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Effects of selection harvest and prescribed burning on nitrogen availability in the mineral soil under ponderosa pine

Kris Zouhar
The University of Montana

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Effects of Selection Harvest and Prescribed Burning on Nitrogen Availability in the Mineral Soil Under Ponderosa Pine

by

Kris Zouhar

presented in partial fulfillment of the requirements

for the degree of

Master of Science

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Approved by:

[Signatures]

Chairperson,
Dean, Graduate School

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Prior to 1900 ponderosa pine forests experienced non-lethal fires at intervals of 13-50 years resulting in what has been described as open, park-like, uneven-aged stands dominated by ponderosa pine. Today, as a result of reduced fire frequency and selective logging, these stands are characterized by thickets of small firs, impacted by insects and disease and increasingly susceptible to stand destroying wildfires. Efforts to restore these ecosystems to pre-1900 stand densities, structures and species compositions are being attempted using selection cutting and underburning. There is currently little information available regarding treatment effects on long-and short-term soil nutrient availability. In this study, we compared the effects of single tree selection cutting and prescribed underburning on available N, N mineralization potential, and microbial biomass and activity, at two depths in the mineral soil under ponderosa pine, 1, 2, 3, 11 and 12 years following disturbance at two separate sites in western Montana. Both study sites consist of a control, a selection cut without burn, and selection cut(s) with underburning. Composite samples were taken from 0-7 cm and 7-15 cm depths at each treatment plot during the summer and fall of 1995, and the spring and summer of 1996. Samples were analyzed fresh for N mineralization potential, microbial biomass, soil respiration rate, amino-N, NH$_4^+$-N, NO$_3^-$-N, and pH. Extractable inorganic N was increased by burning in the 1 year burn plots, was no different from control in the 2 year burn plot, and was lower than the control 11 years after burning in both summer and fall of 1995, although variation within treatments was greater in the fall. In contrast, the N mineralization potential and microbial activity were reduced in burn plots at 1, 2, and 11 years following treatment in both summer samplings. The fall sampling showed a recovery of N mineralization potential in all burn plots. Spring samples had higher magnitude in all measures on all plots, but greater variability within treatments. The cut only plots were similar to control plots on all sample dates. Long-term N availability may be significantly reduced by the combination of harvest and prescribed burning.
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CHAPTER 1: INTRODUCTION

Pre-1900 conditions in ponderosa pine/Douglas-fir forests in western Montana have been described as open, park-like, uneven-aged stands dominated by ponderosa pine (Habeck 1990, Gruel 1983). Today, largely as a result of reduced fire frequency and selective logging, these ecosystems are characterized by thickets of small firs, impacted by insects, disease and stand destroying wildfires (Arno et al. 1995, Fiedler et al. 1992). Methods to restore “ecological structure and process” include silvicultural prescriptions designed to emulate natural disturbance regimes (Arno 1996), and may include prescribed fire (Harrington 1996). Target stands are defined by specific numeric designations of stand density, structure, and species composition based on those that existed in the past according to historical descriptions, old photos, and forest inventory records (Fiedler 1996).

Along these lines, forest management should be based on an understanding of disturbance processes (Rogers 1996). The importance of disturbance has long been recognized, although there has been resistance to maintaining or simulating natural disturbance regimes in managed landscapes until the advent of the ecosystem management approach (DellaSala et al. 1996). Ecosystem management requires “our best understanding of the ecological interactions and processes necessary to sustain ecosystem composition, structure and function” (Christensen et al. 1996). Cook (1995) concluded that most sites will never support a climax stage and that disturbances, past land use, and stochastic events can lead to multiple pathways on a single “type” of site.
Disturbance agents such as fire and logging activities impact the forest in a number of ways including roads, skid trails, invasions of exotic plants, slash piles and spilled fuel, changes in vegetative succession, and disturbed soils. Impacts to soil include compaction, disruption, and removal and redistribution of surface organic matter and mineral soil, resulting in changes in the habitat and activities of soil flora and fauna. The “activities” of soil organisms include the breakdown of organic materials and subsequent mineralization and recycling of nutrients to the living biomass.

When comparing the current, “unhealthy” condition, to the past, “healthy” condition of southwestern ponderosa pine forests, Covington et al. (1997) assume that the rates of decomposition, nutrient cycling, and net primary production will be higher in an ecosystem that approximates the natural structure and natural disturbance regime extant in “presettlement” times. If we choose to accept this assumption, then evaluating the effects of soil disturbance in terms of decomposition rates and nutrient cycling rates is a critical part of an evaluation of a treatment designed to improve forest health. More generally, a healthy forest should have healthy soils. Soil quality and soil health have been defined as “the capacity of a soil to function within ecosystem boundaries to sustain biological productivity, maintain environmental quality, and promote plant and animal health” (Doran and Parkin 1994). There is a growing consensus of the usefulness of biological properties as indicators of soil quality, or soil “health” (e.g. Karlen et al. 1997, Pankhurst et al. 1995, Gregorich et al. 1994). Particularly promising are properties associated with soil organic matter and its decomposition (Gregorich et al. 1994).

Of particular interest for crop production is the cycling and availability of nitrogen. Nitrogen availability is a good indicator of the impact of land use on soil
quality and productivity, because it is dependent on the complex interactions of a number of biotic and abiotic factors. Nitrogen availability is inextricably linked to the organic matter component of the soil, including the micro and macro flora and fauna, both as a source and sink of N, and as a provider of other beneficial properties and nutrients that promote N cycling. Therefore, one must consider a number of parameters in an assessment of the effects of disturbance on N availability.

As part of the Bitterroot Ecosystem Management Research Project, different silvicultural and prescribed fire treatments for restoring ponderosa pine forests to presettlement conditions were established at Lick Creek Resource Demonstration Area in western Montana (Arno 1996). This project, along with an older example of the same silvicultural prescription at E/L Ranch, provided an opportunity to examine the effects of selection cutting and prescribed underburning on selected N availability indices. Potentially mineralizable N (PMN), microbial biomass N, 3-day soil respiration, inorganic and amino N, and soluble anthrone reactive carbon (ARC) were analyzed seasonally in the mineral soil on control plots and on harvested plots treated with and without prescribed fire, for a one-year period.

Because our understanding of the processes and mechanisms involved in N mineralization and availability in forest soils is incomplete, a literature review of the pools, processes and disturbance effects on N availability is included in Chapter 2. The final chapter is a description of the objectives, methods and results of the Lick Creek, E/L Ranch study.
Overview

Nitrogen availability limits production in many forest ecosystems (Keeney 1980, Binkley and Hart 1989). The concept of availability can be defined as the rate at which N is converted into forms usable by plants, or as the degree to which plant production is constrained by a limited N supply (Binkley and Hart 1989). In agroecosystems, N supply rate and N limitation are often closely linked (Binkley and Hart 1989). In forest ecosystems, indices of N availability have had mixed success in predicting growth responses. This is due, in part, to differences in the components of the N cycle each method assesses (Binkley and Hart 1989). It is also due to differences in the relative sensitivity of each method to the factors influencing soil N transformations (Binkley and Hart 1989). Soils in general, and particularly forest soils, are notoriously heterogeneous (Keeney 1980, Wollum 1994, Walley et al. 1996), therefore care must be taken in sampling design and interpretation of results. In forest ecosystems, the availability of N in the forest floor, the nonuniform rooting depth and distribution of trees, and the changes in nutrient supply and demand over the long life of the crop need to be considered.

Raison and Stottlemyer (1991) list important features to consider when modelling changes in the N cycles of temperate forests. They include: atmospheric inputs, N fixation, litter transfers and decomposition, soil processes, N uptake and effects on productivity and litter quality, and N outputs, as well as aspects of the carbon (C) and water cycles. They go on to say that a generic model of forest N cycling is not currently
possible because the links and interactions among the individual processes are not well understood. Clearly, assessment of N availability in forests is neither simple nor straightforward.

Traditionally, available N has been represented by the size of the inorganic N pool over short time scales (within one day). Recently, more attention has been paid to the importance of amino and dissolved organic N and the ability of some plants to utilize these forms either directly or via association with mycorrhizal fungi (e.g. Northup et al. 1995). For the purposes of this discussion, we will assume N availability to be regulated by the size of the inorganic pool, and the fluxes (i.e. mineralization and immobilization) into and out of these pools.

It is useful to look at a conceptual model of the N cycle in terms of pools and flows when addressing the applicability of various availability indices. The model shown in Figure 1 (adapted from Binkley and Hart 1989) depicts pools of soil N in boxes and flows between pools as arrows.

These pools and processes make up the internal N cycle: The inorganic pools \( \text{NO}_3^- \text{ and } \text{NH}_4^+ \), the sizes of which are influenced by the size and composition of the other pools as well as environmental factors. The microbial pool represents an active phase of soil organic matter. The size of this pool is small relative to the fluxes through it, which are up to five times faster than other organic N pools. The labile pool is also an active pool, with shorter residence times and more readily decomposable than the less labile pool. The recalcitrant pool is characterized by humified organic compounds with a long mean residence time such that it is essentially uncoupled from the inorganic N pools on an annual time scale (Binkley and Hart 1989).
Figure 1. Conceptual framework of the components affecting N availability in forest soils. Adapted from Binkley and Hart (1989).

The absolute amount of organic N in the materials ranging from fresh plant litter, to soil microbes, to humic substances is influenced by climate, topography, vegetation, parent material and time, including disturbance history. The rates of transformations within the internal cycle are influenced by the chemical composition of the organic pools, the spatial compartmentalization of these pools, and temperature and moisture (Binkley and Hart 1989). The chemical composition of the pools is directly affected by the vegetation and by the chemical composition and rates of deposition and decomposition of litter and root exudates.
The first part of this literature review is a discussion of vegetation, litter and soil organic matter (SOM) with respect to their influence on the pools and flows of the internal N cycle, and their response to disturbance. This is followed by a discussion of some of the pools and flows of the internal N cycle, how they relate to N availability, how they are measured, and how disturbance might effect them.

**Vegetation**

The types and amounts of vegetation on a site affect the types and amounts of SOM and subsequent N availability at that site. Feedbacks exist in ecosystems in which the nature of the vegetation present can have a large effect on N supply rate, and N supply rates can, in turn, strongly influence vegetation composition (Wedin and Tilman 1990, Vitousek 1982). Wedin and Tilman (1990) tested the effects of five different grass species on net N mineralization in different soil types. They found that by the third year, identical soils under different species had diverged up to 10-fold in annual net mineralization, and that this divergence corresponded to differences in the tissue N concentrations, below ground lignin concentrations, and below ground biomasses of the species (Wedin and Tilman 1990). They cite this as an example of a positive feedback in which a particular species helps to create an environment in which it has a competitive advantage (i.e. the species that causes low N supply rates via poor litter quality is also the stronger competitor for N) (Wedin and Tilman 1990). Conversely, examples of negative feedbacks involving N-fixing species also exist, in which N-fixers improve the availability of N in the system through deposition of N rich litter, thus decreasing the competitive advantage that they have in situations of low N availability.
Consider the work of Northup (1995) which showed that as you move down an extreme soil acidity / fertility gradient, the main form of N released upon incubation of litter is dissolved organic N. This would facilitate recovery of N for plants capable of utilizing organic N via mycorrhizal associations, which are common in ericaceous shrubs. This capability would add to the competitive ability of these plants in an environment of low SOM quality where they may be able to outcompete the trees for N in their own litter.

