

Effects of Clear Felling and Residue Management on Nutrient Pools, Productivity and Sustainability in a Clonal Eucalypt Stand in South Africa

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DECLARATION

I declare that the research reported in this thesis, submitted for the degree of doctor of philosophy at the University of Stellenbosch, is the result of my own original research, except where otherwise indicated. This thesis has not been submitted for any degree or examination at any other university.

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ABSTRACT

The subtropical ecosystem of the Zululand coastal plain is prized by the South African commercial plantation forestry industry for its rapid clonal *Eucalyptus* growth, short rotations (6 to 7 years) and high yields. This region is typified by sandy soils that are low in clay and organic matter, have small nutrient reserves and are poorly buffered against nutrient loss. The subtropical climate induces rapid decomposition of residues and tree litter resulting in small litter nutrient pools and rapid nutrient release into the soil, particularly after clearfelling. A combination of large nutrient demands through rapid growth, rapid nutrient turnover and small soil nutrient reserves implies that sites in this region are sensitive and may be at risk of nutrient decline under intensive management. The work in this study set out to determine the risk of nutrient depletion through harvesting and residue management on a site within the Zululand region, to assess nutritional sustainability and the risk of yield decline in successive rotations. Some bulk biogeochemical cycling processes of macro-nutrients nitrogen (N), phosphorus (P), potassium (K), calcium (Ca) and magnesium (Mg) were assessed, and assessments also included sodium (Na).

An existing *Eucalyptus* stand was clearfelled and treatments were imposed on the residues after broadcasting to simulate various levels of nutrient loss through levels of harvesting intensity and residue management. These included residue burning (Burn), residue retention (No-Burn), fertilisation (stem wood nutrient replacement), whole tree harvesting and residue doubling. Outer blocks of the stand were not felled, but included as replicates of an undisturbed standing crop treatment. Biogeochemical nutrient cycling processes were assessed primarily in the standing crop, Burn and No-Burn treatments, in the assumption that these represented the furthest extremes of nutrient loss. Data collection commenced a year prior to clearfelling and continued to two years and six months after planting with key data collection over a 20.1 month period from clearfelling to canopy closure (one year after planting). Water related nutrient pools and fluxes were assessed as atmospheric deposition (bulk rainfall, throughfall and stemflow) and gravitational leaching to 1 m soil depth. Drainage fluxes were predicted using the Hydrus model and real-time soil moisture data. Zero tension lysimeters collected soil solution for chemical analysis. Sequential coring in the 0 to 30 cm soil layer was used to determine in situ soil N mineralisation. Soil chemical and physical properties were assessed over the first meter of soil at clearfelling and new crop canopy closure to determine soil nutrient pools sizes. Biomass nutrient fluxes were assessed from litterfall, residue and litter decomposition, and above ground accretion into the tree biomass. Leaching and N mineralisation were monitored in the No-Burn, Burn and standing crop treatments only.

Atmospheric deposition, while variable, was shown to be responsible for large quantities of nutrients added to the *Eucalyptus* stand. Nitrogen and K additions were relatively high, but within ranges reported in previous studies. Rapid tree canopy expansion and subsequent soil water utilisation in the standing crop permitted little water to drain beyond 1 m resulting in small leaching losses despite a sandy well drained soil. Further leaching beyond this depth was unlikely under the conditions during the study period. Mineralisation and immobilisation of N also remained low with net immobilisation occurring. The standing crop was shown to be a relatively stable system that, outside of extreme climatic events, had a relatively balanced or positive nutrient budget (i.e. nutrient inputs minus outputs).

Large quantities of nutrients were removed with stem-wood-only harvesting in the No-Burn treatment leaving substantial amounts on the soil surface in the harvest residues. Whole tree removal increased losses of all nutrients resulting in the largest losses of P and base cations compared to all other treatments. This was mostly due to high nutrient concentrations in the removed bark. Loss of N in the Burn treatment exceeded whole tree N losses through combustion of N held in the harvest residues and litter layer. The majority of K leached from the residues prior to burning and a relatively small fraction of the base cations were lost from the partially decomposed residues during burning. Ash containing substantial amounts of Ca and relatively large amounts of N and Mg remained after burning. Surface soil Ca and Mg was significantly increased by the ash which moved into the soil with rainfall directly after burning.

Rapid soil moisture recharge occurred within a few months after clearfelling, increasing leaching from the upper 50 cm of soil. Clearfelling increased net N mineralisation rates, increasing mobile NO_3^- -N ions in the soil surface layers. Nitrate concentration peaked and K concentration dipped in the upper soil layers of the Burn treatment directly after burning. Deep drainage and leaching (beyond 1m depth) over the 20.1 month period was, however, not significantly different between the Burn and No-Burn treatments. Rapid soil moisture depletion and nutrient uptake with new crop growth reduced leaching fluxes to levels similar to the standing crop by six months after planting. Taking the full rotation into account, clearfelling induced a short-lived spike in N and cation leaching compared with the low leaching losses in the undisturbed standing crop. Soil N mineralisation over the 20.1 month period in the burnt treatment was half that of the No-Burn treatment.

Growth and nutrient accumulation was significantly higher in the fertilised treatment than in other treatments up to 2.5 years of age. Growth in the Burn treatment was greatest compared to

other treatments during the first few months, but slowed thereafter. No significant growth differences were found between all other treatments from a year to 2.5 years after planting. Early growth was therefore apparently not limited by N supply despite large differences in N mineralisation between Burn and No-Burn. Foliar vector analysis indicated that fertilisation improved growth initially through increased foliar N and P at six months after planting followed by Mg and Ca at one year. The Burn treatment was not nutrient limited. These growth results contrasted with similar international research on sandy tropical sites where growth was reduced after residue removal and increased after residue doubling. The combined nutrients released from pools in the litter layer or ash and soil in addition to atmospheric inputs were sufficient to provide most nutrients required to maintain similar growth rates across all treatments. This demonstrated the importance of residue derived nutrients to early growth nutrient supply. Reduced N mineralisation through a lack of substrate may limit N supply later in the rotation where residue had been removed.

Construction of a nutrient budget for the system revealed that high levels of atmospheric inputs have the potential to partially replenish a large proportion N, K, and Ca lost during clearfelling, provided losses are constrained to stemwood removal only. However, loss of Mg that occurred primarily through leaching may not be replaced under the low Mg inputs recorded in this study. Larger nutrient removals (i.e. stemwood plus other plant parts) placed a heavier reliance on the small soil nutrient pools at this site which can limit future productivity. More intense harvesting and residue management practices dramatically increased the risk of nutrient depletion. Losses of specific nutrients depended on a combination of clearfelling biomass removal, residue burning and subsequent leaching. Nitrogen losses due to harvesting and burning were more substantial than those due to leaching. Mg and K losses depended most strongly on the time after clearfelling before re-establishment of the new crop and rainfall patterns, while Ca and P losses depended directly on the amount of biomass removed. Depletion risk was the greatest for Mg and K through rapid leaching, even after stem wood only removal. Deep root uptake and deep drainage with associated cation loss needs to be investigated further to quantify ecosystem losses and recovery of cations displaced beyond 1 m.

Atmospheric deposition is one of major factors countering nutrient losses. However, atmospheric inputs may not be reliable as these may lessen in future through pollution control legislation and climate change. Changes in growth rate under poor nutrient management practices are small and difficult to detect relative to the large impacts of changing weather patterns (drought), wildfire and pest and disease. This makes it difficult to prove nutrient related growth decline. It may be possible that improvements in genetics, silvicultural technologies and

atmospheric inputs may also be masking site decline (in general) and in part explain the lack of evidence of a growth reduction in the region.

As the poorly buffered sandy soils on the Zululand Coast are at risk of nutrient depletion under the short rotation, high productivity stands, it may be necessary to stipulate more conservative harvesting and residue management practices. A more conservative stem-wood only harvesting regime is recommended, retaining all residues on site. Residue burning should be avoided if N losses become a concern. The length of the inter-rotation period must be kept short to reduce cation leaching losses. Site nutrient pools need to be monitored and cations may eventually need to be replenished through application of fertilisers or ash residues from pulp mills. Management practices therefore need to be chosen based on the specific high risk nutrients in order to maintain a sustainable nutrient supply to current and future plantation grown *Eucalyptus*.

DEDICATION

To my late grandmother Mavis Mawman who always encouraged and inspired me to study.

Pointing to her head she would always tell me:
“it’s what is in *here* that counts”



A family of elephants passing the experimental area en route to a nearby indigenous forest

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LIST OF TERMS AND ABBREVIATIONS**Chemical terms**

TKN	Total Kjeldahl nitrogen
O-N	Organically bound nitrogen
CEC	Cation exchange capacity
OC	Organic carbon
EC	Electrical conductivity

Hydrus model

Q_r	Residual soil moisture content
Q_s	Saturated soil moisture content
α	Parameter in the soil moisture retention function
n	Parameter in the soil moisture retention function
K_s	Saturated hydraulic conductivity

Statistical terms

ANOVA	Analysis of Variance
CV	Coefficient of Variation
MSE	Mean Square Error
RCB	Randomized Complete Block
R^2	R-square; coefficient of determination
SD	Standard Deviation
LSD	Least significant difference
CI	Confidence interval
SE	standard error
ns	Not significant
p	Probability
*	$0.01 < p < 0.05$
**	$0.001 < p < 0.01$
***	$p < 0.001$

Treatment abbreviations

Standing crop (SC) Undisturbed crop – no harvest

Whole-Tree	Removal of stem wood, bark, branches and foliage at clearfelling
Burn	Residues broadcast and burned
No-Burn	Residues broadcast and retained
Fertilised (Fert)	No-Burn and replacement of stem wood macronutrients
Double	No-Burn and residues added from Whole-Tree plots

Plant/tree related terms

GPP	Gross primary productivity
NPP	Net primary productivity
<i>Supply</i>	Refers to the available supply of tree growth resources
<i>RCE</i>	Resource capture efficiency
<i>RUE</i>	Resource uptake efficiency
dbh	Diameter of mature tree stem at 1.3 m above ground level
gld	Ground-line diameter of juvenile tree stem
LAI	Leaf area index.
SLA	Specific leaf area.
SI ₅	Site index - 80th percentile of tree heights projected to 5 years of age
LOI	Loss on ignition
<i>k</i>	Decay constant

Climate terms

MAP	Mean annual precipitation
MAT	Mean annual temperature
APAN	Refers to Class A Pan Model of Evaporation

Other

HDPE	High-density polyethylene
CIFOR	Centre for International Forestry Research
LAN	Limestone ammonium nitrate
TDR	Time domain reflectometry

CHAPTER 1: GENERAL INTRODUCTION AND STUDY OBJECTIVES

1.1. INTRODUCTION

The development of science and management strategies in commercial plantation forestry requires the merging of two seemingly contradictory approaches: that of a plantation forest as an agricultural production system and as a managed ecosystem. The agricultural approach (historically popular in some plantation forestry circles in South Africa) classically considers the use of available tools and technology to maximise biological production through increased or optimised resource use. A managed ecosystem approach considers the plantation forest as an integration of biotic and abiotic components that are managed to ensure a sustainable supply of products. Each approach has merit, although the managed ecosystem approach is usually preferred by forest scientists and managers alike as it considers a broad-sense, holistic view of plantation forestry. The lack of scientific data describing many aspects of South African plantation forests as ecosystems has however, led to management decisions based more on empirical evidence or experience-based intuition than on scientific evidence or process understanding. Plantation forest managers have therefore expressed a need to integrate this intuitive approach with a quantitative, evidence-based understanding to ensure the maintenance or improvement of present and future plantation productivity without compromising site resources, neighbouring ecosystems, human activities and profitability. Scientists and managers therefore need to gather data and evidence that documents the impacts of various management practices on sustainable plantation forest productivity across all forest plantation sites while not forgoing the wealth of experience held by the South African and international forestry industry.

1.2. COMMERCIAL FORESTRY AND LONG-TERM SITE PRODUCTIVITY

Commercial plantation forestry, consisting of 51% *Pinus* spp., 40% *Eucalyptus* spp., and 8% *Acacia* sp., presently covers 1.3 million hectares or 1.1% of South African land area (FSA 2009b). The plantations, originally established mainly into grassland ecosystems (Acocks 1953; Mucina and Rutherford 2006), are spread across a relatively large range of site types and soil forms, each having a unique site history with respect to previous land management. Playing a vital role in supplying South African timber and fibre needs (approximately 17 million air dry tons per annum of which 63% is short rotation pulpwood) the commercial forestry industry

contributes 1.27% to the GDP and employs 0.5 million people. A constant supply of commercially produced wood decreases the pressure on native forests for wood production (and other forest products) while creating much awareness around social and environmental concerns. The *Eucalyptus* species supplies around 60% of South African pulp fibre (FSA 2009b) from plantations and is mainly concentrated in the KwaZulu-Natal and Mpumalanga forestry regions, covering approximately 0.5 million hectares of land (FSA 2009b).

In response to much criticism that has been received over the years, particularly around water and environmental concerns, the commercial forestry industry has become one of the most regulated (self and government) agricultural practices in South Africa (Dye and Versfeld 2007). These regulations, an increased demand for wood products, and numerous other political, economic and environmental factors have pressurised the industry to seek innovations to enable larger quantities of timber and biomass to be produced from a relatively small and shrinking land-base (FSA 2009b). This shrinkage of land area (9% between 1999 and 2009 (FSA 2009b)) has occurred through *inter alia*: wetland delineation, successful land-claims that have subsequently been converted to non-forestry activities and the establishment of wildlife corridors through plantation forest areas. However, increases in productivity of 19% through the same (above) ten year period have been realised through management improvements that include: the selection of faster growing trees in tree breeding programs, improved matching of tree species and hybrids to sites for optimal timber growth, planting of disease resistant clonal varieties, shorter rotation lengths, more intensive silvicultural practices at establishment, mechanisation and integrated pest and disease management programs (Schutz 1982; Schönau 1984; Burger 1996; Bauhus et al. 2002; du Toit et al. 2010). In addition to sawtimber, pulpwood, mining timber and poles, plantation forestry has in recent years been considered a source of renewable bio-energy, as the harvest residues are seen as an easily available biomass source. This has also substantially increased the rate and quantity of biomass removal per unit land area of plantation forests and consequently increased pressure on water and nutrient resources on the remaining land area (Nambiar 1996,1999). It is therefore crucial that the negative impact of a greater demand on water and nutrients be understood and quantified so that long-term sustainable productivity can be maintained and promoted.

1.3. A NEED FOR PROCESS-BASED SUSTAINABILITY RESEARCH

Monitoring changes in plantation productivity over numerous rotations using measures of tree growth and forest product output (narrow sense sustainability, (Evans 1999)) may be a poor indication of ecological processes and functioning within the plantation forest when used alone. This is due to the effects that climatic variability, genetic improvements, pests and diseases outbreaks, wildfires and non-uniform silvicultural practices have on tree growth based productivity determinants of a site. It is therefore necessary to understand the factors that influence the complex biotic and abiotic functioning of ecosystem processes that drive sustainable productivity, while excluding, or accounting for as many of the confounding effects as possible. The principles and processes driving sustainable productivity are not fully understood or quantified in commercial plantation forestry, particularly around factors that dictate site fertility (Blanco et al. 2005; Laclau et al. 2005a; Watt et al. 2005). Research therefore needs to become more focused on understanding the key processes that determine the nutritional sustainability of productive forest plantations, by quantifying the effects of intensive management practices on key nutrient processes in productive plantation forests grown on sensitive sites.

1.4. EARLY INTENSIVE SILVICULTURE DURING THE INTER-ROTATION

The early stage of a short-rotation pulpwood plantation growth cycle, from felling to post canopy closure of the subsequent crop represents the most intensive silvicultural management period of a plantation forest. The operations during this period often determine the growth and productivity of the remainder of that rotation (Nambiar 2008; du Toit et al. 2010; Smethurst 2010), while having the potential to damage or improve ecosystem productivity, which can affect the growth of subsequent rotations (Schönau 1984; Brais et al. 2002). Although the tree growth responses to management practices may vary widely across sites, the impact that management practices have on sustainable nutrient supply and growth may become more important on infertile sites, i.e. sites that have small nutrient reserves. Sites characterised by rapid growth and fast nutrient turnover will be far more sensitive to management interventions than cooler, dryer, slow growth sites.

1.5. ZULULAND COASTAL ECOSYSTEM

The Zululand coastal plains (northern KwaZulu-Natal, South Africa) consist of extensive areas of sandy albic arenosols with single-grain structure (FAO 2006) and a sub-tropical climate that results in areas of low nutrient reserves (Hartemink and Hutting 2005; Fey 2010), but high productivity potential (Smith and du Toit 2005). These soils are unable to buffer large chemical, physical and organic (biological) changes, placing them at risk of degradation, ultimately limiting tree productivity under poor management practices. This has been confirmed through tree growth responses to applications of N fertiliser at planting (du Toit et al. 2001; du Toit and Oscroft 2003). Clonal *Eucalyptus* is grown primarily as a short rotation (6-7 years) pulpwood crop across the area. Site management includes residue broadcasting, windrowing and, where necessary, residue burning. Standard fertilisation practices recommend the addition of around 70 kg ha⁻¹ of N as limestone ammonium nitrate (LAN). Plantations grown in this region have given little conclusive evidence towards a nutrient related productivity decline despite having undergone numerous rotations of intensive silvicultural management practices and frequent wildfires (de Ronde 1992; Oscroft and Little 2008).

1.6. STUDY APPROACH

This thesis describes research that was carried out to determine the impact of clearfelling and two opposing extremes in standard management practices (residue retention and residue burning) undertaken during the fallow inter-rotation period after harvesting, on pools, losses and gains of selected elements. This was also done to describe the impact of site management on long-term sustainable productivity of clonal *Eucalyptus*, with respect to natural and management induced nutrient losses and gains. The study primarily compares the fluxes of nitrogen (N) within the water, soil and organic components of a mature clonal *Eucalyptus* stand with those after felling upon which residue burn and no-burn management practices were imposed. The elements potassium (K), calcium (Ca), magnesium (Mg), sodium (Na) and phosphorus (P) are included in various components of the research. Data collection began prior to clearfelling and continued to canopy closure of the new crop growth, though some aspects of the data collection continued beyond canopy closure. Additional treatments were included that involved manipulation of residues and fertiliser addition so that the interpretation of the results could be improved.

The main objective of the study was to:

Compare major nutrient pools and fluxes in a high productivity, clonal *Eucalyptus* stand grown on a nutrient sensitive site that was left undisturbed with a clearfelled stand subjected either to residue retention or residue burning, through planting to canopy closure; relating this to current and future nutrient supply and productivity.

1.7. HYPOTHESIS

Nutrient flux processes and losses on a productive, but sensitive site are accelerated by clearfelling and subsequent residue burning resulting in a significant reduction in soil and biomass nutrient pools that are not replenished through natural additions.

This study will be achieved through a number of component studies with the following objectives:

- To track changes in soil moisture fluxes, parameterise a soil moisture model and predict gravitational drainage fluxes within the first meter of soil.
- Determine the order of magnitude of atmospheric nutrient additions to the site and the region.
- Describe the fate of nutrients held in tree, litter and residue biomass components as a consequence of clearfelling and residue management by comparing decomposition, litterfall, growth, nutrient accretion in the undisturbed stand with the newly planted stand.
- Determine changes in top-soil N mineralisation and how this contributes to soil mineral N fluxes throughout the study between clearfelling and canopy closure.
- To use predicted drainage fluxes and collected soil solution data to calculate nutrient leaching within the first meter of soil.
- To determine net nutrient gains and losses induced directly and indirectly through harvesting and residue management operations, quantifying net losses, gains and pools of nutrients in the system through a nutrient budget approach.

1.8. THESIS STRUCTURE

This thesis comprises a literature review (Chapter 2) followed by general site descriptions and the experimental methods and layout (Chapter 3). The main body of this thesis is a compilation of papers covering the first five component studies prepared for peer-reviewed publications

(Chapters 4 to 8). The final chapter (Chapter 9) is a general discussion and synthesis of the work. The significance of the results and their implications for future research are highlighted. As much of the research presented in this thesis has been published or is currently under review, Chapters 4 to 8 have been presented as close to their published format as possible and include relevant literature and objectives specific to each component study. Repetitive materials and methods content has mostly been removed and included in Chapter 3 to enhance readability. Table and figure numbering has been altered in accordance with this thesis structure. The papers were written to stand as separate entities in each publication and often necessitated the use of data outside of the study described in this thesis. Some data presented in the results of each chapter repeats or draws upon data derived from other chapters. Chapters also include additional comments and data as appendices.

CHAPTER 2: GENERAL LITERATURE REVIEW

2.1. INTRODUCTION

Globally, plantation forests are an important renewable resource that in addition to wood, provide a wide range of products and ecosystem services (Sampson et al. 2005; Nambiar 2008). Both worldwide, and in South Africa, plantation forest management aims at producing an economically profitable, sustainable wood supply through intensive management by improving genetics, silviculture and harvesting (Morris and Smith 2002; Binkley et al. 2004; Goncalves 2004; Glavič and Lukman 2007). Despite increasing productivity of South African plantation forests (consisting of *Pinus*, *Eucalyptus* and *Acacia* species; (FSA 2009b)) concerns around the long-term sustainability of yields are still being raised. This is due to the increased productivity and a limited understanding of the ecological principles and processes driving the productivity of plantation ecosystems (Nambiar 1996; Laclau et al. 2003a; Watt et al. 2005; du Toit et al. 2010).

Increased productivity through intensive management practices has placed an increased demand on plantation sites, altering soil physical properties, soil organic matter fluxes and nutrient dynamics (Smethurst and Nambiar 1990; Blanco et al. 2005; du Toit et al. 2010; Rietz 2010). These intensive practices, that often include shorter rotation lengths, frequent harvesting and extreme silvicultural operations (residue burning and whole tree harvesting) result in more frequent and higher biomass removal while providing less time for the plantation site (and ecosystem) to recover (Tiarks and Ranger 2008; du Toit et al. 2010). The survival of the forestry industry is ultimately dependent on sustainable management practices derived from sound research, and is imperative not only to ensure continued profitable yields, but impacts on broader social, environmental and economic sustainability issues (Attiwill and Leeper 1987; Costanza and Patten 1995; Nambiar 1996; Toman and Ashton 1996; Olbrich et al. 1997; Evans 1999; Nambiar 1999; Glavič and Lukman 2007; Nambiar and Kallio 2008).

2.2. SUSTAINABILITY

The definitions of sustainability are numerous and lack common consensus, often formulated according to human agendas or within a narrow context. These definitions, in the most part,

describe an outcome based prediction of sustainable resource supply and utilise a variety of measured indicators that may include or exclude environmental, economic and social components, covering various time scales and endpoints (Attiwill and Leeper 1987; Costanza and Patten 1995; Nambiar 1996; Toman and Ashton 1996; Evans 1999; Nambiar 1999; Glavič and Lukman 2007). Sustainability in plantation forestry has in a similar manner focussed on predicting the supply of wood (fibre) and other products from plantation forests into the future, and uses measures of these forest products over time to determine sustainability (Evans 1999; Morris and Smith 2002). This approach, termed narrow sense sustainability, gives little consideration to ecosystem processes and is often masked by confounding factors such as silviculture, genetic improvements, the effects of weather patterns, pest and disease and other factors (Smith et al. 2005a; du Toit 2006). Sustainability in the broader sense encompasses an understanding of the ecological processes that drive the productivity of the plantation and includes whole tree plus ecosystem productivity as opposed to the marketable components considered by narrow sense sustainability (Smith et al. 2005a). Although broad sense sustainability offers improved understanding and prediction, to be complete, it must acknowledge the natural finite lifespan or renewal cycles of an ecosystem. Natural systems when left undisturbed have cyclical lifespans of finite duration (du Toit 2006; Voinov and Farley 2007). A broader definition of sustainability in the plantation forestry context to encompass definitions and constraints found in literature is suggested as follows:

Management of plantation forest ecosystem resources to ensure the ecological productivity of the system for the maximum expected lifespan while maintaining continued economic, social and environmental benefits derived from the plantation ecosystem.

As it is not possible to meet all the requirements laid out by these objectives, management practices are often formulated to allow for acceptable tradeoffs between the realised benefits derived from the plantation and the various sustainability criteria (Attiwill and Leeper 1987; Costanza and Patten 1995; Nambiar 1996; Toman and Ashton 1996; Evans 1999; Nambiar 1999; Morris and Smith 2002; Glavič and Lukman 2007; Voinov and Farley 2007).

This review considers the interactions between management and biogeochemical nutrient cycling and the role that this plays in maintaining the ecological sustainability of plantation forest

ecosystems for maintaining economically viable yields into the future. Although this approach tends towards sustainability in both the narrow and broad sense, it provides a more in-depth understanding of ecosystem processes that drive plantation productivity. This literature review attempts to introduce this idea, while not repeating details and literature derived data given in each of CHAPTERS 3 to 8.

2.3. PRODUCTIVITY AND PLANT GROWTH RESOURCES

Plantation forest gross primary productivity relies on the ability of trees to capture light energy and utilise this to convert carbon dioxide (CO₂) and water into biomass (Battaglia et al. 1998; Powers 1999). The ability of trees to capture light is dependent on tree leaf area, which in turn is reliant on the availability of the resources of light, air, water, and nutrients (not ignoring time and land area/growing space). Gross primary productivity (*GPP*) is the product of resource supply (*Supply*), the proportion or efficiency of resource capture (*RCE*) and the efficiency at which resources are utilised (*RUE*) (Monteith (1977), expressed as:

$$\text{Productivity (GPP)} = \text{Supply} \times \text{RCE} \times \text{RUE}.$$

Gross primary production is allocated to respiration (including symbiotic organisms) and net primary production (NPP), and NPP is further allocated to plant structural components (including stemwood) and growth functions (such as root and canopy turnover). Losses of NPP can occur through pest and disease interactions and adverse weather. Gross primary production, NPP and allocation patterns are species dependant, can change over the duration of a rotation (physiological age), and can be affected by site factors that are altered by natural or man induced shifts in resource availability or balances (Cannell and Dewar 1994; Binkley et al. 2004; Stape et al. 2004; Ryan et al. 2008). Productivity will be reduced if there is any single resource limitation or imbalance through management practices or natural phenomenon that alters *Supply*, *RCE* or *RUE* of tree growth resources (Binkley et al. 2004; Ryan et al. 2008). It is logical that permanent alterations of any of these will alter the inherent ecological productivity of a site and impact on the tree productivity in the long-term (sustainability). The ecological productive potential of a plantation forest is therefore determined through the physical and biological interactions between

individual trees, the trees and other biota, the chemical, biological and physical nature of soils (including depth) and climate (Waring and Ludlow 2008).

2.4. PLANTATION MANAGEMENT AND SUSTAINABILITY

Plantation management uses various strategies to enhance or maintain productivity through manipulation of site properties to alter resource *Supply*, *RCE* and *RUE*. Management practices have little influence over natural shifts in global weather patterns that dramatically alter the productive potential of any given rotation of trees (Pretzsch 2009). Such changes can only be adaptively managed through timing of silvicultural operations to coincide with seasons, following rainfall events or planting genetically tolerant tree species. The supply of water and nutrients from soil and organic reserves are therefore the main biotic and abiotic factors that can be manipulated through management practices, and that determine the productivity of plantation grown *Eucalyptus* (Laclau et al. 2003a; Stape et al. 2004; Nambiar 2008; Smethurst 2010). Current plantation management therefore uses a wide variety of early site preparation and silvicultural practices that can enhance plantation growth or reduce production costs. These include harvesting methods, the management of residues, site and soil preparation, fertilisation, site-species matching, planting practices, tree spacing, thinning, pruning, pest/disease control and weed control (Nambiar and Kallio 2008; Tiarks and Ranger 2008).

Although management practices can enhance the growth of a current tree rotation, such practices if done without care, can have a negative impact on plantation productivity in the long term. This is through negative impacts on site resources soil and organic matter that can be lost or degraded. Poor or incorrect management has the potential to severely degrade or deplete soil structure, chemistry, carbon (micro-organisms included) and site nutrient supply and storage (du Toit et al. 2010; Rietz 2010). The risk of these resources becoming degraded or depleted is dependent on site climatic and edaphic properties, tree species, management practices and the capture, storage and release (fluxes) of carbon and nutrients between atmospheric, soil and biological pools (Laclau et al. 2003a; Nambiar 2008; Smethurst 2010). The risk and rate of such loss or degradation is however, site and management dependant. Binkley (1986b) for example, proposed that nutrient losses are most relevant on sites that have small nutrient reserves in relation to large removals and hence most at risk of nutrient depletion. A number of international studies also confirmed this concept in tropical *Eucalyptus* plantations, which suffered significant productivity

declines after excessive biomass removal (Carlyle and Nambiar 2001; Nambiar 2008). This was particularly evident in stands grown on sandy soils (arenosols) and dystrophic loams (oxisols), with low nutrient buffer capacities and low organic matter, where declines in growth were attributed to a reduction in nutrient availability or nutrient pool depletion (Mendham 2003; Nambiar 2008; Saint-André et al. 2008; Tiarks and Ranger 2008; Laclau et al. 2010a). The sustainable supply of nutrients for tree growth at a specific site can only be determined through understanding the size of the various nutrient reserves or pools, the fluxes of nutrients into (additions), between (cycling) and out of (losses) each pool.

2.5. NUTRIENT POOLS AND FLUXES

Nutrients in plantation forest ecosystems are found in organic and inorganic forms in soil, in soil organic material, tree biomass (stem, roots, canopy and reproductive structures) and forest floor litter layers (Morris 1986; Laclau et al. 2003a). Nutrients move (flux) between these pools and are added to and lost from the ecosystem (**Figure 1**). Fluxes of nutrients are driven primarily by energy and matter flows that are regulated by site environmental factors. These factors principally include water, temperature, plant available radiation, wind, phenology, time, physical and chemical nature of each component of the ecosystem (air, plant, organic material, soil and water). An understanding of these various pool sizes and flux rates and magnitudes during each plantation growth phase relative to site types and management practice is necessary to determine the sustainability of plantation forests from a nutritional perspective (Wells and Jorgenson 1978; Attiwill and Leeper 1987; Attiwill and Adams 1993a; Morris 1997; Nzila et al. 2002; Laclau et al. 2005a; Guo et al. 2006). A more comprehensive review of these pools and fluxes is presented in CHAPTERS 5 to 9 below. Since water is one of the main drivers of nutrient fluxes in plantation forests (Smethurst and Nambiar 1990; Carlyle and Nambiar 2001), management practices that alter the hydrological properties of soils or even surface properties can have a dramatic effect on nutrient flux processes. Residue burning, for example, can increase soil water repellency which in turn can reduce infiltration rates and increase runoff and soil erosion on sloped lands (Scott and Van Wyk 1990; Scott 1993; DeBano 2000; Scott 2000). The dependence of many nutrient cycling processes on water fluxes is obvious and implied in the remainder of this review. Soil water is central to this study and a more comprehensive literature review on soil water in forest ecosystems is presented in CHAPTER 4.

2.5.1. Nutrient fluxes

Natural inputs of nutrients occur through mineral weathering, nitrogen (N) fixation, atmospheric deposition and a number of other, often minor processes (animal and colluvial movement) (**Figure 2.1**). A short review of the literature on atmospheric inputs of N, potassium (K), calcium (Ca) and magnesium (Mg) with rainfall, throughfall and stemflow in forest ecosystems is presented in CHAPTER 5. Anthropogenic inputs are typically associated with fertilisation, while nutrient losses occur, directly and indirectly, through harvesting and silvicultural management practices. Fertiliser is used in commercial forests to increase productivity by improving nutrient availability (even on inherently fertile sites). Anthropogenic losses occur directly through harvesting and burning and indirectly through soil erosion and acceleration of natural loss processes. Natural losses occur through leaching, erosion, oxidation during fires and volatilization (denitrification of N) (Jorgenson et al. 1975; Binkley 1986a; Attiwill and Adams 1993a; Kimmins 1994; Ranger and Turpault 1999; Blanco et al. 2005).

Nutrients are cycled within the system by retranslocation, canopy exchange, biomass decomposition and soil exchange, which redistributes nutrients from and between various ecosystem nutrient pools (soil, detritus, plant, atmosphere) and includes the immobilisation and mobilisation of these nutrients (Ranger and Turpault 1999; du Toit and Scholes 2002). Nutrient lock-up in the forest floor or its release through decomposition is dependent on factors that include temperature and moisture regimes, lignin and N contents and ratios, soil faunal characteristics and litter pH (Hobbie and Gough 2004; Prescott 2005). Warmer and wetter conditions, for example, induce more rapid decomposition and nutrient release (Jorgenson et al. 1975; Attiwill and Leeper 1987).

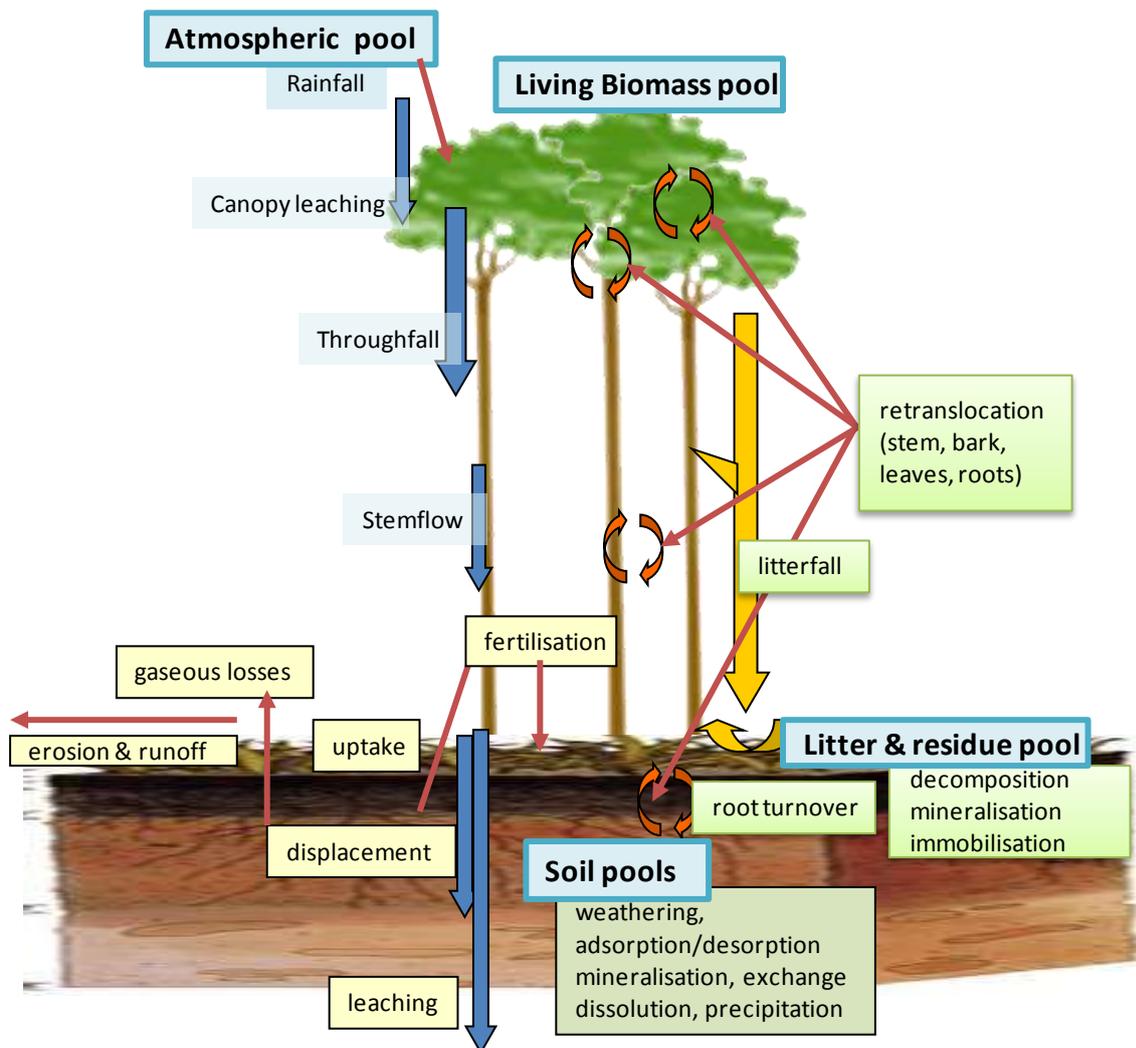


Figure 2.1: Representation of some major nutrient pools, additions, losses and cycling fluxes within a mature plantation ecosystem.

2.5.2. Nutrient pools in the soil

The soil generally contains a disproportionately large total nutrient quantity relative to plantation biomass nutrient pools, although a very small portion of this is accessible to the trees at any one time (Fisher and Binkley 2000). The size of the soil nutrient pool is generally dependent on the volume of soil available for root exploration, the depth to which nutrient uptake can occur, soil density and the concentrations of plant available soil nutrients (Turner and Lambert 2011). Nutrients may be held in various organic or inorganic forms in the soil that may be readily available for root uptake or be transformed into plant available forms within a rotation or over many rotations. The fluxes between these pools and availability forms are complex and specific to each nutrient element in each soil pool, but also their interactions as this will control the

thermodynamic processes affecting exchange, sorption, precipitation, etc. (Fisher and Binkley 2000). Nutrients held in organic forms are either microbially mineralised or immobilised. Inorganic nutrients transfer between the soil solution and the soil solid phase through ion exchange redox reactions (cation or anion exchange), are reduced or oxidised and undergo fixation or dissolution processes (Fölster and Khanna 1997).

2.5.2.1. Nitrogen

Nitrogen, which is generally held as organic forms in the soil carbon pool, can be very large relative to mineral forms, but mineralisation of the organic forms is dependent on microbial activity which may limit the rate and amount of N released for uptake by trees (Binkley and Hart 1989). The soil carbon pool is made up of plant and microbial matter in active (readily oxidisable), intermediate (slowly oxidisable) and passive (recalcitrant) fractions that can turnover at rates of days, years, decades, centuries or millennia (Attiwill and Leeper 1987). Organic additions to the soil N pool occur through root and litter decomposition and leaching of organic compounds from the tree canopy. Fertiliser, N fixation (symbiotic and non-symbiotic) and atmospheric deposition add inorganic N to this pool (Ranger and Turpault 1999; Son 2001). Nitrogen mineralisation by soil micro-organisms converts organic soil N from decomposed plant material to ammonium, while further oxidative nitrification can convert a portion of this ammonia to nitrite and then nitrate (Raison et al. 1987). A loss of N from the soil pool can occur through plant uptake, leaching of ammonium and nitrate, volatilisation under alkaline conditions, oxidation during high intensity fires, or through denitrification under anaerobic conditions (Groffman 1995). Release of N bound in organic forms through N mineralisation can provide a large proportion of tree uptake requirements (Fölster and Khanna 1997; Scholes 2002). The release of N from soil organic material through mineralisation depends on the physical and chemical nature of the soil organic matter, soil moisture and soil temperature which includes aeration, C: N ratio, soil pH and soil P availability (Tisdale et al. 1985; Gunderson et al. 1988; Harris 1988; Carlyle et al. 1990; Van Miegroet et al. 1990; Rice and Havling 1994; Goncalves and Carlyle 1994; Barnes et al. 1997; Sierra 1997; O'Connell and Rance 1999). Changes in these factors with soil depth results in decreased N mineralization with increased soil depth (Cabrera 1993). Understanding these variables has allowed N release rates to be predicted at both temporal and spatial scales using process-based models parameterised with a minimal set of site and soil data (O'Connell and Rance 1999; Paul et al. 2002). Net immobilisation of N can

occur under conditions of low soil N availability, relative to available organic carbon (C) (Smith and du Toit 2005). Nitrogen and N mineralisation are discussed further in CHAPTER 7..

2.5.2.2. Phosphorus

Phosphorus (P) is often abundant in the soil, existing in non-labile organic (microbial, plant detritus and humus), primary apatite minerals, secondary compounds, as P that is chemi-sorbed or occluded onto mineral surfaces (iron and aluminium oxides, carbonates and clays), labile (exchangeable) P and soil solution P (Walker and Syers 1976; Frossard et al. 2000). Phosphorus fluxes occur between solid and solution phases in the soil through adsorption, desorption of primarily forms of P ions on to functional groups of soil sesquioxide surfaces and dissolution and precipitation of P compounds (Harley and Gilkes 2000; Breemen and Buurman 2003b,2003a). Organic to inorganic transformations occur through biological mineralisation and immobilisation of organic and labile P (Kalbitz et al. 2000). Although having a far larger pool in the soil than the forest floor litter layer, organic forms of P in the forest floor tend to be far more available for plant uptake than soil P forms, resulting in the bulk of P uptake being derived from and recycled back to the organic P pools (Attiwill and Adams 1993a). The loss of the forest floor litter often results in a notable deficiency of P in the trees (Neary et al. 1999). Natural additions of P occur through mineral weathering and losses through leaching (Breemen and Buurman 2003b). P is not a particularly mobile soil element so leaching is often only in soils that lack sorption sites. Phosphorus leaching rates are often very low in undisturbed forest ecosystems. Greater losses occur through loss of particulates (erosion) where P is sorbed or fixed on the fine material. South African plantation forests are grown on highly weathered soils that contribute insignificant weathering P and are often prone to P fixation (Bainbridge et al. 1995; du Toit and Scholes 2002). Mycorrhizal associations with tree roots can enhance P uptake from the soil, labile P pools and from (often inaccessible) soil sources (Plassard and Dell, 2010). This may be important on P deficient soils where fungal excretion of organic anions such as oxalic acid can release insoluble P.

2.5.2.3. Base cations

Base cations (K, Ca and Mg) are predominantly held in the soil on the soil exchange complex. The ability of soils to bind cations depends on the cation exchange capacity (CEC) of the soil, which is dependent on soil clay type and content, organic matter (humus) content and pH.

Potassium can be held in the interlayer positions if illitic clays are present in the soil (Sawhney 1972). Mineral weathering and atmospheric inputs are the most important long-term sources of natural base cation inputs to plantation forests. Mineral weathering is dependent on soil mineral composition and exposed surface area, soil depth, climate (which determines temperature and moisture conditions), soil CO₂ and soil acidity (Sverdrup and Warfvinge 1993). As plant growth affects the last four factors to a variable degree, it can alter mineral weathering rates and exchange processes (Kelly et al. 1998). Although mineral weathering can be an important input of cations in the long-term, the short-term supply is often insignificant from a plant supply perspective (Jorgenson et al. 1975; Ranger and Turpault 1999). The general lack of long-term soil studies has made it difficult to quantify the rate of base cation release from mineral weathering (Richter and Markewitz 2001). However, cation release from mineral weathering plays a minor role in the highly weathered Southern African soils (Owens and Watson 1979; du Toit and Scholes 2002). Other factors affecting base cation status are mineralisation of organic material, root exudation and turnover, and canopy leaching. These contribute to the cycling of base cations through the soil, organic material and plant environment (Likens and Bormann 1995; Ouimet and Duchesne 2005). Losses of base cations from the soil occur primarily through plant uptake and subsequent biomass removal or through leaching, especially in acidic soils and soils that have a low buffering ability (low carbon and low clay soils).

2.5.3. Nutrient pools in the biomass

Considering a rotation, the greatest potential for site manipulation and nutrient loss occurs during the period between clearfelling and canopy closure (Nambiar 2008; du Toit et al. 2010; Smethurst 2010). A number of harvest techniques and residue management practices are used in plantation forestry resulting in partial or full retention, removal or burning of residues. The choice is driven by wildfire risk mitigation, economics, residue markets, environmental considerations and operational practicality. Residue can be left where it falls after harvesting, distributed (broadcast) over the soil surface, stacked in rows, mulched or incorporated into the soil (du Toit et al. 2004; Smith and du Toit 2005). The combination of residues and the previous crop litter layer on the forest floor is often referred to as “slash”. Harvesting intensity, often referred to as residue removal, can involve partial or complete removal of various tree components in addition to harvested stem wood. Further removals might also include the forest floor litter layer. Mechanical whole tree harvesting, off-site debarking, residue burning and fuel

wood collection can incorporate both high intensity harvesting and residue removal (Nambiar 2008; Saint-André et al. 2008; Tiarks and Ranger 2008).

Harvesting and residue management result in direct nutrient loss that occurs as a consequence of biomass removal and burning, as well as indirect loss through accelerated leaching, runoff and soil erosion following harvesting (Fisher and Binkley 2000; Pretzsch 2009). These losses are increased by whole-tree harvesting and off-site debarking, as further biomass is removed in addition to the stemwood (Nzila et al. 2002; Gonçalves et al. 2008a; Saint-André et al. 2008). The quantity of nutrients lost is dependent on the amount of biomass removed and the tree component that is removed; the more metabolically active components (leaves and bark) containing the highest nutrient concentrations followed by branches and stem wood (Dames et al. 2002; Nzila et al. 2002; du Toit et al. 2004; Blanco et al. 2005; Safou-Matondo et al. 2005; Saint-André et al. 2008; Dovey 2009). Often neglected in harvesting loss assessments is the timing of removal after clearfelling as some nutrients (potassium and phosphorus) may leach from biomass prior to collection if left to stand in field.

Burning of harvest residues includes combustion of the litter layer and surface soil organic matter. This can cause substantial nutrient losses through oxidation, volatilization, ash and particulate transport, followed by further leaching and erosion (Fisher and Binkley 2000). These losses increase as layer mass, nutrient content and fire intensity increase; which in turn depend on weather conditions during burning, residue size distribution and arrangement, and moisture content (Attiwill and Leeper 1987; Nzila et al. 2002; O'Connell et al. 2004; Nadel 2005; du Toit et al. 2010; Zavala et al. 2010).

Indirect nutrient losses occur from the soil, forest floor and harvest residues prior to and after planting. This is due to residue and soil temperatures and soil water content becoming elevated through direct sunlight exposure and minimal water and nutrient uptake. This results in more rapid residue decomposition and nutrient displacement leading to accelerated leaching (assuming adequate moisture), as well as volatilisation and soil erosion in specific areas (Morris and Miller 1994; Fisher and Binkley 2000; Fernández et al. 2004; Blanco et al. 2005; Gómez-Rey et al. 2008). Shading and uptake of water and nutrients by the newly planted crop gradually reduces the impact of these nutrient loss mechanisms.

The negative impact of large nutrient losses on soil nutrient and organic matter content is most important on infertile sites. This can reduce nutrient availability and pool size and manifest as a reduction in productivity in subsequent rotations and in the long-term (Spangenberg et al. 1996; Corbeels et al. 2005; Goncalves et al. 2007; Deleporte et al. 2008; Saint-André et al. 2008).

A network of international studies funded through the Centre for International Forestry Research (CIFOR) investigated the effects of residue management after clearfelling on subsequent productivity across tropical plantation forests. Eucalypts growth was reduced after residue removal on the low-fertility sites in Australia (Saint-André et al. 2008), Brazil (Gonçalves et al. 2008b), China (Xu et al. 2008) and the Republic of Congo (Deleporte et al. 2008). More fertile sites did not show the same productivity declines in Australia (Saint-André et al. 2008) or decreased only after complete residue and litter layer removal in South Africa (du Toit et al. 2008). However, growth decline occurred with little evidence of soil chemical changes diagnosed using conventional soil chemical analyses.

Despite evidence of nutrient related growth decline in international forests (Nambiar 2008; Saint-André et al. 2008; Tiarks and Ranger 2008), only a few studies in southern Africa have linked the loss of specific nutrients through certain harvesting and residue management practices with subsequent long-term productivity decline. This has been undertaken for *Pinus patula* grown on Gabbro derived soils (Morris 1986; Crous et al. 2008) and *Eucalyptus grandis* grown on shale/dolerite derived soils (Evans 1999; du Toit and Scholes 2002). Successive rotation tree growth, fertiliser response, foliar diagnostics, nutrient pool assessment and nutrient budget balances have been used separately or in combination as evidence of nutrient depletion and growth decline in each case. Early growth responses in both international and South African studies may have been due to the removal of the nutrient pool (residue and litter layer) that provided a large proportion of readily available nutrients during early growth at these study sites (Nambiar 2008; Saint-André et al. 2008; Tiarks and Ranger 2008). This increased short-term nutrient availability after clearfelling or burning that masked the long-term effects of the large nutrient losses has been termed the “assart effect” (Kimmins 1994). Data describing the ability of the various South African plantation sites and soils to supply the nutrient demand of trees is also limited to only a few studies (Morris 1986; du Toit and Scholes 2002; Crous et al. 2008).

Although some South African studies have associated nutrient loss data with subsequent growth and uptake into the new crop, they are limited to specific soils, do not fully describe all the nutrient loss and addition mechanisms and may confound the effects of climatic variation, genetics and site conditions with the interpretation of results when forecasting to subsequent

rotations. As a consequence it has been difficult to link this early growth reduction to long-term nutrient related growth decline as many study sites lack long-term evidence of a response.

2.5.4. Nutrient budgets

To be able to fully describe and understand these processes in the plantation forestry context, the interaction between management practices with plantation nutrient uptake and bio-chemical cycling of nutrients needs to be understood (Ranger and Turpault 1999; Laclau et al. 2005a). The benefit of such understanding is the potential to develop a budget of nutrient inputs and losses from a site, using a budget deficit to highlight nutrients that are being rapidly depleted (Ranger and Turpault 1999; Laclau et al. 2005a). A nutrient budget however, is only useful if viewed in relation to ecosystem pool size and nutrient demand and supply (Binkley 1986b; du Toit and Scholes 2002). This was shown in du Toit and Scholes (2002) where a nutrient budget was used with measures of available and potentially available nutrient pool sizes as an approach towards developing a nutrient sustainability index applicable to any forest system. The index has potential to highlight sites sensitive to nutrient decline under intensive management practices, provided data is available across multiple sites. The outcome of the budget therefore depends on the fertility of the soil relative to the budget outcome and tree rooting depth. Low or high soil fertility may also be predetermined by past agricultural practices, lowered through repeated burning or raised through fertilisation. Deep rooted trees will have access to a larger body of soil nutrients than shallow rooted trees. Certain tree species can also be better adapted to low soil nutrients than others and better able to tolerate nutrient stress.

Some recognition has been given to nutrient cycling, in South African plantations in the past (Morris 1992; Olbrich et al. 1997; Dames et al. 2002; du Toit and Scholes 2002; Louw and Scholes 2002; Lowman 2004; Nadel 2005; Champion et al. 2006; Dovey 2009), but comprehensive quantitative studies dealing with nutrient cycling are scarce and often limited to unpublished work. While this is primarily due to the financial and labour resources required for such work, it is also due to the difficulty in understanding the flux processes and obtaining the information required. Given these constraints, much of the sustainability research performed internationally has focused on the effects of nutrient removal on tree growth and soil properties. Nutrient budgeting, discussed in Ranger and Turpault (1999) highlights the need for and benefits of carefully constructed nutrient budgets that can be used for early warning of negative impacts of management practices on tree productivity.

2.6. CONCLUSIONS FROM THE GENERAL LITERATURE REVIEW

The productivity potential of a plantation forest is determined through the physical and biological interactions between the trees, climate, the chemical, physical and biological nature of soils, and soil depth (Waring and Ludlow 2008). These site quality factors qualitatively and quantitatively determine the supply and uptake of resources available for tree growth throughout the life of a plantation. Productivity and the allocation of tree growth to stemwood can change over the duration of a rotation and are altered by natural or man induced shifts in resource availability (Cannell and Dewar 1994; Stape et al. 2004). Productivity will be reduced when a tree growth resource becomes limited or imbalanced (Binkley et al. 2004; Ryan et al. 2008). Management practices that alter the supply, uptake, use or balance of tree growth resources will impact on productivity, while a permanent resource alteration will impact on long term sustainable productivity (Monteith 1977). It is therefore important to understand the factors that affect the quantity and rate of supply of resources to trees, factors that limit or enhance the uptake of the resources and the optimal balance that affords maximum growth in current and future rotations. The risk of these resources becoming degraded or depleted is dependent on site climatic and edaphic properties, tree species, management practices and the capture, storage and release (fluxes) of carbon and nutrients between the various atmosphere, soil and biological pools (Laclau et al. 2003a; Nambiar and Kallio 2008; Tiarks and Ranger 2008; Smethurst 2010).

Sustainable productivity research and management decisions are crucial to the survival of the forestry industry, as not only do they affect profits from wood yield, but impact on broader social, environmental and economic sustainability issues that ultimately constrain management decisions. Monitoring nutrient cycling processes through the hydrological, biological and soil components of a plantation grown during the intensive inter-rotational and early growth phase of a plantation is therefore necessary as it will enable a closer assessment of the impacts that silvicultural management practices have on soil and biological nutrient pools and fluxes of a site.

CHAPTER 3: SITE DESCRIPTION AND EXPERIMENTAL TREATMENTS

3.1. STUDY SITE

The site selected was located on the Zululand coastal plains (northern KwaZulu-Natal, South Africa) where extensive areas of sandy soil and sub-tropical climate result in areas of high productivity potential on soils with small nutrient reserves. The experiment was initiated at the end of 2007 in a 17.6 hectare compartment of seven-year-old clonal *E. grandis* x *E. camaldulensis* in the Siyaqhubeka owned Dukuduku plantation (**Figure 3.1**). The site was converted (circa 1955) from a mosaic of indigenous lowland coastal forest and grassland to commercial forestry. Using the climatic data of (Schulze et al. 1997), the study area is characterised by a relatively high and variable seasonal (summer) rainfall, (mean annual precipitation (MAP) of 920 mm), mean annual temperature (MAT) of 21.7°C, APAN reference evaporation of 1814.5 mm. This was in agreement with rainfall (**Figure 3.2**) and temperature (**Figure 3.3**) data observed over the past 50 years at a nearby weather station. Wind patterns are predominantly in north-easterly and south-westerly directions along the coast (Swewcuk and Prinsloo 2010). The site is 62 m above sea level, 15 km inland from the coast (Indian ocean) with its centre at 28° 17' 51" S and 32° 18' 55" E. Although the nearest perennial river is found approximately 6 km to east, the site is 1.3 km from nearest delineated wetland area, the centre of which is at 45 m above sea level. The slope of the land is relatively flat with an incline of 0.17°. Soils are deep (>30 m), free draining sandy soils (<5% clay) with a low organic carbon content (<1%). Some basic soil physical and chemical properties are presented in **Table 3-1**, showing the low soil organic C and fertility status, with low cation exchange capacity and high soil acidity of the sandy soils at the site. These data were collected at 16 points in a stratified grid across the experimental site using 10 cm steel coring tubes.

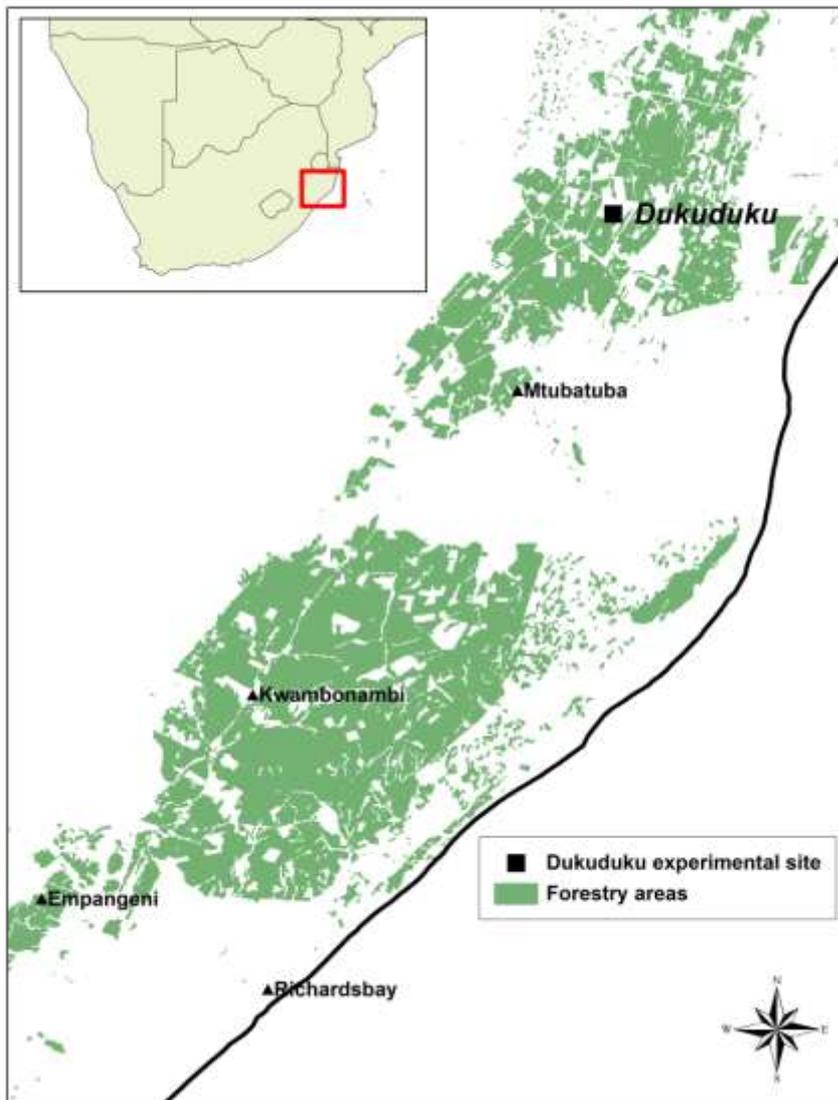


Figure 3.1: Map showing the location of the plantation area in southern Africa and the study site location within KwaZulu-Natal commercial plantation forestry areas.

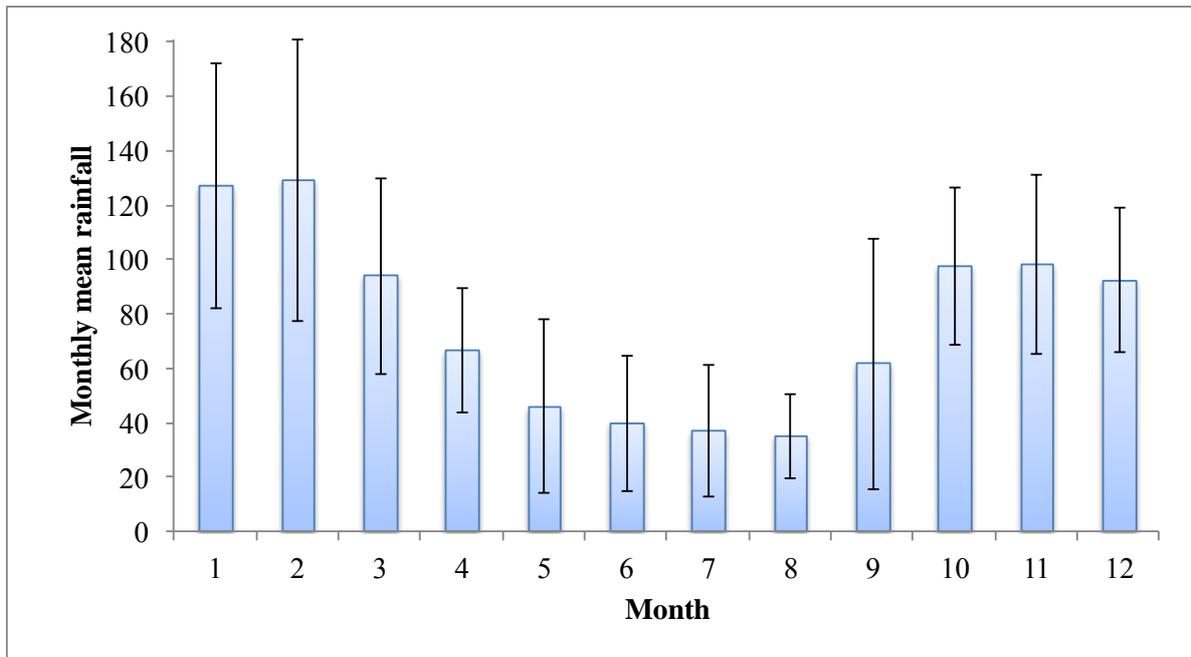


Figure 3.2: 50 year mean monthly rainfall observed from a nearby weather station. Single standard deviation is shown as I-bars.

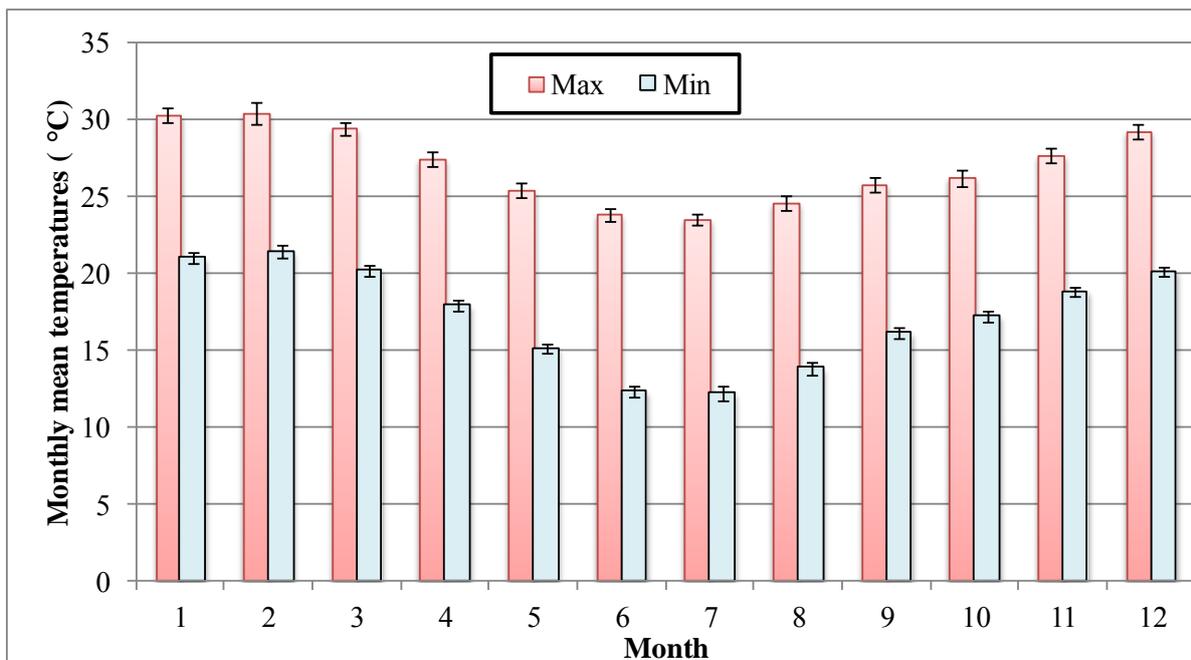


Figure 3.3: 50 year mean monthly maximum (Max) and minimum (Min) temperatures observed from a nearby weather station. Standard errors shown as I-bars.

Table 3-1: Basic soil physical and chemical properties of the top 100 cm (in 20 cm increments) at the start of the study and chemical properties at 50 cm increments thereafter

Depth (cm)	Bulk Density ($g\ cm^{-3}$)	Clay (%)	Sand (%)	Silt (%)	pH (KCl)	OC (g kg^{-1})	TKN (g kg^{-1})	Bray 2 P (mg kg^{-1})	K ⁺	Ca ²⁺	Mg ²⁺	Na ⁺ (cmol _c kg^{-1})	Exch. acidity	ECEC	Acid Sat. %
0-20	1.53	2.5	94.5	3.0	4.35	4.3	0.39	2.61	0.02	0.45	0.20	0.03	0.31	1.01	25
20-40	1.55	2.3	95.3	2.4	4.31	2.7	0.37	1.6	0.03	0.46	0.22	0.03	0.31	1.05	29
40-60	1.57	2.2	96.0	1.8	4.33	2.1	0.28	1.49	0.04	0.49	0.24	0.04	0.33	1.14	32
60-80	1.59	1.8	95.9	2.3	4.36	1.8	0.35	1.95	0.03	0.91	0.28	0.04	0.36	1.62	34
80-100	1.61	2.0	96.2	1.7	4.37	1.6	0.30	3.04	0.03	0.51	0.23	0.04	0.36	1.17	36
100-150	1.61				4.56	1.75	0.31	0.72	0.03	0.31	0.22	0.05	0.36	0.97	40
150-200	1.64				4.69	2.14	0.36	0.71	0.04	0.12	0.14	0.03	0.21	0.53	35
200-250	1.67		nd		4.71	1.12	0.31	0.70	0.03	0.00	0.11	0.02	0.16	0.31	36
250-300	1.64				5.01	0.83	0.31	0.71	0.03	0.44	0.13	0.02	0.12	0.73	28
300-350	1.42				4.49	0.49	0.21	0.78	0.12	1.50	0.66	0.06	0.27	2.61	17

OC (WB): Organic carbon (Walkley-Black); TKN: Total Kjeldahl nitrogen; nd: not determined

Standing crop prior to clearfelling

The clonal *E. grandis* x *E. camaldulensis* (GxC) trees were originally planted at a spacing of 3 m x 3 m in a rectangular compartment (92 x 213 trees) that was chosen as having its long axis as close to a north/south orientation as possible (**Figure 3.4**). Tree volume at 7 years was derived from tree height and diameter at 1.3 m above ground level (dbh) using tree volume and taper functions of Morley and Little (2011) as 143 m³ ha⁻¹ for 1 102 trees ha⁻¹ (99% survival). The site index at base age 5 (SI₅, 80th percentile of tree heights projected to 5 years of age) for this compartment was estimated as 19.9 using the data of Coetzee (1992). Site index, a tree growth-based measure of site quality, was within normal ranges typical for this area (Smith et al. 2005b). The mature tree canopies in this stand (and trees of the entire region) were also infested with *Thaumastocoris peregrinus*, peaking during the summer months of January to March each year (Nadel et al. 2010). This insect pest reduces photosynthetic activity in the canopy and may cause increased litterfall (Nadel et al. 2010). Evidence of *Gonipterus spp.*, a leaf-eating insect that reduces leaf area and apical growth, was also found.

