Understanding

sediment dynamics and hydrology

to manage water resources

in a tropical montane forest of Kenya

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This thesis is submitted in fulfillment of the regulations for the degree of

Doctor of Philosophy

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Declaration

I declare that the thesis presented is my own work, except where references are made to other work and that it has not been submitted, in whole or in part, in any previous application or award for a higher degree elsewhere. Contributions by other researchers are properly acknowledged.

Jaqueline Stenfert Kroese

Immenstaad, Germany, September, 2020

Statement of Authorship

This thesis was prepared as a thesis by papers. The chapters are presented in the format of the accepted papers or intended for submission to peer-reviewed journals. Each paper's reference list can be found in a combined reference list at the end of the thesis.

Chapter 1 provides a general introduction and outlines the objectives of the thesis and is not intended for publication.

Chapter 2 comprises the literature review and is not intended for publication.

Chapter 3 comprises the description of the study area and methods and is not intended for publication.

Chapter 4 is published in *Water Resources Research*
Stenfert Kroese, J., S. R. Jacobs, W. Tych, L. Breuer, J. N., Quinton and M. C. Rufino (2020). Tropical forest conversion is a critical driver for sediment supply in East African catchments. *Water Resources Research,* doi: 10.1029/2020WR027495.

JSK, MCR and JNQ designed the research. JSK conducted sampling, laboratory work and analysed results. JSK and WT performed modelling. JSK prepared the manuscript with input from MCR, JNQ, LB, SRJ and WT.

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Stenfert Kroese, J., P. V. G. Batista, S. R. Jacobs, L. Breuer, J. N. Quinton and M. C. Rufino (2020). Agricultural land is the main source of stream sediments after conversion of an African montane forest. *Scientific Reports*, *10, 1-15,* doi: 10.1038/s41598-020-71924-9.

JSK, MCR and JNQ designed the research. JSK conducted sampling, laboratory work and analysed results. JSK and PVGB performed modelling. JSK prepared the manuscript with input from MCR, JNQ, LB, SRJ and PVGB.

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JSK, MCR and JNQ designed the research. JSK conducted sampling, laboratory work and analysed results. JSK prepared the manuscript with input from MCR, JNQ, LB and SRJ.

Chapter 7 comprises a general discussion and conclusions and is not intended for publication.

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Abstract

Montane forests are unique ecosystems in the tropics and they regulate soil and water functions at the landscape scale. Their conservation is important because forests contribute with their abundant above and belowground biomass to increased soil stability and reduced soil erosion. However, tropical forests are the hotspots of land use change mainly due to the fertile soils and the mild climatic conditions where they grow. In the East African highlands, the demand for agricultural land by an increasing human population and the cultivation on steep hillslopes put pressure on forest and water resources. Streams in these montane catchments are often enriched in suspended sediments, which affect water quality and represent a loss of soil capital. However, these tropical montane ecosystems are understudied, especially in sub-Saharan Africa. Having a scarcity of studies on sediment dynamics and hydrology makes it more difficult planning for future sustainable land use. To contribute towards closing this knowledge gap, this thesis uses three study catchments (27-36 km^2) under different land use (e.g. natural forest, tea-tree plantations and smallholder agriculture) as a ‘microcosm’ within which to understand process response to disturbance within the headwaters of the Sondu River Basin (3,470 km^2) in the Mau Forest Complex of Kenya. A four-year high-resolution sedimentological time series recorded the highest sediment concentrations in the smallholder agriculture catchment, followed by the tea-tree plantation and the natural forest catchment, caused by increased surface runoff. Rainfall-runoff modelling showed that soils of the natural forest catchment had high permeability reflected in the deep water flow pathways, in contrast to the compacted soils in the smallholder agriculture catchment with a dominance of shallow sub-surface flow to surface runoff. Sediment response to rainfall (up to 3.5 hr) was delayed in the smallholder agriculture catchment compared to the nearly instantaneous response (<1.5 hr) in the forested catchment due to sediment supply from near-by sources. Sediment fingerprinting conducted at the smallholder agriculture catchment unravelled the relative contribution of four different sediment sources. Agricultural land accounted for the largest contribution (75% with 95%-confidence interval 63-86%) of the total sediment, while channel banks, gullies and unpaved tracks were shown to be local sediment hotspot sources. Suspended sediment collected with time-integrated, manual- and automatic-event based sediment samplers at the outlet of the three catchments over a period of up to four months, demonstrated that particulate carbon and nutrient concentrations were up to three times higher in the natural forest compared to the smallholder agriculture catchment. The low particulate macronutrient concentrations point to the fast impoverishment of agricultural soils after deforestation. The findings of the study clearly show that land use change has an important impact on sediment dynamics and hydrological pathways, which can affect the water balance of the whole ecosystem and deteriorate downstream water supplies and the water quality of Lake Victoria. The findings of this study further contribute to the wider knowledge of other tropical montane systems facing similar pressures like agricultural expansion and deforestation.

List of Contents

List of Figures

List of Tables

1 General introduction

Globally, 35.9 billion tonnes of soil is estimated to erode annually (Borrelli *et al.* 2017). A significant 60% of this eroded soil ends up in aquatic systems and is permenantly lost from terrestrial ecosystems (Zimdahl 2012). Soil erosion processes are strongly linked to the loss of soil health and the degradation of water resources. The topsoil, rich in nutrients and organic matter, is the main resource for agricultural production, and therefore soil loss is a serious constraint to sustain future agricultural productivity. In addition, soil erosion occurs at an unsustainable alarming rate, which is recognized to be much faster than it can be replenished through natural soil formation processes (Evans *et al.* 2020). Current projection indicates that the world population is likely to reach 8.5 billion people by 2030 (UN 2019), adding on the challenge of feeding this population with deteriorating soil resources. To address these global concerns, the UN calls for the '*protection, restoration and promotion of sustainable use of terrestrial ecosystems and sustainably managed forests, combating desertification, halting and reversing land degradation and halting biodiversity loss*' as one of the sustainable development goals (SDG 15) of the 2030 agenda (UN 2015).

Overall, the rate of soil erosion varies globally on average by around 1.4 t ha^{-1} yr^{-1} between neighbouring countries (Wuepper *et al.* 2020). Latin America, Africa and Asia are in the group of the highest predicted soil erosion rates (3.53, 3.51 and 3.47 t ha^{-1} yr^{-1}, respectively). Other parts of the world, such as North America, Europe and Oceania have lower predicted rates (2.23, 0.92 and 0.90 t ha^{-1} yr^{-1}, respectively). The soil erosion rates estimated based on land use change projections for the African continent exceeds those of Latin America by about 10%, thus turning Africa into the hotspot for soil erosion (Borrelli *et al.* 2017). In sub-Saharan Africa, in particular in the heterogeneous and fragmented landscapes of the highlands of East Africa, soil erosion is more severe compared to other regions (Place *et al.* 2006; Wuepper *et al.* 2020).

Tropical montane regions are particularly prone to surface soil erosion due to their topographic position on steep terrain (Wohl 2006; Grangeon *et al.* 2012) and the occurrence of rainfall of high erosivity (Guzman *et al.* 2013; Nishigaki *et al.* 2017). In addition to these natural underlying conditions, soil erosion in montane ecosystems is accelerated by land use changes. The removal of a protective vegetation cover and the loss of the dense rooting system of woody ecosystems are among the key factors that enhance erosion processes, and consequently suspended sediment contribution. Such land use changes may lead to further erodibility of the susceptible soils on fragile ecosystems under steep slopes. Tropical headwaters under shifting cultivation may become major contributors of suspended sediments because of high surface connectivity between hillslopes and the stream network due to steep slopes and the loss of natural barriers (Fan *et al.* 2012; Fryirs 2013). Few studies in tropical montane regions have found evidence for enhanced turbidity and suspended sediment concentrations due to increased soil erosion following the conversion of natural forests to agriculture, as was shown in North and South Brazil (Figueiredo *et al.* 2010; Didoné *et al.* 2014; Minella *et al.* 2018), the Ecuadorian Andes (Harden 2001) and East Africa (Nyssen *et al.* 2009; Tamooh *et al.* 2012; Guzman *et al.* 2013).

Land use change increases sediment loads, and therefore to manage the impacts there is a need to understand where the sediment comes from, and which could be the most effective measures. However, unravelling sediment dynamics is complex due to their highly spatial and temporal variability (Vercruysse *et al.* 2017). Studies have shown that 80-95% of the annual sediment load is generated during high flows of short duration occurring storm peaks (De Girolamo *et al.* 2015; Sun *et al.* 2016). The smallest proportion of sediment is carried during baseflow or dry seasons. Suspended sediment concentrations also change throughout individual storm events (Sun *et al.* 2016). How soil erosion and hydrological dynamics in a catchment respond to land use or land use changes, depends on a number of catchment-related parameters, climatic conditions and land management practices. Tropical montane ecosystems are relevant to study sediment and hydrology dynamics under land use change because of erosive rainfall they are exposed to and their irregular topography.

The Mau Forest Complex in the western highlands of Kenya is such an ecosystem under threat of land use conversion. The Mau is the largest tropical montane forest of East Africa and it represents an important catchment area for Kenya. However, over the last four decades 25% of the forest cover has been lost to commercial and smallholder agriculture (Brandt *et al.* 2018). The Mau Forest is the headwaters to twelve rivers, one of which is the Sondu River. The montane headwater catchments of the Sondu River Basin are characterised by high surface connectivity due to steep hillslopes. Land use changes and anthropogenic disturbance in these headwaters are expected to lead to deterioration of the water quality and changes in hydrology. For the development of targeted and sustainable land management practices to lower suspended sediment fluxes and their negative impact on aquatic habitat, people, livestock and the economy, there is need to understand the dynamics of suspended sediments and source areas in these catchments.

To understand the high temporal and spatial variability of sediment dynamics and hydrological response requires long-term monitoring of hydrology and sediment dynamics. However, based on a review of 84 publications of sediment studies in Africa, it was found that only 20 catchments included gauging station measurements in Kenya. Most of these data points were classified as of ‘poor’ data quality due to a low sampling frequency or based on rating curves with less than 50 observations (Vanmaercke *et al.* 2014). In addition, modelled soil erosion rates of highlands are underestimated compared to field measurements, likely because of the underrepresentation of field measurements for cross-validation in highlands (Borrelli *et al.* 2017). With the absence of a knowledge base on sediment dynamics and hydrology in tropical headwater catchments, such as in the Sondu River Basin, the design and application of sustainable land management practices will be a challenging task. This study aimed to address this knowledge gap in sediment source areas and hydrological flow pathways under contrasting land use in montane ecosystems and to improve the understanding of these under-researched environments, which can be used in the future to develop targeted soil and water conservation strategies.

1.1 Aim and objectives of the thesis

To address the knowledge gap of sediment dynamics and hydrology in tropical headwater catchments, the aim of this research is

> *to assess the effect of land use on hydrological and environmental quality aspects in tropical montane catchments used as a 'microcosm' within the headwaters of the Sondu River Basin in the South-West Mau, Kenya.*

This study contributes to the current knowledge on water and nutrient cycles for tropical montane forests and adds to previous studies in the headwaters of the Mau Forest Complex that investigated the key drivers of forest disturbances (Bewernick 2016; Brandt *et al.* 2018), soil hydraulic conditions and soil quality (Owuor *et al.* 2018), soil organic matter (Chiti *et al.* 2018), greenhouse gas emissions (Arias-Navarro *et al.* 2017a, b; Wanyama *et al.* 2018), hydrological pathways and water sources (Jacobs *et al.* 2018a), dynamics of dissolved nitrogen and nitrate (Jacobs *et al.* 2017, 2018b) and extends the knowledge base on land use, hydrology and sediment dynamics, source areas and particulate macronutrient concentrations.

This research had the following specific objectives:

1. To quantify rainfall, runoff and suspended sediment transport dynamics in order to assess the influence of land use on suspended sediment yield and their seasonal responses (Chapter 4),
2. To assess the timing of the response of suspended sediment to rainfall and discharge and to improve the understanding of the dominant water flow pathways under different land use (Chapter 4),
3. To identify the best sediment tracer composite of a large pool of biogeochemical and geochemical elements for sediment provenance determination (Chapter 5),
4. To estimate the relative sediment contribution from four source areas: agricultural land, gullies, unpaved tracks and channel banks (Chapter 5),
5. To quantify sediment-associated macronutrient (total carbon, total nitrogen and total phosphorus) fluxes from catchments under natural forest, tea-tree plantations and smallholder agriculture (Chapter 6); and
6. To draw conclusions on the effect of land use on sediment dynamics in the headwaters of the Sondu River Basin and reflect suggestions for sustainable land management practices (Chapter 7).

1.2 Outline of the thesis

This thesis is based on three peer-reviewed articles; either intended for submission or already published (Chapter 4-6):

Chapter 4 addresses objectives 1 and 2 and provides an assessment of the impact of land use on sediment and hydrological dynamics based on a four-year time series of rainfall, runoff and suspended sediment concentrations. Hydrological flow pathways are key in generating suspended sediment. The difference in hydrological pathways under different land uses is described with a rainfall-runoff model under the *Data-Based Mechanistic* (DBM) modelling philosophy. Linear continuous time-transfer function models explain the response of water flow pathways to rainfall within the natural forest, tea-tree plantation and smallholder agriculture catchment. In addition, cross-correlation functions identify the timing of sediment response to runoff and rainfall to assess sediment storage and supply at catchment-scale.

In Chapter 5, which addresses objectives 3 and 4, a sediment fingerprinting modelling is used to investigate relative contributions of four sediment source areas within the smallholder agriculture catchment. The focus here is on the smallholder agriculture catchment because it is the catchment under the highest disturbances generating six times more suspended sediment yield compared to the natural forest catchment. Agricultural land, unpaved tracks, gullies and channel banks are selected as the four main sediment source areas. To unravel the relative contribution of each sediment source, a Mix-SIAR un-mixing modelling approach is used under a Bayesian framework.

Chapter 6 addresses objective 5 and explores particulate macronutrient fluxes and continues with the comparison of the three catchments dominated by natural forest, tea-tree plantations and smallholder agriculture. Sediment-associated total carbon, nitrogen and phosphorus fluxes are quantified at the catchment outlets during two sampling periods in 2018 and 2019. The impact of land use on the different macronutrient concentrations, ratios and relationships in suspended sediments is further assessed.

Chapter 7 concludes the thesis with a perspective of the key findings and limitations encountered during the course of this research and on implications for future research. A reflection is drawn on to the challenges of the application of sustainable agricultural management practices in the rural highlands of Western Kenya coinciding with high poverty rates.

2 Literature review

2.1 Tropical montane forest catchments under threat

Forests and, in particular tropical montane forests, are characterised by a rich above and belowground biomass and biodiversity, and provide a wide range of ecosystem services (Martínez *et al.* 2009). They regulate climate and the carbon (C) cycle, absorb C through photosynthesis and store the C in the abundant above and belowground biomass (Malmer *et al.* 2010). Tight nutrient cycles associated with litter input and an active soil biological community such as fungi or bacteria generates soils rich in soil organic matter and nutrients (Neill *et al.* 1997; Robinson *et al.* 2020). Forests play a vital role in the hydrological cycle by generating rainfall and contributing to controlling floods and streamflow peaks through evapotranspiration, interception and soil infiltrability (Bruijnzeel 2004; Ogden *et al.* 2013). Springs originate in montane forest catchments and build important headwaters for river systems of high water quality (Buytaert *et al.* 2006; UNEP 2012). Native forest soils with increased permeability generate deep water pathways to recharge groundwater storage, especially during wet seasons (Malmer *et al.* 2010; Ouyang *et al.* 2019). This increased permeability together with a dense vegetative cover with a diverse strata and complex rooting system controls soil erosion and reduces discharge of suspended sediments to streams (Chaves *et al.* 2008; Owuor *et al.* 2018). Among these ecosystem services, tropical montane forests also provide adjacent communities and the global population with natural resources such as timber, medicinal plants (Masese *et al.* 2012) or grazing areas for livestock (Russell *et al.* 2001).

Forests are under severe threat due to a number of complex interconnected factors, which can be broadly categorized into: (1) agricultural expansion, (2) infrastructure and (3) urban expansion and (4) mining. Population growth is the core driver of deforestation interlinked with each of the four drivers (Defries *et al.* 2010). Deforestation has a long history in temperate regions. In Europe forest conversion started before the industrial revolution at the end of the 18th century and in North America with the first European settlers due to settlement and cultivation (Williams 2000; Kaplan *et al.* 2009). Today's focus is on tropical and sub-tropical regions where alarming deforestation rates are observed. In 2019, for example, 11.9 million hectares of tree cover was lost in the tropics (GFW 2020).

Commercial and subsistence agricultural expansion accounted for an estimated total of around 73% of all deforestation across tropical and sub-tropical countries between 2000 and 2012 (Hosonuma *et al.* 2012). Forests are converted to provide land for agriculture to produce commodities such as palm oil, soybeans, rice, cocoa, tea and tree plantation woodlands (e.g. eucalyptus, cypress or acacia) (Nepstad *et al.* 2006; Carter *et al.* 2018). In addition, the expansion of commercial agriculture is driven by the growing global demand of beef, where livestock ranches replace natural ecosystems, as is happening in Brazil (Neill *et al.* 1997; Nepstad *et al.* 2006). On top of large-scale agricultural expansion, scarcity of arable land and denseley populated tropical areas urges people to convert forest ecosystems to subsistence agriculture usually cultivated by local smallholders (Delve & Ramisch 2006; Himeidan & Kweka 2012).

The second most important cause of deforestation is infrastructure (10%) and urban (10%) expansion, which leads to forest fragmentation. This is due to road expansion which is often linked to logging. Commercial and illegal logging for timber, pulp and paper often also goes together with agricultural expansion, as the land is usually cultivated for agricultural purposes after conversion. In some countries like Peru, Bolivia, the Democratic Republic of Congo or Indonesia, more than half of the timber exports originate from illegal logging. Furthermore, African forests are degraded by the extraction of fuelwood and charcoal production (Hosonuma *et al.* 2012; Brandt *et al.* 2018). The illegal extraction of timber is coinciding with complex, contradictory and poorly implemented regulations in forested areas (Laurance 1999; Klopp & Sang 2011). Expansion of the road network also brings in settlement and people. In Africa and Asia, urban expansion is projected to increase due to the vast growth of urban population which outpaces rural growth (Montgomery 2008).

Finally, important mineral resources of demand for global trade such as bauxite (aluminium), copper, tin, manganese, iron or gold are found under tropical forests (Sonter *et al.* 2017). Mining-induced deforestation accounts for 7% of the tropical forest loss (NYDF 2019).

2.2 *Effects of land use on soils, hydrological flow pathways and water resources*

The disturbance and conversion of forests may consequently affect the provision of direct and indirect ecosystem services and result in permanent changes of soil physical properties, hydrological flow pathways and quality of water resources.

2.2.1 Soil physical properties

Studies have shown that land cultivation following clearance of natural forests impacts soil physical properties and the water retention capacity of the soil (Bonell 2005; Zimmermann *et al.* 2006; Arnhold *et al.* 2015). Lower field capacity and plant available water content as well as higher wilting point were found in tropical agricultural soils compared to native forest soils (Owuor *et al.* 2018). Agricultural cultivation and land management practices (e.g. using machinery or draft power) contribute to the degradation of soil physical properties, soil structural damage and reduction of macropores in soils, resulting in compaction (Cisneros *et al.* 1999; Price *et al.* 2010). Furthermore, trampling by livestock on pastures also contribute to the compaction of the topsoil (Tollner *et al.* 1990). Compared to the deep and complex rooting system of forest vegetation, vegetation on agricultural land generally has a lower root diversity and depth reducing the development of macropores (Billings *et al.* 2018).

Native forest soils are rich in organic matter and nutrients due to a tighter nutrient cycle through an active soil fungal community, soil biota and fresh organic matter input of the dense biomass (Neill *et al.* 1997). The loss of a dense vegetation cover results in a decline in soil organic matter and soil nutrients (Chiti *et al.* 2018; Owuor *et al.* 2018). Following the clearance of natural forests, the reduction in fresh plant material input leads to an increased mineralization and thus loss of soil organic matter (Chiti *et al.* 2018). A reduction in soil organic matter will also lead to enhanced compaction of the soil and loss of water retention capacity (Owuor *et al.* 2018). These physical changes consequently impede infiltration rates into the soil matrix (Giertz & Diekkrüger 2003; Nyberg *et al.* 2012). A study in Western Kenya showed that infiltration rates were reduced by 60% within 40 years of cultivation (Nyberg *et al.* 2012). Lower infiltration rates on compacted soils lead to increased surface runoff (Figure 1).

In addition to the beneficial effect of vegetation on the soil physical properties and infiltration rates, vegetation cover protects the ground from erosive rainfall through buffering the kinetic energy of rainfall (Bochet *et al.* 1998; Zuazo *et al.* 2004) and the roots lead to a high soil stability by holding the soil together (Gyssels *et al.* 2005). Ground vegetation cover further traps potentially erodable sediment (Rey 2003; Martínez Raya *et al.* 2006). Consequently, agricultural cultivation as a replacement of a dense forest vegetation is an important factor in promoting surface runoff and soil erosion due to the changes in soil physical characteristics, the loss of soil organic matter and the reduced vegetation cover.

Surface runoff is a mechanism for the detachment of soil particles, leading to surface soil erosion. Soil erosion is another reason for the enhanced loss of the topsoil rich in organic matter and nutrients (Martínez-Mena *et al.* 2002), besides increased mineralization, lower input of soil organic matter and nutrient mining through crop uptake under disturbed soils. Nutrients and soil organic matter are essential for plant growth, and, as such, poses limitations for soil health and productivity (Morgan 2005; Liniger *et al.* 2011; Saiz *et al.* 2016). Reduction in soil fertility constrain crop yields of low input agricultural systems in the tropics. This might lead to further deforestation in need for more arable fertile land which may consequently result in more soil erosion.

2.2.2 Hydrological pathways

The changes in soil physical properties following the removal of a natural forest cover impacts catchment hydrology (Giertz & Diekkrüger 2003; Ogden *et al.* 2013). These changes can favour different hydrological flow pathways, such as deep subsurface flow, shallow subsurface flow or surface runoff (Geris *et al.* 2015; Muñoz-Villers *et al.* 2016). Forest ecosystems are characterized by a deep subsurface hydrological connectivity, while a shift from deep 'slow' flow pathways to superficial 'quick' flow pathways are observed in disturbed agricultural catchments (Elsenbeer 2001). These preferential flow pathways can lead to an increase in runoff ratios and peak runoff (Ogden *et al.* 2013). Other studies have found decreased mean transit times for converted forested catchments (Timbe *et al.* 2014; Muñoz-Villers *et al.* 2016). The mean transit time is a fundamental indicator to assess changes in catchment hydrology. Land use might affect flow pathways or catchment storage capacities due to their interrelating changes in soil hydraulic properties (Figure 1) (Mosquera *et al.* 2016; Muñoz-Villers *et al.* 2016).

Tropical natural forests have a so-called 'sponge-effect' with enhanced wet season infiltration rates, reduced direct runoff and flood peaks and increased dry season base flow (Bruijnzeel 2004; Ogden *et al.* 2013). Studies have shown that the dry season runoff recedes more slowly and the runoff rate is greater at the end of the dry season in natural forest catchments compared to agricultural catchments (Ogden *et al.* 2013). The total runoff can be 35% lower with smaller peak runoff rates during floods from natural forests compared to cultivated catchments (Ogden *et al.* 2013). Natural forest catchments lead to flood peak reduction, while the disturbed catchments show higher mean annual streamflow with the occurrence of surface runoff, as it was observed by studies in tropical forest ecosystems in North and South America and East Africa (Muñoz-Villers & McDonnell 2013; Ogden *et al.* 2013; Gebru *et al.* 2019). Hydrological flow pathways are key drivers for the susceptibility to soil erosion in catchments. The shift from deep hydrological flow pathways to surface runoff enhances the potential to deliver water with an increased amount of suspended sediment to the stream network.

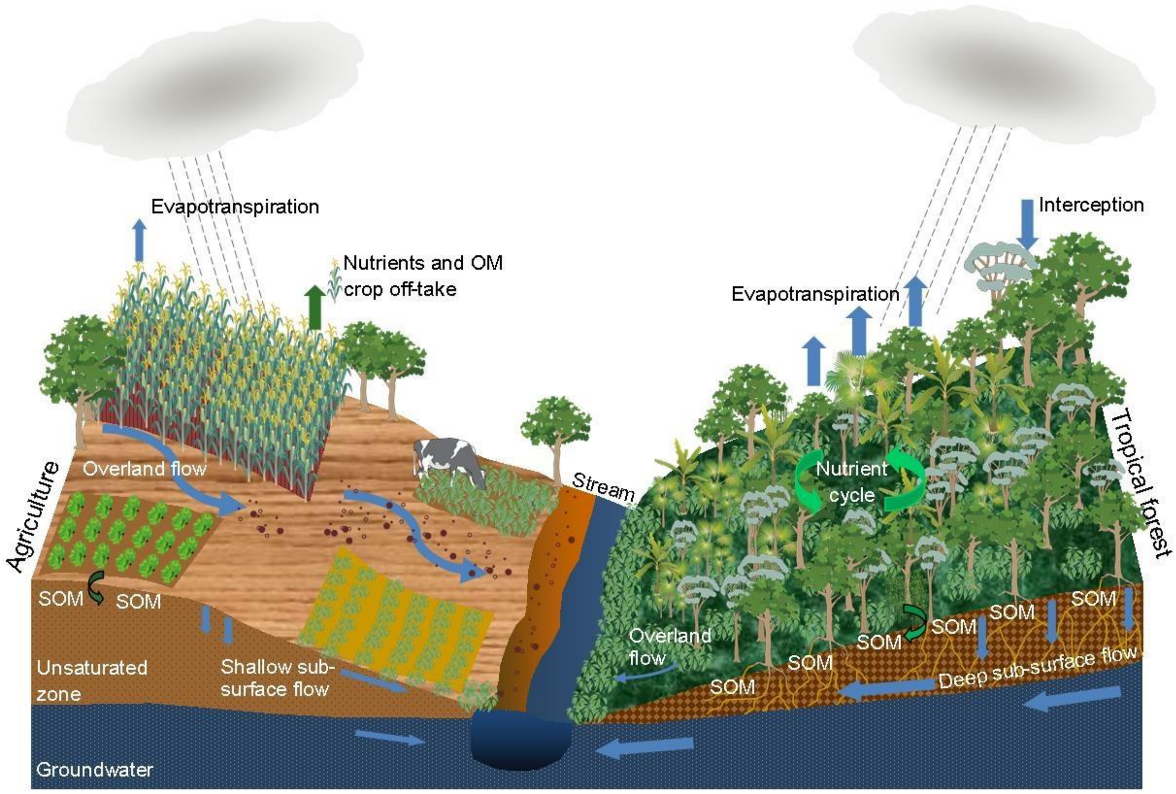

Figure 1 Hydrological and sediment flow pathways in an agricultural and tropical forest ecosystem. SOM = soil organic matter; OM = organic matter.

2.2.3 Suspended sediments in surface water

Increased soil erosion due to changes in soil physical properties and hydrological flow paths are also relevant from the water resource perspective due to increased suspended sediment fluxes. Suspended sediment refers to the fine-grained fraction of the sediment in suspension, generally smaller than 63 µm (Vercruysse *et al.* 2017). Suspended sediments in streams lead to the physical, chemical and biological degradation of water resources (Owens *et al.* 2005; Horowitz 2008). Fine sediment may lower the lifespan and water storage capacity of lakes, water reservoirs or dams through siltation and deposition of sediment (Mogaka *et al.* 2006; Foster *et al.* 2012; Wangechi *et al.* 2015). These changes may impact the economy by reducing capacity for hydropower generation.

Furthermore, the fine sediments and attached pollutants (Domagalski & Kuivila 1993; Thomas *et al.* 2015) or macronutrients (e.g. carbon, nitrogen or phosphorus) (Kronvang *et al.* 1997; Quinton *et al.* 2001; Horowitz 2008) can impact the aquatic ecosystem (Owens *et al.* 2005; Kemp *et al.* 2011) and turn water bodies into an eutrophic state (Foy & Bailey-Watts 1998; Hilton *et al.* 2006). Disturbances and poor land management practices in headwaters can consequently degrade the water quality or affect the primary production for downstream communities and livestock.

The composition of suspended sediments depends on their originating terrestrial source areas (Evrard *et al.* 2013; Laceby *et al.* 2017). The conversion of natural forest ecosystems to agricultural land often leads to an increased number of sediment source areas due to exposure of areas susceptible to soil erosion (hillslopes) caused by removal of the protective vegetation cover. The relative contribution of individual sediment source areas to the total suspended sediment budget at the catchments outlet depends on their activity (e.g. gullies) and connectivity to the stream network (Poesen *et al.* 2003; Fryirs 2013). The increase in surface catchment connectivity as well as the number of active sediment source areas can consequently lead to increased generation of suspended sediments. In order to develop soil conservation strategies to address the negative impacts of suspended sediments on water resources, a thorough understanding is needed about the key drivers and sources underlying sediment dynamics.

2.3 Highlands of the Mau Forest Complex, Kenya

The East African highlands are located above 1,200 m above sea level (a.s.l.) and have daily mean temperatures below 20°C. They are among the most populated areas in East Africa, hosting more than 64% of the total population, while covering only 23% of the total land area of East Africa. Population density ranges between 158 persons per km^2 in Ethiopia and 410 persons per km^2 in Rwanda, with a few exceptions such as Vihiga district, Kenya with 1,200 persons per km^2 (Delve & Ramisch 2006; Himeidan & Kweka 2012). Favourable climatic conditions and inherent fertile volcanic soils make these areas suitable for the development of productive agricultural areas. Scarcity of arable and fertile land together with population pressure pushes people to the most fertile areas located in the proximity of natural ecosystems, but also to the most fragile environments prone to soil erosion. East African montane forest ecosystems on steep hillslopes are thus typically converted to land for agricultural purposes in response to growing population densities (Krhoda 1988; Pellikka *et al.* 2004). Besides forests, wetlands and grasslands are also under extensive conversion (Carter *et al.* 2018).

One such ecosystem is the Mau Forest Complex (400,000 ha), the largest remaining tropical closed-canopy indigenous forest ecosystem of East Africa. The Mau Forest is a complex of sixteen contiguous forests and six separated satellite forests. It covers the western highlands on the western side of the Kenyan Rift Valley at an altitude range between 1,200 and 3,000 m.a.s.l (UNEP 2008; Owino *et al.* 2009). The forest ecosystem builds an important headwater catchment for 12 rivers that feed into Lake Baringo, Lake Nakuru, Lake Natron, Lake Turkana and Lake Victoria. The transboundary Lake Victoria is the second largest fresh water lake in the world and it contributes to the Nile River Basin. The catchment area of the Mau Forest Complex supports livelihoods of approximately 5 million people, livestock, wildlife and the economy (energy, industries, agriculture and tourism) with water resources (UNEP 2008). The natural forest is also home for the indigenous Ogiek people and provides resources for adjacent communities, such as firewood, charcoal, fiber and timber. The forest ecosystem also provides raw material (e.g. timber and pulp) for commercial use and international trade.

The Mau Forest Complex has undergone significant land use changes due to the conversion of natural forest to subsistence and commercial agriculture. The conversion started as early as the colonial time under the British rule (1863-1963), where the Mau Forest was controlled under the central state originally managed by local communities (Ogiek people) (Kimaiyo 2004; Klopp 2012). The colonial state mapped forest boundaries appropriated for the government. The land appropriation process was followed by the establishment of tea and tree plantations, using mainly exotic species (e.g. *Pinus patula*, *Cupressus lusitanica*, *Eucalyptus* spp.), and settlements in 1930 (Binge 1962; Kimaiyo 2004). After independence during post-colonial times, the alteration of forest boundaries continued in order to legitimize areas for natural resource extraction by large-scale logging (Klopp 2012). During the last few decades, around 25% of the area of the Mau Forest Complex, representing 107,707 ha, has been converted to agriculture and settlements (Owino *et al.* 2009). Today's forest cover loss can be mainly attributed to the expansion of subsistence agriculture due to land scarcity and population density (Were *et al.* 2013). Continuous and unsustainable extraction of natural resources, encroachment through settlement and livestock grazing lead to the degradation of the Mau Forest Complex.

2.4 Review of studies within the Lake Victoria Basin and Kenya

Soil erosion in Kenya was documented as early as 1935, whereas sediment yield monitoring was limited to a few major catchments between 1948 and 1965 (Ongwenyi *et al.* 1993). Plot level sediment and turbidity studies in the Sondu River, which originates in the Mau Forest Complex, linked agricultural activities to increased sediment and turbidity values (Shepherd *et al.* 2000; Masese *et al.* 2012; Njue *et al.* 2016). Masese et al. (2012) reported an increase in turbidity in the lower part of the Sondu River Basin between 1986 and 2011. However, the studies are restricted to small-scale experimental plots or to low sampling frequency at catchment level. The absence of a turbidity-suspended sediment relationship and low sampling frequency without accompanied flow measurements makes calculations of annual load uncertain. Other sediment studies focused on large river basins such as the Tana River (Tamooh *et al.* 2012, 2014), including sediment input to reservoirs (Brown *et al.* 1996) or at the Tana Estuary (Kitheka *et al.* 2005). These studies draw the importance onto floodplains and hydro-power reservoirs in trapping and depositing increased suspended sediments at downstream reaches of the Tana River. A discontinuous sediment study over 3.5 years in the Mara River Basin demonstrated that catchment disturbances by wildlife and domestic livestock were linked to increased sediment loads in a semi-arid catchment. This is in contrast to lower stream sediments originating in the Mau Forest Complex coinciding with higher annual rainfall (Dutton *et al.* 2018). These studies have in common that turbidity and siltation are recognized as the principal causes of water resource degradation in Kenya and partially linked to land use change. However, a platform for long-term sedimentological and hydrological measurements in comparison with different land use is missing.

Several modelling studies applied in the Lake Victoria Basin observed that land use change resulted in higher surface runoff (Githui *et al.* 2010), increased stormflows and reduced dry season flows (Mati *et al.* 2008; Mango *et al.* 2011), increased soil erosion and sediment yields (Ssegane *et al.* 2008; Defersha & Melesse 2012). However, empirical models tend to be highly parameterized which may restrict modelling performance in areas of limited data availability and digital elevation models of low resolution (Jayakrishnan *et al.* 2005).

Impacts of land use change or anthropogenic disturbance on hydrological functions are catchment-dependent and may vary based on a number of factors including climatic conditions, and hydrogeological and catchment physical properties. This highlights the importance to improve the understanding on hydrological regimes and sediment dynamics of under-researched or typically un-gauged tropical montane catchments such as the headwaters in the Mau Forest Complex, that are under pressure of natural forest conversion to agricultural systems. Sustainable management practices can be applied to also decouple sediment source areas in these headwater catchments to sustain the water quality for downstream environments, such as Lake Victoria. Land use changes and poor agricultural management in the headwaters of the Mau Forest Complex, both directly and indirectly, affect fishing activities, tourism, industry, biodiversity and the livelihood of local communities in the lake basin (Wangechi et al., 2015) and impose additional economic costs. Lake Victoria exhibit high sediment accumulation rates of 2.3 mm yr^{-1} based on historical records (Verschuren *et al.* 2002) and increased signs of eutrophication (Sitoki *et al.* 2010) stressing the need to mitigate sediment generation and their associated macronutrient inputs in headwater regions.

3 Description of the study area and methods

3.1 Study area

The three catchments in which this study was carried out are located in the headwaters of the Sondu River Basin (3,470 km^2) in the Mau Forest Complex, Kenya between the longitudes 34°04'E-34°49'E and latitudes 0°17'S to 0°22'S (Figure 2). The Sondu River feeds into Lake Victoria, the second largest fresh water lake in the world, an important water resource for about 30 million people in five countries (Mogaka *et al.* 2006). The river basin ranges from 2,935 m a.s.l. on the top of the Mau Escarpment in the east to the flood plains of Lake Vicotria at 1,134 m a.s.l. in the west. The catchments are part of an on-going monitoring programme funded by the German Federal Ministry for Economic Cooperation and Development (Grant 81206682) and the German Science Foundation (Deutsche Forschungsgemeinschaft DFG, Grant BR2238/23-1), which is aimed at assessing the effect of land use on different water quality and quantity parameters and started in October 2014. The study catchments were used as a 'microcosm' to understand process response to disturbance where the findings are transferable to other tropical montane systems under similar pressures. The catchments were chosen based on the criteria of different land use and comparability between the catchment characteristics, such as surface area, shape, morphology, geology, pedology and climate to assess the effect of land use on sediment dynamics and hydrology. The catchments are dominated by (1) natural forest vegetation (35.9 km^2), (2) tea-tree plantations (33.3 km^2) and (3) smallholder agriculture (27.2 km^2) (Figure 3).

