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Slash burning and the effects on nutrient dynamics and soil faunal compostition in an Eucalyptus grandis plantation in South Africa

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Slash burning and the effects on nutrient dynamics and soil faunal composition in an *Eucalyptus grandis* plantation in South Africa.

By

Ryan Leslie Nadel.

A dissertation submitted to the Faculty of Science, University of the Witwatersrand, Johannesburg, in fulfilment of the requirements for the degree of Master of Science.

School of Animal, Plant and Environmental Sciences,
Johannesburg, November 2005.

Declaration.

This thesis was supervised by Prof. M.C. Scholes and co-supervised by Prof. M.J. Byrne.
I declare that this dissertation is my own, unaided work. It is being submitted for the Degree of Master of Science in the University of the Witwatersrand, Johannesburg. It has not been submitted before for any degree or examination in any other University.

_____

Ryan Leslie Nadel.
November 2005.

Abstract.

Sustainability of commercial plantations is dependent on the conservation of soil nutrients, especially on the ancient, leached soils of the tropics. This is particularly important when limited fertilization is practiced. To assess the sustainability of commercial plantations, various sustainability indicators have been suggested. One such indicator is that of biodiversity. The measurement of soil faunal diversity is important as soil fauna play an active role in the cycling of nutrients through the decomposition of plant residues and organic matter. The diversity and different feeding activities of soil fauna influence the rate at which nutrient cycling occurs either through microbial grazing, faecal deposition, the mixing of litter with the mineral soil as well as through the spreading of microbial inoculum. Management practices may, however, negatively influence nutrient availability through losses associated with the burning and removal of slash residues, thus changing the soil faunal community composition. In *Eucalyptus grandis* plantations in South Africa, the site is usually burned following the harvest and prior to replanting, which may markedly disrupt soil faunal function and nutrient availability. Soil biological processes, nutrient loss and soil faunal composition were the focus of this study. Nutrient losses associated with fire and slash management practices were determined by comparing the quantity of nutrients lost from the slash that was either spread or piled following burning. The *in situ* nitrogen and phosphorus mineralization rates and soil faunal composition were measured in six randomly located plots, three burned and three unburned, prior to and after a low intensity fire. Results indicate that 78 – 99% of phosphorus, nitrogen and carbon were lost from the slash as a result of burning, with a greater percentage loss occurring from the slash that was spread. It was further shown that within the burned plots the nitrogen availability was enhanced four fold within the first month following the fire. Phosphorus availability was also enhanced ten fold as a result of the burning, however, these effects on nitrogen and phosphorus availability were short lived. Soil faunal diversity was low both prior to and after burning. Ants are the dominant soil fauna. Soil faunal composition was more markedly influenced by season than by fire, with millipedes occurring in all plots in spring, prior to the fire and ants dominating in summer, post fire.

To my family,
for all their love,
patience and support.

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Chapter 1.

Introduction.

Forestry is a multi million rand industry in South Africa with the forestry and forest products industry in 2002/2003 contributing R14.828 billion or 1.2% of the gross domestic profit (GDP) (F.S.A. 2005). The forestry industry relies on 1.331 million hectares (1.1% of South Africa's land cover) of productive exotic plantations for its survival (F.S.A. 2002). The majority of these plantations occur within the Mpumalanga and KwaZulu-Natal provinces of South Africa contributing to 40.2 % and 38.9% of the total forestry plantation area respectively (F.S.A. 2002). One of several, fast growing, exotic tree species planted in these plantations is *Eucalyptus grandis* Hill ex Maiden. occupying 280 823 hectares of plantation land in 2000 and grown predominantly for pulpwood and mining timber (F.S.A. 2002).

Forests have an important function as they provide goods, services and economic benefits to the population. Globally, the ever increasing demand for forest products has resulted in the need for the forestry industry to adopt the concept of sustainability (Sayer *et al.* 1997). The concept of sustainability has increased in popularity as public and scientific concerns over the fate of the earth have increased (Townsend *et al.* 2000). The term sustainability within a forestry context was stated by Sayer *et al.* (1997) as "maintaining or enhancing the contribution of forests to human well being, both of present and future generations, without compromising the ecosystem resilience, function and biological diversity." There are, however, many costs incurred by the forestry industry to maintain and ensure sustainability within their plantations (Spence 2001). It has thus become increasingly important that research in this field is done so as to guide forest management in implementing successful, cost effective, adaptive management techniques (Schoenholtz *et al.* 2000; Spence 2001).

The concept of sustainability within the forestry industry must be adopted so as to enhance and maintain tree growth into the future (Sayer *et al.* 1997). Tree growth is dependent on water, light and nutrient availability with certain management practices, such as fire and biomass removal, affecting the nutrient availability of a site (du Toit and Scholes 2002). The continual harvesting of vegetation and the use of various management practices, such as the use of fire, can lead to the

long term deterioration of both the soil nutrient pool quality and quantity if nutrients are not returned to the soil in the form of organic matter or fertilizers. The cycling of nutrients within the soil nutrient pool is, however, often enhanced by soil biodiversity, either directly through digestion of organic matter or indirectly by influencing microbial activity within the soil (Shaw *et al.* 1991). The evaluation of relationships between management practices and that of nutrient status and tree growth at a particular site will enable forestry managers to develop sound sustainable forestry management practices. The aim of this project was to improve our understanding of how fire affects both the nutrient dynamics and changes in the soil faunal composition associated with slash burning in an *Eucalyptus grandis* plantation.

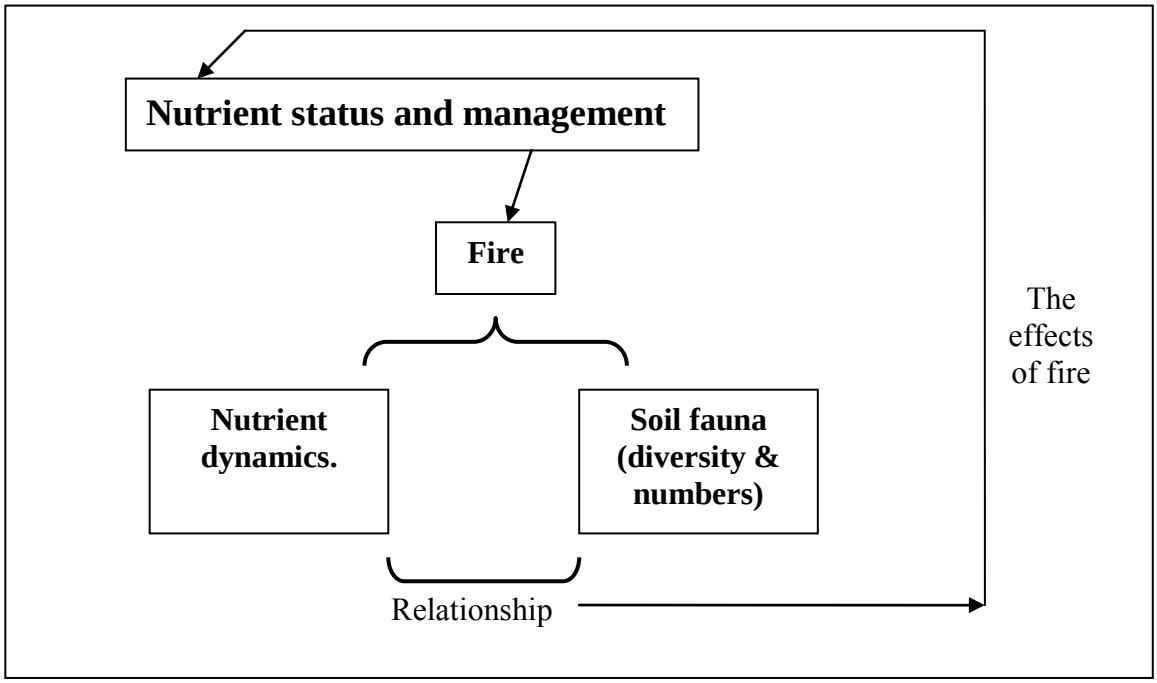


Figure 1.1: A conceptual diagram showing the link between the nutrient status of a site that can be controlled by management, through the use of fire as a management tool, and how it affects both the soil faunal and nutrient dynamics with their inter-relationship feeding back to affect the nutrient status of a site.

The following key questions were addressed in this study;

1. What quantity of nutrients (carbon, nitrogen, phosphorus, potassium, calcium, magnesium and sodium) are lost from slash following burning?
2. How do soil nitrogen mineralization rates and extractable soil phosphorus levels change with season in both the burned and unburned plots?
3. How does the soil faunal diversity change in the slash and mineral soil (0–30cm) in a felled *Eucalyptus grandis* plantation prior to and after burning?
4. What seasonal changes occur in the soil faunal diversity in the slash and mineral soil (0–30cm) in a felled *Eucalyptus grandis* plantation in both the burned and unburned plots?

Chapter 2.

Literature review.

Nutrient cycling processes involve an understanding of the quantity of material present in the system (pool size) as well as the rate of movement between the various pools also referred to as the flux (Figure 2.1). This study focussed on the interactions between fire, soil fauna and changes in the dynamics of nitrogen and phosphorus cycling. The loss of cations from the slash as a result of burning was also investigated.

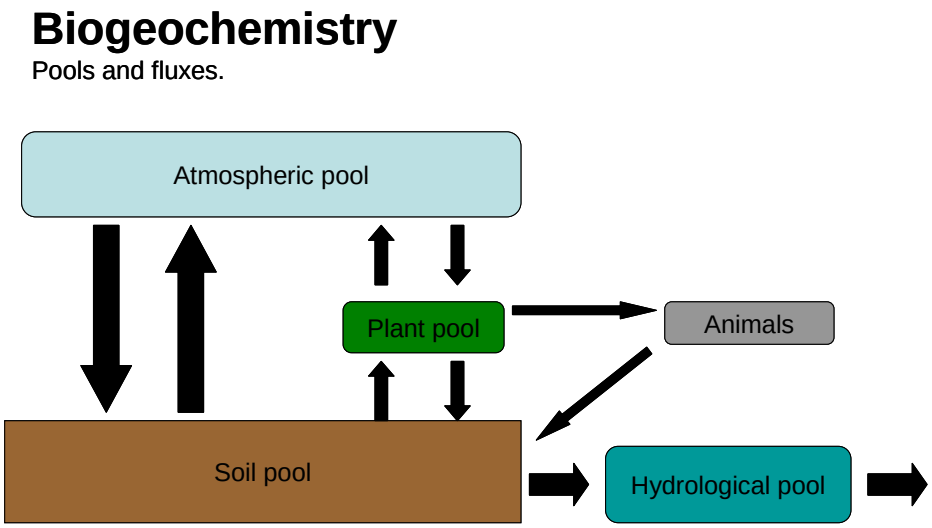


Figure 2.1: A conceptual diagram illustrating the concept of biogeochemistry with its associated pools and fluxes (Arrows represent fluxes between the various pools (blocks)).

2.1. Nutrient dynamics.

One of the major contributing factors for successful plant growth is ensuring that adequate quantities of plant nutrients are made available for plant uptake. These nutrients are primarily derived from weathered bedrock, the decomposition of plant and animal material, precipitation, as well as atmospheric deposition (Sumner 2000). These nutrients can broadly be classified into two groups namely macro and micro nutrients. These groups are classified based on the quantity of nutrients required by the plant for growth to occur. Macronutrients are those nutrients required

by the plant in large quantities whereas micronutrients are required in much smaller quantities (Thompson and Troeh 1978).

2.1.1. Macronutrients.

Macronutrients can further be classified depending on their origin. These nutrients may originate from mineral matter or from the decomposing soil organic matter (Thompson and Troeh 1978). The macronutrients are carbon, hydrogen, oxygen, nitrogen, sulphur, phosphorus, potassium, calcium and magnesium. The carbon, hydrogen and oxygen originate predominantly from both the air and water. These elements form carbohydrates during the process of photosynthesis and are a major component of all organic compounds formed. Nitrogen is one of the most limiting nutrients found within the soil and is required for the synthesis of proteins and vitamins (Miller and Donahue 1990). Nitrogen in the form of nitrates are obtained predominantly through decomposition of organic matter through a process known as nitrogen mineralization (Thompson and Troeh 1978). Calcium is used by the plant in the growth of new roots and stems, whereas magnesium is an essential component of chlorophyll (Miller and Donahue 1990; Adams and Attiwill 1991; Peverill *et al.* 1999). Potassium aids plants in the uptake and movement of several nutrients and is found in greatest concentration in rapidly growing tissue (Thompson and Troeh 1978; Miller and Donahue 1990; Marschner 1999). Phosphorus is an essential nutrient found in numerous organic compounds. Plants can, however, only take up this phosphorus in the form of inorganic phosphates (Marschner 1999).

These plant nutrients are stored and held within the soil organic matter and mineral soil on the negatively charged exchange sites that must undergo processes such as decomposition and mineralization prior to becoming available for plant uptake (Rab 1996; Neary *et al.* 1999). Nutrients that are locked away in the bedrock of a particular site must undergo the process of weathering so as to gradually make these locked away nutrients available to the plant (Thompson and Troeh 1978). The rate at which the mineralization, decomposition and weathering occurs is determined by chemical, physical and biological soil properties(Wilson *et al.* 2002; du Toit 2003). These soil properties are controlled by several factors such as soil temperature, moisture

and texture, geology and both the quantity and quality of organic matter that occurs at a particular site(Wilson *et al.* 2002; du Toit 2003).

2.1.2 Biological processes.

The availability of the nutrients, namely those of nitrogen and phosphorus, are dependent on the rate of mineralization before nutrients are made available for plant uptake. Microbial activity influences both the rate at which decomposition and nutrient mineralization occurs. The rate of microbial activity is, however, sensitive to disturbance and changes in soil physical and chemical properties particularly that of moisture and temperature (Morris 1986; Giardina and Rhoades 2001).

2.1.2.1 Nitrogen mineralization.

As nitrogen is essential for plant growth it has become one of the most thoroughly studied plant nutrients in the world (Palm *et al.* 1996a; Kamprath 2000). The majority of nitrogen is found in the soil organic matter (Power 1994). This organic nitrogen must undergo a process known as nitrogen mineralization prior to the formation of inorganic nitrogen that can be taken up by the plants for growth. During nitrogen mineralization the soil organic matter is first converted to ammonium through the process of ammonification controlled by ammonifying bacteria. Once ammonium is formed a small quantity is used directly by the plant whereas the majority is converted to nitrates through the process of nitrification, where bacteria in the soil convert ammonium into nitrite which is then oxidised by bacteria to form nitrates (Kamprath 2000; Sumner 2000).The uptake of nitrates enables the plant to produce proteins and nucleic acids thus enabling growth. Nitrogen, however, can be lost from the system as denitrifying bacteria convert some nitrates into nitrogen gas which is returned to the atmosphere (Figure 2.2).

Small losses of nitrogen may occur in the natural ecosystems if plant uptake of nitrate is not synchronous with the rate at which nitrogen is mineralized (Palm *et al.* 1996a). Nitrate is easily leached further down the soil profile where it becomes inaccessible for uptake by plant roots (Sumner 2000). In managed ecosystems, such as in an eucalyptus plantations, disruptions affect this natural synchrony and productivity of the ecosystem because only small quantities of

nitrogen are mineralized in comparison to that of the plants demand for nitrogen. Fire affects the quantity of organic matter remaining and thereby affects the potential rate of nitrogen mineralization that may occur.

A fire event, however, often stimulates the rate of nitrogen mineralization by affecting soil temperature and moisture conditions which play a fundamental role in determining the rate at which inorganic nitrogen is mineralized within an ecosystem (Maithani *et al.* 1998). Nitrogen mineralization rates have been shown to increase for up to 90 days following a fire event due to the heat produced from the fire, affecting the microbial activity in the soil (Neary *et al.* 1999; Wilson *et al.* 2002). This increase in the rate of nitrogen mineralization is coupled with an increase in the availability of ammonium and nitrate ions. The ammonium ion concentration within the soil increased three fold following a fire event (Vitousek and Matson 1985; Palm *et al.* 1996a; Weston and Attiwill 1996). Fire may, however, negatively affect the rate of nitrogen mineralization if there is insufficient organic matter or soil moisture to allow for the survival of soil micro organisms. The rate of nitrification is dependent on soil temperature and moisture conditions and has been found to decrease with decreasing soil temperature (Weston and Attiwill 1996).

As nitrogen mineralization is a crucial biological process providing inorganic nitrogen required for plant growth, the sustainable functioning of both managed and natural ecosystems is dependent on the maintenance of a rate of nitrogen mineralization equal to that required by a plant to sustain growth (Tiessen *et al.* 1994). A fire event usually results in large quantities of nitrogen been removed from the ecosystem in the form of plant and organic matter, without nutrient replacement occurring. The loss of these nutrients affects both the synchrony of plant nutrient uptake and the rate nitrogen mineralization in these managed ecosystems. Fertilizer application is therefore needed so as to ensure the availability and replacement of the nutrients that were removed (Scholes 2002).

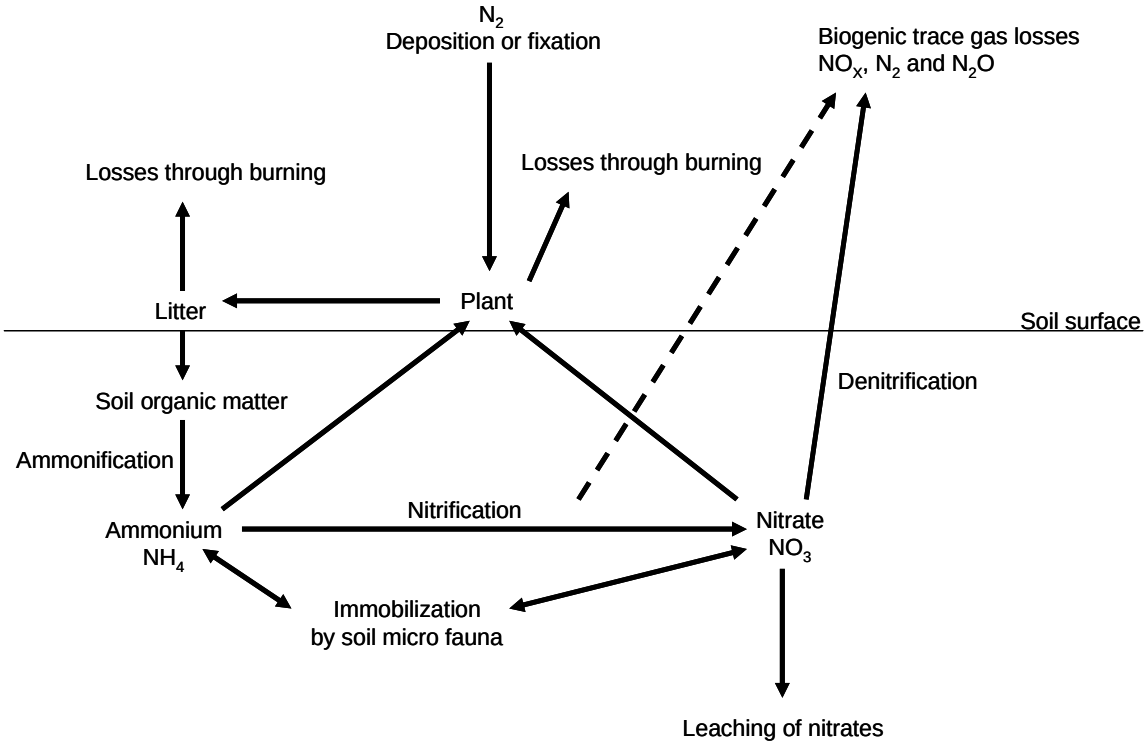


Figure 2.2: A conceptual diagram illustrating the nitrogen cycle.

