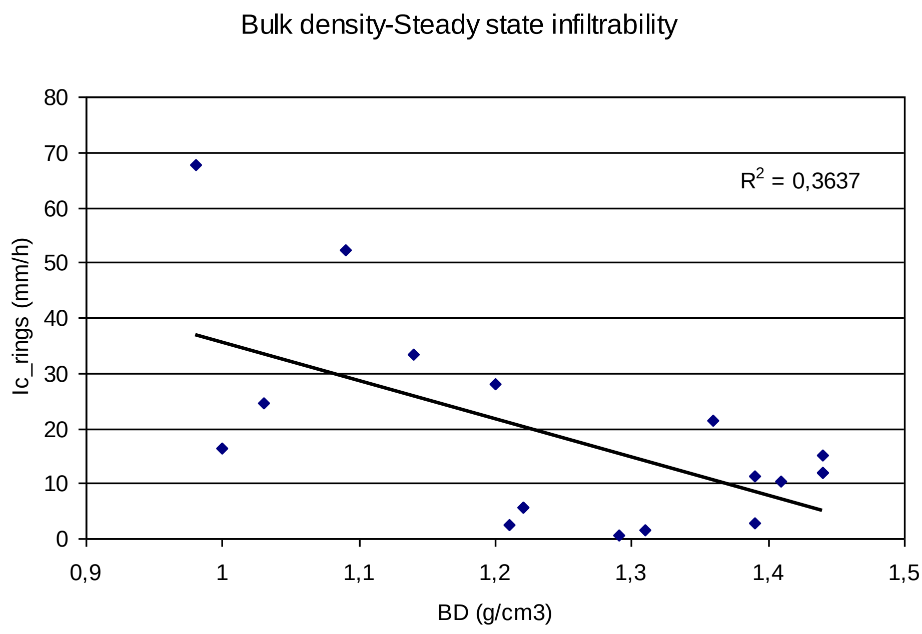


*Figure 16. Regression trend between bulk density (g/cm$^3$) and average steady-state infiltrability measured with the double-ring infiltrometer (mm/h).*

*3.6 Relationship between sediment generation rate and vegetation cover*

Significant (P=0.0136 and P=0.0143) negative correlations were observed between the total vegetation cover and the sediment generation rates, for both 20 mm/h and 60 mm/h rainfall intensities (Table 3, Figure 17), confirming that vegetation cover plays an important role in controlling the amount of generated sediments under a rainfall episode of a given intensity (Rose, 1994; Stocking, 1994). Moreover, according to Stocking (1994), vegetation cover is the single most important factor in soil erosion control in the tropics.

The effective control of erosion by water consists on minimizing the impact of raindrops and the velocity of water on the soil surface (Hillel, 2004). In this context, surface contact cover provides an important protection against erosion due to overland flow, which is typically the major cause of erosion, since it prevents the movement of water over the surface and generates ponding areas (Rose, 1994). Contact cover protects as well the soil from erosion by intercepting raindrops and absorbing their kinetic energy (Stocking, 1994). Above-ground or aerial cover, on the other hand, can provide protection only against the impact of raindrops (Rose, 1994). Some authors like Young (1989), however, highlight that raindrop energy is not substantially reduced by a high tree canopy and that it can actually increase erosion. In our case, the observed reduction in erosion by an increased vegetation cover is significantly (P=0.005 and P=0.013) explained by grass cover, i.e surface contact cover, for rainfall simulations of both 20 mm/h and 60 mm/h intensities. The direct effect of trees and shrubs in reducing the raindrops impact could not be assessed with our methodology but, since an increment in infiltrability was observed under increased vegetation cover, as commented previously, it is reasonable to affirm that these vegetation components are likely to reduce erosion indirectly by diminishing overland flow. Other indirect effects of trees and shrubs in reducing soil erosion could be related to the increase in organic matter inputs into the soil that would improve the stabilization of its structure, thereby preventing the detachment of its particles (Grimaldi, 2003; Rose, 1994). The greater variability in the sediment generation rate in the soils with lower vegetation cover (Figure 17), may be due to the difference in the stability of soil aggregates, since the control plots that where located in the middle-slope gully had suffered a lot of erosion and horizons with low organic matter content were now exposed at the surface, while some of the other control plots, even those with low vegetation covers, had not probably lost so much soil carbon.