Kimmins (1994) indicated that mosses, herbs and shrubs can play major, but differing roles in nutrient circulation in a forest ecosystem, depending on site and stage of development. For example, they may slow down organic matter decomposition and nutrient cycling, compete with trees for nutrients and moisture, or they may promote more rapid nutrient circulation. Understory vegetation is often viewed as competition rather than companion to the crop. This is due to numerous studies which have demonstrated that vegetation control (i.e. pesticides, burning, or scarification) improved growth rates in the early stages of stand (Beets et al. 1994). Morris and Miller (1994) point out that all available data show that early growth of planted seedlings is greatest on sites where biomass removals were greatest and where organic and mineral soil horizons were most severely disturbed. However, these effects are of short-term duration, as most height growth advantage occurs before age 6 and reduces as the forest matures.

In a long-term study (following 35 years of growth) in Oregon, Busse et al. (1996) showed that while ponderosa pine growth was reduced in the presence of understory vegetation in the first 12-20 years, during the last 15 years growth was better in stands with understory. They also found higher levels of soil C, N, and microbial biomass in the
presence of understory vegetation, which they speculate may have contributed to difference in growth. This demonstrates that in considering time scales of ~120 years rotation length, long term changes in vegetation and their effects on nutrient cycling need to be considered.

The effects of early successional species on N availability to later successional species are poorly understood. Several studies have shown improved growth in crops grown with or after N-fixing species (Morris and Miller 1994). Schimel et al. (1996) focused on understanding the specific mechanism by which poplar tannins and phenolics affect microbial activity and subsequent N cycling in river floodplains in Alaska, where the distribution of plant communities is controlled by disturbance coupled with a consistent successional pattern. Specifically, the transition between alder and poplar is characterized by a shift from an open N cycle, with rapid N fixation, mineralization, and nitrification to a closed cycle dominated by mineralization and immobilization of NH$_4^+$. This shift is attributed to changes in litter quality, specifically to concentrations of secondary chemicals (tannins and phenolics) in the poplar foliage. By inhibiting mineralization and stimulating immobilization, poplar secondary compounds may reduce soil N-availability during the transition between alder and poplar successional stages. This has implications with regard to mutualistic relationships of plant populations and physiologies, without the level of specificity found in symbiosis (Schimel et al. 1996).

In fire-adapted ecosystems, microenvironmental changes due to fire initiate successional changes often favoring N-fixing plants and microbes. Symbiotic N fixation by Alnus, Ceanothus, and legume species may be a significant N source after a fire. This may help explain why an actual decrease in site N is not always shown even though N is
volatilized during burning (Wells et al. 1979). Although the N that is fixed is used by the plant, the plant becomes part of the organic matter pool via litterfall and plant senescence, and thus may provide a significant contribution to the N budget.

While effects of one species on another is often explained by differences in litter quality (e.g. Schimel et al. 1996), fewer studies have looked at the effects of variations in root exudation on soil nutrient cycling. For example, paper birch has a well established reputation as a soil improving species, although the mechanism has not been identified. Bradley and Fyles (1995) found that rhizosphere activity was one order of magnitude higher in the rhizosphere of paper birch seedlings as compared with five other tree species. They suggested that high amounts of root labile C compounds in conjunction with rapid mineral-N uptake by birch roots can stimulate microbial communities to acquire nutrients from the native soil, thus increasing the pool of available soil N in the rhizosphere (Bradley and Fyles 1995). They questioned the importance and/or elasticity of the C:N alone in predicting N mineralization rates.

In summary, when the plant community changes, so does the N status of the site. This is more complicated than can be described by C:N and lignin:N (Prescott 1996). Secondary chemistry, rhizosphere dynamics and questions of competitive advantage and mutualism all come into play. These ideas and studies have implications in disturbance and successional ecology. For example, biological invasions by exotic species can alter properties of whole ecosystems through changes in nutrient cycling and hydrology and subsequent changes in productivity (Vitousek 1990). Similarly it has been suggested that changes in native plant community compositions due to management may
also have implications with regard to N availability at the ecosystem or landscape scale

**Litter**

Changes in plant community composition, growth and nutrition, and changes in
nutrient distributions within trees as they age (a larger proportion of the resource is
diverted to the production of N-free material required for structural support) affect N
cycling by affecting litter quantity and chemical composition. The undisturbed forest
floor is an important reservoir of N. It typically consists of layers of organic debris (plant
and animal residues, litter, wastes, root exudates) composed of various organic
compounds, deposited upon and within the mineral soil and subsequently broken down
and incorporated into various pools of soil organic matter (SOM) by soil animals and
microorganisms. The result is an organic horizon with a litter layer (O<sub>l</sub>) only slightly
decomposed, over a fermentation layer (O<sub>e</sub> or duff) that is permeated with fungal mycelia
and fine roots and where decomposition is very active, over a humus layer (O<sub>j</sub>) that is
unrecognizable, dark brown or black, amorphous organic material. These O horizons
overly the A horizon where the bulk of the remaining SOM resides in combination with
mineral soil.

Soil animals such as mites, worms, termites, ants, bark beetles, and wood borers
play important roles in the first steps of decomposition (Edmonds 1991). Fresh litter
falling to the forest floor is also decomposed by fungi and bacteria, which typically
immobilize any N in the decomposing substrate for a period of time before releasing it to
the soil where it becomes available for plant uptake (Edmonds 1991). The length of this
period of immobilization depends on the concentration of N in the initial substrate and may range from a few minutes (red alder leaves) to over 100 years (logs) (Edmonds 1991). At C:N of <20:1, net N mineralization should occur (Stevenson 1986), and this appears to be the case for leaf and needle litter, but it does not appear to be the case for woody substrates which may have critical C:N >100:1 (Edmonds 1991). This indicates that the critical C:N for N release may increase as the substrate decomposition rate decreases. Hart and Firestone (1991) observed mean residence times of 13 and 34 years for organic matter and N in the forest floor, respectively, suggesting that that forest floor layer is a site of net N immobilization in that ecosystem (mixed conifer forest in California). Preston and Mead (1994) used $^{15}$N to label foliage of Douglas-fir to follow the transformations of litterfall-N in a forest ecosystem. They found that after 7.5 years, approximately one-third of the $^{15}$N was found in mineral soil horizons. Because the litter layer is usually considered a site of net N immobilization it is rarely sampled or considered in N mineralization studies, except, perhaps to estimate stock of organic N on site.

Hart et al. (1992) observed litter decomposition in a 9-year-old clearcut (slash raked and burned) and an old growth forest using reciprocal litter treatments to try to separate changes in quality from changes in microenvironment. Although the old-growth litter was initially lower in indices of litter quality, there was no difference in decomposition rates between the two when placed at either site. These results suggest that changes in litter quality within a tree species that occur during forest development may not be responsible for decline in decomposition rates observed in mature relative to young forests (Hart et al. 1992). However, decomposition rates of both age classes were
lower at the old-growth site, suggesting that differences in microenvironmental factors between sites might account for lower rates of decomposition in the old-growth site relative to the young-growth site (Hart et al. 1992).

The activity of the decomposer organisms is limited in cool temperate regions, so rates of deposition often exceed rates of decomposition and organic materials accumulate over time. In the case of fire adapted ecosystems, fire may act as a decomposing agent (Habeck and Mutch 1973). In a recent study Monleon and Cromack (1996) looked at the effects of low-intensity burning on the rates of litter decomposition and N and P release in ponderosa pine stands in central Oregon on ash soils 0.3, 5, and 12 years after burning. Prescribed burning significantly increased the release of N and P from the litter on the sites burned 5 years earlier, while it reduced decomposition rates on sites burned 0.3 and 12 years earlier (Monleon and Cromack 1996). These short-term (0.3 years) results are contrary to what one might expect shortly after a fire, since needles that return to the litter due to scorching by fire have a higher N content than needles dropped naturally. Natural litterfall is preceded by a retranslocation of nutrients from the needles into the tree before the needles fall. Grier (1989) measured needle litter fall and % N in needle litter fall one year following the prescribed burn. The highest needle fall and %N were found in the heavy burn, followed by the light burn followed by the control. There was nearly three times as much N in the needle fall in the heavily burned plot.

Aber et al. (1990) noted that decomposition in litter and in soil organic matter (SOM) have been studied as separate processes in forest ecosystems, using different techniques, and that few attempts have been made to link these two. This is because most litter decay research is short-term. Aber et al. (1990) present long-term (77-month) decay
data and describe two phases of decomposition. They conclude that litter chemistry and/or short-term litter decay rates can be used to predict the chemistry and length of time necessary to convert litter into SOM (Aber et al. 1990). Initial litter chemistry (i.e. lignin:N ratio) has been looked at for the prediction of decomposition rates, with mixed results (e.g. Melillo et al. 1982 and McLaugherty et al. 1985). Stump and Binkley (1993) found a relationship between initial litter chemistry (lignin:N and C:N) and net N mineralization, as well as a relationship between N released from litter decomposition and total (mineral soil plus forest floor) net N mineralization.

Many studies (e.g. Edmonds 1991, Hart and Firestone 1991, Klemmedson et al. 1985) look at the decomposition rates of litter and twigs and branches of the dominant tree species, because nutrient cycling in forest soils is generally considered to be characteristic of the dominant plant species (Bradley and Fyles 1995). Lefevre and Klemmedson (1980) and Klemmedson (1994) looked at differences in ponderosa pine with and without New Mexico locust and Gamble oak (respectively) and found that with increasing oak or locust in the stand, percentage of N increased in the forest floor and mineral soil. Clearly, changes in vegetation on a site will affect the quality and subsequent rate of decomposition of organic matter deposited on and in the soil environment.

Disturbances such as fire and harvest affect not only vegetative succession, but they can also directly effect the distribution of organic matter in the surface O and A horizons. The effects of harvest on the quantity and distribution of SOM depends on the amount of biomass removed, the amount left on site, the type of site preparation used,
and the distribution and severity of surface disturbance. Fire will remove some amount of the surface organic matter depending on fire severity.

**Soil Organic Matter**

Soil organic matter is present in the O<sub>e</sub>, O<sub>a</sub>, and A horizons, although the O and A horizons are often studied independently. Soil organic matter augments soil structure, nutrient storage, and biological activity, which in turn affects N cycling (e.g. Page-Dumroese et al. 1991). In agricultural systems, Stevenson (1986) discussed the value of keeping a rotating fund of decomposable organic matter in the soil through the frequent and periodic return of crop residues (including legumes). The beneficial effect is due to the maintenance or increase of organic matter levels thereby insuring an adequate supply of N by slow decomposition of stable organic matter over time (Stevenson 1986). In a forest ecosystem, this rotating fund of decomposable organic matter comes from litter deposition, and root turnover and exudation, and is critical to site productivity, especially on droughty, coarse textured soils. Studies measuring growth of ponderosa pine trees in response to harvest and site preparation treatments in the intermountain west show that seedlings grow better when ample organic matter is left in place (Harvey et al. 1988).