2.1.2.2. Phosphorus mineralization.

One of the most growth limiting nutrients for plants is that of phosphorus (Sumner 2000). Phosphorus is an element required for plant growth and forms a vital component of plant DNA (deoxyribonucleic acid), RNA (ribonucleic acid) and ATP (adenosine triphosphate) (Coyne 1999). Phosphorus is found within the soil and can be divided into several fractions namely that of soil organic phosphorus, iron and aluminium bound phosphorus; calcium bound phosphorus and available phosphorus. The mineralization of phosphorus from the other phosphorus pools is an extremely slow process as it requires the weathering of soluble bedrock (Figure 2.3) (Sumner 2000). The cycling of the available pool of phosphorus within the organic matter is much simpler than that of the nitrogen cycle as there are fewer steps that are dependent on specific bacteria.

The pool of available phosphorus that can be taken up by the plant is, however, extremely small and usually immobilised within the litter and organic matter layer. The mineralization of this available pool of phosphorus occurs predominantly in the rhizosphere, the transition zone between the soil and the root (López-Gutiérrez *et al.* 2004). Only a minimal amount of phosphorus is made available for plant uptake when this available phosphorus pool cycles from the plant to the soil in the organic form of residues. These organic forms of phosphorus are then converted to inorganic phosphate through the decomposition process, prior to being made available to the plants for uptake (Figure 2.3)(Coyne 1999).

This rate of extractable (available for plant uptake) phosphorus mineralization occurs when phosphorus is released from the roots and soil organisms (López-Gutiérrez *et al.* 2004). Management practices such as fire increase the availability of phosphorus at a particular site due to the deposition of phosphorus from the ash on to the soil surface while increasing soil temperature and pH that will in turn stimulate microbial activity (de Ronde and Stock 1994; Ellis and Graley 1994; Romanyà *et al.* 1994; Geldenhuys *et al.* 2004). The persistence of this increase in available phosphorus has been found to be dependent on soil texture and acidity (Coyne 1999). Sandy soils allow for the easy leaching of available phosphorus further down the soil profile in comparison to more clayey soils that hold the phosphorus ions. The pH of the soil plays an important role in increasing the availability of phosphorus within the system as slightly basic soils allow for the release of phosphorus from the iron and aluminium bound phosphorus pool.

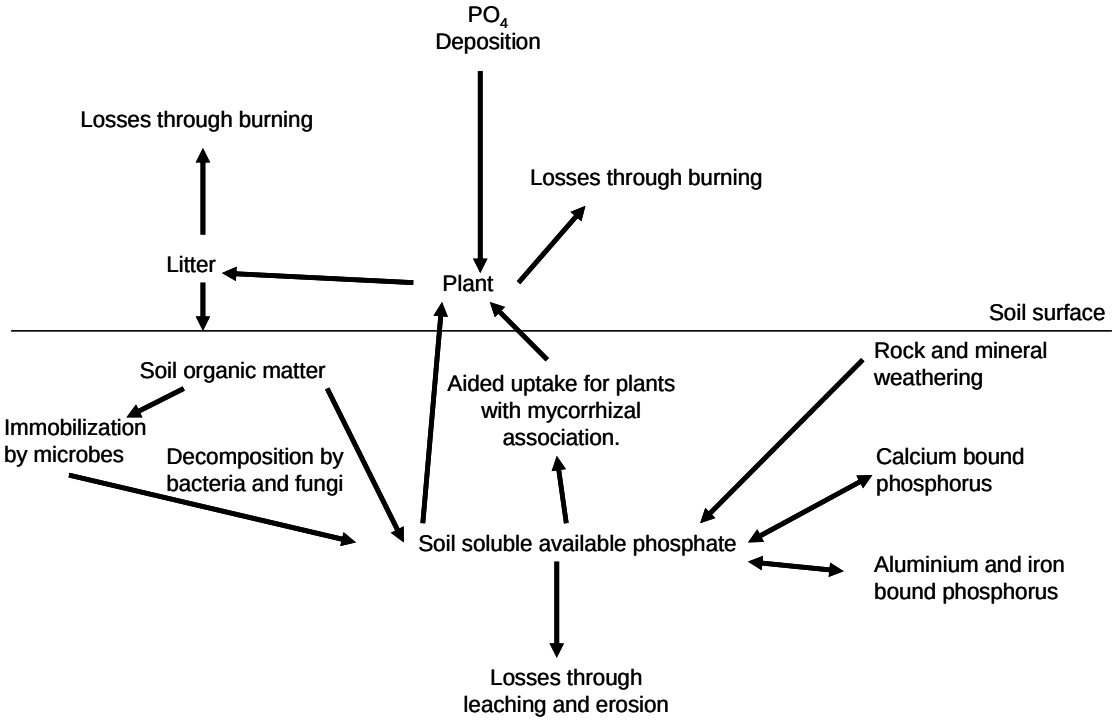


Figure 2.3: A conceptual diagram illustrating the phosphorus cycle.

2.1.2.3 Decomposition.

The cycling of plant nutrients from the plant detritus to the soil nutrient pool occurs during the decomposition process (Hou *et al.* 2005). The release of nutrients during this process are dependent on the seasonal cycling of nutrient immobilization and mineralization processes, controlled predominantly by temperature, moisture conditions, soil parent material and slash quality (Arunachalam *et al.* 1998; Hou *et al.* 2005). The quantity and quality of slash that decomposes to form this organic matter, determines the size of the soil organic matter available nutrient pool (Swift 1999).

Soil fauna aid in the decomposition of plant residues and in doing so release nutrients such as nitrogen into the mineral soil (Warren and Zou 2002). Soil and litter invertebrate fauna play an integral role in the movement and cycling of numerous plant nutrients, needed for growth, between the various nutrient pools found within the ecosystem (Decaëns *et al.* 2001; Susilo *et al.* 2004; Rossi and Blanchart 2005). Invertebrate soil fauna mobilise nutrients within the organic

matter and in so doing increase both the pool size and flux rate of these nutrients that enter and become available for plant uptake (Warren and Zou 2002; Parfitt *et al.* 2005). The release of these nutrients by soil fauna is either directly through digestion of organic matter or indirectly by influencing the microbial activity within the soil (Shaw *et al.* 1991; Lavelle *et al.* 1997).

2.2. Fire.

Fire in plantations is thought of quite differently to that of fire in natural systems. Fire can be used as a management tool for the burning of slash or otherwise plantations can be devastated by wild fires if a fire management policy has not been rigorously applied.

2.2.1 Prescribed burning.

2.2.1.1 Fire as a management tool.

The management of the inter rotational period (Figure: 2.4) in forestry plantations has been found to impact the ecological sustainability and productivity of a site (du Toit 2003). It has become vital for forest managers to understand the impacts that various management tools, such as fire, have on sustainability and productivity when used during this crucial period. Fire is an economic and simple management tool that can be used after felling, to remove plant residues for site preparation, or during the rotation as a fuel management tool (Figure 2.4) (Mackensen *et al.* 1996; Bird 2001; Fernández *et al.* 2004; Bird and Scholes 2005). Alternative methods of removing slash residues have been sought, however, when compared to mechanical land clearing options that impact the soil physical properties it has been suggested that the best possible system for removing slash was still that of burning (Alegre and Cassel 1996).

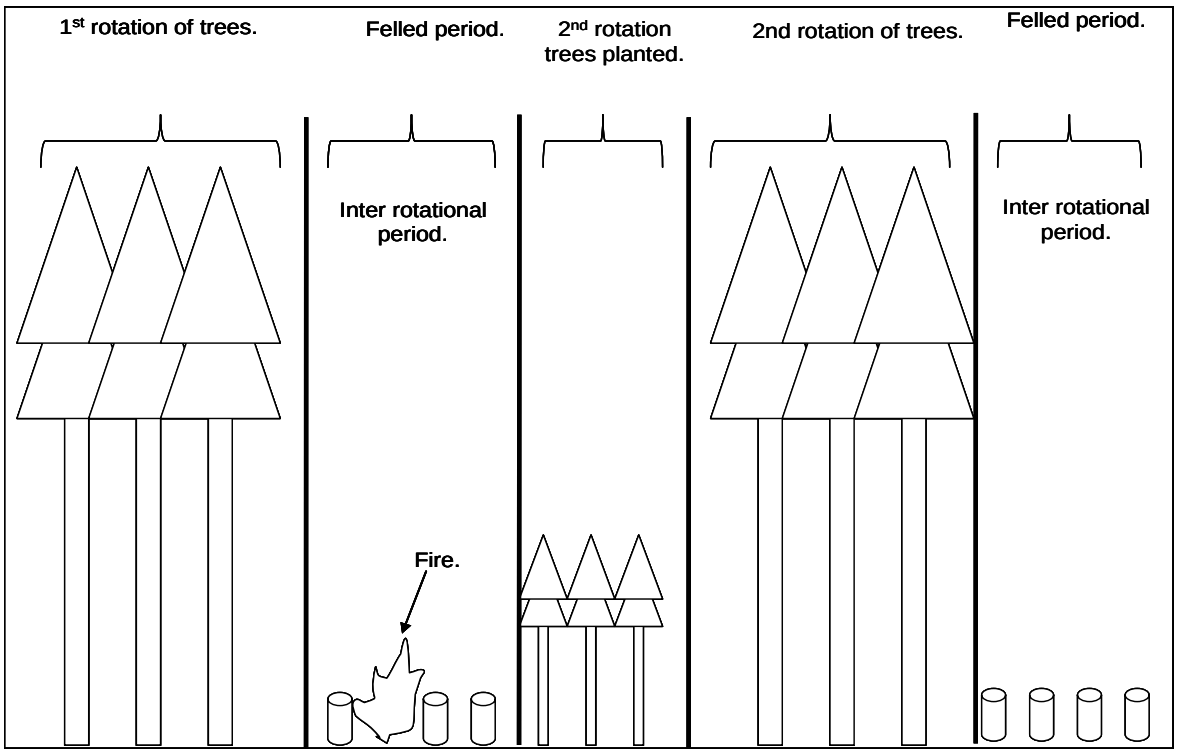


Figure 2.4: The rotational and felled (inter rotational) periods that occur within a plantation.

2.2.1.2 Fire intensity.

Fires vary in intensity and can broadly be classified into three categories, low, medium and high intensity. Forest managers use fires of different intensities when burning and managing slash and/or weeds within a plantation. The intensity of each fire is affected and partly determined by local weather conditions and slash to be burned. The intensity of a fire aids in determining the remaining pool size of nutrients needed for growth (Rab 1996; Gibbons *et al.* 2000). Recommendations in regulating fire intensity have thus been made to forest managers, including that slash should only be burnt under conditions when the slash is moist, decreasing the intensity of a fire (Fernández *et al.* 2004).

The burning of slash using low intensity fires causes fewer nutrient losses and changes in soil physical properties within the ecosystem in comparison to high intensity fires that are usually avoided due to their ability to change soil physical and organic matter properties (Rab 1996;

Fernández *et al.* 2004). These soil physical property changes include alterations in the soil aeration, bulk density and infiltration rates. High intensity fires remove the soil organic matter layer as well as changing the particle size distribution by reducing soil clay content while either increasing the silt or sand texture content, due to the loss of the top soil (Rab 1996; Gimeno-García *et al.* 2000). These changes in soil texture are, however, dependent on the initial soil texture, soil organic matter content, soil moisture content in addition to the duration and the absolute intensity of the fire (Rab 1996; Gibbons *et al.* 2000).

2.2.1.3 Effect of burning slash on soil processes.

Soil temperature is increased during the burning of slash, creating favourable conditions (if adequate moisture is available) for microbial activity (Palm *et al.* 1996b; Neary *et al.* 1999; Wilson *et al.* 2002). Soil ammonium concentrations within the soil are often enhanced by the increased rates of decomposition brought about by increased microbial activity, releasing ammonium and other plant nutrients such as phosphorus (Knoepp *et al.* 2004). The release of these nutrients after a fire is followed by their rapid uptake, by the plant, as they are required for growth (Christensen and Abbott 1989).

Increasing ammonium concentrations are often coupled with declining nitrate concentrations within the soil following a fire event, due to soil organic matter being transformed into ammonium at temperatures greater than 210°C (Gimeno-García *et al.* 2000). In contrast Jensen *et al.* (2001) found that soil nitrate concentrations increased significantly while soil ammonium levels remained constant or declined due to ammonium being rapidly nitrified.

2.2.2 Volatilization and loss of nutrients.

When slash is burned the decomposition process is skipped and nutrients are abruptly released from the slash (Jensen *et al.* 2001). Various factors, such as ambient temperature, moisture content and slash distribution, during the burning event influences the quantity of ash produced, nutrients lost and the degree of soil heating (Gimeno-García *et al.* 2000; Giardina and Rhoades 2001; Yang *et al.* 2005). The fire intensity influences the quantity of nutrients that are lost from

the site through volatilization (changing into vapour) and in smoke as particulate matter (Jensen *et al.* 2001; Tucker 2003; Knoepp *et al.* 2004).

Slash burning decreases the pool size of nutrients (nitrogen, sulphur, phosphorus and potassium) due to their volatilization during the burning process (Kato *et al.* 1999; Neary *et al.* 1999; Wilson *et al.* 2002; Fernández *et al.* 2004). Increased fire intensity increases nutrient losses with larger quantities of carbon, nitrogen and sulphur being lost in the smoke; being volatilized or become ash (Gimeno-García *et al.* 2000; Tucker 2003; Knoepp *et al.* 2004; Yang *et al.* 2005). Nitrogen has been found to be partially volatilized at temperatures exceeding 200°C and completely volatilized at 500°C (Jensen *et al.* 2001).

2.2.3. Ash deposition.

Not all nutrients are, however, lost from a system after a fire has occurred, as the burning of slash often leads to several plant nutrients, held within decomposing plant residues, being made available for plant uptake (Morris 1986). These nutrients become available for plant uptake as burning of the slash and organic matter releases certain of these nutrients to the soil from the remaining ash (Ludwig *et al.* 1998; Kato *et al.* 1999; Jensen *et al.* 2001; Wilson *et al.* 2002).

The availability of these previously inaccessible plant nutrients (particularly calcium, magnesium, phosphorus and to some degree potassium) increase in their pool size and mineralization rates within the top soil after a fire event has occurred, thereby often increasing the site's fertility for short periods of time (Morris 1986; Tucker 2003). The short lived increase in soil fertility after a fire event is due to the cations that initially enriched the ash, being either taken up by the plant or leached further down the soil profile (Kato *et al.* 1999). Phosphorus and various cations such as potassium, magnesium and calcium can also be depleted from the remaining ash if sufficient rainfall occurs, causing these nutrients to be leached down the soil profile (Harwood and Jackson 1975; Jensen *et al.* 2001; Knoepp *et al.* 2004).

2.2.4. Effect of fire on soil and litter invertebrate faunal composition.

The removal of plant residues through various disturbances has been suggested to affect the abundance and diversity of soil and litter fauna. Species richness was reduced within sites that had been burned, although spring fires had no effect on surface active arthropod species richness or community evenness when compared to that of unburned reference sites (Andrew *et al.* 2000; Decaëns *et al.* 2001; Collett 2003). In investigating the effect of fire on arthropod fauna within an indigenous *Eucalyptus* forest it was found that certain taxa took up to a year to recover to their original densities (Collett 2003).

It has, however, been suggested that species dominance and density are either positively or negatively effected by fire (Christensen and Abbott 1989; Parr *et al.* 2002). Dress and Boerner (2004) found that the frequency of the fire affected the soil faunal abundance in an oak- hickory forest. The abundance of soil fauna was reduced within annually burned forests whereas no significant difference in periodically burned sites occurred when comparisons were made with that of an unburned reference site (Dress and Boerner 2004).

2.3. Invertebrate soil and litter fauna.

To assess the sustainability of an ecosystem such as a plantation forest, various sustainability indicators have been suggested, including biodiversity (van Straalen 1997; Spence 2001). The decomposition of plant residue and organic matter is influenced by the activity of soil fauna, predominantly in the top 10cm of the soil surface (Swift 1999).

To aid in the monitoring of biodiversity, the soil invertebrate fauna can be classified into three categories based on size. The micro fauna are organisms less than 0.2mm in size; the meso fauna vary between 0.2 – 10mm and the macro fauna are those invertebrates that are greater than 1cm in size. The macro invertebrate soil fauna modify the above ground litter that eventually enters the soil (Brussaard 1998). Other than size, soil fauna can be classified into functional groups based on their feeding habits. Epigeic species are litter dwelling fauna that feed on the soil surface, anecic species are burrowers that remove litter from the soil surface due to their feeding

activities. The endogeic or soil feeding species live in the soil and feed on organic matter and dead roots (Anderson and Ingram 1996; Lavelle *et al.* 1997). In forestry plantations soil faunal composition and diversity are based on functional groups that vary over time, depending on both the organic matter quality and quantity (Lavelle *et al.* 1997). It is thus important to measure the species richness, community structure and functional feeding type of soil fauna, so as to determine the biodiversity and thus the health and sustainability of a particular site (van Straalen 1997; Spence 2001).

As soil arthropods play a key role in the terrestrial food web they are useful indicators to determine the impacts of forest management practices on ecosystem function as they have specialist feeding and habitat preferences (Greenberg and McGrane 1996).

2.3.1 The role soil and litter invertebrate fauna have in nutrient cycling.

The rate of nutrient cycling has an effect on the soil fertility and has been shown to either be maintained or enhanced by the presence of soil and litter invertebrate fauna at a particular site (York 2000; Decaëns *et al.* 2001; Tucker 2003; Dress and Boerner 2004). The enhanced flux size between soil organic matter and soil nutrient pools is due to the increase in decomposition of the soil organic matter. The decomposition process is often enhanced by soil structural changes, increased aeration and soil water infiltration, induced by the various activities of the soil and litter invertebrate fauna present in both the soil and organic matter. The mixing of soil and nutrients during the digestive process of both the soil and litter invertebrate fauna is another example of how the size of nutrient fluxes are enhanced (Bird *et al.* 2000; Reynolds *et al.* 2003). The grazing of certain soil invertebrate fauna on soil microbes often stimulates the microbial community indirectly and thus the rate of mineralization and flux of certain nutrients essential for plant growth (Bird and Chatarpaul 1986; Coyne 1999; Decaëns *et al.* 2001; Reynolds *et al.* 2003; Dress and Boerner 2004).

It is vital that sufficiently varied food types for the different soil and litter faunal functional groups based on their feeding habits are available in order to ensure optimal nutrient cycling. A diverse assemblage and compositional structure of soil and litter invertebrate fauna within an

environment usually ensures optimal faunal enhanced nutrient cycling. Disturbances, both natural and anthropogenic within a particular site disrupt the soil and litter invertebrate faunal assemblages, favouring certain functional groups or species. Changes to the soil and litter invertebrate faunal diversity present within a particular site, can fundamentally change the size of nutrient fluxes between the various nutrient pools found within a particular site (Bellocq *et al.* 2001).