As shown in Figure 17, the polinomical trend line fits better than the linear relation in the scatter plot of measured vegetation cover and sediment generation rates. In fact, according to Stocking (1994), experimental evidence indicates that the erosion-cover relationship is curvilinear. This implies, in our case, that a vegetation cover around 125 % would have similar effects in preventing soil erosion than one close to 225%. This finding is crucial when planning management objectives in conservation and rehabilitation projects, since more feasible and realistic vegetation covers can be reached causing more or less the same effect than higher ones. Furthermore, enhancement of vegetation cover will result in less erosion compared to the original soil in those cases where initial cover is lower than in those where it is already high. Therefore, improving the vegetation cover of the bare areas around gullies would represent a significant step in order to prevent their further expansion (Figure 18).

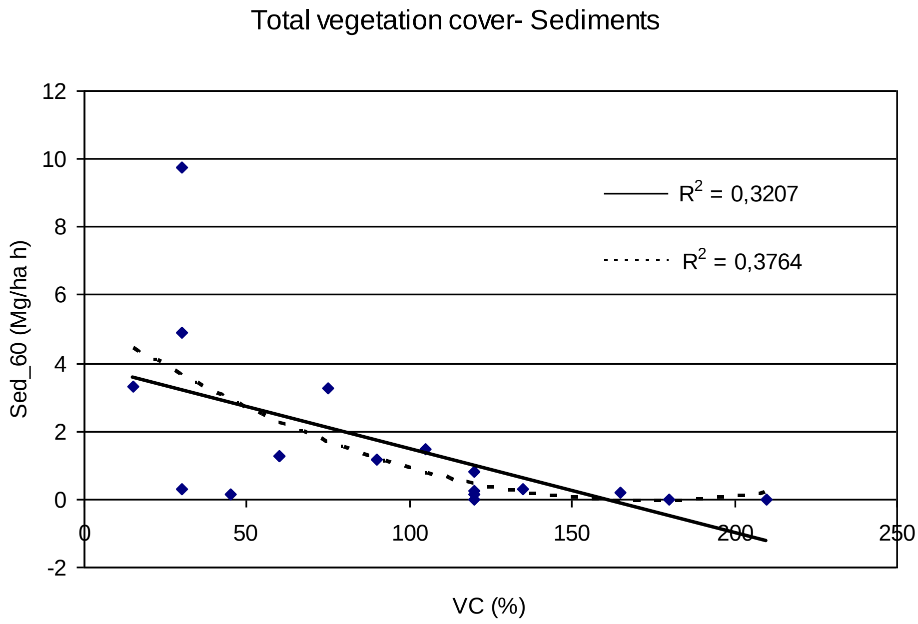


*Figure 17. Regression trend between total vegetation cover (%) and sediment generation rate for 60 mm/h intensity rainfall simulations (Mg $ha^{-1}$ $h^{-1}$).*

*Figure 18. Further expansion of the gully along the bare land around it.*

*3.7 Relationship between sediment generation rate and steady-state infiltrability*

No significant (P=0.077 and P=0.084) impact of infiltrability on the amount of generated sediments under 20 mm/h and 60 mm/h rainfall intensities was obtained (Table 3, Figure 19). Nevertheless, the trend indicated a negative correlation between the two variables, meaning that less erosion takes place in those soils with higher infiltrabilities. This behaviour, which has been observed by other authors as well (Rose, 1994), seems logical since the potential amount of generated sediments under water erosion depends, among others, on the amount of available runoff water, which is inversely proportional to infiltrability. However, only potential erosion is controlled by infiltrability, and changes in the actual amount of generated sediments can take place under different vegetation covers, as shown previously. The scatter plot in Figure 19 seems to follow an asymptotical trend, where very low to zero sediment generation rates occur approximately from 20 mm/h steady-state infiltrability, despite the simulated rain had an intensity of 60 mm/h. This is probably the result of the relatively high vegetation cover that exists on those soils with higher infiltrability. On the other hand, larger variability in the sediment generation rates occurs at lower steady-state infiltrabilities (Figure 19), perhaps due to the existence of different vegetation covers in soils with similar infiltrability.

