Investigating the processes of erosion and sediment yield at different scales in commercial forestry
– A case study at Two Streams, KwaZulu-Natal

by

Jarryd Steven Gillham

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FRONTISPIECE

Abstract

Soil erosion is the detachment and transportation of soil particles from one location to another and has on- and off-site impacts which jeopardize the capacity of ecosystems to deliver environmental services. A possible off-site impact of soil erosion is eutrophication of water bodies, a major concern in water scarce South Africa. Previous studies have outlined the role that agriculture contributes to soil erosion. This study investigates the role of commercial plantations in contributing to soil erosion, which in South Africa occupy over three million hectares. This study considers the processes of erosion and sediment loss at different temporal and spatial scales in a commercial forestry land use.

The research study was undertaken in a mature *Acacia mearnsii* afforested catchment at Two Streams situated near Seven Oaks, Greytown. The first objective of the study was to set-up an appropriate experimental design by using 5x2 m^2 runoff plots (n=9) and 1x1 m^2 micro-plots (n=9) located at three landscape positions. Automatic tipping buckets were used to measure runoff intensity. Runoff from 1 m^2, 10 m^2 plots and 34 ha catchment were assessed from January 2015 to March 2016. At the catchment outlet there was a V-notch weir which measured stream flow, weir samples were taken using an ISCO automatic sampler. Runoff was measured and water samples were collected from the nested scales after selected rainfall events (n=15). The runoff samples were analysed in the laboratory to determine sediment volume, phosphate, nitrate and soil organic carbon.

Sediment loss, for the 1 m^2 and 10 m^2 plots averaged similar amounts per event (0.901 gl^{-1} and 0.809 gl^{-1} respectively) with an average of 0.793 gl^{-1} of sediment loss measured from the weir. The results highlight that the increase in spatial scale did not have an influence on the sediment and nutrient loss, with rain splash and runoff providing similar results (g/m^2). There was a low degree of spatial variation in sediment yield due to low variation in rainfall throughout the catchment and the increase in spatial scale did not have a significant influence in sediment yield. Temporally, higher intensity rainfall events led to high intensity runoff, which led to higher volumes of sediment loss. This was evident on the 18th December 2015 with an intense rainfall event (114 mm) leading to a significant increase in sediment yield compared to the study average. There was a inverse relationship between rainfall/runoff and phosphate, nitrate and dissolved organic carbon concentrations. With higher rainfall/runoff events resulting in lower nutrient volumes this due to the process of dilution compared to smaller rainfall/runoff events which resulted in higher nutrient concentrations.

The results from this study showed the link between rainfall/runoff and soil erosion and the vital role of vegetation interception in reducing the impact of rainfall and water erosion. The results suggest that the *Acacia mearnsii* catchment was effective in reducing the impact of water erosion, which demonstrates that a mature commercial forest with low human impact (harvesting) has manageable soil erosion rates. With the potential increase in the rate of soil erosion due to climate change, more research needs to be undertaken, so that mitigation measures can be designed for the future.

Declaration

I JARRYD STEVEN GILLHAM declare that:

i. The research reported in this dissertation, except where otherwise indicated, is my original work.
ii. This dissertation has not been submitted for any degree or examination at any other university.
iii. This dissertation does not contain other persons' data, pictures, graphs or other information, unless specifically acknowledged as being sourced from other publishers.
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Jarryd Steven Gillham

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List of Abbreviations

ACRU - The Agricultural Catchments Research Unit

AWS – Automatic Weather Station

BSI - Bare Soil Index

DBH – Diameter at Breast Height

DOC - Dissolved Organic Carbon

GCM - General Circulation Model

GIS – Geographic Information Systems

GLASOD - The Global Assessment of Human-induced Soil Degradation

HRU - Hydrologic Response Unit

K - Potassium

MUSLE - Modified Universal Soil Loss Equation

N- Nitrogen

NO_3^- - Nitrate

NPS - Non-Point Source Pollution

P - Phosphorus

POC- Particulate Organic Carbon

RUSLE - Revised Soil Loss Equation

SOC - Soil Organic Carbon

SWAT - The Soil and Water Assessment Tool

TM – Thematic Mapper

TOC - Total Organic Carbon

USDA - US Department of Agriculture

USLE - Universal Soil Loss Equation

WEPP - Water Erosion Prediction Project

WRC - Water Research Commission

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Chapter One

Introduction

1.1 Introduction

Sixty-eight percent of Africa south of the equator is affected by accelerated soil erosion, with South Africa experiencing a high level of soil erosion with the country suffering from an annual soil loss of 450 – 500 metric tonnes of top soil (Beckedahl and de Villiers, 2000; Meadows and Hoffmann, 2003). Approximately seventy percent of South Africa is affected by varying intensities of soil erosion with a high risk of larger portions of the country becoming affected in the future (Laker, 2004). The projected increase in frequency of high intensity rainfall events and extreme events such as floods due to global climate change, is set to cause even higher levels of soil erosion (Bates *et al*, 2008). The rate of soil formation in South Africa is estimated to be approximately one millimetre every forty years, with the rate of soil loss being ten times that of the rate of soil formation (Van Zyl *et al.*, 1996).

The key driver for soil erosion in South Africa is rainfall, with the eastern portion of the country receiving the majority in the summer months (Le Roux *et al*, 2008). The erosional processes depends on a combination of interactive effects of erosion factors namely; rainfall erosivity, soil erodibility, slope steepness and slope length, crop management and support practice. Soil erosion not only involves the loss of fertile topsoil, reduction of soil productivity and reduction in crop yield over time, but also causes water management problems, in particular in semi-arid regions such as South Africa where water scarcity is frequently experienced. Although soil erosion is a natural process, it is often accelerated by human activities such as clearing of vegetation by overgrazing. Smith *et al.*, (2000, pg 355) states that "overgrazing is the main human induced factor causing accelerated water erosion in South Africa".

As different soil erosion processes occur at different spatial and temporal scales, the assessment of soil erosion at the landscape scale is recognized as an issue within the environmental sciences (Chaplot and Poesen, 2012). Water erosion occurs predominantly from precipitation through rain splash, un-concentrated flow as sheet erosion. Rain splash is instrumental in detaching the top-soil and transporting the soil short distances. Runoff and flow detachment also detach soil particles but are generally able to transfer soil greater distances. While flow detachment and transport by splash occur at local level, larger surface areas are required for sedimentation to be the dominant process (Chaplot and Poesen, 2012). Micro-plots can be used to inform on the contribution of splash and rain-impacted flow on sediment mobilisation (Chaplot and Poesen, 2012), with multi-scale studies being a promising approach to detect and quantify the relative contribution of erosion processes (e.g. splash, sheet, concentrated flow, stream bank and stream bed mobilization) that dominate at various spatial scales.

For example, plots of several m^2 are useful tools to evaluate interrill erosion, however they provide little information on the dominant erosive processes and their interactions (Chaplot and Le Bissonnais, 2003).

An associated impact of soil erosion is that it is a key driver to the loading of Nitrogen (N) and Phosphorous (P) into surface waters (Carpenter *et al*, 1998), which can have profound effects upon the quality of receiving waters. The most common effect is an increase in eutrophication leading to an the abundance of algae and aquatic plants (Carpenter *et al*, 1998), which can result in loss of the amenities or services that these aquatic resources provide. Eutrophication leads to increased productivity and biomass of phytoplankton and suspended algae, shifts in phytoplankton composition to bloom-forming species, many of which may be toxic, or which may not be consumed effectively by aquatic grazers (Smith *et al.*, 1999).

The effects of inappropriate land-use practices in South Africa have led to land degradation (Mills and Fey, 2004). Natural forests, maintained as nature reserves, can be stable against erosion (Laker, 2004), however planting of commercial forests can promote erosion (Sherry, 1964). Forestry operations such as timber harvesting, road constructions, clear-cut logging and burning of forest residues have shown to impact on catchment sediment yields and to reduce water quality for downstream users (Van Dijk and Keenan, 2007). There is a tendency for conversion of grassland by afforestation with the commercial *Pinus* and *Eucalyptus* species, which negatively affects the quantity of catchment runoff (Turpie *et al.*, 2008). Commercial forests intercept stream flow in particular when close to watercourses. Siltation of storage dams is acknowledged as a major problem in South Africa and better understanding of erosion and sediment yield is important to limit the cause of siltation. As an example, due to siltation, the storage capacity of the Welbedacht Dam near Dewetsdorp in the Free State reduced rapidly from the original 115 to approximately 16 million cubic metres within twenty years since completion in 1973 (Le Roux, 2011).

1.2 Research Rationale

There are over three million hectares of commercial forestry in South Africa with the consequential erosion and sedimentation contributing to sedimentation concerns in South Africa's rivers and dams, which significantly reduces the nutrient and carbon stock within the soils (Turpie *et al.*, 2008). South Africa is a water scarce country and any form of erosion may have indirect impacts on the water quality and quantity of the waterbodies in the country. Therefore it is a necessity to address the impact of soil erosion in forestry and its accompanied impacts.

This research investigates the impact of water erosion at varying spatial scales and temporal scales. This study formed an integral part of a larger research project (WRC K5/2402) funded by the Water

Research Commission entitled ‘assessing the impact of erosion and sediment yield from different land uses in farming and forestry systems and their effect on water resources in selected catchments of South Africa’. It is intended that these findings will aid forest managers in their understanding of impacts of erosion in commercial forest and associated nutrient transportation. The field and laboratory data from this study will be used for verification of soil erosion models including MIKE-SHE, ACRU and SWAT. Relatively few studies conducted research in forest plantations so this study can help fill a research gap.

1.3 Research Aim and Objectives:

To investigate the processes of erosion and sediment loss at different temporal and spatial scales in a commercial forestry land use.

To achieve this aim, the objectives were:

i. Set-up appropriate experimental design that is able to measure the temporal and spatial variability of soil erosion
ii. Measure soil loss in an afforested catchment at different spatial and temporal scales and to determine the sediment yield at the end of the study.
iii. Determine the spatial and temporal variations in nutrient concentration.

Although gully erosion is evident in parts of South Africa, it is not in the scope of this study as gullies do not occur in the study area. Furthermore, runoff plots and rainfall simulators capture soil loss from rill and interill erosion and not from gully erosion which occurs at larger scales (>0.03 km^2).. Natural ecosystems provide important resources for the well-being of societies. These ecosystems are in jeopardy due to land degradation and, in particular, soil erosion, which not only effects the productivity of land but also has off-site impacts. Transportation of sediments and nutrients impacts water bodies through eutrophication and siltation. South Africa experiences high rates of soil erosion, and with the future threat of climate change, soil erosion may dramatically increase, which a water scarce country such as South Africa can ill-afford. This project investigated erosion and sediment yield at different temporal and spatial scales. To meet this aim, key objectives were created, all with the intention of understanding the impacts of erosion in commercial forests.

Chapter Two

Literature Review

2.1 Soil Erosion

Water availability is predicted to be the single greatest and most urgent development constraint facing South Africa with poor water quality in rivers and streams exacerbating the issue for water users (Nilsson and Malm-Renöfält, 2008). The national water resources strategy (DWAF, 2004) estimates that at current usage and price levels, available water resources will be unable to meet demands by 2025. According to Turpie *et al*. (2008) surface water is heavily committed for use, water is imported from neighbouring countries, and the limited groundwater resources do not offer much of a reprieve. In the future, the growth in human population will lead to an increased need for food and forest production, which will lead to an increased competition for water between different water users (Clulow, 2007).

The rate of soil erosion worldwide, and in South Africa, are likely to increase given the projected increase in frequency of high intensity erosive rainfall events and extreme events such as floods due to global climate change (Van Oost *et al.*, 2000; Meadows, 2006; Mullan *et al.*, 2012). Erosion has both an on and off-site effects. The loss of fertile topsoil on-site decreases soil productivity and reduces crop yields (Ning, 2006) which leads to increased costs to maintain the level of agricultural production. Off-site, soil erosion impacts negatively on the natural water storage capacity of catchments areas, service of man-made reservoirs and dams, quality of surface water, aesthetics and ecological balance (Doody *et al.,* 2012). Sedimentation in rivers and dams not only causes water quality concerns such as eutrophication but also increases wear and tear to nozzles and hydrologic pumps for irrigation. Soil erosion results in siltation of reservoirs which reduces their storage capacity and might result in flooding. Whilst silt can act as a host for pathogens increasing health risks in degraded water systems (Carr and Neary, 2008).

Soil erosion is the process of detachment and transportation of soil materials by wind or water and negatively affects land productivity through loss of soil nutrients and soil organic matter (Lal, 2003). This loss of fertile top soil has negative consequences for agriculture and forestry through loss of production and affects water catchments through sedimentation and eutrophication of waterways. According to Rickson (2014), forestry operations such as timber harvesting, road constructions, clear-cut logging and burning of forest residues have significant impacts on catchment sediment yields reducing the water quality for downstream users.

Best management strategies that will maintain the quantity and quality of fresh water for downstream users need to be developed and adopted to sustain South Africa's scarce water resources (Rockstrom, 2000). It enables decision makers to select appropriate measures to reduce the rate of soil erosion, thus a better understanding of the erosion process and the factors that accelerate soil erosion and determine the sediment delivery ratio in watercourses is required.

Due to the interrelated nature of the research this chapter is divided into ten sections all linked to soil erosion with a primary focus on soil erosion in South Africa. The sections are; defining soil erosion, explaining the types of erosion, outlining the factors that determine the rate of soil erosion, describing overland flow and how soil erosion varies temporally and spatially, considering the implications of soil erosion, the threat of climate change, the impact of soil erosion on forestry and water catchments, strategies that could possibly mitigate soil erosion and finally, modelling soil erosion.

Soil erosion is a physical process of soil degradation and is the most common type of land degradation (Morgan, 1988). According to Le Roux *et al.* (2008) soil erosion can be defined as the detachment and transportation of soil particles from one location to another, the degree of soil erosion ranges from splash erosion to the alarming stage of gully formation. The process of soil erosion can be described as a loss of nutrient rich clay and organic matter, which impoverishes the upper top soil and leads to the upper soil layers being removed through erosion. The intense and increased pressure on the land to provide goods and services leads to its degradation and loss of its productive capacity. Land degradation is the loss in ability of the land to create benefits from the land use that falls under a specified form of land management (Meadows and Hoffman, 2003). Erosion results in the degradation of a soils productivity in a number of ways: it reduces the efficiency of plant nutrient use, damages seedlings, decreases plants' rooting depth, reduces water-holding capacity, decreases permeability and infiltration rates and increases runoff.

It is estimated that approximately seventy-five billion tons of fertile soil are lost from agricultural systems each year (Pimentel and Burgess, 2013). The Food and Agriculture Organization (FAO) estimate that approximately five to seven million hectares of productive topsoil are lost annually through erosion while other estimates state losses of more than ten million hectares per year (Sun *et al*., 2014). In South Africa, erosion is a problem which is worsening according to the Land Degradation Assessment in Dryland Areas (DA, 2008) and unless erosion mitigation and control efforts are encouraged the situation will continue to worsen. Approximately three hundred to four hundred million tons of soil are estimated to be lost annually in South Africa (Ning, 2006). According to the State of Environment Report of South Africa, soil erosion costs are estimated to be two billion rand annually which includes off-site costs for purification of silted dam water (Le Roux *et al*., 2008).

The on-site effect of erosion is a reduction in soil quality through the removal of topsoil and the loss of nutrients and applied fertilizers. In addition, soil erosion has the potential to remove light-weight organic matter and organic residues which reduces the water holding capacity of the eroded soils making them less suitable for plant grow. (Fournier, 2011). Off-site some of the sediments can be trapped by the vegetation in the riparian zone before reaching the stream. Sediments which enter the watercourses may block drainage ditches and stream channels and silt reservoirs and dams. Sediments delivered to the stream can significantly increase the turbidity of water and deteriorate downstream water quality through increased eutrophication which leads to increased water purification costs. Eroded particles, in particular the smaller size fractions such as clays, silts and organic matter have high specific surface areas and charge densities, thus increasing the potential for adsorption of nutrients, agrochemicals and heavy metals onto sediment.

According to Miller *et al*. (2009), concentrations of pollutants in sediment are highly dynamic because of transformations between particulate and solute phases, which exacerbates water quality problems associated with sedimentation. Sharpley *et al.* (1981) described sediment as a multiple stressor in terms of water pollution as concentrations of pollutants such as phosphorus can be higher in sediments than in the original soil. A study carried out by Abagale *et al.* (2012) in Northern Ghana found that soil and nutrient (N, P, K) loss and loss of organic carbon and organic matter were greater on non-vegetated areas of farmlands than on those that had been vegetated. They observed that erosion and soil nutrient loss rates increased with quantity of rainfall over a period of time and with rainfall intensity. According to Ngcobo *et al*. (2012), nitrogen (N) and phosphorus (P) dynamics have not been adequately assessed in South Africa. This is especially relevant in the predominantly agricultural and rural catchments of the country where non-point source pollution (NPS) is widespread. The mechanisms that govern sediment yield, N and P distribution are anticipated to change under conditions of higher temperature and rainfall however the magnitude and direction of that change is not well understood.

Since erosion takes place predominately on land that is being utilized, the limited amount of high potential agricultural land is at high risk of degradation as a result of erosion (Morgan, 1988). South Africa’s population is growing at just under two percent per year and to feed the growing population, food production needs to increase (WWF, 2009). The loss of productive agricultural soil as a result of erosion threatens food security and sustainable development and thus requires attention.

2.2 Types of Soil Erosion

The degree of soil erosion ranges from splash erosion to gully formation. The agents that transport the soil comprise those which contribute to the removal of a relatively uniform thickness of soil and those which concentrate their action in channels. The former consist of rain splash and surface runoff in the form of shallow flows of infinite width, sometimes known as sheet flow but more correctly categorized as overland flow, while the latter covers water flow in small channels, known as rills or deep channels known as gullies (Morgan, 1988).

Sheet erosion, which is a uniform removal of soil from the surface, is the second phase of the erosion process after rain spash. As erosion becomes increasingly severe, rill erosion begins (Toy *et al*., 2002). According to Ritter (2012), rill erosion results when surface water runoff concentrates and forms small yet well-defined channels. While it is widely accepted that rills are initiated at a critical distance downslope, where overland flow becomes channelled (Morgan, 1988), rill erosion can occur on steep land and on land that slopes more gently. At one time it was thought that gullies develop as enlarged rills however studies of gullies have revealed that their initiation is a more complex process (Morgan, 1988).

Sealing and crusting has been reported to increase soil erosion by enhancing surface runoff and the detachment of soil particles from the soil surface (Le Bissonnais and Singer, 1993; Wakindiki and Ben-Hur, 2002). Soil crusting refers to the formation of a thin layer at the soil surface which is characterized by reduced porosity and high penetration resistance whilst surface sealing is the initial phase or wetting phase in crust formation (Valentin and Bresson, 1998). Surface crusting, particularly on bare surfaces, is driven by raindrop impact however compaction of the soil affects the formation of a crust (Neave and Rayburg, 2007).

2.3 Processes Determining the Rate of Soil Erosion

Soil erosion process consists of two main phases; the detachment of individual particles from the soil mass and the transportation of these particles by erosive agents such as running water and wind. When there is no energy to transport the particles a third phase occurs, deposition (Salles and Poesen, 2000). Rain splash is the most common detaching agent which occurs through raindrops hitting the bare soil surface, which has the ability to loosen and detach the soil particles. If rain splash had considerable impact then soil particles may be thrown through the air over distances of several centimetres. Soils that are continually exposed to heavy rainfalls are considerably weakened and erosion is most prominent in areas with high levels of rainfall (Prasuhn, 2012). Any form of soil erosion that can be perceived to be detrimental is known as accelerated erosion (Beckedahl and De Villiers, 2000).

Soil is broken up by various weathering processes; mechanical weathering which takes place when rocks are broken down by physical force and chemical weathering which breaks down the bonds holding the rocks together, causing them to fall apart and to form smaller and smaller pieces. Chemical weathering is more common in locations where there is abundant water. Soil is broken up by alternate wetting and drying, freezing and thawing action, wind, tillage processes and by trampling of people and livestock. (Prasuhn, 2012).

All these processes loosen the soil, allowing it to be removed by the agents of transport, which act and contribute to the removal of a relatively uniform thickness of soil. These agents are split into two main groups: the first group consists of rain splash, surface runoff sometimes known as sheet flow better known as overland flow. The second group covers water flow in small channels, known as rills (Morgan, 1988). According to Ritter (2012) rills are shallow drainage lines less than 30cm deep, which develop when surface water concentrates in depressions or low points and erodes the soil.

The factors which influence the rate at which soil erosion occurs are wind and rainfall intensity, soil erodibility, topography, vegetation cover, soil management practises and conservation measures. According to Morgan (1988), factors which affect erosion can be grouped into three categories: energy, resistance and protection. Energy refers to the potential ability of rainfall, runoff and wind to lead to erosion. This category is described by the term erosivity. Fundamental to the resistance category is the erodibility of the soil which depends on its mechanical and chemical properties while the category protection focuses on factors relating to plant cover. Vegetation cover provides varying levels of protection by intercepting the impact of rainfall and reducing the velocity of runoff and wind (Morgan, 1988). Vrieling *et al*. (2014) point out the high variability of rainfall erosivity and vegetation cover through space and time and concluded that spatial and temporal variability of erosivity need to be accounted for, in combination with vegetation cover, when monitoring soil erosion.

2.3.1 Rainfall Intensity and Runoff

The severity of erosion depends upon the quantity of soil material supplied by detachment and the capacity of the eroding agents to transport it. The two forms of energy available for erosion are potential and kinetic energy (Morgan, 1988). Raindrops physically break down soil aggregates and disperse the soil material which increases the susceptibility of the suspended material to be transported by runoff. Rainfall has high energy and soil detachment through splash erosion contributes significantly to soil erosion. The raindrops erosive energy is proportional to its size, whilst the rate of soil erosion and the amount of soil eroded is proportional to the quantity of runoff. Runoff occurs when the rainfall intensity exceeds the infiltration rate of the soil. The amount of runoff is greater during short duration high intensity storms which provide sufficient energy to detach and disperse soil aggregates. The amount of runoff generally increases with increasing soil compaction and soil crusting and decreases with increasing plant canopy and basal cover. Erosion caused by long-lasting low intensity rainfall can, however, cause significant soil loss when accumulated over time (Fournier, 2011). Droughts are important as this effects vegetation cover and increases the amount of bare soil, droughts are usually broken by flood events when ground cover is low which may cause severe erosion..

2.3.2 Soil Erodibility

Soil erodibility refers to the ability of the soil to resist erosion and is based on the soil's physical and chemical properties (Bissonnais, 1996). Soils with a higher organic matter content have an improved structure and relative faster infiltration rate and thus show greater resistance to erosion due to reduced runoff. Tillage practices that lower organic matter content, destroy soil structure and compact soil surface can significantly increase soil erodibility. According to Morgan (1988), silt loams, loams, fine sands and sandy loams are the most detachable. Finer particles are more difficult to erode due to the cohesiveness of the clay minerals of which they are comprised, unless they have been previously detached and, as a result, lost their cohesion, in which case they can then be moved at low shear velocities. Aggregate stability also depends on the type of clay mineral present.

Soil dispersion is a process that occurs in soils that are vulnerable to erosion by water. In soil layers where clays are saturated with sodium ions ("sodic soils"), soil can break down very easily into fine particles and wash away. This can lead to a variety of soil and water quality problems,

2.3.3 Topography

Soil erosion by water is proportional to the steepness of the slope of the field and to the slope length due to the greater accumulation of volume and velocities of surface runoff. Water erosion, especially gully erosion occurs on level land where flow accumulation is high (Morgan, 1988).

2.3.4 Vegetation

Fournier (2011) described vegetation cover as the most significant factor determining the severity of the soil erosion process. Vegetation cover and litter provide protection to the soil surface against the impact of erosive energy from raindrops while plant roots bind the soil particles into aggregates resulting in improved soil structure with high infiltration rates and less surface runoff (Mohammed and Adam, 2010). Residual roots provide channels that help to improve the hydraulic conductivity of the soil. Bare soils and soils with little vegetative cover or crop residues are highly susceptible to soil erosion. However, the erosion reducing effectiveness of vegetation or litter depends significantly on the type, extent and quantity of cover (Podwojewski *et al*., 2011). The effectiveness depends on how much vegetation cover is available at various periods during the year, relative to the amount of erosive rainfall that falls during these periods. In addition, the spatial distribution of vegetation along the slope has a significant impact on catchment sediment yield. Groundcover is the most important form of vegetation cover to reduce erosion, especially in forestry plantations it reduces the impact of rain splash and flow.

2.3.5 Soil Management

Good management practices such as planting along the contours can significantly reduce the energy of surface runoff and thus sediment transport. Fournier (2011) considered several soil management practices as adequate to reduce or prevent soil erosion: the prevention of the direct impact of raindrops on soil through mulching and plant cover, the management of soil surface in a way that the infiltration rate is improved and surface runoff is minimised, the shortening of the slope length to reduce surface runoff accumulation and the diversion of excess runoff in a controlled manner through waterways and graded channels. Poor land management practices, on the other hand, such as the inappropriate placement of roads or unsuitable timber extraction methods, in particular in areas prone to soil movement, can lead to high levels of soil erosion (McGarry, 2011).

2.5 Spatial and Temporal Variations of Runoff

In landscapes, there are spatial and temporal variations of water and nutrient fluxes and this can be useful for improving land management (Laznik *et al.*, 1999). There have been various studies conducted to improve understanding of rainfall-runoff processes and many hydrologic studies to improve understanding of hydrologic processes (Beven, 1989). Yet there is still a need to improve methods to describe runoff generation mechanisms occurring over hillslopes. This will lead to increased knowledge of how catchments generate flow and how runoff generation mechanisms impacts on nutrient and sediment transportation. Soil erosion rates measured at one scale are not representative for sediment yield at another scale level (De Vente and Poesen, 2005).

There are two different mechanisms used to describe overland flow generation (Horton, 1933; Hewlett and Hibbert, 1967). The first mechanism is Hortonian flow, which occurs when rainfall intensity exceeds the infiltration capacity of the soil. The second is when saturation exceeds surface runoff (this is when the perched water table rises, saturating the whole soil profile and ultimately creates a seepage face at the soil surface). Saturation excess overland flow occurs typically in areas where saturation occurs (i.e. bottomlands and seepage faces) (Sen *et al.*, 2010; Van de Giesen *et al*., 2011). As a result, runoff will vary spatially and there is a need to improve the understanding of spatial and temporal variations of runoff (Sen *et al*., 2010; Van de Giesen *et al*., 2011). It is important to note that weather has temporal variability, in particular, rainfall and that soil erosion and nutrient totals can, in some circumstances, be dominated by a few extreme events (Renschler and Harbor, 2002),

Surfaces in a catchment differ in response to rainfall and thus it cannot be assumed that there is uniform overland flow generation within a landscape. Overland flow generation is a spatially-variable process and is complicated to a large degree by temporal variation (Bergkamp, 1998; Cammeraat, 2004). Overland flow generated within a catchment is influenced by the interaction between; topography, soil and land cover, rainfall event characteristics, soil surface conditions, antecedent soil moisture conditions, infiltration rates, soil hydraulic properties and the depth to water table (Casenave and Valentin, 1992; Hernandez *et a*l., 2003). It is important to investigate the soil surface characteristics (environmental factors) which control the generation of overland flow. Groundcover was found to enhance infiltration and ultimately decrease the amount of overland flow generated (Bartley *et al*., 2006; Bautista *et al*., 2007; Podwojewski *et al*., 2011). There is a general trend of increasing sediment yield with increasing spatial scale (De Vente and Poessen, 2005).