2.3.2 Factors affecting arthropod diversity.

In South Africa many hectares of indigenous high altitude grasslands and indigenous forests have been converted to exotic, commercial forestry plantations. Such anthropogenic changes to the environment will disturb and change the soil and litter invertebrate fauna that are capable of surviving in such an environment (McGeoch 2002). As the minority of land in South Africa is under conservation protection it has become vital for land owners, such as the forestry industry, to play an active and crucial role of conserving existing biodiversity within their sites (McGeoch 2002). To aid in this aim particularly that of soil and litter invertebrate fauna diversity, it has become important to determine the effect that forestry management techniques have on diversity (Samways *et al.* 1996; Bird *et al.* 2000). It has been suggested that forestry management techniques should try to copy those of natural disturbances as species diversity is usually more ecologically resilient to such disturbances (Bellocq *et al.* 2001). It is crucial, however, to bear in mind when analysing such studies that both species and functional groups are affected differently by various forestry management practices and natural disturbances (Bird *et al.* 2000).

2.3.2.1 The effect of management techniques.

The abundance and diversity of soil fauna within managed ecosystems is dependent on certain management techniques such as the cropping system used, plant density and pesticide usage (Shah *et al.* 2003).The harvesting of trees and site preparation prior to replanting has been shown to affect the soil faunal species richness and diversity that occur within a particular site (Bird and Chatarpaul 1986; Bird *et al.* 2000; Bellocq *et al.* 2001; Decaëns *et al.* 2001; Rossi and Blanchart 2005). It is of importance to understand the effect that various silviculture management practices, have on the functioning and composition of soil and litter invertebrate faunal populations (Baker

1998). Research into soil faunal management worldwide has increased as researchers have recognised the importance of soil fauna in improving soil properties such as fertility and structure (Baker 1998).

Clear felling of a coniferous forest in Finland was found to affect abundance rather than species richness as the same species dominated both prior to and after clear felling (Huhta *et al.* 1967). Directly after the harvesting of trees there is a temporary increase in abundance before a longer lasting gradual decline in soil and litter invertebrate fauna (Huhta *et al.* 1967; Huhta 1976; Bird and Chatarpaul 1986). This effect has been attributed to the quantity of litter that remains within a particular site, providing a food source and refuge for the soil and litter invertebrate fauna, following the clear cutting (Huhta *et al.* 1967; Huhta *et al.* 1969; Huhta 1976; Bird and Chatarpaul 1986). Recently harvested sites with decaying wood and litter had a greater species richness and abundance of detritivores (Bellocq *et al.* 2001). After a short period the availability of this food source, however, decreases followed by similar pattern for both the soil and litter invertebrate fauna (Bird and Chatarpaul 1986). Huhta *et al.* (1969) found that several years following a clear felling event, various soil faunal species returned to their original levels whereas other species had lower abundances compared to that of the control sites.

In a Colombian savanna it was shown that after the burning of grass the species density reached levels achieved prior to the fire with a much lower species richness when compared to that of an unburned site (Decaëns *et al.* 2001). The recovery of soil and litter invertebrate fauna to levels prior to harvesting and litter removal, aided by fire, have been shown to take over twenty-seven years to achieve (Bird and Chatarpaul 1986). These low densities for many years following a fire have been suggested as being due to the availability of soil organic matter and species specific nutritional requirements from food sources such as fungus (Huhta *et al.* 1967; Bird and Chatarpaul 1986).

In studies looking at the effect of fire on ant diversity and abundance in an Australian blackbutt, *Eucalyptus pilularis*, forest a significant difference in ant species composition and assemblage occurred between the burned and unburned sites (Andrew *et al.* 2000; York 2000). Andrew *et al.*

(2000) and Shaw *et al.* (1991), however, both indicated that the burning of slash in Australian forests showed variable effects on the insect densities and diversity. Soil fauna that increase in density following a fire prior to a gradual decline is due to these species obtaining their nutrient requirements from micro-organisms aiding in the process of decomposition. An increase in the rate of bacterial growth within the soil surface as a result of an increase in pH and nutrient availability following a burning event has been found to increase the density of bacteria feed organisms (Huhta *et al.* 1967).

Shaw *et al.* (1991) indicated that having low intensity infrequent fires, with unburned plots in adjacent forests in close proximity, allowed soil invertebrate fauna to find refuges in these sites and in so doing showed minimal effects of fire and the burning of slash on arthropod densities and diversity. Unburned refuges found on adjacent plots allow source arthropod populations the chance to re colonise the plots that were burned (Horgan 2005; Hou *et al.* 2005). The rate at which re colonization occurred was noted as being determined by the intensity of the fire and the quantity of slash remaining after a burning event has occurred (Dress and Boerner 2004).

2.3.2.2 The effect of soil moisture and temperature.

In a *Eucalyptus pilularis* forest it has been found that plant residue, such as logs, stems and leaves are commonly used by soil fauna for shelter and serve as a refuge for their survival (Andrew *et al.* 2000). Lavelle *et al.* (1997) and Spence (2001) noted that trees and their litter of varying palatability and quality sustain significantly different invertebrate communities thus increasing the biodiversity of a particular site. Litter quality has been shown as being a good indicator for soil fauna density and diversity (Warren and Zou 2002). Monoculture forestry plantations severely impact the biodiversity that can occur at a site. Little difference in litter quality for soil fauna sustenance occurs within forestry plantations compared to the varying litter qualities that are found within natural ecosystems (Warren and Zou 2002).

Shaw *et al.* (1991) indicated that the management of plant residues influenced the life cycle and habitat of the soil fauna that in turn affected their survival, activity and thus their contribution to certain nutrient cycling processes. It was, however, further noted by Bengtsson *et al.* (1998) that

they had no knowledge of any study that has shown that increasing the amount of plant litter had an effect on the soil food web. It was found that decreasing the amount of plant litter had an effect on the diversity of the soil fauna, although when the plant litter was doubled no change in soil fauna were noted (Bengtsson *et al.* 1998).

Climatic and microclimatic conditions such as soil moisture and temperature were found to affect the movement of the soil and litter invertebrate fauna within the soil profile and played an important role in determining the soil and litter invertebrate faunal community (Bird *et al.* 2000; Sharon *et al.* 2001; Rossi and Blanchart 2005). Soil and litter invertebrate fauna present at a particular site have varying moisture and temperature requirements and also undergo seasonal variation in their position to be found within the soil profile (Huhta *et al.* 1967; Rossi and Blanchart 2005). Soil fauna have been shown to burrow within the soil profile during the dry months whereas within the wetter months the invertebrate fauna spent the majority of their time within the leaf litter layer (Sharon *et al.* 2001). Sharon *et al.* (2001) found that soil inhabiting invertebrate communities were related to soil textural classes and mineral properties within the mediterranean ecosystems only under similar climatic conditions.

The removal of the tree canopy affects the micro climatic conditions of a site as more radiant energy is able to penetrate the soil, and in so doing increase the variability of the soil temperature and moisture content (Bird and Chatarpaul 1986). Interestingly it was found that fire also affects the movement of soil invertebrate fauna within the soil profile as prior to the fire the majority of the invertebrate fauna were found in the litter and top layer of the soil, however, following the fire they were found further down the soil profile (Decaëns *et al.* 2001). Further reduction in species richness can occur due to the increased soil temperature conditions as well as the decrease in soil moisture. It has been suggested that higher soil temperature and dryness could possibly stimulate ant activity (Collett 2003). Certain species of soil and litter invertebrate fauna often thrive during periods of increased soil moisture as a result of increases in the quantity of nutrients to be leached into the soil following rainfall events (Bird and Chatarpaul 1986).

2.3.2.3 Comparison between indigenous vegetation and exotic plantations.

Several studies have focussed on determining the differences in soil and litter invertebrate fauna between indigenous vegetation and that of exotic plantations (Samways *et al.* 1996; Ratsirarson *et al.* 2002; Attignon *et al.* 2005). Leaf litter invertebrate fauna generally occur at a non significantly reduced species richness within exotic plantations when compared to that of indigenous vegetation (Samways *et al.* 1996; Ratsirarson *et al.* 2002). One should, however, be aware that no distinct differences could be determined when comparisons at the order and family levels were made. However, it was found that when comparisons were made at the lower species levels the distinct effects of exotic vegetation could be observed (Samways *et al.* 1996).

Ratsirarson *et al.* (2002) indicated that within the Newlands forests in Cape Town, ant species richness within an indigenous forest was not significantly different to that of a pine or an eucalyptus plantation. It was further reported that ant species richness within a pine plantation was slightly greater to that of an indigenous forest (Ratsirarson *et al.* 2002). As ants are not entirely dependent on leaf litter for their survival, leaf litter quality therefore did not appear to act as a determinant of species richness for a particular site (Ratsirarson *et al.* 2002). The authors, however, cautioned that one should bear in mind that indigenous forests provide a refuge to many of the rare species, compared to those that are found within exotic plantations (Ratsirarson *et al.* 2002). Attignon *et al.* (2005) found that termite species richness was significantly higher in semi deciduous forests when compared to that of teak plantations. Greater densities of termites were, however, encountered within the teak plantations due to greater densities of a predatory ant occurring within semi deciduous forests (Attignon *et al.* 2005).

2.3.3 The reason for measuring species diversity.

As soil and litter invertebrate fauna are sensitive to change and affect the rate of nutrient cycling, monitoring their composition prior to and after various silviculture management practices can act as an early warning system to indicate changes in site productivity, functioning and sustainability (Bird *et al.* 2000; Dale and Beyeler 2001; McGeoch 2002; Hamilton 2005). Measurements of soil faunal diversity aid in determination of conservation requirements for a particular site so as to ensure ecosystem functioning and sustainability (Horgan 2005). Particular species of soil and

litter invertebrate fauna, namely that of termites, ants and earthworms have been noted as being excellent indicators of soil fertility (Decaëns *et al.* 2001). Exotic, commercial forestry plantations are, however, often referred to as green deserts with the perception that minimal micro-faunal diversity is supported in such environments. Studies such as this one hope to determine whether such statements are true and to improve our understanding of how a low intensity fire, when used as a silviculture management tool, affect both soil and litter faunal community structure and abundance.

2.3.4 Diversity indices.

The measurement of species richness and diversity helps scientists to determine population and community dynamics within a particular site (Hamilton 2005). To aid in this challenging process of measuring species diversity, various diversity indices have been developed in conjunction with species richness, abundance and other diversity measurements (Hamilton 2005; Rossi and Blanchart 2005). Two of the most common non parametric measures of diversity that emphasise the species richness component of the diversity is that of the Shannon-Wiener and Simpson's index (Magurran 2004).

The Shannon-Wiener index measures the amount of order that occurs within a population and is an estimation of how difficult it would be to predict the next individual to be sampled within a population (Magurran 2004). The index value represents the degree of uncertainty in determining the next individual to be sampled. Larger values for this indicate a larger degree of uncertainty with no uncertainty represented by a value of 0 (Krebs 1989). The Simpson's index measures the heterogeneity and is regarded as one of the best dominance determining indices as it consistently ranks assemblages even if sample sizes vary in size (Magurran 2004). The probability that two individuals that are sampled within a population belong to the same species is determined by the diversity index, values that range from 0 to 1 with 0 representing low diversity estimates (Krebs 1989).

One should, however, be cautioned that for meaningful comparisons between treatments it is vital to identify trends in a population. This can, however, be difficult to achieve as numerous

indices require the number of samples and/ or individuals caught within a site or treatment to be standardised so as to avoid sampling size biases (Magurran 2004). It is therefore extremely important that one standardises all data sets to a common number of individuals for the purpose of comparing species richness between treatments or sites (Gotelli and Colwell 2001). This problem can, however, be alleviated through the use of the rarefaction curves to determine the species richness for smaller populations (Magurran 2004). Certain statistical software such as Colwell's EstimateS have the ability to calculate rarefaction curves in addition to various species richness and diversity indices such as the Shannon-Wiener and Simpson's index (Ratsirarson *et al.* 2002; Magurran 2004; Colwell 2005).

Chapter 3.

Materials and methods.

Site description:

This project was conducted at the CSP2 compartment located next to the Karkloof Project site, run by the Institute for Commercial Forestry Research (ICFR). This site is located in the eastern regions of South Africa in the KwaZulu - Natal Midlands close to the town of Howick (29° 24' S; 30° 12' E) (du Toit 2002). The site was first planted in the early 1960's with the 3rd rotation of *Eucalyptus grandis* being harvested in 2003. The site is located approximately 1260m above sea level with an underlying geology of dolerite. The mean annual temperature is approximately 15°C with an annual precipitation of 950mm. A detailed description of soil and weather conditions of the adjacent Karkloof project site has been published by du Toit (2003).

This project formed part of a larger project controlled by the ICFR that assesses the nutrient optimisation at the Karkloof trial cluster. In order to address the key questions outlined for this project an *Eucalyptus grandis* plantation was felled in September 2003 and was sampled throughout the duration of this project from September 2004. The larger project outlined by the ICFR consists of a total of thirty plots (on homogenous soil). Fifteen of these plots were burned in November 2004 and fifteen remained unburned (Figure 3.1). Of the fifteen plots in each category, twelve plots were fertilized and three plots were left unfertilized. The three plots that received no fertilization were sampled throughout the duration of this study. The burned plots were sampled in conjunction with the remaining unburned plots so as to determine the effect of fire as well as the re-colonization rates of soil fauna after a fire event.

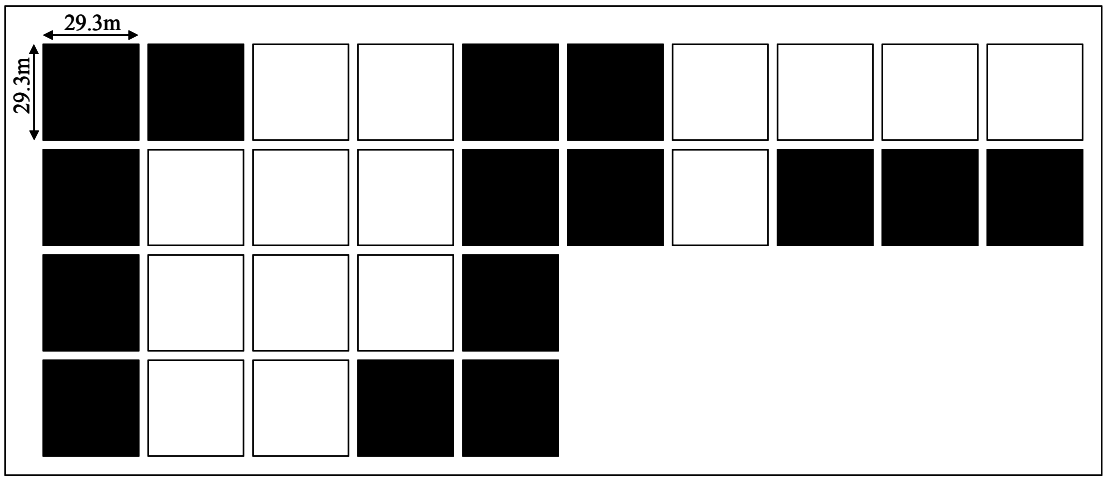


Figure 3.1: An illustrative diagram (not to scale) of a felled *Eucalyptus grandis* plantation consisting of thirty plots. The plant residues in fifteen of these plots were burned ■ with the remaining fifteen plot's plant residues remaining unburned.

When felling takes place the big logs are removed, the remaining slash can either be spread evenly over the felled area or it can be piled in stacks. At this particular site in each plot the slash was dealt with in both ways, part of it was distributed across the area, this will be referred to in this project as spread slash and the rest was piled into twelve 29.3m x 1m piles and will be referred to as piled slash (Figure 3.2).

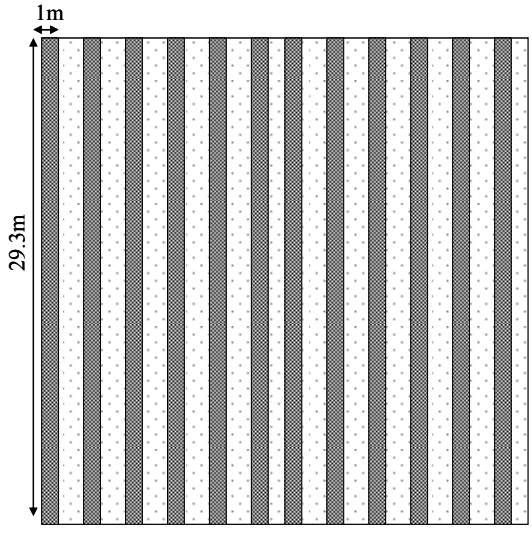


Figure 3.2: An illustrative diagram (not to scale) of each 858.5m^2 plot within a felled *Eucalyptus grandis* plantation. Slash within each plot was found to be spread in the open ⁝ or piled into twelve piles ▒ (each of 29.3m^2 in size).

Data collection:

Prior to burning of the plant residues:

Prior to burning, the potential fuel (plant residue) loads of only the spread slash for each of the plots of 858.5 m^2 were determined by randomly sampling each of these plots using five 0.25m^2 quadrats (Figure: 3.3). All potential fuel in each of the random quadrats was bagged and returned to the laboratory and sieved for arthropods. The residue was sorted into different size fractions, prior to being weighed to determine fresh mass. These fractions included leaves, stems with a diameter of greater than 5cm, stems with a diameter less than 5cm and bark. A sub sample of each of the fractions (± 25 grams) was separately weighed prior to and after being oven dried at less than 70°C to determine the percentage moisture. All oven dried fractions, were milled to 1mm and chemically analysed by the ICFR laboratories to determine the concentration of carbon, nitrogen, phosphorus, potassium, calcium, magnesium and sodium in each of the fractions for each quadrat sampled. The nitrogen concentration was determined by kjeldahl digestion and titration. Potassium, calcium, magnesium and sodium concentrations were determined by dry ashing and atomic absorption spectrometry. Dry ashing and segmented auto analysis were used to determine phosphorus concentrations. Details of the methods that the ICFR laboratories used have been published (Donkins *et al.* 1993a).

The slash pile fuel load, comprising predominantly woody material, was determined by weighing five randomly located slash piles located within the felled *Eucalyptus grandis* plantation. The different fractions namely leaves, bark, stems with a diameter greater than 5cm and stems with a diameter of less than 5cm, as for the spread slash, were individually weighed. Although the nutrient concentration for these slash pile fuel loads was not directly measured, it was assumed that the nutrient concentrations of the individual fractions were similar to that of the spread slash, originating from the same source.

To determine the fuel loads prior to the burning of the plots within the felled plantation, five ash pans were placed in each plot within the litter layer just above the moist humus layer of the spread slash, in the centre of a 0.25m^2 quadrat adjacent to the original quadrat sampled to determine the fuel loads (Figure: 3.3). An additional five ash pans were also placed randomly

above the moist humus layer beneath the slash piles. The pans allowed the ash from the burned plant residues to be collected for further analysis.

