The Effects of Burning or Mulching of Harvest Residues on Selected Soil Properties in a *Eucalyptus* Plantation in Northern KwaZulu-Natal

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Declaration

I hereby certify that the research reported in this dissertation is the result of my own investigation, except where otherwise indicated, and that it has not been submitted for a higher degree at any other university or institution.

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As the candidate's supervisor I have/have not approved this dissertation for submission

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As the candidate's co-supervisors we have/have not approved this dissertation for submission

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Abstract

The sustainable management of commercial forest resources is required to ensure long-term soil fertility and the productivity of later rotations; this includes soil nutrient retention and the protection of soil structure. A major factor in the protection of soil structure is the distribution of soil organic carbon (SOC) and nutrients, as well as the stability of soil aggregates. These are influenced by forest harvest residue management practices such as the removal, displacement on the soil surface, or burning of residues. Soil aggregate stability is an important soil property that gives a reliable estimation of the ability of soils to respond to external forces such as rainfall, wind, and land management. The objectives of the study were to investigate the effect of forestry residue management methods on selected soil properties and their effect on soil structural stability under the previous stump-line and in the current inter-row. The results obtained from the study will contribute to understanding the effects of forest harvest residue management on some soil properties.

The study site was near Paulpietersburg, northern KwaZulu-Natal, South Africa. A trial was established by the Institute for Commercial Forestry Research (ICFR) in 2010 to investigate the effect of tree harvest residue management, soil compaction and its amelioration through ripping, on the growth of *Eucalyptus dunnii* grown on a Magwa soil form. This trial has a factorial treatment design and consists of three soil disturbance treatments (no compaction, compaction, and compaction with amelioration) and two residue management practices (burning and residue mulching), with four replicates (24 treatment plots). The effect of burning and mulching harvest residue on selected soil properties and soil aggregation in relation to other soil properties was studied two years after the treatments were established.

For the purposes of the present study, the no-compaction treatment and both residue management treatments (burning and mulching) were used (eight plots). Bulk samples, 192 in total, were collected from each of the eight plots, from both the current inter-row and the old stump-line, at a depth of 0 to 0.1 m and 0.1 to 0.2 m using a spade. Samples were air dried and sieved to collect soil aggregates between 2 and 8.5 mm for soil aggregate stability determinations by determination of the mean weight diameter (MWD). Some of the bulk sample was analysed for SOC, pH, exchangeable bases (Ca^{2+}, Mg^{2+}, K^+, and Na^+), and particle size distribution. Statistical analyses were carried out using Genstat and the results were regarded as significant if $p < 0.05$ (or 5%).

Residue management had no significant ($p \geq 0.05$) effect on exchangeable Mg^{2+}, K^+, and Na^+, soil pH, exchangeable acidity, effective cation exchange capacity and texture. There was, however, a significant effect of residue management on SOC and exchangeable Ca^{2+}.

Under the mulched plot treatment, SOC at the 0 to 0.1 m depth was significantly higher in soils when compared to the burned plot treatment. Under the burned plot treatment, exchangeable Ca^{2+} concentrations were significantly higher than the mulched treatment. Furthermore, under the burned plot treatment; exchangeable Ca^{2+} and Mg^{2+} concentrations were significantly higher in the 0 to 0.1 m depth than at the 0.1 to 0.2 m depth. Neither residue management strategy had any significant effect on MWD. However, there was a significant effect on MWD relating to soil depth and sampling position. Under both treatments, the MWD was higher at 0 to 0.1 m than at the 0.1 to 0.2 m depth, in both inter-row and stump-line samples. The average MWD of 2.45 mm (with a standard error of 0.04) and values that ranged between 0.8 and 3.5 mm was indicative of stable aggregates. Thus, surprisingly, no significant correlation was found between SOC and MWD. However, it was concluded that changes in residue management may alter SOC, thus impacting on the productivity of the soil. Furthermore, it was suggested that soil aggregate stability was driven by exchangeable Ca^{2+} and Mg^{2+}.

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Declaration: Plagiarism

I, Siphamandla Madikizela, declare that

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Dedication

I dedicate this dissertation to the memory of my beloved father; may your soul wish us the best as the family; see you when we get there.

Zingisile Mthokwana Madikizela

1955 – 2003

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Chapter 1: Introduction

1.1 Introduction

Plantation forestry is expanding worldwide, especially in the tropics and subtropics where fertility is largely supported by soil organic matter (SOM) (Lal, 2004). However, it is not expanding in South Africa (SA) owing to an unchanging supply of land and current government legislation (Aitken, 2004). There is the need, therefore, to increase and to protect site productivity in the future (Dovey *et al.*, 2011). The continued success of commercial forest plantations in SA is highly dependent on maintaining or improving tree growth through improved silvicultural and management techniques.

South African plantation forestry areas are distributed over a wide range of climatic zones, with a total commercial timber plantation area of 1 274 869 ha (DWAF, 2009). The majority of these plantations are in the Mpumalanga and KwaZulu-Natal provinces of SA, accounting for 40.2 % and 38.9 % of the total forestry plantation area, respectively (FSA, 2013). Generally plantations are on acidic soils in the SA summer rainfall region. The commercial forestry industry contributes 1.27 % to the gross domestic product (GDP) and employs 0.5 million people (FSA, 2009). This important forest plantation resource is managed through the intensive silviculture of fast growing, exotic trees with the aim of optimising yield for the saw timber and fibre markets. Two of these, *Pinus* and *Eucalyptus,* are planted extensively on selected areas in SA with 478 000 ha planted to eucalypts; eucalypts thus account for 57 % of the total area planted for industrial pulpwood (FSA, 2010). The plantations originally are established into grassland ecosystems and these are spread across a relatively large range of site types and soil forms (Dovey, 2012). South African forest plantations occur on soils which are predominantly kaolinitic and thus do not swell and shrink, or freeze and thaw (Smith and Johnston, 2001). Most of the soils are susceptible to soil compaction caused by use of machinery (Rietz, 2010).

The *Eucalyptus* was introduced into SA in 1828 and is used primarily for pulpwood, mining timber, and pole production (O'Connell *et al*., 2000). The *Eucalyptus* grown for pulpwood production has a typical rotation length of eight to fifteen years under intensive management (Brown *et al*., 1997; Fernandez *et al*., 2004) before harvesting. These short rotation spans have raised concerns that soil fertility may decline under plantations owing to losses of nutrients and organic matter during the harvesting phase, both directly through export in harvest material and indirectly through leaching (Mendham *et al*., 2002, Titshall *et al*., 2013).

At harvesting and during the inter-rotation phase, management typically involves the conservation, displacement, or loss of harvest residues through mulching, removal or burning, all of which can potentially impact a site (Dovey *et al*., 2011). These harvest residue management practices can have both direct and indirect impacts on soil organic carbon (SOC) and changes thereof. They have the potential to affect or alter soil structure, carbon content (micro-organisms included) and site nutrient supply and storage (du Toit *et al*., 2010; Rietz, 2010). The effects of harvest residue management practices are thus an important focus of research in the long term sustainability of commercial forestry sites.

The practice of burning harvest residues prior to the planting of a new rotation is wide spread in SA. The main reasons for this are to reduce fuel loads and wildfire risks (Dovey, 2012), and to facilitate re-establishment through the cost-effective clearing of residues prior to planting (Norris, 1992; Monleon and Cromack, 1996). However, fire can induce both short-term and long-term changes on soil properties, depending primarily on the type of soil property and the severity and frequency of fire (Certini, 2005). Burning of residues has a number of impacts on the biological, chemical, and physical properties of soils (Kim *et al*., 1999). A number of studies have shown that burning can cause significant loss of SOC and associated nutrients (du Toit, 2003; Mendham *et al*., 2003; du Toit *et al*., 2008; Dovey, 2012). These include changes in the nutrient availability of a site (du Toit and Scholes, 2002; Nadel, 2005) and the loss of SOM (Mendham *et al*., 2002). An added complication is the effect of burning residues on the environment; this practice contributes directly to the “greenhouse effect” that is considered to be allied to global warming (Basanta *et al*., 2003).

The retention of harvest residual biomass (branches, foliage, bark, and litter from the previous rotation) on site, on the other hand, is an expensive operation. *In situ*, this biomass can be windrowed, broadcast, or mulched. Mulching has the advantage over residue broadcasting of reducing fire risk and positively facilitating operations associated with the establishment of a new stand. More importantly, it has been shown to affect positively the long-term SOC level that is a key element in sustaining site productivity (Vance, 2000). It has also been shown to result in an increase in SOM (Smith *et al*., 2000), a reduction in erosion, the conservation of soil nutrients (Blumfield and Xu, 2003), and the conservation of soil moisture (O’Connell *et al*., 2004); all of these may affect tree establishment and early growth in the subsequent stand. Mulches and surface residues also decrease evaporation rates (Norris, 1993; Mulumba and Lal, 2008) and moderate soil surface temperature (Rathore *et al*., 1998).

The effect on soil structure and aggregate stability of the practices of burning or mulching of residues, through the loss or improvement of SOC, respectively, is of particular importance.

Favourable soil structure and high aggregate stability are important in improving soil fertility, increasing agronomic productivity, enhancing porosity and decreasing erodibility (Bronick and Lal, 2005). Soil structural and aggregate stability are often directly correlated with organic carbon (Tisdall and Oades, 1982). Soil organic matter improves the structural stability of soils at various scales, reduces bulk density, and improves water holding capacity. Measurements of SOC content and soil aggregate stability assist in the assessment of the risk of soil structural degradation. The evaluation of relationships between management practices, soil nutrient status and tree growth at a particular site can thus enable forestry managers to develop sound, sustainable forestry management practices. The key research questions were:

- Does mulching or burning of harvest residues affect soil structural stability?
- Do these effects differ between soil in an old stump-line and soil under planted trees?
- What are the main drivers of structural stability under these different residue management practices and at the different locations within each treatment?

The above questions form the basis for this study.

1.2 Objectives and justification for the study

The primary objectives of this study were, therefore, to investigate, two years after treatment implementation, the effects of:

a) Burning or mulching harvest residues on soil structural stability; and
b) Other soil properties that may also affect the structural stability of soil under a specific eucalypt plantation.

Despite previous studies, the literature on these effects under eucalypt forestry management is sparse and provided the impetus for this study which investigates the impact of forestry harvest residue management on soil structural stability and selected indicators of structural stability under a eucalypt plantation.

1.3 Outline of the study

The dissertation is structured as follows:

- **Chapter 2** reviews the literature on the factors affecting soil aggregate stability and the effect of burning or mulching of harvest residues on soil properties.
- **Chapter 3** describes the materials and methods used to carry out the research.

- **Chapter 4** outlines the results of the investigation into the effect of harvest residue burning and mulching on selected soil physico-chemical properties and discusses their importance under the mulched and burned plot treatments.
- **Chapter 5** outlines the results for soil aggregate stability and discusses the relationships between the measured soil properties and aggregate stability.
- **Chapter 6** presents the general discussions, summarises the conclusions of the study, and gives recommendations for future research.

Chapter 2: The impact of selected soil properties and forestry residue management practices on the structural stability of soil: A literature review

2.1 Introduction

Forest soil properties, including the quantity and quality of soil organic carbon (SOC) are influenced by complex interactions between soil type, climate, tree species, and human forest management practices (Lal, 2005). By burning harvest residue during intensive site preparation, commercial forestry may deplete the soil of SOC and nutrients, disrupting the soil structure. The mulching of harvest residue, on the other hand, may have the converse effect.

Soil structure is important to soil functioning as it affects water, gas and nutrient fluxes and storage, and therefore influences the activity and growth of living organisms (Angers and Caron, 1998); erosion, crusting, soil water movement and retention, crop yield and root penetration (Bronick and Lal, 2005). One of the measures of good structure is the stability of soil aggregates in water (Six *et al*., 2000). When structure is good, water infiltration into the soil is fast and also a structured soil is high in biological activity, root growth and absorbs enough water for plant growth (Piccolo and Mbagwu, 1999). If soil structure breaks due to land use and soil/crop management practices (Bronick and Lal, 2005), particle detachment and runoff are increased as infiltration and water retention are reduced. Additionally, poor soil structure may increase the potential for soil erosion (Sarah, 2005). The destruction of SOC and loss of nutrients will affect the water-holding capacity of the soil structure and hence the long-term sustainability of commercial plantations, particularly where site preparation is intensive.

This chapter reviews background information to the present study on the effects of burning or mulching of harvest residues on selected soil properties in a *Eucalyptus* plantation in northern KwaZulu-Natal. Also the objective of this review is to summarise the different factors that affect soil structural stability.

2.2 Soil properties that affect the structural stability of soil

Soil structure is a combination of aggregates of bonded inorganic particles with associated pores. These bonded particles are often operationally divided into micro-aggregate (< 250 μm diameter) and macro-aggregate (> 250 μm diameter) size fractions. Aggregation involves interactions between soil constituents, such as clay minerals, SOC, and the soil microbial community together with the impacts of land use and site management practices (Wei *et al*., 2006).

The structure of the soil can be characterised by its form, resiliency, and stability. Structural stability refers to soil resistance in maintaining its own structure when it is subjected to external forces, particularly those resulting from moisture, any particular dispersive process, and the impact of raindrops (Mataix-Solera *et al*., 2011). This stability, influenced by land use and management, is an important indicator of soil physical quality. According to Brady and Weil (2008), rainfall is the main natural agent responsible for the breakdown of soil aggregates and its effect is threefold: (i) raindrop impact destroys aggregation; (ii) splash detaches soil aggregates and particles; and (iii) runoff removes soil.

The aggregate stability of soils is influenced mainly by SOC, pH, texture, clay mineralogy, cation exchange capacity, aluminium and iron oxides, soil micro-organisms, and microbial biomass and activity (Figure 2.1) (Amezketa, 1999; Six *et al*., 2002; Bronick and Lal, 2005). In particular, SOC, clay content, and mineralogy play key roles in controlling soil structural stability through their influence on water sorptivity and repellency as well as on the strength of bonds between particles.

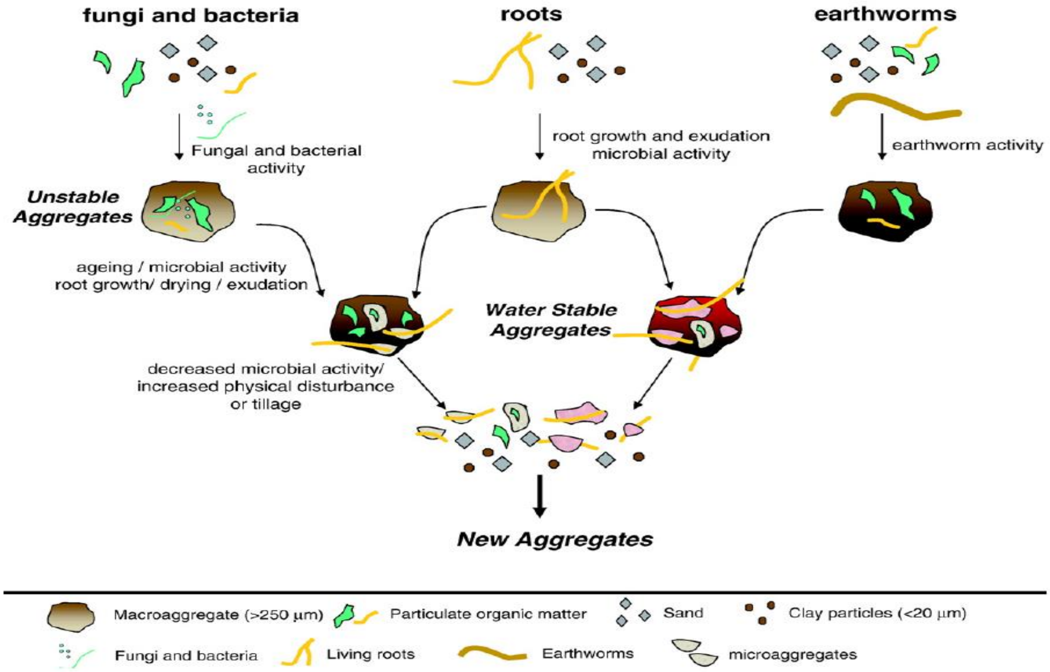

Figure 2.1: Aggregate formation and degradation mechanisms in soils (Six *et al*., 2002).