The N containing fraction of SOM is composed primarily (90-95%) of protein and nucleic acids, around 5% soluble amino compounds, and less than 2% inorganic molecules (Tate 1995). N dynamics in SOM can be estimated by the mean residence time (MRT) of N in the various pools or by the rates of fluxes into and out of these pools.

Pools of SOM defined in models (e.g. figure 1) are often conceptual and have eluded definition by physical, chemical or biological methods (Jarvis et al. 1996). They
are linked by a number of interacting, competing and sometimes antagonistic processes, and are stabilized against mineralization to varying degrees by molecular recalcitrance, physical separation from the soil microbial biomass (SMB), and/or direct association with inorganic ions and clay surfaces (Jarvis et al. 1996). For the purposes of the prediction of N availability over the short to mid term, the pools with mean residence times (which reflect the properties of the substrate and its interaction with the ambient environment) in accord with the time period of interest could be considered.

**Total N**

Total N includes the inorganic, microbial, labile, less labile and recalcitrant pools combined. At a regional scale, where differences in total N are large, available N may be indexed adequately by measures of total N (Binkley and Hart 1989). On smaller scales (e.g. within a forest stand), where difference in total N are small, differences in substrate quality (e.g. C:N, lignin:N) and microenvironment may be more important for indexing N availability (Binkley and Hart 1989).

Quantities of total organic N and C in SOM are determined by methods with longstanding use and acceptance. According to Stevenson (1986), the N content as reflected in the C:N of the soil organic matter is of primary importance in regulating the magnitude and direction of mineralization-immobilization, where: <20 = mineralization, 20-30 = balance, >30 = immobilization. The time required to reach a C:N favorable for mineralization varies with the amount of organic matter, lignin content, degree of comminution and level of respiration of the soil microflora (Stevenson 1986).
The C:N of decomposing substrates continuously declines with time and with depth in the soil profile (Edmonds 1991). This is because during decomposition, part of the C is lost as CO2, while N tends to be conserved, particularly when C:N is initially high, resulting in a lowering of C:N as decomposition proceeds. Increasingly lower C:N suggests more readily mineralizable substrates, but these materials may also become more humified. The net effect of humification is condensation of the N of amino acids into the complex structures of humic and fulvic acids (Stevenson 1986). Humified materials in this recalcitrant pool are generally less mineralizable, with a long MRT (~250-2500 years). With a large portion of organic N in this pool, measures of total N include a large amount of N that is essentially uncoupled from the processes of mineralization and immobilization on annual time scales (Binkley and Hart 1989). The assumption is often made that only 1-3% of the total soil organic N is mineralized during the course of the growing season (Stevenson 1986).

Some research on the effects of disturbance on soil N have used measures of total C and N as indicators of change. In a recent study at Hubbard Brook, Johnson (1995) addressed changes in C:N, C:organic matter, and N:organic matter in the mineral soil and organic horizons as indicators of changes in the decomposition regime and in the properties of soil organic matter, 8 years after whole-tree clear-cutting. Total N in organic and mineral horizons on clear-cut plots averaged 17% lower 8 years after logging, while the mineral soil N pool alone remained unchanged. The C:N and C:OM decreased in mineral soil suggesting increased rates of decomposition. The N:OM ratio was unchanged in the mineral soil, but increased in the O1 + O2 horizon. This increase was likely attributable to the dominance of pin cherry and raspberry, both of which have
relatively high tissue N concentrations, in the understory after harvest. This study concluded that soil nitrogen pools are far from uniform, that losses were greatest in the steepest part of the watershed, and that preservation of organic matter is likely to promote N retention.

In another mixed conifer forest in California, Black and Harden (1995) constructed a recovery sequence using a range of evenly aged timber stands on soils that received similar post-harvest site preparation treatments after clear-cutting, with slash piled and burned. Sites represented 2yr, 7yr, 12yr, and 17 year-old second-growth clearcuts, a 79 year-old clear-cut, and an old growth, uncut control. They found that timber harvest and site preparation dramatically altered soil C and N distribution. Although total stocks of C and N varied considerably among the plots and did not change consistently as a function of recovery age, the C:N varied systematically with recovery age. The C:N were initially high throughout the upper 20cm following site preparation and became lower with depth and with recovery age. The 79-yr-old cut, representing pre-harvest conditions, had the highest C:N, the 2- and 7-year-old cuts had lower C:N and still lower C:N on the 12- and 17-year-old plots, suggesting that considerable decomposition had occurred since harvest (Black and Harden 1995). At this stage of recovery, however, C losses are offset by deposition and accumulation in new organic horizons resulting in increased C:N. They hypothesize that the amount of C ultimately stored in the soil at steady state depends largely on reserves of soil N left after forest management which appears to vary with erosion, intensity of burning and site treatment (Black and Harden 1995). These authors noted the variability in horizon thickness within plots as a result of the mineral soil and organic debris being first rearranged by site
preparation, then eroded in some places and accumulated in other places, and point out the importance of both process-related trends and soil variability in the interpretation of results (Black and Harden 1995).

Similarly, the effects of fire on SOM and total N can be highly variable within a treatment plot because burns are not often uniform across an area. The effects of fire on soil properties has been studied extensively over the years (Wells et al. 1979, Raison 1979, Boyer and Dell 1980, Hungerford et al. 1991). Wells et al. (1979) note that N is the main element lost during a fire and claim that site quality would decrease in areas subjected to repeated fires if a replacement mechanism were not present.

Fire effects on N dynamics are a function of the degree and duration of soil heating. A major problem with predicting soil heating is the inability to quantitatively relate prefire conditions and fire characteristics to the downward heat pulse (Boyer and Dell 1980, Hungerford et al. 1991). Soil heating is a function of the interactions of fuel consumption, and soil properties such as porosity, mineral content, water content, and organic matter content (Hungerford et al. 1991). Fuel consumption is dependent mainly on moisture content, but also on fuel loading, size distribution, mineral soil incorporation, degree of consumption of adjacent fuels, weather conditions, and ignition technique (Albini 1976, Brown et al. 1985). Although duff consumption has been predicted by preburn amounts and moisture contents (Harrington 1987), dependence on so many variables makes total fuel consumption and subsequent soil heating and effects on soil N difficult to predict. Furthermore, a wide range of fuel consumption can be expected in any given fire, the spatial distribution of the burn pattern having both short and long-term effects on soil N.
When the organic matter does burn, how it is altered chemically depends upon the temperatures reached during the fire. When soil temperatures reach 300°C and above, >50 percent of the N can be lost to volatilization (White et al. 1973). DeBell and Ralston (1970) found that N was lost as N\textsubscript{2}, by burning litter in the lab and measuring the amount of total N before and after burning (in the ash). How this effect translates to that seen in situ is complicated by the fact that most of the O horizon that is burned in a fire isn’t always composed of litter (O\textsubscript{e}), but of duff (O\textsubscript{e} and O\textsubscript{u}), and this overlies a mineral horizon whose specific physical properties may affect N transformations during burning.

In light to moderate prescribed burns under ponderosa pine, Nissley et al. (1980) reported a consistent decrease in total N in the forest floor. Kovacic et al. (1986) and Jurgensen et al. (1981), however, found total N unchanged by a ponderosa pine underburn and a Douglas-fir /western larch clearcut burn, respectively. Klemmedson et al. (1962) measured 140 kg total N/ha lost from the forest floor in a prescribed burn under ponderosa pine, but they also measured a small increase in total N in the underlying mineral soil. This may be explained by a redistribution of organic matter from the forest floor or slash layers into the mineral soil with light to moderate burning (Raison 1979, Wells et al. 1979), or by other research (e.g. Ryan and Covington 1986, Covington and Sackett 1992) indicating increases in mineral soil NH\textsubscript{4}\textsuperscript{+} following burning.

Claims that severe fires have been found to decrease total N but increase the power to supply N (Raison 1979) should be interpreted with caution. Duff and large woody material make up the bulk of the fuel at most northern Rocky Mountain sites and are important as fuels, and also because of their role in nutrient cycling and site productivity (Harvey et al. 1989). Duff, rotten wood and upper mineral soil are essential
substrates for maintaining forest productivity because they are the main source of N to sustain plant growth as well as the main sources of inoculum of soil microbes (Amaranthus et al. 1989, Powers 1989).

**Microbes**

Soil microbes, as well as soil animals, are the key players in the decomposition of organic matter and nutrient cycling in forest ecosystems. The significance of soil organisms is best shown by their density and diversity: in 1 m² of forest one may encounter over 1000 species and several million individuals (Powers 1989). Soil organisms form an intricate web to capture and assimilate N into complex organic compounds and then slowly release it into the ecosystem (Amaranthus et al. 1989).

The soil microbial biomass pool represents an active phase of soil organic matter. In most soils, no more than 3% of the N present occurs in the biomass at any one time (Stevenson 1986). However, since turnover of N from dead microbial cells is about 5 times faster than that for native soil organic N, it is an important pool (Binkley and Hart 1989). Microbial biomass is the center of the internal soil N cycle as an agent of change and decomposition, as a sink, and as a source of potentially labile N.

Soil microbes are concentrated in the surface horizons and so are susceptible to disturbance. Clearcutting with slash left in place has been found to increase microbial biomass, but burning has been found to decrease biomass (e.g. Entry et al. 1986). Chang et al. (1995) found an increase in microbial biomass from 3-year-old to 10-year-old plantations to old-growth forest, following clearcutting and slash burning in a western red cedar/western hemlock forest. This suggests a decline after treatment followed by
recovery over time. A growth decline observed in the 10-year-old plantations was explained by lower amounts of available N (mineralizable N was generally lowest in the 10-year-old plantations), as well as greater competition for available N from the microbial community (microbial C:N were higher in the 10-year old plantations indicating the greater potential for net immobilization) (Chang et al. 1995).

One might expect potentially mineralizable N to be lower in recently burned sites due to removal of organic matter and probable decline in microbial populations. Bacteria and fungi are killed at temperatures of 50°-160°C (Hungerford et al. 1991). The duration of these effects, if any, is uncertain, and undoubtedly highly variable. With the flush of nutrients and microclimatic changes (i.e. warmer, moister, higher pH, reduced competition for water and nutrients from vegetation), N mineralization can increase as long as sufficient carbon products are available (Hungerford et al. 1991, Covington and Sackett 1990). White (1986) found no change in N mineralization rates until 6 months after burning at which time increased rates were observed. If too much organic matter is consumed, however, N mineralization potentials can be greatly reduced (Hungerford et al. 1991).

**Inorganic N**

The internal N cycle is characterized by biological interchange of inorganic forms of N with organic N, or mineralization-immobilization (Stevenson 1986). The organic pool is the larger of these two pools of N, with estimates of 93-97% of N in the soil surface layer existing in organic combinations (Stevenson 1986). The inorganic pools (NH$_4^+$, NO$_3^-$, NO$_2^-$) can be considered ephemeral, because they are produced and used
concurrently and according to conditions which affect the activities of certain microorganisms.