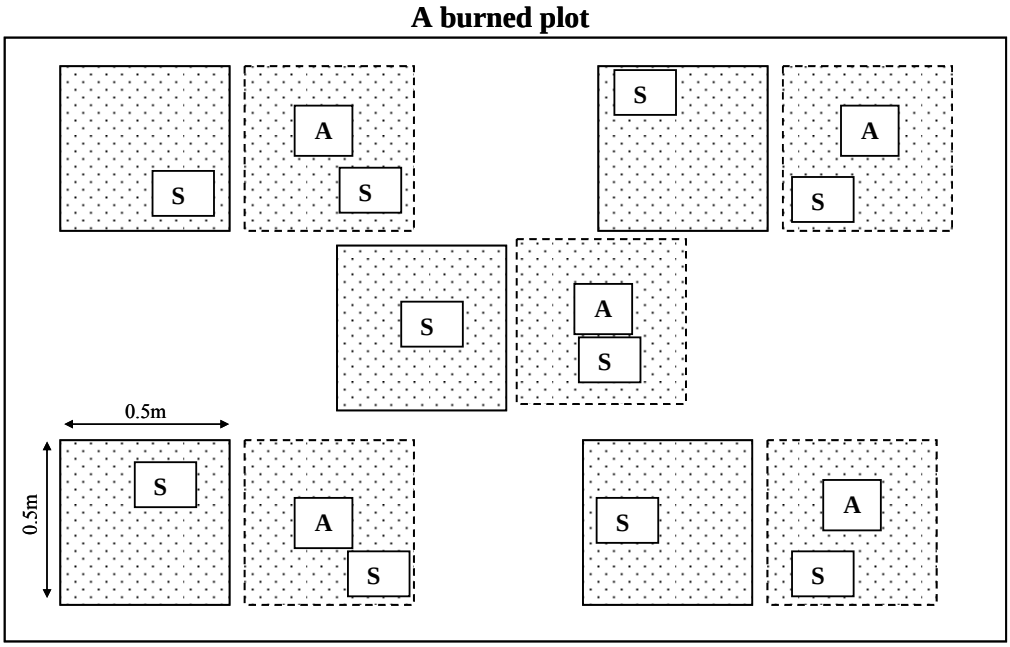


Figure 3.3: Illustrating how plant residues [ ], ash [A] and soil [S] monoliths were sampled within the spread slash inside a 0.25m^2 quadrat within the burned plots, each of 858.5m^2 in size, prior to and after burning.

After the burning of the plant residues:

Immediately after the burning of the plant residues, the ten ash pans that were placed in each of the burned plots were collected. The ash and unburned plant residue were once again sorted into different size fractions. Each of the fractions was separately weighed prior to and after being oven dried at less than 70°C. These measurements were used to determine the percentage combustion for each of the fractions. Samples of each of the oven dried fractions were chemically analysed by the ICFR laboratories for carbon, nitrogen, phosphorus, potassium, calcium, magnesium and sodium (Donkins *et al.* 1993a).

The quantity of collected slash and remaining ash and its nutrient concentration were quantified and scaled up to the area of either the spread or piled slash and were reported in kg ha^{-1}. The

results obtained for each of these elements were used for comparison with those acquired prior to burning so as to calculate the percentage loss of each of the nutrients, due to fire.

Soil nitrogen and phosphorus mineralization:

The *in situ* method of determining nitrogen and phosphorus mineralization was used within each of the burned and unburned sites throughout the duration of the study. In each of the burned and unburned plots a total of fifteen stainless steel cores (30cm x 5cm) were placed in the soil to a depth of 20cm. These cores were placed within each of the plots in groupings of three. For each of the six sampling periods (September, October, November, December, February, April) one of the cores in each of the groupings of three, was removed immediately ($time_0$) and analysed in the laboratory for nitrate (NO_3^-), ammonium (NH_4^+) and extractable phosphorus levels. Ammonium and nitrate concentrations of these soils were determined by extraction and colourimetric procedures (Anderson and Ingram 1996). Available phosphorus within the soil samples was determined by the ICFR laboratory using the Bray-2 extraction method (Donkins *et al.* 1993b).

For the remaining two cores in each grouping, one of the cores received no water and the other had 50ml of distilled water added to allow for the accumulation of nutrients due to enhanced microbial activity. These cores were then securely covered with plastic to prevent rain from entering the cores and leaching the nutrients during the incubation period. The remaining two cores in each of the groupings were allowed to incubate in the field for a period of time ($time_1$) before being removed and analysed in the laboratory for NO_3^-, NH_4^+ and extractable phosphorus.

The differences in nitrate and ammonium content in the soil between the two different sampling periods ($time_1 - time_0$) determined the rate of nitrogen mineralization within the soil. The actual nitrogen and phosphorus mineralization rates were determined by the difference in NO_3^-, NH_4^+ and extractable phosphorus at $time_0$ to the content at $time_1$ cores that received no additional water. The potential nitrogen and phosphorus mineralization rates were determined from the cores that received additional water.

Fauna within the slash:

Plant residues collected within the 50cm x 50cm quadrats were returned to the laboratory. These residues were sieved through a 2mm sieve, to collect fauna that occurred within the plant residue. The sieved plant residues were sorted into the different size fractions. A sub sample of the fractions (±25 grams) was separately weighed and placed in a Winkler extractor (Krell *et al.* 2005). The Winkler extractor aided in the extraction of litter invertebrate fauna from the plant residues placed in a gauze bag above a funnel and left to dry (Figure 3.4) (Krell *et al.* 2005). As the plant residues dried, in each of the Winkler extractors, the litter invertebrate fauna were attracted to the water found in a container beneath the gauze bags. The invertebrate fauna crawled through the ±5mm spaces of the gauze and fell down into the funnel into a collecting jar (Krell *et al.* 2005). These jars fauna were identified to family level, and functionally classified into feeding groups. Comparisons between the fauna collected prior to and after burning were then made.

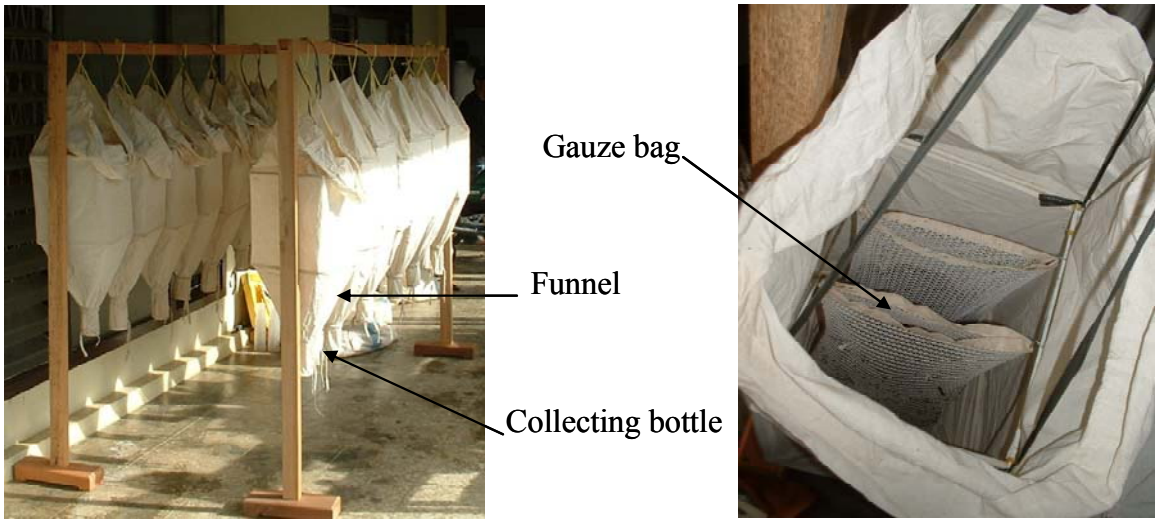


Figure 3.4: A Winkler extractor used to extract litter invertebrate fauna, with various components being indicated (photograph Jeroen Huising).

Fauna within the mineral soil:

To determine the soil fauna that occurred within the mineral soil, sweep netting and the excavation of five (25cm x 25cm by 30cm deep in the mineral soil) monoliths in the soil occurred for each of the burned plots, prior to and immediately after burning. The monoliths were excavated in the centre of each 50cm x 50cm quadrats used to sample the spread plant residues (Figure: 3.3).

For each of the monoliths the organic matter layer and the mineral soil, to a depth of 30cm, was removed in 15cm layers that was then hand sorted and sieved for all soil fauna with a body length greater than 2 mm in size (Anderson and Ingram 1996). All soil fauna collected from each of the soil layers were placed in collecting jars with the soil invertebrate fauna being classified to family level and into functional feeding groups.

The sweep netting method around each of the monoliths prior to sampling of each monolith allowed for the collection of the larger soil fauna that have the ability to move swiftly away from disturbances. For the sweep netting samples, 5 sweeps around each prescribed monolith occurred, with all invertebrate fauna caught, being classified and separated into functional groups.

Every alternate month from October 2004 to June 2005 the soil fauna in both the burned and unburned plots were sampled using the monolith and sweep netting methods as described above. Voucher specimens for all invertebrate fauna collected, throughout the duration of the study, have been sent for housing at the Natal Museum, Pietermartizburg, South Africa.

Data analysis:

The computer programs SAS Enterprise Guide 3 (SAS Institute Inc., 2004) and STATISTICA (StatSoft, version 6, 2002) were used for all statistical procedures. There were three true replicates with five pseudo-replicates for each treatment. These have been treated as fifteen replicates for statistical purposes. This was the only way in which sample size could be increased

due to limited plot availability in an operational forestry site. Prior to conducting parametric testing, data were tested for normality both between and within treatments. A repeated measures analysis of variance (ANOVA) and post ANOVA testing were done to test whether significant differences between soil nitrogen mineralization rates, phosphorus mineralization rates, ammonium, nitrate and extractable phosphorus occurred between the burned and unburned plots existed between time intervals. A multi factorial ANOVA (MANOVA) and post ANOVA testing was done to determine whether significant differences existed in nutrient concentrations before and after burning for the various litter fractions and ash sampled.

Non parametric Yates corrected Chi-square tests with Bonferroni adjustment were used to compare the total number and abundance of morpho-species between the different soil layers, humus and slash within the soil and between the burned and unburned treatment plots. A non parametric Wilcoxon rank test with Bonferroni adjustment was used to determine whether significant differences in species richness occurred between the burned and unburned plots. Shannon Wiener and Simpson diversity indices for each of the burned and unburned plots throughout the duration of the study were calculated using Colwell's EstimateS computer program (Colwell 2005). A t-test was used to determine whether a significant difference between the Shannon Wiener and Simpson Indices for the burned and unburned plots occurred.

Chapter 4.

Results.

The results section has been divided into three sections, with the first section of this chapter addressing the key question that involves the loss of nutrients following the burning of an *Eucalyptus grandis* plantation. The nutrient concentration of slash fractions and ash are tabulated separately prior to a summarised table of nutrient losses associated with burning being presented.

The second section focuses on the effect of fire and season on certain soil nutrient concentrations. The monitoring of these nutrient concentrations within the plantation over time allows for an understanding of concentration changes associated with fire and that of season. Soil ammonium, nitrate and extractable phosphorus concentrations are presented, as are nitrogen and phosphorus mineralization rates for the duration of the study.

Arthropod diversity and composition are addressed in the third section of this chapter. Arthropod location and species richness within the treatment plots for the duration of the study are presented prior to monitoring the presence of the dominant taxa. A table of the abundance of the morpho-species (taxa separated by morphological differences) caught and the diversity indices for each of the sampling months conclude the results section.

4.1 Quantity of nutrients lost from slash.

Table 4.1: The mean nutrient concentration of spread slash from a felled *Eucalyptus grandis* plantation in South Africa (± standard deviation; n=15). Different superscripts for each nutrient, indicates significant differences $P < 0.05$ between slash fractions.

		Concentration of nutrients (%)						
Residue.	Dry mass kg ha^{-1}	Carbon	Nitrogen	Phosphorus	Calcium	Magnesium	Potassium	Sodium
Leaves	981	50.01 ± 1.09^{a}	1.69 ± 0.23^{a}	0.10 ± 0.02^{a}	0.85 ± 0.21^{a}	0.21 ± 0.08^{a}	0.34 ±0.10^{a}	0.12 ± 0.05^{a}
Stems diameter:								
> 5 cm.	2046	46.80 ± 0.93^{a}	0.26 ± 0.14^{b}	0.03 ± 0.01^{b}	0.47 ± 0.34^{a}	0.07 ± 0.02^{b}	0.04 ± 0.03^{b}	0.01 ± 0.01^{b}
< 5 cm.	3667	47.10 ± 0.52^{a}	0.41 ± 0.13^{bc}	0.04 ± 0.01^{bc}	0.47 ± 0.13^{a}	0.09 ± 0.01^{b}	0.10 ± 0.08^{b}	0.03 ± 0.03^{b}
Bark	8740	45.06 ± 1.38^{a}	0.51 ± 0.10^{c}	0.04 ± 0.01^{c}	1.32 ± 0.80^{b}	0.11 ± 0.02^{b}	0.05 ± 0.01^{b}	0.01 ± 0.00^{b}

The quantities of several plant residue fractions, which remained following the felling of an *Eucalyptus grandis* plantation, varied between 981 and 8740 kg. ha^{-1} with leaves comprising the smallest and bark from the tree trunks the largest fraction. Leaves possessed a significantly greater ($F_{(21, 127)}$=19.12, $P < 0.001$) concentration of nitrogen, phosphorus, magnesium, potassium and sodium in their tissues compared to stems which had the lowest concentration of these nutrients. Calcium concentrations were significantly greater ($F_{(21, 127)}$=19.12, $P < 0.001$) within the bark compared to the stems with the lowest concentration. The carbon concentration within the various plant residue fractions varied between 45 – 50 % with bark having significantly less ($F_{(21, 127)}$=19.12, $P < 0.0001$) compared to leaves which had the greatest.

Table 4.2: The mean nutrient concentrations in ash remaining after burning of *Eucalyptus grandis* slash (± standard deviation; n=14). Different superscripts for each nutrient, indicates significant differences $P < 0.05$ between remaining ash of spread and piled slash.

		Concentration of nutrients (%)						
Residue.	Dry mass kg ha^{-1}	Carbon	Nitrogen	Phosphorus	Calcium	Magnesium	Potassium	Sodium
Ash from spread slash	1199	22.83 ± 10.29^{a}	0.63 ± 0.29^{a}	0.19 ± 0.19^{a}	13.87 ± 4.77^{a}	1.14 ± 0.54^{a}	0.41 ± 0.19^{a}	0.06 ± 0.03^{a}
Ash from piled slash	6240	7.45 ± 3.83^{b}	0.36 ± 0.22^{b}	0.14 ± 0.13^{b}	10.45 ± 2.72^{b}	1.17 ± 0.27^{b}	0.97 ± 0.57^{b}	0.18 ± 0.14^{b}

The quantity of ash remaining varied between 1199–6240 kg ha^{-1} with more ash occurring within areas that had slash piled prior to burning, compared to areas where slash had been spread out. Higher concentrations of carbon ($F_{(1, 20)}$=54.67, $P < 0.008$), nitrogen ($F_{(1, 20)}$=4.32, $P < 0.009$), phosphorus ($F_{(1, 20)}$=3.62, $P < 0.009$) and calcium ($F_{(1, 20)}$=53.44, $P < 0.009$) occurred within the ash from spread slash compared to the remaining ash from the slash that was piled. The concentration of magnesium ($F_{(1, 20)}$=24.26, $P < 0.007$), potassium ($F_{(1, 20)}$=12.22, $P < 0.007$) and sodium ($F_{(1, 20)}$=7.21, $P < 0.006$) was greater in the slash pile ash compared to the ash from spread slash.

Table 4.3: The estimated quantities of nutrients lost from slash due to burning in a felled *Eucalyptus grandis* plantation (brackets indicate percentage loss of the nutrient; n=14).

			Nutrient loss (kg ha^{-1})						
Treatment	Fuel load kg ha^{-1}	% loss	Carbon	Nitrogen	Phosphorus	Calcium	Magnesium	Potassium	Sodium
Spread slash	15434	92	6737	74	5	38	7	9	3
			[95]	[91]	[82]	[24]	[42]	[73]	[85]
Slash in pile	78911	92	36586	528	31	-39	17	33	19
			[99]	[96]	[78]	[-6]	[18]	[32]	[59]

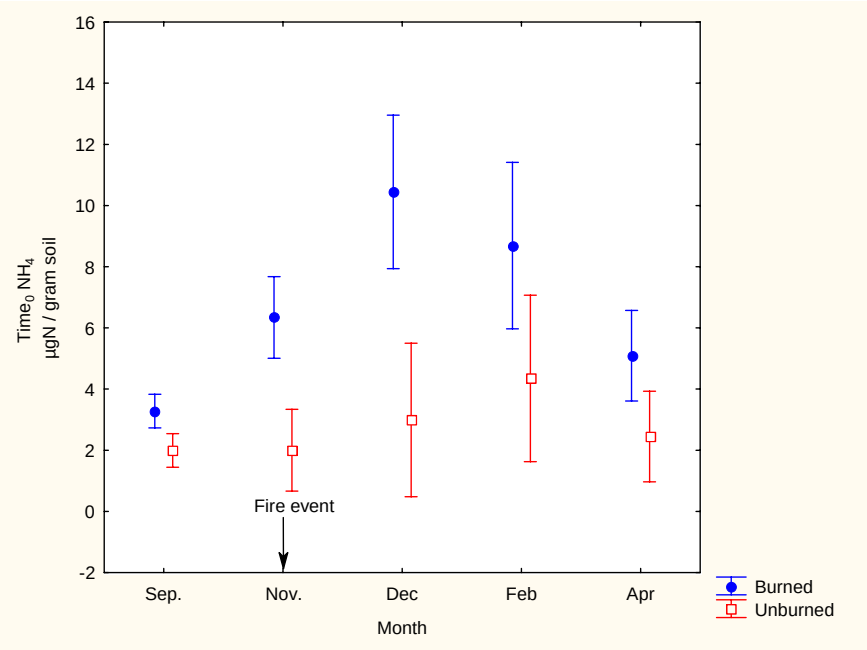
A five fold difference in the quantity of spread slash compared to that of slash distributed in piles was noted. The spread slash comprised approximately of 15 tons ha^{-1} of plant residues in comparison to the 78 tons ha^{-1} of slash that was piled. Biomass losses as a result of the fire were approximately 92% for both treatments. The percentage loss of various nutrients (carbon, nitrogen and phosphorus) was similar between treatments although in general a greater quantity of these nutrients were lost from the piled slash ($F_{(1, 12)}$=3130, $P < 0.0002$). Potassium, magnesium and sodium losses were significantly less ($F_{(1, 12)}$=3130, $P < 0.0002$) from piled slash to that of spread slash. A gain of 39 kg ha^{-1} of calcium occurred when slash was piled whereas losses of 38 kg ha^{-1} occurred in slash that had been spread.

Summary:

- The fuel loss was the same (92%) for both piled and spread slash with 78– 99% of phosphorus, nitrogen and carbon being lost from the slash as a result of burning. Such a large nutrient flux from the plantation may affect the nutrient sustainability of the site depending on the frequency and intensity of burning.
- Less potassium, sodium and magnesium were lost from the slash piles when compared with spread slash. However, calcium concentrations increased in the ash from the slash piles. These data indicate that management of slash affects the loss of nutrients due to fuel compaction, aeration and moisture.