2.2.1 Soil organic carbon

Soil organic carbon is formed from readily decomposable material such as roots, dead and living organisms and humus (Gregorich *et al*., 1994). Soil organic carbon is considered a major binding agent that stabilizes soil aggregates (Haynes and Beare, 1997; Abiven *et al*., 2009). According to Emerson (1967), SOC binds the primary particles in the aggregate and this, in turn, increases the stability of the aggregates and limits their degradation during any wetting process. The relationship between organic carbon content and macro-aggregate stability has been well documented (Tisdall and Oades, 1982; Haynes *et al.*, 1991). For example, the close relationship between soil aggregate stability, estimated by mean weight diameter (MWD), and SOC is presented in Figure 2.2, where MWD shows an increase with increasing SOC in oxisols from Thailand (Trakoonyingcharoen *et al*., 2012).

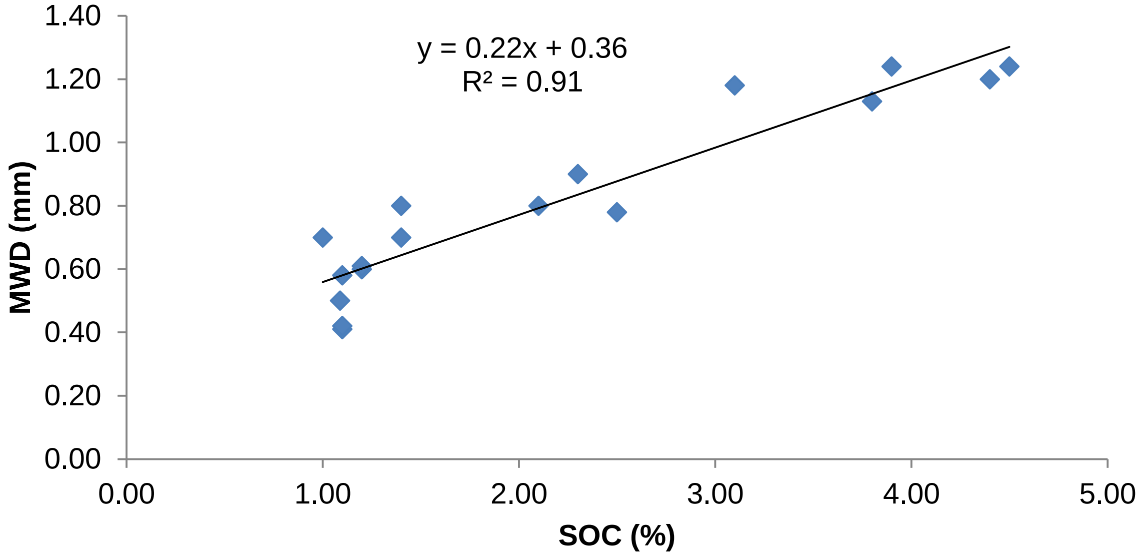

Figure 2.2: The relationship between mean weight diameter (MWD) and soil organic carbon (SOC) (redrawn from Trakoonyingcharoen *et al*., 2012).

2.2.2 Clay mineralogy and soil texture

Soil texture is originally inherited from the parent material and subsequently modified by atmospherically derived dust and chemical and physical weathering. When these inorganic particles aggregate together, the clay content and mineralogy of the soil affect both the degree of aggregation and stability of the aggregates. Clay acts as a cementing material that holds particles together in the aggregate (Boix-Fayos *et al*., 2001) and clay mineralogy has substantial effects on aggregate stability (Opara, 2009). Clay also influences SOM decomposition and turnover (Bronick and Lal, 2005). In the absence of SOC, high clay content plays a major role in promoting soil particle aggregation. For example, Tayel *et al*. (2010), who investigated the effect of some soil characteristics on soil structure in Egyptian topsoil's, found that those with high clay content were more strongly aggregated (Table 2.1).

Table 2.1: The influence of clay content on soil aggregation in some Egyptian top-soils (modified from Tayel *et al*., 2010)

Profile number	Depth (m)	Clay content (%)	Aggregation (%)
1	0-0.25	33.96	63.86
3	0-0.20	70.28	80.50
4	0-0.30	47.35	73.06

Soils dominated by shrink-swell (often smectitic) clays are characterised by unstable aggregation because, as clay particles swell, they separate from other particles, causing dispersion; oxides and kaolin clays, on the other hand, are responsible for very stable aggregation (Six *et al.*, 2004). Six *et al.* (2000) and Lado and Ben-Hur (2004) found that clay soils dominated by kaolinite and vermiculite had the highest aggregate stability when compared to soils dominated by chlorite and illite. Six *et al*. (2000) concluded that the increased aggregate stability of the soil is due to the stabilising effect caused by interaction between the kaolinite and Fe and Al oxides.

2.2.3 Exchangeable bases and pH

Polyvalent cations, notably Ca^{2+} and Mg^{2+}, are important for the stabilization of SOM and aggregates through their role in the formation of clay-cation–organic matter complexes (Clough and Skjemstad, 2000). Calcium is more effective than Mg^{2+} in improving soil structure. Calcium inhibits clay dispersion and the associated disruption of aggregates by replacing Mg^{2+} and Na^{+}. The latter is a highly dispersive agent resulting directly in the break-up of aggregates and indirectly affecting aggregation through a reduction in plant productivity (Bronick and Lal, 2005). Calcium thereby increases aggregate stability (Bronick and Lal, 2005).

On the other hand, high exchangeable sodium percentage (ESP), which is highly correlated with sodium adsorption ratio (SAR), causes the cohesive forces within the soil aggregates to be weak and enhances slaking (Greene *et al*., 2002). Also, an increase in SAR leads to greater dispersion causing soil structural degradation (Marchuk and Rengasamy, 2010).

The chemistry of soils is complex and interactional so changes in one property, such as pH, may affect another property, such as the availability of nutrients. Soil pH affects biological, chemical, and physical soil properties (Brady and Weil, 2008). The pH of soil solution can greatly influence the dispersion/flocculation of clay particles and thus soil aggregation. At low pH, soils are normally flocculated (Haynes and Naidu, 1998) and both Al^{3+} and Fe^{3+} control aggregation in acidic soils (Barral *et al*., 1998). These cations stabilise soil aggregates through cationic bridging and the formation of organo-metallic compounds and gels (Amezketa, 1999). Large aggregates form in soils of high pH and high carbonate concentration (Boix-Fayos *et al*., 2001).

2.3 Residue management effects on soil properties

Residues from the harvesting of eucalypt plantations are either retained on the forest floor area (broadcast, windrowed or mulched), removed during harvesting, or burnt directly on the area (Norris, 1992). These residues represent a significant proportion of the organic matter and nutrient store of a site, and their management through retention may influence longer term plantation productivity through changes in SOC and nutrient supply (Mendham *et al*., 2002). The retention of residues may also act as a buffer against nutrient losses by reducing leaching during the early stages of plantation development when root systems have limited spatial extent and are unable to utilise all available nutrients (Shammas *et al*., 2003).

Burning has both positive and negative effects on soil biological, chemical, and physical properties (Certini, 2005). Carbon and N are lost in volatile forms or as particulates as a result of burning (DeBano, 1990) and so burning of residues can cause a significant loss of SOC and associated nutrients such as N in the long-term (Mendham *et al*., 2002; Prieto-Fernandez *et al*., 2004). This pattern depends on the type of soil and temperatures reached on the soil surface during the burning process. The burning of residues may also induce increased leaching as nutrients locked up in the residues and surface soil are mobilised during combustion (Powers *et al*., 2005).

2.3.1 Soil organic carbon, exchangeable bases and pH

Nutrients occur in forest ecosystems in several pools, namely in the above and below-ground living biomass, on the forest floor, and in the soil. Base cations are held in varying quantities on the exchange complex of soils after their release by mineral weathering and this is an important long-term source of nutrient input to forest systems (Ackerman *et al*., 2013).

Tiarks and Ranger (2008) summarised the effects of slash management and other practices on soil chemical properties that were measured in a network of 16 sites where tropical forest plantations were harvested and replanted under tropical conditions. Retaining residues on the soil surface did not affect SOC at six sites but increased SOC at five other sites. Soil pH changed with residue retention only at four sites out of nine, at three of which an increase in pH was measured. The soil pH was acidic at all the sites and relatively constant with depth, time and treatment. Retention of residues had an effect on exchangeable soil K^+ at only one site out of fourteen.

Mendham *et al*. (2003) investigated the effects of residue management on nutrient contents and growth of second rotation eucalypts in south-western Australia. They found that the retention of residues after harvest resulted in higher exchangeable K^+, Ca^{2+}, and Mg^{2+} in the 0.05 to 0.20 m soil while Ca^{2+} and Mg^{2+} initially increased in the 0 to 0.05 m depth after burning but decreased within two years. Du Toit *et al*. (2008), however, found a significant increase in exchangeable soil Ca^{2+} but no effect on exchangeable soil Mg^{2+} when residues were retained in their South African study.

Fire can modify rates of litter decomposition and nutrient release, and these changes may influence patterns of nutrient availability and long-term productivity (Dovey, 2012). Hot fires can lead to considerable damage of a forest ecosystem. According to Rab (1996), burning alters the physical and hydrological properties of soils under eucalypt plantations, affecting site productivity and increasing the rate of erosion. Forest fires can induce or enhance water repellency in most forest soils which, in turn, can reduce slaking caused by increased pore pressure in aggregates as water enters the aggregate by matric suction (Zavala *et al*., 2010). The most significant responses to fire tend to be concentrated in the upper 0.05 m of the soil with lesser responses occurring at greater depths. Furthermore, the burning practice is associated with loss of N and has a negative impact on the long-term status of SOC (Biederbeck *et al*., 1980). Choromanska and DeLuca (2002) reported that C and N mineralization decreased after fire and did not recover within the nine months of their study period. The burning of residues transforms organically bound nutrients to inorganic forms that are often more readily available for plant uptake (Morris, 1986; Christensen and Abbott, 1989). However, burning residues can also lead to leaching of some nutrients, such as K, below the rooting zone due to increased mobility and will also result in large volatile losses of other nutrients, particularly N (DeBano, 1990). In the longer term, repeated burning of harvest residues has the potential to degrade the fertility of plantation sites (O'Connell *et al*., 2000). The soil pH after burning residues tends to increase due to hydrolysis of the base cation oxides which are abundant in ash (Ballard, 2000). Du Toit *et al*. (2008) in their *Eucalyptus grandis* study in SA found that soil pH was high after burning in soils derived from shale and dolerite parent materials. There was a significant increase in pH that occurred between planting and two years of age for all the treatments before pH declined to initial levels after seven years (Figure 2.3). The increase in pH was attributed to an increased decomposition of residues.

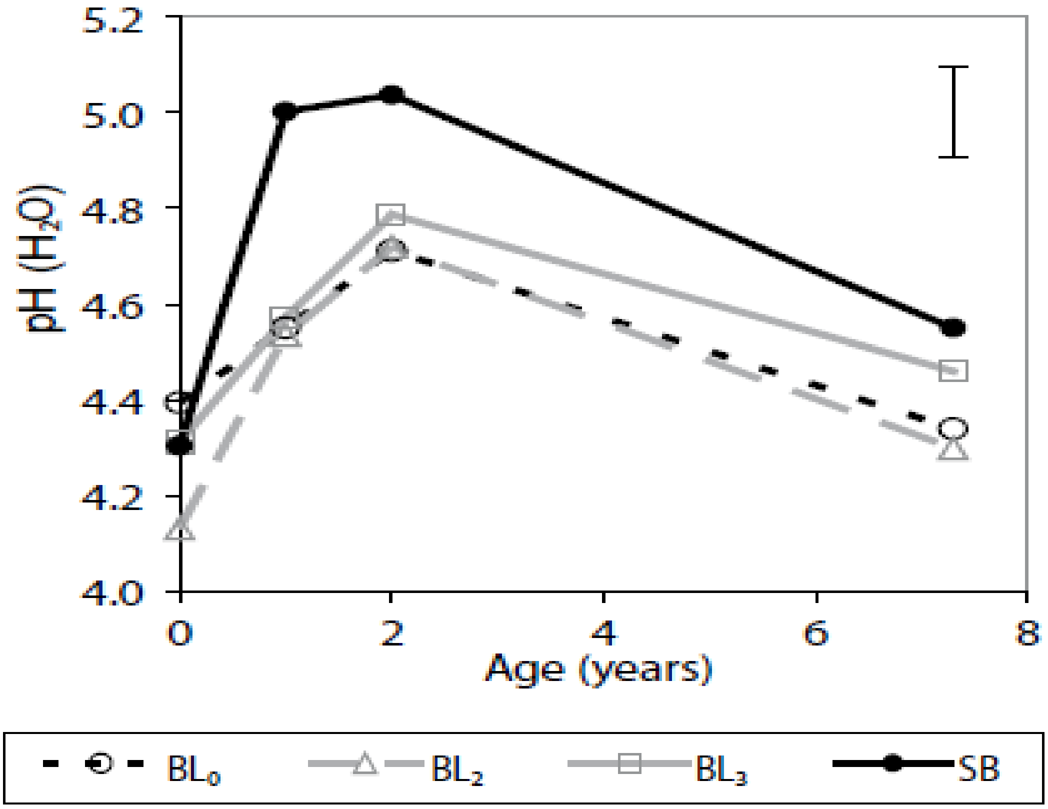

Figure 2.3: Changes in soil pH of the top-soil (0 to 0.1 m) over time for selected treatments. BL_0: all residues and litter removed; BL_2: residues retained and uniformly distributed; BL_3: double the quantity of residues, SB: residues burnt (du Toit *et al*., 2008).

2.3.2 Clay mineralogy and soil texture

Although the texture and mineralogy of a site cannot be easily altered by management (Titshall *et al*., 2013), phyllosilicate minerals and iron oxides may be affected by high temperatures generated at the soil surface by forest fires (Ulery *et al*., 1996). The effect of residue management on clay mineralogy and texture in eucalypt plantations has not been reported in the public domain literature. However, laboratory studies have been done on the influence of temperature on soil properties, where soils were heated at temperatures from 300 to 600ºC (Sertsu and Sanchez, 1978; Ulery *et al*., 1996; Ketterings and Bigham, 2000; Ketterings *et al*., 2000; Fox *et al*., 2007; Varela *et al*., 2010). Generally, only moderate and severe fires have potential to cause strong mineral transformations because dehydroxylation of most minerals occurs above 500ºC (Mataix-Solera *et al*., 2011).

Soil texture influences soil physical (i.e. water, air and strength) and nutrient retention characteristics (Brady and Weil, 2008). The components of soil texture (sand, silt, and clay) have high temperature thresholds and are not usually affected by fire unless they are subjected to high temperatures at the mineral soil surface. Kim *et al.* (1999) analysed topsoil under unburned and burned areas in *Pinus densiflora* stands to determine particle size distribution and they found that there was no significant difference ($p > 0.05$) between the unburned and the burned areas at each soil depth.

2.3.3 Decomposition of residues and nutrient release

Litter decomposition is an important process in forest nutrient cycling and formation of SOC (Bargali *et al*., 1993). Decomposition is a complex phenomenon influenced by the activity and nutrient demand of heterotrophic decomposers (Kaila *et al*., 2012). It is influenced by residue management, litter quality and climatic conditions (Prescott, 1996). Decomposition of plant residues is aided by soil fauna and in doing so they release nutrients such as nitrogen into the mineral soil.