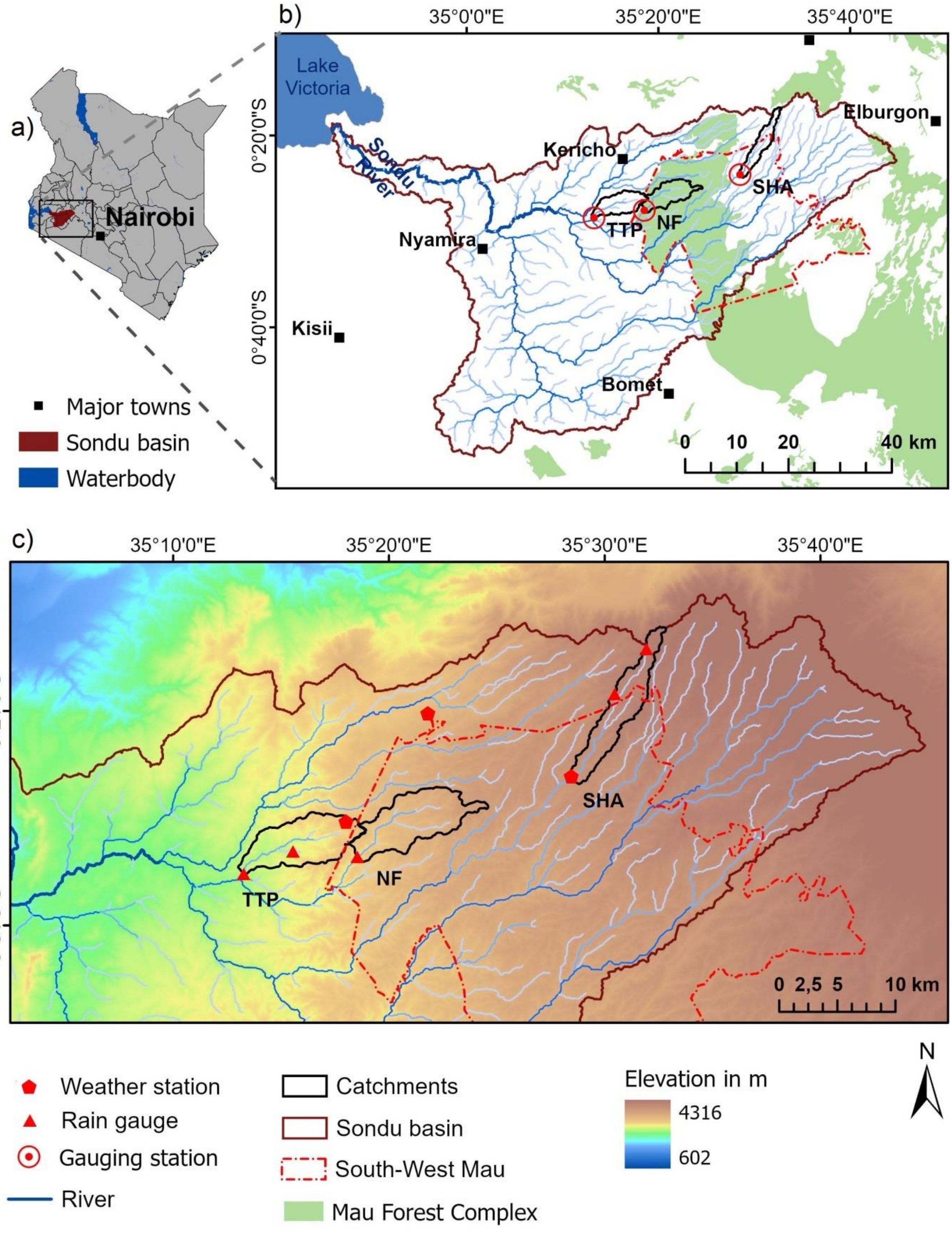

Figure 2 Overview of the a) Sondu River Basin within Kenya showing the b) location of the automatic monitoring equipment (gauging station) at the outlet of the tea-tree plantations (TTP), natural forest (NF) and the smallholder agriculture (SHA) in the headwaters of the Sondu Basin and elevation (SRTM digital elevation model 30 m resolution; USGS, 2000) and c) land use map with weather stations and tipping bucket rain gauges.

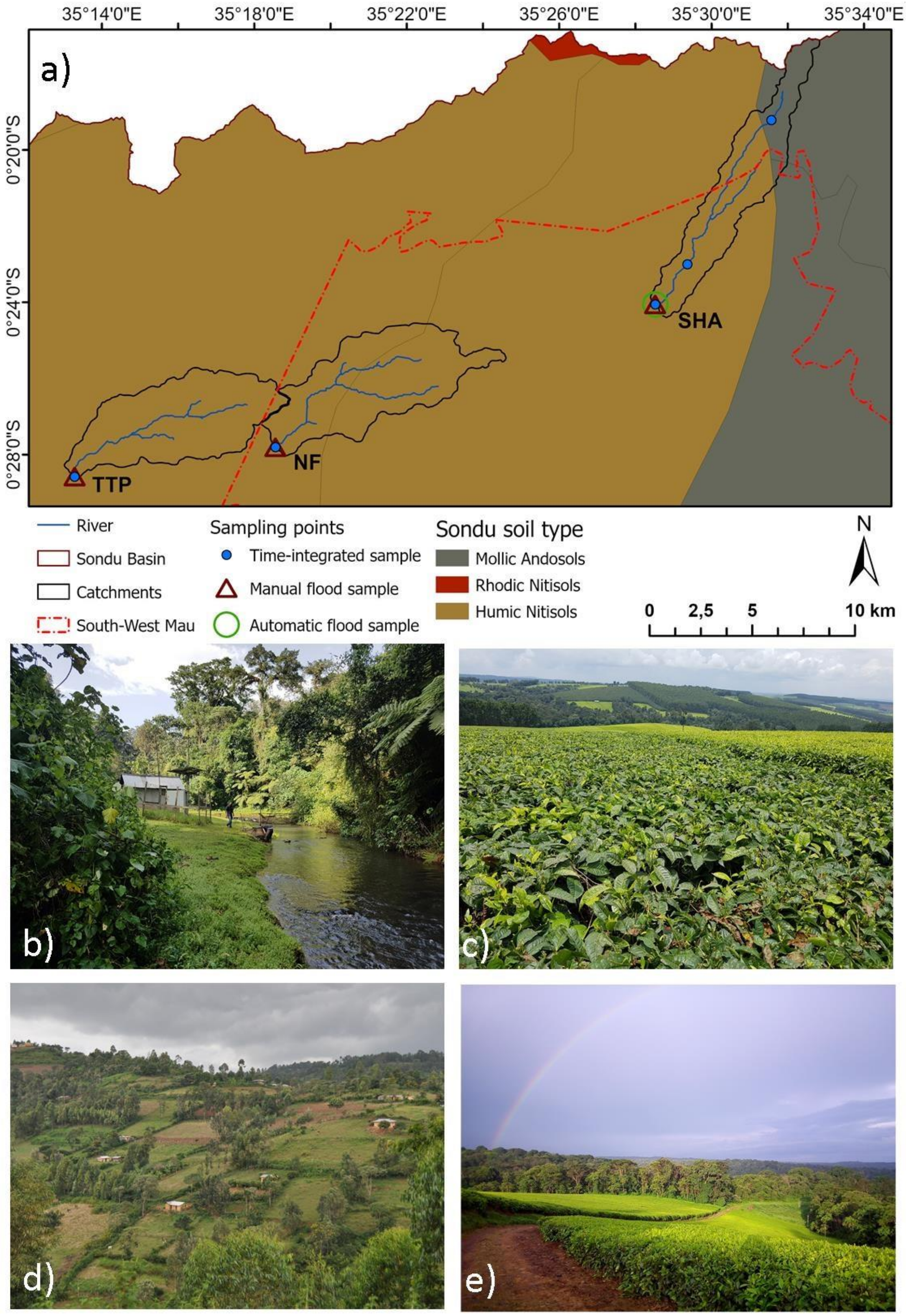

Figure 3 Locations of the a) suspended sediment sampling points in the tea-tree plantation (TTP), natural forest (NF) and smallholder-agriculture (SHA) catchments and dominant soils (Soil and Terrain database for Kenya (KENSOTER) version 2.0; ISRIC, 2007). Photos of the b) outlet of the natural forest, c) the tea plantations and in the background tree plantations, d) hillslopes of the smallholder agriculture and e) boundary between the tea plantations and the South-West Mau.

The natural forest catchment is part of the the South-West Mau, the largest forest block in the Mau Forest Complex. The Mau Forest Complex is one of the last remaining isolated patches of closed-canopy montane forests in Kenya. As an afromontane mixed forest, indigenous broad-leafed evergreen trees and shrubs dominate the area with a complex vegetation pattern. The forest vegetation follows altitudinal zones from west to east: a lower montane forest is found above 1,200 m a.s.l. including species such as *Polyscias kikuyuensis*, *Macarangea kilimandscharica*, *Olea hochstetteri*, *Casearia battiscombei* and *Fagara* spp., transitioning to irregular patches of bamboo forest and grassland vegetation above 2,300 m a.s.l. (Binge 1962). In highly degraded areas natural vegetation is replaced by cultivated land (Owino *et al.* 2009; Kinyanjui 2011). A riparian zone transists from the forest vegetation containing an understorey of a dense cover of shrubs and tree ferns combined with tall indigenous tree species throughout the catchment. The South-West Mau is bordered by tea and tree plantations at the western side and degraded through encroachment by smallholder farmers on the eastern upland side, transitioning to subsistence smallholder agriculture. A buffer belt of governmental tea plantations along the eastern and western forest boundaries have been established to prevent further encroachment of adjacent local communities (NTZD 2015). The natural forest catchment (35.9 km^2) was chosen based on the relatively low levels of forest disturbance in the South-West Mau (Figure 2).

The tea-tree plantation catchment ranges from 1,788 to 2,141 m a.s.l. and borders the South-West Mau to the west. Commercial tea and tree plantations including mainly exotic species (e.g. *Pinus patula*, *Cupressus lusitanica*, *Eucalyptus* spp.) and settlements were established at the beginning of the 20th century (Binge 1962). The catchment is dominated by tea (*Camellia* spp.) plantations alternated with *Eucalyptus saligna* and *Cupressus lusitanica* woodlots that are used for timber production and fuelwood for the tea factories. Some of the tea companies apply soil conservation strategies to control soil erosion, such as mulching, terracing, planting rows of oat grass interplanted between rows of tea and cover trees with mature tea trees during the establishment of new tea bushes. Herbicides are commonly used to control weeds. Aerial application of inorganic fertilizer is conducted two to three times per year on the tea plantations (150-250 kg N ha^{-2} yr^{-1} and 8-13 kg P ha^{-2} yr^{-1}) (Jacobs *et al.* 2018b). In general, unpaved and paved roads are linked to well-maintained sited cut-off drains or open culverts. The design of the well-engineered drainage system takes into account the optimal routing of surface runoff to the riparian zones before entering the streams. The riparian buffer zone of approximately 30 m includes an intact canopy of mixed indigenous tree species with a dense surface cover (Figure 3).

The smallholder agriculture catchment is located in the upper part of the Sondu River Basin above an altitude of 2,400 m a.s.l. A heterogeneous mosaic of farmlands on average smaller than one hectare characterizes the landscape (Figure 3). Subsistence farmers grow vegetable crops such as maize, beans, peas, kale, cabbage, and potatoes, millet and tea (*Camellia* spp.). A combination of hoeing and herbicide application is used for weed control. Inorganic fertilizer is applied manually on potatoes and maize (23-45 kg N ha^{-2} yr^{-1} and 12-23 kg P ha^{-2} yr^{-1}), while manure is used for cabbage and other greens. Tillage in the form of hand-hoe cultivation and ploughing with oxen are the common practices for soil preparation. Soil management strategies such as vegetated buffer strips cultivated with Napier grass (*Pennisetum purpureum*) were only occasionally observed on the mostly steep agricultural hillslopes. Crop residues often remain on the fields. Grasslands for livestock and gazetted forests of *Eucalyptus saligna*, *Cupressus lusitanica* and *Pinus patula* are alternated with croplands. Existing market opportunities for wood products has motivated farmers to grow these tree species to improve livelihoods. Mosaics of bamboo forests, as remains of the natural forest, grow naturally around springs. Road density of the catchment can be considered high, however the quality of the roads is low. The frequent used unpaved roads are highly compacted. They often run parallel to the slope connecting the hillslopes with the stream network. Under these conditions unpaved roads often become deeply incised gullies. Steep hillslopes connect with small floodplains in a steep and narrow valley floor. The riparian zone is mostly absent or replaced by Eucalyptus woodlots or agricultural land, which makes channel banks susceptible to erosion. The rural agricultural highlands face commonly high population densities and severe poverty linked to low agricultural productivity (Place *et al.* 2006).

The three catchments are characterized by long and steep hillslopes with a maximum slope gradient of 72% in the natural forest catchment. The elevation ranges from 1,788 to 2,691 m a.s.l. The streams are first- and second-order perennial streams merging together to the Sondu River (a sixth-order stream) (Figure 2). The drainage density ranges between 0.42 and 0.64 km km^{-2}. The north-south movement of the Intertropical Convergence Zone (ITCZ) results in seasonal shifts and duration in rainfall regimes in Kenya with a bimodal rainfall pattern in the study area (Camberlin 2018). The wet seasons are between March and June and between October and December referred to the ‘long rainy season’ and the ‘short rainy season’, respectively. An intermediate rainy season occurs between the two wet seasons. The mean annual rainfall is 1,979 ± 325 mm yr^{-1} (period 1905-2019) with rainfall peaks in April and May (>260 mm $month^{-1}$). January and February is the dry season with monthly rainfall <95 mm $month^{-1}$. Temperature and evapotranspiration tends to increase with decreasing altitude. The mean annual temperature is 12°C at 2,935 m a.s.l. and increases to

16°C in the town Kericho at 2,100 m a.s.l. The minimum daily temperature is 11°C and exceeds 25°C during the dry season, while June and July are usually the coldest months with maxima around 22°C (Jacobs *et al.* 2017). Evapotranspiration decreases from about 1,800 to 1,400 mm yr^{-1} from lower (1,800 m) to higher (>1,900 m) altitudes, respectively (Krhoda 1988).

Geologically, Kericho Phonolites cover the lower catchment (tea-tree plantations), followed by phonolitic nephelinites with intercalated tuffs and Mau ashes with basal tuff encompassing the natural forest catchment, while phonolitic nephelinites comprises the upper catchment in the smallholder agriculture (Binge 1949; Jennings 1962). The catchments are covered by up to 3-6 m deep and well-drained, volcanic loamy soils (Sombroek *et al.* 1982), characterized as mollic Andosols and humic Nitisols (ISRIC 2004) with moderate to high amounts of organic matter (Figure 3) (Dunne 1979).

3.2 Methodologies of monitoring

This research used a combination of biophysical data and modelling to identify sediment and hydrological dynamics of catchments under contrasting land use. To unravel the complex dynamics of suspended sediments, a four-year time series of hydrological (including runoff and rainfall) and sedimentological data in 10-minute resolution from 2015 to 2018 was analyzed (Chapter 4). To quantify the relative contribution of different sediment sources in the smallholder agriculture catchment, a sediment fingerprinting approach was used and four different sediment sources, such as agricultural land (n=137), unpaved tracks (n=60), gullies (n=19) and channel banks (n=32), and suspended sediment as target samples (n=35) were analysed for their geochemical and biogeochemical elements (Chapter 5). The effect of land use on the different sediment-associated macronutrient (total carbon, nitrogen and phosphorus) concentrations were quantified (Chapter 6).

The outlet of each of the three catchments is equipped with an automatic stream monitoring system recording data on 10-minute resolution. A radar sensor (VEGAPULS WL61, VEGA Grieshaber KG, Schiltach, Germany) records the distance to the water level to determine water level or stage. The estimated water level was related to stream discharge based on a site-specific second-order polynomial stage-discharge relationship. The calibration was checked over a wide range of stream flows using salt-dilution gauging (Shaw *et al.* 2011), an Accoustic Doppler Velocimeter (ADV; FlowTracker, SonTek, San Diego CA, USA) or an Acoustic Doppler Current Profiler (ADCP; RiverSurveyor S5, SonTek, San Diego, USA) depending on river size and discharge (Jacobs *et al.* 2018b) (Chapter 4 and 6). For water quality, a UV/Vis spectrophotometer (spectro::lyser, s::can Messtechnik GmbH, Vienna, Austria) measures *in situ* turbidity in FTU (formazin turbidity unit) by transmitting a beam of light to an optical receptor (Figure 4). With an increase in turbidity, the transmission of light decreases. Turbidity was used as a surrogate for suspended sediment concentrations. An *ex situ* linear turbidity-sediment relationship was used to convert long-term turbidity data into suspended sediment concentrations. The calibration method is described in Chapter 4.3.2 (Chapter 4 and 6). The monitoring equipment is visited on a weekly to bi-weekly basis for maintenance and downloading of the data. Since January 2016, the data is additionally automatically uploaded to an online database, except for the site at the natural forest due to network restrictions (Figure 2).

In addition to the automatic stream monitoring, a total of eight tipping bucket rain gauges (5 tipping bucket rain gauges: Theodor Friedrichs, Schenefeld, Germany and 3 weather stations: ECRN-100 high resolution rain gauge) record rainfall data programmed to measure cumulative precipitation per 10 minutes with a 0.2 mm resolution. One weather station each was installed in the tea-tree plantation and the smallholder agriculture catchment, while the third weather station is located at the boundaries of the South-West Mau. In addition to the weather stations, two tipping bucket rain gauges were evenly distributed in each catchment of the tea-tree plantations and the smallholder agriculture, while one was installed at the outlet of the natural forest catchment. The tipping bucket rain gauges are checked on a monthly basis for maintenance and downloading of the data. The average rainfall over each catchment area is estimated by weighting the contribution of rainfall measured at each tipping bucket using Thiessen polygons (Figure 2) (Chapter 4 and 6).

Suspended sediment was collected for the turbidity-sediment calibration, the sediment fingerprinting and the qualitative analysis of suspended sediment (Chapter 4, 5 and 6). Sediment traps (time-integrated sediment sampler) were installed at the outlet of each catchment and two additional locations upstream of the outlet of the smallholder agriculture catchment following the method by Phillips et al. (2000) (Figure 3 and Figure 4). A polyvinylchloride (PVC) pipe (0.045 m (ID) x 0.30-0.50 m) was attached to two metal or wooden bars, using plastic cable ties or brackets. The bars were screwed to a metal or wooden stand to secure stability on the stream bed. The ends of the PVC pipe were sealed and an inlet and outlet (<2mm) towards the top of each end allows stream water to pass through. A cap on one end allows the frequent removal of trapped sediment (Figure 4). The sediment trap is oriented directly parallel to the flow direction of the stream. Flow velocity reduces within the chamber of the pipe to allow suspended sediment to settle. The suspended sediment samples were collected every three to five days. In addition to the sediment traps, event-based flood samples were manually collected with bulk river water samples (~10 L) at the outlet of each catchment. At the outlet of the smallholder agriculture automatic water samplers were additionally installed (3700 Full-size portable sampler, Teledyne ISCO, Lincoln, USA) to collect 0.5 L samples during the rising and falling limbs of the storm hydrograph (Figure 3).

Figure 4 a) Water monitoring equipment with b) turbidity sensor and c) sediment trap at the outlet of the natural forest catchment.

A flow diagram of the main data chapters is presented with the main aim, objectives, set hypotheses, datasets required and the key message of each data chapter (Figure 5).

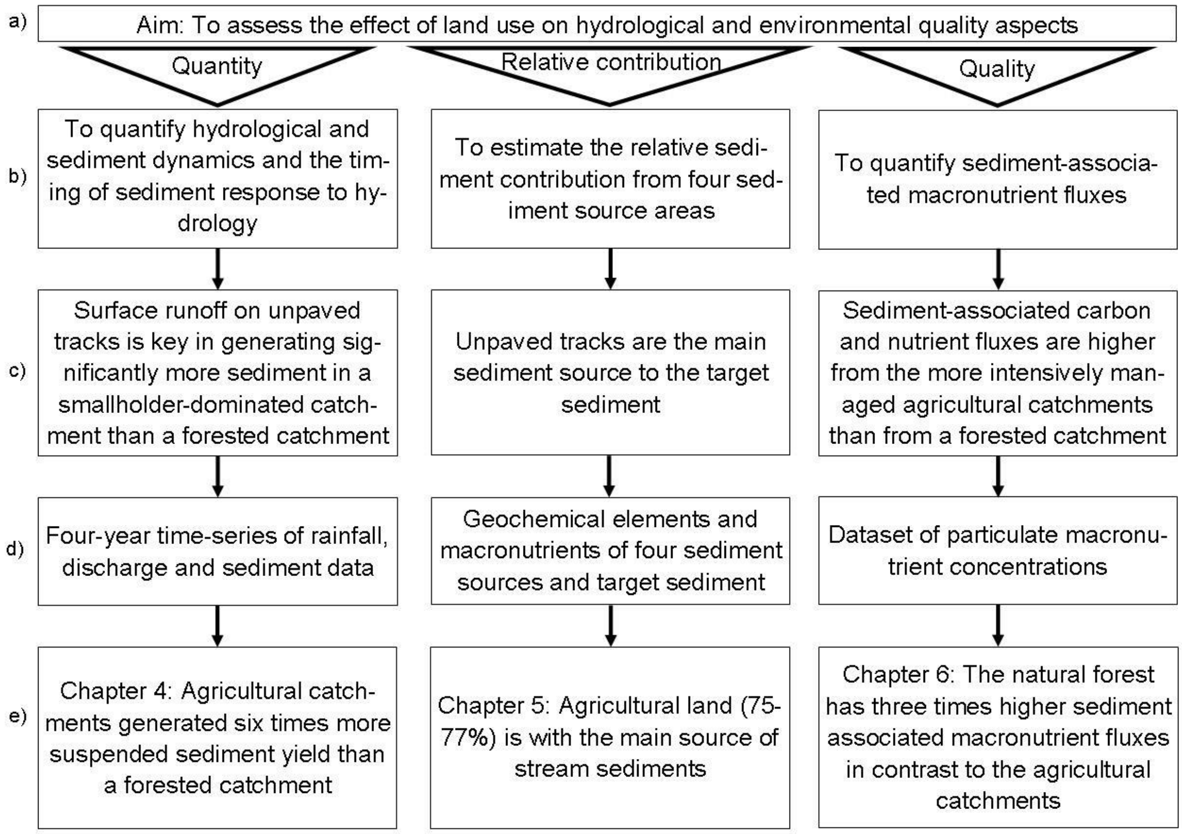

Figure 5 Flow diagram of the study about the (a) aim, (b) objectives, (c) hypotheses, (d) dataset required and (e) results of the data chapters.

4 Tropical montane forest conversion is a critical driver for sediment supply in East African catchments

4.1 Introduction

The conversion of native ecosystems to agriculture leads to the degradation of soil properties (Morgan 2005; Githui *et al.* 2009; Owuor *et al.* 2018), which can increase soil erosion rates (Bruijnzeel 2004). Soil erosion does not only deplete fertile topsoil from agricultural land, but also leads to water quality deterioration caused by an increase in fine suspended sediments (Brown *et al.* 1996; Quinton *et al.* 2001; Horowitz 2008). Hence, suspended sediment physically affects the fluvial network, (Owens *et al.* 2005) polluting drinking water for communities, livestock and wildlife, and impacting downstream water reservoirs and hydropower generation because of the accumulation of the sediments (Mogaka *et al.* 2006; Foster *et al.* 2012; Wangechi *et al.* 2015). Additionally, pollutants such as pesticides (Brown *et al.* 1996) and nutrients (phosphorus or nitrogen) (Fraser *et al.* 1999; Quinton *et al.* 2001; Horowitz 2008) can be attached to sediments and can harm aquatic biota (Owens *et al.* 2005; Kemp *et al.* 2011; Gellis & Mukundan 2013). The increase in nutrient concentrations can result in eutrophication of water bodies (Foy & Bailey-Watts 1998; Hilton *et al.* 2006; Mogaka *et al.* 2006).

Land use change in catchments with a strong connectivity between sediment sources and streams can abruptly increase sediment supply to the fluvial system (Fryirs 2013). There may be multiple sediment source areas governing sediment supply, such as hillslope soils (Minella *et al.* 2008; Didoné *et al.* 2014), gullies (Poesen *et al.* 2003; Minella *et al.* 2008; Fan *et al.* 2012), riverbanks (Trimble & Mendel 1995; Lefrançois *et al.* 2007) and unpaved tracks (Ziegler *et al.* 2001; Minella *et al.* 2008; Ramos-Scharrón & Thomaz 2016). The term connectivity can be further sub-divided into two distinct types, such as structural and functional connectivity. Structural sediment connectivity is often defined as a contiguous and physical linkage (e.g. topography, surface cover) within the landscape, while the latter is described as a process-based concept (e.g. rainfall characteristics, runoff) that links landscape units (Zingaro *et al.* 2019). However, the functional connectivity can also include geomorphic processes that drive fluxes of water and sediment (e.g. processes leading to sediment transport and availability) (Wainwright *et al.* 2011; Masselink *et al.* 2016). Unravelling sediment dynamics is complex and requires continuous high-frequency monitoring because suspended sediment concentrations change rapidly throughout individual storms (De Girolamo *et al.* 2015; Sun *et al.* 2016; Vercruysse *et al.* 2017).

Alternatively, turbidity observations can be used as a surrogate to determine in-stream suspended sediment concentrations, which allow for establishing continuous *in situ* suspended sediment datasets, even for remote sites (Lewis 1996; Ziegler *et al.* 2014; Minella *et al.* 2018).

Streams in montane headwaters are major contributors to suspended sediment yield because of the steep terrain that leads to a strong hillslope to channel connectivity (Wohl 2006; Grangeon *et al.* 2012; Morris 2014). Significant increases in suspended sediment yield can be expected from tropical montane headwater catchments, which are heavily affected by deforestation followed by cultivation of erosion prone areas, often without soil conservation measures (Wohl 2006; Ramos-Scharrón & Thomaz 2016). Land use change may impact catchment hydrology and runoff mechanisms in the tropics (Muñoz-Villers & McDonnell 2013; Ogden *et al.* 2013), which are key processes determining sediment yields.

In East Africa, land use change in tropical montane forests are mainly driven by scarcity of arable land (Pellikka *et al.* 2004) with the most fertile lands located in the proximity of natural ecosystems (Krhoda 1988). The Mau Forest Complex exemplifies this case, with one quarter of the forest converted to agricultural land over the last four decades (Brandt *et al.* 2018), in addition to the clearances at the beginning of the 20th century to establish commercial tea plantations (Binge 1962). Due to its location in the highlands of Kenya, the Mau Forest is a critical catchment area for the country; it is the headwater to twelve rivers, one of which is the Sondu River, a tributary of Lake Victoria (UNEP *et al.* 2005; Mogaka *et al.* 2006).

Eutrophication and sedimentation are major environmental problems affecting Lake Victoria, where sediments are estimated to accumulate at a rate of 2.3 mm yr^{-1} (Verschuren *et al.* 2002). Although authorities in Kenya acknowledge the need to reduce sediment pollution, the linkages between land use change and changes in sediment dynamics in the headwater catchments are not well quantified (Nyssen *et al.* 2004; Vanmaercke *et al.* 2010, 2014). There is limited data on sediment export for montane catchments in sub-Saharan Africa in general, and in East Africa in particular (Walling & Webb 1996; Ntiba *et al.* 2001). Clearly, this is a significant gap in knowledge of these environments that requires empirical measurements to address it. Not only will these measurements improve the understanding of these under-researched environments, but they will also assist in the development of targeted soil and water conservation strategies to disconnect sediment source areas in the upper catchments of the Mau Forest from the fluvial system and downstream environments, including Lake Victoria.

The overall aim of this study was to elucidate the spatial and temporal dynamics of suspended sediment and to quantify suspended sediment loads in tropical montane streams under contrasting land uses using a four year high-temporal resolution dataset. The main objectives were: (a) to quantify rainfall, streamflow and suspended sediment transport dynamics, (b) to compare the seasonal responses in suspended sediment yield, (c) to assess the timing of the response of suspended sediment to rainfall and discharge and (d) to improve the understanding of the dominant water flow pathways. The hypothesis was set out that surface runoff diverted on unpaved tracks is key in generating significantly more sediment in a smallholder-dominated catchment than a forested catchment.

4.2 Materials and Methods

4.2.1 Catchment characteristics and site description

The three catchments studied are located in the headwaters of the Sondu River Basin (3,470 km^2) in the western highlands of Kenya (Figure 6). Each catchment is dominated by a distinct land use: (1) natural forest (NF; 35.9 km^2), (2) smallholder agriculture (SHA; 27.2 km^2) and (3) tea-tree plantations (TTP; 33.3 km^2). The Sondu River drains into Lake Victoria, which is the second largest fresh water lake in the world, an important water and economic resource for five countries and one source of the Nile River.

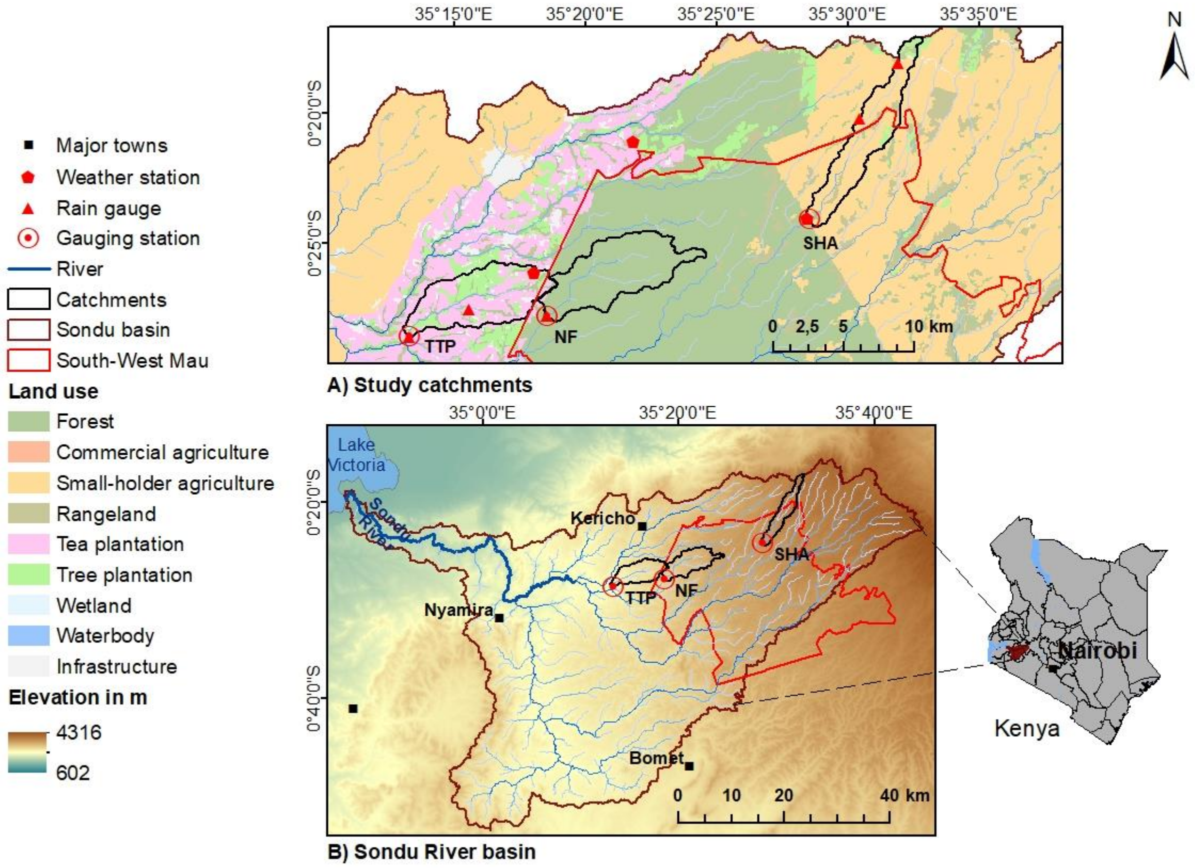

Figure 6 Overview of the A) study catchments: tea-tree plantations (TTP), natural forest (NF) and the smallholder agriculture (SHA), showing locations of gauging and weather stations, tipping bucket rain gauges and land use in the B) Sondu River Basin and its outlet to Lake Victoria (SRTM digital elevation model 30 m resolution; USGS, 2000) in Western Kenya.

The three catchments (Table 1) are characterized by steep hillslopes with a maximum slope gradient of 72% in the natural forest catchment. The streams are mostly first- and second-order perennial streams that merge together to form the River Sondu (a sixth-order stream). The rainy seasons are bimodal with a long rainy season between March and June, and a short rainy season between October and December with a continued intermediate rainy season between the two wet seasons. Mean annual precipitation is 1,988 ± 328 mm (period 1905-2014) with rainfall peaks in April and May (>260 mm month^{-1}). January and February (<95 mm month^{-1}) are the driest months.

Table 1 Physical characteristics of the three catchments under different land use natural forest, tea-tree plantations and smallholder agriculture in the South-West Mau, Kenya.

	Natural forest	Tea-tree plantations	Smallholder agriculture
Outlet coordinates[a]	35°18'32.0472"E 0°27'47.592"S	35°13'17.22"E 0°28'34.9176"S	35°28'31.7316"E 0°24'4.0248"S
Area (km^2)	35.9	33.3	27.2
Elevation range (m a.s.l.)	1,968-2,385	1,788-2,141	2,389-2,691
Mean slope ± SD (%)	15.7±8.4	12.4±7.6	11.6±6.7
Basin order (Strahler)	2	2	2
Drainage density (km km^{-2})	0.48	0.42	0.64
Soil infiltration rate (mm hr^{-1})[b]	760±500	430±290	401±211
Geology[c]	Igneous rock (Volcanic) (100%)	Igneous rock (Volcanic) (100%)	Igneous rock (Volcanic) (72%) & Pyroclastic (28%)
Dominant soils[c]	Humic Nitisols (100%)	Humic Nitisols (100%),	Humic Nitisols (72%) & mollic Andosols (28%)
Vegetation	Afromontane mixed forest, grassland, bamboo, broad-leafed evergreen trees and shrubs	Tea plantations with woodlots of *Eucalyptus* spp., *Cypress* spp. and *Pinus* spp.	Perennial & annual crops (maize interspersed with beans, potatoes, millet, cabbage and onions), woodlots, grassland
Riparian vegetation	Forest vegetation	>30 m buffer with indigenous vegetation	Degraded riparian vegetation, Eucalyptus woodlots

[a]WGS 1984 UTM Zone 36S
[b]Owuor et al. 2018
[c]KENSOTER Geology data from the Soil and Terrain database for Kenya (KENSOTER) version 2.0

The natural forest catchment is located in the South-West Mau block of the Mau Forest Complex. The Mau Forest is an afromontane mixed forest dominated by indigenous broad-leafed evergreen trees and shrubs with a complex vegetation pattern. Riparian forests with a mixture of indigenous vegetation are present throughout the catchment. The natural forest catchment is characterized by high infiltration rates with the occurrence of shallow to deeper subsurface water pathways, whereas the tea-tree plantation and the smallholder agriculture catchments have lower infiltration rates with a dominance of surface runoff (Jacobs *et al.* 2018a; Owuor *et al.* 2018) (Table 1).

In the smallholder agriculture catchment, subsistence farmers grow maize interspersed with beans, potatoes, millet and cabbage on small farms (circa 1 ha). Small-scale tea plantations, eucalyptus (*Eucalyptus* spp.), cypress (*Cypressus* spp.) and pine (*Pinus* spp.) woodlots are interspersed with crop fields and grazing land (Table 1). A combination of hand weeding, hoeing and herbicides is used for weed control. Bamboo (*Bambusa* spp.) is generally found around natural springs. The whole catchment is connected by a dense network of unpaved tracks either bare or sparsely covered by grass; stream crossings rarely have bridges. The heavily travelled unpaved tracks have commonly become eroded gullies (Figure 7a-b), that run down the slope to rivers, connecting surface runoff from surrounding fields with the stream network (Figure 7c-e). Cattle entrance points to the stream are generally highly disturbed and have degraded riverbanks (Figure 7d-f). The natural riparian vegetation is in many areas replaced by Eucalyptus woodlots or small bushes. In some places, riparian wetlands are found.

Figure 7 a-c) Incised and unpaved tracks provide a direct connection with the stream, d-e) degraded and disturbed riverbank from livestock entering the streams and f) eroded suspended sediments in streams within the smallholder agriculture catchment.

The tea-tree plantation catchment has tea fields alternated with *Eucalyptus* spp. and *Cypress* spp. woodlots that are used for fuelwood at the tea factories. Some of the tea companies use mulch and rows of oat grass between rows of tea to control soil erosion during the establishment of new tea bushes. Herbicides are commonly used to control weeds. Cover crops with mature tea trees during the establishment of a new tea crop, terracing and sited cut-off drains are also used within the catchment to control soil erosion. The catchment is covered by a network of well-maintained paved and unpaved roads, linked to drainage systems, such as open culverts along the roads that connect them to the streams. The riparian vegetation includes a mix of indigenous tree species that cover densely the ground and form a buffer of approximately 30 m (Table 1).

The study area is underlain by folded volcanics from the early Miocene times. Porphyritic phonolites, a member of the sequence of basic and intermediate lavas (igneous rocks) are predominant in the study area (Binge 1962), where pyroclastic rocks cover the upper part of the catchment (ISRIC 2004). The study area comprises well-drained, very deep (>1.8 m) dark-red and dark-brown loamy soils (Sombroek *et al.* 1982), with moderate to high amounts of organic matter under the forest cover (Dunne 1979).