3.2. SETUP OF EXPERIMENTAL AREA

At the end of 2007 the experimental design was overlain onto the entire standing crop compartment as zones for felling and zones to be left undisturbed as standing crop areas (**Figure 3.4**). Installation of equipment commenced a full year prior to felling, allowing adequate time for soil and equipment stabilisation. Photographic images of various equipment installations, events and the site in general are given in **Appendix 3.4 (Image 3-1 to Image 3-18)**.

3.2.1. Treatment implementation

Treatments were applied to the study site after clearfelling, as a four replicate randomised complete block design with two replicates of un-felled trees forming a standing crop treatment (**Figure 3.4**). The standing crop areas were delineated as 46 x 92 tree (138 x 276 m) zones at the northern and southern end of the compartment with a surrounding buffer zone of 51 m width (17 trees) which was clearfelled (**Figure 3.4**). Standing crop plots were delineated in central positions of the northern and southern areas, with buffer zones of 17 trees. The north/south orientation and the large buffer areas were designed to minimise the intrusion of additional shading and/or solar penetration between felled and standing crop plots. The central portion of the compartment was felled, and divided into 36 square experimental plots 42 x 42 m, each with an internal 15 x 15 m plot used for tree growth measurement. Felling of the buffer zone began in September 2008. The buffer areas were used as extraction routes for the main felling operation at the end of November 2008. Timber was manually stacked in early January 2009 and mechanically extracted.

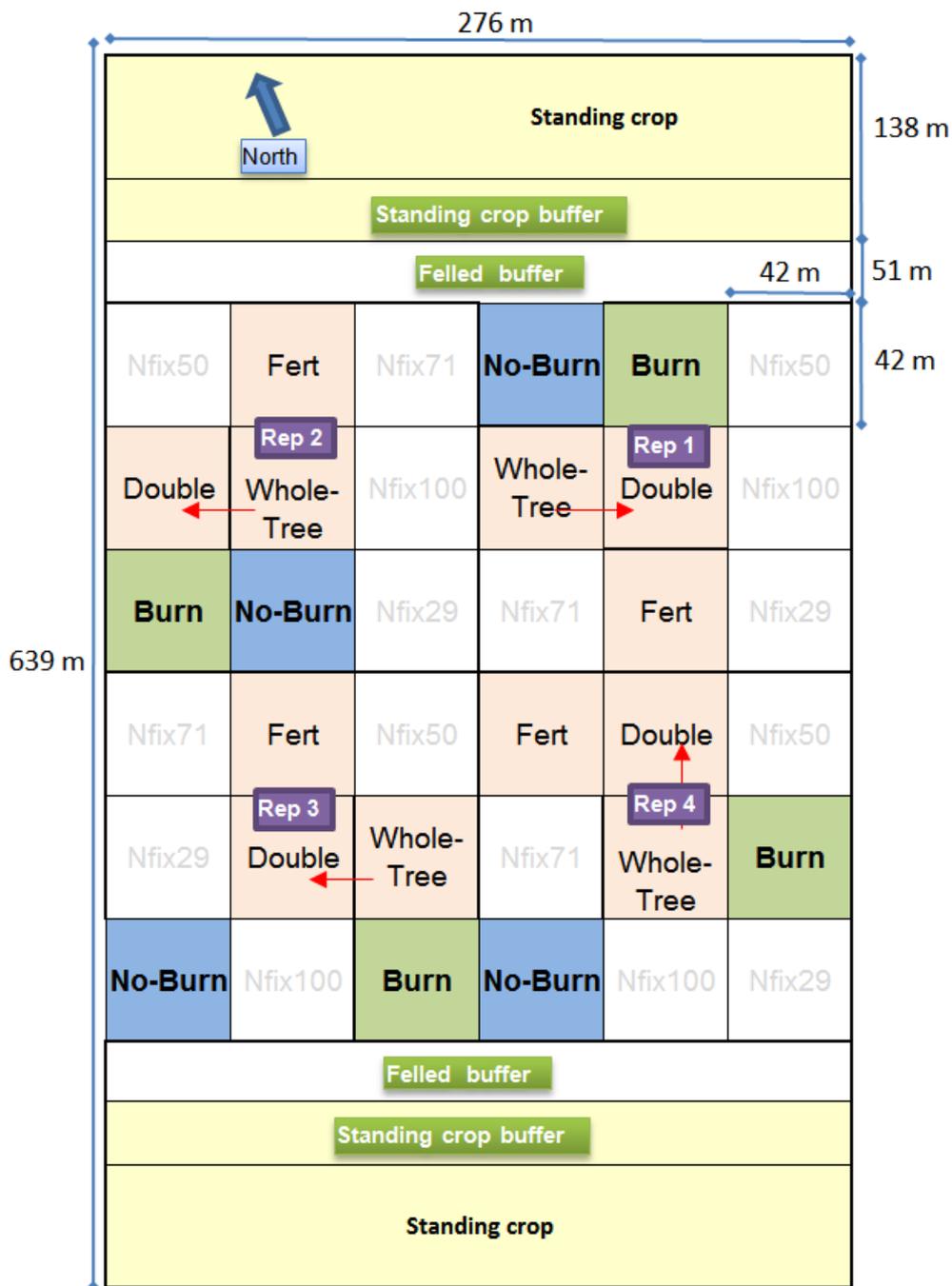


Figure 3.4: Experimental layout of treatment plots showing the standing crop zones and internal felled Burn and No-Burn treatment plots with dimensions given in metres. Additional treatment codes apply to further components of this study not reported here (Whole-Tree as whole tree harvest, Fert. – No-Burn plus fertiliser replacement, Double - double residue layer).

Residues were manually broadcast across the entire felled area over the old crop litter layer after clearfelling, except for the whole tree harvesting treatment (Whole-Tree) from which whole trees were removed, leaving the litter layer intact. The remaining treatments included a single residue layer (No-Burn), a double residue layer (Double) (created using the residues removed from the Whole-Tree plots), a broadcast and burn treatment (Burn) and a macro-nutrient fertiliser replacement treatment (Fert). Fertiliser was used to supply a majority of macro nutrients lost

with stemwood removal during harvesting calculated using seven year old tree stem data from the biomass sampling described later in this chapter. Fertiliser was applied shortly after planting to each tree in a 15 cm radius ring. Granulated fertiliser was applied per tree was 179 g limestone *ammonium* nitrate (28), 128 g superphosphate (10.5), 102 g KCl (50), 140 g CaNO₃ (19, 15.5) and 94 g of MgSO₄ (10). This added N, P, K, Ca and Mg to the soil in amounts of around 120, 22, 85,68, 16 kg ha⁻¹ respectively. The Burn treatment was implemented in early March 2009 on a hot cloudless day under light, windy conditions to induce a rapid complete burn. Weed and coppice re-growth was initially manually slashed, then eliminated using *glyphosphate and* Triclopyr. The site was manually pitted (to 30 cm depth using a mattock) in July 2009 and planted at a 3 x 2 m espacement to a *G x C* clone (GC 514) in accordance with a company policy change on spacing.

Summary of treatments and acronyms:

- **Standing crop** (SC) Undisturbed crop – no harvest
- **Whole-Tree** (W) Removal of stem wood, bark, branches and foliage at clearfelling
- **Burn** Residues broadcast and burned
- **No-Burn** Residues broadcast and retained
- **Fertilised** (Fert) Residues broadcast and retained, Replacement of macronutrients removed with stem wood
- **Double** (2S) Residues broadcast and retained with addition of an extra residue load that had been removed from Whole-Tree plots.

The treatments selected for intensive process monitoring in this study were the Burn and No-Burn (residue retention) treatments, as these were considered to represent the most extreme practices of the typical residue management operations, and the standing crop areas were used to represent the site without a harvesting disturbance.

3.3. DATA COLLECTION

3.3.1. Weather Data

A fully-automatic Campbell Scientific weather station was installed in an adjacent open area to record rainfall, temperature, relative humidity, solar radiation, wind speed and wind direction. Data were collected at hourly intervals and summarised daily.

3.3.2. Water fluxes

Water fluxes were monitored as precipitation, canopy drainage (throughfall and stemflow), soil moisture content and drainage to 100 cm below the soil surface throughout the felling, residue management, and re-establishment phases of the plantation cycle until the newly planted trees were 1.5 years old.

3.3.3. Rainfall throughfall and stemflow equipment

Three white plastic funnels (14.1 cm diameter) were installed at 1.2 m above the ground (4 replicates) in the felled plots to collect precipitation, and similarly in the standing crop plots to collect throughfall (**Figure 3.5**). The tree diameter at breast height (dbh, i.e. measured 1.3 m above ground level) was used to construct the size class distribution, ranging between 13 and 18 cm. The majority of trees (75 %) were between 15 and 17 cm in diameter. The distribution of tree size classes was considered in selecting 4 x 4 tree plots into which the monitoring equipment was installed. This was done to capture the heterogeneity of the tree canopy while representing the full tree diameter distribution at each sampling site. This design was utilised considering the uniform canopy of the clonal *Eucalyptus* crop, the intended use of the samples in water quality assessments and acceptable error limits of the design (Kimmins 1973; Lawrence and Fernandez 1993; Stape et al. 2004; Mululo Sato et al. 2011). Water drained from the funnels via polyethylene tubing was collected and stored in 5 ℓ white high-density polyethylene (HDPE) bottles. A small piece of plastic mesh was placed in each funnel to prevent detritus from entering the tubing. The bottles were collectively housed in a 25 ℓ white HDPE bucket, buried to protect the bottles from heat and sunlight, with only the lid exposed. Four size classes of tree diameter were selected using the tree diameter distribution, and a single tree was selected from the midpoint of each class, onto which stemflow collectors were installed. The stemflow collectors (**Figure 3.6**) were slotted 20 mm polyurethane tubes attached to each of the 4 trees

draining into separate 5 ℓ bottles contained in a partially buried 30 ℓ bucket (Levia and Frost 2003). The tubes spiralled each tree twice from 0.6 to 1.4 m above ground level, held fast to the tree with plastic packaging belt threaded through the tube. Contact between the tree and tube length was achieved using a 100 % silicone sealant formulation.

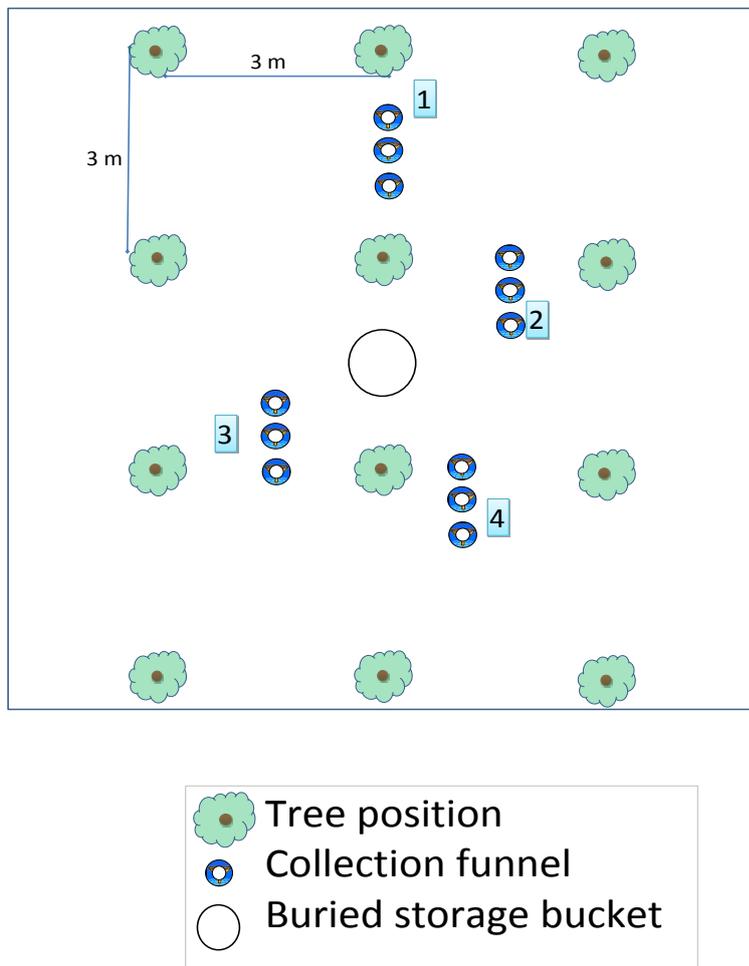


Figure 3.5: Layout of a funnel cluster and storage container relative to tree positions.

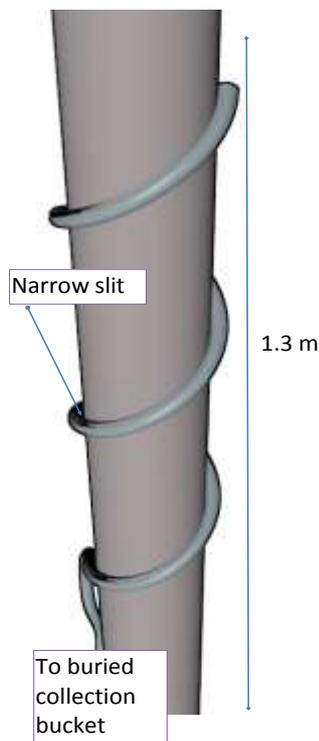


Figure 3.6: Side view of a narrowly slit stemflow collection tube attached to a tree stem.

3.3.4. Lysimeter installation

A crescent-shaped trench was excavated to a depth of 1.2 m to allow measurement of soil moisture status and flux on the site (**Figure 3.7**). The trench was situated between the trees so that the wall of the pit gave access to the inter-row of 6 trees, the front of the crescent facing 2 trees, the arms turning diagonally across the intra-row of 2 trees each. This was done to allow access to undisturbed soil at the pit face while allowing the pit face to cover as much soil and tree size variability as possible. Four replicates of zero tension plate lysimeters (30 cm x 30 cm) with 3 cm walls on 3 sides were inserted horizontally at depths of 15 cm, 50 cm and 100 cm, into the wall of a trench, by opening a narrow envelope into the pit face at each depth. Horizontal distances between plates were maintained at no less than 60 cm. This installation was repeated in each treatment block.

The lysimeter plates were used as a cost-effective method to collect a sample of drainage water for water quality diagnosis, although they are known to have a collection efficiency of about 10% (Weihermuller et al. 2007). This was assumed to give an integrated sample of gravity-drained soil solution, but necessitated the prediction of actual drainage using the soil moisture flux model described later. The depths chosen here were intended to reflect normal fine root length and mass density distributions characteristic of the soil under *Eucalyptus*. Highest

densities in clonal *Eucalyptus* have been shown to occur in the first 15 cm of soil, decreasing markedly with depth to 50 cm and gradually thereafter until very few fine roots are found (Knight 1999; Laclau et al. 2001; Gonçalves and Mello 2004; O'Grady et al. 2005). Prior to insertion, soil from the envelope was placed as slurry into the plate, the plate pressed to the top of the envelope to make best contact, and the gap under the plate was filled with the remaining soil. The plates were installed at a slight angle to allow for drainage to a corner plug connected to a polyurethane tube. A trench dug from the centre of the crescent led into a pit excavated to 1.4 m, which allowed the 4 plates at each depth to gravity drain to plastic collection bottles. The crescent trenches and inter-leading trenches were carefully backfilled ensuring no disturbance of the area above the installed plates. The walls of the access pits were reinforced with timber and the pits were covered with a corrugated lid to offer protection from sunlight and rainfall. A second soil solute collection system was installed as suction cups between each plate. These were held at a constant vacuum of -50 bars using a vacuum pump regulated through vacuum switches and Campbell CR10x data loggers. This system failed to collect a sufficient soil solution for analysis and has been excluded from this study.

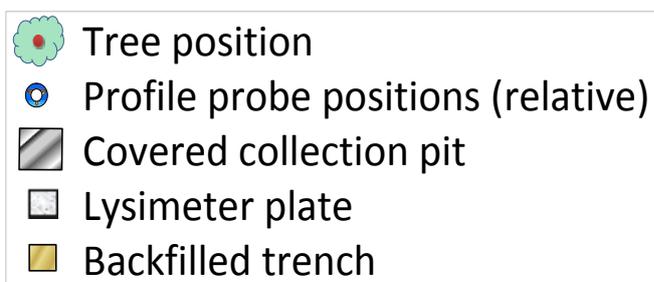
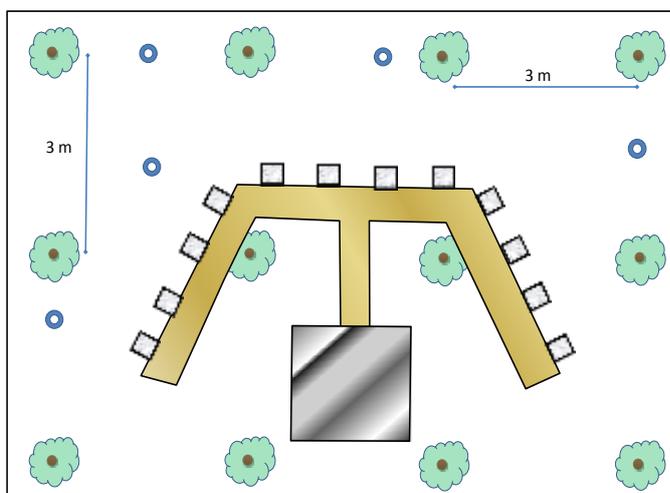


Figure 3.7: Top view layout of trench and storage pit relative to tree positions, showing lysimeter plates that are alternatively buried 15, 50 and 100 cm. Profile probe positions shown relative to trees.

3.3.5. Water Sample collection and analysis

Water collection began in December 2007, with weekly collection until early 2011. Sample volume was recorded and sub-samples were decanted into 250 ml Teflon bottles and transported in a cool box. The sub-samples were assessed for pH and electrical conductivity (EC), refrigerated at around 5°C for no more than four weeks and bulked through weighted bulking prior to chemical analysis. Weighted bulking, representing four weeks sample collection, was achieved using the fraction of each weekly volume over the total four week volume. Bulking was necessary to reduce laboratory analytical costs. Measurements of pH and EC were repeated after bulking.

Water samples were analysed for organic and inorganic N as Total Kjeldahl-N [4500-N_{org} method (Clesceri et al. 1998)], ammonia N (NH₄⁺-N) and nitrate (NO₃⁻-N) [flow injection (Patton and Crouch 1977; APHA 1995)], the difference between Kjeldahl-N and (NH₄⁻-N + NO₃⁻ N) was assumed to be organic N (O-N). The TKN method included a modification where salicylic acid was added to the catalyst mixture. This converts NO₃-N to nitric acid which nitrates the salicylic acid. The nitro-compounds produced were then reduced to ammonia and distilled by adding excess alkali. Included in the analyses were the cations K⁺, Ca²⁺, Mg²⁺ and sodium (Na⁺) [Auto analyser, (EPA 1984)]. Phosphorous was initially determined, but later abandoned due to cost and laboratory equipment failure. Cost also excluded the analysis of sulphur and other micronutrient elements. Field blanks (passing deionised water through the field equipment) and standard samples were submitted for analysis to ensure analytical accuracy and test for field contamination.

The volumes of water collected were used with funnel area to calculate precipitation and throughfall in mm. Stemflow was calculated (mm) using the volume data for each tree size to predict volume collection by all trees and scaling this back to the area represented by the trees on the stand. Total canopy drainage was calculated as the sum of stemflow and throughfall. Solute concentrations were multiplied with total volume collected by each collector for each rainfall event to derive masses of elements contained in rainfall, throughfall and stemflow.

Concentrations that were unreasonably high relative to the other ion concentrations (particularly K^+ and organic-N for large rainfall events) were excluded as possibly contaminated.

3.3.6. Soil moisture monitoring

A soil capacitance profile probe was used to measure volumetric soil moisture content at weekly intervals at 10, 20, 30, 40, 60, 80, 100 cm below the soil surface via 5 access tubes installed into each plot. A micro-topography was apparent over the entire site, possibly caused by lifting of soil near to trees, relative to the soil level of the spaces between trees (5 to 10 cm amplitude). The access tubes were therefore placed within and between the tree rows to represent the variability of soil moisture in each plot (**Figure 3.5C**). Time-domain reflectometry (TDR) probes were installed in one of each of the treatment plots at 15, 50, 75 and 100 cm, automatically recording volumetric soil moisture content at hourly intervals via a custom-built data-logger. Soil samples for gravimetric moisture analysis (oven drying at 105°C until constant mass) were collected periodically to validate soil moisture contents measured with the electronic devices

3.3.7. Soil chemical properties

Soil chemical analyses were undertaken at the start of the study, just prior to felling and again at the end of the study when the new trees were 1.5 years old. Soils were collected from five points in each plot using a PVC core that was driven into the soil. Soils were separated into incremental layers to represent layers sampled by the lysimeter plates (0 - 15 cm, 15 – 50 cm and 50 – 100 cm) and bulked across each depth for each plot. Additional samples collected monthly as 0 – 5 cm, 5- 15 cm and 15 – 30 cm layers using the same methods above were included in the analysis. The samples were air dried and passed through a 2 mm sieve and analysed for pH, N, P, K Ca, Mg and Na using the procedures described in Hunter (1975), Farina (1981) and Donkin et al. (1993). Nitrogen was determined using the Kjeldahl method (Nelson and Sommers 1980; Donkin et al. 1993; Mulvaney 1996). Bray-2 extractable P was determined using the Bray and Kurtz (1945) methodology, the filtrate automatically analyzed colorimetrically at 880 nm using a segmented flow autoanalyser (SAN^{plus} SYSTEM; Skalar Analytical, Breda, The Netherlands) with ascorbic acid colour development (Murphy and Riley 1962). Cation (Ca, Mg, K and Na) extractants (Donkin et al. 1993) and Helmke and D.L. (1996) diluted with an ionisation suppressant (strontium or caesium) were determined by atomic absorption spectrophotometry

(AAS; SpectrAA-10, Varian Techtron Pty. Ltd., Mulgrave, Victoria, Australia). Soil pH was determined in 1: 2.5 soil: 1M potassium chloride solution ratio (Thomas 1996) with using a standard glass electrode (Metrohm Hershau E396B; <http://products.metrohm.com>).

3.3.7.1. Water repellency samples

Soil samples collected to assess N mineralisation described in CHAPTER 7 were used for a further determination of soil moisture and water repellency. A simple semi-quantitative assessment of water repellency was conducted using a water dropper to apply 6 droplets of deionised water to a smoothed surface of a sub-sample of the air-dried soils collected from each of the treatments. The time taken until the water drops were absorbed (infiltration) into the soil was recorded and the average time used as an indicator of water repellency (Letey et al. 2000; Scott 2000).

3.3.8. Growth and biomass sampling

Tree growth was determined at six monthly intervals after planting (August 2009) up to two years and six months after planting. Tree height and ground line diameter (*gld*) were initially recorded in the new crop followed by diameter at 1.3 m above ground level (*dbh*) once the trees were of sufficient size for these measurements. Above ground biomass and nutrient contents were initially determined at six months, one and two years after planting. The two year-old sampling only included biomass data as nutrient data was not available at the time of publication. Canopy closure was approximated at the point where canopies of adjacent trees started to overlap and occurred at about one year after planting.

3.3.9. Stand biomass and nutrient measurements

Aboveground tree biomass was estimated using 20 destructively harvested trees across the entire site. Destructive harvesting utilised a *dbh* size class distribution, which was divided into five size classes from which trees were selected to represent the full range of tree sizes across the site. Trees were selected from each treatment in the new crop to capture nutrient concentration differences, while reducing analytical costs. Harvested trees were separated into foliage, branches (live and dead), bark and stem wood. The initial harvest in the new crop was separated

into foliar and non-foliar (woody) material (stem, bark and branch). Field wet mass of each component and subsamples moisture contents after drying to constant mass at 60°C were used to derive the total biomass of each tree component. Additional wood disk samples were used to determine wood density by mass displacement in water. Specific leaf area (*SLA*) was determined by scanning leaves from each tree on a flat-bed image scanner with a 5 mm calibration grid. Analysis of individual leaf area was performed on the scanned images using ImageTool 3.0 software (UTHSCSA 2002). Oven dry mass and area of a batch of leaves was used to calculate *SLA* which was multiplied by total foliar dry mass to derive canopy leaf area.

Stand level biomass and leaf area index (*LAI*, m² m⁻²) was estimated from single tree biomass components using correlations with *gld* and *dbh*. A polynomial function produced the best predictive equations with *dbh* in the mature crop, while a natural log transformed *gld* and *dbh* gave the best fits for all log transformed tree components of the new crop. All models had R² values greater than 0.96. Statistical probabilities for predictive equations were all significant (p < 0.001), as were all predictive parameter estimates (p < 0.001). Growth efficiency was calculated in the old and new crop with respect to incremental above ground woody biomass production per unit *LAI* and expressed thus: Mg ha⁻¹ yr⁻¹ LAI⁻¹. *LAI* was taken as the mean value between biomass assessment points. Treatment-specific allometric relationships were also tested for nutrient contents, but lacked significance and did not improve the predictions.

3.3.10. Litterfall and residue

Standing crop litterfall was collected weekly using 50% shade net held at 15 cm above the ground in steel framed (1.0 x 1.5 m) litter traps. Five litter traps were installed in each treatment positioned at various distances from the trees. Litter was separated into foliar, branch and bark components and bulked four weekly. Litterfall was not assessed in the newly planted crop as litter fall was negligible up to canopy closure and tree canopies were also too low for effective litter collection. Forest litter layer and residue biomass was collected at four weekly intervals using a metal ring (34 cm diameter) across five random points in each treatment plot. A loss on ignition (*LOI*; samples ashed at 450°C for 12 hours) was used to estimate mass without of soil contamination. The same ring method and analyses were used before and after burning to assess biomass and nutrient loss after burning. In addition, 10 steel plates (40 x 40 cm) were inserted under the residue (on the soil surface) prior to burning to collect ash remaining after burning.

This additional method was attempted to avoid soil contamination. Annual decay rate constants (k) were calculated from percent residue biomass, using a single negative exponential decay model $X_t/X_0 = e^{-kt}$ where X_t is the residue biomass remaining at time t , X_0 is the initial residue biomass and k is the monthly decay constant Olson (1963).

Dried subsamples from each tree component, litter, residue and ash were individually ground and homogenised then analysed for nutrient concentration (N, P, K, Ca, Mg, Na) using the methods described in Kalra and Maynard (1991) at the Cedara laboratories in South Africa. Nutrient concentrations were assessed in four weekly bulked litterfall and residue. Nutrient concentrations for each tree component in each treatment were used with total stand component biomass to derive nutrient content per hectare. Residue, litter and ash nutrient contents were calculated without LOI adjustment. The quantity of nutrients released into the soil due to decomposition was assumed to be the difference between nutrient content at each time of assessment. Nutrient release in the standing crop was calculated as the sum of change in forest floor nutrient content and litterfall nutrient content.

Vector analyses after Weetman (1989) developed further in Salifu and Timmer (2001) were performed for nutrient contents and concentrations in the foliage using the No-Burn treatment as the relative control. The weighted mean concentration was calculated from foliar nutrient content and foliar mass. (Valentine and Allen 1989) described this method as a more reliable method to predict growth response to fertilisation than foliar diagnostics alone as it considers both concentration and growth in the assessment. This method, for example, accounts for the confounding effects of nutrient dilution that may occur after a fertilisation induced growth response.

3.3.11. Statistical analysis

The effect of the treatments on aboveground biomass, nutrient content, and accumulation and growth rates at each age were compared by general ANOVA, with least significant difference ($LSD_{5\%}$) used to show treatment differences where significance was found. As only two reps could be produced for the standing crop treatment, two reps were treated as missing data. Tests for normality using the Shapiro-Wilk test and homogeneity of variance using the Bartlett's test

were performed on the data where applicable. Differences in concentrations were tested using a Student's t-test. All statistical analyses were performed using Genstat[®] for Windows[™] 12th Edition (Payne et al. 2011).

3.4. APPENDIX OF PHOTOGRAPHIC IMAGES



Image 3-1: The sandy soil and litter layer



Image 3-2: Wood structure preventing soil collapsing into pit



Image 3-3: Plates and suction cups inserted into pit face



Image 3-4: Slotted stemflow collector



Image 3-5: Automatic weather station in nearby clearing



Image 3-6: Wood disk samples collected for mass, nutrient and density determination



Image 3-7: Standing crop litter pre-clearfelling



Image 3-8: Clearfelling of the buffer strips



Image 3-9: Clearfelled compartment showing standing crop in background



Image 3-10: Opening a soil pit cover



Image 3-11: Whole-Tree plot surface after clearfelling



Image 3-12: Residue burning



Image 3-13: TDR probes inserted into the Burn treatment the day after burning



Image 3-14: Soil N-min cores in Burn treatment



Image 3-15: Ash layer in soil at four weeks after burning



Image 3-16: A Burn treatment plot at six months after planting



Image 3-17: A No-Burn treatment plot at six months after planting



Image 3-18: A Whole-Tree treatment plot at six months after planting

CHAPTER 4:
**A COMPARISON OF SOIL MOISTURE RELATIONS BETWEEN STANDING AND
CLEARFELLED PLOTS WITH BURNT AND UNBURNT HARVEST RESIDUE
TREATMENTS¹**

4.1. ABSTRACT

The effects of clearfelling and subsequent residue retention or burning on water and nutrient balances needs to be understood and quantified on forest sites that are sensitive to loss, so that the long-term sustainable productivity of such sites can be maintained and promoted. An experimental site was established in a clonal *Eucalyptus* compartment on the Zululand Coastal Plain, to compare changes in water fluxes through the mature undisturbed *Eucalyptus* stand with those after felling and re-planting, under 2 conditions: burning, and retention of the harvesting residues. The study was located in an area of high rainfall and high stand productivity, with sandy soils and low soil carbon and nutrient status; chosen so that the effects of intensive demands on water and nutrient fluxes on a potentially sensitive site could be investigated. This chapter presents only the hydrological component of the study. Data collection included weekly determination of rainfall, throughfall, stemflow and soil moisture fluxes from the surface to a depth of 1 m. Drainage rates through the profile were established using time domain reflectometry probes while water drainage volumes were assessed using shallow plate lysimeters. Despite slow growth in the standing crop during the monitoring period (attributed to a pest infestation), soil moisture depletion remained rapid and drainage below 1 m remained low. Soil moisture was recharged within a few months after clearfelling, but became rapidly depleted as the canopy of new crop developed and approached canopy closure. A decreased wetting-front velocity and a marginally higher field capacity were proposed as evidence of pore clogging that appeared to occur during the inter-rotation period. The soil profile under the unburnt residue maintained a marginally higher soil moisture status and lower drainage than the soil profile under the burnt residue. Although soil moisture and drainage in the burnt and unburnt residue treatments became similar to the standing crop from canopy closure onwards, rainfall additions to soil moisture were depleted faster under the new crop during the first few months after canopy closure. Small differences in soil moisture status between the burnt and unburnt residue treatments presented here may not be sufficient to influence residue management decisions. The length of the inter-rotation period and practice of residue burning may, however, need

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consideration where soil carbon and nutrient loss or displacement may negatively affect the sustainability of the site.

4.2. INTRODUCTION

Water use under commercial plantation forestry has been extensively studied in South Africa with the outcomes used to formulate guidelines for water use charges and regulations (Dye and Versfeld 2007). These regulations, an increased demand for wood products, and numerous other political, economic and environmental factors, have pressurised the industry to seek innovations to enable larger quantities of timber and biomass to be produced from a decreasing land base (FSA 2009b). The loss of land area, 9% between 1999 and 2009 (FSA 2009b) has occurred through, *inter alia*: wetland delineation, successful land-claims that have subsequently resulted in land being converted out of forestry, and the establishment of wildlife corridors through plantation forest areas. Increases in productivity of 19%, for the same 10-year period, have been realised through the selection of faster growing trees, improved matching of tree species to sites for optimal timber growth, planting of disease-resistant clonal varieties, shorter rotation lengths, more intensive silvicultural practices, and mechanisation. This has also substantially increased the rate and quantity of biomass removal per unit land area of plantation forests and consequently increased pressure on water and nutrient resources on the remaining land area. It is therefore crucial that the negative impact of a greater demand on water be understood and quantified so that long-term sustainable productivity can be maintained and promoted.

Soil moisture is rapidly utilised under forestry with little evidence of long-term soil or ground water recharge under full-canopied *Eucalyptus* crops, as dynamic equilibrium is maintained between stomata and leaf area index controls of evapotranspiration and soil moisture availability (Kienzle and Schulze 1992; Dye 1996; Laclau et al. 2001). Tracking the processes within the hydrological component of a commercial plantation forest requires knowledge of the magnitudes of the water fluxes, tree water use, and the impact of site management on these fluxes. Site management at harvesting, and the conservation, displacement or loss of harvest residues through removal or burning during the inter-rotation phase, can potentially impact a site. As the hydrological cycle is a key driver of forest nutrient fluxes and cycles, the effect of clearfelling and subsequent residue management on soil moisture and drainage may impact on water availability to the subsequent crop and nutrient loss or displacement from the active fine

root zone (top 50 cm of soil) (Smethurst and Nambiar 1990; Carlyle and Nambiar 2001). This is in addition to nutrient and carbon losses that occur through harvesting and residue burning. Water repellency of soils, which is common under *Eucalyptus* plantations, may be increased with residue burning, reducing infiltration rates, increasing surface runoff and soil erosion on sloped lands (Scott and Van Wyk 1990; Scott 1993; DeBano 2000; Scott 2000).

The deep sandy soils of the Zululand coastal ecosystem (Northern KwaZulu-Natal, South Africa) are low in clay, organic carbon content, nutrient and water storage capacity, and have high drainage rates (Hartemink and Hutting 2005). These soils are unable to buffer chemical, physical and organic (biological) changes, placing them at risk of degradation, ultimately limiting or reducing tree productivity under poor site-management practices. The sustainability of management practices on sites which have high productivity but low soil moisture and nutrient storage capacity is increasingly questioned, particularly under currently prescribed residue-burning operations.

A study was initiated to compare soil moisture content and drainage volumes (using *in situ* measurement techniques), in a mature clonal *Eucalyptus* stand, with those after felling, and upon which residue burn and no-burn management practices were imposed, and for the first 18 months of new crop growth. In addition, a soil moisture model (Hydrus 1D) was calibrated with soil moisture content measured at each depth to verify the predictive ability of the model used. This study forms part of a larger study that aimed to determine the impact of site management on long-term sustainable productivity with respect to natural and management-induced nutrient losses and gains. The model will be discussed in future publications as a tool to predict drainage fluxes. Water fluxes will be multiplied by measured dissolved nutrient concentrations to estimate nutrient fluxes due to gravity drainage, in a similar manner to that used by Laclau et al. (2007).

4.3. MATERIALS AND METHODS

Materials and methods for this chapter that are common to other chapters are given in CHAPTER 3. Methods used in soil moisture content are given here.

4.3.1. Model prediction of water moisture and drainage

The 1-dimensional Hydrus 1D soil moisture model (Šimůnek et al. 2008) was parameterised for the study site and used to predict daily soil moisture content and drainage at each depth. Hydraulic conductivity parameters were estimated using a neural network prediction after Schaap and Bouten (1996) with matrix potential measurements of a similar soil taken from Rietz (2010) and soil textural fractions and bulk density determined from samples taken at the site (**Table 4-1**). Soil hydraulic parameters are presented in **Table 4-1** as residual and saturated soil moisture content (Q_r , Q_s , respectively), parameters in the soil moisture retention function ($Alpha$, n), and saturated hydraulic conductivity (K_s). The van Genuchten-Mualem soil hydraulic model was used as the hydraulic conductivity component of the model (van Genuchten 1980). Evapotranspiration was estimated using the Penman-Monteith equation for the model and meteorological data collected at the site, and also evaluated using TDR data recorded at the site. Leaf area index recorded at the site through destructive methods was allowed to fluctuate according to optically-determined LAI-2000 measurements and values given in Dye et al. (2004). A radiation extinction coefficient of 0.4 and an albedo of 0.2 were used after Stape et al. (2004) for the standing crop trees. An albedo of 0.3 was used during the fallow period in the felled area (ten Berge 1986). Rainfall was adjusted for interception using a relationship developed between weekly rainfall and weekly canopy drainage from data collected at the site. Interception of rainfall by the litter and residue layers was estimated for the standing crop tree litter layer and the felled crop residue layer using relationships between residue mass and interception, after Paul et al. (2003). Root distributions were taken from literature-derived values (Knight 1999; Laclau et al. 2001; Nouvellon et al. 2002; Gonçalves and Mello 2004; O'Grady et al. 2005) and were validated from cores driven into the soil to a depth of 350 cm at 5 points in each of the 6 plots. Roots separated from the soil were dried and weighed to determine root mass per soil volume in layers of 0-5, 5-15, 15-30, 30-50, 50-100, and 50 cm increments thereafter, to 350 cm. A root water uptake model Feddes et al. (1978) was used to simulate root water uptake assuming transpiration to range between 1 and 6 mm per day (Dye 1996; Nouvellon et al. 2002).

Table 4-1: Mean values of soil organic carbon, texture, and bulk density taken at incremental depths and soil moisture retention and conductivity parameters

Sample depth (cm)	Organic carbon %	Clay %	Sand %	Silt %	Bulk Density g cm^{-3}	Q_r $\text{cm}^3 \text{cm}^{-3}$	Q_s $\text{cm}^3 \text{cm}^{-3}$	α cm^{-1}	n -	K_s cm day^{-1}
0-20	0.43	2.5	94.5	3.0	1.53	0.0373	0.3778	0.0499	2.4853	728
20-40	0.27	2.3	95.3	2.4	1.55	0.0374	0.3732	0.0498	2.4657	733
40-60	0.21	2.2	96.0	1.8	1.57	0.0375	0.3683	0.0497	2.4473	719
60-80	0.18	1.8	95.9	2.3	1.59	0.0376	0.3628	0.0496	2.4297	688
80-100	0.16	2.0	96.2	1.7	1.61	0.0375	0.3569	0.0495	2.4129	639
100-120	0.15	2.6	96.2	1.2	1.64	0.0374	0.3505	0.0493	2.3969	572

4.4. RESULTS

4.4.1. Rainfall and canopy drainage

Annual rainfall for the first 2 years of the study was below the site long-term average (920 mm). Rainfall measured between October and October of each year was 696 mm for 2009, 721 mm for 2010, and 918 mm for 2011. Linear regression analysis between rainfall, throughfall, stemflow and canopy drainage (**Table 4-2**) indicated that around 90% of rainfall penetrates the canopy as throughfall after the initial 1.4 mm of interception, while a further 5% of rainfall penetrates as stemflow. Interception as a percentage of rainfall was dependant on rainfall intensity, with large rainfall events resulting in a low interception percentage (0.25% with 218 mm) and small rainfall events resulting in a high interception percentage (100% with 1 mm). Coefficient of variation (CV) between throughfall collectors averaged at 6.2% and remained below 15% for the duration of the study. Smaller rainfall events (below 3.5 mm) produced the largest variability between collectors. Stemflow volume increased with tree size (dbh) and was correlated with rainfall (a statistical correlation (R) of 0.93 to 0.95 between collectors). On a few occasions a small amount of precipitation was collected under the tree canopy without rainfall (possibly from dew or mist interception).

Table 4-2: Relationships between rainfall collected in an open area and that collected with throughfall and stemflow derived through linear regression

Statistic	Throughfall	Stemflow	Canopy drainage
Constant	-1.404***	0 ^{ns}	-1.462***
Slope	0.9076***	0.04975***	0.9584***
R ²	99.1	94.1	99.2
Regression mean square	39562.9***	237.9043***	44120.489***
Residual mean square	5.206	0.1129	5.342
SE Constant	0.386	0.00108	0.391
SE Slope	0.0104	0.0000	0.0105

A mass of 26 Mg ha⁻¹ (2.6 kg m⁻²) of harvest residue remained on the site after harvesting. This was added to the pre-existing 24.6 Mg ha⁻¹ (2.5 kg m⁻²) of forest floor during harvesting to yield 50.6 Mg ha⁻¹ (5.1 kg m⁻²) of residue. The 3-month delay between felling and burning resulted in a large portion (28%) of the residues decomposing prior to burning. Burning reduced the remaining residue from 36.4 Mg ha⁻¹ (3.6 kg m⁻²) to 4.2 Mg ha⁻¹ (0.4 kg m⁻²), a layer of ash and char that did not persist on the soil surface after the first week after burning, during which 15 mm of rain fell. Only a very small quantity of coarse char remained on the soil surface in small mounds, the remainder leached into the soil with larger particles creating a distinctive char horizon between 5 and 10 cm from the soil surface. The soil surface thereafter was predominantly ash-free with only white sand visible.

4.4.2. Change in soil moisture after harvesting

Soil moisture content is presented in **Figure 4.1 A - C** as the mean and standard deviation of data sets from 10 probes' data in each treatment recorded at 10 cm increments (**Appendix 4-3A - F**). Soil moisture recharge in the felled plots began within 1 month after felling (and 72 mm of rainfall) at shallow depths and after 3 months at 100 cm with a further 230 mm of rainfall. Recharge in the standing crop plots, although often less than in the felled areas, was rapidly reduced after rainfall during the first few summer months after felling (**Figure 4.1 A- C**). During each winter of the study in the standing crop treatment water content near 100 cm depth remained at a level slightly above wilting point despite being reduced to wilting point at the shallower depths. This may indicate a reduction in root activity at this depth during winter. Soil moisture contents in the No-Burn and Burn treatments were consistently higher than the standing crop treatments and seemed to remain above field capacity for a period of approximately 2

months during the inter-rotational period, even during prolonged periods without rainfall. Differences in soil moisture content between residue management treatments, although relatively small, remained consistent throughout the inter-rotational period. The No-Burn treatment remained moister than the Burn treatment with up to 3% higher water content after burning during the period from felling to about 6 months of age of the new crop. Differences in soil moisture content between the standing crop and both the Burn and No-Burn treatments decreased as the new crop approached canopy closure. Soil moisture measurements after canopy closure were slightly higher in the No-Burn treatment than the Burn treatment, while soil moisture in both Burn and No-Burn plots tended to remain below that of the standing crop treatment.

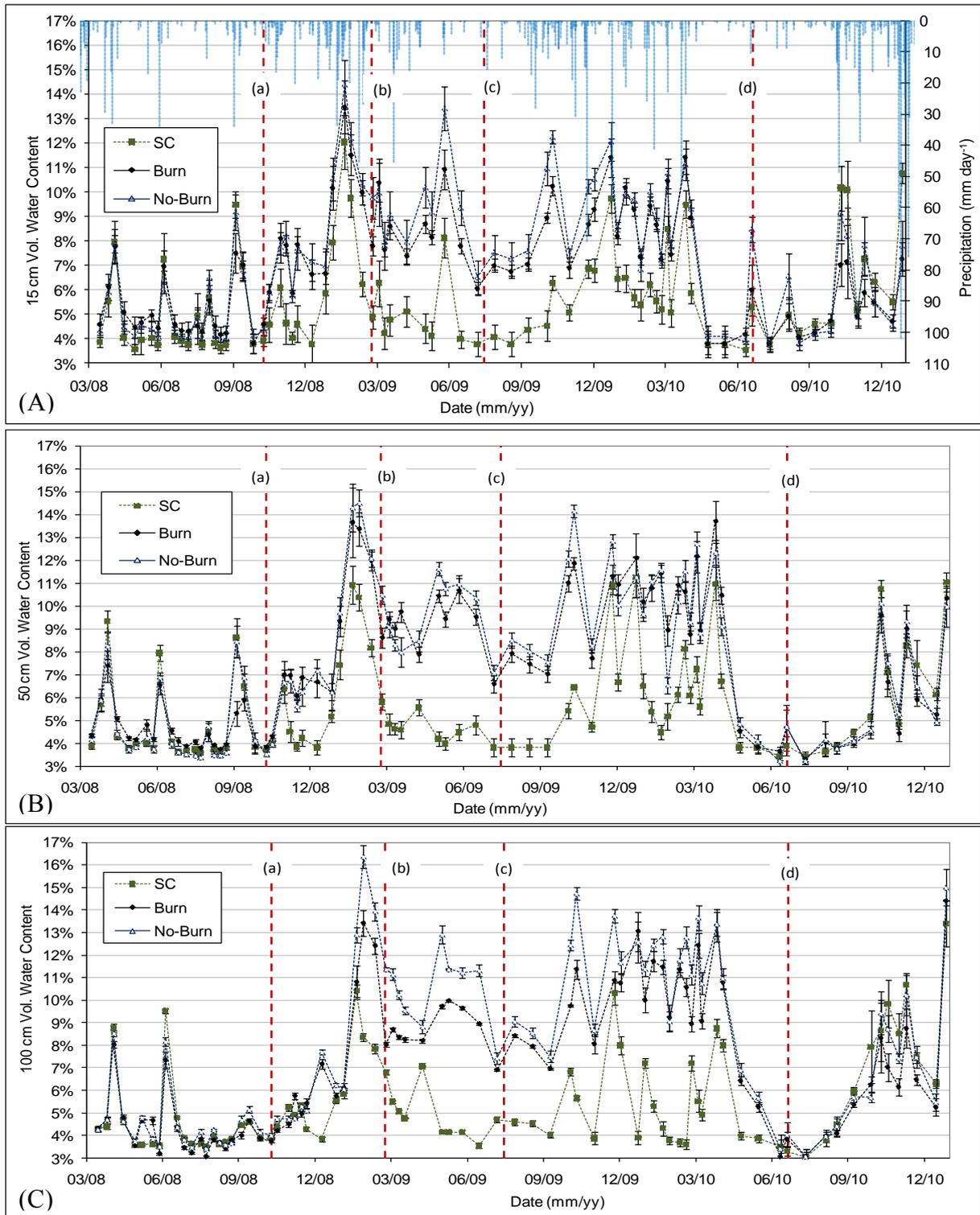


Figure 4.1(A-C): Volumetric soil moisture content as means of the standing crop (SC), residue burned (Burn) and residue retained (No-Burn) treatments at 15, 50 and 100cm from 200 days before harvesting to 18 months after planting. Dashed lines represent (a) felling; (b) residue burning; (c) planting and (d) canopy closure. Rainfall events are shown as vertical lines scaling to an inverted secondary Y axis in 5a. Single standard deviations are given as vertical I bars.

Variability (standard deviation) between probes and CV within each treatment plot was linked primarily to the probe position relative to the trees. Probes at the centre of 4 trees reflected greater wetting after rainfall and more rapid subsequent drying than probes positioned near to trees. Standard deviations and CV across all probes at each depth within each residue management treatment were also greater after rainfall as differences in soil moisture content between probe positions relative to the trees became more exaggerated. This was most apparent in the standing crop treatment and least apparent in the felled treatment plots during the inter-rotation period, variability being greater in the Burn treatment at the soil surface. Variability between probe positions began to decrease to similar levels as the standing crop treatments as new crop growth progressed. The CV between probe positions remained below 15% at the soil surface, decreasing with depth to below 5% as differences between probe positions became less. Measured water contents obtained via probes and gravimetric determination agreed well, indicating adequate probe calibration for the site. Differences in wetting and drying patterns between probe positions may have been due to limited runoff and run-on across the micro-topography of the site. Rapid drying may have been through higher root densities in the wetter depressed zones (Laclau et al. 2001).

Although water use was apparently high in the standing crop, very little growth occurred during the study period, with the biomass increasing by 4.6 % (4.2 Mg ha^{-1}) during the first year after planting the new crop. While stem wood accounted for 3.2 Mg ha^{-1} of this growth litterfall added a further 7.7 Mg ha^{-1} to above-ground biomass production in the standing crop during this period. Leaf area index was calculated as $1.68 \text{ m}^2 \text{ m}^{-2}$ across the site at felling, fluctuating between 1.75 in the wet season and 1.42 in the dry season of the standing crop plots. Water-use efficiency for stem wood production was therefore poor, given the high rate of water loss and limited growth. New crop growth was initially greater in the Burn treatment than in the No-Burn treatment, but slowed in the Burn treatment after canopy closure. Differences in biomass in the Burn treatment compared with the No-Burn treatment amounted to 29% more at 6 months, 17% more at 1 year and 5% more at 18 months. Although biomass in the Burn and No-Burn treatments was not statistically different from 18 months onwards, growth in the Burn treatment continued to slow relative to the No-Burn treatment. Leaf area index was measured as 1.9 and 1.5 in the Burn and No-Burn treatments, respectively, at 6 months of age and 2.7 and 2.4, respectively, at 1 year (canopy closure). An above-ground biomass of 4.1 and 3.1 Mg ha^{-1} was estimated in the Burn and No-Burn treatments, respectively, at 6 months of age, and 11.5 and 9.9 Mg ha^{-1} at 1 year after planting.

During the inter-rotation to canopy closure period at 3 to 14 months after clearfelling (February 2009 to April 2010), hourly TDR and weekly profile probe soil moisture data gave some evidence indicating an alteration in soil hydraulic conductivity (infiltration rate) and field capacity in the felled No-Burn and Burn treatments. These parameters were not measured directly during this period as this change was not expected. Hourly TDR data produced consistent wetting-front advance rates on each test plot for a number of rainfall events prior to felling across all treatments, and throughout the study period for the standing crop treatment. **Figure 4.2** presents evidence of a wetting front (using TDR probe measurements) after a prolonged dry period and the onset of evening rainfall. An evening sample of the TDR data was selected to reduce the potential effects of daytime evapotranspiration. A wetting front was detected at 100 cm depth within 2 to 3 h of detection of increased moisture at 15 cm, in agreement with the expected time for water to move this distance through the profile for the given soil hydraulic conductivity of the soil. After felling and during the inter-rotational period (**Figure 4.2**), wetting front rates slowed in the felled Burn and No-Burn treatment plots while remaining unchanged in the standing crop treatment plots. The decrease in rate of the wetting front velocity was most apparent from around 3 months after felling in the Burn and No-Burn treatments, becoming less apparent over the months that followed. Penetration time in the water drop repellency test was found to vary by a few seconds, though no significant difference ($P < 0.05$) was found between treatments. This suggested that the water repellency of the soil did not alter water infiltration rates between treatments, though the infiltration rate may have been affected by soil surface condition (presence or absence of residue and site micro-topography).

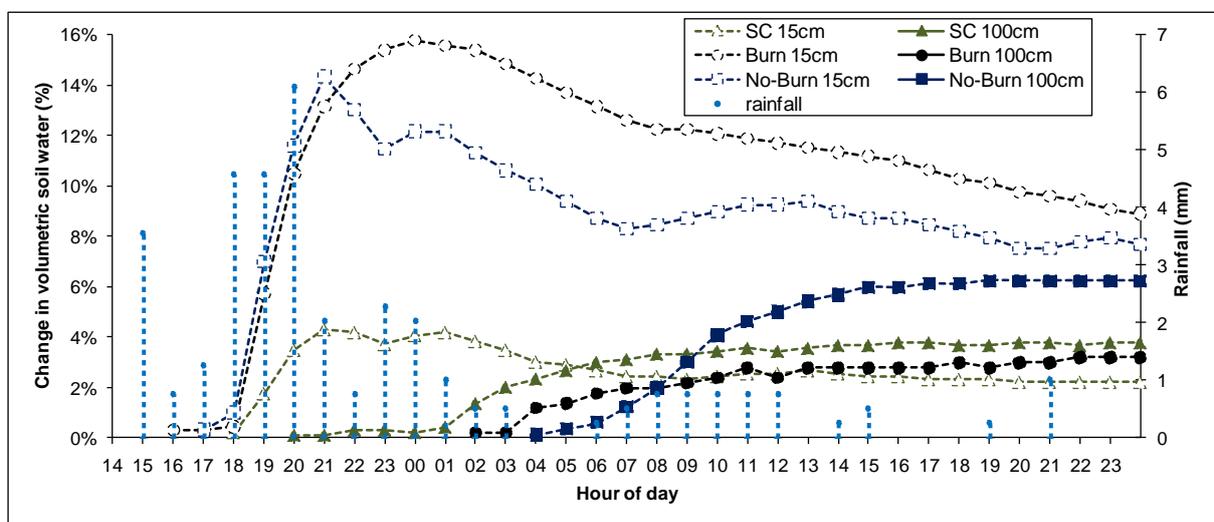


Figure 4.2: Demonstration of the time delay between wetting front detection at 15 cm and 100 cm as percentage change in soil moisture content after onset of rainfall in the standing crop (SC), residue burned (Burn) and residue retained (No-Burn) treatments. Calculated as difference from pre-rainfall soil moisture content, (zero values not shown).

4.4.3. Water collected by plate lysimeters

The volumes of water collected were dependent on soil moisture content prior to rainfall events and the intensity of each rainfall event (**Figure 4.3A-C**). Low-intensity rainfall events resulted in more water collection than similar-quantity high-intensity rainfall events, as the lower intensity of rainfall may have reduced the fraction of water lost by plates overflowing. Cumulative rainfalls during each time period were 418, 211, 641, 286 and 727 mm, respectively (**Figure 4.4**). Volumes of water collected decreased with depth and were generally the lowest in the standing crop treatment (**Figure 4.3A – C, Figure 4.4**). Larger total volumes of water were collected under the Burn treatment at each depth than under the No-Burn treatment. Despite the trends of differences between treatments, no significant ($P < 0.05$) differences between treatments were found.

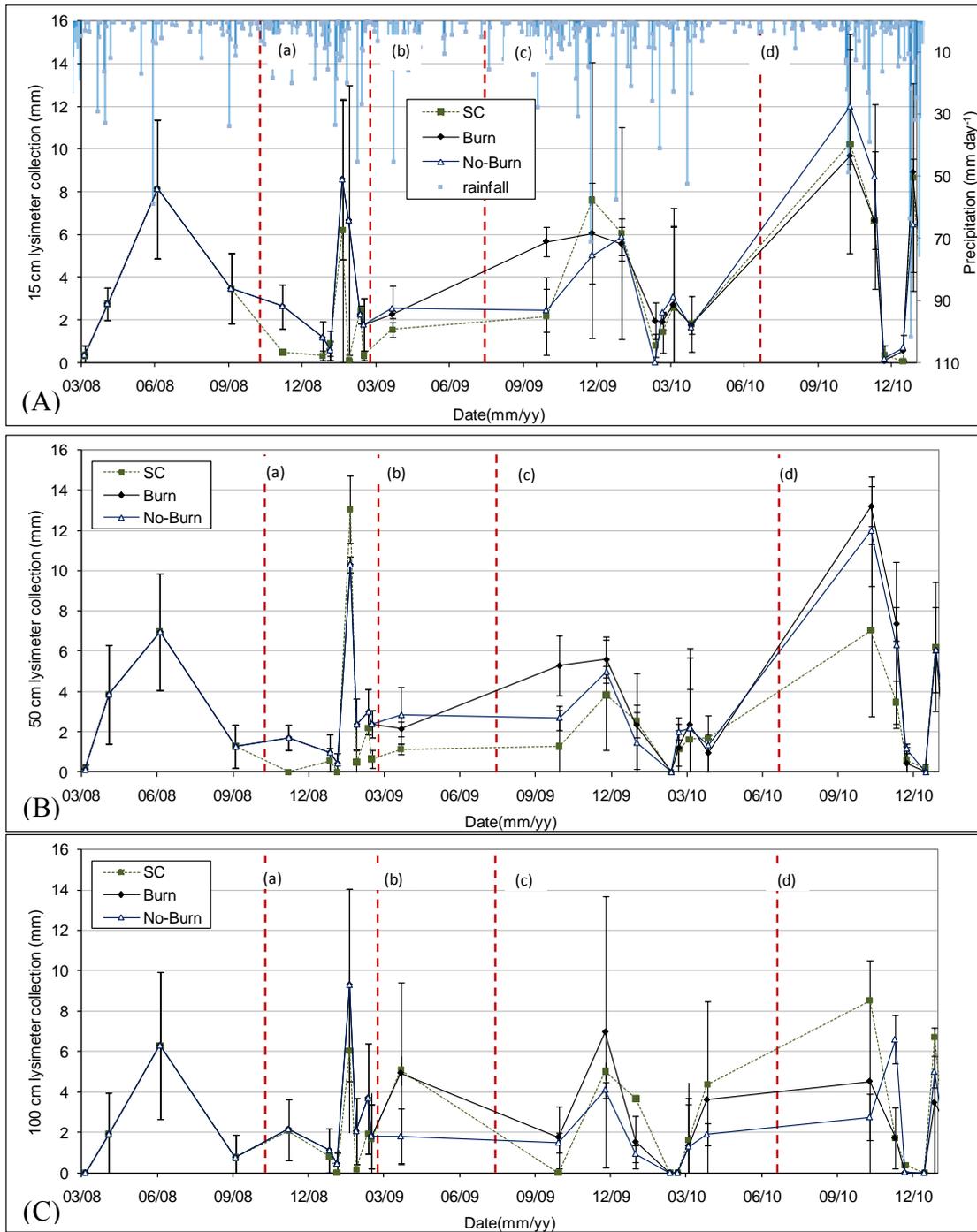


Figure 4.3(A-C): Mean volumes (\pm SD) of water collected at 15, 50 and 100 cm depth with plate lysimeters in the standing crop (SC), residue burned (Burn) and residue retained (No-Burn) treatments. Dashed lines represent (a) felling; (b) residue burning; (c) planting and (d) canopy closure. I bars represent single standard deviations. Rainfall events are shown as vertical lines scaling to an inverted secondary Y axis in 6(A).

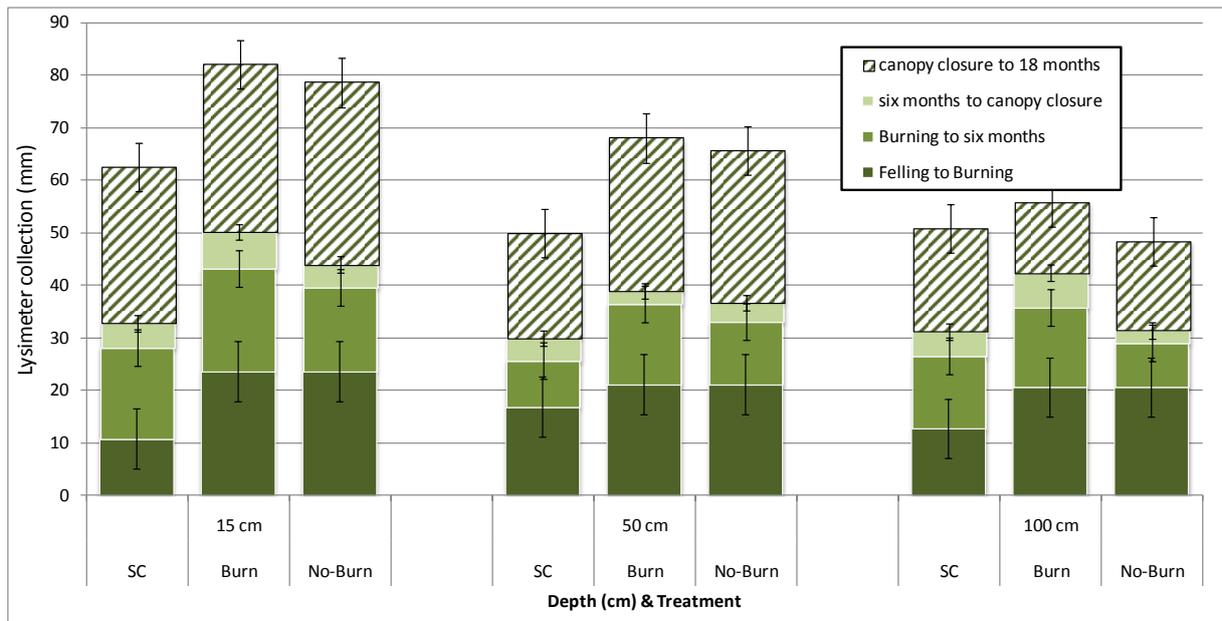


Figure 4.4: Total volume of water collected at 15, 50 and 100 cm depth with plate lysimeters in the standing crop (SC), residue burned (Burn) and residue retained (No-Burn) treatments from felling to 18 months after planting. Stacked bars represent time periods from felling to burning, and six monthly time periods from planting to canopy closure onwards. I bars represent 5% least significant differences for treatment and collection period.

4.4.4. Predicted water content using HYDRUS model

Table 4-3 presents the linear regression variables and relationships between water content predicted with the Hydrus model and that measured within the profile probe at each depth for the standing crop plots. The model adequately estimated water content at all depths with a near 1:1 relationship between predicted and observed values. Model estimates were poorer at the soil surface, but improved with soil depth; this is reflected in the increase in the R^2 values with increasing depth. Soil surface estimations may have been less reliable as the surface soils had more variable organic carbon content and lower bulk density, especially in the 0 to 10 cm soil layer. A further estimation of soil moisture content to 10 m depth was performed for the standing crop treatment, assuming constant soil properties to depth and the absence of a water table above 10 m depth. The model estimated no drainage beyond 2 to 3 m depth, assuming all water to be taken up by the standing crop. However, this was not validated with actual measurements.

Table 4-3: Relationships between water content predicted with the Hydrus 1D model and measured with the profile probe at each depth for the in the standing crop (SC) treatments.

Statistic	10 cm	20 cm	30 cm	40 cm	60 cm	100 cm
Constant	-0.00016 ^{ns}	0.00002 ^{ns}	0.00002 ^{ns}	-0.00009 ^{ns}	-0.00034 ^{ns}	-0.00003 ^{ns}
Slope	1.0046 ^{***}	1.0023 ^{***}	1.0023 ^{***}	1.004 ^{***}	1.0081 ^{***}	1.0000 ^{***}
R ²	79.9	80.1	80.1	81.9	80.3	86.3
Regression mean square	0.01512 ^{***}	0.02145 ^{***}	0.02145 ^{***}	0.02333 ^{***}	0.02309 ^{***}	0.01673 ^{***}
Residual mean square	0.00007	0.0001	0.0001	0.00009	0.0001	0.00005
SE Constant	0.00394	0.00421	0.00421	0.00395	0.00406	0.00302
SE Slope	0.0684	0.0678	0.0678	0.064	0.0678	0.0541

^{***} Superscript implies F- or T- test significance at $p < 0.001$, ns superscript implies no significance

Model predictions of water content were poorer than the standing crop for the No-Burn and Burn treatments during the inter-rotation to canopy closure period, 3 to 14 months after clearfelling. Predicted water content during this period was significantly below values measured in the field. Reducing soil hydraulic conductivity and increasing field capacity in the Hydrus 1D model, for the period in question, to levels determined with the probe measurements improved the model's predictive capabilities. An inverse model prediction of soil hydraulic properties was also used to produce improved soil moisture estimation. Although adjusting soil hydraulic conductivity in the Hydrus 1D model allowed for a much closer fit between predicted and observed soil moisture content during this time period, changes in other hydraulic parameters could only be estimated. **Figure 4.5** and **Figure 4.6** present the Hydrus 1D predicted total volumes of water passing the 15, 50 and 100 cm soil depths for the duration of the study for the standing crop and felled treatments areas. Values in **Figure 4.6** are total volumes from felling to 18 months after planting, subdivided into time periods similar to **Figure 4.4**. Drainage volumes as predicted by Hydrus (**Figure 4.6**) were higher than those measured in the field (**Figure 4.4**). Actual drainage was 4 to 6% of predicted drainage in the felled plots, and 6 to 27% of predicted values in the standing crop plots. Plate drainage rate may have been too slow and water may have been diverted around the plates or by plates overflowing, an occurrence common in such lysimetry studies (Weihermuller et al. 2007).

