University of Alberta

Short-term Post-Fire Nitrogen Dynamics in a Riparian Mixedwood Stand in Alberta

by

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1. INTRODUCTION

Most of Alberta's boreal forest lies within the Boreal Plain Ecozone where mixedwood stands comprised of both deciduous and coniferous species are common. Fire is an important mechanism of natural disturbance in mixedwood stands of the Western Boreal Forest (WBF) and has a significant effect on soil microbiological, physical, and chemical properties that affect nutrient cycling (Ahlgren and Ahlgren 1960; Raison 1979; MacLean et al. 1983; Neary et al. 1999; Chanasyk et al. 2003). Of the six essential macronutrients required for plant growth, nitrogen (N) is considered most limiting in the boreal ecosystem (Kimmins 1996). Fire alters the soil environment by heating the soil, oxidizing the forest floor and surface vegetation, and depositing ash; these changes have a significant effect on post-fire N dynamics (Raison 1979; MacLean et al. 1983; Neary et al. 1999). The greatest changes in soil N cycling occur in the short-term (< 1 year) after fire, however, few have focused on the short-term changes in N dynamics after an understory fire with little canopy combustion. Instead, short-term post-fire N dynamics have been been determined by experimental or prescribed burning in open areas (e.g. DeBano et al. 1979; Jensen et al. 2001; Frey et al. 2003), laboratory heating experiments (e.g. Dunn et al. 1979; Choromanska and DeLuca 2002), or post-hoc comparison of stands that experienced high crown fire severity with similar unburned control stands (e.g. Khanna and Raison 1986; Dyrness et al. 1989; Dumontet et al. 1996). Research indicates that in forest ecosystems such as the WBF, low severity, understory fires dominate the fire regime (e.g. Wright and Agee 2004); this evidence suggests that further examination of their effect on N dynamics is needed.

In the landscape, the riparian zone is commonly defined as that area adjacent to a water body that is influenced by either elevated water tables or frequent flooding and may play a key role in the regulation of stream N concentrations (Cirmo and McDonnell 1997; Naiman and Decamps 1997). Little research on riparian zones along forested headwater streams has been conducted in the Boreal Plain Ecozone, where geologic and climatic conditions (a thick layer of glacial drift over bedrock, low precipitation and high potential evapotranspiration) create different hydrological patterns from those in the Boreal Shield Ecozone or humid climates (Devito et al. 2004). In this region, ephemeral or intermittent streams comprise a large (>50%) proportion of the surface drainage network however, few have examined whether the riparian zones of these streams differ in terms of N availability (e.g. net and gross rates of mineralization and nitrification) from the rest of the stand. Devito et al. (2004) indicated that ephemeral streams could be a source of runoff in the Boreal Plain Ecozone; high N availability in ephemeral riparian zones may therefore indicate greater potential for downstream transfer of N during storm events. Therefore, study of N dynamics in ephemeral or intermittent riparian zones may further the understanding of their role in the surface water drainage network.

Adoption of the natural disturbance paradigm to guide forest management practices in the WBF is problematic; through this paradigm riparian zone harvesting may be justified by the acceptance of fire, or severe disturbance, as an important mechanism that maintains the ecological functionality of riparian zones. Research suggests that fire is not uncommon in the riparian zone of ephemeral or intermittent streams (Alberta Sustainable Resource Development 2001; Andison and McCleary 2002; Macdonald et al. 2004). However, high moisture content in the riparian zone of an ephemeral or intermittent riparian valley may reduce fire severity (heating intensity and duration) and favour rapid post-fire microbial recovery compared with an upper slope position in the same stand (Dunn et al. 1979; Nelson 2001). Therefore, the post-fire change that occurs in soil properties, including N availability, in an ephemeral riparian zone may be less than those changes that occur after fire in the surrounding, drier landscape (Raison 1979; Neary et al. 1999). This trend of a lesser impact in the riparian compared to the upland area within a stand may be even more apparent in an understory fire simulated by experimental burning.

This thesis assessed the influence of experimental burning, differences in moisture content due to slope position, and the interaction of these factors on the soil N cycle. Several parameters were assessed including total, extractable and microbial biomass carbon (C) and N contents, inorganic N (NH_4^+ and NO_3^-) content, net and gross rates of mineralization and nitrification, foliar and total plant N content, soil pH, soil moisture content, and soil temperature. I hypothesized that net and gross rates of mineralization and nitrification as well as total, extractable and microbial biomass C and N contents would be higher in the lower slope, riparian position compared with the upper slope position due to the positive effect of moisture content on microbial activity. I further hypothesized that high moisture content would reduce the penetration of heat energy into soil and therefore forest floor removal in the riparian compared with the upper slope position, resulting in lesser differences in the soil N cycle in the riparian position between the unburned and experimentally burned treatments.

This thesis is organized into five chapters. Following this introductory chapter, the next chapter (Chapter 2) summarizes the scientific literature on the soil N cycle, focusing on commonly assessed biotic and abiotic components within mature mixedwood stands and the short, mid and long term effects of fire on some of these soil properties. Hypotheses regarding the expected differences between the riparian, lower slope and upper slope positions within a riparian mixedwood stand in the WBF are also presented in Chapter 2. Chapter 3 presents and discusses the results of the effect of slope position and experimental burning on several soil properties including soil temperature, soil moisture content, soil pH, and total, extractable and microbial biomass C and N contents. Chapter 4 focuses on the impact of slope position and experimental burning on NH_4^+ and NO_3^- contents, net N mineralization and nitrification rates, gross N mineralization, immobilization and nitrification rates, and foliar and total plant N content. The last chapter (Chapter 5) synthesizes the results of Chapters 3 and 4 into two conceptual models that illustrate the effect of high moisture content in the lower slope, riparian position on the N cycle in mature and recently burned areas.

1.1. References

Ahlgren, I.F. and Alhgren, C.E. 1960. Ecological effects of forest fires. Bot. Rev. **26**: 483-533.

Alberta Sustainable Resource Development. 2001. Chisholm fire reports [online]. Available from http://www3.gov.ab.ca/srd/forests/chisholm [accessed January 20 2003].

Andison, D.W. and McCleary, K. 2002. Disturbance in Riparian Zones on Foothills and Mountain Landscapes of Alberta. Alberta Foothills Disturbance Ecology Research Series Report Number 3. Bandaloop Landscape-Ecosystem Services, Belcarra.

Chanasyk, D.S., Whitson, I.R., Mapfumo, E., Burke, J.M. and Prepas, E.E. 2003. The impacts of forest harvest and wildfire on soils and hydrology in temperate forests: a baseline to develop hypotheses for the Boreal Plain. J. Environ. Eng. Sci. **2**: S51-S62.

Choromanska, U. and DeLuca, T.H. 2002. Microbial activity and nitrogen mineralization in forest mineral soils following heating: evaluation of post-fire effects. Soil Biol. Biochem. **34**: 263-271.

Cirmo, C. P. and McDonnell, J.J. 1997. Linking the hydrologic and biogeochemical controls of nitrogen transport in near-stream zones of temperate-forested catchments: a review. J. Hydrol. **199**: 88-120.

Devito, K.J., Creed, I.F and Fraser, C.J.D. 2004. Controls on runoff from a partially harvested aspen forested headwater catchment, Boreal Plain, Canada. Hydrol. Proc. In Press.

DeBano, L.F., Eberlein, G.E. and Dunn, P.H. 1979. Effects of burning on chaparral soils: I. Soil nitrogen. Soil Sci. Soc. Am. J. **43**: 504-509.

Dumontet, S., Dinel, H., Scopa, A., Mazzatura, A. and Saracino, A. 1996. Post-fire soil microbial biomass and nutrient content of a pine forest from a dunal Mediterranean environment. Soil Biol. Biochem. **28**: 1467-1475.

Dunn, P.H., DeBano, L.F. and Eberlein, G.E. 1979. Effects of burning on chaparral soils: II. Soil microbes and nitrogen mineralization. Soil Sci. Soc. Am. J. **43**: 509-514.

Dyrness, C.T., Van Cleve, K. and Levison, J.D. 1989. The effect of wildfire on soil chemistry in four forest types in interior Alaska. Can. J. For. Res. **19**: 1389-1396.

Frey, B.R., Lieffers, V.J., Munson, A.D. and Blenis, P.V. 2003. The influence of partial harvesting and forest floor disturbance on nutrient availability and understory vegetation in boreal mixedwoods. Can. J. For. Res. **33**: 1180-1188.

Jensen, M., Michelsen, A. and Gashaw, M. 2001. Responses in plant, soil inorganic and microbial nutrient pools to experimental fire, ash, and biomass addition in a woodland savanna. Oecologia. **128**: 85-93.

Khanna, P.K. and Raison, R.J. 1986. Effect of fire intensity on solution chemistry of surface soil under a Eucalyptus pauciflora forest. Aust. J. Soil Res. **24**: 423-434.

Kimmins, J.P. 1996. Importance of soil and the role of ecosystem disturbance for sustained productivity of cool temperate and boreal forests. Soil Sci. Soc. Am. J. **60**: 1643-1654.

Macdonald, E., Burgess, C.J., Scrimgeour, G.J., Boutin, S., Reedyk, S. and Kotak, B. 2004. Should riparian buffers be part of forest management based on natural disturbance? For. Ecol. Manage. **187**: 185-196.

MacLean, D.A., Woodley, S.J., Weber, M.G. and Wein, R.W. 1983. Fire and nutrient cycling. *In* The Role of Fire in Northern Circumpolar Ecosystems. *Edited by* R.W. Wein and D.A. MacLean. John Wiley & Sons Ltd., New York. pp. 111-132.

Naiman, R.J. and Decamps, H. 1997. The ecology of interfaces: riparian zones. Annu. Rev. Ecol. Syst. **28**: 621-658.

Neary, D.G., Klopatek, C.C., DeBano, L.F. and Ffolliott, P.F. 1999. Fire effects on belowground sustainability: a review and synthesis. For. Ecol. Manage. **122**: 51-71.

Nelson, R.M. Jr. 2001. Water relations of forest fuels. *In* Forest Fire Behaviour and Ecological Effects. *Edited by* E.A. Johnson and K. Miyanishi. Academic Press, San Diego. pp. 79-145.

Raison, R.J. 1979. Modification of the soil environment by vegetation fires, with particular reference to nitrogen transformations: a review. Plant Soil. **51**: 73-108.

Wright, C.S. and Agee, J.K. 2004. Fire and vegetation history in the eastern Cascade Mountains, Washington. Ecol. App. **14**: 443-459.

2. LITERATURE REVIEW

2.1. Introduction

Frequent and intense forest fires shape the forested landscape of the Boreal Plain Ecozone within the Canadian boreal forest (Johnson 1992; Barnes et al. 1998). Fire has a significant impact on soil and site properties; the degree of post-fire change has been related to forest floor (LFH) removal, which is a function of pre-burn forest floor depth, bulk density, and moisture content (Miyanishi and Johnson 2002). Of the variables affecting forest floor combustion, moisture content is known to increase along a hillslope gradient from the dry upper knoll to the wet lower slope (Samran et al. 1995; Huang and Schoenau 1997; Offord 1999). Within a stand, variation in moisture content may influence the pattern of forest floor combustion and therefore the changes that occur in nutrient, including nitrogen (N), cycling after fire (Fyles et al. 1991; Brais et al. 2000; Miyanishi and Johnson 2002). Study of the impact of fire on the N cycle, as a proxy for post-fire change in soil fertility, is extensive because this macronutrient is considered most limiting to forest productivity and most subject to oxidation and volatilization loss during fire (MacLean et al. 1983; Raison et al. 1985; Kimmins 1996).

Riparian zones are generally defined as moist, stream or lakeside aquatic-terrestrial transition zones that may play a key role in the regulation of stream N concentrations (Cirmo and McDonnell 1997; Naiman and Decamps 1997). Little research has been conducted on riparian zones along forested headwater streams in the Boreal Plain Ecozone, where ephemeral or intermittent streams are common. In these streams, the riparian zone may be wetter than an upper slope position within the same stand; this difference in soil moisture content may influence soil properties and N availability in the two positions.

Fire is not uncommon in the riparian zone of small, ephemeral or intermittent streams in the Boreal Plain Ecozone (Alberta Sustainable Resource Development 2001). Higher soil moisture content and relative humidity combined with a decline in air temperature and wind speed in ephemeral or intermittent riparian valleys may reduce fire severity (heating intensity and duration) in the riparian zone compared with the drier upland area (Samran et al. 1995; Nelson 2001; Dwire and Kauffman 2003). This suggests that the changes in N dynamics that occur after fire may be different in the two positions; this hypothesis merits further investigation.

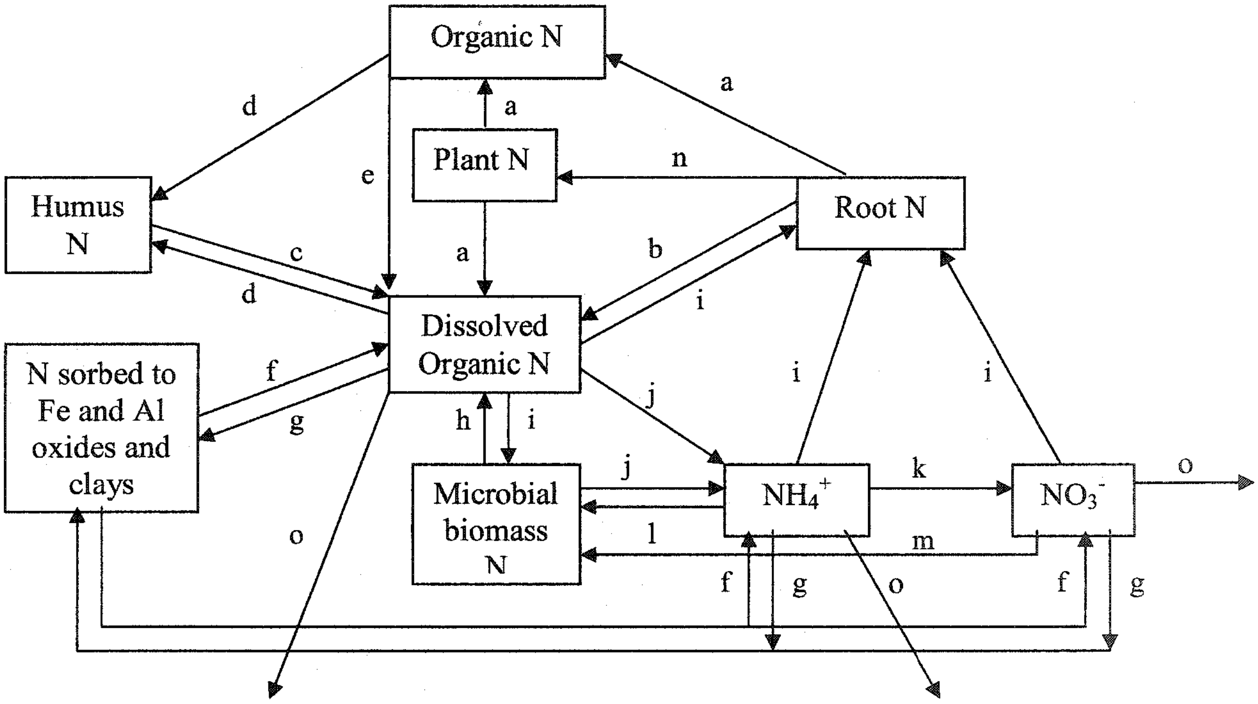
The objective of this literature review was to provide the background information necessary to hypothesize the impact of differences in moisture content due to slope position within an ephemeral or intermittent valley in the WBF on pre- and post-fire trends in the soil N cycle. The review focuses on N cycling in mixedwood stands in the WBF where possible; these stands are common within the forested landscape of the Boreal Plain Ecozone but have received relatively little research attention. The review is separated into two sections: the first section discusses commonly measured indices of N dynamics within the internal N cycle and the relationships between them. The influence of soil acidity, substrate quality, soil temperature and soil moisture content on the internal N cycle are also reviewed. The second section discusses the short-, mid- and long-term effects of fire on some components of the soil N cycle and the post-fire change that occurs in soil acidity, substrate quality, and soil temperature and moisture content. Each section ends with hypotheses on the trends expected in net N mineralization and nitrification rates, gross N mineralization, NH_4^+ and NO_3^- consumption, and nitrification rates, microbial biomass C and N, microbial C/N ratio, soluble organic N, foliar and total plant N content, pH, soil C/N ratio, soil temperature, and soil moisture content between a riparian, lower slope and a non-riparian upper slope position of a mixedwood stand in the WBF. The first section presents the hypotheses for differences without fire while the second section presents the hypotheses for short-term changes after fire.

2.2. Nitrogen In Riparian Mixedwood Stands

2.2.1. Commonly Measured Components in the Soil N Cycle

The biogeochemical N cycle includes an external component where N is added or removed from the forest ecosystem, and an internal cycle that involves the conversion or transfer of N from one ecosystem pool to another (Figure 2.1).

Figure 2.1. The internal N cycle in forest ecosystems. The processes involved are denoted by lowercase letters and include: (a) litterfall and plant death; (b) root exudation; (c) decomposition; (d) humification; (e) solubilization and decomposition; (f) desorption; (g) adsorption; (h) microbial death; (i) assimilation; (j) ammonification; (k) nitrification; (l) immobilization of NH_4^+; (m) immobilization of NO_3^-; (n) translocation; (o) N export by leaching, denitrification, volatilization, etc. Net N mineralization equals (j + k) – (l – m), net nitrification equals k – m, gross N mineralization equals j + k, gross NH_4^+ consumption equals m+l, gross nitrification equals k, and gross NO_3^- consumption equals m. Based on Davidson et al. (1992), Hart et al. (1994a), Offord (1999), and Qualls (2000).



2.2.1.1. Net N Mineralization and Net Nitrification Rates

The net N mineralization rate is measured as the amount of NH_4^+ and NO_3^- present in a soil incubated over a certain period of time minus that present in the soil before incubation. The net nitrification rate is similarly measured by subtracting the initial NO_3^- pool from the post-incubation NO_3^- pool (Figure 2.1). Net rates of mineralization and nitrification can be assessed in the laboratory using aerobic incubations (Stanford and Smith 1972) or *in situ* using various techniques including the buried bag method (Eno 1960), and the ion exchange resin bag (Adams et al. 1989) or core methods (Hart and Firestone 1989). Net rates can be positive, zero, or negative (Wang et al. 2001).

Low contributions of N from wet and dry deposition (2 to 4 kg N ha^{-1} yr^{-1}) and nitrogen fixation (symbiotic 2 to 6 kg N ha^{-1} yr^{-1}; non-symbiotic 1 to 5 kg N ha^{-1} yr^{-1}) suggest that mineralization is the rate-controlling step of inorganic N supply in the boreal forest (Binkley 1981; Fyles and McGill 1987; Shaw et al. 1989; Huang and Schoenau 1997; Ohrui et al. 1999; Bhatti et al. 2002). Net N mineralization rates vary from 9 to 125 kg N ha^{-1} yr^{-1}, typically falling between 15 to 25 kg N ha^{-1} yr^{-1}, in Canadian boreal forests (Fyles et al. 1991; Bhatti et al. 2002). In mixedwood stands, the net N mineralization rate is related to the dominant tree species; generally deciduous stands demonstrate greater rates of net N mineralization than coniferous stands (MacLean and Wein 1978; Paré and Bergeron 1996; Reich et al. 1997; Evans et al. 1998; Côté et al. 2000; Giardina et al. 2001; Vance and Chapin 2001; Ollinger et al. 2002). Research from pure and mixedwood stands in the WBF supports this conclusion. Carmosini et al. (2003) reported mean cumulative net N mineralization rates of 5.02 kg N ha^{-1} in the forest floor in July and August in mature trembling aspen (*Populus tremuloides* Michx.) stands while Walley et al. (1996) demonstrated median cumulative net N mineralization of 47 to 78 kg N ha^{-1} in the surface 15 cm of soil over an 8 week incubation in a trembling aspen-white spruce (*Picea glauca* (Moench) A. Voss) mixedwood stand in Saskatchewan, and Little et al. (2002) reported that cumulative net N mineralized in July and August in a white spruce-trembling aspen mixedwood stand ranged from 0.28 to 4.03 kg N ha^{-1} in the forest floor and 2.84 to 7.32 kg N ha^{-1} in the 0 to 10 cm mineral soil layer.

Once released by mineralization, one of the fates of NH_4^+ is conversion to NO_3^- autotrophically by *Nitrosomonas*, *Nitrosolobus*, *Nitrospira*, *Nitrosovibrio*, and *Nitrobacter* species or heterotrophically by the oxidation of organic amino N via organic nitroso compounds (Barraclough and Puri 1995; Havlin et al. 1999). Typically, heterotrophic nitrification produces less than 8 percent of total nitrate (Barraclough and Puri 1995) and autotrophic nitrification dominates the nitrification process (Stark and Hart 1997). In mature mixedwood stands of the WBF, the annual net nitrification rate is close to zero (Walley et al. 1996; Little et al. 2002). Nitrification may be inhibited by low pH (<4.5), low temperature, anaerobic conditions, low C and NH_4^+ availability, low nitrifier populations, poor competitive ability of nitrifying organisms for NH_4^+, or the presence of allelopathic phenolic compounds (Hart et al. 1994b; Walley et al. 1996; Neary et al. 1999; Ste-Marie and Paré 1999). However, measurement of gross nitrification rates has demonstrated that nitrification, although less important than mineralization in forest soils, does occur and that NO_3^- is quickly immobilized by soil microbes (Davidson et al. 1992; Stark and Hart 1997).

2.2.1.2. Gross N Mineralization, Consumption, and Nitrification Rates

Gross rates of N mineralization, NH_4^+ and NO_3^- consumption, and nitrification are measured using isotope pool dilution or enrichment methods (Davidson et al. 1991; Barraclough and Puri 1995). Gross rates of mineralization and nitrification do not include consumption of NH_4^+ or NO_3^- (Figure 2.1) and can be much greater than net rates (Hart et al. 1994a); they have been suggested as a reference to evaluate other assessment methods for N mineralization and nitrification (Wang et al. 2001). In a trembling aspen stand in Alberta, Carmosini et al. (2002) reported that, in the forest floor and 0-10 cm mineral soil respectively, gross ammonification rates were 29.5 and 0.61 g N kg^{-1} yr^{-1}; gross NH_4^+ immobilization rates were 31.3 and 0.53 g N kg^{-1} yr^{-1}; gross nitrification rates were –0.10 and 0.0 g N kg^{-1} yr^{-1}; and gross NO_3^- immobilization rates were 0.63 and 0.12 g N kg^{-1} yr^{-1}. This study suggested that a significant proportion of NH_4^+ and NO_3^- is microbially immobilized in forest soils in the Western Boreal Forest (WBF); further research is required to validate these results.

Gross and net rates of mineralization and nitrification are generally poorly correlated due to the reflection of both N mineralization and immobilization processes over a longer time period in net rates. Several authors have illustrated contrasting trends in gross and net N mineralization or nitrification rates between treatments (Davidson et al. 1992; Hart et al. 1994b; Stark and Hart 1997; Stottlemyer and Toczydlowski 1999b; Verchot et al. 2001). The net rate depends on the difference between gross mineralization and immobilization rates; when gross mineralization exceeds gross immobilization rates, net and gross rates of mineralization may be positively correlated (e.g. Verchot et al. 2001; Zaman and Chang 2004). In general, gross rates of N transformations more accurately reflect the dynamics of the microbial population while net rates indicate the rate of inorganic N production available for plant uptake.

2.2.1.3. Microbial Biomass C and N

Soil microbial populations perform C and N transformations and their biomass stores large amounts of energy and nutrients; microbial biomass is perceived as a relatively labile component of soil organic matter. Microbial biomass C (MB-C) and N (MB-N) are generally measured to indicate the effect of changes in substrate quality, quantity, and environmental conditions on the soil organic matter pool, to which they are positively related (Chang et al. 1995; Li et al. 2004). However, there are few studies on either MB-C or MB-N in either pure or mixedwood stands in the WBF. In the upper 15 cm of forest soil in a trembling aspen-white spruce mixedwood stand in Saskatchewan, Walley et al. (1996) reported median values of MB-C, which varied between slope positions, of 560 to 590 kg C ha^{-1} and MB-N, which did not vary with slope position, of 100 kg N ha^{-1}.

Analysis of the correlation between microbial biomass and net and gross N mineralization rates is often conducted in order to elucidate the effect of microbial biomass on N immobilization and release. Research on this relationship is inconclusive; negative (Vance and Chapin 2001; Zaman and Chang 2004), positive (Davidson et al. 1992; Hart et al. 1994b; Bohlen et al. 2001; Smolander and Kitunen 2002), and no (Holmes and Zak 1994; Hart et al. 1994b; Puri and Ashman 1998) correlation between MB-C and MB-N and either net N mineralization rate, gross N mineralization rate or inorganic N content have been demonstrated. This suggests that in some situations, the microbial biomass may act as a nutrient sink and immobilize nutrients, while in others it may act as a nutrient source and release soluble C and N compounds for forest soil microorganisms. Stark and Hart (1997) suggested that where the microbial biomass turns over quickly, such as in young forest stands, nutrient retention by the microbial biomass is promoted; however, this hypothesis was not corroborated in a study of forested stands of varying age in coastal British Columbia (Chang et al. 1995). Clearly, additional research is required on MB-C, MB-N, and net and gross rates of mineralization in order to understand the nature of their relationship and roles in the internal N cycle.

The microbial C/N ratio can be used to indicate changes in the microbial population structure that may affect mineralization-immobilization processes. Low microbial C/N ratios, generally associated with bacteria, have been shown to produce net N mineralization, and high microbial C/N ratios, generally associated with fungi, produce net N immobilization (Chang et al. 1995; Tate 1995). Fungi are known to predominate in soils containing recalcitrant organic matter with a high lignin content which is slowly decomposed because the microorganisms are generally nutrient limited (McClaugherty et al. 1985; Little et al. 2002). Few studies have related the microbial C/N ratio to soil N cycling in the WBF; however, Walley et al. (1996) reported that the microbial C/N ratio varied only 0.5 units between different slope positions within a mixedwood stand in Saskatchewan. This suggests that in a riparian mixedwood stand, the microbial C/N ratio may not explain a significant proportion of the variation in soil N cycling.

2.2.1.4. Soluble and Dissolved Organic N

Soluble organic matter (e.g. soluble organic C, N and P) includes organic molecules able to pass through a 0.45 μm filter that are potentially soluble but adsorbed on oxyhydroxides, clays, or solid organic matter surfaces, and those which are dissolved in soil solution (e.g. dissolved organic C, N, and P) (Qualls 2000; Kalbitz et al. 2000). Measurement of soluble organic N (SON) has shown that it varies seasonally and by horizon and may represent from 85 to 95 percent of total soluble N in forest soils, however only a small proportion of SON is dissolved in soil solution (Chang et al. 1995; Huang and Schoenau 1998; Devito et al. 1999; Hannam and Prescott 2003). Dissolved organic C, N, and P (DOC, DON, and DOP respectively) primarily originate in the upper forest floor via gradual leaching from senescing fresh litter, desorption, dissolution or hydrolysis of humus, microbial death, and root exudation (Qualls et al. 1991; Huang and Schoenau 1998; Qualls 2000; Kalbitz et al. 2000); concentrations in soil solution depend on the balance between these additions and removal via microbial hydrolysis, root uptake, and adsorption reactions (Qualls et al. 1991; Kalbitz et al. 2000; Qualls 2000). DON leaches from the forest floor to allow humus formation in mineral soil (Qualls et al.

1991), is a source of free proteins and amino acids for certain plants (Chapin et al. 1993; Kielland 1994; Raab et al. 1996), and can be exported from soils that are either low in oxyhydroxides or clay, dominated by macropore flow, or periodically saturated (Kalbitz et al. 2000). There is no published research on either SON or DON in mixedwood stands of the WBF; however, Huang and Schoenau (1998) reported that water soluble N (similar to SON) in a Saskatchewan trembling aspen stand ranged from 0.06 to 0.10 g N kg^{-1} in the L, 0.03 to 0.07 g N kg^{-1} in the F, 0.02-0.05 g N kg^{-1} in the H, and 0.002-0.02 g N kg^{-1} in the Ae.

Hart et al. (1994b) suggested that soluble organic C (SOC) and N were major C and N sources for forest soil microorganisms and therefore should be inversely correlated to MB-C and MB-N when SOC and SON are neither limiting nor in over-supply. Other research has demonstrated positive correlations between soluble and dissolved forms of C and N and MB-C and MB-N in the forest floor and upper mineral soil (0-10 cm) (Chang et al. 1995; Smolander and Kitunen 2002; Hannam and Prescott 2003), implying that either the microbial biomass was a significant source of SOC and SON or microbial growth was limited by the lack of SOC and SON. Myrold (1987) suggested that MB-N was positively correlated to site fertility; these trends may therefore indicate that where the microbial biomass is large, microorganisms produce soluble C and N faster than they are able to use it.

2.2.1.5. Foliar and Total Plant N Content

N cycling research is generally conducted to examine the availability of N for plant uptake and growth (e.g. Reich et al. 1997; Kimmins 1996). Roots take up NH_4^+, NO_3^-, and soluble free and combined amino acids that are converted into NH_4^+; these N compounds are then combined with C to form amino acids and subsequently proteins (Havlin et al. 1999). Analysis of the total N content in the whole plant or part of the plant is based on the idea that total N content indicates N supply in the soil (Havlin et al. 1999); this type of analysis is generally performed on current year tree foliage in forest ecosystems (van den Driessche 1974; Chang et al. 1995; Macdonald et al. 1998; Schade et al. 2002). Ollinger et al. (2002) found a positive relationship between foliar N and the net N mineralization rate in deciduous but not coniferous stands. In addition to total N content, understory plant composition and percent cover may also be used to indicate N availability in boreal forest stands (Fyles and McGill 1987; Strong et al. 1991; La Roi et al. 1998; Driscoll et al. 1999; Prescott et al. 2000b). Few studies have examined whether differences in foliar N or total plant N content of understory plants may indicate changes in N availability within a forest stand, however, the ease of plant tissue sampling compared to soil sampling suggests that this hypothesis merits further investigation.

2.2.2. Abiotic Influences on the Soil N Cycle

2.2.2.1. Soil Acidity

The acidity of the soil solution is based on the activity of the hydrogen ion, H^+, and is commonly expressed in pH units. Acidity in soils originates from organic matter, aluminosilicate clays, Fe and Al hydrous oxides, exchangeable aluminum, soluble salts, and carbon dioxide (Havlin et al. 1999); acidic soils favour fungi because at low pH most bacteria are unable to maintain their internally neutral pH (Tate 1995). In the boreal forest, soil pH generally declines from trembling aspen to white birch (*Betula papyrifera* Marsh.) to white spruce to black spruce (*Picea mariani* (Mill.) BSP) stands (Dyrness et al. 1989; Huang and Schoenau 1996; Bauhus et al. 1998; Driscoll et al. 1999; Vance and Chapin 2001). Therefore, the pH of deciduous-dominated mixedwood stands should generally be greater than that of conifer-dominated mixedwood stands in the WBF. Research from the WBF supports this hypothesis; median soil pH was 5.6 in the upper 15 cm of soil in a trembling aspen-white spruce mixedwood stand in Saskatchewan, and ranged from 4.2 to 6.1 in the forest floor and from 4.7 to 5.6 in the Ae horizon in white spruce-trembling aspen mixedwood stands in Saskatchewan and Alberta (Walley et al. 1996; Offord 1999).

The relationship between pH and mineralization and nitrification rates may vary with stand type. Ste-Marie and Paré (1999) showed that net N mineralization rates were stimulated by positive and negative changes to the original soil pH in jack pine (*Pinus banksiana* Lamb.), white spruce, trembling aspen and white birch stands; nitrification rates increased with pH in the deciduous stands but did not change in the coniferous stands. This suggests that an increase in pH in trembling aspen-dominated mixedwood stands may have a positive effect on both net N mineralization and nitrification rates in the WBF.

2.2.2.2. Substrate Quality

Substrate quality commonly refers to the N content of leaf, fine root, and woody organic substrate present in the forest floor and mineral soil in comparison to the C content; it controls microbial utilization and is commonly assessed using either the C/N ratio or lignin/N ratio (McClaugherty et al. 1985; Palm and Sanchez 1991). When readily soluble C compounds and cellulose are abundant, C/N ratio is the best predictor of N availability. Once these compounds are exhausted and microbes are C limited, lignin/N ratio or polyphenol/N ratio may be a more appropriate measure of substrate quality (McClaugherty et al. 1985; Palm and Sanchez 1991; Giardina et al. 2001; Prescott et al. 2000a). In mixedwood stands of the WBF, C/N ratios in the forest floor vary with dominant tree species while the C/N ratio of the A horizon generally ranges from 11 to 16 (Offord 1999; Little et al. 2002). In white spruce-dominated mixedwood stands, the C/N ratio varies from 23 to 34 in the forest floor, tends to increase in both the forest floor and A horizon of the lower slope position of a hillslope, and has been negatively correlated to specific net (net rates normalized to a N basis) and net rates of N mineralized (Offord 1999; Little et al. 2002). In trembling aspen-dominated mixedwood stands, the C/N ratio is much lower in the forest floor (17 to 18) compared with conifer-dominated mixedwood stands (Walley et al. 1996; Offord 1999); Offord (1999) demonstrated a lower C/N ratio in the forest floor of the lower slope position that was negatively correlated to amount of specific net N mineralized but found no relationship between C/N ratio and N mineralization in the A horizon (Walley et al. 1996; Offord 1999). This suggests that even a small increase in the soil C/N ratio may reduce the net N mineralization rate in some WBF soils.

2.2.2.3. Soil Temperature and Soil Moisture Content

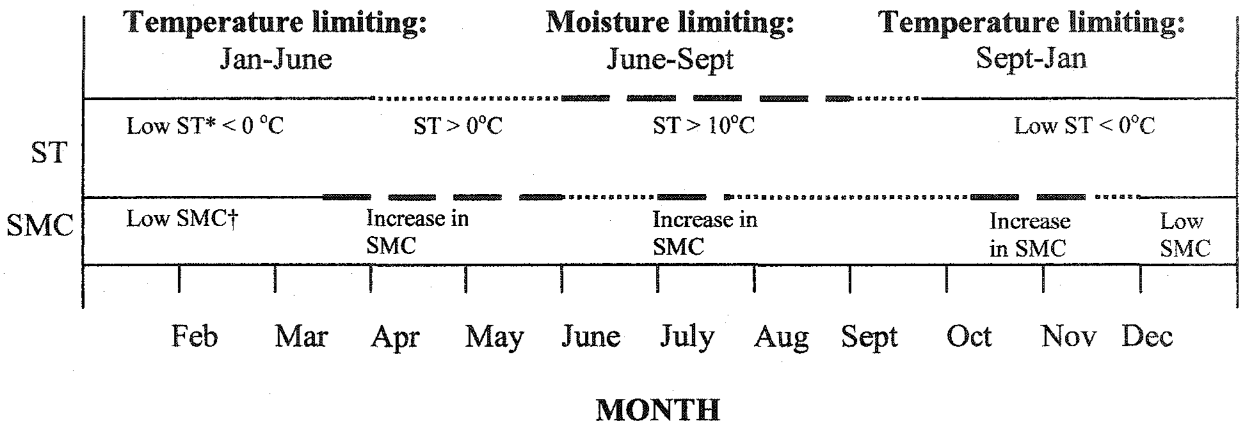
Of the abiotic factors affecting microbial activity, soil temperature and moisture content may exert the greatest control over N cycling; research suggests that their effects may be linked. The relationship between soil temperature, soil moisture content, and the

N cycle seems to involve two aspects: (1) reduction in microbial activity and N transformations due to low temperature, low soil moisture content, excessively high temperature, or excess moisture (Hanks 1992; Goncalves and Carlyle 1994; Puri and Ashman 1998; Havlin et al. 1999; Zaman and Chang 2004); and (2) exponential increase in rates of N transformations between minimum and maximum temperatures and soil moisture contents for microbial activity (Stanford et al. 1973; Stanford and Epstein 1974; Ellert and Bettany 1992; Goncalves and Carlyle 1994; Verburg et al. 1999; Zaman and Chang 2004).