4.2.2 Automated hydrological and sediment monitoring

This study uses a four year dataset (January 2015 to December 2018) on rainfall, discharge and turbidity with a 10 minute resolution (Figure 6). A radar sensor (VEGAPULS WL61, VEGA Grieshaber KG, Schiltach, Germany) collected continuous water level measurements. Water level (‘stage’) was used to determine stream discharge based on a site-specific second-order polynomial stage-discharge relationship (Jacobs *et al.* 2018b). The calibration was checked over a wide range of stream flows using salt-dilution gauging (Shaw *et al.* 2011), an Acoustic Doppler Velocimeter (ADV; FlowTracker, SonTek, San Diego CA, USA) or an Acoustic Doppler Current Profiler (ADCP; RiverSurveyor S5, SonTek, San Diego, USA) depending on river size and discharge (Jacobs *et al.* 2018b). Specific discharge [mm day^{-1}] was determined by integrating instantaneous discharge taken at 10 minute intervals over a day and relating it to the catchment area. Precipitation was measured using eight automatic tipping bucket rain gauges calibrated to measure cumulative rainfall every 10 minutes with a 0.2 mm resolution (5 tipping bucket rain gauges: Theodor Friedrichs, Schenefeld, Germany, and 3 weather stations: ECRN-100 high resolution rain gauge). Using Thiessen polygons, the weighted contribution of rainfall of every tipping bucket in each catchment was estimated. A more detailed description of the study sites and instrumentation can be found in Jacobs et al. 2018b. Turbidity was measured *in situ* as a surrogate for suspended sediment concentrations using a UV/Vis spectroscopy sensor (spectro::lyser, s::can Messtechnik GmbH, Vienna, Austria). Turbidity is measured in FTU (formazin turbidity unit) by transmitting a beam of light to an optical receptor. With an increase in water turbidity the transmission of light decreases. To calculate sediment concentrations, a site-specific turbidity-suspended sediment calibration was established (section 4.2.3.1). Before each turbidity measurement, the window of the sensors was automatically cleaned by compressed air to remove any interfering particles. The sensors were additionally cleaned manually on a weekly basis using a specific cleaning agent recommended by the manufacturer to reduce biofouling on the measurement window and by manually removing debris and sediment.

4.2.3 Calibration, quality assurance and analysis

4.2.3.1 Sediment-turbidity rating curve

We used a site-specific, *ex situ* incremental suspension calibration to convert long-term turbidity records into an estimate of instantaneous suspended sediment concentrations (mg L^{-1}). A river water-sediment suspension with 16 to 18 concentration increments was established to simulate changing stream water suspended sediment concentrations occurring from low flow conditions (minimum 0 mg L^{-1}) to storm events (maximum 4,607 mg L^{-1}). The sediment suspension consisted of fine suspended sediment collected from sediment traps (time-integrated Phillips samplers) and fine soil material mixed with turbid river water collected during storm events. To ensure that only the clay size fraction remained in the suspension, the sediment suspension was decanted twice after the settling time for coarse particles (particle size >2 µm) had elapsed (Stokes law: 98 sec). The spectro::lyser probes from each monitoring station measured each concentration increment starting with river water representing low flow conditions (0 to 8 FTU). Small quantities of the synthetic sediment suspension were added at each concentration increment until the maximum measurable turbidity of 1,500 FTU was reached. The exact concentration was then determined gravimetrically from a 250 mL sub-sample at each increment. Total suspended sediment load was determined by multiplying suspended sediment concentration by discharge. Suspended sediment yield was calculated by integrating the sediment load over time and relating it to the catchment area. The sediment mass is reported in tonnes (t=megagrams) to conform with other published values.

4.2.3.2 Data quality assurance

Quality assurance of the turbidity, discharge and precipitation dataset was performed in two different ways. First, during equipment maintenance and manually downloading of the data any observed anomalities were recorded in a log book. Potential causes of anomalous values included (i) sensor above water level, (ii) turbidity sensor completely buried by deposited sediment during storm periods, (iii) biofilm or other phenomena on the measurement window due to malfunctioning of automatic cleaning with compressed air, (iv) measurement gaps due to incidents of power supply failure or (v) counting of number of tips by the rain gauges restricted by blocked funnel or spiderwebs. The readings for these periods were flagged with Not-a-Number (NaN).

After anomalous values were replaced by NaN, the median absolute deviation (MAD) was used to detect local outliers. The MAD has the following form:

$$MAD_i = b\ M_{i2}(|x_i - M_{i1}(x_i)|) \quad (1)$$

where x_i is the whole dataset, M_{i1} is the median of the dataset and M_{i2} is the median of the absolute deviation from the dataset from its median. The constant *b* estimates the standard deviation and was set to 1.4826 for normal distribution (Leys *et al.* 2013).

A moving window of *k* measurements around observation x_i at time t_i was used to detect local outliers with $x_j = (x_{i-k/2} \dots x_{i-1}, x_{i+1} \dots x_{i+k/2})$:

$$\frac{x_i - M_{j,i}}{MAD_{j,i}} > a \qquad (2)$$

where *a*=6 is the threshold for outlier selection, $M_{j,i}$ is the median and the $MAD_{j,i}$ is the MAD for x_j, while the moving window *k* was set to 16. Missing sediment data was interpolated using a linear function.

4.2.3.3 Data analysis

All data were tested for normality with the Shapiro-Wilk test. Significant differences were tested on suspended sediment, rainfall and discharge values among the different land uses using Kruskal-Wallis test for analyses of variances. To detect the significance of the effect of land use on the hydrological and sedimentological parameters, and within and among seasons on suspended sediment load the pairwise Wilcoxon rank sum test was used.

Five seasons, dry season, start of long rains, long rains (long rainy season), intermediate rains (season between the long and short rainy season) and short rains (short rainy season), were identified to calculate their contribution to annual suspended sediment yield (Jacobs *et al.* 2018b). The periods were chosen based on exceeding a threshold of monthly specific discharge for each catchment. The seasons for each year vary in length and timing due to variations in the onset of the rains and monthly streamflow (Figure 8).

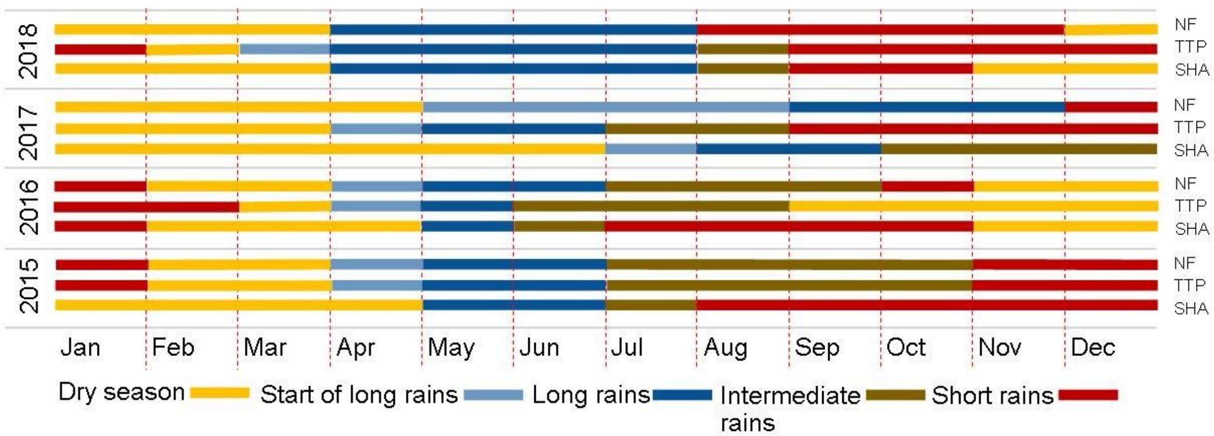

Figure 8 Timing of the five hydrological seasons for the natural forest (NF), tea-tree plantation (TTP) and smallholder agriculture (SHA) catchment during the observation period January 2015 to December 2018.

4.2.3.4 Modelling flow pathways

A linear continuous-time (CT) transfer function (TF) model with rainfall-runoff non-linearity was used to identify the dynamics that explain the response of water flow pathways to rainfall within a catchment by conceptualising a *Single-Input, Single-Output* (SISO) system (Young & Garnier, 2006). These types of models are equivalent to systems of linear differential equations and can be applied in numerous mass and energy transport as well as chemical or biochemical proccess applications, including flow and sediment delivery models (Chappell *et al.* 2006). The transfer function modelling process follows the *Data-Based Mechanistic* (DBM) modelling philosophy, searching through a range of model structures, ordering them according to statistical criteria, then retaining models that have a physical explanation (Young & Beven 1994). The DBM approach produces parsimonious models describing rainfall-runoff relationships, that include very few tuneable parameters (Lees 2000). Hourly time series of rainfall [mm hr^{-1}] was used as input, and the corresponding hourly time series of discharge [m^3 sec^{-1}] as output (the units conversion is absorbed into the model coefficients, in case of Eq. 6 – into coefficient b_0). These parameters have hydrological interpretation, describing hydrological pathways, so while these are not strictly hydrological models derived from the process models, they still apply to hydrological systems (Beven 2012).

Hydrological processes are known to be non-linear (Beven 2012), with the effectiveness of rainfall (amount of rainfall converted into discharge) dependent upon the state of saturation of the catchment. Therefore, the rainfall-runoff non-linearity is modelled using the Hammerstein model structure, with the input (rainfall) transformed using a non-linear function into what is termed ‘effective rainfall’, which then drives the linear dynamics of the transport process model. This study uses a power law relationship between measured rainfall and effective rainfall as surrogate for soil moisture to translate rainfall to effective rainfall (see text S1 for more details). Effective rainfall is the dynamically changing proportion of rainfall representing the volume of streamflow generated after soil moisture storage is deducted from the total rainfall (Beven 2012). The RIVCBJ (Refined Instrumental Variable Continuous Time Box-Jenkins Identification, for continuous models, Young and Garnier, 2006) algorithm was used to estimate model parameters. RIVCBJ is a component of the CAPTAIN toolbox which runs within MATLAB® (Taylor *et al.* 2007). The linear CT transfer function model has the following form:

$$Y(s) = \frac{B(s)}{A(s)} U(s) e^{-s\tau} + E(s) \qquad (3)$$

$A(s)$ and $B(s)$ characterise the dynamic relationship between the input and the output signals in the Laplace operator domain. The Laplace operator is the Laplace frequency domain equivalent of the time derivative operator $s \sim \frac{d}{dt}$. Functions $A(s)$ and $B(s)$ are constructed as polynomials in the *s* domain as follows, with *m* and *n* being the respective orders of the numerator and denominator polynomials:

$$A(s) = s^n + a_1 s^{n-1} + \cdots + a_n s^0 \tag{4}$$

$$B(s) = b_0 s^{m-1} + b_1 s^{m-2} + \cdots + b_m s^0 \tag{5}$$

In Eq. (3) $Y(s)$ denotes the Laplace transform of the output signal as hourly streamflow [m^3 sec^{-1}], $U(s)$ is the Laplace transform of the input signal hourly rainfall [mm hr^{-1}], $E(s)$ are the model residuals and $e^{-s\tau}$ is the Laplace transform of time delay τ representing the pure time delay (as opposed to the dynamic lag resulting from the system's dynamics) in time units between the input and output signals.

In this study, up to third order models were tested for all the sites and model fit was evaluated according to the coefficient of determination (R_t^2) (also known as the Nash-Sutcliffe efficiency) (see text S2) and the Young Identification Criterion (YIC) (see text S3 and S4). First order transfer function models were selected for the three catchments, where each system has a different depletion time, determined by its time constant. A first order continuous time transfer function model is written as:

$$Y = \left(\frac{b_0}{s+a_1}\right) e^{-s\tau} U = \left(\frac{SSG}{sTC+1}\right) e^{-s\tau} U \tag{6}$$

where $1/a_1$ is the time constant and the parameter b_0/a_1 represent the Steady State Gain (SSG) of the hypothetical pathway of rainfall through the catchment with a and b as the dynamic response characteristics. The time constant (TC) reflects the response between the input (rainfall) and the output (runoff or streamflow) (Young & Garnier 2006). Linear CT transfer function models were identified for each year between 2015 and 2018 for all three catchments.

4.2.3.5 Sediment response to hydrological variables

We used the cross-correlation function (CCF) to identify the statistical correlation between two sets of time series at different time lags (Lee *et al.* 2006; Mayaud *et al.* 2014). Rainfall and discharge time series were cross-correlated with the suspended sediment concentration time series. The peak response time between either precipitation or discharge to sediment concentrations was calculated as the delay time in time lags together with its correlation strength between these variables. Cross-correlation functions were calculated as:

$$CCC(\tau) = \frac{\frac{1}{n}\sum(x_i - \bar{x})(y_{i-\tau} - \bar{y})}{\sigma_x \sigma_y} \quad (7)$$

where $CCC(\tau)$ is the cross-correlation coefficient at time lag τ, $\tau = 0, \pm 1 \pm 2 \ldots \pm m$ between the two time series (sampled every 10 minutes), where x_i is observed rainfall or positive derivative of discharge at sample number i and $y_{i-\tau}$ is the suspended sediment concentration at sample number $i - \tau$, $\bar{x}$ is the mean rainfall or positive derivative of discharge, $\bar{y}$ is the mean suspended sediment concentration, σ_x is the standard deviation of rainfall or estimated positive derivative of discharge and σ_y is the standard deviation of suspended sediment and n is the number of data points. At the 95%-confidence interval, lag-time correlations are significant when $CCC(\tau)$ exceeds the standard error of $2/\sqrt{N}$, where N is the length of the dataset (Diggle 1990). The positive derivative of discharge, i.e. the estimated rate of change on the rising limb of the hydrograph, was selected because the main sediment pulses are mostly generated during the rising limb (Alexandrov *et al.* 2003; De Girolamo *et al.* 2015). A similar derivative effect has been observed in dynamic sediment load models by Walsh et al. (2011). The CCF analysis was carried out for each year between 2015 and 2018 for all three catchments.

4.3 Results

4.3.1 Hydrological response of the three catchments

Mean annual rainfall for the study period was 1,842, 1,730 and 1,554 mm yr^{-1} with maximum hourly rainfall over the whole observation period of 37.4, 33.1 and 27.5 mm hr^{-1} for the natural forest, tea-tree plantations and smallholder agriculture catchments, respectively. The wettest year for the smallholder agriculture catchment was 2018 with 1,823 mm yr^{-1} of rainfall, while for the natural forest and tea-tree plantations precipitation was highest in 2015 with 1,986 and 1,928 mm yr^{-1}, respectively. The annual mean specific discharge was 632±157, 610±153 and 621±224 mm yr^{-1} for the natural forest, tea-tree plantations and smallholder agriculture catchments, respectively. The catchment runoff coefficient was similar for the natural forest and the tea-tree plantations with a mean of 0.34 and 0.35, respectively, and 0.39 for the smallholder agriculture (Table 2).

Table 2 Hydrological characteristics and total suspended sediment (and 95%-confidence interval) for the three catchments under different land use natural forest (NF; 35.9 km^2), tea-tree plantations (TTP; 33.3 km^2) and smallholder agriculture (SHA; 35.9 km^2) in the South-West Mau, Kenya. Different capital letters indicate significant differences between the different land uses (p<0.05).

Site	Year	Annual rainfall (mm yr^{-1})	Annual specific discharge (mm yr^{-1})	Runoff coefficient [a]	Total suspended sediment load (t yr^{-1})	Total suspended sediment yield (t km^{-2} yr^{-1})
NF	2015	1,986	714 (693-738)	0.36 (0.35-0.37)	407 (378-439)	11.3 (10.5-12.2)
	2016	1,846	518 (497-542)	0.28 (0.27-0.29)	667 (615-724)	18.6 (17.1-20.2)
	2017	1,783	483 (466-502)	0.27 (0.26-0.28)	673 (622-729)	18.7 (17.3-20.3)
	2018	1,755	812 (783-844)	0.46 (0.45-0.48)	1,337 (1,228-1,457)	37.2 (34.2-40.6)
	Mean	1,842 A	632 (610-656) A	0.34 (0.33-0.36) A	771 (711-837) A	21.5 (19.8-23.2) A
TTP	2015	1,928	768 (730-820)	0.40 (0.38-0.43)	2,376 (2,185-2,603)	71.4 (65.6-78.2)
	2016	1,655	593 (555-642)	0.36 (0.34-0.39)	1,397 (1,277-1,539)	42.0 (38.4-46.2)
	2017	1,478	408 (372-468)	0.28 (0.25-0.32)	780 (701-880)	23.4 (21.0-26.4)
	2018	1,858	673 (634-728)	0.36 (0.34-0.39)	1,042 (952-1,151)	31.3 (28.6-34.5)
	Mean	1,730 A	610 (573-665) A	0.35 (0.33-0.38) A	1,399 (1,279-1,543) A	42.0 (38.4-46.3) A
SHA	2015	1,607	561 (539-582)	0.35 (0.34-0.36)	2,324 (2,161-2,494)	85.4 (79.5-91.7)
	2016	1,369	479 (456-503)	0.35 (0.33-0.37)	2,271 (2,088-2,464)	83.5 (76.8-90.6)
	2017	1,416	492 (471-514)	0.35 (0.33-0.36)	2,440 (2,237-2,653)	89.7 (82.2-97.5)
	2018	1,823	953 (920-986)	0.52 (0.50-0.54)	7,273 (6,774-7,790)	267.4 (249.1-286.4)
	Mean	1,554 A	621 (596-646) A	0.39 (0.38-0.41) A	3,577 (3,315-3,851) A	131.5 (121.9-141.6) B

[a]Annual specific discharge as proportion of annual rainfall

Discharge in all catchments was flashy and varied seasonally. Rising limbs were generally steep and had variable falling limbs depending on event size. The highest discharge peaks were measured during the long rainy seasons between April and July in 2015, 2016 and 2018. In contrast, 2017 was the driest year with a late onset of the rains and the highest discharge peaks between August and November for the smallholder agriculture and the natural forest catchments. However, in the tea-tree plantation catchment the rains started in May lasting until November 2017, resulting in discharge peaking in May and September 2017. High discharges were also recorded in January 2016 because the 2015 rains continued through November and December and were followed by an unusually wet January (Figure 9).

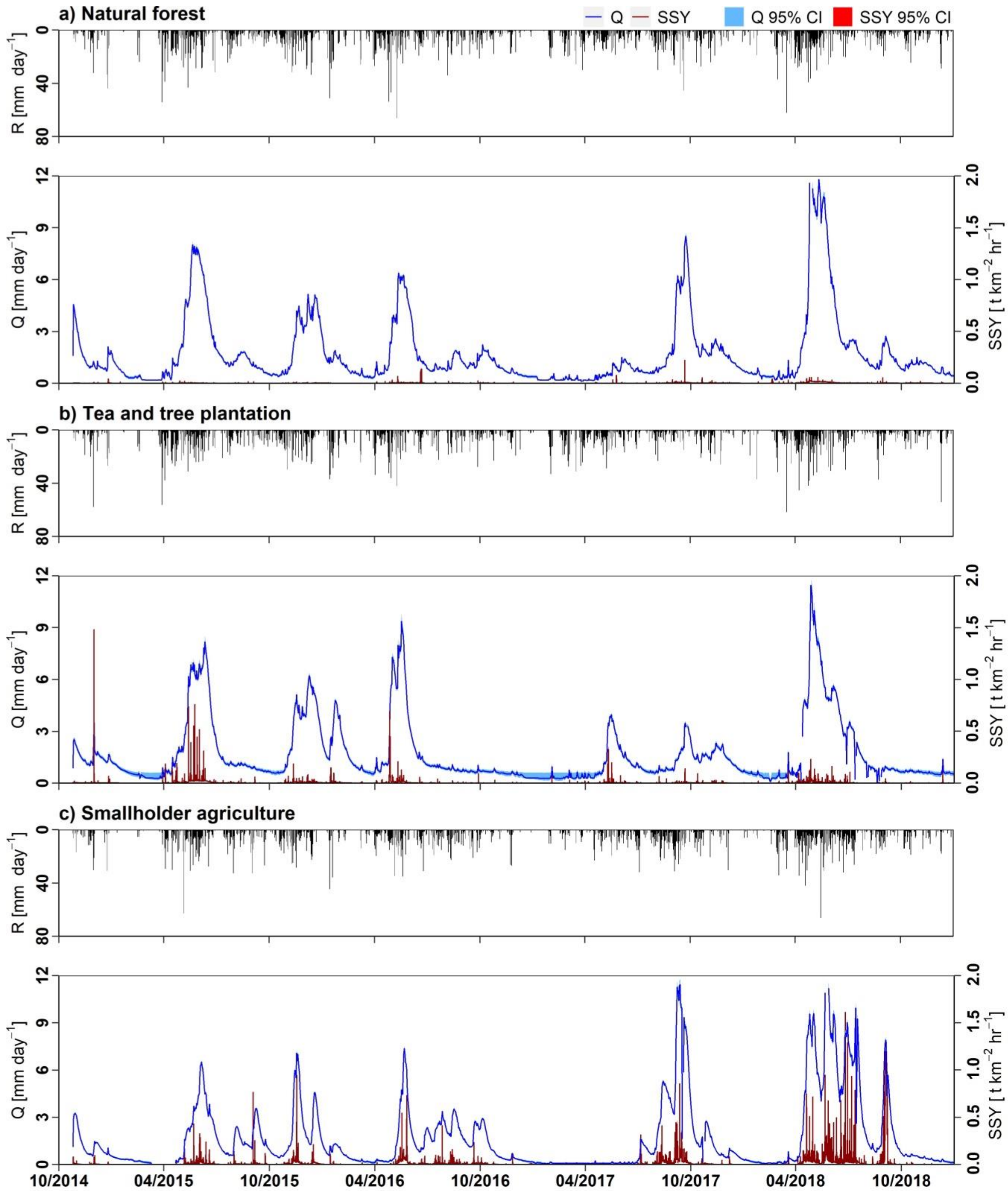

Figure 9 Time series of daily accumulated rainfall (R) [mm day^{-1}], daily specific discharge (Q) [mm day^{-1}] and hourly suspended sediment yield (SSY) [t km^{-2} hr^{-1}] aggregated from 10 minute resolution with 95%-confidence interval of the a) natural forest, b) smallholder agriculture and c) tea-tree plantation catchments in the South-West Mau, Kenya, between October 2014 and December 2018.

4.3.2 Relationship between turbidity and suspended sediment concentration

We obtained one rating curve for all three catchments to predict suspended sediment concentration from the measured turbidity values. A linear model provided the best fit between the *in situ* turbidity and suspended sediment concentrations, and there was no significant difference between slopes for each site-specific calibration (p-value>0.1). The intercept of the linear model was forced through the origin to prevent negative sediment concentrations at low turbidity, yielding an equation of the form TSS = 2.4*turbidity (R^2=0.98, p-value<0.001, n=50; Figure 10). This equation was used to convert the turbidity data to suspended sediment concentrations.

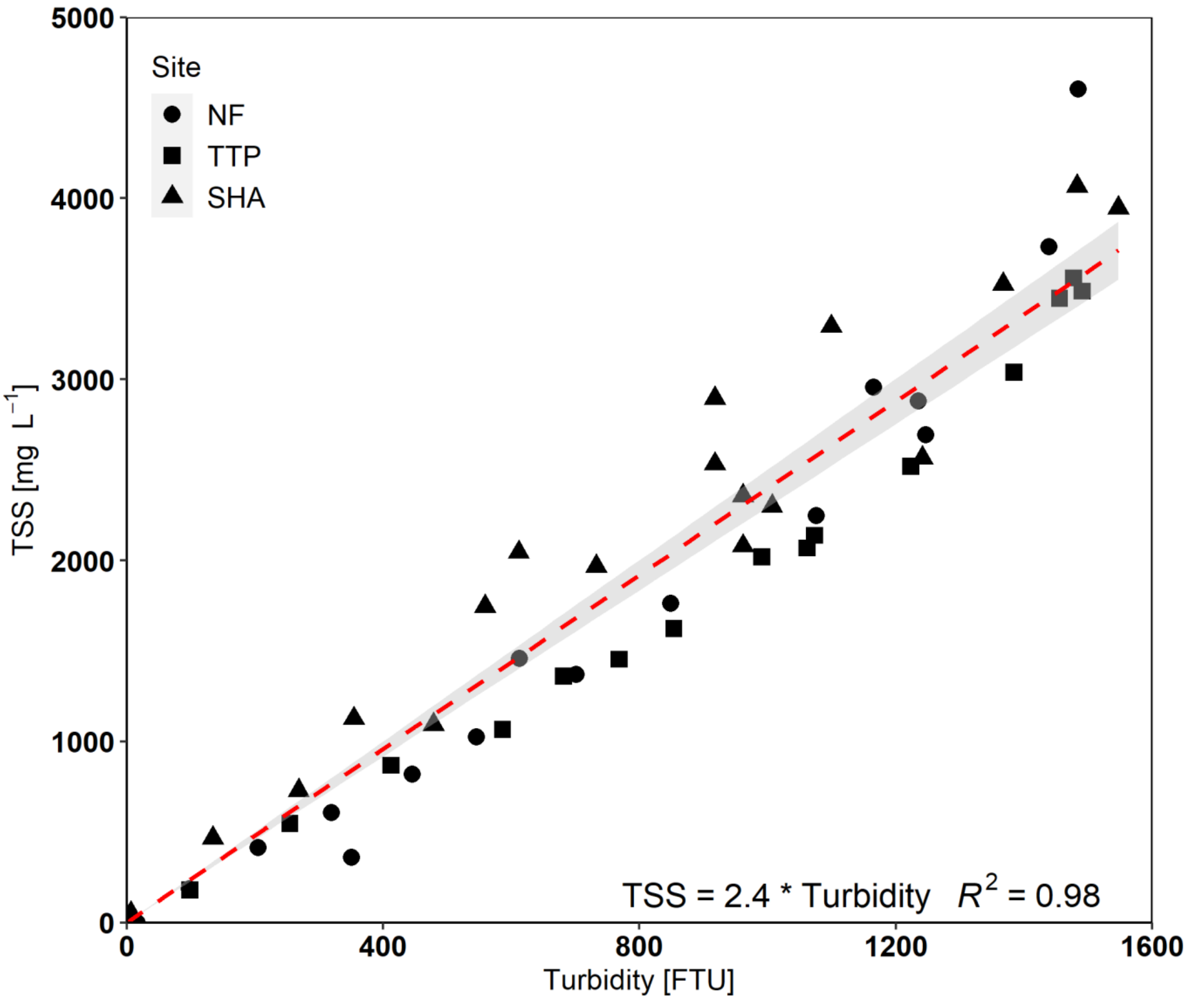

Figure 10 Relation between total suspended sediment concentrations (TSS) [mg l^{-1}] and turbidity [FTU=Formazin Turbidity Unit] measurements for three catchments: natural forest (NF), tea-tree plantations (TTP) and smallholder agriculture (SHA) and the fitted linear model (grey shaded area: 95%-confidence interval).

4.3.3 Suspended sediment dynamics

Sediment yield for the natural forest was lower than for the other two catchments (Figure 9). The sedigraph of the natural forest had smaller peaks with a short event time, whereas the smallholder agriculture and tea-tree plantations showed a steep increase followed by a flat recession with a long event time. The sedigraph of the smallholder agriculture catchment showed a flashy sediment response to rainfall and increased discharge. The maximum sediment yield in the natural forest catchment was 0.2 t km^{-2} hr^{-1}, followed by the tea-tree plantations with 1.5 t km^{-2} hr^{-1} and the smallholder agriculture with a maximum sediment peak of 1.6 t km^{-2} hr^{-1} (Figure 9). The mean annual suspended sediment yield was significantly higher for the smallholder agriculture catchment (131.5±90.6 t km^{-2} yr^{-1}) than for the tea-tree plantations (42.0±21.0 t km^{-2} yr^{-1}) and the natural forest (21.5±11.1 t km^{-2} yr^{-1}) ($p<0.05$) (Table 2). The lowest mean suspended sediment concentration (33.8±73.8 mg l^{-1}) was also lowest for the natural forest catchment, followed by the tea-tree plantations (47.4±90.7 mg l^{-1}). Concentrations were three to four times higher at the outlet of the smallholder agriculture catchment (128.6±233.4 mg l^{-1}) than at the other catchments ($p=0.04$). The daily mean suspended sediment load from the natural forest was the lowest, followed by the tea-tree plantations and the smallholder agriculture (2.1, 3.9 and 10.0 t day^{-1}, respectively). The total suspended sediment load for the entire study period (2015-2018) for the smallholder agriculture (14,308 t) was four times higher than that of the natural forest (3,083 t), whereas sediment load for the tea-tree plantations (5,595 t) was only twice that of the natural forest. These loads represent a mean of 771, 1,399 and 3,577 t yr^{-1} for the natural forest, tea-tree plantations and smallholder agriculture, respectively. Suspended sediment yield increased from 2015 to 2018 in the natural forest and smallholder agriculture catchments. In the tea-tree plantations, a similar sediment and rainfall pattern was observed with a decline from 2015 to 2017 and then an increase again in 2018 (Table 2). The natural forest had the longest period of missing data lasting for 95 days in November 2015 to February 2016, followed by a shorter gap in the smallholder agriculture of 50 days from March to April 2015 and the tea-tree plantations had the shortest period of missing sediment data of 13 days between March and April 2017. Besides these periods, minor gaps were usually of less than 24 hours with a total of missing sediment data of 7% for the natural forest, 2% for the tea-tree plantations and 4% for the smallholder agriculture catchments between 2015-2018.

4.3.4 Seasonal variations in suspended sediment

During the study period, suspended sediment yield showed pronounced seasonal variability, with most sediment being transported during the long rains in all catchments, and the highest monthly yields being recorded for the smallholder agriculture catchment (Figure 11).

Overall, more than half of the sediment yield (45-52%) was attributed to the long rains, which cover less than one third of the year. The sediment contribution during the long rains, intermediate rains and short rains in the smallholder agriculture was significantly greater than in the natural forest and tea-tree plantations ($p<0.05$). For the natural forest the streams carried significantly more material during the long rains (mean yield of 4.3±1.8 t km^{-2} $month^{-1}$) than during any other season. In the tea-tree plantations and smallholder agriculture, the sediment yield for the long rains (mean 10.9±6.9 and 30.7±10.6 t km^{-2} $month^{-1}$, respectively) differed significantly from the dry season (mean 0.4±0.2 and 0.6±0.3 t km^{-2} $month^{-1}$, respectively), the intermediate rains (mean 1.1±0.6 and 5.4±3.3 t km^{-2} $month^{-1}$, respectively) and the short rains (mean 3.2±2.2 and 23.9±26.1 t km^{-2} $month^{-1}$, respectively), while there was no difference between the sediment yield for the start of the long rains (mean 5.7±6.5 and 7.8 t km^{-2} $month^{-1}$, respectively) and the long rains ($p<0.05$).

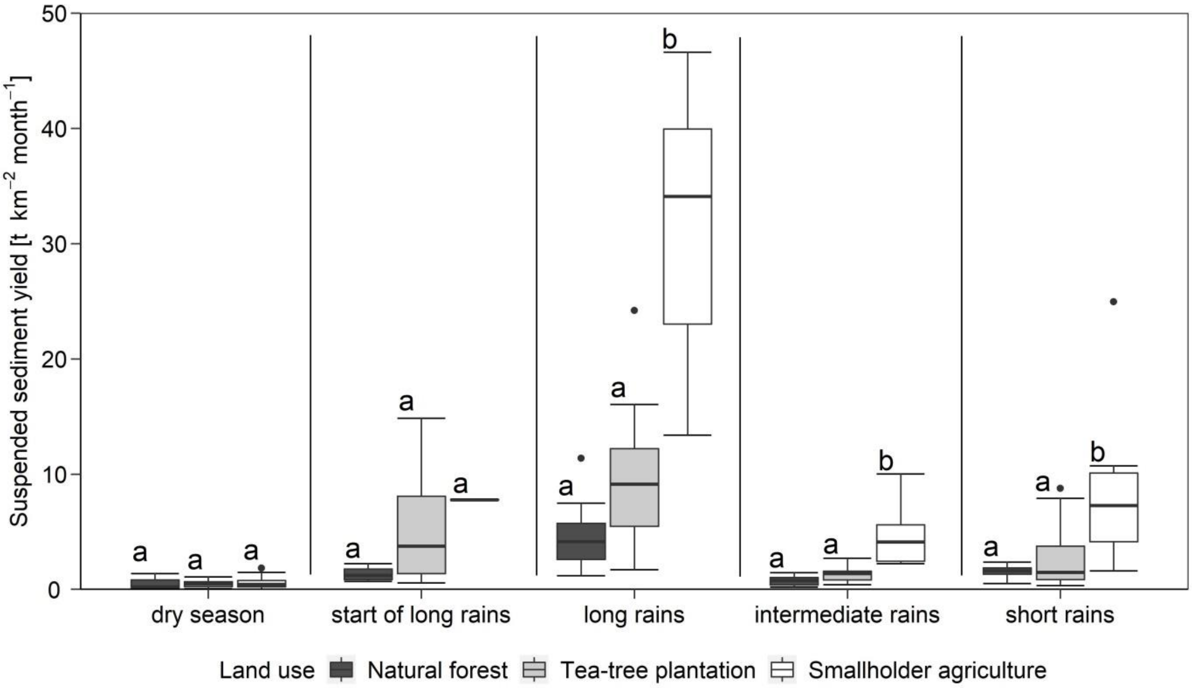

Figure 11 Boxplots of the monthly total suspended sediment yield [t km^{-2} $month^{-1}$] for different seasons for the natural forest (NF), tea-tree plantations (TTPs) and smallholder agriculture (SHA) catchments in the South-West Mau, Kenya. Seasons: dry season, start of the long rainy season, long rainy season, intermediate rainy season and short rainy season between January 2015 and December 2018. Different letters above the box plot indicate significant differences between the land uses within each season ($p<0.05$).

4.3.5 Flow pathways and streamflow dynamics

We compared hydrological flow pathways for each catchment as these are key in delivering sediments to the streams. Rainfall-runoff response was modelled over a continuous period of one year for each monitoring year 2015-2018 (Figure 13). In all three catchments, first order linear models (Eq. 6) were selected because these had the highest coefficients of determination (R_t^2) ranging between 86 and 93% and explained the data with the most negative YIC ranging between -12.5 and -10.4. The tea-tree plantations had a lower model performance in 2017 compared to the other years with a R_t^2 value of 54% and a YIC of -8.3, where the first order model was identified as the optimal model (Table 3). A simple first order model was used to derive the time constant to compare the dynamic relationship between rainfall and runoff response among the three catchments. The interpretation of the continuous time transfer function model suggests a slower rainfall-runoff catchment response in the natural forest and tea-tree plantation catchments in contrast to the fast flow response to rainfall in the smallholder agriculture catchment. The time constants calculated ranged from 8.4 to 11.5 days in the natural forest catchment and from 9.6 to 13.0 days in the tea-tree plantations. The smallholder agriculture had time constants between 6.2 to 8.6 days (Table 3).

Table 3 Summary of the linear continuous-time transfer function models for the natural forest, tea-tree plantation and smallholder agriculture catchment in the South-West Mau, Kenya for four years (study period 2015-2018). YIC=Young Identification Criterion, R_t^2=coefficient of determination, model structure [*n*=denominator polynomial, *m*=numerator polynomial, τ=pure time delay].

Site	Year	Time constant [days]	YIC	R_t^2	Model structure [*n*, *m*, τ]
Natural forest	2015	11.5	-11.2	0.92	[1 1 2]
	2016	12.6	-11.1	0.92	[1 1 2]
	2017	8.4	-11.2	0.92	[1 1 2]
	2018	9.8	-12.1	0.93	[1 1 2]
Tea-tree plantations	2015	12.1	-10.4	0.86	[1 1 2]
	2016	9.6	-11.5	0.90	[1 1 2]
	2017	13.0	-8.3	0.54	[1 1 2]
	2018	10.5	-11.4	0.92	[1 1 2]
Smallholder agriculture	2015[a]	8.6	-10.7	0.86	[1 1 2]
	2016	8.7	-11.3	0.88	[1 1 3]
	2017	6.2	-12.5	0.96	[1 1 2]
	2018	6.9	-11.4	0.92	[1 1 2]

[a]Data used in the analysis 1May-31Dec2015

4.3.6 Time lags between rainfall and discharge to sediment concentrations

The analysis using cross-correlation functions (CCF) (Eq. 7) showed statistically significant correlations between rainfall and discharge to suspended sediment exceeding the 95%-confidence interval of 0.001 for a sample size of 61,201 in all three catchments for the four years. The CCF indicates the impulse response time between the peak of rainfall and discharge to the suspended sediment peak (Figure 14). The natural forest had almost instantaneous (<10 minutes) to rapid responses (1.5 hrs) in suspended sediment to both rainfall and discharge in all four years. A period shorter than a year was used for the natural forest in 2015 and 2016 because of missing sediment data (Table 4). The rainfall to sediment cross-correlogram in the tea-tree plantations differed from the other two catchments, with a first fast peak within one hour followed by a delayed second peak after 5 to 8.5 hours. The discharge to sediment response in the tea-tree plantations was similar to that of the smallholder agriculture catchment with a time lag of around two to three hours. The smallholder agriculture had variable time lags between either rainfall or discharge to sediment ranging between 1.5 to 3.8 hours. The impulse response time between the peaks of discharge and sediment concentration was in general longer compared to the rainfall peak (Table 4).

Table 4 Summary of cross-correlation functions (CCF) between rainfall/ positive derivative of discharge to suspended sediment with time lag (in hours) and peak cross-correlation coefficients reported for the natural forest, tea-tree plantation and smallholder agriculture catchment in the South-West Mau, Kenya over four years (study period 2015-2018).