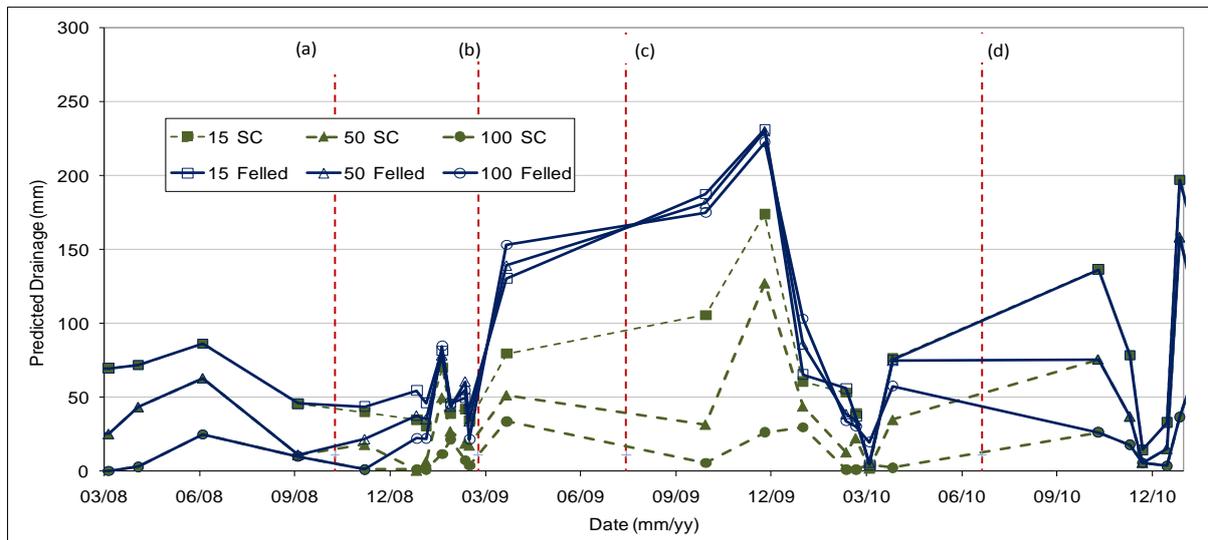


Figure 4.5: Predicted drainage volumes at 15, 50 and 100 cm for felled and standing crop (SC) areas of the study site. Vertical dashed lines represent (a) felling; (b) residue burning; (c) planting and (d) canopy closure.

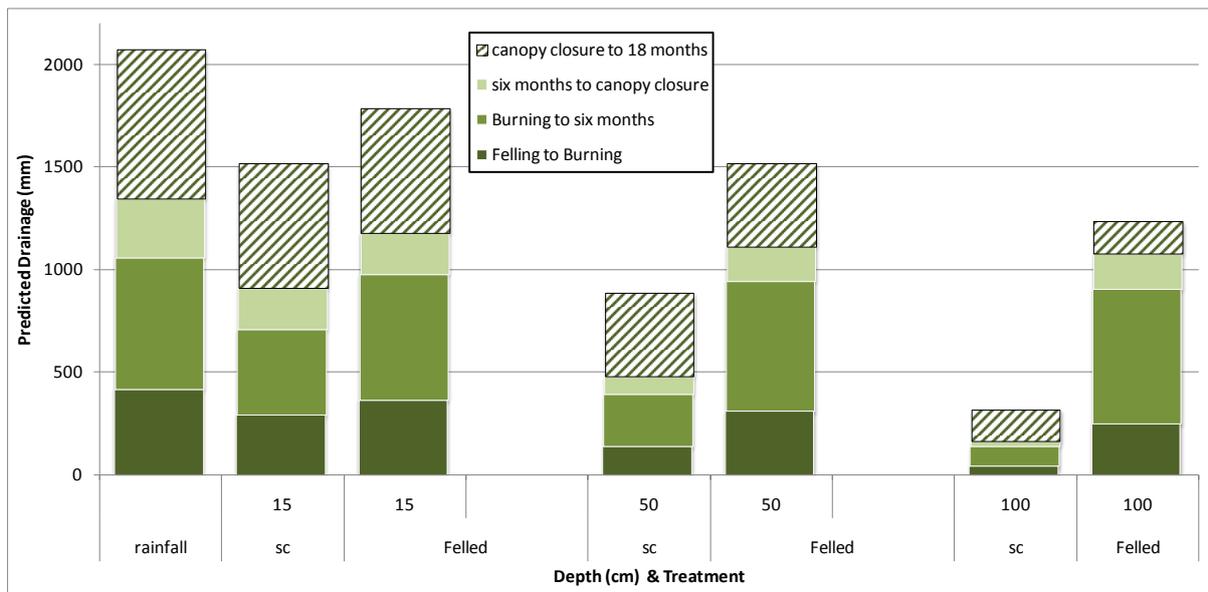


Figure 4.6: Rainfall and Predicted drainage volumes at 15, 50 and 100 cm for felled and standing crop (SC) areas of the study site. Each portion of the bars represents drainage predicted for each period in **Figure 4.5**.

4.5. DISCUSSION

This study shows the changes in soil moisture regimes after clearfelling a clonal *Eucalyptus* crop on the well-drained sandy soils of the Zululand coastal plain. While soil moisture recharge did occur to field capacity under the living tree canopy, soil moisture was rapidly reduced over the entire measured soil profile through evapotranspiration. Changes in soil moisture after clearfelling indicated rapid soil moisture recharge with rainfall after clearfelling; with surface evaporation and weed and coppice re-growth reducing soil moisture content. The increased soil

moisture status after harvesting was temporary as the new trees had developed a relatively large leaf area by the time of canopy closure that afforded a rapid reduction in soil moisture in the upper soil layers. The residue layer on the No-Burn treatment and the absence of this layer on the Burn treatment (with exposed sand) may have been responsible for the small differences in soil moisture content. The retention of residue results in both higher interception of rainfall prior to entering the mineral soil and lower soil surface evaporation rates. Burning eliminated interception of rainfall prior to entering the mineral soil, but increased soil surface evaporation rates through surface exposure. These may have effectively reduced overall measured differences between Burn and No-Burn treatments.

Although soil moisture levels in the new crop were similar to those in the standing crop by canopy closure, relatively faster soil moisture depletion occurred under the larger canopy of the new crop, particularly in the Burn treatment. Despite the increased canopy leaf area after burning having a relatively larger water demand, the additional growth in this treatment may imply increased water use efficiency with respect to biomass production. This early biomass increase is often used by plantation managers as an incentive towards residue burning. The faster-developing canopy reduces weed growth, thereby reducing operational weeding costs. This increase in biomass was short-lived in this study, as growth in the burned plots began to decline relative to the unburnt plots after canopy closure. An increased leaf area after burning may have reduced the new crop's ability to cope with water deficits or drier soil conditions after canopy closure, hence reducing growth rates. This may imply that trees that have developed a larger canopy in response to residue burning may have a reduced ability to cope with drought conditions, should these occur.

Not all processes that affect soil hydrology are clearly understood or accounted for in plantation forest hydrological models. Stemflow has been shown to channel water and nutrients captured during rainfall or mist events directly to tree root systems, providing water to trees that is often not detected in the bulk soil or accounted for in hydrological models (Johnson and Lehmann 2009). Preferential flow along old tree root channels as a mechanism of sub-surface wetting or ground water recharge may also be poorly accounted for (Le Maitre et al. 1999). The interactions between pest infestation and water use efficiency may also need investigation, particularly where pests cause no reduction in leaf area, but increase leaf water loss through damage to epidermal leaf tissue or photosynthetic capacity. The continued water use during pest

infestation and the drastic reduction of growth in this study clearly demonstrates the negative impact and cost of pest infestation to productivity and water use efficiency. Organic substances released from the rapidly-decomposing residues may have played a role in altering the soil during the inter-rotation. Organic exudates, root decomposition, proliferation of fungal hyphae and microbial biomass in the soil may have temporarily altered soil physical properties and disrupted hydraulic flow paths through biological soil-pore clogging. Research has shown that organic matter has a strong influence on water retention and saturated hydraulic conductivity (Rawls et al. 2003). Blanco-Canqui and Lal (2007) presented evidence of mulches added to the soil surface causing an increase in soil porosity and soil moisture content at wilting point and a decrease in soil bulk density. Changes in hydraulic conductivity induced through microbial activity have also been shown in column studies that simulated saturated soils (Seki et al. 1998), although these effects associated with forest residue breakdown need to be confirmed with further study. A water repellent layer in the soil produced by fine char particles may also act to increase the water retention characteristics of a soil (Moore 1996; Woods and Balfour 2010; Onodera and Stan 2011).

Large quantities of nutrients locked up in the residues are rapidly released through residue decomposition into the soil solution during the inter-rotational period. Solutes may be displaced beyond the shallow fine-rooting zone or lost where large quantities are further leached. This may be a concern where nutrient replacement through natural processes occurs at insignificant rates and where fertilisation is not possible. Although high evapotranspiration rates and water use of *Eucalyptus*, which can lead to decreased stream flow, are a disadvantage on water-limited sites where there are other competing human, agricultural, industrial or environmental requirements for water, high evapotranspiration rates and water use may limit nutrient loss from plantation forests. This may be of importance to infertile sites where loss or displacement of nutrients with drainage can negatively affect growth. This principle was demonstrated in studies conducted on the sandy soils of Congo and Brazil where water did not drain beyond a depth of 5 m, despite a far higher rainfall than that recorded for the present study site (Nouvellon et al. 2002). The loss of nutrients in that study had a far more negative effect than the loss of water. Sites at risk of nutrient loss may therefore require special management. Allowing temporary coppice re-growth and planting to achieve earlier canopy closure may limit drainage and nutrient leaching loss that may occur if plantation areas are left fallow for prolonged periods.

4.6. CONCLUSION

Despite the disadvantages of high water use, an implied reduction in nutrient leaching or displacement under a full-canopied crop may allow more nutrients to be retained in the active rooting zones under the undisturbed forest stand. The impact of pest damage on water use efficiency in the full-canopied crop may also need further investigation, as the loss to productivity, coupled with the high cost of water, compounds economic losses related to water use. Although soil moisture and drainage increased after felling, differences between residue retention and residue burning presented here may not be enough to affect the decision between burn and no-burn management policies. Based on data presented here, it is also unlikely that the residue management practices presented here will have any further direct impacts on soil moisture content and drainage beyond the period presented in this study. Reducing water use through increases in productivity as a consequence of superior genetics and clonal forestry may still be one of the best strategies for improving water use efficiency, essentially producing more timber per unit water use (Stape et al. 2004). The possibility of increased water use resulting from a potentially larger tree canopy, induced in response to planting after residue burning, may however increase the drought susceptibility of the new crop around the time of canopy closure. Deeper drainage and leaching or displacement of nutrients during the period directly after clearfelling may be of concern to forest managers for sites that are sensitive to soil nutrient losses. This may give reason for plantation forest managers to shorten the inter-rotation length, with earlier planting to enable a more rapid depletion of soil moisture through earlier canopy closure, thereby retaining more nutrients in the soil and in the growing trees. This, however, may only be necessary for sandy soils with a low nutrient retention capacity. Rapid replanting is obviously also positive from a financial point of view. In addition to soil carbon and nutrient loss with biomass removal and burning, reducing drainage loss or the displacement of nutrients on sites with low water and nutrient storage capacity may be an important factor driving the efficient, and ultimately the sustainable, use of water and nutrient resources of such sensitive sites.

4.7. APPENDICES

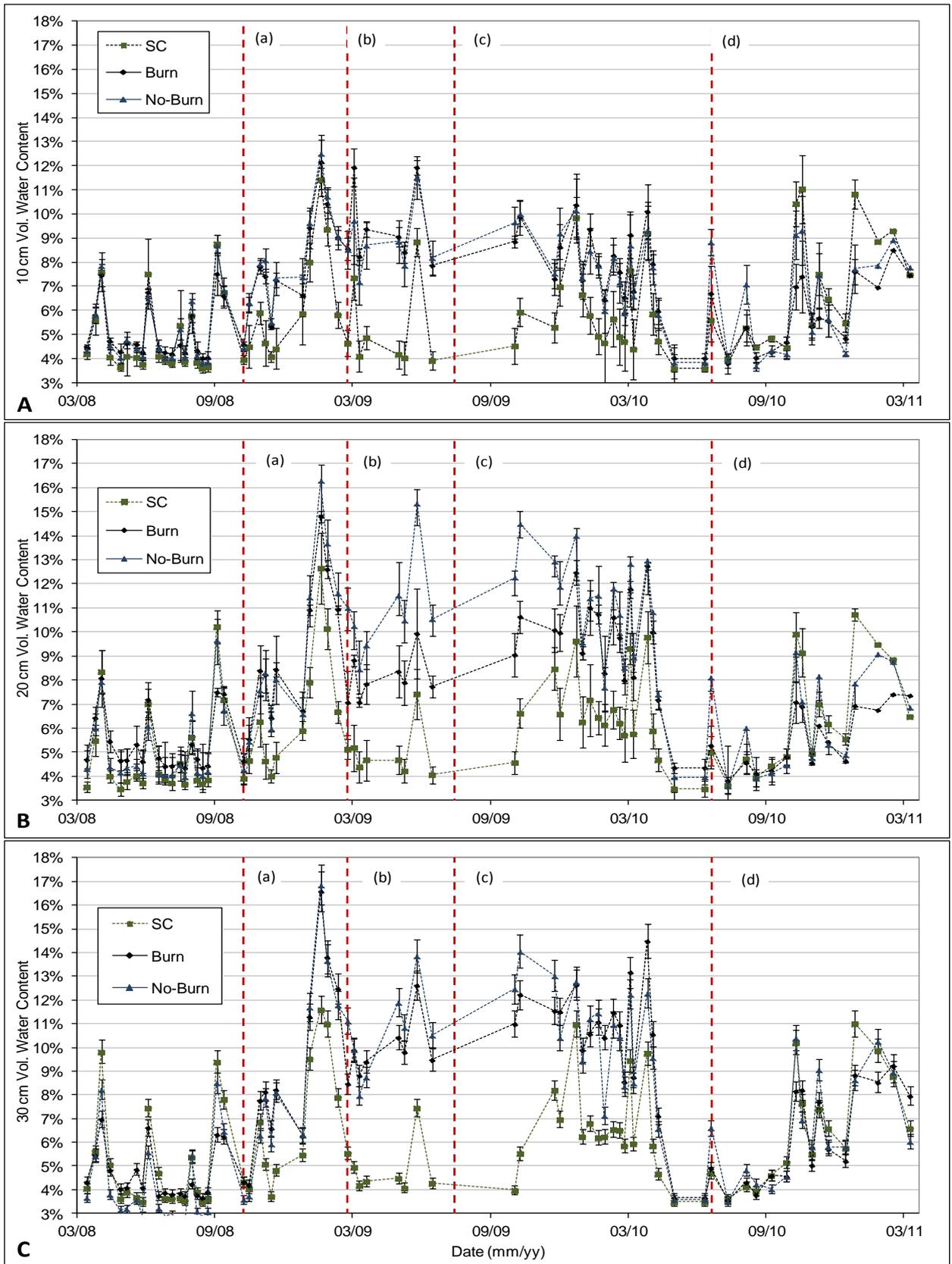
Appendix 4-1: Further Comments about tree root Systems

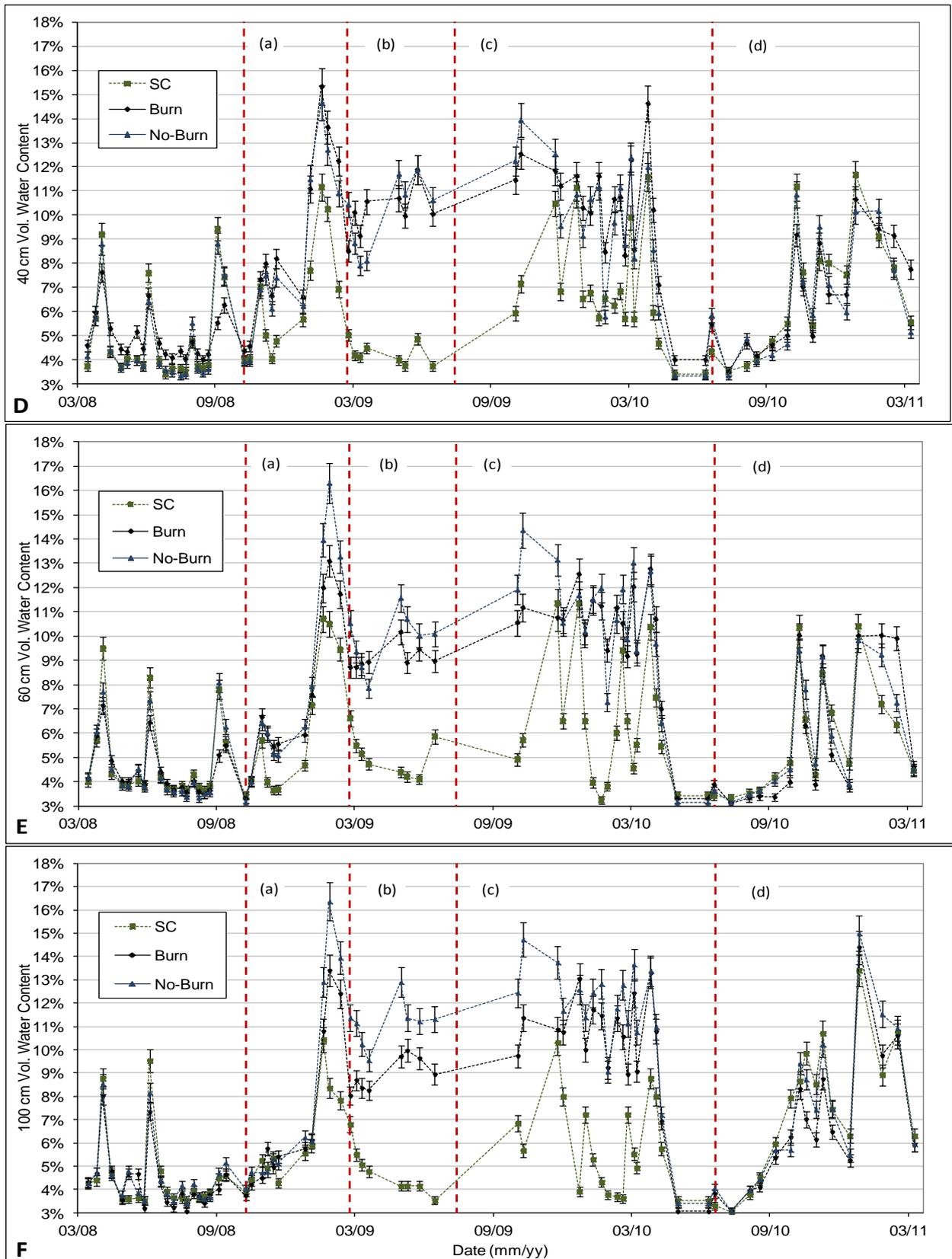
Although tree root systems have been extensively studied, the vast majority of these studies have been with seedling planted trees (Knight 1999). A pot study of cutting propagated *E. globulus* trees were found by Sasse and Sands (1996) to have a lower water use under water stress than seedling propagated trees. Although decreased water use in clonal forestry has not been shown, increases in productivity as a consequence of superior genetics and clonal forestry has been shown to increase water use efficiency, essentially producing more timber per unit water use (Stape et al. 2004). The shallower root system and the absence of a well defined taproot in cutting propagated trees (Maria João et al. 2005) may result in a decreased in water uptake at depth, particularly from the capillary fringe above the water table. This may change the view of water use by clonal forests in Zululand as previous model estimations of ground water use (Kienzle and Schulze 1992) assume the presence of a taproot.

Appendix 4-2: Further Comments about water use efficiency

South African research over several decades has shown that plantations commonly use more water per unit of time and space than indigenous vegetation that it replaces (Scott and Smith 1997; Gush et al. 2002). However, Plantations usually have higher water use efficiency than indigenous forests (Gush and Dye 2009; Gush et al. 2011), mainly because they produce large quantities of biomass per unit of water transpired. Binkley et al. (2004) also demonstrated that resource use efficiency at the landscape scale is greater when there is an increase in total resource use. Rather than afforesting more land, incentives should perhaps revolve around maintaining and ideally increasing productivity on currently afforested sites allowing for more timber to be produced on a limited land area. Commercial plantation owners may therefore need to accept the high water use and focus on achieving high productivity on this limited land base while sustaining soil carbon and nutrient stocks, particularly on nutrient poor sites. This may be realised through investing in a number of key areas in plantation forestry research and management that include: (a) improved genetics, not only to produce higher value products, but to improve tolerance to unfavourable climatic conditions, pest and disease, (b) Improved silviculture and harvesting practices that maintain or build carbon and nutrient stocks, and (c) improvements in pest, disease and wildfire prevention. Essentially maintaining nutrient and

carbon resources and improving water use efficiency are the only options that have any practical feasibility in commercial plantation forestry, allowing the greatest economic value to be obtained from an uncontrollably high water use while conserving all other biological reserves.





Appendix 4-3(A-F): Volumetric soil moisture content as means of each depth in the standing crop (SC), residue burned (Burn) and residue retained (No-Burn) treatments for the duration of the study. Dashed lines represent (a) felling; (b) residue burning; (c) planting and (d) canopy closure. Single standard deviations are given as vertical I bars.

CHAPTER 5: NUTRIENT FLUXES IN RAINFALL, THROUGHFALL AND STEMFLOW²

5.1. ABSTRACT

Atmospheric deposition was assessed at two sites over a four year period in post canopy closed (mature) *Eucalyptus* stands in the northern Dukuduku and southern KwaMbonambi commercial plantation forestry areas of Zululand, South Africa. The aim of this study was to determine the magnitude and relevance of nutrient addition with rainfall, throughfall and stemflow to commercial forestry in this region. Canopy cation exchange was used with rainfall and canopy drainage to derive wet, dry and total atmospheric deposition. Nutrient concentrations measured in the rainfall, throughfall and stemflow varied widely throughout the study period, and between sources and sites. Rainfall was slightly acidic at both sides, but became less acidic upon passing through the tree canopies. Canopy exchange and collection of dry deposition resulted in increased cation concentrations under the tree canopy, while the canopy generally absorbed nitrogen (N), from the rainfall, reducing the below canopy concentrations. Atmospheric deposition was shown to be responsible for large quantities of nutrients added to the *Eucalyptus* stands at each site. Annualised deposition averaged across all years at each site amounted to N, calcium (Ca) magnesium (Mg) and potassium (K) fluxes of 11.0, 6.0, 2.7 and 10.2 kg ha⁻¹ yr⁻¹ at Dukuduku and 10.6, 7.5 and 18.8 kg ha⁻¹ yr⁻¹ at KwaMbonambi. Organic N fluxes contributed a further 8.1 kg ha⁻¹ yr⁻¹ at Dukuduku and 7.1 kg ha⁻¹ yr⁻¹ at KwaMbonambi to the total N deposition. Although K deposition values were high, additions of all other nutrients, although also high, were within the ranges reported in local and international research. Over the course of a full rotation, the atmospheric deposition levels recorded at these sites may have potential to supply a large proportion of the nutrients that are lost during stem-wood harvesting. This study adds value to understanding of nutritional sustainability of fast growing plantation forests, demonstrating the importance of atmospheric deposition as a nutrient addition source to plantation grown *Eucalyptus* along the Zululand coastal plain.

5.2. INTRODUCTION

The South African forestry industry contributes significantly to South Africa's gross domestic product and employment from a relatively small and decreasing portion of South Africa's total

² Published (2011) in *Southern Forests* 73(3-4):193-206.

land area (FSA 2009a). The loss of land area, 9% between 1999 and 2009 (FSA 2009a) has occurred through, *inter alia*: wetland delineation, successful land-claims that have subsequently resulted in land being converted out of forestry, and the establishment of wildlife corridors through plantation forest areas. Increases in production of wood from South Africa's intensively managed plantations (19% for the same 10-year period) has occurred through improvements in genetics, silviculture and harvesting (Schutz 1982; Burger 1996; Bauhus et al. 2002). Although gains in productivity have been realised through these improvements, questions are still being raised around the long term sustainability of yields. The principles and processes driving nutritional sustainability (maintenance of site fertility) in particular are not fully understood or quantified in commercial plantation forestry (Blanco et al. 2005; Laclau et al. 2005b; Watt et al. 2005). The impact that nutritional sustainability has on plantation productivity is dependent on the biogeochemical cycling of nutrients within the forest ecosystem and the ability of trees to acquire, use and cycle these nutrients (Rennenberg and Schmidt 2010).

Nutrient cycling is important in plantation forestry where addition of nutrients through processes such as mineral weathering, nitrogen (N) fixation and atmospheric deposition may partially offset losses incurred through harvesting and indirectly through other management practices. The role nutrient cycling processes plays in site fertility has been described in natural forests and a few plantation forests in international literature (Jorgenson et al. 1975; Wells 1976; Binkley 1986a; Attiwill and Adams 1993b; Ranger and Turpault 1999; du Toit and Scholes 2002; Blanco et al. 2005; Laclau et al. 2005b; Smethurst 2010). Here, the balance of additions, losses and fluxes within the ecosystems has been used to develop nutrient budgets that scrutinise present and future sustainable nutrient supply (Laclau et al. 2003b). Natural nutrient addition may become more important in offsetting nutrient losses where high costs and environmental concerns have reduced the use of fertiliser. That is, provided that management practices do not promote or accelerate other loss mechanisms, such as soil erosion and nutrient leaching. Frequent removal of large quantities of timber on low fertility sites may lead to nutrient depletion under conditions where nutrients are not replaced by natural processes or fertilisation. In addition, further biomass removal has the potential to exacerbate nutrient loss. Little work has been done in South Africa to directly quantify nutrient cycling processes for commercial plantation forestry. This is alarming since many highly productive sites are located on soils that are both highly weathered and limited in nutrient storage capacity (du Toit and Scholes 2002).

Although the contributions made by most natural processes are often too small to make any significant contribution (Jorgenson et al. 1975; Binkley 1986a; du Toit and Scholes 2002) some research data indicates the potential of atmospheric deposition to significantly contribute certain elements in South Africa (van Wyk 1990; Olbrich 1993; Zunckel et al. 2000; Lowman 2004). Atmospheric deposition is the addition of nutrients as wet deposition (precipitation, including mist interception), dry deposition (dry particulate fallout) and as foliar gaseous uptake (EPA 2001). Compounds found in atmospheric deposition are derived primarily from industrial pollution (mainly fossil fuel burning), biomass burning, lightning and coastal windblown sea-spray or mist (Hill et al. 2005). Wet and dry atmospheric deposition can add N as organic-N compounds and mineral forms (ammonium (NH_4^+), nitrate (NO_3^-) and nitrite (NO_2^-) to the ecosystem (Zimmermann et al. 2003; Hill et al. 2005). In addition, calcium (Ca^{2+}), magnesium (Mg^{2+}) and potassium (K^+) are typically added in large amounts, while only small amounts of phosphorus (P) may be added through atmospheric deposition (Binkley 1986a; Ranger and Turpault 1999; de Vries et al. 2003). As rainfall penetrates the tree canopy, ions are exchanged between the canopy and the rainfall, resulting in either uptake of ions or leaching of ions from the tree canopy (Draaijers and Erisman 1995). Interception and interaction of rainfall with tree canopies, which South African research has neglected in the past, may produce significantly different dry deposition fluxes (Lindberg et al. 1986) to those derived through past methods. In addition, studies carried out in South Africa are generally not carried out in commercial plantations and seldom include canopy exchange processes.

This paper describes nutrient additions with rainfall, throughfall and stemflow of two clonal *Eucalyptus* stands on the east coast of South Africa over three successive growing seasons (2007 to 2011). The objective of this study was to determine the order of magnitude of nutrient addition through atmospheric deposition to the northern and southern commercial forestry regions of the Zululand Coastal plains of South Africa. These coastal plains were selected due to the inherently low fertility of their Aeolian soils (Fey 2010) and the high productivity potential and hence large harvesting nutrient demand and loss potential of the sites. These factors increase the risk of rapid site fertility decline and a consequential drop in future productivity. The continued growth of plantations in these areas with respect to nutrient sustainability has been a major concern in recent years as there has been no conclusive evidence of a nutrient related growth decline. This is despite these plantations undergoing many rotations of intensive nutrient removal management practices, such as burning and additional biomass removal at harvesting. A study was initiated in early 2007 at two sites in this region to gauge the magnitude of

atmospheric deposition as a nutrient source and the degree to which this flux compensates for nutrient losses incurred through harvesting.

5.3. MATERIALS AND METHODS

5.3.1. Second study sites

The Zululand Coastal plains and Dukuduku study site are described in CHAPTER 3. A second study site was chosen in the southern forestry region of Zululand to generate an earlier comparative data set and to enable an understanding of sample handling and laboratory processes. Water collection began in April 2007 at the southern site. Both sampling sites were selected centrally and representatively of the commercial forestry areas in the northern and southern regions of the Zululand coastal plains (**Figure 3.1**). For the two sites, growth and site information data existed and both carried mature (post canopy closure) clonal *Eucalyptus spp.* (**Table 5-1**).

Table 5-1: General characteristics of the sampling sites selected for atmospheric deposition monitoring

	Northern site	Southern site
Name	Dukuduku	KwaMbonambi
Stand age at start	6 years	5 years
Clonal hybrid	<i>E. grandis x E. camaldulensis</i>	<i>E. grandis x E. urophylla</i>
Spacing	3 m x 3 m	2.6 m x 2.6 m
Altitude	62 m	60 m
Distance from Ocean	15 km	15 km
Latitude	28° 17' 51" S	28° 37' 54" S
Longitude	32° 18' 55" E	32° 07' 43" E
MAP	920 mm	1196 mm
MAT	22 °C	22 °C
Site Index (5 years)	19.9 m	27.1 m

5.3.2. Calculation of deposition components

Dry deposition and canopy exchange nutrient fluxes were derived algebraically through canopy ion exchange models described in Draaijers and Erisman, (1995) and Erisman et al. (2002) first reported in (Bredemeier 1988). Rainfall nutrient content was assumed to closely approximate wet deposition while the sum of throughfall and stemflow (canopy drainage) contents was

assumed to approximate total deposition plus canopy exchange. The exchange of Na^+ ions through the tree canopy was assumed to be negligible and so Na^+ in canopy drainage was taken as the sum of rainfall (wet deposition) and dry deposition. This assumption was confirmed through evidence of Na substitution for K not being found in the foliage. The ratio between Na^+ dry deposition and Na^+ wet deposition calculated as $(\text{throughfall}_{\text{Na}} - \text{rainfall}_{\text{Na}})/\text{rainfall}_{\text{Na}}$ was assumed to be the same for Ca^{2+} , Mg^{2+} and K^+ . Multiplying this ratio by the wet deposition of each of these elements yielded a derived value of dry deposition for each element after Draaijers and Erisman, (1995) and Erisman et al. (2002). The difference between canopy drainage and derived dry deposition was assumed to be canopy exchange. Canopy exchange of NH_4^+ was calculated assuming the sum of Ca^{2+} , Mg^{2+} and K^+ exchange to be equal to the sum of NH_4^+ and six times hydrogen ions (H^+) exchange minus the canopy leaching of Ca^{2+} , Mg^{2+} and K^+ associated with the excretion of weak acids from the tree canopy (Draaijers and Erisman 1995). Canopy leaching of NO_3^- was assumed to be negligible. Results were summed over six month periods to reflect the wet summer period October to March (S) and the dry winter period April to September (W). This was done to represent seasonality as experienced in the area with the wetter summer period having a higher rainfall, lower vapour pressure deficit and moderately higher temperatures than the drier winter period. Canopy exchange of O-N was assumed to be negligible and calculated as the difference between rainfall O-N and canopy drainage O-N. Research describing the occurrence of uptake or exchange of O-N by tree canopies is scarce (Hill et al. 2005).

5.4. RESULTS

Rainfall and canopy drainage (stemflow and throughfall) are shown in **Figure 5.1a** and **b** for each seasonal period. Rainfall at Dukuduku (**Figure 5.1a**) does not include the first two months owing to the measurements having commenced in December 1997. Although rainfall at both sites was below the long term mean for the majority of the study it was significantly higher at the southern KwaMbonambi site (**Figure 5.1b**, 1072 mm) than at the northern Dukuduku site (**Figure 5.1a**, 809 mm). Around 85% of rainfall penetrated the canopy as throughfall at each site, while a further 5% of rainfall penetrated as stemflow at the Dukuduku site and 2% at the KwaMbonambi site. Interception as a percentage of rainfall was dependant on rainfall intensity, with large rainfall events resulting in a low interception percentage (0.25% with 218 mm) and small rainfall events resulting in a high interception percentage (100% with 1 mm). Coefficient

of variation (CV) between throughfall collectors averaged at 6.2% and remained below 15% for the duration of the study. Smaller rainfall events (below 3.5 mm) produced the largest variability between collectors. Stemflow volume increased with tree size (dbh) and was correlated with rainfall (a statistical correlation (R) of 0.93 to 0.95 between collectors). On a few occasions a small amount of precipitation was collected under the tree canopy without rainfall (possibly from dew or mist interception). A regression relationship between rainfall and canopy drainage at each site gave slightly less canopy drainage relative to rainfall at the KwaMbonambi site ($canopy\ drainage = 0.915 \times rainfall - 1.386; R^2 = 0.991$) than at the Dukuduku site ($canopy\ drainage = 0.958 \times rainfall - 1.462; R^2 = 0.992$). Regression relationships between canopy drainage and rainfall were significant at both sites, having F and t probabilities <0.001 for each parameter estimate. Overall interception was around 12% of rainfall at KwaMbonambi and 10% of rainfall at Dukuduku. The differences in rainfall between summer and winter period divisions are clearer at the Dukuduku site, with a single high rainfall event during winter 2009 at KwaMbonambi (**Figure 5.1a, b**). Very high rainfall occurred during the summer 2010 period at both sites.

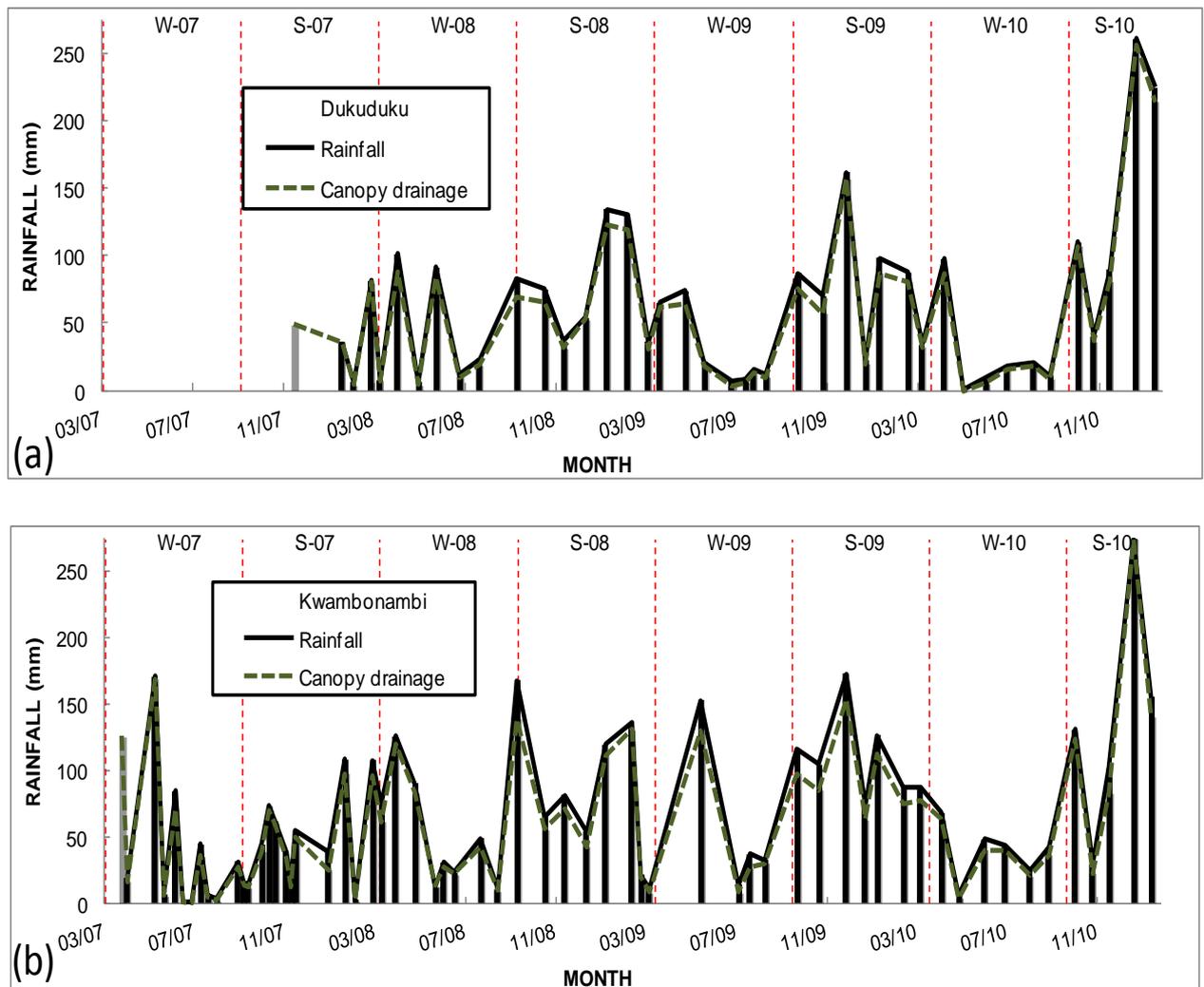


Figure 5.1: Monthly rainfall and canopy for each month at the northern Dukuduku site and southern KwaMbonambi site. Vertical dashed lines represent the start and end of each six month wetter summer (S) and dryer winter (W) periods for each year.

5.4.1. pH

Volume weighted pH (weighted as H^+ concentration) is shown in **Figure 5.2** for rainfall and throughfall at each site using the same six month periods as **Figure 5.1**. Rainfall and throughfall were slightly acidic, with a pH tending to remain below 6 for most measurements. The rainfall pH (CV 10%) was more variable than the throughfall pH (CV 7%) at each site. Rainfall was buffered upon passing through the tree canopy resulting in the narrower throughfall pH range. Dryer winter period pH tended to be higher than the wetter summer period pH values for each source. Rainfall collected at KwaMbonambi had a lower pH than that collected at Dukuduku ($p < 0.001$ using the unpaired Student's T-test on individual event data), while throughfall pH did not followed any site trends.

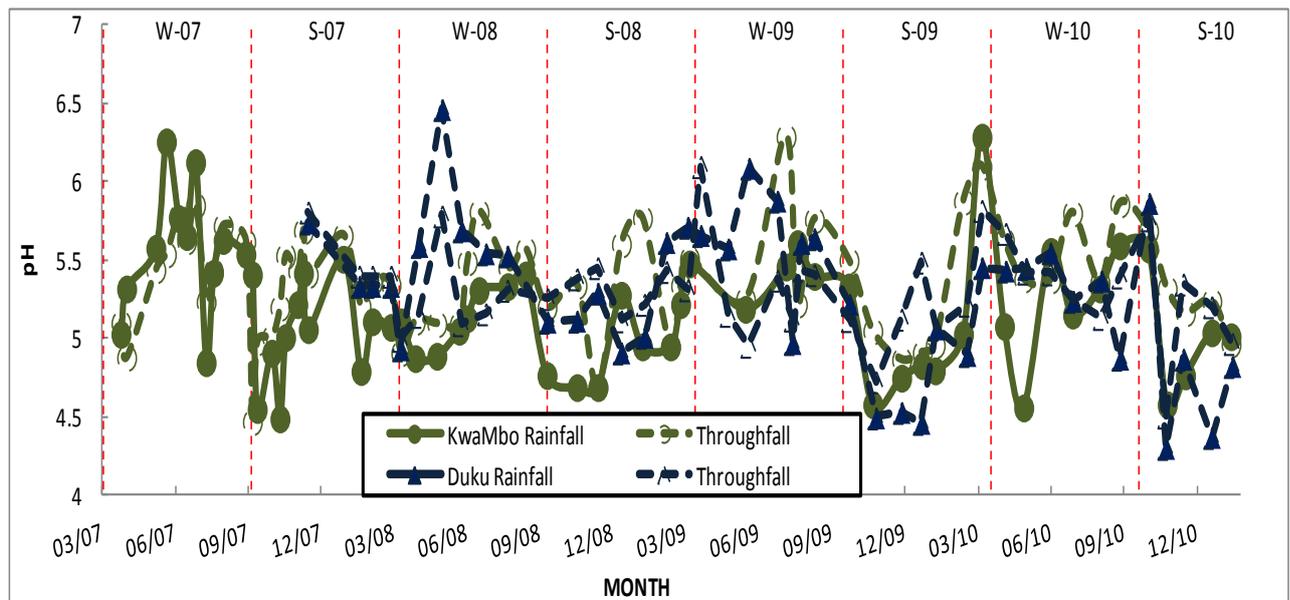
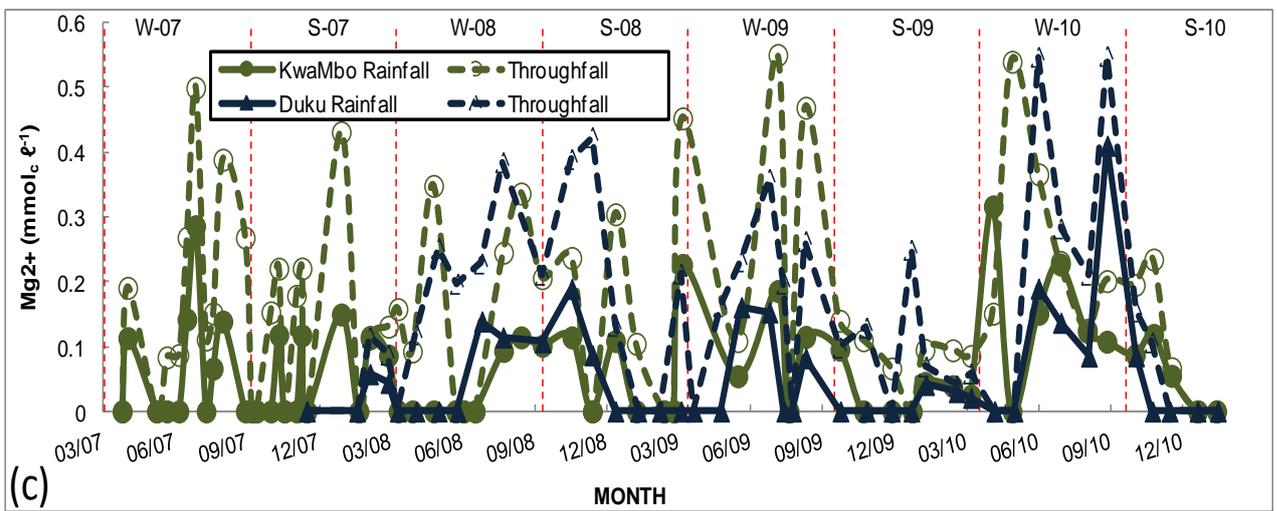
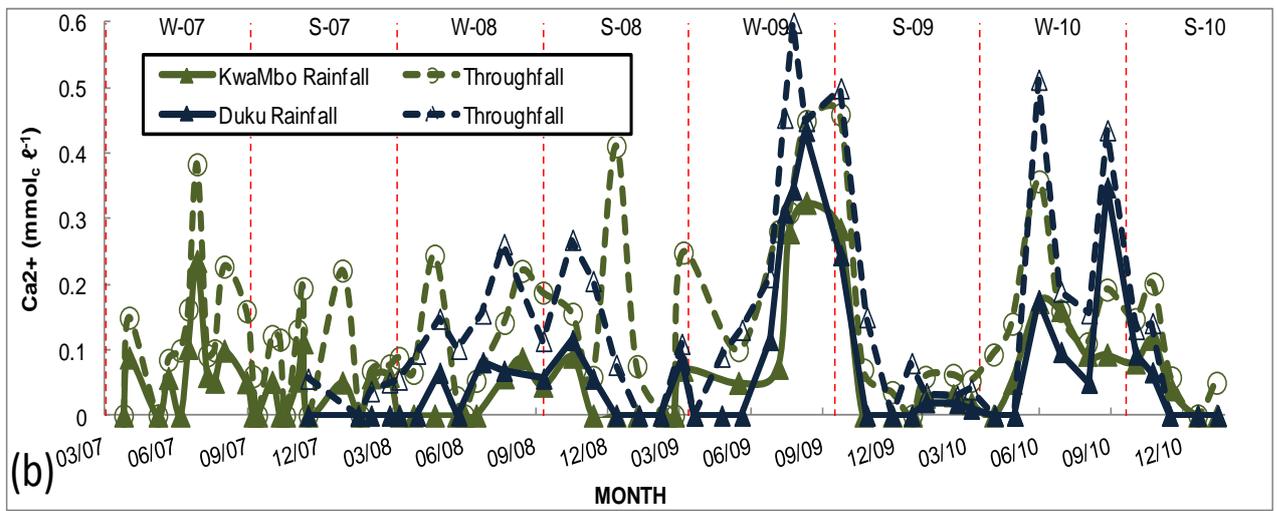
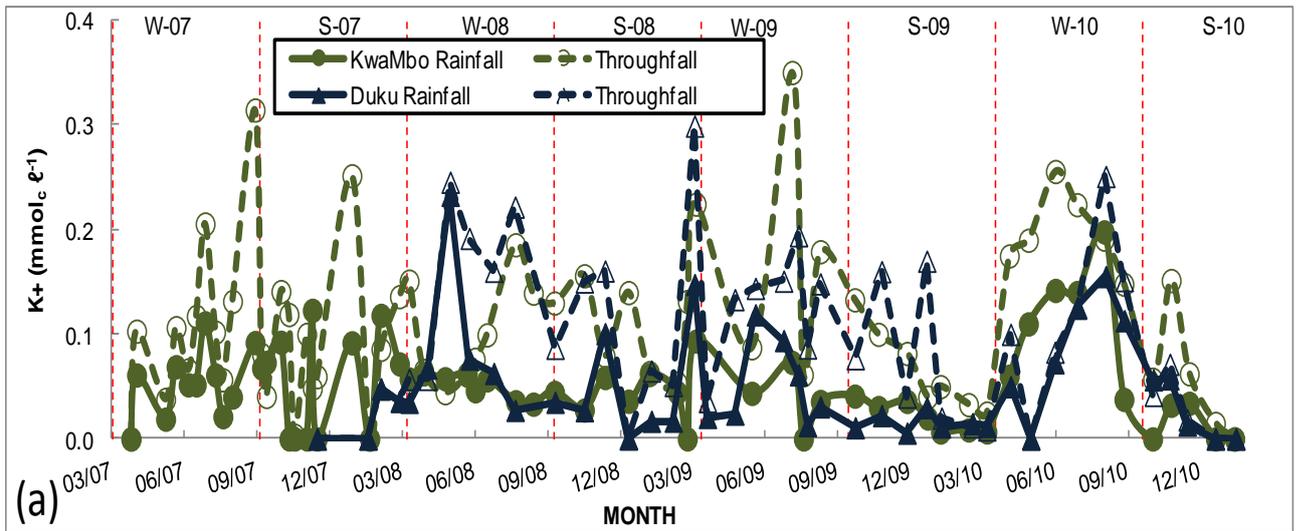


Figure 5.2: pH of rainfall and throughfall solutions volume weighted for each month at the northern Dukuduku (Duku) site and southern KwaMbonambi (KwaMbo) site. Vertical dashed lines represent the start and end of each six month wetter summer (S) and dryer winter (W) periods for each year. (Volume weighting was calculated using pH derived H^+ concentration).

5.4.2. Cation concentrations in throughfall and rainfall

Volume weighted concentrations are shown in **Figure 5.3a to d** for the elements assessed in rainfall and throughfall at each site for each six month period. Volume weighted mean concentrations of K^+ , Ca^{2+} and Mg^{2+} (**Figure 5.3a, b, c**) were often doubled as rainfall passed through the tree canopy after undergoing canopy exchange processes and collecting dry deposition from the tree canopies. K^+ was on average more concentrated than Ca^{2+} and Ca^{2+} more than Mg^{2+} in the rainfall and throughfall at both sites. Although Na^+ (**Figure 5.3d**) was the most concentrated of all the elements in the rainfall, its concentration did not increase by the same extent in passing through the canopy. Concentrations of the cations tended to be higher in smaller rainfall events and increased as the length of time between rainfall events increased. Concentrations of cations were also higher in all sources during the six-month winter periods at both sites. The largest differences in cation concentrations between rainfall and throughfall occurred during the winter periods as spikes in throughfall cation concentrations. Cation concentrations tended to be lower at KwaMbonambi than at Dukuduku in both the rainfall and throughfall, except for Mg^{2+} in the rainfall and K^+ in the throughfall (students T-test $p < 0.001$). Although cations were highly correlated with EC and each other, no significant relationships could be found between the ion concentrations in the rainfall and any other site factor.



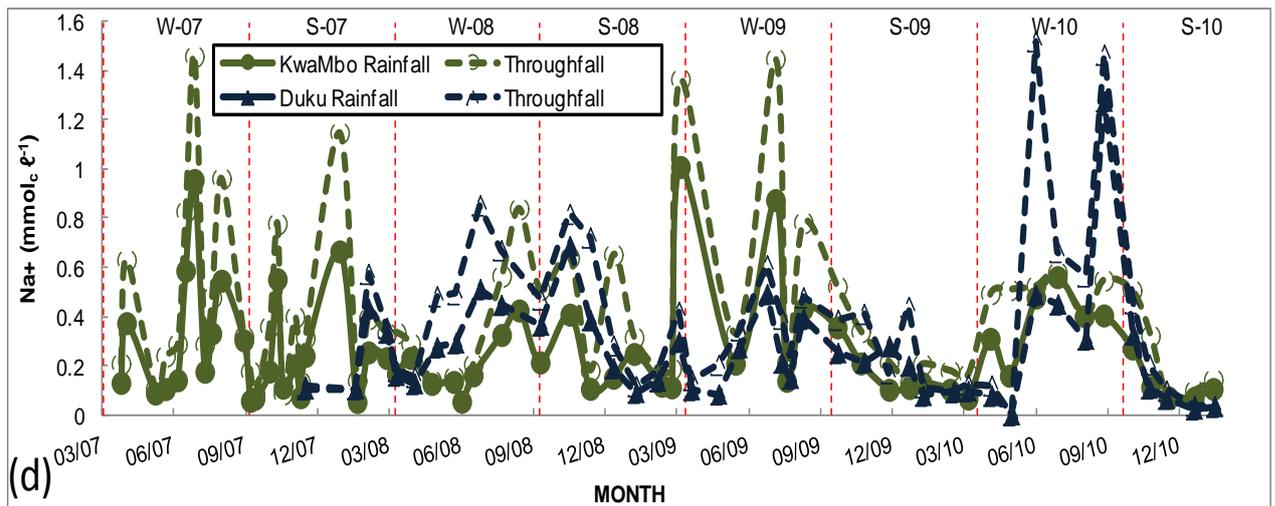


Figure 5.3: Concentrations of (a) K, (b) Ca, (c) Mg and (d) Na ($\text{mmol}_c \ell^{-1}$) in rainfall and throughfall solutions volume weighted for each month at the northern Dukuduku (Duku) site and southern KwaMbonambi (KwaMbo) site. Vertical dashed lines represent the start and end of each six month wetter summer (S) and dryer winter (W) periods for each year.

5.4.3. Nitrogen concentrations in throughfall and rainfall

The three forms of N assessed in the rainfall and throughfall are shown as weighted mean concentrations for each monthly period (**Figure 5.4a to c**). The units $\text{mg} \ell^{-1}$ are used here rather than ionic concentrations to compare the three forms. Mean concentrations of O-N, and NO_3^- -N were slightly higher in throughfall than in the rainfall at both sites, while NH_4^+ -N concentrations were often lower in the throughfall than the rainfall, being reduced through canopy exchange processes. O-N concentrations in the throughfall were on average the highest followed by NH_4^+ -N and NO_3^- -N at both sites. Rainfall NO_3^- -N concentrations were lowest at both sites with rainfall NH_4^+ -N and O-N concentrations being relatively similar. Concentrations of the three N forms (**Figure 5.4a-c**) were generally higher during the winter periods than during the summer periods at both sites, with throughfall N generally higher than rainfall throughout all periods. O-N, NH_4^+ -N and NO_3^- -N were less concentrated at KwaMbonambi (students T-test $p < 0.001$), with NO_3^- -N often and NH_4^+ -N occasionally below analytical detection limits. Although stemflow volumes were comparatively very low, concentrations of all elements were an order of magnitude higher in the stemflow solutions than in the rainfall and throughfall (**Appendix 5-1: and Appendix 5-2:**).

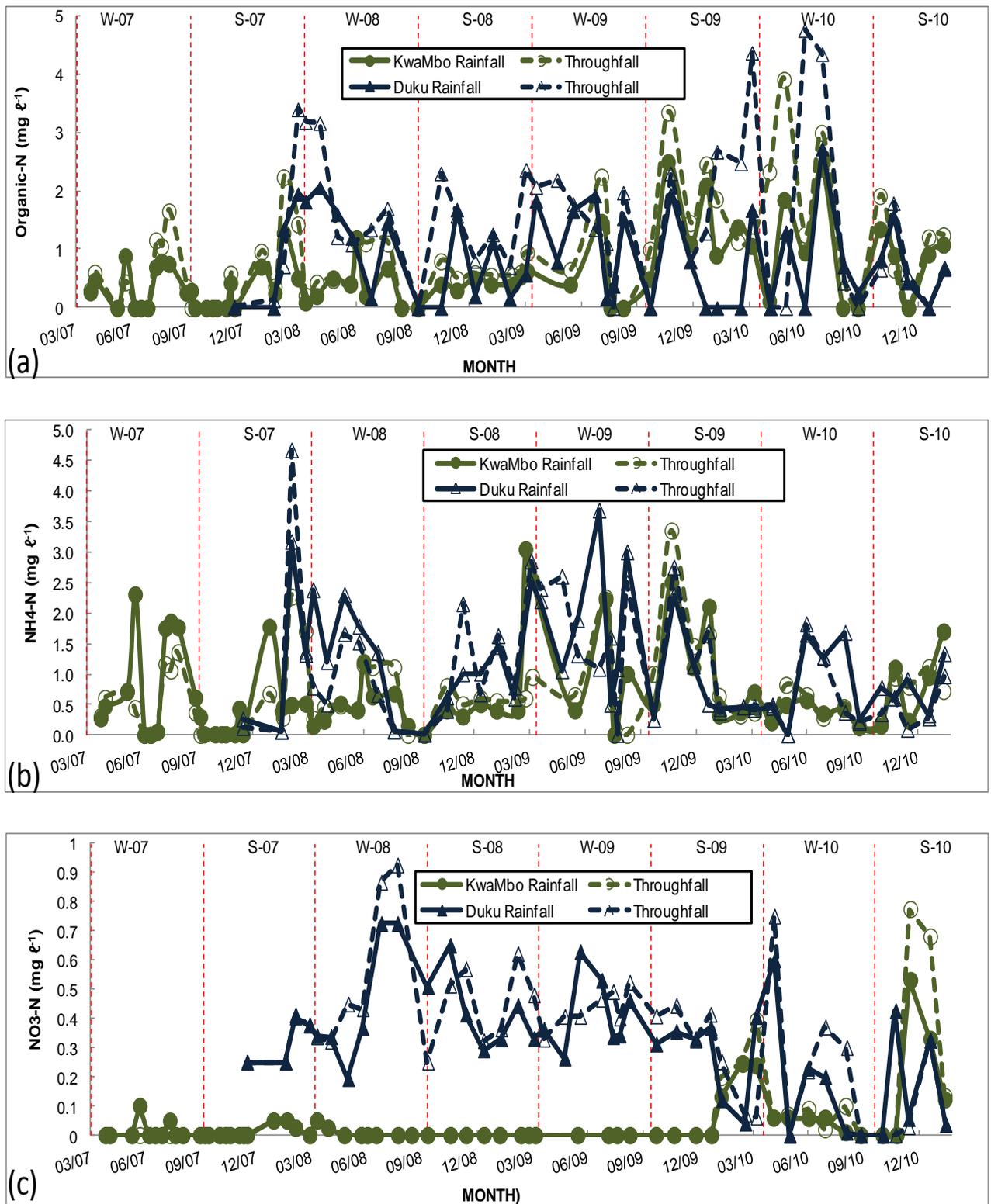


Figure 5.4: Concentrations of (a) Organic-N, (b) NH₄-N and (c) NO₃-N (mg l⁻¹) in rainfall and throughfall solutions volume weighted for each month at the northern Dukuduku (Duku) site and southern KwaMbonambi (KwaMbo) site. Vertical dashed lines represent the start and end of each six month wetter summer (S) and dryer winter (W) periods for each year. Note the different scale for the Y axes.

5.4.4. Cation mass fluxes

The six monthly mass additions of cations with rainfall (bulk wet deposition) and canopy drainage are shown for each site in **Table 5-2** with calculated canopy exchange, dry deposition and annualised totals. Total deposition as well as the sum of wet and dry deposition, is shown as annualised values only. Stemflow (**Appendix 5-1** and **Appendix 5-2**) contributed a small fraction to the total nutrients added with canopy drainage, around 10% of the cations and 5% of N at each site. The canopy exchange models yielded results suggesting substantial exchange between nutrient deposition and canopies which differed for N species and the base cations (**Table 5-2**).

Large quantities of Na^+ dominated cation fluxes at both sites (**Appendix 5-1** and **Appendix 5-2**) followed in decreasing order by K^+ , Ca^{2+} and Mg^{2+} at both sites. Cations addition with canopy drainage (**Table 5-2**) was larger than addition through rainfall, and rainfall larger than derived dry deposition at both sites. Rainfall, canopy drainage and dry cation addition was higher at the KwaMbonambi site than at the Dukuduku site throughout the study period. Canopy exchange processes and collection of dry deposition on the tree canopy greatly increased the quantities of cations in the rainfall after canopy drainage at both sites. This increase was largest for Mg^{2+} and least for K^+ at both sites.

Large quantities of base cations were lost from the tree canopies through exchange processes (positive numbers for canopy exchange in **Table 5-2**). Although the trends were not consistent, larger quantities of Ca^{2+} and K^+ and lower quantities of Mg^{2+} were exchanged at KwaMbonambi than at Dukuduku. These values, in most cases, tending to be higher during the six month summer periods than the winter periods at both sites. Summer Mg^{2+} exchange was on average higher at Dukuduku than at KwaMbonambi, while Ca^{2+} and K^+ were on average higher at KwaMbonambi than at Dukuduku during both seasons. The canopy exchange models on average calculated larger cation dry deposition at KwaMbonambi than at Dukuduku.

5.4.5. Nitrogen mass fluxes

Organic-N was a major contributor to N fluxes and more N was added through wet deposition in each form than through dry deposition at both sites. Rainfall and canopy drainage addition of

NO_3^- -N was lower than O-N and NH_4^+ -N rainfall and canopy drainage addition at both sites. Although the rainfall at KwaMbonambi was higher than Dukuduku (**Figure 5.1**), less N was added through wet deposition and canopy drainage at KwaMbonambi than Dukuduku (**Table 5-2**); owing to differences in concentrations between the two sites (**Figure 5.4**).

The predominant canopy exchange process with respect to N was that of uptake (negative numbers for canopy exchange in **Table 5-2**). Uptake of NO_3^- -N within the tree canopies was most apparent at Dukuduku, with minimal exchange at KwaMbonambi due to undetectable concentrations at that site for most of the study. The average uptake of NH_4^+ -N was slightly higher at KwaMbonambi and tended to be higher during the summer periods seasonal periods (most notable at KwaMbonambi). Derived dry deposition of NH_4^+ -N was however higher at KwaMbonambi than at Dukuduku. Similar NO_3^- -N dry deposition was derived for both sites.

The addition to rainfall O-N after canopy drainage was lower during the six-month winter periods at Dukuduku (**Table 5-2**) than during the summer periods. Although the six-month averages show KwaMbonambi O-N to increase in the rainfall after passing through the tree canopy, seasonal patterns were weak, with slight increases during the summer periods. Higher quantities of O-N were observed more frequently in canopy drainage at Dukuduku than at KwaMbonambi.

Table 5-2: Addition of N species, Ca²⁺, Mg²⁺ and K⁺ by wet deposition, canopy drainage, and dry deposition inferred from canopy exchange calculations for six month wetter summer (S) and dryer winter (W) periods for each year at the northern Dukuduku site and southern KwaMbonambi sites.

Season-year	DukuDuku						Kwambonambi						
	NH ₄ ⁺ -N	NO ₃ ⁻ -N	O-N	Ca ²⁺	Mg ²⁺	K ⁺	NH ₄ ⁺ -N	NO ₃ ⁻ -N	O-N	Ca ²⁺	Mg ²⁺	K ⁺	
	kg ha ⁻¹												
Wet deposition	W-07	n/a	n/a	n/a	n/a	n/a	n/a	3.0	0.0	1.0	1.4	0.4	5.9
	S-07	1.4	0.5	1.6	0.0	0.5	1.3	1.9	0.1	1.2	3.0	3.2	7.8
	W-08	3.3	1.3	3.3	1.2	1.2	5.8	1.7	0.1	1.6	1.6	1.9	10.6
	S-08	4.7	2.0	2.6	2.9	2.5	6.4	2.7	0.0	2.0	1.5	2.1	9.3
	W-09	3.8	0.9	2.6	4.5	0.7	2.6	1.4	0.0	1.0	8.3	2.2	4.1
	S-09	3.3	1.3	2.1	3.5	0.9	2.4	6.6	0.6	7.9	5.9	2.2	4.4
	W-10	1.2	0.6	0.7	1.6	1.3	4.8	0.7	0.1	1.8	4.4	5.3	9.4
	S-10	2.4	0.7	0.9	0.8	0.2	1.6	5.6	1.4	6.5	3.2	2.1	1.0
Dry deposition	W-07	n/a	n/a	n/a	n/a	n/a	n/a	0.5	0.0	0.1	0.4	0.1	2.2
	S-07	0.1	0.0	1.3	0.0	0.0	0.1	1.9	0.1	0.6	1.3	0.9	3.8
	W-08	0.6	0.3	1.0	0.1	0.1	1.1	2.7	0.0	0.5	1.4	1.7	5.0
	S-08	1.6	0.4	2.6	0.4	0.3	1.5	1.6	0.0	0.6	0.6	2.3	5.0
	W-09	1.6	0.4	1.0	1.4	0.0	1.1	0.7	0.0	0.5	3.3	0.8	1.4
	S-09	1.6	0.5	5.2	1.2	0.3	0.9	1.8	0.4	1.2	2.1	1.0	1.8
	W-10	0.0	0.2	0.4	0.5	0.4	2.0	0.6	0.0	1.7	0.3	1.1	1.2
	S-10	0.5	0.0	0.9	0.9	0.3	1.5	2.7	1.3	0.3	3.8	2.7	2.5
Canopy drainage	W-07	n/a	n/a	n/a	n/a	n/a	n/a	2.3	0.0	1.1	3.8	2.8	10.3
	S-07	1.3	0.5	3.0	1.5	1.0	1.4	2.1	0.1	1.8	7.3	6.8	17.5
	W-08	2.0	1.0	4.3	6.1	5.9	9.4	2.2	0.0	2.2	12.1	9.8	18.7
	S-08	4.8	2.0	5.2	7.0	7.0	14.7	2.6	0.0	2.6	9.4	6.1	16.8
	W-09	3.9	0.9	3.5	8.0	3.1	8.6	1.5	0.0	1.5	10.8	4.7	10.3
	S-09	3.5	1.4	7.3	8.7	4.7	11.1	6.4	0.7	9.1	11.0	5.6	12.7
	W-10	0.8	0.8	1.1	2.6	2.0	6.7	0.7	0.1	3.4	7.9	5.8	14.8
	S-10	1.7	0.6	1.8	2.6	2.2	3.8	5.9	2.8	6.8	8.5	6.2	7.6
Canopy exchange	W-07	n/a	n/a	n/a	n/a	n/a	n/a	-1.2	0.0	0.0	1.9	2.3	2.2
	S-07	-0.2	0.0	0.0	1.5	0.5	0.1	-1.7	-0.1	0.0	3.0	2.7	5.9
	W-08	-2.0	-0.5	0.0	4.8	4.5	2.5	-2.2	0.0	0.0	9.1	6.3	3.1
	S-08	-1.6	-0.4	0.0	3.7	4.2	6.8	-1.7	0.0	0.0	7.3	1.7	2.5
	W-09	-1.4	-0.4	0.0	2.1	2.4	4.9	-0.6	0.0	0.0	-0.7	1.6	4.8
	S-09	-1.4	-0.4	0.0	4.0	3.5	7.7	-2.1	-0.3	0.0	3.0	2.4	6.5
	W-10	-0.5	-0.1	0.0	0.4	0.3	-0.1	-0.6	0.0	0.0	3.3	-0.6	4.2
	S-10	-1.2	0.0	0.0	0.9	1.7	0.7	-2.4	-0.4	0.0	1.5	1.4	4.1
mean annual	wet	6.2	2.3	4.3	4.5	2.3	7.7	5.9	0.5	5.8	7.3	4.8	13.1
	dry	1.9	0.5	3.8	1.4	0.4	2.5	3.1	0.5	1.4	3.3	2.7	5.7
	canopy	5.5	2.2	8.1	11.3	8.0	17.3	5.9	0.9	7.1	17.7	11.9	27.1
	exchange	-2.6	-0.6	0.0	5.4	5.3	7.0	-3.1	-0.2	0.0	7.1	4.4	8.3
	Total	8.1	2.8	8.1	6.0	2.7	10.2	9.0	1.0	7.1	10.6	7.5	18.8

Negative values imply a loss from rainfall and canopy uptake, positive implies a gain to rainfall and a loss from the canopy

5.5. DISCUSSION

Many international and local studies and monitoring programmes have recorded atmospheric deposition for various purposes and have yielded broad ranges in nutrient quantities added. Ranges in total additions across Europe, North and South America, Asia, Oceania, Africa and South Africa are presented in **Table 5-3**. These studies cover an array of wet, dry and total deposition across numerous regions, include forestry of multiple species (both natural and planted), and represented many replicates of samples collected for periods of between five and 20 years. Data from South Africa only includes wet and dry deposition outside of forestry canopies. The ranges in nutrient addition through atmospheric deposition were strongly influenced by atmospheric pollution levels, distance from industrial areas, wind direction, distance from the ocean, wind fetch and rainfall patterns. The addition of macro-nutrients by atmospheric deposition in our study falls within the ranges described in **Table 5-3** for Ca^{2+} , Mg^{2+} , NO_3^- -N, NH_4^+ -N at both sites. Addition of K^+ at both sites was at the higher end of the ranges presented in **Table 5-3**. Organic-N was also above these ranges, although very few sites assessed O-N. Sites upon which O-N was assessed in **Table 5-3** only recorded dissolved O-N and not particulate O-N.