2.2.2.3.1. Effect of Soil Temperature and Moisture Content on Net and Gross N Mineralization and Nitrification Rates

The mesophilic organisms involved in the internal N cycle are active between 0 and 35°C (Stanford et al. 1973; Tate 1995); the minimum and maximum temperatures for optimum net N mineralization, gross N mineralization, and gross nitrification range from 5 to 10°C and 25 to 35°C, respectively (Ellert and Bettany 1992; Goncalves and Carlyle 1994; Verburg et al. 1999; Stottlemyer et al. 1999b; Zaman and Chang 2004). These minima and maxima vary with soil type; Offord (1999) demonstrated that in the forest floor of a boreal mixedwood stand, specific net NH_4^+ and NO_3^- mineralization rates reached an optimum at 12°C. N mineralization rate increases linearly between permanent wilting point and field capacity (Stanford and Epstein 1974), however, the minimum and maximum soil moisture contents required to limit net and gross N mineralization rates are 20 to 40% field capacity and more than 100% field capacity, respectively (Goncalves and Carlyle 1994; Stottlemyer and Toczydlowski 1999b; Zaman and Chang 2004). The maximum moisture content required to limit the net nitrification rate was 75% field capacity while the gross nitrification rate showed no clear relationship to moisture content in the range of moisture contents studied by Zaman and Chang (2004). Together, the annual fluctuations in soil moisture content and soil temperature may explain much of the seasonal variation in mineralization rates (Ellert and Bettany 1992; Goncalves and Carlyle 1994). The generally observed annual fluctuations in soil moisture content and temperature in the boreal forest, and when each will be most limiting for microbial activity and N transformations, are summarized in Figure 2.2.

Figure 2.2. Expected annual fluctuations in soil moisture content and soil temperature in the boreal forest. The bold lines indicate periods of high moisture content or temperature. Based on Offord (1999), Huang and Schoenau (1997), and Stottlemyer and Toczydlowski (1999a, 1999b).



*ST = soil temperature; †SMC = soil moisture content.

The general trends illustrated in Figure 2.2 may be influenced by factors affecting soil moisture content and soil temperature within and between forest stands such as slope position, soil texture, and soil organic matter content. From May to November, greater soil temperatures in upper slope positions such as knolls and crests and high soil moisture contents in lower slope positions have been demonstrated (Samran et al. 1995; Huang and Schoenau 1997; Offord 1999; Little et al. 2002; Wilson et al. 2002). Presence of clay often results in poor drainage and thereby reduces soil temperature because of higher soil moisture content (Giardina et al. 2001; Bhatti et al. 2002) whereas accumulation of soil organic matter in the forest floor may allow greater temperature and moisture fluctuation as compared to mineral soil (Huang and Schoenau 1997).

In the boreal forest, the maximum NH_4^+ pool sizes, gross N mineralization rates, and net N mineralization rates occur between May and August while NO_3^- pool sizes, net nitrification rates, and gross nitrification rates show little consistent seasonal variation (Huang and Schoenau 1997; Stottlemyer and Toczydlowski 1999a; Stottlemyer and Toczydlowski 1999b). Adequate soil temperature for microbial activity during the summer months suggests that soil moisture content may exert the main control on mineralization and nitrification rates during this period. Research on the effect of slope position on soil N cycling has demonstrated a negative relationship between moisture content and specific net and net N mineralization rates in mixedwood stands in the WBF (Walley et al. 1996; Offord 1999; Little et al. 2002), while a positive correlation was demonstrated in mature and logged trembling aspen stands in the WBF (Carmosini et al. 2003) and in oak-hickory forests in Ohio (Boerner et al. 2000). Positive (Ohte et al. 1997; Evans et al. 1998; Stottlemyer and Toczydlowski 1999a), and negative (Devito et al. 1999) relationships between soil moisture content and nitrification rate and NO_3^- pool size have been reported. Research suggests that NO_3^- release corresponds to periods of reduced heterotrophic microbial abundance that allows the poorly competitive (for NH_4^+) autotrophic nitrifiers to proliferate (Hart et al. 1994b); nitrification may also be stimulated by base rich groundwater discharge (Venterea et al. 2003). This evidence suggests that it is difficult to predict how soil moisture content will affect net and gross rates of mineralization and nitrification in the WBF during the summer months. Carmosini et al. (2002) suggested that the relationship could be masked by high soil C/N ratios that enhance N immobilization in Boreal Plain soils; further research is required to confirm or refute this hypothesis.

2.2.2.3.2. Effect of Soil Temperature and Moisture Content on Microbial Biomass C and N

Zaman and Chang (2004) suggested that microbial biomass C and N were more strongly affected by soil temperature than moisture content in the range of soil temperature and moisture conditions they studied; MB-C and MB-N declined at 40°C as compared to 5 and 25°C while their weak positive correlations with soil moisture content appeared to be influenced by substrate quality. In other ecosystems, MB-C and MB-N may increase slightly from June to September (Puri and Ashman 1998; Bohlen et al. 2001) or show little change between months in the growing season (Holmes and Zak 1994; Chang et al. 1995). Several studies have suggested a positive correlation between microbial biomass size and moisture content, demonstrating a reduction in MB-C and MB-N with increasing soil depth in the dry season and in years of low annual precipitation (Hossain et al. 1995; Bohlen et al. 2001; Hannam and Prescott 2003). Walley et al. (1996) reported that MB-C and MB-N show a high degree of variability, making within-stand detection of seasonal differences or trends due to soil moisture content or temperature difficult.

2.2.2.3.3. Effect of Soil Temperature and Moisture Content on Soluble and Dissolved Organic N

Soil temperature and moisture content may influence both DON and SON similarly, suggesting that SON measurement may indicate changes in DON pools. Greater DOC concentrations, thought to correspond to DON concentrations, with increasing temperature have been observed in the laboratory but not in the field (Kalbitz et al. 2000; Qualls 2000). Similarly, a laboratory study by Zaman and Chang (2004) illustrated that soluble organic C (SOC) was highest at 40°C. High soil moisture content may increase DOC (and therefore DON) fluxes due to greater microbial release of dissolved organic compounds (Kalbitz et al. 2000); a positive correlation between soil moisture content and SON concentrations was demonstrated in the forest floor and upper mineral soil (0-10 cm) of two separate sites in coastal British Columbia (Hannam and Prescott 2003). However, other studies have demonstrated no relationship between soil moisture content and other proxies of soluble organic matter pools such as water soluble N, extractable C or extractable N (Chang et al. 1995; Huang and Schoenau 1998; Devito et al. 1999). These studies suggest positive correlations between soil temperature and soil moisture content and SON and DON pools; these relationships merit further investigation.

2.2.3. Hypotheses on N Dynamics in Riparian Mixedwood Stands

Research suggests that an increase in moisture content from the upper knoll position to the lower slope riparian position within an intermittent or ephemeral riparian valley in the WBF will affect gross and net rates of N mineralization and nitrification, microbial biomass, soluble organic nutrients, and either foliar or total plant N content. These effects may be augmented by changes in soil temperature, substrate quality, or soil pH between the riparian and upland positions. Measurement of several indices of N availability may provide a better indication of the difference in microbial mineralization-immobilization processes in the riparian versus the upland position. Table 2.1

summarizes the differences expected (in both the forest floor and 0-10 cm mineral soil) between a lower slope, riparian position of a mixedwood stand in the WBF as compared with an upper slope position within the same stand based on the review and discussion in this section.

Table 2.1. Hypothesized differences in net N mineralization rate, net nitrification rate, gross N mineralization rate, gross NH_4^+ consumption rate, gross nitrification rate, gross NO_3^- consumption rate, microbial biomass C and N, soluble organic N, soil pH, soil temperature, and soil moisture content in the forest floor (LFH) and upper mineral soil (0-10 cm) between a lower slope, riparian and an upper slope position of a mixedwood stand in the Western Boreal Forest.

Component of the Internal N cycle	**Riparian Lower Slope versus Upper Slope**
Net N mineralization rate	H
Net nitrification rate	H
Gross N mineralization rate	H
Gross NH_4^+ consumption rate	H
Gross nitrification rate	H
Gross NO_3^- consumption rate	H
Microbial biomass C	H
Microbial biomass N	H
Microbial biomass C/N ratio	NC
Soluble organic N	H
Total plant N	H
Soil pH	NC
Soil C/N ratio	NC
Soil temperature	L
Soil moisture content	H

Note: H indicates a higher rate/pool in the riparian position; L indicates a lower rate/pool in the riparian position; and NC indicates no significant change expected.

2.3. Post-Fire Soil Nitrogen Dynamics

Fire combustion of the forest floor has a significant effect on soil nitrogen dynamics. High moisture content in the riparian zone of an ephemeral or intermittent riparian valley in the WBF may affect the post-fire soil N cycle by influencing the rate of forest floor combustion or post-fire microbial activity (Van Wagner 1983; Neary et al. 1999; Miyanishi 2001; Nelson 2001). Changes in the N cycle after fire herein are separated into three categories: short-term changes that occur within one year, mid-term changes that occur between 1 and 15 years, and long-term changes that occur more than 15 years after fire.

2.3.1. Net N Mineralization and Nitrification Rates

Immediately after fire, the deposition of ash and incompletely combusted organic matter, heat-disruption of organic matter, microorganisms and organo-clay complexes, and greater soil temperature may cause an increase in NH_4^+ content and mineralization rates (Ahlgren and Ahlgren 1960; Raison 1979). Nitrification and subsequently NO_3^- content may also rise due to stimulation of autotrophic nitrifiers by high pH, an increase in cation availability, reduced competition for NH_4^+ and the deposition of charcoal that can adsorb nitrification-inhibiting phenolic compounds (Ahlgren and Ahlgren 1960; Raison 1979; Bauhus et al. 1993; Zackrisson 1996). An increase in the NH_4^+ content of the forest floor and upper mineral soil after fire has been observed in various ecosystems (Khanna and Raison 1986; Fenn et al. 1993; Hernández et al. 1997; Monleon et al. 1997; Grogan et al. 2000; DeLuca and Zouhar 2000; Romanyá et al. 2001; Jensen et al. 2001; Choromanska and DeLuca 2001). This initial pulse may endure for up to one year after fire (Grogan et al. 2000; DeLuca and Zouhar 2000) and varies in magnitude with maximum fire temperature (DeBano et al. 1979; Raison 1979; Giovannini et al. 1990). Below 400°C, destruction of organic N in the forest floor including proteins and amino acids commonly releases NH_3 that may be condensed to NH_4^+ while above 400°C most organic N is oxidized as N_2 and N oxides that are released to the air (Raison 1979; DeBano et al. 1979; Giovannini et al. 1990). Therefore, a higher fire temperature results in a lesser increase in NH_4^+ content; Giovannini et al. (1990) demonstrated that experimentally heating mineral soil at 170 and 220°C increased its NH_4^+ content while temperatures from 460 to 900°C reduced its NH_4^+ content. In the field, high surface temperature during fire may favour greater NH_3 condensation into cooler mineral soil; DeBano et al. (1979) reported that a moderate burn increased the NH_4^+ content in the forest floor and upper mineral soil while an intense burn reduced the NH_4^+ content in the forest floor but raised the upper mineral soil NH_4^+ content. Therefore, fire severity may have a significant influence on the post-fire NH_4^+ increase.

High soil NH_4^+ content may also, for a short period after fire, be attributed to an increased rate of N mineralization. Both an initial rise (Prieto-Fernández et al. 1993; Choromanska and DeLuca 2001; Giardina and Rhoades 2001; Wilson et al. 2002; Nardoto and Bustamante 2003), and no change (Monleon et al. 1997; Romanyá et al. 2001) in the rates of post-fire net N mineralization have been reported in the literature. A rise typically lasts for 2 to 4 months but may endure up to 1 year depending on the availability of readily metabolizable C and N compounds (Monleon et al. 1997; Giardina and Rhoades 2001; Romanyá et al. 2001) that increase after fire as a result of plant and microbial death and then decline (Pietikäinen and Fritze 1993; Dumontet et al. 1996; Hernández et al. 1997; Prieto-Fernández et al. 1998). Once these labile compounds are depleted, the rate of net N mineralization is generally reduced in burnt soils that contain a significant proportion of insoluble recalcitrant C compounds (Almendros et al. 1990; Hernández et al. 1997). A reduced rate of net N mineralization has been observed as early as one month after fire in various ecosystems (Fenn et al. 1993; Hossain et al. 1995; DeLuca and Zouhar 2000; Frey et al. 2003; Nardoto and Bustamante 2003; Choromanska and DeLuca 2001; Romanyá et al. 2001). Fire severity and post-fire moisture content, in addition to other factors such as pre-fire N availability, may explain some of the changes in net N mineralization rate after fire. Choromanska and DeLuca (2002) noted an increase in potentially mineralizable N (PMN) after heating mineral soils at 380°C from a ponderosa pine (*Pinus ponderosa* Laws.)-Douglas fir (*Pseudotsuga menziesii* var. *glauca*) forest, but a decline after heating at 160°C. PMN showed a u-shaped relationship to moisture content after fire; the greatest increase in PMN occurred at the moderate (c.a. -1.0 MPa) moisture content while the greatest decline occurred at the lowest moisture content (c.a. -1.5 MPa) (Choromanska and DeLuca 2002). This suggests that a greater increase in the rate of N mineralization will occur in a low severity fire on moderately moist soil compared with a high severity fire on dry soil.

In the short term, fire may stimulate nitrification in some ecosystems; there may be a short lag between burning and NO_3^- production as nitrifying organisms recover from the effect of heating (Hernández et al. 1997; DeLuca and Zouhar 2000; Grogan et al. 2000; Romanyà et al. 2001; Frey et al. 2003; Andersson et al. 2004). Enhanced post-fire nitrification may be explained by reduced microbial competition for NH_4^+ and stimulation of autotrophic nitrifiers by a post-fire increase in pH (Bauhus et al. 1993; Ste-Marie and Pare 1999), less plant competition for NH_4^+ (Kaye and Hart 1997), or charcoal absorption of nitrification-inhibiting phenolic compounds (Zackrisson 1996). A higher post-fire nitrification rate for up to 1 year after fire has been observed in the forest floor and upper mineral soil of burned, and clearcut and burned sites in various ecosystems (Pietikäinen and Fritze 1993, 1995; Giardina and Rhoades 2001; Romanyà et al. 2001; Frey et al. 2003; Andersson et al. 2004).

In the mid-term, a lower rate of net N mineralization and potentially mineralizable N in the forest floor and upper mineral soil have been reported that, in some cases, persisted up to 14 years after fire (Fyles et al. 1991; Walley et al. 1996; Monleon et al. 1997; Driscoll et al. 1999; DeLuca and Zouhar 2000; Choromanska and DeLuca 2001). However, this trend may not be universal across forest stands; Simard et al. (2001) demonstrated a higher net N mineralization rate in the forest floor and upper mineral soil of burned stands compared with control stands 2, 14, and 21 years after fire in black spruce stands in Quebec. This suggests that in the mid-term, net N mineralization rates may be higher or lower in burned as compared with unburned stands; further investigation is required to identify factors determining this change.

There are few studies that have examined the long-term effect of fire on net N mineralization or nitrification rates in the boreal forest. In the eastern United States, net N mineralization and nitrification rates were higher in 80 to 110 year old burned and logged stands as compared to old growth stands (Goodale and Aber 2001). Further research is required to confirm the hypothesis that disturbance increases net rates of N mineralization and nitrification in the long term, however this thesis research will not address the long-term effect of fire.

2.3.2. Total Soil N

The change in total soil N after fire depends on the balance between N losses and additions. N loss occurs during and after fire due to the oxidation of vegetation and surface organic matter, volatilization of solid compounds, convective transfer of particulate matter, smoke and ash, wind and water erosion, and leaching (Raison 1979; DeBano et al. 1979; Vitousek et al. 1982; Raison et al. 1985; Baird et al. 1999; Fisher and Binkley 2000). N may be added to the soil in a burned ecosystem via incompletely combusted plant necromass (Almendros et al. 1990; Baird et al. 1999) and ash (Raison 1979; Raison et al. 1985; Fisher and Binkley 2000; Grogan et al. 2000), site colonization by symbiotic and non-symbiotic N-fixing organisms (MacLean and Wein 1977; Raison 1979; MacLean et al. 1983; Haynes 1986; Fyles and McGill 1987; Barnes et al. 1998; Havlin et al. 1999; Ohrui et al. 1999), and turnover of understory biomass (MacLean and Wein 1977; MacLean and Wein 1978; Lynham et al. 1998; Johnston and Elliott 1998; Grogan et al. 2000). Depending on the balance of losses versus inputs, total N may increase, decline, or remain stable; change varies depending on fire severity and soil horizon.

Total N loss by oxidation is linearly related to the decline in fuel weight due to the similarity of the oxidation temperature of N and destructive distillation and carbonization temperature of organic matter residue, suggesting that greater fire severity increases N loss (Raison 1979; Raison et al. 1985). Decrease in total forest floor N has been documented after slash burning in spruce stands, after severe fire in white spruce, trembling aspen, white birch, jack pine, ponderosa pine-Douglas fir, and lodgepole pine (*Pinus contorta* Loud.)-Engelmann spruce (*Picea engelmannii* Parry ex Engelm.) stands, and immediately after fire in a chapparal ecosystem (Grier 1975; DeBano and Conrad 1978; MacAdam 1987; Dyrness et al. 1989; Baird et al. 1999; Brais et al. 2000).

The effect of fire on total N in the upper 5 to 10 cm of mineral soil is less conclusive than in forest floor; increases in subsoil total N may be caused by leaching of nitrogenous humates from the decomposing forest floor while subsoil N may decline if complete combustion converts the forest floor to low N charcoal and ash (Klemmedson et al. 1962). Post-fire increase in total N in subsoil has been observed in ponderosa pine-Douglas fir, black spruce, white birch, and jack pine stands (Grier 1975; Dyrness et al.

1989; Baird et al. 1999; Brais et al. 2000); post-fire reduction in mineral soil N has been observed in trembling aspen and lodgepole pine-Engelmann spruce stands (Dyrness et al. 1989; Baird et al. 1999). Some studies found no effect of fire on total N in the upper mineral soil (MacAdam 1987; Dyrness et al. 1989; Brais et al. 2000). This information suggests that the post-fire decline in total forest floor N is positively related to fire severity and that the change in total N in the upper mineral soil may be unrelated to fire severity.

The total N content of the forest floor may recover within as little as 2 or as many as 10 years after fire depending on the ecosystem (MacAdam 1987; Pietikäinen and Fritze 1993; Lynham et al. 1998; Driscoll et al. 1999; Choromanska and DeLuca 2001; Simard et al. 2001; Mabuhay et al. 2003). This suggests that the long-term effects of fire on total soil N content may be minimal.

2.3.3. Microbial Biomass C and N

The size of the post-fire MB-C and MB-N pools largely depends on the penetration of heat energy during the fire and the duration of heating since that determines the number of microorganisms killed (Ahlgren and Ahlgren 1960; Raison 1979; Neary et al. 1999). Soil sterilization occurs above 127°C; the temperature at which certain microbial species decline depends on soil moisture content (Neary et al. 1999). It is well known that steam heat results in microbial death at lower temperatures than dry heat (Dunn et al. 1985); in wet soils fungi and bacteria decline at 60°C and 70°C, respectively, while in dry soil, fungi and bacteria decline above 80°C and 90°C, respectively (Choromanska and DeLuca 2002). Soil moisture content after fire has been positively correlated to microbial proliferation and recovery (Dunn et al. 1979). Together, this information suggests that the microbial population will be most severely affected by a hot fire on moist soil, and that microbial recovery will be faster under moist soil conditions.

Heat sterilization by fire has resulted in an immediate decline in MB-C and MB-N after fire in various ecosystems (Dunn et al. 1979; Pietikäinen and Fritze 1993; Fritze et al. 1994; Hossain et al. 1995; Pietikäinen and Fritze 1995; Dumontet et al. 1996; Hernández et al. 1997; Prieto-Fernández et al. 1998; Choromanska and DeLuca 2001;

Choromanska and DeLuca 2002; Wüthrich et al. 2002; Mabuhay et al. 2003). In contrast, some have noted a short-term increase in MB-C and MB-N after fire that is often followed by a decrease (DeLuca and Zouhar 2000; Jensen et al. 2001; Wilson et al. 2002; Nardoto and Bustamante 2003). This temporary rise may be due to the release of labile C and N compounds during fire that are quickly utilized by a rapidly growing microbial population (DeLuca and Zouhar 2000). It may also reflect differences in the microbial population structure that determine the rate of microbial recovery between ecosystems; fire generally causes a temporary decline in bacterial biomass, while most fungi, except for heat resistant species such as *Aspergillus*, *Gelasinospora* and *Cylindrocarpon* species, repopulate slowly (Widden and Parkinson 1975; Raison 1979; Dunn et al. 1979). A reduced fungal population, which may comprise a large proportion of the microbial biomass in coniferous forests, persisted for more than 10 years in burned Scots pine (*Pinus sylvestris*) stands in Finland (Pietikäinen and Fritze 1993; Fritze et al. 1993). In contrast, in a burned sub-tropical soil in India, the total number of fungi recovered within 30 days after fire, possibly due to high post-fire moisture content (Deka and Mishra 1983). This information suggests that higher post-fire MB-C and MB-N may be due to either a reduced proportion of fungi within the microbial population or rapid microbial recovery after fire.

MB-C and MB-N may remain lower in burned soils for less than 4 to more than 11 years after fire (Pietikäinen and Fritze 1993; Fritze et al. 1993; Dumontet et al. 1996; Walley et al. 1996; Prieto-Fernández et al. 1998). The persistence of low MB-C and MB-N after fire may be due to lower post-fire moisture content or altered soil organic matter structure that inhibits microbial proliferation (Pietikäinen and Fritze 1995); further research is required to relate soil organic matter structure to microbial population dynamics.

No consistent change in the microbial C/N ratio has been demonstrated after fire. Pietikäinen and Fritze (1993) noted an increase in the microbial C/N ratio in the first month after burning in Scots pine and Norway spruce (*Picea abies*) stands in Finland while Dumontet et al. (1996) observed a decline in the first month after burning in Aleppo pine (*Pinus halepensis*) stands in Italy. In eucalyptus (*Eucalyptus pauciflora*) stands in Australia, Hossain et al. (1995) showed that the microbial C/N ratio in the upper mineral soil was lower in a frequently burned (2- to 3-year frequency) compared with a stand burned once 20 years earlier, and that the microbial C/N ratio in both was lower than that of a regularly burnt (7 year frequency) stand. Further research is required to investigate the utility of microbial C/N ratio as an indicator of the microbial population structure.

2.3.4. Foliar and Total Plant N Content

Changes in the size of plant available inorganic N pools after fire may be reflected in the foliar N concentration or total N content of understory plants. Three to 4 months after a low severity fire, increased foliar N concentration was found in Aleppo pine needles and the leaves of a dominant understory shrub species (Gillon et al. 1999). This rise was attributed to the deposition of N-rich, scorched needles on the forest floor after fire that provided the input for increased N availability in the soil. In another study, the total N content of 4 common understory species 3 years after fire was found to be higher following severe burning but not after light or moderate burning (Stark and Steele 1977). Lloyd (1971) found that foliar N concentration was significantly greater 18 and 26 months after fire in some species but total N content was lower in the aboveground biomass of the burned stand due to biomass combustion during fire. This suggests that the foliar N concentration may initially be higher in the regenerating species due to high post-fire N availability but reduction in total aboveground biomass will result in an overall decline in total plant N content after fire.

2.3.5. Abiotic Influences on the Post-fire N Cycle

2.3.5.1. Soil Acidity

As surface vegetation and the forest floor are oxidized, base cations (e.g. K^+, Ca^{2+}, Mg^{2+}) are released from organic matter, compounds including litter and soil organic acids are combusted, and base-rich ash is deposited on the soil surface (Raison 1979). The decline in soil organic acids and neutralization of H^+ in soil solution by base cations results in an increase in pH that depends on the quantity of organic acid and anion consumption, original soil pH, and soil buffering capacity (Raison 1979; Fisher and

Binkley 2000). Generally, the forest floor pH is most affected by fire; the increase in post-fire pH depends on fire severity. A light burn may raise the forest floor pH by 0.2 to 0.8 units; this rise may persist for more than one year (Dyrness et al. 1989; Brais et al. 2000; Jensen et al. 2001). A severe burn may increase forest floor pH 1 to 5 units depending on the degree of forest floor and biomass consumption and may remain higher in burned than in unburned stands for up to 14 years after fire if the ash is retained (MacAdam 1987; Dyrness et al. 1989; Giovannini et al. 1990; Lynham et al. 1998; Brais et al. 2000; Simard et al. 2001). The effect of fire on upper mineral soil (0-10 cm) pH also depends on fire severity; light burning has little effect on mineral soil pH but severe burns may increase mineral soil pH from 0.5 to 2 units for a short time period (MacAdam 1987; Hernández et al. 1997; Mabuhay et al. 2003). This information suggests that the increase in pH after fire is related to fire severity; more severe fire results in a greater increase in pH in both the forest floor and upper mineral soil.

2.3.5.2. Substrate Quality

The change in post-fire C/N ratio of the forest floor and upper mineral soil depends on the balance between C and N losses and additions; both increases and decreases in the C/N ratio after fire have been reported. Deposition of incompletely combusted, N-rich needles combined with organic C loss reduced the C/N ratio in Aleppo pine forest (Gillon et al. 1999); a similar reduction was reported in severely burned white and black spruce stands in Alaska (Dyrness et al. 1989). In the black spruce stand, total soil N increased while organic C declined; in the white spruce stand the change in C/N ratio was attributed to greater C as compared with N oxidation from combusted forest floor (Dyrness et al. 1989). In contrast, an increase in the C/N ratio was observed one month after fire in the forest floor of ponderosa pine-Douglas fir and lodgepole pine-Engelmann spruce stands in Washington state, USA (Baird et al. 1999) and one year after fire in the surface 5 cm of mineral soil in Aleppo pine stands in Italy (Dumontet et al. 1996). Clearly, the change in substrate quality after fire, as indicated by the C/N ratio, may depend on the change in the total soil N content.

2.3.5.3. Soil Temperature and Soil Moisture Content

An increase in soil temperature after fire has been repeatedly documented and is attributed to the combustion of overstory and understory vegetation and the deposition of incompletely combusted black organic matter that promotes greater radiation absorption at the soil surface (Ahlgren and Ahlgren 1960; Raison 1979; Neary et al. 1999; Fisher and Binkley 2000). The soil temperature increase is generally greatest in the surface soil but may also occur at depth (Raison 1979); a light burn in a cleared mixedwood stand in Alberta increased soil temperature by 0.5°C at 20 cm below the mineral soil surface between June and August (Frey et al. 2003). Iverson and Hutchinson (2002) demonstrated that the increase in soil temperature after burning was higher and tended to last longer on xeric, south-facing sites compared with mesic, north-facing sites in mixed oak forest in Ohio, USA. Wilson et al. (2002) also demonstrated a greater increase in soil temperature on frequently burned xeric longleaf pine-wiregrass sites in Georgia, USA compared with intermediate and wet-mesic sites. This information suggests that the magnitude of the increase in soil temperature after fire is negatively related to the post-fire soil moisture content, which also tends to rise initially after fire.

In the short term, post-fire soil moisture content also tends to increase due to reduced transpiration and interception by overstory vegetation, while in the mid term moisture content may decline due to reduced water retention ability and greater evaporation (Ahlgren and Ahlgren 1960; Minshall et al. 1997; Fisher and Binkley 2000; Iverson and Hutchinson 2002; Mabuhay et al. 2003).

The relationship between the rise in soil temperature and soil moisture content and net rates of N mineralization and nitrification remains unclear. Wilson et al. (2002) demonstrated higher net N mineralization and nitrification rates on a frequently burned xeric site and attributed this trend to a greater increase in soil temperature after fire. Few studies have related the post-fire moisture content to N availability, although soil microbial biomass has been shown to respond favourably to water addition in a burned soil (Pietikäinen and Fritze 1995). Almendros et al. (1990) demonstrated that high microbial respiration after fire was related to the increase in easily decomposable C and N compounds immediately after fire; once these compounds were exhausted, microbial activity declined. Therefore, further research is required to establish a causal link between changes in soil temperature and moisture content after fire and N availability.

2.3.6. Hypotheses on Post-fire N Dynamics in a Riparian Mixedwood Stand

Research suggests that in the WBF, high moisture content in the riparian zone of an ephemeral or intermittent riparian valley compared to a drier upper slope position may influence fire severity and post-fire microbial recovery, soil pH, and soil temperature and subsequently, changes in the internal N cycle in the short-term (< 1 year) after fire. Table 2.2 summarizes the expected changes after fire in the forest floor and upper mineral soil (0-10 cm) in a lower slope, riparian position and an upper slope position of a mixedwood stand in the WBF based on the review and discussion presented in this section.

Table 2.2. Hypothesized differences in net N mineralization rate, net nitrification rate, gross N mineralization rate, gross NH_4^+ consumption rate, gross nitrification rate, gross NO_3^- consumption rate, microbial biomass C and N, soluble organic N, soil pH, soil temperature, and soil moisture content in the forest floor (LFH) and upper mineral soil (0-10 cm) of a lower slope, riparian compared with an upper slope position of a mixedwood stand in the Western Boreal Forest in the short-term after fire.

Component of the Internal N cycle	**Riparian Lower Slope**	**Upper Slope**
Net N mineralization rate	Initial H+ then L	Initial H then L+
Net nitrification rate	H	H
Total Soil N	L	L+
Gross N mineralization rate	Initial H+ then L	Initial H then L
Gross NH_4^+ consumption rate	Initial L then H	Initial L then H+
Gross nitrification rate	H	H
Gross NO_3^- consumption rate	H	H
Microbial biomass C	L	L+
Microbial biomass N	L	L+
Microbial biomass C/N ratio	L	L
Soluble organic N	Initial H then L	Initial H+ then L+
Total Plant N Content	L	L
Foliar N Content	H	H
Soil pH	H	H+
Soil C/N Ratio	H	H
Soil temperature	H	H+
Soil moisture content	H	H

Note: H indicates a higher rate/pool; L indicates a lower rate/pool; and + indicates when a greater change is expected in one of the positions.

2.4. References

Adams, M.A., Polglase, P.J., Attiwill, P.M. and Weston, C.J. 1989. In situ studies of nitrogen mineralization and uptake in forest soils: some comments on methodology. Soil Biol. Biochem. **21**: 423-429.

Ahlgren, I.F. and Alhgren, C.E. 1960. Ecological effects of forest fires. Bot. Rev. **26**: 483-533.

Alberta Sustainable Resource Development. 2001. Chisholm fire reports [online]. Available from http://www3.gov.ab.ca/srd/forests/chisholm [accessed January 20 2003].

Almendros, G., Gonzalez-vila, F.J. and Martin, F. 1990. Fire induced transformation of soil organic matter from an oak forest: an experimental approach to the effects of fire on humic substances. Soil Sci. **149**: 158-168.

Andersson, M., Michelsen, A., Jensen, M. and Kjøller, A. 2004. Tropical savannah woodland: effects of experimental fire on soil microorganisms and soil emissions of carbon dioxide. Soil Biol. Biochem. **36**: 849-858.

Baird, M., Zabowski, D. and Everett, R.L. 1999. Wildfire effects on carbon and nitrogen in inland coniferous forests. Plant Soil. **209**: 233-243.

Barnes, B.V., Zak, D.R., Denton, S.R. and Spurr, S.H. 1998. Forest Ecology. 4th Edition. John Wiley & Sons, Inc., New York.

Barraclough, D. and Puri, G. 1995. The use of ^{15}N pool dilution and enrichment to separate the heterotrophic and autotrophic pathways of nitrification. Soil Biol. Biochem. **27**: 17-22.

Bauhus, J., Khanna, P.K. and Raison, R.J. 1993. The effect of fire on carbon and nitrogen mineralization and nitrification in an Australian forest soil. Soil Biol. Biochem. **31**: 621-639.

Bauhus, J., Paré, D. and Côté, L. 1998. Effects of tree species, stand age and soil type on soil microbial biomass and its activity in a southern boreal forest. Soil Biol. Biochem. **30**: 1077-1089.

Bhatti, J.S., Apps, M.J. and Jiang, H. 2002. Influence of nutrients, disturbances, and site conditions on carbon stocks along a boreal forest transect in central Canada. Plant Soil. **242**: 1-14.

Binkley, D. 1981. Nodule biomass and acetylene reduction rates of red alder and Sitka alder on Vancouver Island, B.C. Can. J. For. Res. **11**: 281-286.

Boerner, R.E.J., Morris, S.J., Sutherland, E.K. and Hutchinson, T.F. 2000. Spatial variability in soil nitrogen dynamics after prescribed burning in Ohio mixed-oak forests. Land. Ecol. **15**: 425-439.

Bohlen, P.J., Groffman, P.M., Driscoll, C.T., Fahey, T.J. and Siccama, T.G. 2001. Plant-soil-microbial interactions in a northern hardwood forest. Ecol. **82**: 965-978.

Brais, S., Paré, D. and Ouimet, R. 2000. Impacts of wild fire severity and salvage harvesting on the nutrient balance of jack pine and black spruce boreal stands. For. Ecol. Manage. **137**: 231-243.

Carmosini, N., Devito, K. and Prepas, E.E. 2002. Gross nitrogen transformations in harvested and mature aspen-conifer mixed forest soils from the Boreal Plain. Soil Biol. Biochem. **34**: 1949-1951.

Carmosini, N., Devito, K.J. and Prepas, E.E. 2003. Net nitrogen mineralization and nitrification in trembling aspen forest soils on the Boreal Plain. Can. J. For. Res. **33**: 2262-2268.

Chang, S.X., Preston, C.M. and Weetman, G.F. 1995. Soil microbial biomass and microbial and mineralizable N in a clear-cut chronosequence on northern Vancouver Island, British Columbia. Can. J. For. Res. **25**: 1595-1607.

Chapin, F.S. III, Moilanen, L. and Kielland, K. 1993. Preferential use of organic nitrogen for growth by a non-mycorrhizal arctic sedge. Nature. **361**: 150-153.

Choromanska, U. and DeLuca, T.H. 2001. Prescribed fire alters the impact of wildfire on soil biochemical properties in a Ponderosa Pine forest. Soil Sci. Soc. Am. J. **65**: 232-238.

Choromanska, U. and DeLuca, T.H. 2002. Microbial activity and nitrogen mineralization in forest mineral soils following heating: evaluation of post-fire effects. Soil Biol. Biochem. **34**: 263-271.

Cirmo, C. P. and McDonnell, J.J. 1997. Linking the hydrologic and biogeochemical controls of nitrogen transport in near-stream zones of temperate-forested catchments: a review. J. Hydrol. **199**: 88-120.

Côté, L., Brown, S., Pare, D., Fyles, J. and Bauhus, J. 2000. Dynamics of carbon and nitrogen mineralization in relation to stand type, stand age and soil texture in the boreal mixedwood. Soil Biol. Biochem. **32**: 1079-1090.

Davidson, E.A., Hart, S.C., Shanks, C.A. and Firestone, M.K. 1991. Measuring gross nitrogen mineralization, immobilization, and nitrification by ^{15}N isotopic pool dilution in intact soil cores. J. Soil Sci. **42**: 335-349.

Davidson, E.A., Hart, S.C. and Firestone, M.K. 1992. Internal cycling of nitrate in soils of a mature coniferous forest. Ecol. **73**: 1148-1156.

DeBano, L.F. and Conrad, C.E. 1978. The effect of fire on nutrients in a chaparral ecosystem. Ecol. **59**: 489-497.

DeBano, L.F., Eberlein, G.E. and Dunn, P.H. 1979. Effects of burning on chaparral soils: I. Soil nitrogen. Soil Sci. Soc. Am. J. **43**: 504-509.

Deka, H.K. and Mishra, R.R. 1983. The effect of slash burning on soil microflora. Plant Soil. **73**: 167-175.

DeLuca, T.H. and Zouhar, K.L. 2000. Effects of selection harvest and prescribed fire on the soil nitrogen status of ponderosa pine forests. For. Ecol. Manage. **138**: 263-271.

Devito, K.J., Westbrook, C.J. and Schiff, S.L. 1999. Nitrogen mineralization and nitrification in upland and peatland forest soils in two Canadian Shield catchments. Can. J. For. Res. **29**: 1793-1804.

Driscoll, K.G., Arocena, J.M. and Massicotte, H.B. 1999. Post-fire soil nitrogen content and vegetation composition in Sub-Boreal spruce forests of British Columbia's central interior, Canada. For. Ecol. Manage. **121**: 227-237.

Dumontet, S., Dinel, H., Scopa, A., Mazzatura, A. and Saracino, A. 1996. Post-fire soil microbial biomass and nutrient content of a pine forest from a dunal Mediterranean environment. Soil Biol. Biochem. **28**: 1467-1475.

Dunn, P.H., DeBano, L.F. and Eberlein, G.E. 1979. Effects of burning on chaparral soils: II. Soil microbes and nitrogen mineralization. Soil Sci. Soc. Am. J. **43**: 509-514.

Dunn, P.H., Barro, S.C. and Poth, M. 1985. Soil moisture affects survival of microorganisms in heated chaparral soil. Soil Biol. Biochem. **17**: 143-148.