Vegetation on the soil surface has an inverse relationship to generation of runoff. Sanjari *et al.* (2009) stressed that the linkages between the soil surface characteristics and the generation of overland flow is multi-factoral. Bergkamp (1998) states that effective infiltration rates on grassland hillslopes vary with rainfall intensity and flow depth, due to the interaction between rainfall, runoff, and vegetated micro-topography. Environmental factors vary temporally and spatially, as such, this affects overland flow generation. Soil surface crusting has been found to play a key role in the amount of overland flow generated, as it decreases infiltration which increases overland flow (Bautista *et al*., 2007).

2.7 Global Climate Change

A plethora of studies have been undertaken to investigate the impact of climate change on the hydrological cycle, whilst few have been conducted to examine the impact of climate change on water erosion. This is predominately due to the high uncertainty associated with climate change modelling caused by the coarse scale of General Circulation Models (GCMs). Climate change is likely to worsen the impact of water erosion through its effects on rainfall intensity, soil erodibility, vegetative cover and patterns of land use (Nearing *et al.,* 2005). The GCMs can provide a range of climate scenarios, however these alone are not sufficient to predict future erosion risk, particularly as GCMs are currently poor predictors of changes in rainfall intensity and surface wind-speed. In addition to more regionally reliable GCMs, accurate and reliable databases of parameters such as vegetation cover, soil properties, land use, and management systems are required (Nearing *et al.*, 2005).

With respect to climate change and erosion, much will depend on the future pattern, intensity, and seasonality of rainfall events. Important to emphasize the threats of increasing intensity will lead to an increase in erosion. Enhanced biomass production and increased vegetation cover and soil organic-matter content resulting from elevated CO_2 concentrations could potentially have a positive effect that could lead to a decline in soil erosion risk (Brinkman and Sombroek, 1993). However, the more widely predicted higher temperatures, low rainfall and soil moisture suggest that few areas will receive benefits from global climate change. Instead, projected declines in levels of soil organic matter and the weakening of soil structure will make soils increasingly prone to erosion. Modelled estimates of the effect of climate change on soil erosion depend on assumptions regarding the frequency and intensity of precipitation (Phillips *et al*., 1993). Future erosion risk is more likely to be influenced by an increase in population density, the intensive cultivation of marginal lands and the use of resource-based and subsistence farming techniques than by changes in climate (Nearing *et al*., 2004).

2.8 Soil Erosion and Forestry

Forest ecosystems constitute an important component of the global carbon cycle holding 1240 Pg C (Dixon, *et al.* 1994; Lal, 2005), with most of the carbon (67%) being held in the soil, especially in the top-soil, while (33%) is contained in the above ground biomass. Consequently, any disturbance of forests has great potential to impact the global carbon cycle. South Africa's forest resources are classified into three forest types, i.e., indigenous forests, savanna woodlands and commercial timber commercial forests. All three types play an important environmental role in soil protection and act as carbon sinks, thereby mitigating the effects of climate change. South Africa's natural forests are highly fragmented and represent the smallest forest biome (Mucina *et al.*, 2007), the savanna woodlands form the bulk of South Africa's forest land, covering approximately thirty nine million hectares (DAFF, 2015), the majority of this biome occurs in communal areas. In addition to its protective functions, wooded savanna provides a variety of forest goods and environmental services on which rural poor communities depend. Basic demand, particular for fuel wood, pose a threat to the sustainability of these biomes and it is evident that forest degradation and deforestation is taking its toll in the woodland biome (DEA, 2012). Commercial timber forests occupy an area of 1.27 million hectares in South Africa (Godsmark, 2014), predominantly in high rainfall areas which are characterised by frequent high intensity storms and are mostly located in relatively steep or hilly terrain where the potential for erosion is high. Large proportions of commercial forests are on marginal, highly erodible land and erosion is a major hazard to operations (Musto, 1994).

In stable forest ecosystems, where soil is protected by vegetation, erosion rates are relatively low. Tree leaves and branches intercept and diminish rain and wind energy (Pimentel and Kounang, 1998). Forested areas are generally undisturbed and their soils are covered by litter. During thinning operations, skid paths develop along which the logs are moved to the road side. Timber harvesting at the end of the timber rotation involves a considerable disturbance and exposure of the soil surface (Scott *et al.*, 1998). It can thus be assumed that for long periods of time commercial forests have the ability to provide protection against soil erosion, but they can become sources of erosion and sedimentation when disturbed by thinning and harvesting operations and by site preparation for tree establishment.

The erosion-protective action of a commercial forest is dependent on the development stage of the forest. Oliveira *et al.* (2013) evaluated soil, nutrients and organic carbon losses caused by water erosion in *Eucalyptus* forests at different development stages. They found that soil loss decreased with increasing age of the trees. The loss was influenced by soil type and planting system. Furrow planting caused greater soil loss than pit planting and higher losses in nutrients and carbon. The results of this study highlight the need for improved soil conservation practices to prevent soil erosion at the earlier stages of *Eucalyptus* commercial forests. A study by Bulcock and Jewitt (2012) at the Mistley Canema Estate situated in Sevenoaks where they measured the canopy and litter interception, and for Black Wattle they observed a canopy interception of 27.7% and a litter interception rate of 6.6%, these rates have an impact on reducing the effect of rain splash and runoff.

Soil erosion in commercial forestry is directly related to the intensity of silvicultural operations and consequently the extent of surface organic matter removal and soil disturbance. Roads and compacted areas are main contributors of soil erosion (Musto 1994). According to Swank and Johnson (1994), forest management activities such as forest cutting and harvesting interrupt the natural recycling of nutrients and there is concern that nutrients released may affect downstream uses or reduce site productivity. Small scale catchment studies have produced a large body of information on streamwater quality changes in response to forest management, particularly clearcutting. Changes in streamwater nutrient concentrations following cutting vary considerably between localities, even within a physiographic region. Sediment load, dissolved nutrient concentrations are affected by forest management activities (Binkley and Brown, 1983). Changes in these parameters vary, depending on forest ecosystem, management activity, e.g., harvesting and associated logging methods, site preparation methods and stand improvement between initial re-establishment and harvesting which may involve the use of fire, herbicides or fertilizer. A major concern in harvest and regeneration practices is the impact on stream sedimentation (Campbell and Doeg, 1989).

Forest roads and skid trails which are used in log extraction are recognized as a source of erosion and sedimentation of surface waters.

Early studies on soil erosion under commercial forestry by Sherry (1954, 1961, 1964), conducted in the province of KwaZulu-Natal, showed that fires in afforested areas can affect soil erosion rates, a result confirmed later on by Norris (1993) in the same province and by Scott and Van Wyk (1990) and Scott *et al.* (1998) in the Western Cape province. The study by Scott *et al.* (1998) showed that high intensity wildfire in timber commercial forests in the late dry season caused significantly increased sediment yield due to the formation of fire-induced water repellency in the burned soils. Only small increases in sediment yields were observed following prescribed burning of catchments covered in fynbos. The study remarked on the significance of riparian zones in keeping sediment delivery ratios to watercourses low. Among the products of forest fires are partially or minimally altered carbon compounds all the way through to black carbon, which represents highly stable, recalcitrant and biologically inert soil organic carbon (Gimeno-García *et al.*, 2000), associated with an increase in hydrophobicity. The increased fire-induced hydrophobicity on the soil surface and the decreased protection of the soil surface to the action of raindrops by the suppression of the canopy and the litter, leads to a decreased water infiltration (DeBano, 2000) in soils together with an increase in soil disaggregation and soil detachment, all potentiating runoff and soil erosion (Cerdà and Lasanta, 2005; Cerdà and Doerr, 2008; Jackson and Roering, 2009). This potentially affects downstream water quantity and quality as organic compounds such as black carbon have been shown to be preferentially removed by runoff (Rumpel *et al.*, 2006). Soil erosion negatively impacts on soil fertility, productivity and reduces water quality and lead to pollution of watersheds with nutrients and sediments. Forest are susceptible to soil erosion, when established on marginal land. McGarry (2011) has suggested a soil erosion monitoring programme which employs simple, globally applicable and field-usable indicators and measurements to obtain qualitative and quantitative information regarding soil erosion in commercial forests or on recently deforested sites.

Black Wattle is a fast growing, nitrogen fixing tree which is often used in commercial agroforestry (Moyo *et al.*, 2009). In the mid-nineteenth century the species was imported to South Africa from Australia where it originates from where is has been widely planted (De Wit *et al.*, 2001). Currently the species supports a small but very valuable industry, for building materials, charcoal, and firewood for rural people, commercial forestry and associated industries for example tanning products (De Wit *et al.*, 2001: Moyo *et al.*, 2009). Black wattle thrives in areas that exceed 500 mm annually. There is currently over 130 000 hectares of black wattle commercial forests in South Africa, these are specifically located in the provinces of KwaZulu-Natal, Mpumalanga, and previously the Eastern Cape which have subsequently been abandoned. According to Moyo *et al*. (2009) riparian ecosystems are highly threatened by *A. mearnsii* due to their nutrient availability and their ability to disperse. According to the South African Plant Invaders Atlas Datasheet, tall trees are one of the most common invaders of riparian areas, with *A. mearnsii* being the most recorded invader and is one of the top ten most invasive species in South Africa, with an estimated 2.5 million hectares being invaded (Holmes *et al*., 2008, Moyo *et al*., 2009). *A. mearnsii* is said to have negative impacts upon the functionality of riparian ecosystems and further impacts on biodiversity and water resources (De Wit *et al.*, 2001). A conflict of interest therefore exists, where there is the damaging invasive effect on one hand which gives rise to future costs to society, and a commercial value on the other that provides economic value. Commercial forestry has destructive impacts that are often unavoidable and allows invader species to encroach into areas zoned for water production and conservation (De Wit *et al.*, 2001). Black Wattle is one of a number of invasive species in South Africa that is considered to have increased river bank erosion because it is less well adapted to flash floods than native plants (Macdonald and Richardson, 1986). species has been effective in controlling soil erosion on steep slopes and improving soil fertility (NAS, 1980; Waki, 1984).

2.9 Impacts of Soil Erosion on Water Catchments

Water scare countries such as South Africa are becoming gradually more threatened by pollution and sedimentation of water bodies (Le Roux *et al.*, 2013). The extraction of freshwater for industry, agriculture or cities places the health of aquatic ecosystems and the lives they support at risk (Postel, 2000). With the ever expanding population it is crucial to find methods to fulfil humanities water demands sustainably and too protect the life-support functions these aquatic ecosystems provide. The functions include: water as a provisional service, regulatory service and cultural services, functions vital for human well-being and important is sustaining freshwater-dependant ecosystems. Soil erosion impacts negatively on the natural water storage capacity of catchments areas, service of man-made dams, quality of surface water, aesthetic landscape beauty and ecological balance (Doody *et al.*, 2012). The suspended sediments in streams affect water use and ecosystem health. Furthermore, the loss of soil or sediments from land surfaces reduces not only the productivity of agricultural and forestry ecosystems but also leads to silting of dams and eutrophication of water bodies. Off-site impacts include increased flooding due to reduced river channel capacities and the deterioration of river health because of increased turbidity and pollution with pesticides and fertilisers contained in the sediment-laden flows (Van Zyl and Lorentz, 2004).

Sediments are rich in nutrients such as phosphorus and nitrogen which lead to eutrophication of the receiving water bodies and promote excess growth of algae. Areas of excessive algae growth, called algae blooms, deplete oxygen in the water resulting in the death of aquatic animals. According to Le *et al.* (2014), these algae blooms can cause severe water quality problems such as unpleasant odours, dissolved oxygen depletion, increased pH and dissolved organic carbon concentrations and reduced transparency. Several of the bloom forming species (e.g. *Microcystis* sp.) can release toxins which have adverse impacts on livestock, wildlife and human health. The main cause of eutrophication is phosphorus (P) which is transported in solution with eroded soil from agricultural land that applies fetilizer (Ekholm and Lehtoranta, 2012).

Agriculture is currently the main source of sediment input into rivers (Rickson, 2014). According to Collins and Anthony (2008), there is a widespread concern for the environmental problems associated with erosion and subsequent sediment transport into water catchments. Sediment represents a carrier of nutrients, trace and heavy metals, micropollutants and pathogens. High suspended sediment loadings encourage accelerated channel bed siltation and the siltation of reservoirs. Sediment deposition in lakes and rivers increases water turbidity making it difficult for light to penetrate the water, causing problems for aquatic plants that require sunlight for photosynthesis (Palmer *et al*., 2000).

The effect of agricultural systems on sedimentation and P loss is reflected in the results of a long-term study carried out by Bechmann *et al.* (2005) who monitored two subcatchments in Norway under different agricultural management. They observed that the mean annual concentration of suspended sediments in a stream situated in a cereal-growing area with mixed livestock production was 20 times higher than that of the corresponding stream in a grass and dairy cow production system. While suspended sediment losses and losses of total P increased significantly during the monitoring period in the former subcatchment, a significant downward trend in total P loss was observed in the latter.

2.10 Sediment Yield Modelling

Sediment yield may be defined as the mass of sediment leaving a catchment per unit of catchment area. This is crucial for our comprehension of global denudation rates, biogeochemical cycles, fluvial sedimentary archives and human impacts on sediment fluxes (Vanmaercke *et al.,* 2014). High spatial and temporal variability associated with sediment delivery from eroded hillsides to adjacent watercourses must be consistent when estimating sediment yield (Rickson 2014). In addition, the sediment delivery ratio is dependent on complex relationships between sediment characteristics and availability, erosion, transport and deposition process, climate characteristics, landscape, land use, land management practices and the spatial distribution and density of the receiving watercourses, all these factors need to be considered when modelling sediment yield. McHugh *et al*. (2002) suggested that the discrepancies between erosion rates and sediment loads in watercourses result from a failure to account for deposition of eroded material within the catchment. According to Govers (2011), problems arise when erosion data are extrapolated both to longer time-scales than the period of measurement and to larger areas than those of runoff plots. Although erosion control measures need to be implemented at the field or hillslope scale, allocation of scarce conservation resources and development of policies demand regional scale assessment. Slope-scale measurements include; field rainfall simulation studies and the use of delineated runoff plots which provide valuable data on erosion rates under different crop covers and for different soil types. Field data are essential for the calibration and verification of soil loss models, however, field experiments often only apply to one or a few hillslopes and cannot be directly generalised to monitor and determine the soil erosion for an entire catchment (Le Roux *et al*., 2007). Field erosion measurements are not always feasible, in particular in developing countries such as South Africa, due to financial and equipment constraints. However, sediment yield can still be simulated in such areas through the use of mathematical models and empirical methods (Ning, 2006).

"Soil erosion modelling does have limitations including; data variability, over-parameterisation, unrealistic input requirements, unsuitability of model assumptions and misleading parameter values in local context and lack of verification data" (Le Roux *et al*., 2007, pg 330). Assessments of the quality of erosion models have suggested that spatial patterns of erosion are poorly predicted and that the models can rarely be relied upon to provide accurate predictions of absolute amounts of erosion. Without adequate input data and calibration, models can only be expected to provide a relative ranking of the effects of land management. Soil erosion models have been manipulated and used in regional scales for scenario analysis and to deliver objective comparisons to guide strategy and aid in soil conservation efforts in South Africa (Le Roux *et al*., 2007). There is an increasing range of decision-making tools available which attempt to close the gap between management and research (Van Zyl and Lorentz, 2003) and can assist managers in directing their conservation efforts. These tools include; sediment yield models to predict future changes in erosion and sediment delivery in catchments and streams due to human interventions. The models can be experienced-based or they could be sophisticated numerical models (Rutherford *et al*., 1996). As many types of sediment yield models have been developed for specific conditions and purposes, it is important to document their limitations to understand and to choose the appropriate model for an intended purpose.

Models generally fall into three main categories; empirical, conceptual and physically based models (Le Roux *et al*., 2007). The most widely implemented empirical models for estimating soil loss at the regional scale are the Universal Soil Loss Equation (USLE) developed in the 1970s by the United States Department of Agriculture and its upgraded version the Revised Universal Soil Loss Equation (RUSLE). Conceptual models are accurate in terms of incorporating the underlying transfer mechanism of sediment and runoff generation in their structure, representing flow paths in a catchment as a series of storages (Arnold and Fourier, 2005). Physically based models have an even more sophisticated model structure, being based on the solution of fundamental physical equations which describe stream flow and sediment on a hillslope or in a catchment (de Vente and Poesen, 2005).

The data requirements of models are significantly greater when adding spatial and temporal data. Le Roux *et al*. (2007) state that the description of water fluxes over and through the soil is the base of an erosion model. Additional information such as the information regarding agricultural activities greatly improves the quality of the results. Complex models tend to be restricted to research catchments and are prohibitive in terms of time required for implementation on a regional basis as required by government policies (Merritt *et al.,* 2003). Empirical models are frequently preferred to more complex models, in particular at a regional scale. They can be implemented in areas with limited data and are particularly useful as a first step in identifying sources of sediment. Techniques involving Geographic Information Systems (GIS) and algorithms for digital terrain analysis are readily available and are currently improving the hydrological process description in models (Moore *et al.*, 1991).

In South Africa, national based studies have been summarized in terms of their method and scale of application since 1991 (Le Roux *et al.*, 2007). The Global Assessment of Human-induced Soil Degradation (GLASOD) was one of the first major regional-scale degradation studies. Areas were divided into relatively uniform units based on the most important erosion processes and ranked according to their soil erosion risk and a soil erosion risk map was produced at a continental scale (Sonneveld and Dent, 2009). In 1993 remote sensing was used to monitor soil erosion at a national scale and the Bare Soil Index (BSI) was created with data from Landsat Thematic Mapper (TM). The index was accurate in identifying rural settlements, overgrazed areas and eroded areas in Mpumalanga and the Eastern Cape but was unable to differentiate between ploughed fields and sandstone outcrops from eroded areas. Due to the low resolution of Landsat TM, single gully, rills or sheet erosion could not be delineated by remote sensing (Le Roux *et al*., 2007). Although South Africa has large scale maps to identify broad areas where the risk of erosion is high, these maps are of limited use for erosion prediction and control at the scale of small catchments (<10 km^2). Erosion predictions at this scale were predominately conducted using the Universal Soil Loss Equation (USLE) technology which, however, is not suited to predict the off-site impacts of erosion (van Zyl and Lorentz, 2004). Since most regional-based soil erosion studies in South Africa focus on assessing the factors controlling erosion such as rainfall erosivity, soil erodibility, slope length and conservation practices, USLE and RUSLE, which take these factors into account, are the most widely applied models.

The Soil and Water Assessment Tool SWAT is a catchment-scale, continuous time model operating on a daily time-step developed by the US Department of Agriculture (USDA) Agricultural Research Service (Arnold and Fohrer, 2005). It can be used to simulate water, sediment and chemical fluxes in large catchments with varying climatic conditions, soil properties, stream channel characteristics, land use and management practices. It has gained international acceptance and has been applied to support various large catchment (10–10 000 km^2) modelling studies with minimal or no calibration effort. SWAT is often paired with geographical information systems which allows improved manipulation and organisation of spatial data (Le Roux *et al*., 2013).

The Agricultural Catchments Research Unit (ACRU) model is a daily time step, multi-purpose integrated physical conceptual model developed by the Agricultural Catchments Research Unit within the previous Department of Agricultural Engineering of the University of Natal in Pietermaritzburg, South Africa (Schulze, 1995; Smithers and Schulze, 2004). The physically based model requires data input for meteorological parameters such as daily rainfall, historical flow records, topographical information and information about soils, land use and vegetation cover. The model takes into account the greater spatial and temporal variability associated with soil erosion rate by dividing the catchment into Hydrological Response Units (HRU), based on the assumption that these units are hydrologically relatively homogenous as the result of assumed homogeneity in terms of climate, soil type and land cover. The ACRU model therefore has the potential to effectively indicate which sub-catchments contribute most to the sediment yield. ACRU has the capability to utilize long-record daily rainfall data available for research catchments of interest as input parameters. It can simulate stream flow, sediment yield, total evaporation, and land cover/management and abstraction impacts on water resources at a daily time step and has been verified in semi-arid regions. Warburton *et al*., (2010) demonstrated that the ACRU agrohydrological model is useful in predicting the impact of land use change on the hydrological response.

The ACRU model uses the Modified Universal Soil Loss Equation (MUSLE) module (Williams, 1975) to simulate soil erosion. MUSLE was developed by Williams and Berndt (1977) to be used under semi-arid conditions based on the Universal Soil Loss Equation (USLE). While USLE is limited to the estimation of average annual soil loss, MUSLE can be used to estimate both annually soil loss and soil loss caused by a single storm event (Clutario and David, 2014). In addition, MUSLE eliminates the need to determine sediment delivery ratios which were used by USLE to estimate the proportion of eroded soil which leaves the catchment (Basson, 2004). However, MUSLE has the tendency to significantly overestimate sediment yield in large catchments (> 2000 km^2).

The Water Erosion Prediction Project (WEPP) model was developed by the United States Department of Agriculture. The WEPP model is based on modern hydrological and erosion science and simulates the erosion processes of detachment and transport by raindrop impact on interrill areas, the detachment, transport and deposition by overland flow in rill channels, the detachment, transport and deposition by concentrated flow in channels and the deposition in impoundments (Laflen *et al*., 1991).

2.11 Field Assessment Techniques

Soil erosion monitoring can be carried out on-site (at plot level) and off-site (at sub-catchment and catchment levels). The advantages and limitations of these two monitoring approaches are currently the subject of debate (Hartanto *et al*., 2003). Many studies on soil erosion have been conducted at sub-catchment or catchment levels. Although this approach can better describe the response of a catchment to certain management practices, instream monitoring is expensive and time consuming as monitoring should include a calibration period. On-site monitoring is generally easier to conduct and is less costly. This type of monitoring is best suited to observing soil erosion processes and soil disturbances. Periodical sampling is usually adopted to estimate nutrient losses, however it often underestimates nutrient losses as storm events are more critical for nutrient losses, in particular in the subtropics (Tang *et al*., 2008).

2.11 Strategies to Reduce Soil Erosion

Global concerns regarding environmental disturbances as a consequence of erosion call for concerted efforts to improve the management of ecosystems to minimise soil, nutrient and soil organic carbon (SOC) losses and reduce sedimentation.

According to Van Zyl and Lorentz (2004), it is becoming increasingly better understood that erosion control should be linked to both soil (on-site erosion) and water (off-site sedimentation) conservation initiatives. Several countries have incorporated ‘clean water strategies’ into agricultural policies, legislation and programmes (Parry, 1998). Ekholm and Lehtoranta (2012) mention that methods such as the establishment of buffer strips, riparian zones and wetlands and the construction of settling ponds are recommended for the protection of water and thus the reduction of P loads and eutrophication. They suggest, however, that the link between erosion and aquatic eutrophication is more complex than previously thought and needs to be examined from a wider perspective than merely accounting for the loading and bioavailability of soil-bound P. When studying the effect of soil erosion and its control, not only the processes occurring in the water phase should be considered but also those which take place after the soil particles have settled to the bottom which are driven by microbes in the aquatic sediments (Ekholm and Lehtoranta, 2012).

Erosion is a natural process which cannot be stopped. It can, however, be minimized to an acceptable rate. The maximum acceptable rate of erosion is known as the soil loss tolerance. According to Morgan (1988), a mean annual soil loss of 1.1 kg/m^2 is generally accepted as the maximum permissible, however values as low as 0.2 to 0.5 kg/m^2 are recommended for particularly sensitive areas where soils are shallow and highly erodible. Although preservation efforts should aim to reduce soil loss to those acceptable values, this objective may be under some circumstances unrealistic, in particular in mountainous areas which receive high rainfall. The recommendations on soil loss tolerance are however based on agricultural considerations and ignore problems of pollution and sedimentation, in particular nitrogen, phosphorus and organic matter, and pesticides leave a field either in solution in the runoff or attached to sediment particles (Morgan, 1988).

In South Africa, there is an increasing move toward more sustainable ways of living and food production. Farmers have at their disposable a number of conservation practices which can significantly decrease soil erosion rates (WWF, 2009). Combining a number of these practices is often more effective. The ideal goal is to reduce the soil loss rate to 6.7 t/ha^{-yr}. This is approximately the rate at which soil can rejuvenate itself (DA, 2008).

Ensuring that the soil is always covered with vegetation and that the soil is rich in organic matter are two key methods to prevent soil erosion. Organic matter content influences soil erosion through its effect on the stability of aggregates (Tisdall and Oades, 1982; Guerra, 1994). Organic matter in soil can be increased with crop rotation or by incorporating organic fertilizers. Other methods which can be used by farmers to reduce soil erosion are mulching, which involves the spreading of plant residues over a field, reduced tillage, cultivation of cover crops and contour cropping.

Soil conservation measures can range from covering the soil to protect it from rain splash, improving the infiltration capacity of the soil to reduce runoff, improving the aggregate stability of the soil and increasing surface roughness to reduce the velocity of runoff and wind (Morgan, 1988). Agronomic or biological measures utilize vegetation to reduce soil erosion and afford protection to the soil. Introducing vegetation cover is generally the preferred method since it is comparatively cheap to implement and reduces the impact of rain splash, increases infiltration, reduces runoff volume and decreases wind and water velocities. Mechanical or physical methods (for example contour bunds, terraces, waterways, silt fences) attempt to control the energy available for soil erosion. Mechanical methods are effective in controlling the transport phase but do little to reduce soil detachment and are costly to install and maintain. Agronomic measures combined with sound soil management, can reduce erosion in the soil detachment and transportation phases (Morgan, 1988).

Akbarimehr and Naghdi (2012) suggest two methods to reduce erosion and sediment movement on forest roads and skid trails and prevent off-side impacts: post-harvest water diversion through the use of drainage culverts on forest roads and water bars on skid trails, and the rehabilitation of forest roads and skid trails through the establishment of vegetation cover

For conservation measures to be efficient and cost effective, the identification of areas susceptible to erosion, which have the potential to be the main sources of sediment, is critical. The conservation practices must be closely related to the nature of the erosion problem and must consider the intricacy of the erosion process. Sumner (1995) states that strategies for erosion control in areas where accelerated erosion presents a problem to land management can only be achieved through an understanding of the soil erosion processes and their interaction with different conservation practices.

2.12 Conclusion

Soil erosion is the process of detachment and transportation of soil materials by wind or water. It leads to the loss of fertile topsoil on-site, decreases soil productivity and reduces crop yields over time. It affects water catchments through sedimentation and eutrophication of waterways. The degree of soil erosion ranges from splash erosion to gully formation with the factors that influence the rate at which soil erosion occurs being wind and rainfall intensity, soil erodibility, topography, vegetation cover, soil management practises and conservation measures. Anthropogenic processes and certain land use types have led to accelerated soil erosion in South Africa. Soils are generally fragile: they have low organic matter and are susceptible to high rates of erosion, with the dominant agent causing erosion being water through rainfall and runoff (Van Zyl *et al*., 1996, Le Roux *et al*., 2008). It estimated that approximately five to seven million hectares of productive topsoil are lost annually through erosion and the rate of soil erosion in South Africa is likely to increase due to a projected increase of extreme events such as floods caused by global climate change (Van Oost *et al*., 2000).