4.2 Soil nutrient dynamics.

4.2.1 Ammonium

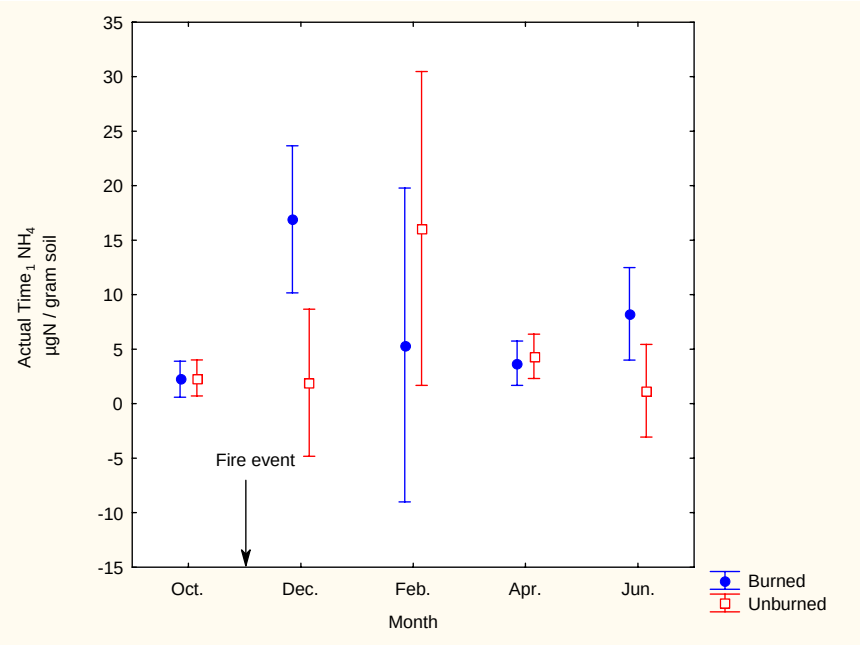


Repeated measures ANOVA
Month x Treatment: $P = 0.03$

		Treatment	
		Burned	Unburned
Month	Sep.	a	a
	Nov.	b	a
	Dec.	b	a
	Feb.	b	a
	Apr.	a	a

Figure 4.1: $Time_0$ soil ammonium concentrations in a felled *Eucalyptus grandis* plantation, over an eight month period in plots where the slash had either been burned or remained unburned (vertical bars represent 95% confidence levels; n=15 per month/treatment). Different lower case lettering indicates significant differences $P < 0.05$ between treatments for each specific month and between months within a treatment.

Figure 4.1 shows that greater concentrations of $Time_0$ ammonium were present in the burned plots throughout the duration of the study. Ammonium levels were highest in the burned plots in December and lowest within the unburned plots during September. There was no significant difference between the treatment plots in September, prior to burning. Significant differences for ammonium $Time_0$ concentrations existed between the burned and unburned plots $F_{(1, 28)}$=111.40, P=0.0001 and between the various months sampled $F_{(4, 112)}$=6.5052, P=0.0001.

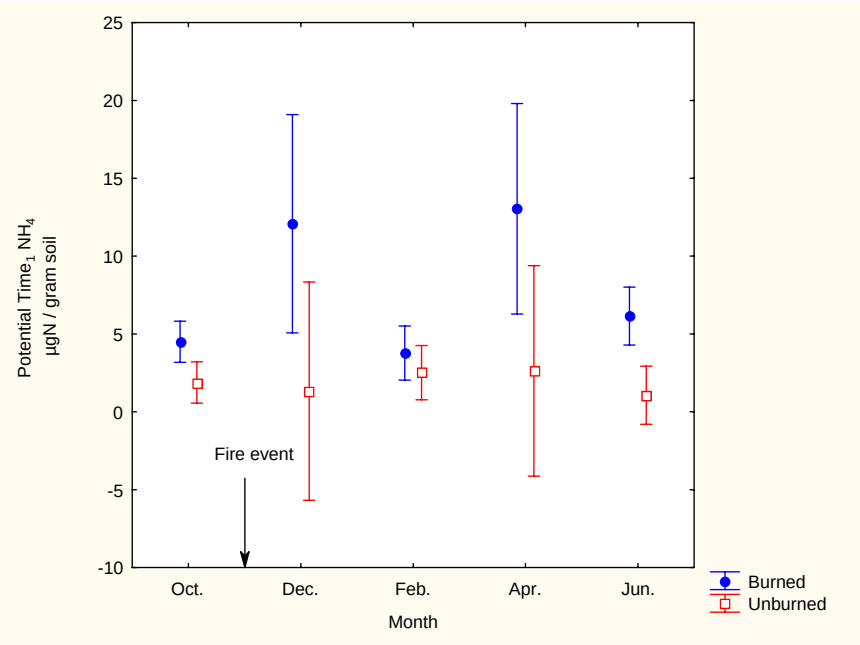


Repeated measures ANOVA
Month x Treatment: P = 0.09

		Treatment	
		Burned	Unburned
Month	Oct.	a	a
	Dec.	a	a
	Feb.	a	a
	Apr.	a	a
	Jun.	a	a

Figure 4.2: Actual $Time_1$ soil ammonium concentrations in a felled *Eucalyptus grandis* plantation, over a nine month period in plots where the slash had either been burned or was unburned (vertical bars represent 95% confidence intervals; n=15 per month/treatment). Different lower case lettering indicates significant differences $P < 0.05$ between treatments for each specific month and between months within a treatment.

Actual $Time_1$ soil ammonium concentrations were highest in the burned plots in December and lowest in the unburned plots during June. Unburned plots experienced a peak in ammonium concentration during February and reverted back to previous levels for the duration of the study in April. In October, prior to burning, there was no significant difference in actual $Time_1$ ammonium concentrations between the treatments. No significant differences occurred between the various treatments $F_{(1, 28)}$=0.76958, P=0.38 and sampling events $F_{(4, 112)}$=2.0566, P=0.09 throughout the duration of the study.



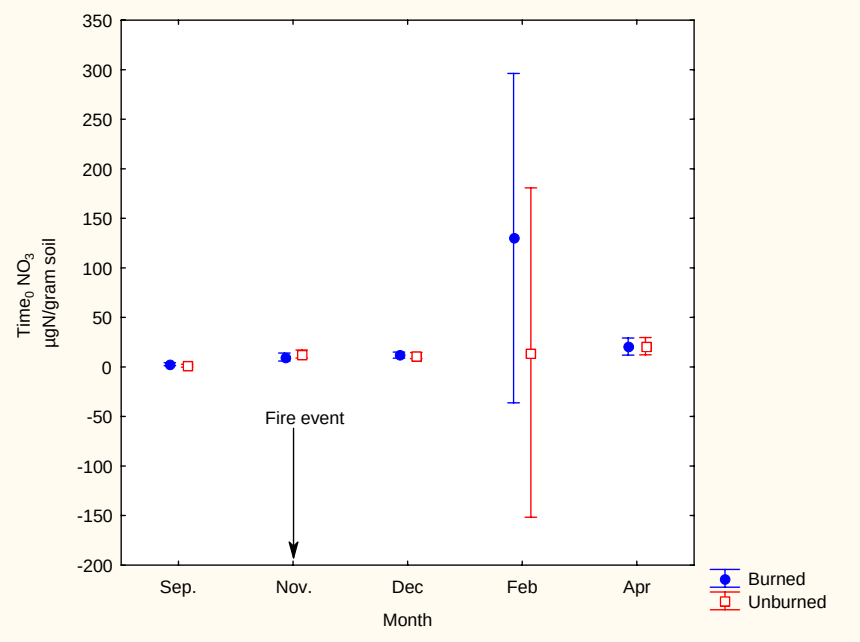
Repeated measures ANOVA
Month x Treatment: $P = 0.04$

		Treatment	
		Burned	Unburned
Month	Oct.	ab	ac
	Dec.	ab	ac
	Feb.	a	ac
	Apr.	b	bc
	Jun.	ab	c

Figure 4.3: Potential $Time_1$ soil ammonium concentrations in a felled *Eucalyptus grandis* plantation, over a nine month period in plots where the slash had either been burned or remained unburned (vertical bars represent 95% confidence intervals; n=15 per month/treatment). Different lower case lettering indicates significant differences $P < 0.05$ between treatments for each specific month and between months within a treatment.

The potential $Time_1$ ammonium concentrations that occurred within the unburned plots were similar throughout the duration of the study. There was no significant difference, prior to burning, between the treatment plots in October. A significant difference $F_{(4, 112)}=2.6802$, $P=0.035$ between the sampling months was revealed. A significant difference between burned and unburned plots $F_{(1, 28)}=9.0972$, $P=0.005$ occurred with the highest ammonium concentrations occurring within the burned plots and lowest in the unburned plots throughout the duration of the study.

4.2.2 Nitrate.

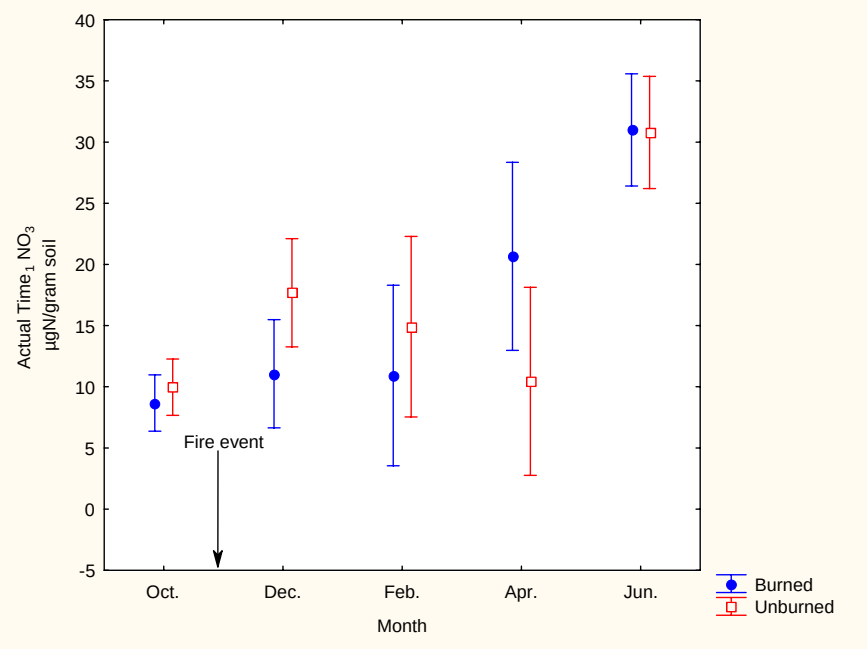


Repeated measures ANOVA
Month x Treatment: $P = 0.4$

		Treatment	
		Burned	Unburned
Month	Sep.	a	a
	Nov.	a	a
	Dec.	a	a
	Feb.	a	a
	Apr.	a	a

Figure 4.4: $Time_0$ soil nitrate concentrations in a felled *Eucalyptus grandis* plantation, over an eight month period in plots where the slash had either been burned or remained unburned (vertical bars indicate 95% confidence intervals; n=15 per month/treatment). Different lower case lettering indicates significant differences $P < 0.05$ between treatments for each specific month and between months within a treatment.

Figure 4.4 shows that $Time_0$ nitrate concentrations in the soil had a greater variation within the burned plots in February and smallest within the unburned plots in September. Up to an eleven fold increase in nitrate concentration occurred within the burned plots in February compared to that of the same plots in December. In September, prior to burning, there was no significant difference in soil nitrate concentrations between the two treatments. $Time_0$ nitrate concentrations were similar for all months sampled except for February. Statistical analyses revealed that no significant differences occurred between sampling months $F_{(4, 112)}=1.1881$, $P=0.32$ and the two treatments $F_{(1, 28)}=0.96395$, $P=0.33$.

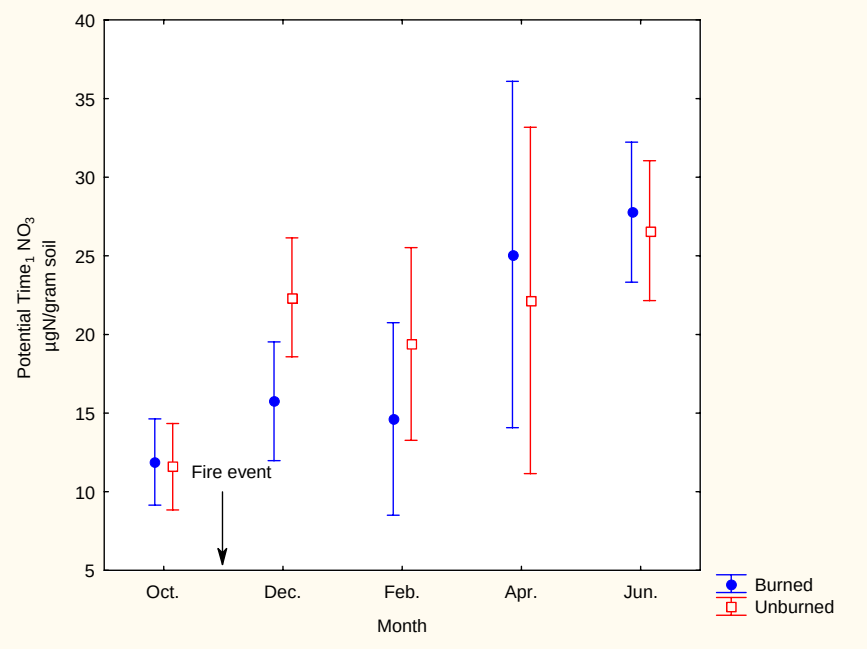


Repeated measures ANOVA
Month x Treatment: $P = 0.04$

		Treatment	
		Burned	**Unburned**
Month	Oct.	a	a
	Dec.	a	a
	Feb.	a	a
	Apr.	ab	a
	Jun.	b	b

Figure 4.5: Actual $Time_1$ soil nitrate concentrations in a felled *Eucalyptus grandis* plantation, over a nine month period in plots where the slash had either been burned or remained unburned (vertical bars represent 95% confidence intervals; n=15 per month/treatment). Different lower case lettering indicates significant differences $P < 0.05$ between treatments for each specific month and between months within a treatment.

Within all plots sampled the minimum actual $Time_1$ soil nitrate concentration was measured during October with the maximum in June. Prior to burning there was no significant difference between the burned and unburned treatment plots. The burned plots' nitrate concentrations gradually increased throughout the duration of the study compared to the variable concentrations found within the unburned plots. There was no significant difference between burned and unburned plots $F_{(1, 28)}$=0.03113, P=0.86 although a significant difference did occur between the individual months sampled $F_{(4, 112)}$=17.828, P=0.0001.



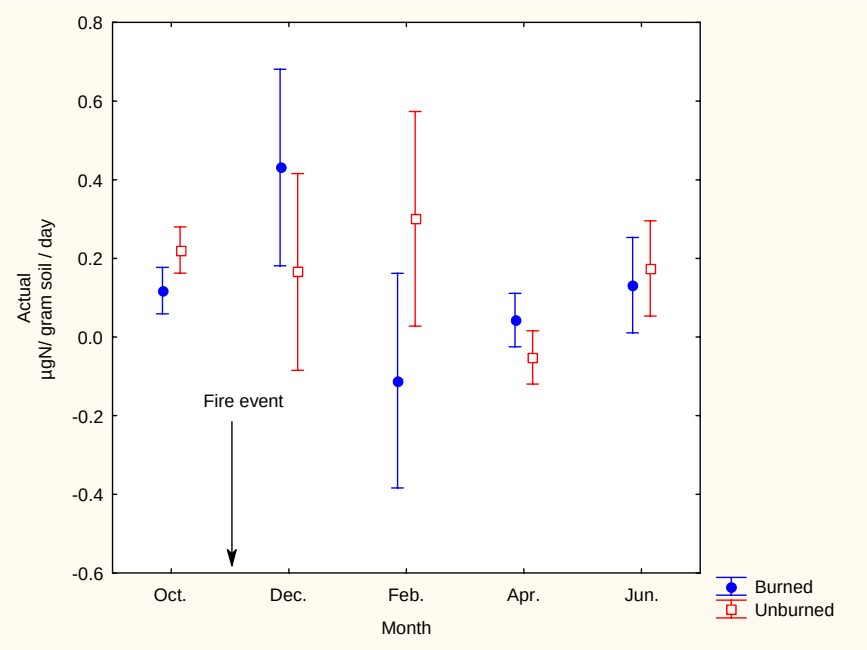
Repeated measures ANOVA
Month x Treatment: $P = 0.37$

		Treatment	
		Burned	**Unburned**
Month	Oct.	a	a
	Dec.	a	a
	Feb.	a	a
	Apr.	b	a
	Jun.	b	b

Figure 4.6: Potential $Time_1$ soil nitrate concentrations in a felled *Eucalyptus grandis* plantation, over a nine month period in plots where the slash had either been burned or remained unburned (vertical bars indicate 95% confidence levels; n=15 per month/treatment). Different lower case lettering indicates significant differences $P < 0.05$ between treatments for each specific month and between months within a treatment.

The lowest concentration of $Time_1$ potential nitrate in the soil was measured in October with the highest during June. There were no significant differences $F_{(1, 28)}=0.29499$, $P=0.59$ between the treatments throughout the duration of the study although a significant difference $F_{(4, 112)}=9.1014$, $P=0.0001$ in potential nitrate concentrations between the sampling months did occur.

4.2.3 Nitrogen mineralization rates.

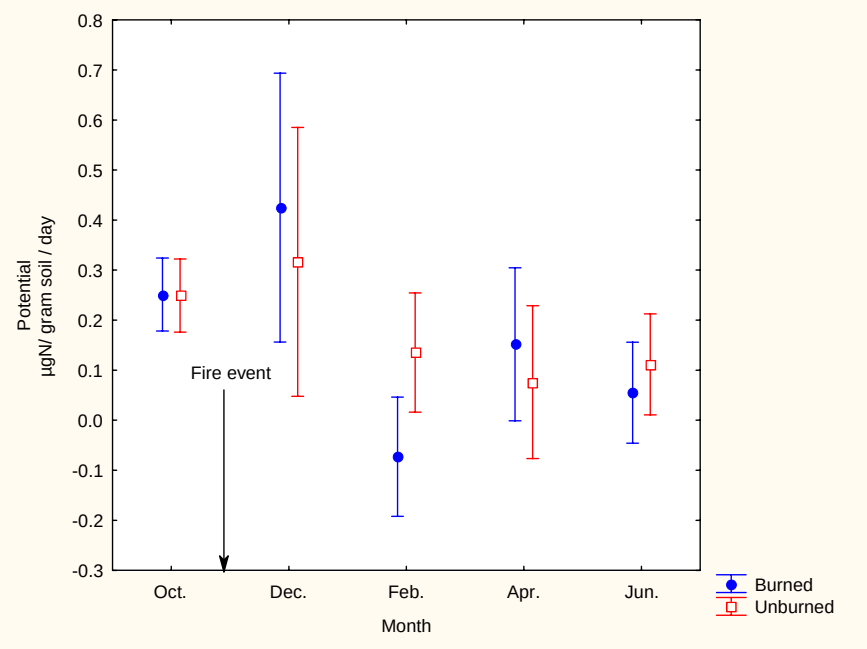


Repeated measures ANOVA
Month x Treatment: $P = 0.01$

		Treatment	
		Burned	Unburned
Month	Oct.	ab	ab
	Dec.	a	ab
	Feb.	b	a
	Apr.	ab	b
	Jun.	ab	ab

Figure 4.7: Actual nitrogen mineralization rates in a felled *Eucalyptus grandis* plantation over a nine month period in plots where the slash had either been burned or remained unburned (vertical bars indicate 95% confidence intervals; n=15 per month/treatment). Different lower case lettering indicates significant differences $P < 0.05$ between treatments for each specific month and between months within a treatment.