Steady state infiltrability - Sediments

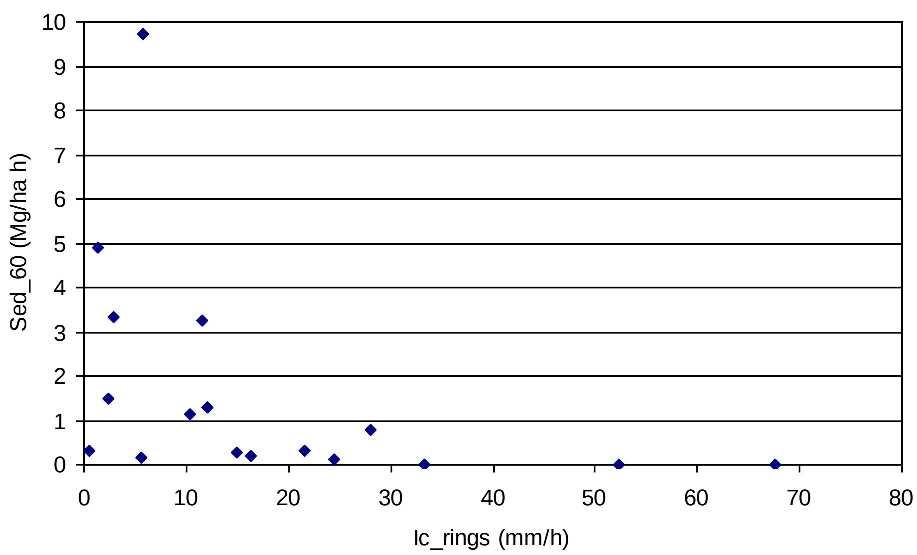


*Figure 19. Regression trend between sediment generation rate for 60 mm/h intensity rainfall simulations (Mg $ha^{-1}$ $h^{-1}$) and average steady-state infiltrability measured with the double-ring infiltrometer (mm/h).*

*3.8 Relationship between steady-state infiltrability from double-ring infiltrometers and rainfall simulations*

A significant ($P<0.0001$) one-to-one relationship with an intercept not significantly different from zero ($P=0.371$ and $P=0.269$) was found between the steady-state infiltrability measured with the double-ring infiltrometers and that obtained from rainfall simulations at 20 mm/h and 60 mm/h (Table 3, Figure 20), contrarily to what most of the existent literature advocates. Bhardwaj and Singh (1992), Touma and Albergel (1992), and Sidiras and Roth (1987) all agree when stating that the double-ring infiltorometer tends to give higher values of infiltration than those obtained with the rainfall simulator (between 2 to 3 times higher according to Bhardwaj and Singh (1992) and 2 to 5 according to Sidiras and Roth (1987)). Touma and Albergel (1992) explain these differences as a consequence of the occurrence of surface sealing during the rainfall simulation experiments. Raindrop energy is a prime factor in causing aggregate breakdown and the formation of crusts or seals over the bare soil, causing a marked reduction in infiltration (Bonell and Williams, 1986). In our case, as it can be observed in Figure 18, there are some plots where a steady-state infiltrability of 0 mm/h was attained with the 60 mm/h rainfall simulations while the values for the same variable measured using the double-ring infiltrometer where slightly higher, indicating the possible occurrence of sealing. In fact, these plots correspond mainly to controls where, as mentioned previously, the vegetation cover is low and thus the soil is more susceptible to raindrop impact. However, the set of data as a whole indicates that the steady-state infiltrabilities measured using both methodologies are very similar, as it was observed by Julander and Jackson (1983) in fine grained, poorly structured, low permeability soils. Therefore, when working with low infiltration soils, according to Julander and Jackson (1983), or in our specific case, if the only variable of concern is the steady-state infiltrability, it is not worth working with a rainfall simulator since the double-ring infiltrometer gives similar results and is much more practical. However, if it is necessary to have data on rainfall infiltration at a given intensity or data concerning erosion, then rainfall simulations will be needed.