A first step in mineralization of organic N is ammonification, whereby organic N is converted to ammonia (NH$_3$). It is carried out exclusively by heterotrophic organisms, both aerobic and anaerobic. A wide array of enzymes are involved, each acting on a specific type of organic compound (Stevenson 1986). N containing macromolecules (proteins, nucleic acids, aminopolysaccharides) are converted to amino acids, purines, pyrimidines, and amino sugars which are, in turn, broken down by amidases, thus releasing NH$_3$.

Following ammonification, NH$_3$ may be converted to NO$_2^-$ and ultimately NO$_3^-$ by the process of nitrification. Nitrification is a much more specific process, carried out predominantly by gram-negative, chemolithotrophic, aerobic bacteria. It is thought, however, that autotrophic nitrifiers are less important in acid environments, wherein heterotrophic fungi convert amino-N directly to NO$_3^-$ in acid coniferous soils (Barraclough and Puri 1995).

The controls of ammonification are less restrictive than those of autotrophic nitrification. That is why nitification is thought to be a limiting process and is therefore much studied. Both processes occur within a range of environmental conditions and have optima.

Management activities such as timber harvest and prescribed burning affect the soil environment in many ways that have the potential to affect the rates and magnitude of mineralization and immobilization. They may cause changes in soil porosity (via disruption and/or compaction), changes in SOM (via redistribution, deposition, plant
succession, and changes in microenvironment), and changes in microbial community
distribution (via physical disruption of habitat and redistribution of resources). The
intensity, severity, distribution and frequency of surface disruption, and the quantity and
distribution of live and dead biomass removed or left on site are important considerations
in the evaluation of these disturbance effects on short and long-term N availability.

Changes in microbial habitat, such as changes in porosity, cause changes in the
flow of gases, fluids, nutrients and other solutes. Soil mixing (e.g. tillage) is likely to alter
mineralization rates by disrupting these flows, whereas compaction is likely to reduce
mineralization by reducing these flows. Soil disturbance has been found to stimulate N
mineralization and nitrification (e.g. Birch 1958, Johnson et al. 1995, Van Miegroet
1995). Nitrogen mineralization in disturbed and undisturbed samples has been measured,
looking for a predictable relationship between the two (e.g. Cabrera and Kissel 1988,
Stenger et al. 1995), but to date, none has been observed. It is thought that the
stimulation of N mineralization is brought about by exposing previously protected
organic matter to soil microbes, and this physical protection has been modeled as a
function of the capacity of clay particles and aggregates to hold organic matter (Hassink
and Whitmore 1997). Compaction, on the other hand has been found to reduce tree
volume (e.g. Helms and Hipkin 1986, in ponderosa pine), perhaps as a result of reduced
N mineralization due to increased physical protection of organic materials in compacted
soil (Breland and Hansen 1996), or due to reduced aeration and WHC.

Vegetation removal reduces water uptake and opens the canopy thereby
increasing solar radiation and increasing soil moisture and temperature. The temperature
effect is exaggerated after fire by the blackening of the soil surface. The effects of
changes in water availability are dependent on the magnitude of the change. It has been shown that optimum water potential for N mineralization occurred between -0.3 and -3 bars (Sommers 1981), and it is often assumed that this water potential occurs at a moisture content equivalent to about 60% of a soil's water holding capacity, but this will vary with different soils. Additional water availability is likely to improve mineralization especially on drier sites and in combination with increased temperatures. The majority of the microbes involved in N mineralization are mesophyllic with optima in the range of 25°-37° C (Jarvis et al. 1996), with net mineralization increasing and becoming less variable at higher temperatures (Stanford et al. 1973). Therefore, one would expect mineralization rates to increase with increases in temperature, all else being equal. However, moisture and temperature do not interact in a simple linear way, therefore variation in microsite temperature and moisture needs to be considered.

A great deal of research has focused on the effects of timber harvest on nutrient fluxes in forest ecosystems. Early research in the 60's and 70's focused on losses of $\text{NO}_3^-$ as a result of logging, usually clearcutting. These losses were thought to occur because harvest disturbance temporarily decreases plant uptake of N, increases mineralization of organic matter by mixing soil surface and organic horizons, allows for higher soil temperatures and greater moisture availability, and thus increases the potential for nitrification and $\text{NO}_3^-$ losses (Pritchett and Fisher 1987). The magnitude of such losses varies with different ecosystems. One of the best known studies was that of the Hubbard Brook watershed in New Hampshire (see Bormann and Likens 1979). Here they measured a loss of about 340 kg/ha of $\text{NO}_3^-$ in streamwater in the first three years following cutting and herbicide treatment. Similar studies reported smaller or no loss of
NO$_3^-$, with greatest losses usually occurring on sites initially high in N. Vitousek and Melillo (1979) conclude that there is a wide disparity in the literature on the effects of disturbance on N losses, and that there was at that time no satisfactory general explanation for the disparity.

At another hardwood site, Matson and Vitousek (1981) used incubations to predict potential nitrification and mineralization in a clear-cut chronosequence. Preliminary results indicated that mineral soil produced at least five times more mineral N on an areal basis than did forest floor, which is consistent with other studies that have found the forest floor to be a site of net N immobilization (e.g. Hart and Firestone 1991). Matson and Vitousek (1981) found that nitrification was always greater in clearcut soils than in control forest soils even under identical conditions of temperature and moisture, while mineralization was significantly greater only in the 4-year-old clearcut. In laboratory studies, they found that increasing moisture increased rates of mineralization and nitrification in both clearcut and control soils, and warmer soil temperatures increased rates of mineralization and nitrification in the youngest clearcut. However they found very little difference in soil moisture content or temperatures in the field (Matson and Vitousek 1981). They suggest that the litter of early successional herbaceous and woody plants which invade clearcuts may have higher N contents and may therefore be more readily decomposable than later successional species particularly where N-fixing symbionts are important to revegetation.

In a mixed conifer forest dominated by ponderosa pine, Frazer et al. (1990) found that net N mineralization rates were greater in 5- and 17-yr-old regenerated clearcuts than in an adjacent 100-yr-old uncut forest. They attribute these effects to higher substrate
availability and to moisture and temperature conditions favoring microbial activity in the clearcut units. The soils in their study were very productive, deep alfisols. The clearcut units were harvested using crawler tractors and slash was piled into windrows and burned. The units were sampled away from windrows. Bulk densities were found to be similar between treatments, indicating no effects of compaction. Most mineralization occurred in the surface soil and was greatest in the clearcut where disturbance and incorporation of organic residues were most recent. Timber harvesting usually increases ammonification and nitrification by altering the main controlling factors, as previously discussed, how long this effect lasts is not known, and the answer may vary with the measurement technique (Frazer et al. 1990), as well as the climate, vegetation and soil characteristics.

Prescribed burning has often been found to cause a decrease in total N on a site. If calculations are based only on losses of specific organic components, and unless increases in mineral soil N after burning are accounted for, one may overestimate the amount of N lost by volatilization to the atmosphere (Harvey et al. 1987). According to Covington and Sackett (1990), some of the nutrients volatilized from the forest floor and slash, such as NH$_4^+$ produced during fire through pyrolysis of organic N compounds, are transferred to the mineral soil. If N is redistributed as NH$_4^+$ during the fire, it would show up immediately following a burn, which has been verified by studies that measured NH$_4^+$-N in the mineral soil immediately following a burn (e.g. Jurgensen et al. 1981, Kovacic et al. 1986, Ryan and Covington 1986, Covington and Sackett 1992).

Jurgensen et al. (1981) measured N losses from the surface organic layers of a clearcut burn in western Montana, but also measured increases in NH$_4^+$ two days
following the fire, which remained elevated for a year. They then observed rapid nitrification after a three-week lag-period, as well as an increase in nitrifying bacteria in the mineral soil. Nitrate is a plant available form but it is also more readily lost from the soil system via denitrification or leaching. One might therefore expect to find higher NO$_3^-$ concentrations in the lower depths of burned sites, if it is not lost to ground water.

Kovacic et al. (1986) tested three burning intensities: Low 980, moderate 1760, and high 2280 kJ s$^{-1}$ m$^{-1}$. They measured increases in NH$_4^+$, one day post burn, that were highest in the most intense burn and were three times that of preburn levels. They attribute this to the release of NH$_4^+$ from “protein-like components of soil clay-organic complexes” and organic matter when heated above 100°C. They also found increases in NO$_3^-$, thirty days post burn.

Covington and Sackett (1992) found similar increases in NH$_4^+$ immediately following prescribed burning under ponderosa pine, and an increase in NO$_3^-$ one year later accompanied by a similar decrease in NH$_4^+$. They found that the NH$_4^+$ increases were proportional to the amount of the forest floor burned, supporting the idea that N volatilized from organic matter can recondense in the underlying soil (DeBano 1990).

Covington and Sackett (1986) found that a burn interval of two years maintained high levels of available N, and that after four years available N was returned to pre-burn levels. They recommend a two-year burn interval at that site. However, under similar conditions with different soils, one might see a decrease in available N after one year, because it was nitrified and lost through leaching or denitrification. Burning intervals that are too frequent may cause too rapid N depletion from some systems.
If the plant-available N released during burning is actually used by the plants, we would expect to find higher concentrations of it in the understory vegetation and pine needles. In a long term study in northern Arizona, Harris and Covington (1983) found nutrient concentrations in understory vegetation were consistently higher on burned than on unburned sites. They also found understory biomass to be twice as high in burned plots. At the same study site Peterson et al. (1994) compared tree growth in plots subjected to different burn intervals. Because the response of ponderosa pine growth to fire is complex, they were able to conclude only that the use of prescribed fire at intervals of 4 to 6 years in forests in Arizona appears to effectively reduce fuels without detrimental impacts on tree growth. Haase (1986) found that prescribed burning created a more favorable seedbed and resulted in a twenty-fold increase in the number of seeds germinating on burned compared to unburned sites. Stark and Steele (1977) studied the nutrient content in shrubs under mature larch/Douglas-fir forests in western Montana three years post-burn from hot, medium and lightly burned sites. They found the greatest increases in biomass and nutrient contents of shrubs on the hotly burned sites (Stark and Steele 1977).

Studies of the effects on ponderosa pine growth in Oregon have suggested that repeated fire can reduce long-term productivity (Grier 1989, Landsberg et al. 1984, Cochran and Hopkins 1991). Landsberg and others (1984) found that needle mass and foliar N were lower in the high and low fuel consumption burns than in controls. This was measured in pounds per acre, and so could be accounted for by scorch damage from the fire. This decrease was evident the first growing season after the burn, and still more pronounced after the fourth growing season in the heavy burn although needle mass
increased slightly by the fourth growing season after the light burn. Eight years following the prescribed burns, the basal area periodic annual increment was lower by 9 percent in the moderate fuel consumption burn and by 15 percent in the high fuel consumption burn (Landsberg 1988). These studies were conducted on ash and pumice soils, whereas the Arizona studies were conducted on high clay basaltic soils. So, there may be some factors related to certain soil characteristics or burn parameters that can account for the variation.