Table 5-3: Means and ranges in nutrient additions with bulk rainfall derived from numerous global studies (kg ha⁻¹ yr⁻¹). Ranges are shown in round brackets and numbers of studies are shown in square brackets

Region	NO ₃ ⁻ -N	NH ₄ ⁺ -N	DON	TN	Ca ²⁺	Mg ²⁺	K ⁺	Na ⁺	P	References
Africa	2.7 (0.3-7.2) [17]	13.2 (0.9-41.7) [17]	nd (nd) [0]	15.8 (1.7-48.9) [18]	2.5 (1.1-7.3) [7]	1 (0.2-3.2) [7]	9.8 (0.9-36.2) [10]	6.5 (0.9-21.7) [7]	0.9 (0.6-1.2) [2]	(Adeniyi 2006; Galy-Lacaux et al. 2009; Laclau et al. 2010b; Josipovic et al. 2011)
Europe	7.2 (1.2-14.2) [11]	10.5 (2.2-25.7) [16]	1.4 (1.4-1.4) [1]	16.8 (3.4-33) [20]	8.8 (1.9-24.7) [17]	2.4 (0.5-4.6) [12]	4.4 (2-9) [13]	13.4 (3.2-33.5) [8]	0.4 (0.4-0.4) [2]	(Jordan et al. 1980; Gundersen et al. 1998; Andersen and Hovmand 1999; Jenssen et al. 2002; Zimmermann et al. 2002; de Vries et al. 2003; Talkner et al. 2010)
North America	3.5 (1.6-4.6) [5]	2.1 (1-3.7) [4]	1.8 (1-2.7) [2]	9.3 (4.8-22.7) [10]	5.9 (0.4-19.3) [21]	1.9 (0.4-8.8) [11]	2 (0.9-8.1) [11]	2.8 (1-10.5) [9]	0.2 (0.1-0.4) [10]	(Scheider et al. 1979; Jordan et al. 1980; Johnson et al. 1995; Williams and Melack 1997)
South America	1.2 (0.7-2.5) [5]	9.5 (0.5-21.7) [8]	1.3 (1.3-1.3) [1]	10.5 (1.2-21.7) [9]	17.1 (1.3-29.3) [13]	2.5 (0.3-5.5) [9]	10.7 (1.1-24.6) [8]	5.9 (2-16.6) [5]	10.4 (0.3-24.9) [5]	(Jordan et al. 1980; Mayer et al. 2000)
Oceania	1.3 (0.5-1.8) [3]	0.7 (0.5-1) [3]	3.9 (2.7-5.1) [3]	5.8 (3.7-7.5) [3]	7.3 (0-12.5) [5]	3.4 (2.5-5.4) [4]	3.7 (2-5.1) [4]	15.8 (10.3-22.4) [4]	0.2 (0.1-0.5) [3]	(Jordan et al. 1980; Adams and Attiwill 1991)
Asia	7.5 (7.5-7.5) [1]	14.1 (14.1-14.1) [1]	nd (nd) [0]	14.8 (8-21.6) [2]	14.5 (4.9-24.7) [3]	4.6 (4.6-4.6) [1]	5.7 (4.4-6.9) [2]	15 (3.2-26.8) [2]	0.4 (0.4-0.4) [1]	(Zhang et al. 2006)
South Africa	6.1 (0-24.9) [14]	2.4 (1-7.2) [12]	nd.0 (nd) [0]	7.4 (1.1-13.7) [12]	8.0 (0.9-20.6) [18]	1.3 (0.1-5.4) [18]	2.9 (0.5-6.7) [18]	nd.0 (nd) [6]	0.8 (0.5-1) [3]	(van Wyk 1990; Olbrich 1995; Zunckel et al. 2000; Lowman 2004; Galy-Lacaux et al. 2009; Josipovic et al. 2011)

nd = no data

The Zululand plantations are likely to receive deposition from a number of sources including the ocean, industrial pollution, biomass burning, vehicle emissions, rural fires and possibly inland coal fired power stations (Annegarn 2005). The southern plantation area is in close proximity to the Richards Bay industrial area while the northern area is surrounded by plantation forests and sugarcane plantations that are burned on a regular basis. Isotopic analyses of air borne dust for an inland site showed that cations were derived primarily from biomass burning (46%) and coal burning (47%) (Piketh and Annegarn 1994), while the contribution of ocean sources was small, being distant from the ocean. Annegarn (2005) presented some evidence that pollution is carried from as far as central southern Africa, passing over the northern Zululand region, recirculation from the ocean back onto the land. This is a substantial pollution source from inland biomass burning that has not been accounted for in the past. Lara et al. (2005) shows sugarcane burning to contribute a major proportion of fine particulate material in Southeastern Brazil, followed by dust. Soil particles, oil combustion and sugarcane burning made up the major proportions of coarse particulate material in that study.

The values for below canopy cation deposition were 2 to 4 times that added by rainfall. Potassium had the largest mass addition of the cations in our study. Mayer et al. (2000) also presented data with large increases in cations under the tree canopy (2 to 13 times) in a tropical rain forest, with K^+ enrichment being the largest. Canopy exchange in Talkner et al. (2010) under a Germany beech stand was also high and the largest for K^+ . Canopy exchange was however far lower in two clonal *Eucalyptus* studies (Congo and Brazil) with a zero K^+ exchange at the Congo site (Laclau et al. 2010b). The infestation by the sap-sucking *Thaumastocoris* insects in our study sites may have induced larger levels of K^+ leaching from the tree canopy through the epidermal damage to the leaves, possibly explaining the high cation loss from the tree canopies.

Quantities of O-N measured in this study are substantial and provided it is in a mineralisable form, may provide a significant source of N to the trees. The contribution of O-N to deposition has been questioned in recent years by a number of authors who have expressed its neglect and importance (Cornell 2011). These authors cite O-N to make up around 30% of total N deposition (Flessa and Hunt 2001; Neff et al. 2002). Long range dust and smoke transport and marine (sea spray) have been noted as possible sources of atmospheric O-N. Studies have shown sea spray to be highly enriched with organic matter, particularly in areas of high marine

biological activity (Rinaldi et al. 2010). This work is fairly recent and numerous authors state the problem of lack of data and highly variable results (Cornell 2011).

Growth on infertile sites characterised by relatively low soil organic carbon and total N levels (in Brazil, Congo, China, Australia, similar to our sites), was reduced where large quantities of residues were removed Nambiar et al., (2004) summarised in Tiarks and Ranger, (2008). Residue removal did not result in tree growth responses in the newly planted crop up to two years of age (Chapter 6). The high O-N concentrations found in our study and the high concentrations that were excluded also seemed to coincide with surface soil N mineralisation spikes recorded at the Dukuduku site (unpublished data). This may give further evidence supporting the high levels of N recorded at our study sites, which will be discussed in future publications.

Results presented in our study may however be an artefact of the canopy exchange model used, a model produced for ion exchange processes. The lesser increase in Na^+ concentrations in passing through the canopy gives some agreement toward the assumption that Na^+ present in the rainfall is not taken up by the tree canopy to any significant degree. In development of this method, there may have been an exchange of Na^+ in the tree canopy that was kept in balance through Na^+ uptake by the root system resulting in the throughfall Na^+ levels seeming to remain unaffected. Canopy uptake of Na^+ can occur during leaf development as reported for deciduous trees in Staelens et al. (2007) which is a particularly problem in high Na^+ marine environments (Staelens et al. 2008). An uptake of Na^+ ions by the tree canopy will therefore result in this model underestimating dry deposition and overestimating canopy exchange of cations.

5.5.1. Significance of the data

A scenario of nutrient loss with harvesting is presented in **Table 5-4** considering an average oven dry stem-wood harvest of 150 Mg ha^{-1} over a seven year rotation and using a simplistic assessment of nutrient loss for plantation grown *Eucalyptus* (Dovey 2009). Assuming similar total deposition values to those in **Table 5-2** over the seven year rotation (excluding the temporary unplanted period between felling and planting) additions with atmospheric deposition have been calculated in **Table 5-4**. The scenario given in **Table 5-4** depicts an average to high

site productivity range for the Zululand region with many sites having productivities falling above or below this value within the Zululand region. Productivity ranges reported by Smith et al. (2006) showed mean annual increments to range between 18 and 58 m³ ha⁻¹ yr⁻¹, with site index at five years ranging between 18.0 to 27.3 m with rotation lengths between five and eight years.

Table 5-4: A hypothetical calculation of nutrient loss by removal of various tree components assuming 150 Mg ha⁻¹ of stemwood and potential addition of nutrients over a seven year rotation (Mineral and organic N shown separately)

Component	biomass Mg ha⁻¹	Total N	Ca	Mg	K
		kg ha⁻¹			
stem-wood	150	128	239	32	156
Bark	18	45	177	56	79
Branches	18	49	68	23	71
Total	186	222	483	111	306
Site	Total Deposition				
	Mineral-N	Organic-N	Ca	Mg	K
Dukuduku	76.4	56.6	41.7	18.8	71.6
KwaMbonambi	70.2	49.8	74.1	52.7	131.8

These data suggest atmospheric deposition to play a major role in the addition of macronutrients to commercial forest plantations in this region. Of the cations K⁺ and Mg²⁺ may be replaced in quantities exceeding removals with stem-wood harvesting at the KwaMbonambi site, while only a small fraction of Ca²⁺ is replaced from atmospheric sources. The KwaMbonambi site, with an apparently larger cation flux, may therefore regain cation losses more rapidly than Dukuduku, particularly for Mg²⁺ and K⁺. The addition of N as NO₃⁻-N and NH₄⁺-N alone has potential to replace significant quantities of N that may be lost with harvesting. Although the fate of O-N is unknown within the forest nutrient cycle, it is deposited in quantities sufficient to outweigh N losses associated with stem-wood harvesting when added to inorganic N.

Higher productivity sites will incur a greater nutrient loss with harvesting, relative to the gains for a given rate of atmospheric deposition. Many other management scenarios that increase nutrient losses must also be included in the assessment, such as removal of tree foliage, burning of residues and the burning and removal of the forest litter layer. Significant improvements in stand growth have been observed after fertilization with N after re-establishment on the Zululand coastal plain (du Toit and Oscroft 2003). On degraded sites, even larger responses have been recorded to fertilization containing mainly N (du Toit et al. 2001). This supports the findings of

the current study, namely that the system remains reliant on inputs of N (and possibly Ca) as supplement to atmospheric deposition inputs, if high levels of productivity is to be maintained. Other natural loss mechanisms (such as leaching), not included here, should also be added to the nutrient loss calculation and should be included in any nutrient budget assessment. It is also clear from **Table 5-4** that the removal of bark and/or branches for off-site debarking, or as an energy fuel source has potential to significantly increase nutrient losses, especially Ca^{2+} . Under this scenario it is unlikely that Ca^{2+} will be replaced over a single rotation under the levels of atmospheric deposition recorded in this study.

5.5.2. Further comments

This study needs to be continued and replicated across a wider range of sites and include various tree species and planting densities. Carbon, sulphur and micronutrient deposition may also need to be included in future studies in order to draw more complete canopy exchange models, nutrient balances and assess deposition rates of other nutrient elements. Such work also needs to continue into the long term as the results presented here represent the period of collection and may not represent future deposition patterns. Furthering this work will create an extremely valuable dataset of atmospheric nutrient flux patterns, contributing significantly to nutrient budget calculations used in assessing sustainable nutrient management practices, while potentially explaining or predicting responses to fertiliser addition. Atmospheric deposition should first be assessed on highly productive, short-rotation, nutrient-poor forestry sites given the potential for these sites to export large quantities of nutrients with harvesting. This is important when attempting to understand the balance between nutrient loss and gain under various management regimes. Such information, in conjunction with soil and biomass nutrient pool data, may be used to indicate the key processes affecting nutritional sustainability and consequently the site specific long-term productivity of forestry plantations in South Africa.

5.6. CONCLUSION

Although this study considers just three years of data at two sites, it gives evidence towards atmospheric deposition being an important nutrient source within the commercial forestry regions of the Zululand coastal plains of South Africa. Given the rates recorded here atmospheric deposition is likely to play a major role in nutrient supply to the plantations within this area, provided that management practices do not accelerate nutrient losses through increased

biomass removal, shorter rotations or more nutrient demanding management practices. Under the conditions of these fast-growing plantation forests it may however become necessary to apply nutrients through fertilisation (Ranger et al. 2002) (organic, mineral, added N₂ fixing species) to compensate for the nutrients lost by stemwood and additional biomass removal after stand harvesting.

5.7. APPENDICES

Appendix 5-1: Dukuduku rainfall, throughfall and stemflow solution concentration means (Mean), standard deviation (SD) and 5% confidence intervals (CI). Total N is the sum of ammonium, nitrate and organic N.

Source	Stats	pH	Ec $\mu\text{S cm}^{-1}$	Na	Ca	Mg mg L^{-1}	K mg L^{-1}	Total N	NH ₄ -N $\mu\text{g L}^{-1}$	NO ₃ -N $\mu\text{g L}^{-1}$
Rainfall	SD	0.53	87.47	5.40	1.92	1.03	2.14	1.60	0.93	0.19
	Mean	5.29	79.30	6.29	1.23	0.66	1.96	2.28	1.19	0.36
	CI	0.11	19.93	1.15	0.41	0.22	0.46	0.34	0.21	0.04
Throughfall	SD	0.36	137.92	7.89	3.03	1.84	3.20	1.89	1.06	0.23
	Mean	5.28	124.70	9.22	2.76	1.94	3.85	2.96	1.06	0.37
	CI	0.08	32.31	1.68	0.64	0.39	0.68	0.40	0.23	0.05
Stemflow	SD	0.36	245.01	11.51	5.29	3.98	5.15	2.62	1.35	0.41
	Mean	4.89	197.10	12.50	5.52	4.41	6.83	3.36	1.22	0.51
	CI	0.08	56.59	2.45	1.12	0.85	1.10	0.56	0.29	0.09

Appendix 5-2: Kwambonambi rainfall, throughfall and stemflow solution concentration means (Mean), standard deviation (SD) and 5% confidence intervals (CI). Total N is the sum of ammonium, nitrate and organic N.

Source	Stats	pH	Ec $\mu\text{S cm}^{-1}$	Na	Ca	Mg mg L^{-1}	K mg L^{-1}	Total N	NH ₄ -N $\mu\text{g L}^{-1}$	NO ₃ -N $\mu\text{g L}^{-1}$
Rainfall	SD	5.13	69.88	5.73	1.09	0.82	1.87	1.39	0.70	0.12
	Mean	5.13	69.88	5.73	1.09	0.82	1.87	1.39	0.70	0.12
	CI	0.09	11.67	0.94	0.30	0.20	0.31	0.24	0.14	0.03
Throughfall	SD	0.40	103.57	6.61	2.34	1.65	2.92	1.50	0.72	0.05
	Mean	5.38	108.30	9.02	2.57	1.84	4.17	1.78	0.75	0.18
	CI	0.10	25.18	1.76	0.59	0.40	0.71	0.37	0.18	0.05
Stemflow	SD	0.32	94.54	16.43	6.89	10.60	12.50	1.46	0.66	0.98
	Mean	5.09	176.44	16.71	8.07	5.64	11.11	1.68	0.79	0.33
	CI	0.08	22.98	4.34	1.73	2.58	3.04	0.35	0.16	0.24

Appendix 5-3: Rainfall, pH and nutrient addition with bulk rainfall at Dukuduku

Date	Volume mm	pH	Na	K	Ca	Mg kg ha⁻¹	O-N	NH₄-N	NO₃-N
Dec-07	47.6	5.7	1.15	0.00	0.00	0.00	0.00	0.12	0.12
Feb-08	38.6	5.4	1.22	0.05	0.00	0.00	0.02	0.12	0.10
Mar-08	84.9	5.4	6.48	1.21	0.00	0.52	1.62	1.17	0.32
Apr-08	94.8	5.2	2.81	2.42	0.00	0.00	1.94	1.23	0.32
May-08	20.5	6.4	0.84	0.89	0.08	0.00	0.36	0.31	0.07
Jun-08	91.3	5.7	6.13	1.21	0.00	0.00	0.67	1.62	0.34
Jul-08	11.9	5.5	1.45	0.29	0.19	0.20	0.02	0.16	0.09
Aug-08	23.4	5.5	2.47	0.25	0.32	0.32	0.34	0.01	0.17
Sep-08	55.5	5.0	5.20	0.76	0.63	0.72	0.00	0.01	0.28
Oct-08	40.0	5.1	4.36	0.51	0.60	0.64	0.00	0.08	0.22
Nov-08	77.1	5.3	11.21	1.22	1.62	1.60	0.24	0.53	0.47
Dec-08	31.0	5.0	2.37	0.89	0.26	0.24	0.40	0.31	0.12
Jan-09	95.3	5.0	3.03	0.20	0.00	0.00	0.65	1.18	0.30
Feb-09	167.2	5.5	4.36	1.28	0.00	0.00	1.07	1.72	0.65
Mar-09	72.1	5.7	3.19	2.34	0.43	0.00	0.22	0.91	0.29
Apr-09	76.3	5.7	2.28	0.71	0.19	0.00	1.26	1.71	0.27
May-09	74.1	5.6	1.43	0.64	0.00	0.00	0.58	0.78	0.20
Jun-09	25.8	6.0	1.85	0.57	0.11	0.50	0.44	0.57	0.16
Jul-09	10.2	5.4	0.67	0.28	0.53	0.05	0.06	0.14	0.04
Aug-09	27.4	5.6	1.61	0.22	2.11	0.12	0.24	0.53	0.11
Sep-09	32.3	5.3	1.88	0.14	1.59	0.00	0.00	0.10	0.10
Oct-09	55.0	5.2	3.19	0.23	2.69	0.00	0.00	0.17	0.17
Nov-09	70.5	4.5	3.48	0.62	0.00	0.00	1.37	1.62	0.25
Dec-09	161.7	4.5	10.41	0.35	0.00	0.00	0.12	0.46	0.53
Jan-10	82.9	5.0	2.12	0.54	0.25	0.30	0.00	0.37	0.16
Feb-10	84.0	4.7	1.67	0.44	0.33	0.36	0.00	0.38	0.07
Mar-10	76.2	5.2	1.66	0.27	0.23	0.24	0.57	0.34	0.16
Apr-10	86.6	5.8	1.55	1.69	0.00	0.00	0.00	0.39	0.52
May-10	11.5	5.1	0.20	0.21	0.00	0.00	0.00	0.05	0.07
Jun-10	20.1	5.5	2.16	0.80	0.53	0.39	0.30	0.29	0.04
Jul-10	7.4	5.1	0.76	0.36	0.14	0.12	0.20	0.09	0.01
Aug-10	25.1	5.0	2.76	1.45	0.52	0.43	0.16	0.36	0.00
Sep-10	6.3	5.1	1.84	0.28	0.44	0.31	0.02	0.01	0.00
Oct-10	126.2	5.3	1.62	0.72	0.52	0.17	0.32	0.18	0.07
Nov-10	113.2	4.7	1.63	0.89	0.31	0.00	0.37	0.72	0.14
Dec-10	42.8	4.6	0.38	0.00	0.00	0.00	0.00	0.24	0.23
Jan-11	413.4	4.6	0.95	0.00	0.00	0.00	0.22	1.12	0.26
Feb-11	29.8	4.7	0.09	0.00	0.00	0.00	0.00	0.11	0.00

Appendix 5-4: Throughfall, pH and nutrient addition with throughfall at Dukuduku

Date	Volume mm	pH	Na	K	Ca	Mg kg ha⁻¹	O-N	NH₄-N	NO₃-N
Dec-07	46.0	5.8	1.22	0.00	0.52	0.00	0.01	0.06	0.11
Feb-08	35.5	5.5	1.10	0.04	0.03	0.03	0.09	0.08	0.09
Mar-08	76.9	5.6	6.32	1.09	0.77	0.83	2.58	1.12	0.29
Apr-08	79.8	5.0	2.88	1.78	1.45	1.05	2.53	0.42	0.26
May-08	15.1	5.7	0.86	0.65	0.33	0.29	0.40	0.12	0.05
Jun-08	78.9	5.1	8.91	2.66	1.61	1.89	0.86	1.20	0.34
Jul-08	9.1	5.2	1.78	0.57	0.28	0.25	0.12	0.06	0.08
Aug-08	17.3	5.3	2.66	1.50	0.90	0.81	0.29	0.01	0.16
Sep-08	44.0	5.3	4.76	1.49	1.00	1.13	0.00	0.00	0.11
Oct-08	30.3	5.3	4.05	1.27	0.99	0.99	0.22	0.04	0.10
Nov-08	63.4	5.4	11.65	3.77	3.25	3.06	1.37	0.47	0.33
Dec-08	24.3	5.3	3.45	1.16	0.85	1.05	0.34	0.44	0.12
Jan-09	87.5	5.1	4.06	1.10	0.69	0.72	0.90	1.00	0.30
Feb-09	148.9	5.3	5.02	3.38	0.00	0.00	1.25	1.99	0.73
Mar-09	58.1	5.5	3.48	3.11	0.44	0.52	0.94	0.67	0.33
Apr-09	68.2	5.7	2.82	1.91	0.20	0.24	1.44	1.68	0.24
May-09	60.1	5.2	2.82	2.67	1.08	1.19	1.32	1.56	0.24
Jun-09	20.6	5.1	1.84	1.17	0.60	0.64	0.44	0.18	0.09
Jul-09	5.8	5.2	0.54	0.45	0.53	0.14	0.07	0.09	0.03
Aug-09	20.7	5.4	1.62	0.91	2.23	0.27	0.17	0.21	0.09
Sep-09	25.7	5.1	2.29	0.77	2.57	0.31	0.00	0.08	0.11
Oct-09	43.7	5.1	3.89	1.31	4.37	0.52	0.00	0.14	0.18
Nov-09	53.5	4.7	5.05	3.35	1.61	0.84	1.23	1.47	0.24
Dec-09	147.4	5.1	5.76	2.31	0.00	0.00	0.10	0.41	0.48
Jan-10	72.2	5.2	3.31	1.68	0.67	1.03	1.67	0.52	0.21
Feb-10	70.5	5.0	1.67	0.42	0.45	0.46	1.80	0.30	0.11
Mar-10	66.4	5.6	1.65	0.26	0.46	0.40	2.22	0.31	0.04
Apr-10	73.1	5.8	2.02	2.86	0.00	0.00	0.00	0.36	0.55
May-10	8.1	5.4	0.22	0.32	0.00	0.00	0.00	0.04	0.06
Jun-10	14.3	5.4	3.23	0.61	0.89	0.66	0.64	0.21	0.04
Jul-10	4.7	5.1	0.72	0.23	0.18	0.16	0.21	0.06	0.02
Aug-10	19.8	5.3	3.19	1.83	0.79	0.63	0.07	0.07	0.05
Sep-10	4.7	5.4	1.57	0.28	0.41	0.31	0.00	0.01	0.00
Oct-10	114.7	5.3	1.58	0.52	0.61	0.34	0.27	0.11	0.00
Nov-10	95.6	5.1	2.16	0.93	0.54	0.25	0.34	0.17	0.02
Dec-10	37.1	5.0	0.53	0.00	0.00	0.00	0.00	0.17	0.20
Jan-11	385.4	5.0	1.06	0.00	0.00	0.00	0.22	0.79	0.23
Feb-11	25.8	5.3	0.08	0.00	0.00	0.00	0.00	0.09	0.00

Appendix 5-5: Stem flow, pH and nutrient addition with stem flow at Dukuduku

Date	Volume mm	pH	Na	K	Ca	Mg kg ha⁻¹	O-N	NH4-N	NO3-N
Dec-07	3.2	4.4	0.11	0.07	0.03	0.04	0.01	0.00	0.01
Feb-08	2.8	4.6	0.16	0.04	0.02	0.03	0.01	0.02	0.01
Mar-08	4.6	4.6	0.44	0.17	0.10	0.11	0.27	0.03	0.02
Apr-08	4.3	4.7	0.14	0.09	0.05	0.09	0.08	0.09	0.01
May-08	0.7	4.5	0.02	0.01	0.01	0.01	0.01	0.01	0.00
Jun-08	5.0	4.9	0.35	0.49	0.29	0.20	0.03	0.05	0.02
Jul-08	0.5	5.0	0.05	0.04	0.03	0.02	0.01	0.00	0.00
Aug-08	1.8	4.6	0.25	0.00	0.00	0.00	0.00	0.00	0.01
Sep-08	3.0	4.9	0.41	0.13	0.13	0.11	0.00	0.00	0.00
Oct-08	2.0	4.7	0.32	0.11	0.11	0.09	0.00	0.00	0.00
Nov-08	4.5	5.0	0.89	0.31	0.33	0.29	0.00	0.01	0.02
Dec-08	0.7	4.8	0.08	0.02	0.02	0.02	0.01	0.01	0.00
Jan-09	4.4	5.1	0.42	0.23	0.15	0.18	0.08	0.05	0.02
Feb-09	7.8	5.0	0.30	0.17	0.11	0.07	0.05	0.07	0.03
Mar-09	3.1	5.2	0.18	0.09	0.07	0.03	0.00	0.01	0.01
Apr-09	3.4	5.4	0.10	0.06	0.03	0.02	0.00	0.01	0.01
May-09	3.8	5.0	0.13	0.16	0.07	0.05	0.05	0.07	0.01
Jun-09	1.1	4.9	0.08	0.16	0.18	0.13	0.02	0.03	0.01
Jul-09	0.1	5.0	0.02	0.01	0.01	0.01	0.00	0.00	0.00
Aug-09	2.2	5.0	0.26	0.15	0.26	0.07	0.04	0.05	0.01
Sep-09	2.3	5.5	0.32	0.18	0.25	0.05	0.00	0.00	0.01
Oct-09	3.2	5.4	0.45	0.25	0.36	0.07	0.00	0.00	0.02
Nov-09	3.6	5.3	0.38	0.25	0.14	0.08	0.08	0.10	0.02
Dec-09	10.0	4.9	0.94	0.59	0.32	0.70	0.00	0.04	0.05
Jan-10	2.5	5.0	0.28	0.21	0.10	0.06	0.06	0.07	0.01
Feb-10	3.3	4.7	0.32	0.20	0.11	0.18	0.02	0.04	0.02
Mar-10	4.4	4.6	0.41	0.26	0.14	0.31	0.07	0.09	0.02
Apr-10	4.5	5.4	0.15	0.19	0.08	0.06	0.06	0.01	0.04
May-10	0.4	4.6	0.01	0.02	0.01	0.01	0.01	0.00	0.00
Jun-10	1.2	4.4	0.24	0.12	0.10	0.07	0.04	0.02	0.01
Jul-10	0.3	4.4	0.06	0.03	0.03	0.02	0.01	0.00	0.00
Aug-10	1.9	4.9	0.31	0.18	0.08	0.06	0.02	0.00	0.00
Sep-10	0.2	4.7	0.04	0.01	0.01	0.01	0.00	0.00	0.00
Oct-10	7.2	5.2	2.36	0.66	0.86	0.60	0.32	0.03	0.03
Nov-10	7.1	4.9	0.47	0.33	0.21	0.16	0.13	0.01	0.00
Dec-10	1.1	5.2	0.08	0.10	0.03	0.06	0.03	0.00	0.01
Jan-11	20.4	5.3	1.20	1.26	0.37	0.74	0.44	0.27	0.08
Feb-11	1.2	4.8	0.05	0.04	0.01	0.02	0.01	0.03	0.00

CHAPTER 6:
**EFFECTS OF CLEARFELLING AND RESIDUE MANAGEMENT ON NUTRIENT
POOLS, EARLY GROWTH AND ABOVE GROUND NUTRIENT
ACCUMULATION³**

6.1. ABSTRACT

Research on sandy tropical soils with low soil nutrient retention and storage capacity has shown growth reductions through nutrient loss after various levels of residue removal. Growth and above ground nutrient pools and fluxes were therefore compared on a sandy low nutrient clonal *Eucalyptus* site on the Zululand Coastal Plain in South Africa after clearfelling to a year after canopy closure. Clearfelled areas were subjected to residue management treatments that included retention with and without fertilisation, residue burning, whole tree removal and residue doubling. An additional treatment was included as a point of reference by leaving undisturbed replicates of trees on-site. Stem only removal retained large quantities of nutrients on-site, while whole tree removal induced the greatest nutrient loss, exceeded only by N lost during residue burning. Growth and nutrient accumulation was initially most rapid in the burned and fertilised treatments. However, growth slowed in the burned treatment resulting in no significant difference from the remaining treatments from a year after planting onwards. Fert. treatment growth remained high while growth in all other treatments remained statistically similar. Nutrient supply in the Burn treatment remained adequate. Foliar vector analysis suggested fertilisation to have overcome N and P at six months and Mg and Ca deficiencies at one year after planting compared to the No-Burn treatment. Results in our study conflicted with research on similar sandy tropical sites that experienced a growth reduction after residue removal and a growth increase after residue doubling. Nutrient pools in the residues, litter layer and soil at our site supplied nutrients in excess of early growth demands. Mature crop data indicated nutrient recycling to limit demand later, allowing high atmospheric deposition rates to partially replenish nutrients lost during clearfelling. These factors may slow the rate of nutrient decline, particularly nitrogen under low nutrient removal management practices. The risk of these sites becoming depleted in subsequent rotations is increased as larger nutrient removal places a heavier reliance on belowground pools.

³ Submitted for publication to Southern Forests.

6.2. INTRODUCTION

Plantation forest management has aimed at producing a sustainable wood supply from intensive management through improvements in genetics, silviculture and harvesting (Goncalves 2004). Despite productivity gains, sustainability of yield in the long-term remains a major concern to management and scientists, and research into principles and processes driving site productivity is ongoing (Nambiar 1996; du Toit and Scholes 2002; Laclau et al. 2003a; Watt et al. 2005; du Toit et al. 2010). Poor management has potential to cause degradation of soil structure, chemistry, carbon (micro-organisms included) and site nutrient supply and storage (du Toit et al. 2010).

The early stage of a short-rotation pulpwood plantation growth cycle, from felling to post canopy closure of the subsequent crop, represents the period of greatest opportunity for site manipulation, as the most silvicultural operations take place during this period (Nambiar and Kallio 2008; Tiarks and Ranger 2008; du Toit et al. 2010; Smethurst 2010). This period also presents opportunities to undertake management practices that promote sustainability, as the silvicultural operations undertaken here have potential to impact on future site resource supply and productivity (Nambiar and Kallio 2008; Tiarks and Ranger 2008). A few studies in southern Africa have demonstrated large quantities of nutrients potentially lost through certain harvesting and residue management practices, particularly residue burning (Morris 1986; du Toit and Scholes 2002; Nadel 2005). There are some data describing the effects of post harvesting residue management on subsequent growth of *Eucalyptus* in South Africa (Nadel 2005; Smith and du Toit 2005; du Toit et al. 2010; Rietz 2010). These are among a limited number of studies for which nutrient loss data has been associated with subsequent growth and uptake into the new crop. In addition, few of these studies have been carried out on sites considered sensitive or susceptible to degradation.

The sandy structureless soils of the Zululand coastal ecosystem (in KwaZulu-Natal) may be susceptible to nutritional degradation and productivity decline. These soils have small nutrient reserves and buffering capacity as they are low in clay and organic carbon content and have high drainage rates (Hartemink and Hutting 2005; Fey 2010). The sub-tropical climate and relatively high rainfall in this region results in areas of high productivity potential (Smith and du Toit 2005). This implies a large nutrient removal on soils with low nutrient reserves. A sustained production therefore, may not be possible in this region; particularly if nutrient removal over

each rotation is increased. Fertilisation responses on old agricultural (sugarcane) sites that have been converted to plantations may be evidence of nutritional degradation in such cases (du Toit et al. 2001; du Toit and Oscroft 2003). Growth decline observed across the broad sugarcane growing regions has been attributed to a loss of soil organic matter and nutrients, decreased soil pH and increased soil bulk density (Qongqo and van Antwerpen 2000; Dominy et al. 2001; Dominy et al. 2002). However, there is little conclusive evidence towards a nutrient related productivity decline on plantation sites despite numerous rotations of intensive silvicultural management practices and frequent wildfires.

A study was implemented to assess the changes in above ground biomass growth, nutrient pools and fluxes in a *Eucalyptus* stand during the period from felling to post canopy closure, through the imposition of a range of site management practices. An undisturbed crop is included to compare mature crop nutrient pools and fluxes with the new crop. The objective was to determine the risk of nutrient depletion and productivity decline in subsequent rotations. The hypothesis is that the low organic matter Aeolian sand derived soils in the subtropical environment of the Zululand coastal ecosystem are at risk of nutrient depletion and yield decline of successive rotations.

6.3. MATERIALS AND METHODS

Materials and methods for this chapter that are common to other chapters are given in CHAPTER 3.

6.3.1. Nutrient Audit

A nutrient audit of data in this chapter was calculated as part of the publication of this chapter prior to availability of leaching data presented in CHAPTER 8..

An audit of above ground nutrients fluxes between felling to canopy closure was calculated as nutrients released between felling and planting. Nutrients addition to the soil included release with residue decomposition between planting and canopy closure with ash-bed contents in the Burn treatment and fertiliser addition in the Fert. treatment. Fertiliser was considered to add nutrients to the soil in addition to residue decomposition during the planting to canopy closure

phase. Nutrient release in the Burn treatment was calculated between felling and burning, adding the nutrient content of the remaining ash to the nutrients released through decomposition prior to burning. Standing crop values were calculated to include litterfall accumulated to the end of each period, which was added to the litter layer mass at the start of each period. Nutrient uptake into the aboveground tree biomass between felling and canopy closure was used with release to calculate the difference between above ground uptake and potential nutrient supply. These calculations assume all nutrients released from the residue, added by fertiliser or remaining as ash after burning to have entered the soil. This calculation was done to indicate the potential supply of nutrients under each residue management practice relative to aboveground nutrient demand. It gives an indication of the extent to which trees rely on belowground nutrient pools and atmospheric inputs.

6.4. RESULTS

Temperatures and rainfall were within the normal mean long-term range of the site. Monthly total rainfall was however often below the long-term mean for the first two years of the study, but rose sharply at the end of the study (**Appendix 6-1**). Rainfall and temperature divided the year into two distinct seasons, a wet season and a dry season. The wet summer season was the six month period from October to March and the dry winter period from April to September.

6.4.1. Standing crop biomass and nutrients

Nutrients held in the standing crop were within ranges given by Dovey (2009) for *Eucalyptus* grown in South Africa, except for the bark Ca concentration, which was three fold the reported values. Branch Ca and stem wood N were also at the higher end of the reported value ranges. Growth rates in the standing crop increasing over the study period and leaf area index remained in a narrow range. A 3.5 Mg ha⁻¹ increase in above ground biomass occurred between the time of felling and planting of the new crop, followed by 4.2 Mg ha⁻¹ during the first year of new crop growth and 9.6 Mg ha⁻¹ in the second year. Standing crop litterfall summed over the whole study period (**Table 6-2**) was smaller than decomposition (**Table 6-1**) resulting in a net loss of forest floor biomass and nutrients. Foliage dominated the litterfall for the duration of the study, with branch-fall only greater than foliar litterfall on a few occasions (**Table 6-2, Appendix 6-2**). Nitrogen and Ca, which had similar content in the litterfall, was most dominant, followed by Mg, K and P which were an order or magnitude lower than N and Ca. Retranslocation and canopy

exchange (during rainfall) account for the large differences between fresh equivalent and litterfall foliage contents. High Ca in the litterfall was through mass loss increases in Ca concentrations prior to litterfall, Ca not being retranslocated (Fife et al. 2008).

Table 6-1: Biomass and nutrients contained in the litter layer and residues for treatments at times of felling, burning, planting and canopy closure. Residue and litterfall are calculated from felling to pre-burning, and from felling to planting and canopy closure, respectively.

Point in time	Treatments and effects	Mass	N	P	K	Ca	Mg
		Mg ha ⁻¹	kg ha ⁻¹				
Felling (October 2008)	Residue	26.0	101.9	18.8	107.2	439.9	67.7
	Standing crop	24.6	201.6	8.4	14.2	202.4	41.9
	No-Burn, Burn, Fert ¹	50.6	303.5	27.3	121.4	642.4	109.6
	Whole-Tree	22.6	211.1	9.4	17.4	292.1	54.0
	Double	76.5	405.4	46.1	228.7	1082.3	177.4
Burning (March 2009)	Pre burn	36.4	232.3	11.0	23.0	448.6	60.9
	Post burn ring Ash	4.2	111.1	8.2	19.7	413.9	39.3
	Post burn plate Ash	4.3	7.7	6.6	27.1	640.8	42.9
Planting (August 2009)	Standing crop litterfall ²	6.9	57.2	2.1	7.3	55.3	13.3
	Standing crop	20.0 ^a	177.3 ^a	7.3 ^a	14.5 ^a	174.7 ^a	35.5 ^a
	No-Burn, Fertilised	28.6 ^b	268.6 ^b	11.3 ^b	18.2 ^a	445.3 ^b	52.4 ^b
	Whole-Tree	18.8 ^a	154.9 ^a	6.9 ^a	15.1 ^a	376.0 ^{ab}	35.8 ^a
	Double	40.5 ^c	267.7 ^b	14.8 ^c	31.7 ^b	978 ^c	80.5 ^c
Canopy closure (August 2010)	Standing crop litterfall ³	8.2	59.5	2.5	15.0	63.7	16.2
	Standing crop	15.5 ^a	169.9 ^{ab}	8.9 ^a	13.6 ^a	182.7 ^a	31.5 ^a
	No-Burn, Fertilised	21.9 ^b	176.0 ^a	7.7 ^a	11.3 ^{ab}	510.7 ^b	35.0 ^a
	Whole-Tree	13.5 ^a	145.5 ^b	7.8 ^b	9.1 ^b	306.2 ^{ab}	26.9 ^a
	Double	28.2 ^c	221.9 ^a	9.7 ^a	14.2 ^{ac}	643.8 ^c	44.1 ^b

¹ Estimate for the three treatments from summing residue and litter layer.

^{2,3} Standing crop litterfall for the period (1) October 2008 to August 2009 and (2) August 2009 to August 2010. Letters (a,b,c) that are different indicate significant treatment differences at each stage (LSD5%, $p < 0.001$).

Table 6-2: Annualised mass of foliar and canopy macro-nutrients deposited as litterfall in the standing crop during the period between felling and canopy closure of the new crop. Fresh foliage mass is the litterfall equivalent mass of nutrients using foliar concentrations and indicates potential retranslocation.

	Mass Mg ha ⁻¹ year ⁻¹	N	P	K	Ca	Mg
		kg ha ⁻¹ year ⁻¹				
Leaves	4.1	48.3	1.7	6.8	42.3	9.0
Bark	0.9	5.5	0.2	1.4	6.2	3.1
Branch	3.8	13.7	0.7	4.7	20.3	4.9
Total	8.8	67.5	2.7	12.9	68.8	17.1
Fresh foliage	4.1	58.4	4.4	29.2	37.3	11.0

6.4.2. Litterfall results (not included in publication)

Although small branch-fall made up the bulk of branch-fall, a less frequent and more spatially spread large branch-fall resulted in a higher variability between the litter-traps (**Appendix 6.2**). Some extreme weather conditions (wind, hail and heavy rainfall) also resulted in increased litterfall variability. Bark litterfall generally occurred at the beginning of the wet seasons, and as it was limited to the regions around each tree, was least effectively caught in the litter traps. Nutrient concentrations in the litterfall changed with rainfall and the nature of the litterfall (foliage, bark or branch) (**Appendix 6.2A-F**). Total litterfall concentrations of N, P Ca and Mg remained in a relatively narrow range, though were generally higher during the wet season and lower during the dry (**Appendix 6-4** and **Appendix 6-5**). Potassium had the largest fluctuations in concentration following both seasonal patterns and rainfall patterns. The daily quantities of macro-nutrients falling with litterfall were influenced by litterfall mass and nutrient concentration, which were dependant on the ratios of tree components making up the litterfall.

6.4.3. Post felling residue nutrient fluxes

Harvesting and residue retention (stem wood removal) left large quantities of macro-nutrients across the site; particularly Ca and N (**Table 6-1**). Residue biomass was almost double forest floor biomass. Uneven mass transfer and distribution resulted in the Whole-Tree and Double treatments having slightly lower residue loads than expected. Decomposition was initially rapid in residue retained treatments; faster during wet than during dry periods (**Figure 6.1**). Residue decay half life in the No-Burn treatment was calculated as 16 months with an annual k factor of 0.42 ($R^2=0.965$). Half life was longer in the Double treatment (12 months) with an annual k value of 0.55 ($R^2=0.948$). Decomposition of the old litter layer in the Whole-Tree treatment was slow after felling, having an 18.5 month decomposition half life. The delay between felling and planting (39 weeks) resulted in a significant loss of residue biomass (**Table 6-1**) and nutrients. Prior to burning at 19 weeks after the start of felling, 28% of the residue biomass had decomposed (**Table 6-1**) and further losses of N, P, and Mg had occurred from the residues. The following 48 weeks between planting and canopy closure resulted in larger N and Ca residue decomposition losses than prior to planting.

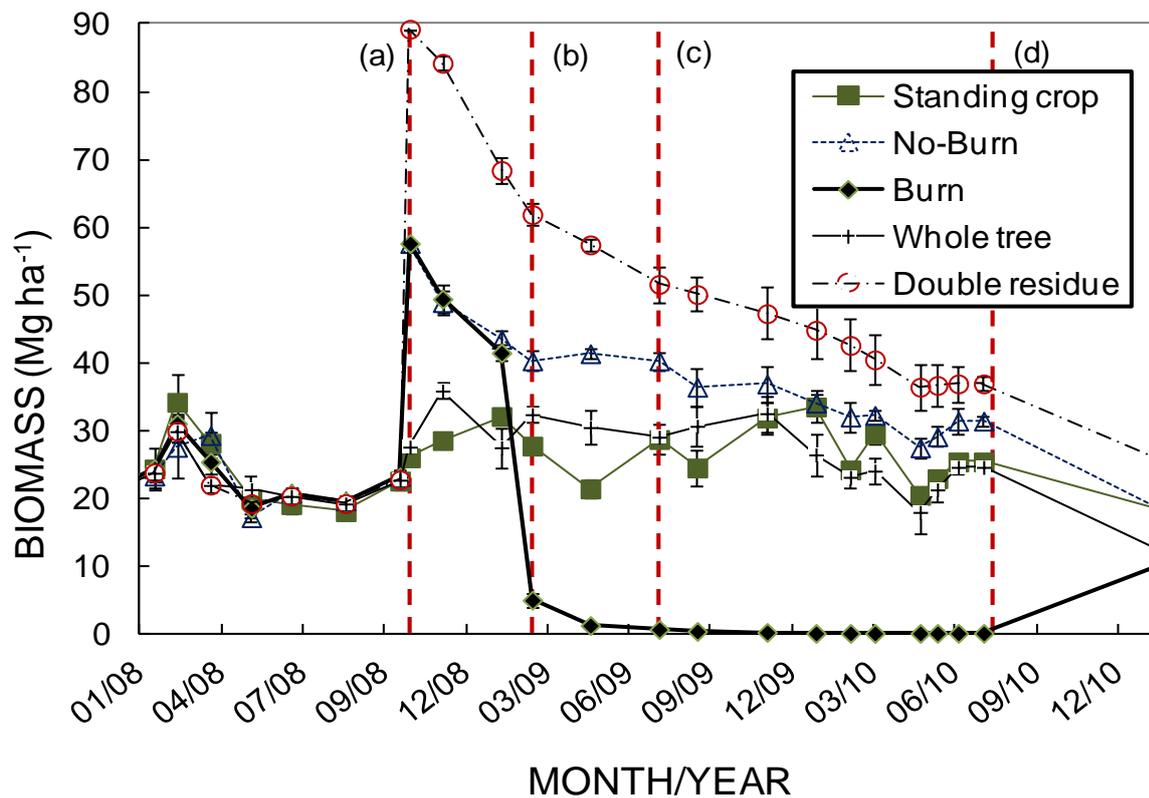


Figure 6.1: Forest floor litter layer and residue biomass from prior to felling, through felling and up to canopy closure for each treatment. Error-bars represent a single standard deviation between sampling points. Dashed lines represent (a) felling; (b) residue burning; (c) planting and (d) canopy closure.

Burning resulted in a large loss of nutrients from the residues through oxidation, in smoke and wind erosion of fine ash. The smallest loss being 8% of the Ca and the highest being 52% of the N compared with pre-burn pools (**Table 6-1**). Larger biomass and N loss occurred through residue burning than through whole tree removal. More P, K, Ca and Mg were lost through whole tree removal than through burning. The two ash assessment methods gave similar biomass losses when calculating LOI adjusted ring mass and unadjusted plate masses, but produced different nutrient losses (**Table 6-1**). A hotter burn was visually evident on the plates as less black carbon and more white ash on the plates than on bare soil. Ring ash samples were therefore used in assessing burning losses.

6.4.4. Residue and litter results (not included in publication)

Although the reduction in residue biomass of different residue treatments was less than 10% in all treatments during the first four weeks after felling, concentrations of nutrients were significantly reduced. Nitrogen, P, K, Ca, and Mg concentrations were reduced to 88%, 46%, 20%, 72% and 75% and 55% of their original values, respectively. In the period after the first four weeks, concentrations remained relatively range bound across all treatments, following

different release patterns with decomposition (**Appendix 6-3 B-F**). N concentration, although remaining stable from felling onwards, was highest in the standing crop fluctuating slightly in all other treatments over time (**Appendix 6-3 A**). K and Mg concentrations decreased while Ca concentrations increased in all treatments between felling and canopy closure, except the standing crop treatment in which the concentrations of these elements remained within a narrow stable range. P remained range bound in all treatments with small fluctuations in concentration.

6.4.5. New crop growth

During the first part of the wet season (October to March) basal area growth rates followed the treatment order Burn > Fert. > No-Burn > Whole-Tree > Double (**Figure 6.2**), with statistical significances overlapping ($p < 0.001$). At 1.0 years of age the Fert. treatment had the highest basal area followed by Burn and the remaining treatments ($p < 0.001$). Burn and Fert. treatment differences were not significant at this point. At 1.5 years, and into the next wet season, trees in the Fert. treatment became significantly larger than the other treatments ($p < 0.001$). Differences between all other treatments were not significant. This pattern of significant difference continued to 2.5 years after planting ($p < 0.001$). Burn treatment growth rates tended to decrease over the 2.5 year growth period relative to No-Burn and all other treatments (**Appendix 6-7**). Height growth differences were not significant prior to two years after planting (**Figure 6.3**). Height became significantly larger in the Fert. treatment at two ($p = 0.049$) and two years six months years after planting ($p = 0.030$).

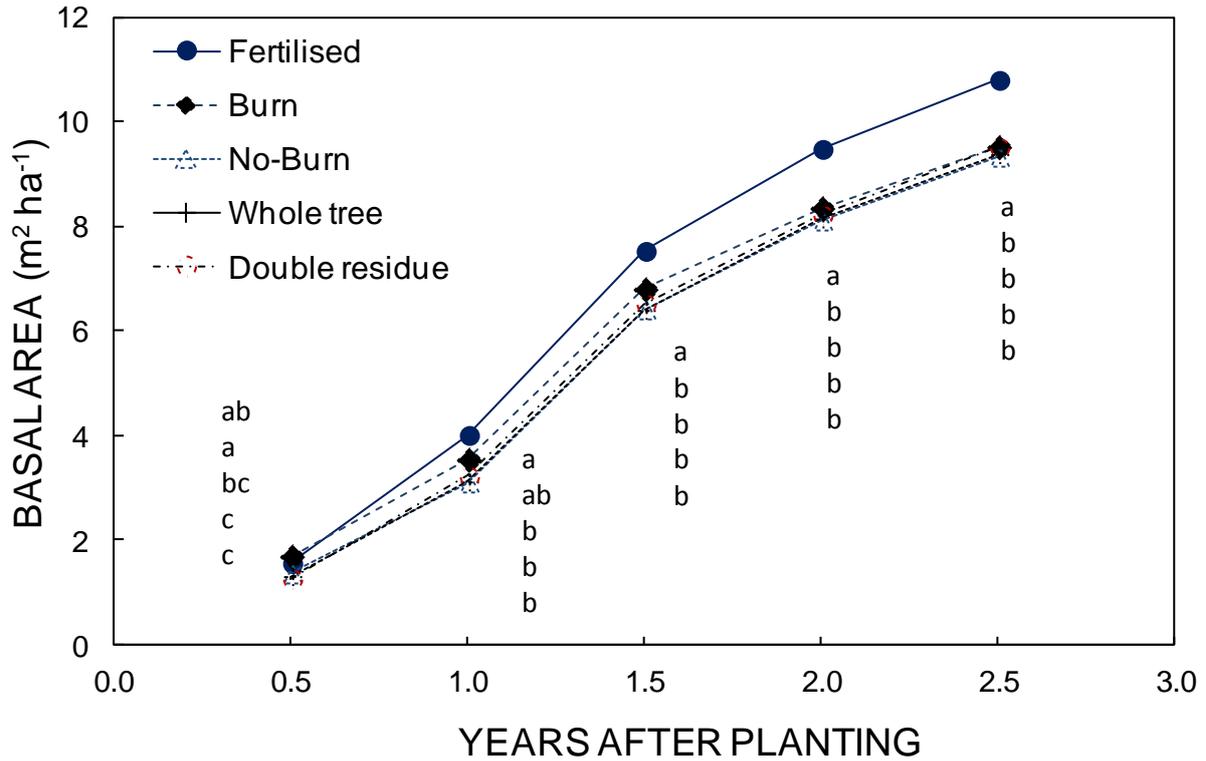


Figure 6.2: Basal area of each treatment at six monthly intervals after planting. 0.5 year basal area data calculated using *gld*, others calculated using *dbh*. Different a, b, c superscripts denote significant differences between treatments at each age in the treatment order of the legend (LSD_{5%}).

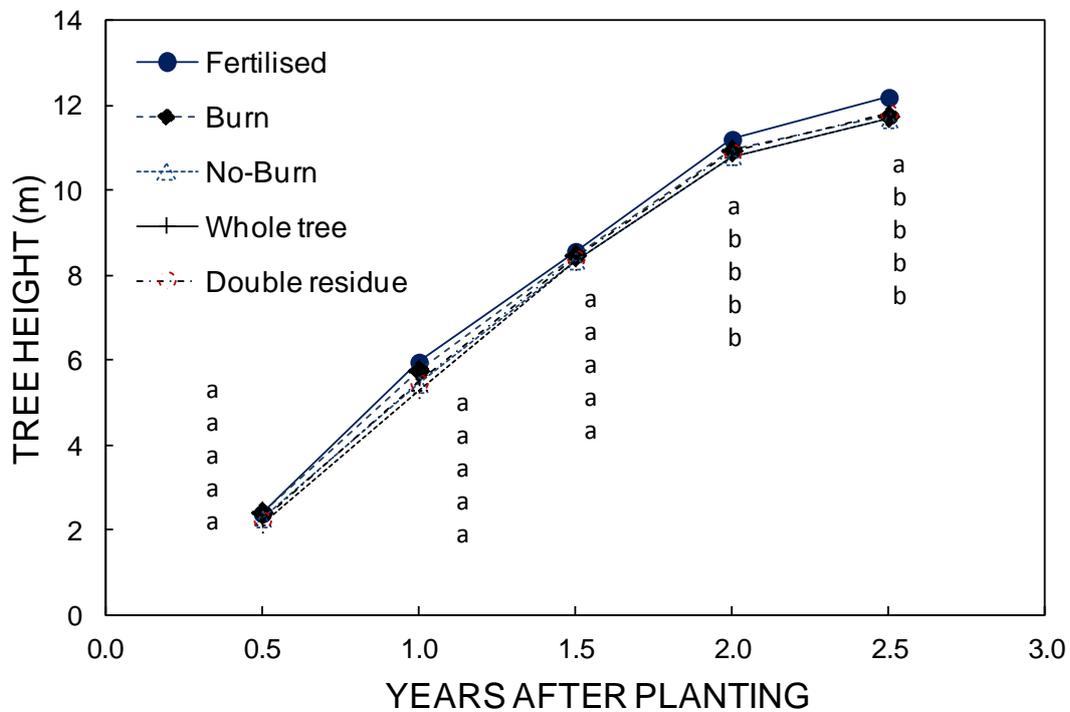


Figure 6.3: Mean heights of trees in each treatment at six monthly intervals after planting. Different a, b superscripts denote significant differences between treatments at each age in the treatment order of the legend (LSD_{5%}).

6.4.6. Above ground biomass and nutrient accumulation

Woody biomass, LAI and foliar biomass were all significantly larger ($p < 0.01$) in the Fert. and Burn treatments at six months than in all other treatments **Figure 6.4**. The Fert. and Burn treatment were not significantly different from each other. The remaining treatments were not significantly different from each other either. At one year after planting (canopy closure) woody biomass, foliar biomass and LAI followed the same order of significance. These measures remained larger in the Fert. and Burn, and larger than all other treatments ($p < 0.05$).

Above ground biomass and LAI in the Fert. treatment was 44.7 Mg ha^{-1} and $3.2 \text{ m}^2 \text{ m}^{-2}$ at two years after planting, significantly more than all other treatments, including the Burn treatment ($p < 0.001$). No significant differences occurred between all other treatments in two year biomass and LAI between all other treatments which had a mean aboveground biomass of 37.3 Mg ha^{-1} and LAI of $2.7 \text{ m}^2 \text{ m}^{-2}$. Leaf area growth efficiency at six months was $3.6 \text{ Mg ha}^{-1} \text{ yr}^{-1} \text{ LAI}^{-1}$ in the Burn and Fert. treatments which were significantly different from an average $3.5 \text{ Mg ha}^{-1} \text{ yr}^{-1} \text{ LAI}^{-1}$ for all other treatments ($p < 0.01$). Although growth efficiency increased to 9.3, 9.0, 8.6, 8.7 and $8.1 \text{ Mg ha}^{-1} \text{ yr}^{-1} \text{ LAI}^{-1}$ in the Fert, Double, No-Burn, Whole-Tree and Burn respectively, differences were not significant.

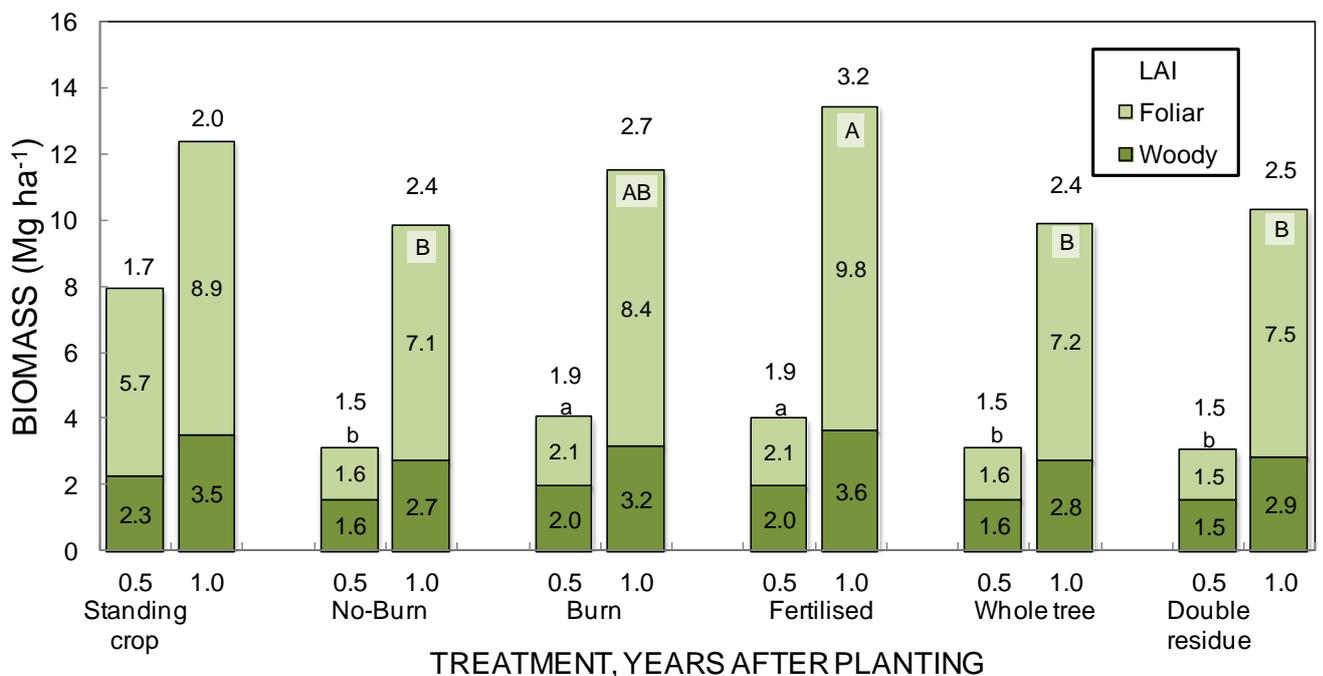


Figure 6.4: Biomass accumulated in the above ground tree components between the planting to 0.5 years, and planting to canopy closure (1.0 years). Values shown on bars with LAI ($\text{m}^2 \text{ m}^{-2}$) on the top. (standing crop is the sum of growth and litterfall) Different a, b, c superscripts denote

significant differences between treatments; lowercase for 0.5 years and uppercase for 1.0 years (LSD_{5%}).

Total above ground nutrient accumulation followed the same treatment rankings as differences in biomass (**Table 6-3**). However, significance at six months of age was found only for N. The Fert. treatment had the highest N contents followed by Burn, No-Burn, Double, and Whole-Tree with significant differences overlapping (**Table 6-3**). At one year after planting, the Fert. treatment had accumulated significantly more N than all other treatments ($p < 0.01$). Nitrogen in the woody biomass was the highest in the Fert. treatment, significance overlapping across all other treatments. There were no further significant treatment differences in woody biomass contents. The Fert. treatment accumulated significantly more P ($p < 0.01$), K ($p < 0.01$) and Mg ($p < 0.05$) into the foliage than all other treatments (**Table 6-3**). Foliar accumulation of Ca was highest in the Burn. Foliar accumulation of these elements was less in the Double and No-Burn, and least in the Whole-Tree treatment with overlapping orders of significant differences. In addition to Ca, the Burn treatment accumulated significantly more P and K than the No-Burn, Whole-Tree and Double treatments. There were no significant differences between W, No-Burn and Double treatments.

Table 6-3: Macro-nutrients accumulated in the above ground tree biomass at 0.5 and 1.0 years after planting (canopy closure). Superscript and subscript values are foliar and woody component masses respectively. Letters that are different indicate a significant difference between treatment mean above ground mass at each age (LSD_{5%}).

Treatment	0.5 years after planting										1 year after planting									
	N		P		K		Ca		Mg		N		P		K		Ca		Mg	
Standing crop*	46.6	25.6	2.2	0.8	10.7	3.1	56.3	23.2	13.5	4.9	67.7	39.1	3.9	1.5	22.2	7.9	82.6	34.3	19.4	7.7
		21.1		1.4		7.6		33.1		8.6		28.6		2.4		14.4		48.3		11.7
No-Burn	32.7 ^{bc}	27.9	2.2 ^a	1.7	31.8 ^a	23.0	19.1 ^a	14.5	6.4 ^a	4.8	64.9 ^b	46.6	4.3 ^{bc}	2.8	57.6 ^c	34.9	42.5 ^c	20.1	14.8 ^b	6.5
		4.8		0.5		8.8		4.7		1.6		18.3		1.5		22.7		22.4		8.3
Burn	39.6 ^{ab}	33.1	2.8 ^a	2.2	41.8 ^a	30.7	24.0 ^a	17.5	8.2 ^a	6.0	75.1 ^b	53.7	5.0 ^b	3.2	66.0 ^{ab}	41.7	50.9 ^a	26.0	16.8 ^{ab}	7.5
		6.5		0.6		11.1		6.5		2.2		21.4		1.8		24.3		24.9		9.2
Fertilised	47.8 ^a	40.4	3.1 ^a	2.5	38.7 ^a	27.8	21.4 ^a	16.1	7.6 ^a	5.6	89.3 ^a	65.6	5.4 ^a	3.7	65.7 ^a	44.5	46.0 ^{ab}	24.9	18.7 ^a	8.8
		7.4		0.7		10.9		5.3		1.9		23.8		1.7		21.2		21.1		9.9
Whole tree	31.2 ^c	26	2.2 ^a	1.6	32.2 ^a	22.8	18.3 ^a	13.0	6.4 ^a	4.8	64.8 ^b	47.4	3.9 ^c	2.5	54.0 ^c	30.7	39.7 ^c	19.4	15.0 ^b	6.9
		5.2		0.5		9.4		5.3		1.6		17.4		1.4		23.3		20.3		8.1
Double	31.7 ^{bc}	26.7	2.2 ^a	1.7	30.5 ^a	22.0	17.5 ^a	12.9	6.0 ^a	4.5	68.0 ^b	49.3	4.3 ^{bc}	2.8	56.6 ^{bc}	35.1	41.7 ^{bc}	21.0	15.1 ^b	6.9
		5.0		0.5		8.5		4.6		1.5		18.6		1.5		21.5		20.8		8.2

*standing crop is the sum of growth and litterfall occurring during each period after planting the new crop.

6.4.7. Foliar nutrient diagnostics

No foliar nutrient deficiencies were found at any stage from planting to canopy closure using standards for eucalypts (Boardman et al. 1997) and a ratio method (Linder 1995). However, vector analysis showed fertilisation to alleviate both N and P deficiencies at six months (**Appendix 6-8**). All other macro-nutrients were diluted in the Fertilised treatment. The burn treatment was not nutrient limited at six months. Weak depletion responses occurred in the Whole tree treatment for N, P, K, and Ca, vector lengths less than 10 at six months for all macro-nutrients. Calcium and Magnesium deficiencies were strongly alleviated by fertilisation in the Fert. treatment at one year after planting. All other elements were between sufficiency and dilution ranges in the remaining treatments. Deficiencies of N and Ca were alleviated in the Burn treatment at one year after planting; all other macro-nutrients were diluted. Depletion of P and K was also found through vector analysis in the Whole tree treatment for the same period. Two-year data showed a N deficiency alleviation in the fertilised treatment and no nutrient limitations in any other treatment.

6.4.8. Nutrient Audit

Large quantities of nutrients were potentially added to the soil during the 39 week felling to planting fallow period (**Table 6-4 i**). Residue nutrient release was lower during the planting to canopy closure period thereafter (48 weeks) for all elements and treatments except N in the No-Burn treatment (**Table 6-4 ii**). Fertilization resulted in a gain of all macro-nutrients to the soil. Nutrient supply from decomposition in the Whole-Tree treatment was less than above ground uptake of P, K, and Ca throughout the entire felling to canopy closure period. Negative Ca values (gain) in the Whole-Tree treatment were through an increase in residue Ca concentrations. However, a net loss of Ca occurred between planting and canopy closure. Residue decomposition nutrient release was larger than above ground nutrient accumulation in all other felled treatments up to a year after planting.

Atmospheric inputs played a major role in nutrient supply to the study site. Combined throughfall and stemflow in the standing crop contributed 16.3 kg ha⁻¹ of N (6.8 kg ha⁻¹ mineral and 9.6 kg ha⁻¹ organic) and 17.2, 15.1 and 6.2 kg ha⁻¹ of K, Ca and Mg respectively between planting and one year after planting (CHAPTER 5). Rainfall contributed 10.6 kg ha⁻¹ of N (6.8

kg ha⁻¹ mineral and 3.8 kg ha⁻¹ organic) 5.5, 7.7, and 1.4 kg ha⁻¹ of K, Ca and Mg. Phosphorus addition was not recorded (below detection). Changes in the supply rates of readily available macronutrient species from the soil pool as a consequence of the different treatments could not be included as these were difficult to measure. However, sequential coring estimates of N mineralisation for the period from felling to one year after planting yielded 46.7 and 24.5 of N mineralised in the No-burn and Burn respectively (unpublished data). Immobilization occurred in the standing crop treatment (-5.7 kg ha⁻¹ yr⁻¹).