Dyrness, C.T., Van Cleve, K. and Levison, J.D. 1989. The effect of wildfire on soil chemistry in four forest types in interior Alaska. Can. J. For. Res. **19**: 1389-1396.

Dwire, K.A. and Kauffman, J.B. 2003. Fire and riparian ecosystems in landscape of the western USA. For. Ecol. Manage. **128**: 61-74.

Ellert, B.H. and Bettany, J.R. 1992. Temperature dependence of net nitrogen and sulfur mineralization. Soil Sci. Soc. Am. J. **56**: 1133-1141.

Eno, C.F. 1960. Nitrate production in the field by incubating the soil in polyethylene bags. Soil Sci. Soc. Am. Proc. **24**: 277-299.

Evans, C.A., Miller, E.K. and Friedland, A.J. 1998. Nitrogen mineralization associated with birch and fir under different soil moisture regimes. Can. J. For. Res. **28**: 1890-1898.

Fenn, M.E., Poth, M.A., Dunn, P.H. and Barro, S.C. 1993. Microbial N and biomass, respiration and N mineralization in soils beneath two chapparal species along a fire-induced age gradient. Soil Biol. Biochem. **25**: 457-466.

Fisher, R.F. and Binkley, D. 2000. Ecology and Management of Forest Soils. John Wiley & Sons Inc., New York.

Frey, B.R., Lieffers, V.J., Munson, A.D. and Blenis, P.V. 2003. The influence of partial harvesting and forest floor disturbance on nutrient availability and understory vegetation in boreal mixedwoods. Can. J. For. Res. **33**: 1180-1188.

Fritze, H., Pennanen, T. and Pietikäinen, J. 1993. Recovery of soil microbial biomass and activity from prescribed burning. Can. J. For. Res. **23**: 1286-1290.

Fritze, H., Smolander, A., Levula, T., Kitunen, V. and Malkonen, E. 1994. Wood ash fertilization and fire treatments in a Scots-pine forest stand: effects on the organic layer, microbial biomass, and microbial activity. Biol. Fertil. Soils. **17**: 57-63.

Fyles, J.W. and McGill, W.B. 1987. Nitrogen mineralization in forest soil profiles from central Alberta. Can. J. For. Res. **17**: 242-249.

Fyles, J.W., Fyles, I.H., Beese, W.J. and Feller, M.C. 1991. Forest floor characteristics and soil nitrogen availability on slash-burned sites in coastal British Columbia. Can. J. For. Res. **21**: 1516-1522.

Giardina, C.P. and Rhoades, C.C. 2001. Clear cutting and burning affect nitrogen supply, phosphorus fractions and seedling growth in soils from a Wyoming lodgepole pine forest. For. Ecol. Manage. **140**: 19-28.

Giardina, C., Ryan, M.G., Hubbard, R.M. and Binkley, D. 2001. Tree species and soil textural controls on carbon and nitrogen mineralization rates. Soil Sci. Soc. Am. J. **65**: 1272-1279.

Gillon, D., Houssard, C., Valette, J.C. and Rigolot, E. 1999. Nitrogen and phosphorus cycling following prescribed burning in natural and managed Aleppo pine forests. Can. J. For. Res. **29**: 1237-1247.

Giovannini, G., Lucchesi, S. and Giachetti, M. 1990. Effects of heating on some chemical parameters related to soil fertility and plant growth. Soil Sci. **149**: 344-350.

Gonçalves, J.L.M. and Carlyle, J.C. 1994. Modelling the influence of moisture and temperature on net nitrogen mineralization in a forested sandy soil. Soil Biol. Biochem. **26**: 1557-1564.

Goodale, C.L. and Aber, J.D. 2001. The long-term effects of land use history on nitrogen cycling in northern hardwood forests. Ecol. App. **11**: 253-267.

Grier, C.C. 1975. Wildfire effects on nutrient distribution and leaching in a coniferous ecosystem. Can. J. For. Res. **5**: 599-607.

Grogan, P., Bruns, T.D. and Chapin, F.S. III. 2000. Fire effects on ecosystem nitrogen cycling in a Californian bishop pine forest. Oecologia. **122**: 537-544.

Hanks, R.J. 1992. Applied soil physics: soil water and temperature applications. 2nd edition. Springer-Verlag, New York.

Hannam, K.D. and Prescott, C.E. 2003. Soluble organic nitrogen in forests and adjacent clearcuts in British Columbia, Canada. Can. J. For. Res. **33**: 1709-1718.

Hart, S.C. and Firestone, M.K. 1989. Evaluation of three in-situ soil N availability assays. Can. J. For. Res. **19**: 185-191.

Hart, S.C., Stark, J.M., Davidson, E.A. and Firestone, M.K. 1994a. Nitrogen mineralization, immobilization, and nitrification. *In* Methods of Soil Analysis. Part 2: Microbiological and Biochemical properties. *Edited by* S.H. Mickelson. Soil Science Society of America, Madison, WI. pp. 985-1018.

Hart, S.C., Nason, G.E., Myrold, D.D. and Perry, D.A. 1994b. Dynamics of gross nitrogen transformations in an old-growth forest: the carbon connection. Ecol. **75**: 880-891.

Havlin, J.L., Beaton, J.D., Tisdale, S.L. and Nelson, W.L. 1999. Soil Fertility and Fertilizers: an Introduction to Nutrient Management. 6th edition. Prentice Hall, Upper Saddle River, NJ.

Haynes, R.J. 1986. Mineral Nitrogen in the Plant-Soil System. Academic Press, Inc., Orlando, FL.

Hernández, T., García, C. and Reinhardt, I. 1997. Short-term effect of wildfire on the chemical, biochemical and microbiological properties of Mediterranean pine forest soils. Biol. Fertil. Soils. **25**: 109-116.

Holmes, W.E. and Zak, D.R. 1994. Soil microbial biomass dynamics and net nitrogen mineralization in northern hardwood ecosystems. Soil Sci. Soc. Am. J. **58**: 238-243.

Hossain, A.K.M.A., Raison, R.J. and Khanna, P.K. 1995. Effects of fertilizer application and fire regime on soil microbial biomass carbon and nitrogen, and nitrogen mineralization in an Australian subalpine eucalypt forest. Biol. Fertil. Soils. **19**: 246-252.

Huang, W.Z. and Schoenau, J.J. 1996. Forms, amounts, and distribution of carbon, nitrogen, phosphorus, and sulfur in a boreal aspen forest soil. Can. J. Soil Sci. **76**: 373-385.

Huang, W.Z., and Schoenau, J.J. 1997. Seasonal and spatial variations in soil nitrogen and phosphorus supply rates in a boreal aspen forest. Can. J. Soil Sci. **77**: 597-612.

Huang, W.Z. and Schoenau, J.J. 1998. Fluxes of water soluble nitrogen and phosphorus in the forest floor and surface mineral soil of a boreal aspen stand. Geoderma. **81**: 251-264.

Iverson, L.R. and Hutchinson, T.F. 2002. Soil temperature and moisture fluctuations during and after prescribed fire in mixed-oak forests, USA. Nat. Areas J. **22**: 296-304.

Jensen, M., Michelsen, A. and Gashaw, M. 2001. Responses in plant, soil inorganic and microbial nutrient pools to experimental fire, ash, and biomass addition in a woodland savanna. Oecologia. **128**: 85-93.

Johnson, E.A. 1992. Fire and Vegetation Dynamics: Studies from the North American Boreal Forest. Cambridge University Press, Cambridge, UK.

Johnston, M. and Elliott, J. 1998. The effect of fire severity on ash, and plant and soil nutrient levels following experimental burning in a boreal mixedwood stand. Can. J. Soil Sci. **78**: 35-44.

Kalbitz, K., Solinger, S., Park, J.H., Michalzik, B. and Matzner, E. 2000. Controls on the dynamics of dissolved organic matter in soils: a review. Soil Sci. **165**: 277-304.

Kaye, J.P. and Hart, S.C. 1997. Competition for nitrogen between plants and soil microorganisms. TREE. **12**: 139-143.

Khanna, P.K. and Raison, R.J. 1986. Effect of fire intensity on solution chemistry of surface soil under a Eucalyptus pauciflora forest. Aust. J. Soil Res. **24**: 423-434.

Kielland, K. 1994. Amino acid absorption by arctic plants: implications for plant nutrition and nitrogen cycling. Ecol. **75**: 2373-2383.

Kimmins, J.P. 1996. Importance of soil and the role of ecosystem disturbance for sustained productivity of cool temperate and boreal forests. Soil Sci. Soc. Am. J. **60**: 1643-1654.

Klemmedson, J.O., Schultz, A.M., Jenny, H. and Biswell, H.H. 1962. Effect of prescribed burning of forest litter on total soil nitrogen. Soil Sci. Soc. Am. Proc. **26**: 200-202.

La Roi, G.H., Strong, W.L. and Pluth, D.J. 1988. Understory plant community classifications as predictors of forest site quality for lodgepole pine and white spruce in west central Alberta. Can. J. For. Res. **18**: 875-887.

Li, Q., Allen, H.L. and Wollum II, A.G. 2004. Microbial biomass and bacterial functional diversity in forest soils: effects of organic matter removal, compaction, and vegetation control. Soil Biol. Biochem. **36**: 571-579.

Little, T.I., Pluth, D.J., Corns, I.G.W. and Gilmore, D.W. 2002. Post-fire forest floor development along toposequences of white spruce-trembling aspen mixedwood communities in west-central Alberta. Can. J. For. Res. **32**: 892-902.

Lloyd, P.S. 1971. Effects of fire on the chemical status of herbaceous communities of the derbyshire dales. J. Ecol. **59**: 261-273.

Lynham, T.J., Wickware, G.M. and Mason, J.A. 1998. Soil chemical changes and plant succession following experimental burning in immature jack pine. Can. J. Soil Sci. **78**: 93-104.

Mabuhay, J.A., Nakagoshi, N. and Horikoshi, T. 2003. Microbial biomass and abundance after forest fire in pine forests in Japan. Ecol. Res. **18**: 431-444.

MacAdam, A.M. 1987. Effects of broadcast slash burning on fuels and soil chemical properties in the Sub-boreal Spruce Zone of central British Columbia. Can. J. For. Res. **17**: 1577-1584.

Macdonald, S.E., Schmidt, M.G. and Rothwell, R.L. 1998. Impacts of mechanical site preparation on foliar nutrients of planted white spruce seedlings on mixed-wood boreal forest sites in Alberta. For. Ecol. Manage. **110**: 35-48.

MacLean, D.A. and Wein, R.W. 1977. Changes in understory vegetation with increasing stand age in New Brunswick forests: species composition, cover, biomass, and nutrients. Can. J. Bot. **55**: 2818-2831.

MacLean, D.A. and Wein, R.W. 1978. Weight loss and nutrient changes in decomposing litter and forest floor material in New Brunswick forest stands. Can. J. Bot. **56**: 2730-2749.

MacLean, D.A., Woodley, S.J., Weber, M.G. and Wein, R.W. 1983. Fire and nutrient cycling. *In* The Role of Fire in Northern Circumpolar Ecosystems. *Edited by* R.W. Wein and D.A. MacLean. John Wiley & Sons Ltd., New York. pp. 111-132.

McClaugherty, C.A., Pastor, J. and Aber, J.D. 1985. Forest litter decomposition in relation to soil nitrogen dynamics and litter quality. Ecol. **66**: 266-275.

Minshall, G.W., Robinson, C.T. and Lawrence, D.E. 1997. Postfire responses of lotic ecosystems in Yellowstone National Park, U.S.A. Can. J. Fish. Aquat. Sci. **54**: 2509-2525.

Miyanishi, K. 2001. Duff consumption. *In* Forest Fire Behaviour and Ecological Effects. *Edited by* E.A. Johnson and K. Miyanishi. Academic Press, San Diego. pp. 437-475.

Miyanishi, K. and Johnson, E.A. 2002. Processes and patterns of duff consumption in the mixedwood boreal forest. Can. J. For. Res. **32**: 1285-1295.

Monleon , V.J., Cromack, K. Jr. and Landsberg, J.D. 1997. Short- and long-term effects of prescribed underburning on nitrogen availability in ponderosa pine stands in central Oregon. Can. J. For. Res. **27**: 369-378.

Myrold, D.D. 1987. Relationship between microbial biomass nitrogen and a nitrogen availability index. Soil Sci. Soc. Am. J. **51**: 1047-1049.

Naiman, R.J. and Decamps, H. 1997. The ecology of interfaces: riparian zones. Annu. Rev. Ecol. Syst. **28**: 621-658.

Nardoto, G.B. and Bustamante, M.M. da C. 2003. Effects of fire on soil nitrogen dynamics and microbial biomass in savannas of central Brazil. Pesq. Agropec. Bras. **38**: 955-962.

Neary, D.G., Klopatek, C.C., DeBano, L.F. and Ffolliott, P.F. 1999. Fire effects on belowground sustainability: a review and synthesis. For. Ecol. Manage. **122**: 51-71.

Nelson, R.M. Jr. 2001. Water relations of forest fuels. *In* Forest Fire Behaviour and Ecological Effects. *Edited by* E.A. Johnson and K. Miyanishi. Academic Press, San Diego. 79-145.

Offord, J.L. 1999. Nitrogen Mineralization Patterns of Boreal Forest Soils. M.Sc. Thesis. Department of Renewable Resources, University of Alberta, Edmonton, AB.

Ohrui, K., Mitchell, M.J. and Bischoff, J.M. 1999. Effect of landscape position on N mineralization and nitrification in a forested watershed in the Adirondack Mountains of New York. Can. J. For. Res. **29**: 497-508.

Ohte, N., Tokuchi, N. and Suzuki, M. 1997. An in situ lysimeter experiment on soil moisture influence on inorganic nitrogen discharge from forest soil. J. Hydrol. **195**: 78-98.

Ollinger, S.V., Smith, M.L., Martin, M.E., Hallett, R.A., Goodale, C.L. and Aber, J.D. 2002. Regional variation in foliar chemistry and N cycling among forests of diverse history and composition. Ecol. **83**: 339-355.

Palm, C.A. and Sanchez, P.A. 1991. Nitrogen release from the leaves of some tropical legumes as affected by their lignin and polyphenolic contents. Soil Biol. Biochem. **23**: 83-88.

Paré, D. and Bergeron, Y. 1996. Effect of colonizing tree species on soil nutrient availability in a clay soil of the boreal mixedwood. Can. J. For. Res. **26**: 1022-1031.

Pietikäinen, J. and Fritze, H. 1993. Microbial biomass and activity in the humus layer following burning: short-term effects of two different fires. Can. J. For. Res. **23**: 1275-1285.

Pietikäinen, J. and Fritze, H. 1995. Clear-cutting and prescribed burning in coniferous forest: comparison of effects of soil fungal and total microbial biomass, respiration activity and nitrification. Soil Biol. Biochem. **27**: 101-109.

Prescott, C.E., Maynard, D.G. and Laiho, R. 2000a. Humus in northern forests: friend or foe? For. Ecol. Manage. **133**: 23-36.

Prescott, C.E., Vesterdal, L., Pratt, J., Venner, K.J., de Montigny, L.M. and Trofymow, J.A. 2000b. Nutrient concentrations and nitrogen mineralization in forest floors of single species conifer plantations in coastal British Columbia. Can. J. For. Res. **30**: 1341-1352.

Prieto-Fernández, A.,Villar, M.C., Carballas, M. and Carballas, T. 1993. Short-term effects of a wildfire on the nitrogen status and its mineralization kinetics in an Atlantic forest soil. Soil Biol. Biochem. **25**: 1657-1664.

Prieto-Fernández, A., Acea, M.J. and Carballas, T. 1998. Soil microbial and extractable C and N after wildfire. Biol. Fertil. Soils. **27**: 132-142.

Puri, G. and Ashman, M.R. 1998. Relationship between soil microbial biomass and gross N mineralisation. Soil Biol. Biochem. **30**: 251-256.

Qualls, R.G., Haines, B.L. and Swank, W.T. 1991. Fluxes of dissolved organic nutrients and humic substances in a deciduous forest. Ecol. **72**: 254-266.

Qualls, R.G. 2000. Comparison of the behaviour of soluble organic and inorganic nutrients in forest soils. For. Ecol. Manage. **138**: 29-50.

Raab, T.K., Lipson, D.A. and Monson, R.K. 1996. Non-mycorrhizal uptake of amino acids by roots of the alpine sedge *Kobresia myosuroides*: implications for the alpine nitrogen cycle. Oecologia. **108**: 488-494.

Raison, R.J. 1979. Modification of the soil environment by vegetation fires, with particular reference to nitrogen transformations: a review. Plant Soil. **51**: 73-108.

Raison, R.J., Khanna, P.K. and Woods, P.V. 1985. Mechanisms of element transfer to the atmosphere during vegetation fires. Can. J. For. Res. **15**: 132-140.

Reich, P.B., Grigal, D.F., Aber, J.D. and Gower, S.T. 1997. Nitrogen mineralization and productivity in 50 hardwood and conifer stands on diverse soils. Ecol. **78**: 335-347.

Romanyà, J., Casals, P. and Vallejo, V.R. 2001. Short-term effects of fire on soil nitrogen availability in Mediterranean grasslands and shrublands growing in old fields. For. Ecol. Manage. **147**: 39-53.

Samran, S., Woodard, P.M. and Rothwell, R.L. 1995. The effect of soil water on ground fuel availability. For. Sci. **41**: 255-267.

Ste-Marie, C. and Paré, D. 1999. Soil, pH, and N availability effects on net nitrification in the forest floors of a range of boreal forest stands. Soil Biol. Biochem. **31**: 1579-1589.

Schade, J.D., Marti, E., Welter, J.R., Fisher, S.G. and Grimm, N.B. 2002. Sources of nitrogen to the riparian zone of a desert stream: implications for riparian vegetation and nitrogen retention. Ecosystems. **5**: 68-79.

Shaw, R.D., Trimbee, A.M., Minty, A., Fricker, H. and Prepas, E.E. 1989. Atmospheric deposition of phosphorus and nitrogen in central Alberta with emphasis on Narrow Lake. Water Air Soil Pollut. **43**: 119-134.

Simard, D.G., Fyles, J.W., Paré, D. and Nguyen, T. 2001. Impacts of clearcut harvesting and wildfire on soil nutrient status in the Quebec boreal forest. Can. J. Soil Sci. **81**: 229-237.

Smolander, A. and Kitunen, V. 2002. Soil microbial activities and characteristics of dissolved organic C and N in relation to tree species. Soil Biol. Biochem. **34**: 651-660.

Stanford, G. and Smith, S.J. 1972. Nitrogen mineralization potentials of soils. Soil Sci. Soc. Am. Proc. **36**: 465-472.

Stanford, G., Frere, M.H. and Schwaninger, D.H. 1973. Temperature coefficient of soil nitrogen mineralization. Soil Sci. **115**: 321-323.

Stanford, G. and Epstein, E. 1974. Nitrogen mineralization-water relations in soils. Soil Sci. Soc. Am. Proc. **38**: 103-107.

Stark, N. and Steele, R. 1977. Nutrient content of forest shrubs following burning. Am. J. Bot. **64**: 1218-1224.

Stark, J.M. and Hart, S.C. 1997. High rates of nitrification and nitrate turnover in undisturbed coniferous forests. Nature. **385**: 61-64.

Stottlemyer, R. and Toczydlowski, D. 1999a. Seasonal relationships between precipitation, forest floor, and streamwater nitrogen, Isle Royale, Michigan. Soil Sci. Soc. Am. J. **63**: 389-398.

Stottlemyer, R. and Toczydlowski, D. 1999b. Nitrogen mineralization in a mature boreal forest, Isle Royale, Michigan. J. Env. Qual. **28**: 709-720.

Strong, W.L., Pluth, D.J., La Roi, G.H. and Corns, I.G.W. 1991. Forest understory plants as predictors of lodgepole pine and white spruce site quality in west central Alberta. Can. J. For. Res. **21**: 1675-1683.

Tate, R.L. III. 1995. Soil Microbiology. John Wiley & Sons, Inc., New York.

Vance, E.D. and Chapin, F.S. III. 2001. Substrate limitations to microbial activity in taiga forest floors. Soil Biol. Biochem. **33**: 173-188.

van den Driessche, R. 1974. Prediction of mineral nutrient status of trees by foliar analysis. Bot. Rev. **40**: 347-394.

Van Wagner, C.E. 1983. Fire behaviour in northern conifer forests and shrublands. *In* The Role of Fire in Northern Circumpolar Ecosystems. *Edited by* R.W. Wein and D.A. MacLean. John Wiley & Sons Ltd., New York. pp. 65-80.

Venterea, R.T., Lovett, G.M., Groffman, P.M. and Schwarz, P.A. 2003. Landscape patterns of net nitrification in a northern hardwood-conifer forest. Soil Sci. Soc. Am. J. **67**: 527-539.

Verburg, P.S.J., Van Dam, D., Hetting, M.M. and Tietema, A. 1999. Microbial transformations of C and N in a boreal forest floor as affected by temperature. Plant Soil. **208**: 187-197.

Verchot, L.V., Holmes, Z., Mulon, L., Groffman, P.M. and Lovett, G.M. 2001. Gross vs net rates of N mineralization and nitrification as indicators of functional differences between forest types. Soil Biol. Biochem. **33**: 1889-1901.

Vitousek, P.M., Gosz, J.R., Grier, C.C., Melillo, J.M. and Reiners, W.A. 1982. A comparative analysis of potential nitrification and nitrate mobility in forest ecosystems. Ecol. Mon. **52**: 155-177.

Walley, F.L., Van Kessel, C. and Pennock, D.J. 1996. Landscape scale variability in N mineralization in forest soils. Soil Biol. Biochem. **28**: 383-391.

Wang, W., Smith, C.J., Chalk, P.M. and Chen, D. 2001. Evaluating chemical and physical indices of nitrogen mineralization capacity with an unequivocal reference. Soil Sci. Soc. Am. J. **65**: 365-368.

Widden, P. and Parkinson, D. 1975. The effects of a forest fire on soil microfungi. Soil Biol. Biochem. **7**: 125-138.

Wilson, C.A., Mitchell, R.J., Boring, L.R. and Hendricks, J.J. 2002. Soil nitrogen dynamics in a fire-maintained forest ecosystem: results over a 3 year burn interval. Soil Biol. Biochem. **34**: 679-689.

Wüthrich, C., Schaub, D., Weber, M., Marxer, P. and Conedera, M. 2002. Soil respiration and soil microbial biomass after fire in a sweet chestnut forest in southern Switzerland. Catena. **48**: 201-215.

Zackrisson, O., Nilsson, M.C. and Wardle, D.A. Key ecological function of charcoal from wildfire in the Boreal forest. Oikos **77**: 10-19.

Zaman, M. and Chang, S.X. 2004. Substrate type, temperature, and moisture content affect gross and net N mineralization and nitrification rates in agroforestry systems. Biol. Fertil. Soils. **39**: 269-279.

3. THE EFFECT OF EXPERIMENTAL BURNING AND SLOPE POSITION ON SOIL PROPERTIES IN AN EPHEMERAL RIPARIAN VALLEY IN ALBERTA

3.1. Introduction

In the western Boreal Plain subregion of Canada's boreal forest, mixedwood stands of deciduous and coniferous species occur on a thick layer of glacial drift over bedrock and receive low amounts of precipitation in comparison to potential evapotranspiration (Devito et al. 2004). These unique climatic and geologic characteristics differentiate the Western Boreal Forest (WBF) from other North American forest ecosystems. In the WBF and in mixedwood stands in particular, fire is an important mechanism of natural disturbance (Johnson 1992). Predictions of increasing fire frequency with climate change (Bhatti et al. 2002) and adoption of the natural disturbance paradigm to guide forest management practices (Macdonald et al. 2004) have renewed interest in fire research. It is well known that fire may induce dramatic changes in soil physical, chemical, and microbiological properties in the short-term (< 1 year) after fire (reviewed in Ahlgren and Ahlgren 1960; Raison 1979; MacLean et al. 1983; Neary et al. 1999; Fisher and Binkley 2000; Chanasyk et al. 2003). However, in the WBF, few have studied short-term post-fire soil properties after a low severity understory fire, even though these fires are an important component of the fire regime.

Oxidation of the surface organic layer during an understory fire results in carbon (C) (as CO_2 and CO) and nitrogen (N) (as N_2, NO, N_2O and NO_2) release, ash deposition, and heat energy production (Raison 1979; Neary et al. 1999). These processes may affect several soil properties including temperature, moisture content, pH, and total, extractable and microbial biomass C and N contents; the change in soil properties that occurs after fire greatly depends on fire severity, a measure of the duration and intensity of heating. Fire severity determines the degree of forest floor combustion, loss in forest floor mass, and penetration of heat energy into the surface organic layer and upper mineral soil (Raison 1979; Raison et al. 1985; Neary et al. 1999). Within-stand variation in forest floor and soil moisture content, forest floor bulk density, and forest floor depth may influence fire severity patterns (Cumming 2001; Miyanishi and Johnson 2002). This suggests that an increase in soil moisture content from the upper slope to the lower slope position along a hillslope may reduce fire severity in the lower slope position (Dwire and Kauffman 2003; Miyanishi and Johnson 2002), resulting in less dramatic, short-term post-fire changes to soil properties in this slope position.

In streams, the riparian zone can be defined as the adjacent area including the floodplain and part of the terrace that is influenced by either elevated water tables or frequent flooding (Naiman and Decamps 1997). In the WBF, few have examined the soil properties of forested riparian zones along ephemeral or intermittent streams that account for a substantial proportion of the surface drainage network, may occur in moderately sloped valleys and may be wetter than drier slope positions within the same stand. Study of the soil properties in these riparian zones is important in order to understand and explain their nutrient (including N) dynamics and potential to contribute soluble nutrients in runoff into the surface drainage network. In addition, little research has been conducted with regards to fire effects on soil properties in these ephemeral riparian valleys even though the occurrence of fire in such riparian locations is not uncommon (Alberta Sustainable Resource Development 2001; Andison and McCleary 2002; Macdonald et al. 2004). It is possible that high moisture content in the riparian, lower slope position of such a valley may reduce fire severity in comparison to drier parts of the stand and thereby affect the post-fire changes that occur in soil properties. This suggests that the soil in the riparian zone may naturally be less disturbed by fire than the surrounding upland, and has implications for forest management based on the natural disturbance paradigm in the WBF.

Generally, the post-fire increase in temperature, moisture content, and pH is positively correlated to the loss in forest floor mass (Ahlgren and Ahlgren 1960; Raison 1979; Fisher and Binkley 2000), as is the post-fire reduction in forest floor total C and N contents (Dyrness et al. 1989; Pietikäinen and Fritze 1993; Johnston and Elliott 1998; Brais et al. 2000). However, the relationship between fire severity and extractable C, which generally declines after fire, and extractable N, which generally increases after fire, remains unclear (Almendros et al. 1990; Pietikäinen and Fritze 1993; Hernández et al. 1997; Prieto-Fernández et al. 1998). Many studies have reported a post-fire decline in microbial biomass C and N contents due to the effect of soil heating (Pietikäinen and Fritze 1993; Fritze et al. 1994; Hossain et al. 1995; Dumontet et al. 1996; Hernández et al. 1997; Prieto-Fernández et al. 1998; Choromanska and DeLuca 2001; Choromanska and DeLuca 2002; Wüthrich et al. 2002), however, few have examined whether this trend varies with differences in fire severity due to within-stand variation in moisture content.

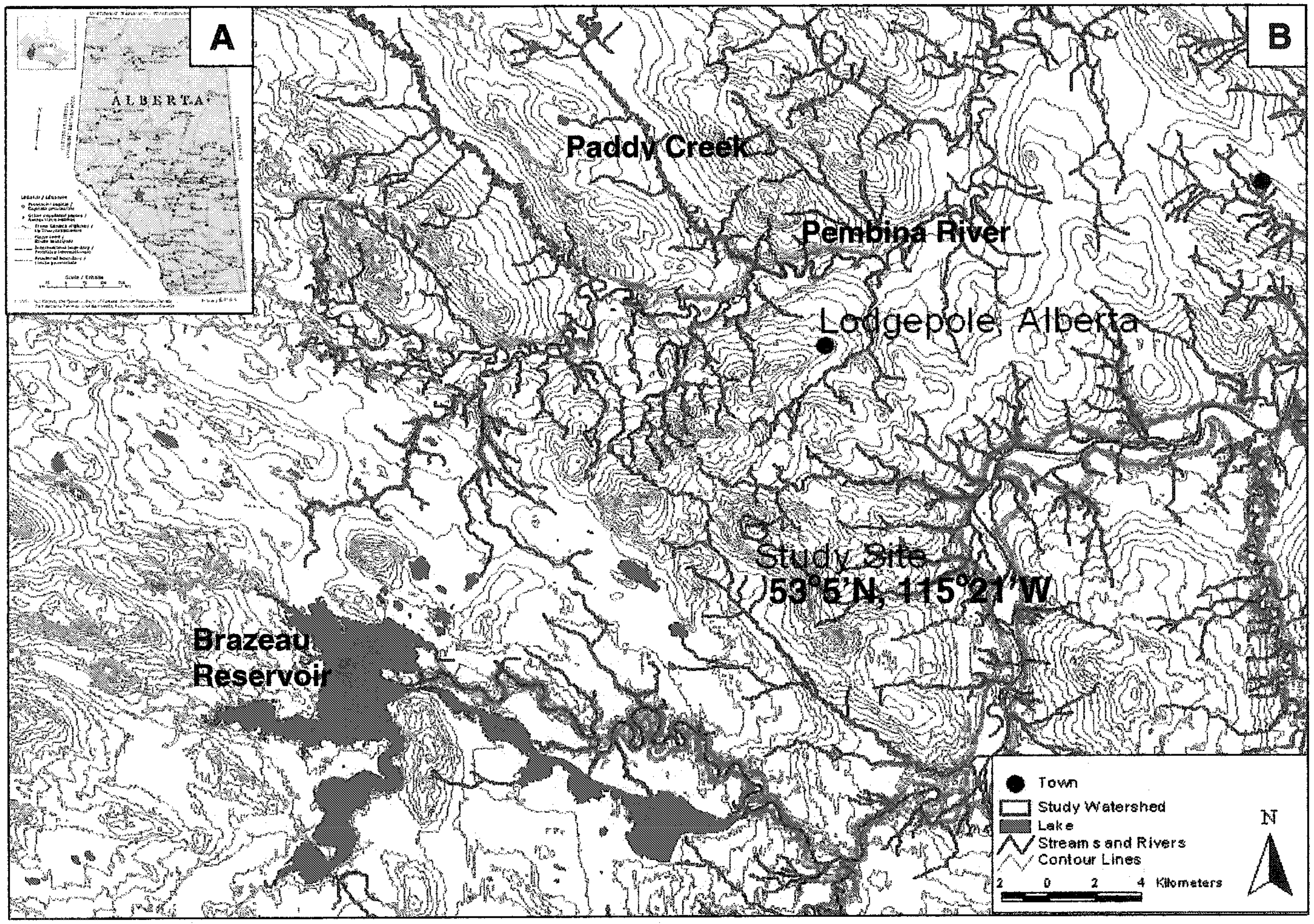
The objectives of this study were: (1) to examine the difference in soil properties between the upper slope and lower slope, riparian positions of an ephemeral valley, and (2) to examine the short-term changes in soil properties after a simulated understory fire in the upper slope and lower slope, riparian positions of the same valley. The study was conducted in an ephemeral riparian valley of a trembling aspen (*Populus tremuloides* Michx.) dominated mixedwood stand in the WBF. Soil properties measured in the forest floor and surface 10 cm of mineral soil included moisture content, pH, total, extractable and microbial biomass C and N contents, total soil C/N ratio, microbial C/N ratio, microbial biomass C/total soil C ratio, microbial biomass N/total soil N ratio, and soil temperature (mineral soil only).

3.2. Materials and Methods

3.2.1. Study Site

The study was conducted from June 2003 to October 2003 in a 90-year-old trembling aspen dominated mixedwood stand (53°5'N, 115°21'W; 914 m a.s.l.) that is approximately 5 km southwest of Lodgepole, Alberta (Figure 3.1). This site is located within the Lower Foothills Natural Subregion (Beckingham et al. 1996) of the Boreal Plain Ecozone (Ecological Stratification Working Group 1996) and the Cynthia Upland Land District of the Western Alberta Plain Physiographic Region (Peters 1981). Relief in the study site varies from 10 to 50 metres and the area has 1.5 to 20 metres of clay loam till underlain by undulating sandstone, siltstone, and mudstone of the Paskapoo (Paleocene) formation (Alberta Energy and Natural Resources 1975; Peters 1981).

Figure 3.1. Map of the study site location (53°5'N, 115°21'W) within Alberta (A) and within the local drainage network (B).



Note: The digital data used to create map B was provided by Weyerhaeuser Company Ltd.

Regional climate of the study area is continental with short, cool summers and cold winters. There is little long-term climate data available for this area; Rocky Mountain House (52°25'N, 114°55'W, 988 m a.s.l.), Alberta, is the nearest full-time weather station to the study site. From 1971 to 2000 annual precipitation averaged 535 mm (123 mm or 23% as snow) and mean annual temperature averaged 2.3°C at Rocky Mountain House (Environment Canada 2004a). In January, temperature averaged -11°C while in July average temperature was 15°C; 49% of annual precipitation occurred as rain from June to August (Environment Canada 2004a). In Drayton Valley (53°13' N, 114°57'W, 883 m a.s.l.), Alberta, a weather station closer to the study site that has been collecting climate data since 2000, average air temperature was 15°C and total rainfall was 263.5 mm from June to October 2003 during this study (Environment Canada 2004b); daily air temperature and precipitation data during the study period are shown in Figures 3.5 and 3.6. In this region, average annual potential evapotranspiration is 508 mm (The National Atlas of Canada 1974). No runoff data exists for the study site. However, mean annual runoff at Pembina River below Paddy Creek (53°7'47''N 115°19'30''W) from 1956 to 2003 approximated 166 mm yr^{-1} (Environment Canada 2004c).

The study site is located within a 97 hectare watershed. An ephemeral stream with a vegetated channel less than 1 metre wide and poorly defined banks dissects the mixedwood stand to the north. An ephemeral draw with poorly defined banks and a vegetated channel less than 1 metre wide occurs in a valley that dissects the stand to the west.

The Hubalta soil series characterizes the area; soils are moderately well drained Orthic Gray Luvisols on the upper and mid slopes and imperfect to very poorly drained Gleyed Gray Luvisols in the depressions (Peters 1981). Soils at the study site are Orthic Gray Luvisols on the upper slopes and range from Gleyed Black Chernozems and Gray Luvisols to Orthic Gleysols in the lower slope riparian areas (soil classification according to Soil Classification Working Group (1998)). No total N deposition data exists for the study site. The closest N deposition data available is from Narrow Lake (54°35'N, 113°37'W) within the Boreal Plain Ecozone; average deposition was 4.24 kg N ha^{-1} yr^{-1} from 1983 to 1986 (Shaw et al. 1989).

The study site includes two distinct plant communities: one characteristic of the mid and upper slopes, and another characteristic of the lower slope, riparian area. Trembling aspen, balsam fir (*Abies balsamea*), and white spruce (*Picea glauca*) are the dominant tree species on the upper slope position; the understory is dominated by bracted honeysuckle (*Lonicera involucrata*), wild red raspberry (*Rubus idaeus*), wild sarsasparilla (*Aralia nudicaulis*), bunchberry (*Cornus canadensis*), tall lungwort (*Mertensia paniculata*), and arrow-leaved coltsfoot (*Petasites sagittatus*). The plant community in the upper slope position can be categorized as Lower Foothills Ecophase f3, bracted honeysuckle Aw-Sw-Pl (Beckingham et al. 1996). In the lower slope riparian position white spruce, balsam fir, trembling aspen, and white birch are the main tree species; bracted honeysuckle, river alder (*Alnus rugosa*), cow parsnip (*Heracleum lanatum*), horsetail (*Equisetum* spp.), and oak fern (*Gymnocarpium dryopteris*) predominate as the understory species. The plant community in the riparian, lower-slope position corresponds to Lower Foothills Ecophase i2, horsetail Pb-Sw (Beckingham et al. 1996). All plant species names are according to Johnson et al. (1995).

Specific characterization of the soils and stratigraphy at the study site was completed. Soil classification was determined from a representative soil pit (1 m deep) established in each landscape position in each block and described according to the Canadian System of Soil Classification (Soil Classification Working Group 1998); the soil profile descriptions are shown in Table 3.1. Each horizon was sampled and analyzed for total and extractable C and N, NH_4^+ and NO_3^- concentrations (Table 3.2).