**General Considerations in Nitrogen Availability Estimates**

A number of approaches have been used to estimate N losses, N mineralization, and N availability following harvest and fire disturbances at the site level. The difficulty of extrapolating results of other studies was noted by Vitousek and Melillo (1979), because the body of research encompasses a wide variety of disturbances (e.g. differences in harvest equipment and site preparations), sampling methods, measuring techniques, time scales, site characteristics, climates, and vegetation.

Estimates of rates of net mineralization represent the sum of competing consumptive and productive processes (mineralization, immobilization) in the internal soil N cycle. Matson and Vitousek (1981) suggest that the higher net nitrification rates in clearcut soils may be due to a larger initial population size of nitrifying bacteria. However, recent studies using $^{15}$N isotope dilution show that gross rates and net rates are not always correlated (Hart et al. 1994, Davidson et al. 1992). Hart et al. (1994) used $^{15}$N isotopic dilution to measure gross rates of nitrification in a laboratory incubation of an old-growth coniferous forest soil. They found a significant increase in the NO$_3^-$ pool size
after 140 days, with a concurrent decrease in the microbial biomass (probably due to 
decline in available C). Gross rates of nitrification were substantial throughout the entire 
incubation and were poorly correlated with changes in NO$_3^-$ pool sizes. This suggests 
that the increase in the NO$_3^-$ pool size may arise from reductions in the rate of microbial 
immobilization of NO$_3^-$ rather than from mechanisms such as increases in autotrophic 
nitrifier populations (Hart et al. 1994). So, perhaps greater nitrification rates (really an 
increase in the NO$_3^-$ pool over time) in clearcut soils are a result of reduced microbial 
biomass caused by limited C availability, or by increased autotrophic competition for 
NH$_4^+$. Measurement of microbial biomass and nitrifier populations and activity are 
necessary to make this interpretation.

This research has great significance in challenging the assumption that 
mineralization and nitrification rates are inherently low in mature forests because of the 
build-up of organic materials which is thought to cause a stagnation of nutrient cycling 
because of high C:N or the build-up of allelopathic compounds in the litter (White 1986). 
Stark and Hart (1997) measured gross mineralization and nitrification in undisturbed and 
disturbed forests in Oregon and Arizona. They found consistently higher rates of 
transformations in the undisturbed ecosystems. Hart et al. (1994) suggest that the mean 
residence time of N in soil pools is a better indicator of the dynamics of N within a given 
pool than are changes in pool size or fluxes into or out of the pool alone.

If measures of net change in pool sizes are an incomplete picture of potentially 
available N, one may question the usefulness of methods that employ these measures. 
However, since some measures of net rates have been found to be well correlated with 
vegetation parameters, they may be useful in some context in some systems.
CHAPTER 3: RESEARCH SUMMARY

INTRODUCTION

Pre-1900 conditions in ponderosa pine/Douglas-fir forests in western Montana have been described as open, park-like uneven-aged stands dominated by ponderosa pine (Habeck 1990, Gruel 1983). Today, largely as a result of reduced fire frequency and selective logging, these ecosystems are characterized by thickets of small firs, impacted by insects, disease and stand destroying wildfires (Arno et al. 1995, Fiedler et al. 1992). Methods to restore "ecological structure and process" include silvicultural prescriptions designed to emulate natural disturbance regimes (Arno 1996), and may include prescribed fire (Harrington 1996). Target stands are defined by specific numeric designations of stand density, structure, and species composition based on those that existed in the past according to historical descriptions, old photos, and forest inventory records (Fiedler 1996). One such project in a ponderosa pine/Douglas-fir forest in western Montana, was initiated in 1991 on the Lick Creek Resource Demonstration Area, Bitterroot National Forest, with the objective of testing the effects on site productivity of different uneven-aged harvest treatments with and without prescribed underburning (Fiedler et al. 1992).

The productivity of any forest site is affected largely by the nutrient status of the soil, and N is commonly a limiting nutrient (Keeney 1980). Of particular interest for crop production is the cycling and availability of nitrogen. Nitrogen availability is a good indicator of the impact of land use on soil quality and productivity, because it is dependent on the complex interactions of a number of biotic and abiotic factors. Nitrogen availability is inextricably linked to the organic matter component of the soil,
including the micro and macro flora and fauna, both as a source and sink of N, and as a provider of other beneficial properties and nutrients that promote N cycling. Therefore, one must consider a number of parameters in an assessment of the effects of disturbance on N availability.

The effects of underburning on N cycling in Ponderosa pine forests has been studied in some detail (e.g. Covington and Sackett 1990, Landsberg 1984). In studies in central Oregon under ponderosa pine on volcanic ash soils, productivity was decreased as a result of prescribed fire (e.g. Landsberg 1992) and vegetation removal (e.g. Busse et al. 1996). Studies under ponderosa pine in basaltic clay soils in Arizona, recorded large increases in available N (e.g. Covington and Sackett 1992), with moderate long-term reductions in growth following 1, 2, 8, and 10 year burning intervals, and improved growth with 4 and 6 year burning intervals (Peterson et al. 1994). Despite this conflicting evidence, it is fairly well agreed upon that light to moderately severe fires (Ryan and Noste 1985) have a positive effect on N cycling by adding energy to the cycle which serves to aid in the decomposition of organic matter which is the main source of plant available N (Harvey et al. 1989). There is, however, little information regarding the combined effect of selection cutting on N cycling in uneven-aged Ponderosa pine forests.

The Lick Creek project, along with an older example of the same silvicultural prescription at E/L Ranch, provides an opportunity to examine the effects of selection cutting and prescribed underburning on selected N availability indices. Powers (1980) found that N mineralized in 14-day anaerobic incubations in the lab correlated well with N mineralized during 6-month field incubations, which is assumed to be directly related to actual N available during the course of a growing season. Furthermore, results by this
method have been positively correlated with growth response variables of trees (Powers 1980, Kraske and Fernandez 1990, Klinka et al. 1994).

OBJECTIVES

The purposes of this study were: 1) to evaluate the effects of single tree selection harvest with and without prescribed underburning on indices of soil organic matter (SOM) quality and N availability at two depths in the mineral soil under ponderosa pine, 1, 2, 3, 11, and 12 years following disturbance; and 2) to look for seasonal changes in these indices. Potentially mineralizable N (PMN), microbial biomass N, 3-day soil respiration, inorganic and amino N, and soluble anthrone reactive carbon indices were analyzed seasonally on control plots and on harvested plots treated with and without prescribed fire, for a one-year period.

MATERIALS AND METHODS

Site Description

The two sites are located in western Montana. 1) E/L Ranch, located in the Blackfoot Valley, northeast of Missoula, Montana, and 2) Lick Creek Resource Demonstration Area on the Bitterroot National Forest, northwest of Darby, Montana.

The E/L ranch site is on an old stream terrace, 1210 m elevation, with level (<3% slope) terrain at the base of a hill, bordering an open grassland. The habitat type is
35

*Pseudotsuga menziesii/Calamagrostis rubescens-Arctostaphylus uva-ursi.* The soils are a complex of Typic and Udic Ustochrepts derived from belt meta-sediments redeposited as alluvial sediments 5-10,000 years ago (Nimlos 1986). A summary of soil physical and chemical characteristics are presented in Table 1.1. Mean annual precipitation recorded at nearby Lubrecht Experimental Forest Headquarters (station formerly located at the Greenough Park Office) is 45.5 cm, with 44% falling in the winter, mostly as snow, and 24% falling in the summer (Nimlos 1986). Mean annual air temperature recorded at the station is about 4°C, mean winter air temperature is -6°C, and mean summer air temperature is about 15°C (Nimlos 1986). The site consists of three 1-ha treatment plots: 1) single tree selection cut, harvested in fall of 1983, 2) single tree selection cut harvested in fall of 1983 and broadcast burned in spring of 1984, and 3) untreated control.

<table>
<thead>
<tr>
<th>Site</th>
<th>Treatment</th>
<th>Sand</th>
<th>Silt</th>
<th>Clay</th>
<th>WHC</th>
<th>Total C</th>
<th>Total N</th>
<th>C:N</th>
<th>pH</th>
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</thead>
<tbody>
<tr>
<td>E/L Ranch</td>
<td>Control</td>
<td>620</td>
<td>280</td>
<td>100</td>
<td>290</td>
<td>29</td>
<td>1.9</td>
<td>15.3</td>
<td>5.2</td>
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<tr>
<td></td>
<td>Cut</td>
<td>480</td>
<td>390</td>
<td>130</td>
<td>348</td>
<td>32</td>
<td>2.0</td>
<td>16.0</td>
<td>5.2</td>
</tr>
<tr>
<td></td>
<td>Cut+Burn</td>
<td>620</td>
<td>300</td>
<td>80</td>
<td>288</td>
<td>24</td>
<td>1.6</td>
<td>15.0</td>
<td>5.4</td>
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<tr>
<td>Lick Creek</td>
<td>Control</td>
<td>700</td>
<td>270</td>
<td>30</td>
<td>247</td>
<td>17</td>
<td>1.0</td>
<td>17.0</td>
<td>4.7</td>
</tr>
<tr>
<td></td>
<td>Cut</td>
<td>720</td>
<td>240</td>
<td>40</td>
<td>243</td>
<td>17</td>
<td>1.1</td>
<td>15.5</td>
<td>4.9</td>
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<tr>
<td></td>
<td>Cut+2yr Burn</td>
<td>680</td>
<td>280</td>
<td>40</td>
<td>246</td>
<td>17</td>
<td>1.1</td>
<td>15.5</td>
<td>5.3</td>
</tr>
<tr>
<td></td>
<td>Cut+1yr Burn</td>
<td>690</td>
<td>270</td>
<td>40</td>
<td>227</td>
<td>18</td>
<td>1.2</td>
<td>15.0</td>
<td>5.0</td>
</tr>
</tbody>
</table>

The Lick Creek site is on a moderately steep slope (~30%) with a south aspect at an elevation of 1460-1525 m. The habitat type is *Pseudotsuga menziesii/Calamagrostis rubescens.* The soils are shallow to moderately deep, derived from highly weathered granitic parent material, and classified as Totelake family, sandy-skeletal, mixed, frigid,
Typic Ustochrepts. A summary of soil physical and chemical characteristics are listed in Table 1.1. The mean annual precipitation (at Darby) is 40 cm, with 27% falling in the winter, mostly as snow, and 21% falling in the summer. The mean annual temperature is 7°C, ranging from 40°C in July to -28°C in January. The site consists of four treatment plots: 1) single tree selection cut, harvested in the fall of 1992, 2) single tree selection cut harvested in the fall of 1992, and broadcast burned in the spring of 1993, 3) single tree selection cut, harvested fall of 1993, and broadcast burned spring of 1994, and 4) untreated control.