Nutrient release during the planting to canopy closure period was far less than aboveground nutrient accumulation for most treatments (**Table 6-4** ii and iii). This implies that the trees were reliant on nutrients released into the soil prior to planting, mineralised or exchanged from the soil pools, and added through atmospheric inputs. Fertilisation in addition to residue decomposition supplied an excess of nutrients relative to aboveground biomass accumulation during the year after planting. Decomposition supplied only a small proportion of aboveground nutrient requirements in the Whole-Tree treatment (**Table 6-4** iv). More P, Ca and Mg were released from the residues in the Double treatment than were accumulated in aboveground growth, while less N and K was released than accumulated in aboveground growth. A small shortfall of P, K and Ca was apparent in the standing crop (**Table 6-4** iv), but this was small relative to the atmospheric inputs given above.

Table 6-4: An audit of aboveground nutrient loss, release (as release of nutrients with decomposition and fertiliser addition) and above ground uptake for each treatment between felling and canopy closure (kg ha^{-1}). It Includes a calculation of the difference between release and uptake.

	Potential nutrient release to soil									
	Felling to planting (i)					Planting to canopy closure (ii)				
	N	P	K	Ca	Mg	N	P	K	Ca	Mg
Standing crop	82	3	7	83	20	67	1	16	56	20
No-Burn	26	14	94	160	52	93	4	7	(65)	17
Burn	174	23	109	571	82	0	0	0	0	0
Fertilised	26	14	94	160	52	212	26	92	3	33
Whole-Tree	56	2	2	(84)	18	9	(1)	6	70	9
Double	120	28	179	31	85	46	5	17	334	36

	Aboveground crop uptake (iii)					Release and uptake difference (iv)				
	Felling to canopy closure									
	N	P	K	Ca	Mg	N	P	K	Ca	Mg
Standing crop	127	6	32	144	34	21	-2	-9	-5	6
No-Burn	65	4	58	43	15	54	14	43	52	54
Burn	75	5	66	51	17	98	18	43	520	66
Fertilised	89	5	66	46	19	149	35	120	117	66
Whole-Tree	65	4	54	40	15	1	-2	-46	-54	12
Double	68	4	57	42	15	98	29	139	323	107

Values in parenthesis indicate an increase in residue nutrient content; all remaining nutrients are assumed available after burning.

6.5. DISCUSSION

6.5.1. Standing crop nutrient fluxes

Standing crop data demonstrates a relative balance between nutrient additions and losses with above ground nutrient cycling, (retranslocation, litterfall and forest floor decomposition) responsible for a significant proportion of growth related nutrient demands. Atmospheric deposition supplied excess nutrients to the standing crop over the study period, while soil leaching losses beyond the root zone were unlikely due to high water use (CHAPTER 4). These data provide a good baseline against which to compare nutrient fluxes in disturbed and undisturbed plantation systems. It demonstrates that growth prior to site disturbance was not limited by the nutrients assessed in our study. However, standing crop growth was possibly limited by the pest infestation. This was evident in the high canopy turnover relative to biomass accumulation and a growth increase as infestation levels visually declined.

6.5.2. Treatment effects on nutrient capital and early growth

Large quantities of nutrients lost through harvesting, residue removal and residue burning reduced the quantities of nutrients available to the next crop and site nutrient pools. However, residue management had very little effect on early growth and biomass development at this site up to 2.5 years after planting. Despite significant nutrient losses from residue and forest floor, the initial growth and nutrient accumulation response in the Burn treatment may have been through increased mineral nutrient availability after burning. Calcium remained in quantities that exceeded early growth demand. Early effects diminished over time, the initially higher growth after burning slowing sufficiently for above ground biomass to match the other residue management treatments. Whole-Tree removal did not reduce growth to 2.5 years, despite large nutrient losses. A combination of atmospheric inputs, soil nutrient supply that included decomposition of the old root system may have provided early growth nutrient requirements to canopy closure. Loss of nutrients from the whole tree harvest treatment may limit growth at a later stage. Although the lack of early growth response to residue doubling was unexpected, nutrient lockup in the residues may have reduced soil nutrient availability. The long delay between felling and burning may also have permitted nutrient return to the soil through decomposition and leaching prior to burning (**Figure 5 i**). However, management usually limits the duration of the fallow period after clearfelling. This will allow smaller quantities of readily available nutrients to be present at establishment. It also underscores the reason for significant responses obtained with small fertilizer applications in close proximity to young trees during the period immediately after planting.

The initial fertiliser response may have been through increased nutrient availability in close proximity to the roots of the young trees. Fertilisation at planting exceeded the N, P and K recommended rates of up to 70, 17, 17 kg ha⁻¹, respectively du Toit and McLennan (2000). Past fertiliser experiments used to derive these rates in the same region have responded to nutrient addition at planting, even after residue burning. Depending on site type, N and/or P and/or S application at planting improved volume growth by 40-50 m³ ha⁻¹ over a 6 year rotation across five site types (du Toit and Oscroft 2003). Although fertilisation significantly increased accumulation of foliar and woody N and P at our study at one year, there was no evidence that these elements were responsible for the growth response. The vector analysis pointed to Ca and Mg having improved growth past canopy closure. This may indicate a shift in nutrient requirements, N and P being adequate in our study after the establishment phase. Although

growth response was through an increase in tree canopy size (*LAI*), growth efficiency may have played a small role during early growth. Growth efficiencies were similar to values in clonal *Eucalyptus* at two years in Congo Laclau et al. (2009) and at three years in Brazil (Stape et al. 2008).

6.5.3. Comparison with similar international studies

A network of international studies co-ordinated through the Centre for International Forestry Research (CIFOR) used similar treatments to investigate the effects of residue management after clearfelling on subsequent productivity across tropical plantation forests (Tiarks and Ranger 2008). Growth was reduced in *Eucalyptus* plantations after residue removal on sites characterised by relatively low soil organic carbon and low total soil nutrient levels in Australia (Saint-André et al. 2008), Brazil (Gonçalves et al. 2008b), China (Xu et al. 2008) and the Republic of Congo (Deleporte et al. 2008). More fertile sites either did not experience a productivity decline (Australia (Saint-André et al. 2008)) or experienced decreased growth only after complete residue and litter layer removal (South Africa (du Toit et al. 2008)). The low organic carbon and low nutrient sites in Brazil, Congo, China and Australia were similar to our site, but demonstrated growth improvement with residue doubling and retention and growth reduction after residue removal. Our study site experienced a similar zero response to the more fertile CIFOR network sites. Soil carbon and nutrient levels reported at these fertile sites were an order of magnitude higher than at our study site.

Increased nutrient availability can also increase above ground growth through a reduction in below ground carbon allocation and may explain past and present responses. The proportion of NPP allocated to below-ground fine root production and respiration is decreased at higher nutrient availabilities (Cannell et al. 1988; Sands and Mulligan 1990; Dickmann et al. 1996; Misra et al. 1998; Giardina et al. 2004; du Toit 2008). Coyle and Coleman (2005) disagree with this approach and argue that belowground changes are rather through an acceleration of physiological growth maturity.

6.5.4. Treatments used and management

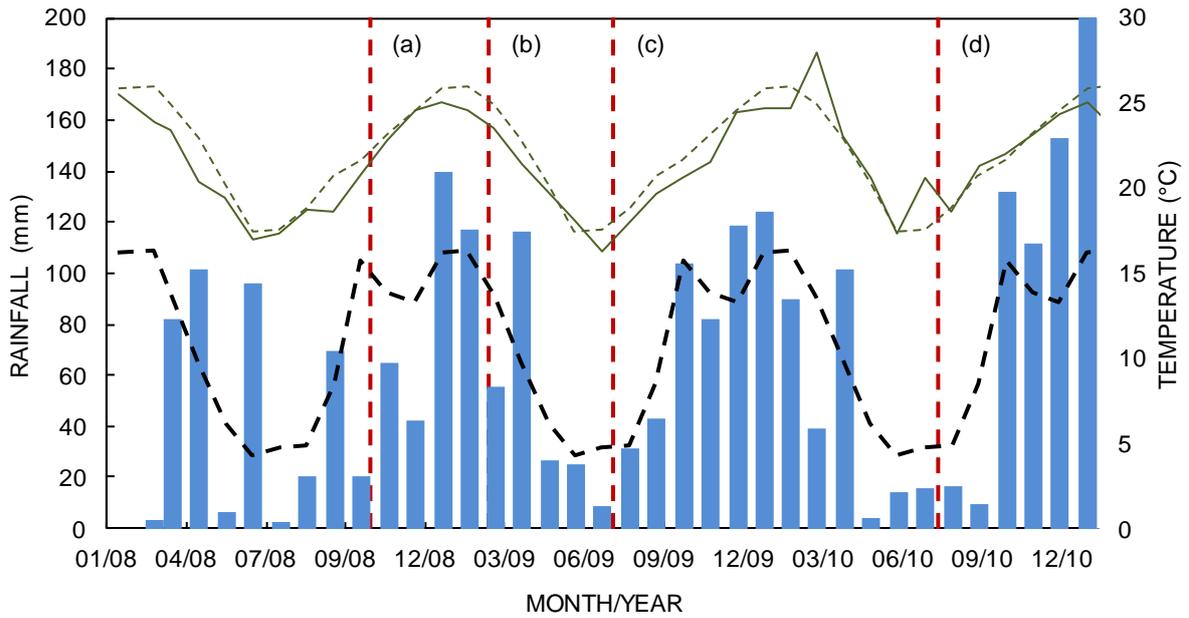
Although most of the residue management practices described in this study are representative of management practices commonly used the SA forestry industry, there is nowadays a strong trend

towards practices involving greater biomass removal. These less conservative practices may be used to reduce fire risk, to facilitate mechanised harvesting or to allow the utilisation of tree residues for energy production. Mechanised whole tree harvesting may become a more common practice in South Africa. Where whole-tree residue harvesting or burning are required, it may be beneficial to leave residues on site for as long as possible to allow nutrients (particularly K and P) to move into the soil. This is provided the more mobile K ions can be stored on the soil exchange and not leached beyond the rooting zone.

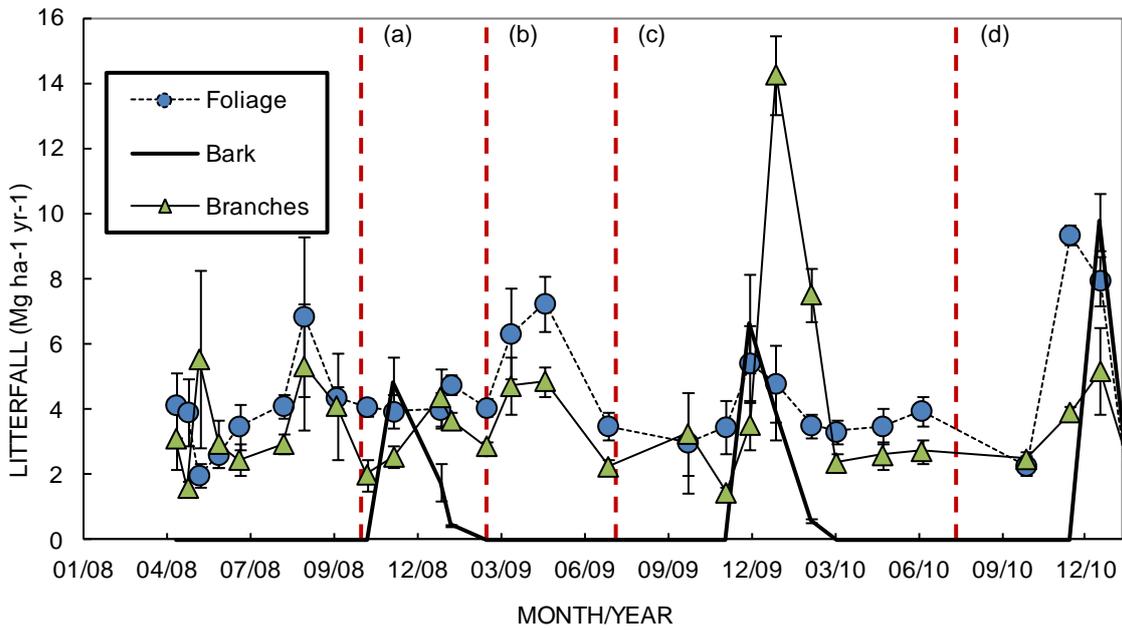
6.6. CONCLUSION

This site occurs on a dystrophic sandy soil with low nutrient retention and storage capacity and small belowground nutrient reserves. Despite this, there was little evidence of an early growth reduction after large nutrient removal. This agrees with the lack of evidence towards a nutrient related decline across the Zululand region. The results of this study may be deceptive if viewed in terms of tree responses growth alone. It therefore accounts for the consequences of management practices on nutrient loss and demonstrates the effect of plantation management on nutrient reserves and future nutrient supply. Nutrient pools in the residues and litter layers at felling play a major role in maintaining nutrient reserves on this site. Further to this, nutrient recycling processes in the mature crop limits nutrient demand while the large atmospheric inputs partially replenish nutrients lost at clearfelling. Increased nutrient loss (whole tree removal and burning) removes above ground pools and places a more heavy reliance on belowground (soil and soil organic) nutrient reserves. Given the small soil nutrient pools at our site, it is likely that soil nutrients will become depleted under management practices that induce larger losses than natural and anthropogenic replacements. Retention of residues may therefore be necessary to conserve nutrients on such sites in Zululand and fertilisation may be required to replenish nutrients lost in quantities that exceed natural replacements. Although this study needs to continue to rotation end, fertilisation responses and smaller atmospheric inputs may point to a depletion of Mg already in effect. However, this needs further testing through fertilisation studies. Further work using more intensive nutrient removal must be undertaken to determine the threshold of nutrient removal at which soil nutrient depletion occurs across sites in Zululand. This must also be extended across broad site types and species ranges to extrapolate effects spatially and into the long term. Strategies to reduce nutrient losses and ameliorate sites where nutrient depletion has already occurred may then be required.

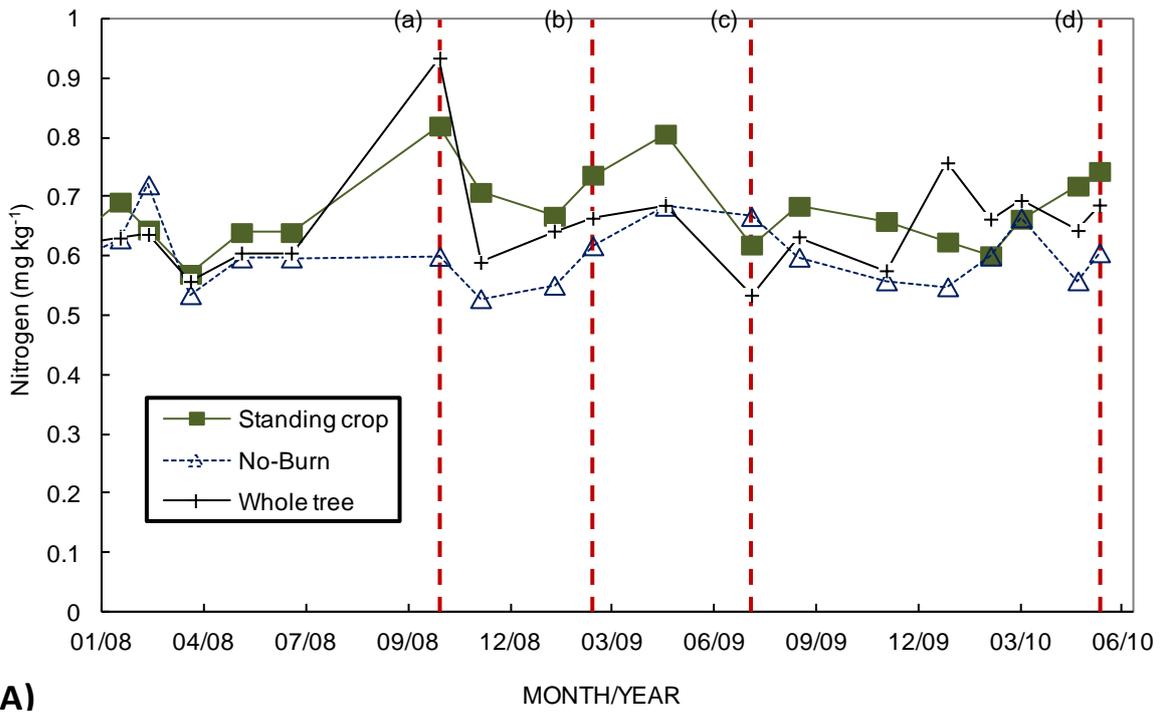
6.7. APPENDICES



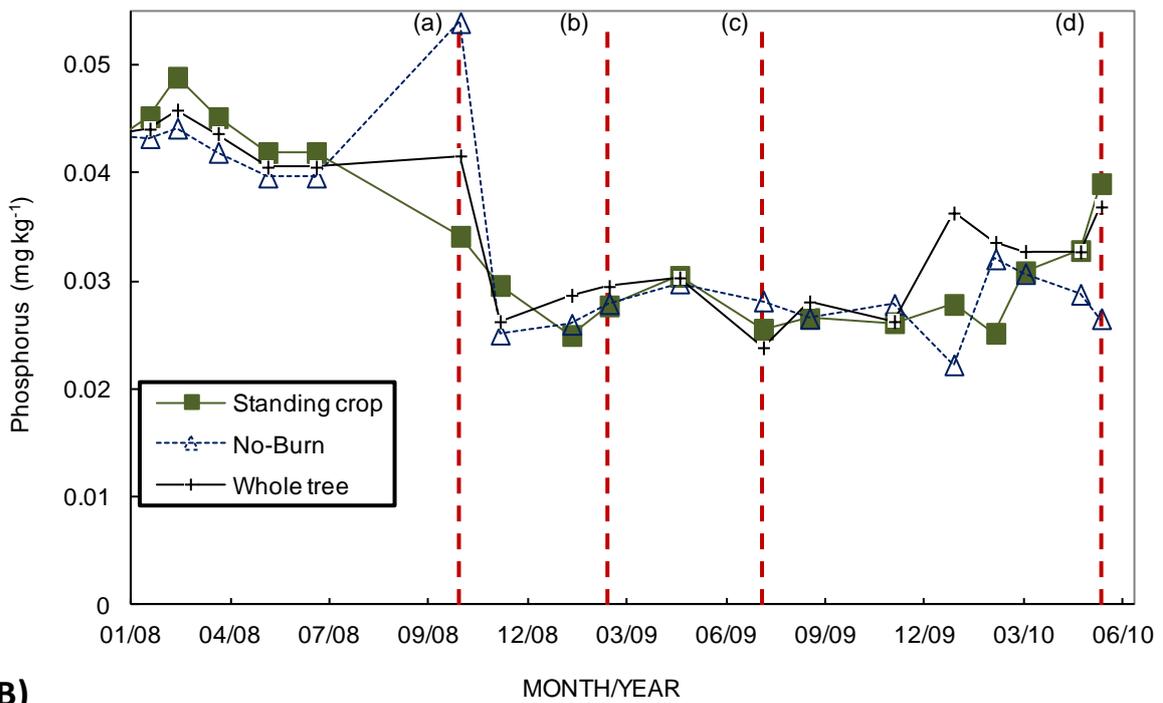
Appendix 6-1: Total monthly rainfall (solid bars) and long term means (dotted line on bar); mean monthly temperature (line) and long term means (fine dotted line) recorded from felling to post canopy closure. Vertical dashed lines represent the times of (a) felling; (b) residue burning; (c) planting and (d) canopy closure.



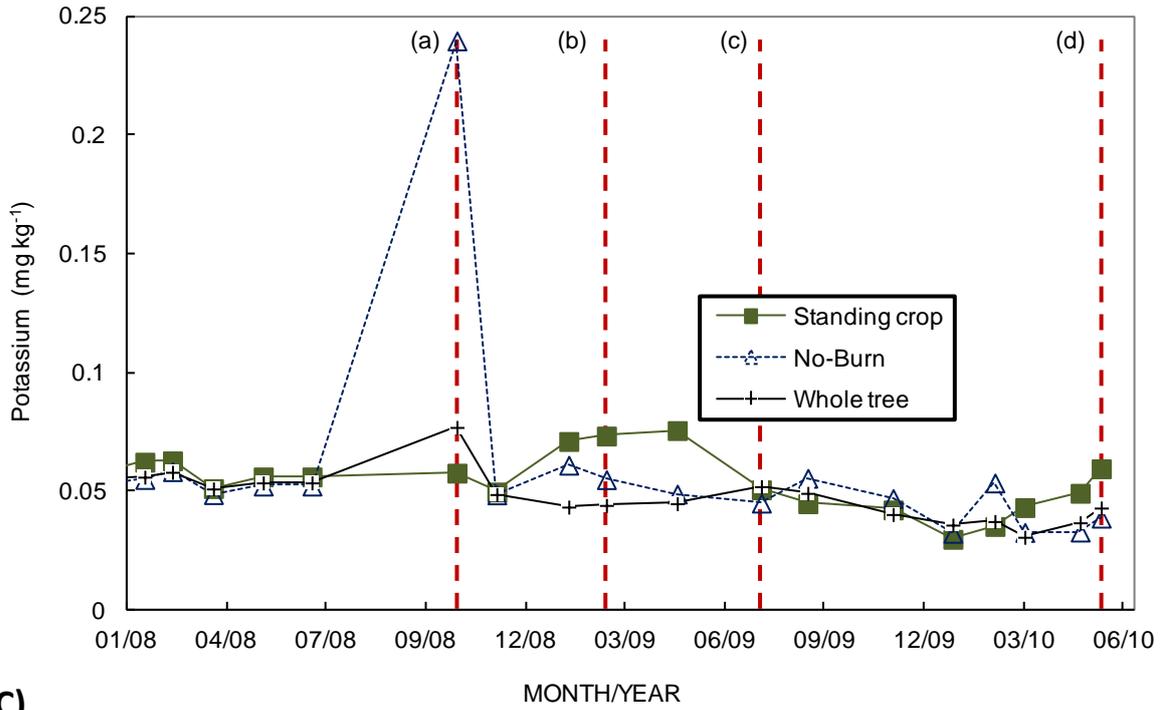
Appendix 6-2: Annual foliar, branch and bark litterfall rates per litter trap collection, averaged across the standing crop plots. Error-bars represent the single standard deviation between all litter traps. Vertical dashed lines represent the times of (a) felling; (b) residue burning; (c) planting and (d) canopy closure.



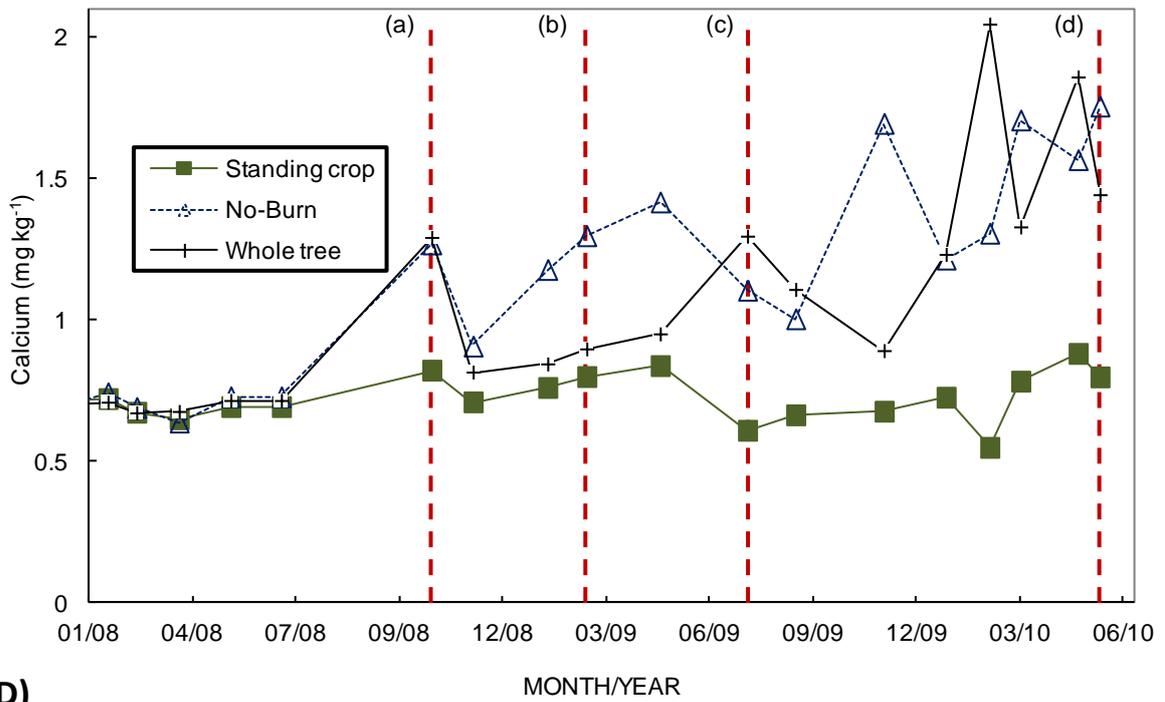
(A)



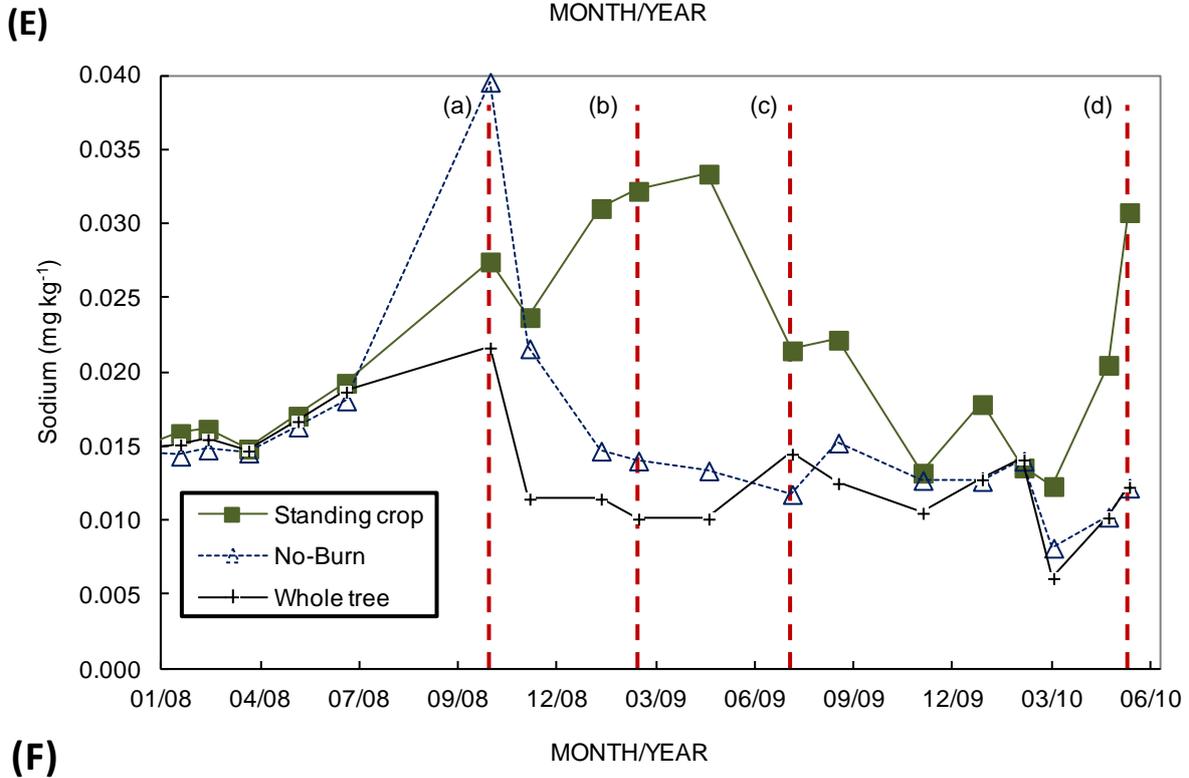
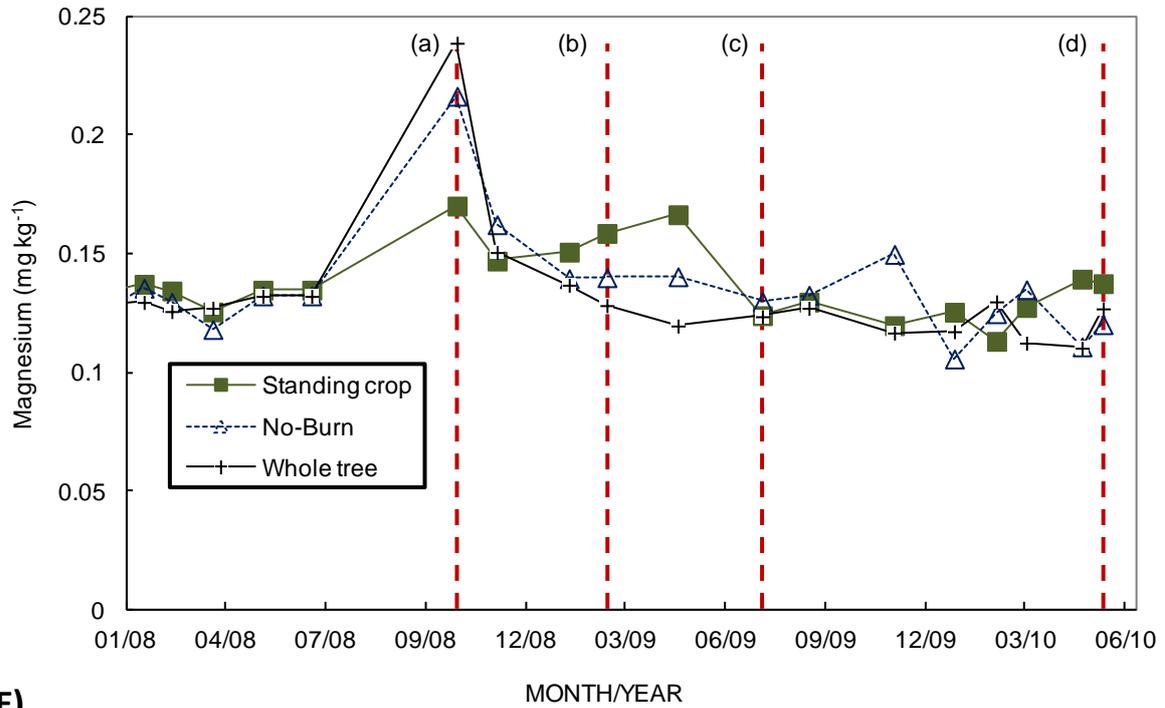
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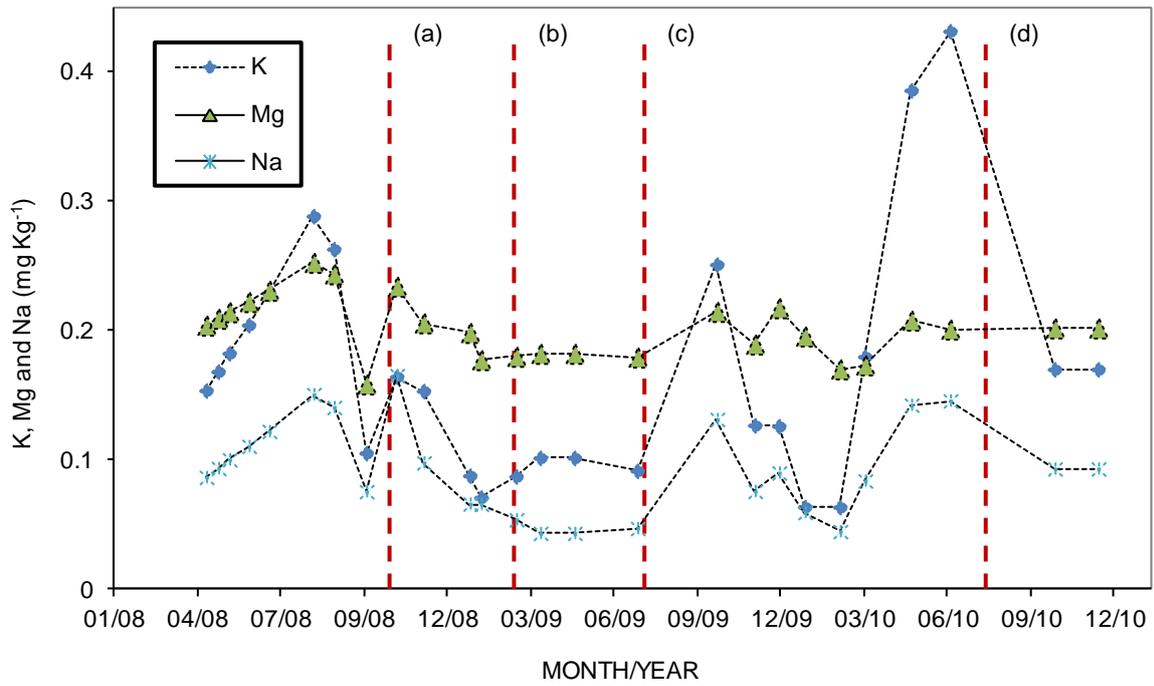
(C)



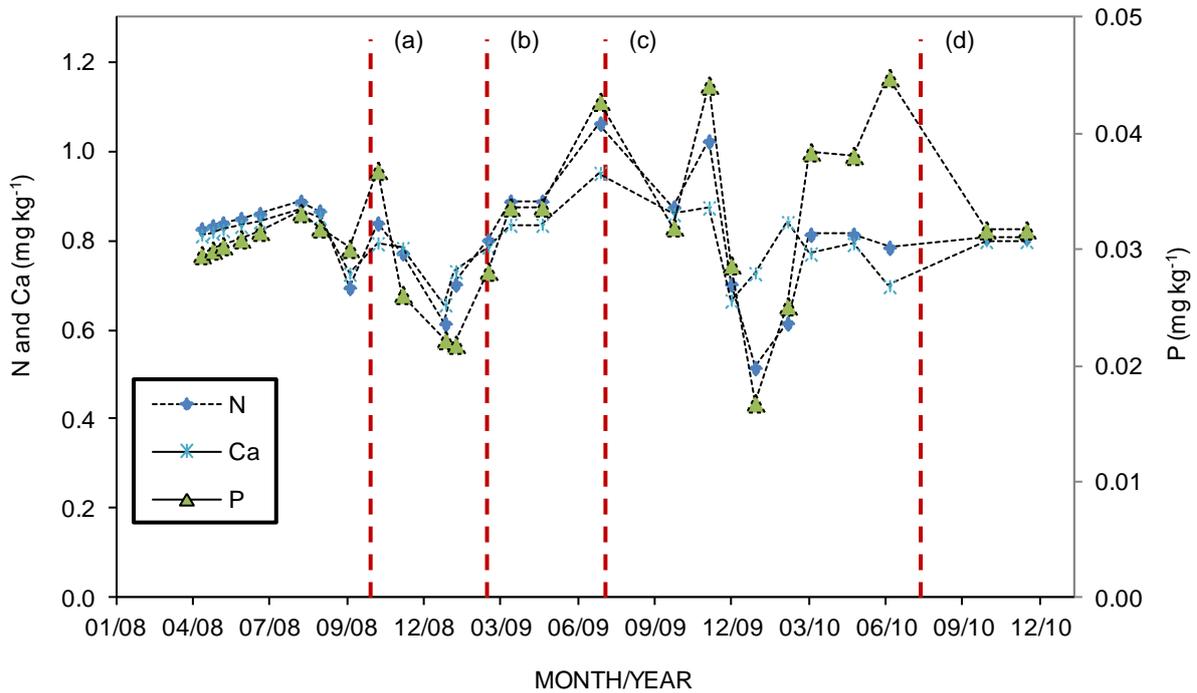
(D)



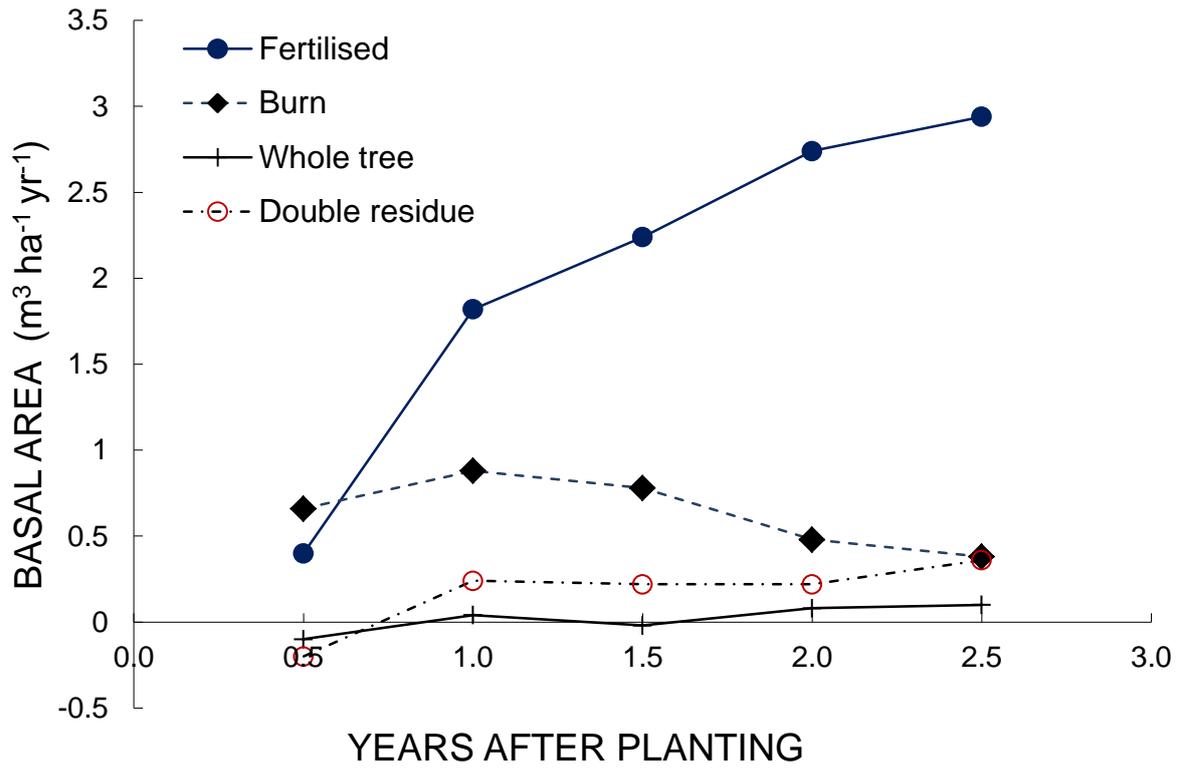
Appendix 6-3(A-F): Concentrations of N, P, K, Ca, Mg and Na in the litter and forest floor of the standing crop, Whole-Tree and No-Burn treatments. Vertical dashed lines represent the times of (a) felling; (b) residue burning; (c) planting and (d) canopy closure.



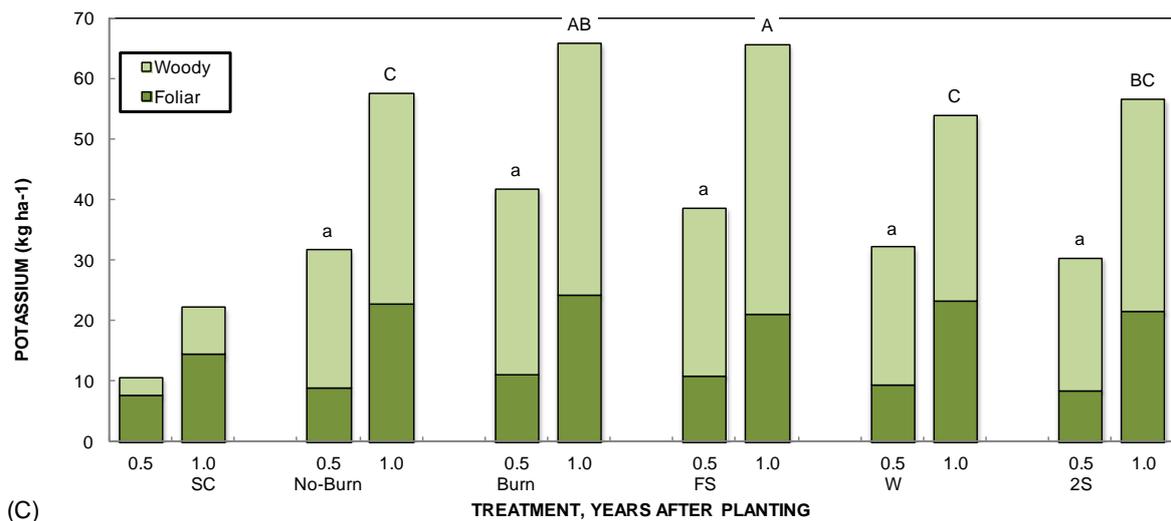
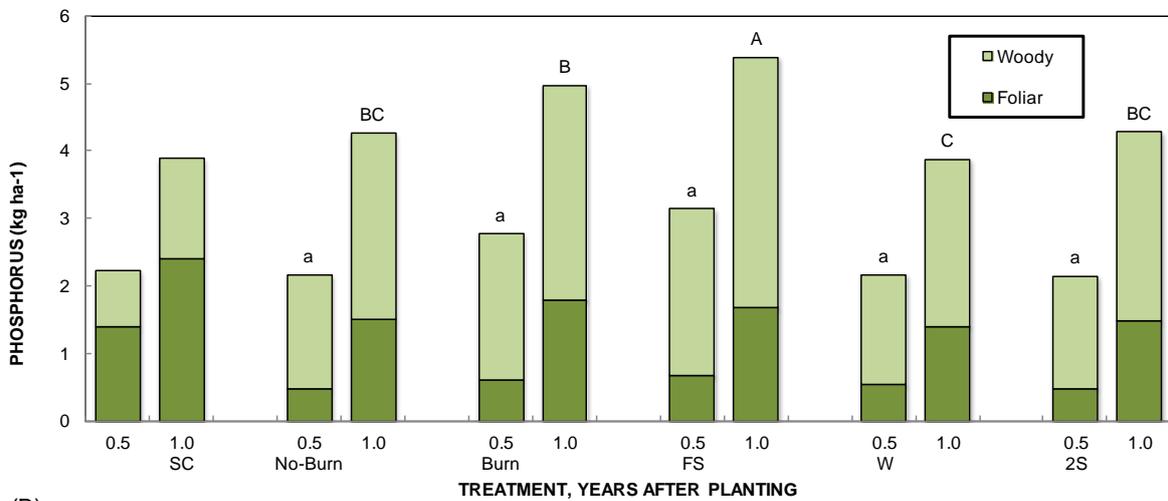
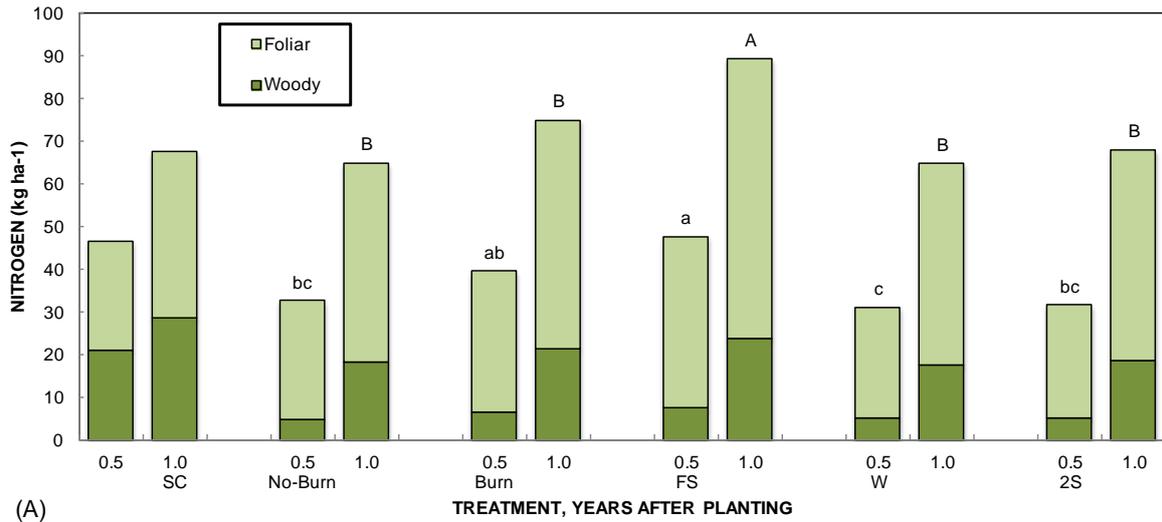
Appendix 6-4: Concentrations of K, Mg and Na in the litterfall collected under the standing crop. Vertical dashed lines represent the times of (a) felling; (b) residue burning; (c) planting and (d) canopy closure.

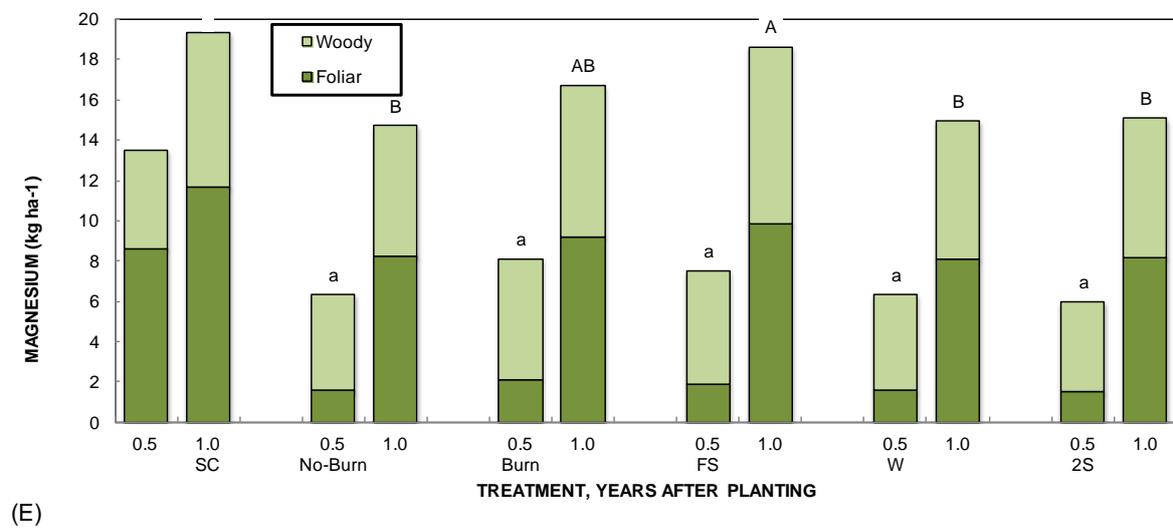
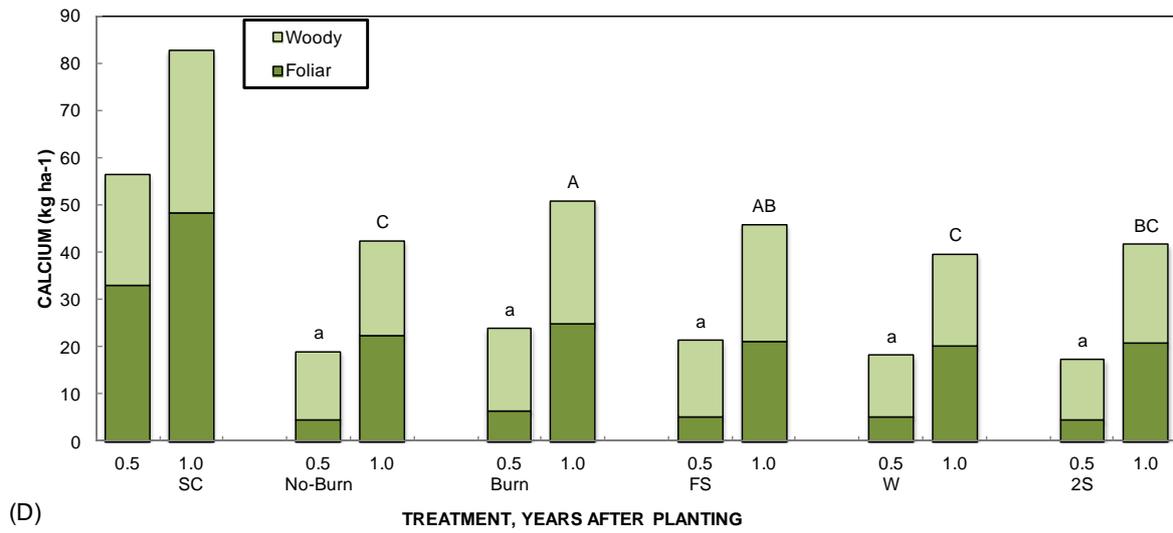


Appendix 6-5: Concentrations of N, Ca and P in the litterfall collected under the standing crop. Vertical dashed lines represent the times of (a) felling; (b) residue burning; (c) planting and (d) canopy closure.

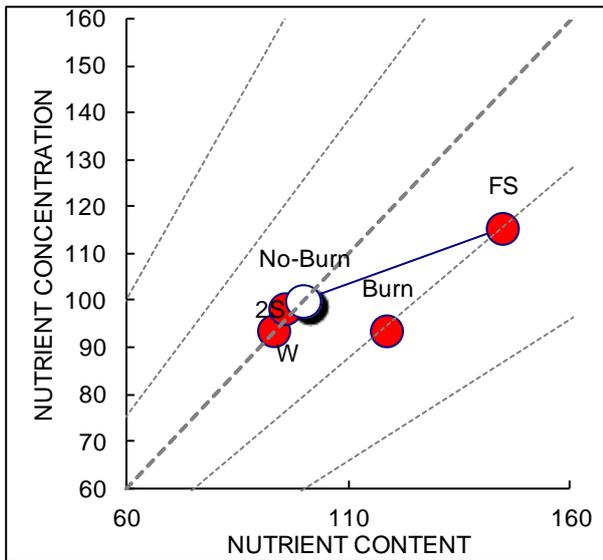


Appendix 6-6: Relative growth rates as the difference between basal area increment of each treatment and basal area increment of No-Burn treatment at six monthly intervals after planting.



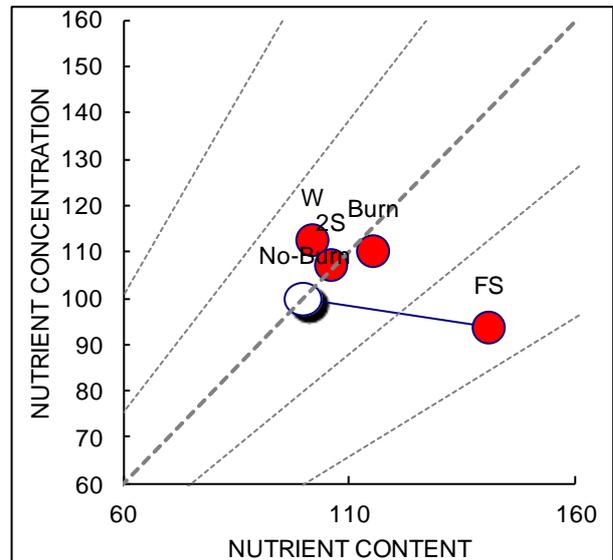


Appendix 6-7 (A-E): Nutrients accumulated in the above ground tree components between the planting to 0.5 years, and planting to canopy closure (1.0 years). Values shown on bars with LAI (m² m⁻²) on the top. Standing crop (SC) is the sum of growth and litterfall) Different a, b, c superscripts denote significant differences between treatments; lowercase for 0.5 years and uppercase for 1.0 years (LSD5%).



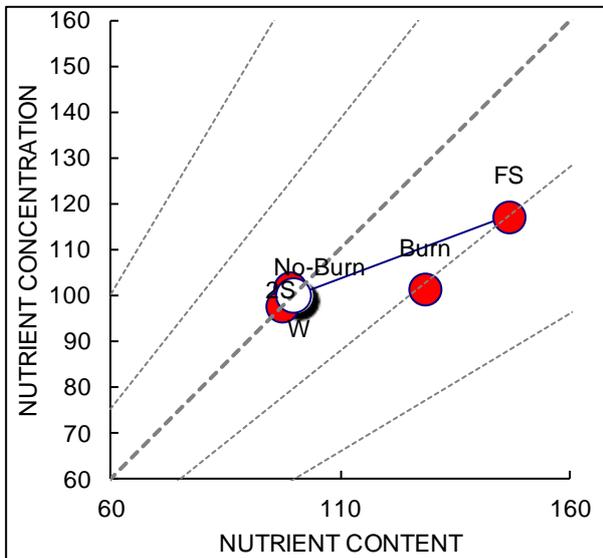
(A) Nitrogen (0.5 years)

Treatment	Mass	Conc	Content	Vector length
Burn	127	94	119	33
Fertilised	125	116	145	54
No-Burn	100	100	100	0
WholeTree	100	94	93	9
Double	97	99	96	5



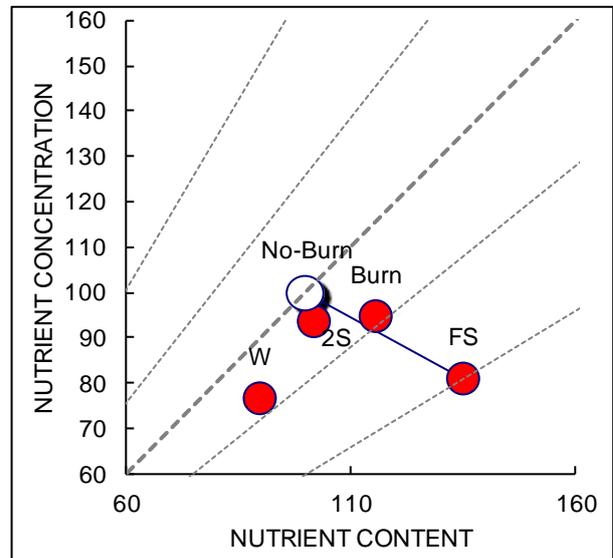
(B) Nitrogen (1.0 years)

Mass	Conc	Content	Vector length
116	110	115	24
133	94	141	53
100	100	100	0
101	113	102	13
104	107	106	10



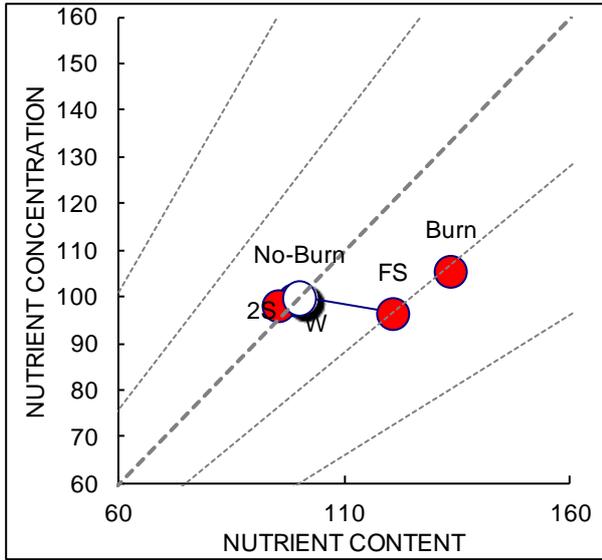
(C) Phosphorus (0.5 years)

Treatment	Mass	Conc	Content	Vector length
Burn	127	101	128	39
Fertilised	125	117	147	56
No-Burn	100	100	100	0
WholeTree	100	98	97	4
Double	97	102	99	3



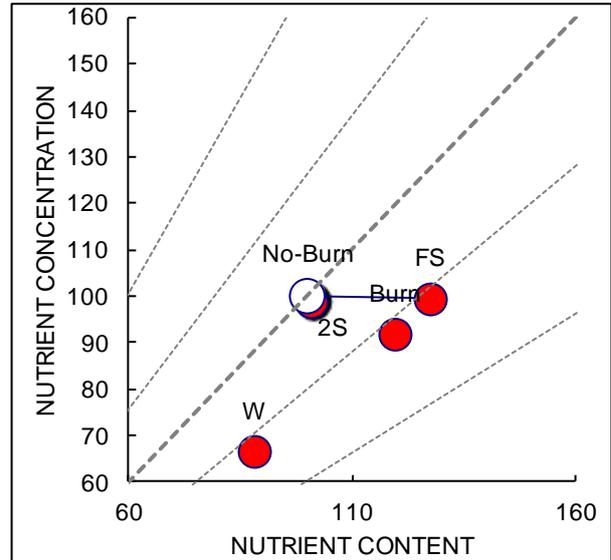
(D) Phosphorus (1.0 years)

Mass	Conc	Content	Vector length
116	95	115	22
133	81	135	52
100	100	100	0
101	77	90	25
104	94	102	8



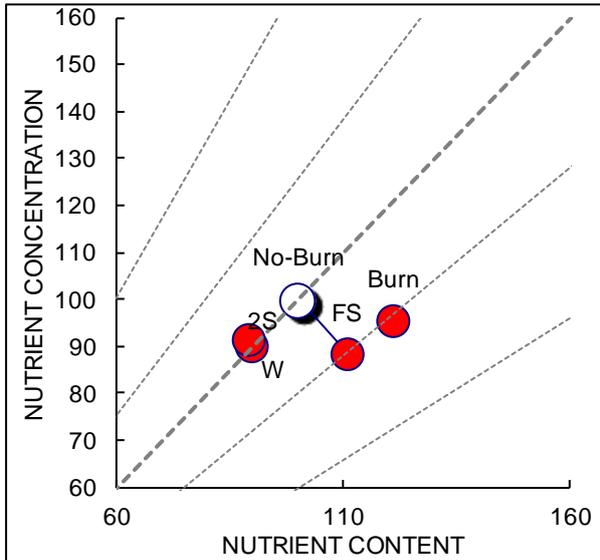
(E) Potassium (0.5 years)

Treatment	Mass	Conc	Content	Vector length
Burn	127	106	133	43
Fertilised	125	96	121	33
No-Burn	100	100	100	0
WholeTree	100	99	99	1
Double	97	98	95	6



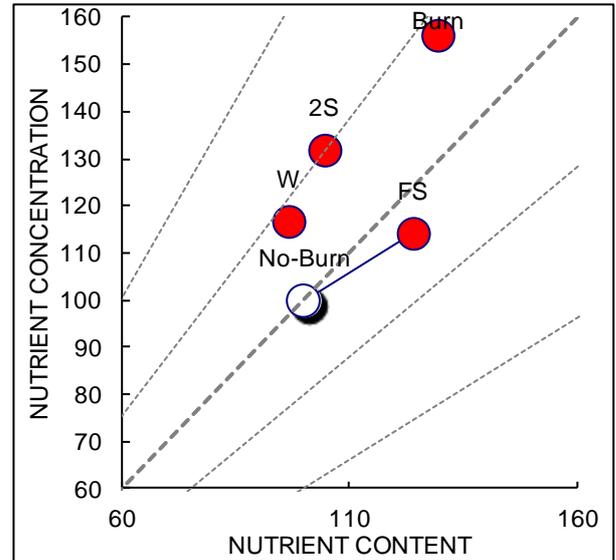
(F) Potassium (0.5 years)

Mass	Conc	Content	Vector length
116	92	119	26
133	99	127	43
100	100	100	0
101	67	88	35
104	99	101	4



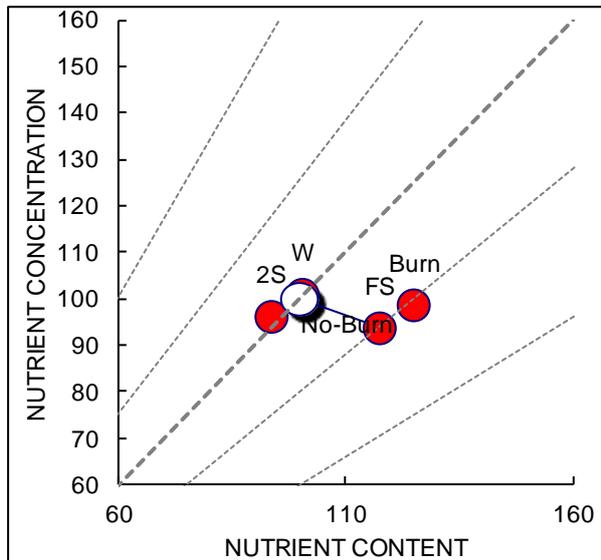
(G) Calcium (0.5 years)

Treatment	Mass	Conc	Content	Vector length
Burn	127	96	121	34
Fertilised	125	89	111	30
No-Burn	100	100	100	0
WholeTree	100	90	90	14
Double	97	92	89	14

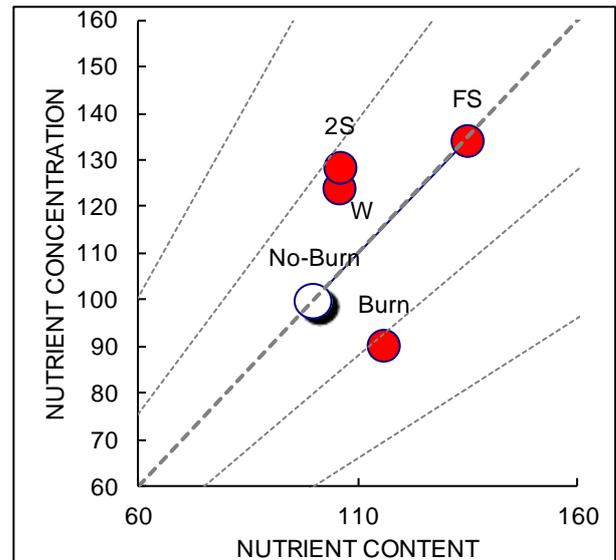


(H) Calcium (1.0 years)

Mass	Conc	Content	Vector length
116	156	129	65
133	114	124	43
100	100	100	0
101	117	96	17
104	132	104	33


(I) Magnesium (0.5 years)

Treatment	Mass	Conc	Content	Vector length
Burn	127	99	125	37
Fertilised	125	94	117	31
No-Burn	100	100	100	0
WholeTree	100	101	100	1
Double	97	96	94	8


(J) Magnesium (1.0 years)

Mass	Conc	Content	Vector length
116	90	116	24
133	134	135	59
100	100	100	0
101	124	106	25
104	129	106	29

Appendix 6-8(A-J): Vector analysis figures for each macro-nutrient element at 0.5 and 1.0 years after planting using No-Burn as the (100%) control and interpretations after Salifu and Timmer (2001) (units are as a percentage of the control). (FS =fertilised; W = whole tree; 2S=double). Vector table is given below each figure showing foliar mass, nutrient concentration (conc) and nutrient content relative to No-Burn (set at 100) and vector length.

Appendix 6-9: Equations used to predict tree component masses and leaf area (LA) at 6 using *gld* (cm) and 12 months after planting using *dbh* (cm).

Component	6 months		12 Months		Standing crop (at felling) $a \times dbh^2 + b \times dbh + c$			
	Equation	R^2	Equation	R^2	<i>a</i>	<i>b</i>	<i>c</i>	R^2
LA (m ²)	$0.002544 \times gld^{2.191603}$	0.967	$0.417 \times dbh^{2.2239}$	0.983	0.0601	0.1432	-0.6161	0.961
Foliage (kg)	$0.0002387 \times gld^{2.2152}$	0.982	$0.0482 \times dbh^{2.2239}$	0.983	0.0156	-0.0777	0.3126	0.977
Woody (kg)	$0.0000523 \times gld^{2.6213}$	0.988	$0.0801 \times dbh^{2.5014}$	0.979	0.3363	2.7031	-22.622	0.999
Branch (kg)			$0.0254 \times dbh^{2.3383}$	0.978	0.1067	-1.7175	11.112	0.982
Stem (kg)			$0.055 \times dbh^{2.5597}$	0.97	0.2002	3.7713	-28.702	0.998
Bark (kg)					0.0308	0.5978	-4.7126	0.999

Appendix 6-10: Equations using dbh to predict tree component masses and leaf area (LA) of the standing crop

Tree Component	Equation	R ²
LA (m ²)	$0.2212 \times dbh^{2.1268}$	0.901
Foliage (kg)	$0.0255 \times dbh^{2.1268}$	0.901
Branch (kg)	$-0.0995 \times dbh^2 + 3.037 \times dbh - 14.169$	0.860
Bark (kg)	$0.0443 \times dbh^2 - 0.1435 \times dbh + 0.2925$	0.961
Stem (kg)	$0.2307 \times dbh^2 - 0.2609 \times dbh - 1.6926$	0.988
Woody (kg)	$0.1756 \times dbh^2 + 2.6326 \times dbh - 15.57$	0.991

Appendix 6-11:: Biomass and macro nutrients in the tree biomass and litter layer at felling

	Pre-harvest component values					
	Biomass (Mg ha ⁻¹)	N	P	K	Ca	Mg
Litter layer	24.6	201.6	8.4	14.2	202.4	41.9
Stem	90.5	118.1	19.9	89.4	85.1	18.1
Bark	12.2	32.2	11.0	55.0	363.1	46.9
Branches	10.8	27.7	4.7	31.3	49.9	12.9
leaves	2.9	41.9	3.1	21.0	26.8	7.9
Residues ¹	50.6	303.9	27.3	121.7	643.1	109.8

¹Residues are the sum of the litter layer and tree remains after stem wood removal

Appendix 6-12: Basal area and mean heights of each treatment at six monthly intervals after planting.