Site	Year	Discharge to sediment		Rainfall to sediment	
		Time lag [hours]	**CCF coefficient**	**Time lag [hours]**	**CCF coefficient**
Natural forest	2015[a]	1.0	0.10	1.5	0.16
	2016[a]	<0.2	0.03	<0.2	0.05
	2017	0.5	0.07	<0.2	0.12
	2018	0.5	0.10	0.2	0.03
Tea-tree plantations[b]	2015	2.5	0.34	1st 0.5 and 2nd 6.5	1st 0.08 and 2nd 0.17
	2016	2.3	0.18	1st 1.0 and 2nd 7.0	1st 0.08 and 2nd 0.16
	2017	3.2	0.32	1st 0.8 and 2nd 8.5	1st 0.08 and 2nd 0.14
	2018	1st 0.5 and 2nd 2.0	1st 0.17 and 2nd 0.17	1st 0.3 and 2nd 5.0	1st 0.18 and 2nd 0.15
Smallholder agriculture	2015	2.8	0.27	0.7	0.13
	2016	2.2	0.18	1.5	0.08
	2017	3.8	0.17	1.5	0.11
	2018	2.2	0.21	3.5	0.15

[a]Data used in the analysis 1Jan-22Nov2015 and 1Mar-31Dec2016
[b]1st and 2nd: identifies the first and second time lag and CCF coefficient of the CCF in the tea-tree plantations

4.4 Discussion

4.4.1 Suspended sediment dynamics

This study shows that the annual suspended sediment yield is around six times greater for the drier smallholder agriculture catchment, and twice greater in the tea-tree plantation catchment compared to the wetter natural forest catchment. The sediment yield difference is likely the result of more vegetation cover and low structural and process-based connectivity in the natural forest, where consequently erosion processes (including mass wasting) are not as active. Similar findings were reported for the neighbouring Mara River Basin, where a semi-arid catchment had higher suspended sediment yields (44 t km^{-2} yr^{-1}) with half of the annual rainfall in contrast to a wetter, but less populated and less disturbed catchment (33 t km^{-2} yr^{-1}) (Dutton *et al.* 2018) (Table 5).

Table 5 Overview of different studies reporting annual suspended sediment yields (SSY) in Kenyan headwater streams and tropical montane forest catchments worldwide derived from gauging station measurements.

Montane catchment	Area (km^2)	Land use	Study period (year)	Annual rainfall ($mm\ yr^{-1}$)	SSY ($t\ km^{-2}\ yr^{-1}$)	Reference
Sub-catchments of Sondu Basin (West Kenya)	35.9	Natural forest	2014-2018	1,842	21	This study
	33.3	Tea-tree (Eucalyptus) plantations	2014-2018	1,730	42	
	27.2	Smallholder agriculture	2014-2018	1,554	131	
Different catchments throughout southern half of Kenya	n.a.	Natural forest (100%)	1948-1968	n.a.	20-30	Dunne, 1979
	n.a.	Natural forest (>51%)	1948-1968	n.a.	10-100	
	n.a.	Agricultural land (>50%)	1948-1968	n.a.	10-1,500	
	n.a.	Grazing land	1948-1968	n.a.	5,000-20,000	
Athi (Kenya)	510	Agriculture, grazing land and settlements	1985	n.a.	109	Kithiia, 1997
Upper Mara-Emarti (South-West Kenya)	2,450	Smallholder agriculture and urban development	2011-2014 (3-4 months)	1,400	33	Dutton et al., 2018
Middle Mara-Talek (South-West Kenya)	4,050	Grazing land	2011-2014 (3-4 months)	600	44	Dutton et al., 2018
Ruharo (Uganda)	2,121	Grazing land, agriculture, <20% Papyrus wetlands	2009-2010	1,535	106	Ryken et al., 2015
Koga (Uganda)	379	Grazing land, agriculture, >80% Papyrus wetlands	2009-2010	1,330	37	Ryken et al., 2015
Andit Tid (Ethiopia)	4.77	Agriculture (30%)	1989-1996	1,467	522	Guzman et al., 2013
Anjeni (Ethiopia)	1.13	Agriculture (80%)	1989-1996	1,675	2,470	
Maybar (Ethiopia)	1.12	Agriculture (60%)	1989-2001	1,417	740	
May Zegzeg (Ethiopia)	1.87	Agriculture and grazing land	2000	774	850	Nyssen et al., 2009
		Agriculture and grazing land with soil conservation practices	2006	708	190	
Arvorezinha (South Brazil)	1.23	Agriculture with traditional soil management & natural forest	2002-2003	2,051	298	Minella et al., 2018
		Agriculture with soil conservation practices & natural forest	2004-2008	1,655	68	
		Agriculture with traditional soil management & cultivated forest	2009-2016	2,102	163	
Conceição (South Brazil)	800	>85% agriculture (soybean, wheat, oats, ryegrass) with soil management, <15% gallery forest, wetlands and urban areas	2000-2010	1718	140	Didoné et al., 2014
			2011	1,422	242	
			2012	1,463	41	
Guaporé (South Brazil)	2,000	Agriculture (soybean, tobacco, maize, oats, ryegrass), grazing land and cultivated forest	2000-2010	1550	140	Didoné et al., 2014
			2011	1,195	390	
			2012	1,660	158	
Baru (Borneo)	0.44	Disturbed logged natural forest	1989	3,205	1,632	Douglas et al., 1993
		Natural forest immediately after logging	1991	2,609	1,017	Douglas et al., 1993
		Natural forest after logging	1995-1996	2,956	592	Chappell et al., 2004
W8S5 (Borneo)	1.7	Natural forest	1989	3,205	118	Douglas et al., 1993
			1991	2,609	117	
Mae Sa (Thailand)	74.2	Natural forest (62%) and agriculture (>20%)	2006-2008	1,743	323	Ziegler et al., 2014
Basper (Philippines)	0.32	Grassland and shrubs	2013	2,660	2,740	Zhang et al., 2018

n.a.=no data available

Missing sediment data and linear interpolation to fill these gaps could have increased the uncertainty in the sediment yields calculated. However, data gaps during dry periods, such as those in the sediment data for the natural forest during 2015 and 2016, would not have a large influence on yield calculations because of the small amount of material transported. Gaps during periods of heavy rainfall, which occasionally occurred in the smallholder agriculture catchment due to siltation, could have contributed to underestimation of the sediment yield, as peaks in the sediment concentration could be missing.

For Africa, suspended sediment yield was estimated to be 634 t km^{-2} yr^{-1} at continental scale based on 682 catchments and rivers with larger drainage areas (mean >1,000 km^2) (Vanmaercke *et al.* 2014). Compared to this, and other sediment studies in Kenya with varying catchment sizes of 24-42,000 km^2 and 8.2 to 6,330 t km^{-2} yr^{-1} (Dunne 1979; Vanmaercke *et al.* 2014), the annual suspended sediment yields (21-131 t km^{-2} yr^{-1}) are within the lower reported ranges. Suspended sediment yields from the tea-tree plantations and natural forest catchment of this study are comparable to those observed in the neighbouring upper Mara River Basin (33 t km^{-2} yr^{-1}; Dutton et al., 2018), which is dominated by small-scale farming and urban development. The smallholder agriculture catchment in this study had slightly higher suspended sediment yields than the Athi catchment in Kenya (109 t km^{-2} yr^{-1}; Kithiia, 1997) and had lower suspended sediment yields than disturbed agricultural catchments in montane headwaters such as the May Zegzeg catchment in Ethiopia (850 t km^{-2} yr^{-1}; Nyssen et al., 2009) or the Arvorezinha, Conceição and Guapore catchments in South Brazil (140-298 t km^{-2} yr^{-1}; Didoné et al., 2014; Minella et al., 2018). Guzman et al. (2013) found that in Ethiopia the highest suspended sediment yields (2,470 t km^{-2} yr^{-1}) in small catchments were in those with the largest proportion of agricultural land. Their reported annual yields were significantly higher than the suspended sediment yields observed in this study with similar annual rainfall. Annual suspended sediment yields of 117 up to 2,740 t km^{-2} yr^{-1} from undisturbed to highly disturbed forest or upland grassland catchments were measured in South-East Asia (Borneo, Thailand and the Philippines) subjected to mass wasting during typhoon or post-typhoon events (Douglas *et al.* 1993; Ziegler *et al.* 2014; Zhang *et al.* 2018) (Table 5).

4.4.2 Factors controlling sediment yield

Vegetation cover

The low annual suspended sediment yield measured in the Mau Forest shows that forest vegetation is the most effective surface cover to limit soil erosion despite the steepest slopes of the forested catchment (Table 1). The dense vegetation, diverse strata and complex rooting systems prevent soil detachment and trap potentially erodible material. Similarly, a dense perennial tea vegetation covers the soil surface in the tea-tree plantations, which can buffer erosive rainfall (Edwards & Blackie 1979). Nevertheless, the annual suspended sediment yield for the tea-tree plantation catchment was twice that of the natural forest despite the soil conservation practices applied by tea companies such as mulching, planting of buffer strips (oat grass) between rows of young tea bushes or cover trees on newly planted tea plots. This indicates that high sediment loads originate from unprotected bare surfaces during renovation of tea plantations or logging activities of woodlots. Logging activities trigger overland flow and erosion processes and lead sediments to the streams, when there are no buffer strips (Douglas *et al.* 1993; Chappell *et al.* 2004). The dense vegetation contrasts with the land management in the smallholder agriculture catchment, where steep slopes tend to be bare between crop harvest and the start of the next cropping season. During that period, bare surfaces are prone to soil erosion, although cropland surface erosion was not observed, which may be explained by the high infiltration rates previously measured on these croplands (401±211 mm hr^{-1}) (Owuor *et al.* 2018). The routing of main flow paths was observed on compacted gullied tracks which act as ephemeral channels during a storm event. Based on these observations, the hypothesis was set that rural unpaved tracks in the smallholder agriculture generate a larger contribution to the total sediment load than agricultural land, but further work is required to confirm this. A potential reason for the annual increase in suspended sediment yield in the smallholder agriculture and the natural forest catchments during the study period could be the reduced tree cover and increasing areas under annual crops and forest disturbance indicated by the study of Brandt et al. (2018).

Connectivity between sediment sources and the streams

The tea-tree plantations and the smallholder agriculture have a higher structural and process-based connectivity than the natural forest, which may be causing higher sediment transfer by connecting multiple source areas with the streams. Lateral linkages (tracks, gullies or drains) can be recognized as process-based concepts that connect sediment source areas at catchment-scale with the stream network (Lane & Richards 1997), which can be an important driving force for the total sediment load into the rivers (Sidle & Ziegler 2010).

In the smallholder agriculture catchment, unpaved tracks are the main pathways for people and livestock to access streams, thus being frequently used and heavily trafficked also by motorbikes (Figure 7). This activity generates highly compacted surfaces, where soil infiltration is impeded. Ziegler et al. (2001) observed that unpaved rural roads, similar in appearance to the unpaved tracks of this study, generate significantly more overland flow compared to adjacent hillslopes. As a consequence of low infiltration rates and downslope-orientated tracks, surface runoff energy increases generating more volume and velocity of flow that can transport large quantities of soil, eventually eroding tracks into gullies (Svoray & Markovitch 2009; Sidle & Ziegler 2010). Other researchers found a strong influence of subsurface water tables in valleys on gully formation and the development of large scale sediment mobilization (Tebebu *et al.* 2010; Zegeye *et al.* 2018). High sediment loads at the outlet of the smallholder agriculture catchment are thought to originate from the eroded unpaved tracks and its connecting adjacent source areas. In addition, a combination of a high structural and process-based connectivity might be the key process for the significant higher sediment yields. Catchment drainage density is higher in the smallholder agriculture (0.64 km km^{-2}) compared to the natural forest and tea-tree plantations (0.48 and 0.42 km km^{-2}, respectively) suggesting a link to increased erosion rates. The tea-tree plantation catchment is hydrologically connected through a network of tracks and well-engineered paved and unpaved drains in between the tea fields. The design of the well-engineered drains took into account the appropriate routing of surface runoff to the riparian zones before entering the streams, suggesting a higher hydrological connectivity than in the smallholder agriculture catchment. However, the drains are well-maintained with a densely forested riparian zone, which reduces sediment export, thereby reducing sediment transport connectivity. The strong hydrological connectivity of the tea-tree plantation catchment could lead to high sediment transport and loads with poor maintenance of the drainage network.

Riparian zones

Dense riparian vegetation can trap sediments before they reach the stream (Pavanelli & Cavazza 2010). An intact forested riparian zone in the natural forest and a riparian buffer of 30 m, as pre-described by the Kenya's Water Act (Republic of Kenya 2012), of mixed indigenous vegetation in the tea-tree plantations seem to be reducing sediment delivery to the streams by trapping eroded soil. In contrast, high sediment loads are expected in the smallholder agriculture catchment, where the riparian vegetation is highly degraded or replaced by crops or woodlots planted on the river banks. Small floodplains in a steep, narrow valley floor provide limited space for sediment storage. In the same river basin, other studies reported that highly degraded riparian zones adjacent to areas cultivated by smallholder agriculture lead to increased suspended sediment concentrations (Masese *et al.* 2012; Njue *et al.* 2016). In the smallholder agriculture catchment, livestock access the streams through the riparian area for watering (Figure 7d-f), which damages the riverbank and the riparian vegetation and further increases sediment supply.

4.4.3 Water pathways are key for sediment production

Hydrological pathways such as surface runoff or subsurface flow are key in determining sediment response in catchments. The analysis showed that the natural forest and tea-tree plantation catchments, with lower suspended sediment yields had the longest streamflow response time to rainfall using the CT transfer function model (Eq. 6). This supports the hypothesis that faster pathways indicate that surface runoff mobilizes soil particles causing six times more suspended sediment yields.

The number of pathways and their response time depend on catchment characteristics (Chappell *et al.* 2006; Ockenden & Chappell 2011). Forest ecosystems are generally characterized by complex catchment behaviour (Chappell *et al.* 1999), where soils with high infiltration rates promote infiltration to deeper subsurfaces. These pathways can be divided into shallow water pathway and deep groundwater pathway (Chappell & Franks 1996). The high infiltration rates (760±500 mm hr^{-1}, Owuor *et al.* 2018) and the long time constants derived through modelling for the natural forest catchment (Table 3) point to subsurface flow pathways. Jacobs et al. (2018a) reported the occurrence of shallow to deeper subsurface flow by using an endmember mixing analysis in the same natural forest catchment. Groundwater seemed to be an important stream water source (Jacobs *et al.* 2018a), which agrees with the calculated long response time (Table 3). The findings corroborate those of other studies in tropical forest catchments, which demonstrated that subsurface flow is the main water pathway of forest ecosystems (Noguchi *et al.* 1997; Boy *et al.* 2008; Muñoz-

Villers & McDonnell 2013). Consequently, low suspended sediment yields are associated with limited surface erosion and sediment delivery to the streams in the natural forest catchment.

The main pathways in the tea-tree plantations with slightly shorter time constants (Table 3) and almost half the infiltration rate (430±290 mm hr^{-1}, Owuor *et al.* 2018) compared to the natural forest suggest that shallow subsurface flow and surface runoff may dominate. Overland flow was observed to be routed through the well-engineered drainage network along roads and surface water drains between tea plantations to the well buffered fluvial network. Overland flow was also thought to be significant by Jacobs et al. (2018b), where nitrate concentrations in stream water seemed to be diluted by surface runoff. The well-maintained drains explain the lower suspended sediment yields, despite the prevalence of surface runoff observed in the tea-tree plantations.

The analysis of the smallholder agriculture catchment showed a relatively fast pathway. However, Owuor et al. (2018) measured infiltration rates of 401±211 mm hr^{-1} on croplands in the catchment, suggesting that subsurface flow is the most likely pathway. Nevertheless, field observations of runoff along with poorly maintained highly compacted tracks, and the shape of a classification of hysteresis loops by Jacobs et al. (2018b) provides a contrasting insight suggesting that surface runoff in the smallholder agriculture is an important vector for sediment. The hypothesis is that the tracks act as ephemeral streams and receive water from the surrounding areas as shallow lateral flow, thus explaining the shorter water response times. The natural forest and tea-tree plantation catchment with high tree cover showed similar time constants, whereas the much lower tree cover in the smallholder agriculture catchment lead to faster pathways. The generally slow pathway component in each model can be explained by the presence of deep and well-drained soils in all catchments (Sombroek *et al.* 1982).

4.4.4 Sediment response times and event duration

The shorter time lag between the peaks of rainfall to sediment than to discharge can be attributed to exposed and easy erodible material adjacent to the outlet. The almost instantaneous sediment response to rainfall in the natural forest and tea-tree plantations suggest sediment supply from readily available, nearby sediment sources (Francke *et al.* 2014; Tena *et al.* 2014; De Girolamo *et al.* 2015) (Figure 12). Near-channel or in-channel sediment sources can originate from the stream bank or the stream bed (Kronvang *et al.* 1997; Chappell *et al.* 1999; Lenzi & Marchi 2000). Temporarily stored sediment is thought to be mobilized very quickly during the first stages of a storm event (Eder *et al.* 2010). Fast response systems of short duration and low mass magnitude, as those observed in the natural forest catchment, are characterized by rapid sediment flushing and fast depletion of sediment supply (Chappell *et al.* 1999), due to limited availability of eroded material from the protected surface (Fang *et al.* 2008; Fan *et al.* 2012; De Girolamo *et al.* 2015). The paved drain network in the tea-tree plantation catchment can act as conduits transporting sediment instantaneously from nearby logged plantations or tracks to the stream. The delayed sediment response after a rainfall event in the smallholder agriculture and the second sediment response in the tea-tree plantation catchment suggest a long travel distance between the sediment source and the catchment outlet. The delayed response can be related to the magnitude of mass, where more distant sediment source areas from the wider catchment accumulate more mass over a longer period. These responses may also indicate the breaching of barriers such as hedges, fences or grazing land, especially in the smallholder agriculture catchment. The long recession limb in the smallholder agriculture and the tea-tree plantation catchments is typically explained as a slow depletion of sediment supply (Francke *et al.* 2014; Tena *et al.* 2014; De Girolamo *et al.* 2015). The more pronounced sedigraph in the smallholder agriculture catchment can be associated with the wide range of accumulated sediment source areas (Figure 12).

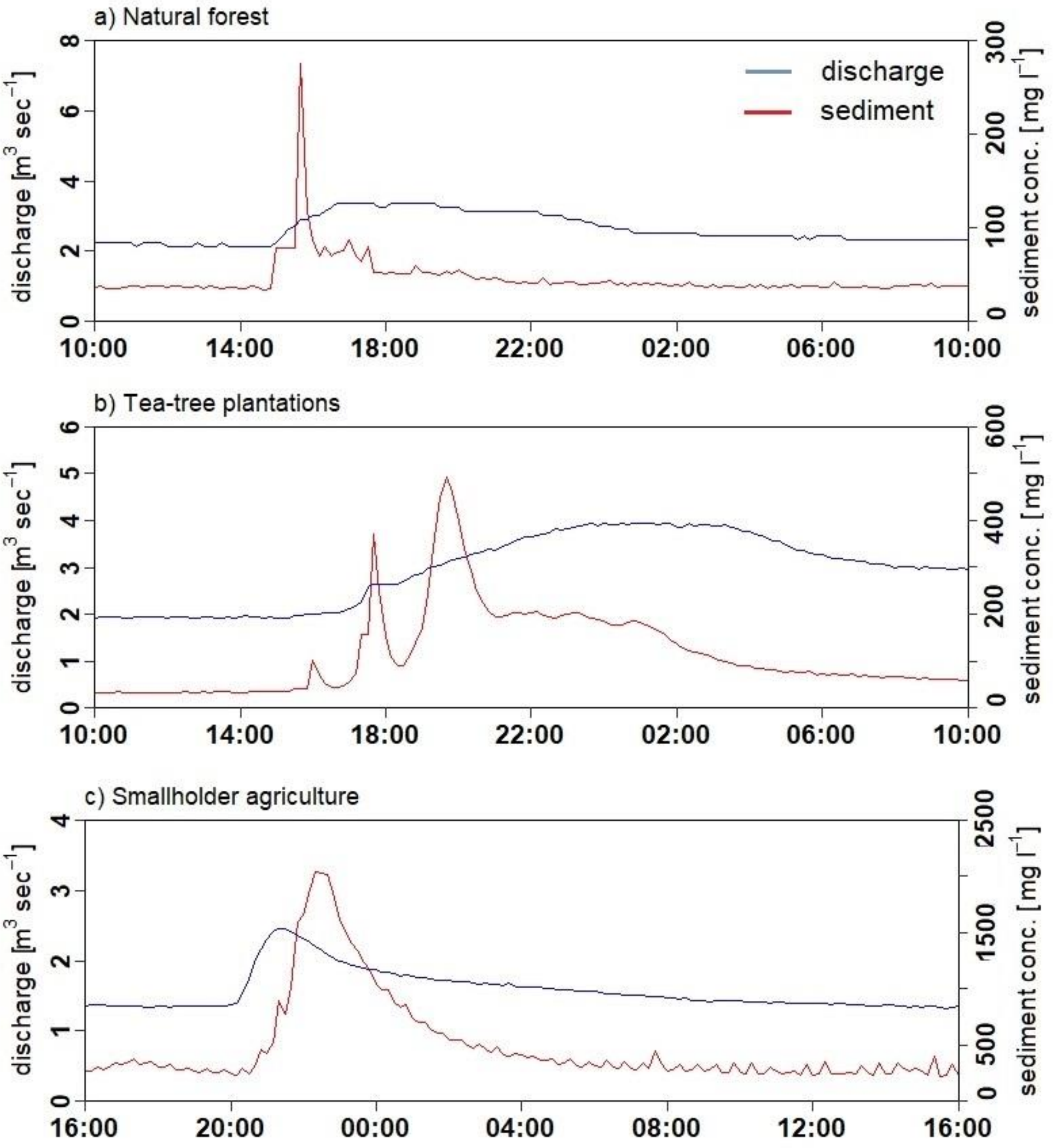

Figure 12 Typical shape of the sedigraph (sediment conc.=concentration [mg l^{-1}] and hydrograph (discharge [m^3 sec^{-1}] (10 minute resolution) for events in the natural forest (02/06-03/06/2018 10:00), the tea-tree plantation (24/04-25/04/2018 10:00) and the smallholder agriculture (17/05-18/05/2018 16:00) catchment in the South-West Mau, Kenya.

4.4.5 Seasonal variability of suspended sediment

Throughout the study period, distinct seasonal variability was observed in suspended sediment yield for the natural forest, tea-tree plantations and smallholder agriculture with higher yields in periods of high discharge and rainfall. In climates with strong seasonality of rainfall, seasons can explain sediment dynamics (Horowitz 2008; De Girolamo *et al.* 2015), as was observed in this study. Wet seasons or high-flow events generate the largest proportion (80-95%) of the annual sediment load, as observed by De Girolamo et al. (2015), Sun et al. (2016) and Vercruysse et al. (2017) in other areas, while the sediment load is the smallest in the dry season with less than 5% in this study. The seasonal differences were more pronounced in the smallholder agriculture catchment compared to the other two catchments explained by the high catchment surface connectivity and low vegetation cover.

4.5 Conclusions

This study presents the first long-term high-resolution sediment dataset of East Africa in Kenya. The four years of continuous data for the natural forest, tea-tree plantation and smallholder agriculture catchments provide critical insights in contrasting sediment dynamics of the tropical montane Mau Forest Complex. The analysis revealed that land use is a critically important driver for sediment supply, where smallholder agriculture generates six times more annual suspended sediment yield than a catchment dominated by natural forest. Besides vegetation cover, a strong catchment surface connectivity through unpaved tracks and gullies from hillslopes to the fluvial network is thought to be the main reason for the differences in sediment yields. However, further work is required to test this hypothesis. Catchments with a high tree cover, such as the natural forest and the tea-tree plantations seem to have similar water pathways with a dominance of subsurface flow. In contrast, in the highly disturbed landscape such as that of the smallholder agriculture catchment, surface runoff dominates and soil erosion increases suspended sediment yield. This superficial water pathway results in the more pronounced seasonal impact of rainfall in the smallholder agriculture compared to the other two catchments, also due to varying vegetation cover. Delayed sediment response to rainfall and a slow depletion in sediment supply in the smallholder agriculture and tea-tree plantations suggests that the wider catchment area is supplying sediment from a range of sediment sources, especially in the catchment dominated by smallholder farming. In contrast, the fast depletion in sediment supply in the natural forest suggests the importance of nearby sediment sources and temporarily stored sediment.

Land scarcity and population growth bring enormous pressure on natural forest ecosystems. Forest conversion will increase sediment production, which will affect, not only many people in the Sondu River Basin who rely on the rivers for drinking water, but also Lake Victoria which is already affected by increased sediment supply. The implementation of catchment management, such as soil conservation measures and better engineering of rural trackways is essential to reduce sediment supply to water bodies. However, a detailed sediment source fingerprinting analysis is necessary to identify the main contributing sediment sources. This will support the application of better management strategies at the source to prevent sediment entering the stream network. Sediment yields reported in other sediment studies in montane smallholder agriculture catchments were higher than in this study, which provide a warning of potentially higher sediment loss in the future unless mitigation strategies are implemented.

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4.6 Supporting information

This study uses a four year (2015-2018) high-temporal resolution dataset on rainfall, discharge and calibrated suspended sediment of a natural forest, tea-tree plantation and smallholder agriculture catchment in the South-West Mau, Kenya. Sediment dynamics and hydrological flow pathways were compared between the three catchments as these are key in delivering sediments to the streams. Therefore, rainfall-runoff was modelled using a linear continuous time (CT) transfer function model over a continuous period of one year for each monitoring period. Figure 13 exemplifies the output of the simulated discharge using the linear CT transfer function model for 2018. Cross-correlations were used to further assess the timing of the response of suspended sediment to rainfall and discharge. An example is given in Figure 14 between discharge and suspended sediment for 2018 for each catchment.

The supporting information provides the equation for the power law relationship between measured rainfall and effective rainfall as surrogate for soil moisture. Effective rainfall is used as input for the linear CT transfer function model. In addition, the coefficient of determination (akin to Nash Sutcliffe Efficiency) and the Young Information Criterion (YIC) are explained as the model assessment criteria for the linear CT transfer function model. The figures show an example of the output of the simulated discharge using the linear CT transfer function model within the CAPTAIN toolbox in MATLAB (Taylor *et al.* 2007) and cross-correlograms between discharge and suspended sediment for the period of 2018.

Text S1.

Effective rainfall is calculated with the power law which is given by:

$$\mathrm{R_e(t) = \ R_{obs}(t) * (Q_{obs}(t-1))^{\alpha}} \tag{S1}$$

where $\mathrm{R_e(t)}$ is the effective rainfall at time t, $\mathrm{R_{obs}}$ the observed rainfall, $\mathrm{Q_{obs}}$ is the observed runoff (discharge) with flow being used as the index of antecedent wetness, this process produces the power law parameter α that determines the fraction of the rainfall that generates runoff and is a constant exponent which is optimized from the observed data. $\mathrm{R_e}$ is the effectiveness of rainfall (amount of rainfall converted into discharge), to fit into the linear CT transfer function model.

Text S2.

Model assessment criteria based on the coefficient of determination:

Model fit was evaluated according to the coefficient of determination $\mathrm{R_t^2}$ (akin to Nash Sutcliffe Efficiency) and the Young Information Criterion (YIC) (Young & Beven 1994) contained in the Captain Toolbox for Matlab (Taylor *et al.* 2007):

$$\mathrm{R_t^2} = 1 - \frac{\sum_{\mathrm{i=1}}^{\mathrm{N}}(\mathrm{Q_m^i} - \mathrm{Q_o^i})^2}{\sum_{\mathrm{i=1}}^{\mathrm{N}}(\mathrm{Q_o^i} - \overline{\mathrm{Q_o}})^2} \tag{S2}$$

where $\mathrm{Q_o^i}$ is observed discharge, $\mathrm{Q_m^i}$ is modelled discharge at i. $\overline{\mathrm{Q_o}}$ is the mean of the observed discharge series. The coefficient of determination can range between $1 > \mathrm{R_t^2} > -\infty$, with 1 being a perfect predicted model.

Text S3-4.

A measure of parameterisation is tested with the Young Information Criterion (YIC):

$$\mathrm{YIC} = \ln\frac{\sigma_r^2}{\sigma_o^2} + \ln\{\mathrm{NEVN}\} \quad \text{(S3)}$$

where σ_r^2 and σ_o^2 are the variances of the residual series and observed series, respectively and NEVN (normalised error variance norm) is given by:

$$\mathrm{NEVN} = \frac{1}{np}\sum_{i=1}^{np}\frac{\sigma_r^2 p_{ii}}{a_i^2} \quad \text{(S4)}$$

where np is the number of estimated parameters, p_{ii} is the ith diagonal of the parameter covariance matrix and a_i^2 is the square of the ith parameter.

Generally, a large negative value indicates a good fit with the lowest amount of parameters necessary to represent the dynamics of the system. A higher order model shows a better fit, but the parameters have a greater uncertainty. The final model choice was based on a R_t^2 as high as possible, balanced by a large negative YIC.

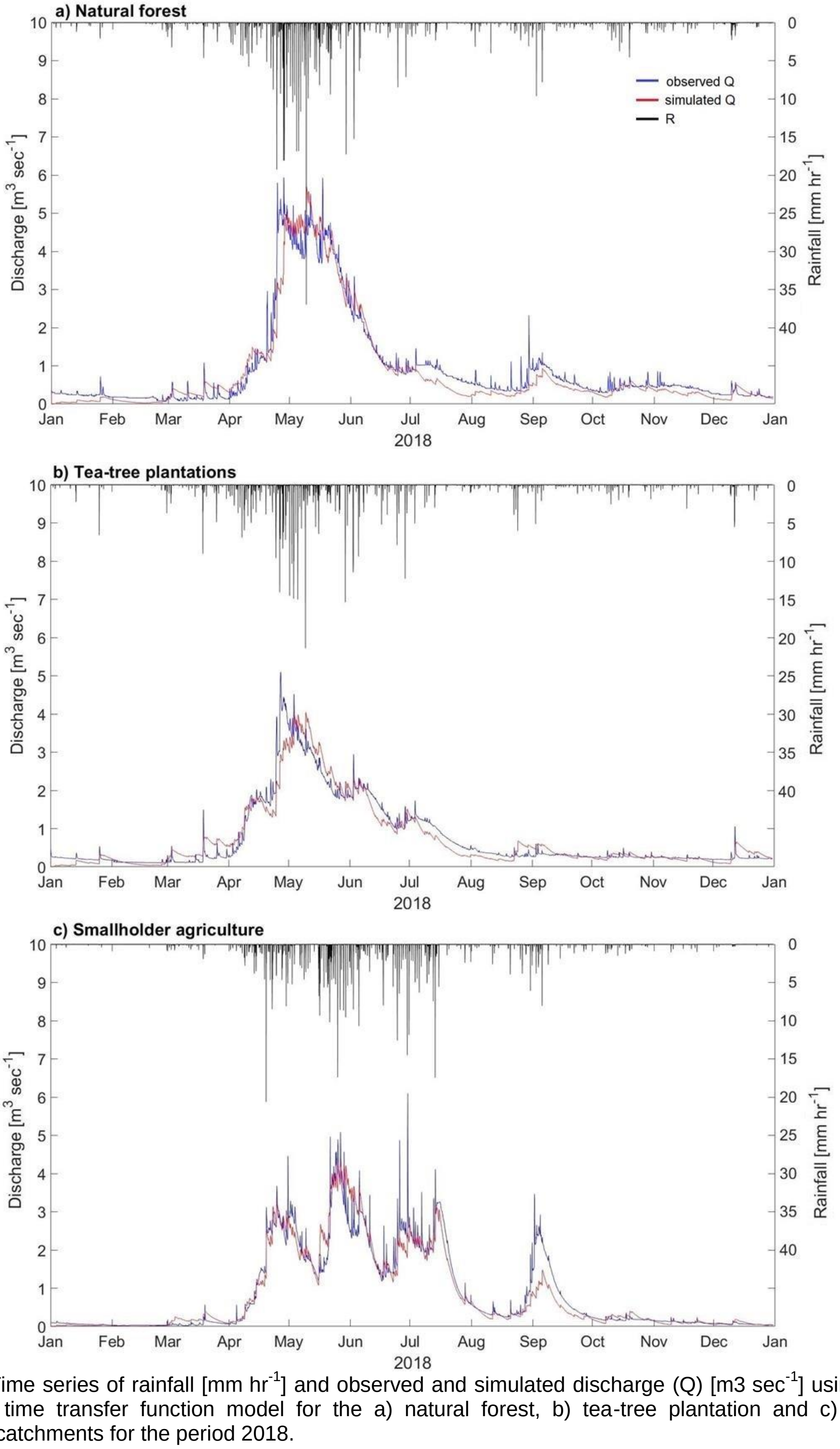


Figure 13 Time series of rainfall [mm hr^{-1}] and observed and simulated discharge (Q) [m3 sec^{-1}] using the linear continuous time transfer function model for the a) natural forest, b) tea-tree plantation and c) smallholder agriculture catchments for the period 2018.

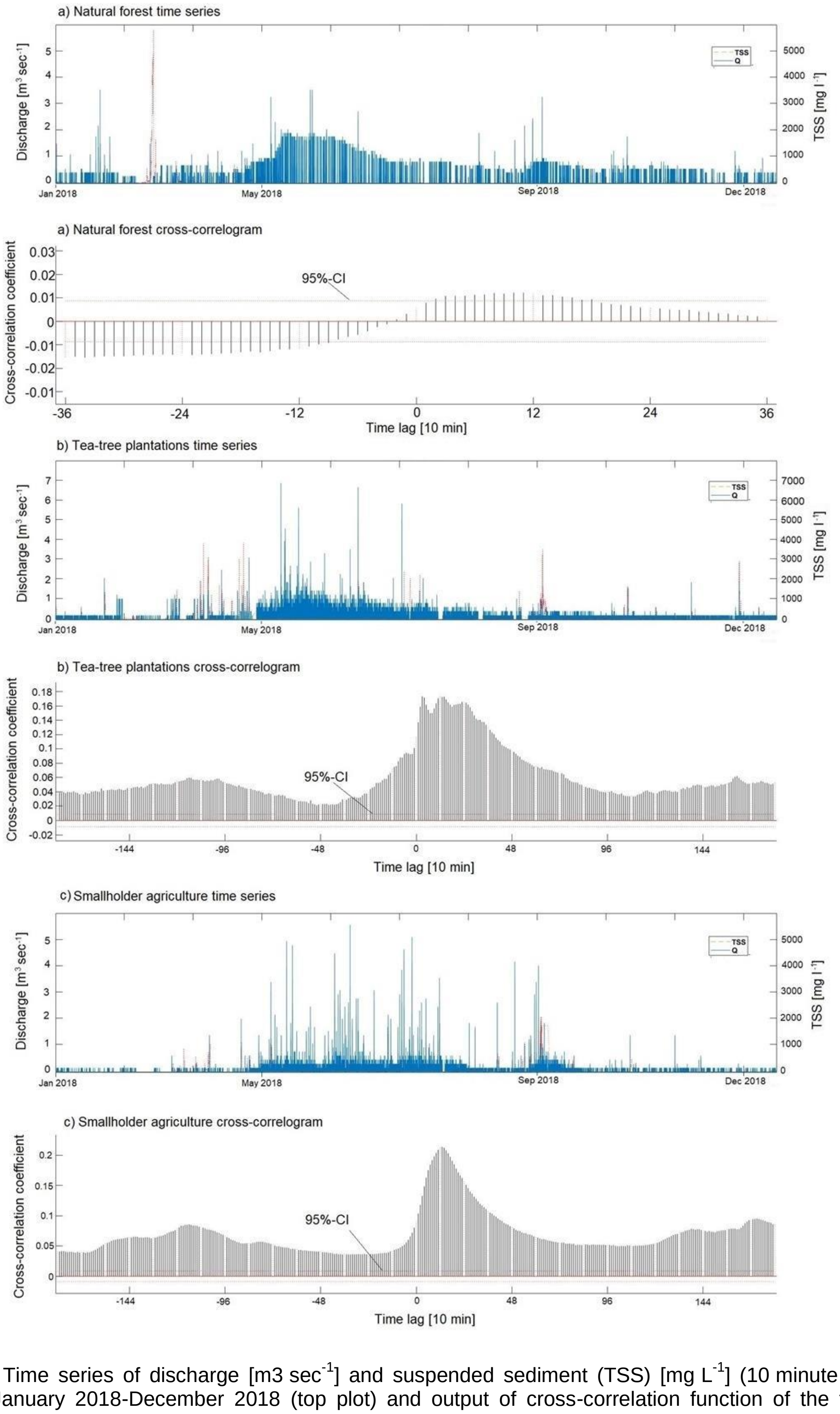


Figure 14 Time series of discharge [m3 sec^{-1}] and suspended sediment (TSS) [mg L^{-1}] (10 minute resolution) between January 2018-December 2018 (top plot) and output of cross-correlation function of the time series between discharge and suspended sediment of 2018 with time lag (10 min) and 95%-confidence interval (CI) (bottom plot) of the a) natural forest, b) tea-tree plantations and c) smallholder agriculture. The cross-correlation function between two time series (e.g. discharge and suspended sediment), is a function of the time lag measured in samples. For each time lag a cross-correlation coefficient is calculated. The peak correlation of the cross-correlogram shows the peak response time between discharge to suspended sediment concentration in time lags (10 minutes). The time lag is for the natural forest at 30 minutes, for the tea-tree plantations at 30 minutes with a delayed response at 120 minutes and for the smallholder agriculture at 132 minutes.

5 Agricultural land is the main source of stream sediments after conversion of an African montane forest

5.1 Introduction

Montane headwater catchments are susceptible to soil erosion due to their topographic position on generally steep hillslopes (Wohl 2006; Morris 2014; Nishigaki *et al.* 2017). Soil erosion is accelerated on land where erosive rain falls on landscapes deforested and converted to agriculture (Defersha & Melesse 2012; Gessesse *et al.* 2015; Nishigaki *et al.* 2017). This phenomenon is evident in many places in sub-Saharan Africa in general and in East Africa in particular (Penny 2009). Soil particles are eroded and transported to downhill areas. The loss of topsoil reduces not only the arable soil depth, but also the content of soil organic matter, nutrients, and trace elements (e.g. organic carbon, nitrogen, phosphorus, magnesium, potassium) (Quinton *et al.* 2001; Morgan 2005), hampering agricultural productivity (Morgan 2005; Liniger *et al.* 2011; Saiz *et al.* 2016). Element concentrations are generally enriched in the fine particle fraction (particle size <63 µm) (Quinton *et al.* 2001; Rawlins *et al.* 2010; Laceby *et al.* 2015b), which are easily transported through water erosion (Morgan 2005). In addition to these on-site effects, soil erosion degrades waterways as suspended sediments reduce the physical, biological and chemical water quality of streams (Owens *et al.* 2005; Horowitz 2008). These off-site effects increase adversely water treatment costs (Clark 1985), and cause the siltation of water reservoirs, which can affect water supply and hydropower generation through reduced water storage capacity (Mogaka *et al.* 2006; Devi *et al.* 2008; Kondolf *et al.* 2014). Furthermore, sediments can be contaminated with heavy metals, nutrients and pesticides, degrading water quality (drinking water), affecting primary production and damaging aquatic habitats (Quinton *et al.* 2001; Horowitz 2008; Gellis & Mukundan 2013). Agricultural intensification together with poor land management practices accelerate soil erosion (Liniger *et al.* 2011) and increase the number of source areas that contribute sediment to the stream network. This highlights the importance of identifying sediment sources, so that efficient management strategies can be implemented to reduce soil erosion and sediment delivery to the streams, thus reducing on- and off-site impacts.