Commercial timber forests occupy an area of 1.27 million hectares in South Africa (Godsmark, 2014). The forestry industry provides employment and is a significant contributor to the country's economy. Environmentally, commercial forests have the ability to provide protection against soil erosion with the commercial forests canopy cover reducing runoff and soil loss and the litter cover protecting the soil from erosion during intense rainfall. However, they can become sources of erosion and sedimentation when disturbed by thinning and harvesting operations and by site preparation for tree establishment.

Soil erosion has off-site impacts which affect water bodies, which is a major problems for water scarce countries such as South Africa who are becoming increasingly threatened by pollution and sedimentation of water bodies. Soil erosion impacts negatively on the natural water storage capacity of catchments areas, service of man-made reservoirs and dams and quality of surface water. Sediment carry nutrients, trace and heavy metals, micro-pollutants and pathogens which may lead to eutrophication of water bodies. With the on-site and off-site impacts of soil erosion, it is becoming increasingly necessary to reduce soil erosion as erosion control is linked to both soil (on-site erosion) and water (off-site sedimentation) conservation initiatives. Erosion is a natural process which cannot be stopped, however it can be minimized to an acceptable rate through management practices. Soil erosion modelling has become a necessary tool used in estimating the amount of soil loss in areas and this allows decision makers to create mitigation measures to reduce the amount of soil loss. This is necessary as the threat of global climate change is predicted to increase soil erosion.

Field data is vital in calibrating and verifying soil erosion models, but, they cannot be generalised to monitor and quantify soil erosion for an entire catchment (Le Roux *et al*., 2007). This research included collecting field-based data which can be used to help populate and verify models. This is a good opportunity as field erosion measurements are not always feasible, in particular in developing countries such as South Africa, due to financial and equipment constraints (Ning, 2006).

Chapter Three

Methods

3.1 Introduction

This chapter is divided into two sections: site description and the experimental methods. The site description provides a description of the environmental conditions of the study site, the geology, vegetation and the climate, and provides information on the experimental infrastructure within the catchment. The experimental methods section outlines the instruments that were used to collect data and how samples were collected; and details the subsequent laboratory methods.

3.2 Site Description

KwaZulu-Natal a province in the eastern part of the country, has large areas of moderate to extremely high potential erosion risk (90%) but relatively low actual erosion risk (18%) due to vegetation cover (le Roux *et al.*, 2008). The rate of soil loss varies geographically and according to land use practices. The study catchment was the Two Streams catchment which is part of the Mistley Canema Estate and is situated near Sevenoaks, 20 km outside Greytown (Figure 3.1). The urban and peri-urban areas of the town of KwaDukuza (previously known as Stanger), Zinkwazi, Darnall and Groutville are located within this region. This catchment's location is a one hour drive from the University of KwaZulu-Natal Pietermaritzburg campus, and it accessible for regular data collection by being close enough to respond quickly to rainfall events.

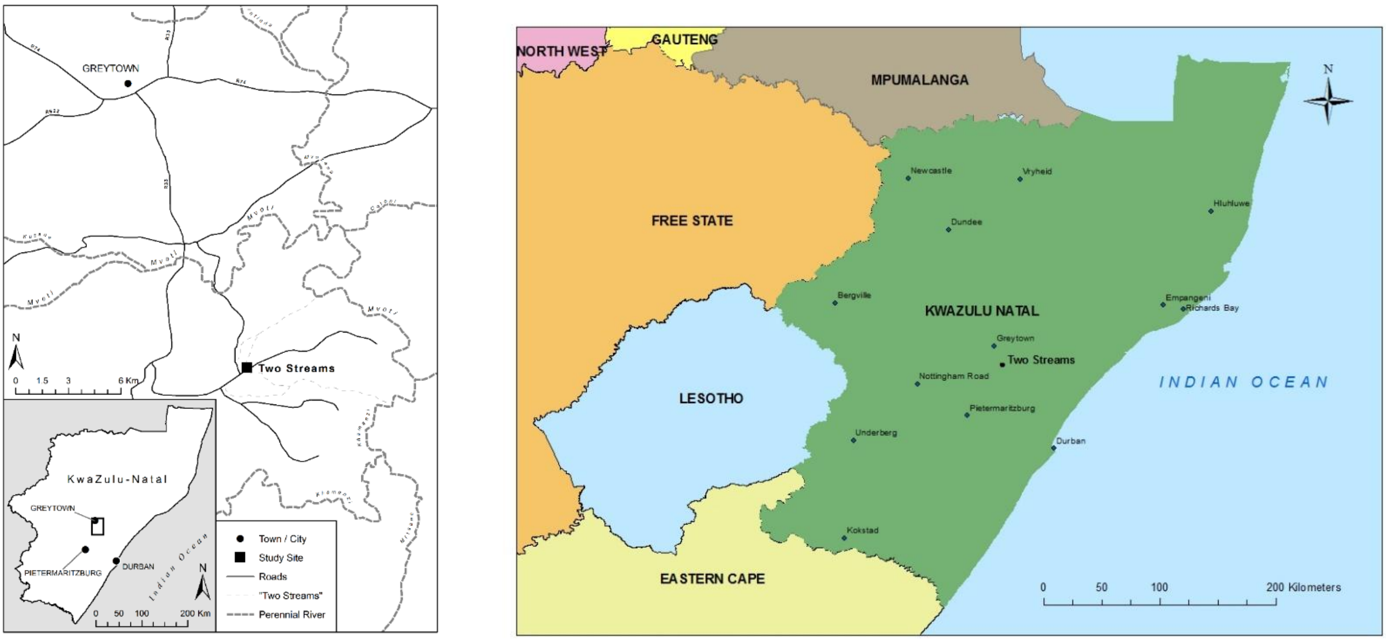

Figure 3.1 Locality maps of the Two Streams Catchment

The bioregion for the area is ‘midlands mistbelt grassland’, and is characterised by an undulating rolling landscapes, with a large proportion of the land being arable (Clulow *et al.*, 2012). It is dominated by forb-rich, tall, sour *Themeda triandra* grasslands of which only a few patches remain due to invasion of native *Aristida junciformis.* The soil formations are apedal and plinthic and are derived from the Ecca Group with dolerite dykes and sills. The land cover consists primarily of communal land in the inland areas, commercial timber in the upper reaches of the Mvoti catchment and dryland and irrigated sugar cane along the coastal strip. Summer thunderstorms or cold fronts cause most of the rain with an annual rainfall ranging from 659 to 1139 mm (Clulow *et al.*, 2012). Mist can be heavy and frequent and might add significantly to precipitation. Moderate frosts, droughts, hail and berg winds are common and the average number of heavy frost days per annum range from 31 to 60 days for inland areas.

The soils are underlain by well weathered sandstone saprolite generally to a depth of 4-5 meters. The Inanda profile is situated in a lower midslope position with a slope of approximately 4%. The Magwa profile is situated on a slope of 0.25% at the footslope just above the valley bottom, where the Katspruit soil is located. There is a high humus content in the A horizons. This is attributed to the hydrophobic nature of the A horizons in this catchment. In most cases water repellence in soils can be attributed to coatings on the soil particles of hydrophobic substances of organic origin, especially under wattle plantations. A soil map for the area can be retrieved from (Le Roux *et al,* 2015).

Two Streams is a thirty-four hectare catchment which has had a number of study sites established across the catchment.Over the last fifteen years the catchment has been intensively instrumented and the hydrology of the catchment monitored. Some sites have been well established during the course of previous research projects. For example, a weir was constructed in 1999, an Automatic Weather Station (AWS) was setup in 2006 and boreholes were drilled in 2001 and 2007. This current project benefited significantly from the established sites which were refurbished and maintained during the course of the project. However, to fulfill the specific objectives of this project a number of new sites and monitoring strategies were implemented.

Following a previous clear felling of the catchment in 2005 there was a single rainfall event of nearly 90 mm observed (Clulow *et al.*, 2012) which caused widespread erosion across the exposed areas and sedimentation of the weir and riparian areas. The hypothesis contributing factors were the intensity of the rainfall, the large areas of bare soil, water repellent soils, slope and lack of management strategies to reduce runoff. In 2010, widespread burning in the catchment during winter for firebreaks and burning of slash piles caused severe damage to the soil due to the heat of the fires. For six months following these burns, severe erosion and sedimentation was observed in the catchment (Clulow *et al.*, 2012). The monitoring and research from this site was to establish results prior to harvesting and the intention is for more research to continue post-harvest to analyse the land use change and management.

The study site had runoff plots and micro-plots installed in different slope locations at Two Streams (Figure 3.2). Measurements from the plots are taken prior to clear-felling to determine the sediment loads of slopes in an afforested catchment. At the outlet of the catchment a weir was constructed and maintained, an ISCO sampler and an automatic sampler were installed.

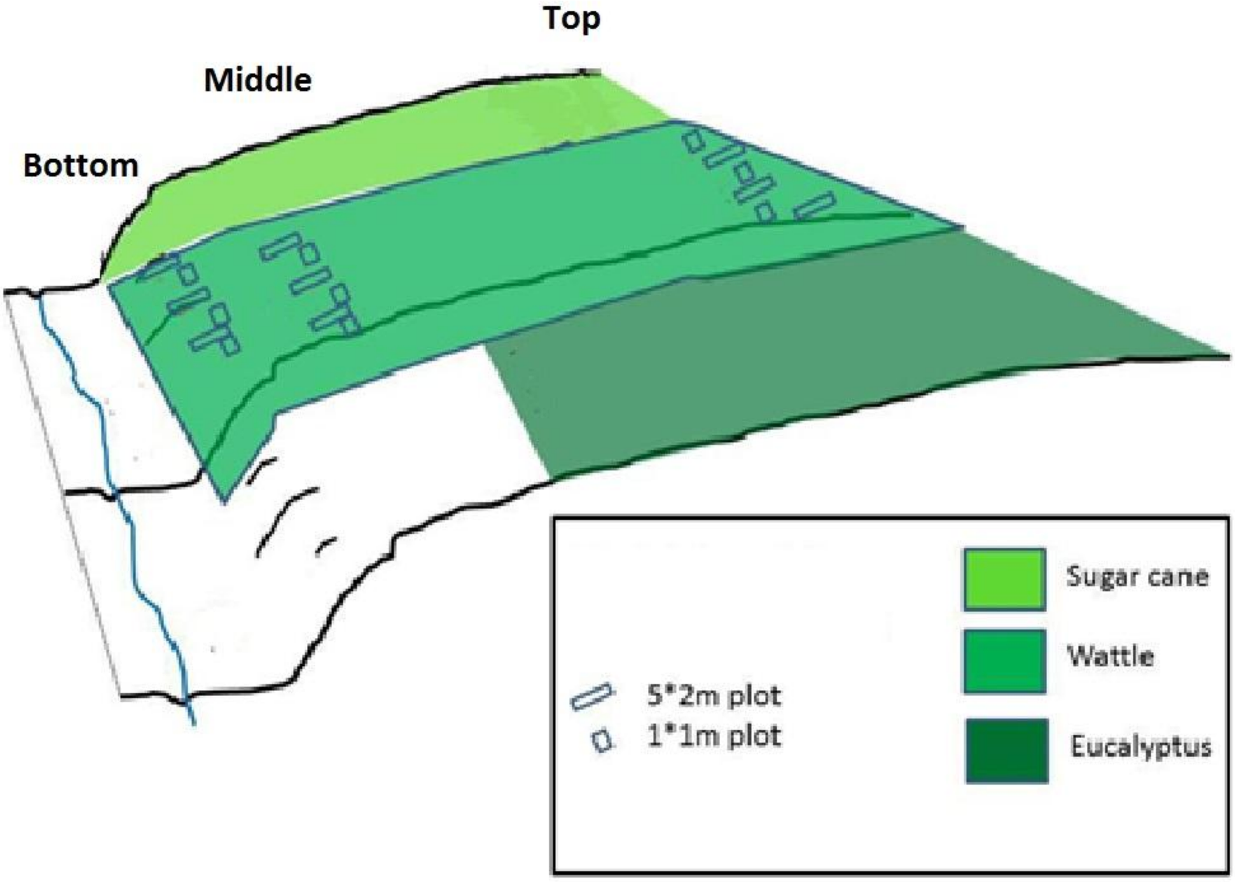

Figure 3.2 Experimental design of the study site

3.3 Field Based Methods

3.3.1 Rainfall Data

Rainfall data were obtained from manual rain gauges which were located adjacent to the runoff plots (Plate 3.1). These were installed to measure the spatial variability of rainfall within the catchment. The rain gauges were set up 1.5m above the ground under tree cover. This was done purposefully to determine the amount of interception and to aid in determining the proportion of that rainfall reached the surface. Water samples were collected from the rain gauges at each location and analysed in the lab to ensure that there was no high nutrient concentration in the rainfall that might impact the runoff results. To gain accurate measurement of rainfall for the area, rainfall data were retrieved from an

Automatic Weather Station (AWS) that was set-up 500m outside of the commercial forest. The AWS provided temporal data of each rainfall event, allowing for event duration to be calculated.

Plate 3.1: Manual rain gauge to measure rainfall under the canopy

3.3.2 The Nested Scales Used for Water and Nutrient Fluxes Evaluation

Three scales of spatial analysis occurred:

a) Micro-plots (nine 1m^2 runoff plots were installed at three different hillslope positions).

b) Runoff plots (nine 10m^2 runoff plots were installed at three different hillslope positions).

c) 34 ha catchment (An ISCO sampler was installed, from which flow height is recorded by a data logger. An automatic water sampler was located at the outlet of the catchment).

3.3.2.1 Micro-plots

Nine 1m x 1m (1 m^2) runoff micro-plots were installed within the catchment with three replicates per slope position (Plate 3.2). The micro-plots were installed at three topographical positions. The metal borders surrounding the micro-plots were inserted to a depth of 0.1 m in the soil and installed parallel to the slope direction. This allowed for any generated overland flow to be directed down the slope and into the gutter of the micro-plot. The gutter was designed to channel and concentrate water into the bottom of the gutter. The gutter fed into the outlet of the micro-plot, connected to a pipe, which fed into a bucket to capture the water. After each site visit, total overland flow volume (R) from each micro-plot replicate was measured with a measuring cylinder and a 500 ml representative sample of the water collected. The sediment in the gutters was flushed down into the bucket with the sample water. Micro-plots provide information on the contribution of rain splash and rain-impacted flow on sediment mobilisation. Cognisance needs to be taken that these plots are capable of over- and under-estimating the overall soil water erosion (Govers and Poesen, 1988)

Plate 3.2: 1 m^2 micro-plot (1x1 m)

3.3.2.2 Plots

Nine 5m×2m (10 m^2) runoff plots were installed adjacent to the micro-plots with three replicates per slope position (Plate 3.3) (Appendix A).The metal borders surrounding both the micro-plots and runoff plots were inserted in the soil to a depth of 0.1 mm, and were installed parallel to the slope direction. This allowed for any overland flow that was generated to be directed down the slope and into the gutter of the plot. The gutter had rain shields to prevent direct rainfall into the gutter which would compromise the results. The gutter was designed to channel and concentrate water into the bottom of the gutter, into the outlet of the plot, connected to a pipe which fed into a 300 l collection tank which stored the overland flow water. Buckets were placed where the water fed straight into, making it easier to measure the volume of small rainfall events. Runoff plots are useful tools to evaluate interill erosion as they provide information on the impact that generated runoff flow has on sediment loss. (Chaplot and Le Bissonnais, 2003).

Plate 3.3: 10 m^2 plot (5x2 m)

During heavy rainfall events the runoff water overflowed out of the bucket and into the tank (Plate 3.4 and 3.5). At each site visit, total overland flow volume (R) from each plot replicate was measured using a measuring cylinder if the water was in the bucket. When the water overflowed into the tank the volume of overland flow was measured by using the volume of the tank and calculating the depth of the water in the tank. A 500ml sample of the water was collected. The sediment in the gutters was flushed down using the collected water.

Plate 3.4: Runoff fills the bucket before the JOJO tank

Plate 3.5: JOJO tank gathers the runoff

A single plot at each of the three different slope positons were connected to a pipe which first fed into a tipping bucket system before being collected in the bucket inside the tank. At the three locations, the tipping bucket mechanism was connected to a HOBO event-logger (Pendant logger) (Plate 3.6). The tipping bucket mechanism was calibrated to tip after every two litres for the 5m x 2m runoff plots. Each tipping bucket was individually calibrated. In addition, the specific time at which the tip occurred, was logged. This was to ensure that the temporal response of each individual plot location (in terms of overland flow) was measured after the onset of a rainfall event.

Plate 3.6: Tipping bucket connected to the hobo logger to measure amount of runoff

3.3.2.3 Catchment Monitoring

To study catchment scale processes an ISCO sampler was installed at the gauging weir outlet to integrate the sedimentation load from the catchment. At the catchment outlet (approximately 34 ha) there is a V-notch weir with a 13-year record of stream flow. The existing logger was coupled to an ISCO 6712 and 3700 series, automatic sampler (Plate 3.7). The height of flow at the catchment outlet was logged by a data logger. Catchment water quality (nutrients and sediments) during both baseflow and stormflow events were measured by collecting samples at the appropriate locations on the hydrograph curve.

Plate 3.7: ISCO 6712 and Automatic Sampler

3.3.3 Spatial and Temporal Variation

The installation of the plots and micro-plots at different locations and in conjunction with a tipping bucket connected to a data logger, was to account for the spatial and temporal variations of overland flow in the catchment. Fifteen representative runoff events were selected for detailed study of the temporal and spatial variations of overland flow. These events were chosen, as all plot locations recorded overland flow and it was assumed to be representative of events, when overland flow was generated within the catchment.

3.3.4 Site Visits

The frequency of site visits depended on the frequency and intensity of the rainfall. High frequency and high intensity rainfall required regular visits to the Two Streams Catchment to collect samples. The summer season required regular visits as rainfall was high, whilst the winter months required fewer site visits. The design and procedure for collecting each samples remained constant. There was a total of fifteen site visits during the course of this study from January 2015 and ending in March 2016.

3.4 Water Quality

Water samples were collected at the different spatial scales and used to assess water quality of the runoff. Water samples were collected manually (from runoff collecting buckets at the micro-plots and plots). The water quality constituents are the Nitrates-Nitrogen (NO_3-N), Total phosphorus (P), Dissolved Organic Carbon (DOC) and Particulate Organic Carbon (POC). POC was defined as the fraction of carbon which had been bonded onto soil particles and then subsequently eroded, POC included any organic matter which had been eroded. DOC was defined as the fraction of carbon which has been dissolved into solution by rainfall and soil water. Water samples were collected in the field by taking 500 ml samples and stored in a cooler box on-site. Once back at the laboratory, samples were stored in a fridge, which was kept at a constant temperature of 4^{o}C until completion of analysis.

3.4.1 Sediment

Water samples were filtered using Ø47 mm filter paper, the filtered sample was dried at 110°C for 24 hours. Samples were placed in a furnace at 550°C for 2 hours to burn off the organic matter. This was then multiplied by the volume of water (l) to determine the sediment concentration (g/l). The sediment yields for each nested scale were calculated by multiplying the sediment concentration (gl^{-1}) by the runoff flux per unit area (l/m^2).

3.4.2 Nitrates and Phosphate Measurement

NO_3^- and P concentration in the water samples was obtained using an AQUALYTIC spectrophotometer AL800. The absorbance of the water samples was read, using an AQUALYTIC spectrophotometer AL800 and converted to concentrations (given as mgl^{-1}). The accuracy of all nutrient analyses were within 10% of the concentrations. Nitrite was initially measured in the study but the values were below detection limit.

3.4.3 Dissolved Organic Carbon

Dissolved organic Carbon (DOC) was performed by Umgeni Water, using a Shimadzu TOC-5000 analyzer with an ASI-5000 autosampler and Balston 78-30 high purity total organic carbon (TOC) gas generator. In this technique, the organic solutes are converted to CO_2 and the CO_2 produced is measured as DOC (in gl^{-1}). Concentrations were converted to yields (in mgl^{-1})

3.4.4 Particulate Organic Carbon

Sediments were dried at 110°C for 24 hours. The sediments were weighed to determine sediment concentration in runoff to compute sediment losses. The samples were placed in a furnace at 550^{o}C for 2 hours to burn off the particulate organic carbon. The samples were then weighed and the difference in mass determined POC. The data were converted to (gl^{-1}).

3.5 Water Repellence

Water repellence was measured at all the sites at the runoff plots and the micro-plots during sampling. This was done by dropping a drop of deionised water (approximately 6 mm diameter) from a height of 1.5 cm on to the surface of the soil. The length of time the drop remains on the surface is taken as the index of water repellence. This procedure was repeated three times at each micro-plot and runoff plot to obtain a consistent and accurate measurement.

3.6 Measurement of Slope, Soil Properties and Vegetation

To determine the variation in slope between the different slope positons. The slope in degrees were measured, using an inclinometer and a ranging rod. The slope of each plot position was measured and was done by standing in front of the plot with the inclinometer and the ranging rod was held at the back of the plot, the change in slope was then read from the inclinometer. To measure soil properties, the top 10 cm of the soil at each plot position was taken by hammer a 7 cm wide steel ring 10 cm into the soil. The samples were then taken to Cedara soil analytical laboratory to provide a chemical analysis of the soil. The Braun Blanquet classification method was used to determine the vegetation cover and abundance at each 10 m^2 and 1 m^2 runoff plot. The classification measured the amount of trees, the average tree diameter at breast height, aerial cover, litter cover and grass cover at each runoff plot.

3.7 Statistical Methods

The statistical methods include; creating comparative tables between the different runoff plot sizes and the different spatial scales. Averages and standard errors are provided to allow for a comparison between the different spatial scales. Scatter graphs allow for a visual comparison between the different plot sizes. To determine if there was any variability between the plot sizes, t tests were run using Microsoft Excel 2013.

3.8 Conclusion

The study site is the Two Streams catchment located 20 km outside of Greytown in KwaZulu-Natal. The Bioregion for the region is ‘midlands mistbelt grassland’ and the soil formations are apedal and plinthic and are derived from the Ecca Group with dolerite dykes and sills. The annual rainfall ranges from 659 to 1139 mm.

The study site is instrumented with nine runoff plots (10 m^2) and nine micro-plots (1 m^2) at specific slope positions within the afforested catchment. These plots determined the spatial variation within the catchment. Rain gauges were installed adjacent to the plots and at three of the sites, tipping bucket rain gauges were installed to determine the amount and rate of runoff leaving the runoff plots. Rain gauges were installed to inform on the amount of rainfall reaching the surface whilst an AWS was installed adjacent to the study site to provide accurate climatic data. To study catchment scale processes an ISCO sampler had been installed at the gauging weir outlet to integrate the sedimentation load from the catchment, an ISCO 6712 and 3700 series automatic sampler was used to collect samples intermittently. After a rainfall event, the amount of runoff was quantified and samples were taken back to the laboratory, with data collection taking place for two rainfall seasons. Samples were measured for sediment, phosphate and nitrate concentration, dissolved and particulate organic carbon. The objective of the methods were to set up an appropriate experimental design to answer the research question.

Chapter Four

Results

4.1 Introduction

The chapter details the relationship between rainfall and runoff and the impact on sediment yield and nutrient concentrations (nitrate, phosphate and carbon) at the different spatial scales.

Rainfall, slope, soil properties and vegetation on-site are described. Plot slope was measured to determine variation in slope steepness as plots that have a steeper slope are able to generate higher flow velocity (van Oost *et al.*, 2000). Soil properties were measured to determine any variation in the erodibility of the soil which is based on the soil's physical and chemical properties (Bissonnais, 1996). Vegetation was described to determine if there was any variation in canopy or litter cover as change in vegetation may affect the volume of runoff (Fournier, 2011).

4.2 Priori Results

4.2.1 Rainfall

The study took place over fourteen months which included two summer rainfall seasons and a low winter rainfall period (Table 4.1). A rainfall season can be defined when rainfall is frequent and when most of a region's average annual rainfall occurs and a non-rainfall season, when rainfall is less frequent and not expected (Wang, 2002). The cumulative total rainfall over the study period was 1135.2 mm over 428 days. Total rain for the rainfall season of 2014-2015, November - February (four months) was 422.5 mm; 275.7 mm for the non-rainfall season of 2015 (eight months) and 437 mm for the rainfall season of 2015-2016 (four months) (Table 4.1) (Appendix B). The annual cumulative rainfall for 2015 was 782 mm. When comparing the amount of rainfall to previous years; 2012 received (958 mm), 2013 (871 mm) and 2014 (712 mm). This study was undertaken in a relatively dry year (average rainfall 659-1139 mm) (Clulow *et al.*, 2011). The highest monthly rainfall occurred in the summer months, with an uncharacteristically high rainfall in July 2015 (Figure 4.1). Site visits took place after high rainfall events, with the majority of site visits taking place during the summer months (Figure 4.2). The AWS recorded an intense rainfall event on the 18th December 2015 with a total 114.6 mm of rainfall falling in the space of a few hours

Table 4.1: Rainfall characteristics for the different rainfall seasons (2014-2016). Cumulative annual rainfall amount (Cum).

Season	Cum
	mm
November 2014-February 2015	422.5
March 2015-October 2015	275.7
November 2015-March 2016	567.0

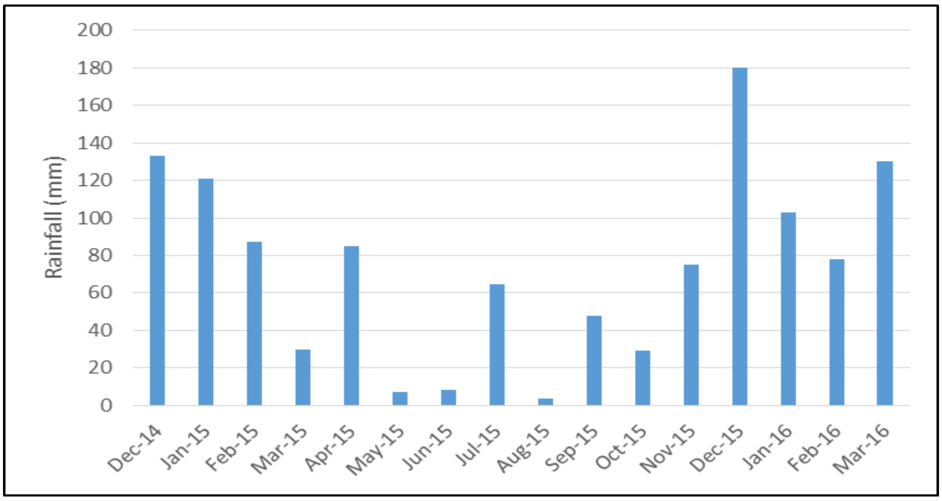

Figure 4.1: Monthly rainfall at Two Streams from December 2014 to March 2016

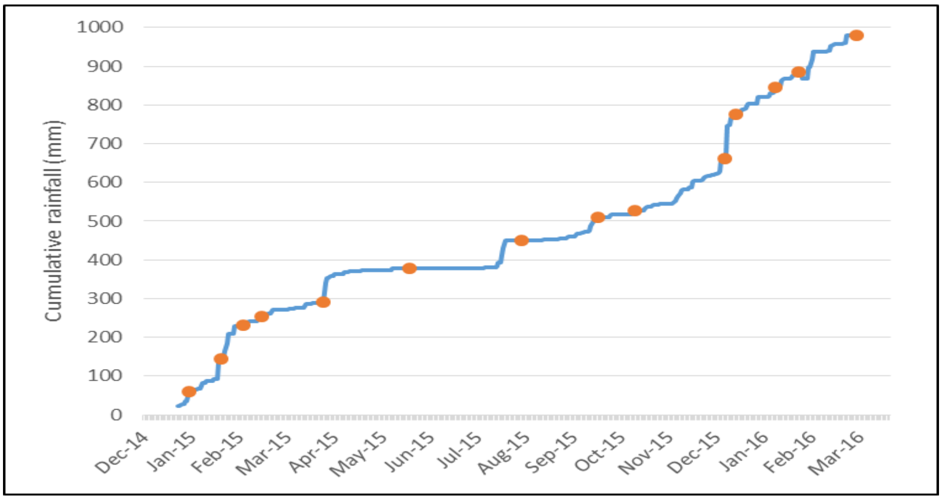

Figure 4.2: Cumulative rainfall and site visits represented by points

Rain gauges were set-up to determine the spatial variation in rainfall within the study site (assuming no evaporation and a consistent canopy cover), high rainfall events had greater variation between the rain gauges compared to small rainfall events, with an average variation of 2 mm of rainfall for a rainfall event (Figure 4.3).