Rates of decomposition are generally faster in soils that are high in clay, with higher content of bases, pH and water retention, and that are warmer. Plant residues have variable C/N ratios, lignin and poly-phenol contents (Berg, 2000). Nutrient concentrations of eucalypt harvest residues vary with species, age, site characteristics, decomposition period and rate and residue management operations (Dovey, 2005). Hernandez *et al*. (2009) found that during decomposition of *Eucalyptus dunnii* harvest residues under a temperate climate in Uruguay, there was a decrease in the biomass of the different components (bark, leaves, surface litter [mix of all residues] and branches) over 24 months (Figure 2.4). At the end of their study they found significant differences in the decomposition rates for the different components, except for branches and non-commercial logs which did not differ. Residue decomposition showed that the leaves lost biomass more rapidly when compared to other components and bark was most resistant to decomposition in the field (Figure 2.4) (Hernandez *et al*., 2009).

The same trend was found by Jones *et al*. (1999) in their study where they investigated the effect of organic matter management on the productivity of *Eucalyptus globulus* stands in Spain and Portugal. Harvest residues have the potential to make a major contribution to nutrient cycling in plantations (O'Connell *et al*., 2004). The decomposition of harvest residues left on the site releases or stores large quantity of nutrients, which are likely to increase soil carbon pools (Mendham *et al*., 2002). Changes in SOC or cations due to harvest residues might lead to changes in soil structural stability since they play a large role in influencing stability.

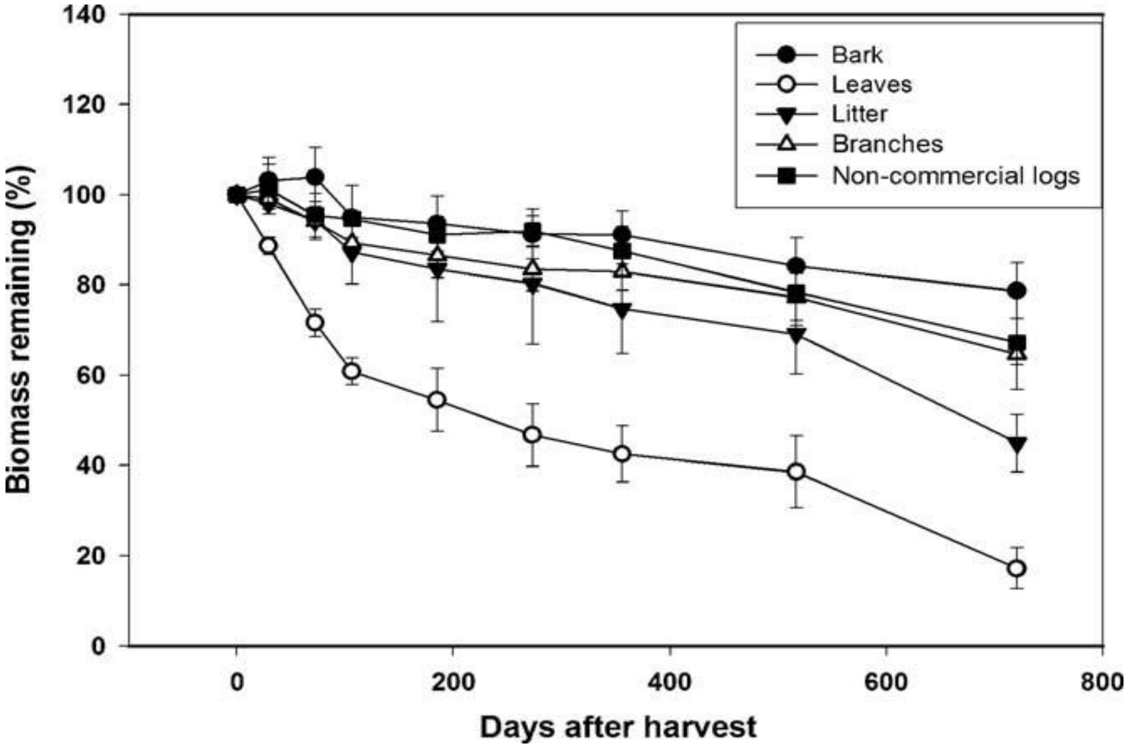

Figure 2.4: Decomposition rates of harvest residue components of *Eucalyptus dunni*. The vertical bars indicate standard errors for each sampling (Hernandez *et al*., 2009)

Eucalypt plantations produce large quantities of biomass, which can be a significant store for nutrients, with most of the nutrients in harvest residues concentrated in the branches and bark (Jones *et al*., 1999; Dovey, 2005). In eucalypt, quantities of K^+ are higher in large branches, while Ca^{2+} and Mg^{2+} are higher in bark than in other components (Dovey, 2005; Sankaran *et al*., 2005). Leaves are important pools of organic matter and nutrients in *E. grandis* plantations.

2.4 Effect of residues and soil organic matter on soil aggregate stability

The measurement of the degree of aggregate stability of a soil is often reported as being related to SOC content and that a high degree of aggregate stability is desirable for adequate plant growth (Bronick and Lal, 2005). Soil aggregate stability in soils under *Eucalyptus* has been reported by Lawal *et al*. (2009); Avanzi *et al*. (2011); Zi-Cheng *et al*. (2011) and Mohanty *et al*. (2012) and forest soils tend to have stable aggregates (Mataix-Solera and Doerr, 2004). Mohanty *et al*. (2012) investigated soil aggregate stability under different vegetation covers in a vertisol of Central India and found that stability of aggregates (MWD) was higher in the soil under *Eucalyptus* compared to cultivated soil. The higher MWD was due to significantly higher SOC in soils and a significant and positive correlation ($p <$

0.05, $R^2 = 0.61$) between MWD and SOC was observed. In contrast with other vegetation cover or land use, the effect of organic inputs (sludge, manure, crop residues, compost and straw) on soil aggregate stability is commonly reported and is used as a point of reference here. Haynes and Beare (1997) investigated the influence of six crop species (barley, wheat, prairie grass, Italian ryegrass, white clover and lupin) on soil aggregate stability, and found significant effects of crop species on aggregate stability. Mulumba and Lal (2008) found that wheat straw mulch application increased total porosity and soil aggregation with increasing mulching rates (0, 2, 4, 8 and 16 Mg ha^{-1} $year^{-1}$).

According to Andreu *et al*. (2001) binding agents of aggregation are affected by fire temperatures and thus there is a change in the aggregate stability of the soil. Mataix-Solera and Doerr (2004) found that structural stability can be increased by low to moderate intensity fires because of the formation of a hydrophobic film on the external surface of aggregates, whereas stability decreases dramatically at high temperatures due to the disruption of organic cements (Badia and Marti, 2003). In contrast to hot fires, cool burns result in less damage to a soil because they are controlled fires used to burn broadcast residues when conditions are moist, resulting in a partial reduction of litter loads (Norris, 1992). Arcenegui *et al*. (2008) investigated the immediate effects of wildfires on water repellency and aggregate stability in some Spanish Mediterranean calcareous soils. In their study, aggregate stability was significantly higher ($p < 0.001$) in burned samples when compared to unburned samples in all the aggregate size fractions they investigated (Figure 2.5).

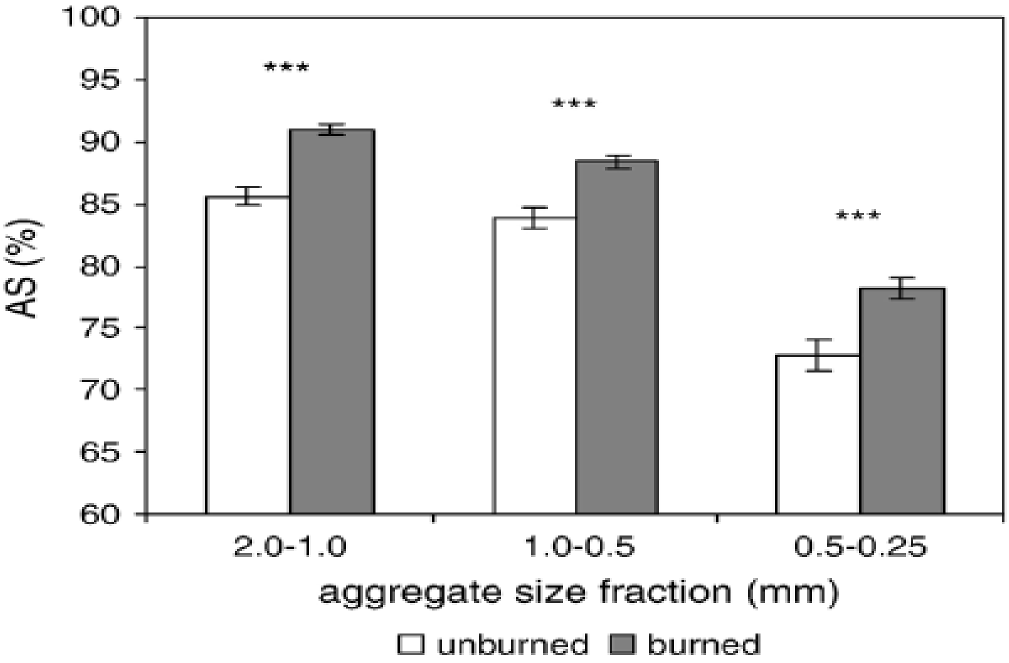

Figure 2.5: Aggregate stability (AS) in unburned and burned samples for different size fractions in Mediterranean calcareous soils, Spain. Mean ± standard error (n = 100). *** Significant difference at p < 0.001 (Arcenegui *et al*., 2008).

2.5 Conclusions

Favourable soil structure and high aggregate stability are important for improving soil fertility, increasing agronomic productivity, enhancing porosity and decreasing erodibility. Soil structural stability is an important indicator of soil physical quality and is influenced by a range of soil properties including clay type, organic matter, texture, exchangeable bases and pH, as well as land use and management. The extent to which SOC improves soil aggregate stability depends on the type of residues present. Returning residues to soil increases SOC, water retention and infiltration, and reduces runoff. Thus SOC improves the structural stability of soils at various scales, reduces bulk density and improves water holding capacity. Fire can modify rates of litter decomposition and nutrient release, and these changes may influence patterns of nutrient availability and long-term productivity. The burning of residues is associated with loss of nitrogen and has a negative impact on the long-term status of SOC (Biederbeck *et al*., 1980). Loss of organic cements due to combustion results in a decrease of structural stability and reduces the diameter of stable aggregates. The soil surface is exposed to high temperatures and rainfall leading to degradation of soil structure due to raindrop impact.

Chapter 3: Materials and methods

3.1 Site description

The study site is near Paulpietersburg, northern KwaZulu-Natal (KZN), South Africa at 270 19’ 54.38” S and 300 45’ 31.0” E (Figure 3.1) and at an altitude 1129 m above sea level. The climate is warm temperate, with a warm summer and a dry winter (Smith *et al*., 2005). The mean annual rainfall is 918 mm per year, mostly during summer, and a mean average midday temperature ranges between 9.7°C and 24.0°C (Camp, 1995). The soil at the site is derived from granite parent material and the soil form is classified as Magwa 1100 with a loam texture (Soil Classification Working Group, 1991). The soils are intensely weathered and the clay mineral composition is dominated by kaolinite, gibbsite and iron oxides (Fey, 2010). The site was originally a private farm, and no records were kept of nutrient status.

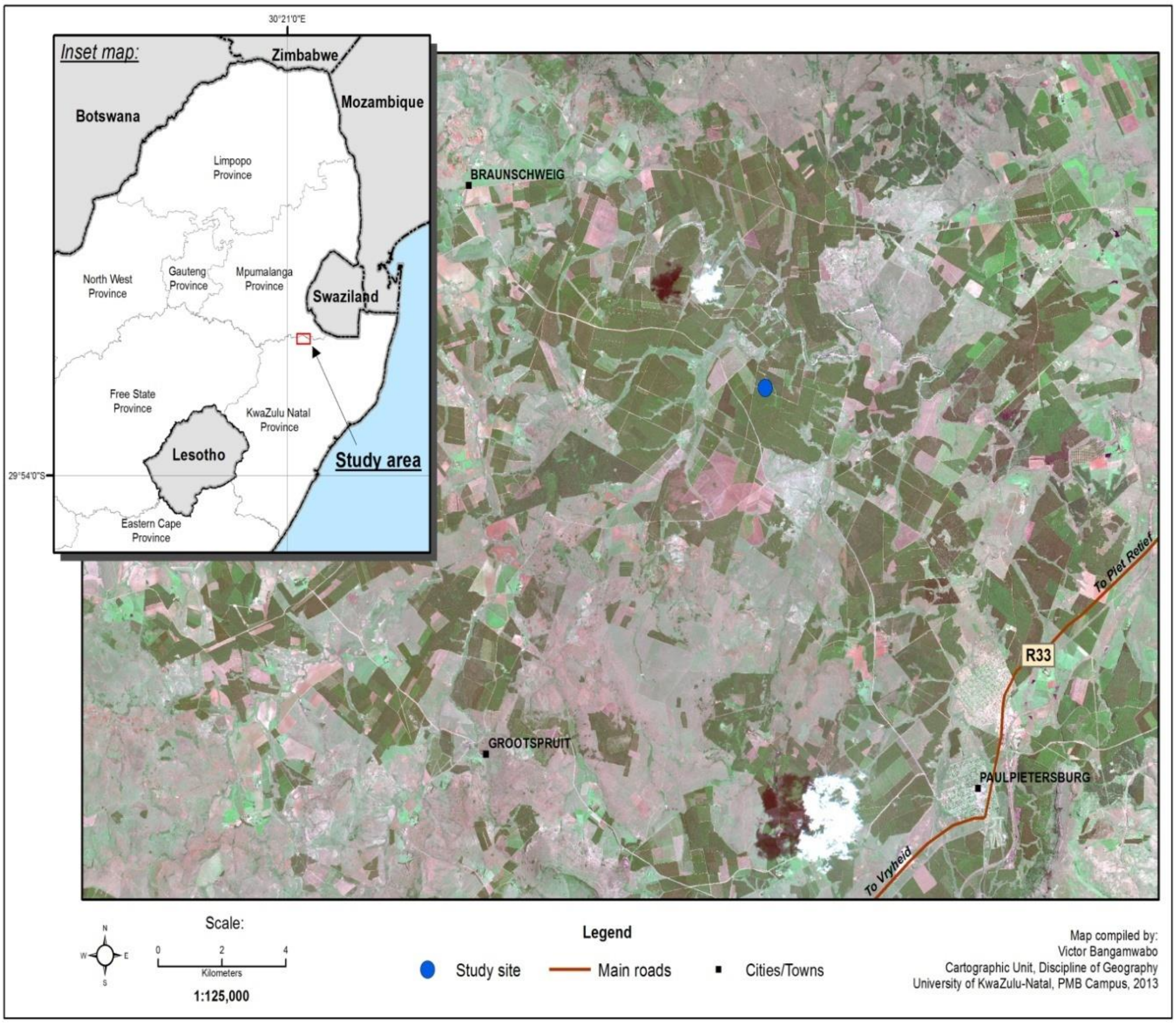

Figure 3.1: Location of the study site near Paulpietersburg, KwaZulu-Natal, South Africa.

3.2 Experimental Design

The Institute for Commercial Forestry Research, based in Pietermaritzburg, KwaZulu-Natal, established a trial, K10, at the study site in May 2010 to investigate the effect of eucalypt harvest residue management, soil compaction and amelioration treatments on the growth of *Eucalyptus dunnii*. The site was planted for the previous three rotations to *E. grandis* (D.N. Rietz *pers. comm.*[1]). The trial area was motor-manually clear-felled in March 2010 and 24 treatment plots were laid out, each measuring 17.0 x 43.2 m in size. Trees had been planted with 3.6 x 1.7 m spacing that resulted in treatment plots containing 10 x 12 trees.