Age (years)	No-Burn	Burn	Fertilised	WholeTree	Double	LSD _{5%}	<i>p</i>
	Basal area (m² ha⁻¹)						
0.5*	1.36 ^{bc}	1.69 ^a	1.56 ^{ab}	1.31 ^c	1.26 ^c	0.23	<0.001
1	3.1 ^b	3.54 ^{ab}	4.01 ^a	3.12 ^b	3.22 ^b	0.5	<0.001
1.5	6.41 ^b	6.80 ^b	7.53 ^a	6.40 ^b	6.52 ^b	0.59	<0.001
2	8.11 ^b	8.35 ^b	9.48 ^a	8.15 ^b	8.22 ^b	0.66	0.007
2.5	9.33 ^b	9.52 ^b	10.80 ^a	9.38 ^b	9.51 ^b	0.66	<0.001
	Mean tree height (m)						
0.5	2.26	2.41	2.38	2.12	2.23	0.21	0.059
1	5.43	5.75	5.95	5.29	5.47	0.52	0.099
1.5	8.34	8.47	8.56	8.31	8.40	0.20	0.112
2	10.80 ^b	10.94 ^b	11.20 ^a	10.78 ^b	10.90 ^b	0.29	0.048
2.5	11.67 ^b	11.75 ^b	12.19 ^a	11.70 ^b	11.83 ^b	0.32	0.027

*0.5 year basal area data calculated using *gld*, others calculated using *dbh*

Different a, b, c superscripts denote significant differences between treatments at each age (LSD_{5%}).

CHAPTER 7: NITROGEN MINERALISATION IN A CLONAL *EUCALYPTUS* PLANTATION ON SAND AS AFFECTED BY CLEARFELLING AND RESIDUE MANAGEMENT⁴

7.1. ABSTRACT

A study was set out to compare *in situ* soil N mineralisation, surface leaching displacement and root uptake fluxes in an undisturbed *Eucalyptus* crop, clearfelled and re-established after residue burning (Burn) and residue retention (No-Burn). This was carried out using a sequential open and closed 0 to 30 cm soil layer coring method on a fast growth, sandy low soil N site. Mineralisation and immobilisation in the undisturbed standing crop remained near zero with a net immobilisation of 3.4 kg ha⁻¹ year⁻¹. Surface N uptake was estimated at 71.3 kg of N ha⁻¹ year⁻¹ while N displacement was limited by high throughfall (27.4 kg of N ha⁻¹ year⁻¹) and high water use. Clearfelling increased N mineralisation rates, mobile NO₃-N concentrations in the soil and surface displacement. 121 kg ha⁻¹ of N was lost during residue burning leaving 111 kg ha⁻¹ of N in ash/char. Net N mineralisation was reduced after burning by nearly 53% over the 20.1 month period. Net N mineralisation in the No-Burn plots increased over time. Net N mineralisation in the No-Burn plots (45.7 kg ha⁻¹) was greater than in the Burn plots (24.5 kg ha⁻¹) over the 20.1 month period. Burning had no significant impact on 0 to 30 cm soil N displacement or root uptake. As growth was improved over the first six months after burning it was suggested that factors other than N supply were limiting early growth. Loss of N and organic substrates through residue burning can exacerbate N loss after clearfelling. A reduced mineralisation can limit N supply to trees later in the rotation. Atmospheric N inputs were substantial in this study site and may offset losses to some degree. Residue retention may be necessary to conserve soil N and maintain N supply to trees on these highly productive but low-N soils.

7.2. INTRODUCTION

Mineralisation of nitrogen (N) is crucial in forests soils where limited N availability has the potential to reduce tree growth rates during periods of high N demand (Binkley and Hart 1989; Maithani et al. 1998; Smethurst et al. 2004). The supply of N through mineralisation, relative to

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tree N demand, is an important aspect in forest nutrient management as it is also a good indicator of ecosystem health, having a strong influence on plantation tree growth (Reich et al. 1997; Morris and Boerner 1998). Nitrogen mineralisation involves the ammonification of organic-N compounds to $\text{NH}_4\text{-N}$ and the oxidative nitrification (consumption) of a portion of this ammonia to nitrite and then nitrate ($\text{NO}_3\text{-N}$) by soil micro-organisms also occurs. Net N mineralisation or immobilisation is the balance of all these processes (Raison et al. 1987). Plant uptake, nitrate leaching displacement or losses and adsorption onto soil exchange surfaces can result in further reductions in soil N, while N fixation, atmospheric deposition and cultural additions (e.g. fertilizers) can increase soil N (Binkley 1986a). The retention of added N strongly depends on soil organic matter content and the form of N deposited to the site or released through microbial processes after being deposited (Zhu and Wang 2011).

Management of harvesting and post-harvest residues plays an important role in N losses during the inter-rotation period as soil temperature, soil moisture content, organic matter content and soil chemistry are affected by residue management (Li and Herbert 2004; Goncalves et al. 2007). The potential for N losses from plantation forest nutrient pools during periods when net N mineralisation exceeds plant uptake is of particular concern during the inter-rotational (fallow) period, from clearfelling to re-establishment. Cessation of water and nutrient uptake combined with higher soil temperatures and soil moisture contents can result in increased losses through enhanced mineralisation leaching and denitrification (Fisher and Binkley 2000). Further losses of organic carbon and N associated with burning (varying according to fire intensity) can also lead to an increase in soil $\text{NH}_4\text{-N}$ release through heating and ash deposition (Weston and Attiwill 1996). Although biomass removal and burning has been shown to contribute a major portion of N loss during the inter-rotation period (Morris and Miller 1994; Spangenberg et al. 1996; Goncalves 2004; Corbeels et al. 2005; Sankaran et al. 2005; Goncalves et al. 2007), further N leaching or displacement losses resulting from accelerated N mineralisation and increased leaching can compromise the sustainability of future N supply (Carlyle et al. 1998b; Piatek and Allen 1999).

Given the increasing demand for forest biomass (Crickmay 2005), the fate of N following clearfelling disturbance and residue management is key to the conservation of N and to site recovery on N limited sites (Weston and Attiwill 1996; Gomez-Rey et al. 2007). Conservation of N following clearfelling can be achieved by maintaining a high C : N ratio after clearfelling to

induce N immobilisation and rapidly re-establishing vegetation cover (Weston and Attiwill 1996). Impacts of management practices on N supply and subsequent future growth need to be considered more carefully as part of N management strategies (Brais et al. 2002). Studies investigating N mineralisation processes in South African plantation forestry are limited predicting fertiliser requirements (Louw and Scholes 2002) and do not chronologically describe changes after clearfelling and residue management.

Our study aimed to assess soil N fluxes during the inter-rotation period of a *Eucalyptus* stand on a site characterised as having small N pools and rapid N-flux rates. The objective of this study was to determine the effect of clearfelling and two extremes of standard management practices (residue retention and residue burning) on in situ mineralisation and nitrification; giving an indication of leaching and root uptake N fluxes. A larger impact was expected after harvesting and residue burning than after residue retention.

7.3. MATERIALS AND METHODS

Materials and methods for this chapter that are common to other chapters are given in CHAPTER 3. Methods used in sequential coring and soil temperature determination are described here separately.

7.3.1. Sequential coring and analysis

A sequential coring method was used to assess *in situ* 0 - 30 cm N-mineralisation rates, including the net effects of leaching, denitrification and atmospheric deposition (Adams and Attiwill 1986; Raison et al. 1987; Adams et al. 1989). Sequential core sampling commenced a year before clearfelling (September 2007). Fifteen PVC cores (5 cm internal diameter and 40 cm in length) were inserted into the soil, to depth of 30 cm, in each sample plot. Five cores were collected immediately to represent a time-zero undisturbed sample, while five closed (capped) and five open (uncapped) cores were left *in situ* for a 28 day incubation. This duration was chosen as it was assumed to be an adequate period to allow small differences in physical conditions while enabling detection of N concentration changes (Jussy et al. 2004). At the end of each incubation period the closed and open cores were collected and transported in cool storage for further analysis. Fifteen new cores were inserted to replace the extracted cores at the end of each 28 day

incubation, and included the collection of new time-zero core samples. Cores were inserted at various positions between the trees to represent the variability of soil micro-topography, while avoiding timber extraction routes and areas disturbed by prior sampling. This generated one sample per layer per treatment in each replicated block.

The collected cores were divided into three layers of 0 - 5 cm, 5 - 15 cm, and 15 - 30 cm, bulked according to core type and soil layer, a homogenised sub-sample was placed into sealed plastic vials that were refrigerated at 5°C until chemical analysis the following day. A sub-sample of each soil sample was used to determine gravimetric water content by oven drying at 105°C for 24 hrs. The samples were analysed for extractable NH₄-N and NO₃-N after shaking 25 g soil with 50 ml 2 M KCl for 1 hour and filtering the extracts through 42 µm (Whatman No. 42) filter paper. The concentrations of NH₄-N and NO₃-N were assessed colorimetrically using segmented flow analysis with a Perstorp Flow Solution III auto-analyser. The sodium salicylate-sodium nitroprusside-hypochlorite method was used for NH₄-N (Alves et al. 1993) and the sulphanilamide-naphthyl-ethylenediamine method for NO₃-N plus NO₂-N after reducing nitrate to nitrite with copperized cadmium wire (Willis and Gentry 1987). Nitrogen mineralisation, nitrification (and immobilisation) were calculated for each incubation period as the difference between the closed core (after *in situ* incubation, day 28) and the time-zero core (pre *in situ* incubation, day 0) NH₄-N and NO₃-N concentrations. Nitrogen mineralisation was calculated in each core set as follows:

$$\Delta Closed = Closed_{(t+1)} - NI_t$$

$$\Delta Open = Open_{(t+1)} - NI_t$$

$$\Delta NI = NI_{(t+1)} - NI_t$$

Where $\Delta Closed$ is the N mineralisation in closed cores over each incubation period, NI_t is the non-incubated bulk soil core at t (time of core insertion). $Closed_{(t+1)}$ is the closed core sample taken after incubation at $t+1$. $\Delta Open$ is N fluxes in the open cores assumed to be mineralisation minus leaching plus deposition. ΔNI is N fluxes in the bulk soil assumed to be mineralisation minus leaching plus deposition minus root uptake. The difference between closed and open core N fluxes was used to estimate leaching and the difference between open cores and bulk soil N fluxes was used to estimate root uptake (Raison et al. 1987; Smethurst and Nambiar 1989). A

calculation of differences between core fluxes was therefore used to estimate leaching plus deposition and root uptake:

$$\text{Net losses (Leaching minus deposition)} = \Delta Open - \Delta Closed$$

This should be root uptake = $\Delta Open - \Delta NI$.

Alternatively, As Raison formulates it: $Closed_{(t+1)} - NI_{(t+1)} - (\text{net losses})$

Which is the same as $Closed_{(t+1)} - NI_{(t+1)} - [\Delta Open - \Delta Closed]$ The contribution of residue and litter mineralisation (and leaching) to soil N was excluded from this study due to the size of the material and financial constraints, but is included in Chapter 8 to a depth of 1 m. The effect of residue management on soil N was therefore prioritised. Net N fluxes were calculated as the sum of NH_4-N and NO_3-N . Mean soil bulk density at each depth (using five samples per depth in each plot) was used with measured N concentrations to scale nutrient content to a per hectare basis.

7.3.2. Atmospheric deposition

Atmospheric inputs described in CHAPTER 3 and CHAPTER 5 were cumulated over each incubation period and correlated with the N flux data taken from the core samples.

7.3.3. In field measurements

Soil temperatures were measured using copper constantan thermocouple probes at depths of 2.5 cm, 10 cm and 22.5 cm which represented the midpoint of each core sample depth. The access tubes and thermocouples were placed within and between the tree rows to represent the variability of soil moisture in each plot. To increase the monitoring area, soil temperature probes were installed as three clusters per depth in each plot, each cluster comprising four to five interconnected probes, giving an averaged reading from the interconnected probes. Campbell *Scientific* Cr10x data loggers measured temperatures across each probe cluster at one minute intervals.

7.3.4. Statistical analyses

Differences between treatments, soil sample depths and sampling dates were compared using a repeated measures ANOVA across time measurements and an ANOVA was used for cumulative N measures, growth and biomass with least significant difference ($LSD_{5\%}$) used to determine significance of treatment differences. Pearson correlation was used to test for correlation between N fluxes and soil measures. All statistical analysis was performed using GenStat for Windows 12th Edition (Payne et al. 2011).

7.4. RESULTS

7.4.1. Changes in system N pools during the study period

Effects of clearfelling and burning on macro-nutrient pools and subsequent tree growth are given in CHAPTER 3. A brief summary is given here for N and in **Table 7-1**. Clearfelling removed 90.5 Mg ha^{-1} of stem wood containing 118.1 kg ha^{-1} of N leaving 50.6 Mg ha^{-1} of residue and forest floor containing 303.5 kg ha^{-1} of N (**Table 7-1**). During the three month delay between clearfelling and burning the residue biomass and N content decreased by 14.2 Mg ha^{-1} and 71.2 kg ha^{-1} of N. Burning reduced the remaining residue to a 4.2 Mg ha^{-1} layer of ash and char with a loss of 121.2 kg ha^{-1} of N. Only a very small quantity of coarse char remained on the soil surface at a week after burning. The remainder leached into the soil with larger particles creating a distinctive char horizon between 5 and 10 cm from the soil surface.

Growth in the Burn treatment was initially more rapid than in the No-Burn treatment, but slowed relative to the No-Burn treatment after canopy closure (CHAPTER 6). Differences between the Burn and No-Burn treatments were no longer significant from one year to 2 years six months after planting (time of publication). Height growth was not significantly different between the Burn and No-Burn treatments at any age. The standing crop accrued 1.3 Mg ha^{-1} of biomass and 2.5 kg ha^{-1} of N into the above-ground tree components during the period between clearfelling to planting of the new crop (**Table 7-1**). Litterfall during this period was far greater than biomass accretion, totalling 6.9 Mg ha^{-1} containing 57.2 kg ha^{-1} of N. The No-Burn treatment accrued significantly less biomass and N than the Burn treatment at six months after planting. At canopy

closure a large quantity of N and biomass remained in the No-Burn treatment residues. The standing crop treatments had reduced forest floor biomass and N compared to that at clearfelling.

Table 7-1: Nitrogen pool sizes and accretion at and between the times of clearfelling, burning, and canopy closure for standing crop, Burn and No-Burn treatments. Change in forest floor/residues pools given in parenthesis.

Component	Nitrogen (kg ha ⁻¹)		
	Standing crop	No-Burn	Burn
Post-Clearfelling Pools (October 2008)			
Forest floor/Residues	201.6	303.5	303.5
Clearfelled treatments implemented but prior to Burning (March 2009)			
Forest floor/Residues	203.8	232.3	111.1*
Accretion ¹	33.1	-	-
Burning treatment implemented but prior to planting (August 2009)			
Forest floor/Residues	177.3	268.6	0.0
Accretion ¹	30.8	-	-
New crop planted and grown up to canopy closure (June 2010)			
Forest floor/Residues	169.9	176	
Accretion ¹	67.7	64.9	75.1

¹ Above-ground standing crop accretion includes litterfall; * is ash remaining after burning.

7.4.2. Soil and air temperature

Mean daily air temperature (**Figure 7.1**) remained below soil temperature for most of the study period and was characterised by a larger variability than soil temperature. Soil temperature was higher at the surface, decreasing with depth (for the depths recorded) during the summer months **Appendix 7.7**. Surface temperatures were moderately lower during winter with warmer temperatures a few centimetres below the soil surface. Soil temperature variation was slightly greater at the surface, with the largest variation occurring in the Burn treatment. Temperature differences between Burn and No-Burn were significant at all measured depths from burning to a four months after planting ($p < 0.001$) (Figure 7.1; Appendix 7.7). Differences between the standing crop and No-Burn treatments were between -2.0 and 2.0 °C after clearfelling, decreasing to differences of between -1.0 °C and 1.0 °C at planting. Mean daily soil temperatures increased by a maximum of 10.0 °C higher in the Burn than the No-Burn treatment immediately after burning, but rapidly decreased to a maximum difference of around 3.0 °C from two weeks after burning (Figure 7.1, period (b) to (c)). This initial difference was due to the albedo effect of the black ash, the subsequent reduction in differences occurred with the ash

being lost into the soil with rainfall (and possibly wind). Temperature differences between treatments continued to decrease over time becoming increasingly similar (non-significantly different) as the new crop approached canopy closure.

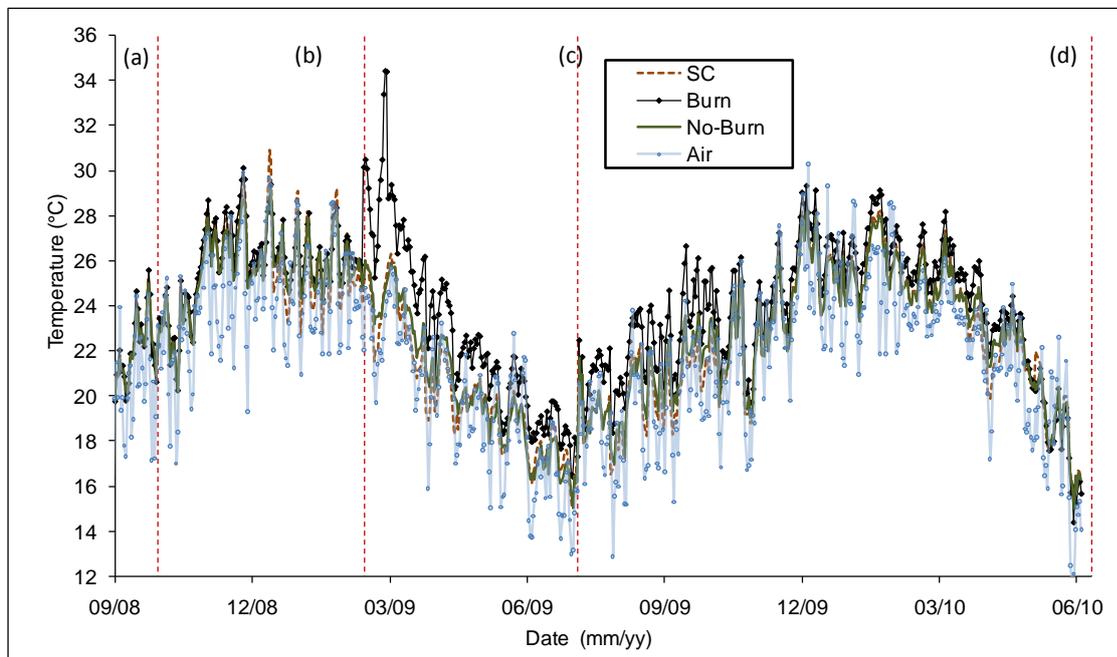


Figure 7.1: Mean daily air and soil (10 cm depth) temperatures for the standing crop (SC), single residue (No-Burn) and burned residue (Burn) treatments. Dashed lines represent (a) clearfelling; (b) residue burning; (c) planting and (d) canopy closure.

7.4.3. Soil moisture

Soil moisture content (CHAPTER 4) increased in the Burn and No-Burn treatments relative to the standing crop treatment after clearfelling (**Figure 7.2**), the moisture contents becoming similar near canopy closure of the new crop. Soil moisture content in the No-Burn treatment was slightly higher than in the Burn treatment from May 2009 to December 2009, with the largest differences occurring shortly after heavy rainfall events. Water contents determined in the core samples after extraction were moderately higher in the open cores than in the closed and time-zero samples. Differences in soil moisture contents between cores types were largest where heavy rainfall occurred during core incubation. Soil moisture content was lower at the surface (0 - 5 cm), increasing with depth in the Burn and No-Burn treatments. Water contents at 0 - 5 cm were similar between the Burn and No-Burn treatments from clearfelling to canopy closure.

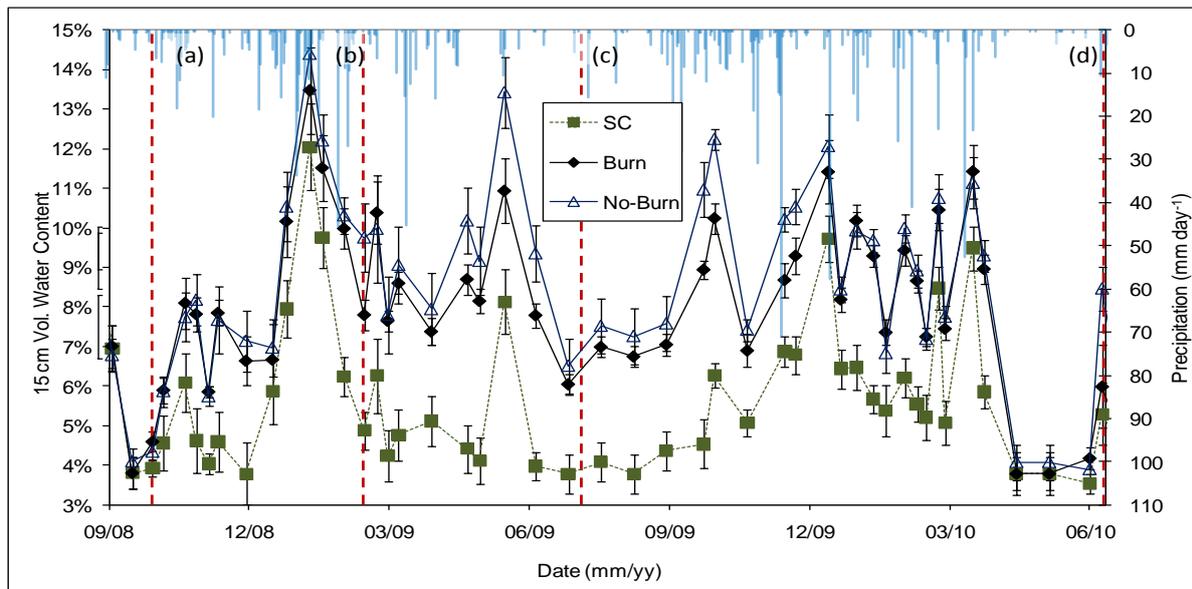


Figure 7.2: Weekly volumetric soil moisture content at 15 cm depth in the standing crop (SC), single residue (No-Burn) and burned residue (Burn) treatments. Dashed lines represent (a) clearfelling; (b) residue burning; (c) planting and (d) canopy closure. Rainfall events are shown as vertical lines scaling to an inverted secondary Y axis. Single standard deviations are given as vertical I bars.

7.4.4. Mineralisation, nitrification and immobilisation

Mineralisation and immobilisation of $\text{NH}_4\text{-N}$ followed similar patterns across all treatments throughout the study period (**Figure 7.3A**). Although p values cannot be shown for each time point in **Figure 7.3A** and **B**, significance differences are given at $p < 0.05$, ($\text{LSD}_{5\%}$). Treatment differences were variable over time. The $\text{NH}_4\text{-N}$ fluxes in the No-Burn and Burn treatments were similar while differing slightly from the standing crop treatment. Mineralisation and $\text{NH}_4\text{-N}$ immobilisation tended to be significantly greater in the standing crop treatment for most incubation periods ($p < 0.03$ in each case). $\text{NO}_3\text{-N}$ fluxes remained consistently less in the standing crop treatment than in the Burn and No-Burn treatments ($p < 0.04$ in each case), although nitrification or immobilised $\text{NO}_3\text{-N}$ gave similar patterns and changed order of significance across the three treatments over time (**Figure 7.3B**). $\text{NO}_3\text{-N}$ fluxes in the No-Burn treatment remained slightly higher than in the Burn treatment from just prior to planting, coinciding with the early onset of rainfall (**Figure 7.2**). Periods of $\text{NH}_4\text{-N}$ and $\text{NO}_3\text{-N}$ immobilisation were correlated with soil moisture content ($R = 0.406$, $p < 0.01$), immobilisation occurring during periods of low soil moisture content. Immobilisation of $\text{NH}_4\text{-N}$ and nitrification was significantly higher at 0 - 5 cm throughout the study period than at 5 - 15 cm or 15 - 30 cm. Net N mineralisation in the standing crop treatment (**Figure 7.3C**) was significantly below the clearfelled treatments on a number of occasions prior to planting ($p < 0.04$). The Burn treatment also peaked significantly in the last summer prior to canopy closure ($p = 0.037$).

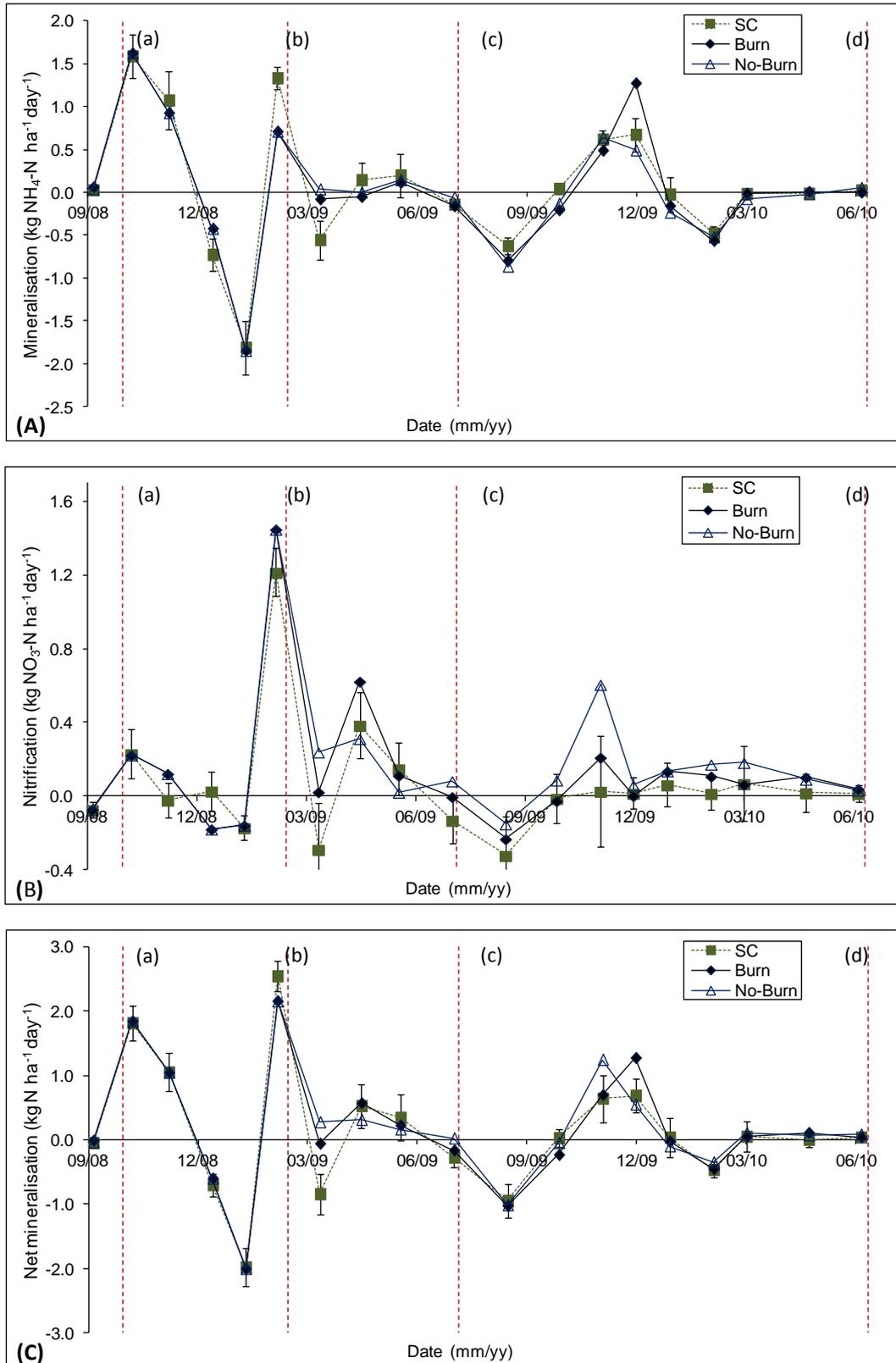


Figure 7.3(A-C): N mineralisation fluxes in the 0 - 30 cm soil layer after in situ incubation of closed cores: (A) mineralisation to NH₄-N (B) nitrification to NO₃-N and (C) net N mineralisation. Treatments are standing crop (SC), residue retention (No-Burn) and burned residue (Burn). Dashed lines represent the times of (a) clearfelling; (b) residue burning; (c) planting and (d) canopy closure. I-bars represent least significant differences (LSD_{5%}).

7.4.5. Leaching and atmospheric inputs

NH₄-N (**Figure 7.4A**) was gained by the open cores in the standing crop treatment during most incubation periods, losses only occurring after high rainfall events. Gains in the standing crop treatment occurred primarily during the dry season, beginning at the onset of the second wet season (October 2009). NH₄-N was lost from the open cores in both felled treatments after clearfelling. Differences between Burn and No-Burn were often significant ($p < 0.03$), but interchanged over time. One large loss occurred from the Burn treatment (during 12/09) that coincided with a loss from the standing crop treatments and a gain in the No-Burn treatment.

NO₃-N (**Figure 7.4B**) was also gained in the open cores of the standing crop treatment for much of the study period.

A loss of NO₃-N occurred through leaching in the No-Burn and Burn treatments from three months after clearfelling. These losses decreased in intensity after the trees were six months old. After six months losses continued to a small extent in the No-Burn treatments, but remained near zero in the Burn treatment. More NO₃-N was initially lost during the second incubation period after burning, but losses became larger in the No-Burn treatment thereafter ($p < 0.04$). Largest losses NO₃-N losses coincided with high rainfall events.

A net N gain occurred with throughfall in the standing crop treatment for most incubation periods (**Figure 7.4C**), leaching losses occurring only after high rainfall events. Net N loss through leaching occurred from both felled treatments from around two months after clearfelling, persisting to six months after planting. Losses became smaller treatments, with some small gains with rainfall. Losses from the No-Burn between the time of burning and up to six months after planting were significantly larger than from the Burn treatment for incubation periods where large NO₃-N losses occurred ($p < 0.02$).

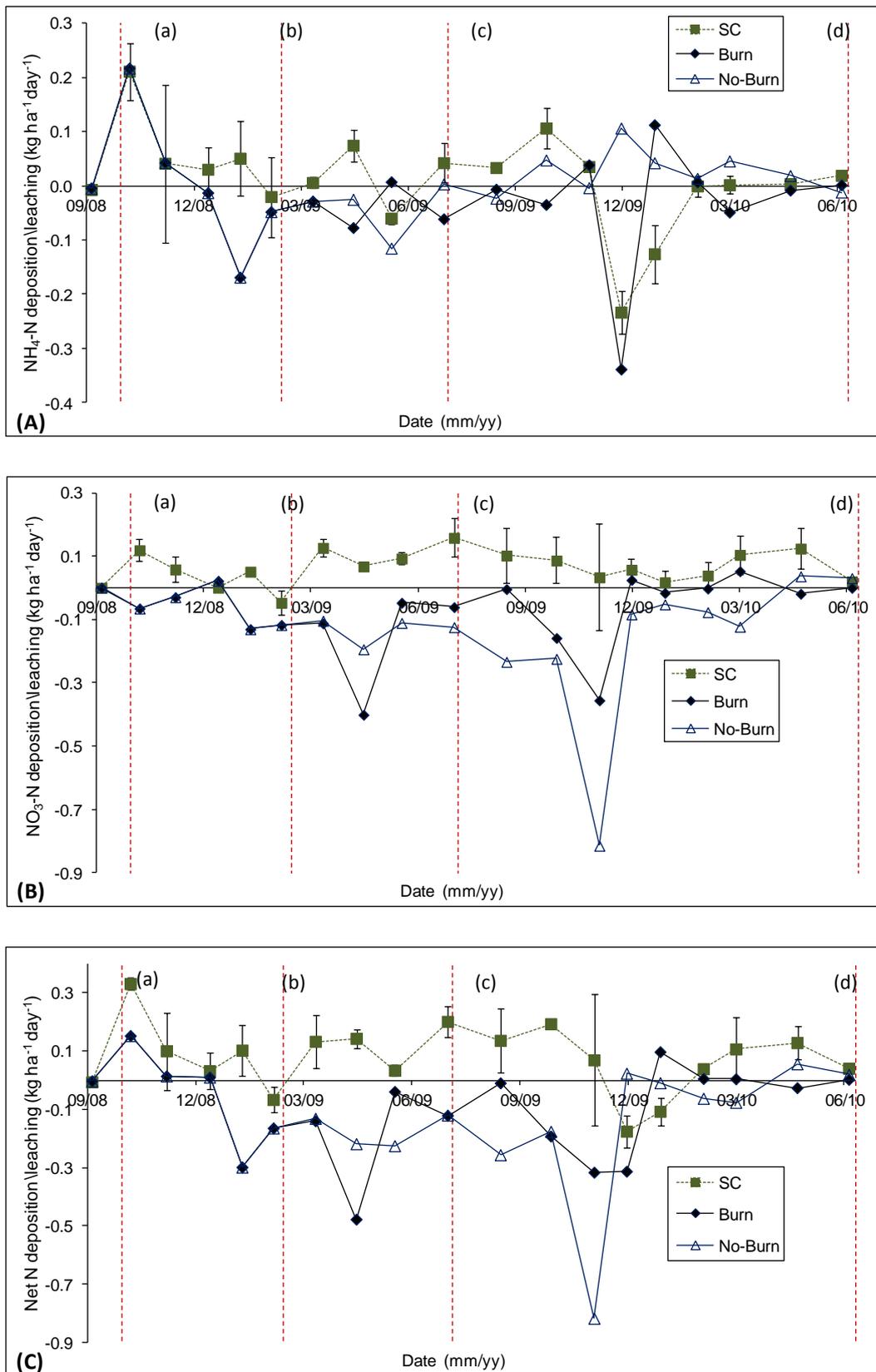


Figure 7.4(A-C): Mineral N added to the 0 - 30 cm soil layer through atmospheric deposition (positive values) or lost through leaching (negative values), estimated by the sequential soil coring method for (A) $\text{NH}_4\text{-N}$, (B) $\text{NO}_3\text{-N}$ and (C) net N. Treatments are standing crop (SC), residue retention (No-Burn) and burned residue (Burn). Dashed lines represent the times of (a) clearfelling; (b) residue burning; (c) planting and (d) canopy closure. I-bars represent least significant differences ($\text{LSD}_{5\%}$).

7.4.6. Root uptake

Root uptake in **Figure 7.5** included N leached or added from decomposition from the litter layer of the standing crop treatment and residues of the felled treatments. Some minor weed and coppice growth occurred for the first few months after clearfelling despite rigorous control measures. Net N uptake occurred in the standing crop throughout the study (**Figure 7.5**), which remained relatively consistent from the time of clearfelling to canopy closure of the new crop. A relatively large gain occurred in the standing crop treatment just prior to burning. A loss occurred in the felled treatments after clearfelling, during three incubations prior to burning. Some small gains occurred in the felled treatments from just prior to burning to planting. Losses began after planting as the new crop grew and with the onset of the second wet season. These losses continued to the end of the study. Differences between Burn and No-Burn treatments were interchangeable and significant for a few incubation periods.

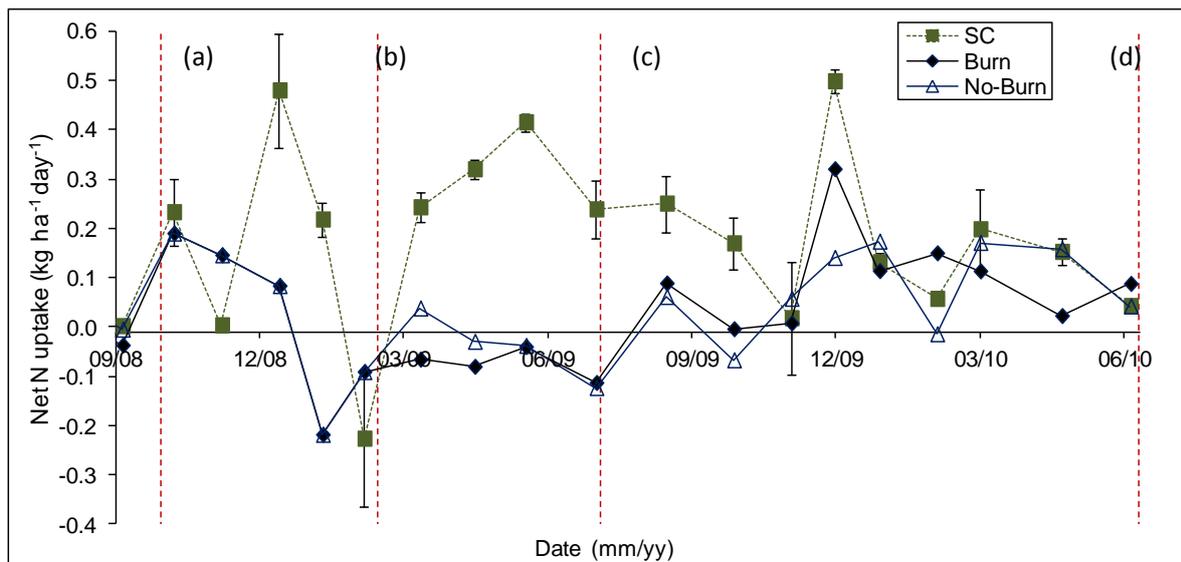


Figure 7.5: Changes in mineral N in the 0 - 30 cm soil layer through root activity (negative values) and residue/litter additions (positive values). Treatments are standing crop (SC), residue retention (No-Burn) and burned residue (Burn). Dashed lines represent the times of (a) clearfelling; (b) residue burning; (c) planting and (d) canopy closure. I-bars represent least significant differences ($LSD_{5\%}$)

7.4.7. Atmospheric deposition

Addition of N through atmospheric inputs (Chapter 5 and Figure 7.6) followed similar trends to mineralisation (**Figure 7.3**) and core measured atmospheric inputs (**Figure 7.4**). A spike in atmospheric N deposition resulted in a spike in N detected in the cores, particularly organic-N deposition. Correlations here reflect atmospheric inputs prior to each incubation. Soil NO_3-N

concentrations were positively correlated with the quantity of $\text{NH}_4\text{-N}$ added by throughfall in the standing crop treatment ($R=0.438$; $p < 0.01$) and rainfall in the clearfelled treatments ($R=0.486$; $p < 0.01$). This was most notable for $\text{NH}_4\text{-N}$ and organic-N. Although rainfall is not shown in **(Figure 7.6)**, rainfall additions were greater than throughfall, throughfall 89% of $\text{NO}_3\text{-N}$ in rainfall and 96% of $\text{NH}_4\text{-N}$ in rainfall. Uptake of N occurred through canopy exchange processes. The quantity of throughfall organic-N deposition was positively correlated with nitrification rate ($R=0.633$; $p < 0.01$). Throughfall organic-N deposition was also correlated with $\text{NO}_3\text{-N}$ ($R=0.399$, $p < 0.01$). The quantity of throughfall $\text{NH}_4\text{-N}$ deposition was negatively correlated with nitrification rate ($R=-0.502$; $p < 0.01$). Nitrification rates were negatively correlated with the quantity of $\text{NH}_4\text{-N}$ added with rainfall ($R=-0.325$; $p < 0.01$).

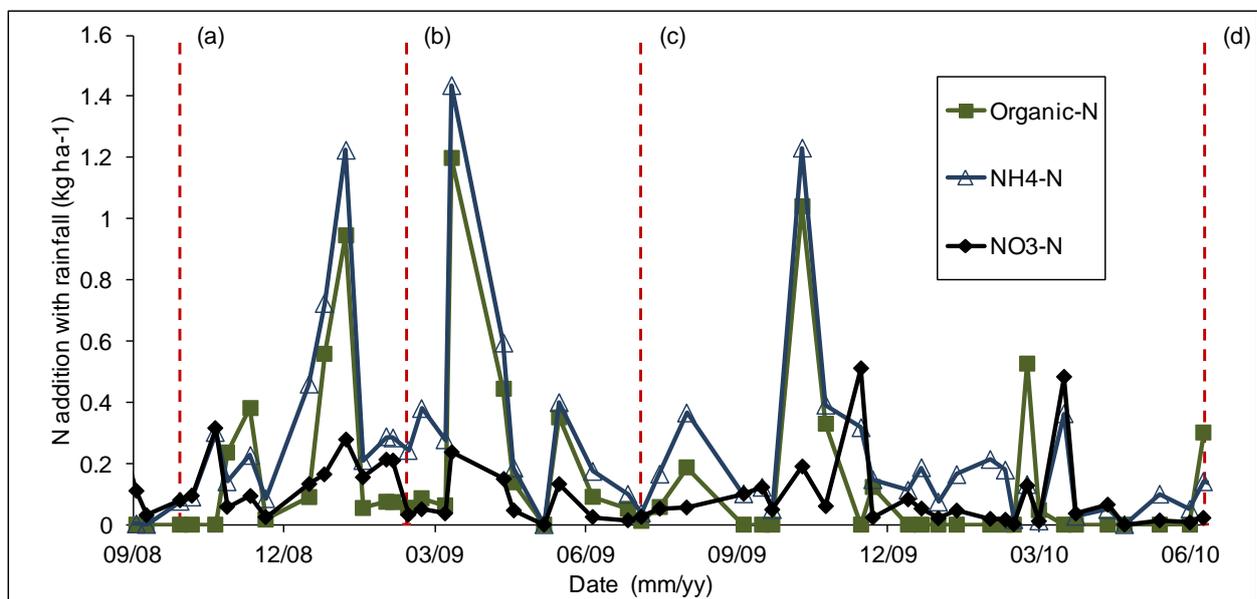


Figure 7.6: $\text{NO}_3\text{-N}$, $\text{NH}_4\text{-N}$ and Organic-N added throughfall (weekly). Dashed lines represent the times of (a) clearfelling; (b) residue burning; (c) planting and (d) canopy closure using data from CHAPTER 5.

7.4.8. Cumulative net nitrogen fluxes

Cumulative and annualised net nitrogen fluxes estimated from core samples are shown in Table 7.2. The period between clearfelling and planting exhibited net N mineralisation in all treatments, with statistically similar levels in the No-Burn and Burn treatments, but with significantly lower levels in the standing crop treatment. Planting to canopy closure gave a cumulative net immobilisation of N in the Burn and standing crop treatments, which were statistically similar. The No-Burn treatment yielded net N mineralisation, statistically different to the other treatments. All treatments were significantly different for cumulative net N

mineralisation between clearfelling and canopy closure, which showed the largest net N mineralisation in the No-Burn, followed by the Burn, with a net immobilisation occurring in the standing crop treatment. The nine month period between clearfelling and planting was dominated by immobilisation of $\text{NH}_4\text{-N}$, and nitrification to $\text{NO}_3\text{-N}$ (**Figure 7.3A and B**). Treatment nitrification differences were significant during each period of the study at $p < 0.001$. No-Burn treatment immobilisation of $\text{NH}_4\text{-N}$ was significantly larger than in the Burn treatment, which was significantly larger than the standing crop treatment ($p < 0.001$). Immobilisation of $\text{NH}_4\text{-N}$ did not change significantly with depth between felling and planting ($p > 0.08$) and planting and canopy closure ($p = 0.232$). Nitrification however tended to be greater at the surface than at depth between felling and planting ($p < 0.001$) and planting and canopy closure ($p < 0.001$). Overall nitrification was 2.7 fold greater at 0 to 5 cm than 5 to 15 cm depths and 7.0 fold greater at 5 to 15 cm than 15 to 30 cm depths ($p < 0.001$).

The combined effect of N deposition and leaching could also be estimated from the core sample data (Table 7-2). A net N gain occurred in the standing crop treatment throughout the study through atmospheric deposition (**Table 7-2**) with a slightly higher deposition rate during the first nine months of the study. Although deposition also occurred in the felled treatments, it was significantly outweighed by leaching losses. An overall net N loss occurred in the No-Burn and Burn treatments (**Table 7-2**). These were statistically similar for the first period (felling to planting) and larger in the No-Burn during the second period (planting to canopy closure). Differences between the felled treatments were not significant over the entire period (felling to canopy closure). Differences between treatments in **Table 7-2** were attributed to treatments differences in $\text{NO}_3\text{-N}$ during the first, second and full period $p < 0.001$ in each case. Deposition of $\text{NH}_4\text{-N}$ was not significantly different between treatments during any period ($p = 0.24, 0.06, 0.09$ respectively). Leaching of $\text{NO}_3\text{-N}$ decreased significantly with depth for all periods ($p = 0.01, 0.04, 0.01$ respectively), whereas $\text{NH}_4\text{-N}$ was only significant with depth during the first period ($p = 0.00, 0.21, 0.11$ respectively). This significant difference occurred as a greater leaching at 15 cm than at 30 cm depth. A total of $4.1 \text{ kg ha}^{-1} \text{ yr}^{-1}$ of $\text{NH}_4\text{-N}$ and $41.8 \text{ kg ha}^{-1} \text{ yr}^{-1}$ $\text{NO}_3\text{-N}$ was gained with rainfall in the standing crop during the clearfelling to canopy closure period. A total of 3.6 and $19.1 \text{ kg ha}^{-1} \text{ yr}^{-1}$ of $\text{NH}_4\text{-N}$ and 73.6 and $39.6 \text{ kg ha}^{-1} \text{ yr}^{-1}$ $\text{NO}_3\text{-N}$ was leached during the clearfelling to canopy closure period in the No-Burn and Burn treatments respectively. Less N was recorded as throughfall above ground during each period than was recorded by open core calculations.

A large quantity of N was lost from the soil surface of the standing crop with root uptake (**Table 7-2**) across both periods. Root uptake was significantly larger in the standing crop treatment than in the felled treatments throughout the study. Calculated root uptake was small prior to planting, with no significant differences between felled treatments. Post planting root uptake was larger, but differences were not significant between felled treatments. Root uptake prior to planting occurred as a loss of $\text{NO}_3\text{-N}$ in the felled treatments, occurring primarily from 0 to 5 cm depth. A small gain in $\text{NH}_4\text{-N}$ occurred in the felled treatments at the 0 to 5 cm soil depth during this period. Root uptake of $\text{NO}_3\text{-N}$ and $\text{NH}_4\text{-N}$ was largest in the standing crop treatment, but not significant different between felled treatments. More $\text{NO}_3\text{-N}$ was taken up than $\text{NH}_4\text{-N}$ throughout the entire study period. Uptake did not change with depth prior to planting ($p=0.25$), but decreased with depth after planting ($p<0.001$). These differences occurred through significant differences in $\text{NO}_3\text{-N}$ uptake across each period, $p<0.001$ in each case. Although differences in $\text{NH}_4\text{-N}$ uptake did occur, these were as a smaller uptake from the 0 to 5 cm depth than from the other two depths ($p<0.001$ for each period). Root uptake amounted to $42.8 \text{ kg ha}^{-1} \text{ yr}^{-1}$ of $\text{NH}_4\text{-N}$ and $76.5 \text{ kg ha}^{-1} \text{ yr}^{-1}$ $\text{NO}_3\text{-N}$ in the standing crop treatment for the entire study period. A total of 8.6 and $4.6 \text{ kg ha}^{-1} \text{ yr}^{-1}$ of $\text{NH}_4\text{-N}$ and 35.2 and $44.1 \text{ kg ha}^{-1} \text{ yr}^{-1}$ $\text{NO}_3\text{-N}$ was leached during the clearfelling to canopy closure period in the No-Burn and Burn treatments respectively. Aboveground N accumulation (accretion estimated from biomass studies) was larger than 0 to 30 cm root uptake calculated using core methods. However, standing crop aboveground N accumulation (including litterfall) was of a similar order of magnitude to core calculated uptake. Loss of N from the litter and residues was large. This loss could not be directly related to soil core fluxes as fine root growth occurred near the surface (visual observation) in direct contact with the humus layer component of the residues and litter layer.

Table 7-2: Cumulative (kg ha^{-1}) and annualised ($\text{kg ha}^{-1} \text{ year}^{-1}$) net nitrogen fluxes in-field cores of 0 – 30 cm soil from clearfelling to canopy closure of the new crop.

		Cumulative (kg ha^{-1})			<i>p</i>	Annualised ($\text{kg ha}^{-1} \text{ year}^{-1}$)		
		No-Burn	Burn	Standing crop		No-Burn	Burn	Standing crop
Clearfelling to planting (9 months)	Net Mineralisation	38.1 ^a	28.8 ^a	6.0 ^b	0.011	50.6	38.2	8.0
	Deposition - leaching	-28.8 ^a	-31.9 ^a	24.5 ^b	<.001	-38.2	-42.3	32.5
	Uptake	11.1 ^a	6.1 ^a	63.1 ^b	<.001	14.7	8.1	83.8
	Rainfall/Throughfall	15.3	15.3	19.2		20.3	20.3	25.5
	N accretion	0.0	0.0	63.9*				
Planting to canopy closure (11 months)	Net Mineralisation	7.7 ^a	-4.3 ^b	-11.6 ^b	0.02	8.4	-4.7	-12.6
	Deposition - leaching	-48.5 ^a	-26.6 ^b	21.3 ^c	<.001	-52.7	-28.9	23.1
	Uptake	32.6 ^a	42.7 ^a	56.2 ^b	<.001	35.4	46.4	61.1
	Rainfall or Throughfall	9.5	9.5	15.0		10.3	10.3	16.3
	N accretion	64.9	75.1	67.6*				
Clearfelling to canopy closure (20.1 months)	Net Mineralisation	45.7 ^a	24.5 ^b	-5.7 ^c	0.004	27.3	14.6	-3.4
	Deposition - leaching	-77.2 ^a	-58.6 ^a	45.8 ^b	<.001	-46.1	-35	27.4
	Uptake	43.7 ^a	48.8 ^a	119.3 ^b	<.001	26.1	29.2	71.3
	Rainfall/Throughfall	24.7	24.7	34.2		14.8	14.8	20.4
	N accretion	64.9	75.1	131.5*				
	Residue/litter mass loss	118.7	62.4	148.4*				

Mineralisation, nitrification and deposition gains are positive; immobilisation and net loss are negative. Different a, b, c superscripts denote significant differences (LSD5%). * includes litterfall, but excludes canopy exchange

7.5. DISCUSSION

Denitrification was likely to have been zero or negligible at our study site due to the well drained nature of the soil (sandy texture) and the water contents measured over the course of the trial CHAPTER 3, (Dovey et al. 2011) remaining at or well below field capacity (Groffman 1995; Færge and Magid 2004). Net mineral N content in cores was therefore affected by a combination of some or all of the following processes: microbially mediated N mineralisation and nitrification fluxes, N addition with rainfall/throughfall, litter decomposition and N leaching. It is therefore important to realise that the methodology used in the study can have a pronounced influence on N additions, losses and transformations in the soil. In the open cores, inputs of water, N and other nutrients and leaching continued throughout the incubation period, but water uptake by plant roots was excluded. This may lead to higher soil moisture contents than that experienced under ambient field conditions. Closed-top cores receive no atmospheric N inputs and leaching is negligible, but cores can become somewhat drier than ambient field conditions, which can affect microbial activity negatively. These factors, when compared to the closed

cores and the bulk soil *in situ* incubation fluxes may have altered the comparative basal mineralisation rates between the cores, possibly violating the assumptions given in the methodology (Smethurst and Nambiar 1989; Jussy et al. 2004). This may have increased soil N turnover rates (Wienhold et al. 2009) possibly increasing the N mineralised and nitrified (and subsequently leached out), of the open-topped cores and bulk soil, particularly in the clearfelled treatments. Despite these drawbacks, the methods give a large amount of information about N fluxes in the surface soil where the majority of fine root growth occurs (Smethurst and Nambiar 1989). This is the most important N supply region of the soil; therefore leaching out of this zone constitutes a displacement of N from this primary supply zone.

Taken over the two study periods, data in **Table 7-2** show net mineralisation for the clearfelled treatments, but a small immobilization effect in the standing crop. The most plausible explanation for this observation is that lower temperatures throughout the incubation period (**Figure 7.1**), coupled to significantly lower soil moisture contents (**Figure 7.2**) and significantly lower mineral N contents at the onset of most incubation periods, all contributed to lowered N mineralisation rates in the standing crop treatments. Leaching process could have been most pronounced in the two felled treatments because the soil was wetter than the standing crop at the start of each incubation period (**Figure 7.2**), and N mineralisation and nitrification process would have been favoured by higher temperatures in the felled treatments (**Figure 7.1**) and higher pH in the topsoil of the burnt treatment. pH (KCl) at the end of the study (0 to 15 cm) averaged at 4.42, 6.37 and 5.21 in the standing crop, Burn and No-Burn treatments respectively. Furthermore, despite canopy capture and uptake of $\text{NH}_4\text{-N}$ and $\text{NO}_3\text{-N}$, the standing crop received more net N (through organic-N inputs in throughfall) than N inputs received through atmospheric deposition in the two clearfelled treatments (CHAPTER 5). Higher moisture content, coupled to higher nitrification and leaching rates was possibly an overriding mechanism in the open-top cores of the clear-felled treatments. Similarly, different conditions in the bulk soil fluxes may have resulted in a false N uptake. Immobilisation, higher leaching and some limited weed and coppice growth may have accounted for positive root uptake values prior to planting.

7.5.1. Standing crop N fluxes

Net N mineralisation (**Table 7-2**) remained relatively stable in the standing crop throughout the study period alternating between N release and N immobilisation (or consumption). The relatively stable soil temperatures and consistently low soil moisture contents (between rainfall events) gave little opportunity for rapid mineralisation to occur. The low levels of soil moisture (**Figure 7.2**) and the negative number for net N mineralisation over the study period (**Table 7-2**) shows that the standing crop was sub-optimally supplied with both water and soil derived N. This implies a potential growth limitation through both factors. However, N accretion in the standing crop (**Table 7-1**) continued despite being limited by negative soil mineralisation N fluxes. As root uptake may be overestimated using this technique (Smethurst and Nambiar 1989), a large proportion of above-ground tree N accretion was most likely supplied from atmospheric inputs (throughfall), litter decomposition (**Table 7-1**) and deeper soil layers. Throughfall and stemflow recorded at this study site (CHAPTER 5) contributed a large amount of N between clearfelling and canopy closure of the new crop. The difference between addition and accretion resulted in more N added to the soil surface than was utilised and immobilised (**Table 7-2**).

7.5.2. Felled crop N fluxes

Post clearfelling N dynamics were different from that of the standing crop (**Table 7-2**). Changes in N-fluxes became more apparent at around three months after clearfelling, coinciding with the time that rainfall induced soil moisture contents began to diverge. Soil moisture and drainage fluxes presented in (CHAPTER 4; Dovey et al. (2011) show these differences to gradually diminished with growth and increasing water demand of the new crop. The low N mineralisation rates under rapidly drying soils in the standing crop (**Figure 7.2**) and the increased N mineralisation rates with soil moisture recharge after clearfelling show the reliance of N mineralising soil micro-organisms on a stable and wet soil moisture regime. A lack of significant temperature differences between the standing crop and the felled treatments (and later No-Burn) occurred through shading of the soil by the harvest residues while the open canopy architecture of the standing crop permitted high levels of solar penetration between individual tree canopies, thereby increasing surface temperatures. Leaching and root uptake estimates were not statistically different between treatments despite larger mineralisation rates in the No-Burn treatment. This implies residue burning to have minimal impact on soil leaching processes on

this site. It also implies, given the initial growth improvement after burning that factors other than N were more limiting to growth.

The long lead time between clearfelling and burning may have reduced initial differences between the Burn and No-Burn plots by allowing N and organic compounds to enter the soil from residue decomposition prior to burning. Immobilisation of N (**Table 7-2**) in the burnt plots after burning indicates a soil depletion of organic substrates required for N mineralisation and though unfavourable soil surface moisture and temperature conditions (high diurnal variation). A substantial quantity of N was released through N mineralisation and potentially through burning, but this occurred before planting. Owing to this delay, N was not available to the new crop, but lost through leaching and wind erosion prior to being beneficial. The large loss of N during burning and later increased leaching will not be economically beneficial if the cost of N replacement is greater than tree growth gains realised after burning. The burn treatment in Table 7-2 also shows the largest change between periods (from clearfelling to planting) and (from planting to canopy closure): Nitrogen mineralization is large and positive in the former period (28.8 kg ha^{-1}) and becomes negative (immobilized) in the second period. These factors may have compromised growth later in the rotation while potentially negatively impacting the sustainability of future N pools.

Higher levels of net N mineralisation in the No-Burn treatments were likely related to the larger amounts of organic substrate on the soil surface and more stable soil surface temperature and moisture regimes. As a result, N mineralisation remained positive (although less) after planting, contrasting with the Burn treatment. While early growth was less in the No-Burn treatment than in the Burn treatment, a continued N supply will satisfy a larger proportion of tree growth N demand later in the rotation, given the reduced N supply in the Burn treatment. Net mineralisation rates may peak again in the No-Burn treatments later in the rotation as substrate provision (large levels of residue remaining on the No-Burn as opposed to the Burn treatment), soil moisture retention and temperature stabilisation continue enhance net N mineralisation. This was found to occur in a similar study on sands (Nzila et al. 2002).

Treatment effects on cumulative net mineralisation (**Table 7-2**) were similar to those in a comparable study in Brazil (Goncalves et al. 2007), where higher rates of N mineralisation

occurred after minimal disturbance (residue retention) than after residue burning. Similar quantities of N were mineralised in the Brazil study, 58 kg ha⁻¹ in the minimum disturbance treatment and 28 kg ha⁻¹ in the burned treatment over a 21 month period, to our study (**Table 7-2**). Mineralisation in the standing crop treatment of the Brazil study was different, 77 kg ha⁻¹; and was explained by solar penetration from the edges of the standing crop plots from the adjacent felled plots and higher rainfall of the Brazil study. Solar penetration did not occur from the edges in our study. Mean monthly net mineralisation rates (**Table 7-2**) were also within a similar order of magnitude to those in studies reported in Nzila et al. (2002) under similar crop, climate and soil conditions to our study.

7.5.3. Consequence of atmospheric N inputs

This study suggests a large amount of atmospheric N input to the site. High levels of atmospheric N addition may have increased both the rate and quantity of nitrification and mineralisation. Atmospheric deposition may affect the rate and quantity of N released from N mineralisation through changes in C:N ratios, soil pH and litter quality (Månsson and Falkengren-Grerup 2003; Rao et al. 2009). Nitrogen mineralisation rates were increased after N fertilisation in certain studies (Aarnio and Martikainen 1992; Prescott et al. 1995; Fox 2004), although large N additions reduced organic matter quality and mineralisation rates in one study as a result of increased nitrification inducing base cation stripping (Fox 2004).

The addition of organic and inorganic compounds from the decomposing residues and from atmospheric deposition in particular, may have altered the basal mineralisation rates by changing the quantity and quality of mineralisable soil organic compounds. This may also be a constraint in some N mineralisation models that assume a basal mineralisation rate that is altered by water and temperature regimes alone (Paul et al. 2002). Such models do not allow N to become immobilised in the soil and do not account for the effects of N additions from above the soil surface either. This may be evidence towards N mineralisation fluxes being overridden by atmospheric organic N inputs. Although literature does not report on the interaction between atmospherically derived organic-N and N-mineralisation, it is suggested in our study to have contributed to N-fluxes. A lack of large differences between the treatment extremes during the period under study (**Figure 7.3**) may also indicate net N mineralisation to be largely dependent on intrinsic site factors rather than upon soil and site management differences. Intrinsic site

differences overriding treatment effects has been observed in a number of similar studies, reviewed in Carlyle et al. (1998b). The role that the quantity and form of atmospheric N inputs plays in N mineralisation processes needs further investigation as this may drive the ability of a site to immobilise and conserve N during the inter-rotation.

7.6. CONCLUSION

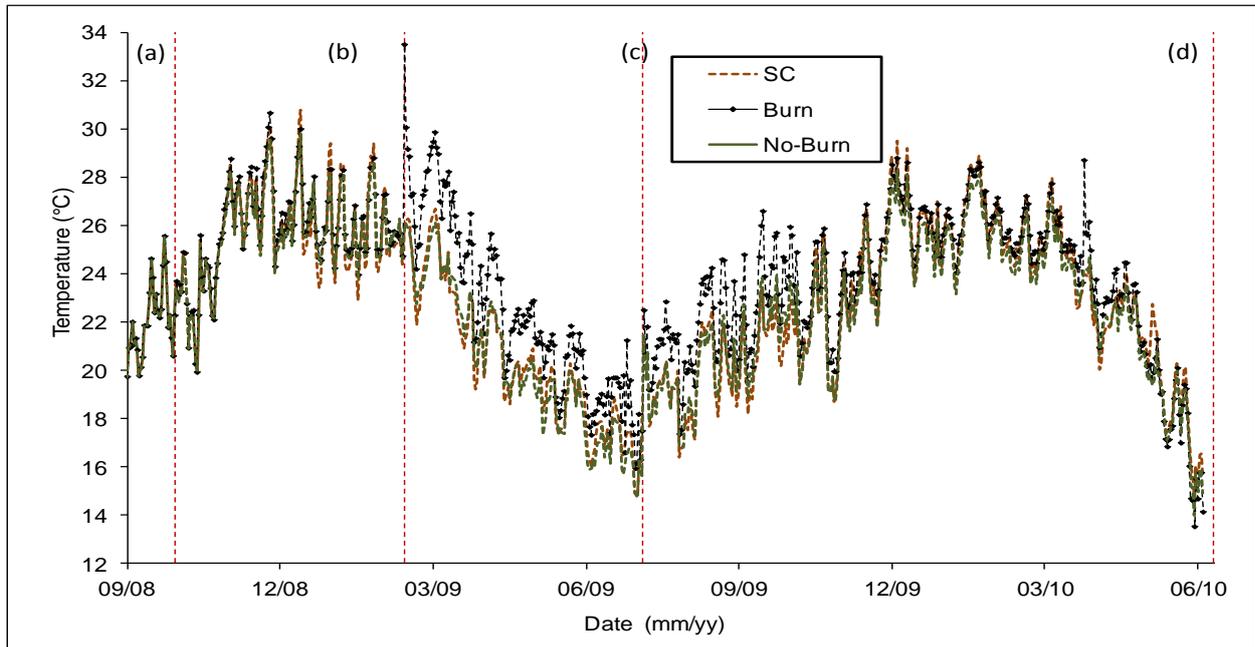
Our study shows a relatively balanced N fluxes in the undisturbed standing crop where release and consumption of N occurs at low levels, increasing during high soil moisture conditions. Nitrogen gained with rainfall under the canopy (throughfall) appeared to be rapidly mineralised, nitrified and possibly removed with tree uptake. Site disturbance through clearfelling increased the rate of N release through mineralisation processes through an improvement in soil moisture and temperature regimes and the provision of additional substrate. Increased soil moisture and NO₃-N concentration in the soil of the felled plots induced an increase in NO₃-N leaching, displacing N from upper to lower soil layers.

Burning of residues in this low-N system reduced soil net N mineralisation by nearly 53% over the 20 month period. The first six months of tree growth was initially improved on the burnt plots, which suggest that factors other than N supply were more limiting to growth during this phase of crop growth. Burning was also less N conservative. The loss of N from the Burn treatment through burning and reduced mineralisation may limit the availability of N to the trees in the burnt plots later in the rotation. Evidence of a large reduction in N mineralisation in the burnt treatment from clearfelling until canopy closure can be seen in Table 7-2. This loss of N during burning and the large quantities of more mobile N in the ash after burning can increase the risk of N loss through erosion (wind and water). This will reduce future N availability to the current crop and negatively impact on soil N pools and fluxes in the long-term (O'Connell et al. 2004; Smaill et al. 2010).

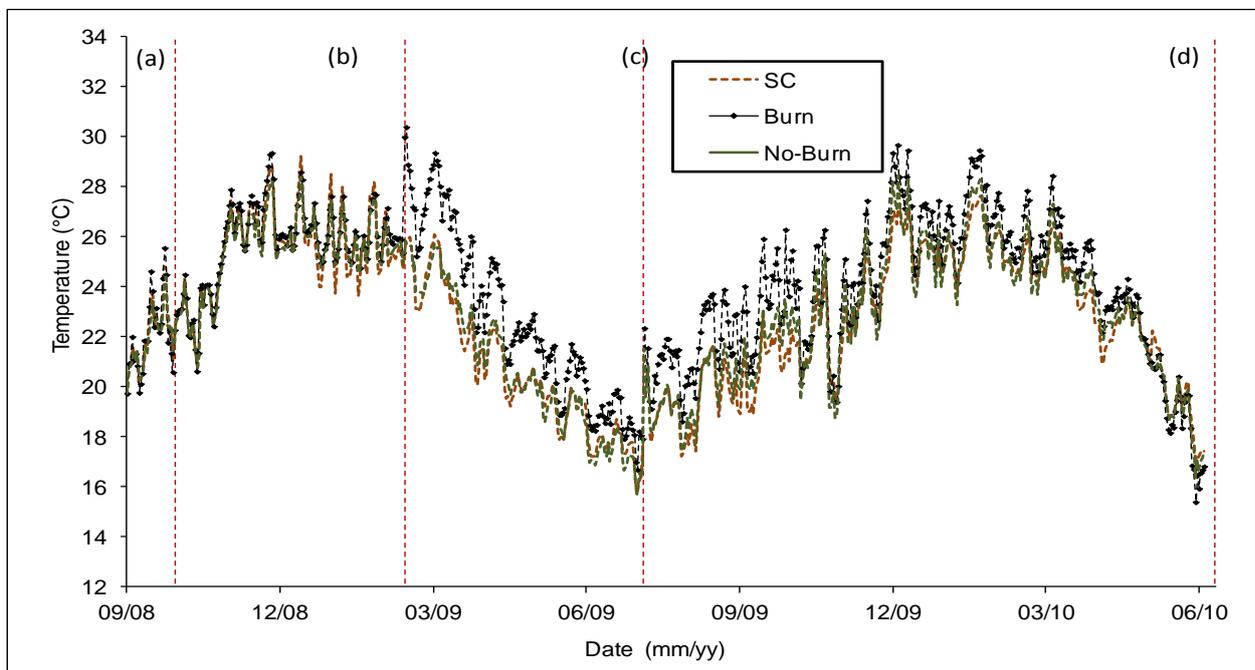
The higher N mineralisation rates, remaining at a net positive after residue retention will enable a more sustained supply of N to the new crop throughout the rotation. Conservation of the residues and the subsequent slower release of N in the No-Burn treatment is a more sustainable practice on this site than residue burning as this allows more time for N that mineralised under residue retention to be taken up and stored in the tree biomass. Losses of N in this low-N system can also be substantially off-set by atmospheric deposition. Despite residue retention being a

more N conservative practice, N is still lost from the system during harvesting and potential deep drainage leaching losses. Further residue management strategies aimed at N retention in the soil and rapid recovery need to be investigated for N conservation on such sites.