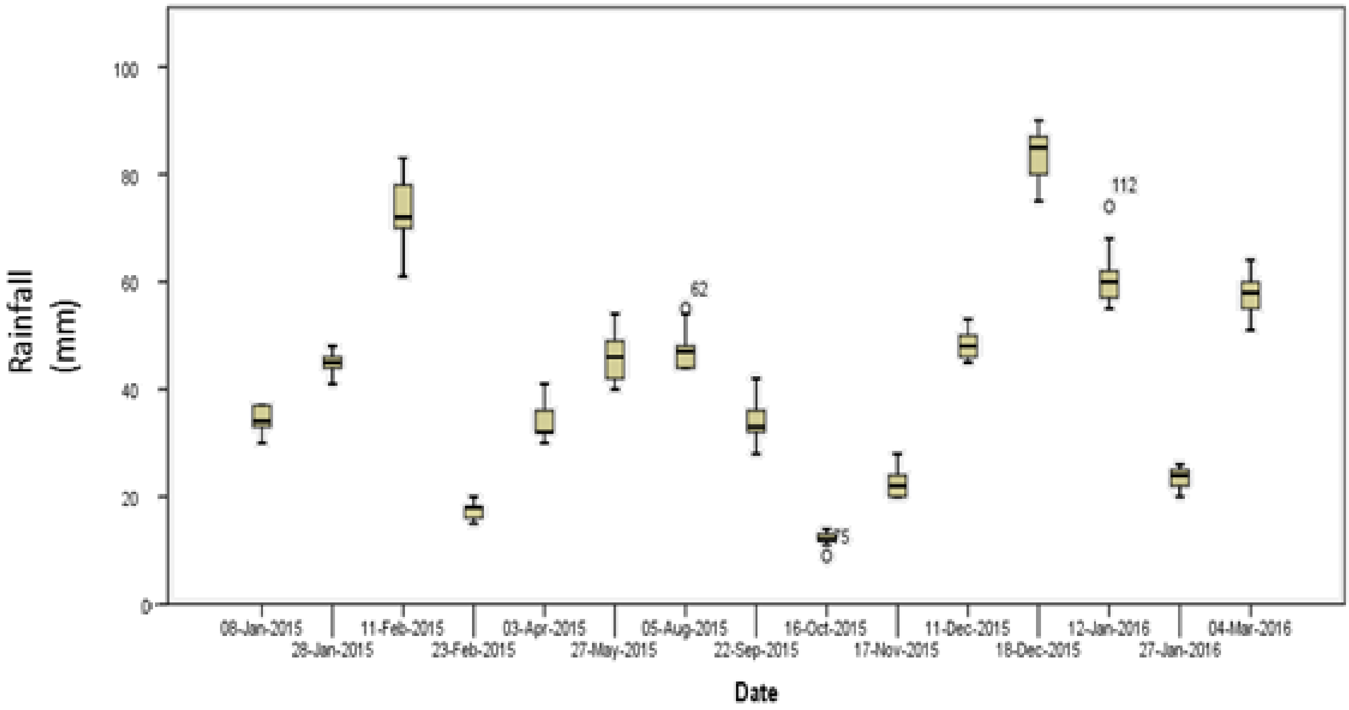

Figure 4.3: Box and whisker plots of average rain gauge readings for study duration

A comparison between the Automatic Weather Station (AWS) and the in-field rain gauges, illustrates the vital role of canopy cover in intercepting rainfall. The AWS consistently recorded higher rainfall then the rain gauges (Table 4.2). There was an average interception of 34.1%, assuming no evaporation from the rain gauges. Seasonality did not influence interception rates as the rainfall season 2014-2015 had an interception rate of 31.68%, the non-rainfall season of 2015 had an interception rate of 32.88% and the rainfall season of 2015-2016 had an interception rate of 36.98%. There were two site visits 3rd April 2015 and 12th Jan 2016 that had slightly lower interception rates than what was usually experienced 6.2% and 9.6% respectively this may have been due to evaporation from the rain gauges or human error by taking higher rain gauge readings then what actually was experienced.

In a study by Bulcock and Jewitt (2012) at the Mistley Canema Estate situated in Sevenoaks (the same commercial forest as the present study) they measured the canopy and litter interception, from April 2008 to March 2011. For *Acacia mearnsii* they observed a canopy interception of 27.7%. This study measured slightly higher canopy interception rate possibly due to evaporation from the rain gauges.

Table 4.2: Summary rainfall table

Site visit date	AWS (mm)	Rain gauge (mm)	Interception (%)
08-Jan-15	60.1	34.1	43.1
28-Jan-15	84.0	44.8	46.7
11-Feb-15	88.0	73.3	16.7
23-Feb-15	22.0	17.6	20.2
03-Apr-15	36.2	33.8	6.2
27-May-15	85.0	45.8	46.1
05-Aug-15	71.9	47.8	33.5
22-Sep-15	58.9	34.1	42.1
16-Oct-15	18.9	12,0	36.5
17-Nov-15	54.2	22.7	58.2
11-Dec-15	79.1	48.4	38.8
18-Dec-15	114.6	83.6	27.1
12-Jan-16	67.6	61.1	9.6
27-Jan-16	48.3	23.6	51.2
04-Mar-16	89.9	58.0	35.5
Average total	**68.6**	**42.7**	34.1

4.2.2 Slope

The runoff plots at the bottom positions had the steepest gradient (9.3^0), with the top and middle slope positions having similar gradients (4.0^0 and 5.6^0 respectively). The exception being site 6 on the middle slope position with a 7.9^0 gradient (Table 4.3).

Table 4.3: Slope steepness of each runoff plot

Slope Position	Site number	Slope (°)	Average Slope(°)
Top	Site 1	4.1	4.03
	Site 2	4.2	
	Site 3	3.8	
Middle	Site 4	4.8	5.6
	Site 5	4.1	
	Site 6	7.9	
Bottom	Site 7	8.9	9.33
	Site 8	8.4	
	Site 9	10.7	

4.2.3 Soil

Soil analysis was undertaken by the Cedara soil analytical laboratory which provided information on the total carbon and nitrogen in the soil (Table 4.4.). The total carbon (0.45%) and nitrogen (6.37%) concentrations were consistent for all runoff plots. Soil fertility provided information on the following elements: phosphorous, potassium, calcium, magnesium, zinc, manganese and copper. Furthermore, the amount of cations, acid saturation and pH was measured (Table 4.5). The data were consistent across the catchment, with only site nine recording any notable change, with higher calcium and cations values and lower acid saturation. The soil texture was sandy silt at all runoff plots.

Table 4.4: Total percentage of soil nitrogen and carbon

Slope Position	Sample No.	Total % Nitrogen	Average Total % Nitrogen	Std. dev Total % Nitrogen	Total % Carbon	Average Total % Carbon	Std. dev Total % Carbon
	Site 1	0.43			5.96		
Top	Site 2	0.45	0.46	0.042	6.68	6.67	0.71
	Site 3	0.51			7.38		
	Site 4	0.58			7.3		
Middle	Site 5	0.32	0.44	0.132	4.55	5.86	1.38
	Site 6	0.41			5.73		
	Site 7	0.47			6.8		
Bottom	Site 8	0.48	0.46	0.026	6.86	6.59	0.423
	Site 9	0.43			6.1		
		Total Average	0.45	0.072		6.37	0.892

Table 4.5: Chemical analysis of soils

Slope Position	Sample ID	Sample density (g/mL)	P (mg/L)	K (mg/L)	Ca (mg/L)	Mg (mg/L)	Exch. acidity cmol/L	Total Cations cmol/L	Acid sat. %	pH (KCl)	Zn (mg/L)	Mn (mg/L)	Cu (mg/L)
	Site 1	0.86	14	37	216	32	2.56	4	64	3.71	4.8	21	1.8
Top	Site 2	0.86	11	61	151	30	2.9	4.06	71	3.67	2.3	14	1.5
	Site 3	0.81	14	53	155	22	3.02	4.11	73	3.58	3.9	17	1.6
	Site 4	0.83	12	40	131	17	2.27	3.16	72	3.82	0.6	28	1.4
Middle	Site 5	0.88	5	40	103	22	2.74	3.54	77	3.68	0.2	19	1.7
	Site 6	0.87	8	32	91	8	2.72	3.32	82	3.66	0.3	29	1.6
	Site 7	0.9	5	67	202	32	3.14	4.58	69	3.76	0.5	14	1.7
Bottom	Site 8	0.87	14	47	172	14	3.58	4.67	77	3.71	0.5	25	2.1
	Site 9	0.94	6	67	555	107	2.11	5.92	35	3.86	0.7	19	2.2

4.2.4 Vegetation

The Braun Blanquet classification method was used to determine the vegetation cover and abundance at each plot (Table 4.6). In the commercial forest the litter consisted of twigs and leaves that fell from the *Acacia mearnsii*. The species richness of the area was low with the dominant grass species in the commercial forest being *Eragrostis tef*.

Table 4.6: Vegetation abundance at each site using the Braun Blanquet method

	10 m²					1 m²		
Site number	Individual Trees	Average tree diameter at breast height (dbh) (mm)	Aerial Cover	Litter Cover	Grass Cover	Aerial Cover	Litter Cover	Grass Cover
Site 1	4	140,1	4	5	2	3	5	1
Site 2	4	148.4	4	5	r	3	5	r
Site 3	4	130.4	3	5	1	3	5	+
Site 4	3	124.2	3	5	2	3	5	3
Site 5	4	139.0	4	5	2	4	5	1
Site 6	4	118.5	3	5	2	3	5	2
Site 7	4	122.5	4	5	1	2	5	3
Site 8	4	139.4	4	5	1	3	5	5
Site 9	4	136.6	5	3	5	3	5	5

Average tree dbh in mm (n=100)	167.1

Note: The ratings for the Braun Blanquet are: r = very small cover, rare occurrence, + = cover less than 1%, 1 = cover between 1-5%, 2 = cover between 5-25%, 3 = cover between 25-50%, 4 = cover between 50-75% and 5 = cover more than 75%.

4.3 Experimental Results

4.3.1 Runoff

This section provides details on the runoff at the different slope positions and spatial scales (Table 4.7). It also provides information of the measured water repellency as this determine if the rainfall was likely to infiltrate into the soil or run directly off the soil.

Table 4.7: Average runoff at the different plot locations. Three plot replicates are located at each site: top slope; middle slope and bottom slope. A total of nine 10 m² plots and 1 m² plots with fifteen rainfall events were recorded.

	Top	Middle	Bottom
(n=15)			
Average 10 m²	9.78 l/m²	6.48 l/m²	10.46 l/m²
Average 1 m²	8.87 l/m²	7.49 l/m²	13.49 l/m²

The plots on the bottom slope position averaged the highest volume of runoff with the 10 m^2 plots averaging 10.46 l/m^2 and the 1 m^2 plots 13.49 l/m^2. There was a decline in runoff at the top slope with the 10 m^2 plots averaging 9.78 l/m^2 and the 1 m^2 plots averaging 8.87 l/m^2. The plots on the middle slope positons averaged the lowest volume of runoff with the 10 m^2 plots recording on average 6.48l l/m^2 and the 1 m^2 plots recording 7.49 l/m^2. In terms of variation of runoff between the runoff plots, for high runoff events there was high variation between the runoff plot compared to small runoff events which had low variation (Figures 4.4 and 4.5). On the 18th December 2015 there was a maximum runoff event of 28 l/m^2 for the 10 m^2 runoff plot and 24.71 l/m^2 for the 1 m^2 runoff plots. The 23th February 2015 recorded the minimum runoff event of 0.42 l/m^2 for the 10 m^2 runoff plot and 2.07 l/m^2 for the 1 m^2 runoff plots, this was due to low rainfall and possibly from antecedent conditions reducing runoff. During the duration of the study (approximately fifteen months), a total of 1327 l (132.7 l/m^2) of runoff ran off a single 10 m^2 plot whilst 139 l (139 l/m^2) of runoff ran off a 1 m^2 runoff (Table 4.8). The commercial forest became increasingly less effective at reducing runoff as the amount of precipitation per storm increased. Pervious surfaces are highly affected by antecedent moisture conditions, as they will produce a greater rate of runoff when they are wet than when they are dry.

For the rainfall season of 2014-2015 (four months), the maximum runoff volume was 14.59 l/m^2 at the 10 m^2 plots and 12.37 l/m^2 at the 1 m^2 plots (both on 11th February 2015), the minimum runoff volume was 0.40 l/m^2 at the 10 m^2 plots and 0.89 l/m^2 at the 1 m^2 plots (both on 23rd February, 2015). For the non-rainfall season of 2015 (eight months), the maximum runoff volume was 21.79 l/m^2 at the 10 m^2 plots and 9.44 l/m^2 at the 1 m^2 plots (both on 27th May 2015), the minimum runoff volume was 0.51 l/m^2 at the 10 m^2 plots and 2.91 l/m^2 at the 1 m^2 plots (both on 3rd April 2015). For the rainfall season of 2015-2016 (four months) the maximum runoff volume was 28.66 l/m^2 at the 10 m^2 plots and 24.71 l/m^2 at the 1 m^2 plots (18th December 2015). The minimum runoff volume was 0.42 l/m^2 at the 10 m^2 plots and 2.06 l/m^2 at the 1 m^2 plots (27th January 2016). Note there were two site visits that had a high amount of runoff the 27th May 2015 and 18th December 2015. The 18th December 2015 was due to an intense rainfall (114.6 mm) and the 27th May 2015 was possibly due to high rainfall (85 mm) but more likely due to antecedent moisture as the rainfall on that date was lower than some other rainfall events, thus it is more likely that rainfall occurred when there was already antecedent moisture present causing greater surface runoff. The AWS data shows that for the 27th May site visit there had been small but consistent rainfall events, this meant antecedent moisture was present and thus when rainfall did occur greater surface runoff was generated.

The threshold rainfall is the amount of rainfall is always required before any runoff occurs may be only in the range of 3 mm while in other catchments this value can easily exceed 12 mm, particularly where the prevailing soils have a high infiltration capacity. The fact that the threshold rainfall has first to be surpassed explains why not every rainstorm produces runoff. This is important to know when assessing the annual runoff-coefficient of a catchment area. This study required generally more than 10 mm of rainfall for runoff to be found in the JOJO tanks.

Table 4.8: Total runoff from the runoff plots (l)

Date	Number of days since previous collection	10 m² (l)	1 m² (l)
8-Jan-15	11	9.68	3.18
28-Jan-15	20	94.50	8.38
11-Feb-15	14	145.89	12.37
23-Feb-15	12	4.05	0.89
3-Apr-15	39	5.17	2.19
27-May-15	54	217.93	9.44
5-Aug-15	70	55.52	4.40
22-Sep-15	48	46.63	6.22
17-Nov-15	56	28.48	5.79
11-Dec-15	24	151.24	17.56
18-Dec-15	7	286.58	24.71
12-Jan-16	25	119.69	20.31
27-Jan-16	15	4.24	2.10
4-Mar-16	37	158.10	21.82
	Total	**1327.64**	**139.34**
	Average	**94.83**	**9.95**

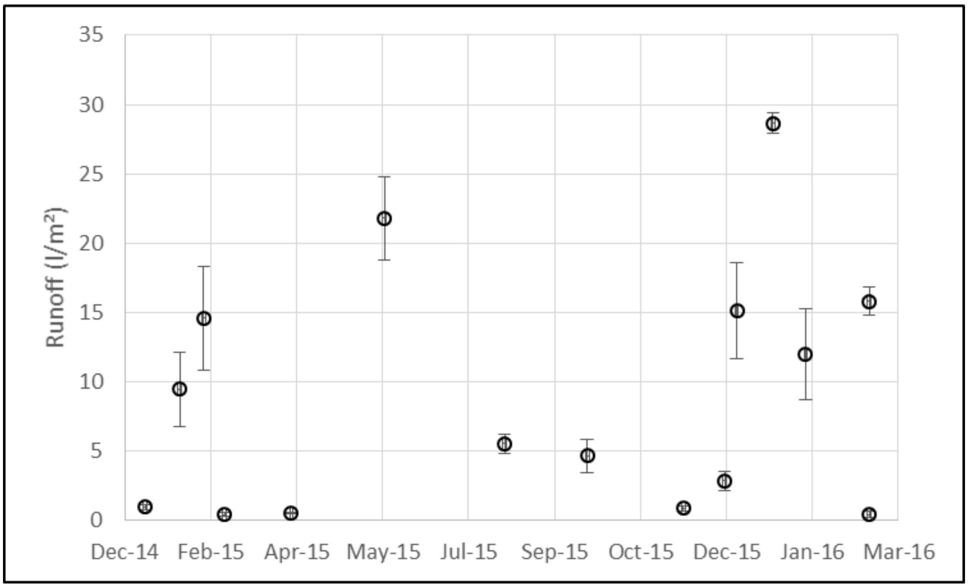

Figure 4.4: Average runoff and standard error for 10 m² plots

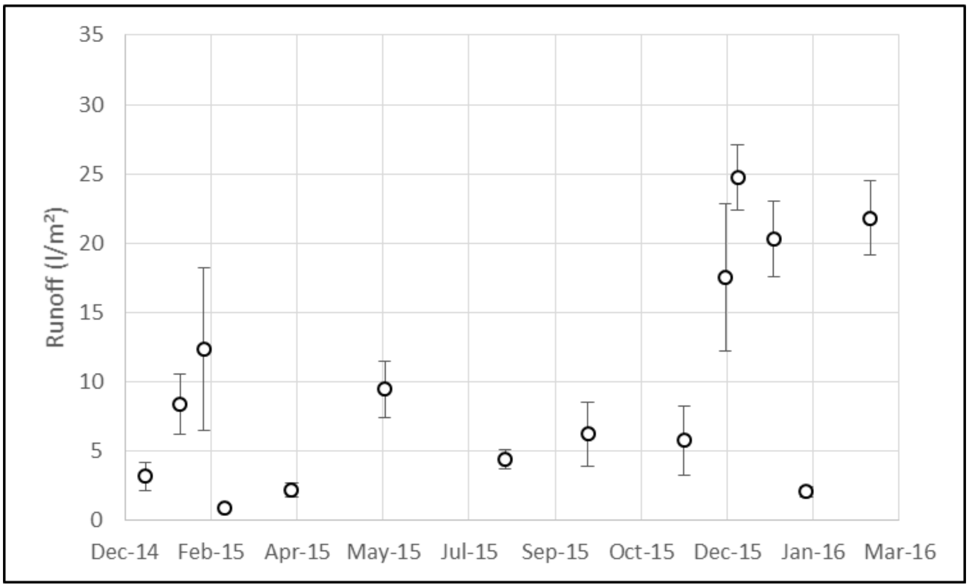

Figure 4.5: Average runoff and standard error for 1 m² plots

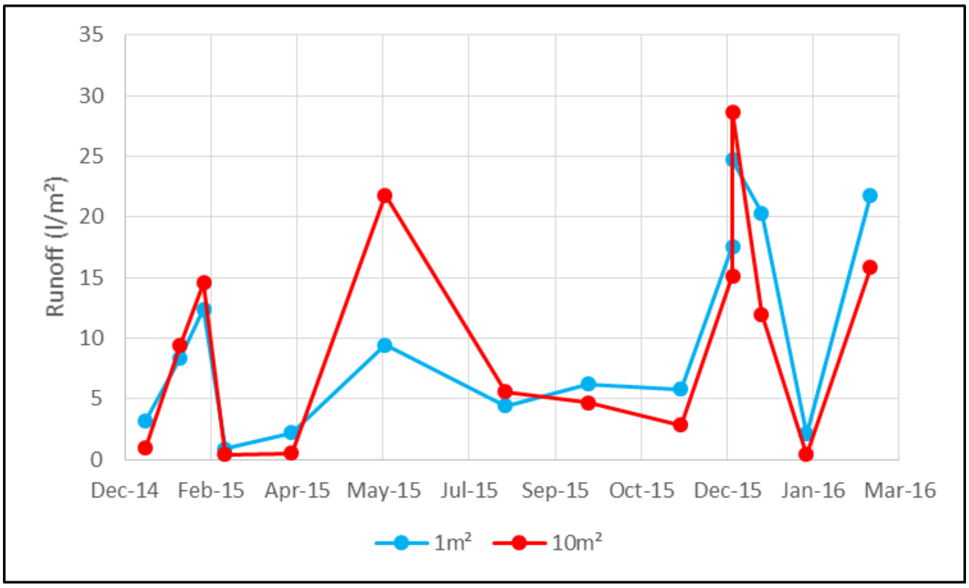

Figure 4.6: Average runoff at the 1 m² and 10 m² runoff plots

The runoff volume (l/m²) off the 10 m² and 1 m² plots were similar during the summer months (Figure 4.6). In the winter period, the 10 m² plots produced a higher volume of runoff compared to the 1 m² plots, possibly as a consequence of the winter months lower rainfall amounts and intensities. The different seasons would have different processes, in summer rain-splash erosion is prominent due to higher intensity rainfall events, in winter the rainfall events are less intense and the process of runoff flow is likely to be prominent. The average runoff in 2015 was 9.50 l/m² for the 10 m² runoff plots and 8.65 l/m² for the 1 m² runoff plots. The full set of runoff results can be found in Appendix C.

Repellency was consistent for the plots over the study duration, with the repellency being tested at every site visit and remaining constant regardless of season. When the soil was dry, the soil was highly repellent, with it taking a drop of water over 5 minutes to infiltrate into the soil (Table 4.9). This highlights the soil being hydrophobic and the conditions in the catchment having low infiltration and high surface runoff, so when rainfall did occur, the rainfall that reached soil would runoff. Important to note that when the water repellency test was done on an area of that plot that had antecedent moisture, infiltration would occur immediately. These tests show that throughout the study, when a rainfall event did occur so in turn did high amounts of runoff, with low amounts being infiltrated. Table 4.9 has the same reading throughout the study.

Table 4.9: Water repellency of the soil at the different slope positions

Date	Top	Middle	Bottom
08-Jan-15	>5min	>5min	>5min
28-Jan-15	>5min	>5min	>5min
11-Feb-15	>5min	>5min	>5min
23-Feb-15	>5min	>5min	>5min
03-Apr-15	>5min	>5min	>5min
27-May-15	>5min	>5min	>5min
05-Aug-15	>5min	>5min	>5min
22-Sep-15	>5min	>5min	>5min
17-Nov-15	>5min	>5min	>5min
11-Dec-15	>5min	>5min	>5min
18-Dec-15	>5min	>5min	>5min
12-Jan-16	>5min	>5min	>5min
27-Jan-16	>5min	>5min	>5min
04-Mar-16	>5min	>5min	>5min

4.3.2 Sediment Yield

This section provides details on the sediment yield at the different slope positions and spatial scales (Table 4.10). The average sediment yield is provided as it illustrates the differences between plot locations and sizes.

The slope position with the highest sediment yield was the middle slope position with the 10 m^2 plot producing 0.909 gl^{-1} and the 1 m^2 plots 0.804 gl^{-1} the middle slope position also recorded the lowest runoff, this may demonstrate that rain splash was the dominant cause of sediment loss and not runoff. There was a decline in sediment removal at the bottom plot position with the 10 m^2 plots producing 0.897 gl^{-1} and the 1 m^2 plots 0.834 gl^{-1} which was the slope position that had recorded the highest runoff volume. The slope position that had the lowest sediment eroded was the top slope positions with the 10 m^2 plots on average producing 0.837 gl^{-1} and the 1 m^2 plots 0.834 gl^{-1}.

Table 4.10: Sediment yield comparison (average volume) at the different plot locations (top, middle, and bottom) for the different plot

	Top	Middle	Bottom
(n=15)			
Average 10 m^2	0.837 gl^{-1}	0.909 gl^{-1}	0.897 gl^{-1}
Average 1 m^2	0.834 gl^{-1}	0.804 gl^{-1}	0.790 gl^{-1}

For the rainfall season of 2014-2015 (four months) (four events), the maximum sediment yield for a rainfall event was 0.811 gl^{-1} at the 10 m^2 plots and 0.802 gl^{-1} at the 1 m^2 plots, the minimum sediment yield for a rainfall event was 0.623 gl^{-1} at the 10 m^2 plots and 0.770 gl^{-1} at the 1 m^2 plots. For the non-rainfall season of 2015 (eight months) (four events), the maximum sediment yield for a rainfall event was 0.801gl^{-1} at the 10 m^2 plots and 0.801 mgl^{-1} at the 1 m^2 plots, the minimum sediment yield for a rainfall event was 0.740 gl^{-1} at the 10 m^2 plots and 0.749 gl^{-1} at the1 m^2 plots. For the rainfall season of 2015-2016 (four months (six events) the maximum sediment yield for a rainfall event was 1.784 gl^{-1} at the 10 m^2 plots and 1.090 gl^{-1} at the 1 m^2 plots. The minimum sediment yield for a rainfall event was 0.814 gl^{-1} at the 10 m^2 plots and 0.740 gl^{-1} at the 1 m^2 plots. Important to note that most of the maximum values recorded were during high rainfall events and the minimum values occurred during low rainfall events, these results point to rainfall as a key driver of sediment loss

During the duration of the study (approximately fifteen months), on average a total of 1.38 kg (138 g/m^2) of sediment was removed from a 10 m^2 plot from runoff, whilst 0.119 kg (119 g/m^2) of sediment had been removed from a 1 m^2 runoff plot through runoff. The amount of sediment removed from the plots was correlated with the rainfall/runoff amount. With high rainfall/runoff associated with high sediment yield. Highest contributor to sediment yield came on the 18th December 2015 which was an intense rainfall event (114.6 mm in 2 hours), led to high runoff which in turn led to high sediment yield (Table 4.11).