Particle size analysis was performed on the surface mineral soil (0-10 cm) of each plot following the hydrometer method of Gee and Or (2002) with readings at 0.5, 1, 90 and 1440 minutes. Soil texture in the upland and riparian plots ranged from silt loam to loam. In the upland plots, the mean (SE) percent of clay, silt, and sand were 23.17 (1.46), 47.80 (3.41), and 29.03 (3.09), respectively. In the riparian plots, the mean (SE) percent of clay, silt, and sand were 18.86 (0.75), 46.59 (1.80), and 34.55 (2.46) respectively.

A dutch auger was used to sample soil to 2 metres depth in the stream channel, 5 metres from the stream channel, and in the upland position (Table 3.3). Depth to water table, depth to mottles, and drainage class in the riparian location were recorded in each block on 17 July 2003: hillslope 1: 84 cm, 48 cm, imperfect; hillslope 2: 92 cm, 0 cm, poor; hillslope 3 > 100 cm, 48 cm, imperfect. Drainage class is identified based on soil texture and depth to redoximorphic features of gley and mottling (Beckingham et al. 1996). Redoximorphic features provide visual evidence of periodic soil saturation due to the influence of water on oxidation, reduction, and translocation of free iron oxides (Simonson and Boersma 1972).

Table 3.1. Soil profile description at each hillslope-landscape position combination within the study site on 17 July 2003.

Horizon	Depth (cm)	Description*
Upland Position-Hillslope 1: Orthic Gray Luvisol		
LF	9–0	
Ae	0–20	Light gray (10YR 7/2 d); loam; moderate medium platy; soft; few, medium and fine, oblique, inped and exped roots; clear, wavy boundary; 19-22 cm thick.
Bt	20+	Brown (10YR 4/3 d); clay loam; coarse medium subangular blocky; hard; few, medium, fine and very fine, oblique, inped and exped roots; diffuse, gradual boundary.
Upland Position-Hillslope 2: Orthic Gray Luvisol		
LF	8-0	
Ae	0–15	Pale brown (10YR 6/3 d); loam; strong coarse platy; soft; few, coarse, horizontal, inped and exped roots and abundant, medium, horizontal and oblique, inped and exped roots; gradual, wavy boundary; 14-18 cm thick.
Bt1	15 –48	Brown (10YR 4/3 d); clay loam; moderate medium subangular blocky; firm; abundant, medium, oblique, inped and exped roots; gradual wavy boundary; 30-35 cm thick.
Bt2	48-100	Brown (10YR 4/3 d); clay loam-clay; strong coarse subangular blocky; very firm; abundant, medium, oblique inped and exped roots.
C	100+	Dark grayish brown (10YR 4/2 d); clay loam.

Upland Position-Hillslope 3: Orthic Gray Luvisol		
LF	8-0	
Ae	0-17	Light brownish gray (10YR 6/2 d); silt loam; coarse medium platy; soft; plentiful, fine, oblique, inped and exped roots; clear, wavy boundary; 15-18 cm thick.
AB	17-23	Brown (10YR 5/3 d); loam; coarse medium subangular blocky; slightly hard; plentiful, medium and fine, oblique, inped and exped roots; clear, wavy boundary; 6-8 cm thick.
Bt	23-100	Dark grayish brown (10YR 4/2 d); clay loam-clay; strong, coarse, subangular blocky; very firm; few, fine, oblique, inped and exped roots; gradual, wavy boundary; 75-90 cm thick.
C	100+	Dark grayish brown (10YR 4/2 d); clay loam.
Riparian Position–Hillslope 1: Gleyed Black Chernozem		
L	4-0	
Ah	0-20	Very dark gray (10 YR 3/1 d); sandy loam; strong fine granular; soft; abundant, medium and fine, vertical and oblique, inped and exped roots; clear, wavy boundary; 16-22 cm thick.
Hb	20-24	
Ahb	24-37	Black (10 YR 2/1 m); loam; few, medium and coarse, horizontal, inped and exped roots; clear, smooth boundary; 6-13 cm thick.
Btgj	37-84	Brown (10 YR 4/3 m); clay loam; strong medium subangular blocky; many, fine, distinct, brownish yellow (10 YR 6/8 m) mottles; firm; few, fine, oblique, inped and exped roots; gradual smooth boundary; 47-54 cm thick.
Cg	84+	Grayish brown (10 YR 5/2 m); clay loam; massive.

Riparian Position-Hillslope 2: Orthic Gleysol		
LH	7-0	
Ahgj	0-6	Very dark gray (GLEY 1 3/N m); loam; few, fine, distinct, red (2.5YR 4/6 m) mottles; strong medium granular; friable; abundant, fine, oblique, inped and exped roots; abrupt, smooth boundary; 6-7 cm thick.
Bg1	6-19	Very dark gray (GLEY 1 3/N m); loam-coarse sandy loam; few, fine, prominent, yellowish red (5YR 5/8 m) mottles; massive; few, fine and medium, oblique, inped and exped roots; abrupt, smooth boundary; 12-14 cm thick.
Bg2	19-46	Very dark gray (GLEY 1 3/N m); clay loam; many, fine, prominent, yellowish red (5YR 5/8 m) mottles; massive; firm; plentiful, fine and medium, oblique, inped roots; abrupt, smooth boundary; 18-26 cm thick.
Cg	46+	Light greenish gray (GLEY 1 7/10Y m); clay loam; many, fine, prominent, yellowish brown (10YR 5/6 m) mottles; massive; firm; few, fine, oblique, inped roots.
Riparian Position-Hillslope 3: Gleyed Gray Luvisol		
LF	14-0	
Ae	0-5	Pale brown (10YR 6/3 d); loam; weak fine platy; soft; few, fine, medium and coarse, horizontal and oblique, inped and exped roots; abrupt, wavy boundary; 5-6 cm thick.
Bt	5-48	Dark grayish brown (10YR 4/2 d); clay loam; weak to moderate fine to medium subangular blocky; hard; few, coarse, horizontal, inped roots and plentiful, very fine, oblique, inped and exped roots; gradual, wavy boundary; 45-55 cm thick.
Btgj	48+	Dark gray (10YR 4/N m), clay loam; many, fine, prominent, brown (7.5YR 5/6 m) mottles; moderate fine subangular blocky; firm.

*Soil profile description follows Soil Classification Working Group (1998).

Table 3.2. Soil chemical properties of each horizon at each hillslope-landscape position combination within the study site on 17 July 2003.

		C	EC	N	NH_4^+	NO_3^-	EN
	Depth	g C	mg C	g N	mg N	mg N	mg N
Horizon	(cm)	kg^{-1}	kg^{-1}	kg^{-1}	kg^{-1}	kg^{-1}	kg^{-1} *
Upland Position-Hillslope 1: Orthic Gray Luvisol							
LF	9 – 0	453	1586	7.88	44.8	0.00	224
Ae	0 – 20	7.19	36.3	0.87	0.42	0.00	1.12
Bt	20+	7.04	11.7	0.93	0.36	0.00	0.36*
Upland Position-Hillslope 2: Orthic Gray Luvisol							
LF	8-0	377	868	12.0	29.8	0.00	137
Ae	0 – 15	9.90	30.8	1.55	0.23	0.35	0.57*
Bt1	15 – 48	6.71	6.43	1.63	0.84	0.00	0.84*
Bt2	48-100	7.44	11.5	0.49	0.07	0.00	0.07*
C	100+		6.89		0.42	0.48	0.90*
Upland Position-Hillslope 3: Orthic Gray Luvisol							
LF	8-0	438	2775	8.15	28.7	0.00	285
Ae	0-17	6.17	59.3	0.89	0.00	0.00	1.25
AB	17-23	4.25	19.3	0.67	0.41	0.11	0.52*
Bt	23-100	7.06	18.2	0.84	0.21	0.00	0.21*
C	100+	6.93	11.7	0.91	0.17	0.20	0.38*
Riparian Position–Hillslope 1: Gleyed Black Chernozem							
L	4-0	444	1927	9.44	90.8	0.00	272
Ah	0-20	83.9	77.9	2.55	3.51	0.85	5.86
Hb	20-24	311	181	7.22	5.76	0.00	4.18
Ahb	24-37	64.6	43.1	3.29	1.30	0.39	3.42
Btgj	37-84	6.67	10.9	0.75	0.15	0.17	0.32*
Cg	84+	14.50	7.40	0.68	0.26	0.63	0.88*

Riparian Position-Hillslope 2: Orthic Gleysol							
LH	7-0	445	552	9.30	64.9	3.04	54.8
Ahgj	0-6	45.7	34.5	2.75	3.26	1.42	4.68*
Bg1	6-19	15.3	12.4	0.77	0.78	0.36	1.14*
Bg2	19-46	27.7	12.7	1.48	0.39	1.13	1.52*
Cg	46+	7.63	5.50	0.91	0.30	1.08	1.37*
Riparian Position-Hillslope 3: Gleyed Gray Luvisol							
LF	14-0	369	678	6.96	16.6	0.00	92.5
Ae	0-5	10.8	27.7	1.19	0.20	0.00	0.20*
Bt	5-48	8.95	13.5	1.02	0.46	0.00	0.46*
Btgj	48+	8.49	7.15	0.90	0.30	0.82	1.1*

*In these cases, EN was below detection limits and was estimated as $\sum$ (NH_4^+ and NO_3^-).

Table 3.3. Soil texture to 2 metres depth in the stream channel, 5 metres from the stream channel, and in the upland area at the study site. Samples collected on 17 July 2003.

In stream		5 metres from stream		Upland	
Depth (cm)	Texture*	Depth (cm)	Texture	Depth (cm)	Texture
0-22	L	0+	C	0-15	L
22-166	CL			15-48	CL
166+	L			48+	C

* Soil texture was determined in the field by hand texturing; L = loam, CL=clay loam, C=clay.

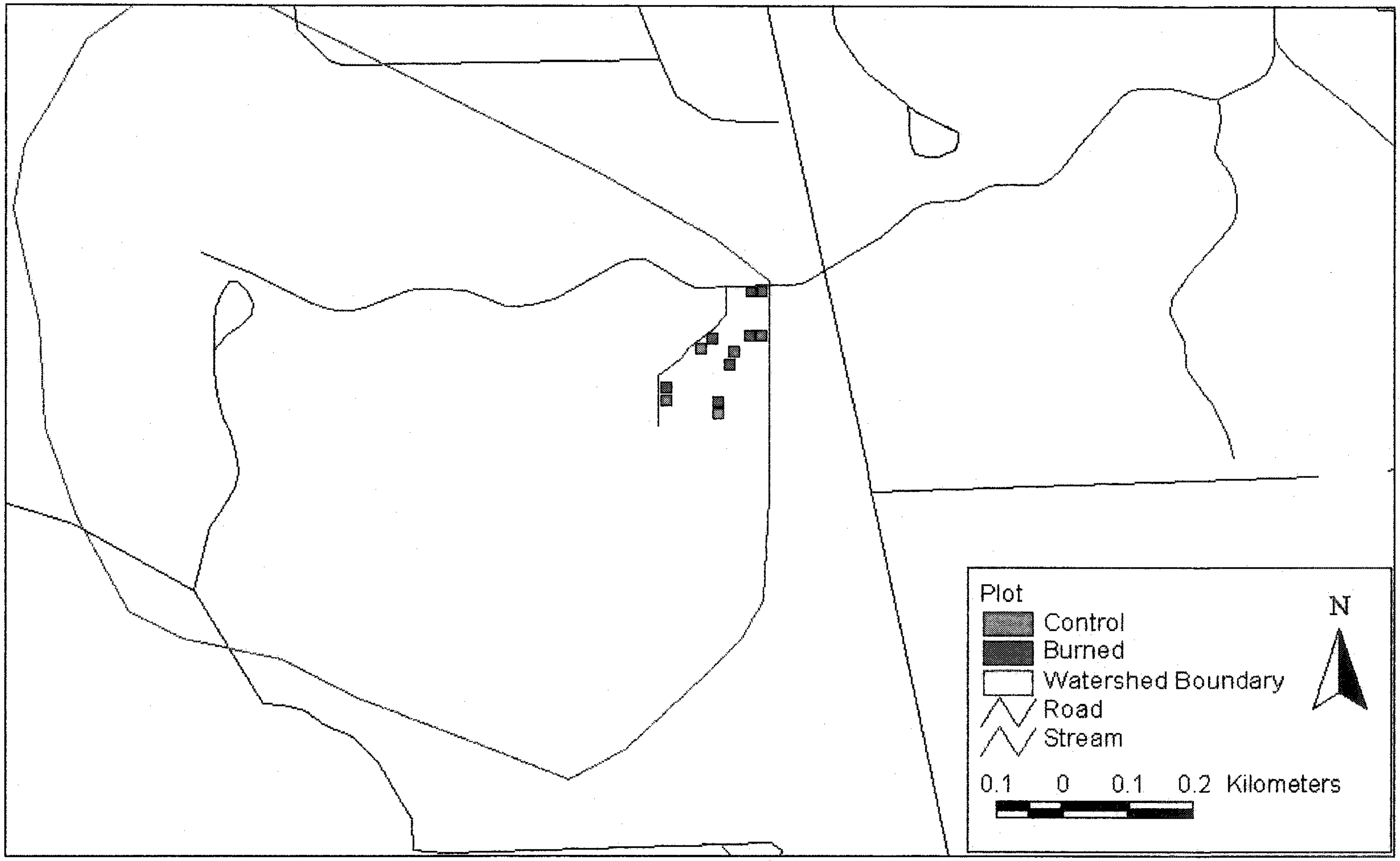
3.2.2. Sampling Design

The study site was selected based on a lack of stand-replacing fire disturbance within the last 60 years; a moderate slope gradient (15-30%) within the stand; the presence of a forested riparian valley with a poorly defined channel and intermittent or ephemeral stream flow; and an obvious increase in soil moisture content from the upland to the riparian position. The riparian position was defined as the toeslope area adjacent to the stream channel; the upland position was defined as the water-shedding upper slope position below the crest.

The study site was divided into three north to northwest facing hillslopes varying in length from 50 to 90 metres and separated laterally by 15 to 200 metres. Each hillslope was identified as a block and stratified into upland and riparian positions. In each block, two adjacent 2 x 2 m plots were established in canopy gaps of each slope position. One of the 2 plots was randomly assigned a burn treatment; the other plot was not disturbed and used as a control (reference) plot. Therefore, the study included 4 plot types: upland control (UC), upland burn (UB), riparian control (RC), and riparian burn (RB), in a 2 x 2 experiment. The experimental design is shown in Figure 3.2.

The burn treatment involved some plot preparation before burning. In mid-June the burn plots were prepared for prescribed burn by clipping the aboveground vegetation and leaving it in place, digging perimeter trenches to mineral soil, severing all root connections, and removing flammable vegetation around the plot. All the plots were covered with two layers of thin plastic sheet for 10 days to dry the forest floor but the material was still too wet to burn. Therefore, the forest floor was removed from the burn treatment plots to approximately 5 cm depth, dried in laboratory ovens at 68°C for 48 hours, and returned to the same plot for burning. The burn was conducted on 4 July 2003 using propane torches (see Figure 3.3) and was extinguished with water; it rained later that evening. After the burn, black ash and charred woody debris covered the surface of the burned plots. The mean dry weight of forest floor replaced on site for burning and depth of burn is shown for the UB and RB plots in Table 3.4.

Figure 3.2. Illustration of the experimental design within the study watershed.



Note: The digital data used to create this map was provided by Weyerhaeuser Company Ltd.

Figure 3.3. Immediately after beginning (A) and after completing (B) the experimental burn in one of the plots on 4 July 2004.

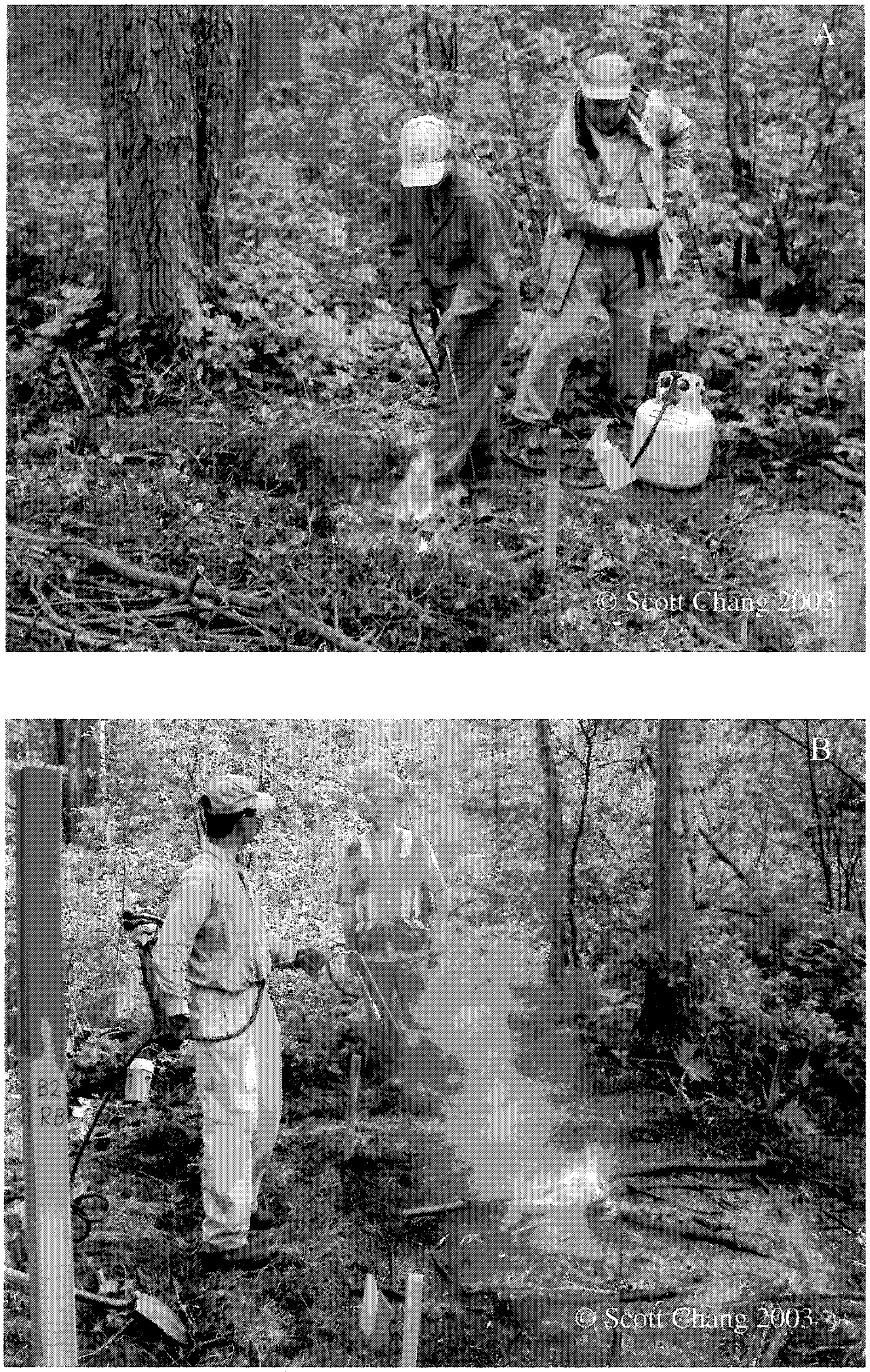


Table 3.4. Mean (SE in parentheses) dry weight of forest floor replaced and depth of burn after prescribed burning on 4 July 2003 in the upland burn (UB) and riparian burn (RB) plots.

Plot Type	Dry Mass of Forest Floor Removed (kg) *	Depth of Burn (cm) †	Depth of Burn (%)‡
UB	32 (3.8)	5.4 (0.35)	64 (4.6)
RB	21 (3.0)	1.8 (0.32)	34 (15)

* Dry weight was estimated based on the weight of one pail of organic matter dried at 68°C for 48 hours multiplied by the number of pails replaced for each plot.

† Depth of burn (cm) is the difference in forest floor depth before and after the burn.

‡ Depth of burn (%) is the depth of burn (cm) divided by the original forest floor depth.

3.2.3. Field and Laboratory Measurements

To assess pre-burn differences in species composition and cover between plots, plant species cover estimates were performed using a 59 cm by 59 cm quadrat (replicated three times per plot) in June 2003.

Soil temperature and moisture content were monitored hourly 5 cm below the mineral soil surface from 16 July to 1 October 2003. To monitor soil temperature, 2 or 3 HOBO® H8 Temp Loggers (Onset Computer Corporation, Bourne, MA, USA) were buried in each plot. Each HOBO® H8 Temp Logger was enclosed in 2 Ziploc® bags with a package of silica gel dessicant, a block of soil to 10 cm depth was cut out to expose the soil profile, a shelf was cut out at 5 cm mineral soil depth in the soil profile, the HOBO® H8 Temp Logger was inserted flat into the shelf, and the shelf and hole were backfilled with original soil material to minimize disturbance. HOBO® H8 Temp Loggers have an accuracy of ± 0.7°C and a resolution of 0.4°C at 21°C (Onset Computer Corporation 2002).

To monitor soil water content, a Campbell Scientific CS616 Water Content Reflectometer (Campbell Scientific, Inc., Logan, UT, USA) was inserted horizontally using a CS615 Installation Kit at 5 cm depth of mineral soil of an exposed soil profile at the edge of each plot. The hole was then backfilled with original soil material in its natural order to minimize disturbance; this installation may alter hydrologic characteristics by altering the structure of the upper mineral soil. The CS616 Water Content Reflectometers have an accuracy of ± 2.5% and a resolution of approximately 0.1% (Campbell Scientific, Inc. 2002). The period reading recorded by a Campbell Scientific CR10X datalogger was converted to volumetric moisture content after calibration of the probes to account for the differences in soil bulk density between the riparian and upland mineral soil.

Soil samples were collected for determination of total, extractable and microbial biomass C and N contents, pH, and gravimetric moisture content from 5 June to 1 October 2003 at 28 to 30 day intervals. On each sampling date, a stainless steel corer (10 cm long, 5 cm diameter) was used to collect 3 cores each of the LFH (hereafter referred to as the forest floor) and mineral soil from 0-10 cm (hereafter referred to as the mineral soil) in each plot. The cores were composited, transported to the laboratory, and processed within 48 hours.

To determine bulk density, mineral soil was sampled 8 times per plot using a small corer (31.42 cm^3) and dried at 105°C for 24 hours; bulk density varied between plots from 0.6 to 1.2 g cm^{-3}. Average forest floor bulk density was estimated as 0.17 g cm^{-3} based on data in the literature (Offord 1999; Carmosini et al. 2003).

Forest floor and mineral soil samples were homogenized by sieving to 4 mm. Gravimetric moisture content was determined by drying two 5 g subsamples at 68°C for 48 hours for forest floor or at 105°C for 24 hours for mineral soil (Kalra and Maynard 1991). Extractable C (EC) and N (EN) were extracted by shaking approximately 5 g fresh forest floor and 10 g fresh mineral soil in 50 mL 2 M KCl for 1 hour (Kalra and Maynard 1991). Extracts were filtered gravimetrically through prewashed Fisherbrand® Q2 filter paper and immediately frozen at -18°C until analysis. In the extracts, EC and EN were determined on a Shimadzu TOC-V CSH/CSN Total Organic Carbon Analyzer (Shimadzu Corporation, Kyoto, Japan).

The chloroform fumigation-extraction method was used to measure microbial biomass C (MB-C) and N (MB-N) within 5 to 7 days of sample collection (Horwath and Paul 1996). Two pairs of each sample (5 g dry weight basis for forest floor, 10 g dry weight basis for mineral soil) were weighed; one was fumigated with ethanol-free chloroform for 24 hours at room temperature (22°C) and the other was extracted by shaking in 60 mL 0.5 M K_2SO_4 for 1 hour, gravity filtered through prewashed Fisherbrand® Q2 filter paper and immediately frozen at -18°C. After the 24 hour incubation period, chloroform was removed from the soil samples and the fumigated samples were extracted using the same procedure. Evaluation of the change in moisture content during fumigation showed that gravimetric moisture content was altered very little; a reduction of 2% occurred in the forest floor and an increase of 3% occurred in the mineral soil. The filtrate was analyzed for total extractable organic C (TEC) and total extractable N (TEN) using a Shimadzu TOC-V CSH/CSN Total Organic Carbon Analyzer (Shimadzu Corporation, Kyoto, Japan). MB-C and MB-N were calculated by the equations $\text{MB-C}=F_C/K_C$ and $\text{MB-N} = F_N/K_N$ where F_C and F_N are the difference in TEC and TEN between fumigated and unfumigated samples, K_C is 0.38 (Vance et al. 1987) and K_N is 0.54 (Brookes et al. 1985). The microbial C/N ratio was calculated for each sample by dividing MB-C by MB-N. Microbial biomass C and N, EC, and EN were converted to an areal basis from a per gram basis using the average soil bulk density.

Forest floor and mineral soil samples collected on June 5, July 5, and October 1 were air dried and ground to a fine powder using a Type MM 200 Retsch Mixer Mill (Retsch GmbH 88 Co., KG, Germany). Total C and N analyses were completed using a Shimadzu TOC-V Series SSM-5000A Solid Sample Module for Total Organic Carbon Analyzer (Shimadzu Corporation, Kyoto, Japan). The C/N ratio of each sample was calculated by dividing total C by total N. The MB-C/TC ratio was calculated by dividing MB-C by total C and the MB-N/TN ratio was calculated by dividing MB-N by total N. Soil C and N contents were converted to an areal basis from a per gram basis using the average soil bulk density.

Air dried forest floor and mineral soil samples were used to determine soil pH in 0.01 M $CaCl_2$ (hereafter referred to as pH_{CaCl2}); a 1:4 soil:0.01M $CaCl_2$ ratio was used for forest floor and a 1:2 soil: 0.01M $CaCl_2$ ratio was used for mineral soil. The use of $CaCl_2$ to suspend the soil displaces exchangeable acid cations thereby resulting in a lower pH value than a water suspension (Nykvist and Skyllberg 1989).

3.2.4. Statistical Analyses

The hypotheses of no significant pre-burn floristic difference between the burn and control plots before burning but a significant difference between the riparian and upland plots before burning was tested using a rank-transformed Multi-Response Permutation Procedure (MRPP) (McCune and Grace 2002). MRPP is a non-parametric procedure similar to discriminant analysis and multivariate analysis of variance that is appropriate for plant community data; rank transformation of the Euclidean distance measures allows greater sensitivity of the analysis and tests the null hypothesis of no difference in average within-group ranked distances (McCune and Grace 2002).

Indicator Species Analysis (Dufrêne and Legendre 1997) was used to identify species that were significantly more frequent and abundant in the riparian compared with the upland plots. This method produces indicator values for each species in each group based on its relative frequency and relative abundance where the perfect indicator (indicator value = 100) is always present in and exclusive to the group. Indicator values are then tested for statistical significance through a Monte Carlo test (1000 runs with randomized data) (McCune and Grace 2002). An α value of 0.05 was used to identify significant indicator species between riparian and upland positions. Both multivariate tests (MRPP and Indicator Species Analysis) were performed using the PC-ORD ver. 4 program (McCune and Mefford 1999).

The hypotheses of no significant differences in temperature, volumetric moisture content, gravimetric moisture content, $\mathrm{pH}_{\mathrm{CaCl2}}$, total C, total N, EC, EN, MB-C, MB-N, microbial C/N ratio, MB-C/TC ratio or MB-N/TN ratio between the upland and riparian positions, burn and control plots, or months was tested using the sample data collected after the burn was conducted (July to October 2003). Data from the forest floor and mineral soil was analyzed separately. Outliers were assessed using boxplots and scatterplots of the data and removed before statistical analysis. The assumptions of ANOVA (normality and homoscedasticity of the residuals between treatment groups) were assessed between treatment groups using the Shapiro-Wilks test statistic (W) in SAS version 8.1 (SAS Institute Inc., Cary, NC, USA) and the Brown-Forsythe modification that uses the median to calculate the Levene test statistic in SPSS for Windows 11.5 (SPSS Inc., Chicago, IL, USA). Natural log transformations were performed for the variables that did not meet the normality criterion; adjustment for heterogeneity of variance was performed using the group= command within the MIXED procedure of SAS version 8.1. Analysis of variance (ANOVA) was used for statistical analysis due to the complexity of the design structure (randomized block split-split plot design with repeated measures); ANOVA was conducted using the MIXED procedure in SAS version 8.1. For each independent variable, the June sampling data was used as a covariable to account for pre-burn differences between adjacent plots. Means were compared using t-tests given statistically significant ANOVA results for month or any interactions; due to the low sample size, an α value of 0.10 was chosen to indicate statistical significance. Spearman correlation coefficients, the distribution-free equivalent of Pearson correlation coefficients, were calculated using SPSS for Windows 11.5 on the raw data to determine which variables co-varied; the significant correlations were verified by plotting the data. Results of the correlation analysis are not shown, however, significant ($P<0.10$) correlation between variables is indicated in the text followed by the correlation coefficient (r), the P value, and the number of samples (n) used in the correlation analysis.

3.3. Results

3.3.1. Species Composition and Cover

The MRPP showed no significant pre-burn floristic difference between the burn and control plots but a significant difference between the riparian and upland plots (Table 3.5). The Indicator Species Analysis identified 11 indicator species in the upland position and 3 indicator species in the riparian position based on a P value of 0.05 (Table 3.6). In the upland position, the significant indicator species were *Aralia nudicaulis*, *Aster* spp, *Cornus canadensis*, *Fragaria virginiana*, *Lathyrus ochroleucus*, *Maianthemum canadense*, *Mertensia paniculata*, *Petasites sagittatus*, *Rubus idaeus*, *Solidago* spp, and *Viola renifolia*. In the riparian position, the significant indicator species were *Equisetum* spp, *Gymnocarpium dryopteris*, and *Heracleum lanatum*.

Table 3.5. Results of the MRPPs testing the null hypotheses: a) no significant difference in pre-burn species composition between adjacent plots and b) no significant difference in pre-burn species composition between slope positions.

Plot type	Average distance	N	MRPP Statistics
a) Burn vs. Control Plots			
Burn	0.4084	18	Observed delta = 0.4920
Control	0.5709	17	Expected delta = 0.5000
			T = -1.2856 A = 0.016 P = 0.1070
b) Riparian vs. Upland Plots			
Riparian	0.4008	18	Observed delta = 0.4558
Upland	0.5140	17	Expected delta = 0.5000
			T = -7.0834 A = 0.0884 P <0.0001

Note: Average distance is the mean Euclidean distance between each combination of quadrats from a particular plot type. N is the number of quadrats sampled in each plot type. The observed delta is the weighted mean within-group distance calculated from the data while the expected delta is derived from a null distribution. T is the test statistic that describes the separation between groups. A is the chance-corrected within-group agreement.

Table 3.6. Indicator values for statistically significant (P<0.05) indicator species from riparian and upland plots at the study site.

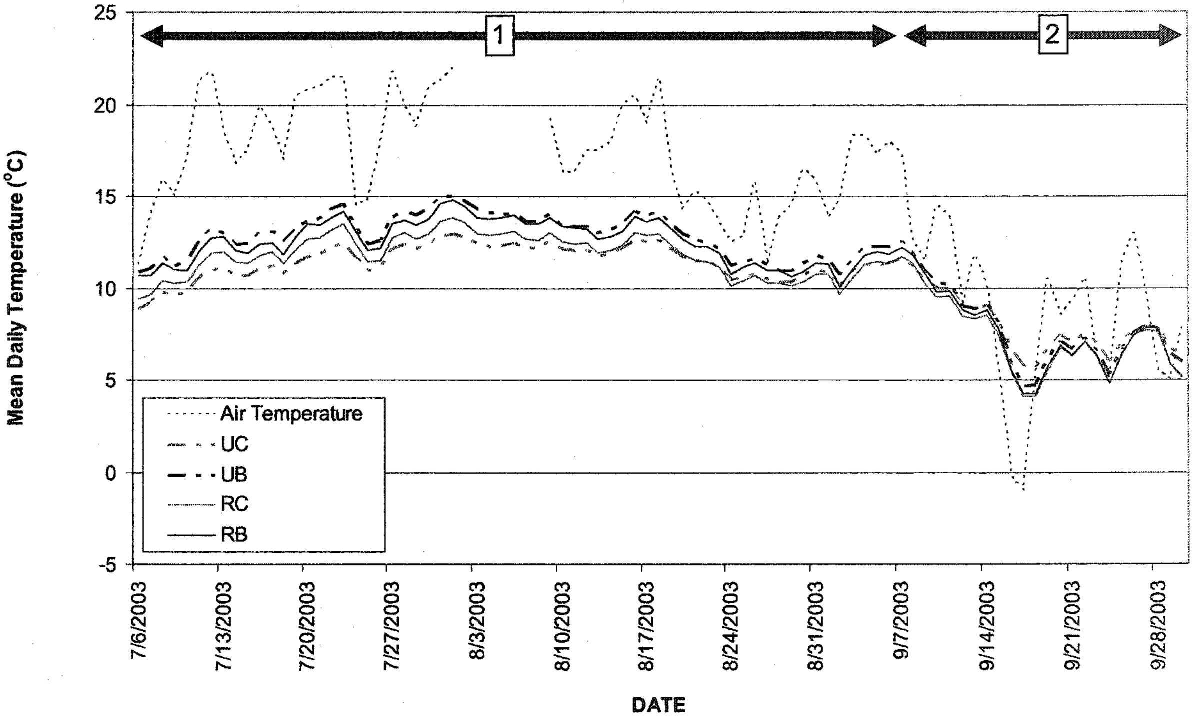
Plot Type	Species	Observed Indicator Value	Mean Randomized Indicator Value	P value
Upland	*Aralia nudicaulis*	50.0	25.0	0.005
	Aster spp.	47.1	19.4	0.001
	Cornus canadensis	72.0	41.9	0.002
	Fragaria virginiana	57.8	40.3	0.031
	Lathryus ochroleucus	41.1	21.4	0.013
	Maianthemum canadense	41.7	28.1	0.047
	Mertensia paniculata	62.2	35.9	0.005
	Petasites sagittatus	45.6	26.0	0.016
	Rubus idaeus	53.9	35.5	0.017
	Solidago spp.	29.4	13.9	0.026
	Viola renifolia	39.8	20.2	0.019
Riparian	*Equisetum* spp.	88.9	31.2	0.001
	Gymnocarpium dryopteris	77.8	28.0	0.001
	Heracleum lanatum	61.1	23.6	0.001

Note: Observed indicator values are calculated by multiplying the relative frequency and relative abundance of the species in a specific group. The mean randomized indicator values are calculated from 1000 Monte Carlo runs with randomized data. The P value is the proportion of randomized runs that had indicator values higher than the observed.

3.3.2. Soil Temperature

Soil temperature in the mineral soil was positively correlated (r=0.84, P<0.0001, n=1032) with air temperature (Figure 3.4) and negatively correlated with forest floor depth (r=-0.18, P<0.0001, n=1032). Soil temperature in the burned plots was significantly (F=33.15, P=0.0289, ANOVA data not shown) greater than that in the control plots; mean (SE) daily soil temperature from July 6 to October 1, 2003 was 10.47 (0.12) °C in the UC plots, 11.48 (0.17) °C in the UB plots, 10.56 (0.16) °C in the RC plots, and 11.14 (0.17) °C in the RB plots. However, the data suggests two separate trends in soil temperature between the burned and control plots: one from 6 July to 10 September (Period 1), and the other from 10 September to 1 October (Period 2). During Period 1, the UB plots were 1.4 °C warmer than the UC plots and the RB plots were 0.7°C warmer than the RC plots. During Period 2, the UB plots were approximately 0.3 °C cooler than the UC plots while the RB plots remained slightly (0.1°C) warmer than the RC plots (Figure 3.4).

Figure 3.4. Daily mean air temperature* and soil temperature (°C) at 5 cm depth of mineral soil in the upland control (UC), upland burn (UB), riparian control (RC), and riparian burn (RB) plots from 6 July to 30 September 2003. Lines 1 and 2 indicate the two periods in which soil temperature trends differed.