Experimental Design and Execution

The experimental unit at Lick Creek is part of a larger research project to implement alternative harvest and prescribed burning regimes whose effects are meant to emulate natural processes (Arno et al. 1995). Specifically, the goals are to reestablish and perpetuate ponderosa pine as the predominant species, to develop and maintain uneven-aged stand structures, and to reduce and recycle forest floor materials. Units to be harvested were randomly selected and harvested in the fall. The prescriptive elements for the individual tree selection prescription used to achieve target stand structure, density and maximum tree size goals are as follows: a residual basal area of 50 ft²/acre, maximum tree diameter of 20 inches, diameter distribution (q) of 1.2 (4 inch class) (see Fiedler 1995 for explanation), a species composition with a maximum of 10-20% Douglas-fir, and a cutting cycle of 20 years. The slash left on site included tops of harvested trees, and all trees less than 7” dbh (Appendix A). On May 19, 1993, one unit was burned with an intermediate intensity burn, between 5:30 and 11am, with an air
temperature of 53-70°F, and a relative humidity of 48-56%. Fuel moisture conditions
and fuel reductions in are listed in Appendix A. Burn information is not available for the
unit burned in the spring of 1994.

Tree growth was measured after five years (end of 1997 growing season). Foliar
samples were collected from selected trees in each plot during the spring of 1995 and
1996, and analyzed for C and N content.

**Sampling Design**

In the summer of 1995, we selected one block at the Lick Creek area that had an
additional unit adjacent to the other three that was harvested in a similar fashion and
burned one year after the others. Fifteen evenly-spaced, permanent plot centers were
located on a grid within each treatment unit for use as reference points for gathering tree
data. Using the plot centers as a guide, five transects were placed along the contour of the
slope across each plot and samples taken from 4 randomly located points along each
transect. Duff and litter layers were removed, and 3 soil cores were taken to a depth of 15
cm. Cores were divided into 0-7 and 7-15 cm depth sections and composited across
transect points by depth, resulting in five composite samples for each depth for each
treatment unit. The samples were placed in a cooler after collection, returned to the lab,
refrigerated overnight, and analyzed the following day. The same procedure was
followed to gather samples at the E/L site. Sampling was repeated in the fall, spring and
following summer, although logistic constraints prevented sampling at the E/L site in the
spring.
Mixed-bed ion exchange resin capsules were buried to depths of 7 and 15 cm at two of the four sample locations in each transect. The bags were left in place for 6-12 weeks at which time two more sets were put in place to be removed 1) before the ground froze and 2) after the spring thaw. The accumulation of inorganic N in the resin capsules was used to estimate net mineralization occurring over the time period that they were in the ground (Hart and Firestone 1989, Binkley and Hart 1989).

Laboratory Analyses

Most of the visible root matter was removed by hand from the composite samples, which were then subsampled for gravimetric moisture content, pH, water holding capacity (DeLuca and Keeney, 1994), and particle size distribution (Gee and Bauder 1986). A subsample was extracted in 2M KCl for inorganic N determination as described by Keeney and Nelson (1982). Potentially mineralizable N was determined following the procedure described by Keeney (1982). Five grams (oven dry equivalent) of soil were placed in a centrifuge tube, 12.5 ml of water added, N$_2$ gas was added to the headspace, and the tube was sealed and incubated for 14 days at 25° C. The samples were then extracted with 12.5 ml of 4 M KCl, shaken for 30 minutes and then filtered through Whatman # 2 filter papers. The extract was then analyzed for NH$_4^+$, using the indophenol blue method (Bundy and Meisinger 1994). Microbial biomass was determined by fumigation/extraction method followed by reaction with ninhydrin (DeLuca and Keeney 1993a). Anthrone reactive carbon (ARC) was determined as described by DeLuca and Keeney (1993b).
Three-day respiration rates were measured by the alkaline trap method (Zibilske 1994). Fifty grams (oven dry equivalent) of fresh mineral soil was placed in a 0.96 L glass jar and water was added to bring soils to approximately 60 percent water holding capacity. An open scintillation vial containing 30 ml of 1.0 N NaOH was placed in the jar, the jar sealed and lace in a constant temperature chamber at 25 C for 3 days. The vials of NaOH were then quantitatively transferred to 250 ml flasks, 30 ml of 2 N BaCl₂ and 5 drops of phenylthaline indicator solution added, and titrated to a white endpoint with 1.0 N HCl.

Statistical Analyses

Because this study is unreplicated, application of statistics to test an hypothesis regarding treatment effects could be considered a case of psuedoreplication (Hurlburt 1984, Hargrove and Pickering 1992). Ecological inferences can still be drawn even though treatments are not replicated (Hurlburt 1984, Hawkins 1986), but this is best done descriptively, since it is philosophically inappropriate to apply statistical tests to results if “replicates” are only samples from a single experimental unit, because these are not independent and at best can only demonstrate differences between and variability within locations (Hurlburt 1984).

Data were analysed using the t-test procedure of Cochran and Cox (1950) (SAS Institute Inc. 1989) at p<0.10 to determine differences between control and treated plots and between burned and unburned treatment plots. This allows a starting point from which to discuss results, but should not be considered a conclusive test of treatment
effects. Results are also discussed descriptively following the presentation of T-test results, and inferences and possible explanations are based on these descriptions.
RESULTS

Analysis by Depth

Results from all analyses on treatment plots on all dates at both sites showed consistently lower values (and higher pH) for the 7-15 cm depth than for the 0-7 cm depth, with only a few exceptions. This suggests that most of the nutrients and biological activity occur in the top 7 cm, which includes the A horizon. One exception to note is the ratio of Biomass C/total C, which is consistently higher in the 7-15 cm depth. The remainder of the results will focus on comparisons of measurements taken from the surface 7 cm.

Results from E/L Ranch

Total C and N

Total C and total N, are highest in the cut plot and lowest in the cut and burned plot (Table 3.2). The C:N is also lowest in the cut and burned plot. The soil pH does not differ significantly between sites (Table 3.2).

<table>
<thead>
<tr>
<th>Table 3.2. Total N and C, E/L Ranch.</th>
</tr>
</thead>
<tbody>
<tr>
<td>Total C</td>
</tr>
<tr>
<td>---------</td>
</tr>
<tr>
<td>Control</td>
</tr>
<tr>
<td>Cut</td>
</tr>
<tr>
<td>Cut + 11yr Burn</td>
</tr>
</tbody>
</table>

* Differs significantly from control as determined by t-test of five replicates at p<0.10.
Labile C and N

Samples taken in the summer of 1995 and analyzed for potentially mineralizable N (PMN), microbial biomass, three-day microbial respiration rates, ninhydrin reactive N (NRN) and anthrone reactive C (ARC) showed similar patterns in all measures (Figure 3.1). Potentially mineralizable N, microbial respiration, and ARC were highest in the cut plot and lowest in the burn plot with significant differences in PMN and microbial respiration between the cut and burn plots (Figure 3.1). Biomass N and NRN were highest in the control plot and lowest in the burn plot, with significant differences in biomass N between control and both treated plots, and in NRN between control and burn plots (Figure 3.1). Differences were small and variability high in some cases but the patterns were consistent. Samples from summer 1996 showed no significant differences between plots in any of the parameters measured, although the mean values measured in the burned plot were consistently lower. PMN was much larger than the previous year with very little difference between plots (Figure 3.1) or with depth in the profile. There was also very little difference between plots in measures of biomass N and NRN, while microbial respiration and ARC had patterns more similar to those of the previous summer (Figure 3.1). Water content, expressed as a percentage of water holding capacity (WHC) does not differ significantly between treatment plots on any sample date (Figure 3.1).
The ratio of PMN/total N was significantly higher in the cut plot than either the control or burn plots (Table 3.3). The ratios of biomass N and C to total N and C were significantly higher in the control plot due to exceptionally high biomass values in the control plot the first summer (Table 3.3).
Table 3.3. Ratios, E/L Ranch, summers

<table>
<thead>
<tr>
<th></th>
<th>PMN/totN</th>
<th>BioN/totN</th>
<th>BioC/totC</th>
<th>% of total</th>
</tr>
</thead>
<tbody>
<tr>
<td>1995</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Control</td>
<td>0.53</td>
<td>3.10</td>
<td>1.34</td>
<td></td>
</tr>
<tr>
<td>Cut</td>
<td>0.94*</td>
<td>1.27*</td>
<td>0.50*</td>
<td></td>
</tr>
<tr>
<td>Cut +Burn</td>
<td>0.55</td>
<td>1.34*</td>
<td>0.60*</td>
<td></td>
</tr>
<tr>
<td>1996</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Control</td>
<td>1.41</td>
<td>1.64</td>
<td>0.69</td>
<td></td>
</tr>
<tr>
<td>Cut</td>
<td>1.13</td>
<td>1.45</td>
<td>0.59</td>
<td></td>
</tr>
<tr>
<td>Cut +Burn</td>
<td>1.00</td>
<td>1.59</td>
<td>0.73</td>
<td></td>
</tr>
</tbody>
</table>

* Differs significantly from control as determined by t-test of five replicates at p<0.10, within sample dates.

_Inorganic N_

Extractable inorganic N was low in all plots on both summer sample dates, with little difference between plots, although both extractable and resin-accumulated NH₄⁺ was significantly lower in the burn plot in the summer of 1995 (Table 3.4). Accumulation of inorganic N on resin capsules during the first summer was similarly low, consistently less than 1 ppm (Table 3.4). Resin capsules were not used at E/L in the summer of 1996.
Table 3.4. Inorganic N, E/L Ranch summers.

<table>
<thead>
<tr>
<th></th>
<th>1995 NH₄⁺ (mg/kg)</th>
<th>1995 NO₃⁻ (ug/capsule)</th>
<th>1995 Resin-NO₃⁻</th>
<th>1995 Resin-NH₄⁺</th>
</tr>
</thead>
<tbody>
<tr>
<td>Control</td>
<td>2.51</td>
<td>0.46</td>
<td>9.75</td>
<td>45.00</td>
</tr>
<tr>
<td>Cut</td>
<td>2.30</td>
<td>0.50</td>
<td>11.60</td>
<td>33.00</td>
</tr>
<tr>
<td>Cut + 11yr Burn</td>
<td>1.50*</td>
<td>0.48</td>
<td>10.10</td>
<td>3.10</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1996 Control</td>
<td>3.13</td>
<td>0.68</td>
<td>--</td>
<td>--</td>
</tr>
<tr>
<td>Cut</td>
<td>3.07</td>
<td>0.18</td>
<td>--</td>
<td>--</td>
</tr>
<tr>
<td>Cut + 12yr Burn</td>
<td>2.87</td>
<td>0.14</td>
<td>--</td>
<td>--</td>
</tr>
</tbody>
</table>

* Differs significantly from control as determined by t-test of five replicates at p<0.10, within sample dates.

Potentially mineralizable N and NH₄⁺ were both higher the second summer (Figure 3.1 and Table 3.4). Anthrone reactive C and NRN are very similar between summers, and biomass N is the same with the exception of a much higher value in the control plot the first summer. Respiration rates were lower the second summer.

The fall sampling again showed the same pattern of highest values in the cut only plot and lowest values in the burn plot for PMN, NRN, biomass N, microbial respiration, and ARC, with significant differences between the cut and burn plots; and differences in biomass N, microbial respiration and ARC, between control and burn plots (Figure 3.1). Ratios were not significantly different from one another on the fall sample date (Table 3.5).
Extractable inorganic N concentrations were again low, with few differences between plots except that the burn plot again (as in the first summer) had lower values for available NH$_4^+$ than the other two plots. Ion accumulation in resin capsules was again very low (less than 0.3 ppm) (Table 3.6).