Table 4.11: Total sediment volume from the runoff plots

Date	10 m² (g)	1 m² (g)
08-Jan-15	6.03	2.55
28-Jan-15	75.63	6.62
11-Feb-15	118.32	9.75
23-Feb-15	3.23	0.69
03-Apr-15	4.14	1.64
27-May-15	174.42	7.56
05-Aug-15	41.08	3.29
22-Sep-15	38.24	4.69
17-Nov-15	26.45	4.93
11-Dec-15	119.90	13.67
18-Dec-15	511.29	26.94
12-Jan-16	105.20	18.67
27-Jan-16	3.46	1.53
04-Mar-16	149.63	16.29
Total	**1377.00**	**118.85**
Average	**98.36**	**8.49**

There was generally low variation between the runoff plots (Figures 4.7, 4.8). T tests also confirmed these results. The sediment volume (gl^{-1}) from the 10 m^2 plot and 1 m^2 plots was similar throughout the study. With a sediment yield of 138 g/m^2 at a 10 m^2 plot compared to 119 g/m^2 at a 1 m^2 plot. Only the 18th of December (due to an intense rainfall event) showing a marked difference, with the 10 m^2 plots producing a significantly higher sediment volume (Figure 4.9).

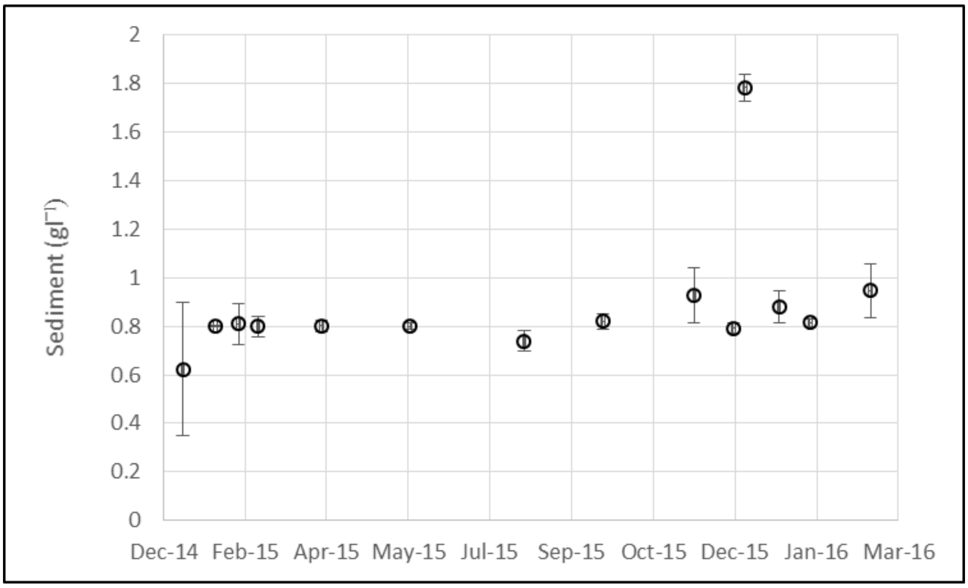

Figure 4.7: Average sediment yield and standard error for 10 m^2 plots

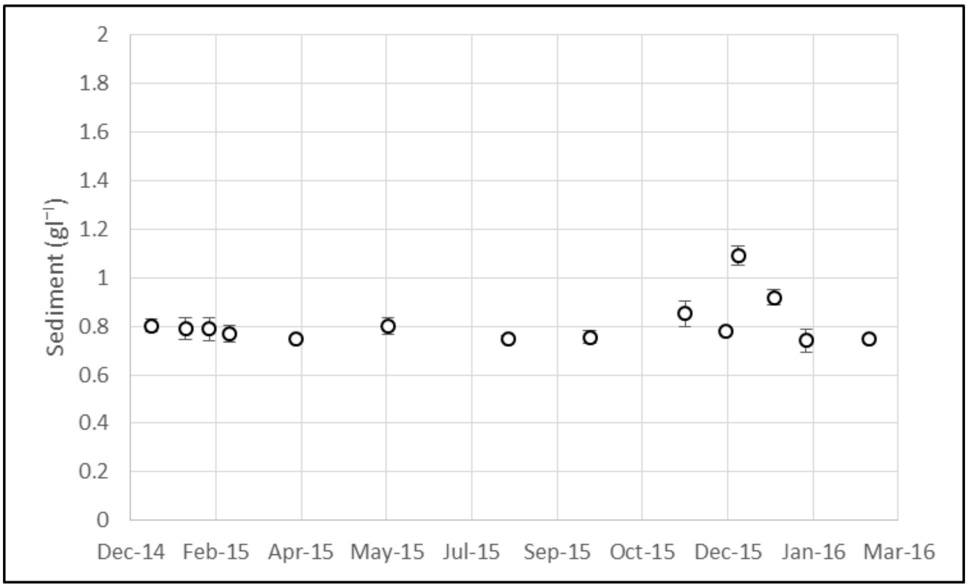

Figure 4.8: Average sediment yield and standard error for 1 m^2 plots

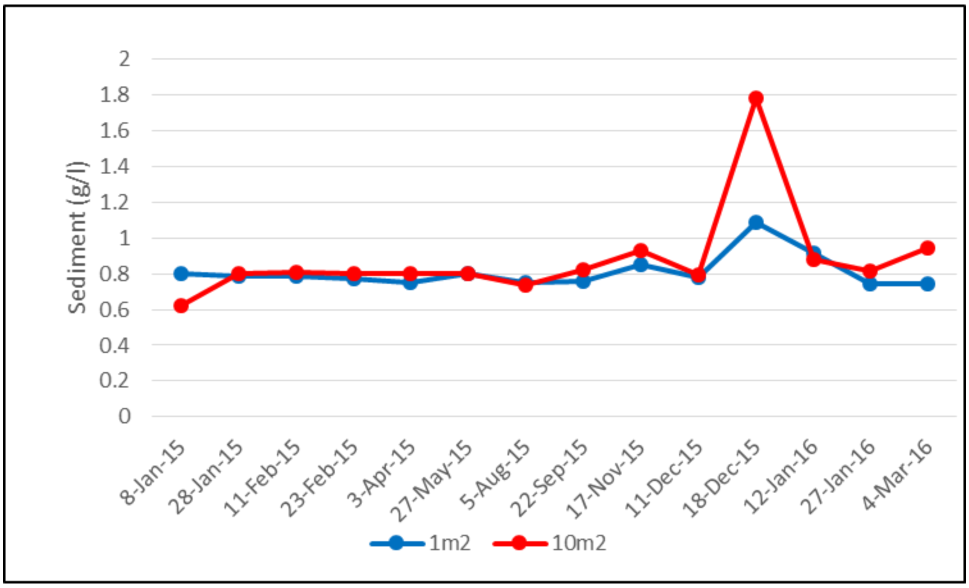

Figure 4.9: Relationship between 1 m^2 and 10 m^2 plots for sediment loss

The highest cumulative sediment yield was from the 34 ha catchment, followed by the 10 m^2 and then the 1 m^2 plots (Figure 4.10). Due to unfortunate circumstances the sampler at the weir was damaged during an intense rainfall event and was not fixed before the study ended, thus the 34 ha section of the graph has not being completed. It can be assumed though that it would continue to increase in the summer months. To obtain an accurate measurement it may be of use in the future to model sediment yield. The summer months had an increase in sediment yield due to the increase in rainfall. The full set of sediment yield results can be found in Appendix D.

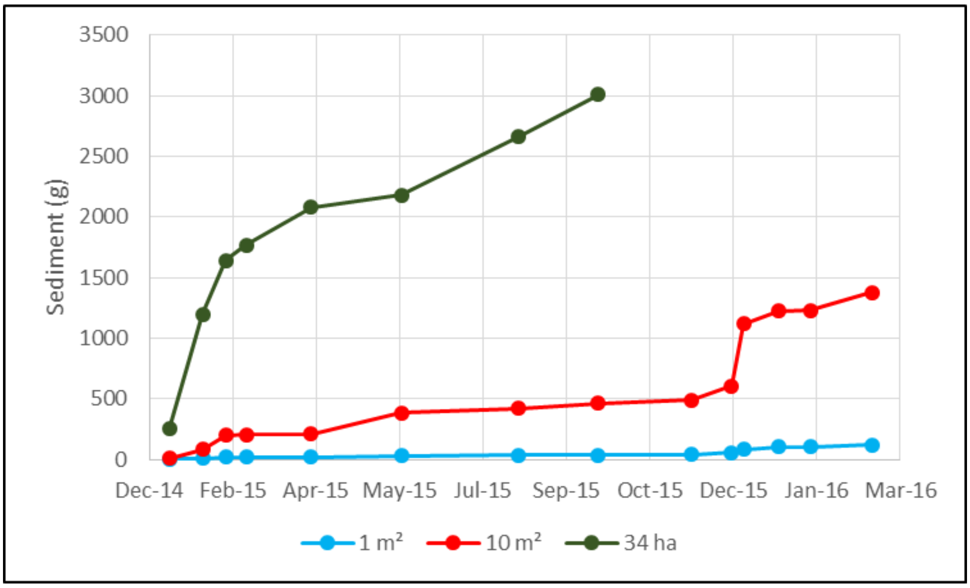

Figure 4.10: Cumulative sediment yield at the various scales (1 m², 10 m² and 34 ha)

4.3.3 Phosphate

Phosphate is regarded as a key contributor to eutrophication. This section provides details on the phosphate concentrations at the different slope positions and the different spatial scales (Table 4.12). The average phosphate concentration is provided as it illustrates the differences between plot locations and sizes.

The highest average concentration of phosphate was measured at the bottom slope position with the 10 m^2 plots recording on average 0.72 mgl^{-1} and the 1 m^2 plots recording 0.33 mgl^{-1} per event. There were lower values recorded in phosphate concentration at the top slope with the 10 m^2 lots recording on average 0.51 mgl^{-1} and the 1 m^2 recording 0.25 mgl^{-1} and the lowest phosphate concentration was recorded off the mid slope with the 10 m^2 plots recording on average 0.39 mgl^{-1} and the 1 m^2 plots recording 0.1 mgl^{-1}.

Table 4.12 Phosphate concentration (average volume in mgl^{-1}) at the different plot locations (top, middle, and bottom) for the different plot sizes.

	Top	Middle	Bottom
(n=15)			
Average 10 m²	0.51 mgl^{-1}	0.39 mgl^{-1}	0.72 mgl^{-1}
Average 1 m²	0.25 mgl^{-1}	0.14 mgl-^{1}	0.33 mgl^{-1}

For the rainfall season of 2014-2015 (four months), the maximum phosphate concentration for a rainfall event was 1.60 mgl^{-1} at the 10 m^2 plots and 0.75 mgl^{-1} at the 1 m^2 plots, the minimum phosphate concentration for a rainfall event was 0.26 mgl^{-1} at the 10 m^2 plots and 0.15 mgl^{-1} at the 1 m^2 plots. For the non-rainfall season of 2015 (eight months), the maximum phosphate concentration for a rainfall event was 1.14 mgl^{-1} at the 10 m^2 plots and 0.58 mgl^{-1} at the 1 m^2 plots, the minimum phosphate concentration for a rainfall event was 0.29 mgl^{-1} at the 10 m^2 plots and 0.08 mgl^{-1} at the 1 m^2 plots. For the rainfall season of 2015-2016 (four months), the maximum phosphate concentration for a rainfall event was 0.81 mgl^{-1} at the 10 m^2 plots and 0.10 mgl^{-1} at the 1 m^2 plots. The minimum phosphate concentration for a rainfall event was 0.07 mgl^{-1} at the 10 m^2 plots and 0.03 mgl^{-1} at the 1 m^2 plots. The maximum values recorded were during low rainfall events and the minimum values occurred during high rainfall events, these results would point to the impact of dilution. The 10 m^2 plots recorded significantly greater concentration of phosphate compared to the 1 m^2 plots

Events that had on average high phosphate concentration also had high variation in phosphate between the runoff plots for both the 10 m^2 and 1 m^2 plots. (Figures 4.11 and 4.12). Evidence of this is the first event measured in December 2014, which on average had a high phosphate but also high variation in phosphate values between the runoff plots, showing very little consistency in the P value measured at the runoff plots. This was common for all events that had on average a high phosphate concentration,

P values were recorded for the collected rainfall from the rain gauges adjacent to each runoff plot. Overall the concentration was low (<0.06 mgl^{-1}) and it is assumed it would not impact upon the reading for each plot. Weir samples had a phosphate concentration of less than 0.06 mgl^{-1} throughout the duration of the study. The phosphate concentration (mgl^{-1}) from the 10 m^2 plot and 1 m^2 plots were similar at the start of the study, they then decline for 1 m^2 plots from February 2015 until the end of the study (Figure 4.13). Interesting to note that the 18th of December which was the highest rainfall event recorded one of the lowest concentration of phosphate 10 m^2 = 0.14 mgl^{-1} and 1 m^2 = 0.08 mgl^{-1} for the different spatial scales, this may be due to dilution. Overall the 10 m^2 plots had a higher concentration of phosphate then the 1 m^2 plots, with a single exemption on the 27th May 2015. The full set of phosphate results can be found in Appendix E.

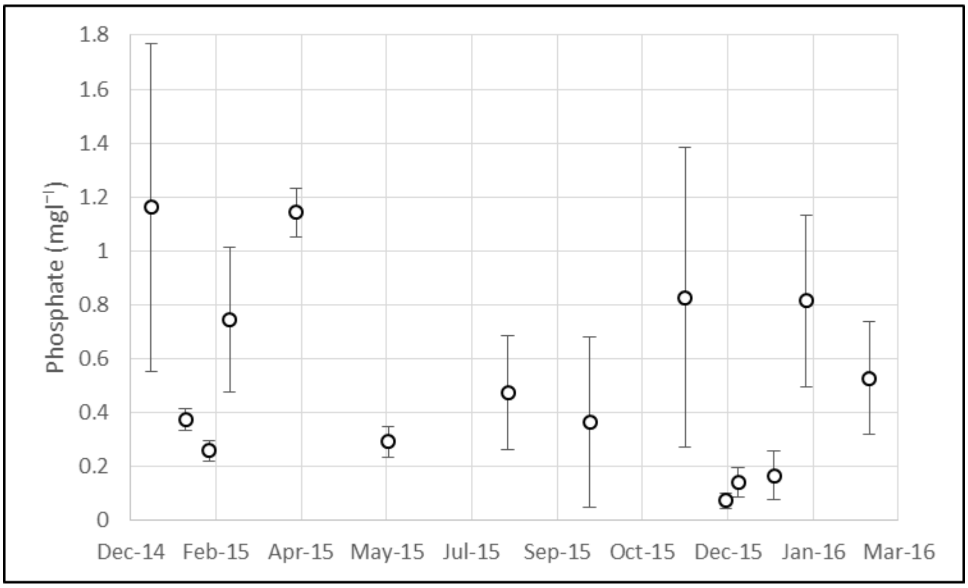

Figure 4.11: Average and standard error of phosphate for 10 m^2 plots

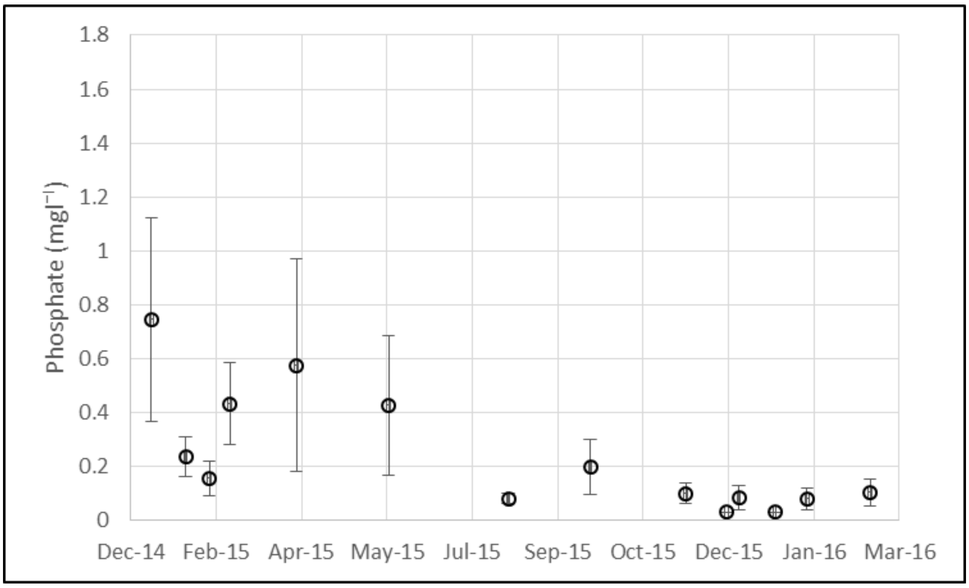

Figure 4.12: Average and standard error of phosphate for 1 m^2 plots

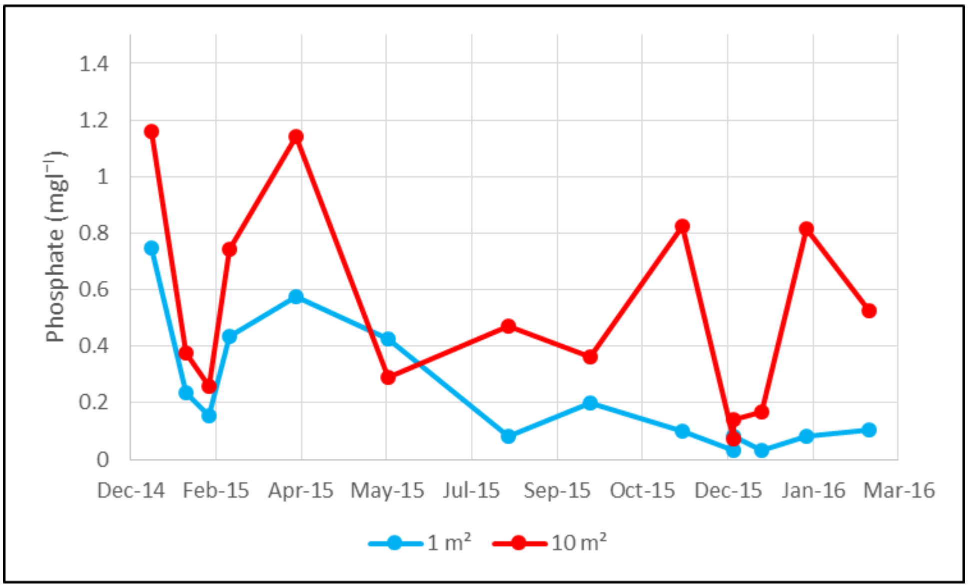

Figure 4.13: Relationship between 1 m^2 and 10 m^2 plots for phosphate

4.3.4 Nitrate

This section outlines the nitrate concentrations at the different slope positions and spatial scales, the impact that high rainfall and low rainfall has on the nitrate concentrations (Table 4.13). The average nitrate concentration is provided as it illustrates the differences between plot locations and sizes.

The highest average concentration of nitrate was measured at the mid slope position with the 10 m^2 plots recording on average 4.471 mgl^{-1} and the 1 m^2 plots 2.246 mgl^{-1} per event. There was a decline in nitrate concentration at the top slope with the 10 m^2 plots recording on average 4.872 mgl^{-1} and the 1 m^2 plots recording 2.246 mgl^{-1}. The lowest nitrate concentration was found at the bottom slope with the 10 m^2 plots recording on average 3.756 mgl^{-1} and the 1 m^2 plots recording 3.56 mgl^{-1}. Events that had high nitrate concentration also had high variation in nitrate between the runoff plots for both the 10 m^2 and 1 m^2 plots (Figure 4.14 and 4.15).

Table 4.13: Nitrate concentration (average volume in mgl^{-1}) at the different plot locations (top, middle, and bottom) for the different plot sizes.

	Top	Middle	Bottom
(n=15)			
Average 10 m^2	4.872 mgl^{-1}	4.471 mgl^{-1}	3.756 mgl^{-1}
Average 1 m^2	2.246 mgl^{-1}	4.155 mgl^{-1}	3.560 mgl^{-1}

For the rainfall season of 2014-2015 (four months), the maximum nitrate concentration for a rainfall event was 5.50 mgl^{-1} at the 10 m^2 plots and 3.52 mgl^{-1} at the 1 m^2 plots, the minimum nitrate concentration for a rainfall event was 1.77 mgl^{-1} at the 10 m^2 plots and 1.60 mgl^{-1} at the 1 m^2 plots. For the non-rainfall season of 2015 (eight months), the maximum nitrate concentration for a rainfall event was 6.27mgl^{-1} at the 10 m^2 plots and 6.19 mgl^{-1} at the 1 m^2 plots, the minimum nitrate concentration for a rainfall event was 2.40 mgl^{-1} at the 10 m^2 plots and 2.55 mgl^{-1} at the 1 m^2 plots. For the rainfall season of 2015-2016 (four months) the maximum nitrate concentration for a rainfall event was 5.33 mgl^{-1} at the 10 m^2 plots and 4.89 mgl^{-1} at the 1 m^2 plots. The minimum nitrate concentration for a rainfall event was 2.35 mgl^{-1} at the 10 m^2 plots and 2.01 mgl^{-1} at the 1 m^2 plots. The maximum values recorded were during low rainfall events and the minimum values occurred during high rainfall events, these results would point to the impact of dilution. The concentration of nitrate found at the different plot sizes were similar these were confirmed by running t tests.

Nitrate values were recorded for the collected rainfall from the rain gauges adjacent to each runoff plot. The concentration was consistently low (<1 mgl^{-1}) and it is assumed would not impact upon the reading for each plot. Weir samples had a nitrate concentration of less than 1 mgl^{-1} throughout the duration of the study. The nitrate concentration (mgl^{-1}) from the 10 m^2 plot and 1 m^2 plots were similar during the study with exceptions on 27th January 2016 and the 4th March 2016 (end of the study). The 18th of December which was the highest rainfall event recorded one of the lowest concentration of nitrate (10 m^2 = 2.35 mgl^{-1} and 1 m^2 = 2.61 mgl^{-1}) (Figure 4.16). During the summer period the 10 m^2 plots had a higher concentration of nitrate then the 1 m^2 plots and in the winter period the 1 m^2 plots had a higher concentration of nitrate then the 10 m^2 plots. The full set of nitrate results can be found in Appendix F.

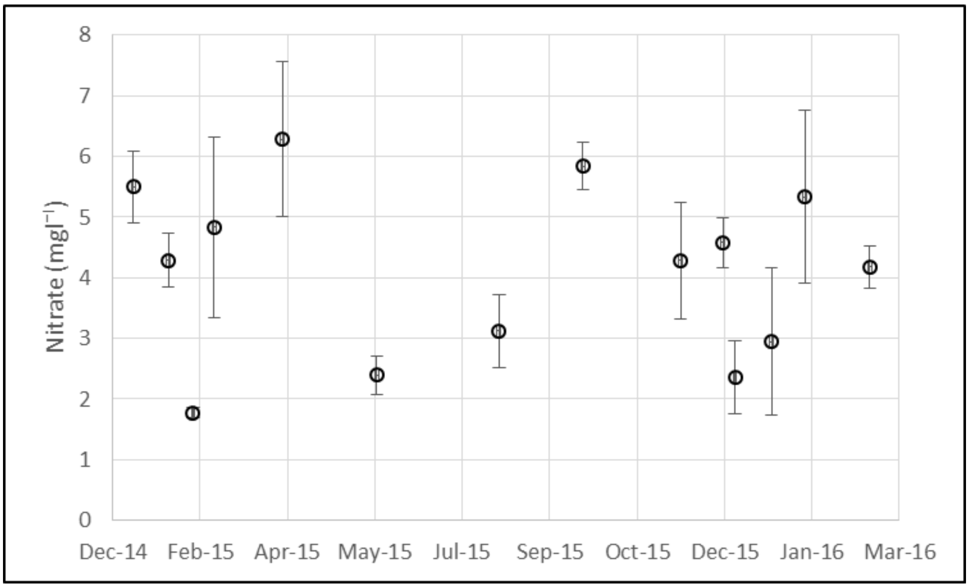

Figure 4.14: Average and standard error for nitrate for 10 m^2 plots

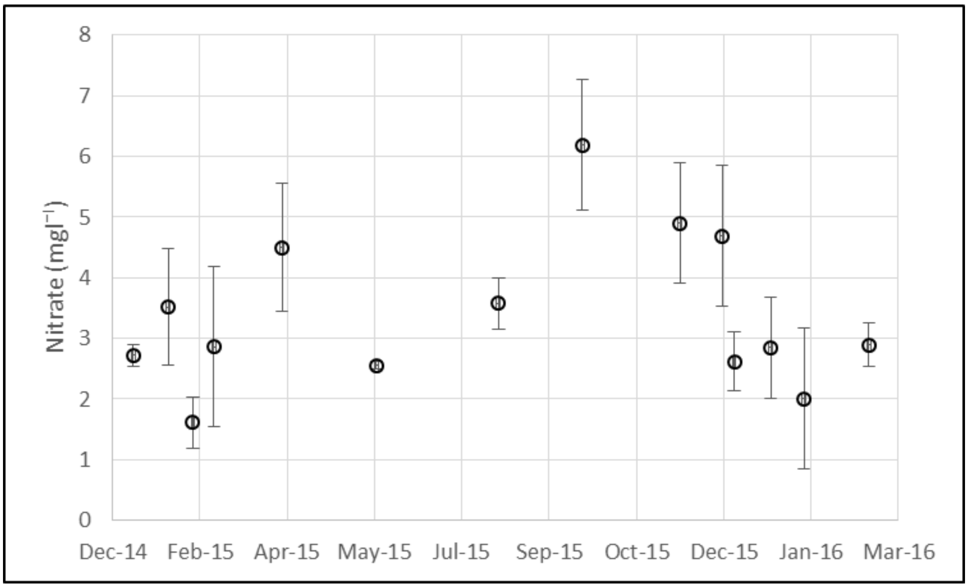

Figure 4.15: Average and standard error for nitrate for 1 m^2 plots

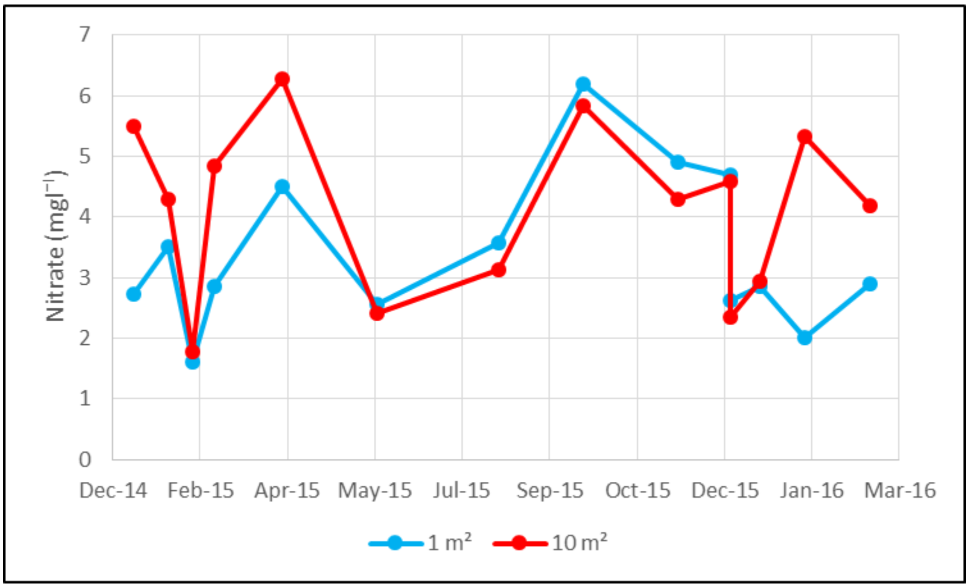

Figure 4.16: Relationship between 1 m^2 and 10 m^2 plots for nitrate

4.3.5 Dissolved Organic Carbon

Soil plays a key role in the carbon cycle with soil organic carbon being the basis of soil fertility. It releases nutrients for plant growth, promotes the structure, biological and physical health of soil, and is a buffer against harmful substances. Soil organic carbon is divided into dissolved organic carbon and particulate organic carbon. This section provides details on the dissolved organic carbon concentrations at the different slope positions and the spatial scales (Table 4.14). The average dissolved organic carbon concentration is presented as it illustrates the differences between plot locations and sizes.