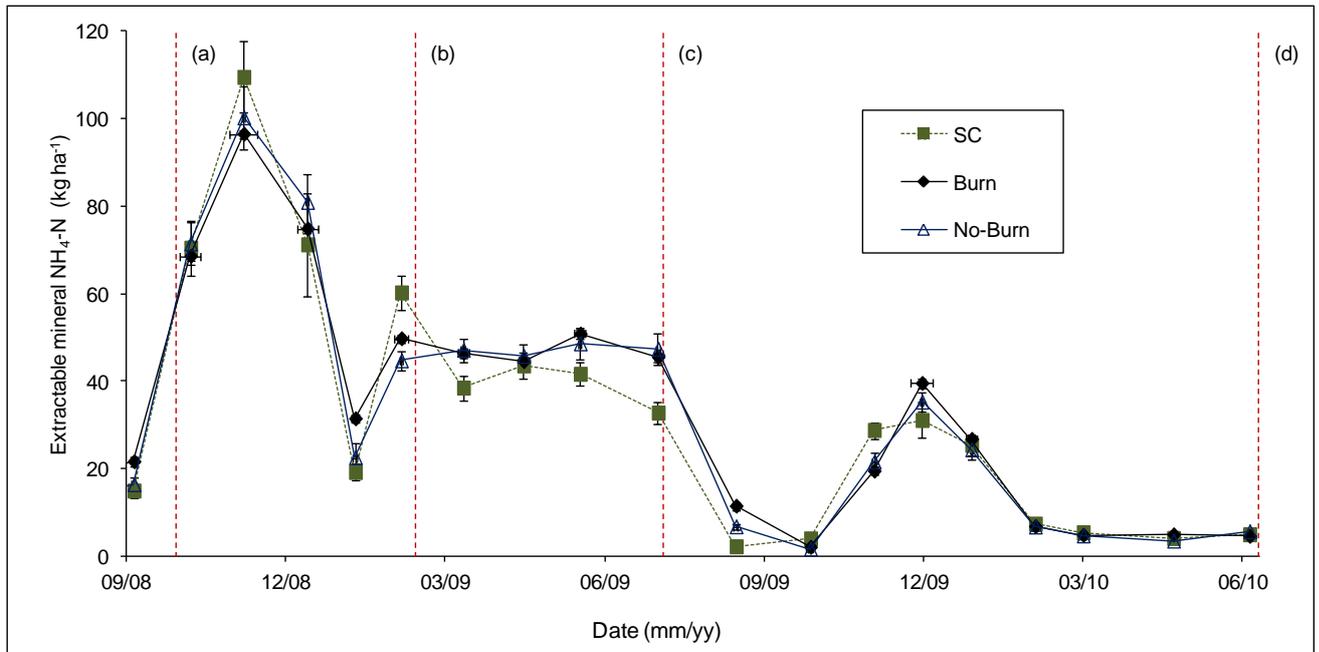
7.7. APPENDIX



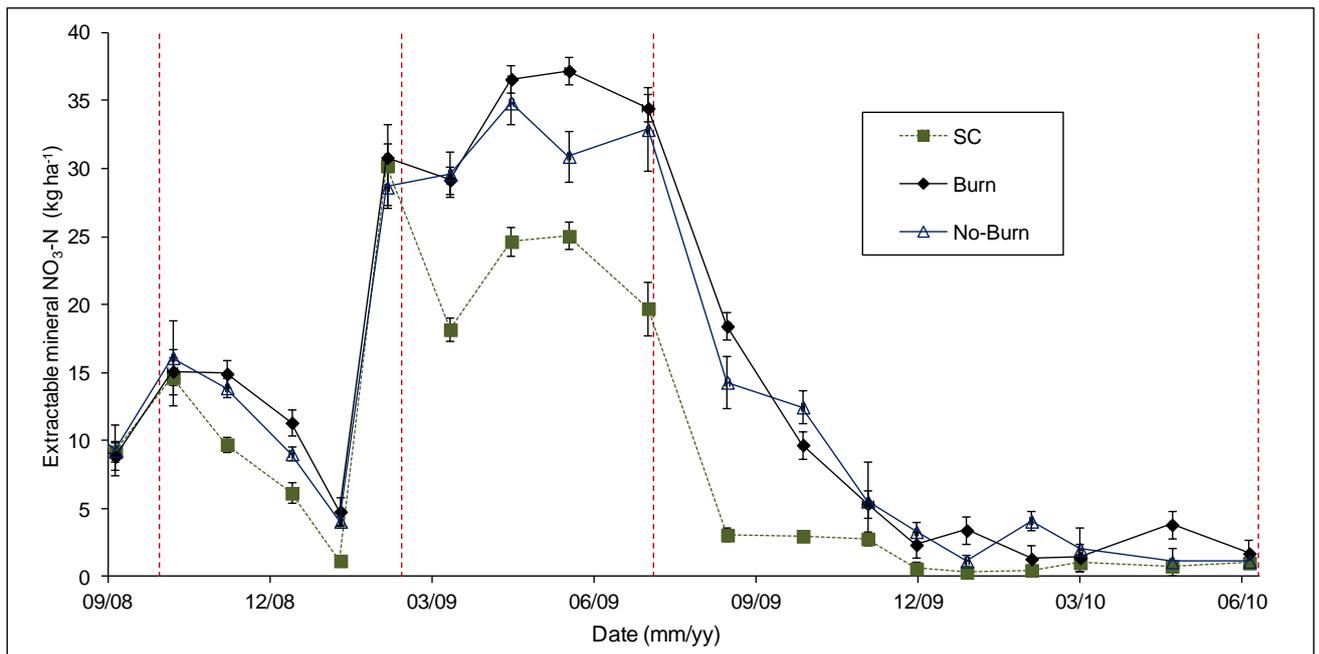
Appendix 7-1: Mean daily air and soil (2.5 cm depth) temperatures for the standing crop (SC), single residue (No-Burn) and burned residue (Burn) treatments. Dashed lines represent (a) clearfelling; (b) residue burning; (c) planting and (d) canopy closure.



Appendix 7-2: Mean daily air and soil (22.5 cm depth) temperatures for the standing crop (SC), single residue (No-Burn) and burned residue (Burn) treatments. Dashed lines represent (a) clearfelling; (b) residue burning; (c) planting and (d) canopy closure.



Appendix 7-3: Mineral NH₄-N in the 0 - 30 cm soil layer. Treatments are standing crop (SC), residue retention (No-Burn) and burned residue (Burn). Dashed lines represent the times of (a) clearfelling; (b) residue burning; (c) planting and (d) canopy closure. I-bars represent least significant differences (LSD_{5%}).



Appendix 7-4: Mineral NO₃-N in the 0 - 30 cm soil layer. Treatments are standing crop (SC), residue retention (No-Burn) and burned residue (Burn). Dashed lines represent the times of (a) clearfelling; (b) residue burning; (c) planting and (d) canopy closure. I-bars represent least significant differences (LSD_{5%}).

Appendix 7-5: Cumulative (kg ha⁻¹) ammonia, nitrate and net nitrogen fluxes in-field cores at 0 - 5, 5 - 15 and 15 - 30 cm soil depths from clearfelling to canopy closure of the new crop.

Flux	Treatment	Depth	Clearfelling to planting			Planting to canopy closure			Clearfelling to canopy closure		
			NO ₃ -N	NH ₄ -N	Net-N	NO ₃ -N	NH ₄ -N	Net-N	NO ₃ -N	NH ₄ -N	Net-N
Net Mineralisation	No-Burn	5	23.81	-3.52	20.29	34.18	-6.60	27.59	58.00	-10.12	47.88
		15	13.56	-0.29	13.29	12.48	-9.68	2.80	26.02	-9.92	16.10
		30	16.55	-12.08	4.49	-5.16	-17.55	-22.71	11.39	-29.61	-18.22
	Burn	5	28.36	1.29	29.63	12.98	-9.76	3.21	41.32	-8.49	32.88
		15	19.30	-5.40	13.90	3.55	-4.94	-1.40	22.82	-10.32	12.51
		30	8.39	-23.19	-14.80	-3.83	-2.31	-6.12	4.58	-25.49	-20.91
	Standing crop	5	16.92	-14.23	2.69	4.56	-0.07	4.49	21.48	-14.30	7.18
		15	8.86	2.88	11.74	-4.17	-1.81	-5.98	4.68	1.08	5.76
		30	3.23	-11.70	-8.47	-8.33	-1.83	-10.15	-5.09	-13.53	-18.62
Deposition minus leaching	No-Burn	5	-11.49	-0.06	-11.49	-29.47	-1.39	-30.88	-40.92	-1.40	-42.37
		15	-4.57	-12.00	-16.59	-16.29	5.13	-11.18	-20.83	-6.88	-27.71
		30	-2.25	1.57	-0.71	-9.62	3.18	-6.42	-11.88	4.71	-7.17
	Burn	5	-5.48	-5.79	-11.30	-10.49	0.98	-9.50	-16.00	-4.80	-20.80
		15	-10.43	-8.12	-18.60	-3.75	-0.41	-4.18	-14.20	-8.58	-22.73
		30	-4.81	2.79	-2.03	-4.51	-8.46	-12.99	-9.37	-5.69	-15.01
	Standing crop	5	4.86	-1.95	2.92	13.09	-0.35	12.73	17.95	-2.30	15.65
		15	3.78	2.15	5.92	5.08	1.24	6.32	8.86	3.39	12.25
		30	10.13	5.57	15.70	4.85	-2.59	2.25	14.97	2.98	17.95
Root Uptake	No-Burn	5	-8.21	0.56	-7.70	-11.62	0.39	-11.22	-19.88	0.91	-18.92
		15	-3.32	2.70	-0.61	-5.77	-8.34	-14.10	-9.10	-5.67	-14.76
		30	-5.72	2.95	-2.77	-0.48	-6.80	-7.31	-6.18	-3.84	-10.02
	Burn	5	-13.27	3.14	-10.08	-13.26	-0.30	-13.55	-26.48	2.85	-23.67
		15	-3.15	2.99	-0.12	-8.14	-6.66	-14.78	-11.25	-3.65	-14.95
		30	-1.15	5.26	4.12	-5.29	-9.03	-14.33	-6.41	-3.76	-10.21
	Standing crop	5	19.31	-11.92	7.39	20.62	4.69	25.31	39.93	-7.23	32.70
		15	9.75	19.59	29.34	6.76	8.66	15.42	16.51	28.24	44.76
		30	13.64	12.73	26.37	6.40	9.04	15.44	20.05	21.76	41.81

Appendix 7-6: Cumulative (kg ha⁻¹) ammonia and nitrate fluxes in-field cores at 0 - 30 cm soil depth from clearfelling to canopy closure of the new crop.

Flux	Treatment	Clearfelling to planting		Planting to canopy closure		Clearfelling to canopy closure	
		NO ₃ -N	NH ₄ -N	NO ₃ -N	NH ₄ -N	NO ₃ -N	NH ₄ -N
Net Mineralisation	No-Burn	53.9	-15.9	41.5	-33.8	95.4	-49.6
	Burn	56.1	-27.3	12.7	-17.0	68.7	-44.3
	Standing crop	29.0	-23.0	-7.9	-3.7	21.1	-26.7
Deposition minus leaching	No-Burn	-18.3	-10.5	-55.4	6.9	-73.6	-3.6
	Burn	-20.7	-11.1	-18.8	-7.9	-39.6	-19.1
	Standing crop	18.8	5.8	23.0	-1.7	41.8	4.1
Root Uptake	No-Burn	17.2	+6.2	17.9	14.8	35.2	8.6
	Burn	17.6	+11.4	26.7	16.0	44.1	4.6
	Standing crop	42.7	20.4	33.8	22.4	76.5	42.8

Appendix 7-7: Ammonia fluxes (kg ha⁻¹) in-field cores at 0 - 30 cm soil depth for each measured date.

Date	Net Mineralisation			Deposition minus leaching			Root Uptake		
	Standing crop	No-Burn	Burn	Standing crop	No-Burn	Burn	Standing crop	No-Burn	Burn
13/02/08	-4.04	-0.61	-0.10	-6.95	-3.81	-2.97	-6.20	-2.49	0.40
27/03/08	-4.09	-0.65	-2.19	-5.72	-2.94	-4.10	1.27	-2.29	0.17
06/05/08	3.83	3.96	-1.33	7.12	7.89	3.33	3.02	-5.54	-3.22
03/10/08	3.65	5.94	8.84	2.51	5.13	8.17	0.99	9.52	13.80
04/11/08	50.78	51.86	51.86	57.53	58.79	58.79	55.48	50.97	50.97
04/12/08	32.14	27.73	27.73	33.37	29.00	29.00	39.02	28.34	28.34
09/01/09	-26.28	-15.28	-15.28	-25.21	-15.76	-15.76	-38.25	-20.48	-20.48
05/02/09	-48.93	-49.92	-49.92	-47.57	-54.48	-54.48	-51.99	-50.85	-50.85
03/03/09	34.58	18.41	18.41	34.05	17.16	17.16	41.01	20.21	20.21
07/04/09	-19.43	1.24	-2.86	-19.23	0.23	-3.86	-21.74	2.39	-3.33
11/05/09	4.86	-0.02	-1.82	7.38	-0.88	-4.46	5.07	-1.38	-1.90
12/06/09	6.29	4.57	3.57	4.36	0.87	3.79	-1.92	2.86	6.30
26/07/09	-6.28	-2.60	-7.12	-4.43	-2.48	-9.82	-8.86	-1.25	-5.31
08/09/09	-27.53	-38.10	-35.08	-26.07	-39.13	-35.38	-30.54	-40.54	-34.01
20/10/09	1.71	-5.61	-8.75	6.15	-3.64	-10.23	1.80	-5.11	-9.40
25/11/09	22.18	23.03	17.51	23.44	22.88	18.88	24.78	19.92	17.41
22/12/09	18.14	13.23	34.33	11.83	16.09	25.19	2.28	13.67	20.05
19/01/10	-0.70	-6.60	-4.44	-4.22	-5.43	-1.30	-5.70	-10.91	-12.79
24/02/10	-17.32	-18.83	-20.39	-17.35	-18.36	-20.16	17.99	17.62	19.94
23/03/10	-0.49	-2.08	-0.41	-0.45	-0.86	-1.73	1.97	2.03	1.98
13/05/10	-0.72	-1.05	0.29	-0.55	-0.11	-0.16	1.30	1.18	-0.10
25/06/10	1.02	2.18	-0.07	1.82	1.66	-0.02	-0.85	-2.14	0.34

Appendix 7-8: Nitrate fluxes (kg ha^{-1}) in-field cores at 0 - 30 cm soil depth for each measured date.

Date	Net Mineralisation			Deposition minus leaching			Root Uptake		
	<i>Standing crop</i>	<i>No-Burn</i>	<i>Burn</i>	<i>Standing crop</i>	<i>No-Burn</i>	<i>Burn</i>	<i>Standing crop</i>	<i>No-Burn</i>	<i>Burn</i>
13/02/08	12.76	11.02	13.44	22.12	22.08	22.51	15.48	13.15	16.07
27/03/08	3.59	3.07	3.89	5.41	4.77	6.89	0.22	-7.75	4.99
06/05/08	-19.77	-20.45	-19.96	-18.57	-17.35	-18.04	-27.77	-16.15	-22.54
03/10/08	-11.07	-11.40	-10.76	-10.91	-11.11	-10.59	-11.12	-16.16	-12.07
04/11/08	7.33	7.15	7.15	11.16	5.09	5.09	5.45	6.53	6.53
04/12/08	-0.66	3.66	3.66	1.11	2.79	2.79	-4.93	-1.19	-1.19
09/01/09	0.95	-6.43	-6.43	1.01	1.80	1.80	-3.57	-4.22	-4.22
05/02/09	-4.65	-4.26	-4.26	-3.25	-7.77	-7.77	-4.97	-5.75	-5.75
03/03/09	31.57	37.67	37.67	30.36	34.64	34.64	29.04	25.31	25.31
07/04/09	-10.15	8.38	0.79	-5.70	4.77	-3.11	-12.04	0.96	-1.66
11/05/09	13.05	10.62	21.18	15.37	4.05	7.58	6.48	5.23	7.43
12/06/09	4.71	0.67	3.60	7.70	-2.86	2.12	0.40	-3.93	0.60
26/07/09	-5.81	3.62	-0.15	1.18	-1.81	-2.80	-5.32	1.98	-2.74
08/09/09	-14.21	-6.63	-10.20	-9.69	-16.89	-10.34	-16.66	-18.58	-16.03
20/10/09	-0.56	3.66	-1.07	3.10	-5.72	-7.69	-0.08	-1.82	-8.74
25/11/09	0.92	21.82	7.57	2.14	-7.47	-5.19	-0.20	-6.92	-4.33
22/12/09	0.40	1.63	0.06	1.97	-0.61	0.75	-2.20	-2.24	-3.01
19/01/10	1.69	3.83	3.78	2.21	2.40	3.36	-0.27	-2.16	1.06
24/02/10	0.50	6.23	3.90	1.92	3.49	3.85	-0.15	-2.93	2.10
23/03/10	1.81	4.93	1.71	4.64	1.64	3.15	-0.55	2.06	-0.08
13/05/10	0.90	4.65	5.24	7.26	6.55	4.33	0.25	0.92	-2.44
25/06/10	0.60	1.38	1.70	1.52	2.74	1.74	-0.23	-0.03	2.12

CHAPTER 8: CONTRIBUTION OF GRAVITATIONAL LEACHING TO INTER-ROTATION NUTRIENT FLUXES⁵

8.1. ABSTRACT

Little is known about the fate and sustainability of nutrients during the intensively managed inter-rotation period of highly productive short rotation clonal *Eucalyptus* grown on the sandy soils of the Zululand coastal ecosystem in South Africa. A study was designed to compare gravitational nutrient leaching within the top meter of soil in an undisturbed standing crop of mature clonal *Eucalyptus* with adjacent clearfelled areas subjected either to prescribed residue burning or to residue retention (broadcasting).

Leaching in the undisturbed crop decrease strongly with depth as the mature trees utilised most of the nutrients and water resulting in small losses beyond 100 cm depth.

Clearfelling overall induced a short-lived spike in nitrogen and cation leaching compared with the low leaching losses in the undisturbed crop. The increased felled treatment leaching beyond 100 cm was only significant during the period after clearfelling through the temporary unplanted period to six months after the new crop was planted. Soil water and nutrient uptake with new crop growth gradually reduced soil moisture content and soil solution nutrient concentrations reducing leaching in the felled plots to levels similar to the undisturbed crop.

Residue burning increased the pools of Ca and Mg in the soil, but did not significantly alter their leaching. Although nitrate leaching increased during the first few months after burning and potassium leaching decreased, these changes were small compared to the effect of clearfelling on leaching. Losses of nutrients associated with biomass loss during clearfelling and residue burning in addition to clearfelling induced leaching loss may increase the risk of nutrient depletion at our site. These high nutrient losses, relative to small pools sizes at this site may act as a severe limitation to future tree productivity. The duration of increased post-felling leaching losses may however be reduced through nutrient conservation strategies, such as early re-establishment and fertilisation. This study may indicate similar soils afforested across the Zululand region to be at risk of nutrient depletion if not managed to conserve nutrients during the clearfelling and early establishment phase.

⁵ Submitted for publication

8.2. INTRODUCTION

The residues and litter layer remaining after clearfelling a plantation forest contains large quantities of nutrients, their retention viewed as necessary to ensure present and future nutrient supply (Carlyle et al. 1998b; Piatek and Allen 1999). While a number of residue management options are available for *Eucalyptus* plantations, including retention, displacement and removal (du Toit et al. 2004; Smith and du Toit 2005) residue burning after clearfelling is often practiced as a means of creating a clean site that has a lower wildfire risk, and is easier to plant and manage. However, concerns are being raised within the South African forestry industry about the impacts of residue burning on tree productivity in current and future rotations. Large losses of plant nutrients associated with residue burning, particularly nitrogen (N) and carbon, can be detrimental to nutrient poor ecosystems (Morris 1986; du Toit and Scholes 2002; Nadel 2005), while beneficially reducing N in polluted ecosystems (Gomez-Rey et al. 2007). Furthermore, wind and water displacement of ash from burning followed by increased soil erosion and leaching due to loss of soil surface cover and increased soil hydrophobicity, may exacerbate nutrient losses and eventually deplete soil nutrients (Pilkington et al. 2007). Residue retention is sometimes viewed in a negative light as it increases the fuel-load of a site, increasing the risk of wildfire. This risk is greatest on cooler, drier sites where residues persist through slow breakdown. The fallow inter-rotation period after clearfelling may also be a time of increased nutrient leaching or loss with drainage, because large quantities of nutrients are released through residue leaching and more rapid residue decomposition, along with increased soil moisture and a lack of plant uptake (Fisher and Binkley 2000; Goncalves et al. 2007; O'Hehir and Nambiar 2010). Losses through leaching or displacement of nutrients may be greater after burning as the additional mobile nutrients released in the ash add to nutrients released during decomposition of the old crop root system (Powers et al. 2005).

Residue and litter stores of carbon and nutrients are more important on sandy soils that are characteristically low in clay, organic carbon and fertility (Rosenstrauch 1938; Noble et al. 2005). The retention, displacement or loss of residues during the inter-rotation phase can significantly impact the nutrients dynamics of a site (O'Hehir and Nambiar 2010). Impacts of such losses can be large on sandy soils where soil nutrient pools are small within the region where fine-root density is greatest (top 50 cm of the soil (Smethurst and Nambiar 1990; Carlyle and Nambiar 2001)) and the ability of soil to retain nutrients is low. The sandy soils of the

Zululand coastal ecosystem in South Africa are characterised by low clay and organic carbon content, low nutrient and water storage capacity and high drainage rates (Hartemink and Hutting 2005). These are characteristics that may ultimately limit nutrient availability and consequently reduce stand productivity. Such soils have limited ability to buffer chemical, physical and organic (biological) changes and are at risk of becoming degraded. In addition, the Zululand coast is also characterised by high temperatures and rainfall and low vapour pressure deficit that results in rapid growth of trees, short rotation lengths (six to seven years), a high nutrient demand and rapid litter and residue decomposition. This may lead to accelerated loss of organic matter and nutrients, as nutrients that are not taken up after mobilisation may be displaced through leaching to greater depths in the soil profile or completely lost to deep drainage.

This study aimed at comparing displacement of nutrients due to leaching with gravitational soil water drainage within the top meter of the soil between a standing crop of mature clonal *Eucalyptus* and adjacent clearfelled areas subjected to burning and residue retention (broadcasting slash). This study draws from a number of component studies within a larger experiment that aimed to determine the impact of site management on long-term sustainable productivity of clonal *Eucalyptus*, with respect to natural and management induced nutrient losses and gains.

8.3. MATERIALS AND METHODS

Materials and methods for this chapter that are common to other chapters are given in CHAPTER 3.

8.4. RESULTS

8.4.1. Rainfall and drainage volumes

Water collected as rainfall and canopy drainage (throughfall and stemflow) (**Appendix 8-1**) with predicted soil water drainage for each treatment and each depth (CHAPTER 4) Data was cumulated across the time periods felling to burning, burning to six months after planting (i.e. stand age 0.5 years), from stand age six months to one year (canopy closure) and from stand age

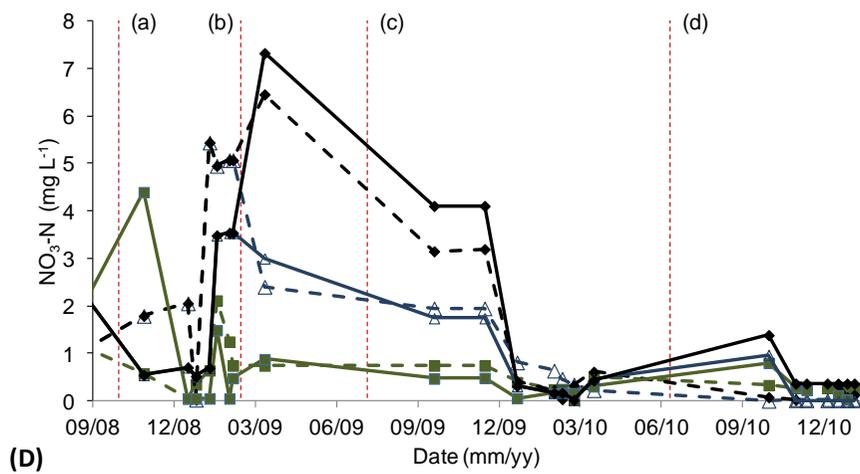
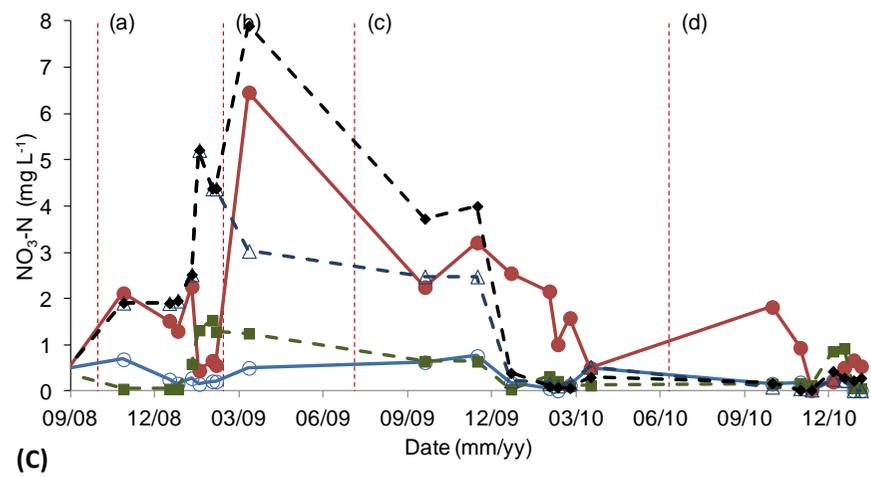
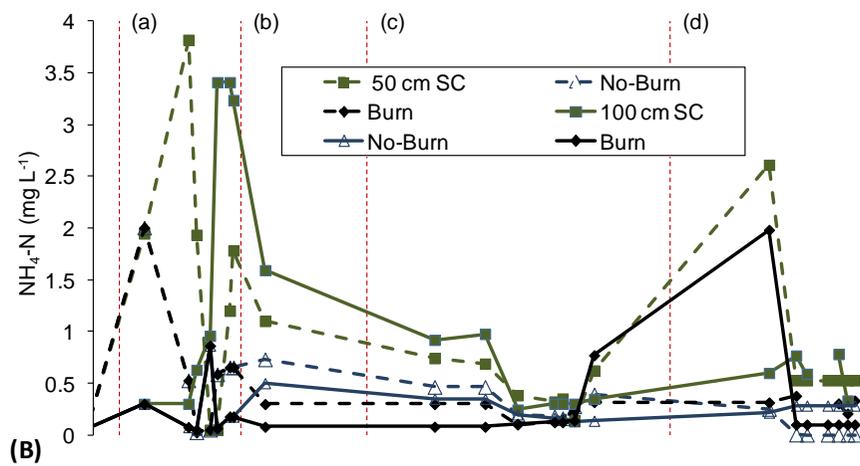
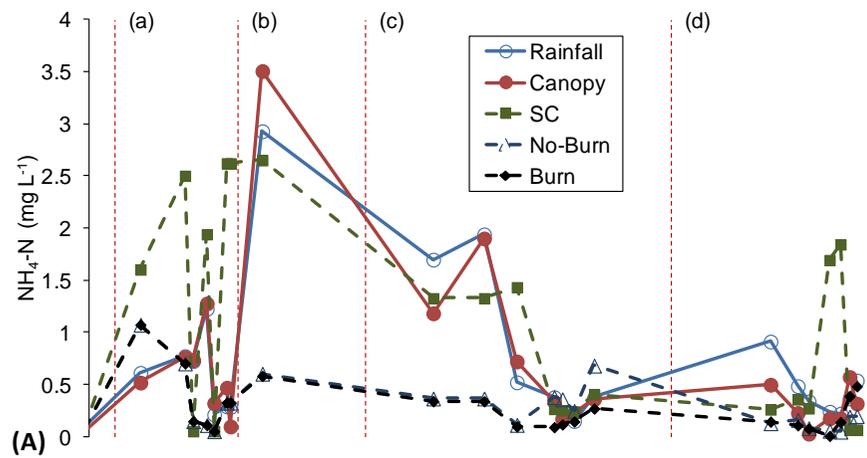
one year to 18 months (1.5 years). The duration of these 4 monitoring periods were 4, 11, 6 and 6 months respectively, thus totalling to 27 months for the project. Only one drainage event occurred between burning and planting, which has been included as part of the burning to 0.5 years cumulative total. Due to interception under the tree canopy, canopy drainage was slightly lower than rainfall. Total drainage predicted at each depth was initially lower in the standing crop treatment at each depth, but as tree growth commenced drainage in the felled treatments became similar to drainage in the standing crop treatment. The cumulative drainage in the standing crop treatment was considerably lower at depth than that in the felled treatments between felling and canopy closure.

8.4.2. Nutrient concentrations

All 15 cm standing crop treatment N concentrations tended to follow canopy drainage N concentrations (**Figure 8.1**). Soil solution NH_4^+ concentrations (**Figure 8.1A and B**) were generally higher in the standing crop treatment than in the felled treatments at all depths, for the period between felling and canopy closure of the new crop (1 year old). Soil solution NH_4^+ concentrations were similar between the Burn and No-Burn treatments at all depths. Concentrations decreased with depth across all treatments. Soil solution ammonium and nitrate, as well as volume weighted NH_4^+ and NO_3^- in rainfall and canopy drainage had particularly low concentrations during Jan-Feb 2010.

NO_3^- concentrations tended to be lower in the standing crop than the felled treatments at all depths (**Figure 8.1C and D**). A rise in NO_3^- concentrations at all depths after felling was maintained until around six months after felling. Concentrations of NO_3^- were lowest at 100 cm depth in the Burn and No-Burn treatments. NO_3^- was higher in the Burn treatment between planting and 0.5 years after planting at all depths. After this, NO_3^- concentrations were similar in all treatments.

Organic-N concentrations (**Figure 8.1E and F**) generally decreased with depth, with concentrations in the No-Burn treatment tending to be higher on average than the Burn treatment at all depths. Organic-N levels were initially low in the felled treatments around the time of planting, but increased to levels similar to the standing crop treatment towards the end of the study.



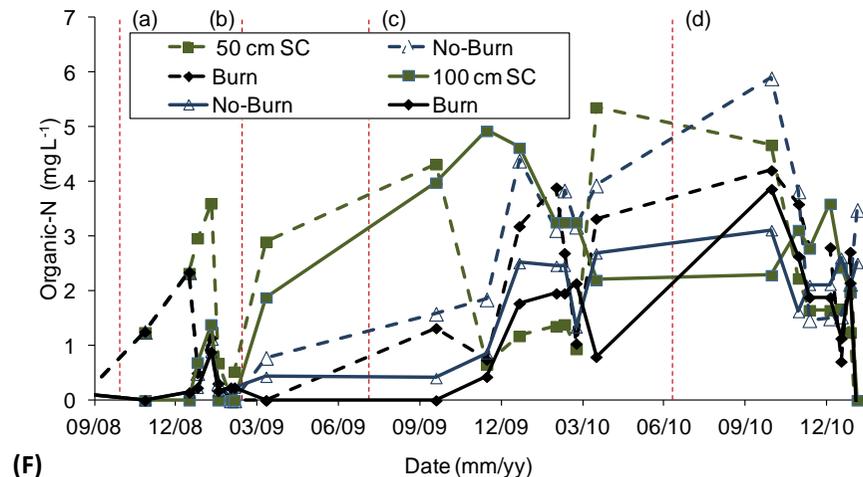
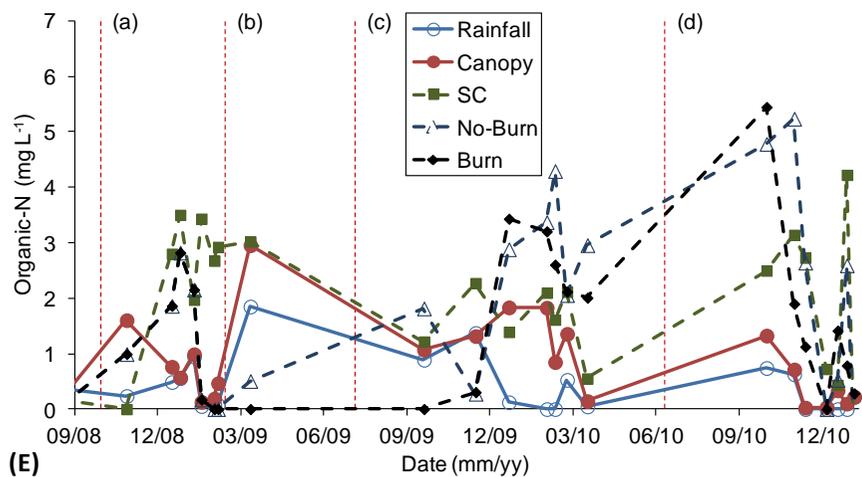


Figure 8.1(A-F): Soil solution and volume weighted rainfall and canopy drainage NH₄-N (a), NO₃-N (c) and organic-N (e) concentrations measured at 15 cm depth and NH₄-N (b), NO₃-N (d) and organic-N (f) concentrations measured at 50 cm and 100 cm depths for felled and standing crop (SC) areas of the study site. Dashed vertical lines represent (a) felling; (b) residue burning; (c) planting and (d) canopy closure.

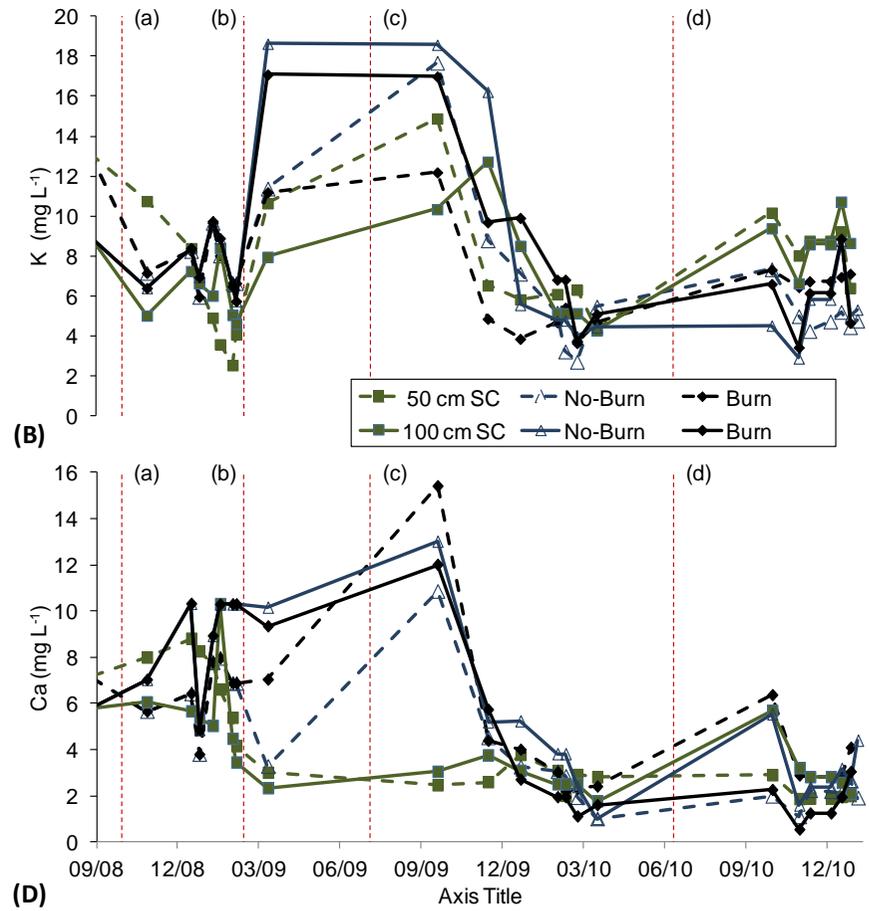
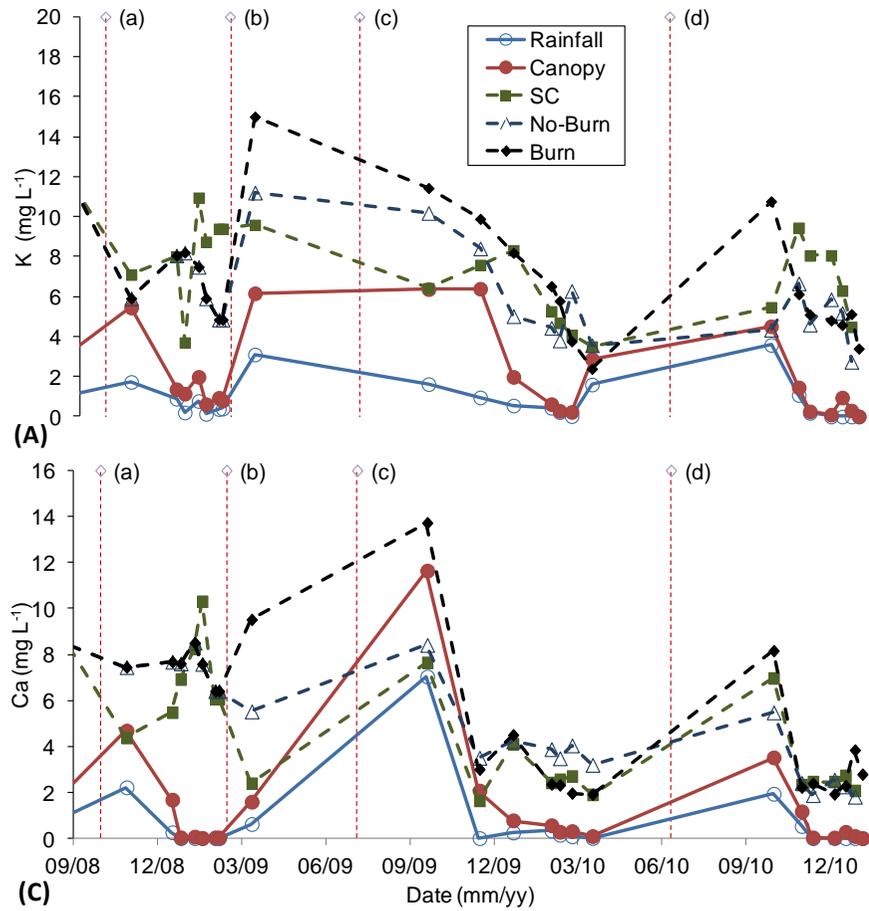
Rainfall was enriched with base cations K^+ , Ca^{2+} , Mg^{2+} and Na^+ in passing through the tree canopy (**Figure 8.2**), but was markedly more concentrated in the soil solution at all depths than in canopy drainage. Soil solution base cation concentrations were altered after felling and again after residue burning. With few exceptions, cation concentrations were two to eight-fold higher in the soil solution at all depths than in the canopy drainage and rainfall.

Soil solution K^+ tended to be lower in the standing crop treatment than in the felled treatments at each depth (**Figure 8.2A and B**). On average, differences in K^+ concentration between the standing crop and Burn treatment were small at the 15 cm depth, while K^+ concentration increased with depth for the felled treatments. Concentrations of K^+ were, on average, higher in the Burn than in the No-Burn treatment at 15 cm depth, becoming lower in the Burn at 50 and 100 cm depth. K^+ concentrations tended to decrease from planting towards the end of the study period in all treatments, decreasing in the felled treatments to below the standing crop treatment concentrations.

Soil solution Ca^{2+} concentrations (**Figure 8.2C and D**) were mostly lower in the standing crop than the felled treatments across all depths. Calcium concentration decreased with depth in the standing crop treatment, but increased with depth in both felled treatments. The concentration of Ca^{2+} peaked at 15 and 50 cm depths after burning and remained higher than the No-Burn until six months after planting. Concentrations in the felled treatments became similar to those in the standing crop treatment from around six months after planting.

Soil solution Mg^{2+} concentration (**Figure 8.2E**) increased at 15 cm depth in the Burn treatment after burning, while remaining similar between the standing crop and No-Burn treatments at 15 cm depth. Soil solution Mg^{2+} concentration increased with depth in the felled treatments (**Figure 8.2F**). These concentrations peaked at 100 cm depth in both felled treatments prior to burning, but gradually reduced to lower than the standing crop treatment a few months after planting.

Soil solution Na^+ concentrations decreased relative to the standing crop treatment from a few months after felling (**Figure 8.2G and H**). Concentration of Na^+ increased with depth in all treatments, particularly in the standing crop treatment. Concentrations were similar in the Burn and No-Burn treatments throughout.



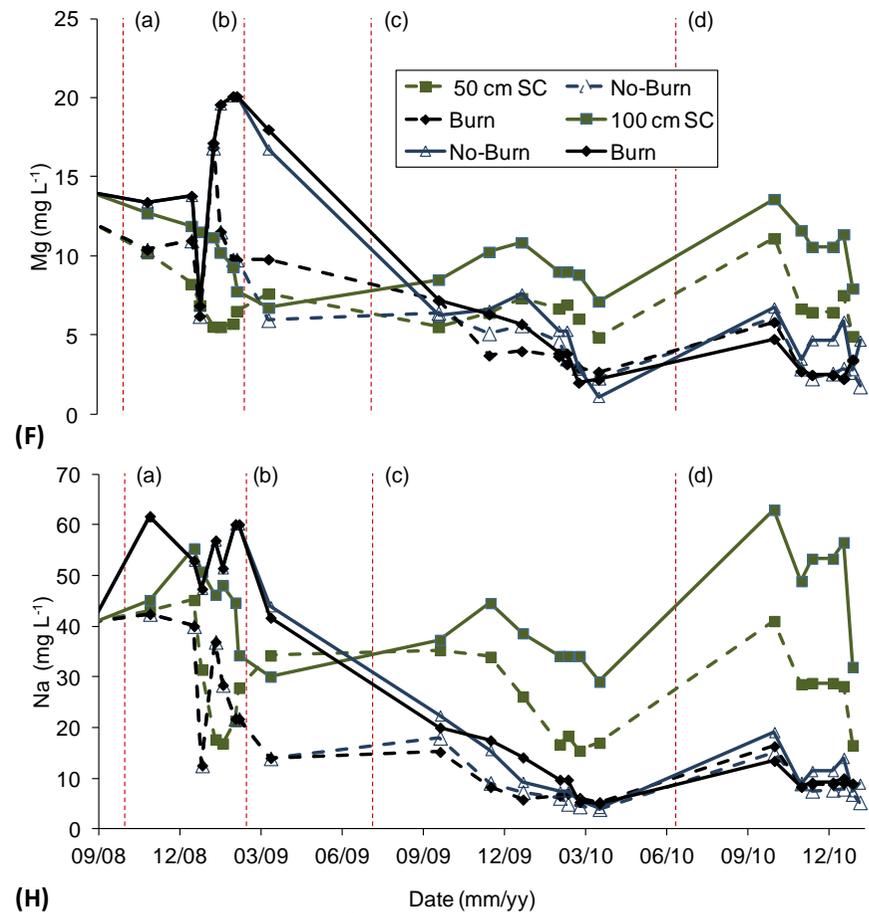
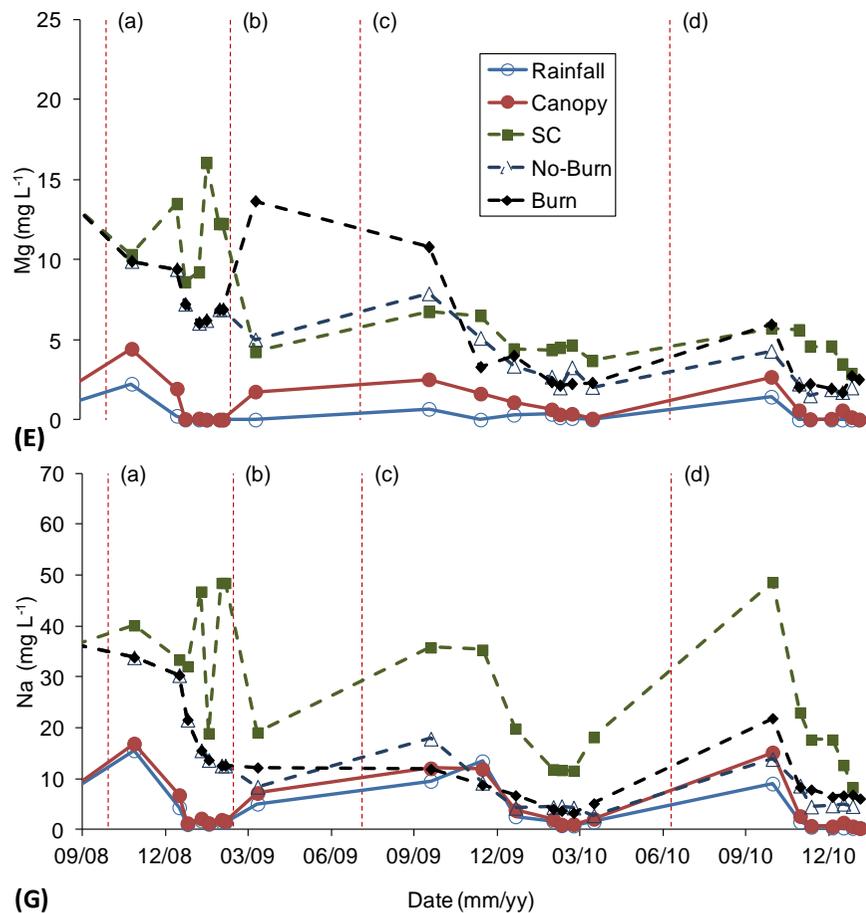


Figure 8.2(A-H): Soil solution and volume weighted rainfall and canopy drainage K^+ (a), Ca^{2+} (c), Mg^{2+} (e) and Na^+ (g) concentrations recorded at 15 cm depth and K^+ (b), Ca^{2+} (d), Mg^{2+} (f) and Na^+ (h) concentrations recorded at 50 and 100 cm depths for felled and standing crop (SC) areas of the study site. Dashed vertical lines represent (a) felling; (b) residue burning; (c) planting and (d) canopy closure.

8.4.3. Soil solution mass flux

NH₄-N leaching at 15 cm in the standing crop treatment was more than that added with canopy drainage (**Appendix 8-2a**), decreasing strongly with depth (**Appendix 8-2:a**). In contrast, less NH₄-N leached to 15 cm depth in the felled treatments than added by rainfall. Significantly less NH₄-N leached to 15 cm depth under the felled treatments than under the standing crop treatment. From felling to six months after planting more NH₄-N leached to 50 cm in both felled treatments than did in the standing crop treatment. This larger NH₄-N leaching persisted to canopy closure at 50 and 100 cm in the No-Burn treatment. Fluxes thereafter became similar in felled treatments at 15 and 50 cm depth and across all treatments at 100 cm.

Leaching of NO₃-N (**Appendix 8-2b**) at the plates contrasted NH₄-N leaching and NO₃-N leaching values were much larger in magnitude than NH₄-N leaching (**Appendix 8-2a**). Canopy drainage NO₃-N was similar to 15 cm standing crop leaching, and this was significantly lower than leaching under the felled treatments. Far more NO₃-N was leached in the felled treatments than was added with rainfall. Leaching of NO₃-N tended to decrease with depth across all treatments. Nitrate leaching in burnt and unburnt plots were statistically similar across all times and depths with one notable exception: the values at all depths for the period between burning and six months after planting, where burnt plots showed significantly greater NO₃-N leaching. Nitrate leaching also become similar between treatments at each depth from around six months after planting.

More organic-N was leached past 15 cm than added by canopy drainage to the standing crop and rainfall in the felled treatments. Organic-N decreased with depth across all treatments. It was significantly less at depth in the standing crop treatment than in both felled treatments (**Appendix 8-2c**). Organic-N leaching was initially lower in the felled treatments than the standing crop treatment at 15 cm depth. Significantly less organic-N was leached to 50 and 100 cm in the Burn treatment than the No-Burn treatment after burning and up to six months after planting.

The quantity of base cations K (**Appendix 8-3a**), Ca (**Appendix 8-3b**) and Mg (**Appendix 8-3c**) leached to 15 cm in the standing crop treatment was more than that added by canopy drainage, and more at 15 cm in the felled treatments than that added by rainfall. Canopy cation drainage was larger than rainfall cation addition. Felled treatments recorded highest levels of potassium leaching beyond 100 cm, most of the significant differences relative to the standing crop

occurring during the period from burning to 6 months. Leaching of K decreased strongly with depth in the standing crop treatment, in contrast to the felled treatments.

More Ca was leached at all depths in the felled treatments than the standing crop treatment from felling to six months after planting. Similar quantities of Ca were leached in the felled treatments for most periods and depths, with the total for burnt treatments being slightly greater than the unburnt treatment. Leaching of Ca took place in similar quantities at each depth in the felled treatments, but in strong contrast, diminished with depth under the standing crop treatment.

Magnesium leached in the felled treatments was only higher than that leached in the standing crop treatment at 15 cm depth between burning and six months after planting, but lower for all other periods. Magnesium leaching strongly decreased with depth in the standing crop treatment. Magnesium leaching increased with depth in the felled treatments with large increases during the burning to six months period. Leaching fluxes in Burn and No-Burn treatments at 50 and 100 cm depth were similar.

Nitrogen speciation changed from being largely comprised of NH_4^+ ions in the standing crop treatment to NO_3^- ions in the felled treatments between felling and six months after planting (**Table 8-1**). This speciation in the felled treatments reverted back to become similar to the standing crop treatment after the trees reached six months of age. In total, a significantly greater quantity of N was leached beyond 100 cm from the felled treatments than from the standing crop treatment (**Table 8-1**). Burning resulted in more NO_3^- ions and marginally less NH_4^+ ions leaching beyond 100 cm in than in the No-Burn treatment. Significantly more N was leached beyond 100 cm in the Burn treatment than in the No-Burn treatment between felling and six months after planting (**Table 8-1**).

Leaching of K^+ , Ca^{2+} , Mg^{2+} and Na^+ ions was significantly larger in the felled treatments than in the standing crop treatment at each depth between felling and six months after planting (**Table 8-1**). Significantly more K^+ ions and less Ca^{2+} ions were leached beyond 100 cm in the No-Burn and than in the No-Burn treatment between felling and six months after planting (**Table 8-1**). These differences became less apparent after the trees were six months of age, however, differences in Ca^{2+} and Na^+ ion leaching remained significant. Relative to the Standing crop, Ca^{2+} leaching was marginally higher and Na^+ lower in the felled treatments from six to 18 months of tree age.

Table 8-1: Leaching (kg ha^{-1}) beyond 100 cm depth for felled and standing crop (SC) areas of the study site over the felling to 0.5 years after planting period (15-months), and the 0.5 to 1.5 years after planting period (12-months, after canopy closure). Different a, b, c superscripts denote significance difference between treatments at $\text{LSD}_{5\%}$

	Standing crop	Burn	No-Burn	Treatment F. probability	Standing crop	Burn	No-Burn	Treatment F. probability
	Felling to 0.5 years after planting				0.5 to 1.5 years after planting			
Organic-N	3.7a	3.5a	7.3b	0.004	4.1a	5.3a	7.0a	0.044
NH₄-N	1.6a	1.1b	2.4c	<.001	0.7a	0.7a	0.3a	0.140
NO₃-N	0.6a	22.4a	11.6b	0.026	0.3a	0.65a	0.8a	0.002
K	3.2a	28.3b	32.2c	0.004	4.2a	4.6a	4.0a	0.002
Ca	3.2a	45.6b	39.4b	<.001	2.6a	4.0a	4.7a	<.001
Mg	11.8a	88.5b	85.6b	0.001	15.2a	9.0a	10.1a	0.507
Na	25.2a	129.9b	126.4b	<.001	36.5a	12.4b	11.9b	<.001

Accumulation of nutrients into the above ground biomass and nutrient storage in the litter layer remained large in the standing crop treatment (**Table 8-2**). Nutrient uptake into the trees of the standing crop treatment underwent rapid turnover through high rates of litterfall. Litterfall had not commenced by canopy closure in the felled treatments. Residue burning resulted in large losses of nutrients (particularly N) from the residues (**Table 8-2**). Very little ash was present in the Burn treatment at canopy closure (visual observation), all nutrients remaining in the ash were assumed to have entered the soil with rainfall or lost through wind erosion. The No-Burn treatment had large aboveground N, Ca and Mg nutrient pools at the end of the study (**Table 8-2**). Nitrogen and Ca above-ground biomass pools at one year after planting in the No-Burn plots were 60 and 80% of the standing crop pools respectively. Residue retention also maintained large amounts of P and Mg in the above-ground biomass relative to the standing crop treatment. Burn treatment N and Ca above-ground biomass pools were 20% and 7% of the standing crop pools respectively at canopy closure, consisting of live tree biomass alone. Limited K retention in the residues of the No-Burn treatment yielded similar K above-ground biomass pools to the Burn treatment (**Table 8-2**).

Table 8-2: Above -ground nutrient pools at felling and canopy closure and litterfall, burning loss and tree biomass nutrient content occurring between felling and canopy closure of the new crop.

Pool or Flux	Treatment	N	P	kg ha ⁻¹			
				K	Ca	Mg	Na
Pools at felling	Standing crop	403.1	43.9	194.2	683.5	120.5	27.0
	Felled plots	294.7	25.7	112.2	605.4	103.9	18.9
Standing crop	Litter fall	116.7	4.6	22.3	119.0	29.5	12.2
	Tree biomass	10.6	1.8	9.5	24.7	4.1	1.1
Burn	Burning loss	121.2	2.8	3.3	34.7	21.6	1.2
	Tree biomass	75.1	5.0	66.0	50.9	16.8	2.7
No-Burn	Tree biomass	64.9	4.3	57.6	42.5	14.8	2.4
Pools at Canopy closure	Standing crop	381.9	46.2	203.1	688.4	114.2	28.3
	Burn*	75.1	5.0	66.0	50.9	16.8	2.7
	No-Burn	241.0	12.0	68.9	553.2	49.7	5.9

*Surface ash was washed into the soil and no longer present and remnants were too small and variable for accurate sampling

8.4.4. Soil nutrient pools and fluxes

Concentrations of total N, extractable P and exchangeable base cations decreased with depth and exchangeable Na concentration was highest at each depth in the standing crop plots. A reduction in soil total N pool sizes (**Table 8-3**) occurred after felling at all depths, the total N pool size in the felled treatments being significantly smaller than in the standing crop treatment. The total N pool size in the pre-felling assessment seemed lower than later on in the standing crop treatment. Exchangeable K pools were significantly larger in the felled treatments than the standing crop treatment, accounted for by large differences in the surface to 15 cm soil layer. These pools were significantly larger in the No-Burn treatment than the Burn treatment. Potassium levels in No-Burn treatment at the end of the study were however similar to those recorded at the start. Calcium pool sizes were only significantly larger in the Burn treatment compared to the No-Burn and standing crop treatments. The larger total Ca pool in the Burn treatment than the other two treatments was solely due to a very large pool in the surface 15 cm layer. Differences between Burn and No-Burn were not significant. Pre-felling Ca and Mg levels remained relatively constant in the standing crop treatment. The Na pool was significantly smaller in the felled treatments at the end of the study than in the standing crop treatment. The reduced Na pool size in the felled treatments compared with the standing crop treatments was due to a pronounced decrease in Na concentrations at depth, particularly in the 50 to 100 cm layer. Sodium concentrations in the 50 to 100 cm layer of the No-Burn and the Burn treatments were around 9% and 31% of standing crop treatment values respectively at the end of the study.

Table 8-3: Soil nutrient pools at incremental depths between 0 and 100 cm soil depth at the start and end of the study (kg ha^{-1}). Significant differences ($\text{LSD}_{5\%}$) are shown between total pool sizes (bold text) at the end of the study.

	Depth range (cm)	Bray 2		Salt-Exchangeable base cation				
		Total N	extractable P	K	Ca	Mg	Na	
Pre-Felling	0-15	1106.2	6.6	32.3	246.5	64.7	22.2	
	15-50	1228.5	8.5	64.0	251.6	80.5	37.7	
	50-100	2133.3	17.2	62.6	235.1	139.3	122.6	
	0-100	4468.0	32.3	158.9	733.3	284.6	182.4	
Post canopy closure	Standing crop	0-15	1248.0	8.4	34.6	350.0	79.5	29.8
		15-50	2760.0	3.4	16.7	152.0	78.8	90.4
		50-100	2653.0	9.4	14.2	249.0	143.6	95.6
		0-100	6661^a	21.2^a	65.5^a	751^a	301.9^a	215.8^a
	Burn	0-15	1130.0	11.6	67.5	940.0	112.1	28.8
		15-50	2245.0	4.1	8.3	247.0	82.2	63.6
		50-100	1732.0	8.0	17.3	224.0	72.4	30.0
		0-100	5107^b	23.7^a	93.1^b	1411^b	266.7^a	122.4^b
	No-Burn	0-15	927.0	9.7	105.3	532.0	98.3	24.4
		15-50	1778.0	6.4	23.2	234.0	72.9	57.4
		50-100	2086.0	8.0	36.1	176.0	71.5	8.8
		0-100	4791^b	24.2^a	164.6^c	942^a	242.7^a	90.6^b
Fprob		0.3%	83%	1.5%	4.2%	53%	<.001	

8.5. DISCUSSION

8.5.1. Leaching in an undisturbed standing crop

Nutrient leaching, occurring to a small extent in the undisturbed standing crop, was dependant on rainfall volume and intensity, with drainage volumes constrained by high tree water use (CHAPTER 4). An annualised nutrient leaching beyond 100 cm (felling to 1 year) amounted to $4.2 \text{ kg ha}^{-1} \text{ N}$ (1.3 kg ha^{-1} of $\text{NH}_4\text{-N}$, $0.5 \text{ kg ha}^{-1} \text{ NO}_3\text{-N}$, 2.3 kg ha^{-1} organic-N), $7.8 \text{ kg ha}^{-1} \text{ K}$, $4.0 \text{ kg ha}^{-1} \text{ Ca}$, $9.0 \text{ kg ha}^{-1} \text{ Mg}$ and 36.3 kg ha^{-1} of Na. Rapid water use under the mature *Eucalyptus* crop reduced drainage loss (CHAPTER 4) and the possibility of deeper soil leaching losses. A study in Congo on a site with sandy soils similar to our study site showed rapid nutrient uptake by *Eucalyptus* tree roots to limit drainage volumes under mature *Eucalyptus* (Laclau et al. 2005a), resulting in a relatively small fraction of nutrients leached below 6 m relative to those entering the soil.

Nutrient leaching at 15 cm was far higher than nutrients added by canopy throughfall. Mobile anions to facilitate the leaching of base cations would have been present in relative abundance in the form of sulphates and chlorides (not monitored in our study) due to proximity of the study

site to the ocean. Although leaching was most notable for the cations, it was the largest for Mg^{2+} followed by K^+ and Ca^{2+} . High levels of litterfall and rapid decomposition at the soil surface (data submitted for publication) added to nutrients in the soil solution, increasing leaching at 15 cm. Further additions may have occurred through rapid fine root turnover, although not recorded in our study. Nitrogen leaching was also larger at 15 cm depth than in the throughfall, but not to the same extent as N was immobilised through soil microbial activity under the mature trees (data submitted for publication). Laclau et al. (2003a) demonstrated that large a proportion of nutrient uptake by clonal *Eucalyptus* are derived from litterfall and root turnover. These nutrients, depending on their mobility in the soil, may leach through the upper soil layers prior to uptake. Despite high leaching at the soil surface under the standing crop, leaching decreased strongly with depth (**Appendix 8-2, Appendix 8-3**). This demonstrates a displacement of nutrients occurring under the mature trees from the surface to relatively deeper layers. These nutrients are not lost from the soil, but are either taken up by the trees, accumulated in the soil at depth or immobilised in organic complexes.

8.5.2. Post felling

Soil nutrient leaching increased a few months after clearfelling as a lack of water uptake afforded soil moisture recharge and more rapid drainage with rainfall. This leaching (and high cation loss) beyond 100 cm in felled treatments was far greater than rainfall input. This was brought about through aboveground biomass additions, accelerated mineralisation processes, increases in mobile accompanying anions such as NO_3^- and a lack of uptake further increasing soil solution concentrations. Increases in soil temperature and moisture following harvesting triggers a more rapid mineralisation and nitrification of soil organic matter (Carlyle 1993), a process that occurred at our site and can be inferred from the abundance of NO_3^- after felling in **Figure 8.1**. The more mobile K^+ and NO_3^- ions were rapidly leached whereas Ca^{2+} and NH_4^+ ions leached slower, being less mobile in the soil. This lower mobility reduced Ca^{2+} leaching losses after burning, as was evident by higher calcium concentrations in the upper soil layers of the burnt plots. Felled treatments had a high leaching of N, K, Ca, Mg and Na beyond 100 cm, which was most significant during the period after felling and burning to 6 months after the new crop was planted. The increased leaching during the fallow and early growth period occurred when tree uptake was not sufficient to reduce drainage and utilise the concentrated soil solution.

Nitrogen leaching beyond 100 cm soil depth increased during the first few months after burning through an increase in NO_3^- ions in the soil solution (**Figure 8.1b**). Although K^+ leaching was relatively lower in the Burn than the No-Burn, Ca^+ and Mg^{2+} leaching were not substantially altered following burning. A reduction in K^+ leaching loss beyond 100 cm in the Burnt plots may have been as a result of more rapid early tree growth in the Burn treatment. Leaching at 100 cm in the felled plots however differed strongly from standing crop plots (**Table 8-1**) as increased soil water drainage carried nutrients from shallower layers to deeper layers. In addition, nutrients accumulated in the soil at depth were released during increased drainage and cessation of root uptake. Nutrient and water uptake gradually became large enough to reduce leaching during the rapid tree canopy development phase of all felled plots after planting (**Table 8-1**).

8.5.3. Atmospheric inputs

Atmospheric inputs (CHAPTER 5) played a major role in soil solution chemistry under the standing crop. High $\text{NH}_4\text{-N}$ concentrations in canopy drainage translated into high $\text{NH}_4\text{-N}$ concentration in the soil leachate of the standing crop treatment. This is typical of N rich sites (fertile soils) that receive large N inputs (Slesak et al. 2009). Nitrogen mineralisation (formation of $\text{NH}_4\text{-N}$) was dominant in the standing crop throughout the study period (data submitted for publication). High litterfall and litter layer decomposition also played a role. In contrast with the standing crop, peaks in rainfall $\text{NH}_4\text{-N}$ corresponded with peaks in soil solution $\text{NO}_3\text{-N}$ of the felled treatments. Increased N mineralisation followed by rapid nitrification may have been responsible for the large increases in $\text{NO}_3\text{-N}$ leaching in felled plots.

8.5.4. Soil cation accumulation

The increase in soil cation concentrations with depth (particularly Na) in the standing crop treatment of our study (**Table 8-3**) may indicate an accumulation of cations in the deeper soil through limited drainage. These cations gained through atmospheric deposition, canopy exchange and litterfall may explain the high Na concentrations in the soil and soil solution (Mayer et al. 2000; de Vries et al. 2003). Cation deposition in our study was less than that lost through leaching beyond 100cm in the mature crop. This was likely through a combination of water and nutrient uptake that limited the movement of these nutrients through the soil profile. Lower rainfall (dry periods) immobilise soil nutrients and result in an accumulation of nutrients

in the soil, whereas higher rainfall periods promote leaching of mobile nutrients, increasing losses and reducing soil nutrient pools (Johnson et al. 2002). A portion of the cations in the soil solution under the felled treatments may therefore have originated from the release of cations accumulated in the soil under the previous crop. This is shown as an initial increase in the leaching of these elements (particularly Na) as a consequence of increased drainage after felling. It is also likely that release of nutrients from the previous crop tree root system was delayed, increasing soil solution concentrations a few months after clearfelling. A short delay in decomposition of old tree roots may therefore have contributed to a delayed increase in soil solution concentrations (Stevens et al. 1993; Powers et al. 2005).