The trial has a factorial treatment design, consisting of no compaction, compacted, compacted plus ameliorated, and two residue management practices, burned and mulched (Plates 3.1 and 3.2), with four replications (24 treatment plots) and arranged in a randomised complete block design (Figure 3.2). The no compaction treatment was implemented by ensuring no machinery movement over these plots and the use of manual felling and timber extraction. The burned plot treatments were implemented with a cool burn, while the mulched plot treatments used a tractor drawn Ahwimulcher M500 (AHWI Maschinenbau GmbH, Germany). Residue management treatments were applied after the compaction treatments had been applied.

The present study used both residue management treatments, burned and mulched, from the no compaction treatments with samples collected from plots 1, 2, 8, 11, 20, 21, 23, and 24 (Figure 3.2); plots 8, 11, 23, and 24 are on the up-slope, and plots 1, 2, 20, and 21 are on the down-slope.

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[1] Dr D.N. Rietz, Senior Research Scientist, Sustainable Forest Productivity Programme, Institute for Commercial Forestry Research, South Africa.

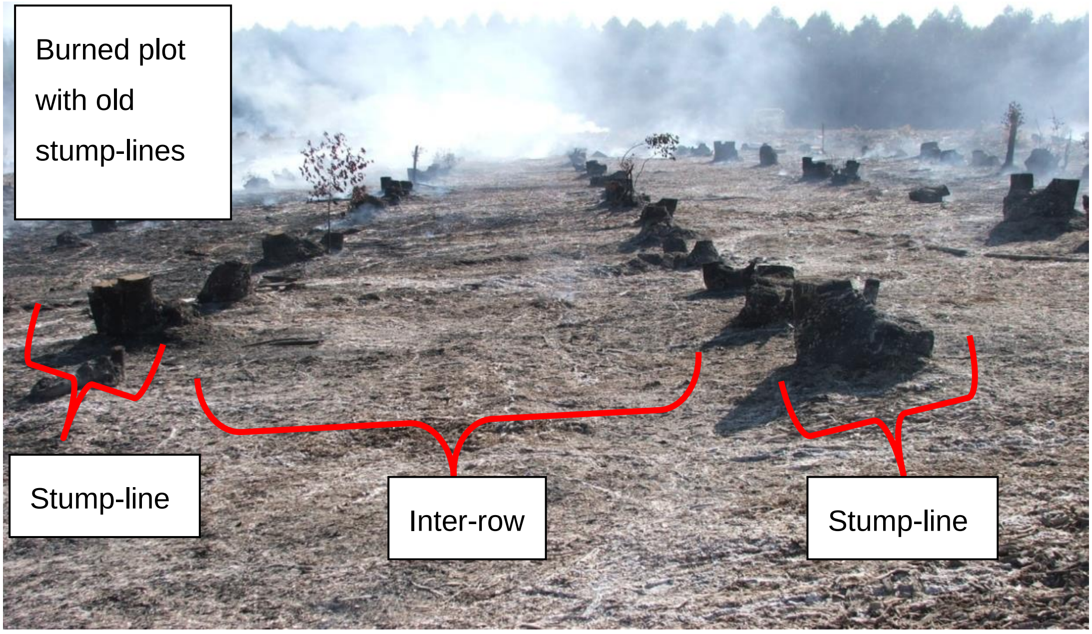

Plate 3.1: An example of the burned residue management treatment shortly after implementation of treatments (photo supplied by the Institute for Commercial Forestry Research ©, 2012).

Plate 3.2: An example of the mulched residue management treatment shortly after implementation of treatments (photo supplied by the Institute for Commercial Forestry Research ©, 2012).

Figure 3.2: Plot and treatment layout of the K10 trial with UTM co-ordinates. Plots 1, 2, 8, 11, 20, 21, 23 and 24 were used in the present study (figure overlay courtesy of Dr Gaghik Hoyhannissian (IRD), image Google Earth).

▨ Mulch (M)
☐ Burn (B, plots not shaded)

3.3 Soil sampling and sample preparation

Samples for this study were collected in February 2012, about two years after the trial was established. Samples were collected from inter-row (IR- areas either side of the stump-line) and from between the old stumps within the stump-line (SL- the line of stumps from the previous rotation) (Plates 3.1 and 3.2) from the mulched and burned plots of the no-compaction treatment. Samples were collected at 0 to 0.1 m and 0.1 to 0.2 m depths. Six samples were collected from the inter-row and six samples from the stump-line, at each depth per plot, which gave twelve samples from inter-row and twelve samples from stump-line per plot. The total number of samples per plot was 24, with a total of 192 representative soil samples for the study (8 plots x 24 samples). The samples were collected by spade to minimise the shearing and kneading effects of an auger that might alter the natural aggregation of the soil. Samples were transported in plastic bags, then air-dried, and a portion used to test structural stability; another portion was milled to pass through a 2 mm sieve for the analysis of organic carbon (on a sub-sample milled to < 0.5 mm), pH, exchangeable acidity, exchangeable bases and particle size distribution.

3.4 Laboratory analyses

3.4.1 Soil aggregate stability

A sub-sample of the air-dried soil was sieved using sieves of mesh size 2.8 and 5 mm. The collected aggregates were oven dried at 40ºC for 24 h and aggregate stability was measured according to the French norm NF X 31-515 (AFNOR, 2005). This method consists of three treatments: water treatment (WT), ethanol treatment (ET), and slow capillary wetting-ethanol treatment (SCWET). The WT (slaking due to fast wetting) was chosen for the present study as it is the most aggressive of the three treatments, mimicking rainfall with > 50 mm h^{-1} intensity on dry soil (to represent the effect of rapid water slaking) (AFNOR, 2005). A 5 to 10 g oven dried sample of aggregates was immersed in 50 mL distilled water for ten minutes. The water was siphoned off with a pipette, and the remaining soil material was gently transferred to a 53 µm sieve immersed in ethanol using an ethanol-filled wash bottle. The sieve was then gently swirled, five times to the right and five times to the left, by hand. The remaining aggregates on the 53 µm sieve were oven dried and gently sieved by hand on a nest of sieves of 2000, 1000, 500, 200, 106, and 53 µm mesh size. The weight of each fraction was then measured. The < 53 µm fraction was calculated as the difference between the initial weight and the sum of the weights of the other six fractions. The aggregate stability was expressed as the mean weight diameter (MWD) of the soil particle fractions obtained after wetting (Equation 3.1).

MWD = ((3.5 x [%> 2mm]) + (1.5 x [% 1 to 2 mm]) + (0.75 x [% 0.5 to 1 mm]) + (0.35 x [% 0.2 to 0.5 mm]) + (0.15 x [% 0.1 to 0.2 mm]) + (0.075 x [% 0.05 to 0.1 mm]) + (0.025 x [%< 0.05mm])) /100

[(% >2), (%1 to 2), (%0.5 to 1), (%0.2 to 0.5), (%0.1 to 0.2), (%0.05 to 0.1) and (%< 0.05)] = mass of aggregates as percentage aggregates per sieve (mm). **(Equation 3.1)**

3.4.2 Particle size distribution

Particle size distribution was determined by a modified pipette method (Gee and Bauder, 1986) on samples from each treatment (mulched and burned) at both sampled depths of 0 to 0.1 m and 0.1 to 0.2 m. Four samples from each plot were analysed; in total 32 samples were analysed. Air dry (< 2 mm) material was dispersed by the addition of 10 mL Calgon (35.7 g sodium hexametaphosphate and 7.9 g sodium carbonate per litre) and 15 mL distilled water with subsequent blender treatment for seven minutes. The dispersed sample was passed through a 0.053 mm sieve into a 1 L sedimentation cylinder and made up to the 1 000 mL mark with distilled water. The > 0.053 mm fraction (sand) was dried at 105°C then cooled in a desiccator before it was weighed and the < 0.053 mm fraction in the cylinders was brought into suspension by agitation with a custom-made plunger. The coarse silt (0.02-0.053 mm), fine silt (0.002-0.02 mm) and clay (< 0.002 mm) were determined by sedimentation and pipette sampling after appropriate settling times for each size fraction, according to Stoke's Law. The pipetted sample was dried overnight at 105°C. Separate sub-samples of all < 2 mm materials were dried at 105°C overnight for moisture content determination. All fractions are given as a mass percentage of oven-dry soil, after correcting for moisture content of air-dry samples.

3.4.3 Soil organic carbon, exchangeable bases, exchangeable acidity and pH

Soil organic carbon (on a < 0.5 mm sub-sample) was determined by acid dichromate oxidation (Walkley, 1947). Calcium, K^+, Mg^{2+}, and Na^+ were extracted with ammonium acetate (Thomas, 1982). Calcium and Mg^{2+} were measured by atomic absorption spectrophotometry and Na^+ and K^+ by flame emission spectrometry using a Varian (AA280FS) atomic absorption spectrophotometer. Exchangeable acidity was determined by titration after extraction with 1M KCl (Thomas, 1982). Soil pH was determined in a soil: solution ratio of 10 g: 25 mL using both 1M KCl solution and distilled water (Thomas, 1996). The pH of the supernatant was read, after the samples had stood for an hour, using a standard glass electrode (MetrohmHersiau E396B). Effective cation exchange capacity (ECEC) was determined and expressed in $\text{cmol}^+ \text{kg}^{-1}$:

$$\text{ECEC} = \Sigma\ (\text{Ex. Acid} + Ca^{2+} + Mg^{2+} + K^+ + Na^+)$$

3.5 Statistical analysis

Statistical analysis was carried out on data sets using Genstat 14th Edition (Payne *et al.*, 2011). Analysis of variance (ANOVA), using a split-split plot design was used to test for treatment effects. The main plots were either burned or mulched, with the split-plot treatment consisting of stump-line or inter-row, and the split-split plot treatments consisting of the soil depth. The average value of the six samples collected for any given sample position or depth in a plot was used to represent the mean of that plot. The mean of the four replicates (n=4) and standard error of the mean for each treatment, sample position and sample depth are reported. The results were regarded as significant if a p value of less than 0.05 (or 5%) was obtained. To determine potential relationships between soil properties, a correlation matrix was produced. Principal component analysis (PCA) was conducted to highlight the variables that were most affecting aggregate stability and to assess the similarity between the mulched and burned treatments (Canoco Version 4.56; terBraak and Smilauer, 2009).

Chapter 4: Effect of harvest residue burning and mulching on selected soil physico-chemical properties

4.1 Introduction

Forest harvest residue management practices can have a large impact on the productivity and long-term sustainability of short-rotation plantation forests (Powers *et al*., 1990; Mendham *et al*., 2002; Deleporte *et al*., 2008); this is primarily due to their impact on the quality and quantity of soil organic carbon (SOC) left on a site. Decreasing SOC resulting from forest harvest residue management has been identified as a substantial threat to the long-term site productivity of forest plantations (Powers *et al*., 1990; Kelting *et al*., 1999).

Soil organic carbon is a critically important component of soil quality because it affects soil physical, chemical, and biological properties (Gregorich *et al*., 1994; Carter, 1996; Certini, 2005) such as nutrient mineralization potential, cation exchange capacity (CEC), aggregate stability, and water retention (Mulumba and Lal, 2008; Abiven *et al*., 2009). Harvest residue management practices such as residue retention on site (broadcasting, windrowing, mulching) generally increase SOC content; this enhances soil moisture and CEC, improves soil tilth, and increases soil nutrient availability (Norris, 1993). On the other hand, residue removal or burning tends to lower SOC and soil nutrients are lost through leaching and volatilisation (Morris, 1986; Norris, 1993; du Toit and Scholes, 2002). The unprotected soil surface on burnt areas may accelerate losses associated with soil erosion. Burning of harvest residues includes combustion of the litter layer and the surface SOC (Dovey, 2012). The objective of this section of the study was to characterise the soil physico-chemical properties and determine the effect of residue management on these. Soil properties assessed included:

1. Particle size distribution; and
2. Chemical properties (SOC, pH, exchangeable acidity, exchangeable bases and effective cation exchange capacity).

4.2 Materials and methods

Chapter 3 presented the site description and materials and methods used in the study. Sections 3.4.2 and 3.4.3 describe the physical and chemical analyses, respectively, that are reported in this chapter.

4.3 Results and discussion

Table 4.1 presents a summary of soil properties for treatments, depths, and positions. Generally the exchangeable base cations (Ca^{2+}, Mg^{2+}, Na^{+} and K^{+}), pH and ECEC values were higher than values reported by Titshall (2011) for other granite-derived soils and within the ranges reported by Donkin (1993) for South African forestry soils. The site values were also higher than those reported by Rietz (2013) for samples taken six months after treatment implementation. The pH (KCl) is acidic and ranged between 3.8 and 4.8.

4.3.1 Soil particle size distribution

There was no significant interaction effect of residue by sampling position by depth on the clay ($p = 0.804$), silt ($p = 0.816$), and sand ($p = 0.99$) and no marked difference in clay content between the mulched and burned plot treatments (Appendix 4.1a, b and c). However, a significant effect was found on soil depth ($p = 0.008$) and sampling position ($p = 0.022$) with clay (Appendix 4.1a). The clay content was higher in the 0.1 to 0.2 m depth than in the 0 to 0.1 m depth. In the inter-row, clay content was higher when compared to the stump-line. There was a significant ($p = 0.007$) effect between the interaction of soil depth and sampling position on silt (Appendix 4.1b). There was no significant effect found with sand on residue management ($p = 0.713$), soil depth ($p = 0.552$) and sampling position ($p = 0.142$) (Appendix 4.1c). The soils from plots 8, 11, 23, and 24 (upper-slope) had a relatively high amount of clay when compared to plots 1, 2, 20, and 21 (lower-slope) (Table 4.2). However, as texture of the soil cannot be easily altered by forest harvest residue management practices (Titshall *et al.*, 2013) this is presumably a function of natural site variation.

Table 4.1: Mean soil chemical properties under the burned and mulched treatments (n=4±SE) at 0 to 0.1 m and 0.1 to 0.2 m; IR: Inter-row; SL: Stump-line; B: Burn; M: Mulch. Ca, Mg, Na and K refer to exchangeable Ca^{2+}, Mg^{2+}, Na^{+} and K^{+}

Residue treatment	Position	Depth	pH (H_2O)	pH (KCl)	Ca^{2+}	Mg^{2+}	Na^{+}	K^{+}	EA	ECEC
	(m)				---$cmol_c\ kg^{-1}$					--
B	IR	0-0.1	4.75(±0.12)	4.32(±0.06)	1.61 (±0.40)	0.50 (±0.04)	0.28 (±0.04)	0.18 (±0.05)	1.07(±0.19)	3.64(±0.58)
B	IR	0.1-0.2	4.42(±0.12)	4.07(±0.19)	0.84 (±0.16)	0.26 (±0.02)	0.27 (±0.05)	0.16 (±0.04)	1.66(±0.19)	3.19(±0.53)
B	SL	0-0.1	5.12(±0.12)	4.65(±0.45)	1.90 (±0.27)	0.73 (±0.11)	0.28 (±0.05)	0.21 (±0.04)	0.46 (0.43)	3.58(±0.59)
B	SL	0.1-0.2	4.52(±0.09)	4.15(±0.00)	1.26 (±0.24)	0.40 (±0.06)	0.26 (±0.04)	0.19 (±0.05)	1.41(±0.16)	3.52(±0.56)
M	IR	0-0.1	4.65(±0.23)	3.85(±0.40)	0.92 (±0.26)	0.38 (±0.06)	0.30 (±0.11)	0.14 (±0.02)	1.49(±0.22)	3.23(±0.52)
M	IR	0.1-0.2	4.47(±0.18)	3.95(±0.21)	0.47 (±0.13)	0.22 (±0.03)	0.28 (±0.11)	0.11 (±0.01)	1.45(±0.21)	2.53(±0.43)
M	SL	0-0.1	4.72(±0.18)	3.87(±0.40)	1.76 (±0.45)	0.38 (±0.08)	0.30 (±0.11)	0.14 (±0.01)	1.05(±0.10)	3.63(±0.59)
M	SL	0.1-0.2	4.60(±0.20)	3.85(±0.25)	0.64 (±0.22)	0.27 (±0.04)	0.32 (±0.10)	0.11 (±0.01)	1.25(±0.23)	2.59(±0.42)

EA: Exchangeable acidity.