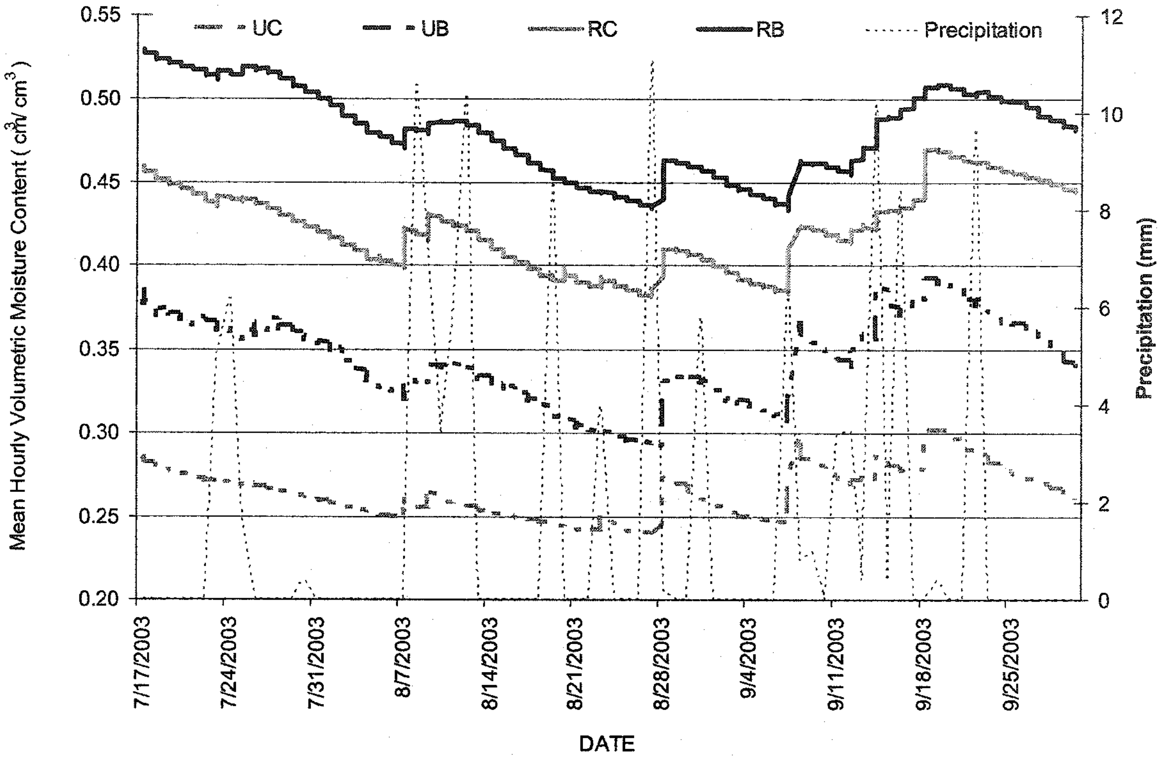
* Daily mean air temperature data from the Drayton Valley (53°13'N 114°57'W) weather station; no data was available from August 3 to August 10, 2003 (Environment Canada 2004b).

3.3.3. Volumetric and Gravimetric Soil Moisture Contents

In the mineral soil, peaks in hourly volumetric soil moisture content from 17 July to 30 September appeared to correspond to precipitation events (Figure 3.5). Mean monthly volumetric soil moisture content was not significantly different between the riparian and upland ($P=0.1339$) or burned and control plots ($P=0.1354$), however July and September had significantly higher moisture contents ($P=0.0004$) than August (ANOVA data not shown). The mean (SE) hourly volumetric moisture content tended to be highest in the RB plots (0.480 (0.000616) cm^3 cm^{-3}) followed by the RC (0.421 (0.000566) cm^3 cm^{-3}), UB (0.343 (0.000603) cm^3 cm^{-3}) and UC plots (0.264 (0.000355) cm^3 cm^{-3}) (Figure 3.6).

In the forest floor, gravimetric moisture content did not differ between burned and control plots but was significantly higher in the riparian plots than in the upland plots in July. In the mineral soil, gravimetric moisture content was unaffected by burn treatment or position (Table 3.7). In both the forest floor and mineral soil, gravimetric moisture content was significantly greater in July than in August, September, and October and was significantly lower in August (forest floor only) and September compared to October (Figure 3.6 A and B; Table 3.7). The gravimetric moisture content data, in addition to data for many of the response variables, is illustrated in a box plot (Figures 3.6 A and B). In a box plot, the middle line in the box represents the median value; the bars extend up to the maximum value and down to the minimum value.

Figure 3.5. Daily precipitation (mm)* and hourly volumetric soil moisture content (cm^3/cm^3) at 5 cm depth of mineral soil for the upland control (UC), upland burn (UB), riparian control (RC), and riparian burn (RB) plots from 17 July to 30 September 2003.



* Daily precipitation data from the Drayton Valley (53°13'N 114°57'W) weather station (Environment Canada 2004b).

3.3.4. $\mathrm{pH}_{\mathrm{CaCl2}}$

In the forest floor, $\mathrm{pH}_{\mathrm{CaCl2}}$ was significantly higher in the burned plots compared with the control plots. On average, $\mathrm{pH}_{\mathrm{CaCl2}}$ was 1.1 units higher in the UB plots compared with the UC plots and was 0.6 units higher in the RB plots compared with the RC plots (Figure 3.6 C). The RC plots tended to have a higher $\mathrm{pH}_{\mathrm{CaCl2}}$ than the UC plots in July, September, and October (Figure 3.6 C). In the mineral soil, $\mathrm{pH}_{\mathrm{CaCl2}}$ was 0.6 pH units higher in the UB plots than in the UC plots but did not differ between the RB and RC plots. Soil $\mathrm{pH}_{\mathrm{CaCl2}}$ was positively correlated to gravimetric moisture content ($r=0.518$, $P<0.0001$, $n=48$) and differed significantly between months (Table 3.7; Figure

3.6 D). Soil pH_{CaCl2} tended to be higher in the RC plots than in the UC plots in July, August, and September.

Figure 3.6. Gravimetric moisture content and pH_{CaCl2} in the forest floor and mineral soil (0-10 cm) of samples collected in July, August, September, and October from the upland control (UC), upland burn (UB), riparian control (RC), and riparian burn (RB) plots.

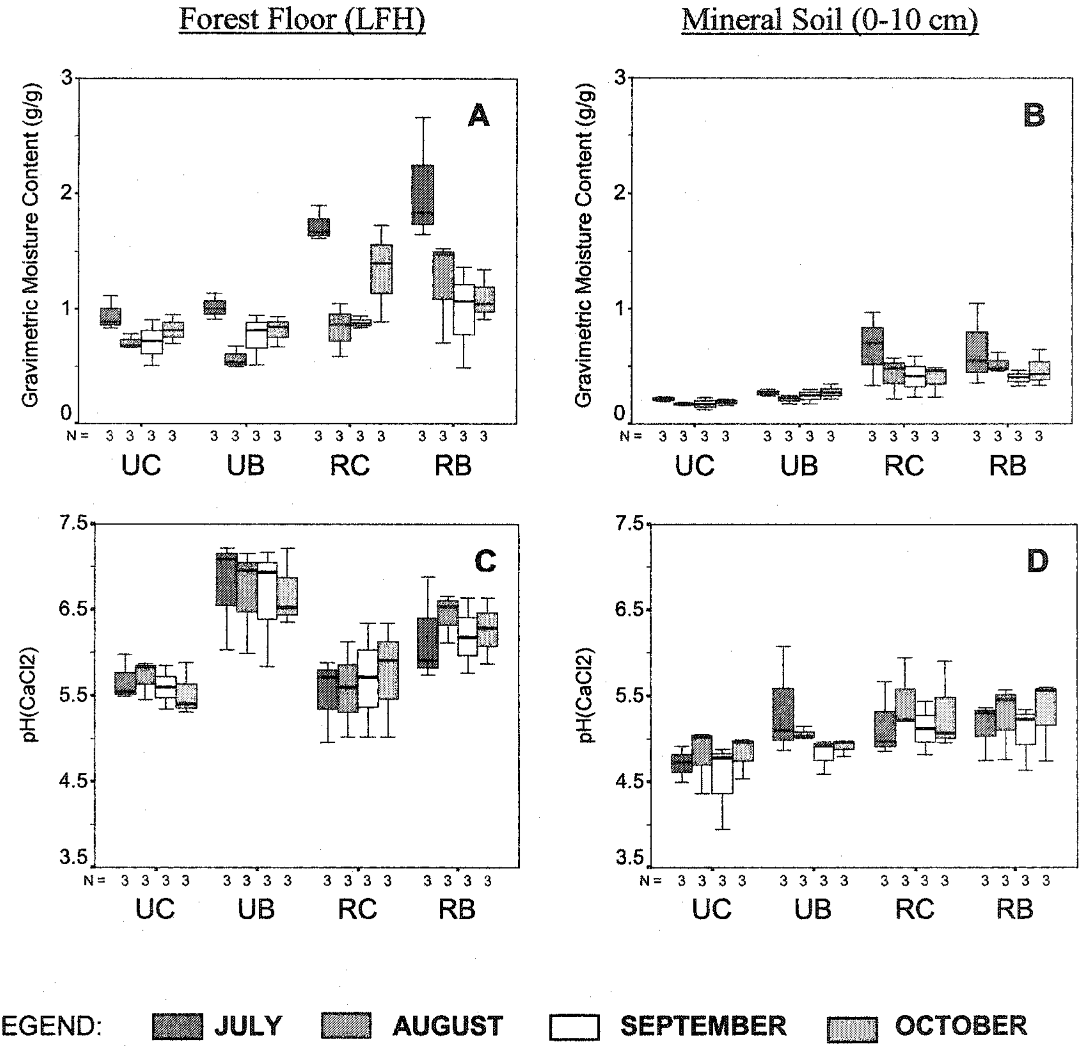


Table 3.7. F and P values from the analysis of variance for gravimetric moisture content and pH_{CaCl2} in the forest floor (LFH) and mineral soil (0-10 cm).

	Gravimetric Moisture Content		pH_{CaCl2}	
Effect	F	P value	F	P value
FOREST FLOOR (LFH)				
Position (Pos)	5.61	0.254	2.64	0.351
Burn	0.45	0.625	34.95	0.107
Pos*Burn	0.73	0.550	0.42	0.635
Month	22.51	<0.0001	0.50	0.687
Pos*Month	6.36	0.003	1.32	0.290
Burn*Month	1.32	0.292	0.39	0.760
Pos*Burn*Month	1.82	0.171	1.35	0.282
MINERAL SOIL (0-10 cm)				
Position (Pos)	1.84	0.404	0.00	0.970
Burn	8.64	0.209	1.21	0.470
Pos*Burn	2.57	0.355	1.08	0.488
Month	10.58	0.0001	7.42	0.001
Pos*Month	2.28	0.105	1.30	0.298
Burn*Month	0.96	0.426	1.37	0.275
Pos*Burn*Month	1.50	0.239	0.78	0.517

3.3.5. Total C and N Contents and C/N Ratio

In the forest floor, total C and N contents were positively correlated (r=0.944, P<0.0001, n=24) and were significantly lower in the burned plots than in the control plots (C: F=11.68, P=0.0637; N: F=7.63, P=0.0967, ANOVA data not shown). On average, the total C and N contents in the burned plots in the upland were 34% and 32%, respectively, of those in the control plots; in the riparian position the burned plots contained 42% of the total C content and 46% of the total N content compared with the control plots (Table 3.8). The UB plots had a significantly lower total N content than both the UC and RC plots. Total C and N contents tended to be higher in the riparian plots than in the upland plots (Table 3.8). In the forest floor, the C/N ratio was not correlated to either total C or N contents and was not significantly affected by position or the burn treatment (Table 3.8).

In the mineral soil, total C content was significantly correlated with total N content ($r=0.430$, $P=0.036$, $n=24$) and the C/N ratio ($r=0.863$, $P<0.0001$, $n=24$). Total C content tended to be lower (89 and 86% of the control values in the riparian and upland positions, respectively) in the burned plots than in the control plots and tended to be higher in the riparian plots that contained, on average, 43% more C than in the upland plots (Table 3.8). The total N content was slightly (7%) greater in the RB plots compared with the controls; the UB and UC plots and the upland and riparian plots did not differ in total N content (Table 3.8). The C/N ratio tended to be lower in the upland plots than in the riparian plots and in the burned plots compared with the control plots.

Table 3.8. Mean (SE in parentheses) total C and N content, and soil C/N ratio of the upland control (UC), upland burn (UB), riparian control (RC), and riparian burn (RB) plots in the forest floor (LFH) and mineral soil (0-10 cm). Treatment means followed by the same letter for the forest floor or mineral soil are not significantly different ($P>0.1$).

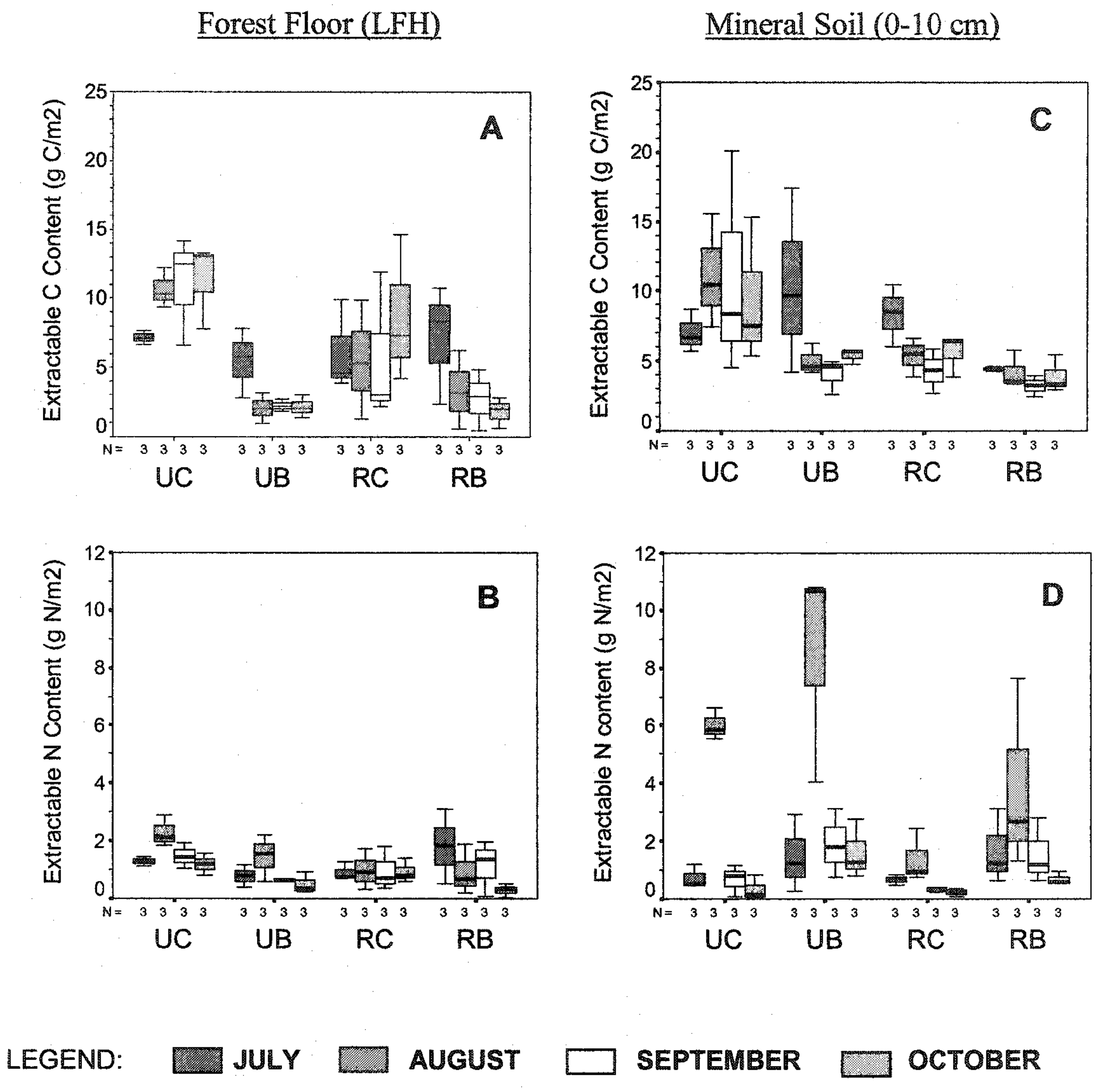
Plot Type	Total C content (g C m^{-2})	Total N content (g N m^{-2})	C/N Ratio
FOREST FLOOR (LFH)			
UC	3397.22 (98.66) a	88.70 (6.75) a	39.75 (3.83) a
UB	1158.95 (129.76) b	27.73 (2.70) b	41.46 (1.40) a
RC	4602.94 (1245.49) a	109.85 (31.97) a	43.42 (3.84) a
RB	1937.46 (477.61) b	50.02 (11.00) b	37.43 (3.90) a
MINERAL SOIL (0-10 CM)			
UC	3040.71 (689.47) a	191.33 (23.52) a	15.46 (2.10) a
UB	2641.89 (327.74) a	190.19 (11.80) a	13.99 (1.62) a
RC	5294.70 (1016.39) a	188.40 (13.87) a	27.66 (4.14) a
RB	4699.33 (607.90) a	202.61 (9.80) a	23.86 (3.53) a

3.3.6. Extractable C and N Contents

In the forest floor, EC and EN contents were significantly correlated ($r=0.780$, $P<0.0001$, $n=48$). The EC content in the UB plots was 30% of that in the UC plots and in the RB plots was 58% of the EC content in the RC plots. The EC content was significantly lower in the burned plots compared with the control plots in August, September, and October. EC content was significantly higher in the UC plots than in the UB, RC, or RB plots in August, September and October (Figure 3.7 A; Table 3.9). EN content tended to be lower in the UB plots than in the UC plots but there was no trend apparent between the RB and RC plots. EN content was significantly lower in October than in July, August, and September (Figure 3.7 B; Table 3.9).

In the mineral soil, the mean EC content did not differ significantly between the burned and control or the upland and riparian plots (Figure 3.7 C; Table 3.9). The UC plots had a significantly higher EC content than the UB, RC, and RB plots in August and September; the EC content was significantly higher in the RC compared with the RB plots in July (Table 3.9; Figure 3.7 C). The EN content tended to be higher in the burned plots (43% and 67% for the upland and riparian positions, respectively) as compared with the control plots. The EN content was significantly higher in August than in July, September and October and was significantly lower in October than in other months (Table 3.9; Figure 3.7 D).

Figure 3.7. Extractable C and N contents in the forest floor and mineral soil (0-10 cm) of samples collected in July, August, September, and October from the upland control (UC), upland burn (UB), riparian control (RC), and riparian burn (RB) plots.



3.3.7. Microbial Biomass C and N Contents and Microbial C/N Ratio

In the forest floor, microbial biomass C (MB-C) and N (MB-N) were significantly correlated ($r=0.909$, $P<0.0001$, $n=48$). Both MB-C and MB-N were significantly lower in the burned plots (MB-C 12% and 26%, MB-N 11% and 23% of the control values in the upland and riparian positions, respectively) than in the control plots (Figure 3.8 A and B; Table 3.9). In the UC and RC plots, MB-C was significantly higher in September than in the other months; MB-N was significantly higher in September and October than in

July and August. In the burned plots, MB-C was significantly greater in July and August than in September and October; MB-N was similar from July to October. MB-N in the UB plots was significantly lower than in the RB plots from July to October (Figure 3.8 A and B; Table 3.9). Microbial C/N ratio was significantly higher in the burned plots than in the control plots in July and was significantly lower in the burned plots in August and September (Figure 3.8 C; Table 3.9). In October, the microbial C/N ratio was significantly higher in the RB and UC plots as compared with the RC plots (Figure 3.8 C).

In the mineral soil, MB-C was significantly correlated to MB-N ($r=0.873$, $P<0.0001$, $n=48$); both MB-C and MB-N tended to be lower in the burned (MB-C 67% and 69%, MB-N 85% and 81% of the control values in the upland and riparian positions, respectively) as compared with the control plots. Both MB-C and MB-N tended to be higher in the riparian compared with the upland plots (Figure 3.8 D and E). Microbial C/N ratio was significantly lower in the burned plots than in the control plots (Figure 3.8 F; Table 3.9).

Figure 3.8. Microbial biomass C and N contents, and the microbial C/N ratio in the forest floor and mineral soil (0-10 cm) of samples collected in July, August, September, and October from the upland control (UC), upland burn (UB), riparian control (RC), and riparian burn (RB) plots.

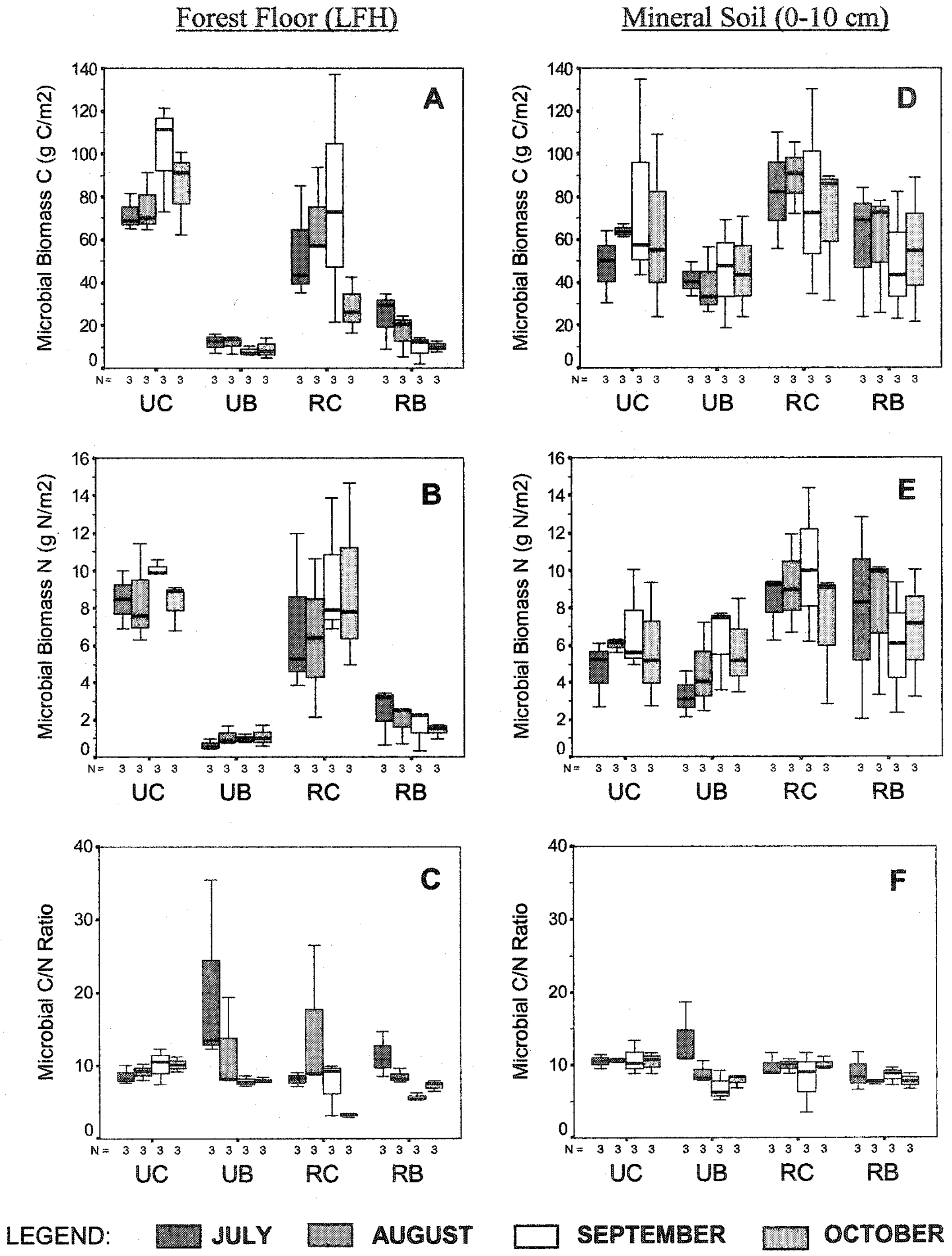


Table 3.9. F and P values from the analysis of variance for extractable C (EC), extractable N (EN), microbial biomass C (MB-C), microbial biomass N (MB-N), and microbial C/N ratio in the forest floor and mineral soil (0-10 cm).

	EC		EN		MB-C		MB-N		Microbial C/N Ratio	
Effect	F	P value	F	P value	F	P value	F	P value	F	P value
FOREST FLOOR (LFH)										
Position (Pos)	0.73	0.549	2.30	0.371	0.15	0.766	3.68	0.306	10.29	0.192
Burn	8.01	0.216	1.55	0.431	158.17	0.051	132.93	0.055	0.28	0.689
Pos*Burn	38.04	0.102	7.23	0.227	20.15	0.140	11.44	0.183	3.79	0.302
Month	6.82	0.002	3.58	0.029	2.21	0.113	2.97	0.052	7.19	0.001
Pos*Month	1.82	0.170	1.25	0.313	2.01	0.139	0.82	0.495	4.41	0.013
Burn*Month	16.83	<0.0001	1.67	0.201	3.46	0.032	4.96	0.008	5.96	0.004
Pos*Burn*Month	2.64	0.072	0.98	0.421	1.21	0.329	2.87	0.058	2.61	0.075
MINERAL SOIL (0-10 CM)										
Position (Pos)	7.71	0.109	2.99	0.334	0.36	0.656	0.34	0.664	2.06	0.288
Burn	7.96	0.106	12.55	0.175	12.26	0.177	0.40	0.641	14.59	0.062
Pos*Burn	0.08	0.807	0.27	0.696	2.45	0.362	0.32	0.673	1.09	0.405
Month	2.64	0.073	22.16	<0.0001	0.29	0.833	1.99	0.142	1.28	0.305
Pos*Month	0.30	0.825	3.28	0.038	1.19	0.336	2.05	0.133	0.15	0.930
Burn*Month	0.78	0.519	1.14	0.353	0.65	0.593	1.35	0.280	0.67	0.577
Pos*Burn*Month	3.04	0.049	0.39	0.765	0.20	0.895	1.37	0.276	1.67	0.199

3.3.8. MB-C/TC Ratio and MB-N/TN Ratio

In the forest floor, position, burn, or the position-burn interaction did not have a significant effect on the average MB-C/TC ratio. The mean (SE) ratio of MB-C/TC in the forest floor was 0.023 (0.0018) in the UC plots, 0.0090 (0.0011) in the UB plots, 0.0088 (0.0021) in the RC plots, and 0.011 (0.0024) in the RB plots. In contrast, the mean MB-N/TN ratio was significantly lower in the burned plots compared with the control plots (F=16.63, P=0.0552, ANOVA data not shown). The mean (SE) ratio of MB-N/TN was 0.096 (0.0049) in the UC plots, 0.032 (0.0051) in the UB plots, 0.082 (0.019) in the RC plots, and 0.040 (0.0043) in the RB plots. In the mineral soil, there were no significant differences between plot types for the MB-C/TC ratio or the MB-

N/TN ratio; the mean (SE) ratios of MB-C/TC and MB-N/TN were 0.018 (0.0015) and 0.027 (0.0035) in the UC plots, 0.016 (0.0012) and 0.024 (0.0046) in the UB plots, 0.015 (0.00089) and 0.042 (0.0068) in the RC plots, and 0.011 (0.0015) and 0.037 (0.0089) in the RB plots, respectively. The MB-C/TC ratio tended to be lower in the RC plots than in the UC plots while the MB-N/TN ratio showed the opposite trend.

3.4. Discussion

3.4.1. Differences between the Riparian and Upland Positions

Among the soil and vegetation properties measured in this trembling aspen dominated mixedwood stand in the WBF, only plant community composition, gravimetric moisture content in the forest floor in July, EC content in the forest floor in August, September and October, EC content in the mineral soil in August and September, and microbial C/N ratio in the forest floor in October differed significantly between the riparian and upland positions. Sporadic flood events and higher moisture content in the riparian area are the likely causes of the change in plant community composition (Naiman and Decamps 1997; Lamb et al. 2003); the indicator species *Gymnocarpium dryopteris* and *Equisetum* spp. indicate moist conditions while *Heracleum lanatum* suggests disturbance by ephemeral flooding (Johnson et al. 1995; Beckingham et al. 1996). Significantly wetter conditions in the forest floor of the RC plots compared with the UC plots in July may be due to a combination of high rainfall and low air temperature in June (Environment Canada 2004b) that limited evapotranspiration and allowed groundwater recharge. In the Boreal Plain, the combination of low precipitation and high evapotranspiration, the high soil water storage capacity and low transmissivity of Gray Luvisolic soils, and the presence of a thick layer of glacial substrate over bedrock generally allows little runoff or interflow from the upper slope areas to the lower slope areas (Whitson et al. 2004; Devito et al. 2004). The lack of runoff and absence of a fine-textured or frozen soil horizon in the riparian soil profiles suggests that the higher moisture content in the RC plots may be due to a groundwater contribution that varies annually and seasonally in magnitude (Devito et al. 2004). This hypothesis is supported by significant monthly variation in soil moisture content; July and September were significantly wetter than August.

EC content, which is generally used to approximate the water soluble C content, was only higher in the UC plots than in the RC plots in August, September and October (Figure 3.7 A and C). This result may be partly explained by an increase in litterfall inputs in late summer and fall that exacerbated the difference between positions (Huang and Schoenau 1996a). EC content was positively correlated to MB-C content in the forest floor (r=0.627, P=0.001, n=24) which also tended to be higher in the UC plots compared with the RC plots (Figure 3.8). A positive relationship between EC and MB-C contents has previously been reported (e.g. Chang et al. 1995), suggesting that the microbial biomass and microbial metabolites (e.g. hexose- and deoxyribose sugars) may be an important source of potentially soluble C compounds (Kalbitz et al. 2000). However, this relationship may also be an artifact created by the use of KCl to extract organic C; Huang and Schoenau (1996a) demonstrated that K_2SO_4 extraction of organic C resulted in partial lysis of microbial cells, releasing more soluble C into solution. In the mineral soil, MB-C and EC were not related; MB-C was positively correlated to gravimetric moisture content (r=0.556, P=0.005, n=24). This suggests that there is no obvious explanation for the higher EC content in the UC compared with the RC plots.

The microbial C/N ratio may indicate changes in the microbial population structure; the C/N ratio of bacteria is lower (4:1 to 6:1) than that of fungi and mycorrhizae (> 10:1) (Tate 1995). In this study, the median microbial C/N ratio in the forest floor of the control plots from July to October was higher (8.9) than that reported in a trembling aspen-white spruce mixedwood stand in Saskatchewan (~6.0) (Walley et al. 1996). This suggests that the microbial population structures between these two stands may be different. However, the forest floor microbial C/N ratios generally fell within the range (6.4 to 12.4) reported for other boreal forest stands in Quebec and Alaska (Bauhus et al. 1998; Vance and Chapin 2001). The significant reduction in the microbial C/N ratio in the forest floor of the RC plots in October only was caused by the decline in forest floor MB-C content in October while the MB-N content remained constant (Figure 3.8), shown by the positive correlation between the microbial C/N ratio and MB-C content (r=0.515, P=0.010, n=24). There is no ready explanation for this reduction however it does suggest a shift in the microbial population structure in October.

In addition to these significant differences, there were several trends that appeared to be masked by large heterogeneity in soil type; forest floor depth and drainage class showed the greatest variation between RC plots (Table 3.1). Forest floor depth differed by 10 cm between the RC plots of hillslopes 1 and 3 and was positively correlated to total C and N, MB-C, MB-N, EC, and EN contents in the forest floor of the RC plots (data not shown). Studies from large rivers have demonstrated that forest floor thickness in the riparian zone is inversely correlated to frequency of flooding; on the Peace River floodplain in northern Alberta, Schwendenmann (2000) reported a thinner forest floor on the active floodplain compared with terrace soils. This evidence suggests that frequency of flooding disturbance in the riparian position increased in order of hillslope 1 > hillslope 2 > hillslope 3 and that frequent disturbance favoured lower total C and N, MB-C, MB-N, EC, and EN contents in the forest floor. In the mineral soil, total C and N, MB-C, and MB-N contents and the C/N ratio were highest in the RC plot of hillslope 2, which had an elevated water table relative to the other RC plots that resulted in its higher volumetric moisture content and poor drainage (data not shown). A positive relationship between moisture content and total C and N contents and the C/N ratio has been demonstrated in white spruce-trembling aspen stands in Alberta (Offord 1999; Little et al. 2002); moisture content and microbial biomass size have also been shown to be positively correlated (Hannam and Prescott 2003; Bohlen et al. 2001). Together, this information suggests that large longitudinal heterogeneity in the soil types of the riparian position within this trembling aspen dominated mixedwood stand may have masked some biologically significant differences in their soil characteristics compared with the relatively homogeneous soils in the upland position.

3.4.2. Effects of the Experimental Burn

In all plots burning had the greatest impact on soil chemical properties in the forest floor; total C and N, MB-C, MB-N, and EC contents, and MB-N/TN and microbial C/N ratios were significantly lower and pH_{CaCl2} was significantly higher immediately after fire in the burned plots compared with the control plots.

In the control plots, the total forest floor C and N contents fell within reported values (total C: 1.8 to 5.0 kg C m^{-2}; total N: 0.07 to 0.28 kg N m^{-2}) based on data from 80 to 100-year-old pure trembling aspen and mixed trembling aspen-white spruce stands in the WBF (Huang and Schoenau 1996b; Nalder and Wein 1999; Offord 1999; Little et al. 2002). Compared with the control plots, forest floor C in the burned plots was 57 to 78% (1.8 to 2.8 kg C m^{-2}) lower in the upland position and 35 to 61% (0.76 to 1.5 kg C m^{-2}) lower in the riparian position; N was 62 to 76% (0.048 to 0.070 kg N m^{-2}) lower in the upland and 31 to 44% (0.013 to 0.044 kg N m^{-2}) lower in the riparian position. The average reduction in forest floor C and N (C: 58%; N: 53%) in all the plots was similar to that reported one month after fire in a trembling aspen stand in Alaska where total forest floor organic matter and N contents were both 52% (1.8 kg m^{-2} and 0.039 kg N m^{-2}, respectively) lower in the burned compared with the control stands (Dyrness et al. 1989).

The difference in total C and N contents tended to be greater between the UC and UB plots than between the RC and RB plots; this was likely a function of the difference in average percent forest floor consumed in the upland (64%) compared with the riparian (34%) plots (Table 3.4). A greater decline in fuel weight generally results in more gaseous C and N release due to the similarity of the oxidation temperature of N and the destructive distillation and carbonization temperature of organic matter residue (Raison 1979; Raison et al. 1985). In white spruce stands in Alaska, Dyrness et al. (1989) reported a significantly lower total Kjedahl N content and organic matter quantity in the heavily burned compared with the lightly burned stand. Similarly, in a trembling aspen dominated mixedwood stand in Ontario, greater reduction in forest floor depth tended to favour lower forest floor total N content (Johnston and Elliott 1998). Proportionate loss of C and N after burning suggests little change in the C/N ratio immediately after fire, which was observed in this study and is supported by fire research in a pine forest in Italy (Dumontet et al. 1996).

The increase in forest floor pH_{CaCl2} after fire was consistent with published fire studies (MacAdam 1987; Dyrness et al. 1989; Hernández et al. 1997; Brais et al. 2000; Simard et al. 2001), appeared to be correlated to the percent forest floor consumed and was greater in the UB plots than in the RB plots (Figure 3.6 C). Fire combustion of litter and soil organic acids and neutralization of H^+ in soil solution by base cations contained in the ash deposited on the surface are the main mechanisms responsible for the post-fire rise in pH (Raison 1979; Fisher and Binkley 2000). Other authors within the circumpolar boreal forest have positively correlated the post-fire reduction in forest floor mass to the post-fire increase in forest floor pH (Dyrness et al. 1989; Brais et al. 2000), however, this is the first study to report the same result for trembling aspen dominated mixedwood stands.

In this study, the median MB-C and MB-N contents in the forest floor of the control plots were slightly higher and lower, respectively, than the median contents reported for MB-C (58 g m^{-2}) and MB-N (10 g m^{-2}) in the surface 15 cm soil of a trembling aspen-white spruce stand in Saskatchewan (Walley et al. 1996) but fell within the range of MB-C and MB-N contents reported in boreal forest stands from Quebec and Alaska (Bauhus et al. 1998; Vance and Chapin 2001). After fire, MB-C and MB-N contents were 84 to 92% (68 to 82 g C m^{-2}) and 86 to 92% (7.2 to 8.4 g N m^{-2}) lower, respectively, in the UB plots than in the UC plots and were 58 to 83% (28 to 71 g C m^{-2}) and 63 to 87% (4.1 to 10.3 g N m^{-2}) lower, respectively, in the RB plots compared with the RC plots. Large variation in the reduction in MB-C and MB-N contents in the RB plots is partly explained by the difference between plots in the percent of forest floor consumed; consumption was greatest in the RB plot of hillslope 1 which also showed the most reduction in MB-C and MB-N contents among the riparian plots. No recovery of MB-C or MB-N contents was observed within 4 months after burning in any of the burned plots. Other studies have observed that MB-C and MB-N contents may remain lower in burned areas for more than 11 years after fire (Pietikäinen and Fritze 1993; Fritze et al. 1993; Dumontet et al. 1996; Prieto-Fernández et al. 1998), however, Walley et al. (1996) reported that MB-C and MB-N contents were higher in a trembling aspen-white spruce stand that had been burned 4 years earlier compared with the control stand.