8.5.5. Study outcome

The increase in leaching was short lived and constrained to the period directly following clearfelling, decreasing in magnitude after planting to become similar to the mature crop by the canopy closure stage. Assuming mean standing crop data to represent potential future leaching and atmospheric deposition, **Figure 8.3**, a hypothetical scenario based on K data demonstrates this short lived yet relatively substantial post-harvest leaching effect. A similar pattern emerges for NO_3^- and cations recorded in this study, except for the increased NO_3^- and decreased K^+ leaching after burning. The additional increase in NO_3^- after felling and burning was smaller than the increase caused through clearfelling alone and also short-lived. Atmospheric addition of all nutrients was then larger than leaching losses under the mature crop at our site, which can be related to an accumulation of nutrients in the soil that are not taken up or immobilised in the mature crop.

It is apparent from the data in this study that the effect of clearfelling on leaching was more substantial than the effect of residue management on leaching. A combination of harvesting nutrient losses, oxidative and particulate losses during burning, when added to the small increase in leaching losses after burning may rapidly reduce nutrient pools on our site if accumulated over successive rotations. These losses will primarily occur from organic or biomass nutrient pools, which are crucial to maintaining nutrient supply to trees on such infertile sites (Laclau et al. 2003a). Further to this, a loss of nutrients from the top 100 cm of soil is critical after clearfelling and during the establishment phase of the new crop, given the small soil nutrient pools, rapid decomposition of residue after clearfelling, increased leaching losses and large early tree growth

demands. Nutrient leaching is therefore a concern since it occurs during a small portion of the rotation and may be further reduced through minimising the duration of the temporary unplanted period after felling and through accelerating early tree growth.

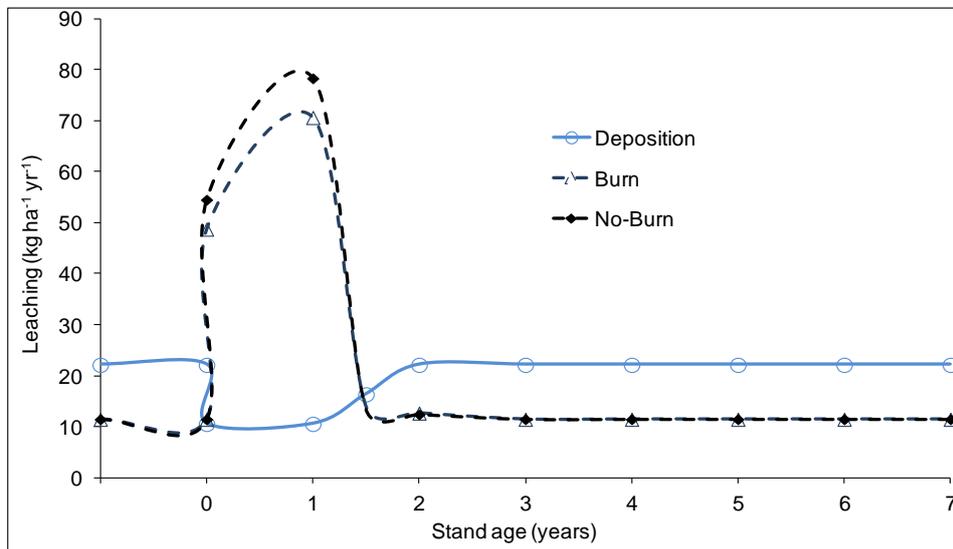


Figure 8.3: A hypothetical projection of annual leaching and atmospheric inputs to rotation end using mature crop data for projected years (based on K data).

Although root growth may extend beyond 3m depth after the first year (Bouillet et al. 2002) this soil layer carries the bulk of nutrient scavenging fine roots (Laclau et al. 2001). Deeper exploration to depths of around 85% of the tree height under non-limiting soil conditions (Christina et al. 2011) may imply a rooting depth in the new crop of around 10 m in our study. Nutrients lost to deeper layers may therefore be extracted by deep roots and redistributed to shallower soil layers through litterfall and root turnover (McCulley et al. 2004). da Silva et al. described relative uptake of nitrate tracers and Ca and K analogue tracers, giving evidence of significant nutrient uptake at 3 m depth. Additional increases in N losses from the system as a whole may also occur as a consequence of organic matter lost during burning diminishing the soils ability to store N (Pilkington et al. 2007). Conservation of residue N pools with reduced N leaching loss beyond 100 cm may help to conserve N in the long term. Residue retention was shown to reduce N leaching in *Pinus radiata* grown on sandy soils (Carlyle et al. 1998a) and under *Eucalyptus* (Gómez-Rey et al. 2008), and was further improved with residues break-up and soil mixing. This was as a result of an improved soil carbon status.

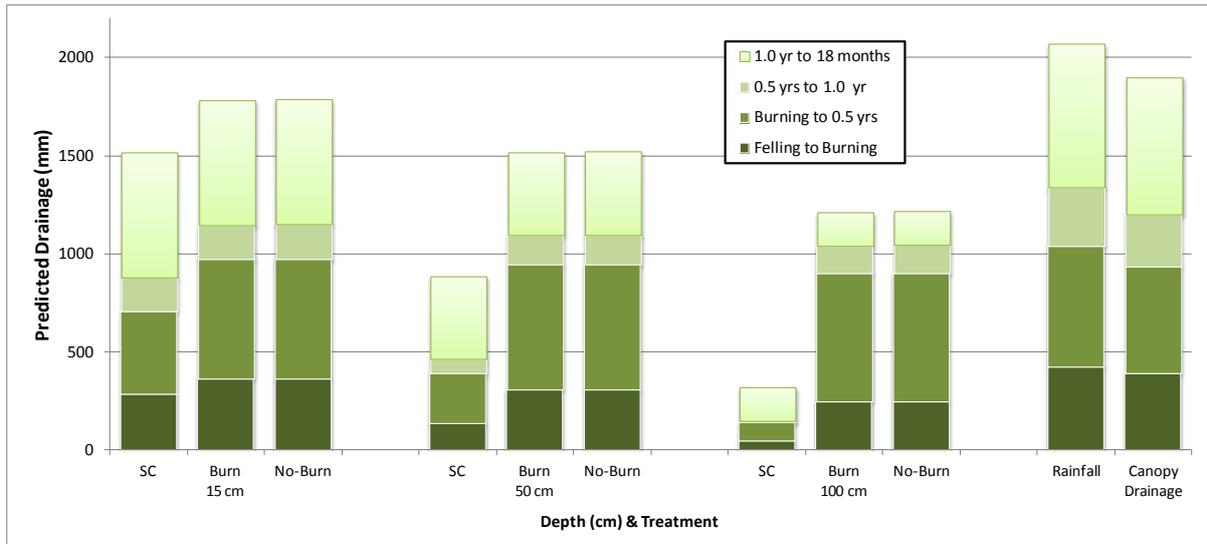
8.6. CONCLUSION

Leaching losses in general were relatively small under the undisturbed mature crop and may be of little consequence to the nutrient status of the site if such levels occur throughout the rotation. Data presented in this study demonstrated a large, short duration increase in leaching as a result of clearfelling which was rapidly reduced to pre-felling levels by the newly planted crop. Residue management had little effect on leaching. Prescribed burning, an extreme residue management practice resulted in a temporary increase in nitrate leaching which was small compared to losses associated with clearfelling alone. The most substantial nutrient losses in this study were incurred through harvesting biomass removal, residue burning (oxidation) and clearfelling induced leaching.

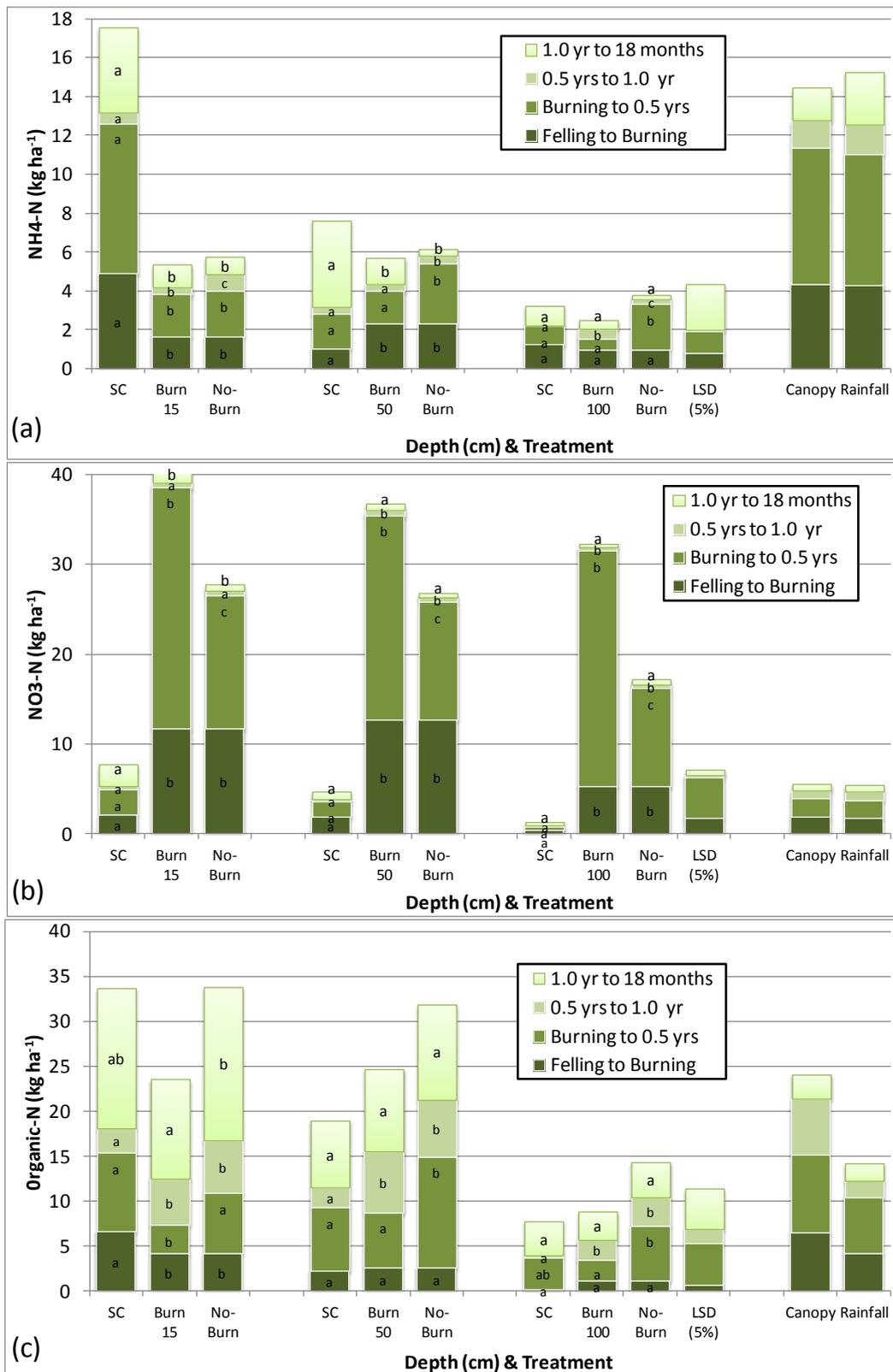
Given the size of the soil nutrient pools relative to nutrient losses at our study site it is likely that increased nutrient losses will accelerate a decline in these site nutrient pools. Residue burning on such nutrient poor soils should be avoided, not so much for the increase in leaching, but for the oxidative and particulate loss during burning in addition to clearfelling losses. This may result in a long-term site nutrient decline and reduce the inherent productivity potential of the site.

Increased leaching of N and cations after clearfelling may be of concern to plantation managers as this will decrease short-term nutrient availability to the new crop. Given the high nutrient demand during early tree growth and the high risk of additional nutrient loss during the temporary unplanted period, management practices that reduce losses and promote nutrient retention need to be encouraged (Nzila et al. 2002). Rapid re-establishment and rapid early growth may facilitate the conservation of soil nutrients during this high loss period. Harvesting and biomass removal will however, eventually deplete the nutrients on these infertile soils, unless nutrients are replaced through fertilisation (mineral, organic or N-fixation) or rotation lengths are extended to allow for natural replenishment.

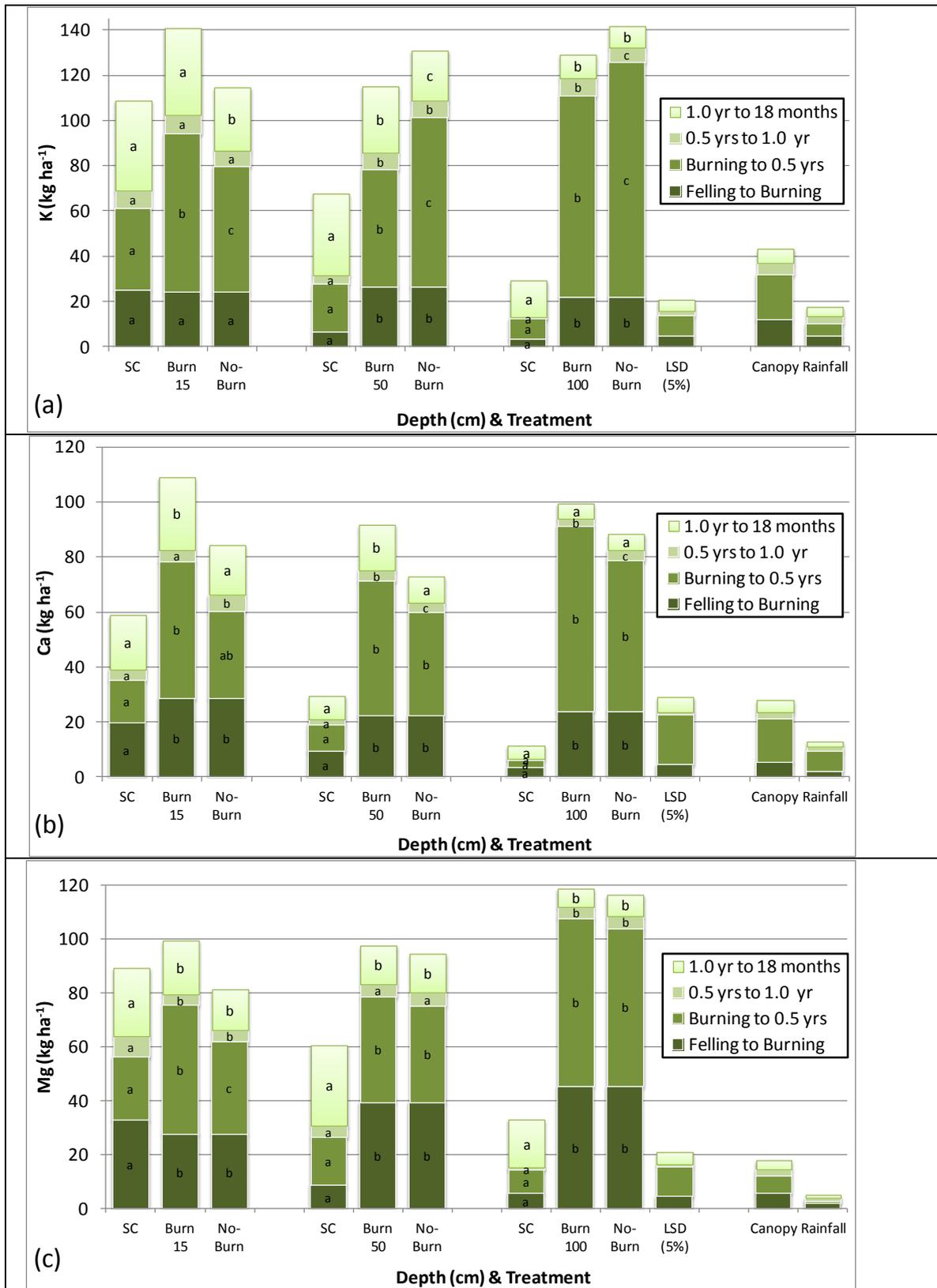
8.7. Appendix



Appendix 8-1: Rainfall, canopy drainage and Hydrus predicted drainage volumes at 15, 50 and 100 cm for felled and standing crop (SC) areas of the study site. Each portion of the bars represents drainage predicted for each period.



Appendix 8-2: Total N leached during each time period at 15, 50 and 100 cm and added with rainfall and canopy drainage for felled and standing crop (SC) areas of the study site. Different a, b, c superscripts denote significance difference between treatments at each depth for each time period at LSD_{5%}. A LSD_{5%} stacked bar is given to show significant differences.



Appendix 8-3: (a) K, (b) Ca, and (c) Mg leached during each time period at 15, 50 and 100 cm and added with rainfall and canopy drainage for felled and standing crop (SC) areas of the study site. Different a, b, c superscripts denote significance difference between treatments at each depth for each time period at LSD_{5%}. A LSD_{5%} stacked bar is given to show significant differences.

Appendix 8-4: Ammonium, nitrate and organic nitrogen deposition and leaching at 15, 50 and 100 cm soil depths cumulated for each key period. Least significant difference (LSD) and ANOVA *F* probability given for leaching differences between treatments (kg ha⁻¹).

Time period (years)	Standing crop Canopy	Felled Rainfall	Standing crop	Burn 15 cm	No-Burn	Standing crop	Burn 50 cm	No-Burn	Standing crop	Burn 100 cm	No-Burn	LSD	F pr.
NH₄-N													
Felling to Burning	4.4	4.3	4.9	1.6	1.6	1.0	2.3	2.3	1.2	1.0	1.0	0.8	0.10
Burning to 0.5	7.0	6.8	7.8	2.2	2.4	1.8	1.7	3.1	1.0	0.5	2.4	1.1	0.01
0.5 to 1.0	1.4	1.5	0.5	0.3	0.8	0.3	0.3	0.4	0.0	0.5	0.2	0.1	<.001
1.0 to 1.5	12.8	12.6	4.4	1.2	0.9	4.5	1.3	0.3	1.0	0.4	0.3	2.4	0.06
Felling to 0.5	11.4	11.0	12.6	3.8	4.0	2.8	4.0	5.4	2.2	1.5	3.3	1.1	0.00
0.5 to 1.5	14.2	14.1	4.9	1.5	1.7	4.8	1.7	0.8	1.0	1.0	0.5	2.4	0.07
NO₃-N													
Felling to Burning	1.9	1.8	2.2	11.8	11.8	1.9	12.8	12.8	0.4	5.2	5.2	1.8	<.001
Burning to 0.5	2.1	2.0	2.8	26.7	14.7	1.7	22.6	13.0	0.5	26.2	10.9	4.6	<.001
0.5 to 1.0	0.9	0.9	0.3	0.5	0.5	0.3	0.6	0.6	0.0	0.4	0.4	0.2	0.04
1.0 to 1.5	4.9	4.7	2.4	1.4	0.7	0.8	0.7	0.5	0.4	0.4	0.7	0.7	0.19
Felling to 0.5	3.9	3.8	5.0	38.5	26.5	3.6	35.4	25.8	0.9	31.4	16.2	4.8	<.001
0.5 to 1.5	5.8	5.5	2.7	1.9	1.2	1.1	1.3	1.0	0.4	0.8	1.0	0.6	0.49
Organic N													
Felling to Burning	6.5	4.3	6.7	4.2	4.2	2.2	2.7	2.7	0.2	1.1	1.1	1.2	0.72
Burning to 0.5	8.6	6.2	8.7	3.1	6.7	7.1	6.1	12.3	3.5	2.4	6.2	2.8	0.02
0.5 to 1.0	6.2	1.7	2.7	5.1	5.9	2.2	6.7	6.3	0.2	2.1	3.2	1.5	0.00
1.0 to 1.5	21.4	12.2	15.6	11.1	17.0	7.4	9.1	10.6	3.8	3.2	3.8	4.5	0.42
Felling to 0.5	15.2	10.5	15.4	7.3	10.9	9.3	8.8	14.9	3.7	3.5	7.3	2.2	0.00
0.5 to 1.5	27.6	13.9	18.3	16.2	22.9	9.6	15.8	17.0	4.1	5.3	7.0	3.8	0.04

Appendix 8-5: Potassium, calcium and magnesium deposition and leaching at 15, 50 and 100 cm soil depths cumulated for each key period. Least significant difference (LSD) and ANOVA *F* probability given for leaching differences between treatments (kg ha⁻¹).

Time period (years)	Standing crop Canopy	Felled Rainfall	Standing crop	Burn 15 cm	No-Burn	Standing crop	Burn 50 cm	No-Burn	Standing crop	Burn 100 cm	No-Burn	LSD	F pr.
K													
Felling to Burning	11.9	4.8	25.0	24.3	24.3	6.4	26.4	26.4	3.4	21.8	21.8	2.5	<.001
Burning to 0.5	19.8	5.4	36.1	69.9	55.5	21.4	51.7	75.1	9.3	88.8	104.2	9.1	<.001
0.5 to 1.0	5.4	3.5	8.0	8.1	6.8	3.6	7.3	7.3	0.4	8.0	6.0	1.4	<.001
1.0 to 1.5	37.1	13.6	39.5	38.2	28.1	36.0	29.2	22.0	16.1	10.1	9.6	5.3	0.01
Felling to 0.5	31.7	10.1	61.1	94.2	79.8	27.8	78.2	101.6	12.7	110.6	126.0	10.4	<.001
0.5 to 1.5	42.5	17.1	47.5	46.3	34.9	39.6	36.5	29.4	16.6	18.1	15.6	6.0	0.03
Ca													
Felling to Burning	5.6	2.0	19.9	28.8	28.8	9.5	22.4	22.4	3.5	23.9	23.9	5.0	<.001
Burning to 0.5	15.8	7.7	15.5	49.7	31.6	9.5	48.8	37.8	2.9	67.4	55.1	18.5	0.00
0.5 to 1.0	2.4	1.2	3.7	4.0	6.0	1.9	3.8	3.0	0.2	2.3	3.5	0.7	<.001
1.0 to 1.5	23.7	10.9	19.6	26.1	17.8	8.5	16.6	9.7	4.9	5.7	5.9	5.7	0.12
Felling to 0.5	21.4	9.7	35.4	78.5	60.4	19.0	71.2	60.2	6.4	91.3	79.0	19.2	<.001
0.5 to 1.5	26.1	12.0	23.3	30.2	23.8	10.4	20.4	12.7	5.1	8.1	9.4	6.0	0.09
Mg													
Felling to Burning	5.6	1.8	33.1	27.5	27.5	8.8	39.4	39.4	5.9	45.5	45.5	4.7	<.001
Burning to 0.5	6.6	0.9	23.3	47.8	34.6	17.8	39.2	35.7	8.4	62.1	58.6	10.8	<.001
0.5 to 1.0	2.3	1.1	7.5	3.9	3.9	4.2	4.6	4.8	0.7	4.1	4.6	0.7	0.54
1.0 to 1.5	14.5	3.8	25.5	20.1	15.3	29.9	14.1	14.6	17.8	6.8	7.7	4.6	<.001
Felling to 0.5	12.2	2.7	56.4	75.3	62.1	26.6	78.6	75.1	14.4	107.6	104.1	7.8	<.001
0.5 to 1.5	16.8	4.9	33.0	23.9	19.2	34.1	18.7	19.5	18.5	10.9	12.3	4.4	<.001
Na													
Felling to Burning	28.3	23.1	115.4	74.6	74.6	32.6	101.6	101.6	22.7	141.4	141.4	10.4	<.001
Burning to 0.5	33.7	29.7	129.2	61.7	72.8	91.5	72.7	80.1	35.3	157.1	149.3	17.6	0.17
0.5 to 1.0	10.4	7.5	25.9	7.0	6.4	12.5	9.1	7.2	2.8	10.1	8.2	1.4	<.001
1.0 to 1.5	72.4	60.3	129.5	64.7	46.4	117.5	43.1	38.3	81.2	18.3	19.1	18.0	<.001
Felling to 0.5	62.0	52.8	244.6	136.4	147.5	124.2	174.2	181.7	58.0	298.5	290.6	20.4	<.001
0.5 to 1.5	82.8	67.9	155.4	71.7	52.8	130.0	52.2	45.5	84.0	28.4	27.3	18.0	<.001

CHAPTER 9: GENERAL DISCUSSION AND CONCLUSION

9.1. ABSTRACT

The risk of nutrient depletion and yield decline of successive rotations was assessed on the low nutrient and organic matter Aeolian sand derived soils in the subtropical environment of the Zululand coastal ecosystem. The effects of harvesting intensity, residue management and fertilisation on pools and fluxes of nutrients in the residues and soil and in a new crop were measured. Residue management included retention, burning and doubling of residues and whole tree removal. Plots of the old crop were left undisturbed to enable comparison of nutrient pools and fluxes. Nutrient recycling and atmospheric additions (particularly N and K) provided substantial nutrient supply to the undisturbed crop, while minimal leaching losses beyond 1 m soil depth enabled nutrients to accumulate in the biomass and first meter of soil. Substantial nutrient (particularly N) and biomass losses occurred during harvesting and residue burning. Soil pools of Ca and Mg increased after burning, but soil N mineralisation was reduced by 46%. Leaching beyond 1 m increased dramatically after clearfelling then decreased when trees were 6 months old, but was not affected by residue burning. Nutrient losses due to harvesting and burning were more substantial than leaching losses. Nutrient uptake during the early rapid growth phase was met primarily through residue and litter decomposition. Early growth was not altered by harvesting intensity and residue management, but increased after fertilisation. The risk of a nutritional productivity decline (projected over a seven year rotation) was countered in this system through atmospheric inputs that replaced a large proportion N, K, and Ca lost through stem wood removal. However, Mg may be at risk as deposition was too small to replace losses. While not reflected in subsequent growth, evidence from nutrient flux studies suggested that increased nutrient removal through more intense harvesting and residue management can increase the risk of subsequent rotation productivity decline.

9.2. INTRODUCTION

The Zululand coastal plain is a productive and rapid timber growing area of great value to the South African forestry industry (Louw 1997; du Toit et al. 2001). High productivity over consecutive six to seven year rotations has been realised on the nutrient poor (dystrophic) soils across this region due to intensive management. An apparent lack of productivity decline on these low nutrient sites has continued despite increased losses induced through high productivity. This has been questioned and discussed by plantation managers and researchers for many years. Evidence linking harvesting intensities and residue management practices with productivity decline in the subsequent rotation and in the long-term has been presented across a number of international study sites with low litter and soil N (Saint-André et al. 2008) (CHAPTER 2). These productivity declines were dependent on climatic (fast growth sites) and edaphic properties linked with soil organic matter and low nutrient availability at each site (nutrient poor sites) (Corbeels et al. 2005). Evidence of productivity responses to soil carbon and nutrient losses on nutrient poor (dystrophic) South African sites has been limited to fertilisation at planting (particularly N) studies. Substantial positive responses occurred on a number of infertile sites across Zululand that were degraded through intensive agricultural practices or on sites that possessed physiographical constraints (du Toit et al. 2001; du Toit and Oscrift 2003). These past responses (discussed in CHAPTER 6) are perhaps anecdotal evidence that sites across this region were at one time and still may be at various levels of agriculturally induced nutrient degradation. It also demonstrates the potential for these sites to become degraded through intensive agriculture.

9.2.1. The Study

The study presented in this thesis was therefore commissioned by members of the South African forestry industry who were interested in understanding the major processes driving the sustainable supply of nutrients to the trees grown in the Zululand coastal region on nutrient poor soils. This included the effects of harvesting and residue management practices on these processes. Describing each component of the biogeochemical nutrient cycling processes at a high level of resolution is complex and requires substantial human and financial resources. This study therefore considered the effects of extremes in management practices on the bulk processes driving nutritional sustainability at the site. The effects of clearfelling and residue management on the biomass and pools and fluxes of nutrients N, P, K, Ca and Mg in the new tree crop, residues and soil were compared between a clearfelled stand and an undisturbed stand. The clearfelled stand had a number of management operations applied that resulted in treatments with various quantities of residues and

nutrient loads on the soil surface. Residues were broadcast across most treatments at clearfelling, the exception being removal of all residues through whole tree harvesting (Whole-Tree), leaving the old litter layer intact. The residues from the Whole-Tree treatment were used to create a residue (and nutrient load) doubling treatment (Double). A burning treatment (Burn) was applied and contrasted with residue retention (No-Burn) and the undisturbed crop (standing crop). The Burn treatment, No-Burn and standing crop were assumed to be at the extremes of nutrient flux processes by virtue of their impact on carbon and nutrients. A further treatment (Fert) used fertiliser (applied at planting) to replace nutrients lost through stemwood harvesting with residue retention. These treatments were designed to enable an in-depth understanding of changes in growth and nutrient fluxes under various levels of nutrient availability. An additional atmospheric deposition monitoring site was also included within the region (CHAPTER 5), against which atmospheric inputs were compared. The study was carried out from prior to clearfelling and was completed at canopy closure of the new crop (one year after planting). However a number of measurements were continued beyond canopy closure. In completion this study has given many insights into the interactions between site nutrient dynamics, management and productivity.

The study was separated into the following sub-studies that determined:

- Describing treatment effects on soil moisture and prediction of drainage fluxes (CHAPTER 4)
- The magnitude of atmospheric addition in rainfall and canopy drainage (CHAPTER 5 CHAPTER 5: above).
- Changes in residue, litter, tree growth and biomass nutrient pools and fluxes (CHAPTER 6).
- Changes in top-soil N mineralisation fluxes (CHAPTER 7).
- The impact on nutrient leaching to one meter soil depth using predicted drainage fluxes and collection of free drainage soil solution (CHAPTER 8).

This final chapter is a summary of key findings and a synthesis of the data presented in the previous chapters. This chapter will give insights into the work, offer some speculation and suggest future research to refine and expand the findings of this study.

9.3. METHODS OF DATA SUMMARY:

Nutrient flux data derived from each thesis chapter is summarised between felling and canopy closure (a 20.1 month period) using a graphical representations of the Standing crop, Burn and No-Burn treatments. Soil and biomass pools are given at the time of canopy closure of the new crop, totalled to one meter depth. Annualised fluxes and growth based nutrient accretion of the Standing Crop are calculated using the same 20.1 month period. Nutrient accretion is calculated as the sum of tree growth and litterfall. Decomposition is calculated as the sum of litterfall and nutrients lost from the litter layer through decomposition. Where data was not available for various treatments, it was estimated from literature or other treatments. Further to this, a balance of nutrients entering and leaving the soil was calculated. A nutrient budget was also calculated for the full rotation, projecting all measured additions and losses to seven years after planting.

9.3.1. Short-term nutrient balance

The balance of short term (clearfelling to canopy closure) nutrient fluxes was calculated as the sum of nutrient addition at the soil surface and losses to the atmosphere or below 1 m soil depth for each treatment. This calculation assumes the balance of all nutrient addition and losses to the soil to contribute to soil nutrient supply (and pools) during the 20 month clearfelling to canopy closure period. Additions considered included atmospheric inputs, residue and litter decomposition, fertilisation and post burning ash (that indirectly accounts for burning losses). Losses considered were leaching and nutrient uptake through tree growth. These were assumed to be the only factors reducing soil nutrient supply, essentially reducing short term soil nutrient availability. Other losses (runoff, erosion and denitrification) were assumed to be negligible. Aboveground nutrient uptake through growth was calculated as the sum of aboveground growth accretion and litterfall. Residue decomposition prior to burning was added to ash layer nutrient content after burning as an estimate of short term soil nutrient supply in the Burn treatment. Fluxes not recorded in the other treatments were estimated. Leaching in the Fert. treatment was assumed to be equal to that in the No-Burn treatment. Whole-Tree treatment leaching was estimated assuming the ratio between No-Burn leaching loss : residue decomposition to be similar to Whole-Tree treatment leaching loss : residue decomposition. This did not include the contribution of belowground old root system decomposition to leaching. Atmospheric inputs and leaching of P were not included in the short-term balance (not assessed at the site).

These calculations were done to determine the balance of nutrient additions and losses to the soil so that the contribution of the balance of these nutrient fluxes to soil nutrient supply could be determined. This was contrasted with growth uptake during the rapid early growth phase of the new crop. A negative balance implies an increased uptake of readily plant-available below ground nutrient reserves during the early tree growth phase, while a positive balance suggests a nutrient surplus. A true reflection of nutrient availability is more complex and needs to take into account, for example, concurrent processes of mineralisation and immobilisation in the case of N and P, as well as sorption-desorption and redox reactions in the case of P. However, despite these shortcomings, this estimation does give an indication of magnitude and role that various aboveground nutrient sources play in supplying or supplementing early tree growth nutrient demands. The belowground nutrient reserves comprise soil nutrients held on the soil exchange, in the soil solution and released from the old tree root system. As the old root system was not measured, it was assumed to form part of the soil nutrient pool. Decomposition of the old root system was assumed to contribute similar quantities of nutrients to the soil solution across all treatments.

9.3.2. Projected long-term nutrient budget

A seven year forward projected budget was estimated as the cumulative sum of felling to canopy closure fluxes by combining the 20 months clearfelling to canopy closure fluxes with standing crop fluxes thereafter. This generated an algebraic balance of all nutrient additions and losses occurring in each treatment. Similar assumptions were used as above (Section 9.3.1), while all other fluxes (including mineral weathering and runoff) were also assumed to be negligible. Phosphorus deposition was estimated from literature derived values in CHAPTER 5 as $0.5 \text{ kg ha}^{-1} \text{ yr}^{-1}$. Nutrient budgets were contrasted with soil pool sizes to 1 m soil depth using total N, extractable P and salt extractable base cations. Although these pools reflect a close approximation of soil nutrient reserves given the small cation exchange of these soils and the negligible weathering additions that can occur, it can only be taken as an index of soil nutrient supply. Total N includes a fraction of recalcitrant pools, a small fraction of which becomes available through N mineralisation. Site nutrient pools were taken as the sum of soil, litter and tree biomass pools. The nutrient budget was calculated as additions and losses to site nutrient pools, so only included atmospheric additions, leaching losses, burning losses, fertiliser additions and biomass removal from the site. Exchange and cycling between the trees, litter layer and soil was therefore excluded.

9.4. RESULTS AND DISCUSSION

The short term nutrient balance (defined in section 9.3.1) is discussed in sections 9.4.1 to 9.4.3

9.4.1. Main findings: The undisturbed standing crop

Accumulation of nutrients into the above ground biomass and nutrient storage in the litter layer remained large in the standing crop treatment during the 20.1 month intensive monitoring period.

Large amounts of nutrients (particularly N and K) were added through atmospheric deposition over the 20 month study period (**Figure 9.1**). Organic sources were noted as a potential major contributor to N deposition and noted as requiring further investigation (CHAPTER 5).

Leaching in the undisturbed crop decreased markedly with increasing depth with small losses beyond 1 m depth. This was attributed to high water use in the undisturbed crop. More N and less K, Ca and Mg were supplied to the soil in the standing crop during the 20 month period than was taken up in aboveground tree growth (**Table 9-1**). The quantity of N supplied through deposition was larger than the combined loss through leaching past 1 m soil depth, aboveground accretion and N immobilisation. Larger cation losses relative to additions occurred primarily through leaching past 1m soil depth. It is however, unlikely that leaching loss through deep drainage occurred under the standing crop. Model estimates indicated drainage only to occur to a depth of three meters under the conditions recorded during the study period (CHAPTER 4). This can limit leaching loss as tree roots would have access to nutrients at this soil depth. A zero leaching loss may therefore be assumed under the standing crop under normal climatic conditions and result in a positive cation balance in the undisturbed crop.

Recycling of nutrients (internal translocation and litterfall decomposition), atmospheric addition and high water use (low nutrient mobility in dry soil) limited tree growth demand on soil nutrient reserves (**Figure 9.1**). The undisturbed crop therefore maintained a more positive nutrient balance than the post felling crop, accumulating nutrients in the tree biomass and soil.

9.4.2. Main findings: Clearfelled and replanted crop

Growth up to 2.5 years after planting was not altered by any level of harvesting intensity or residue management. Replacement of stem wood nutrient losses through fertilisation improved early growth to 2.5 years after planting. Vector analysis at 6 months suggested that this occurred through responses to N and P supply and at one year after planting to Mg and Ca supply.

Biomass removal during harvesting and residue burning was responsible for a large proportion of the large nutrient losses observed. (**Figure 9.1**). Residue burning left large quantities of Ca and Mg on the soil surface. Calcium pool size in particular was significantly increased in the top soil layer, persisting beyond canopy closure (**Figure 9.1, Table 9-1**). Burning induced the largest N (and organic matter) loss of all treatments. Cation losses were however greatest after whole tree removal.

Net N mineralisation in the top 30 cm of the soil was reduced by 46% relative to the No-burn treatment during the 20.1 months following burning.

Rapid soil moisture recharge that occurred after clearfelling dramatically increasing leaching of N and the cations beyond 1 m. Losses of N were increased primarily through more rapid nitrification raising soil nitrate concentrations following clearfelling. Soil moisture content and leaching losses were however, not significantly altered by residue burning compared to residue retention. A large proportion of Mg leaching (42%) occurred prior to planting. A majority of K (79%) and Ca (67%) and similar Mg (54%) leaching occurring between planting and six months after planting (CHAPTER 8). The post clearfelling increase in leaching was therefore short-lived as new crop growth rapidly reduced soil moisture content and soil solution concentrations to levels similar to the undisturbed crop by around six months after planting.

9.4.3. Nutrient balance

Decomposition and atmospheric inputs over the felling to canopy closure period resulted in an excess of N in the No-Burn treatment relative to aboveground accretion (**Table 9-1**). In contrast, K, Ca and Mg were added to the soil in quantities far smaller than aboveground tree accretion. Large losses of Mg with leaching beyond 1 m resulted in a net loss of Mg from the soil during the 20 month period. Similarly, leaching of K in addition to growth uptake resulted in a large net K deficit. This deficit also occurred for Ca, but was due to rapid Ca uptake during early growth rather than leaching loss. Short term soil balance was most negative for K and least for Ca, meaning that stand nutrient uptake of particularly K had to be supplied from readily available soil storage pools. Phosphorus seemed to remain in excess of aboveground growth uptake, but leaching and deposition were not specifically recorded.

Above ground organic nutrient pools in the Burn treatment comprised live tree biomass alone as no residue pools was present on the soil surface. Differences between the soil N and Mg balances and

tree uptake were more negative for the Burn treatment than for the No-Burn treatment (**Table 9-1**). A positive Ca balance occurred in the Burn treatment as a result of the ash layer on the soil surface, but this assumed all the ash remained (no wind or water erosion) and nutrients it contained in it were in a plant available mineral form. Fertiliser addition (assuming similar leaching to No-Burn) created a positive nutrient balance for all elements except Mg.

Whole-Tree aboveground nutrient pools comprised the partially decomposed previous rotation litter layer and residual tree biomass. Although these pools were marginally smaller than No-Burn pools, except for Ca pools that were much smaller (48% of pre-clearfelling standing crop pools). Whole tree removal resulted in a P, K, Ca and Mg deficit assuming similar leaching losses. No growth depression occurred in the Whole-Tree treatment as nutrients released from the litter layer in addition to the soil and the old root system were most likely sufficient to supply early growth demands. Only soil P and K were significantly lower in the Whole-Tree treatment than the No-Burn treatment at canopy closure. Vector analysis indicated that these elements were in decline (CHAPTER 6).

Pools of K were similar across all treatments due to low K retention in the soil and residues which was leached out shortly after clearfelling. Residue retention maintained large amounts of nutrients in the aboveground biomass pools to canopy closure relative to residue burning and whole tree removal. Nutrients remaining in residues create a more favourable condition as these can become available later in the rotation and add to supply for later tree growth demand. This may, however, have less benefit to tree growth as nutrient demand is usually small in healthy mature stands and semi-mature stands (Laclau et al. 2003a). Due to residue and litter layer retention aboveground biomass N, Ca and Mg pools at the end of the study were larger in the No-Burn treatment than in the Burn treatment. The No-Burn treatment had attained 57% and 76% of the standing crop pre-clearfelling N and Ca biomass pools respectively by one year after planting. In contrast Burn treatment N and Ca biomass pools attained only 18% and 7% of the pre-clearfelling standing crop biomass nutrient pools by canopy closure.

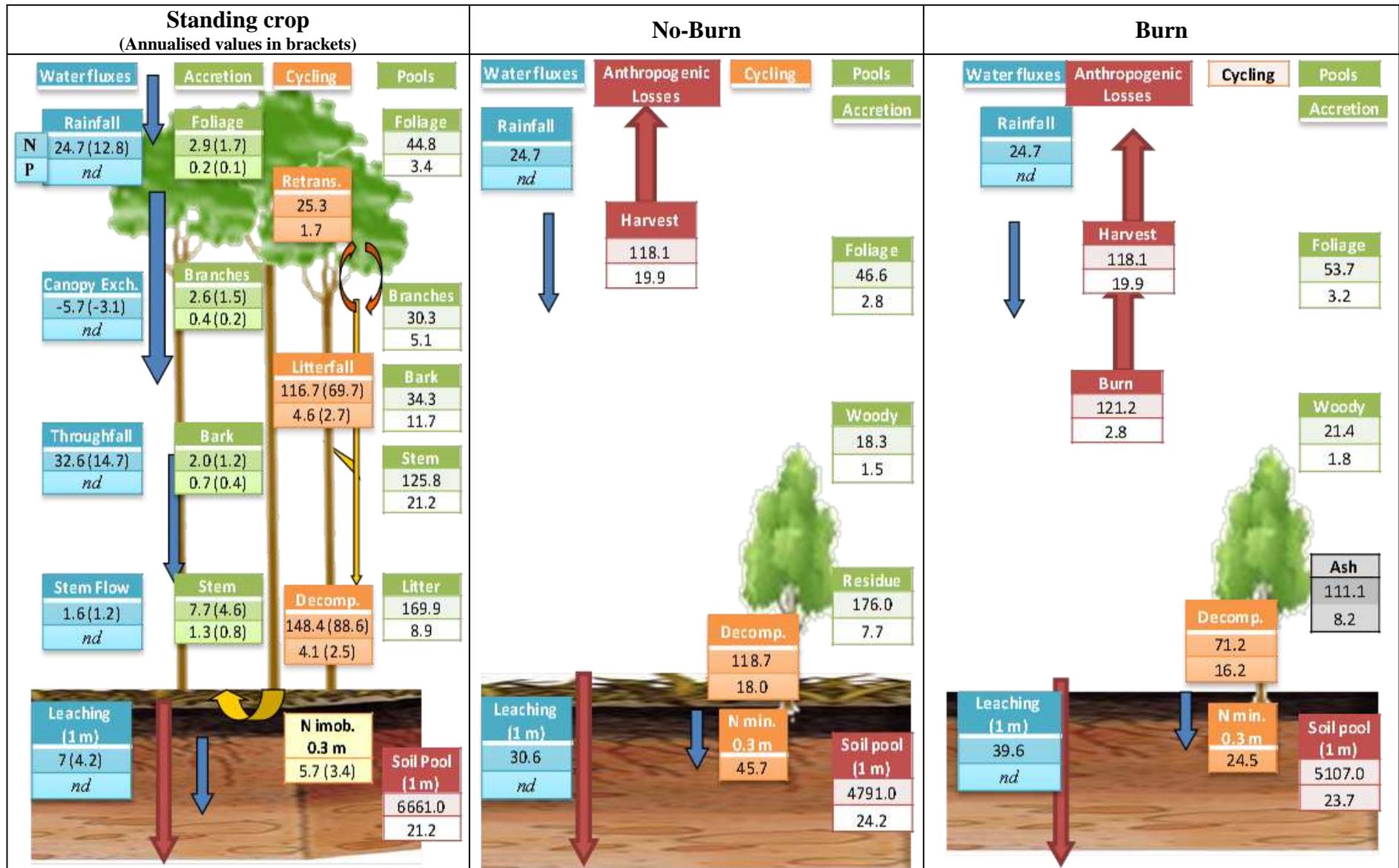
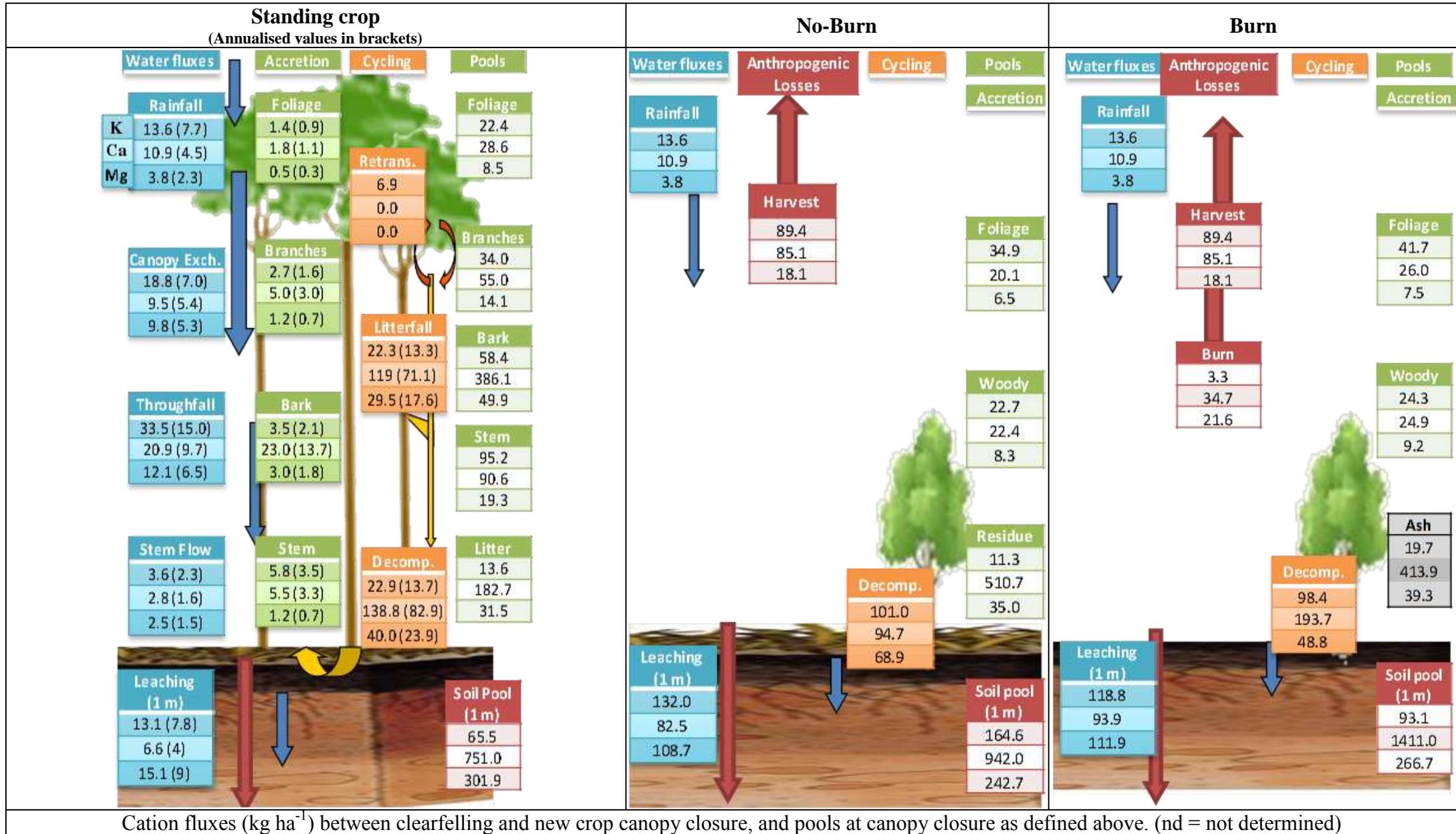


Figure 9.1: Nitrogen and phosphorus fluxes (kg ha^{-1}) between clearfelling and new crop canopy closure, and pools at canopy closure as defined above. (nd = not determined)



Cation fluxes (kg ha^{-1}) between clearfelling and new crop canopy closure, and pools at canopy closure as defined above. (nd = not determined)

Table 9-1: Short term fluxes of nutrients into the soil. Supply is atmospheric deposition, decomposition, post-burn ash and fertiliser less leaching losses. Uptake is aboveground accretion. See section 9.3.1 for detailed explanation of calculations.

Treatment	Flux/pool	N	P	K	Ca	Mg
Standing	Supply	181	5	28	147	30
	Uptake	132	7	36	154	35
	Supply - Uptake	50	-2	-8	-8	-6
	Soil pools at 1 year	6661	21	66	751	302
	Biomass pools at 1 year	405	50	224	743	123
No-Burn	Supply	113	18	-17	23	-36
	Uptake	65	4	58	43	15
	Supply - Uptake	48	14	-75	-19	-51
	Soil pools at 1 year	4791	24	165	942	243
	Biomass pools at 1 year	241	12	69	553	50
Burn	Supply	56	16	-7	111	-59
	Uptake	75	5	66	51	17
	Supply - Uptake	-19	11	-73	60	-76
	Soil pools at 1 year	5107	24	93	1411	267
	Biomass pools at 1 year	75	5	66	51	17
Fert	Supply*	241	42	77	128	-15
	Uptake	89	5	66	46	19
	Supply - Uptake	152	37	11	82	-33
	Soil pools at 1 year	5428	22	166	809	216
	Biomass pools at 1 year	265	13	77	557	54
Whole-Tree	Supply*	75	2	12	6	-9
	Uptake	65	4	54	40	15
	Supply - Uptake *	10	-2	-42	-34	-24
	Soil pools at 1 year	4408	18	109	897	207
	Biomass pools at 1 year	210	12	63	346	42

*Based on hypothetical or assumed leaching losses

9.4.4. Projected whole rotation nutrient budget

A nutrient budget projected over a seven year rotation **Table 9-2** produced net losses of all elements in each treatment, except N and P which were gained in the Fert. treatment.

Nitrogen

Residue burning resulted in a more negative N budget than the No-Burn treatment. This was followed closely by whole tree harvesting. These treatments also removed a large proportion of the aboveground carbon pool which can possibly result in N deficiency or depletion later in the rotation. Nitrogen leaching losses played a relatively small role compared to harvesting removals and burning losses. As a percentage of total losses, leaching comprised 21% in the No-Burn, 14% in the Burn and 7% in the Whole-Tree treatments. Despite relatively large total soil N pools, much of this N pool may not be available for tree uptake, but released slowly through N mineralisation. Mineral N in the top 30 cm soil depth ranged between 10 and 75 kg ha⁻¹ in the undisturbed crop, which was between 0.5 and 4.5 % of the total N pool to 30 cm soil depth (CHAPTER 7). Overall, N mineralisation and immobilisation resulted in near zero net N mineralisation in the undisturbed standing crop throughout the study. The readily available soil N pool was therefore a small fraction of the total N pool. The readily available N pool is potentially at risk of future depletion should N held in the soil and litter carbon pools be lost or be recalcitrant to breakdown. This will also be affected by atmospheric inputs and any changes to this contribution

Cations

Cation losses projected over a seven year rotation were very high relative to exchangeable soil pools, particularly for K and Mg. All budgets were less negative after residue retention and least negative after fertilisation. Potassium losses were similar across each of the residue management treatments, hence unaffected by residue management. Fertilisation replaced a small amount of this K, resulting in a moderately less negative K budget. Calcium budgets were most affected by residue management. Whole tree harvesting produced the most negative Ca budget. This was followed by residue burning which was around 60% less than projected Whole-Tree losses. Residue burning induced the most negative seven year Mg budget. The Mg budget was similar across other treatments, only marginally more positive after fertilisation.

Table 9-2: Projected seven year additions, losses and budget ($\text{kg ha}^{-1} \text{ yr}^{-1}$) with and an index of number of rotations to residue and soil nutrient pool depletion. Pool sizes are defined in section 9.3.2.

		N	P	K	Ca	Mg
No Burn	Losses	24.4	2.8	37.6	26.9	25.0
	Addition	18.0	0.5	9.7	6.1	2.6
	Budget	-6.4	-2.3	-27.9	-20.8	-22.4
	Rotations	15.4	0.7	0.3	1.4	0.4
Burn	Losses	43.0	3.2	36.2	33.5	28.5
	Addition	18.0	0.5	9.7	6.1	2.6
	Budget	-25.0	-2.7	-26.5	-27.4	-25.9
	Rotations	4.0	0.6	0.3	1.1	0.3
Whole-Tree	Losses	36.9	5.5	35.5	76.8	24.7
	Addition	18.0	0.5	9.7	6.1	2.6
	Budget	-18.9	-5.0	-25.8	-70.6	-22.1
	Rotations	5.3	0.3	0.3	0.4	0.4
Fertilised	Losses	24.4	2.8	37.6	26.9	25.0
	Addition	35.1	3.7	21.9	15.9	4.8
	Budget	10.6	0.9	-15.7	-11.1	-20.1
	Rotations	gain	gain	0.4	2.7	0.4
Soil and biomass pools pre-felling		4890	79	370	1461	412

(Assuming deposition of $0.5 \text{ kg P ha}^{-1} \text{ yr}^{-1}$).

Early growth was not affected by harvesting and residue management on the low nutrient Zululand soils, despite large nutrient removals and losses. This was expected for Zululand given weak growth responses to burning in past studies (Germishuizen and Smith 2008), but contradicted similar international work presented in CHAPTER 2 and CHAPTER 6. Despite the large losses and substantial reductions in N mineralisation following burning, early growth data suggested that N was not limiting at this low-N site. A lack of response was due to nutrient demands during the rapid early growth phase being met through nutrients released through a combination of sources. These sources were soil nutrient reserves, decomposition of the old litter layer and old tree root system, atmospheric deposition and the ash bed remaining after burning. Growth responses occurred through fertilisation, vector analysis pointing initially to N and P, then to Ca and Mg being responsible for much of the response at one year after planting. Despite increased Ca availability in the Burn treatment, a growth increase was not realised possibly due to a lack of Mg. The onset of K and P depletion was evident in the Burn treatment, which also occurred in the Whole-Tree treatment.

The relevance of losses used in calculating the nutrient budget assume that nutrients lost from the first meter of soil are no longer accessible to the tree crop later in the rotation. Leaching losses from the first meter of soil are more accurately a displacement of nutrients out of the active fine root zone. This soil layer is most important during the early growth re-establishment phase and for the bulk of nutrient supply during later growth. A lack of nutrients in the top soil will therefore compromise tree growth during the early growth phase and most likely impact on the productivity of the whole rotation. Uptake of nutrients leached during the inter-rotation to deeper soil layers may occur during post-canopy closure growth until soil moisture becomes depleted. Nutrients displaced below one meter and extracted by deeper roots will then be recycled to the soil surface (through litterfall and root turnover). This will enrich the topsoil, replenishing soil nutrients (da Silva et al. 2011). However, rapid soil moisture depletion makes prolonged deep uptake unlikely under a mature crop (below three meters soil depth), other than after heavy rainfall events. There may also be insufficient water in deep soil layers for roots to survive or to extract large amounts of nutrients. These arguments need to be tested through deep soil sampling and possibly isotope studies of various elements placed in deep soil layers. This will confirm accumulation of nutrients at depth and determine whether or not tree uptake occurs from such depths.

9.4.5. Soil nutrient analytical methods

The analytical methods used to evaluate nutrient content and availability in the various soil pools may not give an absolutely true reflection of actual availability and transformations. As such, soil nutrient pools sizes measured are only estimation based on the selected analytical methods, and also sampling protocols used, and may have degree of error inherent in them. Similarly, the effects of residue management on soil chemical properties in similar international tropical plantation studies did not adequately explain subsequent growth effects (Tiarks and Ranger 2008). A nutrient budget can therefore be of limited value or difficult to interpret without a robust indicator of soil nutrient supply potential. Further work is therefore needed to determine short and long-term soil supply potential so that the life expectancy of soil nutrients can be determined under each plantation management system.

Using extractable soil pools, K and Mg are indicated as being at greatest risk of depletion, even under conservative (stem wood only removal) management practices. Burning (at high intensity)

dramatically increases N loss, not only placing N at risk of depletion, but the soil carbon pool in which it is held. Fire intensity is linearly related to N loss (Fisher and Binkley, 2000) and high intensity fires are responsible for large N losses. Whole tree removal during harvesting also increases the risk of N loss. Massive increases in Ca losses (held primarily in tree bark) place Ca at a very high depletion risk with whole tree harvesting. This system as with most plantations grown on sand soils relies on nutrients held in organic (biomass) pools to maintain site sustainability. This dependency on the retention of the organic nutrient pool was demonstrated on the sandy tropical soils in the Congo where a large proportion of cations (particularly Mg) were held in the residues and litter layer relative to the soil (Deleporte et al. 2008). Although N was also at greatest risk of depletion in that system, residue retention was advised as crucial to a sustainable supply of all elements.

9.4.6. Residue burning

Fire is an important residue management tool used to reduce wildfire risk through fuel load reduction and creation of fire breaks, but the use of intensive fires is viewed as an unsustainable practice that is advised against in South African plantation forestry management. Losses of cations may not be of major consequence under low intensity residue burning, provided leaching losses are reduced by planting soon after burning operations. The exclusion of high intensity fires to conserve N and carbon is far more relevant on the sandy soils in the Zululand ecosystem than the more buffered system described in du Toit (2006). Fire may yet be a valid management tool in this study area, provided it is not practiced every rotation on the same plantation compartment after felling and is carried out at a low burning intensity to reduce nutrient losses (Binkley 1986b; Neary et al. 1999; Guinto et al. 2001).

9.4.7. Whole tree removal

Whole tree removal or partial removal (branches and bark) may be practiced and used to provide biomass for energy production (Ghaffariyan et al. 2011). This is under investigation in the Zululand region due to high costs and unreliability of energy supply. Such practices undertaken on *Eucalyptus* plantation sites in Australia and have been deemed unsustainable though evidence of negative effects on productivity (Nambiar 2010). Whole tree removal has the potential deplete nutrients more rapidly than residue burning due to the large amount of Ca extracted with bark removal. Selective removal of bark or branches may not improve the situation given the

large nutrient quantities held in these tree components. If biomass in excess of stem wood is to be removed, the application of fertilisers containing additional base cations or soil ameliorants rich in base cations (such as pulp mill residue or boiler ash) should be investigated to maintain nutritional sustainability.

9.4.8. Leaching

The relevance of leaching to total nutrient losses diminished as nutrient removal increased. This was through smaller inputs to the soil solution from smaller decomposing residue loads remaining after residue burning or whole tree removal. Leaching is therefore a secondary loss mechanism that only becomes important where residues have been retained. The rapid reduction in soil moisture and drainage with tree growth can indicate that leaching losses may be reduced through earlier re-establishment of the next crop. Shortening the inter-rotation period may be used to reduce soil moisture sooner, thereby reducing leaching losses.

9.4.9. Nutrient budgets

Data presented here demonstrates the reliance of fast growing short-rotation *Eucalyptus* plantations on biological nutrient cycling to provide early tree growth nutrient demands. This is similar to trends given in Laclau et al. (2010b) for tropical *Eucalyptus* plantations in Congo, Brazil and Australia. Seven year budgets calculated for plantation sites given in Laclau et al. (2010b) gave far larger N deficits than this study, attributed to smaller atmospheric inputs. Cation budgets in this study were an order of magnitude more negative than the budgets in the above tropical sites. This was due to the long inter-rotation and the relatively shallow depth (1 m) to which leaching was recorded in this study. These studies also emphasised the importance of organic matter conservation and fertilisation for maintaining a balance with long-term nutrient losses in addition to providing shorter-term nutrient demands. This emphasises the importance of a short inter-rotation length to allow root growth to progress at a similar rate to the nutrient rich wetting front with soil depth Laclau et al. (2010b). This will promote nutrient uptake rather than deeper leaching losses.

9.4.10. Management options

The loss of every nutrients in this study cannot be completely reduced by changing a single management system or mechanism alone as each nutrient presented different sensitivities to harvesting, residue management practice and time. Magnesium and K losses were relatively insensitive to residue management, but seemed to depend rather on time after clearfelling and rainfall. Nitrogen losses were highly dependent on residue management, while Ca and P losses were dependant on the level of biomass removed during clearfelling. Losses of nutrients may therefore only be holistically reduced through a combination of changes in harvesting and residue management as well as the timing of subsequent re-establishment. This is provided that rotation lengths are not reduced as the weight of the inter-rotation losses increases as rotation length decreases.

9.4.11. Natural inputs

The decline in nutrient supply may be too slow to cause visible or detectable nutrient deficiencies. Where residues are retained, a large proportion N, K, and Ca lost through stem wood removal can be replaced through atmospheric deposition over a seven year rotation. Atmospheric deposition may therefore be maintaining the apparent sustained timber supply in the Zululand region under current residue retention management practices. However, atmospheric inputs of Mg were comparatively low and inadequate for replacing stem wood harvesting losses. The role of atmospheric inputs is also uncertain and may increase or decrease depending on pollution levels and legislative pollution controls. Losses due to residue burning and whole tree harvesting cannot be replaced through natural inputs.

9.4.12. Expansion of work

The data described in this thesis comprises part of a larger set of data collected under tropical and sub-tropical plantation forestry. Within the south African context it is the first study of its kind to holistically combine a number of pools and fluxes into a single study, albeit for a single site and species. It adds to the international dataset by describing nutrient pools and fluxes in an apparently more fertile system than those found in *eucalyptus* plantations in Congo and Brazil. Used within the context of such international datasets and an understanding of other South African plantation sites the findings of this study can be extended further. This will require collection of site classification (soils and biomass, etc.) and monitoring data (weather,

deposition, tree productivity, etc) and using a process based model of the major fluxes to extrapolate in space and time.

Further work is required to determine the magnitude of some of the key nutrient pools and fluxes across a wider range of sites throughout the Zululand Coastal region and the entire South African plantation forestry areas. These include atmospheric inputs, nutrient losses associated with different tree species, levels of biomass removal and fire intensities. Atmospheric inputs can be a major input into forest systems, but can also induce larger nutrient losses or create nutrient imbalances. A measure of soil nutrient supply potential needs to be established to determine the link between nutrient pool sizes determined through soil chemical properties used in this study and actual tree available soil nutrients. Deeper soil exploration is required to understand tree rooting depth and the nutrient supplying ability of soils with depth. Such work and work carried out in this thesis needs to be continued through long term monitoring to enable a real time indication of changes in the health of the plantation forest over many rotations. This must involve investigating other soils and sites representative of the variability found in Zululand and further into South African plantations. Means of site rehabilitation after nutrient depletions also need to be investigated. In addition, extrapolation of this budget to a seven year rotation may not adequately describe inter-year variability (Ranger and Turpault 1999). It does however describe the most representative stages of stand growth which include the inter-rotation and the mature growth phase.

9.5. CONCLUSION

The hypothesis of this thesis was accepted as nutrient flux processes and losses on this productive, but sensitive site were accelerated after clearfelling and through subsequent residue burning. This resulted in significant reductions in soil and biomass nutrient pools. High level losses from these pools cannot be replenished through natural addition processes.

The proportion of nutrient uptake from soil and organic material reserves is greatest during the period from planting to post canopy closure. It is reduced thereafter as internal retranslocation and redistribution and litterfall with the associated decomposition processes begin to provide the majority of tree nutrient requirements (Laclau et al. 2003a; Smethurst 2010). The inter-rotational and early stage of a short-rotation pulpwood plantation growth cycle represents the period of greatest opportunity for site manipulation. The most intensive silvicultural operations during this period can affect growth for the remainder of the rotation (Nambiar 2008; du Toit et al. 2010;

Smethurst 2010) , and possibly for subsequent rotations ([Nambiar, 2008](#)). As such, this period also presents opportunities to undertake more sustainable management practices. Silvicultural operations undertaken during this period have potential to impact on future site resource supply and productivity (Nambiar 2008).

This study helped to identify the most important processes that need monitoring in understanding nutrient supply and demand and nutritional sustainability on the tropical *Eucalyptus* sites in Zululand. Organic matter fluxes are important as these can easily be manipulated through harvesting and residue management resulting losses that can overshadow most other fluxes. Organic matter turnover is also a critical nutrient supply mechanism for early tree growth nutrient demands, especially on the unbuffered sandy Zululand soils. Atmospheric deposition is important at this study site as this is the only major natural input mechanism that can aid in maintaining site nutrient capital under conservative management practices. Leaching fluxes can become an important loss mechanism on sandy sites where inter-rotation lengths are prolonged. It is also important to note that soil erosion can be a major and unrecoverable loss mechanism on steep slopes under poor management practices.

The physical, chemical and biological characteristics of the sandy soils on the Zululand Coast may act as a severe limitation to future tree productivity given their low organic carbon and nutrient content, high drainage rates and generally limited resilience to nutrient loss. The short rotation lengths, high productivity and choice of management practices therefore have a crucial role to play in determining the sustainability of plantation forestry on this site and similar sites along the Zululand Coast. It may therefore be necessary to stipulate conservative stem-wood only harvesting practices on sensitive sites through a combination of:

- On site residue retention utilising stem wood only harvesting.
- Avoidance of high intensity harvest residue burning on low N soils.
- Rapid re-establishment of a new crop after clearfelling.
- Monitoring of site nutrient pools (cations in particular) and maintaining soil nutrient status. This can be done through fertilisation or through the co-application of alternative soil ameliorants such as pulp mill residue.

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