ECEC: Effective cation exchange capacity (Sum of Ex. Acid, Ca^{2+}, Mg^{2+}, K^{+} and Na^{+}).

Standard errors (SE) are in brackets.

Table 4.2: Particle size distribution on selected plots which represent the (a) mulched and (b) burned residue treatments at 0-0.1 and 0.1-0.2 m depths and different sampling positions IR: Inter-row; SL: Stump-line; B: Burn; M: Mulch. Plots 8, 11, 23 and 24 are on up-slope; plots 1, 2, 20 and 21 are on down-slope.

(a)

Plot	Treatment	Depth (m)	Position	Coarse sand (%)	Medium sand (%)	Fine sand (%)	Coarse silt (%)	Fine silt (%)	Clay (%)
1	M	0-0.1	IR	28.5	22.5	9.0	8.0	27.0	5.0
1	M	0.1-0.2	IR	15.5	18.0	9.5	17.5	29.2	10.4
1	M	0-0.1	SL	21.2	18.1	11.8	9.9	31.9	7.0
1	M	0.1-0.2	SL	22.2	17.7	8.3	9.6	32.2	10.0
8	M	0-0.1	IR	14.0	11.5	9.7	12.0	25.3	27.5
8	M	0.1-0.2	IR	15.5	11.6	9.1	10.4	25.8	27.5
8	M	0-0.1	SL	18.5	13.9	11.6	8.5	32.5	15.0
8	M	0.1-0.2	SL	16.6	13.9	8.1	10.0	26.3	25.0
20	M	0-0.1	IR	16.6	12.1	9.6	7.4	31.7	22.5
20	M	0.1-0.2	IR	14.6	11.6	11.6	12.5	27.1	22.5
20	M	0-0.1	SL	16.3	12.9	9.1	10.0	26.7	25.0
20	M	0.1-0.2	SL	15.7	16.5	7.9	12.2	25.3	22.5
21	M	0-0.1	IR	14.4	16.8	7.9	8.5	27.5	25.0
21	M	0.1-0.2	IR	16.5	15.5	9.0	5.9	28.2	25.0
21	M	0-0.1	SL	18.8	16.8	6.9	5.7	29.3	22.5
21	M	0.1-0.2	SL	16.6	17.9	6.3	5.5	23.8	30.0
Average				17.6	15.4	9.1	9.6	28.1	20.2
Total					**42.1**		**37.7**		**20.2**

Table 4.2 (continued): Particle size distribution on selected plots which represent the (a) mulched and (b) burned residue treatments at 0-0.1 and 0.1-0.2 m depths and different sampling positions IR: Inter-row; SL: Stump-line; B: Burn; M: Mulch. Plots 8, 11, 23 and 24 are on up-slope; plots 1, 2, 20 and 21 are on down-slope.

(b)

Plot	**Treatment**	**Depth (m)**	**Position**	**Coarse sand (%)**	**Medium sand (%)**	**Fine sand (%)**	**Coarse silt (%)**	**Fine silt (%)**	**Clay (%)**
2	B	0-0.1	IR	16.5	18.9	8.8	7.7	30.6	17.5
2	B	0.1-0.2	IR	14.0	11.2	12.0	10.0	35.3	17.5
2	B	0-0.1	SL	18.5	18.8	10.6	7.4	32.2	12.5
2	B	0.1-0.2	SL	20.2	17.4	8.6	8.5	30.4	15.0
11	B	0-0.1	IR	17.7	16.2	7.8	7.2	26.2	25.0
11	B	0.1-0.2	IR	18.9	15.3	8.8	7.0	25.1	25.0
11	B	0-0.1	SL	14.8	12.8	9.0	20.5	25.5	17.5
11	B	0.1-0.2	SL	14.2	15.3	6.7	12.7	25.9	25.0
23	B	0-0.1	IR	14.2	11.6	6.1	7.5	28.0	32.5
23	B	0.1-0.2	IR	16.9	10.5	7.9	5.6	24.1	35.0
23	B	0-0.1	SL	18.5	15.4	7.8	7.5	30.7	20.0
23	B	0.1-0.2	SL	24.0	17.7	5.3	5.0	23.0	25.0
24	B	0-0.1	IR	17.2	13.8	6.5	6.0	29.1	27.5
24	B	0.1-0.2	IR	17.9	13.5	3.5	10.0	25.1	30.0
24	B	0-0.1	SL	16.4	14.4	7.3	10.0	29.5	22.5
24	B	0.1-0.2	SL	14.7	16.3	3.7	15.0	20.3	30.0
Average				17.2	14.9	7.5	9.2	27.6	23.6
Total				**39.6**			**36.8**		**23.6**

4.3.2 Exchangeable bases and cation exchange capacity

There was no significant interaction effect of residue by sampling position by depth on the concentrations of exchangeable Ca^{2+} ($p = 0.235$), Mg^{2+} ($p = 0.327$), Na^{+} ($p = 0.234$), and K^{+} ($p = 0.834$) (Appendix 4.2a, b, c and d). However, residue burning significantly ($p = 0.043$) affected soil Ca^{2+}, and it was higher when compared to residue retention by 0.45cmol$_c$ kg^{-1}. The exchangeable Ca^{2+} and Mg^{2+} concentrations were significantly ($p < 0.001$) higher in the 0 to 0.1 m depth in both treatments than in the 0.1 to 0.2 m depth (Appendix 4.2a and b). This may be due to the mineralisation of nutrients held in the residue biomass during burning as observed by Mendham *et al*. (2003) and Deleporte *et al*. (2008) and the consequent addition of ash to the soil surface (O'Connell *et al*., 2004). These exchangeable cations were not significantly modified by harvest residue management at the lower soil depth.

There was also a significant effect on soil Mg in the sampling position ($p = 0.044$) and residue by depth interaction ($p = 0.027$). Soil Mg was higher in the stump-line when compared to the inter-row by 0.10 cmol$_c$ kg^{-1}. The highest concentrations of exchangeable Mg^{2+} were also found in the burned plot treatment, though this was not significantly (p = 0.106) different from the residue retention plot treatment (Table 4.3). The lack of significant response of soil Mg^{2+} to residue management may be due to plant uptake of this nutrient.

Residue treatment, soil depth, and sampling position had no significant effect on soil K^{+} (Appendix 4.2d). It is possible this nutrient was leached or taken up by plants since K^{+} is relatively mobile in soil. Deleporte *et al*. (2008), who investigated the effects of slash and litter management practices on soil chemical properties and growth of second rotation eucalypts on sandy soils in the Congo, found that residue management had no significant ($p > 0.05$) effect on soil K in the 0 to 0.1 m depth. Similarly, du Toit *et al*. (2008) reported that residue management had no significant effect on soil K^{+} in the 0 to 0.1 m depth under a *Eucalyptus grandis* plantation on a clay soil in South Africa. Exchangeable K^{+} in the 0 to 0.1 m depth from the study showed similar results where residue treatment had no significant effect. Leaching, bio-geochemical cycling, and nutrient uptake by growing trees can alter the amount of nutrients present. Thus, the time of sampling can affect the amount and availability of nutrients. However, Rietz (2013) also reported that residue management had no significant effect on soil K^{+} and Mg^{2+} between 0 and 0.3 m depth six months after treatment implementation at the same site as that used in the present study. The soil at the study site also had higher exchangeable Ca^{2+}, Mg^{2+}, and K^{+} values when compared to other soils which are derived from granite in the summer rainfall region of South Africa on which eucalypt is grown (Titshall, 2011).

Table 4.3: Chemical and physical soil properties under the mulched and burned treatment in the 0 to 0.1 m and 0.1 to 0.2 m depth and different sampling position IR: Inter-row; SL: Stump-line; B: Burn; M: Mulch

	SOC	**Ca^{2+}**	**Mg^{2+}**	**Na^{+}**	**K^{+}**	**EA**	**ECEC**	**MWD**	**Soil pH**	**Soil pH**
	%	**---cmol$_c$kg^{-1}--------------------------------**						**mm**	**H$_2$0**	**KCl**
Overall mean (n=192)	2.03 (±0.11)	1.18 (±0.08)	0.39 (±0.02)	0.29 (±0.01)	0.16 (±0.01)	1.23 (±0.04)	3.25 (±0.09)	2.45 (±0.04)	4.65 (±0.02)	4.08 (±0.03)
Main effect means										
M (n=96)	2.50 (±0.17)	0.95 (±0.10)	0.95 (±0.09)	0.30 (±0.02)	0.12 (±0.12)	1.31 (±0.06)	3.00 (±0.11)	2.40 (±0.07)	4.60 (±0.02)	3.87 (±0.02)
B (n=96)	1.57 (±0.10)	1.40 (±0.11)	1.40 (±0.11)	1.40 (±0.11)	0.19 (±0.11)	1.15 (±0.06)	3.49 (±0.14)	2.49 (±0.06)	4.71 (±0.04)	4.30 (±0.05)
IR (n=96)	1.95 (±0.15)	0.96 (±0.90)	0.34 (±0.02)	0.34 (±0.02)	0.15 (±0.01)	1.40 (±0.06)	3.16 (±0.11)	2.38 (±0.06)	4.57 (±0.03)	4.05 (±0.04)
SL (n=96)	2.10 (±0.15)	1.39 (±0.12)	0.44 (±0.03)	0.44 (±0.03)	0.16 (±0.01)	1.04 (±0.06)	3.33 (±0.15)	2.51 (±0.06)	4.73 (±0.04)	4.12 (±0.05)
Two-way residue by depth effect means										
0 - 0.1 m (M) (n=48)	2.75 (±0.25)	1.34 (±0.15)	0.38 (±0.03)	0.38 (±0.03)	0.14 (±0.01)	1.32 (±0.09)	3.47 (±0.16)	2.78 (±0.07)	4.60 (±0.03)	3.87 (±0.03)
0.1 - 0.2 m (M) (n=48)	2.23 (±0.24)	0.56 (±0.09)	0.24 (±0.02)	0.24 (±0.02)	0.11 (±0.01)	1.31 (±0.09)	2.54 (±0.12)	2.03 (±0.09)	4.60 (±0.03)	3.87 (±0.03)
0 - 0.1 m (B) (n=48)	1.71 (±0.15)	1.76 (±0.15)	0.62 (±0.04)	0.62 (±0.04)	0.19 (±0.02)	1.15 (±0.08)	4.00 (±0.20)	2.71 (±0.07)	4.71 (±0.06)	4.30 (±0.07)
0.1 - 0.2 m (B) (n=48)	1.44 (±0.15)	1.05 (±0.16)	0.33 (±0.02)	0.33 (±0.02)	0.17 (±0.01)	1.15 (±0.08)	2.98 (±0.18)	2.27 (±0.08)	4.72 (±0.06)	4.28 (±0.06)

There was no significant interaction effect of residue by sampling position by depth on ECEC (Appendix 4.3). However, a significant ($p = 0.003$) effect on ECEC was found with soil depth. The average ECEC of the soils under the burned plot treatment was higher (4.00 (±0.20) and 2.98 (±0.18) $cmol_c\ kg^{-1}$ in the 0 to 0.1 m and 0.1 to 0.2 m, respectively) than the soils under the mulched plot treatment (3.47 (±0.10) and 2.54 (±0.12) $cmol_c\ kg^{-1}$ in the 0 to 0.1 m and 0.1 to 0.2 m, respectively) (Table 4.3). This might be due to residue burning, with this effect resulting from changes in exchangeable base cations and exchangeable acidity (du Toit *et al*., 2008). It could be also due to higher base cations (Ca^{2+} and Mg^{2+}) in soils under the burned plot treatment. Exchangeable acidity was responsible for the high levels of ECEC in soils. Soils had higher values of ECEC than values reported by Titshall (2011) for other granite-derived soils in the region.

4.3.3 Exchangeable acidity, soil pH, and acid saturation

No significant effect of residue management or the interaction between residue management, sampling position, and depth on the exchangeable acidity (EA) was found (Appendix 4.4a). The mean exchangeable acidity values in soils under burn and mulch treatments ranged between 0.46 and 1.66 $cmol_c\ kg^{-1}$. The values from the site were lower than values reported by Titshall (2011) for other granite-derived soils in the region. Also the levels of exchangeable acidity from the study were lower when compared to levels reported by du Toit *et al*. (2008).

No significant effect of residue management or the interaction between residue management, depth, and sampling position on soil pH (H_2O and KCl) was found (Appendix 4.4b and c). This might be a result of soil pH levels having returned to their initial levels by the time the soil samples were taken, two years after implementation. A number of studies have investigated the effect of residue management on soil pH under eucalypt plantations and have found an increase in pH of one or more units a few months after burning (Tiarks and Ranger, 2008; Rietz, 2010; Dovey, 2012). Where increases in pH after burning are found, it is generally considered to be due to the combustion of organic matter, ash production, and denaturing organic acids (Certini, 2005), which provide large quantities of basic ions, resulting in increased pH (Ballard, 2000; Certini, 2005); and also due to increased decomposition of residues (du Toit *et al*., 2008). For this study, residue management had no significant effect at all. The pH values were within values reported by Donkin (1993) for South African soils.

There was no significant interaction effect of residue by sampling position by depth on acid saturation (Appendix 4.5). However, a significant ($p < 0.001$ and $p = 0.031$, respectively) effect on depth and sampling position, was found (Appendix 4.5). The 0.1 to 0.2 m depth had a higher acid saturation when compared to 0 to 0.1 m depth under both residue treatments. This inevitably results in a decrease in exchangeable basic cations. Sampling position in the IR had higher acid saturation than in the SL under both residue treatments, for example, BIR = 29 % and BSL = 13 % (Table 4.4). Acid saturation values were lower than values reported by Titshall (2011) for other granite-derived soils in the region.

Table 4.4: Acid saturation under the burned and mulched treatments (n = 4) at 0 to 0.1 m and 0.1 to 0.2 m; IR: Inter-row; SL: Stump-line; B: Burn; M: Mulch.

Residue treatment	Depth	Position	Acid saturation (%)
B	0-0.1	IR	29
B	0.1-0.2	IR	52
B	0-0.1	SL	13
B	0.1-0.2	SL	40
M	0-0.1	IR	46
M	0.1-0.2	IR	57
M	0-0.1	SL	29
M	0.1-0.2	SL	48

4.3.4 Soil organic carbon

There was no significant (p = 0.497) interaction effect of residue by depth by sampling position on the amount of SOC (Appendix 4.6). However, residue treatments had a significant effect (p = 0.025) on SOC (Appendix 4.6). It was higher in the mulched plot treatment (2.5 %), than in the burned plot treatment (1.6 %). This is associated with the levels of added harvest residues. There was also a significant effect (p = 0.039) of depth on SOC (Appendix 4.6). In the 0 to 0.1 m soil depth, SOC was higher (2.2 %) than in the 0.1 to 0.2 m soil depth (1.8 %). The results also show that the effect of the retention of harvest residue on SOC was higher at both 0 to 0.1 and 0.1 to 0.2 m depths when compared to burning of residues (Figure 4.1). The burning of residues might have resulted in loss of SOC (Mendham *et al*., 2003) at these depths. Although not significant, SOC in the stump-line under the mulched plot treatment was higher (2.8 (±0.51) and 2.4 (±0.25) % in the 0 to 0.1 m and 0.1 to 0.2 m, respectively) than under the burned plot treatment (1.8 (±0.37) and 1.4 (±0.39) % in the 0 to 0.1 m and 0.1 to 0.2 m, respectively) (Figure 4.1). Sampling position had no significant effect (p = 0.207) on SOC (Appendix 4.6).