Sediment source fingerprinting is a well-established, valuable technique to identify and apportion target sediments, in this case stream sediments, to their different sources within a catchment (Evrard *et al.* 2013; Owens *et al.* 2016; Laceby *et al.* 2017; Davies *et al.* 2018). Fingerprinting begins with the classification of potential sediment sources based on reconnaissance sampling with the help of, for example, Google Earth imagery (Boardman 2016) and sampling of target and sediment sources across hydrographs (Phillips *et al.* 2000; Evrard *et al.* 2013). Then, potential tracer properties are selected, followed by statistical tracer selection (Collins *et al.* 2012, 2017; Gellis & Noe 2013; Smith *et al.* 2018). Several fingerprinting studies have used different sediment properties to successfully determine their provenance. For example, Singh (2009), Collins et al. (2010) and Hardy et al. (2010) used geochemical elements, while Fox and Papanicolaou (2008), Evrard et al. (2013) and Mckinley et al. (2013) used biogeochemical properties. Mukundan et al. (2010), Owens et al. (2012) and Evrard et al. (2016) included fallout radionuclides and others used a combination of different fingerprinting properties (Froger *et al.* 2018). Fundamental for the selection of trace elements for fingerprinting is that the tracers are present in measurable concentrations, behave in a conservative way through the mixing process (i.e. no change from source to sink) and are representative for the source (Koiter *et al.* 2013; Collins *et al.* 2017; Stock *et al.* 2018).

The advantage of using geochemical elements as tracers is that they provide a useful and inexpensive tool to determine rapidly a substantial number of potential tracer properties (Collins *et al.* 2017). Total nitrogen and total carbon are proven to be good tracers in discriminating between surface soil erosion (e.g. topsoil of agricultural land) and subsoil erosion processes (e.g. unpaved tracks, gullies or channel banks) (Owens *et al.* 2006). As the target sediment originates from upstream hillslope areas, the biogeochemical or geochemical elemental composition in the sediment source should be similar to the mixed sediment at the catchments outlet (Laceby *et al.* 2017). Having a large pool of tracers increases the chance to select statistically the optimum tracer composite to differentiate the target sediment to its originating sources and to quantify their relative contributions to the sediment in the stream water (Collins *et al.* 2017).

A wide variety of Frequentist or Bayesian modelling approaches have been applied in un-mixing modelling (Nosrati *et al.* 2014; Collins *et al.* 2017; Smith *et al.* 2018). The effectiveness of Bayesian sediment fingerprinting models with the use of geochemical fingerprint compositions has been demonstrated by Koiter et al. (2013), Cooper et al. (2014) and Blake et al. (2018) and has been positively evaluated by Davies et al. (2018). Bayesian models apply probability distributions and incorporate prior knowledge. This leads gradually to increased knowledge from one experiment to the next and to strengthening model performance. The prior knowledge of an uncertain quantity is described by the probability models (Davies *et al.* 2018; Stock *et al.* 2018).

The Sondu River Basin in Kenya, one of the headwaters of Lake Victoria, has experienced land use changes over the last four decades, where 25% of the largest remaining tropical montane forest in Kenya, the Mau Forest Complex, has been converted to commercial (tea and tree plantations) and smallholder agriculture (Brandt *et al.* 2018). A four-year sediment monitoring study, revealed that smallholder agriculture generates a five times higher annual sediment yield than a montane forest ecosystem (South-West Mau Forest part of Mau Forest Complex) and almost three times higher than commercial agriculture (tea and tree plantations), where erosion management strategies are implemented (Stenfert Kroese *et al.* 2019). The smallholder agriculture catchment, located in the montane headwater of the Sondu River Basin plays an important role in sediment delivery to the downstream reaches of Lake Victoria. Intensive land use practices without soil conservation techniques can increase sediment yields within the Lake basin, especially in regions with a high structural catchment connectivity (Fryirs 2013). A dense network of unpaved tracks connects hillslope areas with the stream network within the smallholder agriculture catchment and it has been observed delivering sediment-rich water to the streams during rain storms. This, together with a number of sediment studies in the tropics highlighting the importance of unpaved tracks in acting as natural drainages and discharging sediment direct to the stream network (Ziegler *et al.* 2001; Minella *et al.* 2008; Ramos-Scharrón & Thomaz 2016), suggest that these tracks may be a key sediment source.

The overall aim of the study was to apportion the relative contributions of four potential sediment sources: agricultural land, gullies, unpaved tracks and channel banks to suspended sediment yields within a smallholder agriculture catchment in the headwater of the Sondu River Basin. Identifying the major sediment source is critical to develop targeted soil conservation strategies to reduce erosion, to disconnect source areas from the stream network and to decrease sediment delivery to Lake Victoria.

The main objective was to determine the main sediment source within a smallholder agriculture catchment, through (a) identification of the best sediment tracer composite of a large pool of biogeochemical and geochemical elemental properties for sediment provenance determination, and (b) estimation of the relative sediment contribution from agricultural land, gullies, unpaved tracks and channel banks using a Bayesian multivariate un-mixing model. The hypothesis was set out that unpaved tracks are the main sediment source to the target sediment.

5.2 Methods

5.2.1 Catchment description

The study area is a smallholder agriculture catchment (27 km^2) located in the western highlands of Kenya. It is part of the headwater of the Sondu River Basin, which drains into Lake Victoria (Figure 15). The catchment has a dry season (January to March) and a wet season with two rainfall peaks: the long rains from March to May (77-277 mm month^{-1}) and the short rains from June to September (~160 mm month^{-1}) with continued intermitend rainfall events between seasons. The annual rainfall ranged from 1,400 to 1,800 mm (period 2015-2018) (Stenfert Kroese *et al.* 2020b).

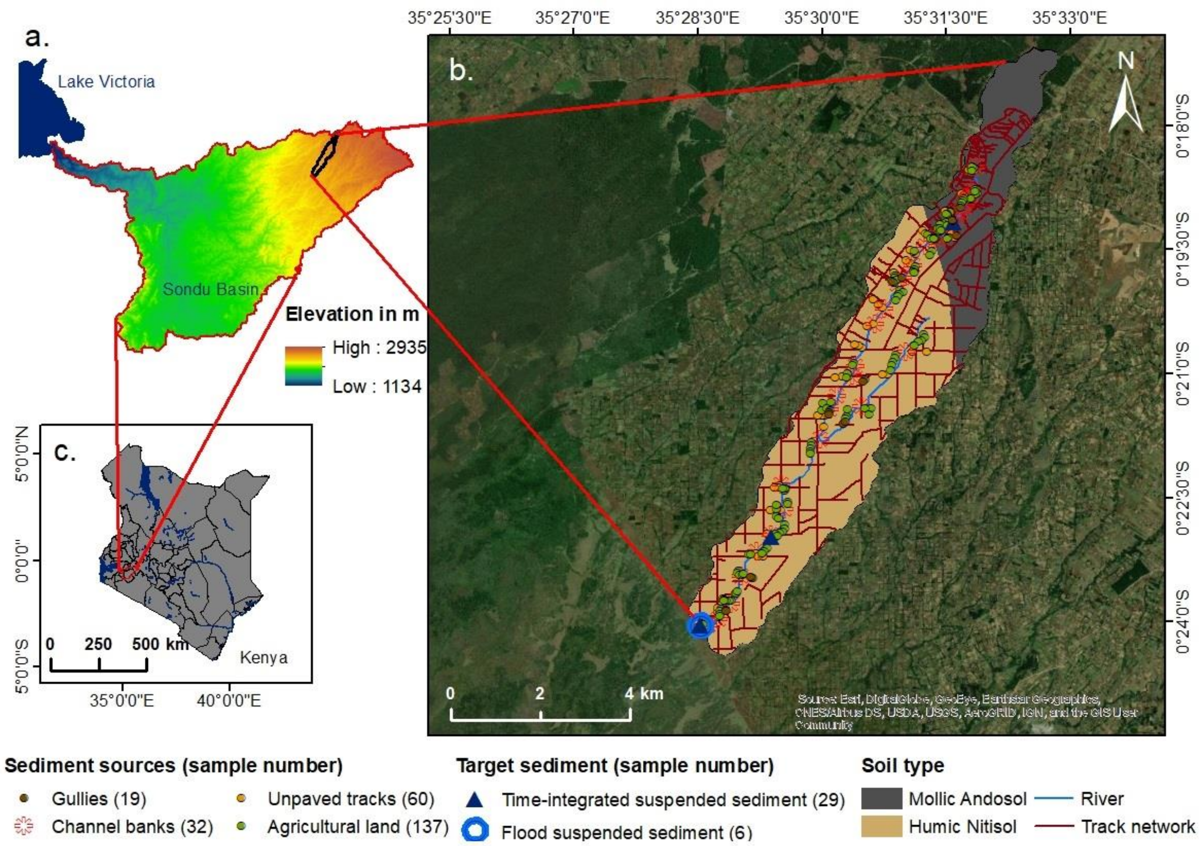


Figure 15 (a) Elevation map (SRTM digital elevation model 30 m resolution (USGS 2000)) of the Sondu River Basin with outlet to Lake Victoria and (b) pedological map with source and target sediment sampling points (Geology data from the Soil and Terrain database for Kenya (KENSOTER) version 2.0 (ISRIC 2004) with imagery basemap (Esri 2020) of the smallholder agriculture catchment in the highlands of (c) Kenya (map generated using ArcMap 10.4 (10.4.1) (ESRI 2016).

Throughout the catchment steep slopes (maximum 52.4%) characterise the montane area with an altitude range from 2,389 m a.s.l. at the catchment outlet to 2,691 m a.s.l. at the source. The geology is characterised by lava flows of volcanic (72%) and pyroclastic (28%) parent material (ISRIC 2004). Phonolites, a member of a group of extrusive igneous porphyritic rocks (lavas), predominate in this area (Binge 1962). There are two soil types: mollic Andosols (28%) in the upper part of the catchment and humic Nitisols (72%) in the middle and lower part (Table 6). They are well drained, deep soils (up to 5-6 m in depth) consisting of dark-brown loamy to clayey soils (Sombroek *et al.* 1982) with moderate to high organic matter content (Dunne 1979). Nitisols are characterised by high concentrations of free iron. Mollic Andosols are formed following rapid weathering of porous volcanic material. Dominant minerals include allophanes, with hydrous aluminosilicates, and ferrihydrite. Aluminium-humus complexes protect the organic matter from bio-degradation (WRB 2015). Due to their high porosity these soils have excellent internal drainage with infiltration rates of 400 mm hr^{-1} (Owuor *et al.* 2018). Nevertheless, desiccated Andosols, common after deforestation, have low water permeability that makes them susceptible to water erosion (WRB 2015).

Table 6 Physical characteristics of a smallholder agriculture catchment in the South-West Mau, Kenya (SD=standard deviation).

Area	km^2	27.2
Outlet coordinates[a]	Longitude	35°28'31.7316''E
	Latitude	0°24'4.0248''S
Stream order (Strahler)		1, 2
Drainage density	km km^{-2}	0.64
Track density	km km^{-2}	4.28
Altitude range	m a.s.l.	2,389-2,691
Mean slope ± SD	%	11.6±6.7
Geology[b]		Igneous rock (Volcanic) (72%) Pyroclastic (28%)
Dominant soils[b]		Humic Nitisol (72%) Mollic Andosol (28%)
Soil profile	Andosol	AC to ABC
	Nitisol	AB(t)C
Land use		Perennial & annual crops, woodlots, grazing land

[a]WGS 1984 UTM Zone 36S
[b]Geology data from the Soil and Terrain database for Kenya (KENSOTER) version 2.0

Historically, most of the catchment was covered by the Mau Forest Complex, which was cleared for agricultural land over the last four decades (Brandt *et al.* 2018). Today their fertile volcanic soils are planted with a variety of crops including maize, interspersed with beans, potatoes, millet, cabbage and tea on small farms (<1 ha). Eucalyptus (*Eucalyptus* ssp.*)*, cypress *(Cupressus* ssp.*)* and pine *(Pinus* ssp.*)* woodlots are interspersed with croplands and grazing land. Tillage in the form of hand-hoe cultivation and ploughing with oxen are the common practices for soil preparation. A dense network of unpaved tracks, organized in a rectangular grid, connect hillslopes with the stream network. These tracks regularly used by people and livestock often become incised to form gullies. Gullies can be found on steep slopes or around unprotected springs. Stream channel banks are susceptible to erosion due to the absence of riparian vegetation (Figure 16). The catchment supplies on average 106±46.8 t km^{-2} yr^{-1} of fine suspended sediment at the outlet (Stenfert Kroese *et al.* 2019).

5.2.2 Source and target sediment sampling design

Source and target sediment sampling were conducted from April to June 2019. A stratified sampling design was used which was based on field reconnaissance and Google earth imagery. Sampling points were restricted to areas where soil mobilization and on-field transport processes from hillslope areas were potentially connected with the stream network. To represent material susceptible to runoff detachment, sediment source samples were collected by scraping the uppermost layer of soil (~2 cm) of agricultural land and unpaved tracks (Figure 16a & c). The agricultural land source group included annual cropping field and fallow land, tea plots and grassland. Grasslands were not distinguishable from the cropland sources based on similar tracer concentrations, because they are used in rotations with croplands (Figure 21). Each agricultural land sampling point (*n*=137) was composed of multiple subsamples of the same land use collected along parallel transects within a radius of 25 m around a sampling point pre-selected visually on recent Google earth imagery. Multiple unpaved tracks samples were collected along transects on the track width, track length and of track walls and combined to a bulk sample (*n*=60). Samples of gullies (*n*=19) and channel banks (*n*=32) (Figure 16b & d) were collected from several points along vertical and horizontal profiles of the subsoil (up-to a depth of 2 m) and combined into a single sample. Channel bank samples were only taken on sites with exposed banks without vegetation cover. Litter or vegetated cover was removed prior to taking soil samples. Each surface and subsoil sample was composed of 10-20 subsamples. A plastic trowel was used for sample collection to avoid metal cross-contamination (Figure 17a & b).

Figure 16 (a) Characteristic landscape of the smallholder agriculture catchment, (b) hillslope gully, (c) unpaved eroded track connecting with the stream and (d) exposed channel bank.

Target sediment samples consisted of flood sediment samples (*n*=6) and time-integrated suspended sediment samples (*n*=29). Flood sediment samples were collected manually and automatically at the catchment outlet. Manual event-based samples were retrieved with bulk river water samples (~10 L). In addition, two automatic water samplers (3700 Full-size portable sampler, Teledyne ISCO, Lincoln, USA) were used to collect 0.5 L samples during the rising and falling limbs of the storm hydrograph. Sediment from the bulk river water samples and the automatic sampling were extracted through the settling and sedimentation method and then air-dried (Figure 17a-c). For time-integrated samples three sediment traps following the method by Phillips et al. (2000) were installed at the outlet and at two locations upstream of the outlet (Figure 15) and samples were collected every three to five days.

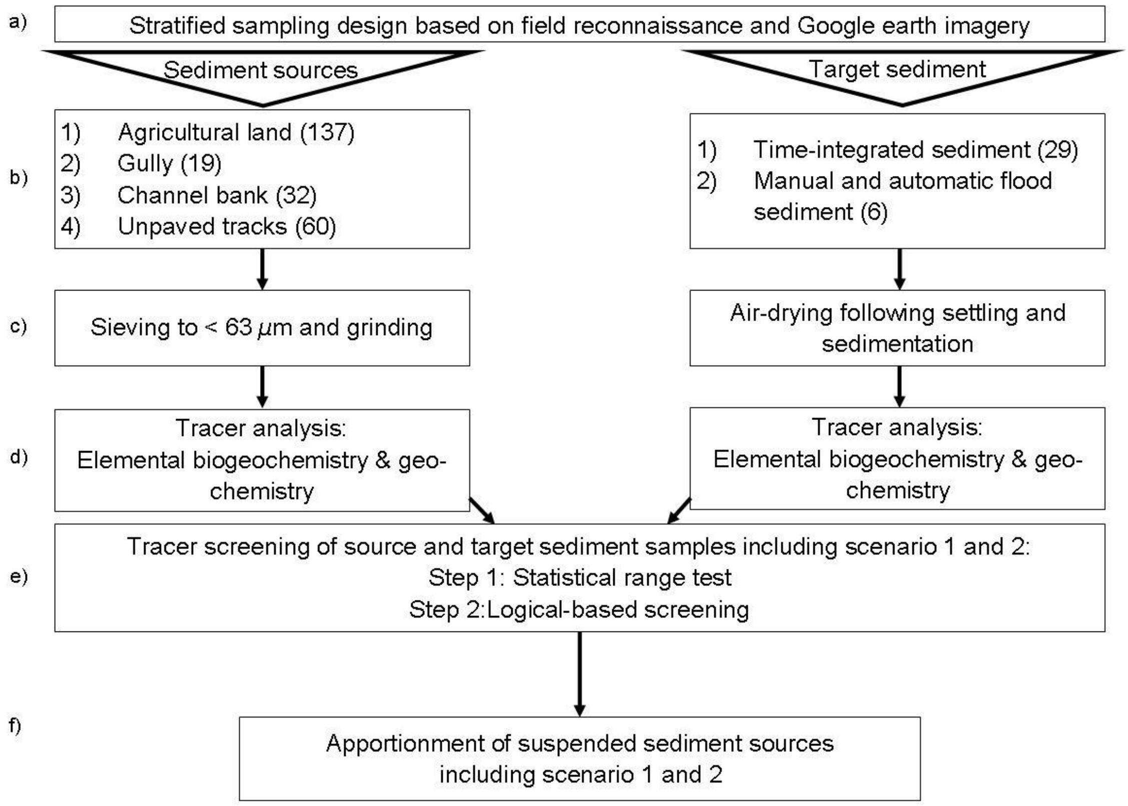


Figure 17 Flow diagram of the sediment fingerprinting sampling, analytical and statistical procedure (in brackets sample number).

5.2.3 Sediment source and target sediment processing and analysis

Sediment source samples were sieved to <63 µm in the field to facilitate direct comparison to the target sediment samples (Walling *et al.* 1999; Collins *et al.* 2017). Sediment source samples were air-dried and ground for further analysis (Figure 17c). All sediment samples (sources and target) were analysed in the laboratory for their chemical properties, including total carbon (TC) and total nitrogen (TN) content (Carter *et al.* 2003; Evrard *et al.* 2013) and major and minor elemental geochemical constituents (Carter *et al.* 2003; Collins *et al.* 2010) as potential sediment tracers. For TC and TN concentration a sub-sample of 15 mg was wrapped in tin capsules and combusted in an elemental micro-analyser (Elementar vario MICRO Cube, Elementar Analysensysteme GmbH, Langenselbold, Germany) at 950°C. A handheld XRF spectrometer (Bruker Tracer IV-SD, Bruker, Kennewick, WA USA) that uses energy dispersive X-ray fluorescence (EDXRF) was employed to determine the following major elements: Al_2O_3, CaO, Fe_2O_3, K_2O, MgO, Na_2O, P_2O_5, SiO_2 and TiO_2 and trace elements: Ba, Cr, Cu, Mn_2O_3, Nb, Ni, Pb, Rb, Sr, Y, Zn and Zr. For this analysis, a sub-sample of 500 mg was placed onto a thin film to measure trace elements at a setting of 40kV and 15.7 µA and major elements at an excitation of 15kV and 35 µA under vacuum with Helium gas. For each sample the mean of three replicates was used for further analysis (Figure 17d).

5.2.4 Tracer selection procedure

To select tracers for optimal discrimination of different sources, a verification procedure is needed to prove the strength of composite tracers for source apportionment. A two-step tracer selection approach was used based on a statistical tracer screening combined with a logical-based selection (step 2) for an optimum range of tracers (Figure 17e). This two-step approach differs from un-mixing models under the Frequentist framework, where commonly a three-stage statistical procedure is aimed to select a minimum range of tracers (Davies *et al.* 2018; Batista *et al.* 2019). The use of several tracers including weak tracers increases the explanatory power of un-mixing modelling under the MixSIAR Bayesian framework (Small *et al.* 2004; Martínez-Carreras *et al.* 2008; Sherriff *et al.* 2015). The covariance structure in the MixSIAR un-mixing modelling reduces redundancy and therefore a discriminant function analysis to create a composite of a minimum of tracers is not required (Stock *et al.* 2018).

The range test was used to exclude elements that do not differentiate sediment sources (Blake *et al.* 2018). A target element concentration should be in range with the source mixing polygon showing whether a tracer on the target sediment is enriched or depleted compared to the sediment sources (Walling *et al.* 1999; Mukundan *et al.* 2010). Consequently, the range test analyses the conservative behaviour of the selected tracers (Collins *et al.* 2017). Elements plotting outside the mixing polygon are removed from the subsequent analysis. Here, the mean and standard deviation of log-transformed concentrations of the target sediment should be within the ranges of the concentrations of the sediment sources. Tracers outside this criterion violate the numerical modelling assumptions and may lead to false results of the un-mixing model (Collins *et al.* 2013). The remaining elements selected by the range test were screened for their discrimination power to distinguish between surface and subsurface sources. The discriminating power of the selected tracer composite was evaluated using the reclassification coefficient and linear discriminant analysis (LDA) bi-plots. Based on the evaluation of the discrimination power, two different scenarios were run: scenario 1 with the inclusion of the unpaved tracks as individual source, and scenario 2 excluding the tracks from the analysis. Tracers were screened independently for each scenario. For statistical analyses, the R software (RStudio 2017) was used together with the packages MASS (Venables & Ripley 2002) and klaR (Weihs *et al.* 2005) for the LDA cross-validation.

5.2.5 Modelling source apportionment

To estimate the relative contribution of each source at catchment level, a Bayesian un-mixing model was applied using the MixSIAR model (Stable Isotope Analysis in R) (Stock & Semmens 2016). The MixSIAR model runs in the JAGS (Just Another Gibbs Software)

software (Plummer 2003) to carry out a Markov chain Monte Carlo (MCMC) sampling together with a Bayesian analysis that produces diagnostics, density plots *a posteriori* and summary statistics. Prior to modelling, element concentrations were log transformed to approach normality. The MixSIAR model uses the mean and standard deviation of the tracers as inputs. The model parameters, i.e. the proportions of tracer compositions of the sediment sources, are treated as random variables. Parameter uncertainty is specified by using three different stages of probabilistic predictions by: (1) using the Dirichlet distributions to determine prior probability distributions for parameters, (2) constructing a likelihood function for the model and (3) using the Bayes rule to adjust prior distributions based on observed data to derive the posterior probability distribution (Bolstad 2007). The basic linear un-mixing model takes the following form:

$$C_{ti} = \sum_{s=1}^{m} P_s C_{Si} \quad (8)$$

with $\sum_{s=1}^{m} P_s = 1$ and $0 \leq P_s \leq 1$. C_{ti} is the concentration of tracer *i* of the target sediment *t*, P_s is the proportional contribution from sediment source *s*, and C_{Si} is the concentration of tracer *i* of the sediment source *s* with the number of sediment source *m*. The estimated discrimination of sediment sources was carried out with the MCMC sampling on three 'long' chains of length 300,000 with a 200,000-sample burn-in and a jump length of 100 to minimize autocorrelation between runs, yielding 3,000 model values of proportional source contributions. Un-mixing model convergences were assessed with the Gelman-Rubin diagnostic, where the chain length was increased when >5% of total variables was above 1.05 (Stock & Semmens 2016). The un-mixing modelling results of the relative contributions of each sediment source are presented as the average with 95%-confidence interval for scenario 1 and 2 (Figure 17f).

5.3 Results

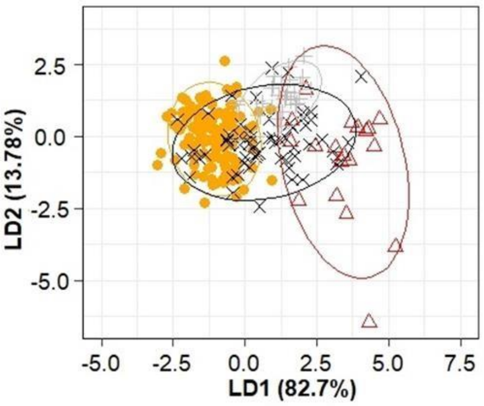
5.3.1 Tracer selection and their discriminative behaviour

Prior to the statistical procedure of tracer selection, P_2O_5 was removed as a potential tracer. Phosphorus concentrations may be highly variable in space due to inorganic fertilizer applications and because phosphorus is prone to transformation. This non-conservative behaviour may influence the fingerprinting modeling results (Granger *et al.* 2007; Collins *et al.* 2017). Highly P_2O_5-enriched sediment sources were observed on samples originating from tea fields within the catchment. Of 23 geochemical and biogeochemical elements measured in each source and target sediment sample, eight (including P_2O_5) were removed due to values below the detection limit (with the exception of P_2O_5) (Table 9). During the tracer screening, with the tracks included as a source (scenario 1), another five elements were removed because these did not comply with the range test (see Methods, step 1). The ten elements remaining that passed the range test were further assessed. Rb and Zr were excluded in step 2 due to their low discrimination power to differentiate between surface and subsurface sources. A final set of eight tracers remained, comprising: TN, Al_2O_3, Fe_2O_3, K_2O, MgO, Mn_2O_3, Sr and Nb. All selected composite fingerprints were able to reclassify correctly 80% of the samples in their source group after the LDA cross-validation. Because a reclassification coefficient of 80% (Table 7, scenario 1) is considered to be weak and because of the overlapping of tracers on the tracks with the other three sediment sources (Figure 18a), a second scenario was run where tracks as individual source were excluded. The tracers remained the same for a reduced number of sediment sources and the exclusion of the tracks increased the discriminatory power of the composite fingerprints to 95% (Table 7, scenario 2).

Table 7 Tracers for two selection steps and reclassification coefficient (%) for scenario 1 with unpaved tracks and scenario 2 without unpaved tracks.

Scenarios	**Selection step**	**Selected tracers**	**% of correctly classified samples**
Scenario 1: Unpaved tracks included	Step 1	TN, Al_2O_3, Fe_2O_3, K_2O, MgO, Mn_2O_3, Sr, Rb, Nb, Zr	
	Step 2	TN, Al_2O_3, Fe_2O_3, K_2O, MgO, Mn_2O_3, Sr, Nb	80
Scenario 2: Unpaved tracks excluded	Step 1	TN, Al_2O_3, Fe_2O_3, K_2O, MgO, Mn_2O_3, Sr, Rb, Nb, Zr	
	Step 2	TN, Al_2O_3, Fe_2O_3, K_2O, MgO, Mn_2O_3, Sr, Nb	95

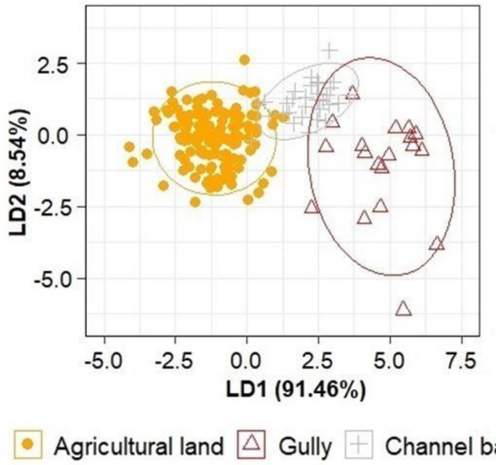
The tracers discriminated agricultural land, gullies and channel banks sediment sources well as shown in the LDA bi-plot using the first and second discriminant functions (LD1 and LD2) (Figure 18a - scenario 1). Agricultural land and gullies do not overlap much unlike the tracer signature for the tracks that was not well discriminated. The confusion matrix shows the performance of the discriminant analysis and predicts the number of overlying samples with the tracks: agricultural land has 22 samples, the channel banks 5 and the gullies source has 3, which is also depicted in the LDA plot with highly distributed points of overlying sources (Figure 18a). When the tracks source is removed from the analysis in scenario 2, the LDA bi-plot shows improved source discrimination for the agricultural land, gullies and channel banks sources (Figure 18b).



a) Scenario 1: unpaved tracks included

Confusion matrix

		Actual			
		Agricultural land	Gully	Channel bank	Track
Predicted	Agricultural land	132	0	4	22
	Gully	0	15	0	3
	Channel bank	0	2	21	5
	Track	5	2	7	30



b) Scenario 2: unpaved tracks excluded

Confusion matrix

		Actual		
		Agricultural land	Gully	Channel bank
Predicted	Agricultural land	135	0	4
	Gully	0	16	0
	Channel bank	2	3	28

Figure 18 Linear discriminant analysis (LDA) for a) scenario 1: unpaved tracks included & b) scenario 2: unpaved tracks excluded showing the first and second discriminant functions (LD1 and LD2) of source reclassification using the selection of the composite fingerprints. Ellipses represent the 95%-confidence interval. The confusion matrix shows predicted (rows) and actual (column) number of samples for each scenario.

5.3.2 Source and target tracer composition

TN was a powerful tracer to discriminate between topsoil (agricultural land) and subsoil sediment sources (gullies, channel banks). The high TN (4.5±0.6 g kg^{-1}) content on the target sediment suggests that most of this sediment originated from the N-enriched agricultural lands (4.5±0.7 g kg^{-1}), where the sources from subsurface soil were depleted. The mean concentrations of Al_2O_3, Fe_2O_3 and MgO in agricultural land were significantly lower than in the gullies, channel banks and tracks sediment sources ($p<0.05$). Low concentrations of these tracers at the agricultural land source corresponded with low concentrations at the target, with no significant difference between Al_2O_3 and MgO. The highest concentrations for Al_2O_3, Fe_2O_3 and MgO were measured in the gullies with 292.5±49.6, 251.1±44.6 and 12.8±3.3 g kg^{-1}, respectively. The mean concentrations of the tracers from the gullies were significantly lower and higher than those in the soil of agricultural land, except in Sr ($p<0.05$). The mean concentrations of the tracks were within the range of those of the agricultural land (Figure 19) and were not analysed further because they cannot be distinguished from the agricultural land.

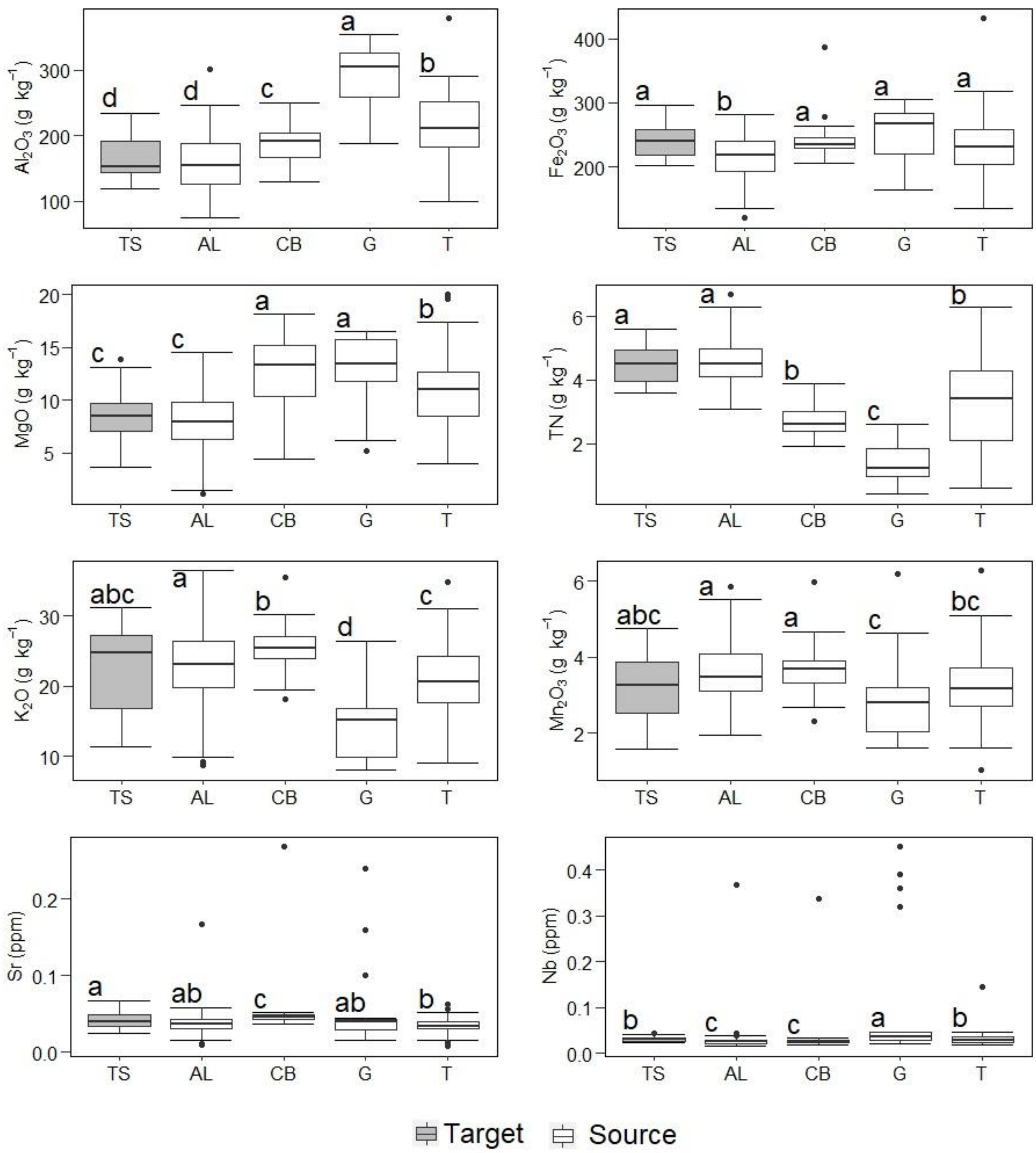


Figure 19 Tracer concentrations (g kg^{-1}) on sediment sources (AL=agricultural land, CB=channel banks, G=gullies and T=tracks) and target sediments (TS) (letters indicate significant difference p<0.05).

5.3.3 Sediment apportionment to their sources

The un-mixing model using the selected eight tracer fingerprints on the sediment sources passed the Gelman-Rubin convergence diagnostic with a long MCMC chain run length. All the potential scale reduction factor values were <1.05, indicating that the chain length of the MCMC was long enough. The fingerprinting results estimate the contributions of the four sediment sources to the target sediment with uncertainty estimated through the MCMC simulation procedure with 3,000 posterior realizations, which are expressed as mean and 95%-confidence intervals (CI). The fingerprinting analysis shows that agricultural land accounted for 75% (95%-CI 63-86%) of the target sediment at the outlet of the smallholder agriculture catchment while the channel banks contributed with 21% (95%-CI 8-32%). The lowest contributions to sediment were generated by the tracks and gullies with 3% (95%-CI 0-12%) and 1% (95-CI 0-4%), respectively. With the removal of tracks as independent sediment source, the relative contributions increased slightly for the agricultural land to 77% (95%-CI 67-87%) and for the channel banks to 22% (95%-CI 11-33%). The apportionment for gullies remained the same at 3% (Figure 20).

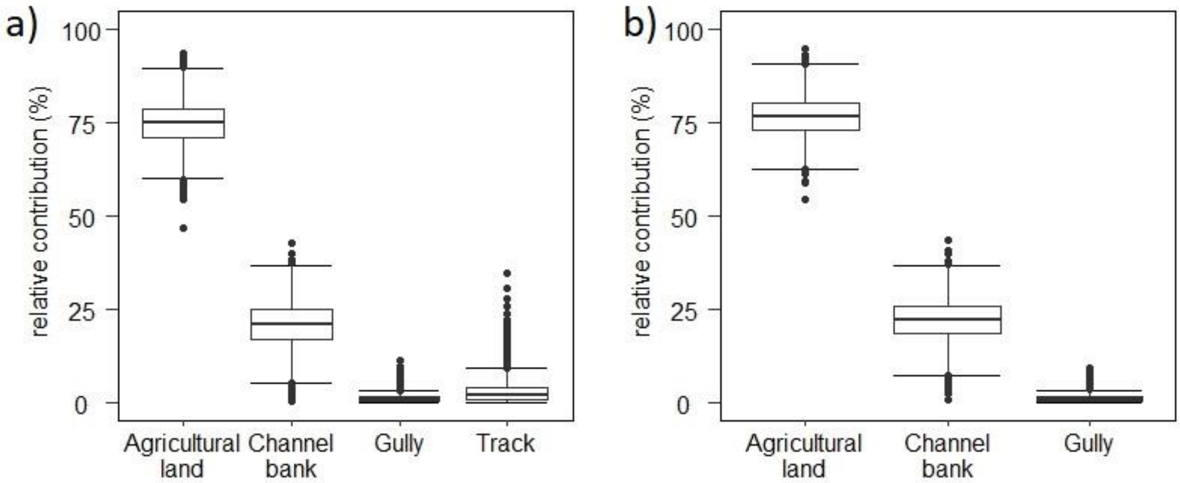


Figure 20 Relative apportionments of agricultural land, channel banks, tracks and gullies sediment sources based on 3,000 MCMC runs with (a) tracks included as a sediment source and (b) excluded tracks.