The highest concentration of dissolved organic carbon was measured at the mid slope position with the 10 m^2 plots recording 23.39 mgl^{-1} and the 1 m^2 plots recording 22.12mgl^{-1} per event. There was a decline in dissolved organic carbon concentration at the top slope with the 10 m^2 plots recording on average 19.25 mgl^{-1} and the 1 m^2 plots recording 17.38 mgl^{-1} and the lowest dissolved organic carbon concentration was found at the bottom slope with the 10 m^2 plots recording on average 16.92 mgl^{-1} and the 1 m^2 plots recording 18.05 mgl^{-1}. Events that had high dissolved organic carbon concentration had greater variation in dissolved organic carbon between the runoff plots for both the 10 m^2 and 1 m^2 plots (Figures 4.17 and 4.18).

Table 4.14: Dissolved organic carbon concentration (average volume in mgl^{-1}) at the different plot locations (top, middle, and bottom) for the different plot sizes.

	Top	Middle	Bottom
(n=15)			
Average 10 m^2	19.25 mgl^{-1}	23.39 mgl^{-1}	16.92 mgl^{-1}
Average 1 m^2	17.38 mgl^{-1}	22.12 mgl^{-1}	18.05 mgl^{-1}

For the rainfall season of 2014-2015 (four months), the maximum DOC concentration for a rainfall event was 14.68 mgl^{-1} at the 10 m^2 plots and 11.79 mgl^{-1} at the 1 m^2 plots, the minimum DOC concentration for a rainfall event was 8.28 mgl^{-1} at the 10 m^2 plots and 8.86 mgl^{-1} at the 1 m^2 plots. For the non-rainfall season of 2015 (eight months), the maximum DOC concentration for a rainfall event was 24.37 mgl^{-1} at the 10 m^2 plots and 24.13 mgl^{-1} at the 1 m^2 plots, the minimum DOC concentration for a rainfall event was 9.18 mgl^{-1} at the 10 m^2 plots and 11.52 mgl^{-1} at the 1 m^2 plots. For the rainfall season of 2015-2016 (four months) the maximum DOC concentration for a rainfall event was 40.93 mgl^{-1} at the 10 m^2 plots and 38.68 mgl^{-1} at the 1 m^2 plots. The minimum DOC concentration was for a rainfall event 12.75 mgl^{-1} at the 10 m^2 plots and 14.51 mgl^{-1} at the 1 m^2 plots. The maximum values recorded were during low rainfall events and the minimum values occurred during high rainfall events, these results coukd point to the role of dilution.

DOC values were recorded for the collected rainfall from the rain gauges adjacent to each runoff plot. Overall the concentration was low ($<2mgl^{-1}$) and it is assumed it would not impact upon the reading for each plot. Weir samples had a low DOC concentration with a maximum concentration of 2.56 mgl^{-1} throughout the duration of the study

The DOC concentration (mgl^{-1}) from the 10 m^2 plot and 1 m^2 plots were similar throughout the study (Figure 4.19). During the summer months the 10 m^2 plots had a higher DOC concentration than the 1 m^2 plots and in the winter months the 1 m^2 plots had a higher DOC concentration than the 10 m^2 plots. However the 18th of December which was the highest rainfall event recorded had a relatively low concentration of DOC (10 m^2 = 12.75 mgl^{-1} and 1 m^2 = 14.15 mgl^{-1}). The full set of DOC results can be found in Appendix G.

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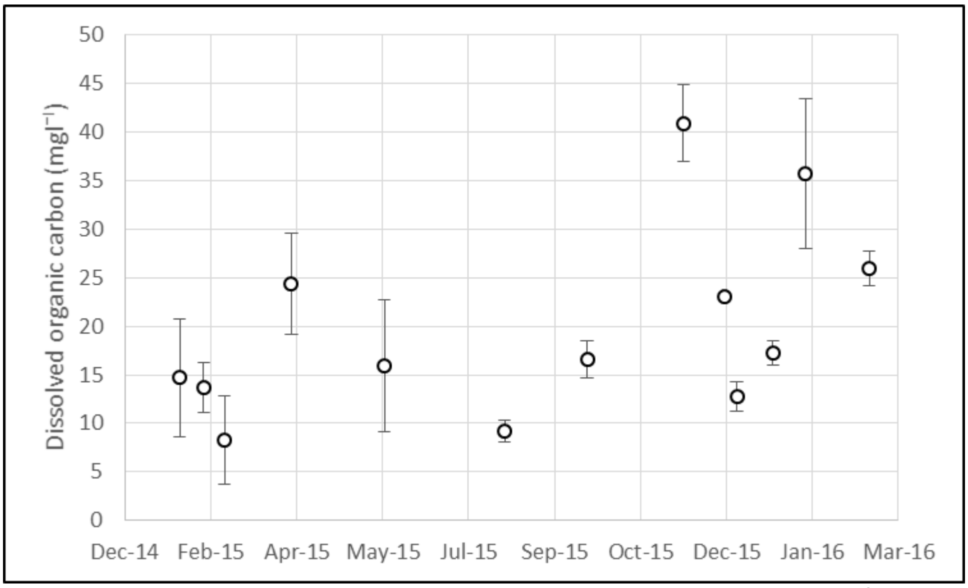

Figure 4.17: Average and standard error of dissolved organic for 10 m² plots

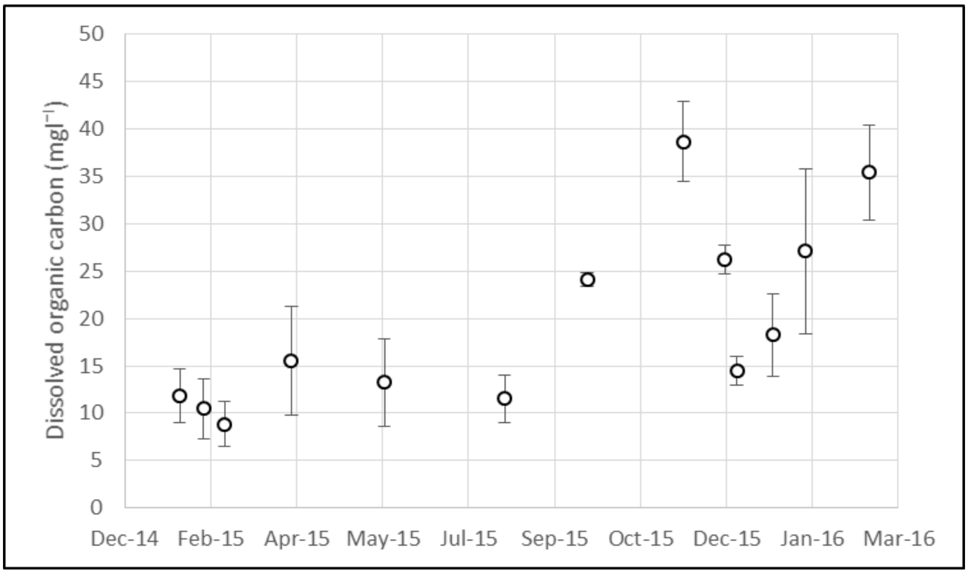

Figure 4.18: Average and standard error of dissolved organic carbon between plots for 1 m² plots

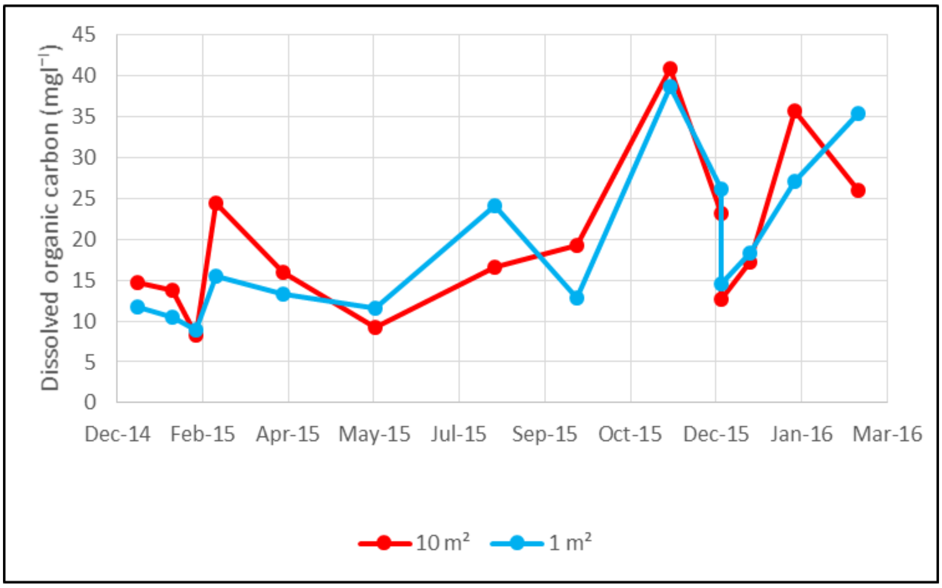

Figure 4.19: Relationship between 1 m^2 and 10 m^2 plots for dissolved organic carbon

4.3.6 Particulate Organic Carbon

This section provides details on the particulate organic carbon concentrations at the different slope positions and spatial scales (Table 4.15). The average particulate organic carbon is presented as it illustrates the differences between plot locations and sizes.

The highest concentration of particulate organic carbon yield from the plots came from the mid slope position with the 10 m^2 plots on average having a 0.116 gl^{-1} yield and the 1 m^2 plots having a 0.92 gl^{-1} yield. There was a decrease in particulate organic carbon removed at the top slope with the 10 m^2 plots having a 0.104 gl^{-1} yield and the 1 m^2 having a 0.094 gl^{-1} yield. The slope position with the lowest particulate organic carbon yield was the bottom slope with the 10 m^2 plots having a 0.076 gl^{-1} yield and the 1 m^2 plots having a 0.074 gl^{-1} yield. Events that had high particulate organic carbon concentration also had high variation in particulate organic carbon between the runoff plots for both the 10 m^2 and 1 m^2 plots (Figures 4.20 and 4.21).

Table 4.15: Particulate organic carbon concentration comparison (average volume in gl^{-1}) at the different plot locations (top, middle, and bottom) for the different plot sizes.

	Top	Middle	Bottom
(n=15)			
Average 10 m^2	0.104gl^{-1}	0.116gl^{-1}	0.076gl^{-1}
Average 1 m^2	0.094 gl^{-1}	0.092gl^{-1}	0.074 gl^{-1}

For the rainfall season of 2014-2015 (four months), the maximum POC yield for a rainfall event was 0.097 gl^{-1} at the 10 m^2 plots and 0.124 gl^{-1} at the 1 m^2 plots, the minimum POC yield for a rainfall event was 0.045 gl^{-1} at the 10 m^2 plots and 0.039 gl^{-1} at the 1 m^2 plots. For the non-rainfall season of

2015 (eight months), the maximum POC yield for a rainfall event was 0.054gl^{-1} at the 10 m^2 plots and 0.069 mgl^{-1} at the 1 m^2 plots, the minimum POC yield for a rainfall event was 0.034 gl^{-1} at the 10 m^2 plots and 0.037 gl^{-1} at the1 m^2 plots. For the rainfall season of 2015-2016 (four months) the maximum POC yield for a rainfall event was 0.500 gl^{-1} at the 10 m^2 plots and 0.173 gl^{-1} at the 1 m^2 plots. The minimum POC yield for a rainfall event was 0.060 gl^{-1} at the 10 m^2 plots and 0.046 gl^{-1} at the 1 m^2 plots. The maximum yields were recorded during high rainfall events and the minimum yields during low rainfall events, these results point to rainfall as a key driver of Particulate Organic Carbon loss.

POC values were recorded for the collected rainfall from the rain gauges adjacent to each runoff plot. Overall the concentration was low (<0.07 gl^{-1}) and it is assumed it would not impact upon the reading for each plot. Weir samples had a low POC volume with a maximum amount of 0.48 gl^{-1} recorded throughout the duration of the study

The particulate organic carbon volume (gl^{-1}) from the 10 m^2 plot and 1 m^2 plots was similar throughout the study (Figure 4.22), only on 18th December 2015 (due to the intense rainfall) was there a significant increase in particulate organic carbon, with the 10 m^2 plots producing significantly more particulate organic carbon then the 1 m^2 plots. The last trip on the 3rd of March had a significant difference in particulate organic carbon, in this instance the 1 m^2 plots produced a significantly higher volume (10 m^2 = 0.499 gl^{-1} and 1 m^2 = 0.173 gl^{-1}). The full set of POC results can be found in Appendix H.

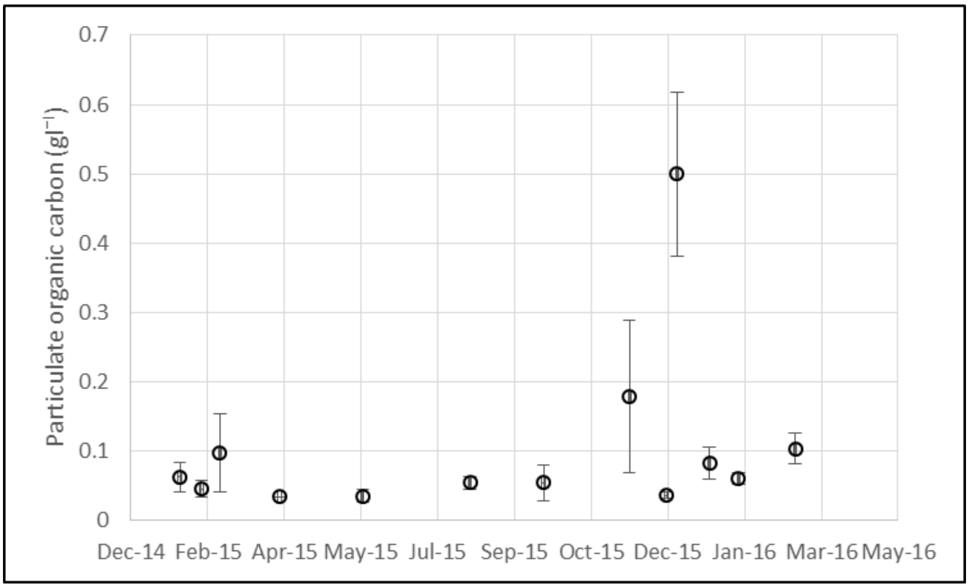

Figure 4.20: Average and standard error of particulate organic carbon for plots (10 m^2 plots)

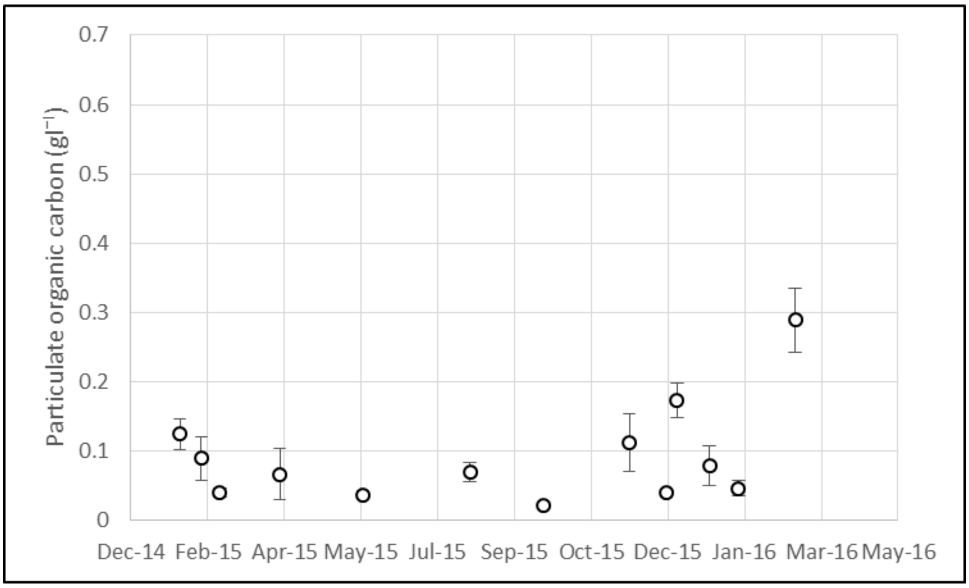

Figure 4.21: Average and standard error of particulate organic carbon for plots (1 m^2 plots)

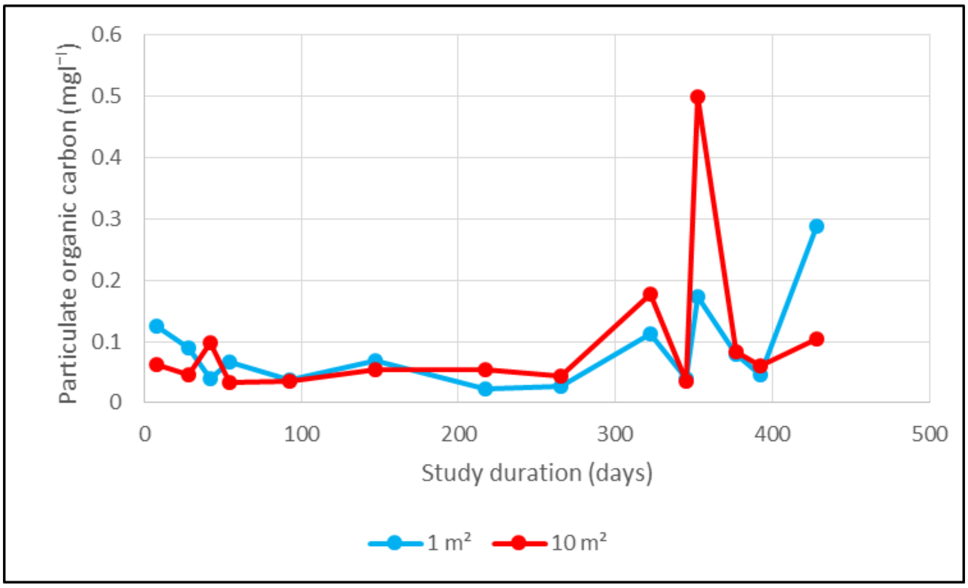

Figure 4.22: Relationship between 1 m^2 and 10 m^2 plots for particulate organic carbon

The cumulative particulate organic carbon yields at the different spatial scales was highest at the 34 ha catchment, followed by the 10 m^2 plots and then the 1 m^2 plots (Figure 4.23). The sampler at the weir was damaged due to an intense rainfall and was not fixed before the study ended, thus the 34 ha section of the graph has not been completed. It can be assumed though that it would continue to increase in the summer months. To obtain an accurate measurement it may be of use in the future to model the POC lost. The summer months experienced the most visible increase in POC due to the increased rain and runoff.

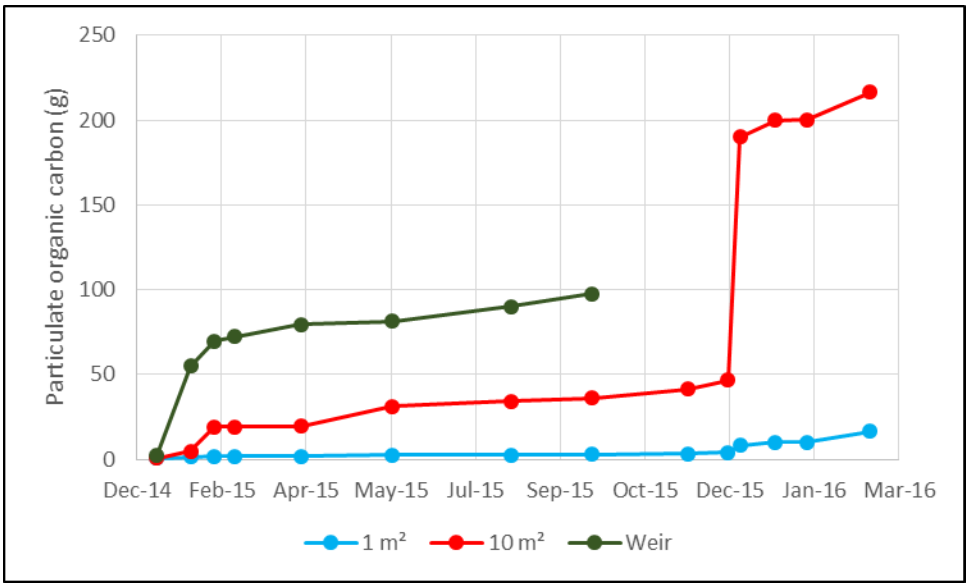

Figure 4.23: Cumulative particulate organic carbon yield at the various scales (1 m^2, 10 m^2 and 34 ha)

4.3.7 Summary of Experimental Results

On average, the highest runoff and P were recorded from the bottom slope position and the highest average sediment, NO_3^-, DOC and POC measurements from middle slope position. The 10 m^2 plots on average recorded higher values in the measurements besides runoff (sediment, P, NO_3^-, DOC and POC) compared to the 1 m^2 plots (Table 4.16). Rain-splash was the main contributor to sediment loss, with the 10 m^2 and 1 m^2 plots recording similar amounts. Dilution was present, with high rainfall events leading to low nutrient concentrations and low rainfall events leading to high nutrient concentrations. Besides phosphate, the nutrient and sediments concentrations recorded at the different plot sizes were similar.

Table 4.16: Summary table of the average measurements taken at the different spatial scales for the study duration

	Top		Mid		Bot	
	10 m^2	1 m^2	10 m^2	1 m ^{2}	10 m^2	1 m^2
Runoff	9.780 l/m^2	8.870 l/m^2	6.480 l/m^2	7.490 l/m^2	10.460 l/m^2	13.490 l/m^2
Sediment	0.837 gl^{-1}	0.834 gl^{-1}	0.909 gl^{-1}	0.804 gl^{-1}	0.897 gl^{-1}	0.790 gl^{-1}
P	0.510 mgl^{-1}	0.250 mgl^{-1}	0.390 mgl^{-1}	0.140 mgl^{-1}	0.720 mgl^{-1}	0.330 mgl^{-1}
NO_3^-	4.870 mgl^{-1}	2.246 mgl^{-1}	4.470 mgl^{-1}	4.155 mgl^{-1}	3.760 mgl^{-1}	3.560 mgl^{-1}
DOC	19.250 mgl^{-1}	17.380 mgl^{-1}	23.390 mgl^{-1}	22.120 mgl^{-1}	16.920 mgl^{-1}	18.050 mgl^{-1}
POC	0.104 gl^{-1}	0.094 gl^{-1}	0.116 gl^{-1}	0.092 gl^{-1}	0.076 gl^{-1}	0.074 gl^{-1}

4.4 Stream Flow

To determine the stream flow per day a rating table was formulated by Gush *et al* (2011) for the catchment. This included using the stream height data measured at the v-notch weir and converting to stream flow. There is a possibility that for some of the data, when downloaded, the offset values were not entered into the logger correctly, as an irregularity in the graph occurred. The 34ha catchment showed a relatively constant stream flow over the study duration, during the period of December 2014 to March 2015 there was a major increase in stream flow, due to the increase in rainfall events. When that summer period ended the stream flow reduced (Figure 4.24). Samples from the ISCO sampler were taken from the sampler and analysed when there was a significant enough change in stream height (± 5cm). As mentioned previously, the concentrations of phosphate (<0.06 mgl^{-1}), nitrate (<1 mgl^{-1}) and dissolved organic carbon (<2 mgl^{-1}) were consistent throughout the study. An average of 0.79 gl^{-1} of sediment and 0.016gl^{-1} of particulate organic carbon per event were measured from the weir.

An unfortunate issue occurred with the logger in November 2015 and this took a few weeks to fix. When it was fixed a flood event occurred caused by the intense rainfall on the 18th December 2015 flooding all the equipment and extensively damaging it. Only in mid-2016 did the loggers begin logging stream height again. With the summer months bringing increased rainfall it can be assumed that there would be an increase in the stream flow similar to what was experienced during the December 2014 to March 2015 period.

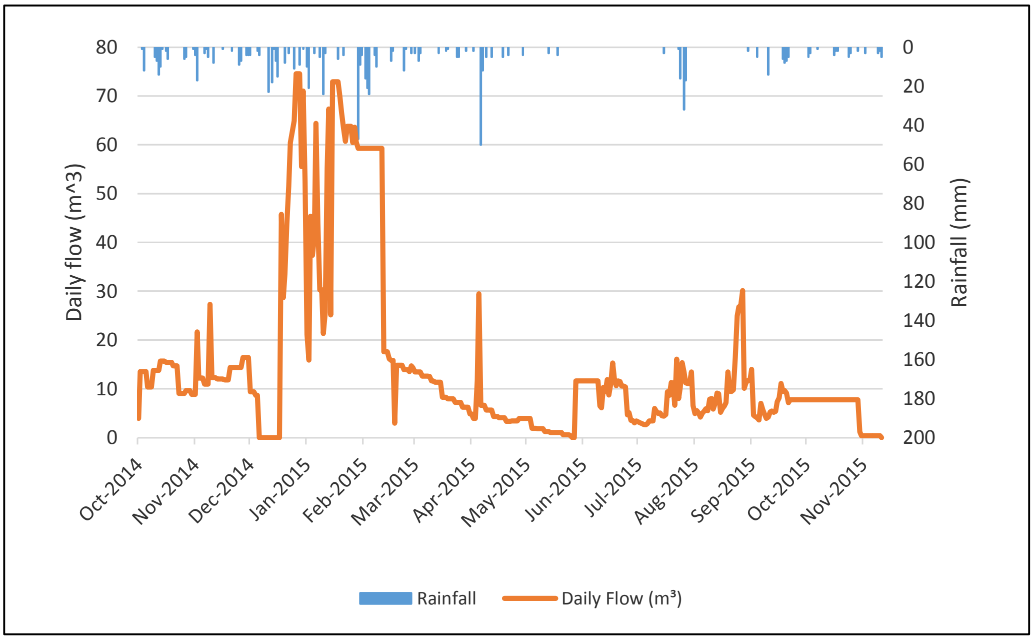

Figure 4.24: Stream flow record from October 2014 to October 2015

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4.5 Conclusion

Rainfall was, as expected, found to be seasonal, high during the summer months and low in the winter months, which impacted upon runoff, sediment and nutrient loss during the study. The canopy cover had an impact in intercepting the rainfall and reducing the amount of rainfall reaching the surface. The rain gauges set up in the catchment generally all had low variation between plots and had a constant interception offset against the AWS. The slope, soil characteristics and vegetation were measured and described in the catchment to aid in understanding soil erosion processes.