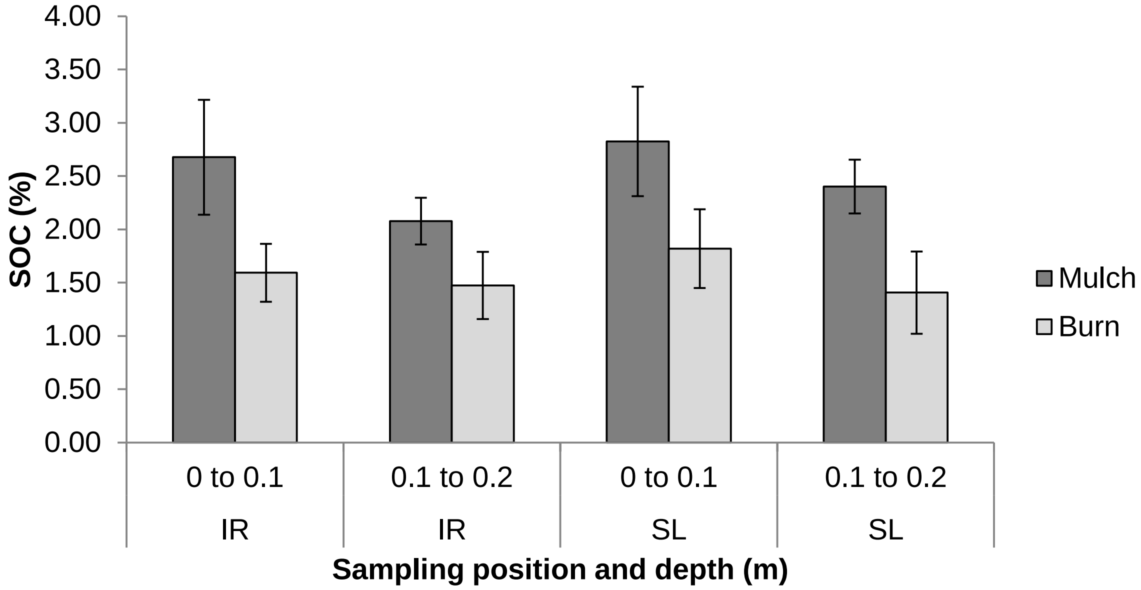

Figure 4.1: The soil organic carbon content (SOC, %) under the burned and mulched plot treatment at 0 to 0.1 m and 0.1 to 0.2 m depths (n=4 ± SE) IR: inter-row, SL: stump-line.

The trend of an increase in SOC at both sampled depths under the mulched plot treatment is probably caused by decomposition of the residues and humus layer with the addition of new substrate and the increase in temperature following clear felling (du Toit *et al*., 2008). The differences are due not only to the added carbon, from the mulching, but also due to the loss of SOC from burning (Mendham *et al*., 2003). The overall increase in SOC (0 to 0.1 m and 0.1 to 0.2 m) measured in soils at the site under the mulched plot treatment is most likely a result of the input of organic matter in the form of harvest residues and tree roots (Rietz, 2010).

4.4 Conclusions

The soil is acidic, with low levels of exchangeable base cations and effective cation exchange capacity. Levels of exchangeable K^+, Ca^{2+}, Mg^{2+}, and Na^+ are high when compared to granite-derived soils (ICFR, 1998; Titshall, 2011), while pH, exchangeable acidity and ECEC are relatively similar.

There was no overall significant interaction effect of residue management by sampling depth by sampling position on soil texture, exchangeable base cations, pH, SOC, ECEC, EA, and acid saturation. However, residue management significantly affected exchangeable Ca^{2+} and SOC. Clay content, exchangeable Ca^{2+} and Mg^{2+}, ECEC, acid saturation and SOC were significantly affected by sampling depth. Also, clay content, exchangeable Mg^{2+} and acid saturation were significantly affected by sampling position. Neither residue management nor the interaction between residue management and sampling position had any significant effect on soil pH at the site.

Residue retention resulted in significantly higher SOC under the mulched plot treatment when compared to burned plot treatment. This is probably due to additional harvest residues from the previous rotations. In the 0 to 0.1 m soil depth, SOC was significantly higher than in the 0.1 to 0.2 m soil depth.

Burning residues resulted in higher exchangeable Ca^{2+} under the burned plot treatment when compared to the mulched plot treatment. The exchangeable Ca^{2+} and Mg^{2+} were significantly higher in the 0 to 0.1 m depth in both treatments than in the 0.1 to 0.2 m depth. This is most likely due to the mineralisation of nutrients during burning. The lack of significant residue management effects on Mg^{2+}, K^{+}, and Na^{+} is most likely a result of either plant uptake or leaching of any additional nutrients supplied by residues or loss of nutrients during the burning of residues.

Residue retention management did affect SOC after two years implementation. Residue retention increased SOC when compared to residue burning. Soil organic carbon assists in the ability of soil to improve aggregation and losses will increase the susceptibility of the site to degradation. Therefore, retaining harvest residues on the site would increase SOC, which has implications for nutrients, and water and SOC plays a role in soil structure by promoting aggregation. Fire is known as a residue management tool but intensive fires are unsustainable. Burning of residues, if continued will result in loss of SOC and base cations. This may affect site productivity, aggregation and these effects can be reduced by retention of residues. Therefore, burning of residues on a site may not be a valid management tool and residue retention is necessary to maintain SOC and cations for productivity.

Chapter 5: The effect of residue management on soil aggregate stability and the relationship with selected soil physico-chemical properties

5.1 Introduction

Soil structure is central to soil functioning as it controls water, air and nutrient fluxes and storage, and therefore influences the activity and growth of living organisms (Bronick and Lal, 2005). Soil structure is characterised by the shape and stability of aggregates and aggregate stability refers to soil structure resilience to external mechanical forces (Mulumba and Lal, 2008; Mataix-Solera *et al*., 2011). The aggregate stability of soils is influenced mainly by soil organic carbon (SOC), pH, texture, and clay mineralogy (Amezketa, 1999; Bronick and Lal, 2005; Abiven *et al*., 2009). The interaction of clay, SOC and aggregates is affected by soil pH, CEC and cations, all of which are related to the amount and type of clay present in the soil (Amezketa, 1999). Quantification of these properties is necessary to better understand the effects of harvest residue management on aggregate stability. Generally, soils that have higher aggregate stability are less prone to structural degradation and erosion (Bronick and Lal, 2005). Soil aggregate stability represents an important characteristic of soil structure, which is closely connected to soil erodibility, soil water regime, soil biota and soil nutrient availability (Mataix-Solera *et al*., 2011). The mean weight diameter (MWD) of the soil is related to aggregate stability and higher MWD corresponds to higher aggregate stability (Le Bissonnais, 1996).

A decline in soil structure is commonly related to land use and management practices (Bronick and Lal, 2005) that involve the use of machinery which decreases soil porosity and results in soil compaction (Smith, 2003; Smith and du Toit, 2005; Rietz, 2010). However, many studies have also investigated the effect of fire on soil aggregate stability under laboratory and field conditions (Andreu *et al.* 2001; Badia and Marti, 2003; Mataix-Solera and Doerr, 2004; Fox *et al*., 2007; Arcenegui *et al*., 2008; Zavala *et al*., 2010), although results have sometimes been found to be contradictory. Arcenegui *et al*. (2008), for example, studied the effect of wildfires on water repellency and aggregate stability in calcareous soils in Spain. They observed a significant ($p < 0.001$) increase in soil aggregate stability under the burned treatments when compared to unburned treatments. In contrast, Badia and Marti (2003) studied the effect of plant ash and heat intensity on the chemical and physical soil properties of two contrasting soils in Spain, observing a decrease in soil aggregate stability after burning.

The objective of this section of the study was to:

- Evaluate the effect of residue management on soil structural stability;
- Determine if any of the other soil properties measured were related to changes in structural stability.

5.2 Materials and methods

Chapter 3 gave the site description and materials and methods used in the study. Section 3.4.1 describes the soil structural stability (MWD) method used.

5.3 Results and discussion

5.3.1 Aggregate stability

No significant difference was found in the interaction between residue treatment, sampling position and soil depth (Appendix 5.1). There was no significant effect (p = 0.439) of residue management on MWD. A possible explanation for this is that under the burned plot treatment, high MWD values might have been due to the formation of a hydrophobic film on the external surface of aggregates (Mataix-Solera and Doerr, 2004). The hydrophobic film is a result of rapid volatilisation of hydrophobic compounds from the litter layer during a wildfire (Mataix-Solera and Doerr, 2004). This causes hydrophobic substances to become strongly cemented, reducing water entry into the aggregates thereby enhancing their stability in water (Piccolo and Mbagwu, 1999). Under the mulched plot treatment, the higher aggregate stability was due to residues left on the surface which increased SOC, thus promoting aggregate stability. However, a significant effect of residue management on MWD was found with soil depth ($p < 0.001$) and sampling position (p = 0.042). The residue by soil depth interaction had also a significant (p = 0.021) effect on MWD (Appendix 5.1). The MWD in the stump-line under the mulched and burned plot treatment was much higher in the 0 to 0.1 m depth mm than in the 0.1 to 0.2 m depth 2.9 (±0.04) mm compared to 2 (±0.10) mm, respectively. Results for the inter-row showed the same trend (Figure 5.1). This might be due to a contribution of the decomposing root systems of the previous rotation.

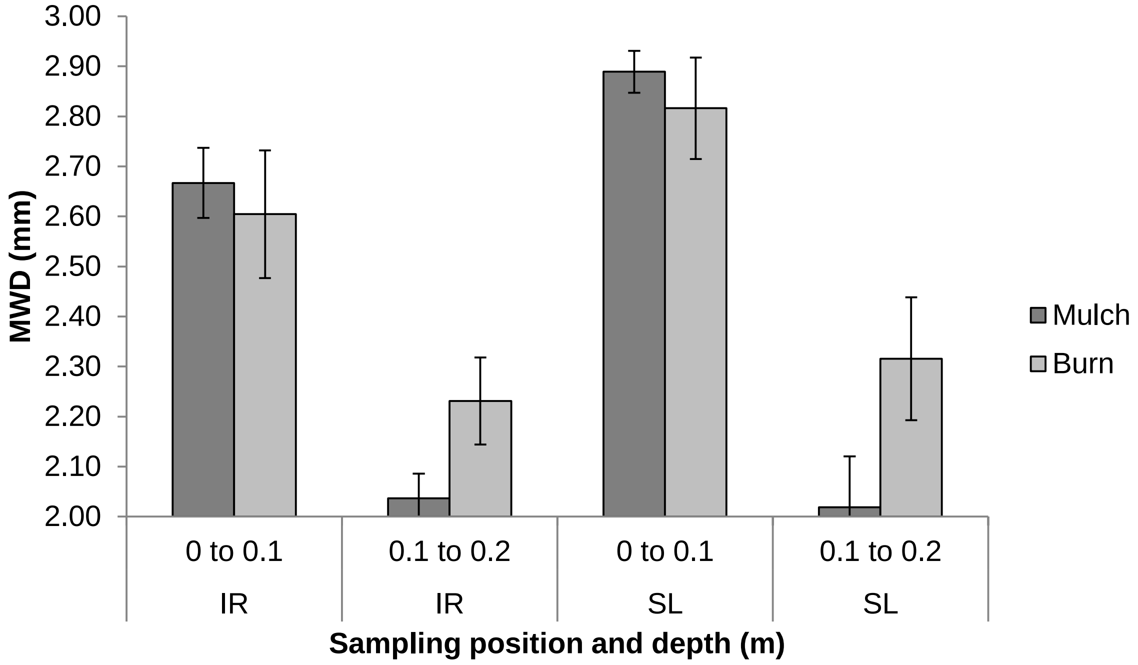

Figure 5.1: The mean weight diameter (MWD) at 0 to 0.1 m and 0.1 to 0.2 m depths under the burned and mulched treatments (n=4 ±SE). IR: inter-row; SL: stump-line.

Mean weight diameter is related to soil aggregate stability with higher MWD values corresponding to higher soil aggregate stability (Le Bissonnais, 1996). The average MWD of 2.45 mm (SE±0.04) and values between 0.8 to 3.5 mm shows that the aggregates are stable. According to Le Bissonnais (1996), soil aggregates from the study site have great stability considering their high mean value above 2 mm. Such good structure favours gas and water transfer in soils and crop rooting, while reducing susceptibility to erosion (Tayel *et al*., 2010).

5.3.2 Relationship between soil aggregate stability and other soil properties

The principal component analysis (PCA) (Figure 5.2) indicates that SOC was responsible for the largest source of variation in the data, followed by cations and MWD. There was no correlation between the MWD and the base cations with the SOC content (vectors at right angles) (Figure 5.2). However, the MWD had a relatively stronger positive correlation with, in particular, Ca^{2+}, and, to a lesser extent, with K^+ and Mg^{2+} than with SOC. The MWD was negatively correlated to Na^+. No clear patterns between the burned and mulched treatments were evident from the ordination distribution, though there was a general increase in the number of mulch samples along the direction of increase in SOC.

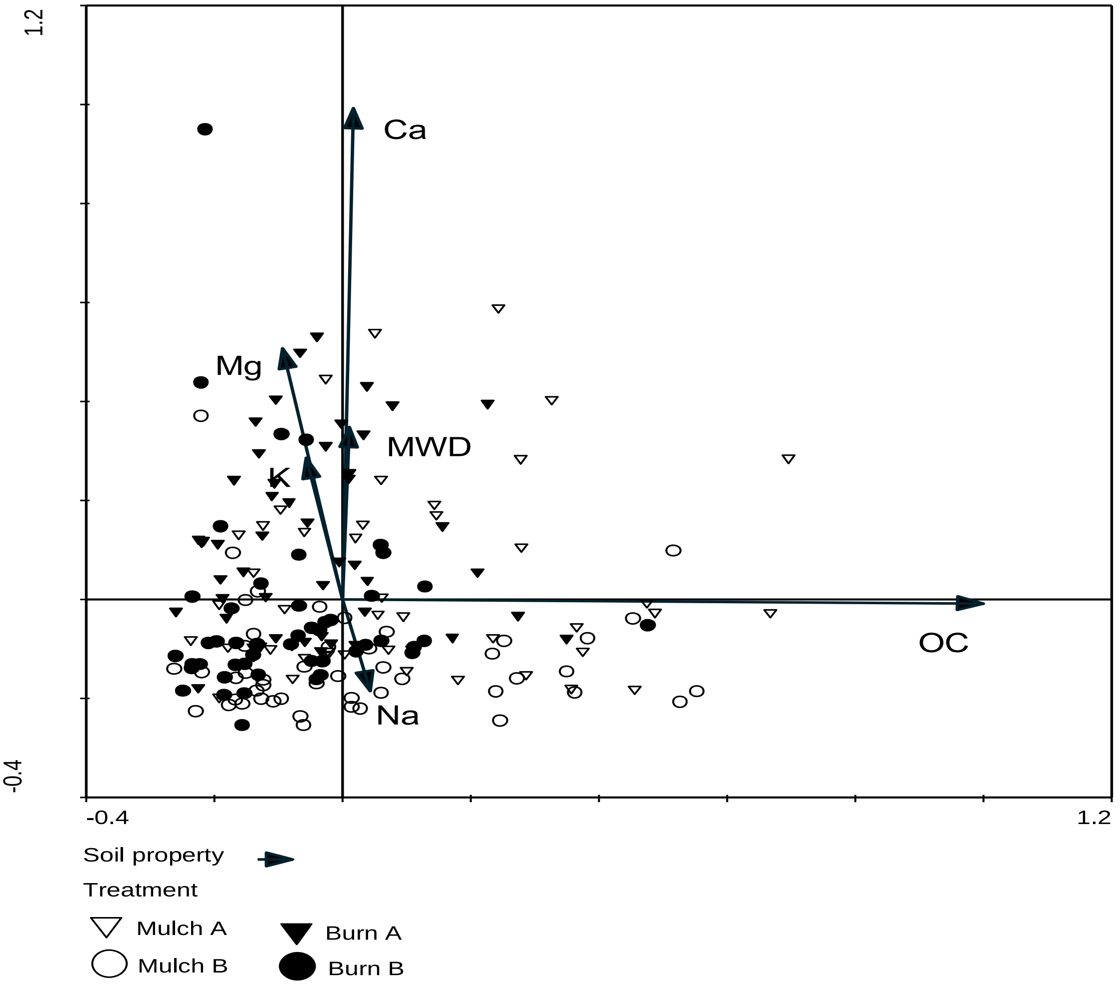

Figure 5.2: Principal component analysis (PCA) of soil properties (arrows) and residue treatment (symbols). Ca^{2+}, Mg^{2+}, K^+ and Na^+: exchangeable bases; OC: soil organic carbon; MWD: mean weight diameter. Eigen values for X-axis and Y-axis were 57.4 and 31.1%, respectively, accounting for 88.5% of the variance in the data set. A: 0 to 0.1 m depth and B: 0.1 to 0.2 m depth.The arrows indicate the direction of increasing amount of the parameter and the length the relative positive amount of a parameter. The angle between the arrows indicates the relative strength of correlation between the parameters, where an acute angle indicates high correlation; a right angle indicates independence or no correlation and an obtuse angle, a negative correlation. The perpendicular position of a symbol to a parameter arrow indicates the relative amount of the parameter for that symbol (Leps and Smilauer, 2003).