5.3.4 Sediment yield of each sediment source

We calculated absolute sediment generated by each sediment source by using the contributions of each source and the sediment yield of 106 t km^{-2} yr^{-1} calculated at the catchment outlet by Stenfert Kroese et al. (2019). The contribution of agricultural land was estimated to be 80 t km^{-2} yr^{-1}, 22 t km^{-2} yr^{-1} by channel banks, 3 t km^{-2} yr^{-1} by tracks and 1 t km^{-2} yr^{-1} by gullies. In addition to the absolute sediment contributions per catchment area (27 km^{2}), the sediment yield was evaluated of each specific sediment source per unit area within the catchment. The total annual sediment load (2,892 t) was multiplied by the relative sediment contribution to obtain the sediment yield per source area (Table 8). This was then divided by the area occupied by the different sources to calculate the sediment yield per unit area, i.e. the amount of sediment originating from one square kilometre of land covered by that particular sediment source. By including the unit area, the sediment yield was highest for channel banks (6,073 t km^{-2} yr^{-1}), followed by gullies (567 t km^{-2} yr^{-1}), tracks (150 t km^{-2} yr^{-1}) and the lowest for agricultural land (85 t km^{-2} yr^{-1}).

Table 8 Relative (%) and absolute sediment contributions (t km^{-2} yr^{-1}) weighted per source area and catchment area for tracks included and excluded of the sediment apportionment within a smallholder agriculture catchment in the Sondu Basin of Kenya. In brackets 95%-confidence interval.

Scenario	Sources	Surface area km^2	Relative contribution %	Absolute contribution per source area t km^{-2} yr^{-1}	Absolute contribution per catchment area t km^{-2} yr^{-1}
Scenario 1 (tracks included)	Agricultural land	25.7	75 (63-86)	85 (71-97)	80 (67-91)
	Gullies	0.1	1 (0-4)	567 (0-2,268)	1 (0-4)
	Channel banks	0.1	21 (8-32)	6,073 (2,314-9,254)	22 (8-34)
	Tracks	0.6	3 (0-12)	150 (0-598)	3 (0-13)
Scenario 2 (tracks excluded)	Agricultural land	26.8	77 (67-87)	83 (72-94)	82 (71-92)
	Gullies	0.1	1 (0-4)	567 (0-2,268)	1 (0-4)
	Channel banks	0.1	22 (11-33)	6,362 (3,181-9,544)	23 (12-35)

5.4 Discussion

5.4.1 Behaviour of selected tracer elements

The tracer composite of scenario 2, excluding tracks, explained 95% of classified sources, which is considered to be robust (Sherriff *et al.* 2015). The selected tracers showed clear differences in their concentrations between sources, caused by underlying biogeochemical and pedological processes related to each element. TN is an important plant nutrient and therefore a powerful tracer to discriminate between surface (agricultural land) and subsurface (gullies and channel banks) sediment sources. The higher concentrations at the soil surface reflect the inputs from atmospheric sources, fertilization, animal manure and biological N fixation (Jobbágy & Jackson 2004; Fox & Papanicolaou 2007). This decrease in TN concentrations with depth helped differentiate between surface and subsoil sources in this study, similarly to previous fingerprinting studies e.g. Russell et al. (2001), Gellis et al. (2009) and Collins et al. (2019). To characterise the natural variability in the distribution of TN among sediment sources, a larger number of source samples was collected than typically obtained as recommended by Laceby et al. (2015a). As suspended sediment was derived during high flow events when sediment travel times are short, the reduction in TN concentrations due to biological processes, such as mineralization are expected to be small (Rose *et al.* 2018).

Geochemical tracers also showed differences between surface and subsoil sources resulting from weathering and pedogenic processes. For example, Al_2O_3, Fe_2O_3 and MgO were enriched in subsoils and depleted in topsoils, whereas the opposite was observed for K_2O and Mn_2O_3. This elemental behaviour was observed in a fingerprinting study by Tiecher et al. (2017), where aluminium was a strong tracer to differentiate subsoil and surface sources in Southern Brazil. Clay content is expected to increase with soil depth with enriched residual Al_2O_3 and Fe_2O_3 concentrations (McLaughlin 1954; Marques *et al.* 2004). The subsoil is expected to be enriched with trace elements (Sr and Nb), because clay acts as a sink for these elements (Horowitz 1991). Tropical climates with high annual rainfall intensify weathering processes of mollic Andosols and humic Nitisols. Less weathered subsoil may therefore exhibit higher concentrations in Al_2O_3, Fe_2O_3 and MgO (Tyler 2004; Bini *et al.* 2011). The elemental composition of the topsoil may not have been as influenced by pedological processes as deeper horizons which are closer to the parent material (Tyler 2004). Igneous rocks are rich in Fe_2O_3 and MgO (Marshak 2008), thus providing a good discriminator for subsoil sources with higher concentrations of these elements. Conversely, K_2O and Mn_2O_3 are plant macro- and micronutrients and are recycled by plants mainly in the topsoil. Vegetation on agricultural land may transport these heavier elements (K_2O and Mn_2O_3) to the topsoil, a process that does not occur in gullies or unpaved tracks which are mostly bare (Jobbágy & Jackson 2001, 2004).

Although the tracers helped to differentiate between surface and subsoil sources, they could not differentiate clearly between the different land uses, such as grasslands and cropping fields. Only 67% of the sources were correctly classified by the tracers, which is why the different land uses were categorized into one sediment source group called agricultural land (Figure 21). Future source apportionment studies could also include the use of an informative prior from independent evidence to help constrain the model given source overlap (Wynants *et al.* 2020).

5.4.2 Sediment sources and hotspots

The results of this study show that agricultural land is the main sediment source in the smallholder agriculture catchment with a contribution of 77%. This is reflected in the chemical composition of the target sediments, with tracer concentrations similar to those expected in agricultural topsoils (Figure 19). A similar source apportionment was observed in two studies in steep cultivated catchments in South-Brazil, where surface erosion originates mainly from agricultural land with smaller to minor contributions from channel banks and unpaved tracks (Minella *et al.* 2008; Tiecher *et al.* 2017). Also in a Zambian catchment, sediments were found to mainly originate from communal land cultivation, which covered about 70% of the catchment area. In that study, a dense network of trackways was thought to connect overgrazed hillslope areas with the stream network delivering sediment from the bush grazing sediment source (Collins *et al.* 2001). These findings further elucidate the need for soil conservation strategies in other tropical catchments with a current trend towards agricultural expansion (Hosonuma *et al.* 2012).

In this study, unpaved and highly compacted tracks often run parallel to the slope, thus acting as conduits during rainfall events and turning into ephemeral streams carrying surface runoff with high loads to the streams. Due to their frequent use and position, it was hypothesised that these tracks would be the main sediment source, as suggested by other sediment fingerprinting studies (Motha *et al.* 2004; Nosrati & Collins 2019). However, the tracks were found to be only a minor contributor to the overall catchment sediment budget. Their role as conduits from the surrounding source areas to the stream may explain the overlap of tracers in the track samples with the remaining sources. Besides, unpaved tracks may have been converted from agricultural land which might also lead to similarity in their tracer characteristics. Consequently, unpaved tracks were excluded as an independent sediment source, because the tracer composite could not clearly distinguish from agricultural land, leading to 80% of correctly classified samples.

Although the sediment yield per unit area of channel banks, gullies and unpaved tracks are large, they are not the main contributors to sediments at catchment scale. However, the sediment contribution increases at a smaller scale, which was also observed by Tiecher et al. (2017) for sub-catchments <3 km². The scale-dependency points to one main finding: that sediment loss from agricultural land per unit area is relatively low, but because it occupies most of the catchment area, it contributes the greatest proportion of the sediment yield. Channel banks, gullies and unpaved tracks are significant local hotspot sources, but their overall contribution is of lower significance. Instead, gullies and unpaved tracks act as significant conduits, connecting hillslopes with the streams. When weighting sediment contributions to the whole catchment, a dilution effect by the catchment area decreases the sediment supply from the scale-dependent source areas. When scaling up to the Sondu River Basin, the large proportion of agricultural areas within the whole catchment raises a concern, as it discharges an increasing amount of sediment at the outlet into Lake Victoria. However, source apportionment might vary depending on sampling locations of target sediment in nested sub-catchments due to catchment-scale dependent sediment dynamics (Koiter *et al.* 2013). A large scale campaign may give additional information of changing source contributions within the whole catchment.

5.4.3 Implications of sediment management strategies

An increased sediment yield in the streams originating from nutrient-rich topsoil of agricultural land contributes to eutrophication of waterbodies (Mainstone & Parr 2002) and threatens crop yields (Liniger *et al.* 2011; Saiz *et al.* 2016). The loss in annual agricultural productivity will decrease farmer's revenue, but, more importantly, it will threaten food production and security for future generations due to slow rates of soil formation (Evans *et al.* 2019). This is a further impediment for the already low agricultural productivity of the country, where soil erosion is a primary constraint to improving yields (Cilliers *et al.* 2018).

Cohen et al. (2006) estimated the magnitude of soil erosion losses for Kenya's economy to be US$ 390 million annually or 3.8% of the gross domestic product. In addition to the on-site impacts, the nutrient rich topsoil ending up in the streams may lead to an oversupply of macronutrients (N and P) (Quinton *et al.* 2001; Horowitz 2008) resulting in an eutrophic state of watercourses, turning rivers from natural to degraded (Kreiling *et al.* 2018). The discharge of nutrient rich sediments from the montane streams to the outlet of the Sondu River Basin is in particular a threat for the already enriched Lake Victoria (Lung'ayia *et al.* 2001) and degrades the lake's water quality. This emphasizes the need for targeted mitigation measures to decrease hidden costs of on- and off-site impacts of soil erosion.

Since the majority of the annual sediment yield in the study area originates from agricultural land, this should be the main target for soil management strategies. The largest proportion of the total sediment budget (59%) is generated during the start of the long rainy season, which coincides with the start of the planting season (March-April), large areas of bare soil or low ground vegetation cover on agricultural land (Stenfert Kroese *et al.* 2020b). Vegetative buffer strips alternated with fields cultivated with perennial crops can reduce the pace of surface runoff and can trap eroded material. This practice could lead to a significant reduction in soil loss from cropping fields, especially during the planting season (Wanyama *et al.* 2012). Buffer strips could also act as source of livestock feed for example to cultivate fodder crops such as Napier grass (*Pennisetum purpureum*). Currently, erosion buffer strips are only occasionally observed on agricultural land within the catchment. The sampling campaign was restricted to one hydrological period (April-June 2019), which limits the characterization of seasonal dynamics of sediment sources. In the future, target sediment could be sampled throughout different seasons (short and long rainy season and dry season) to test whether there are temporal changes in source contributions.

A small floodplain in a steep, narrow valley floor provides limited space for sediment storage in the study area. This increases the need for enhanced on-field erosion mitigation measures to protect soil on hillslopes. Other methods to reduce soil erosion from agricultural land include terracing and the cultivation of cover crops, which can boost crop productivity. Cover crops can protect the surface from erosive rainfall by reducing the energy of raindrops (Morgan 2005; Durán *et al.* 2008), as well as enhancing soil organic matter (Gyssels *et al.* 2005). Terracing on steep slopes, especially in the study area, could reduce surface erosion through decreasing slope length and steepness in dividing the slope into smaller segments (Liniger *et al.* 2011). In Kenya, *fanya juu* terracing is commonly used on moderately steep slopes up-to 20% (Tiffen *et al.* 1994). The use of these terraces on croplands have increased crop yields by 25% in East Africa (Liniger *et al.* 2011) and help the recovery of soil organic matter (Saiz *et al.* 2016) due to reduced soil erosion.

Although the unpaved tracks are not a main source of sediment, according to this study, they show high erosion rates per area, acting as hotspots, and therefore effective management strategies should still be targeted at unpaved tracks. Other studies have emphasized unpaved roads as landscape features with high erosion rates and as significant contributors of sediment to the stream network (Sidle *et al.* 2004; Croke *et al.* 2005; Rijsdijk *et al.* 2007). In Malaysia erosion rates were estimated up to 320±24 t ha^{-1} yr^{-1} originating from steep skid trails (>20% gradient) in logged forests (Sidle *et al.* 2004), while in South-Brazil the need for erosion control programmes were stressed onto unpaved tracks as they form into perennial landscape features (Thomaz *et al.* 2014; Tiecher *et al.* 2017). The potential sediment contribution could increase with the length of a track, where the gullied track length defines the distance of sediment transport (Croke *et al.* 2005). These results stress on the importance of disconnecting rural unpaved tracks from the drainage system to lower the contribution of sediment from surrounding hillslopes in tropical disturbed catchments. Sediment yield could be further reduced by disconnecting cultivated hillslope areas in the smallholder catchment from the highly incised and mainly long unpaved tracks to reduce sediment delivery during storm events. In addition, water pathways on the tracks could be diverted to adjacent agricultural lands to attenuate the routing of surface runoff by unpaved tracks to the streams.

Riparian zones, as described within Kenya’s Water Act, are defined as a buffer of 30 m along the watercourse (Republic of Kenya 2012). Riparian vegetation is efficient in buffering significantly surface runoff, reducing suspended sediments and nutrient discharge from agricultural land (Sheridan *et al.* 1999; Borin *et al.* 2005; Décamps *et al.* 2009), but also prevents the collapse of channel banks (Simon & Collison 2002). A dense riparian vegetation can strengthen channel bank stability through a dense rooting system (Abernethy & Rutherfurd 2000; Small *et al.* 2004). Tiecher et al. (2017) showed that the presence of an intact riparian forest disconnected cultivated areas from the stream network, decreasing the sediment contribution from cropland sources compared to areas with a degraded riparian forest in a catchment in Southern Brazil. The same was found in the Chesapeake Bay watershed, USA, where a forested floodplain increased the amount in trapped sediment (Gellis *et al.* 2009). In the study area, widespread cultivation within the buffer zone and the absence of riparian vegetation may result in the higher transfer from soil particles originating from hillslopes areas. Furthermore, increased pressure on available arable land in the catchment is leading to conversion from wetlands to agricultural lands. With this conversion, another important sediment trap is lost (Kansiime *et al.* 2007; Ryken *et al.* 2015). This emphasises the need for appropriate management of riparian areas to buffer hillslope sediments and to strengthen channel banks.

In the study area, gullies provided the smallest contribution (1%) to the sediment budget. Although the overall sediment contribution of channel banks and gullies remained low, both sources are potential soil erosion hotspots at the local scale. Furthermore, the absence of management strategies might increase the relative sediment contribution with proceeding gully head or wall erosion (Croke *et al.* 2005). Gully rehabilitation could be achieved by replanting trees and increasing vegetation cover. Further gully expansion could be impeded in diverting waterways to avoid lateral runoff water entering through gully walls and keeping away livestock and human movement from highly eroded areas (Liniger *et al.* 2011).

5.5 Conclusions

We aimed to identify the relative contribution of four sediment sources: agricultural lands, unpaved tracks, gullies and channel banks using un-mixing modelling (MixSIAR) to assess the relative contribution to suspended sediment at the outlet of a smallholder agriculture catchment in the highlands of Kenya. Due to the topography of this montane headwater catchment, there is a strong structural catchment connectivity between hillslope areas and the stream, with limited deposition of sediment in the narrow floodplain with a steep valley floor. The sediment fingerprinting demonstrated that topsoil of agricultural lands (77% with 95%-CI of 67-87%) is the main source of suspended sediment. A dense network of unpaved trackways fragments the landscape of the catchment and acts as conduit between hillslope areas and waterways. However, these tracks could not be identified as individual sediment source. The lack of an intact riparian vegetation results in subsoil channel bank erosion as a second major sediment contributor with 22% (11-33% 95%-CI), whereas gullies contribute only a small proportion with 1% (0-4% CI-95) to the sediment yield.

Sediment fingerprinting may be applied in other tropical agricultural catchments to further develop the knowledge base of sediment source areas where problems of soil erosion and the accumulation of sediments are a concern. Based on these findings, it was speculated that in catchments dominated by smallholder agriculture in large parts of East Africa in combination with poor road network and lack of appropriate runoff and sediment management strategies agricultural land is the main source of sediment. These results emphasize the need for targeted erosion mitigation strategies on agricultural land to limit soil erosion and control annual sediment yield. Moreover, the large proportion in agricultural areas in the highly populated highlands of Kenya raises concern in discharging an increasing amount of sediment to the streams, consequently, impacting further the water quality of Lake Victoria.

Acknowledgments

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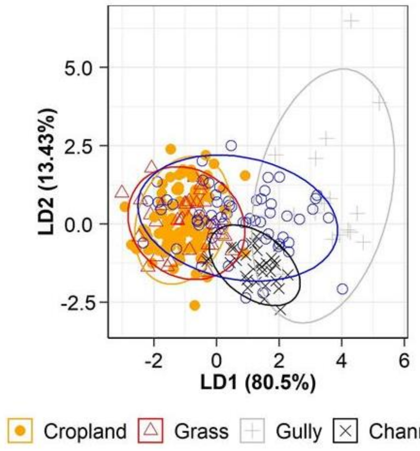
5.6 Supporting information

Table 9 Average concentrations of TN, TC and geochemical elements of source and target sediment samples (* removed prior analysis).

	Agricultural land	Tracks	Gullies	Channel banks	Target sediment
TN [g kg^{-1}]	5	3	1	3	4
TC [g kg^{-1}]	53	41	16	29	50
Na_2O [g kg^{-1}]	32	33	30	30	34
MgO [g kg^{-1}]	8	11	13	12	9
Al_2O_3 [g kg^{-1}]	157	214	293	185	166
SiO_2 [g kg^{-1}]	478	466	454	455	449
P_2O_5 [g kg^{-1}]*	1	1	1	1	1
K_2O [g kg^{-1}]	23	21	14	25	23
TiO_2 [g kg^{-1}]	30	34	36	40	30
Mn_2O_3 [g kg^{-1}]	4	3	3	4	3
Fe_2O_3 [g kg^{-1}]	213	232	251	241	240
CaO [g kg^{-1}]	9	9	7	11	11
Ba [ppm]*	0.03	0.03	0.02	0.02	0.02
Cr [ppm]*	-0.03	-0.01	-0.01	-0.03	-0.02
Ni [ppm]*	0.00	0.00	0.00	0.00	0.00
Cu [ppm]*	0.00	0.00	0.00	0.00	0.00
Zn [ppm]*	0.01	0.01	0.01	0.00	0.00
Pb [ppm]*	0.00	0.00	0.00	0.00	0.00
Rb [ppm]	0.00	0.00	0.00	0.00	0.00
Sr [ppm]	0.04	0.03	0.05	0.05	0.04
Y [ppm]*	0.00	0.00	0.00	0.00	0.00
Zr [ppm]	0.15	0.16	0.52	0.16	0.15
Nb [ppm]	0.03	0.03	0.11	0.04	0.03



Predicted	Actual: Cropland	Grassland	Gully	Channel bank	Track
Cropland	95	31	0	3	17
Grassland	2	1	0	0	1
Gully	0	0	15	0	3
Channel bank	0	0	2	21	5
Track	3	5	2	8	34
		Reclassification coefficient			67%

Figure 21 Linear discriminant analysis (LDA) showing the first and second discriminant functions (LD1 and LD2) of source reclassification using the selection of the composite fingerprints. Ellipses represent the 95%-confidence interval. The confusion matrix shows predicted (rows) and actual (column) number of samples with reclassification coefficient.

6 Particulate macronutrient exports from tropical African montane catchments point to the impoverishment of agricultural soils

6.1 Introduction

In sub-Saharan Africa, streams often have high concentrations of suspended sediments mainly due to the anthropogenic disturbance of natural ecosystems (Mogaka *et al.* 2006; Penny 2009). These sediment concentrations can be particularly high in the steep highlands of East Africa, where surface soil erosion in catchments dominated by agriculture generates significantly more suspended sediments than in native forest catchments (Brown *et al.* 1996; Tamooh *et al.* 2014; Stenfert Kroese *et al.* 2020b). The loss of organic carbon- (C) and nutrient-rich topsoil through surface runoff induced by soil erosion from agricultural surfaces (Quinton *et al.* 2001; Powlson *et al.* 2011) leads to further deterioration of tropical soils (Okalebo *et al.* 2005; Tully *et al.* 2015). Phosphorus (P) and nitrogen (N) are soil nutrients which can limit crop growth and therefore their loss from topsoils should be avoided (Pasley *et al.* 2019). The clays found in many of the tropical soils of East Africa contain amorphous iron (Fe), which can limit plant P availability and consequently the application of fertilizers (inorganic N and P and/ or manure) is often required to improve agricultural productivity (Mutuo *et al.* 1999). Soil-forming processes tend to be much slower than the rates of soil loss through erosion (Amundson *et al.* 2015; Evans *et al.* 2020), which can lead to soil and plant nutrient deficits resulting in low and stagnant crop yields (Lederer *et al.* 2015; Saiz *et al.* 2016). Soil organic matter is a critical determinant of soil fertility, providing nutrients to plants, and playing fundamental roles in soil carbon sequestration and soil water functions (Weil & Brady 2016; Owuor *et al.* 2018). Processes affecting nutrient stocks and pools of soils are tightly connected to the processes controlling riverine C and nutrient fluxes.

Sediments and their associated organic C and nutrients, such as N and P (Quinton *et al.* 2001; Horowitz 2008; Johnson *et al.* 2018) may impact streams by reducing benthic communities, primary productivity and water storage capacity of water reservoirs (Tamene *et al.* 2006; Hunink & Droogers 2011) and by increasing turbidity (Tamooh *et al.* 2014; Stenfert Kroese *et al.* 2020b). In nutrient-limited freshwater systems, an excess of nutrients can cause eutrophication (Smith & Schindler 2009; Smith *et al.* 2017; Jarvie *et al.* 2019), inducing algal blooms and promoting invasive weeds (e.g. water hyacinth) (Lung’ayia *et al.* 2001). When C enters the watercourse, it can be mineralised and emitted as a greenhouse gas (Marx *et al.* 2017b, a). Nutrient-enriched sediments can turn stream waters from sink to source of nutrients through sorption and desorption processes and increased chemical and biological activity (Mainstone & Parr 2002; Palmer-Felgate *et al.* 2009; Kreiling *et al.* 2019).

Several studies investigating the effects of land use change and agricultural intensification on the water quality of river systems have focused on dissolved C and nutrients such as N and P (Drewry *et al.* 2009; Smith *et al.* 2017; Bowes *et al.* 2018). Others have shown that up to 95% of the nutrient loads and up to 40% of the C loads in streams are transported in particulate form, associated to suspended sediments (Scanlon *et al.* 2004; Moran *et al.* 2005; Rodríguez-Blanco *et al.* 2015). Sediment-associated nutrients are spatially variable (Withers *et al.* 2001; Harrington & Harrington 2014) and their concentrations increase during storm events with the rising limb of the hydrograph (Drewry *et al.* 2009; López-Tarazón *et al.* 2016). Walling et al. (1997) and Bender et al. (2018) observed that P loads mainly occur in particulate form, while N is mainly transported in dissolved form (Wang *et al.* 2015). Others have found that the major inputs to rivers occur as dissolved P and N loads (Harrington & Harrington 2014). Currently, there are no studies for East Africa investigating sediment-associated C, N and P exports from different land uses. This is an important gap in the knowledge because the East Africa region has a very dynamic land use system, with intensive conversion of natural ecosystems (forests, wetlands and grasslands) to subsistence and commercial agriculture (Carter *et al.* 2018).

This study aims to fill this knowledge gap by developing an improved understanding of the response of suspended sediments and C and nutrient fluxes associated with sediments under contrasting land use at catchment scale in the headwaters of the Sondu River Basin originating in the Mau Forest Complex, Kenya. Earlier work in the area found increased suspended sediment yields and dissolved nitrate exports in agricultural compared to forested land use types (Jacobs *et al.* 2018b; Stenfert Kroese *et al.* 2020b) and identified agricultural land as the main source of sediments within a smallholder agriculture catchment (Stenfert Kroese *et al.* 2020a). Therefore, the hypothesis was set out that the sediment-associated C and nutrient fluxes would be higher from the more intensively managed agricultural catchments than from a natural forest catchment.

6.2 Materials and methods

6.2.1 Catchment characteristics

This study was conducted in nested catchments in the montane headwaters of the Sondu River Basin (3,470 km^2) located in the South-West Mau, Western Kenya, home to one of the largest closed-canopy montane forest of East Africa (the Mau Forest Complex), and where the inherent fertility of the soils allowed the development of a tea industry in parallel to subsistence agriculture (Binge 1962). Over the last four decades, the western highlands of Kenya have undergone significant land use changes, whereby 25% of the Mau Forest Complex was converted to commercial and smallholder agriculture (Brandt *et al.* 2018). As an important headwater area to tributaries of Lake Victoria, the Mau Forest Complex is a critical catchment area that supplies people (approximately 5 million), livestock, wildlife and the economy with fresh water (UNEP 2008). Lake Victoria, an important freshwater lake for five countries, has shown increased signs of eutrophication over recent years (Zhou *et al.* 2014), stressing the need for mitigation and control of nutrient inputs.

The study catchments are characterized by distinct land uses: (1) natural forest (NF; 35.9 km^2), (2) tea-tree plantations (TTP; 33.3 km^2) and (3) smallholder agriculture (SHA; 27.2 km^2) (Figure 22 & Figure 23). The mean slope gradient of the catchments ranges between 11.6 and 15.7%, up to a maximum of 72% in the natural forest catchment. The streams are first- and second-order perennial streams merging to Sondu River (a sixth-order stream). The region has a bimodal rainfall pattern with a long rainy season (March-June) and a short rainy season (October-December) with a continued intermediate rainy season between the two wet seasons (July-September). The driest months are in January and February. The mean annual rainfall is 1,979 ± 325 mm yr^{-1} (period 1905-2019). Geology is composed of folded volcanics from the early Miocene. Kericho Phonolites cover the lower catchment (tea-tree plantations), followed by phonolitic nephelinites with intercalated tuffs and Mau ashes with basal tuff encompassing the natural forest, while phonolitic nephelinites comprises the upper catchment in the smallholder agriculture (Binge 1949; Jennings 1962). The catchments are covered by deep (>1.8 m) and well-drained, dark-red loamy soils (Sombroek *et al.* 1982), characterized as mollic Andosols and humic Nitisols (ISRIC 2004) with moderate to high amounts of organic matter (Dunne 1979).

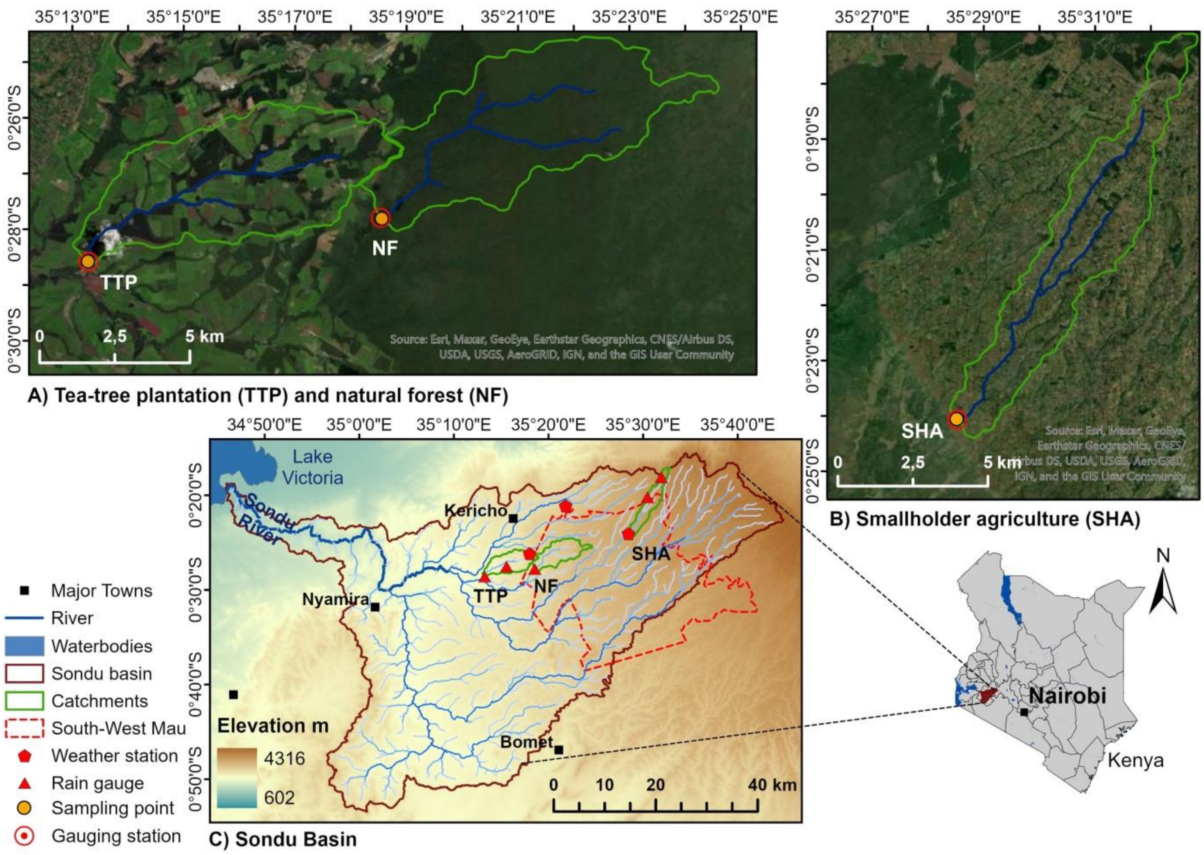


Figure 22 a) Tea-tree plantation (TTP) and natural forest (NF) and b) smallholder agriculture (SHA) catchments with imagery basemap (Esri 2020) as nested catchments of the c) Sondu River Basin with elevation (SRTM digital elevation model 30 m resolution) (USGS 2000) in the South-West Mau, Kenya.

The natural forest catchment falls within the South-West Mau Forest part of the Mau Forest Complex. As an afromontane mixed forest, species such as *Polyscias kikuyuensis*, *Macarangea kilimandscharica*, *Olea hochstetteri*, *Casearia battiscombei* and *Fagara* spp. dominate the vegetation, transitioning to irregular patches of bamboo forest above 2,300 m a.s.l. (Binge 1962). The riparian zone transits from the forest vegetation containing an understorey with a dense cover of shrubs and tree ferns combined with tall indigenous tree species (Table 10).

The tea-tree plantation catchment borders the South-West Mau to the west. The catchment is characterised by tea (*Camellia* spp.) plantations alternated with *Eucalyptus saligna* and *Cupressus lusitanica* woodlots used for fuelwood for tea processing at the tea factories and timber production. The common practices to control soil erosion are mulching and interplanting rows of oat grass between rows of tea during the establishment of new tea fields, cover crop establishment with mature tea trees, and terracing and sited cut-off drains. A well maintained road system along with open culverts as road drainages connects surface runoff with the riparian buffer zone. Herbicides are commonly used to control weeds. Aerial application of inorganic fertilizer is conducted two to three times per year on the tea plantations (150-250 kg N ha^{-1} yr^{-1} and 8-13 kg P ha^{-1} yr^{-1}) (Jacobs *et al.* 2018b). Riparian zones of up to 30 m from the river are commonly vegetated by native tree species densely covering the ground (Table 10).

In the upper smallholder agriculture catchment, subsistence farmers grow maize interspersed with beans, potatoes, millet, cabbage and tea (*Camellia* spp.) on farms usually smaller than one hectare. Grasslands for livestock and woodlots of *Eucalyptus saligna*, *Cupressus lusitanica* and *Pinus patula* are alternated with agricultural land. A combination of hoeing and herbicide application is used for weed control. Inorganic fertilizer is applied manually on potatoes and maize (23-45 kg N ha^{-1} yr^{-1} and 12-23 kg P ha^{-1} yr^{-1} twice a year on potatoes and once a year on maize), while manure is commonly used for cabbage and other greens. Unpaved roads, frequently used by people, livestock and motorbikes, often develop into deeply incised gullies running down slope, coupling hillslopes with the stream network. Degraded river banks with a sparse or absent riparian vegetation are prone to erosion, and in some places degraded riparian wetlands are found (Table 10).

Table 10 Catchment characteristics under different land use natural forest, tea-tree plantations and smallholder agriculture in the South-West Mau, Kenya.

		Natural forest	Tea-tree plantations	Smallholder agriculture
Outlet coordinates[a]		35°18'32.0472"E 0°27'47.592"S	35°13'17.22"E 0°28'34.9176"S	35°28'31.7316"E 0°24'4.0248"S
Area (km^2)		35.9	33.3	27.2
Elevation range (m a.s.l.)		1,968-2,385	1,788-2,141	2,389-2,691
Mean slope ± SD (%)		15.7±8.4	12.4±7.6	11.6±6.7
Basin order (Strahler)		1, 2	1, 2	1, 2
Drainage density (km km^{-2})		0.48	0.42	0.64
Sediment particle size	**Clay %**	81	76	87
	Sand %	19	24	13
Geology		Phonolites	Phonolitic nephelinites	Phonolitic nephelinites and Mau ashes with basal tuff
Dominant soils[b]		Humic Nitisols	Humic Nitisols	Mollic Andosols & humic Nitisols
Vegetation		Afromontane mixed forest with broad-leafed evergreen trees and shrubs, grassland, bamboo	Perennial tea plantations, *Eucalyptus saligna* and *Cupressus lusitanica* woodlots	Perennial & annual crops (maize, beans, potatoes, millet, cabbage and onions, tea), woodlots, grassland
Riparian vegetation		Forest vegetation	>30 m buffer with indigenous vegetation	Degraded riparian vegetation and wetlands, Eucalyptus woodlots

[a] WGS 1984 UTM Zone 36S
[b] KENSOTER Geology data from the Soil and Terrain database for Kenya (KENSOTER) version 2.0

6.2.2 Continuous field monitoring

The outlet of each catchment is equipped with an automatic gauging station to measure continuously (10 minute interval) water level (m) with a radar sensor (VEGAPULS WL61, VEGA Grieshaber KG, Schiltach, Germany) and turbidity (formazin turbidity unit=FTU) using a UV/Vis spectroscopy sensor (spectro::lyser, s::can Messtechnik GmbH, Vienna, Austria) (Figure 22). Stream discharge ($m^3\ s^{-1}$) was obtained by using a site-specific second-order polynomial water level to discharge rating curve (Jacobs *et al.* 2018b). Specific discharge [mm day^{-1}] was determined by integrating instantaneous discharge taken at 10 minute intervals over a day and relating it to the catchment area. *In situ* turbidity measurements were used to estimate suspended sediment concentrations (mg L^{-1}), based on a rating curve between turbidity and suspended sediment concentrations established by Stenfert Kroese et al. (2020b). After obtaining continuous discharge and suspended sediment concentration values, suspended sediment load was determined by multiplying suspended sediment concentration and discharge for each 10 minute interval. Suspended sediment yield was calculated by integrating sediment load over time and relating it to the catchment area.

Precipitation was measured using eight automatic tipping bucket rain gauges calibrated to measure cumulative rainfall every 10 minutes with a 0.2 mm resolution (5 tipping bucket rain gauges: Theodor Friedrichs, Schenefeld, Germany, and 3 weather stations: ECRN-100 high resolution rain gauge) (Figure 22). Thiessen polygons weighted the contribution of rainfall of every tipping bucket to each catchment. A more detailed description of sampling sites and instrumentation can be found in Jacobs et al. (2018). This study uses hydrological and sedimentological data between January 2018 and December 2019.

6.3 Data quality assurance

For quality assurance the turbidity, discharge and rainfall datasets were checked for anomalities. Recorded values were flagged with Not-a-Number (NaN) during malfunctioning such as (i) sensor above water level, (ii) siltation of turbidity sensors, (iii) biofilm or debris on the measurement window due to malfunctioning of automatic cleaning with compressed air, (iv) measurement gaps due to incidents of power supply failure or (v) restricted counting of number of tips by the rain gauges by blocked funnel or spiderwebs.

Once anomalous values were replaced by NaN, local outliers were detected by the median absolute deviation (MAD) for discharge and suspended sediment:

$$MAD_i = b\ M_{i2}(|x_i - M_{i1}(x_i)|) \quad (9)$$

where x_i is the whole dataset, M_{i1} is the median of the dataset and M_{i2} is the median of the absolute deviation from the dataset from its median. The standard deviation is estimated by the constant *b* set to 1.4826 for normal distribution (Leys *et al.* 2013). A moving window of *k*=16 measurements around observation x_i at time t_i was used to detect local outliers with $x_j = (x_{i-k/2} \dots x_{i-1}, x_{i+1} \dots x_{i+k/2})$:

$$\frac{x_i - M_{j,i}}{MAD_{j,i}} > a \quad (10)$$

where *a*=6 is the threshold for outlier selection, $M_{j,i}$ is the median and the $MAD_{j,i}$ is the MAD for x_j. There are gaps for discharge and suspended sediment data for the smallholder agriculture catchment in September until October 2019 due to theft of the power supply. Missing sediment data was integrated using a linear interpolation.