A decline in the forest floor and mineral soil microbial C/N ratio of the burned plots from August to October suggests a change in the microbial population structure. Dumontet et al. (1996) also observed a decline in the microbial C/N ratio one month after fire in a Mediterranean pine ecosystem. Fungi are less heat tolerant than bacteria (Dunn et al. 1985) and were more affected by fire than bacteria in coniferous forests in Finland

(Pietikäinen and Fritze 1993, 1995). Therefore, the reduced microbial C/N ratio may suggest a post-fire decrease in the proportion of fungi in the microbial population.

The decline in microbial biomass after burning has been attributed to heat sterilization, lower moisture content, the alteration of soil organic matter to inhibit microbial activity, and reduced available C supply (Fritze et al. 1994; Pietikäinen and Fritze 1995; Neary et al. 1999; Choromanska and DeLuca 2001; Choromanska and DeLuca 2002). Heat-induced soil sterilization is supported by the immediate decline in MB-C and MB-N contents after burning that tended to be larger in the UB plots and RB plot of hillslope 1 compared with the RB plots of hillslopes 2 and 3. This suggests that the decline in microbial biomass size is positively related to the percent of forest floor consumed. In this study, there was no indication that post-fire microbial recovery was aided by slightly higher moisture content in the riparian position, as suggested in laboratory studies conducted by Dunn et al. (1979) and Choromanska and DeLuca (2002). However, altered humus quality as indicated by the lower MB-C/TC and MB-N/TN in the burned plots compared with the controls may have limited microbial recovery in both the riparian and upland positions (Bauhus et al. 1998; Hernández et al. 1997). Reduced supply of available C compounds is suggested by the decline in forest floor EC content in August and onwards; the initial increase in EC content has been previously observed (Pietikäinen and Fritze 1993; Prieto-Fernández et al. 1998) and is attributed to carbohydrate release by lysed microbes and heat-induced organic matter changes (Prieto-Fernández et al. 1998). I did not observe changes in the rate of microbial respiration (qCO_2) or perform analyses that allowed direct conclusions about microbial community composition to be made. However, other studies suggest that after fire microbial respiration per unit biomass initially increases due to a change in the microbial community from slow growing K-strategists (microbes that use resources efficiently within a highly competitive microbial population) to microbes with a high growth rate termed r-strategists (Fritze et al. 1994; Pietikäinen and Fritze 1995; Wüthrich et al. 2002) but then declines due to the reduction in the availability of C (Choromanska and DeLuca 2001). Further investigation into the organic matter structure to examine its change after fire in the WBF and its relationship to microbial properties is needed.

In the mineral soil, temperature was the only factor other than the microbial C/N ratio that was significantly affected by the burn treatment. Mean daily soil temperature in the mineral soil at 5 cm depth was significantly higher in the UB plots from July 6 to September 10 and in the RB plots from July to October compared with their respective control plots. Frey et al. (2003) also reported a significantly higher mean summer temperature at 20 cm mineral soil depth in burned clearcut and partially cut white spruce dominated mixedwood stands in Alberta. Higher soil temperature immediately after fire is attributed to both the consumption of insulating understory vegetation and forest floor, and the deposition of incompletely combusted, black organic matter and charcoal (Ahlgren and Ahlgren 1960; Raison 1979; Neary et al. 1999; Fisher and Binkley 2000).

A greater percent of forest floor consumption in the UB compared with the RB plots may have created greater soil temperature dependence on air temperature in the UB plots. Greater response of daily soil temperature to changes in air temperature was also reported after clear-cutting in a *Picea abies* stand (Kahkonen et al. 2002). This explanation is confirmed by the larger change in temperature of the UB and UC plots (1.7°C) compared with the RB and RC plots (0.8°C) from July to mid-August combined with the lower soil temperature in the UB plots than the UC plots after September 10.

In this study, it was difficult to achieve a similar depth of forest floor combustion on all plots. After oven drying, the forest floor in the riparian position did not combust as easily as the forest floor of the upland position (personal observation, July 2003). This trend varied between hillslopes; the thin, dry forest floor in the riparian plot of hillslope 1 burned easily and quickly compared with the riparian plots in hillslopes 2 and 3. Greater difficulty in burning the organic layer in the riparian position of hillslopes 2 and 3 may be due to upward capillary movement of water from the underlying mineral soil (Samran et al. 1995) or due to insufficient depth of dried material to propagate smoldering combustion (Miyanishi and Johnson 2002). This evidence suggests that in this trembling aspen dominated mixedwood stand, the forest floor of the riparian plots in hillslopes 2 and 3 would be unlikely to burn without either substantial energy generated by flaming combustion or a prolonged drought period. In contrast, the thin forest floor of the riparian plot on hillslope 1 could dry quickly and burn more readily compared with the other riparian plots. The difference in fire behaviour that would be expected between riparian plots thus suggests that fire severity would not be uniformly lower in an ephemeral or intermittent riparian valley compared to the upland area. This heterogeneity in fire behaviour between riparian plots may explain the lack of significant position-burn interactions in this study.

3.5. Conclusions

The main hypothesis of this study was that, within an ephemeral riparian valley located in a trembling aspen-dominated mixedwood stand in the WBF, a higher moisture content in the riparian, lower slope position as compared to the upper slope position would influence soil properties in both undisturbed and burned treatments. The shift in plant community composition and trends in gravimetric and volumetric moisture content demonstrated that the riparian position of the ephemeral riparian valley studied was wetter compared with the upper slope position. However, few significant differences were found in soil properties between the RC and UC plots.

In the forest floor, total C and N, MB-C, MB-N and EC contents and MB-N/TN and microbial C/N ratios were significantly lower while forest floor pH_{CaCl2} and mineral soil temperature were significantly higher in the burned treatment as compared with the unburned treatment. Similar post-fire trends have been reported in other ecosystems but this is the first study to report the short-term effects of an experimental burn conducted within an intact forest canopy on soil properties in a trembling aspen-dominated mixedwood stand in the WBF. A tendency towards a greater increase in pH_{CaCl2} and soil temperature and a larger reduction in total C and N, MB-C and MB-N in the UB compared with the RB plots suggested greater fire severity in the burned plots of the upland position; this trend was consistent with the depth of burn data. It is possible that some of the preparation required to burn the plots (e.g. vegetation clipping, forest floor drying) combined with large heterogeneity in forest floor consumption within the RB plots may have masked any significant position-burn interaction effects for the measured soil properties.

3.6. References

Ahlgren, I.F. and Alhgren, C.E. 1960. Ecological effects of forest fires. Bot. Rev. **26**: 483-533.

Alberta Energy and Natural Resources. 1975. Ecological Land Classification and Evaluation: Brazeau Reservoir Area. Land Use Study Special Report.

Alberta Sustainable Resource Development. 2001. Chisholm fire reports [online]. Available from http://www3.gov.ab.ca/srd/forests/chisholm [accessed January 20 2003].

Almendros, G., González-Vila, F.J. and Martin, F. 1990. Fire-induced transformation of soil organic matter from an oak forest. An experimental approach to the effects of fire on humic substances. Soil Sci. **149**: 158-168.

Andison, D.W. and McCleary, K. 2002. Disturbance in Riparian Zones on Foothills and Mountain Landscapes of Alberta. Alberta Foothills Disturbance Ecology Research Series Report Number 3. Bandaloop Landscape-Ecosystem Services, Belcarra.

Bauhus, J., Paré, D. and Côté, L. 1998. Effects of tree species, stand age and soil type on soil microbial biomass and its activity in a southern boreal forest. Soil Biol. Biochem. **30**: 1077-1089.

Beckingham, J.D., Corns, I.G.W. and Archibald, J.H. 1996. Field guide to ecosites of west-central Alberta. Canadian Forest Service Special Report 9. Northern Forestry Centre, Edmonton, AB.

Bhatti, J.S., Apps, M.J. and Jiang, H. 2002. Influence of nutrients, disturbances and site conditions on carbon stocks along a boreal forest transect in central Canada. Plant Soil. **242**: 1-14.

Bohlen, P.J., Groffman, P.M., Driscoll, C.T., Fahey, T.J., and Siccama, T.G. 2001. Plant-soil-microbial interactions in a northern hardwood forest. Ecol. **82**: 965-978.

Brais, S., Paré, D. and Ouimet, R. 2000. Impacts of wild fire severity and salvage harvesting on the nutrient balance of jack pine and black spruce boreal stands. For. Ecol. Manage. **137**: 231-243.

Brookes, P.C., Landman, A., Pruden, G. and Jenkinson, D.S. 1985. Chloroform fumigation and the release of soil nitrogen: a rapid direct extraction method to measure microbial biomass nitrogen in soil. Soil Biol. Biochem. **17**: 837-842.

Campbell Scientific, Inc. 2002. CS616 Water Content Reflectometer Instruction Manual. Campbell Scientific Canada Corp., Edmonton, AB.

Carmosini, N., Devito, K.J. and Prepas, E.E. 2003. Net nitrogen mineralization and nitrification in trembling aspen forest soils on the Boreal Plain. Can. J. For. Res. **33**: 2262-2268.

Chanasyk, D.S., Whitson, I.R., Mapfumo, E., Burke, J.M. and Prepas, E.E. 2003. The impacts of forest harvest and wildfire on soils and hydrology in temperate forests: a baseline to develop hypotheses for the Boreal Plain. J. Environ. Eng. Sci. **2**: S51-S62.

Chang, S.X., Preston, C.M. and Weetman, G.F. 1995. Soil microbial biomass and microbial and mineralizable N in a clear-cut chronosequence on northern Vancouver Island, British Columbia. Can. J. For. Res. **25**: 1595-1607.

Choromanska, U. and DeLuca, T.H. 2001. Prescribed fire alters the impact of wildfire on soil biochemical properties in a Ponderosa Pine forest. Soil Sci. Soc. Am. J. **65**: 232-238.

Choromanska, U. and DeLuca, T.H. 2002. Microbial activity and nitrogen mineralization in forest mineral soils following heating: evaluation of post-fire effects. Soil Biol. Biochem. **34**: 263-271.

Cumming, S.G. 2001. Forest type and wildfire in the Alberta boreal mixedwood: what do fires burn? Ecol. Appl. **11**: 97-110.

Devito, K.J., Creed, I.F. and Fraser, C.J.D. 2004. Controls on runoff from a partially harvested aspen forested headwater catchment, Boreal Plain, Canada. Hydrol. Proc. In Press.

Dufrêne, M. and Legendre, P. 1997. Species assemblages and indicator species: the need for a flexible asymmetrical approach. Ecol. Monogr. **67**: 345-366.

Dumontet, S., Dinel, H., Scopa, A., Mazzatura, A. and Saracino, A. 1996. Post-fire soil microbial biomass and nutrient content of a pine forest from a dunal Mediterranean environment. Soil Biol. Biochem. **28**: 1467-1475.

Dunn, P.H., DeBano, L.F. and Eberlein, G.E. 1979. Effects of burning on chaparral soils: II. Soil microbes and nitrogen mineralization. Soil Sci. Soc. Am. J. **43**: 509-514.

Dunn, P.H., Barro, S.C. and Poth, M. 1985. Soil moisture affects survival of microorganisms in heated chaparral soil. Soil Biol. Biochem. **17**: 143-148.

Dwire, K.A. and Kauffman, J.B. 2003. Fire and riparian ecosystems in landscape of the western USA. For. Ecol. Manage. **128**: 61-74.

Dyrness, C.T., Van Cleve, K. and Levison, J.D. 1989. The effect of wildfire on soil chemistry in four forest types in interior Alaska. Can. J. For. Res. **19**: 1389-1396.

Ecological Stratification Working Group. 1996. A national ecological framework for Canada. Centre for Land and Biological Resources Research, Agriculture and Agri-Food Canada and State of Environment Directorate, Environment Canada, Ottawa, ON. 125 pp.

Environment Canada. 2004a. Canadian Climate Normals or Averages 1971-2000 [online]. Available from http://www.climate.weatheroffice.ec.gc.ca/climate_normals/index_e.html [accessed April 23 2004].

Environment Canada. 2004b. Climate Data [online]. Available from http://www.climate.weatheroffice.ec.gc.ca/climateData/canada_e.html [accessed April 23 2004].

Environment Canada. 2004c. Archived Hydrometric Data: Pembina Creek below Paddy River [online]. Available from http://www.msc-smc.ec.gc.ca/wsc/hydat/H2O/index_e.cfm?cname=graph.cfm [accessed May 7 2004].

Fisher, R.F. and Binkley, D. 2000. Ecology and Management of Forest Soils. John Wiley & Sons Inc., New York.

Frey, B.R., Lieffers, V.J., Munson, A.D. and Blenis, P.V. 2003. The influence of partial harvesting and forest floor disturbance on nutrient availability and understory vegetation in boreal mixedwoods. Can. J. For. Res. **33**: 1180-1188.

Fritze, H., Pennanen, T. and Pietikäinen, J. 1993. Recovery of soil microbial biomass and activity from prescribed burning. Can. J. For. Res. **23**: 1286-1290.

Fritze, H., Smolander, A., Levula, T., Kitunen, V. and Malkonen, E. 1994. Wood ash fertilization and fire treatments in a Scots-pine forest stand: effects on the organic layer, microbial biomass, and microbial activity. Biol. Fertil. Soils. **17**: 57-63.

Gee, G.W., and Or, D. 2002. Particle-size analysis. *In* Methods of soil analysis. Part 4: Physical methods. *Edited by* J.H. Dane and G.C. Topp. Soil Science Society of America, Inc., Madison, WI. pp. 255-294.

Hannam, K.D. and Prescott, C.E. 2003. Soluble organic nitrogen in forests and adjacent clearcuts in British Columbia, Canada. Can. J. For. Res. **33**: 1709-1718.

Hernández, T., García, C. and Reinhardt, I. 1997. Short-term effect of wildfire on the chemical, biochemical and microbiological properties of Mediterranean pine forest soils. Biol. Fertil. Soils. **25**: 109-116.

Horwath, W.R. and Paul, E.A. 1994. Microbial Biomass. *In* Methods of Soil Analysis. Part 2. Microbiological and Biochemical Properties. *Edited by* R.W. Weaver. Soil Science Society of America, Inc., Madison, WI. pp. 753-774.

Hossain, A.K.M.A., Raison, R.J. and Khanna, P.K. 1995. Effects of fertilizer application and fire regime on soil microbial biomass carbon and nitrogen, and nitrogen mineralization in an Australian subalpine eucalypt forest. Biol. Fertil. Soils. **19**: 246-252.

Huang, W.Z. and Schoenau, J.J. 1996a. Distribution of water-soluble organic carbon in an aspen forest soil. Can. J. For. Res. **26**: 1266-1272.

Huang, W.Z. and Schoenau, J.J. 1996b. Forms, amounts and distribution of carbon, nitrogen, phosphorus and sulphur in a boreal aspen forest soil. Can. J. Soil Sci. **76**: 373-385.

Johnson, E.A. 1992. Fire and Vegetation Dynamics: Studies from the North American Boreal Forest. Cambridge University Press, Cambridge, UK.

Johnson, D., Kershaw, L., MacKinnon, A. and Pojar, J. 1995. Plants of the western boreal forest and aspen parkland. Lone Pine Publishing, Edmonton, AB.

Johnston, M. and Elliott, J. 1998. The effect of fire severity on ash, and plant and soil nutrient levels following experimental burning in a boreal mixedwood stand. Can. J. Soil Sci. **78**: 35-44.

Kahkonen, M.A., Wittman, C., Ilvesniemi, H., Westman, C.J. and Salkinoja-Salonen, M.S. 2002. Mineralization of detritus and oxidation of methane in acid boreal coniferous forest soils: seasonal and vertical distribution and effects of clear cut. Soil Biol. Biochem. **34**: 1191-1200.

Kalbitz, K., Solinger, S., Park, J.H., Michalzik, B. and Matzner, E. 2000. Controls on the dynamics of dissolved organic matter in soils: a review. Soil Sci. **165**: 277-304.

Kalra, Y.P. and Maynard, D.G. 1991. Methods manual for forest soil and plant analysis. Information Report NOR-X-319. Canadian Forest Service, Northern Forestry Centre, Edmonton, AB.

Lamb, E.G., Mallik, A.U. and Mackereth, R. 2003. The early impact of adjacent clear-cutting and fire on riparian zone vegetation in Northwestern Ontario. For. Ecol. Manage. **177**: 529-538.

Little, T.I., Pluth, D.J., Corns, I.G.W. and Gilmore, D.W. 2002. Post-fire forest floor development along toposequences of white spruce-trembling aspen mixedwood communities in west-central Alberta. Can. J. For. Res. **32**: 892-902.

MacAdam, A.M. 1987. Effects of broadcast slash burning on fuels and soil chemical properties in the Sub-boreal Spruce Zone of central British Columbia. Can. J. For. Res. **17**: 1577-1584.

Macdonald, E., Burgess, C.J., Scrimgeour, G.J., Boutin, S., Reedyk, S. and Kotak, B. 2004. Should riparian buffers be part of forest management based on natural disturbance? For. Ecol. Manage. **187**: 185-196.

MacLean, D.A., Woodley, S.J., Weber, M.G. and Wein, R.W. 1983. Fire and nutrient cycling. *In* The Role of Fire in Northern Circumpolar Ecosystems. *Edited by* R.W. Wein and D.A. MacLean. John Wiley & Sons Ltd., New York. pp. 111-132.

McCune, B. and Grace, J.B. 2002. Analysis of ecological communities. MjM Software Design, Gleneden Beach, Oregon, USA.

McCune, B., and Mefford, M.J. 1999. PC-ORD: Multivariate Analysis of Ecological Data. MjM Software Design, Gleneden Beach, Oregon, USA.

Miyanishi, K. and Johnson, E.A. 2002. Processes and patterns of duff consumption in the mixedwood boreal forest. Can. J. For. Res. **32**: 1285-1295.

Naiman, R.J. and Decamps, H. 1997. The ecology of interfaces: riparian zones. Annu. Rev. Ecol. Syst. **28**: 621-658.

Nalder, I.A. and Wein, R.W. 1999. Long-term forest floor carbon dynamics after fire in upland boreal forests of western Canada. Global Biogeochem. Cycles. **13**: 951-968.

Neary, D.G., Klopatek, C.C., DeBano, L.F. and Ffolliott, P.F. 1999. Fire effects on belowground sustainability: a review and synthesis. For. Ecol. Manage. **122**: 51-71.

Nykvist, N. and Skyllberg, U. 1989. Spatial variation of pH in the mor layer of coniferous stands. Scand. J. For. Res. **4**: 3-11.

Offord, J.L. 1999. Nitrogen Mineralization Patterns of Boreal Forest Soils. M.Sc. Thesis. Department of Renewable Resources, University of Alberta, Edmonton, AB.

Onset Computer Corporation. 2002. HOBO ® H8 Family of Data Loggers [online]. Available from http://www.onsetcomp.com/Products/Product_Pages/HOBO_H08/H08family.html [accessed March 1 2004].

Peters, T.W. 1981. Reconnaissance soil survey of the Brazeau Dam area. Alberta Soil Survey Report No. 40. Alberta Institute of Pedology Report S-81-40. Pedology Consultants and Agriculture Canada. Land Resources Research Institute, Agriculture Canada, Ottawa, ON.

Pietikäinen, J. and Fritze, H. 1993. Microbial biomass and activity in the humus layer following burning: short-term effects of two different fires. Can. J. For. Res. **23**: 1275-1285.

Pietikäinen, J. and Fritze, H. 1995. Clear-cutting and prescribed burning in coniferous forest: comparison of effects of soil fungal and total microbial biomass, respiration activity and nitrification. Soil Biol. Biochem. **27**: 101-109.

Prieto-Fernández, A., Acea, M.J. and Carballas, T. 1998. Soil microbial and extractable C and N after wildfire. Biol. Fertil. Soils. **27**: 132-142.

Raison, R.J. 1979. Modification of the soil environment by vegetation fires, with particular reference to nitrogen transformations: a review. Plant Soil. **51**: 73-108.

Raison, R.J., Khanna, P.K. and Woods, P.V. 1985. Mechanisms of element transfer to the atmosphere during vegetation fires. Can. J. For. Res. **15**: 132-140.

Samran, S., Woodard, P.M. and Rothwell, R.L. 1995. The effect of soil water on ground fuel availability. For. Sci. **41**: 255-267.

Schwendenmann, L. 2000. Soil properties of boreal riparian plant communities in relation to natural succession and clear-cutting, Peace River Lowlands, Wood Buffalo National Park, Canada. Water Air Soil Pollut. **122**: 449-467.

Shaw, R.D., Trimbee, A.M., Minty, A., Fricker, H. and Prepas, E.E. 1989. Atmospheric deposition of phosphorus and nitrogen in central Alberta with emphasis on Narrow Lake. Water Air Soil Pollut. **43**: 119-134.

Simard, D.G., Fyles, J.W., Paré, D. and Nguyen, T. 2001. Impacts of clearcut harvesting and wildfire on soil nutrient status in the Quebec boreal forest. Can. J. Soil Sci. **81**: 229-237.

Simonson, G.H. and Boersma, L. 1972. Soil morphology and water table fluctuations: II. Correlation between annual water table fluctuations and profile features. Soil Sci. Soc. Am. Proc. **36**: 649-653.

Soil Classification Working Group. 1998. The Canadian System of Soil Classification. Agriculture and Agri-Food Canada Publication 1646. NRC Research Press, Ottawa, ON.

Tate, R.L. III. 1995. Soil Microbiology. John Wiley & Sons, Inc., New York.

The National Atlas of Canada. 1974. Average annual potential evapotranspiration. 4th Edition [online]. Available from http://atlas.gc.ca/site/english/maps/archives/4thedition/environment/climate/049_50?l=3&r=0&c=0#map [accessed July 6 2004].

Vance, E.D., Brookes, P.C. and Jenkinson D.S. 1987. An extraction method for measuring soil microbial biomass C. Soil Biol. Biochem. **19**: 703-707.

Vance, E.D. and Chapin, F.S. III. 2001. Substrate limitations to microbial activity in taiga forest floors. Soil Biol. Biochem. **33**: 173-188.

Walley, F.L., Van Kessel, C. and Pennock, D.J. 1996. Landscape scale variability in N mineralization in forest soils. Soil Biol. Biochem. **28**: 383-391.

Wüthrich, C., Schaub, D., Weber, M., Marxer, P. and Conedera, M. 2002. Soil respiration and soil microbial biomass after fire in a sweet chestnut forest in southern Switzerland. Catena. **48**: 201-215.

Whitson, I.R., Chanasyk, D.S. and Prepas, E.E. 2004. Patterns of water movement on a logged Gray Luvisolic hillslope during the snowmelt period. Can. J. Soil Sci. **84**: 71-82.

4. SHORT-TERM EFFECTS OF EXPERIMENTAL BURNING ON NITROGEN DYNAMICS AND THEIR RELATION TO SLOPE POSITION

4.1. Introduction

In the Western Boreal Forest (WBF), fire is an important mechanism of natural disturbance (Johnson 1992). Fire significantly influences short-term (< 1 year) post-fire trends in the nitrogen (N) cycle through oxidation of N stored in organic matter, volatilization of NO_3^- and amino acids, deposition of ash, and alteration of soil microbiological, chemical and physical properties that affect N transformations (Raison 1979; MacLean et al. 1983; Neary et al. 1999). However, in the WBF, few have examined the short-term effect of fire on soil N cycling, focusing instead on mid-term (1-15 years) fire effects (Walley et al. 1996) or N dynamics in mature (Fyles and McGill 1987; Huang and Schoenau 1997; Offord 1999; Little et al. 2002) or logged stands (Carmosini et al. 2002; Frey et al. 2003; Carmosini et al. 2003). In addition, limited research has been conducted on short-term post-fire N dynamics after low severity, understory fires even though these fires are an important component of the WBF's fire regime.

The effect of an understory fire on the soil N cycle largely depends on heating intensity and duration (e.g. fire severity) that determines the degree of forest floor and vegetation combustion, loss in forest floor mass, and penetration of heat energy into the surface organic and mineral soil (Raison 1979; Raison et al. 1985; Neary et al. 1999). Several authors have reported an immediate post-fire increase of variable size and duration in the NH_4^+ content of the surface soil (Khanna and Raison 1986; Giovannini et al. 1990; Fenn et al. 1993; Hernández et al. 1997; Monleon et al. 1997; DeLuca and Zouhar 2000; Grogan et al. 2000; Nardoto and Bustamante 2003). Previous studies suggest that this increase in NH_4^+ content will be greatest in low temperature fires on moist soils (Raison 1979; DeBano et al. 1979; Romanyà et al. 2001; Andersson et al. 2004), suggesting that fire severity affects post-fire NH_4^+ content. After this initial post-fire pulse of inorganic N subsides, mineralization and nitrification rates determine N availability for plant uptake and growth. In general, net N mineralization and nitrification rates are thought to increase for a short period after fire and then decline

(Prieto-Fernández et al. 1993; Monleon et al. 1997; DeLuca and Zouhar 2000; Giardina and Rhoades 2001; Romanyà et al. 2001; Nardoto and Bustamante 2003). Under conditions that promote greater oxidation and volatilization of N stored in the surface organic layer, substrate-limitation may favour microbial immobilization and reduce the net N mineralization rate compared with areas of little forest floor combustion (Romanyà et al. 2001; Choromanska and DeLuca 2002). This suggests that N may be more limiting for plant growth after an understory fire that results in extensive forest floor oxidation; this hypothesis merits further investigation. Gross rates of mineralization, NH_4^+ and NO_3^- consumption, and nitrification may also provide clues on the effect of the degree of forest floor combustion on short-term post-fire N transformations; few have investigated fire's effect on these parameters.

In the WBF, intermittent or ephemeral headwater streams comprise a large proportion of the surface drainage network; their adjacent riparian zones may be important in regulating stream N concentrations (Cirmo and McDonnell 1997). Minimal N inputs into forested ephemeral riparian zones from flooding, groundwater, and surface runoff suggest that mineralization and nitrification processes control N availability in such areas. Research suggests that fluctuating aerobic and anaerobic conditions characteristic of ephemeral riparian zones may favour higher rates of N mineralization compared with drier slope positions in the same stand (Pinay et al. 2002). In the WBF, a negative relationship between moisture content as related to slope position and specific net and net N mineralization rates has been demonstrated in various stand types (Walley et al. 1996; Offord 1999; Little et al. 2002); it is possible that this trend may differ in a lower slope, riparian zone.

In addition, the interaction between differences in moisture content due to slope position and fire requires further study. An increase in moisture content from the upper slope to the lower slope riparian soils could reduce forest floor combustion in the riparian position of an intermittent or ephemeral riparian valley in the WBF (Miyanishi and Johnson 2002; Dwire and Kauffman 2003). This implies that after an understory fire along a slope, N availability may be higher in the lower slope, riparian zone due to a greater pulse of NH_4^+ and relatively higher N mineralization rates.

This paper reports on a study designed to investigate the effect of high moisture content in the lower slope, riparian position and a simulated understory fire on N transformations. The research was conducted in an ephemeral riparian valley of a trembling aspen dominated mixedwood stand in the WBF; parameters used to indicate differences in N cycling included foliar % N, total aboveground plant N content, soil NH_4^+ and NO_3^- contents, NO_3-N/NH_4-N and inorganic N/extractable N ratios, net N mineralization and nitrification rates, and gross N mineralization, NH_4^+ consumption, nitrification and NO_3^- consumption rates in the forest floor and surface 10 cm of mineral soil.

4.2. Materials and Methods

4.2.1. Study Site

The study was conducted from June 2003 to April 2004 in a 90-year-old trembling aspen (*Populus tremuloides* Michx.) dominated mixedwood stand (53°5'N, 115°21'W; 914 m a.s.l.) that is approximately 10 km southwest of Lodgepole, Alberta (Figure 3.1). This site is located within the Lower Foothills Natural Subregion (Beckingham et al. 1996) of the Boreal Plain Ecozone (Ecological Stratification Working Group 1996) and the Cynthia Upland Land District of the Western Alberta Plain Physiographic Region (Peters 1981). Relief in the study site varies from 10 to 50 metres and the area has 1.5 to 20 metres of clay loam till underlain by undulating sandstone, siltstone, and mudstone of the Paskapoo (Paleocene) formation (Alberta Energy and Natural Resources 1975; Peters 1981).

Regional climate of the study area is continental with short, cool summers and cold winters. There is little long-term climate data available for this area; Rocky Mountain House (52°25'N, 114°55'W, 988 m a.s.l.), Alberta, is the nearest full-time weather station to the study site. From 1971 to 2000 annual precipitation averaged 535 mm (123 mm or 23% as snow) and mean annual temperature averaged 2.3°C at Rocky Mountain House (Environment Canada 2004a). In January, temperature averaged -11°C while in July average temperature was 15°C; 49% of annual precipitation occurred as rain from June to August (Environment Canada 2004a). In Drayton Valley (53°13' N, 114°57'W, 883 m a.s.l.), Alberta, a weather station closer to the study site that has been collecting climate data since 2000, average air temperature was 15°C and total rainfall was 263.5 mm from June to October 2003 during this study (Environment Canada 2004b). In this region, average annual potential evapotranspiration is 508 mm (The National Atlas of Canada 1974). No runoff data exists for the study site. However, mean annual runoff at Pembina River below Paddy Creek (53°7'47''N 115°19'30''W) from 1956 to 2003 approximated 166 mm yr^{-1} (Environment Canada 2004c).

The study site is located within a 97 hectare watershed. A second order ephemeral stream with a vegetated channel less than 1 metre wide and poorly defined banks dissects the mixedwood stand to the north. A first order ephemeral draw with poorly defined banks and a vegetated channel less than 1 metre wide occurs in a valley that dissects the stand to the west.

The Hubalta soil series characterizes the area; soils are moderately well drained Orthic Gray Luvisols on the upper and mid slopes and imperfect to very poorly drained Gleyed Gray Luvisols in the depressions (Peters 1981). Soils at the study site are Orthic Gray Luvisols on the upper slopes and range from Gleyed Black Chernozems and Gray Luvisols to Orthic Gleysols in the lower slope riparian areas (soil classification according to Soil Classification Working Group (1998)). No total N deposition data exists for the study site. The closest N deposition data available is from Narrow Lake (54°35'N, 113°37'W) within the Boreal Plain Ecozone; average deposition was 4.24 kg N ha^{-1} yr^{-1} from 1983 to 1986 (Shaw et al. 1989).

The study site includes two distinct plant communities: one characteristic of the mid and upper slopes, and another characteristic of the lower slope, riparian area. Trembling aspen, balsam fir (*Abies balsamea*), and white spruce (*Picea glauca*) are the dominant tree species on the upper slope position; the understory is dominated by bracted honeysuckle (*Lonicera involucrata*), wild red raspberry (*Rubus idaeus*), wild sarsasparilla (*Aralia nudicaulis*), bunchberry (*Cornus canadensis*), tall lungwort (*Mertensia paniculata*), and arrow-leaved coltsfoot (*Petasites sagittatus*). The plant community in the upper slope position can be categorized as Lower Foothills Ecophase f3, bracted honeysuckle Aw-Sw-Pl (Beckingham et al. 1996). In the lower slope riparian position white spruce, balsam fir, trembling aspen, and white birch are the main tree species;

bracted honeysuckle, river alder (*Alnus rugosa*), cow parsnip (*Heracleum lanatum*), horsetail (*Equisetum* spp.), and oak fern (*Gymnocarpium dryopteris*) predominate as the understory species. The plant community in the riparian, lower-slope position corresponds to Lower Foothills Ecophase i2, horsetail Pb-Sw (Beckingham et al. 1996). All plant species names are according to Johnson et al. (1995).

4.2.2. Sampling Design

The study site was selected based on a lack of stand-replacing fire disturbance within the last 60 years; a moderate slope gradient (15-30%) within the stand; the presence of a forested riparian valley with a poorly defined channel and intermittent or ephemeral stream flow; and an obvious increase in soil moisture content from the upland to the riparian position. The riparian position was defined as the toeslope area adjacent to the stream channel; the upland position was defined as the water-shedding upper slope position below the crest.

The study site was divided into three north to northwest facing hillslopes varying in length from 50 to 90 metres and separated laterally by 15 to 200 metres. Each hillslope was identified as a block and stratified into upland and riparian positions. In each block, two adjacent 2 x 2 m plots were established in canopy gaps of each slope position. One of the 2 plots was randomly assigned a burn treatment; the other plot was not disturbed and used as a control (reference) plot. Therefore, the study included 4 plot types: upland control (UC), upland burn (UB), riparian control (RC), and riparian burn (RB), in a 2 x 2 experiment. The sampling design is shown in Figure 3.2.

The burn treatment involved some plot preparation before burning. In mid-June the burn plots were prepared for prescribed burn by clipping the aboveground vegetation and leaving it in place, digging perimeter trenches to mineral soil, severing all root connections, and removing flammable vegetation around the plot. All the plots were covered with two layers of thin plastic sheet for 10 days to dry the forest floor but the material was still too wet to burn. Therefore, the forest floor was removed from the burn treatment plots to approximately 5 cm depth, dried in laboratory ovens at 68°C for 48 hours, and returned to the same plot for burning. The burn was conducted on 4 July 2003 using propane torches (see Figure 3.3) and was extinguished with water; it rained later that evening. After the burn, black ash and charred woody debris covered the surface of the burned plots. The mean dry weight of forest floor replaced on site for burning and depth of burn is shown for the UB and RB plots in Table 3.4.

4.2.3. Field and Laboratory Measurements

Soil samples were collected for analysis from 5 June to 1 October 2003 at 28 to 30 day intervals; the last set of incubated samples was retrieved from the site on 16 April 2004. On each sampling date, a stainless steel corer (10 cm long, 5 cm diameter) was used to collect 3 cores each of the LFH (hereafter referred to as the forest floor) and 0-10 cm mineral soil (hereafter referred to as the mineral soil) in each plot. The cores were composited by horizon for each plot, transported to the laboratory, and processed within 48 hours.

In situ net N mineralization and nitrification rates were measured using the buried bag technique (Eno 1960). Net rates were measured from 5 July 2003 to 1 October 2003 with consecutive 28 to 30-day incubations and from 1 October 2003 to 16 April 2003 with a single incubation. On each sampling date, a stainless steel corer (10 cm long, 5 cm diameter) was used to collect 3 pairs of cores of the forest floor and 3 pairs of cores of mineral soil (0-10 cm) in each study plot. One core of the pair was placed in a tightly sealed polyethylene bag and returned to its original horizon position. The remaining core was composited with the other two unincubated cores of the same horizon (forest floor or mineral soil), transported to the laboratory, and processed within 48 hours. Incubated samples of each horizon from each plot were collected at the start of the next incubation, transported to the laboratory, composited by depth, and processed within 48 hours.

To determine bulk density, mineral soil was sampled 8 times per plot using a small corer (31.42 cm^3) and dried at 105°C for 24 hours; bulk density varied between plots from 0.6 to 1.2 g cm^{-3}. Average forest floor bulk density was estimated as 0.17 g cm^{-3} based on data in the literature (Carmosini et al. 2003; Offord 1999).

Forest floor and mineral soil samples were homogenized by sieving to 4 mm. Gravimetric moisture content was determined by drying two 5 g subsamples at 68°C for 48 hours for forest floor or at 105°C for 24 hours for mineral soil (Kalra and Maynard 1991). Inorganic N (NH_4^+ and NO_3^-) and extractable C (EC) and N (EN) were extracted by shaking approximately 5 g fresh forest floor and 10 g fresh mineral soil in 50 mL 2 M KCl for 1 hour (Kalra and Maynard 1991). Extracts were filtered gravimetrically through prewashed Fisherbrand® Q2 filter paper and immediately frozen at -18°C until analysis. In the extracts, NH_4^+ concentration was determined by the indophenol blue method (Keeney and Nelson 1982), NO_3^- concentration by the vanadium (III) chloride method (Doane and Horwath 2003), and EC and EN on a Shimadzu TOC-V CSH/CSN Total Organic Carbon Analyzer (Shimadzu Corporation, Kyoto, Japan). The NO_3-N/NH_4-N ratio was calculated by dividing the NO_3^--N content by the NH_4^+-N content of each plot for each month; the IN/EN ratio was similarly calculated as inorganic N (IN: NH_4^+ + NO_3^-) divided by EN.