Results of correlation analysis between aggregate stability (MWD) and some selected soil properties are given in Table 5.1. The SOC which generally plays a major role in soil aggregation (Tisdall and Oades, 1982), surprisingly showed a poor positive correlation. Possibly, the type of organic material, for example, mycorrhizal hyphae and polysaccharides, was more important than the total amount in stabilising the aggregates (Tisdall and Oades, 1982). The correlation between the MWD and bases (Ca^{2+} (r = 0.236, $p < 0.05$), Mg^{2+} (r =0.208, $p < 0.05$)) was higher when compared to the correlation between the MWD and SOC. The positive and significant correlation between exchangeable Ca^{2+} and MWD is possibly due to the ability of the basic cation to promote flocculation of soil colloids. Calcium and magnesium improve aggregation, and therefore, soil structure (Bronick and Lal, 2005). The result showing no correlation between MWD and SOC is in contrast to other researchers' findings (Tisdall and Oades, 1982; Amezketa, 1999; Abiven *et al*., 2009). For example, Abiven *et al*. (2009), who investigated the effects of organic inputs over time on soil aggregate stability, reported that soil aggregate stability was positively correlated with SOC.

Aggregation is mainly maintained by the presence of SOC, thus changes in SOC can lead to changes in aggregation (Mulumba and Lal, 2008). Also there was a weak positive correlation (r = 0.211) but significant ($p < 0.05$) between MWD and ECEC. Although there was no strong correlation between MWD and ECEC, generally ECEC may play a role in the improvement of MWD. This is stimulated by the interaction of polycationic bridging in which the repulsive forces between negatively charged clay and/or SOC are reduced (Tayel *et al*., 2010). Soils under the burned and mulched treatment had higher values of ECEC in the 0 to 0.1 m depth than 0.1 to 0.2 m depth; this might explain the high values of MWD at this depth as ECEC is usually related to soil aggregation (Bronick and Lal, 2005; Tayel *et al*., 2010). There were negative, not significant correlations between MWD and soil pH (KCl), exchangeable acidity and acid saturation (Table 5.1). This suggests that these variables are not principal factors influencing MWD.

It is generally accepted that SOC improves the biological, chemical and physical properties of soils attributable, in part, to its bonding effect (Gregorich *et al*., 1994; Carter, 1996; Certini, 2005). In the present study, MWD data showed no correlation with SOC (Figure 5.3), although soils under the mulch treatment had significantly ($p < 0.05$) higher SOC than those under the burn treatment (Figure 4.1, Appendix 4.6). There was no correlation even when the data were separated between burn and mulch treatments (Appendix 5.2a and b).

Table 5.1: Correlation matrix between the mean weight diameter (MWD) and other selected properties

	SOC	*Soil pH (H_2O)*	*Soil pH (KCl)*	*Ca^{2+}*	*Mg^{2+}*	*Na^+*	*K^+*	*EA*	*ECEC*	*AS*	*MWD*
*SOC	1										
Soil pH (H_2O)	-0.139	1									
Soil pH (KCl)	-0.100	-0.037	1								
Ca^{2+}	0.009	0.167	0.163	1							
Mg^{2+}	-0.094	0.121	0.376*	0.477*	1						
Na^+	0.044	-0.176	0.045	-0.183	-0.095	1					
K^+	-0.058	0.212*	0.232*	0.272*	0.399	0.045	1				
EA	-0.045	-0.465	-0.201	-0.155	-0.099	0.128	-0.028	1			
ECEC	-0.031	-0.061	0.136	0.849*	0.563*	0.005	0.368*	0.335*	1		
AS	-0.048	-0.421	-0.331	-0.645	-0.497	0.052	-0.319	0.748*	-0.297	1	
MWD	**0.007**	**0.123**	**-0.005**	**0.236***	**0.208***	**-0.042**	**0.120**	**-0.063**	**0.211***	**-0.228**	1

* $P < 0.05$

SOC: Soil organic carbon

EA: Exchangeable acidity

ECEC: Effective cation exchange capacity

AS: Acid saturation

MWD: Mean weight diameter

Ca^{2+}, Mg^{2+}, Na^+ and K^+: exchangeable base cations

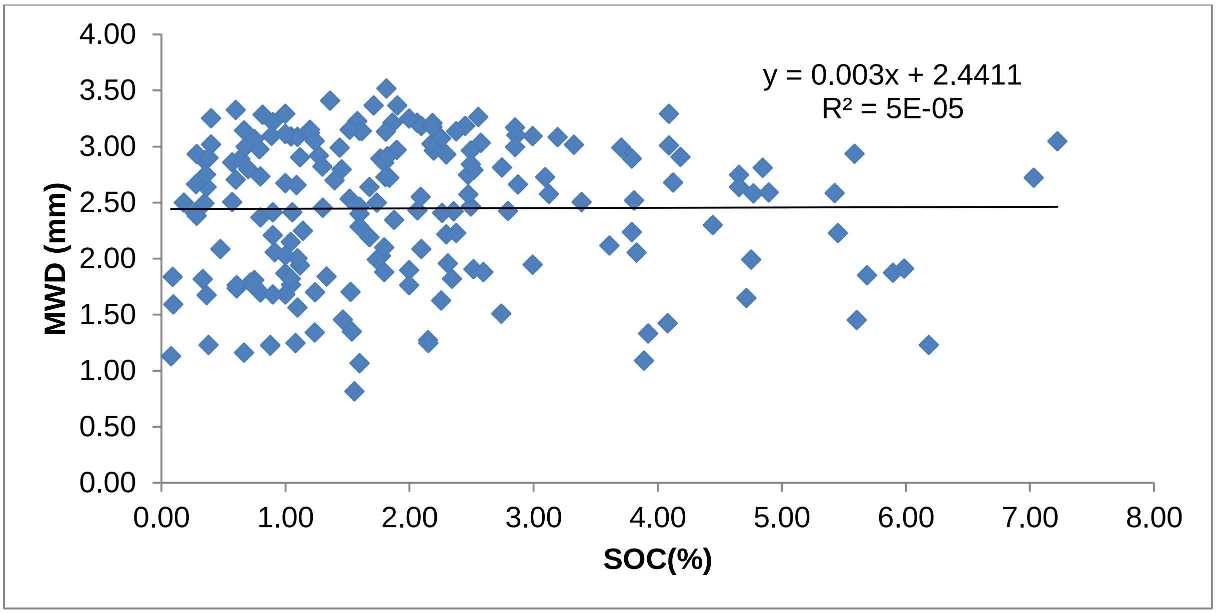

Figure 5.3: The relationship between mean weight diameter (MWD, mm) and soil organic carbon (SOC, %), n=192.

However, the stability of the soil aggregates is a product of the interactive effect of many different edaphic factors (Bronick and Lal, 2005). The result could mean clay content might have played a certain role in the MWD. Generally, clay acts as a cementing material, binding particles together, and influencingsoil organic matter (SOM) decomposition and turnover (Boix-Fayos *et al*., 2001; Bronick and Lal, 2005; Tayel *et al*., 2010). High clay content in soils is often associated with high soil aggregate stability (Tayel *et al*., 2010). This was not the case in this study, probably because the clay content of the soils was not high. Clay content was not included in the PCA analysis as only a few samples were analysed for particle size distribution in order to represent the study site.

5.4 Conclusions

No significant differences were found in the MWD between soils under either the burned or mulched plot treatments. However, a significant difference was found with depth, where soils under the burned and mulched plot treatment had higher MWD in the 0 to 0.1 m than in the 0.1 to 0.2 m depth. Mean weight diameter was also significantly affected by sampling position, as samples from the SL had a higher MWD than those from the IR. It was also affected by the interaction of residue management by depth. According to the method used, soils from the study site have very stable aggregates with a mean of 2.45 (±0.04) mm and this was substantiated by the presence of soil aggregates of ≥ 2.0 mm. No correlation was found between MWD and SOC in soils under both treatments although SOC can be the controlling factor of aggregate stability. Although soils under the mulched plot treatment had significantly higher SOC when compared to soils under the burned treatment, there was no strong correlation. Cations (Ca^{2+}, Mg^{2+}, and K^{+}) were, however, weakly positively correlated with MWD, suggesting that their presence might have had an influence on the aggregate stability of the soils. The cations may be a slightly stronger driver than SOC for MWD. Thus, in terms of residue management, the maintenance of adequate concentrations of the bases is of greater direct importance than the presence of organic carbon, although it is likely the interaction between these properties that drives aggregation and thus MWD.

Chapter 6: General discussions, conclusions and recommendations for future research

This study evaluated the effects of residue management on soil physico-chemical properties, and aggregate stability in soils, and also assessed the link between soil aggregate stability and edaphic factors.

There were no significant effects of the interaction between residue management treatments or depth and sampling position on particle size distribution, soil pH, soil organic carbon (SOC), exchangeable cations; effective cation exchange capacity (ECEC), exchangeable acidity (EA), and mean weight diameter (MWD) at the study site. The levels of these properties were similar across both of the residue management treatments.

Significant effects of residue management were, however, found on exchangeable Ca^{2+}. Exchangeable Ca^{2+} was higher under the burned plot treatment than under the mulched plot treatment two years after implementation. This, however, was most likely due to the mineralisation of nutrients during burning. The higher exchangeable Ca^{2+} under the burned plot treatment when compared to the mulched plot treatment was expected as this has also been observed by other researchers (du Toit *et al*., 2008; Dovey, 2012). The practice of burning of harvest residues is a widespread residue management practice in SA to reduce fuel loads and wildfire risks.

Residue management did, however, have a significant effect on SOC. Soil under the mulched plot treatment had higher SOC values than that under the burned plot treatment. This was likely due to the presence of harvest residues from the previous rotations in the soil under the mulched plot treatment. Burning, by contrast, resulted in a loss of carbon content. Burning of residues has implications for soil nutrient sustainability, and also losses of SOC may increase degradation of the soil. In the future, residue burning, if is continued, may affect soil productivity and so fire may be a questionable management tool at this site. Generally, as SOC is an important binding agent, a decrease in SOC would lead to a decrease in soil aggregate stability. The significant effect of residue management on SOC did not result in a change in any other measured soil properties. However, changes might occur after greater quantities of residues have decomposed. Residue retention thus has an important role in protecting soil by increasing carbon content and conserving moisture. Residue retention may also supply a large proportion of nutrients to the site.

No significant differences were observed in MWD between the burned and mulched plot treatments. The soils from the site have stable aggregates with an average MWD of 2.45 mm (SE±0.04). Mean weight diameter was significantly higher in the 0 to 0.1 m depth than at the 0.1 to 0.2 m depth. Under the mulched plot treatment, this was likely due to the presence of harvest residues which release carbon and thus promote aggregation. In contrast, in the case of the burned plot treatment, this could have been due to the formation of hydrophobic films. This suggests that the higher MWD observed under the mulched and burned plot treatment indicates that the soils at the site are not susceptible to erosion; soil degradation therefore will not be promoted by either treatment.

The comparison between the burned and mulched plot treatments showed that the main influences on the aggregate stability were, in fact, the exchangeable bases (Ca^{2+}, Mg^{2+} and K^{+}, positively; Na^{+}, negatively); SOC was apparently not a main driver. The lack of a direct relationship between SOC and the MWD in this study does not conform to what other researchers have found, even though the MWD was high. The lack of a direct relationship might be due to residues from previous rotations that were not decomposed sufficiently to increase the carbon content that would have promoted aggregation. From this study, it is concluded that no single factor can explain the soil structural stability unaided by other factors.

It can, nonetheless, be concluded that residue retention is an appropriate practice which can be used for residue management on these soils. This is because SOC under the mulched plot treatment was higher than the burned plot treatment. Even though there was no correlation between SOC and MWD, it was possible that SOC played a role in soil aggregate stability and promoted aggregation. Residue retention could also keep the soils less prone to soil erosion and degradation as the soil surface is covered with mulch. Retention of residues can also release nutrients which are essential for plant growth.

It will be important to extend this research to investigate soil parameters that were not measured in the present study and to carry out more soil aggregate stability tests at different sites under eucalypt plantations. Parameters that should be included in future studies are the role of soil biological activity, Al and Fe oxides, clay mineralogy, and root density. Soil biology influences soil physical and chemical properties, thus examining microbial activity and soil faunal activity would also help to understand better the effects of burning and mulching harvest residues. Aluminium and iron play a role in the stability of soil aggregates through cationic bridging and the formation of organo-metallic compounds. Therefore, it would help to understand their effect on aggregate stability on these soils. Clay mineralogy has substantial effects on soil aggregate stability and may influence the infiltration rate and soil loss under rainfall. Roots are considered as binding agents and their positive effect on soil aggregate stability is to enmesh fine particles and thereby stabilise soil aggregates. Sampling time must also be considered in future research.

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Appendices

Appendix 4.1: Analysis of variance of the effect of residue treatments (burned vs mulched), soil depth of measurement and plot position (inter-row and stump-line; IR/SL) on particle size distribution. Blocking factor: replicate (Rep); whole plots: residue treatments; sub-plots: soil depth; sub-sub plots: position (IR/SL).

(a) Clay (%)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Rep stratum	3	1109.81	369.94	14.57	
Rep × Residue stratum					
Residue	1	94.88	94.88	3.74	0.149
Residual	3	76.19	25.40	4.84	
Rep × Residue × Depth stratum					
Depth	1	80.96	80.96	15.44	**0.008**
Residue × Depth	1	0.53	0.53	0.10	0.762
Residual	6	31.46	5.24	0.45	
Rep × Residue × Depth × Position stratum					
Position	1	80.96	80.96	6.94	**0.022**
Residue × Position	1	36.34	36.34	3.11	0.103
Depth × Position	1	28.31	28.31	2.43	0.145
Residue × Depth × Position	1	0.75	0.75	0.06	0.804
Residual	12	140.09	11.67		
Total	31	1680.28			

(b) Silt (%)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Rep stratum	3	397.61	132.54	4.42	
Rep × Residue stratum					
Residue	1	49.55	49.55	1.65	0.289
Residual	3	90.06	30.02	1.34	
Rep × Residue × Depth stratum					
Depth	1	41.09	41.09	1.83	0.225
Residue × Depth	1	17.20	17.20	0.77	0.415
Residual	6	134.80	22.47	1.68	
Rep × Residue × Depth × Position stratum					
Position	1	1.64	1.64	0.12	0.733
Residue × Position	1	23.19	23.19	1.73	0.213
Depth × Position	1	138.28	138.28	10.33	**0.007**
Residue × Depth × Position	1	0.76	0.76	0.06	0.816
Residual	12	160.68	13.39		
Total	31	1054.84			

(c) Sand (%)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.[2]
Rep stratum	3	264.06	88.02	1.97	
Rep × Residue stratum					
Residue	1	7.30	7.30	0.16	0.713
Residual	3	134.07	44.69	2.65	
Rep × Residue × Depth stratum					
Depth	1	6.70	6.70	0.40	0.552
Residue × Depth	1	11.71	11.71	0.69	0.436
Residual	6	101.15	16.86	0.70	
Rep × Residue × Depth × Position stratum					
Position	1	59.57	59.57	2.46	0.142
Residue × Position	1	1.47	1.47	0.06	0.809
Depth × Position	1	41.45	41.45	1.71	0.215
Residue × Depth × Position	1	0.00	0.00	0.00	0.999
Residual	12	290.09	24.17		
Total	31	917.58			

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[2] **d.f** degrees of freedom **s.s.** sum of squares **m.s.** mean sum of squares **v.r.** variance ratio F probability

Appendix 4.2: Analysis of variance of the effect of residue treatments (burned vs mulched), soil depth of measurement and plot position (inter-row and stump-line; IR/SL) on exchangeable base cations. Blocking factor: replicate (Rep); whole plots: residue treatments; sub-plots: soil depth; sub-sub plots: position (IR/SL).