Rainfall is a key variable and high rainfall led to high runoff which led to high volumes of sediment and particulate organic carbon yield. There was minimal variation between the plots in terms of runoff, sediment yield and nutrient concentrations (Table 4.17). In terms of sediment loss and particulate organic carbon, there was similar amount of sediment and POC leaving the different plot sizes (gl^{-1}), but due to the 10 m^2 plots having greater cumulative runoff, there was a greater cumulative sediment yield/POC coming from the 34 ha catchment followed by the 10 m^2 plots and the 1 m^2 plots. The results highlighted the impact of an intense rainfall event such as the one experienced on the 18th of December 2015 can have in terms of significantly increasing the sediment and POC yield. The results from the 1 m^2 and 10 m^2 plots were similar, with only a small degree of variation in the measurements. Only the phosphate measurements had significant variation in plot sizes. This highlights that rain splash and runoff were similarly as effective in detaching and transporting sediment and nutrients. The results from this study shows that an increase in spatial scale does not have a significant impact on sediment yield (g/m^2) as the processes at the 1 m^2 plots (rain splash) are providing similar results as the 10 m^2 plot size (runoff). High rainfall led, in some instances, to low nutrient concentrations, with nutrients being diluted due to the increased runoff volume. Stream flow was measured during the study and was subject to rainfall, with intense rainfall leading to greater velocity stream flows.

Chapter Five

Discussion

5.1 Introduction

Commercial forests are a key land use and contribute approximately 4.4% to the regional GDP of KwaZulu-Natal and 1% to the national GDP (DAFF, 2014). Due to its importance to the economic sector, the environmental impacts need to be considered to ensure that the environmental thresholds for the impacted area are not exceeded and the area can be effectively rehabilitated. Therefore soil erosion and its associated impacts were selected for this study as they are known to have significant on- and off-site impacts, in particular during harvesting. Much research has been undertaken on the impact of soil erosion on agriculture but less on mature commercial forests in particular in South Africa. It is recognized that soil erosion rates are relatively low in stable forest ecosystems, where soil is protected by vegetation cover that intercept and diminish rain and wind energy (Pimentel and Kounang, 1998). At the study site, during the study, the ecosystem was stable, with no major impact on the ecosystem (for example no harvesting took place), with the only influence being the removal of alien invasive species from the catchment.

Field work enables the validation of models and allows for models to be created that can determine the rate of soil erosion and nutrient loss, this allows for mitigation measures to be created to combat the threat of climate change. The field work data from this study can be used to inform models such as The Soil and Water Assessment Tool (SWAT) and the Agricultural Catchments Research Unit (ACRU), which can be used to simulate water, sediment and chemical fluxes in the catchment under varying climatic conditions and management strategies, thus providing valuable information to inform decisions on the future management of the catchment.

5.2 Rainfall

This study took place during a below average rainfall period which affected most of South Africa, with KwaZulu-Natal being heavily affected. If this study had taken place in an average rainfall period, the results may have been different, with more intense rain events possibly leading to greater sediment loss. The rainfall was seasonal, with the summer months having more frequent rainfall events.

There was minimal spatial variation of rainfall throughout the catchment, with approximately 2 mm in rainfall variation between the rain gauges at the different slope positions, possibly due to some rain gauges being impacted more by evaporation or by tree canopy interception. There was a 34% interception rate determined by comparing the rain gauges at each plot to the Automatic Weather Station. The interception rate in this study was greater than what was measured by Bulcock and Jewitt (2012) who, in the same study site measured canopy interception for *Acacia mearnsii* at 27.7%. The interception rate has a significant impact with less rainfall making it to the ground to detach and flow down the slope. Bulcock and Jewitt (2012) observed greater canopy interception for *A. mearnsii* compared to *E. grandis* (14.9%) and *P. Patula* (21.4%).

When the *Acacia mearnsii* is harvested the canopy cover will be removed, increasing the volume of rainfall directly reaching the underlying soil, which will lead to an increase in runoff, which in turn will lead to an increase in sediment loss.

5.3 Runoff

Runoff varied with seasons and in response to rainfall characteristics (Bartley *et al*., 2006; Bautista *et al*., 2007). Runoff is primarily controlled by the response of the soil surface, any variations in rainfall resulted in variability in runoff from a rainfall event (Orchard *et al.*, 2012). Mohammed and Adam (2010) noted that there is a close relationship between rainfall events and the subsequent volume of runoff. In their study, high intensity rainfall events led to higher volumes of runoff, whilst low intensity rainfall events led to lower runoff. This study found similar results, as the more intense rainfall events resulted in less infiltration and a higher volume of runoff.

Sheet erosion and its efficiency to detach and transport soil material is spatially and temporally variable confirming previous investigations (Oakes *et al.*, 2012). The greatest runoff generated for the present study was observed for the micro-plots (1 m^2) with an average of 29.9 l/m^2 compared to 26.7 l/m^2 for the 10 m^2 plot for the study period, the greater average at the micro-plots may be as a result of the edge effect as the smaller plot means greater edge to soil surface ratio. The averages from the plot sizes demonstrate the similarity in runoff between the plot sizes. The total runoff received for the study duration from a 10 m^2 plot was 1327 l compared to 139 l from a 1 m^2 plot. This means that the 10 m^2 plots received an approximate total of 132.7 l/m^2 of runoff which is similar to the 1 m^2 plot which received an approximate total of 139 l/m^2, this demonstrates that an increase in spatial scale does not necessarily mean an increase in runoff. Joel *et al*. (2002) observed that the runoff coefficient decreased with increasing plot sizes and a decrease in the response with increasing area. Runoff will vary spatially and there is a need to improve the understanding of spatial and temporal variations of runoff (Sen *et al*., 2010; Van de Giesen *et al*., 2011). This similarity in runoff (l/m^2) is useful as it allows for a direct comparison between the different plot sizes, as it can be used to determine if similar runoff (l/m^2) meant similar sediment yield or similar nutrient concentrations. It is important to note that weather has temporal variability, in particular rainfall, and that soil erosion and nutrient totals can, in some circumstances, be dominated by a few extreme events (Renschler and Harbor, 2002).

Sanjari *et al.* (2009) stressed that the linkages between the soil surface characteristics and the generation of runoff is multi-factoral. Bergkamp (1998) states that effective infiltration rates on grassland hillslopes vary with rainfall intensity and flow depth, due to the interaction between rainfall, runoff, and vegetated micro-topography. Environmental factors vary temporally and spatially, as such, this affects runoff generation. Vegetation reduces the volume and intensity of runoff. In this study there was limited vegetation cover, the plots consisted predominately of litter and a short grass species *Eragostis tef*, thus there was minimal infiltration or a reduction in runoff.

5.4 Soil Erosion

Soil erosion is determined by rainfall intensity, runoff, erodibility, topography, vegetation and soil management (Morgan, 1988). Gabbard *et al*. (1998) and Truman *et al*. (2007) observe a linear relationship between runoff rate and sediment yield on arable soils and increases in sediment transport as a result of increased runoff rates. Sediment yield is the end production of soil loss after deposition. The present study observed a similar correlation between rainfall intensity/runoff and soil erosion. Throughout the study the sediment yield from the runoff plots was consistent, until the 18th of December site visit which had an intense rainfall event that led to a flash flood and the volume of sediment was significantly higher. The 18th of December site visit recorded a 114.6 mm rainfall event with 62 mm of rain falling in an hour. The tipping bucket recorded 143 tips in thirty minutes and 205 tips overall. The event resulted in a high volume of sediment with an average of 1.87 gl^{-1} coming off the 10 m^2 plots this is significant when compared to a smaller rainfall event such as the 23rd February 2015 (22 mm) which had a lower sediment yield of 0.798 gl^{-1} for the 10 m^2 plots. The yield from this high rainfall event was also significant in comparison to the rest of the study which averaged 0.881 gl^{-1} for the 10 m^2 plots. There was little variation in sediment yield from the different slope positions, with a difference of 8.2% sediment yield between the middle and top slope.

With the soil at Two Streams being highly repellent, rainfall ran rapidly off the soil and down slope, with low infiltration when soils were dry and higher when the soil was slightly saturated. One needs to take cognisance of the fact that the plots were well covered with litter and there was little bare soil, so the impact of runoff was minimal, decreasing erosivity energy. The amount of sediment in the samples taken from the automatic sampler were low, as the river is protected by dense vegetation on both sides and any runoff into the river is reduced. What is a concern is when the *Acacia mearnsii* is harvested there will be bare soil which will be exposed to rainfall and there is a risk of high runoff with associated high sediment yield, which will be carried into the river system. It is imperative that a future study measures the impact that tree harvesting has on the stream flow in terms of the change in sediment concentration. Silvicultural treatments should be applied within one or two years of harvesting, before regrowth makes sediment movement difficult. These silviculture treatments include forest cleaning of undergrowth to reduce competition to existing seedlings, or seedlings that might become established and liberation cutting of dense stands of trees and poles of commercial and non-commercial species.

Vegetation plays a key role in decreasing runoff and sediment yield (Mohammed and Adam, 2010; Chaplot and Poesen, 2012), by means of its canopy, roots and litter components. A study by Mohammed and Adam (2010) in the western slopes of the West Bank/ Palestine, Central Highland showed that natural forests had significantly lower rates of soil erosion and sediment then other land use types. In commercial forestry however there is a reputation for high soil loss as Grey and Jacobs (unpublished notes) measured increases in sediment production and gully erosion from headwall and channel bank retreat caused by forestry skid road construction and use in the southern Cape. In the KZN Midlands, site preparation by ripping, ploughing and harrowing have resulted in substantial increases in soil loss, and rates of up to 7.1 t/ha^{-yr} were recorded. (Moerdyk 1991). Whilst up to 200 t/ha^{-yr} have been recorded where wattle and brushwood have been burned after harvesting (Sherry 1959).

This study found that the vegetation played a significant role in reducing the impact of rainfall and runoff which resulted in low sediment yield. Hartanto *et al.* (2003) found that sapling density, canopy cover, litter layer, and woody debris are important ecological factors that reduce soil erosion. The study found that the presence of organic forest floor material, such as litter layer and woody debris, is important in preventing soil detachment and providing surface roughness, thus reducing runoff and downslope soil particle movement. The present study observed similar phenomenon with the study site having a high litter layer which reduced the volume of runoff and prevented sediment loss. Oliveira *et al*. (2013) found that soil loss decreased with increasing age of the trees. When this study begun the *Acacia mearnsii* were in the mature stage with the trees in the catchment greater than 8 years old. Although this is not a natural forest the trees in this catchment play a key role in reducing erosion. Bulcock and Jewitt (2012) who in the same site as the present study measured the canopy and litter interception, from April 2008 to March 2011 and in their results for *Acacia mearnsii* for the study site the litter cover had an interception rate of 6.6%, which would have a definite impact on reducing the impact of rain splash.

Chaplot and Le Bissonnais (2003) compared interrill erosion in tilled fields under different slope gradients (4 to 8%), plot sizes (1 m and 5 m long) and natural and simulated rainfalls with intensities ranging from 1.5 to 30 mm h^{-1}. Their study found that the rate of erosion increased with slope. Their results suggest that slope steepness had little impact on sediment concentration for the micro-plots (1 m^2). This was due to the reduced length of the micro-plot, as it did not allow for flow velocity to be significantly different between the various slopes. The results from the present study suggest similar findings with a 5.4% difference of sediment lost between the slopes. The sediment yield from Chaplot and Le Bissonais (2003) was significantly higher (2.9 to 49 gl^{-1}) then what was experienced at Two Streams which under natural rainfall events had a maximum sediment yield of 1.784 gl^{-1} at the 10 m^2 plots and 1.090 gl^{-1} at the 1 m^2 plots for a 62 mm h^{-1} rainfall event. These studies are comparable as they are undertaken using similar methods and the results show that a mature commercial forest has significantly lower sediment yield then tilled fields.

This study highlights that spatial scale did not have a significant impact on sediment yield, the 1 m^2 and 10 m^2 plots averaged similar sediment loads 0.809 gl^{-1} and 0.901 gl^{-1} respectively, this is greater than the average volume of 0.793 gl^{-1} taken from the weir. According to Oakes *et al*., (2012) soil erosion tends to increase spatially under the occurrence of runoff connectivity at the soil surface, which is most commonly induced under conditions of reduced rain water infiltration.

With the increase in spatial scale, there is an increase in runoff due to the increased area, this leads to more sediment being lost through water detaching the sediment from the runoff plots. The angle and increase in surface allows for flow to increase velocity and increases the ability to detach and transport sediment. A 10 m^2 plot received a total of 1327 l (132.7 l/m^2) of runoff for the study duration compared to 139 l (139 l/m^2) of runoff from the 1 m^2 plots. A 10 m^2 plots had a total sediment yield of 1377 g for the study duration compared to the yield of a 1 m^2 plot which was 118 g. This means that the 10 m^2 plots had an approximate total of 137.7 g/m^2 sediment yield which is similar to the 1 m^2 plot which received an approximate total of 118 g/m^2. This demonstrates that the increase in spatial scale did not have a significant impact on sediment yield. These values are also under the 1.1 kg/m^2 per annum which is the maximum permissible amount of soil loss stated by Morgan (1988). Even though the value from the 10 m^2 plot can still be considered high, it is important to note that 511 g (51.1 g/m^2) of sediment eroded off the 10 m^2 plots from the 18th of December rainfall event. This extreme rainfall event, led to a flash flood, which meant high intense runoff and substantial soil erosion. The average sediment yield (g/l) between the three spatial scales was similar. The total volume of amount of sediment coming off the different spatial scales was runoff dependent. The highest cumulative sediment yield came from the 34 ha catchment, followed by the 10 m^2 plots due to the increase in length compared to the 1 m^2 runoff plots.

Defersha and Melesse (2012) state that significant erosion and runoff varies with land use. Current studies by Birkett *et al.* (2016) at Okhombe valley observed runoff on degraded and non-degraded cattle access paths and rehabilitated cattle access paths using rainfall simulation. In their study, micro-plots were set up and a rainfall simulator was used to simulate a set amount of rain for a set amount of time. The results showed that the access cattle paths had a sediment yield of 3.26 gl^{-1} for a 27 mmh-^{1} rainfall event and 8.97 gl^{-1} for an 80 mmh^{-1} rainfall event. The rehabilitated cattle access paths had a sediment yield of 1.51 gl^{-1} for a 42 mmh^{-1} rainfall event and 1.21 gl^{-1} for a 56mmh^{-1}. By comparison at Two Streams, the 18th of December 2015 event (62 mmh^{-1} rainfall event) had 1.09 gl^{-1} recorded at the micro-plots the same size plot used in the Birkett *et al*. (2016) study. Although the conditions in their study was different to what was experienced at Two Streams, the results are still comparable illustrating the effectiveness of the commercial forest cover in reducing sediment yield. A similar study was undertaken at the Potshini catchment situated in the KwaZulu-Natal Drakensberg region, by Podwojewski *et al.* (2011) who applied a rainfall simulation to overgrazed pathways. Micro-plots were set up on the pathways with varying amounts of grass cover, from bare soil to grass covering the micro-plot. The results showed that soil loss increased rapidly with increase in bare soil and that crusting is a major process limiting water infiltration, including on plots with high vegetal cover.

Le Bissonnais and Singer, (1993); Wakindiki and Ben-Hur, (2002) and Podwojewski *et al.* (2011), all state the importance that sealing and soil crusting has on soil erosion by enhancing surface runoff and the detaching soil particles from the soil surface. Surface crusting, particularly on bare surfaces, is driven by raindrop impact however it is also impacted by compaction of the soil (Neave and Rayburg, 2007). Ben-Hur *et al*. (1985) similarly found that soil crusting has a large influence on the infiltration process and hence increases overland flow. At Two Streams there was little to no bare soil as litter and vegetation covered the majority of the catchment. No soil crusting was evident so there was no increase in surface runoff and soil detachment or signs of rill forming during the course of the study.

5.5 Rain Splash and Surface Runoff

Water erosion occurs predominantly from precipitation through rain-splash and un-concentrated flow as sheet erosion. Rain-splash is instrumental in detaching the top-soil and is only effective if the rain falls with sufficient intensity. If it does, then as the raindrops hit the bare soil, their kinetic energy is able to detach and move a considerable amount of soil particles a few centimetres, and its effects are solely on-site. When rain hits the surface most of it flows downhill as runoff, the flowing water from the runoff has the ability to detach and transport sediment great distances (Chaplot and Poesen, 2012). The comparison between the 10 m^2 and 1 m^2 plots allowed the study to investigate the main agent of water erosion. Micro-plots were used as they informed on the contribution of splash and rain-impacted flow on sediment mobilisation (Chaplot and Poesen, 2012), whilst the standard 10 m^2 plots were used as they had an increased spatial area and an increase in angle which allowed for flow to be generated. This study highlighted that rain splash (1 m^2 plots) and surface runoff (10 m^2 plots) provided similar results (l/m^2).

5.6 Phosphate

The ecosystems ability to retain nutrients relates to its maturity, with a more mature ecosystem being able to retain nutrients better than a younger ecosystem (Quinton *et al*., 2010). Clearing and disrupting vegetation cover leads to a loss of nitrogen and phosphorous, the harvesting stage of commercial forestry therefore will lead to significant losses in nitrogen and phosphorous.

P is detached primarily by kinetic energy through raindrop impact or from flowing water and the amount of P is not only transferred by the quantity of soil mobilised, but also by the concentration of P in the material transported (Dougherty *et al.*, 2004). The processes of P mobilisation can be classified into physical (detachment and entrainment of particles containing P, including colloids) and chemical (release of phosphate ions into solution) processes. Surface waters receive most P from surface flows rather than in groundwater, since phosphates bind to most soils and sediments. Phosphorus is delivered to aquatic systems as a mixture of dissolved and particulate inputs, each of which is a complex mixture of these different molecular forms of pentavalent P (Dougherty *et al.*, 2004).

The fate of P in runoff is strongly influenced by factors such as stream bank erosion, ground water inflows, and internal cycling within the stream by sediment–water interactions and biological action. The nature and rate of this transfer, and subsequent concentrations of P in runoff, are governed by chemical, biological, physical, and hydrological factors.

Quinton *et al. (*2001) observed a disproportionately high loss of P in sediment from small storm events. They attributed this to preferential erosion of fine clay particles of high P content during small runoff events. Whereas in larger storm events, the greater energy of overland flow transported coarser material containing relatively lower concentrations of P. Lewis (1986) noticed a relationship that showed a pattern close to being the inverse of runoff and phosphorous. The existence of a linear soil P–runoff P relationship has been observed under controlled rainfall simulation The present study had a similar result as Quinton *et al.*(2001) and Lewis (1986), with high rainfall events providing low concentrations of P and low rainfall events higher concentrations of P. This is demonstrated on the 18th of December which was the highest rainfall/runoff event and returned the lowest concentration of phosphate (10 m^2 = 0.14 mgl^{-1} and 1 m^2 = 0.08 mgl^{-1}), whilst a smaller rainfall event such as the 23rd February 2015 (22 mm) had a greater concentration of P (10 m^2 = 0.74 mgl^{-1} and 1 m^2 = 0.43 mgl^{-1}).

The 10 m^2 runoff plots recorded on average higher concentrations of phosphate (0.54 mgl^{-1}) than the 1 m^2 plots (0.24 mgl^{-1}). The phosphate samples from the weir all recorded <0.06mgl^{-1}, this may be due to the impact of dilution which is the loss/extraction of nutrients caused by the increase in water thus reducing the concentration of P. The results showed that P stayed in the system and was unable to reach the river. There was low variation in phosphate concentration between slope positions.

According to Correll (1999) the acceptable concentration of total P is not clear, with there being no widely accepted standard. For most lakes, streams, reservoirs, and estuaries concentrations of 0.1 mgl^{-1} total P/L are unacceptably high (Correll, 1999). The runoff plots recorded higher concentrations of P than what Correll (1999) deems acceptable but samples taken from the stream were all at an acceptable level. It is important to keep the concentration of phosphate in the river low, as high phosphate concentrations speed up the rate of eutrophication. It is recommended that the stream is continually monitored in terms of phosphate level, in particular when the *Acacia mearnsii* is harvested as one predicts an increase in the volumes of runoff which has the potential to transport higher volumes of P into the system.

5.7 Nitrate

Anh *et al.* (2014) showed that the amount of soil organic carbon and nitrogen are strongly related to the understory biomass with a strong correlation between the amount of litter and understory biomass. Their results suggest that understory biomass, surface cover, and bulk density are the most important characteristics influencing soil nutrient status and erosion rates, and these three controlling factors are governed by the specific characteristics of forests types. At Two Streams there is a high volume of litter and understory biomass so high levels of nitrogen and soil organic carbon were anticipated.

Lewis (1986) noted an inverse relationship between nitrate concentration and the time of highest discharge. This may help explain why in this study there was an inverse relationship between rainfall and nitrate concentration, for high rainfall events there were low nitrate, phosphate and dissolved organic carbon concentrations and for low rainfall events the nitrates, phosphate and dissolved organic carbon concentrations were higher. Tang *et al.* (2008) states that high rainfall may cause dilution leading to the loss of nutrients, this seems to have happened during this study, the 18th of December was the highest rainfall/runoff event and returned a low nitrate concentration (10 m^2 = 2.35 mgl^{-1} and 1 m^2 = 2.61 mgl^{-1}), whilst a smaller rainfall event such as the 23rd February 2015 (22 mm) had a higher concentration of nitrate (10 m^2 = 4.83 mgl^{-1} and 1 m^2 = 2.86 mgl^{-1}).

The plot sizes recorded similar concentrations of nitrate with the 10 m^2 runoff plots recording an average concentration of (4.37 mgl^{-1}) compared to the 1 m^2 plots (3.39 mgl^{-1}). The nitrate samples from the weir all recorded <1 mgl^{-1}, this may be due to the impact of dilution. The results indicate that nitrate stayed in the system and did not reach the river. There was low variation in nitrate concentration between slope positions.

According to Behar (1996) natural levels of nitrate are usually less than 1 mgl^{-1} in water. Concentrations over 10 mgl^{-1} will impact on the freshwater aquatic environment and is the maximum concentration allowed in human drinking water by the U.S. Public Health Service and in South Africa which has the same standard (Mamba *et al.*, 2008). For this study, the weir nitrate samples taken were all <1 mgl^{-1}. The nitrate values for the plots are all higher than that which occurs naturally (1 mgl^{-1}) but are not high enough to impact on the freshwater aquatic environment.

Vuorenmaa *et al.* (2002) compared the average specific losses of nitrogen and phosphorous from agricultural land (Cereal crop cultivation) to losses from forest land. The results showed an eight times higher total nitrogen loss and twelve times higher total phosphorus loss from agriculture. Vuorenmaa, *et al.* (2002) stated that in the forested catchments the impact of forestry operations, such as clear-cutting and fertilization, and the impact of atmospheric nitrogen deposition could be seen in nutrient losses. When the trees are harvested one expects the level of nitrate in the catchment to decline as there will be a major loss in understory biomass and as mentioned previously there is a strong correlation between understory biomass and nitrate. It could be important to record the level of nitrate going directly into the river after the trees are harvested. With results similar to Vuorenmaa, *et al.* (2002) expected, it is important to monitor the water quality to ensure the amount of nitrate going into the river is not unacceptably high.

5.8 Total Organic Carbon

Organic matter plays a major role in the aquatic system and is typically measured as Total Organic Carbon (TOC), which is divided into Dissolved Organic Carbon (DOC) and Particulate Organic Carbon (POC). DOC includes active chemical matter with dispersed molecules smaller than 0.45 µm. DOC content in surface waters varies from 0.5 to 0.7 mgl^{-1} and between 12-17 mgl^{-1} in wetland conditions (Thurman, 1985). POC is organic carbon content is frequently defined as organic matter larger than 0.7µm (Fielder *et al*., 2008)

Soil organic carbon exits soils in the form of greenhouse gases as a result of decomposition. It is exported in a particulate form through tillage, wind or water erosion or as Dissolved Organic Carbon by water movements. Soil litter, humus and microbial biomass constitute the most important sources of terrestrial dissolved organic matter (DOC), and recent studies on the dynamics of DOC, over the past 10 to 20 years, have shown that the fate of DOC in the soil (i.e.mineralisation or adsorption) is highly sensitive to key soil properties such as structure, mineralogy, pH, concentration in cations, ionic strength and phosphate concentration. The mobile DOC can move rapidly within hillslopes on the soil surface through runoff or more slowly in the subsurface within soils or within the fractured bedrock. Lal (2003) indicate that a significant portion of eroded TOC is exposed to greater oxidation rates during and after erosion. Soil crustability and erodibility generally increase as organic carbon content decreases (Le Bissonais and Arrouays, 1997). Soils are vital in carbon cycling but due to intensive cultivation or continuous cropping there is a decline in soil organic carbon (Lal and Kimble, 1997). Due to the importance of carbon in soils, the present study was interested in determining the amount of carbon being eroded by runoff at a commercial forest. The results from the present study can be used to model the amount of carbon lost. To determine the total amount of carbon being lost, dissolved and particulate organic carbon was measured.

5.8.1 Dissolved Organic Carbon

There was an inverse relationship between rainfall/runoff and DOC. For high rainfall events low concentrations of DOC were recorded from the runoff plots and for low runoff events higher concentrations of DOC were recorded. This is demonstrated on the 18th of December which was the highest rainfall/runoff event returned a low concentration of dissolved organic (10 m^2 = 12.75 mgl^{-1} and 1 m^2 = 14.15 mgl^{-1}) , whilst a smaller rainfall event such as the 17th November 2015 (79 mm) had a greater concentration of DOC (10 m^2 = 40.93 mgl^{-1} and 1 m^2 = 38.67 mgl^{-1}).

Both the 10 m^2 and 1 m^2 runoff plots had similar average concentration of DOC (10 m^2 = 19.85 mgl^{-1} and the 1 m^2 plots 19.18 mgl^{-1}). There was low variation in DOC concentration between slope positions. Low DOC concentrations were measured at the weir (1.87 mgl^{-1}). This may be explained in a study which was conducted by Chaplot and Ribolzi (2013), which revealed that while large amounts of organic carbon are solubilised during a storm event, only a tiny portion of this DOC reaches the river network. According to Kirchman *et al.* (1991) DOC in undisturbed watersheds generally range from approximately 1 to 20 mgl^{-1} carbon. The maximum concentration of DOC allowed in South African drinking water is 10 mgl^{-1} (Mamba *et al.*, 2008). The runoff plots recorded higher concentrations of DOC then what is acceptable in South African drinking water but samples taken from the stream were of an acceptable level (10 mgl^{-1}).

5.8.2 Particulate Organic Carbon

There was a correlation between rainfall and POC from the runoff plots. Initially the volume of POC was low as the rainfall intensity was low. Intense rainfall events yielded a higher POC count. For example as mentioned previously the 18th of December 2015 was an intense storm which had a high volume of rainfall (114.6 mm) which led to a high POC yield (10 m^2 = 0.50 gl^{-1} and 1 m^2 = 0.17 gl^{-1}), whilst a smaller rainfall event such as the 23rd February 2015 (22 mm) had a lower POC yield (10 m^2 = 0.10 gl^{-1} and 1 m^2 = 0.04 gl^{-1}).

This study highlights the importance of spatial scale, for POC the 1 m^2 and 10 m^2 plots averaged similar particulate organic carbon (0.099 gl^{-1} and 0.087 gl^{-1} respectively) which is greater than the average volume of 0.016 gl^{-1} taken from the weir suggesting that POC stayed in the system and did not reach the river. The POC volume was spatially variable, with different slope locations recording different POC values, although there was low variation between slope positions.