Figure 4.7 shows that prior to burning there was no significant difference in nitrogen mineralization rates between the treatment plots. The mineralization of nitrogen within the burned plots increased four fold within the first month following burning, however, this was not significantly different. The rate of nitrogen mineralization decreased to its lowest level within the burned plots in February. Similar nitrogen mineralization rates were measured for both the burned and unburned plots during October, April and June. There were no significant differences $F_{(1,28)}=0.54941$, $P=0.46$ in the rate of nitrogen mineralization between the burned and unburned plots throughout the duration of this study. Although a significant difference $F_{(4, 112)}=3.1572$, $P=0.01$ in the rate of nitrogen mineralization across the nine months exists.



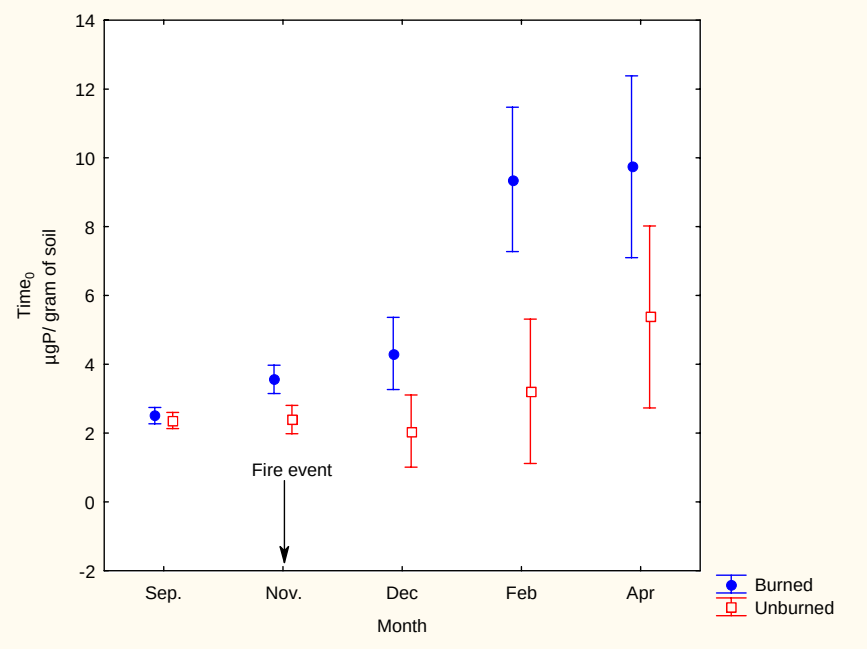
Repeated measures ANOVA
Month x Treatment: $P = 0.27$

		Treatment	
		Burned	Unburned
Month	Oct.	ab	ab
	Dec.	a	ab
	Feb.	b	ab
	Apr.	ab	b
	Jun.	b	ab

Figure 4.8: Potential nitrogen mineralization rates in a felled *Eucalyptus grandis* plantation, over a nine month period in plots where the slash had either been burned or remained unburned (vertical bars indicate 95% confidence intervals; n=15 per month/treatment). Different lower case lettering indicates significant differences $P < 0.05$ between treatments for each specific month and between months within a treatment.

The potential rate of nitrogen mineralization was highest within the burned plots in December and lowest in the same plots in February. No significant difference in the rate of nitrogen mineralization was found between treatments, prior to burning, in October. The rate of potential nitrogen mineralization was greater in the unburned plots compared to that of the burned plots during February. Comparisons between burned and unburned plots revealed no significant difference $F_{(1, 28)}=0.11415$, $P=0.74$ although a significant difference $F_{(4, 112)}=6.2836$, $P=0.001$ was noted between the sampling months for the burned plots.

4.2.4 Extractable phosphorus.

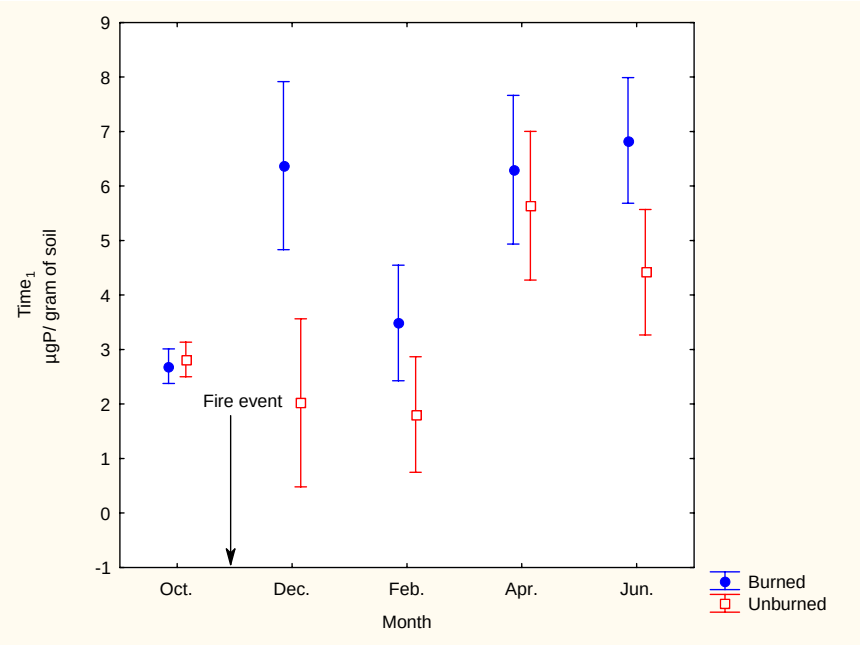


Repeated measures ANOVA
Month x Treatment: $P = 0.0001$

		Treatment	
		Burned	Unburned
Month	Sep.	a	a
	Nov.	a	a
	Dec.	a	a
	Feb.	b	a
	Apr.	b	a

Figure 4.9: $Time_0$ soil extractable phosphorus concentrations in a felled *Eucalyptus grandis* plantation, over an eight month period in plots where the slash had either been burned or remained unburned (vertical bars represent 95% confidence intervals; n=15 per month/treatment). Different lower case lettering indicates significant differences $P < 0.05$ between treatments for each specific month and between months within a treatment.

$Time_0$ soil extractable phosphorus concentrations increased exponentially over time within the burned plots. In September, prior to burning, there was no significant difference in extractable phosphorus concentrations between the burned and unburned treatments. The lowest extractable phosphorus levels occur during December within the unburned plots and highest within the burned plots in April. Extractable phosphorus concentrations were found to be significantly greater $F_{(1, 28)}=29.741$, $P=0.00001$ within the burned compared to the unburned plots. Further significant differences $F_{(4, 112)}=17.673$, $P=0.00001$ were found to occur between the various months sampled.



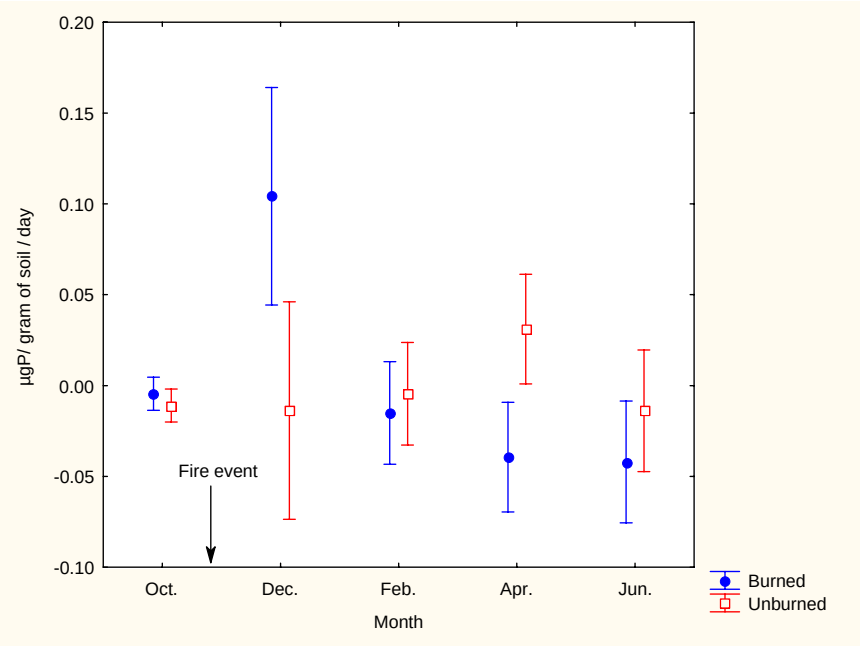
Repeated measures ANOVA
Month x Treatment: $P = 0.001$

		Treatment	
		Burned	**Unburned**
Month	Oct.	a	ac
	Dec.	b	ac
	Feb.	a	a
	Apr.	b	bc
	Jun.	b	bc

Figure 4.10: $Time_1$ soil extractable phosphorus concentrations in a felled *Eucalyptus grandis* plantation, over a nine month period in plots where the slash had either been burned or remained unburned (vertical bars represent 95% confidence intervals; n=15 per month/treatment). Different lower case lettering indicates significant differences $P < 0.05$ between treatments for each specific month and between months within a treatment.

$Time_1$ extractable phosphorus concentrations were similar for both treatments in October, prior to burning. $Time_1$ extractable phosphorus concentration within the burned plot in December significantly increased four fold following the fire event in November. Similar extractable concentrations occurred in the burned plots in December and June with both the burned and unburned plots in April having similar concentrations. The lowest extractable phosphorus concentrations occurred within the unburned plots in December and February. Significant differences occurred between treatments $F_{(1, 28)}=21.328$, $P=0.0001$ and months sampled $F_{(4,112)}=15.578$, $P=0.00001$.

4.2.5 Phosphorus mineralization rates.



Repeated measures ANOVA
Month x Treatment: $P = 0.0001$

		Treatment	
		Burned	Unburned
Month	Oct.	a	a
	Dec.	b	a
	Feb.	a	a
	Apr.	a	ab
	Jun.		a

Figure 4.11: Phosphorus mineralization rates in a felled *Eucalyptus grandis* plantation, over a nine month period in plots where the slash had either been burned or remained unburned (vertical bars represent 95% confidence intervals; n=15 per month/treatment). Different lower case lettering indicates significant differences $P < 0.05$ between treatments for each specific month and between months within a treatment.

Figure 4.11 shows that in October, prior to burning, there were no significant differences in phosphorus mineralization rates between the burned and unburned treatment plots. In the burned plots, phosphorus mineralization rates were significantly increased ten fold within the first month following the fire event. These increased phosphorus mineralization rates, however, were short lived with gradually decreasing rates occurring within the burned plots, for the remainder of this study. Phosphorus mineralization rates within the unburned plots gradually increased from December levels, peaking in April prior to a decrease in June. There was no significant difference between the treatments $F_{(1, 28)}=0.05765$, $P=0.81$ although a significant difference $F_{(4,112)}=5.2474$, $P=0.0006$ between sampling months was found for the burned plots.

Summary:

- $Time_0$ soil extractable phosphorus concentrations for both treatments increased exponentially overtime, as a result of enhanced decomposition.
- Fire increased both nitrogen and phosphorus mineralization for a 30 day period and thereafter the effect was non significant. Increased availability of these growth limiting nutrients need to be synchronised with plant uptake to maximise the benefit of the fire.

4.3 Soil fauna.

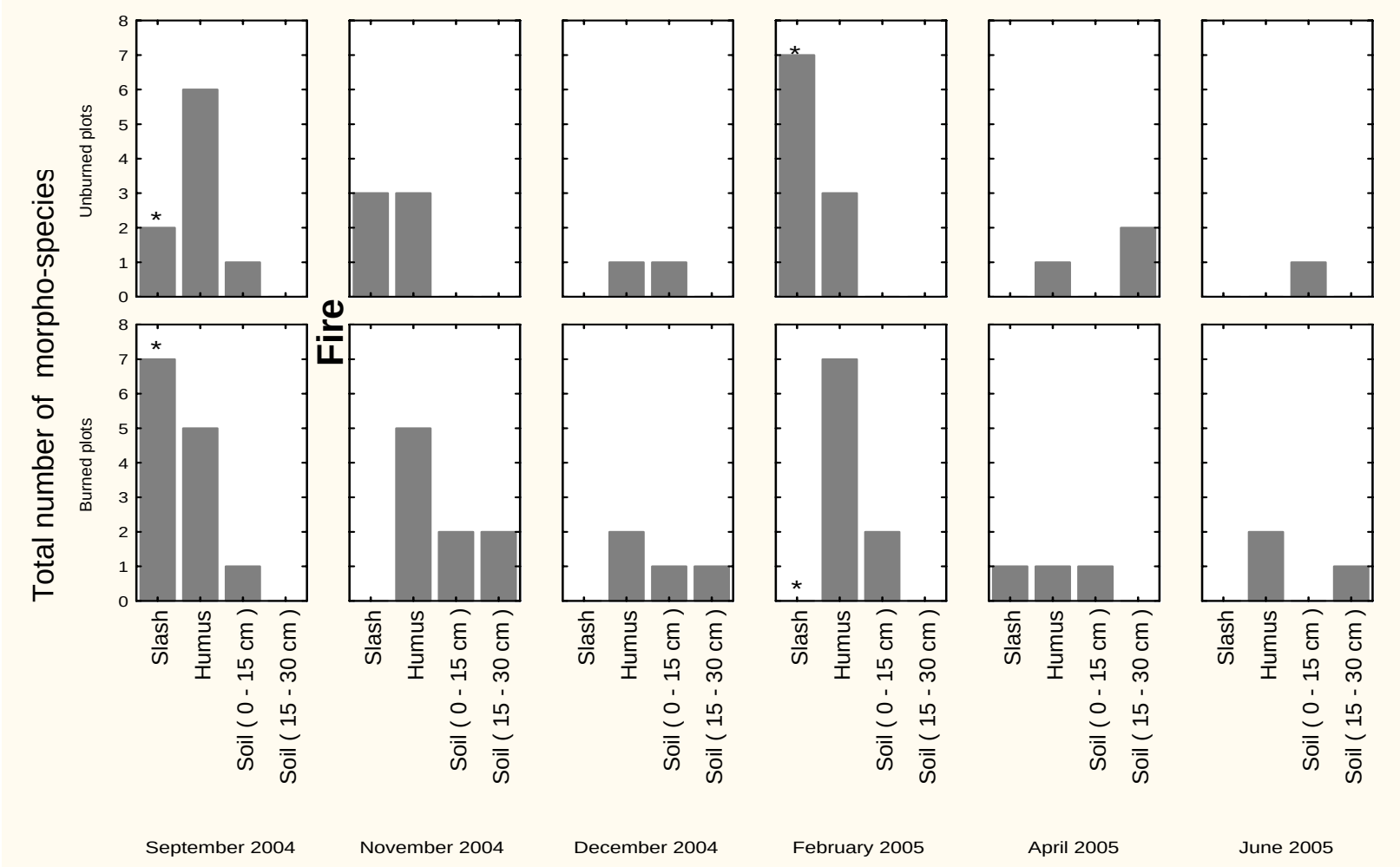


Figure 4.12: The total number of arthropod morpho-species found throughout the study and separated to show, prior to and after burning in different layers of soil and humus within a felled *Eucalyptus grandis* plantation. (n=15 per month/treatment). * indicates significant difference ($P<0.05$) between treatment plots, within soil layers.

The majority of morpho-species, collected throughout the duration of this study (September 2004 to June 2005), were found in the humus layer (χ^2_1=19.47; P<0.001) and the least were found in the 15-30 cm soil layer (χ^2_1 = 12.31; P<0.001) in both treatment plots (Figure 4.12). The total number of morpho-species found within the slash layer was equal for both the burned and unburned plots for the duration of the study.

When comparing the total number of morpho-species collected during September 2004 (prior to the burning) the majority were caught within the slash layer of the plots scheduled for burning and the minority in the 0-15cm soil layer in both treatments. χ^2 tests comparing the total number of individual morpho-species caught revealed a non significant differences between the two treatments for the slash layer; humus layer (χ^2_1 = 0.18; P>0.05) and soil (0-15cm) layer (χ^2_1 = 0.50; P>0.05) with significantly more (χ^2_1 = 2.88; P>0.05) being found within the slash layer within the burned plots.

In November 2004, immediately after the fire event, it was found that the number of morpho-species varied between treatment plots. Comparisons in morpho-species location between treatments through the use of χ^2 tests revealed that no significant differences occurred (slash layer χ^2_1 = 3.33; P>0.05; humus layer χ^2_1 = 1.06; P>0.05 and soil (0-15cm) and (15-30cm) layer χ^2_1 = 2.50; P>0.05).

Species richness in both the burned and unburned plots fluctuated seasonally (Figure 4.12). The lower species richness values for December, April and June were due to large rainfall events and/or a cold front passing through during the week of sampling. The highest numbers of species occurred within the burned plot in September 2004 and the unburned plots during February 2005. The total number of morpho-species found within each of the burned and unburned plots for each month were pooled for a Wilcoxon rank test revealing that no significant difference Z=1.677; P= 0.09 occurred between the burned and unburned plots throughout the duration of

the study. χ^2 tests comparing the total number of morpho-species between the treatment plots revealed no significant differences for all months sampled.

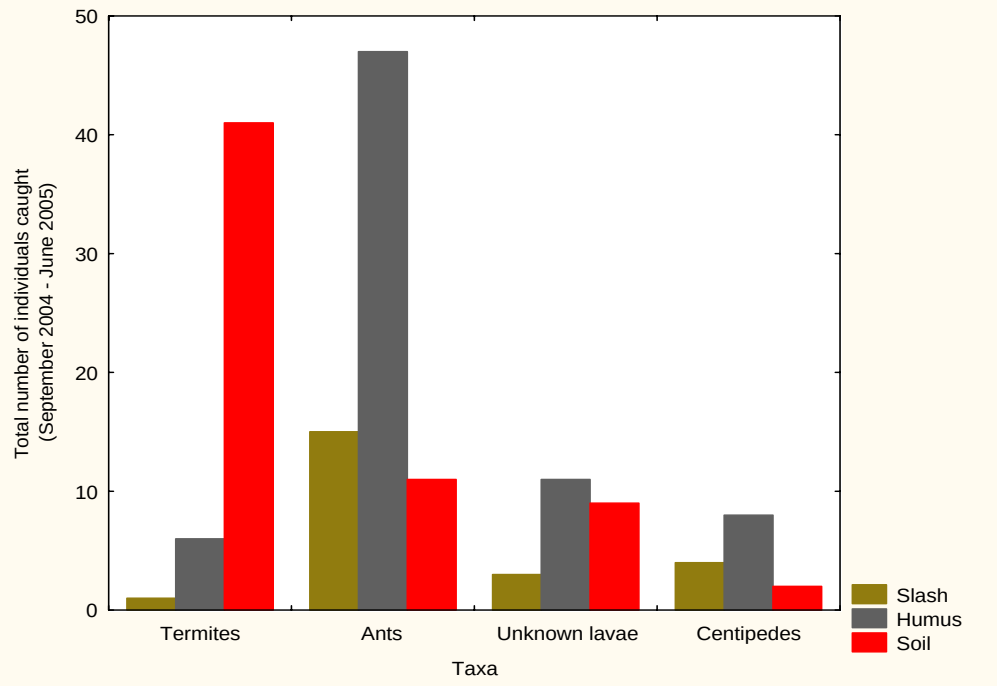


Figure 4.13: The total abundance of arthropod taxa found in slash, humus and soil layers, collected over nine months (September 2004 to June 2005) from a felled *Eucalyptus grandis* plantation.