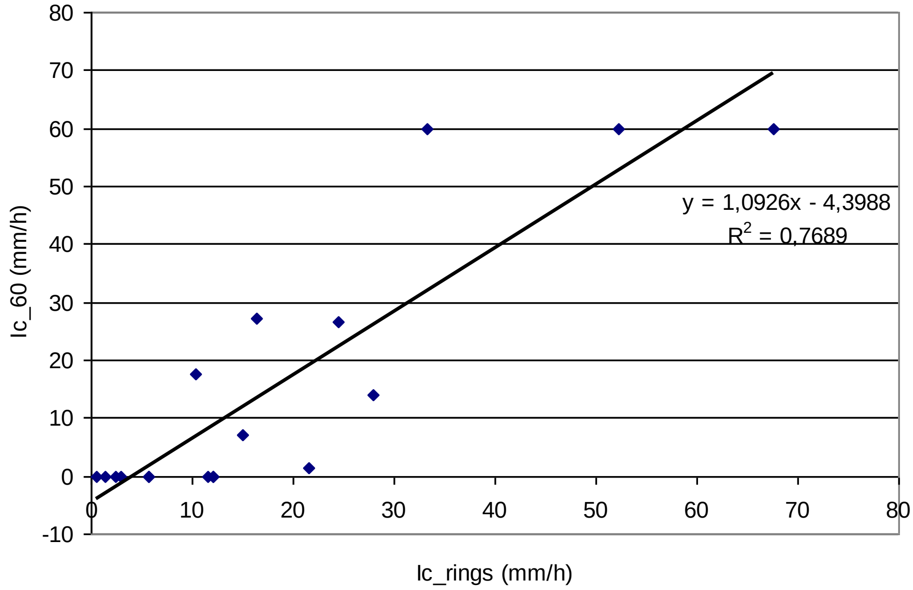


*Figure 20. Regression trend between the average steady-state infiltrability measured with the double-ring infiltrometer (mm/h) and the one obtained from the rainfall simulations at 60 mm/h intensity.*

### *3.9 Relationship between sediment generation rates at 20 mm/h and 60 mm/h rainfall intensities*

A significant (P=0.0006) positive relationship with a slope of 3.3 and an intercept not significantly (P=0.64) different from zero was found between the sediment generation rate under 20 mm/h intensity rainfall and the one under an intensity of 60 mm/h (Table 3, Figure 21), indicating that the amount of generated sediments is directly proportional to rainfall intensity, at least in the range between 20 mm/h and 60 mm/h. The positive relationship between these two variables seems logical and is reflected in the universal soil-loss equation (USLE), where the predicted annual soil loss depends, among other factors, on the rainfall erosivity, which is linked to its intensity.

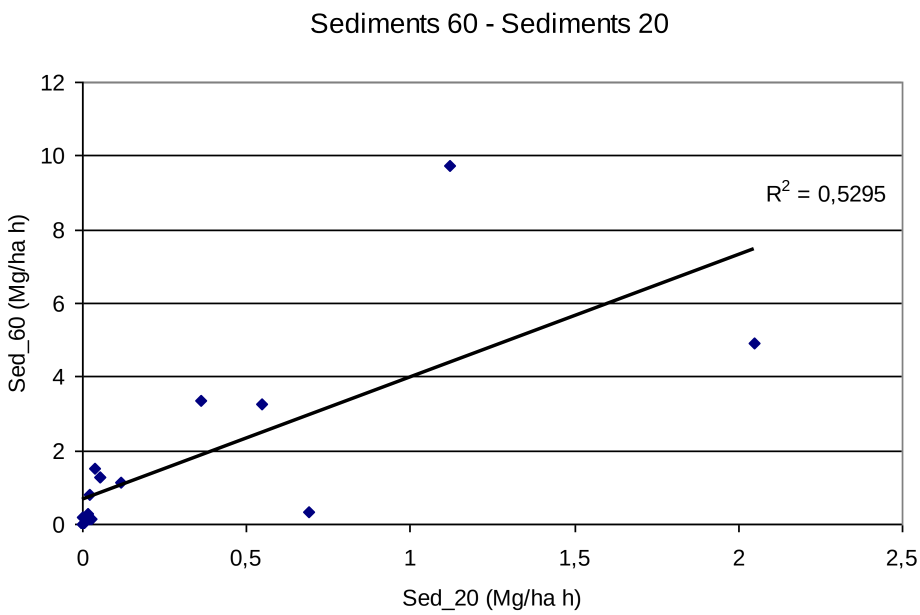


*Figure 21. Regression trend between sediment generation rate for both 20 mm/h and 60 mm/h intensity rainfall simulations (Mg ha$^{-1}$ h$^{-1}$)*