(a) Calcium

Source of variation	**d.f.**	**s.s.**	**m.s.**	**v.r.**	**F pr.**
Rep stratum	3	2.6670	0.8890	6.30	
Rep × Residue stratum					
Residue	1	1.6258	1.6258	11.52	**0.043**
Residual	3	0.4234	0.1411	0.36	
Rep × Residue × Position stratum					
Position	1	1.5062	1.5062	3.84	0.098
Residue × Position	1	0.0416	0.0416	0.11	0.756
Residual	6	2.3545	0.3924	1.91	
Rep × Residue × Position × Depth stratum					
Depth	1	4.4317	4.4317	21.61	**<0.001**
Residue × Depth	1	0.0121	0.0121	0.06	0.812
Position × Depth	1	0.1490	0.1490	0.73	0.411
Residue × Depth × Position	1	0.3201	0.3201	1.56	0.235
Residual	12	2.4610	0.2051		
Total	31	15.9926			

(b) Magnesium

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Rep stratum	3	0.1004	0.0334	0.85	
Rep × Residue stratum					
Residue	1	0.2074	0.2074	5.25	0.106
Residual	3	0.1185	0.0395	3.02	
Rep × Residue × Position stratum					
Position	1	0.0840	0.0840	6.43	**0.044**
Residue × Position	1	0.0536	0.0536	4.10	0.089
Residual	6	0.0784	0.0130	1.73	
Rep × Residue × Position × Depth stratum					
Depth	1	0.3515	0.3515	46.44	**<0.001**
Residue × Depth	1	0.0482	0.0482	6.38	**0.027**
Position × Depth	1	0.0009	0.0009	0.12	0.733
Residue × Depth × Position	1	0.0079	0.0079	1.05	0.327
Residual	12	0.0908	0.0075		
Total	31	1.1421			

(c) Sodium

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Rep stratum	3	0.3852	0.1284	2.19	
Rep × Residue stratum					
Residue	1	0.0056	0.0056	0.10	0.776
Residual	3	0.1762	0.0587	220.36	
Rep × Residue × Position stratum					
Position	1	0.0003	0.0003	1.36	0.288
Residue × Position	1	0.0014	0.0014	5.32	0.061
Residual	6	0.0015	0.0002	0.43	
Rep × Residue × Position × Depth stratum					
Depth	1	0.0002	0.0002	0.35	0.566
Residue × Depth	1	0.0009	0.0009	1.48	0.247
Position × Depth	1	0.0004	0.0004	0.72	0.412
Residue × Depth × Position	1	0.0009	0.0009	1.57	0.234
Residual	12	0.0075	0.0006		
Total	31	0.5806			

(d) Potassium

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.[3]
Rep stratum	3	0.0403	0.0134	0.81	
Rep × Residue stratum					
Residue	1	0.0320	0.0320	1.93	0.259
Residual	3	0.0496	0.0165	28.55	
Rep × Residue × Position stratum					
Position	1	0.0013	0.0013	2.39	0.173
Residue × Position	1	0.0016	0.0016	2.85	0.142
Residual	6	0.0034	0.0005	0.59	
Rep × Residue × Position × Depth stratum					
Depth	1	0.0036	0.0036	3.76	0.076
Residue × Depth	1	0.0001	0.0001	0.11	0.741
Position × Depth	1	0.0000	0.0000	0.00	0.966
Residue × Depth × Position	1	0.0000	0.0000	0.05	0.834
Residual	12	0.0117	0.0009		
Total	31	0.1441			

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[3] **d.f** degrees of freedom **s.s.** sum of squares **m.s.** mean sum of squares **v.r.** variance ratio F probability

Appendix 4.3: Analysis of variance of the effect of residue treatments (burned vs mulched), soil depth of measurement and plot position (inter-row and stump-line; IR/SL) on effective cation exchange capacity (ECEC). Blocking factor: replicate (Rep); whole plots: residue treatments; sub-plots: soil depth; sub-sub plots: position (IR/SL).

Source of variation	**d.f.**	**s.s.**	**m.s.**	**v.r.**	**F pr[4].**
Rep stratum	3	1.1771	0.3924	0.59	
Rep × Residue stratum					
Residue	1	1.8893	1.8893	2.86	0.189
Residual	3	1.9794	0.6598	1.92	
Rep × Residue × Depth stratum					
Depth	1	7.6924	7.6924	22.35	**0.003**
Residue × Depth	1	0.0168	0.0168	0.05	0.833
Residual	6	2.0651	0.3442	0.85	
Rep × Residue × Depth × Position stratum					
Position	1	0.2558	0.2558	0.63	0.442
Residue × Position	1	0.0156	0.0156	0.04	0.848
Depth × Position	1	0.1551	0.1551	0.38	0.548
Residue × Depth × Position	1	0.2044	0.2044	0.50	0.491
Residual	12	4.8648	0.4054		
Total	31	20.3160			

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[4] **d.f** degrees of freedom **s.s.** sum of squares **m.s.** mean sum of squares **v.r.** variance ratio F probability

Appendix 4.4: Analysis of variance of the effect of residue treatments (burned vs mulched), soil depth of measurement and plot position (inter-row and stump-line; IR/SL) on soil exchangeable acidity, pH (H_2O) and pH (KCl). Blocking factor: replicate (Rep); whole plots: residue treatments; sub-plots: soil depth; sub-sub plots: position (IR/SL).

(a) Exchangeable acidity

Source of variation	**d.f.**	**s.s.**	**m.s.**	**v.r.**	**F pr.**
Rep stratum	3	0.2757	0.0919	3.53	
Rep × Residue stratum					
Residue	1	0.2112	0.2112	8.12	0.065
Residual	3	0.0781	0.0260		
Rep × Residue × Depth stratum					
Depth	1	0.0000	0.0000		
Residue × Depth	1	0.0000	0.0000		
Residual	6	0.0000	0.0000	0.00	
Rep × Residue × Depth × Position stratum					
Position	1	1.1400	1.1400	3.08	0.105
Residue × Position	1	0.0242	0.0242	0.07	0.802
Depth × Position	1	0.0000	0.0000	0.00	1.000
Residue × Depth × Position	1	0.0000	0.0000	0.00	1.000
Residual	12	4.4401	0.3700		
Total	31	6.1694			

(b) pH (H_2O)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Rep stratum	3	1.2159	0.4053	1.54	
Rep × Residue stratum					
Residue	1	0.0903	0.0903	0.34	0.599
Residual	3	0.7909	0.2636		
Rep × Residue × Depth stratum					
Depth	1	0.0000	0.0000		
Residue × Depth	1	0.0000	0.0000		
Residual	6	0.0000	0.0000	0.00	
Rep × Residue × Depth × Position stratum					
Position	1	0.1953	0.1953	5.22	**0.041**
Residue × Position	1	0.0528	0.0528	1.41	0.258
Depth × Position	1	0.0000	0.0000	0.00	1.000
Residue × Depth × Position	1	0.0000	0.0000	0.00	1.000
Residual	12	0.4493	0.0374		
Total	31	2.7946			

(c) pH (KCl)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.[5]
Rep stratum	3	1.0734	0.3578	0.50	
Rep × Residue stratum					
Residue	1	1.4878	1.4878	2.07	0.246
Residual	3	2.1534	0.7178		
Rep × Residue × Depth stratum					
Depth	1	0.0000	0.0000		
Residue × Depth	1	0.0000	0.0000		
Residual	6	0.0000	0.0000	0.00	
Rep × Residue × Depth × Position stratum					
Position	1	0.0378	0.0378	1.20	0.294
Residue × Position	1	0.1378	0.1378	4.39	0.058
Depth × Position	1	0.0000	0.0000	0.00	1.000
Residue × Depth × Position	1	0.0000	0.0000	0.00	1.000
Residual	12	0.3768	0.0314		
Total	31	5.2671			

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[5] **d.f** degrees of freedom **s.s.** sum of squares **m.s.** mean sum of squares **v.r.** variance ratio F probability

Appendix 4.5: Analysis of variance of the effect of residue treatments (burned vs mulched), soil depth of measurement and plot position (inter-row and stump-line; IR/SL) on acid saturation. Blocking factor: replicate (Rep); whole plots: residue treatments; sub-plots: soil depth; sub-sub plots: position (IR/SL).

Source of variation	**d.f.**	**s.s.**	**m.s.**	**v.r.**	**F pr[6].**
Rep stratum	3	377.6	125.9	1.07	
Rep × Residue stratum					
Residue	1	846.7	846.7	7.17	0.075
Residual	3	354.1	118.0	4.27	
Rep × Residue × Depth stratum					
Depth	1	1007.8	1007.8	36.42	**<0.001**
Residue × Depth	1	3.3	3.3	0.12	0.741
Residual	6	166.0	27.7	0.11	
Rep × Residue × Depth × Position stratum					
Position	1	1485.7	1485.7	6.00	**0.031**
Residue × Position	1	17.3	17.3	0.07	0.796
Depth × Position	1	1.0	1.0	0.00	0.950
Residue × Depth × Position	1	26.4	26.4	0.11	0.750
Residual	12	2973.0	247.8		
Total	31	7259.0			

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[6] **d.f** degrees of freedom **s.s.** sum of squares **m.s.** mean sum of squares **v.r.** variance ratio F probability

Appendix 4.6: Analysis of variance of the effect of residue treatments (burned vs mulched), soil depth of measurement and plot position (inter-row and stump-line; IR/SL) on soil organic carbon (SOC). Blocking factor: replicate (Rep); whole plots: residue treatments; sub-plots: soil depth; sub-sub plots: position (IR/SL).

Source of variation	**d.f.**	**s.s.**	**m.s.**	**v.r.**	**F pr.**[7]
Rep stratum	3	9.0121	3.0040	7.80	
Rep × Residue stratum					
Residue	1	6.8002	6.8002	17.66	**0.025**
Residual	3	1.1549	0.3850	3.88	
Rep × Residue × Position stratum					
Position	1	0.1989	0.1989	2.00	0.207
Residue × Position	1	0.0490	0.0490	0.49	0.509
Residual	6	0.5953	0.0992	0.44	
Rep × Residue × Position × Depth stratum					
Depth	1	1.2097	1.2097	5.36	**0.039**
Residue × Depth	1	0.1207	0.1207	0.53	0.479
Position × Depth	1	0.0069	0.0069	0.03	0.864
Residue × Position × Depth	1	0.1106	0.1106	0.49	0.497
Residual	12	2.7108	0.2259		
Total	31	21.9690			

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[7] **d.f** degrees of freedom **s.s.** sum of squares **m.s.** mean sum of squares **v.r.** variance ratio F probability

Appendix 5.1: Analysis of variance of the effect of residue treatments (burned vs mulched), soil depth of measurement and plot position (inter-row and stump-line; IR/SL) on mean weight diameter (MWD). Blocking factor: replicate (Rep); whole plots: residue treatments; sub-plots: soil depth; sub-sub plots: position (IR/SL).

Source of variation	**d.f.**	**s.s.**	**m.s.**	**v.r.**	**F pr.**[8]
Rep stratum	3	0.1367	0.0455	0.57	
Rep × Residue stratum					
Residue	1	0.0632	0.0632	0.79	0.439
Residual	3	0.2396	0.0798	4.26	
Rep × Residue × Position stratum					
Position	1	0.1252	0.1252	6.67	**0.042**
Residue × Position	1	0.0042	0.0042	0.22	0.652
Residual	6	0.1126	0.0187	0.67	
Rep × Residue × Position × Depth stratum					
Depth	1	2.8194	2.8194	101.07	**<0.001**
Residue × Depth	1	0.1962	0.1962	7.03	**0.021**
Position × Depth	1	0.0675	0.0675	2.42	0.146
Residue × Position × Depth	1	0.0063	0.0063	0.23	0.643
Residual	12	0.3347	0.0279		
Total	31	4.1059			

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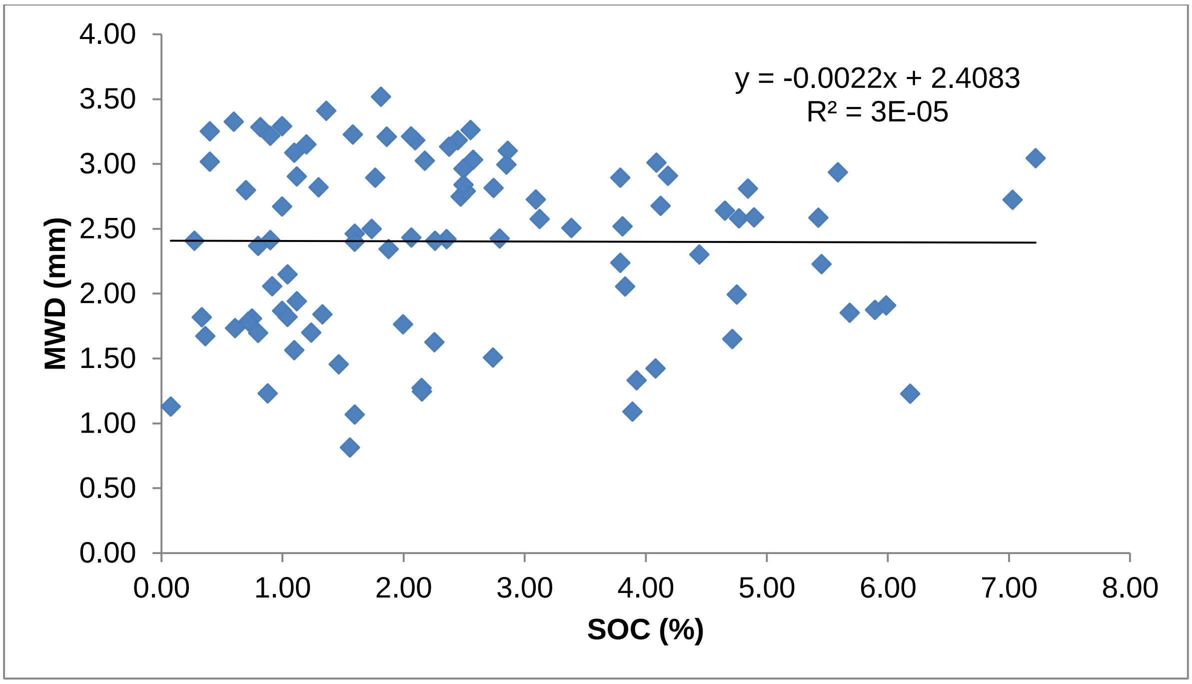
d.f. degrees of freedom **s.s.** sum of squares **m.s.** mean sum of squares **v.r.** variance ratio F probability

Appendix 5.2: Relationship between mean weight diameter (MWD and soil organic carbon (SOC) under residue treatments.

(a) Mulched plot treatment



(b) Burned plot treatment