Net N mineralization rate (mg N kg^{-1} soil) was estimated as the sum of NH_4^+ and NO_3^- in the composite incubated sample minus that in the preincubation sample. Net nitrification was determined as the difference in NO_3^- concentration between the incubated and initial samples. The change in EC and EN concentrations between incubated and initial samples was also calculated. Soil N and EC concentrations in addition to net N mineralization and net nitrification rates, and the change in EC (hereafter referred to as ΔEC) and EN (hereafter referred to as ΔEN) content after incubation were converted to an areal basis from a per gram basis using the average soil bulk density.

Gross N mineralization, gross NH_4^+ consumption, gross nitrification, and gross NO_3^- consumption rates were determined on samples collected on 4 August 2003 using a non-leaching technique (Zaman and Chang 2004). Four replicate samples of forest floor (5 g dry weight basis) and mineral soil (12.5 g dry weight basis) from each plot were weighed into sterilized plastic containers and incubated at 22°C for 14 days at a constant moisture content. After the incubation period, half of the samples were injected with 1 mL of $(^{15}NH_4)_2SO_4$ (60% enriched) and half were injected with 1 mL of $K^{15}NO_3$ (60% enriched), giving 2 μg ^{15}N g^{-1} soil. This application rate minimized the stimulation of microbial activity but still provided enough ^{15}N to allow the samples to be accurately analyzed on a mass spectrometer (Zaman and Chang 2004). Fifteen minutes after ^{15}N was added, 1 of the 2 injected samples was extracted with 50 mL 0.5 M K_2SO_4 according to the previously described procedure. The remaining sample was extracted with 50 mL

0.5 M K_2SO_4 after a 24 hour incubation. The $(^{15}NH_4)_2SO_4$–enriched extracts were steam distilled with MgO and the released NH_3 was trapped in H_2SO_4 instead of H_3BO_3 to determine NH_4^+(Choi et al. 2003). The $K^{15}NO_3$-enriched extracts were steam distilled with MgO to remove the NH_4^+ (not quantified), followed by another steam distillation with Devarda's Alloy to determine NO_3^-; the distillates were also collected in H_2SO_4. To prevent isotopic cross-contamination between samples, acetic acid and ethanol were used to clean the steam distillation apparatus between each distillation. Standard NaOH solution was used to determine NH_4^+ and NO_3^- concentration by titration (Choi et al. 2003). After adjusting pH to 2 to 3 using 0.1N H_2SO_4, the titrated solution was evaporated to dryness in a laboratory oven (Choi et al. 2003). Dried samples with a low inorganic N content were spiked with 0.05 to 0.3 mg N as $(NH_4)_2SO_4$ depending on the pre-spiking inorganic N content before determination of ^{15}N on a Europa Scientific Tracermass continuous flow isotope ratio mass spectrometer linked to a Roboprep-C/N analyzer (Europa Scientific Ltd., Crewe, UK). The ^{15}N in the sample was then determined using a mass balance equation. Gross rates were calculated according to the equations (Hart et al. 1994):

$$m = \{([NH_4^+]_0 - [NH_4^+]_t)/t\} \text{ x } \{\log(APE_0/APE_t)/\log([NH_4^+]_0/[NH_4^+]_t\}$$

$$cA = m - \{([NH_4^+]_t - [NH_4^+]_0)/t\}$$

where m = gross mineralization rate of the soil (mg kg^{-1} d^{-1}), cA = gross NH_4^+ consumption rate of the soil (mg kg^{-1} d^{-1}), t = time (1 day), APE_0 = atom percent ^{15}N excess of NH_4^+ pool at time 0, APE_t = atom percent ^{15}N excess of NH_4^+ at time t, $[NH_4^+]_0$ = total NH_4^+ concentration (mg kg^{-1}) at time 0, $[NH_4^+]_t$ = total NH_4^+ concentration (mg kg^{-1}) at time t. Gross rates of nitrification and NO_3^- consumption are similarly calculated by substituting NO_3^- concentrations and atom percent ^{15}N excesses into the above equations; gross nitrification rate is designated as n and gross NO_3^- consumption rate is identified as cN.

On 3 September 2003 (two months after the burn), the aboveground vegetation was collected from all plots and separated into annual and perennial plants. Perennial plants were further separated into woody and non-woody tissues. Plant samples were dried at 68°C for 48 hours, weighed, and ground to fine powder using a Type MM 200 Retsch Mixer Mill (Retsch GmbH 88 Co., KG, Germany). Total N analysis was completed using a Shimadzu TOC-V Series SSM-5000A Solid Sample Module for Total Organic Carbon Analyzer (Shimadzu Corporation, Kyoto, Japan). Total N was converted to an areal basis from a per g basis using the dry weight of the biomass removed from each plot.

4.2.4. Statistical Analyses

The hypotheses of no significant differences in NH_4^+ and NO_3^- content, net and gross rates of N mineralization and nitrification, ΔEC or ΔEN content, and plant N content between the upland and riparian positions, burn and control plots, or months was tested using the sample data collected after the burn was conducted (July to October 2003). Data from the forest floor and mineral soil was analyzed separately. Outliers were assessed using boxplots and scatterplots of the data and removed before statistical analysis. The assumptions of ANOVA (normality and homoscedasticity of the residuals between treatment groups) were assessed between treatment groups using the Shapiro-Wilks test statistic (W) in SAS version 8.1 (SAS Institute Inc., Cary, NC, USA) and the Brown-Forsythe modification that uses the median to calculate the Levene test statistic in SPSS for Windows 11.5 (SPSS Inc., Chicago, IL, USA). Natural log transformations were performed for the variables that did not meet the normality criterion; adjustment for heterogeneity of variance was performed using the group= command within the MIXED procedure of SAS version 8.1. An appropriate transformation was not found for NO_3^- content and the net rate of nitrification that contained many zeroes as true values. The type I error is therefore somewhat inflated for these variables. Analysis of variance (ANOVA) was used for statistical analysis due to the complexity of the design structure (randomized block split-split plot design with repeated measures); ANOVA was conducted using the MIXED procedure in SAS version 8.1. For NH_4^+ and NO_3^- contents, the June sampling data was used as a covariable to account for pre-burn differences between adjacent plots. Means were compared using t-tests given statistically significant ANOVA results for month or any interactions; due to the low sample size, an α value of 0.10 was chosen to indicate statistical significance. Spearman correlation coefficients, the distribution-free equivalent of Pearson correlation coefficients, were calculated using SPSS for Windows 11.5 on the raw data to determine which variables co-varied; the significant correlations were verified by plotting the data. Results of the correlation analysis are not shown, however, significant ($P<0.10$) correlation between variables is indicated in the text followed by the correlation coefficient (r), the P value, and the number of samples (n) used in the correlation analysis.

4.3. Results

4.3.1. Inorganic N

In the forest floor, NH_4^+ content was significantly higher in the burned plots (23% and 68% higher in the upland and riparian positions, respectively) than in the control plots in July and August. In the riparian position, the burned plots had a significantly lower NH_4^+ content in October than the control plots. In all the plots, the NH_4^+ content was significantly lower in October than in July, August, and September (Figure 4.1 A; Table 4.1). The NH_4^+ content data, in addition to data for many of the response variables, is illustrated in a box plot (Figure 4.1 A). In a box plot, the middle line in the box represents the median value; the bars extend up to the maximum value and down to the minimum value. The NO_3^- content in the forest floor tended to be higher in the burned compared with the control plots in August, September, and October. The NO_3^- content in the forest floor also tended to be higher in the riparian plots compared with the upland plots (Figure 4.1 B; Table 4.1).

In the mineral soil, NH_4^+ content was significantly higher in the burned plots (88% and 74% higher in the upland and riparian positions, respectively) compared with the control plots and was significantly lower in October compared with July, August, and September (Figure 4.1 C; Table 4.1). The NO_3^- content in the mineral soil tended to be higher in the burned plots compared with the controls and in the riparian plots compared with the upland plots (Figure 4.1 D; Table 4.1).

Figure 4.1. NH_4^+ and NO_3^- content in the forest floor and mineral soil (0-10 cm) of samples collected in July, August, September, and October from the upland control (UC), upland burn (UB), riparian control (RC), and riparian burn (RB) plots.

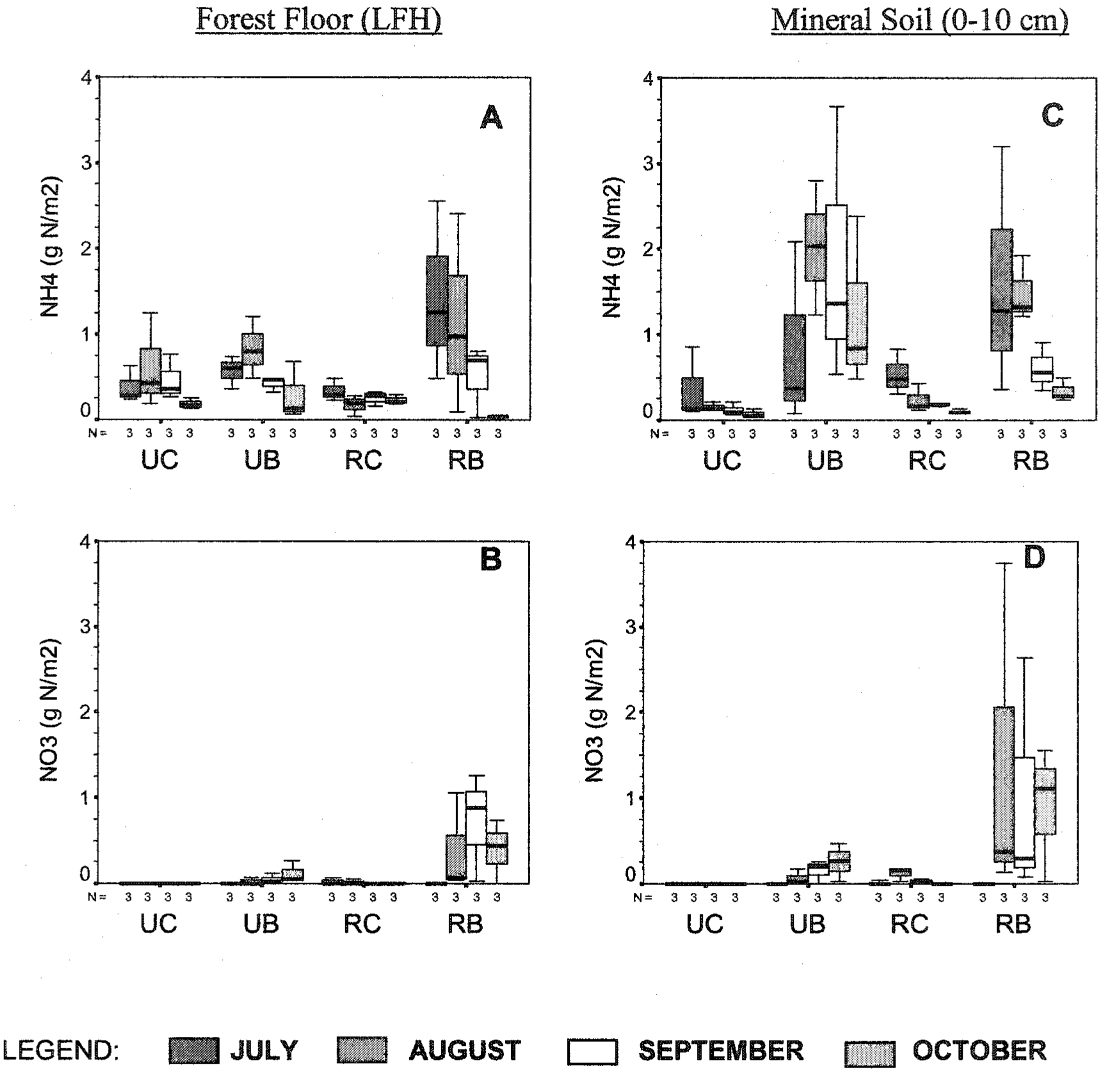


Table 4.1. F and P values from the analysis of variance for NH_4^+ content, NO_3^- content, NO_3-N/NH_4-N ratio, and IN/EN ratio in the forest floor (LFH) and mineral soil (0-10 cm).

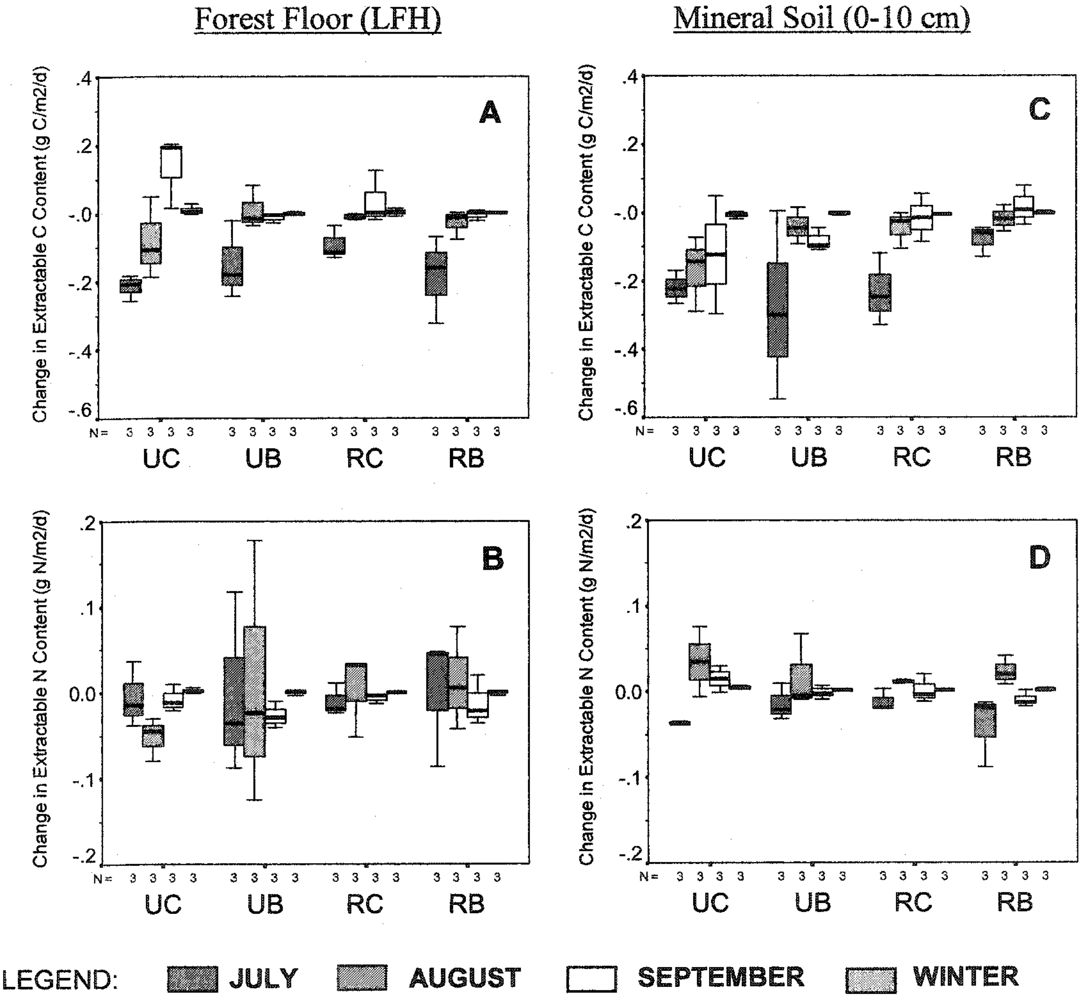
	NH_4^+		NO_3^-		NO_3-N/NH_4 -N Ratio		IN/EN Ratio	
Effect	F	P value	F	P value	F	P value	F	P value
FOREST FLOOR (LFH)								
Position (Pos)	0.65	0.569	3.32	0.210	3.62	0.198	1.80	0.312
Burn	0.75	0.546	4.99	0.155	6.39	0.127	39.38	0.025
Pos*Burn	0.14	0.776	2.87	0.232	3.33	0.210	0.01	0.925
Month	12.01	<0.0001	1.81	0.173	4.36	0.014	2.06	0.132
Pos*Month	2.43	0.090	1.64	0.207	2.21	0.113	0.17	0.915
Burn*Month	6.61	0.002	2.05	0.133	4.48	0.012	4.80	0.009
Pos*Burn*Month	3.90	0.021	1.82	0.170	2.29	0.104	2.64	0.073
MINERAL SOIL (0-10 cm)								
Position (Pos)	0.00	0.998	2.38	0.263	2.97	0.2270	6.70	0.122
Burn	35.32	0.106	3.44	0.205	2.15	0.2801	9.71	0.089
Pos*Burn	0.15	0.762	1.90	0.302	1.66	0.3269	0.25	0.666
Month	4.15	0.017	1.42	0.260	1.80	0.1750	3.04	0.049
Pos*Month	2.08	0.130	1.20	0.330	1.38	0.2739	0.56	0.649
Burn*Month	4.06	0.018	1.09	0.372	2.09	0.1275	0.82	0.497
Pos*Burn*Month	1.02	0.400	0.86	0.474	1.60	0.2166	0.84	0.487

4.3.2. Change in Extractable C and N Contents

In the forest floor, ΔEC was significantly correlated to ΔEN (r=0.716, P<0.0001, n=48); both ΔEC and ΔEN were negatively correlated to the net N mineralization rate (ΔEC: r=-0.393, P=0.006, n=48; ΔEN: r=-0.345, P=0.018, n=48). ΔEC and ΔEN were significantly lower in the July as compared to the August, September, and winter incubations and ΔEC was lower in the riparian plots than in the upland plots in July (Figure 4.2 A and B; Table 4.2).

In the mineral soil, ΔEC and ΔEN were significantly correlated (r=0.676, P<0.0001, n=48). ΔEC was significantly lower in the July than in the August, September, and winter incubations (Figure 4.2 C; Table 4.2); ΔEC tended to be highest in the August incubation. ΔEN was not significantly affected by position, the burn treatment, or month (Figure 4.2 D; Table 4.2).

Figure 4.2. Change in extractable C and N content in the forest floor and mineral soil (0-10 cm) of the July (30 day length of incubation), August (30 day), September (28 day), and winter (197 day) incubations in the upland control (UC), upland burn (UB), riparian control (RC) and riparian burn (RB) plots.



Forest Floor (LFH)

Mineral Soil (0-10 cm)

Table 4.2. F and P values from the analysis of variance for the change in EC content (ΔEC) and change in EN content (ΔEN) in the forest floor (LFH) and mineral soil (0-10 cm).

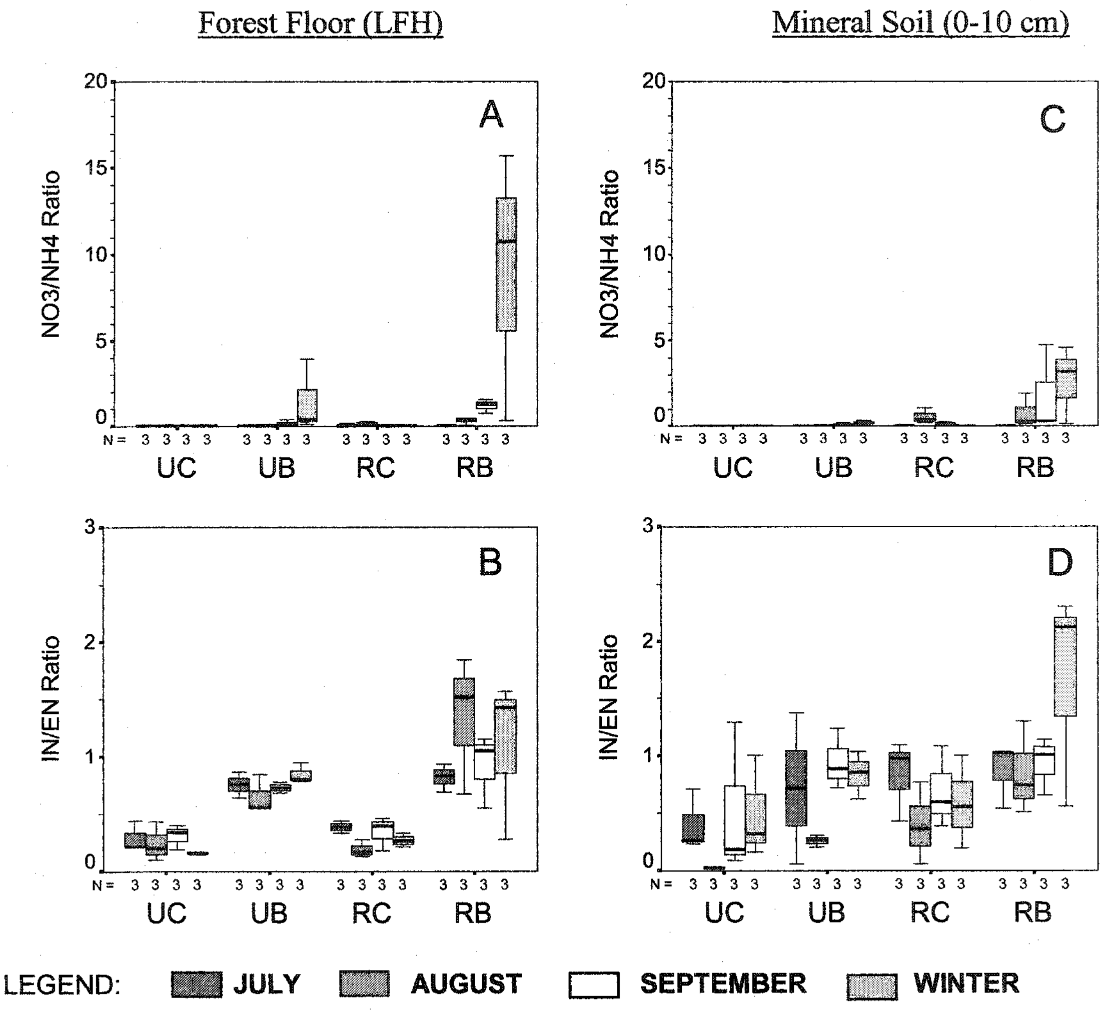
	ΔEC		ΔEN	
Effect	F	P value	F	P value
FOREST FLOOR (LFH)				
Position (Pos)	0.26	0.663	0.34	0.619
Burn	0.11	0.775	0.02	0.908
Pos*Burn	0.18	0.714	0.21	0.689
Month	14.57	<0.0001	11.74	<0.0001
Pos*Month	2.68	0.070	1.56	0.227
Burn*Month	2.34	0.010	1.30	0.298
Pos*Burn*Month	1.06	0.385	1.36	0.281
MINERAL SOIL (0-10 cm)				
Position (Pos)	3.06	0.222	0.20	0.696
Burn	5.79	0.138	0.12	0.767
Pos*Burn	0.04	0.864	0.03	0.888
Month	7.17	0.001	0.22	0.883
Pos*Month	1.09	0.374	0.24	0.866
Burn*Month	0.88	0.464	0.44	0.726
Pos*Burn*Month	0.69	0.568	0.24	0.865

4.3.3. NO_3-N/NH_4-N and IN/EN Ratios

In the forest floor, the NO_3-N/NH_4-N ratio tended to be higher in the burned and riparian plots compared with the control and upland plots, respectively, but the difference was only significant in October; the NO_3-N/NH_4-N ratio was significantly higher in October than in July, August, and September (Figure 4.3 A; Table 4.1). The forest floor IN/EN ratio was significantly higher in the burned plots compared with the control plots in July, August, September, and October but did not differ between the riparian and upland plots (Figure 4.3 B; Table 4.1).

In the mineral soil, the NO_3-N/NH_4-N ratio was not significantly affected by the burn treatment, position, or month (Figure 4.3 C; Table 4.1). The IN/EN ratio was significantly higher in the burned plots compared with the control plots and was significantly lower in August than in September and October (Figure 4.3 D; Table 4.1).

Figure 4.3. NO_3-N/NH_4-N and IN/EN ratios in the forest floor and mineral soil (0-10 cm) of samples collected in July, August, September, and October from the upland control (UC), upland burn (UB), riparian control (RC), and riparian burn (RB) plots.



Forest Floor (LFH)

Mineral Soil (0-10 cm)

4.3.4. Net N Mineralization and Nitrification Rates

In the forest floor, net N mineralization rate was significantly affected by month (Table 4.3); the July incubation was significantly higher than the August, September and winter incubations, and the September and winter incubations were significantly different. Net mineralization rates tended to be lower in the burned than in the control plots in the August, September, and winter incubation periods (Figure 4.4 A). Net nitrification rate in the forest floor was not significantly affected by position, the burn treatment, or month (Table 4.3). However, net nitrification rate tended to be higher in the UB than in the UC plots in the September and winter incubations. Net nitrification rates tended to be higher in the July incubation compared with the August, September, and winter incubations in the UB, RC, and RB plots (Figure 4.4 B).

In the mineral soil, the net N mineralization rate was not significantly affected by position, burn treatment, or month (Table 4.3), however, net N mineralization rates tended to be higher in the RB plots than in the RC plots in the July incubation and lower in the burned plots as compared with the control plots in the August, September, and winter incubations (Figure 4.4 C). Net nitrification rate in the mineral soil was not significantly affected by position, the burn treatment, or month (Table 4.3). Net nitrification rates tended to be higher in the UB than in the UC plots in all the incubation periods. Net nitrification rates in the RB plots tended to be higher than the RC plots during the July and August incubations and lower during the September and winter incubations (Figure 4.4 D).

Figure 4.4. Net N mineralization and nitrification rates in the forest floor and mineral soil (0-10 cm) of the July (30 day length of incubation), August (30 day), September (28 day), and winter (197 day) incubations in the upland control (UC), upland burn (UB), riparian control (RC), and riparian burn (RB) plots.

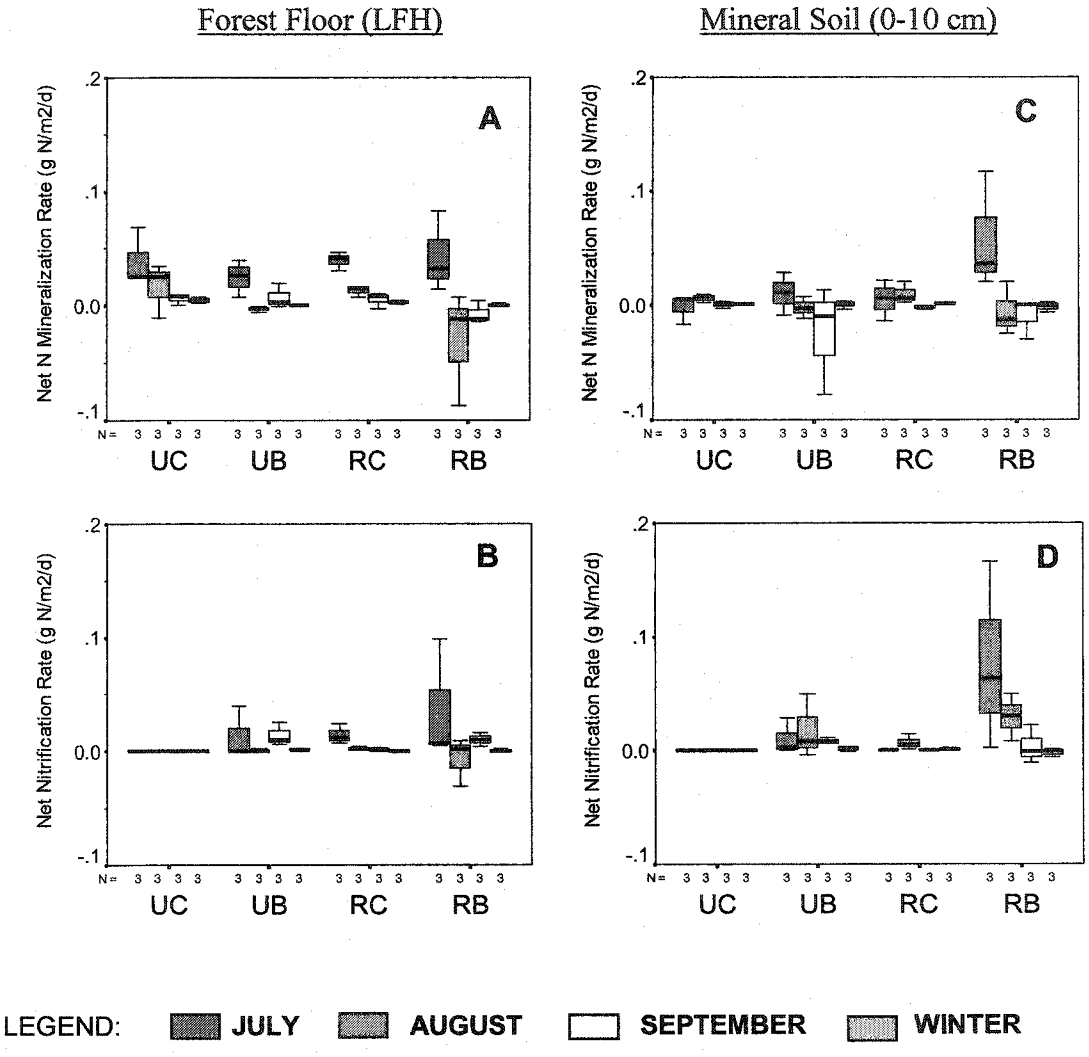


Table 4.3. F and P values from the analysis of variance for net N mineralization rate and net nitrification rate in the forest floor (LFH) and mineral soil (0-10 cm).

	Net N Mineralization Rate		Net Nitrification Rate	
Effect	F value	P value	F value	P value
FOREST FLOOR				
Position (Pos)	0.05	0.844	0.05	0.847
Burn	4.26	0.175	1.81	0.311
Pos*Burn	0.02	0.899	4.53	0.167
Month	17.40	<0.0001	2.22	0.112
Pos*Month	2.02	0.140	0.28	0.839
Burn*Month	2.09	0.129	1.40	0.266
Pos*Burn*Month	1.92	0.155	0.51	0.678
MINERAL SOIL (0-10 cm)				
Position (Pos)	0.02	0.896	1.48	0.349
Burn	0.65	0.504	5.53	0.143
Pos*Burn	0.86	0.452	0.46	0.569
Month	0.99	0.414	1.88	0.159
Pos*Month	0.61	0.613	1.67	0.200
Burn*Month	2.02	0.138	2.28	0.105
Pos*Burn*Month	0.25	0.861	2.17	0.118

4.3.5. Cumulative Net N Mineralization and Nitrification Rates

The cumulative net N mineralization rate in the forest floor from 5 July 2003 to 16 April 2004 was significantly higher in the control compared with the burned plots and tended to be higher in the upland compared with the riparian plots. In contrast, the cumulative net nitrification rate, though not significant, tended to be higher in the burned and the riparian plots compared with the control and the upland plots, respectively (Table 4.4). In the mineral soil, neither the cumulative net N mineralization rate nor the cumulative net nitrification rate was significantly affected by either position or the burn treatment, however the cumulative net nitrification rate tended to be higher in the burned compared with the control and in the riparian compared with the upland plots (Table 4.4).

In the forest floor, the winter (1 October 2003 to 16 April 2004) incubation accounted for an average (SE) of 37 (7.2)% of the cumulative net N mineralization rate in the UC plots, 19 (9.5) in the UB plots, 24 (5.2)% in the RC plots, and 51 (24)% in the RB plots. In the mineral soil, the winter incubation accounted for an average (SE) of 47 (7.5)% of the cumulative net N mineralization rate in the UC plots, 32 (9.7)% in the UB plots, 33 (1.8)% in the RC plots, and 53 (34)% in the RB plots.

Table 4.4. Mean (SE in parentheses) cumulative net N mineralization and nitrification rates from 5 July 2003 to 16 August 2004 in the forest floor (LFH) and mineral soil (0-10 cm) of the upland control (UC), upland burn (UB), riparian control (RC), and riparian burn (RB) plots. Treatment means followed by the same letter within the same column for the forest floor or mineral soil are not significantly different (P>0.1).

Plot Type	Cumulative Net N Mineralization Rate (g N m^{-2})	Cumulative Net Nitrification Rate (g N m^{-2})
FOREST FLOOR (LFH)		
UC	2.8 (0.84) a	0.0 (0.0) a
UB	0.90 (0.31) b	1.1 (0.46) a
RC	2.3 (0.35) a	0.61 (0.14) a
RB	0.29 (0.56) b	1.3 (0.81) a
MINERAL SOIL (0-10 cm)		
UC	0.35 (0.21) a	0.0 (0.0) a
UB	-0.35 (2.7) a	1.4 (0.35) a
RC	0.73 (0.88) a	0.46 (0.30) a
RB	0.97 (3.1) a	3.0 (3.5) a

4.3.6. Gross Mineralization, Immobilization and Nitrification Rates

There were no significant effects of position or burn treatment on gross mineralization rate, gross NH_4^+ consumption rate, gross nitrification rate, or gross NO_3^- consumption rate (Table 4.5; ANOVA data not shown). However, in the forest floor the gross mineralization rate tended to be higher in the upland compared with the riparian plots; the gross mineralization and NH_4^+ consumption rates tended to be higher in the UB than in the UC plots but tended to be lower in the RB than in the RC plots. In the mineral soil, the gross rates of mineralization and NH_4^+ consumption tended to be higher in the burned compared with the control plots and lower in the RC compared with the UC plots. In the forest floor and mineral soil, the gross rates of nitrification and NO_3^- consumption tended to be higher in the burned compared with the control and in the RC compared with the UC plots.

Table 4.5. Mean (SE in parentheses) gross mineralization rate (M), gross NH_4^+ consumption rate (CA), gross nitrification rate (N), and gross NO_3^- consumption rate (CN) in the forest floor (LFH) and mineral soil (0-10 cm) of the upland control (UC), upland burn (UB), riparian control (RC), and riparian burn (RB) plots.

Plot Type	M (mg N kg^{-1} d^{-1})	CA (mg N kg^{-1} d^{-1})	N (mg N kg^{-1} d^{-1})	CN (mg N kg^{-1} d^{-1})
FOREST FLOOR (LFH)				
UC	42.70 (12.70) a	2.23 (13.03) a	-19.13 (21.36) a	1.77 (1.35) a
UB	50.35 (18.41) a	42.82 (21.80) a	37.12 (16.10) a	11.60 (7.27) a
RC	34.28 (17.24) a	38.23 (2.91) a	1.27 (8.69) a	1.85 (4.76) a
RB	23.46 (9.50) a	19.07 (0.51) a	23.83 (25.07) a	4.43 (10.13) a
MINERAL SOIL (0-10 CM)				
UC	3.37 (2.01) a	3.98 (1.55) a	1.53 (0.00) a	-0.51 (0.00) a
UB	4.04 (4.68) a	11.69 (9.50) a	17.31 (15.68) a	26.02 (26.14) a
RC	0.24 (4.22) a	2.24 (0.97) a	-5.81 (5.59) a	2.36 (2.50) a
RB	4.45 (1.71) a	3.38 (3.32) a	-11.90 (13.76) a	6.65 (1.21) a

4.3.7. Foliar % N and Total Plant N Content

The % N in the annual plants tended to be higher in the burned plots than in the control plots but did not differ between positions (Table 4.6). No difference between positions in % N in the leaves or woody tissue of the perennial plants was apparent. The total plant N content, however, was significantly lower in the burned plots compared with the control plots (Table 4.6).

Table 4.6. Mean (SE in parentheses) percent N in annual plants and perennial plant leaves and woody tissues, and total plant N content (g m^{-2}) in the upland control (UC), upland burn (UB), riparian control (RC) and riparian burn (RB) plots. Treatment means followed by the same letter within the same column are not significantly different (P>0.1).

Plot Type	N in Annual Plants (%)	N in Perennial Leaves (%)	N in Perennial Woody Tissue (%)	Total Plant N Content (g N m^{-2})
UC	1.13 (0.12) a	1.16 (0.09) a	0.47 (0.09) a	0.73 (0.11) a
UB	1.29 (0.09) a	-	-	0.18 (0.02) b
RC	1.13 (0.12) a	1.08 (0.07) a	0.49 (0.02) a	0.77 (0.12) a
RB	1.33 (0.19) a	-	-	0.12 (0.01) b

4.4. Discussion

In this study, the range of NH_4^+ (LFH: 0.04 to 1.25 g N m^{-2}; 0-10 cm: 0.05 to 0.86 g N m^{-2}) and NO_3^- (LFH: 0 to 0.07 g N m^{-2}; 0-10 cm: 0-0.18 g N m^{-2}) contents, cumulative net N mineralization rates from July to September (LFH: 0.68-3.1 g N m^{-2};0-10 cm: -0.21-0.85 g N m^{-2}), and cumulative net nitrification rates from July to September (LFH: 0-0.54 g N m^{-2}; 0-10 cm: -0.01-0.43 g N m^{-2}) in the control plots were typical of those reported for soils in the Canadian boreal forest (Fyles et al. 1991; Walley et al. 1996; Little et al. 2002; Carmosini et al. 2003). Gross mineralization, gross NH_4^+ consumption, gross nitrification, and gross NO_3^- consumption rates also fell within the range of values reported for soils in mature stands in the boreal forest (Stottlemyer and Toczydlowski 1999b; Carmosini et al. 2002; Bengtsson et al. 2003).