Differences between summer and fall were such that PMN, ARC, and microbial biomass tended to be higher in the fall in most plots. Inorganic N was no different between summers and fall, although the NO$_3^-$ values were lower in the fall. NRN was much lower in the fall, and microbial respiration was no different. Water content was lower in the fall, especially in the burned plot.
Results from Lick Creek

Total C and N

Analyses of total C and N and extractable cations at Lick Creek showed a great deal of homogeneity between treatment plots in terms of particle size distribution, water holding capacity (Table 3.1), total C, total N, C:N, and extractable K, Mg, and Ca (Table 3.7). The pH of both burn plots was significantly higher than control plot, and that of the older burn plot was also significantly higher than the cut plot (Table 3.7).

<table>
<thead>
<tr>
<th>Table 3.7. Background data, Lick Creek.</th>
</tr>
</thead>
<tbody>
<tr>
<td>Total C</td>
</tr>
<tr>
<td>---------</td>
</tr>
<tr>
<td>Control</td>
</tr>
<tr>
<td>Cut</td>
</tr>
<tr>
<td>Cut + 2yr Burn</td>
</tr>
<tr>
<td>Cut + 1yr Burn</td>
</tr>
</tbody>
</table>

* Differs significantly from control as determined by t-test of five replicates at p<0.10.

Labile C and N

Samples taken in the summer of 1995 and analyzed for PMN, NRN, microbial biomass and respiration, and ARC showed some significant differences between control and burned plots (Table 3.8). PMN, microbial respiration and ARC were significantly higher in control plots as compared with both burn plots, and biomass N was also significantly higher in the control plot than the more recently burned plot. ARC was significantly higher in the cut plot as compared with both burn plots, and microbial respiration was also higher in the cut plot than in the more recently burned plot. In the
second summer sampling there were fewer differences between treatment plots (Table 3.8). Biomass N was significantly higher in the control plot than in all treated plots, and NRN was significantly higher in the control than in the older burned plot. Mean values for PMN and biomass N were lower in both burn plots on both summer sampling dates, although not determined to be significant by t-test. There were differences in microbial respiration and ARC in 1995, but not in 1996. The second summer sample date was the driest date, with moisture contents around 20% of WHC compared with around 80% the previous summer.

Ratios of biomass N/total N and biomass C/total C were similar in all plots, but tended to be highest in the control plot both summers (Table 3.9). Differences between control and treated plots were greater the second summer, corresponding with lower biomass numbers on that date. Biomass N/total N is 1.17-1.47% in all plots the first
summer, but drops to less than 0.66% in all treated plots the following summer while control drops only to 1.17%. Similarly, Biomass C/total C is around 0.42-0.50% the first summer and drops in all treated plots to <0.24% the following summer but stays at 0.41% in control plot. Ratios of PMN/total N were significantly higher in the control than both burned plots and higher in the cut plot as compared with the more recently burned plot the first summer. This same trend is evident the second summer. The first summer the PMN/total N is near 1.00% in the cut and control, and drops to less than 0.60% the following summer. The burn plots remain around 0.33-0.45% both summers. (Table 3.9).

<table>
<thead>
<tr>
<th>Table 3.9. Ratios, Lick Creek Summers</th>
</tr>
</thead>
<tbody>
<tr>
<td>1995</td>
</tr>
<tr>
<td>------</td>
</tr>
<tr>
<td>Control</td>
</tr>
<tr>
<td>Cut</td>
</tr>
<tr>
<td>Cut + 2yr Burn</td>
</tr>
<tr>
<td>Cut + 1yr Burn</td>
</tr>
<tr>
<td>1996</td>
</tr>
<tr>
<td>Control</td>
</tr>
<tr>
<td>Cut</td>
</tr>
<tr>
<td>Cut + 3yr Burn</td>
</tr>
<tr>
<td>Cut + 2yr Burn</td>
</tr>
</tbody>
</table>

*Differs significantly from control as determined by t-test of five replicates at p<0.10, within sample dates.

**Inorganic N**

Resin and extractable inorganic N measurements were very small and variable and had no differences between treatments on either summer sample date (Table 3.10).
Table 3.10. Inorganic N, Lick Creek, summers.

<table>
<thead>
<tr>
<th></th>
<th>NH₄</th>
<th>NO₃</th>
<th>Resin-NH₄</th>
<th>Resin-NO₃</th>
</tr>
</thead>
<tbody>
<tr>
<td>1995</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Control</td>
<td>1.21</td>
<td>0.27</td>
<td>14.80</td>
<td>8.40</td>
</tr>
<tr>
<td>Cut</td>
<td>0.86</td>
<td>0.31</td>
<td>17.90</td>
<td>47.80</td>
</tr>
<tr>
<td>Cut + 2yr Burn</td>
<td>1.24</td>
<td>0.32</td>
<td>12.10</td>
<td>7.90</td>
</tr>
<tr>
<td>Cut + 1yr Burn</td>
<td>1.30</td>
<td>0.55</td>
<td>16.80</td>
<td>8.40</td>
</tr>
<tr>
<td>1996</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Control</td>
<td>1.36</td>
<td>1.82</td>
<td>0.15</td>
<td>--</td>
</tr>
<tr>
<td>Cut</td>
<td>1.58</td>
<td>1.45</td>
<td>0.33</td>
<td>--</td>
</tr>
<tr>
<td>Cut + 3yr Burn</td>
<td>1.23</td>
<td>1.29</td>
<td>0.00</td>
<td>--</td>
</tr>
<tr>
<td>Cut + 2yr Burn</td>
<td>1.42</td>
<td>0.77</td>
<td>4.40</td>
<td>--</td>
</tr>
</tbody>
</table>

Differences between summers include higher moisture content, greater microbial biomass, and greater microbial respiration in all plots the first summer, with differences in the control and cut plots greater than differences in burn plots between summers. PMN was much lower in cut and control plots the second summer, and was consistently lower in burn plots (4.56-5.07 mg/kg) than in cut only (6.00-12.03 mg/kg) and control (6.38-10.36 mg/kg) plots on both sample dates. ARC was much higher the second summer in all plots, and NRN was somewhat higher the second summer in all plots (Table 3.8).

Fall and spring sample dates have fewer differences between treatments and overall higher magnitude in most measurements in most plots. In the fall, ARC was greater in control than any of the treated plots, and biomass N was greater than in the recent burn (Table 3.11). PMN was highest in the control plot, except that it was unusually high in the more recently burned. Biomass N and microbial respiration were higher in control and lowest in the recent burn plot (Table 3.11). There were no differences in NRN.
between treatment plots. These numbers were low and highly variable, similar to inorganic N measurements.

Table 3.11. Lick Creek fall and spring.

<table>
<thead>
<tr>
<th></th>
<th>1995</th>
<th></th>
<th>1996</th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>PMN</td>
<td>NRN</td>
<td>Biomass N</td>
<td>Respiration</td>
</tr>
<tr>
<td>1995</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Control</td>
<td>21.86</td>
<td>0.87</td>
<td>30.50</td>
<td>0.18</td>
</tr>
<tr>
<td>Cut</td>
<td>14.37</td>
<td>1.90</td>
<td>21.24</td>
<td>0.16</td>
</tr>
<tr>
<td>Cut + 2yr Burn</td>
<td>14.01</td>
<td>0.91</td>
<td>25.82</td>
<td>0.12</td>
</tr>
<tr>
<td>Cut + 1yr Burn</td>
<td>25.79</td>
<td>1.19</td>
<td>19.02*</td>
<td>0.15</td>
</tr>
</tbody>
</table>

* Differs significantly from control as determined by a t-test of five replicates at p<0.10, within sample dates.

In the spring the same burn plot was significantly lower in PMN than the cut and control plots, while the older burn plot had the highest value (Table 3.11). Biomass N and microbial respiration were again lowest in this plot, although not significantly different. ARC was significantly higher in the control plot as compared with the cut and recent burn plots. Again there were no significant differences in NRN (Table 3.11).

Moisture content was lower in the recent burn than in the control or the cut.

The ratio of biomass N/total N and biomass C/total C were higher in the fall and spring than in the summers (Table 3.12). The ratios were larger in the control and cut only plots except that the values for the older burn plot were higher than the cut only plot in the fall (Table 3.12). The ratios of PMN/total N were especially high in all plots in the
fall (>1.00%), with the control and cut plots having the larger values, although not significantly different (Table 3.12). These values were lower again in the spring, with the control plot being significantly higher than the recently burned plot.

<table>
<thead>
<tr>
<th>Table 3.12. Ratios, Lick Creek fall and spring</th>
</tr>
</thead>
<tbody>
<tr>
<td>Fall</td>
</tr>
<tr>
<td>------</td>
</tr>
<tr>
<td>Control</td>
</tr>
<tr>
<td>Cut</td>
</tr>
<tr>
<td>Cut + 2yr Burn</td>
</tr>
<tr>
<td>Cut + 1yr Burn</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Spring</th>
</tr>
</thead>
<tbody>
<tr>
<td>Control</td>
</tr>
<tr>
<td>Cut</td>
</tr>
<tr>
<td>Cut + 3yr Burn</td>
</tr>
<tr>
<td>Cut + 2yr Burn</td>
</tr>
</tbody>
</table>

* Differs significantly from control as determined by t-test of five replicates at p<0.10, within sample dates.

Measures of extractable and resin accumulated inorganic N were again low and variable with few notable differences (Table 3.13). NO₃⁻ was higher in the recent burn than the control and resin NO₃⁻ was higher in the recent burn than in the cut or the control (Table 3.13). On most sample dates, the resin accumulated inorganic N was barely detectable with the exception of the spring sample date. This date also had the highest moisture availability (>100% of WHC), as well as higher values of extractable NO₃⁻ (Table 3.13).
Table 3.13. Inorganic N, Lick Creek fall and spring.