Termites, ants, centipedes and unknown larvae occurred within all three soil layers (Figure 4.13). Within the slash and humus layers ants were most abundant. Termites were most abundant within the soil layer, although the termites collected were mostly from one sampling occasion at one site. Ants and centipedes were more abundant in the humus where they might be susceptible to fire.

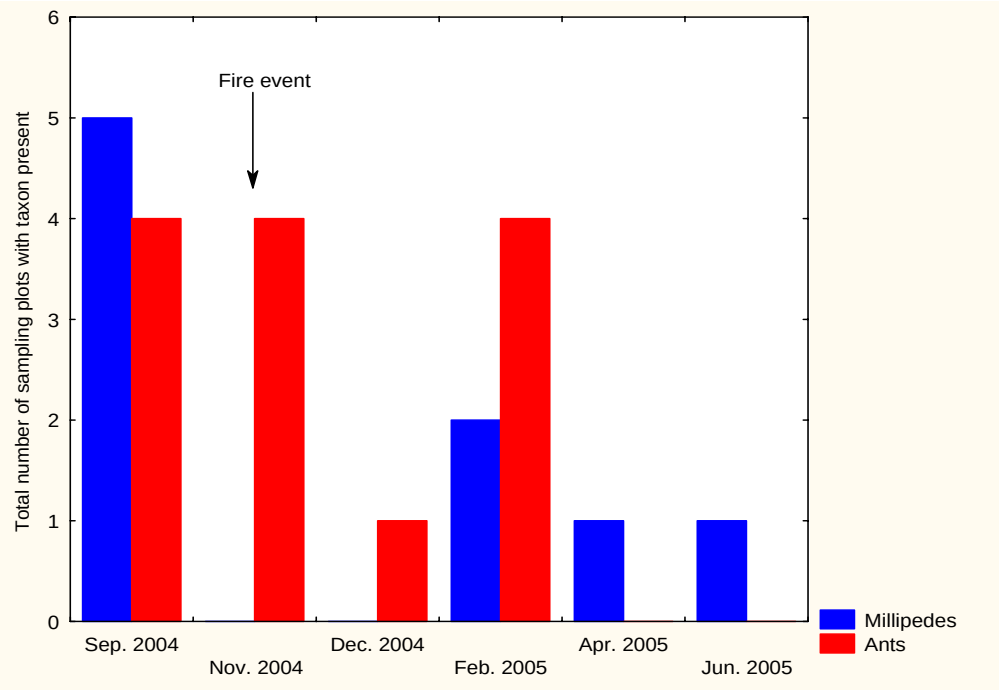


Figure 4.14: Seasonal abundance of the most common arthropod taxa collected from slash, humus and soil monoliths in a felled *Eucalyptus grandis* plantation based on their presence or absence in three sampled plots per treatment (n=6 per month).

Millipedes dominated in spring prior to burning and ants in summer, post burning (November to February). Ants were absent from all plots in April and June whereas millipedes were absent from November, returning in February (Figure 4.14).

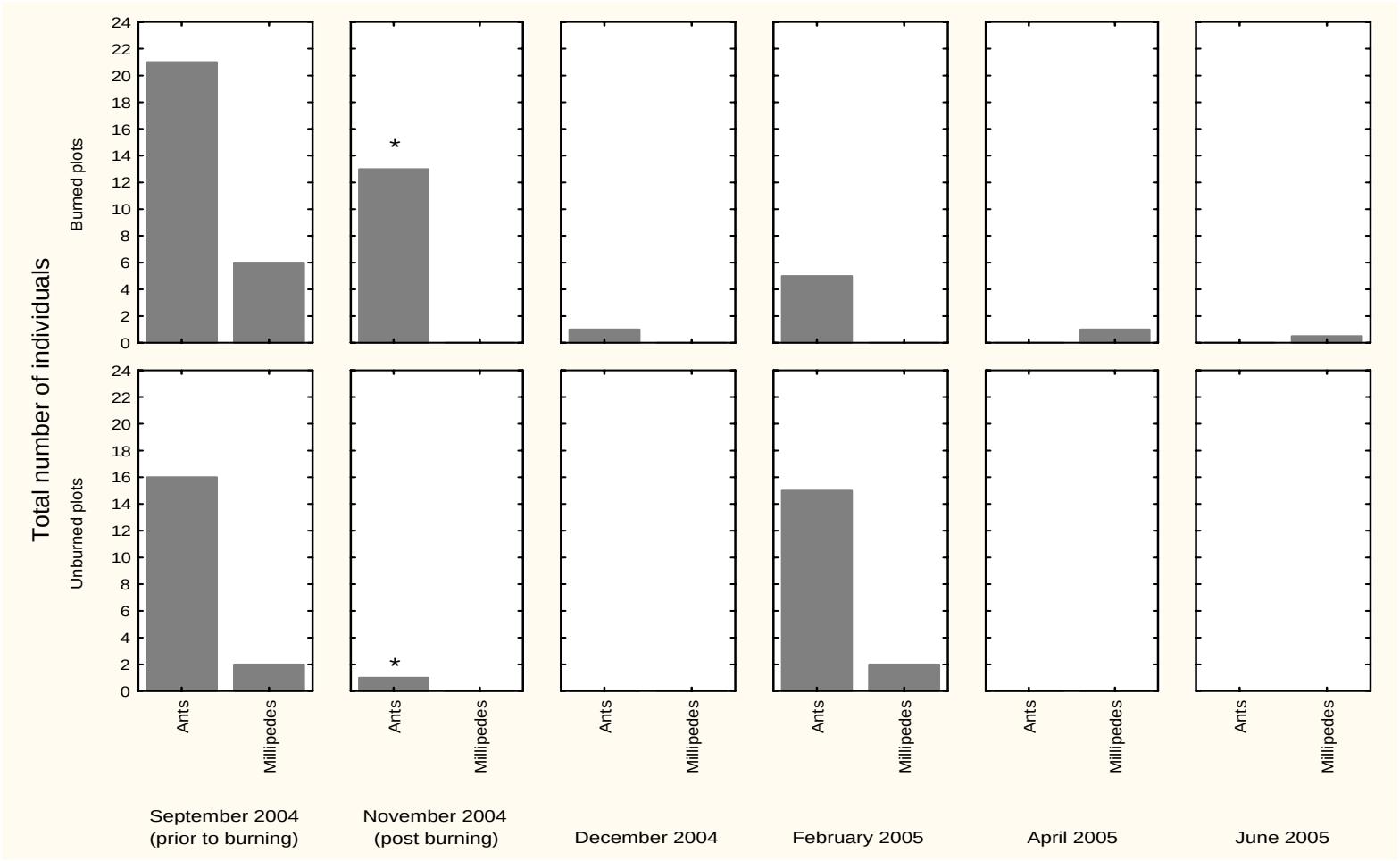


Figure 4.15: Seasonal abundance of ants (Formicidae) and millipedes (Diplopoda) collected in a felled *Eucalyptus grandis* plantation, over a nine month period in plots where the slash had either been burned or remained unburned. Indicates significant difference ($P<0.05$) between burned and unburned plots.

Ants had the greatest number of individuals within both treatments during the month of September (Figure 4.15). A decrease occurred in the number of individuals caught over time within the burned plots whereas in contrast the number of individuals that were caught in the unburned plots decreased and increased. No ants were caught during April 2005 and June 2005 sampling events. Significant differences between the number of ants caught in the burned and unburned plots occurred during the month of November 2004 ($\chi^2_1 = 12.01$; $P<0.001$).

Prior to burning, millipede numbers were greatest within the burned plots during the month of September 2004. Millipedes were absent from all plots during the sampling months of November and December 2004, returning to the unburned plots in February 2005. In April and June 2005, millipedes were only found within the burned plots. χ^2 tests were used to compare the number of individuals between treatments for all months sampled. It was revealed that no significant differences between the burned and unburned plots occurred.

Table 4.4: The cumulative number of arthropod individuals collected in burned and unburned plots within a felled *Eucalyptus grandis* plantation over a nine month period in all plots where the slash had either been burned or remained unburned.

		September		November		December		February		April		June		Total
		Burned	Unburned	Burned	Unburned	Burned	Unburned	Burned	Unburned	Burned	Unburned	Burned	Unburned	
Springtails (Collembola)														
Entomobryoidea										1		1		2
Cockroaches (Blattodea)														
Blattidae									1					1
Termites (Isoptera)														
Termitidae						48								48
Grasshoppers and crickets (Orthoptera)														
Acrididae								1						1
Gryllidae								1						1
Beetles (Coleoptera)														1
Scarabaeidae								1						1
	Cetoniinae								1					1
Lycidae									1					1
Coccinellidae									1					1
Tenebrionidae								1						1
Flies (Diptera)														
Bombyliidae		1												1
Bees (Hymenoptera)														
Apidae					1									1
Ants (Hymenoptera)														
Formicidae														
	Formicinae	4		3	1			5						13
	Myrmicinae	6	16	10	12	1			15					60
	Ponerinae	11												11
Hexapoda														
Scarabaeidae			1	1		1								3
Unknown		4	5	3	2		2	4		1	1	1		23
Centipedes (Myriapoda; Chilopoda)														
Lithobiomorpha														
Henicopidae	*Lamyctes*	1		1				1	3					6
Scolopendromorpha														
	Cormocephalus		4	1	1									6
Geophilomorpha														
Geophilidae											1		1	2
Millipedes (Myriapoda; Diplopoda)														
Polydesmoidea														
Dalodesmidae	*Gnomeskelus*	6	2						2	1		1		12
Other		7	2	1				3	2					15
Diversity indices														
Shannon- Wiener (species)		1.46	1.23	0.96	0.84	0.21	0	1.29	1.24	0.19	0.39	0.81	0	2.23
Simpsons index (species)		0.17	0.32	0.29	0	0.92	0	0.15	0.2	0	0	0	0	0.16

Very few arthropods were caught within each of the treatment plots for all sampling events. Ants of the sub family *Myrmicinae* were the most abundant taxon collected throughout the study. The largest number of individuals caught within a single sampling period was termites in the burned plots in December. Several single arthropod taxa were collected only once, over a nine month period, within the various treatment plots (Table 4.4).

The Shannon Wiener diversity index for both the burned and unburned treatment plots was very low. Values for the burned plots ranged between 0.19 to 1.46, and 0 to 1.33 for the unburned plots. No significant differences $t_{10} = 0.64$; $P = 0.54$ in species diversity occurred between the burned and unburned plots throughout the duration of the study.

Summary:

- Species diversity was low within the felled *Eucalyptus grandis* plantation with ants being the dominant soil fauna.
- Fire did not significantly change species diversity with millipedes occurring in the majority of plots in spring, prior to the fire and ants dominating in summer, post fire.

Chapter 5.

Discussion.

The discussion section of this dissertation is separated into four sections. Each section will address one of the key questions asked in this study.

Increasing productivity in alignment with ecological sustainability is of vital importance if the forestry industry is to ensure site sustainability and consumer demands into the future. The sustainable growth of trees is, however, dependent on the conservation of nutrients, required for plant growth, a major concern specifically when minimal fertilization is practiced. Improvements in the understanding of factors influencing the cycling of nutrients are of vital importance for the achievement of high sustainable yields within managed plantations. The rate at which nutrients cycle within managed exotic plantations is controlled by the size of the soil nutrient pool and the magnitude of the fluxes that either increase or decrease the size of this soil organic matter pool. The magnitude of the nutrient flux leaving the organic matter pool within the soil is dependent on the rate of decomposition and mineralization of the organic matter that in turn produces nutrients needed for plant growth. A better understanding of the impacts of certain management practices on the availability of these nutrients will aid the forestry industry in achieving site sustainability. The management of the inter-rotational period has been found to impact the ecological sustainability and productivity of a site by impacting the availability of nutrients. During this crucial period, fire is often used as a simple management tool to remove plant residues that remain after tree felling (de Ronde *et al.* 2004). Fire, however, results in the loss of nutrients with the quantity of these losses being dependent on the intensity of the fire.

5.1 What quantity of nutrients are lost from the slash following burning?

The majority of elements (carbon, nitrogen, phosphorus, magnesium, potassium and sodium) were found in larger concentrations within the leaf component of the slash compared to that of the stems or bark (Table 4.1). Calcium had greater concentrations associated with the bark. Foliar concentrations were similar to those found within the litter from trees grown both within a mixed *Eucalyptus* forest stand in Tasmania and in an indigenous neotropical dry forest in Brazil (Harwood and Jackson 1975; Kauffman *et al.* 1993). The leaves, make up the smallest

component of the slash but due to their high nutrient concentrations play a crucial role in maintaining the cycling of nutrients within the ecosystem. Leaves are, however, fine fuels that undergo combustion at low fire intensities resulting in the majority of their nutrients being returned to the system (Yang *et al.* 2005). In comparison, stems that will burn at higher temperatures, result in more of the nutrients being volatilized and fewer nutrients returning to the soil in the form of ash.

Phosphorus, calcium, magnesium and potassium concentrations within the ash were found to be greater than those found within the slash prior to burning (Table 4.2). Increases in nutrient concentration are due to the concentrating effect that occurs during the burning and combustion process (Bird 2001). During combustion, carbon in the slash is volatilised at low temperatures to form carbon dioxide whereas other nutrients require higher temperatures before they are volatilized. Therefore 1kg of ash has a greater concentration of nutrients in comparison to 1kg of slash that was composed predominantly of carbon (Bird 2001). In comparison, decreases in nutrient concentrations of carbon and nitrogen within the remaining ash indicate the loss of nutrients through volatilization. The concentrating effect does, however, not fully explain the extent of the increase in calcium concentration within the ash. The large increase in calcium concentration within the ash could be due to the fact that volatilization of calcium is minimal. Calcium has been found to only volatilize at 1200°C therefore all calcium associated with the woody material enriches the remaining ash that occurs within a particular site (Kauffman *et al.* 1993).

The quantities of various nutrients lost from other ecosystems during fire have been found to range between 440-1183 kg ha^{-1} for nitrogen, 10-86 kg ha^{-1} for phosphorus, 48-97 kg ha^{-1} for potassium, 73-100 kg ha^{-1} for calcium and 37-68 kg ha^{-1} for magnesium (Harwood and Jackson 1975; Morris 1986; du Toit 2003). The quantity of nutrients reported by du Toit (2003) as being lost are 6, 5, 11, 2, and 7 times higher for nitrogen, phosphorus, potassium, calcium and magnesium respectively compared to those found to occur within the spread slash within this study. In comparison the quantity of nutrients lost from the piled slash were found to be similar for nitrogen and phosphorus with 3, 19 and 9 times greater for potassium, calcium and

magnesium respectively, to that reported by du Toit (2003). The quantity of nutrients lost from a particular ecosystem is, however, dependent on the original fuel load and the intensity and the duration of the fire used to burn the slash as this would affect the quantity of nutrients that are volatilized. The greater the intensity of the fire the larger the loss of nutrients, as nutrients such as carbon, nitrogen and phosphorus are volatilised at lower temperatures than that of other cations with phosphorus being volatilised at temperatures exceeding 360°C (Romanyà *et al.* 1994; Jensen *et al.* 2001).

Although the quantities (kg ha^{-1}) of nutrients lost from the system are small, the proportion of these nutrients lost from the slash nutrient pool was high as a result of the fire. The proportional losses of nutrients leaving the slash nutrient pool were markedly greater than those found by Harwood and Jackson (1975), Morris (1986) and du Toit (2003) who reported proportional nutrient losses ranging from 42-54% for nitrogen and 18-49% for phosphorus. This study had losses comparable to those reported by Kauffman *et al.* (1993) for a neotropical dry forest with proportional losses ranging from 79–96% for nitrogen and 3–56% for phosphorus.

To understand the extent to which these proportional nutrient losses affect the nutritional sustainability of a plantation the net gains, losses and relative pool sizes for available nutrients to be found within a plantation must be identified. The major nutrient pools noted within a plantation forest are that of litter, standing biomass and the soil nutrient pool (du Toit and Scholes 2002). The soil pool usually constitutes a major reservoir of nutrients, although only a small quantity are available for plant uptake (Scholes and Scholes 1999). Larger quantities of certain available nutrients (nitrogen, phosphorus and potassium) within a *Pinus patula* plantation were, however, found within the litter nutrient pool in comparison to that of the soil nutrient pool (Dames *et al.* 2002). The burning of this litter layer would substantially affect the loss of available nutrients within the plantation, as a large quantity of available nutrients would be lost (Dames *et al.* 2002). The plantation's ability to buffer against nutrient losses is dependent on the flux of nutrients being lost and size of the available nutrient pool. Larger nutrient pools would have a greater buffering capacity against short term nutrient losses from the plantation (du Toit and Scholes 2002).

The greatest loss of nutrients that occur within forestry plantations are determined primarily through management practices such as harvesting and slash burning. Harvesting an *Eucalyptus grandis* plantation has been estimated to remove approximately 63 t ha^{-1} of timber with its associated nutrients, approximately every ten years (Scholes and Scholes 1999). du Toit and Scholes (2002) thus suggested that input and output budgets be evaluated for these specific management practices to determine their impact on a nutrient sustainability.

The proportion of nutrients lost by burning in this study, was estimated using published data on available nutrient pool sizes for an *Eucalyptus grandis* plantation adjacent to the current study site (du Toit and Scholes 2002; du Toit 2003). Potential losses for all nutrients from the plantation for slash that was piled before burning ranged from 1-12%, with magnesium having the lowest and phosphorus the highest potential losses from the plantation. These indicate that the loss of nutrients per rotation as a result of slash burning constituted a small fraction of the available nutrient pool for the plantation studied. It can therefore be assumed that the burning of slash does not pose an immediate threat to the plantation's nutritional sustainability. It is, however, important to realise that the quantity of nutrients lost from the plantation needs to be replaced, by fertilization, in order to ensure the long term sustainability of the site. Large nutrient losses from the plantation will eventually deplete the available nutrient pools if nutrients are not returned in the form of fertiliser.

This study found that approximately 92% of litter was consumed by the fire with 74 and 528 kg ha^{-1} of nitrogen being consumed within the spread and piled slash respectively (Table 4.3). Phosphorus losses were equally large with 5-31 kg ha^{-1} being lost from the spread and piled slash (Table 4.3). Such large losses in nitrogen and phosphorus are expected when the majority of litter is consumed by the fire (Turner and Lambert 1996). Turner and Lambert (1996) suggested that approximately 70-400 kg ha^{-1} of nitrogen and 1-20 kg ha^{-1} of phosphorus should be lost from a system when the majority of litter is consumed by fire. Other nutrients within this study fell within the range of losses as reported by studies conducted by Harwood and Jackson (1975), Morris (1986) and du Toit (2003). It has been reported that within a mixed forest in Tasmania the

proportion of nutrients lost from the system as a result of fire were greatest for magnesium > phosphorus > potassium > calcium (Harwood and Jackson 1975). This is, however, different to what has been found within this study where it has been found that losses are greater for phosphorus > potassium > magnesium > calcium (Table 4.3). The logs found in burned stands within a study conducted by Harwood and Jackson (1975) were not completely consumed by the fire, and the associated magnesium in the heartwood remained within the logs and was not returned to the system in the form of ash. Magnesium losses were therefore overestimated due to the lower quantities of magnesium found within the remaining ash in comparison to that of the other nutrients.