6.3.1 Suspended sediment sampling

Suspended sediment was sampled during the long rainy seasons in 2018 and during the drier period of the start of the long rainy season in 2019 (May-September 2018 and April-May 2019). The sediment sampling covered 20 and 12 sampling days in the natural forest, 22 and 13 days in the tea-tree plantation and 13 and 16 sampling days in the smallholder agriculture catchment in 2018 and in 2019, respectively. Three different methods were deployed for suspended sediment sampling: time-integrated sampling with sediment traps (*n*=88) following the method by Phillips et al. (2000) (Figure 23d), manual (*n*=6) and automatic (*n*=7) (3700 Full-size portable sampler, Teledyne ISCO, Lincoln, USA) storm event-based bulk sampling (Table 11). The manual event-based sampling was conducted next to the installed time-integrated samplers at each catchments outlet (Figure 22). Time-integrated samplers were emptied of accumulated suspended sediment after three to five days. The storm event-based samples were retrieved during a storm manually with bulk river water samples (~10 L). The auto-sampling was only conducted at the outlet of the smallholder agriculture catchment, whereby samples were collected during the rising and falling stages of a storm event at 30 minute interval and composed to a bulk sample. Sediment in suspension from all three sampling methods was extracted through settling and sedimentation followed by air-drying in aluminium trays.

Table 11 Total number of samples of time-integrated, manual and automatic storm event-based sediment sampling of the natural forest, tea-tree plantations and smallholder agriculture catchments in 2018 and 2019.

Catchment	Year	Number of time-integrated sediment samples	Number of manual storm event-based bulk samples	Number of automatic storm event-based bulk samples
Natural forest	2018	20	-	-
	2019	12	2	-
Tea-tree plantations	2018	22	-	-
	2019	11	3	-
Smallholder agriculture	2018	13	-	-
	2019	10	1	7

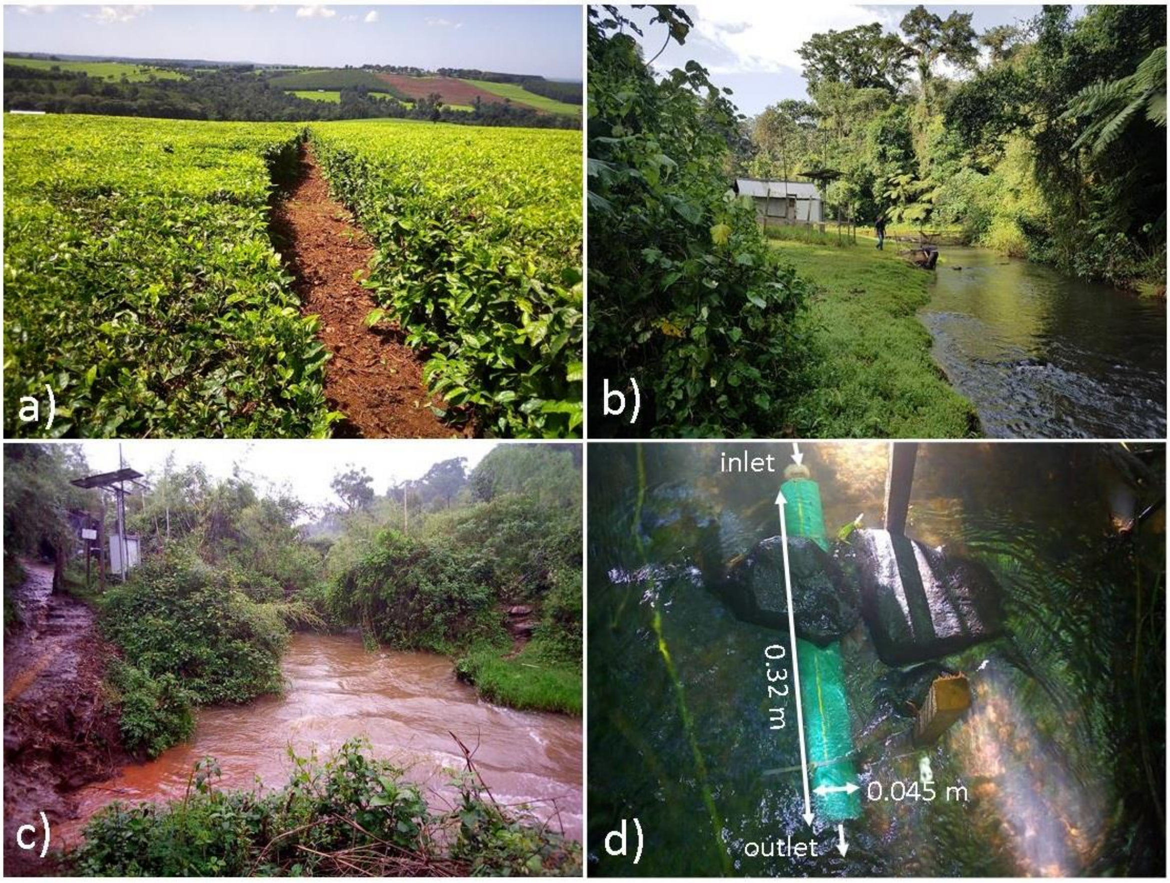


Figure 23 a) Tea-tree plantation catchment, b) outlet of the natural forest and c) outlet of the smallholder agriculture catchment and d) time-integrated sediment trap.

6.3.2 Processing and qualitative analysis of suspended sediment

An aliquot of each sediment sample (>230 mg) was ground using a ball mill grinder for further laboratory analysis. The ground samples were analysed for total carbon (TC), total nitrogen (TN) and total phosphorus (TP) concentrations. In this context, TC, TN and TP refer to the total C, N and P concentration in suspended sediments, which corresponds to particulate C, N and P (expressed in g kg^{-1}). For TC and TN concentration measurements, a sub-sample of 30 mg of ground sediment was wrapped in tin capsules and combusted in an elemental micro-analyser (Elementar vario EL III, Elementar Analysensysteme GmbH, Langenselbold, Germany) at 950°C. For TP concentration measurements, a sub-sample of

200 mg of ground sediment was digested in 4.4 mL of sulphuric acid-hydrogen peroxide digest reagent and heated to 400°C for two hours (Allen *et al.* 1974). After diluting the digestate twice, TP was determined using colorimetry based on a reaction with acidic molybdate in the presence of antimony which forms an antimony-phosphomolybdate complex. Ascorbic acid turns the complex in an intensely blue (phosphomolybdenum blue) which is measured spectrophotometrically at 880 nm in a segmented flow analyser (Auto-analyser 3HQ, SEAL Analytical Ltd., Hampshire, United Kingdom). The remaining sediment samples were analysed for organic matter using gravimetric weight change by loss on ignition at 500°C for 4 hours in a muffle furnace.

To estimate nutrients and C concentrations in stream water (mg L^{-1}), the mean suspended sediment concentrations (mg L^{-1}) for each sampling period were multiplied with the particulate nutrients and C concentrations (g kg^{-1}) per dry weight sediments. For TC, TN and TP load calculations, the mean discharge and suspended sediment concentration was obtained for each sediment sampling period. The concentrations of TC, TN and TP (g kg^{-1}) were multiplied to the mean discharge (m^3 s^{-1}) and mean suspended sediment concentrations (mg L^{-1}) to obtain the sediment-associated loads (t day^{-1}). Annual suspended sediment-associated TC, TN and TP yields were calculated by integrating the mean of the daily loads to annual loads and relating it to the catchment area.

6.3.3 Data analysis

All data were analysed for normality using the Shapiro-Wilk test. Significant differences were tested on particulate TC, TN and TP values among the different land uses between the two years using the Kruskal-Wallis test for analyses of variances. The effect of land use on particulate TC, TN and TP concentrations within each year were tested for significance using the pairwise Wilcoxon rank sum test. To test the linear relationship between two macroelements, the correlation coefficient r was identified between each two of the macroelements. All significant differences reported are at $p<0.05$.

6.4 Results

6.4.1 Hydrological and suspended sediment responses

Mean annual rainfall for the study period (2018-2019) was 1,989, 2,006, and 1,671 mm yr^{-1} for the natural forest, tea-tree plantation and smallholder agriculture catchments, respectively. In the natural forest and the tea-tree plantations, 2019 was wetter than 2018, while the smallholder agriculture catchment was drier in 2019. Mean annual specific discharge was highest in the natural forest catchment followed by the tea-tree plantation and the smallholder agriculture catchments with 806 (778-834) mm yr^{-1}, 678 (642-714) mm yr^{-1}

and 658 (634-682) mm yr^{-1}. The catchment runoff coefficient was highest for the natural forest catchment (0.41) and smaller for the tea-tree plantations and the smallholder agriculture with a mean of 0.34 and 0.39, respectively. The mean annual suspended sediment yield for the two years was highest in the smallholder agriculture with 231 (215-248) t km^{-2}yr^{-1} followed by the natural forest with 57 (54-62) t km^{-2}yr^{-1} and the tea-tree plantation catchments with 48 (44-52) t km^{-2}yr^{-1} (Table 12).

Table 12 Hydrological characteristics and total suspended sediment (TSS) (and 95% confidence interval) of the three catchments under different land use natural forest (NF), tea-tree plantations (TTP) and smallholder agriculture (SHA) in the South-West Mau, Kenya of 2018 and 2019.

Site	Year	Annual rainfall [mm yr^{-1}]	Annual specific discharge [mm yr^{-1}]	Runoff coefficient[a]	TSS load [t yr^{-1}]	TSS yield [t km^{-2} yr^{-1}]
NF	2018	1,881	814 (783-845)	0.43 (0.42-0.45)	1,343 (1,230-1,461)	37 (34-41)
	2019	2,098	798 (772-823)	0.38 (0.37-0.39)	2,730 (1,925-2,234)	76 (73-84)
	Mean	1,989	806 (778-834)	0.41 (0.39-0.42)	2,037 (1,925-2,234)	57 (54-62)
TTP	2018[b]	1,922	677 (637-717)	0.35 (0.33-0.37)	1,067 (972-1,167)	32 (29-35)
	2019	2,089	679 (647-711)	0.32 (0.31-0.34)	2,105 (1,933-2,284)	63 (58-69)
	Mean	2,006	678 (642-714)	0.34 (0.32-0.36)	1,586 (1,453-1,725)	48 (44-52)
SHA	2018	1,870	942 (909-974)	0.50 (0.49-0.52)	5,244 (4,872-5,629)	193 (179-207)
	2019[c]	1,473	375 (359-390)	0.25 (0.24-0.27)	7,336 (6,835-7,854)	270 (251-289)
	Mean	1,671	658 (634-682)	0.39 (0.38-0.41)	6,290 (5,853-6,741)	231 (215-248)

[a]Specific discharge as proportion of annual rainfall
[b]Gaps in discharge and suspended sediment data due to sensor malfunctioning
[c]Gaps in discharge and suspended sediment data due to theft of power supply

In all three catchments, discharge followed the rainfall pattern. The rising limb of the hydrograph was generally steep, followed by either steep or gentle falling limbs, depending on the magnitude of the storm event. Discharge peaked during the long rainy season between April and July in 2018. In contrast, 2019 experienced a delayed onset of the rains and the highest discharge peaks occurred between October and December. Suspended sediment peaks followed the same pattern as discharge (Figure 24).

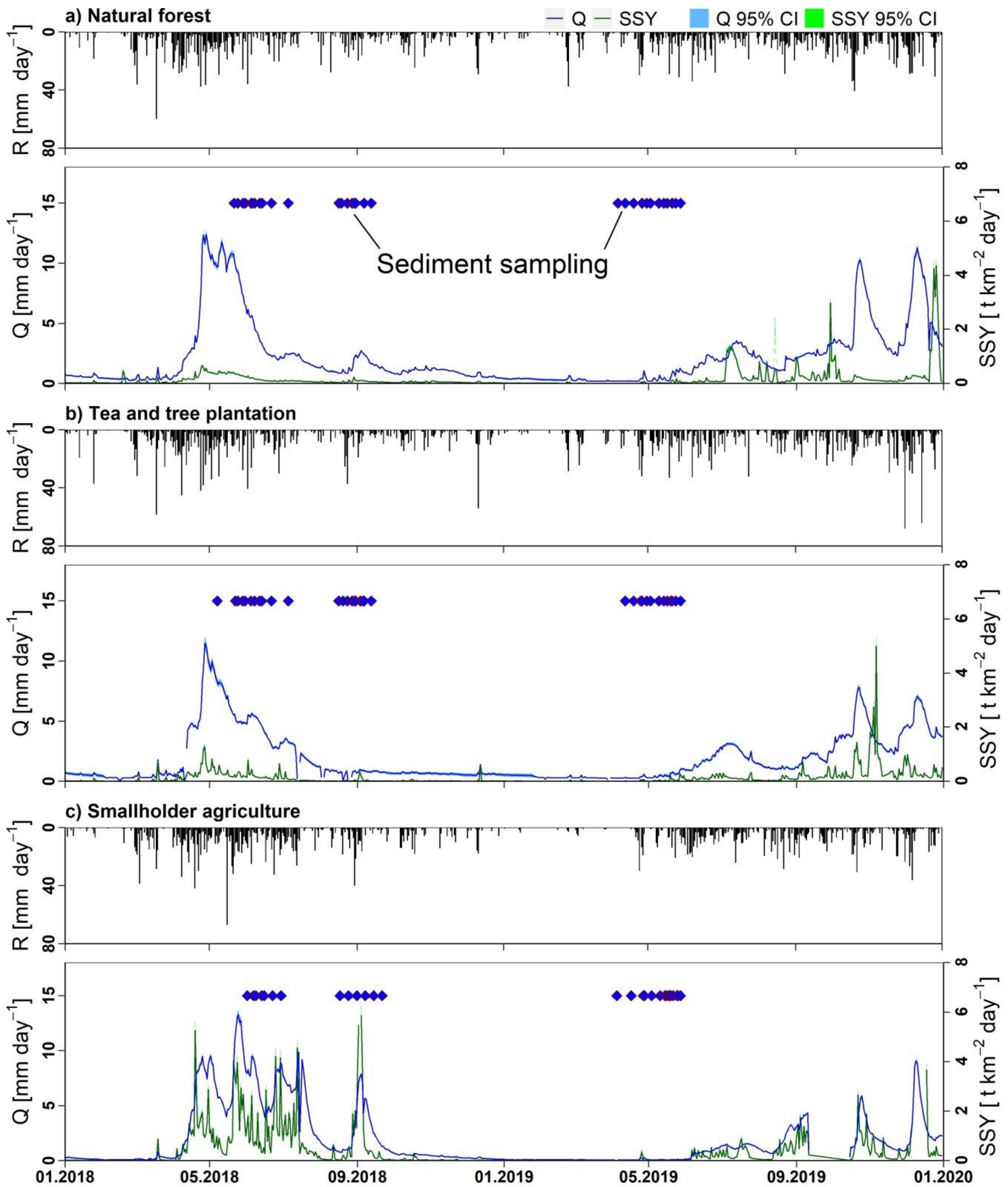


Figure 24 Daily accumulated rainfall (R) [mm day^{-1}], specific discharge (Q) [mm day^{-1}] and suspended sediment yield (SSY) [t km^{-2} day^{-1}] aggregated from 10 minute resolution with 95% confidence interval (CI) of the a) natural forest, b) smallholder agriculture and c) tea-tree plantation catchments in the South-West Mau, Kenya between January 2018 and December 2019.

6.4.2 Macronutrient concentrations on sediment and in the stream

Suspended sediment sampling for the particulate TC, TN and TP analysis was conducted during the high flows in 2018 (May-September), while in 2019 the sampling coincided with the start of the long rainy season during April and May (Figure 24). The TC concentrations were 7% lower in the natural forest catchment and 15% higher in the smallholder agriculture in 2019 than in the previous year. The TN concentrations were 8-20% higher in 2019 than in

2018 for all three catchments. In the natural forest and the tea-tree plantation catchments, the TP concentrations were 25% and 17% higher, respectively, and 38% lower in the smallholder agriculture catchment in 2019 than in 2018 (Figure 25).

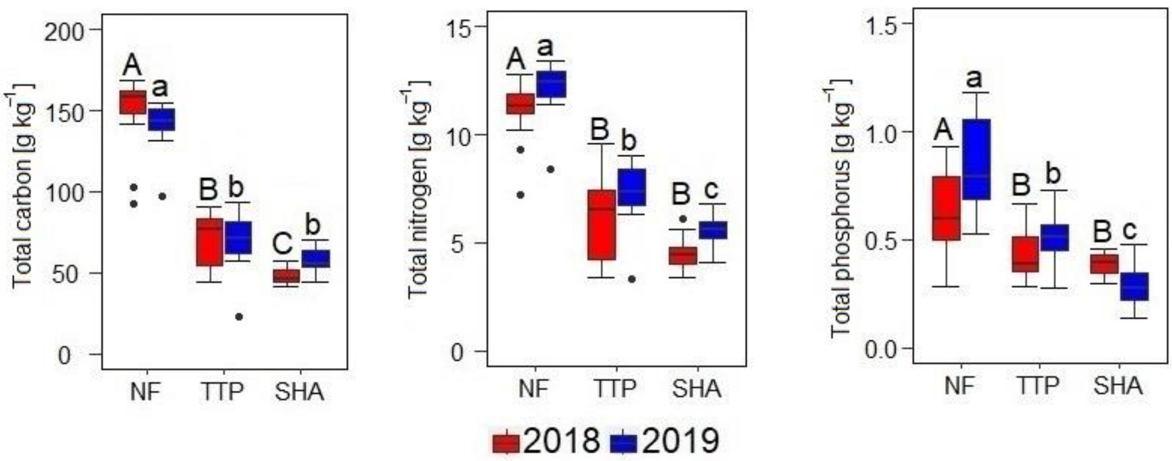


Figure 25 Particulate total carbon, nitrogen and phosphorus concentrations [g kg sediment^{-1}] of time-integrated, manual and automatic collected samples of the natural forest (NF), tea-tree plantation (TTP) and smallholder agriculture (SHA) catchments in the South-West Mau, Kenya in 2018 and 2019. Different letters indicate significant differences between land uses ($p<0.05$).

Particulate TC, TN and TP concentrations were significantly higher in the natural forest catchment than in the tea-tree plantation and smallholder agriculture catchments in both years. The TC concentrations were significantly higher in the tea-tree plantations than in the smallholder agriculture in 2018, while there was no difference in 2019. TN and TP concentrations were significantly higher in the tea-tree plantation than in the smallholder agriculture catchment in 2019, but concentrations were not significantly different between the tea-tree plantations and the smallholder agriculture in 2018 (Table 13).

Table 13 Mean ± standard deviation of particulate TC, TN and TP concentrations [g kg sediment^{-1}] of time-integrated, manual and automatically collected samples, suspended sediment concentrations (TSS) [mg L^{-1}] and TC, TN [mg L^{-1}] and TP [µg L^{-1}] concentrations in water at the outlet of the natural forest (NF), tea-tree plantations (TTP) and the smallholder agriculture (SHA) in the South-West Mau, Kenya based on 13-22 sampling days for the sampling campaign from May-October 2018 and 14-18 sampling days for the period April-June 2019.

		Concentrations in sediment			**Concentrations in stream water**			
Site	**Year**	**TC [g kg^{-1}]**	**TN [g kg^{-1}]**	**TP [g kg^{-1}]**	**TSS [mg L^{-1}]**	**TC [mg L^{-1}]**	**TN [mg L^{-1}]**	**TP [µg L^{-1}]**
NF	2018[a]	151.0±20.5	11.2±1.3	0.6±0.2	43.6±10.9	6.6±1.8	0.5±0.1	28.5±13.9
	2019[b]	141.4±14.6	12.2±1.3	0.9±0.2	43.4±22.0	6.0±2.6	0.5±0.2	36.2±20.5
	Mean	146.2±17.6	11.7±1.3	0.7±0.2	43.5±16.4	6.3±2.2	0.5±0.2	32.4±17.2
TTP	2018[a]	69.2±16.9	6.1±1.8	0.4±0.1	50.8±48.1	3.6±4.3	0.3±0.3	21.9±23.8
	2019[b]	69.5±17.3	7.3±1.5	0.5±0.1	104.7±127.3	6.3±6.9	0.7±0.7	43.4±36.9
	Mean	69.4±17.1	6.7±1.6	0.5±0.1	77.8±87.7	5.0±5.6	0.5±0.5	32.7±30.4
SHA	2018[a]	48.1±4.9	4.5±0.8	0.4±0.1	188.0±151.7	8.7±6.4	0.8±0.6	70.6±52.0
	2019[b]	56.9±7.7	5.6±0.6	0.3±0.1	231.4±441.0	11.9±22.9	1.2±2.5	63.1±123.1
	Mean	52.5±6.3	5.0±0.7	0.3±0.1	209.7±296.4	10.3±14.7	1.0±1.5	66.8±87.6

[a]wet period May-October 2018
[b]drier period April-June 2019

The mean TC, TN and TP concentrations in the stream water for both years, estimated based on the particulate macronutrient concentrations and suspended sediment concentrations, were highest for the smallholder agriculture and lowest for the natural forest catchment and tea-tree plantations (Table 13).

The natural forest catchment had the highest percentage of organic matter in suspended sediments with 31%, followed by the tea-tree plantations with 24% and the lowest percentage was measured in the smallholder agriculture catchment with 16% (Table 10).

6.4.3 Stoichiometric macronutrient ratios and their relationships

The C:N ratio in the sediment from natural forest (12.6±1.0) was significantly higher ($p<0.05$) than those of the tea-tree plantations (10.5±1.2) and the smallholder agriculture (10.6±1.3) for both years, while the C:N ratio was not significantly different between the tea-tree plantations and the smallholder agriculture ($p>0.05$). The C:P ratio in the natural forest (269.8±106.4) was significantly higher than the tea-tree plantations (160.5±39.2) and the smallholder agriculture (126.7±22.6) in 2018, but the C:P ratio was not significantly different between the three catchments in 2019. The N:P ratio was significantly higher in the natural forest (19.9±6.9) than the smallholder agriculture (11.8±3.0) in 2018, while 2019 showed no significant difference between the three catchments ($p>0.05$). The C:N ratio was significantly higher in 2018 than in 2019 in the natural forest and the tea-tree plantations, while the C:P and N:P ratio was significantly higher in 2019 than 2018 in the smallholder agriculture (Figure 26). The sediment-associated C:N:P ratio was highest in the natural forest catchment with 225 C: 17N: 1P, followed by the smallholder agriculture with 172 C: 16N: 1P and the tea-tree plantations with 148 C:14 N: 1P, whereas the N:P ratio was similar for all three catchments.

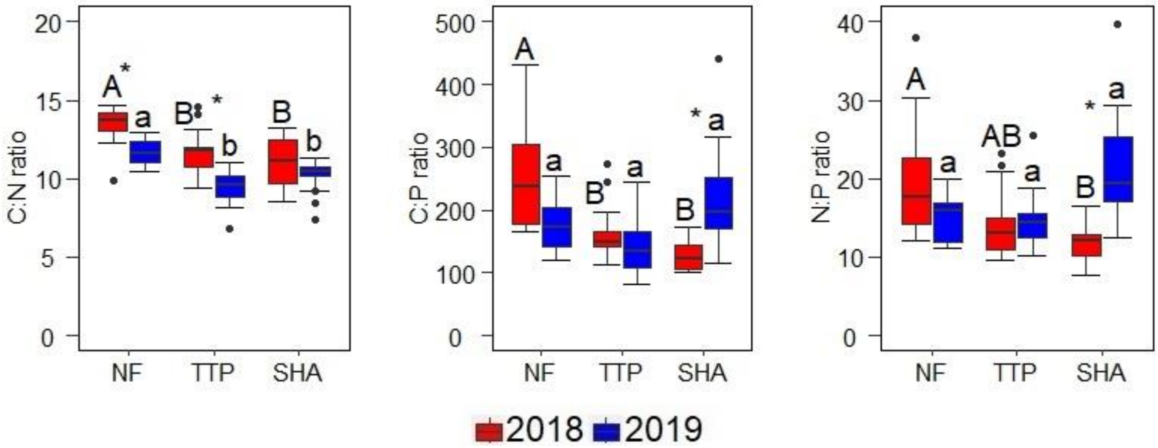


Figure 26 Macronutrient ratios for carbon-nitrogen (C:N), carbon-phosphorus (C:P) and nitrogen-phosphorus (N:P) of the natural forest (NF), tea-tree plantation (TTP) and smallholder agriculture (SHA) catchments in the South-West Mau, Kenya. Different letters indicate significant differences between catchments and the asterisk indicates significant differences within one catchment between years ($p<0.05$).

TC, TN and TP concentrations were correlated against each other for all three catchments (Figure 27). The strongest relationship was observed between TC and TN within the tea-tree plantations (r=0.87), followed by the smallholder agriculture (r=0.71) and the natural forest catchment (r=0.64). The correlation between TN and TP was strongest in the tea-tree plantations (r=0.58). No significant relationships were found between TC and TP in the natural forest and in the smallholder agriculture catchment with low correlation coefficient values ranging between -0.15 and 0.04, and between the correlation of TN and TP in the smallholder agriculture (r=-0.21) ($p>0.05$). The correlations of the natural forest catchment were in general higher than those of the tea-tree plantations and the smallholder agriculture (Figure 27).

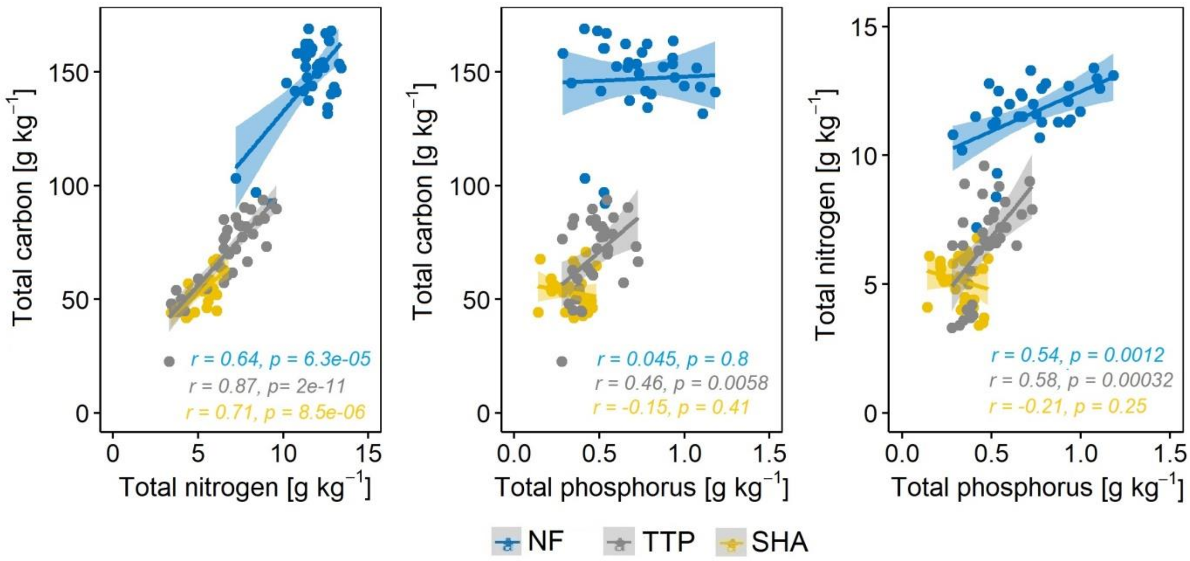


Figure 27 Correlations with correlation coefficient r obtained between total carbon, nitrogen and phosphorus [g kg sediment^{-1}] concentrations of the natural forest (NF), the tea-tree plantation catchments (TTP) and the smallholder agriculture (SHA) in the South-West Mau, Kenya in 2018 and 2019. Significant difference at $p<0.05$.

6.4.4 Sediment-associated macronutrient loads

The mean daily suspended sediment loads for the sampling period were highest in the smallholder agriculture, followed by the natural forest and lowest in the tea-tree plantations (14.6±15.0 t day^{-1}, 4.9±2.5 t day^{-1} and 3.4±3.9 t day^{-1}, respectively). The mean daily TC and TN load in suspended sediment during the sampling periods in 2018 and 2019 was highest for the smallholder agriculture (668.9±661.9 kg day^{-1} and 60.5±59.5 kg day^{-1}), followed by the natural forest (496.9±335.3 kg day^{-1} and 37.2±20.9 kg day^{-1}) and the tea-tree plantation catchment (193.4±178.6 kg day^{-1} and 18.1±18.5 kg day^{-1}). For the sediment-associated TP, the highest daily loads were observed in the smallholder agriculture followed by the natural forest and the tea-tree plantations (5.5±5.1 kg day^{-1}, 1.9±1.3 kg day^{-1}, and 1.4±1.3 kg day^{-1}, respectively). The mean annual sediment-associated TC, TN and TP yields of the sampling period is estimated to be highest for the smallholder agriculture (9.0±8.9 t TC km^{-2} yr^{-1}, 0.8±0.8 t TN km^{-2} yr^{-1} and 73.9±68.9 kg TP km^{-2} yr^{-1}), followed by the natural forest and lowest for the tea-tree plantations (2.1±2.0 t TC km^{-2} yr^{-1}, 0.2±0.2 t TN km^{-2} yr^{-1} and 14.9±13.6 kg TP km^{-2} yr^{-1}) (Table 14).

Table 14 Overview of total suspended sediment-associated (TSS) total carbon (TC), total nitrogen (TN) [t km^{-2} yr^{-1}] and total phosphorus (TP) yields [kg km^{-2} yr^{-1}] and total suspended sediment yields [t km^{-2} yr^{-1}] based on 13-22 sampling days for the sampling campaign from May-October 2018 and 14-18 sampling days for the period April-June 2019.

Site	Year	TSS TC yield [t km^{-2} yr^{-1}]	TSS TN yield [t km^{-2} yr^{-1}]	TSS TP yield [kg km^{-2} yr^{-1}]	TSS yield [t km^{-2} yr^{-1}]
NF	2018 wet period	9.0±5.6	0.7±0.4	32.1±17.4	59.3±41.1
	2019 drier period	1.2±1.2	0.1±0.1	7.0±8.6	8.5±9.3
	Mean	5.1±3.4	0.4±0.2	19.6±13.0	50.2±25.2
TTP	2018 wet period	3.0±2.6	0.3±0.2	20.5±15.5	51.3±44.4
	2019 drier period	1.2±1.3	0.1±0.2	9.3±11.7	24.0±41.9
	Mean	2.1±2.0	0.2±0.2	14.9±13.6	37.7±43.2
SHA	2018 wet period	17.5±16.9	1.6±1.5	145.6±133.2	383.7±385.6
	2019 drier period	0.4±0.8	0.05±0.1	2.3±4.5	8.2±15.9
	Mean	9.0±8.9	0.8±0.8	73.9±68.9	196.0±200.8

6.5 Discussion

6.5.1 Land use affects sediment-associated carbon and nutrient concentrations

This study shows that land use is a key control of suspended sediment-associated TC, TN and TP concentrations in the headwaters of the Sondu River Basin. Most significantly, the natural forest catchment has much higher particulate TC, TN and TP concentrations in contrast to the agricultural catchments. These results refute the hypothesis that particulate-bound TC, TN and TP concentrations per dry weight sediment are highest in agricultural catchments with the majority of suspended sediment (77%) originating from agricultural land (Stenfert Kroese *et al.* 2020a).

The TC concentrations of the natural forest catchment were more than twice the concentrations recorded in the Congo basin under disturbed and undisturbed forest cover (14-95%) (Coynel *et al.* 2005) and of disturbed agricultural catchments in temperate regions (Walling *et al.* 2001; López-Tarazón *et al.* 2016), while the smallholder agricultural and tea-tree plantation catchments of this study fell within the range of the concentrations reported (Table 15). Other catchments of disturbed river basins under mixed land use (Tana River basin, Kenya and Ayeyarwady and Thanlwin River, Myanmar) had lower TC concentrations (Bird *et al.* 2008; Tamooh *et al.* 2012) than the concentrations of all three catchments of this study.

The TN concentrations of the natural forest catchment were in the same range as those of sub-catchments under mixed land use of the Yangtze River, Jialing River and Wujian River, China, during the wet season, but exceeded the concentrations of intensified agricultural catchments in the USA, New Zealand and Spain by up to 12-fold (McDaniel *et al.* 2009; McDowell 2015; López-Tarazón *et al.* 2016). The TN concentrations of the agricultural catchments of this study were within the ranges of concentrations in intensified temperate agricultural catchments (Walling *et al.* 2001; Pavanelli & Selli 2013) (Table 15).

The particulate TP concentrations of the natural forest, tea-tree plantations and smallholder agriculture catchments were lower compared to P-saturated agricultural catchments of temperate regions (Neal *et al.* 2006; Ramos *et al.* 2015; Sandström *et al.* 2020) or of sub-catchments in China during the wet season (Wang *et al.* 2015) (Table 15). A tobacco cultivated catchment in sub-tropical Brazil, with a TP range of 0.09-3.58 g P kg^{-1}, with a similar mean annual rainfall of 1,938 mm yr^{-1} (Bender *et al.* 2018) exceeded the concentrations of this study, most likely due to the high fertilizer application rates.

In contrast to this study with depleted nutrients in agricultural catchments, agricultural land use and catchment disturbance were directly correlated with increased nutrient concentrations in sediment, where usually lower TP concentrations were observed in forested (Haggard *et al.* 2007) and less agriculturally intensive catchments in the UK (Palmer-Felgate *et al.* 2009). The present results also differ from a study in the Fox River Watershed, USA, where the historical accumulation of legacy P in sediments led to no relationship between TP concentrations and land use or management activities (Kreiling *et al.* 2019).

Enriched TC, TN and TP concentrations were found in suspended sediment with a higher percentage of the coarser fraction (60 µm-2 mm) from the natural forest catchment (Table 10), while the smallholder agriculture catchment with the finer suspended sediment showed lower TC, TN and TP concentrations. This is in contrast to the study by Walling et al. (2001), where the catchment with the finest material had higher nutrient concentrations. This enrichment of fine particle fraction has also been observed in other studies (Uusitalo *et al.* 2001; Kreiling *et al.* 2019; Sandström *et al.* 2020). The Severn catchment (in England and Wales) dominated by arable cultivation had unexpectedly low nutrient concentrations, compared to three other South-Western British rivers with a smaller coverage in arable land. The lower nutrient concentrations were explained by the higher amount of coarser material and an upland area characterized by soils with lower nutrient concentrations (Walling *et al.* 2001). The high nutrient concentrations of the natural forest catchment of this study might be associated with the nutrient-rich soils and biologically fresh material contributing largely to the sediment despite the coarser material.

Despite the application of inorganic fertilizers on agricultural land within the tea-tree plantation and the smallholder agriculture catchments, the sediment-associated nutrient concentrations measured were one to three-fold lower than the natural forest catchment. The tea-tree plantation catchment receives higher fertilizer inputs compared to the smallholder agriculture catchment explaining the slightly higher nutrient concentrations in sediments.

Table 15 Overview of particulate total carbon (TC), total nitrogen (TN) and total phosphorus (TP) mean concentrations [g kg^{-1}] of catchment studies around the world. Analysed sample material: SS = suspended sediment, BS = riverbed sediment and WS = water sample. DR Congo = Democratic Republic of Congo, TRPR = Tana River Primate Reserve.

Catchment/ basin	Country	Area [km^2]	Land use	Sample	Study period [year][b]	Rainfall [mm]	TC [g kg^{-1}]	TN [g kg^{-1}]	TP [g kg^{-1}]	Reference
					Tropical catchments					
Sondu basin	Kenya	36	Forest	SS	2018-2019	1,989	147.61	12.06	0.81	This study
Sondu basin	Kenya	33	Agriculture	SS	2018-2019	2,006	83.72	8.00	0.54	This study
Sondu basin	Kenya	27	Agriculture	SS	2018-2019	1,671	53.29	5.02	0.33	This study
Kora (Tana River)	Kenya	22,080	Mixed land use	SS	2009-2011	450-900	25.32[c]	n.a.	n.a.	Tamooh et al. (2014)
Garissa (Tana River)	Kenya	32,500	Mixed land use	SS	2009-2011	450-900	19.92[c]	n.a.	n.a.	
TRPR (Tana River)	Kenya	66,500	Mixed land use	SS	2009-2011	450-900	17.29[c]	n.a.	n.a.	
Oubangui (Congo basin)	DR Congo	489,000	Forest (22%)	SS	1990-1996	1,550	60.61	n.a.	n.a.	Coynel et al. (2005)
Mpoko (Congo basin)	DR Congo	23,900	Forest (14%)	SS	1991-1994	1,550	41.67	n.a.	n.a.	
Ngoko-Sangha (Congo basin)	DR Congo	67,000	Forest (95%)	SS	1991	1,550	61.17	n.a.	n.a.	
Congo/Zaire (Congo basin)	DR Congo	3,500,000	Forest (50%)	SS	1990-1993	1,550	64.64	n.a.	n.a.	
Arroio Lajeado Ferreira	Brazil	1.2	Mixed land use	WS	2011-2015	1,938	n.a.	n.a.	1.22	Bender et al. (2018)
Streams in New Zealand	New Zealand	<20,000	Agriculture	BS	2012 (02-03)	n.a.	2.10	0.21	0.42	McDowell (2015)
Ayeyarwady and Thanlwin River	Myanmar	n.a.	Mixed land use	SS	2006 (05,08,09)	~3,000	14.65[c]	n.a.	n.a.	Bird et al. (2008)
					Temperate and Mediterranean catchments					
Duck Creek, Fox River, Wolf River	USA	<9,666	Mixed land use	BS	2016-2017	748-800	n.a.	n.a.	0.47[d]	Kreiling et al. (2019)
Embarras basin, Vermilion basin	USA	<839	Agriculture	BS	2004	1,270	n.a.	0.34[d]	0.36[d]	McDaniel et al. (2009)
Wye, Welland, Avon	UK	0.4-9.9	Agriculture (more intensive)	BS	2005-2006	671-905	n.a.	n.a.	1.56	Palmer-Felgate et al. (2009)
Wye, Avon	UK	0.4-9.9	Agriculture (less intensive)	BS	2005-2006	671-905	n.a.	n.a.	0.63	
Severn, Avon, Eve, Dart	UK	<6,850	Agriculture	SS	1995-1996	600-2,300	53.70[c]	4.47	1.49	Walling et al. (2001)
Upper Thames River	UK	3,500	Mixed land use	WS	1997 (1-6 years)	n.a.	n.a.	n.a.	2.16-6.87	Neal et al. (2006)
Enxoé River	Portugal	61	Agriculture	SS	2010-2013	500	n.a.	n.a.	4.30	Ramos et al. (2015)
Ésera basin	Spain	1,484	Mixed land use	SS	2011-2012	1,069	60.81	0.80	n.a.	López-Tarazón et al. (2016)
Reno River	Italy	389	Agriculture	SS	2000-2009	950-	n.a.	9.50	0.46	Pavanelli and

					(flood events)	1,015				Selli (2013)
South Sweden	Sweden	<33.1	Agriculture (54-59%)	WS	2004-2017	539-623	n.a.	n.a.	1.07	Sandström et
		<16.3	Agriculture (89-93%)	WS	2004-2017	506-709	n.a.	n.a.	1.78	al. (2020)
Yangtze River (sub-	China	n.a.	Mixed land use	SS	2010 (05-10)	n.a.	n.a.	1.23	0.54	Wang et al.
catchment)					2011 (11-04)	n.a.	n.a.	11.36	4.09	(2015)
Jialing River (sub-	China	n.a.	Mixed land use	SS	2010 (05-10)	n.a.	n.a.	3.29	0.55	
catchment)					2011 (11-04)	n.a.	n.a.	22.50	5.00	
Wujian River (sub-	China	n.a.	Mixed land use	SS	2010 (05-10)	n.a.	n.a.	4.09	0.86	
catchment)					2011 (11-04)	n.a.	n.a.	18.18	4.55	

n.a. = not available
[b]month of the year
[c]particle organic carbon
[d]concentrations presented in median

Even though particulate concentrations in sediments were the highest in the natural forest catchment, the sediment-associated loads and the concentrations of particulate TC, TN and TP in the stream water were highest in the smallholder agriculture due to the overall higher suspended sediment concentrations (Table 14). These higher loads and in-stream concentrations in the smallholder agriculture catchment agree with a study conducted in the same three catchments by Jacobs et al. (2018) revealing the highest dissolved nitrate loads in the agricultural compared to the forested streams.