4.4.1. Seasonal Trends

The net N mineralization rate was significantly higher in the July as compared with the August, September, and winter incubations. A greater net N mineralization rate corresponded to the most negative ΔEC and ΔEN contents in the July incubation, suggesting significant microbial consumption of soluble C and N compounds (Hart et al. 1994). Maximum net and gross N mineralization rates in July have been previously demonstrated in white birch (*Betula papyrifera* Marsh.) and white spruce (*Picea glauca* (Moench) A. Voss) stands in the boreal forest of Michigan (Stottlemyer and Toczydlowski 1999a,b). Higher net N mineralization rates in July did not appear to correspond to mean monthly soil temperature, which was similar in the mineral soil in July (12.1°C) and August (12.4°C), but may be explained by significantly higher gravimetric and volumetric moisture contents in the forest floor and mineral soil at the beginning of the July incubation compared with the August, September, and winter incubations (data not shown). Other studies have suggested an increase in net N mineralization rate with moisture content above optimum temperatures for microbial activity but little effect of moisture content below that point (Gonçalvez and Carlyle 1994; Leirós et al. 1999; Zaman and Chang 2004); in mixedwood stands in the WBF, Offord (1999) demonstrated an optimum temperature of 12°C for the specific net N mineralization rate in the forest floor. This suggests that the high net N mineralization rate in July may be due to the combination of optimal soil temperature and moisture content for microbial activity.

Results from the winter incubation indicate significant N mineralization in the forest floor and mineral soil of this trembling aspen-dominated mixedwood stand between October and mid-April; in the control plots the net N mineralization rate in winter varied from 0.24 to 1.5 g N m^{-2} in the forest floor and 0.13 to 0.48 g N m^{-2} in the mineral soil. To date, research focus has been placed on mineralization and nitrification rates in the WBF from May to October (e.g. Walley et al. 1996; Huang and Schoenau 1997; Little et al. 2002; Frey et al. 2003). This study suggests that microbial activity and therefore N mineralization may persist under snow cover, corroborating the results of Carmosini et al. (2003) and Stottlemyer and Toczydlowski (1999a) who reported positive rates of net N mineralization and nitrification from October to May in the forest floor of mature trembling aspen, white birch, and white spruce stands. In the Boreal Plain, the fate of this accumulated N is unknown, however it may infiltrate soil during spring snowmelt due to little snowmelt runoff (Stottlemyer and Toczydlowski 1999a; Devito et al. 2004; Whitson et al. 2004) and provide an immediate N source for plant growth. Further research is required to determine the importance of winter N mineralization for plant growth in the WBF.

4.4.2. Differences between the Riparian and Upland Positions

In this study no statistically significant differences in soil N cycling between the riparian and upland positions were found. However, the data showed that NO_3^- content, NO_3-N/NH_4-N ratio, and cumulative net nitrification rate were higher in the lower slope, riparian compared with the upland plots. Greater NO_3^- supply was also demonstrated in wetter lower slope compared with drier upper slope positions in a mature trembling aspen stand in Saskatchewan (Huang and Schoenau 1997). A significantly lower extractable C content, higher C/N ratio (data not shown), lower net and gross N mineralization rates, and higher gross NH_4^+ consumption rate in the RC compared with the UC plots suggests that the heterotrophic microbial population may have been smaller, limited by substrate availability and possibly stable or declining in the RC plots. Combined with greater NH_4^+ mobility in moist soils, there may have been greater competition for NH_4^+ by autotrophic nitrifiers in the RC plots (Hart et al. 1994; Evans et al. 1998). Much higher mean cumulative net nitrification rates in the RC plots than those reported for the forest floor [0.086 (SE: 0.14) g N m^{-2}] and mineral soil [0.067 (0.050) g N m^{-2}] of trembling aspen stands in the WBF (Carmosini et al. 2003) suggests that the trend towards greater net nitrification and NO_3^- pools in the RC as compared to the UC plots may have been biologically significant. Indeed, NO_3^- content represented up to 23% and 52% of total inorganic N while net nitrification rates were up to 54% and 508% of net mineralization rates in the forest floor and mineral soil of the RC plots, respectively.

4.4.3. Effects of the Experimental Burn

Burning significantly increased the NH_4^+ content in July and August and the IN/EN ratio in the forest floor and mineral soil while the total plant N content and forest floor cumulative net N mineralization rates were significantly reduced. The initial pulse of NH_4^+ in the forest floor after fire is attributed to condensation of NH_3 that is released from combusted organic N compounds and from heat disrupted organo-clay complexes (Raison 1979; DeBano et al. 1979; Giovannini et al. 1990). In the mineral soil, a higher NH_4^+ content may be due to condensation of NH_3 into the cooler mineral soil during fire and NH_4^+ leaching from the forest floor (Klemmedson et al. 1962; DeBano et al. 1979).

In this study, the difference in forest floor NH_4^+ content appeared to be larger between the RB and RC plots compared with the UB and UC plots, while the difference in mineral soil NH_4^+ content appeared to be greater between the UB and UC plots. This trend suggests a lower penetration of heat energy in the RB plots that reduced oxidation and volatilization of N compounds to gaseous N_2 and N oxides. Average fire temperature and post-fire NH_4^+ content were negatively correlated in the forest floor, but positively related in the mineral soil in experiments that heated soil from chapparal and ponderosa pine (*Pinus ponderosa* Laws.)-Douglas fir (*Pseudotsuga menziesii* var. *glauca*) ecosystems at different temperatures (DeBano et al. 1979; Choromanska and DeLuca 2002). The NH_4^+ content in the forest floor of the burned plots approached that of the control plots 2 months after the burn while the NH_4^+ content in the mineral soil remained slightly higher in the burned plots compared with the controls. High NH_4^+ content in the mineral soil of the burned plots may persist; in a ponderosa pine-Douglas fir ecosystem, higher post-fire NH_4^+ content in the mineral soil endured for more than 16 months after high severity fire (Choromanska and DeLuca 2001).

An increase in the post-fire IN/EN ratio suggests that inorganic N was the main component of EN in the burned soils. Prieto-Fernández et al. (1998) observed an increase in EN content in the mineral soil after fire in pine forests in Spain that mainly depended on the increase in inorganic N. In this study, the EN content was significantly correlated to the NH_4^+ content in the forest floor ($r=0.661$, $P<0.0001$, $n=48$) and mineral soil ($r=0.636$, $P<0.0001$, $n=48$), however, the EN content only tended to be higher after fire in the mineral soil (data not shown). This suggests a decline in extractable organic N in the forest floor after fire.

Fire release of readily metabolizable C and N compounds combined with a post-fire increase in soil temperature and moisture content has been shown to temporarily stimulate microbial activity and N mineralization in the remaining forest floor after fire (Ahlgren and Ahlgren 1960; Raison 1979; Almendros et al. 1990; Prieto-Fernández et al. 1993). In this study, an increase in net N mineralization rate after burning was not observed. In July (immediately after the burn), the net N mineralization rate was similar in the forest floor and slightly higher in the mineral soil of the control as compared to the burned plots but by August the net N mineralization rate had declined in both horizons in the burned plots. A reduction in the forest floor and mineral soil EC and EN contents in August (data not shown) suggests exhaustion of the easily decomposable compounds released by fire one month after burning (Nadelhoffer and Aber 1983; Almendros et al. 1990). In the WBF, a reduced net N mineralization rate has also been reported immediately after burning in the upper 5 cm of mineral soil in a white spruce clearcut stand in Alberta (Frey et al. 2003) and 4 years after fire in the surface 15 cm of soil in a trembling aspen-dominated mixedwood stand in Saskatchewan (Walley et al. 1996). This suggests that a lower net N mineralization rate may be expected after fire both where the canopy remains intact (this study) and where it is removed (Walley et al. 1996; Frey et al. 2003).

The post-fire decline in net N mineralization rate in the forest floor may be due to alteration of the humus to include more stable, recalcitrant organic compounds (Nadelhoffer and Aber 1983; Almendros et al. 1990; Hernández et al. 1997), or a reduction in the post-fire microbial population (Pietikäinen and Fritze 1993; Neary et al. 1999). This study supports both of these hypotheses; significantly lower microbial biomass C (MB-C) and N (MB-N) contents suggest a decline in the microbial population and lower MB-C/total C and MB-N/total N ratios indicate reduced humus quality (Bauhus et al. 1998) in the forest floor of the burned compared with the control plots (data not shown). However, the cumulative net N mineralization rate and gross N mineralization rate in August tended to be higher in the forest floor of the UB plots where the MB-C, MB-N, MB-C/total C, and MB-N/total N ratios tended to be lower compared with the RB plots. This suggests that the higher mean post-fire soil temperature in the UB (11.48°C) as compared with the RB plots (11.14°C) may have stimulated microbial activity (data not shown). Wilson et al. (2002) also observed a positive relationship between soil temperature and the net N mineralization rate in frequently burned longleaf pine-wiregrass ecosystems in Georgia, USA. Further research is required to determine the relationship between humus quality, microbial activity and biomass, soil temperature, and post-fire net N mineralization rates.

Although the changes were not significant, NO_3^- contents in August, September and October, the NO_3-N/NH_4-N ratio in October, the cumulative net nitrification rate, and the gross nitrification and NO_3^- consumption rates tended to be higher in the forest floor and mineral soil of the burned compared with the control plots. DeLuca and Zouhar (2000) also observed a short lag between fire and an increase in NO_3^- content that may be accounted for by delayed nitrification. Increased nitrification rates after fire may be explained by stimulation of autotrophic nitrifiers by a post-fire increase in pH (Bauhus et al. 1993; Ste-Marie and Paré 1999), less plant competition for NH_4^+ (Kaye and Hart 1997), or charcoal absorption of nitrification-inhibiting phenolic compounds (Zackrisson et al. 1996). A higher post-fire nitrification rate has been observed in the forest floor and mineral soil of burned, and clearcut and burned sites in various ecosystems (Pietikäinen and Fritze 1993, 1995; Giardina and Rhoades 2001; Romanyà et al. 2001; Frey et al. 2003). It is not clear how long the increased nitrification rates will persist after fire if a reduced rate of net N mineralization decreases the NH_4^+ supply to autotrophic nitrifiers.

4.5. Conclusions

This study demonstrated no significant effect of higher moisture content in the lower slope, riparian position on NH_4^+ and NO_3^- contents, net N mineralization and nitrification rates, gross rates of mineralization, immobilization, and nitrification, or total plant N content. However, there was a trend towards lower gross mineralization and gross NH_4^+ consumption rates, and higher net and gross nitrification rates in the riparian compared with the upper slope position that may be due to a decrease in substrate quality in the riparian position (Hart et al. 1994). This result suggests that high moisture content may indirectly influence the N cycle by favouring C accumulation and correspondingly, a higher C/N ratio. A negative relationship between substrate quality, as indicated by the C/N ratio, and the net N mineralization rate has been previously demonstrated in coniferous and deciduous-dominated mixedwood stands in the WBF (Offord 1999; Little et al. 2002). Further research on the microbial community in the riparian and upper slope position may explain some of these trends.

Compared with the unburned plots, the burned plots in this stand demonstrated a significantly higher NH_4^+ content in July and August and IN/EN ratio from July to October in the forest floor and mineral soil, a significantly lower cumulative net N mineralization rate in the forest floor from July to April, and a significantly lower total plant N content. These trends indicate that the short-term effect of burning in this trembling aspen-dominated mixedwood stand was to initially increase and then reduce N availability; this is the first study to report such trends after a low severity fire where the canopy remained intact in a trembling aspen dominated mixedwood stand in the WBF. No significant position-burn interactions were identified, however, slightly higher cumulative net and gross N mineralization rates in the forest floor of the UB compared with the RB plots suggest a positive correlation between forest floor removal and N availability in the short term after fire. Examination of the changes in microbial activity, humus quality, and soil temperature and their relative influence on net and gross rates of N mineralization more than one year after fire is required to determine the persistence of this trend.

4.6. References

Ahlgren, I.F. and Alhgren, C.E. 1960. Ecological effects of forest fires. Bot. Rev. **26**: 483-533.

Alberta Energy and Natural Resources. 1975. Ecological Land Classification and Evaluation: Brazeau Reservoir Area. Land Use Study Special Report.

Almendros, G., Gonzalez-vila, F.J. and Martin, F. 1990. Fire induced transformation of soil organic matter from an oak forest: an experimental approach to the effects of fire on humic substances. Soil Sci. **149**: 158-168.

Andersson, M., Michelsen, A., Jensen, M. and Kjøller, A. 2004. Tropical savannah woodland: effects of experimental fire on soil microorganisms and soil emissions of carbon dioxide. Soil Biol. Biochem. **36**: 849-858.

Bauhus, J., Khanna, P.K. and Raison, R.J. 1993. The effect of fire on carbon and nitrogen mineralization and nitrification in an Australian forest soil. Soil Biol. Biochem. **31**: 621-639.

Bauhus, J., Paré, D. and Côté, L. 1998. Effects of tree species, stand age and soil type on soil microbial biomass and its activity in a southern boreal forest. Soil Biol. Biochem. **30**: 1077-1089.

Beckingham, J.D., Corns, I.G.W. and Archibald, J.H. 1996. Field guide to ecosites of west-central Alberta. Canadian Forest Service Special Report 9. Northern Forestry Centre, Edmonton, AB.

Bengtsson, G., Bengtson, P. and Månsson, K.F. 2003. Gross nitrogen mineralization-, immobilization-, and nitrification rates as a function of soil C/N ratio and microbial activity. Soil Biol. Biochem. **35**: 143-154.

Carmosini, N., Devito, K. and Prepas, E.E. 2002. Gross nitrogen transformations in harvested and mature aspen-conifer mixed forest soils from the Boreal Plain. Soil Biol. Biochem. **34**: 1949-1951.

Carmosini, N., Devito, K.J. and Prepas, E.E. 2003. Net nitrogen mineralization and nitrification in trembling aspen forest soils on the Boreal Plain. Can. J. For. Res. **33**: 2262-2268.

Choi, W.J., Ro, H.M. and Lee, S.M. 2003. Natural ^{15}N abundances of inorganic nitrogen in soil treated with fertilizer and compost under changing soil moisture regimes. Soil Biol. Biochem. **35**: 1289-1298.

Choromanska, U. and DeLuca, T.H. 2001. Prescribed fire alters the impact of wildfire on soil biochemical properties in a Ponderosa Pine forest. Soil Sci. Soc. Am. J. **65**: 232-238.

Choromanska, U. and DeLuca, T.H. 2002. Microbial activity and nitrogen mineralization in forest mineral soils following heating: evaluation of post-fire effects. Soil Biol. Biochem. **34**: 263-271.

Cirmo, C. P. and McDonnell, J.J. 1997. Linking the hydrologic and biogeochemical controls of nitrogen transport in near-stream zones of temperate-forested catchments: a review. J. Hydrol. **199**: 88-120.

DeBano, L.F., Eberlein, G.E. and Dunn, P.H. 1979. Effects of burning on chaparral soils: I. Soil nitrogen. Soil Sci. Soc. Am. J. **43**: 504-509.

DeLuca, T.H. and Zouhar, K.L. 2000. Effects of selection harvest and prescribed fire on the soil nitrogen status of ponderosa pine forests. For. Ecol. Manage. **138**: 263-271.

Devito, K.J., Creed, I.F and Fraser, C.J.D. 2004. Controls on runoff from a partially harvested aspen forested headwater catchment, Boreal Plain, Canada. Hydrol. Proc. In Press.

Doane, T.A. and Horwath, W.R. 2003. Spectrophotometric determination of nitrate with a single reagent. Anal. Lett. **36**: 2713-2722.

Dwire, K.A. and Kauffman, J.B. 2003. Fire and riparian ecosystems in landscape of the western USA. For. Ecol. Manage. **128**: 61-74.

Ecological Stratification Working Group. 1996. A national ecological framework for Canada. Centre for Land and Biological Resources Research, Agriculture and Agri-Food Canada and State of Environment Directorate, Environment Canada, Ottawa, ON. 125 pp.

Environment Canada. 2004a. Canadian Climate Normals or Averages 1971-2000 [online]. Available from http://www.climate.weatheroffice.ec.gc.ca/climate_normals/index_e.html [accessed April 23 2004].

Environment Canada. 2004b. Climate Data [online]. Available from http://www.climate.weatheroffice.ec.gc.ca/climateData/canada_e.html [accessed April 23 2004].

Environment Canada. 2004c. Archived Hydrometric Data: Pembina Creek below Paddy River [online]. Available from http://www.msc-smc.ec.gc.ca/wsc/hydat/H2O/index_e.cfm?cname=graph.cfm [accessed May 7 2004].

Evans, C.A., Miller, E.K. and Friedland, A.J. 1998. Nitrogen mineralization associated with birch and fir under different soil moisture regimes. Can. J. For. Res. **28**: 1890-1898.

Fenn, M.E., Poth, M.A., Dunn, P.H. and Barro, S.C. 1993. Microbial N and biomass, respiration and N mineralization in soils beneath two chapparal species along a fire-induced age gradient. Soil Biol. Biochem. **25**: 457-466.

Frey, B.R., Lieffers, V.J., Munson, A.D. and Blenis, P.V. 2003. The influence of partial harvesting and forest floor disturbance on nutrient availability and understory vegetation in boreal mixedwoods. Can. J. For. Res. **33**: 1180-1188.

Fyles, J.W. and McGill, W.B. 1987. Nitrogen mineralization in forest soil profiles from central Alberta. Can. J. For. Res. **17**: 242-249.

Fyles, J.W., Fyles, I.H., Beese, W.J. and Feller, M.C. 1991. Forest floor characteristics and soil nitrogen availability on slash-burned sites in coastal British Columbia. Can. J. For. Res. **21**: 1516-1522.

Giardina, C.P. and Rhoades, C.C. 2001. Clear cutting and burning affect nitrogen supply, phosphorus fractions and seedling growth in soils from a Wyoming lodgepole pine forest. For. Ecol. Manage. **140**: 19-28.

Giovannini, G., Lucchesi, S. and Giachetti, M. 1990. Effects of heating on some chemical parameters related to soil fertility and plant growth. Soil Sci. **149**: 344-350.

Gonçalves, J.L.M. and Carlyle, J.C. 1994. Modelling the influence of moisture and temperature on net nitrogen mineralization in a forested sandy soil. Soil Biol. Biochem. **26**: 1557-1564.

Grogan, P., Bruns, T.D. and Chapin, F.S. III. 2000. Fire effects on ecosystem nitrogen cycling in a Californian bishop pine forest. Oecologia. **122**: 537-544.

Hart, S.C., Nason, G.E., Myrold, D.D. and Perry, D.A. 1994. Dynamics of gross nitrogen transformations in an old-growth forest: the carbon connection. Ecol. **75**: 880-891.

Hernández, T., García, C., and Reinhardt, I. 1997. Short-term effect of wildfire on the chemical, biochemical and microbiological properties of Mediterranean pine forest soils. Biol. Fertil. Soils. **25**: 109-116.

Huang, W.Z. and Schoenau, J.J. 1997. Seasonal and spatial variations in soil nitrogen and phosphorus supply rates in a boreal aspen forest. Can. J. Soil Sci. **77**: 597-612.

Johnson, E.A. 1992. Fire and Vegetation Dynamics: Studies from the North American Boreal Forest. Cambridge University Press, Cambridge, UK.

Johnson, D., Kershaw, L., MacKinnon, A. and Pojar, J. 1995. Plants of the western boreal forest and aspen parkland. Lone Pine Publishing, Edmonton, AB.

Kalra, Y.P. and Maynard, D.G. 1991. Methods manual for forest soil and plant analysis. Information Report NOR-X-319. Canadian Forest Service, Northern Forestry Centre, Edmonton, AB.

Kaye, J.P. and Hart, S.C. 1997. Competition for nitrogen between plants and soil microorganisms. TREE. **12**: 139-143.

Khanna, P.K. and Raison, R.J. 1986. Effect of fire intensity on solution chemistry of surface soil under a Eucalyptus pauciflora forest. Aust. J. Soil Res. **24**: 423-434.

Keeney, D.R. and Nelson, D.W. 1982. Nitrogen – inorganic forms. *In* Methods of Soil Analysis. Part 2. Chemical and Microbiological Properties. *Edited by* A.L. Page. American Society of Agronomy, Inc., and Soil Science Society of America, Inc., Madison, WI. pp. 643-698.

Klemmedson, J.O., Schultz, A.M., Jenny, H. and Biswell, H.H. 1962. Effect of prescribed burning of forest litter on total soil nitrogen. Soil Sci. Soc. Am. Proc. **26**: 200-202.

Leirós, M.C., Trasar-Cepeda, C., Seoane, S. and Gil-Sotres, F. 1999. Dependence of mineralization of soil organic matter on temperature and moisture. Soil Biol. Biochem. **31**: 327-335.

Little, T.I., Pluth, D.J., Corns, I.G.W. and Gilmore, D.W. 2002. Post-fire forest floor development along toposequences of white spruce-trembling aspen mixedwood communities in west-central Alberta. Can. J. For. Res. **32**: 892-902.

MacLean, D.A., Woodley, S.J., Weber, M.G. and Wein, R.W. 1983. Fire and nutrient cycling. *In* The role of fire in northern circumpolar ecosystems. *Edited by* R.W. Wein and D.A. MacLean. John Wiley & Sons Ltd., New York. 111-132.

Miyanishi, K. and Johnson, E.A. 2002. Processes and patterns of duff consumption in the mixedwood boreal forest. Can. J. For. Res. **32**: 1285-1295.

Monleon , V.J., Cromack, K. Jr. and Landsberg, J.D. 1997. Short- and long-term effects of prescribed underburning on nitrogen availability in ponderosa pine stands in central Oregon. Can. J. For. Res. **27**: 369-378.

Nadelhoffer, K.J. and Aber, J.D. 1983. Leaf litter production and soil organic matter dynamics along a N availability gradient in southern Wisconsin (U.S.A.). Can. J. For. Res. **13**: 12-21.

Nardoto, G.B. and Bustamante, M.M. da C. 2003. Effects of fire on soil nitrogen dynamics and microbial biomass in savannas of central Brazil. Pesq. Agropec. Bras. **38**: 955-962.

Neary, D.G., Klopatek, C.C., DeBano, L.F. and Ffolliott, P.F. 1999. Fire effects on belowground sustainability: a review and synthesis. For. Ecol. Manage. **122**: 51-71.

Offord, J.L. 1999. Nitrogen Mineralization Patterns of Boreal Forest Soils. M.Sc. Thesis. Department of Renewable Resources, University of Alberta, Edmonton, AB.

Peters, T.W. 1981. Reconnaissance soil survey of the Brazeau Dam area. Alberta Soil Survey Report No. 40. Alberta Institute of Pedology Report S-81-40. Pedology Consultants and Agriculture Canada. Land Resources Research Institute, Agriculture Canada, Ottawa, ON.

Pietikäinen, J. and Fritze, H. 1993. Microbial biomass and activity in the humus layer following burning: short-term effects of two different fires. Can. J. For. Res. **23**: 1275-1285.

Pietikäinen, J. and Fritze, H. 1995. Clear-cutting and prescribed burning in coniferous forest: comparison of effects of soil fungal and total microbial biomass, respiration activity and nitrification. Soil Biol. Biochem. **27**: 101-109.

Pinay, G., Clement, J.C. and Naiman, R.J. 2002. Basic principles and ecological consequences of changing water regimes on nitrogen cycling in fluvial systems. Env. Manage. **30**: 481-491.

Prieto-Fernández, A.,Villar, M.C., Carballas, M. and Carballas, T. 1993. Short-term effects of a wildfire on the nitrogen status and its mineralization kinetics in an Atlantic forest soil. Soil Biol. Biochem. **25**: 1657-1664.

Prieto-Fernández, A., Acea, M.J. and Carballas, T. 1998. Soil microbial and extractable C and N after wildfire. Biol. Fertil. Soils. **27**: 132-142.

Raison, R.J. 1979. Modification of the soil environment by vegetation fires, with particular reference to nitrogen transformations: a review. Plant Soil. **51**: 73-108.

Raison, R.J., Khanna, P.K. and Woods, P.V. 1985. Mechanisms of element transfer to the atmosphere during vegetation fires. Can. J. For. Res. **15**: 132-140.

Romanyà, J., Casals, P. and Vallejo, V.R. 2001. Short-term effects of fire on soil nitrogen availability in Mediterranean grasslands and shrublands growing in old fields. For. Ecol. Manage. **147**: 39-53.

Ste-Marie, C. and Paré, D. 1999. Soil, pH, and N availability effects on net nitrification in the forest floors of a range of boreal forest stands. Soil Biol. Biochem. **31**: 1579-1589.

Shaw, R.D., Trimbee, A.M., Minty, A., Fricker, H. and Prepas, E.E. 1989. Atmospheric deposition of phosphorus and nitrogen in central Alberta with emphasis on Narrow Lake. Water Air Soil Pollut. **43**: 119-134.

Soil Classification Working Group. 1998. The Canadian System of Soil Classification. Agriculture and Agri-Food Canada Publication 1646. NRC Research Press, Ottawa, ON.

Stottlemyer, R. and Toczydlowski, D. 1999a. Seasonal relationships between precipitation, forest floor, and streamwater nitrogen, Isle Royale, Michigan. Soil Sci. Soc. Am. J. **63**: 389-398.

Stottlemyer, R. and Toczydlowski, D. 1999b. Nitrogen mineralization in a mature boreal forest, Isle Royale, Michigan. J. Env. Qual. **28**: 709-720.

The National Atlas of Canada. 1974. Average annual potential evapotranspiration. 4th Edition [online]. Available from http://atlas.gc.ca/site/english/maps/archives/4thedition/environment/climate/049_50?l=3&r=0&c=0#map [accessed July 6 2004]

Walley, F.L., Van Kessel, C. and Pennock, D.J. 1996. Landscape scale variability in N mineralization in forest soils. Soil Biol. Biochem. **28**: 383-391.

Whitson, I.R., Chanasyk, D.S. and Prepas, E.E. 2004. Patterns of water movement on a logged Gray Luvisolic hillslope during the snowmelt period. Can. J. Soil Sci. **84**: 71-82.

Wilson, C.A., Mitchell, R.J., Boring, L.R. and Hendricks, J.J. 2002. Soil nitrogen dynamics in a fire-maintained forest ecosystem: results over a 3 year burn interval. Soil Biol. Biochem. **34**: 679-689.

Zackrisson, O., Nilsson, M.C. and Wardle, D.A. 1996. Key ecological function of charcoal from wildfire in the Boreal forest. Oikos **77**: 10-19.

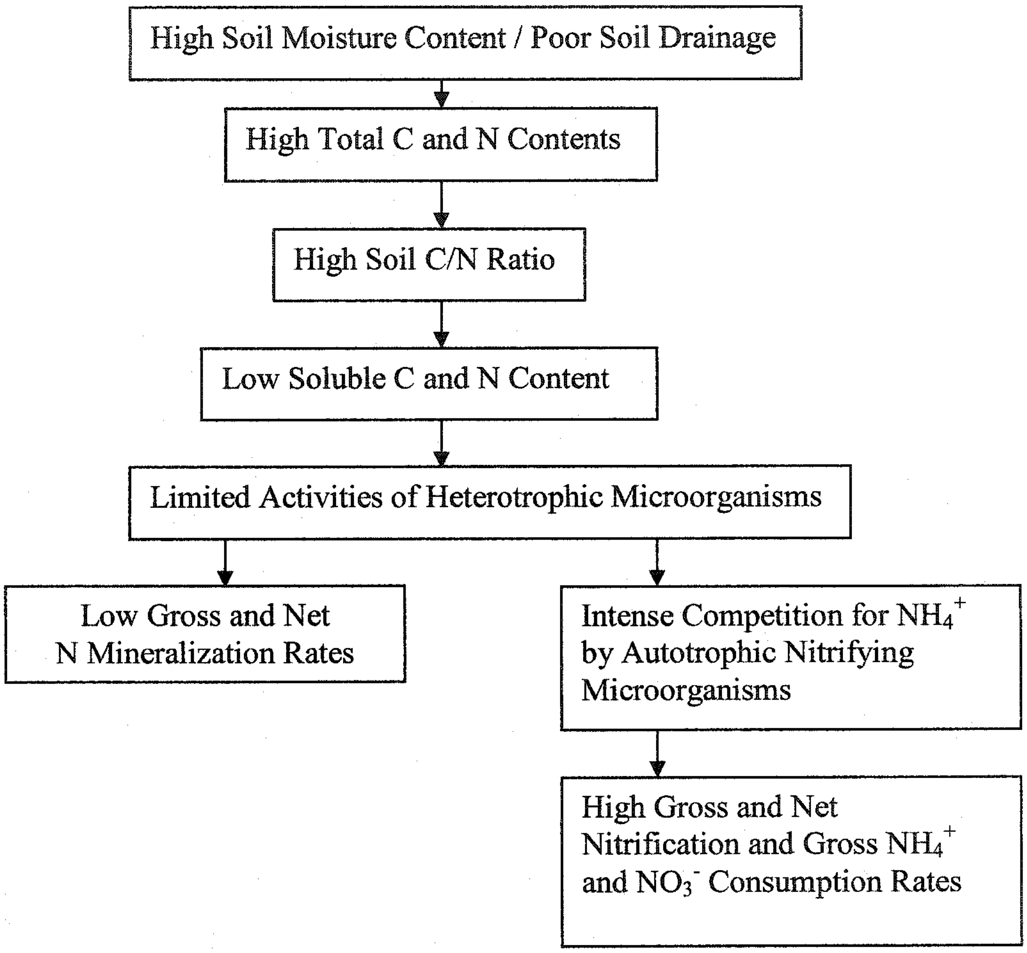
Zaman, M. and Chang, S.X. 2004. Substrate type, temperature, and moisture content affect gross and net N mineralization and nitrification rates in agroforestry systems. Biol. Fertil. Soils. **39**: 269-279.

5. GENERAL DISCUSSION AND CONCLUSIONS

The main objectives of this study were to evaluate the effect of variation in moisture content due to slope position, the short-term effect of an experimental understory burn, and the interaction of these two factors on the N cycle in a mature boreal mixedwood stand. Factors that are known to influence the N cycle (soil temperature, soil moisture content, soil pH, and soil C/N ratio) in addition to commonly assessed components of the N cycle (total, extractable and microbial biomass C and N contents, NH_4^+ and NO_3^- contents, net N mineralization and nitrification rates, gross mineralization, immobilization, and nitrification rates, foliar % N, and total plant N content) were evaluated in a trembling aspen dominated riparian mixedwood stand in the Western Boreal Forest (WBF).

Soil moisture content was only significantly higher in the riparian position compared with the upland position within the stand in July. The riparian position tended to have a higher volumetric moisture content than the upland position in the mineral soil (5 cm) during the measurement period (July to September 2003) and the riparian position contained indicator species (*Gymnocarpium dryopteris*, *Equisetum* spp., and *Heracleum lanatum*) that suggested moist conditions and ephemeral flooding. Large heterogeneity in soil characteristics (e.g. forest floor depth and drainage class) between the riparian plots allowed few significant differences between positions; only EC content in the forest floor in August, September and October and in the mineral soil in August and September, and microbial C/N ratio in the forest floor in October differed significantly between the riparian and upland positions. However, the data suggested that high moisture content and poorer soil drainage in the riparian position was positively correlated to the soil C/N ratio and that lower substrate quality resulted in a lower rate of gross N mineralization and higher rates of net and gross nitrification in the RC plots. These results indicate that poor soil drainage and high soil moisture content in the lower slope position of a hillslope indirectly affect the N cycle through their effect on substrate quality; this conceptual model is presented in Figure 5.1.

Figure 5.1. A conceptual model illustrating the effect of high soil moisture content in the riparian, lower slope position of a hillslope on the N cycle.



Monthly variations in the net N mineralization rate suggested that when substrate quality remained constant, soil temperature and soil moisture content interacted to control N availability. In this study, the highest rate of net N mineralization in July may indicate that during this period soil temperature and soil moisture conditions were the most optimal for microbial activities while sub-optimal soil temperature and soil moisture content in August, September, and winter may have limited the net N mineralization rate (Offord 1999). Positive net N mineralization rates during the winter (October to April) indicated that in the WBF, available N under snow cover may provide a significant N source for spring plant growth; this hypothesis requires further investigation.

Burning with propane torches under an intact forest canopy had its most significant effect on the forest floor (LFH); total C and N, MB-C, MB-N and EC contents, MB-N/TN and microbial C/N ratios, and the cumulative net N mineralization rate were significantly lower and pH_{CaCl2} , NH_4^+ content in July and August, and the IN/EN ratio were significantly higher in the forest floor of the burned plots compared with that of the control plots. In the mineral soil (0-10 cm), only soil temperature, the IN/EN ratio, and NH_4^+ content in July and August were significantly higher in the burned as compared with the control plots. In the first month after burning, the net N mineralization rate in the forest floor was similar in the burned and control plots, possibly due to stimulation of microbial activity by soluble C and N compounds released from combusted organic matter and lysed microbes (Almendros et al. 1990). After the first month, the net N mineralization rate was significantly lower in the burned compared with the control plots; this change may be related to the post-fire decline in substrate quality and exhaustion of readily metabolizable C and N compounds (Almendros et al. 1990). There was also a tendency towards higher rates of net and gross nitrification in the burned plots; this trend has been previously attributed to stimulation of nitrifying microorganisms by high pH and reduced plant competition for NH_4^+ (Bauhus et al. 1993; Kaye and Hart 1997).

Although no significant position by burn interactions were identified, trends in forest floor total and microbial biomass C and N contents, pH_{CaCl2}, NH_4^+ content, and net N mineralization rate and mineral soil temperature suggested greater fire severity in the UB as compared with the RB plots as was indicated by the depth of burn. Depth of forest floor combustion was positively correlated to post-fire forest floor pH_{CaCl2}, total and microbial biomass C and N contents, NH_4^+ content, net N mineralization rate and mineral soil temperature. Overall, the trends suggest a conceptual model of the influence of forest floor and soil moisture content on short-term changes in the N cycle after burning that is summarized in Figure 5.2.

The short data collection period, low within-stand replication, and lack of inter stand replication limit the general conclusions that can be drawn from this study. However, it appears that substrate quality, which may have been indirectly affected by moisture content, exerted the greatest control over mineralization and nitrification processes between slope positions within this mature trembling aspen-dominated mixedwood stand. The tendency towards greater nitrification and NO_3^- pools in the lower slope, riparian position suggests that investigation of N movement from these riparian zones into ephemeral streams during flood events merits further attention. In addition, further study of the soil N cycle in ephemeral riparian zones should examine the importance of N inputs from groundwater, floodwater, runoff, and nitrogen fixing species for plant growth. Seasonal trends suggested that net N mineralization rates were highest when environmental conditions (e.g. soil temperature and soil moisture content) were optimal for microbial activity, and that mineralization of N during the winter was positive. Assessment of the importance of winter N mineralization for spring plant growth requires further exploration. One month after the experimental burn was conducted, the net N mineralization rate was lower in the burned compared with the control plots in both slope positions. Therefore, it appears that the short-term effect of fire may be to reduce N availability in trembling aspen dominated mixedwood stands in the WBF. Further research on short-term change after fire in microbial community composition and its relation to altered humus quality, the relative influence of soil temperature, humus quality, and microbial population structure on net and gross N mineralization rates, and the consequence of a lower net N mineralization rate for plant growth is recommended.

5.1. References

Almendros, G., Gonzalez-vila, F.J. and Martin, F. 1990. Fire induced transformation of soil organic matter from an oak forest: an experimental approach to the effects of fire on humic substances. Soil Sci. **149**: 158-168.

Bauhus, J., Khanna, P.K. and Raison, R.J. 1993. The effect of fire on carbon and nitrogen mineralization and nitrification in an Australian forest soil. Soil Biol. Biochem. **31**: 621-639.

Kaye, J.P. and Hart, S.C. 1997. Competition for nitrogen between plants and soil microorganisms. TREE. **12**: 139-143.

Offord, J.L. 1999. Nitrogen Mineralization Patterns of Boreal Forest Soils. M.Sc. Thesis. Department of Renewable Resources, University of Alberta, Edmonton, AB.

Figure 5.2. A conceptual model that demonstrates the influence of forest floor and soil moisture content on the short term (< 1 year) effect of burning on the N cycle.