With the increase in spatial scale there is an increase in runoff, which led to more POC being lost through water detaching the carbon from runoff plots. Each 10 m^2 plot received, a total of 1327 l (132.7 l/m^2) of runoff compared to 139 l of runoff from the 1 m^2 plots. The 10 m^2 plots recorded a total POC yield of 216.5 g compared to 16.3g from a 1 m^2 plot. This means that the 10 m^2 plots had approximately 21.7 l/m^2 POC yield which is similar to the 1 m^2 plot (16.3 g/m^2). This demonstrates an increase in spatial scale does not significantly impact POC yield. Even though the average concentration of POC from the weir was low (gl^{-1}), the size of the stream meant more litres which means a higher POC yield from this spatial scale. This highlights the key impact that rain splash has on the removal of POC from the soil surface, as the concentration of POC is similar between the different plot sizes. The 1 m^2 plot informs on the impact of rain splash with little spatial area for flow, whilst the 10 m^2 plots has an increased spatial area for the process of runoff. With the values being similar it suggests that rain splash and runoff are similar agents of POC erosion. The question of whether or not these losses of POC are higher than the natural replenishment of the soil organic carbon needs to be addressed. There needs to be further research and investigation into the behaviour of nutrients, such as DOC, POC beyond the fate of the micro-plot, plot and catchment boundaries. Such an approach requires an understanding of the catchment's geochemistry and microbiology. More research and management strategies should consider DOC and POC loss at the microscale.

5.9 Nutrient Summary

The nutrients measured in the study all suggest similar findings, with low spatial variation. There was similar concentrations of nutrients measured at the different plot scales with the 10 m^2 plots on average, producing higher averages, but the differences were low, with only phosphate providing a discernable differences between plot sizes. Measurements taken from the weir were all low, and all under the standard concentration levels. The nutrient results illustrated the impact of dilution as high rainfall events led to low concentrations of nutrients. Overall, a larger rainfall event caused more nutrient loss as increased runoff caused higher volume of nutrients to be detached and mobilised. For both plot sizes, there was more runoff per higher rainfall event, which meant more nutrients being detached at both plot sizes. With the collection tanks having more runoff, the nutrients were then diluted by the larger quantity of runoff water. Only Particulate Organic Carbon increased due to an increase in rainfall as rain splash and runoff would remove the POC from the surface.

5.10 Stream flow

Stream data were available from October 2014 to 2015, providing a year of data which were consistent, in the summer period the increase in rainfall events led to greater volumes of stream water which in turn led to greater stream flow. ISCO samples for the summer months had higher sediment concentration then samples taken in the winter months, due to the higher velocity flow leading to sediment transport. It was unfortunate that an intense rainfall event led to the loss of stream flow data towards the end of the study. It is the author's expectation that the stream flow would have increased in the summer months due to increased rainfall, this would most likely lead to an increase in the soil erosion rate and when the *Acacia mearnsii* is harvested, the flow rate and resultant soil erosion will increase.

5.11 Conclusion

Climate change is likely to affect soil erosion through its effects on rainfall intensity, soil erodibility, vegetative cover and patterns of land use (Nearing *et al.*, 2005). With the possible threat of climate change leading to high intensity rainfall, leading to increased soil erosion it is important to be proactive and protect the soil. There are large areas of KwaZulu-Natal with high potential risk of future erosion especially areas with hill sand steep terrain (Le Roux *et al.*, 2008). "Approximately 50% (61 million ha) of South Africa has a moderate to severe erosion potential (>12 t/ha$^{\cdot yr}$), whereas approximately 20% (26 million ha) of land is classified as having a moderate to severe actual erosion risk" (Le Roux *et al*., 2008, pg 312). The results from this study show the value that trees have on reducing soil erosion through canopy cover intercepting the rainfall and the litter on the ground reducing the volume of runoff. Rain splash is a key driver of detaching soil and transporting it. It is therefore imperative to be proactive and to counter the impending threats of climate change. The author's recommendation is to target areas that are under threat of severe erosion and increase vegetation cover the understory cover by using non-invasive species that can reduce the impact of rain splash and runoff. According to Le Roux *et al.* (2008) the risk of erosion is great with over 26 million ha of South African land at risk of high erosion due to the lack of maintenance of the current vegetation cover. The main issue to note is the increasing intensity of events leading to increased soil erosion.

Findings from this study concur with previous studies that the key driver of soil erosion and sediment yield in this study is rainfall. With increased rainfall one experienced increased sediment yield. Rain was seasonal with the majority of the rain falling in the summer months. The results demonstrate the lack of significance of spatial scale. Spatially, the measurements from the variables at the different sized runoff plots were similar, demonstrating the similarity of impact that rain splash and runoff has in sediment and nutrient detachment and mobilisation. The plot sizes had similar; runoff (l/m^2), sediment, nutrient concentrations and particulate organic carbon yields (g/l^{-1}), which demonstrated that the different processes experienced at the different scale were providing similar results. This also depends on the difference in surface/catchment area. From small to large catchments there is a decline in sediment yield due to the increase in sediment yield due to the increase of barriers and places for deposition to occur (De Vente and Poesen, 2005). Temporally, high intensity rainfall events led to significant sediment and particulate organic carbon loss as illustrated on the 18th of December which recorded a 114.6 mm rainfall event, which led to an average of 1.87 gl^{-1} of sediment coming off the 10 m^2 plots which compared to the rest of the study averaged 0.881 gl^{-1} for the 10 m^2 plots. Phosphate, nitrate and dissolved organic carbon concentrations were low for high rainfall events and high for low rainfall event this seems to infer the process of dilution which is the loss/extraction of nutrients caused by the higher volumes of runoff but, a larger rainfall event caused more nutrient loss as increased runoff caused more nutrients were detached and mobilised but diluted by the increased volume of water in the collection tank.

Compared to other land uses, commercial forestry that is not influenced by human impact (harvesting) has a low rate of soil erosion, due to the aerial, litter and grass cover protecting the soil from the impacts of rain splash and runoff. Agriculture and land that is overgrazed has much higher rates of erosion than at Two Streams, emphasizing the importance that land use has on the rate of erosion and considerations must be given to this for future planners and managers. At Two Streams the nutrient measurements taken from the weir were all of an acceptable concentration, so the concentrations recorded at the plot scale were not influencing the concentrations at the weir, suggesting that the nutrients stayed in the system and did not reach the river. Although the rate of erosion and nutrient transportation are currently acceptable at Two Streams, it is predicted that the sediment yield and nutrient concentration will increase once the commercial forest is harvested.

The results of this study may be used to model the soil and nutrient loss for the catchment and help inform the impacts that climate change will have on the catchment. The current land use has been effective in reducing the impact of rain splash, due to the effect that the tree canopy and litter has on intercepting the rainfall and subsequent runoff. As commercial forestry is a major contributor to South Africa's GDP, the environmental impacts have to be accepted. This study found that a commercial forest pre-harvest has acceptable soil erosion rates however when the harvesting process occurs, not only will heavy machinery increase soil erosion, the loss of canopy cover and litter will increase bare soil and subsequently soil erosion in the catchment until the new crop has reached the same mature stage as the present study. In that time the amount of soil erosion that would have occurred will be significant and post-harvest measurements need to be done to quantify the effects of harvesting.

The key driver of soil erosion, is rainfall intensity as high rainfall intensity resulted in greater soil erosion, so mitigation measures need to be put in place that reduces the amount and intensity of rainfall on the surface, this study has shown that tree cover was effective in rainfall interception and the associated litter reduced the intensity of rainfall. The data set from the present study may be used in the future to populate soil erosion models, so that future climatic conditions can be modelled and mitigation measures can be put in place to reduce soil erosion and nutrient loss.

Chapter Six

Conclusion

Three objectives were formulated to meet the research aim of investigating the processes of erosion and sediment yield at different temporal and spatial scales in a commercial forestry land use. The objectives were:

i. Set-up appropriate experimental design that is able to measure the temporal and spatial variability of soil erosion
ii. Measure soil loss in an afforested catchment at different spatial and temporal scales and to determine the sediment yield at the end of the study.
iii. Determine the spatial and temporal variations in nutrient concentration.

With response to objective one, the context of the study is on commercial forestry with specific focus on soil loss through water erosion. It was important to set up a design that measured soil erosion processes that were highly variable in time and space. One m^2 micro-plots were selected to quantify the impact of local erosion processes associated with splash and rain-impacted flow (Kinnell, 2001; Chaplot and Poesen, 2012), whilst ten m^2 runoff plots, which conformed to the standard dimensions used in other studies in South Africa (Chaplot and Poesen, 2012), were selected to determine the impact of flow (Chaplot *et al*., 2011). Three replicate plots were set up at three different slope locations. At each of the three slope locations tipping bucket mechanism were connected to HOBO event-logger (pendant logger), to allow rainfall event intensity to be calculated. To measure the sediment yield at catchment scale an ISCO sampler was installed at a gauging weir outlet from the catchment. At the catchment outlet (34 ha) there is a V-notch weir. The existing logger was coupled to an ISCO 6712 and 3700 series, automatic sampler. The height of flow at the catchment outlet is logged by a data logger. Sediment and nutrient loads, during both base flow and stormflow events were measured by collecting samples at the appropriate locations on the hydrograph curve. Site visits were conducted after rainfall events, with the runoff at each of the plots being measured and samples being taken for laboratory analysis. To determine the extent of spatial variation in rainfall, rain gauges were set up adjacent to each runoff plot. The rain gauges allowed the determination of canopy interception by comparing rainfall values from an Automatic Weather Station to the rain gauge values.

With an appropriate experimental design in place for the remaining two objectives of the study could be met. The second objective was to measure sediment yield in an afforested catchment at different spatial and temporal scales. The amount of sediment at each plot replicate and spatial scale was measured by filtering the runoff samples and burning off the particulate nutrients until the sediment remained. The soil loss was determined and compared at the different scales. The results showed that an increase in spatial scale did not have a significant impact on sediment yield, as both the 10 m^2 and 1 m^2 plots averaged similar amounts (0.901 gl^{-1} and 0.809 gl^{-1} respectively), samples from the weir averaged 0.793 gl^{-1}. A total of 1377 g (137.7 g/m^2) were measured from a 10 m^2 plot compared to the 118 g (118 g/m^2) from a 1 m^2 plot, these results highlighted that rain splash and runoff were similarly effective in sediment detachment and mobilisation. There was low spatial variation between the different slope positions as the top, middle and bottom slope positions all recorded similar sediment volumes. High intensity rainfall events led to higher sediment volume, with the afforested catchment of mature trees experiencing low sediment loss as the vegetation and canopy cover reduce the impact of rainfall, thus reducing rain splash impact. The soil was protected by leaves and vegetation so, as expected, sediment loss from the catchment was low.

The third objective of this study was to determine the spatial and temporal variations in nutrient concentration. Phosphate and nitrate were selected for this study as they are linked to eutrophication in river systems (Ekholm and Lehtoranta, 2012). Soil organic carbon was selected as previous studies have emphasized its importance and the threat that soil erosion has to the soil carbon stocks. (Chaplot and Poesen, 2012). This study found that there was an inverse relationship between rainfall/runoff and phosphate, nitrate and dissolved organic carbon concentrations, as higher rainfall/runoff events resulted in lower nutrient volumes compared to small f events, possibly due to dilution. Larger rainfall events caused more nutrient loss as increased runoff caused higher volume of nutrients to be detached and mobilised. There was a small degree of spatial variation with small differences in nutrient values with different slope positions receiving similar concentrations of nutrients. The results for particulate organic carbon were similar to that of the sediment yield, as increased rainfall led to greater POC yield. The different plot size had similar concentrations of nutrients thus demonstrating that the increase in spatial scale had a limited impact on nutrient availability.

It can be concluded that the afforested catchment in Two Streams has low rates of sediment loss and the temporal variations perform a key role in soil loss. This was evident through the impact that rainfall intensity has had on increasing runoff and subsequently sediment loss. Furthermore, the findings demonstrate the impact of vegetation has on limiting the impacts of water erosion as it is vital in reducing the impact of rain splash, this is done by the tree cover intercepting rainfall and the ground cover protecting the bare soil from the impact of rain splash. The increase in spatial scale did not influence sediment yield and nutrient loss. Rain splash through high intensity rainfall was the key driver of sediment detachment and mobilisation.

The conditions of the catchment will change significantly once the catchment is harvested, with an expected increase in sediment and nutrient loss. It is important that the changes are recorded and compared to this study to provide key insight on the effects of the forestry land use through a cycle of growth, harvesting and replanting. Further understanding of the processes leading to changes of nutrient and carbon fluxes need to be performed to integrate this study with the ecosystem functioning of a landscape. Such a study requires more field observations and ultimately more data. The field data collected from this study can be used in modelling to determine the rate of soil erosion and nutrient loss, by using models mitigation measures can be put in place. Such an approach can be used to extrapolate the results from this study to larger areas A focus on the fate of the nutrients beyond the observed scales are required as are larger scale observations.

Soil erosion is a significant problem in South Africa with an annual soil loss of 400 million tonnes (Meadows and Hoffmann, 2003). This threat has both on- and off-site impacts that negatively influence the natural ecosystems. This study provided an insight to soil erosion and nutrient loss and transportation within a mature afforested catchment, which had low sediment and acceptable nutrient loss compared to agricultural land use. Measurements need to be done post-harvest and compared to this study as one expects the rate of soil erosion to increase. It is paradoxical as we require the forestry sector as an important contributor to the economy of the region, but we have to accept the consequential environmental harm. We need to manage this by understanding and hence modelling predicted outcomes. Our models are only as good as the data we receive and thus the need for such a study as this provides high spatial and temporal data to populate these models.

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Appendices

Appendix A: Site Photos

Site 1 Runoff plot

Site 1 Micro-plot

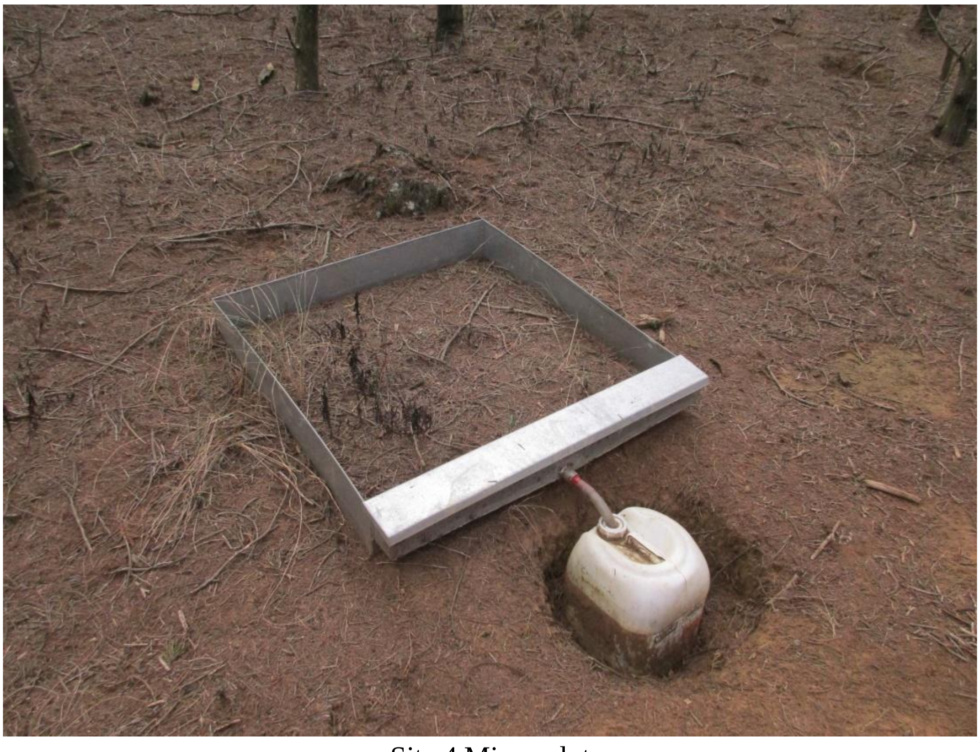
Site 2 Runoff plot

Site 2 Micro-plot


Site 3 Runoff plot


Site 3 Micro-plot

Site 4 Runoff plot



Site 4 Micro-plot

Site 5 Runoff plot

Site 5 Micro-plot

Site 6 Runoff plot

Site 6 Micro-plot

Site 7 Runoff plot

Site 7 Micro-plot

Site 8 Runoff plot

Site 8 Micro-plot


Site 9 Runoff plot


Site 9 Micro-plot

Appendix B: Rainfall

Rain gauge readings at Two Streams (mm)

Date	Top	Middle	Bottom
8-Jan-15	33.0	35.7	33.7
28-Jan-15	44.0	44.3	46.0
11-Feb-15	78.0	70.5	71.5
23-Feb-15	18. 7	17.3	16.7
3-Apr-15	36.3	33.3	31.7
27-May-15	45.0	45. 7	46.7
5-Aug-15	46. 7	47.3	49.3
22-Sep-15	31.3	35. 7	35.3
16-Oct-15	11.3	12.0	12.7
17-Nov-15	24.7	21.3	22.0
11-Dec-15	50.3	48.0	47
18-Dec-15	81. 7	85. 7	83.3
12-Jan-16	59.7	62.0	61.7
27-Jan-16	22.7	24.0	24.0
4-Mar-16	55.3	59.3	59.3

Automatic Weather Station rainfall data for Two Streams (mm)

Date	AWS (mm)
8-Jan-15	60.1
28-Jan-15	84.1
11-Feb-15	88.0
23-Feb-15	22.0
3-Apr-15	36.2
27-May-15	85.0
5-Aug-15	71.9
22-Sep-15	58.9
16-Oct-15	18.9
17-Nov-15	54.2
11-Dec-15	79.1
18-Dec-15	114.6
12-Jan-16	67.6
27-Jan-16	48.3
4-Mar-16	89.9

Appendix C: Runoff

Runoff volume at 10 m^2 plots (l/m^2)

Date	Top	Middle	Bottom
8-Jan-15	0.79	0.80	1.31
28-Jan-15	11.93	4.15	12.27
11-Feb-15	13.34	8.84	21.59
23-Feb-15	0.23	0.31	0.67
3-Apr-15	0.38	0.62	0.55
27-May-15	25.37	15.89	24.12
5-Aug-15	6.39	4.18	6.09
22-Sep-15	4.96	2.42	6.60
16-Oct-15	0.88	0.52	1.19
17-Nov-15	2.13	4.25	2.17
11-Dec-15	19.69	8.25	17.46
18-Dec-15	29.40	27.17	29.40
12-Jan-16	15.21	5.42	15.27
27-Jan-16	0.03	0.47	0.77
4-Mar-16	16.08	13.96	17.38

Runoff volume at 1 m^2 (l/m^2)

Date	Top	Mid	Bot
8-Jan-15	3.48	1.27	4.79
28-Jan-15	5.75	6.79	12.61
11-Feb-15	12.00	2.40	22.69
23-Feb-15	0.95	1.10	0.63
3-Apr-15	2.23	3.00	1.35
27-May-15	9.77	5.72	12.85
5-Aug-15	3.32	5.7	4.17
22-Sep-15	2.72	5.43	10.50
17-Nov-15	3.72	2.90	10.75
11-Dec-15	9.28	15.92	27.48
18-Dec-15	26.67	20.00	27.47
12-Jan-16	21.86	15.05	24.02
27-Jan-16	2.44	1.28	2.49
4-Mar-16	20.02	18.37	27.05

Appendix D: Sediment Yield

Sediment loss at 10 m^2 plot (gl^{-1})

Date	Top	Middle	Bottom
08-Jan-15	0.089	0.997	0.783
28-Jan-15	0.801	0.804	0.796
11-Feb-15	0.870	0.918	0.645
23-Feb-15	0.778	0.735	0.882
03-Apr-15	0.831	0.756	0.818
27-May-15	0.809	0.817	0.775
05-Aug-15	0.797	0.659	0.764
22-Sep-15	0.841	0.755	0.864
17-Nov-15	1.148	0.867	0.771
11-Dec-15	0.756	0.795	0.823
18-Dec-15	1.688	1.876	1.789
12-Jan-16	0.755	0.906	0.976
27-Jan-16	0.831	0.833	0.780
04-Mar-16	0.733	1.013	1.093

Sediment loss at 1 m^2 plot (gl^{-1})

Date	Top	Middle	Bottom
08-Jan-15	0.842	0.809	0.757
28-Jan-15	0.857	0.703	0.810
11-Feb-15	0.726	0.757	0.882
23-Feb-15	0.830	0.762	0.718
03-Apr-15	0.746	0.742	0.756
27-May-15	0.870	0.786	0.746
05-Aug-15	0.765	0.730	0.753
22-Sep-15	0.803	0.749	0.713
17-Nov-15	0.957	0.803	0.796
11-Dec-15	0.775	0.763	0.797
18-Dec-15	1.162	1.080	1.029
12-Jan-16	0.865	0.975	0.912
27-Jan-16	0.741	0.825	0.657
04-Mar-16	0.731	0.778	0.731

Appendix E: Phosphate

Phosphate measurements at 10 m^2 plots (mgl^{-1})

Date	Top	Middle	Bottom
8-Jan-15	0.465	0.640	2.377
28-Jan-15	0.300	0.387	0.437
11-Feb-15	0.180	0.303	0.290
23-Feb-15	1.210	0.737	0.287
3-Apr-15	1.290	0.980	1.157
27-May-15	0.190	0.387	0.297
5-Aug-15	0.390	0.153	0.877
22-Sep-15	0.995	0.057	0.040
16-Oct-15	1.605	0.583	0.030
17-Nov-15	0.030	0.553	1.897
11-Dec-15	0.060	0.030	0.130
18-Dec-15	0.077	0.093	0.253
12-Jan-16	0.077	0.073	0.347
27-Jan-16	0.57	0.427	1.445
4-Mar-16	0.177	0.503	0.903

Phosphate measurements at 1 m^2 plots (mgl^{-1})

Date	Top	Middle	Bottom
8-Jan-15	0.377	0.360	1.500
28-Jan-15	0.170	0.157	0.383
11-Feb-15	0.217	0.030	0.220
23-Feb-15	0.443	0.690	0.165
3-Apr-15	1.355	0.090	0.280
27-May-15	0.273	0.077	0.933
5-Aug-15	0.097	0.043	0.103
22-Sep-15	0.073	0.120	0.400
17-Nov-15	0.117	0.030	0.153
11-Dec-15	0.030	0.030	0.030
18-Dec-15	0.030	0.043	0.177
12-Jan-16	0.030	0.030	0.030
27-Jan-16	0.030	0.160	0.050
4-Mar-16	0.083	0.030	0.200

Appendix F: Nitrate

Nitrate measurements at 10 m^2 plots (mgl^{-1})

Date	Top	Middle	Bottom
8-Jan-15	5.40	6.57	4.53
28-Jan-15	3.90	3.80	5.17
11-Feb-15	1.70	1.93	1.70
23-Feb-15	7.70	4.13	2.67
3-Apr-15	5.00	8.83	5.00
27-May-15	2.10	2.07	3.03
5-Aug-15	3.03	4.20	2.13
22-Sep-15	6.45	5.10	5.97
16-Oct-15	6.15	5.87	1.35
17-Nov-15	2.55	5.90	4.40
11-Dec-15	5.23	4.70	3.80
18-Dec-15	2.00	1.53	3.53
12-Jan-16	0.87	5.07	2.90
27-Jan-16	7.10	2.50	6.40
4-Mar-16	3.90	4.87	3.77

Nitrate measurements at 1 m^2 plots (mgl^{-1})

Date	Top	Middle	Bottom
8-Jan-15	3.02	2.75	2.40
28-Jan-15	1.86	5.20	3.50
11-Feb-15	2.01	2.07	0.75
23-Feb-15	0.94	5.40	2.25
3-Apr-15	2.40	5.50	5.60
27-May-15	2.60	2.63	2.43
5-Aug-15	2.79	3.73	4.20
22-Sep-15	7.34	4.03	7.20
17-Nov-15	2.91	5.85	5.93
11-Dec-15	3.08	6.97	4.03
18-Dec-15	1.82	3.50	2.53
12-Jan-16	1.22	3.37	3.97
27-Jan-16	0.27	4.20	1.55
4-Mar-16	2.23	2.97	3.47

Appendix G: Dissolved Organic Carbon

Amount of dissolved organic carbon at 10 m^2 plots (mgl^{-1})

Date	Top	Middle	Bottom
8-Jan-15	6.93	26.70	10.40
28-Jan-15	18.90	11.10	11.20
11-Feb-15	4.49	17.40	2.95
23-Feb-15	21.90	34.30	16.90
3-Apr-15	8.29	29.50	10.00
27-May-15	11.10	9.18	7.26
5-Aug-15	16.70	19.90	13.30
22-Sep-15	17.70	23.00	17.30
17-Nov-15	30.30	46.60	42.90
11-Dec-15	22.53	23.30	23.53
18-Dec-15	14.67	13.92	9.67
12-Jan-16	18.37	14.75	18.70
27-Jan-16	49.70	34.30	23.20
4-Mar-16	24.93	23.57	29.50

Amount of dissolved organic carbon at 1 m^2 plots (mgl^{-1})

Date	Top	Middle	Bottom
8-Jan-15	16.90	11.40	7.08
28-Jan-15	16.80	7.64	7.05
11-Feb-15	4.29	11.90	10.40
23-Feb-15	25.90	14.80	6.02
3-Apr-15	7.50	22.40	9.80
27-May-15	7.36	11.30	15.90
5-Aug-15	22.70	24.90	24.80
22-Sep-15	6.39	16.80	15.10
17-Nov-15	30.67	44.80	40.57
11-Dec-15	23.53	26.27	28.73
18-Dec-15	15.10	11.60	16.85
12-Jan-16	16.05	26.73	12.00
27-Jan-16	23.45	43.60	14.23
4-Mar-16	26.70	35.50	44.10

Appendix H: Particulate Organic Carbon

Volumes of Particulate Organic Carbon at 10 m^2 plots (gl^{-1})

Date	Top	Middle	Bottom
8-Jan-15	0.049	0.105	0.034
28-Jan-15	0.045	0.066	0.024
11-Feb-15	0.056	0.210	0.026
23-Feb-15	0.034	0.034	0.034
3-Apr-15	0.051	0.034	0.019
27-May-15	0.063	0.063	0.036
5-Aug-15	0.106	0.023	0.033
22-Sep-15	0.082	0.017	0.030
17-Nov-15	0.399	0.057	0.080
11-Dec-15	0.041	0.040	0.028
18-Dec-15	0.292	0.701	0.507
12-Jan-16	0.055	0.128	0.065
27-Jan-16	0.046	0.0743	0.059
4-Mar-16	0.147	0.074	0.090

Volumes of Particulate Organic Carbon at 1 m^2 plots (gl^{-1})

Date	Top	Middle	Bottom
8-Jan-15	0.160	0.129	0.084
28-Jan-15	0.150	0.045	0.073
11-Feb-15	0.030	0.052	0.036
23-Feb-15	0.139	0.035	0.026
3-Apr-15	0.038	0.028	0.045
27-May-15	0.045	0.069	0.094
5-Aug-15	0.017	0.021	0.027
22-Sep-15	0.027	0.021	0.030
17-Nov-15	0.082	0.194	0.059
11-Dec-15	0.041	0.048	0.028
18-Dec-15	0.125	0.207	0.187
12-Jan-16	0.037	0.134	0.067
27-Jan-16	0.044	0.068	0.028
4-Mar-16	0.382	0.232	0.253