The spread slash was dominated by stems with a diameter of less than 5cm and a small amount of bark and leaves. Absolute nutrient losses were low due to the initial nutrient concentration and low fuel loads, whereas proportional losses were high as a result of loose compaction and efficient combustion (Table 4.3). In comparison, piled slash was dominated by stems greater than 5cm, with significant amounts of bark and leaves. These materials contained high initial potassium, magnesium and sodium concentrations. Following burning, absolute losses were high, proportional losses were low, resulting in enriched ash content (Table 4.3). Interestingly there is a gain from calcium in the slash piles with losses occurring within the open areas; however, the concentrations of calcium were similar within both the spread and piled slash ash. The slash pile, however, consisted of more woody material and therefore greater quantities of associated calcium in comparison to that found within the spread slash. It was reported that with increasing biomass consumption by fire there is a decrease in carbon, nitrogen and phosphorus and increases in calcium concentration (Kauffman *et al.* 1993). This pattern has been related to the increases in volatilization of nutrients, with losses to calcium due to volatilization being minimal or even negligible.

The quantity of remaining ash and the loss of various nutrients are dependent on the composition of the material burned and the distribution and compaction of the litter (Trollope *et al.* 2004). Such factors play an influential role in the combustion and intensity under which the slash was burned and thereby affects the loss of nutrients (Geldenhuys *et al.* 2004). The losses of

nutrients as a result of fire, experienced in this study are small relative to what is tied up in the ecosystem (du Toit and Scholes 2002; du Toit 2003).

5.2 How do soil nitrogen mineralization rates and extractable soil phosphorus levels change with season in both burned and unburned plots?

The release of nitrogen and phosphorus from decomposing plant litter and soil organic matter is dependent on seasonal cycling of nutrient mineralization and immobilization processes (Arunachalam *et al.* 1998). The rate of plant litter decomposition that occurs within a particular site is strongly determined by climatic conditions and the chemical composition of the litter that occurs on a particular site (Arunachalam *et al.* 1998). Soil microbial activity may increase after a fire, due to the reduced carbon to nitrogen ratios of deposited ash and remaining organic material (Geldenhuys *et al.* 2004).

Mineralization rates may, however, decline after a fire event due to a reduction in the soil moisture required for the survival of micro organisms. Fire events that consume the majority of fuel, such as occurred in this study (92% fuel loss) will usually result in the remaining ash increasing the soil pH within a particular site. Increases in soil pH usually increase bacterial populations rather than fungal populations, involved in the decomposition of soil organic matter (Geldenhuys *et al.* 2004). Increases in soil nutrient concentrations in a particular site are strongly dependent on the immobilization of nutrients (Arunachalam *et al.* 1998). It has, however, been shown that within unburned sites, through the process of decomposition, a greater quantity of nutrients become available over time in comparison to sites that were burned (Vitousek and Matson 1985).

Ammonium concentrations.

Disturbances such as a fire event have been shown to immediately increase the availability and concentration of ammonium within the soil, as was shown within this study (Figure 4.1 & 4.2). Such increases in ammonium concentration are due to the heating of the soil, causing the release of ammonium through the mineralization process, or from the ash which is deposited on the soil

surface (Weston and Attiwill 1996; Knoepp *et al.* 2004). Soil ammonium concentrations may also be increased due to the fire event that transformed the organic matter into ammonium at temperatures greater than 210 °C (Gimeno-García *et al.* 2000).

If favourable temperature and moisture conditions occur within a particular site following a disturbance event, such as fire, it has been suggested that the ammonium ions would be immobilized due to the growth of microbial organisms that are limited in the availability of carbon, that is required for growth. This was shown with ammonium ion concentrations decreasing within the burned sites in February following the increased ammonium concentrations immediately following the fire event in December (Figure 4.3). The immobilization of ammonium ions following the fire event and an initial rapid increase in microbial activity is one way in which nitrogen can be conserved within the site (Weston and Attiwill 1996).

Increases in ammonium concentrations within the soil, following a fire event, have been shown to last for up to 120 days (Weston and Attiwill 1996). This study, however, found that this increase only occurred for 30 days following a fire event. In an *Eucalyptus regans* forest a fire event increased the ammonium concentration by approximately three times from 60µgN(NH_4)gram $soil^{-1}$ to 160µgN(NH_4)gram $soil^{-1}$ after burning, with high ammonium concentration persisting until the second winter following the burning event (Weston and Attiwill 1996). In this study ammonium concentrations within the soil were also increased approximately three fold following the fire event within the burned plots from 6µgN(NH_4)gram $soil^{-1}$gram to 16µgN(NH_4)gram $soil^{-1}$. In comparison the immobilization of ammonium ions that occurred within the unburned sites throughout the duration of this study may be due to the fact that the carbon to nitrogen ratio within the plant litter and organic matter may have stimulated the immobilization of ammonium ions within the soil (Vitousek *et al.* 1982; Weston and Attiwill 1996). The C:N ratio provides an indication of the limitation of energy (carbon) to that of nutrient availability (nitrogen) as soil microbial fauna usually prefer a low C:N ratio for mineralization to occur (Swift *et al.* 1979; Cadisch and Giller 1997). Carbon is usually abundant in comparison to nitrogen which is often limiting and immobilised until it is no longer a limiting

element, when carbon concentrations are reduced as a result of decomposition (Swift *et al.* 1979; Cadisch and Giller 1997).

Nitrate concentrations.

Nitrification rates within a particular site are dependent on soil temperature and have been found to be more sensitive to soil temperature conditions than that of ammonium (Weston and Attiwill 1996). However, this was not found to affect the rate of nitrification within this study, although it may have caused the increase in the initial soil concentration of nitrate during the sampling month of February.

Soil nitrate has been found to decrease following the burning event (Gimeno-García *et al.* 2000). Although in this study nitrate concentrations were found to increase throughout the duration of the study within the burned sites (Figure 4.5 & .6). Such increases in the rate of nitrification were found to occur within disturbed forest sites due to the availability of ammonium ions that occur within the soil (Weston and Attiwill 1996). As ammonium was quickly nitrified and lead to larger quantities of nitrate compared to that of ammonium being found after a fire event (Jensen *et al.* 2001). It was further found that nitrification rates were always lower within unburned sites in comparison to sites that had been burned, and burned sites showed increases in the rate of nitrification particularly during the warmer summer and autumn months (Weston and Attiwill 1996).

Nitrogen mineralization rates.

The rate of nitrogen mineralization within the unburned sites was found to be highest during the summer months as found by Weston and Attiwill (1996) (Figure 4.7). This study, however, revealed that seasonally induced changes in nitrogen mineralization rates were more dependent on temperature than water, as no significant difference between the rate of potential (addition of water) and actual nitrogen mineralization (Figure 4.7 & 4.8) occurred.

Inorganic nitrogen levels have been shown to increase for up to 90 days following a fire event (Jensen *et al.* 2001). These increases in inorganic nitrogen were short lived in this study and only

lasted for approximately 30 days (Figure 4.7 & 4.8). The increase in the rate of nitrogen mineralization is due to fire affecting the temperature of the soil. The heat from the fire enhanced the biological process when it was transferred below ground, heating the soil and thus stimulating the activity of the microbial fauna and thereby increasing the rate of nitrogen mineralization (Neary *et al.* 1999; Wilson *et al.* 2002). The significant decrease in the rate of nitrogen mineralization within the burned sites in February may be due to the potential short term unavailability of carbon associated with the organic matter, following the increased mineralization rates. The unavailability of carbon results in the immobilization of inorganic nitrogen within the burned plots (Figure 4.7 & 4.8). In an *Eucalyptus regans* forest it was found that the concentration of inorganic nitrogen was increased prior to reverting back to similar concentrations of undisturbed sites approximately 250 days after the fire event had occurred (Weston and Attiwill 1996). In this study the nitrogen mineralization rate and the concentration of inorganic nitrogen were found to reach similar concentrations to that of the unburned sites approximately 150 days following the fire event (Figure 4.7 & 4.8).

Increases in inorganic nitrogen concentrations following a fire event have been suggested as potentially being due to chemical mineralization instead of biological nitrogen mineralization (Weston and Attiwill 1996). Total inorganic nitrogen concentrations have been suggested by Weston and Attiwill (1996) as being mostly due to increases in ammonium concentrations. Resulting from rapid increases in micro fauna immobilization that occurred and decreased the ammonium levels within the soils. In this study, however, in was shown that the increase in the rate of nitrogen mineralization was due to increases in the rate of nitrification, shown by the greater concentrations of nitrates within the soil compared to that of ammonium ions throughout the duration of this study (Figure 4.2 & Figure 4.5). Therefore following the fire event the immediate increase in inorganic nitrogen occurred due to the increase in soil temperature and availability of carbon, and soil moisture due to rainfall.

Phosphorus mineralization.

Increases in phosphorus availability following a fire have been found to occur within several studies after a fire event has occurred (de Ronde and Stock 1994; Ellis and Graley 1994;

Geldenhuys *et al.* 2004). The persistence of phosphorus availability is, however, dependent on the soil texture and acidity that occurs within a particular site as well as the intensity of the fire that occurred (Romanyà *et al.* 1994). The availability of phosphorus has been found to increase in soils that have low phosphorus concentrations, due to the substantial increase in phosphorus from the ash deposited on the soil surface (Romanyà *et al.* 1994).

Sandy soils usually lose the majority of the available phosphorus due to the leaching process in which phosphorus ions are transported further down into the soil profile. Basic clay soils on the other hand have been found to have increased phosphorus availability for several years following a fire as phosphorus that is usually associated with iron oxide and aluminium ions at low pH values becomes available. A basic pH value close to neutrality allows the movement of phosphorus within the soil that would usually be associated with these iron and aluminium ions and oxides (Geldenhuys *et al.* 2004). The soil and ash interactions affecting the pH of the soil as well as the increase in soil temperature and moisture can affect the availability of phosphorus and result in substantial increases (Romanyà *et al.* 1994; Giardina and Rhoades 2001). The increase in temperature and pH increases the activity of the bacterial population that is involved in the decomposition of soil organic matter (Geldenhuys *et al.* 2004).

This change in pH (as a result of ash deposition) coupled with an increase in soil temperature brought about by the fire event explains the 10 fold increase in phosphorus mineralization that occurred within the burned plots (Figure 4.11). This increase was substantially lower (at approximately 0.10µgP.gram $soil^{-1}.day^{-1}$) compared to Bird (2001) (approximately 0.8 µgP.gram $soil^{-1}.day^{-1}$) after burning the litter of a *Pinus patula* plantation. The rate of phosphorus mineralization was also much lower at between -0.04 and 0.1 µgP.gram $soil^{-1}.day^{-1}$ compared to that of Bird (2001) who reported rates between 0.2 and 6 µgP.gram $soil^{-1}.day^{-1}$. The rate of phosphorus mineralization within this study may be lower to that reported by Bird (2001) due to the initial pool size of available phosphorus in this study being smaller due to the clearfelling and removal of trees.

Nitrogen and phosphorus mineralization was increased for a 30 day period within burned plots following a burning event. These increased rates are assumed to be due to increases in temperature, adequate moisture conditions and optimal pH conditions that stimulated the activity of the microbial organisms involved in the biological processes of mineralization and decomposition.

5.3 How does soil faunal diversity change in the slash and mineral soil in a felled *Eucalyptus grandis* plantation prior to and after burning?

Several studies have shown that fire, when used as a management tool, reduced species richness and diversity at a particular site (Huhta *et al.* 1967; Bird and Chatarpaul 1986; Bird *et al.* 2000; Bellocq *et al.* 2001; Decaëns *et al.* 2001; Rossi and Blanchart 2005). This study, however, revealed that the species richness that occurred within the plantation in September 2004, prior to burning, was not affected by fire as the humus layer remained, although in some areas it was scorched as a result of 92% of fuel being lost. Numerous species losses from various other studies were noted to occur when the humus and organic matter layer was removed by a fire event (Huhta *et al.* 1967; Dress and Boerner 2004). This study showed similar species diversities prior to and after a fire event and was comparable to several studies that have found fire, not to affect the surface active arthropod communities species diversity and richness when compared to the unburned reference sites (Andrew *et al.* 2000; Collett 2003).

Species richness being similar within both the burned and unburned plots after the fire event (Table 4.12) may be due to the close proximity of *Themeda* grassland and a 7 year old *Eucalyptus grandis* plantation. These habitats may have acted as either a refuge for various species or may have provided a source population to colonise the burned site. No seasonal data set exists for soil fauna in indigenous grasslands compared to that of exotic plantations. Data that, however, do exist suggest that the grassland would harbour a higher species diversity and species richness to that of an exotic plantations (Samways *et al.* 1996). The re-colonisation of soil and litter invertebrate fauna from source populations and neighbouring vegetation, after a

disturbance event has been reported by several studies (Shaw *et al.* 1991; Horgan 2005; Hou *et al.* 2005)

5.4 What seasonal changes occur in the soil faunal diversity in the slash and mineral soil in a felled *Eucalyptus grandis* plantation in both burned and unburned plots?

The seasonality of soil and litter invertebrate fauna within a plantation forest was determined by monitoring presence, absence and abundance of the most common species that occur within a plantation forest. Millipedes were found to occur within all plots prior to burning and ants were found in all, post burning (Figure 4.14).

Millipedes are detritivores which aid in the process of mineralization and decomposition through increasing microbial activity of the soil by aerating and fragmenting the litter (Lavelle *et al.* 1997; Greyling *et al.* 2001). Millipedes are usually most active after the first rains, although this period of activity is usually short–lived, only lasting from 77 - 111 days (Dangerfield and Telford 1991). This would explain the dominance of millipedes within all plots during spring after the first rains had occurred (Figure 4.14). Decreases in minimum temperatures have been shown to bring about the onset of millipede burrowing (Dangerfield *et al.* 1992). Burrowing and over-wintering within the soil to depths of 30cm allows millipedes to survive the dry and cold months from May to October resulting in their absence from the soil surface (Dangerfield and Telford 1991; Dangerfield *et al.* 1992).

Millipedes feed on fungi and bacteria that are present on the decomposing leaves (Dangerfield *et al.* 1992). Greater species diversity of detritivores, such as millipedes, occurs within sites that have been recently harvested due to the availability of decaying wood and litter on which they can feed (Bellocq *et al.* 2001). As millipedes rely on decomposing litter as a food source for their survival, the fire event removed the some of this food source, which had not yet formed humus, and also removed plants that provided shelter for the millipedes. This resulted in the absence of millipedes from the burned sites until April 2005 (Figure 4.15). The reoccurrence of millipedes within the burned sites in April 2005 may be due to the growth of *Eucalyptus grandis* seedlings and the removal of coppice growth from stumps that remained after felling. Plant re-growth has

been found to increase species richness within a particular site by providing food and shelter for soil and litter invertebrate species (Bird and Chatarpaul 1986; Christensen and Abbott 1989; Samways *et al.* 1996; Bird *et al.* 2000).

Ants in comparison to millipedes are more generalist feeders and are not dependent on leaf litter for their survival (Dumpet and Johnson 1981; Ratsirarson *et al.* 2002). The sub families of ants that occurred within this study have been noted as being predominantly predatory (Skaife *et al.* 1979). Arthropod density and abundance are affected by management practices such as a fire event (Huhta *et al.* 1967; Christensen and Abbott 1989; Parr *et al.* 2002). The abundance and density of ants was found to be affected by the fire event (Figure 4.15). Similarly Dress and Boerner (2004) found that a reduction in the abundance and density of soil fauna was found to increase with fire frequency within an oak – hickory forest, when compared to that of unburned reference sites.

The decline in ant abundance within the burned plots is attributed to quantity of available food that remained on the burned sites. Similar effects have been noted to occur within various studies that related the decreases in invertebrate populations to food availability (Huhta *et al.* 1967; Huhta 1976; Bird and Chatarpaul 1986). In comparison the decline in soil and litter invertebrate fauna within the unburned plots over time may be due to the fact that after a period of time following the clearfelling event the availability of food was diminished (Huhta *et al.* 1967; Bird and Chatarpaul 1986).

Invertebrate fauna were found in the litter and top layers of the soil prior to fire and further down the soil profile after the fire event had occurred (Decaëns *et al.* 2001). This was similar to what was found to occur within this study with soil invertebrate fauna being found to occur further down the soil profile following the fire event, in comparison to that of the unburned sites in which invertebrate fauna were found in the slash and humus layers (Figure 4.12). This movement of soil fauna suggests that climatic and micro climatic conditions affect the movement of soil and litter invertebrate species (Bird and Chatarpaul 1986; Sharon *et al.* 2001; Rossi and Blanchart 2005). Soil temperature and moisture affect the movement of various organisms that are a food

source for predatory and parasitic organisms found within a particular site (Huhta *et al.* 1967). Soil and litter invertebrates undergo a seasonal variation in their position in which they are found within the soil profile being found further down the soil profile during the dry months and within the slash during the wetter months (Huhta *et al.* 1967; Sharon *et al.* 2001).

Invertebrate species diversity was low within all sites both before and after a fire event. Remaining soil fauna, however, were shown to survive within a plantation even after a disturbance event. Fire was shown to merely affect the abundance and density of various invertebrate fauna. Low diversity may occur within the plantation as a result of remaining fuel and vegetation affecting food quality, quantity and shelter on the site (Huhta *et al.* 1967; Huhta 1976; Bird and Chatarpaul 1986).

This study has found that:

- 78 – 99% of phosphorus, nitrogen and carbon were lost from the slash as a result of burning.
- Less potassium, sodium and magnesium were lost from the slash that was piled due to the nature of the material burned, enriching the remaining ash.
- Calcium was enhanced in the ash from the slash piles due to the woody nature of the material.
- Fire increased both nitrogen and phosphorus mineralization in the soil for a 30 day period and thereafter the effect was non significant.
- Invertebrate faunal species diversity was low in all sites; before and after fire.
- Millipedes occurred in the majority of plots in spring, prior to the fire and ants dominated in summer, post fire.
- Fire did not significantly change faunal species diversity.

Recommendations.

Soil nutrient conservation.

1. If the objective of the plantation manager is to improve the nutrient sustainability of the system then it is recommended that forestry managers spread the slash and burn under optimal conditions. This will decrease the intensity of the fire thereby reducing the loss of nutrients during a fire event.
2. If calcium needs conserving then slash should be piled.
3. If nitrogen and phosphorus needs conserving then the slash should be spread.

Invertebrate fauna conservation.

1. In order to enhance the invertebrate faunal community it is recommended that the inter-rotational period is kept as short as possible.
2. It is recommended that this study be complimented by a seasonal study within a plantation versus the grassland, so as to determine the impacts of afforestation on soil faunal diversity and species richness.

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