Our sediment-associated TC, TN and TP yield estimations may be underestimated due to the sampling of the drier start of the long rainy season in 2019. In addition, the short sampling period in both years might have resulted in a higher uncertainty range. For an improved understanding of the particulate TC, TN and TP fluxes, an increased sampling frequency during the long rains, would give additional temporal information.

6.5.2 Suspended sediments from organic and mineral origin

The low TC and nutrient concentrations in sediments from agricultural soils are associated with soils critically lower in nutrients and organic matter compared to nutrient-rich forest soils. The native forest vegetation was converted to smallholder farms during the last few decades (Brandt *et al.* 2018). This conversion leads to reduced organic inputs and increased decomposition rates of soil organic matter following physical and chemical management practices (Freibauer *et al.* 2004). Globally, an average decrease in soil TC of around 30% was found for soils after conversion of forests to croplands (Murty *et al.* 2002; Don *et al.* 2011), while a study of a converted native forest in the highlands of Western Kenya showed a decrease by 30-40% within the first 39 years after conversion (Nyberg *et al.* 2012). Similarly, a decline in soil organic carbon and nutrients was observed following conversion to agricultural cultivation in the same catchments of the Mau Forest Complex (Were *et al.* 2016; Arias-Navarro *et al.* 2017b; Owuor *et al.* 2018; Wanyama *et al.* 2018). The highest TC and TN concentrations were measured in the surface soil (0-0.05 m) of the natural forest catchment (81.1±24.2 g kg^{-1} and 4.9±2.3 g kg^{-1}, respectively), compared to lowest concentrations on croplands in the smallholder agriculture (56.9±11.1 g kg^{-1} and 2.1±1.2 g kg^{-1}, respectively) (Owuor *et al.* 2018). Chiti et al. (2018) found a significant decline in soil organic carbon as a consequence of forest degradation in the same tropical montane forest. In their study, the organic horizon, the litter layer, had the highest TC concentration under primary forest (412.3±23.2 g kg^{-1}) compared to a degraded forest (408.2±21.3 g kg^{-1}) and to cypress (398.6±19.6 g kg^{-1}) and tea plantations (381.7±17.6 g kg^{-1}) that replaced the forest. With the reduction in the organic horizon a decline in soil TC was found in the uppermost mineral soil layer (<0.05 m) (Chiti *et al.* 2018). This implies that suspended sediment is of mineral origin in the agricultural catchments and of organic origin in the natural forest catchment, which is also reflected in the differences in organic matter and the C concentrations in suspended sediment (Table 10 and Table 13).

The higher C:N:P ratios and the higher relationship between TC and TN in the natural forest catchment (Figure 27) further imply due to a tighter nutrient cycle, that TN is of organic origin, where a lower and similar relationship in both agricultural catchments suggests mineral sediment sources. Similarly, the study of Johnson et al. (2017) in a forested catchment in the Piedmont region, Maryland (USA) showed that sediment sources from forest floor litter had the highest TC and TN concentrations compared to near-stream sources such as stream bed and stream banks. The fresher material from the forest floor was likely the least degraded resulting in higher TC and nutrient concentrations (Johnson *et al.* 2018). Old native forest catchments maintain a tight nutrient cycle through higher surface biodiversity and biomass. They accumulate organic matter and nutrients by decomposition of fresh litter material and humus compared to managed land use types (Dawson & Smith 2007). A strong relationship was observed between TC and TP for the tea-tree plantation catchment, which may indicate that the mineralization of soil organic matter contributes to the available TP concentration, as it was observed in the study of Maranguit et al. (2017).

Particulate nutrients, in particular TP with a high affinity to amorphous Fe, can be stored at downstream reaches. However, deposited sediment can turn to a nutrient source in dissolved form. This process might inevitably increase the risk of eutrophication at downstream reaches. Therefore, soil stability through soil organic matter should be improved to reduce soil erosion and suspended sediments to constrain the loss of soil organic carbon and nutrients. Soil management practices should be focused on the retention of soil organic matter through boosting agricultural productivity, mulching or cover crops in the agricultural catchments to maintain soil fertility. Increased crop residues create a positive feedback loop for the accumulation of organic matter (Nyberg *et al.* 2012). Other management practices such as vegetative buffer strips, erosion ditches or *fanya juu* terracing can prevent soil erosion (Tiffen *et al.* 1994; Conelly & Chaiken 2000).

6.5.3 Historical nutrient losses on agricultural soils – impoverished agricultural soils

The low sediment-bound concentrations (TC, TN and TP) in the smallholder agriculture catchment reflect soils low in organic matter and nutrients due to intensive cropping without appropriate practices for restoration of the soil fertility. In addition to the decline in organic matter upon conversion, soils cultivated on steep hillslopes with poor soil conservation strategies and excessive surface runoff are prone to erosion and further nutrient depletion (Stenfert Kroese *et al.* 2020b). This may have been intensified by the cultivation of a major nutrient miner, *Pyrethrum* daisy (*Chrysanthemum cinerariifolium*), following the clearance of the South-West Mau, which is known to promote surface soil erosion, due to poor soil cover (Smaling *et al.* 1993). Similar soil nutrient deficits have been observed in other densely populated tropical agricultural regions cultivated on steep hillslopes in Uganda (Lederer *et al.* 2015), Tigray (Ethiopia) (Girmay *et al.* 2009) or Kisii District (Western Kenya) (Smaling *et al.* 1993) through aggregated nutrient losses, caused in particular by surface erosion and insufficient compensation by fertilizer application. Scarcity of land leads to an overexploitation of soil nutrients on agricultural land due to the absence of fallow periods.

6.5.4 Surface and subsurface sediment sources

In addition to soils low in organic matter and nutrient losses through land use change and historical cultivation, the lowered concentrations of sediment C, N and P in the smallholder agriculture catchment may be explained by sediment originating from the subsurface. Sediment C and nutrient concentrations are often lower in the subsurface (Russell *et al.* 2001; Gellis *et al.* 2009; Wanyama *et al.* 2018). The sediment fingerprinting study of Stenfert Kroese et al. (2020a) demonstrated that the subsoil sources are of increased importance in the smallholder agriculture compared to the natural forest and the tea-tree plantation catchment due to exposure of subsoil to erosion processes.

6.6 Conclusions

Catchments in the headwaters of the Sondu River Basin of the South-West Mau, Kenya, show that cultivated land uses have led to a pronounced decline in sediment-associated TC, TN and TP concentrations compared to native forest ecosystems. The high C:N ratio in the natural forest catchment reveals that most particulate TN is organic matter based. The native forest is rich in biomass, with a tighter nutrient cycle and likely high soil organic matter input. This is in contrast to the smaller nutrient stocks in the agricultural land, reflecting the impoverished agricultural soils due to its low organic matter content. The disturbance of soil through agricultural practices increases soil mineralization of soil organic matter, echoed in the lower organic matter content in suspended sediments. Despite the lower sediment-associated TC, TN and TP concentrations, the smallholder agriculture had the greatest particulate TC, TN and TP yields due to the higher export of suspended sediment yields passing the outlet. Elevated TC, TN and TP concentrations in sediments and in stream water contribute to nutrient pollution and eutrophication impacting the downstream reaches such as Lake Victoria. Management practices should focus on restoring soil organic matter of agricultural topsoils to increase soil organic carbon, soil nutrients and improve agricultural productivity. Practices to manage soil organic matter should also control soil erosion and consequently reduce suspended sediment concentrations and the loss of nutrient-rich topsoil.

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7 Discussion and conclusions

Africa is projected as the continent with the highest soil erosion rates based on land use change modelling (Borrelli *et al.* 2017). The heterogeneous and fragmented landscapes of tropical montane headwaters in East Africa are susceptible to soil erosion due to cultivation on fragile hillslopes coinciding with rainfall of high erosivity. This discussion provides a summary of the key findings of this thesis with an evaluation of the limitations encountered during the course of the research and a perspective on implications for future research. A reflection is drawn on to the importance of the findings for rural farmers, challenges of the application of soil conservation measures and the future of agricultural productivity of soils experiencing severe erosion in East African highlands.

7.1 Summary of key findings

The individual chapters of this thesis contributed to filling the knowledge gaps on the impact of land use on suspended sediment and their particulate macronutrient dynamics and hydrological flow pathways in headwaters of the Sondu River Basin, in Kenya.

Temporal and spatial variability of suspended sediment and hydrology

This thesis explored the relationship between rainfall, runoff and suspended sediment transport dynamics and sediment yield, and the temporal dynamics of sediment supply (Chapter 4). The analysis uses a four-year high-frequency dataset of hydrology and suspended sediment obtained during the observation period between 2015 and 2018 to explore these complex relationships. One of the main findings is that the smallholder agricultural catchment generated six times more suspended sediment yield (131.5±90.6 t km^{-2} yr^{-1}) than the forested catchment (21.5±11.1 t km^{-2} yr^{-1}), which was more than three times the yield of the catchment under tea and tree cultivation (42.0±21.0 t km^{-2} yr^{-1}). Putting these results into the wider context, the annual sediment yield was still at the lower end of the reported ranges for other sediment studies in Kenya (8.2-6,330 t km^{-2} yr^{-1}) and other tropical montane catchments (140-2,470 t km^{-2} yr^{-1}) (Vanmaercke *et al.* 2010; Guzman *et al.* 2013; Didoné *et al.* 2014). All three catchments showed pronounced seasonal variability, where the long rains (March-April) contributed about half of the sediment yield (45-52%). The dry season (January-February) contributed with only 5% to the sediment yield. Moreover, further evidence was presented on the extent to which hydrological flow pathways control suspended sediment variability. It was demonstrated that conversion of forests to agriculture leads to a shift in the dominant water pathways from deep subsurface to shallow subsurface flow and surface runoff. In agricultural land, surface runoff dominates soil erosion processes and therefore, suspended sediment generation. Lastly, this chapter presents evidence for delayed sediment responses observed in the smallholder agriculture catchment, in contrast to fast responses in the forested catchment. This delayed sediment response to rainfall and a slow depletion in sediment supply in both agricultural catchments suggest that a range of sediment sources from the wider catchment area supplies sediment. In contrast, the fast depletion of sediment supply in the natural forest suggests the importance of nearby sediment sources and temporarily stored sediment. The dominant water pathways (blue arrows) and sediment (yellow arrows) pathways in the three different catchments are displayed in Figure 28. The densely vegetated land cover and riparian zone in the natural forest catchment prevents soil erosion with only localized source areas supplying sediment to the stream network. In the tea-tree plantation catchment, only temporarily bare surfaces such as logged areas or new established plantations are thought to be the main sediment contributors. However, a riparian zone throughout the catchment buffers the stream and a well-maintained drainage network reduces sediment supply. In contrast, the smallholder agriculture catchment has several contributing sources areas from unpaved trackways, gullies, agricultural land to bare channel banks, increasing sediment generation at the catchment's outlet. In addition to being a source area, gullies and unpaved trackways are thought to act as conduits between hillslopes and the stream network.

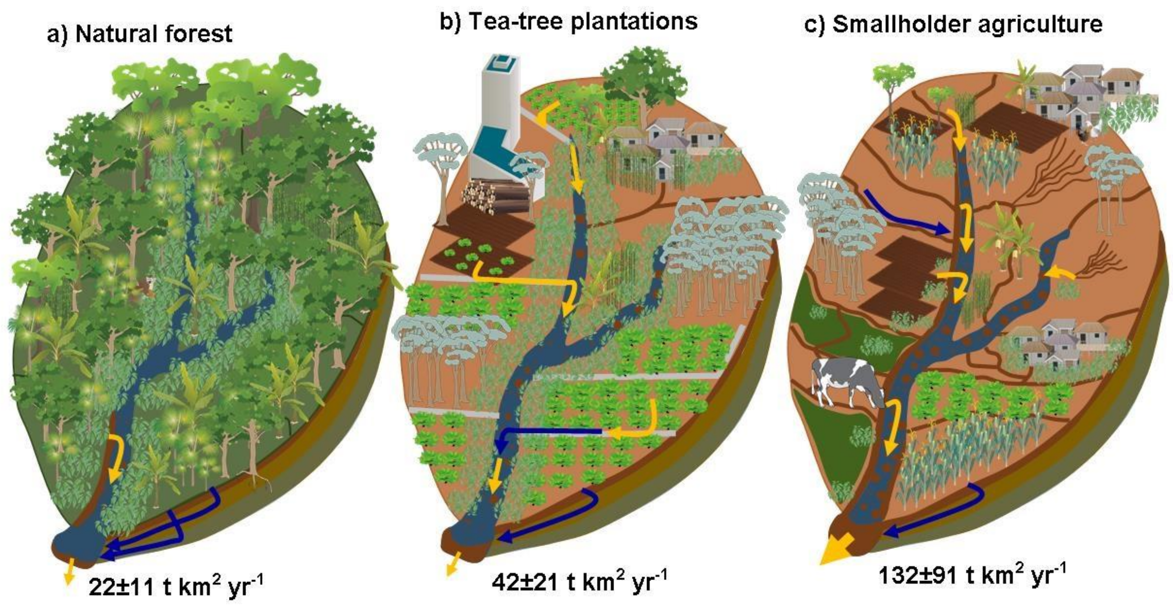

Figure 28 Conceptual model of dominant water (blue arrows) and sediment (yellow arrows) pathways originating from different sediment source areas with annual suspended sediment yield at the outlet of the a) natural forest, b) tea-tree plantation and c) smallholder agriculture catchment in the South-West Mau, Kenya.

Tracer elements and sediment source areas

Chapter 5 identifies the contributions of four sediment source areas in the smallholder agriculture catchment based on the best statistically selected sediment tracer composite. The work presented in Chapter 4 led to the hypothesis that unpaved tracks are the main source areas of the six times higher observed suspended sediment yield in the smallholder agriculture catchment. Therefore, the work under Chapter 5 focused on the smallholder agriculture catchment to address this hypothesis. To develop targeted soil conservation strategies to reduce erosion and suspended sediment delivery to the streams by disconnecting sediment source areas, it is essential to identify sediment source areas by their relative contribution in the smallholder agriculture catchment. Therefore, based on the previously gained knowledge of the delayed sediment response and field reconnaissance, four sediment sources were selected: agricultural land, unpaved tracks, gullies and channel banks to apply a sediment fingerprinting approach. A final set of eight tracers were selected out of 23 geochemical and biogeochemical elements. Two different scenarios were evaluated with the unpaved tracks included (scenario 1) and excluded (scenario 2) as an individual source due to an overlap with tracers. In scenario 1, 80% of the samples were reclassified correctly in their source group, while in scenario 2 an improved reclassification of 95% was achieved. However, both scenarios demonstrated that agricultural land is the main source of suspended sediments (scenario 1: 75% and scenario 2: 77%) for the smallholder-dominated catchment, while channel banks are the second most important contributors. Despite the small relative contribution of channel banks, gullies and unpaved tracks, their sediment yield per unit area was large. The scale-dependency points to one main finding: that sediment loss from agricultural land per unit area was relatively low, but because it occupies most of the catchment area, it contributes the greatest proportion of the sediment yield. Channel banks, gullies and unpaved tracks are significant local hotspot sources, but their overall contribution is of lower significance.

Spatial variability of particulate macronutrient fluxes

The work presented in Chapter 6 focused on the particulate macronutrient (total carbon=TC, total nitrogen=TN and total phosphorus=TP) concentrations originating from the natural forest, tea-tree plantation and smallholder agriculture catchments. The analysis of Chapter 6 builds on the quantification of suspended sediment in Chapter 4, aiming to address the knowledge gap of how much sediment-associated TC, TN and TP end up in the stream under contrasting land use. The application of fertilizers in the agricultural catchments would be expected to lead to high sediment-associated nutrient concentrations. Surprisingly, one of the main findings is that particulate TC, TN and TP concentrations were up to three-fold and significantly ($p<0.05$) higher in the natural forest catchment compared to agricultural catchments. However, the mean macronutrient concentrations in the stream water for the sampling periods in 2018 and 2019, estimated based on the particulate macronutrient concentrations and suspended sediment concentrations, were highest for the smallholder agriculture and lowest for the natural forest catchment and tea-tree plantations. It can be argued that this difference is attributed to the high suspended sediment loads in the smallholder agriculture catchment. The TC concentrations of this study were higher than other reported concentrations (Walling *et al.* 2001; López-Tarazón *et al.* 2016), while the TN concentrations of the natural forest catchment were higher than intensified agricultural catchments, but the values of both agricultural catchments fell within those reported ranges (Walling *et al.* 2001; Pavanelli & Selli 2013). The TP concentrations were lower compared to P-saturated agricultural catchments of temperate regions (Neal *et al.* 2006; Ramos *et al.* 2015; Sandström *et al.* 2020). The high C:N ratio (12.6 ± 1.0 g kg^{-1}) in the natural forest catchment reveals that most particulate nitrogen is organic matter based. Low C:N ratios (10.6 ± 1.3 g kg^{-1}) and low nutrient concentrations point to impoverishment of agricultural soils with a smaller nutrient stock due to its low organic matter content due to poor management practices and low organic matter inputs.

7.2 The future of agriculture in tropical montane ecosystems

The western highlands of Kenya are known for their inherent fertile soils of volcanic origin. The characteristic soils in the study area are humic Nitisols and mollic Andosols (ISRIC 2004). Andosols are formed following the weathering of volcanic material, where aluminium-humus complexes protect soil organic matter from bio-degradation or mineralization (WRB 2015). Deep Nitisols have favourable soil physical characteristics of high permeability. A coarse A-horizon over a fine textured B-horizon with high clay content lead to well-drained soils (Sombroek *et al.* 1982).

Following forest clearance and subsequent cultivation, changes in the soil hydraulic properties cause a loss of permeability of these typically well-drained soils (Owuor *et al.* 2018). This process makes these soils more susceptible to soil erosion, as shown in this thesis, where the smallholder agriculture catchment generated six times more suspended sediments compared to the natural forest catchment (Chapter 4). The continuous loss of topsoil can affect agricultural productivity because of the loss of the nutrients and reduced plant water availability. Since this study has shown that the major sediment source in the smallholder agriculture is agricultural land, much of the natural soil capital is lost to the aquatic system (Chapter 5). The loss of the agricultural topsoil and densely populated areas under continuous cultivation over many decades in tropical countries may lead to the further restriction of productivity. The lack of alternative land for fallow land due to high population densities commonly lead to the overexploitation of agricultural land, as it was shown in tropical agricultural regions cultivated on steep hillslopes in Uganda (Lederer *et al.* 2015), Tigray (Ethiopia) (Girmay *et al.* 2009) or Kisii District (Western Kenya) (Smaling *et al.* 1993), while the annual nutrient depletion rate is >60 kg ha^{-1} (N, P_2O_5 and K_20) for Kenya (Henao & Baanante 2006). Several studies in tropical montane catchments experienced high losses in soil organic carbon and nutrients following conversion to agricultural cultivation (Solomon *et al.* 2000; Nyberg *et al.* 2012; Were *et al.* 2016). The low concentrations of particulate TC, TN and TP in the smallholder agriculture compared to the natural forest catchment reflects the nutrient depleted soils through elevated soil erosion in combination with continuous cultivation over many decades and low input use (Chapter 6).

Compared to agricultural systems, a natural forest ecosystem has a tighter nutrient cycle and high inputs of organic matter through litter input. Consequently, forest conversion to cultivation leads to reduced fresh organic matter input, especially when little crop residue remains on the fields after harvesting (Celik 2005; Chiti *et al.* 2018). Since fertilizer application is extremely low, agricultural production by smallholder farmers in this region is highly dependent on the quantity of soil organic matter. Naturally, tropical soils recycle their nutrients through mineralization of soil organic matter in order to maintain soil fertility. However, if organic matter inputs are too low, this process cannot support agricultural productivity due to low inputs. Nutrient mining through harvest increases the pressure on these already nutrient constrained soils. In addition, the clay in these tropical soils contains amorphous iron (Fe), which can limit plant P availability. A study in a similar montane region of the Munessa-Shashamane Forest in Ethiopia showed that the loss of soil organic carbon of forest origin on converted forest soils is not compensated by the low crop derived soil organic carbon (Lemenih *et al.* 2005). A decrease in soil organic matter reduces soil aggregate stability, which can further accelerate soil erosion. Lower soil water permeability leads to increased susceptibility of surface soil to erosion which further reduces soil organic matter (Chapter 4 and 6). This cycle of organic matter mineralization in a low input system and the additional loss through soil erosion leads to a continuous loss of the already degraded organic matter content, further reducing the productivity of the soil (Figure 29).

The Kenyan highlands are of high importance for agriculture in particular because of the favourable climatic conditions. Since the greatest part of the country relies on rain-fed agriculture due to the lack of irrigation systems. However, climatic factors (e.g. heavy rainfall) and changing vegetation cover can speed up soil chemical weathering processes or mineralization of soil organic matter (Purwanto & Alam 2020). Chapter 5 shows that the surface soil was depleted in macronutrients compared to less weathered subsoils (Stenfert Kroese *et al.* 2020a). The rain-fed agricultural system is vulnerable to climate change and adds on the existing challenges for agricultural productivity. Seasons of droughts or a delayed onset of the rainy season leads to food insecurity, which can increase the dependency on imports. Finally, erosive rainfall makes the already erosion prone hillslopes susceptible to soil erosion. Under climate change the amount of intensive rainfall days is likely to increase in the future in Kenya, which highlights the need to reduce the negative impact on soil erosion to secure agricultural production.

Agricultural productivity is constrained by nutrient deficient soils. Therefore, replenishing soil fertility and boosting productivity is essential to sustain agricultural productivity and the long-term viability of the soils. Finally, an increase in crop residue input could reduce on- and off-site impacts of soil erosion and slow down the negative feedback loop of organic matter mineralization as shown in Figure 29. Land is, as in many other places in the world, the most important resource of Kenya for rural communities (Nyangito *et al.* 2004), this is why it is essential to protect the soil on-site through targeted soil conservation measures.

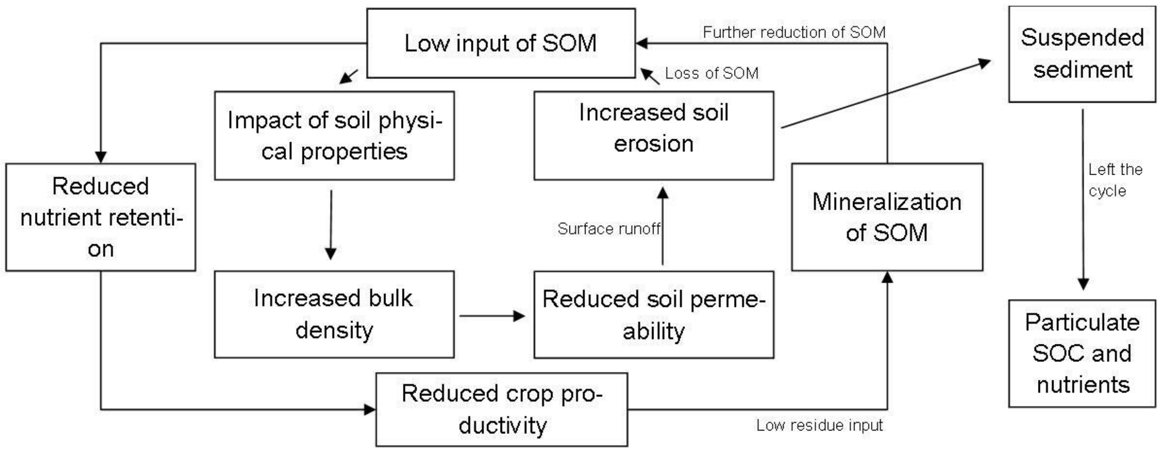

Figure 29 Basic cycle of soil organic matter (SOM) and particulate soil organic carbon (SOC) and nutrients (N and P) in a typical low input agricultural system.

7.3 Soil conservation in rural areas

The agricultural sector in Kenya contributes with 26% to the gross domestic product (GDP). Furthermore, agriculture plays a vital role for the rural economy, employing more than 70% of Kenya's rural population (FAO 2020). Smallholder farming is key in economic growth, food security and poverty alleviation. However, the country's agricultural performance is unsustainable and stays relatively low resulting in a dependency on imported goods. Wheat, palm oil, sugar, corn and rice are the most imported agricultural products in Kenya (FAO 2020). The country turned from a self-sufficient producer to a net importer of most food staples due to a decline in production (Nyangito *et al.* 2004; Henao & Baanante 2006). Rural smallholder farmers practice subsistence agriculture but rely on purchasing food because they cannot meet their staples. Farmers are forced to sell their little surplus such as other marketable vegetables right after harvest independent of a beneficial market price because of the direct need of income. Often this leads to a disadvantage in purchasing their staple during times of higher market prices (Nyangito *et al.* 2004). Additionally, many rural households rely on off-farm incomes. The imbalance between low income and dependency on purchasing food externally explains the vulnerability for food security based on high poverty rates in these rural settings ($360 per capita income) (Place *et al.* 2006). In addition to low productivity and income revenues, soil erosion losses account for 3.8% of the GDP which leads to another constraint for agricultural production (Cohen *et al.* 2006). The intention of this discussion is not to provide solutions to increase agricultural productivity but rather to reflect on the challenges of the application of soil erosion control measures in rural regions under high poverty rates.

Despite these challenges, smallholder farmers remain key in food and nutrition security due to the large proportion (83%) of the total population living in the rural Kenyan highlands (Himeidan & Kweka 2012; Paloma *et al.* 2020). Increasing productivity can be a demanding task when the soil is lost to the aquatic system due to soil erosion. As increased agricultural efficiency is a must for poverty alleviation, there is a fundamental need for protecting the soil from erosion as soil is the natural capital for the people in the rural highlands. However, the interest in applying sustainable agricultural practice is not only restricted by the lack of external resources, but also by restricted property rights in rural areas. Favourable land tenure arrangements for farmers would improve their long-term right to own the land, thus, the motivation would be higher to sustain the health of the soil also for the next generation (Henao & Baanante 2006).

Usually, rural farmers associate soil erosion with soil degradation and link that with a loss in their inherent productivity levels but are restricted to invest in soil conservation measures by the lack of financial, labour and land resources. Accessibility of agro-inputs to boost organic matter inputs is often restricted by the high cost (Sheahan & Barrett 2017). Policies and investment strategies at national level must be implemented to make purchasing agro-inputs more economically attractive and more available for rural farmers. The costs of fertilizers are particularly high in rural areas due to high transportation costs based on poor road conditions. This often leads to higher production costs in relation to the produce revenue (Place *et al.* 2006). The investment in the rural infrastructure can make purchasing of agro-inputs more attractive for rural farmers, whereby an improved quality of road networks can additionally reduce sediment supply to the streams.

The use of leguminous cover crops is another way of strengthening and protecting the soil from runoff and retaining nutrients through N_2-fixation (Matheus *et al.* 2018). However, as crop yields are low, farmers are required to harvest a marketable crop each season. Scarcity of land due to high population density limits the use of leguminous cover crops on fallow land. This means replacing a crop with soil conservation through the fallow system with leguminous cover crops will be challenging. In addition, intercropping systems with leguminous may only increase yields in the long-term and would increase labour needs for weeding, which is inappropriate in the short-living cycle. The financial restrictions would not allow the purchase of seeds other than crop seeds (Rusinamhodzi *et al.* 2011; Franke *et al.* 2018). Vegetated buffer strips cultivated with Napier grass, mulching or mechanical structures such as *fanya juu* terracing to recover soil organic matter due to reduced soil erosion, as mentioned in Chapter 5, might also be economically inefficient due to competition for water, nutrients and space with crops on limited land or increased investment of labour (Rusinamhodzi *et al.* 2011). To trap eroded soil from hillslopes there is need to rehabilitate and revegetate the riparian buffer of 30 m along streams in agricultural catchments, as pre-described by the Kenya's Water Act (Republic of Kenya 2012). However, land scarcity hinders the compliance of this act.

Although rural farming systems tend to be limited in financial or labour resources, they have a strong collective knowledge through networking. They pass on indigenous knowledge using inexpensive technologies and labour-saving practices to adapt technologies to their local needs (Stocking 2003). As a positive perspective, population pressure does not always equally lead to land degradation. If people are the problem for land degradation, they can also become the solution. As Tiffen et al. (1994) has shown that more people equals to less erosion. Capital and security, technological knowledge, economic incentives and market access might contribute to a sustainable agricultural performance. Growth of population can also mean an increase in labour force and agricultural intensification (Tiffen *et al.* 1994).

7.4 Limitations and recommendations for future research

Despite the valuable findings of the thesis, some limitations should be taken into account when interpreting the results, which could be addressed in future research. The four-year sediment time series of Chapter 4 is based upon a one-time *ex situ* turbidity-sediment calibration curve. Due to possible changes in sediment source contributions or in catchment geomorphology, a calibration curve for each year would have been the best-case scenario based on a combination of automatic sampling and manual event-based sampling over the range of storm events during a year. However, the remoteness of the catchments did not allow fast and frequent access to the sites, which restricted manual storm sampling and the frequent collection of the samples collected by automatic samplers. Therefore, the peak of a storm event was generally not covered resulting in a lack of samples for the maximum turbidity of the stream. Therefore, the best approach for such catchments under challenging environments was to set up an *ex situ* rating curve, which allowed the simulation of a storm event. Despite having made every attempt to increase the accuracy of the adapted sediment calibration procedure (e.g. removal of coarse material, sampling of instream suspended sediments or of source sediments in direct vicinity to the outlet), over- or underestimation of suspended sediment concentrations may have increased the uncertainty in the sediment yield calculated. In addition, despite having established a site-specific calibration, the calibration curves for each site were combined to one calibration equation to increase the size of the dataset (n=50) for a more robust relationship. Before combining to one calibration equation, the differences in the slopes were tested for significance to justify using one equation for three catchments. The hydrological modelling in Chapter 4 was an opening for future work. Having provided field measurements to build the sediment rating curve, another step might be to model suspended sediment concentrations based on rainfall or discharge data.

The particulate macronutrient analysis in Chapter 6 was constrained by the low sampling frequency of suspended sediment and also to the delayed onset of the rainy season in 2019. Due to limited flexibility in the planned field sampling campaign, the prolonged dry season restricted the sampling to the start of the rainy season in contrast to the long rainy season in 2018. This may have led to missing important storm events, which could have contributed to a further understanding of the temporal distribution of particulate macronutrient fluxes. In addition, a higher sampling frequency may have provided the opportunity to apportion particulate nitrogen to the biweekly sampled dissolved nitrogen concentrations. This could allow answering the question of how much of the total nitrogen is of particulate and dissolved origin. To strengthen the empirical evidence and to estimate the extend of nutrient losses from agricultural soils in the smallholder agriculture catchment, a targeted soil sampling campaign on the soil nutrient status of the topsoil layer (0-30 cm) would further add to the knowledge of the impact of land use and management.

During the sediment fingerprinting analysis conducted for Chapter 5, it was not possible to differentiate cropland and grassland as two separate source areas with the tracers used. In the smallholder agriculture catchment, grasslands cover a large proportion of the catchment area. Therefore, grasslands might have been an important significant individual source area, as degraded grasslands areas were observed during field work. Based on this observation, grasslands are thought to be important sediment contributors to the stream network.

Although the selected tracers in the sediment fingerprinting supported the differentiation between subsurface and surface sources, they could not differentiate clearly between the different land uses. To address this constraint, the use of isotopic signatures (e.g. $\delta^{13}C$ and $\delta^{15}N$) may support the differentiation of surface sources among land use systems in the future.

Agricultural land is a valuable resource for the rural community for food security and to sustain their livelihoods through farming. Therefore, rural farmers are the main stakeholders to be engaged on the ground during the development phase for options of sustainable agricultural management practices. Therefore, participatory monitoring could be a solution. Soil erosion monitoring as an integrative bottom-up approach that engages the rural community could be used in the future. The integration of the community in soil erosion mapping using simple soil erosion indicators to cover the questions such as which erosion features do occur (e.g. rill erosion, sediment on fields or root exposure), where and why (e.g. changing depth of rural trackways) does soil erosion occur and how active are they, will raise farmer’s awareness what causes soil erosion and why it is important to conserve the soil and the problems related to soil erosion. The farmers, possibly together with agencies or non-governmental organizations can then build on this newly gained knowledge to implement targeted sustainable agricultural management practices using an integrative approach. This will help to identify, together with the farmers, feasible ways for soil conservation management.

7.5 Research output and impact statement

Tropical forests are global hotspots of land use change, and protect the soil from soil erosion at the same time with a dense above and belowground biomass. Despite their importance, tropical montane catchments, in particular in East Africa, are among the least studied in the world. This study has contributed to a unique four-year high-frequency dataset of hydrological and sedimentological measurements collected across three catchments under different land use (natural forest, tea-tree plantations and smallholder agriculture) for a data scarce region in East Africa. It has further produced a database of geochemical element and macronutrient concentrations of four sediment source samples (n=248) and target sediment (n=35) for a smallholder agriculture catchment and particulate macronutrient concentrations (n=101) of three catchments under different land use. This work is relevant not only for African tropical montane environments, but it also contributes to the general understanding of catchments in the tropics facing the challenge of small-scale disturbances and deforestation.

In summary, this study demonstrated the impact of agriculture on terrestrial and aquatic ecosystems in comparison to natural ecosystems in catchments of the largest remaining tropical montane forest of East Africa. By having provided empirical measurements for the magnitude of suspended sediments and their depleted macronutrient fluxes of agricultural catchments, soil conservation practices should be targeted onto agricultural land. The large proportion of agricultural land and the current trend towards agricultural expansion further strengthens the need for the application of management strategies. However, in rural areas with high poverty rates there is a challenge between food production and the implementation of soil conservation strategies, which was discussed in Chapter 7.3. As rural farmers are key in the agricultural sector of East Africa, investment and policies in the rural infrastructure can help to reduce sediment supply from unpaved trackways or their connecting source areas such as agricultural land. Improved quality of roads can further increase market accessibility.

7.6 Conclusions

Investments in sustainable land management should protect the long-term viability of agricultural soils and some of these practices may even provide an alternative source for fodder (e.g. Napier grass) and income to farmers. However, smallholder farmers often have no or restricted financial means and face difficulties to access inputs other than crop seeds to improve soil fertility or to invest in extra labour. In addition to the lack of financial, labour and land resources, land tenure rights in rural areas are often not properly established.

The design and implementation of policies to invest in roads and associated infrastructure would on the one hand reduce transportation costs for e.g. fertilizers to the farmer or produce to markets and improve market access for rural farmers. Increased land productivity enables rural households to grow enough food to generate a surplus and begin selling, which has a positive effect on welfare and poverty reduction. A rise in food supply might provide price stability and a competitive environment for the rural farmer to sell their products. Increased financial capital could be invested in sustainable soil conservation measures. Better engineering of rural trackways could, on the other hand, also have a tremendous effect on the reduction of suspended sediment supply, considering the importance of unpaved tracks as connecting agents between steep hillslopes and the streams (Chapter 5). Chapter 4 showed that a well-engineered road management, as is happening in the tea-tree plantation catchment, can lower suspended sediment contributions by a third compared to the smallholder agriculture catchment.

Preventing soil erosion will arguably restore and rebuild soil productivity, which then can lead to poverty alleviation due to improved agricultural productivity. When farmers fail to install effective conservation measures, the risk of accelerated erosion is high due to the continuous decline in organic matter. Restoring soil organic matter will have a beneficial impact on the water quality through reduced soil erosion and finally will protect Lake Victoria's water quality. Through the improvement of land management, increased productivity of arable land and grazing land will provide alternative income and sources for firewood by the use of cover trees for the rural population. Consequently, this will remove the pressure on natural forest ecosystems, thus, further conversion to smallholder farms and the encroachment to forest margins.

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