Table 3 : Simple regression equations between some of the measured variables: average steady state infiltrability obtained with the double ring infiltrometers (Ic_rings, mm/h), the steady state infiltrability obtained from rainfall simulations with an intensity of 60 and 20 mm/h (Ic_60 and Ic_20, mm/h), sediment generation rates obtained from the rainfall simulations at 60 and 20 mm/h intensity (Sed_60 and Sed_20, Mg ha$^{-1}$ h$^{-1}$), total vegetation cover (%), grass cover (%), bulk density (BD, g/cm$^3$). $\hat{\beta}_0$ and $\hat{\beta}_1$ are the regression equations parameters, intercept and slope respectively, estimators; their p-values, obtained from the individual regression coefficient contrasts using t-tests, are presented as well. y and x indicate the dependant and the explicative variables. $R^2$ is the model predictive capacity and n the sample size.

| **y =** | $\hat{\beta}_0$ | **(p-value)** | $\hat{\beta}_1$ | **(p-value)** | **X** | $R^2$ | **n** |
|---|---|---|---|---|---|---|---|
| Ic_rings | -5.32 $^{ns}$ | 0.3636 | 0.25 ** | 0.0002 | Veg_cover | 0.58 | 18 |
| Ic_rings | 105.20 ** | 0.0023 | -69.62 ** | 0.0081 | BD | 0.36 | 18 |
| Sed_20 | 0.80 ** | 0.0018 | -0.01 * | 0.0136 | Veg_cover | 0.32 | 18 |
| Sed_60 | 3.94 ** | 0.0011 | -0.03 * | 0.0143 | Veg_cover | 0.32 | 18 |
| Sed_20 | 0.89 ** | 0.0006 | -0.01 ** | 0.0047 | Grass_cover | 0.40 | 18 |
| Sed_60 | 4.11 ** | 0.0010 | -0.04 * | 0.0129 | Grass_cover | 0.33 | 18 |
| Sed_20 | 0.51 ** | 0.0078 | -0.01 $^{ns}$ | 0.0772 | Ic_rings | 0.18 | 18 |
| Sed_60 | 2.61 ** | 0.0039 | -0.06 $^{ns}$ | 0.0837 | Ic_rings | 0.18 | 18 |
| Ic_20 | -3.15 $^{ns}$ | 0.3714 | 1.04 ** | <0.0001 | Ic_rings | 0.78 | 18 |
| Ic_60 | -4.40 $^{ns}$ | 0.2594 | 1.09 ** | <0.0001 | Ic_rings | 0.77 | 18 |
| Sed_60 | 0.64 $^{ns}$ | 0.1898 | 3.33 ** | 0.0006 | Sed_20 | 0.53 | 18 |

ns: Not significantly different from 0 ($\alpha \leq 0.05$)
*: Significantly different from 0 at 5%
**: Significantly different from 0 at 1%

## 4. Conclusions

The results showed significant differences in average vegetation cover, top soil bulk density, steady-state infiltrability and sediment generation rates for 20 and 60 mm/h rainfall simulations between treated plots, where trees were present, and controls with no trees.  The total vegetation cover was higher in the treated plots as well as the steady-state infiltrability, while values of sediment generation rates and topsoil bulk density were lower.

Bulk density and vegetation cover significantly explained the differences in steady-state infiltrability. Higher vegetation cover resulted in higher steady-state infiltrability, while reduced bulk density had the same effect.  A significant inverse relationship was found between the sediment generation rate and the vegetation cover. Higher steady-state infiltrabilities also explained the lower sediment generation rates found in the treated plots.

These results show the beneficial effects of tree planting in improving soil physical properties and reducing erosion. Therefore, those land uses that involve the presence of trees, as for instance agroforestry, are interesting to adopt in areas with erosion and soil degradation problems. Furthermore, trees can offer a wide range of products (e.g timber, fuelwood, medicines, fodder, etc.), which, added to the expected improvement in productivity, would positively influence the welfare of the people in these areas.

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