<table>
<thead>
<tr>
<th></th>
<th>NH$_4^+$</th>
<th>NO$_3^-$</th>
<th>Resin-NO$_3^-$</th>
<th>Resin-NH$_4^+$</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Fall</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Control</td>
<td>1.03</td>
<td>0.18</td>
<td>4.20</td>
<td>7.70</td>
</tr>
<tr>
<td>Cut</td>
<td>1.61</td>
<td>0.29</td>
<td>2.60</td>
<td>6.30</td>
</tr>
<tr>
<td>Cut + 2yr Burn</td>
<td>1.14</td>
<td>0.21</td>
<td>5.90</td>
<td>13.10</td>
</tr>
<tr>
<td>Cut + 1yr Burn</td>
<td>1.39</td>
<td>0.49</td>
<td>6.50</td>
<td>10.30</td>
</tr>
<tr>
<td><strong>Spring</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Control</td>
<td>1.19</td>
<td>0.82</td>
<td>66.47</td>
<td>83.40</td>
</tr>
<tr>
<td>Cut</td>
<td>0.89</td>
<td>0.44</td>
<td>81.06</td>
<td>51.20</td>
</tr>
<tr>
<td>Cut + 3yr Burn</td>
<td>1.49</td>
<td>1.52</td>
<td>71.06</td>
<td>48.80</td>
</tr>
<tr>
<td>Cut + 2yr Burn</td>
<td>1.03</td>
<td>1.13</td>
<td>74.16</td>
<td>80.90</td>
</tr>
</tbody>
</table>

Moisture content was fairly consistent between treatment plots within sample dates, with notable differences in water content between sample dates. In the spring, moisture content was fairly high in all plots. Moisture content much lower in the fall than in the spring, corresponding to lower microbial respiration in the fall, but no differences in biomass N, NRN or ARC, and higher PMN in the fall than in the spring.
DISCUSSION

The objectives of this study were to estimate short and long-term effects of cutting and burning on indices of N availability, and look at the seasonal variation of these indices. Indices were measured at 1, 2, 3, 11, or 12 years after treatment at two separate sites. Because site conditions are dissimilar, the sites cannot be directly compared to one another. However, we can assess the contrast between 11-12 years post-treatment and an untreated control at E/L, and 1-3 years post-treatment at Lick Creek. Reference to results from E/L site as “long-term” effects and from the Lick Creek site as “short-term” effects are not meant to imply a continuum.

**E/L Ranch - Long-Term Effects**

**Total C and N**

The burned plot at the E/L site showed a significant reduction in total organic C, but no change in total N (Table 3.2). In contrast, in a similar study on the effects of prescribed burning under ponderosa pine, Monleon et al. (1997) found no change in total N and C pools 12 years after burning. The cut only plot had higher total C, total N, and extractable K and Ca, compared to the cut and burned and control plots (Table 3.2). This may be explained by the fact that this plot retained more understory vegetation than the burned plot and had additional organic matter on the surface and less overstory competition for nutrients than the control. The interpretation of these differences is
confounded, however, by the differences in soil surface texture resulting in higher WHC in the cut only plot as compared with the control and the cut and burned plot (Table 3.1). These site differences are influenced by the pattern of deposition on the landscape which has resulted in a complex of soil types with different surface textures which are unevenly distributed within the different treatment plots. These differences may affect measured parameters, as well as site productivity, by affecting available water. Differences in cation contents could also result from possible differences in mineralogy (not measured).

Labile C and N

Results from the E/L site indicate a reduction in PMN, microbial biomass N, NRN, microbial respiration, and ARC between the cut and burned plot and the control plot on all sample dates (Figure 3.1). Because within-site variability is high, these differences are not always significant. Still, the pattern is consistent and worth noting (Figure 3.1). Monleon et al. (1997) studied the short and long-term effects of prescribed underburning on N availability indices including in situ net N mineralization, under ponderosa pine in central Oregon on low-productivity ash soils. They found a decrease in net N mineralization 12 years after prescribed burning, which is consistent with the findings at E/L. They attribute the decrease to a change in substrate quality which may account for the observed decreases in tree growth at these sites (Grier 1989, Landsberg 1988). In another long-term study, Fritze et al. (1993) observed an initial decrease in microbial biomass C and N, length of fungal hyphae, and soil respiration, followed by recovery of these biomass indicators within 12 years. This study, as well as that of
Monleon et al. (1997), did not show a similar recovery of biomass indicators after 12 years.

The consistently higher values in the cut only plot at E/L, may lead one to conclusions such as those found by Frazer et al. (1990). Frazer et al. (1990) used the chronosequence approach to evaluate the effects of clearcutting on N mineralization rates in a mixed conifer forest in California, and found them to be higher in the regenerated clearcuts than in the 100-yr-old uncut forest. Similarly, at the E/L site, the values for PMN, microbial respiration and ARC tended to be higher in the cut only plot as compared with the uncut control (Figure 3.1). However, these differences may be due, in part to pre-existing site differences between treatment plots.

Biomass and PMN numbers as a percentage of the total (PMN/totN) are consistently higher in the control than in the burn plot at E/L (Table 3.3). Differences in net mineralization expressed as a proportion of total soil N indicate differences in the quality of organic matter if temperature and moisture remain the same (Powers 1990), as is the case with the laboratory measurement of PMN. These differences in N mineralization may be expressed differently under field conditions of moisture and temperature. Percent WHC tended to be only slightly (and not significantly) lower in the burn plot (Figure 3.1), whereas the degree of similarity in temperature between these two plot is unknown.

\textit{Inorganic N}

Inorganic N concentrations were not significantly different between plots. This is not surprising since extractable inorganic N is not considered to be a sensitive indicator of
change except where change is dramatic (Binkley and Hart 1989), such as the immediate effects of fire on NH$_4^+$ concentrations in the mineral soil. Concentrations of NH$_4^+$ were consistently lower in the burn plot (Table 3.4), which corresponded to lower concentrations on resin capsules left on site for the duration of the growing season (Table 3.4). Resin capsules had barely detectable levels of NO$_3^-$ (Table 3.4). This is unusual when we look at other resin capsule studies (Binkley and Hart 1989), but is easily explained by the low levels of NO$_3^-$ measured by KCl extraction.

Seasonal differences

Because plant senescence may lead to greater substrate availability, one would expect PMN, ARC, and microbial biomass and respiration to be higher in the fall, which is what we observed. The burn plot continued to show the lowest values for all measured parameters, with significant differences between control and burn plots in microbial respiration and ARC (Figure 3.1). The ratio of PMN/total N in the fall was higher than the control, suggesting a change in the quality of organic matter in the fall (Table 3.5). This is also evident in the ratios of biomass N and C to total N and C, which were the same in the burn and control plots in the fall. Ammonium concentrations were again higher in the control and lowest in the burn plot, while NO$_3^-$ concentrations were low and variable and no different between plots (Table 3.6). Accumulation of inorganic N on resin capsules was lower in the fall than in the summer.

Lick Creek - Short-Term Effects
**Total C and N**

Total organic N and C and extractable K, Mg, and Ca concentrations were the same in all plots. Soil pH of both burn plots was significantly higher than the control plot. Soil pH tends to rise after fire as a result of the deposition of basic cations in the ash on the soil surface (Wells et al. 1979). Monleon et al. (1997) found an increase in total and inorganic N and total C four months after burning, and a decrease in these values after 5 years, which they attribute to a decrease in SOM quantity after five years. The increase in total C is likely an effect of increased inorganic C associated with the ash. These results differ from those observed at the Lick Creek site where total C and N were unchanged in the burned plots.

**Labile C and N**

We observed greater PMN, biomass N and respiration, and ARC in both the control and the cut plots compared to the burn plots on all sample dates with few exceptions (Tables 3.8 and 3.11). This is probably because there is more organic matter (litter and slash, respectively) remaining on these plots.

In undisturbed systems, soil organic matter (SOM) may be considered to attain a steady state level governed by the soil forming factors (climate, topography, parent material, vegetation, soil flora and fauna, and time) and their interaction (Jenny 1941). This “steady-state” is marked by a balance between inputs of residues and their breakdown by mineralization. Where soils are disturbed by cultivation, the various equilibria involved are not maintained and SOM content tends to decline as
mineralization is enhanced (Jarvis et al. 1996). When soils are disturbed by fire, SOM declines rapidly by direct mineralization via oxidation by fire. With disturbance caused by logging, SOM may decrease in those microsites that have experienced varying degrees of physical disturbance, and in areas experiencing higher temperatures and greater moisture availability as a result of overstory removal, but it is also rapidly replaced by the breakdown of slash and dead roots left after harvest. Following this model, one might predict similar indices of SOM quantity and N availability in the control and cut only sites, with lower values in the cut and burned site.

At the Lick Creek site, on the first summer sampling, PMN, biomass N, microbial respiration, and ARC were all significantly higher in the control as compared with the burn plots (Table 3.8). The cut plot was consistently lower than the control in all measures, and was intermediate in value to the control and burn. This pattern was repeated on the second summer sample date, with the exception that ARC and microbial respiration were higher in the older burn plot, and differences were significant in biomass N only (Table 3.8). All measures tended to be lower the second summer except for NRN and ARC. On the second summer sample date, the soils were very dry (20% WHC, compared with 80% WHC the first summer), and this may explain why NRN and ARC were higher and biomass lower on this date: as the soils dried, microbial biomass was decomposed, leaving higher levels of ARC and NRN, instead of remaining in cells as chloroform labile C and N.

Monleon et al. (1997) found that net N mineralization in the mineral soil surface was unchanged in plots underburned 4 months prior to sampling, and decreased in plots underburned 5 years prior to sampling, as determined by in situ soil core incubation.
This agrees with our measurement of less PMN in the burned plots as compared with control on both summer sample dates, 1-3 years post burn. They speculate that the change in N mineralization rates is a result of decreased SOM quantity after 5 years (Monleon et al. 1997). On the same site, Monleon and Cromack (1996) found that litter decomposition rates were lower 4 months after burning but were unchanged in plots burned 5 years earlier. The release of N and P form litter was, however, increased on sites burned 5 years earlier, a pattern which they attributed to possible changes in microbial ecology of the forest floor (Monleon and Cromack 1996).

In another study on prescribed fire, Pietekainen and Fritze (1993) measured the effects of two fire severities on soil microbial measurements in coniferous forest soil humus layers in Finland. They found that the more severe fire decreased microbial biomass C and N, length of fungal hyphae and soil respiration, and that these did not recover to control levels within the three year study period. The less severe fire did not change microbial biomass N, and the length of fungal hyphae did recover to control levels within the study period (Pietekainen and Fritze 1993). This is further evidence for the argument that a sampling scheme that accounts for the spatial variability of fire severity that is always present in a prescribed springtime fire due to irregular drying patterns, may more accurately reflect the overall effects of the burn.

Ratios of PMN/total N were lower in the burn plots both summers (Table 3.9), suggesting lower quality organic matter in these plots. Similarly, biomass N/ total N and biomass C/total C ratios were consistently higher in control than either the cut or burned plots, with the cut only plot being intermediate in value. These ratios were unchanged between summers in the control and cut only plots, while they decreased substantially in
the burn plots. Biomass drops proportionately less in the cut only and control plots, perhaps because the surface organic layer provides insulation and a more stable microenvironment, less affected by changes in moisture availability and temperature.

**Inorganic N**

In many studies of the effects of fire on N availability the focus has fallen on the increase in NH$_4^+$ in the mineral soil immediately following a burn. These immediate effects last anywhere from 30 days (Kovacic et al. 1986) to one year (Covington and Sackett 1992). The mechanism even seems pretty clear where heating lyses the microbial cells and denatures proteins, thus releasing NH$_4^+$ (Hungerford et al. 1991). Harrington (1995) documented this effect at the Lick Creek sites, with an initial large increase (from 1mg/kg to 19 mg/kg), that remained at 16 mg/kg throughout the first postburn year, and decreased to 6 mg/kg after two years. The sampling method used to arrive at these numbers was such that only burned patches were sampled after the fire. Since our sampling approach was different (we included unburned areas to get at average values for the site as a whole) we cannot directly compare our results to those of Harrington (1995).

Also the purpose of our study was to try to get at what happens after this pulse of N has passed, and plant succession is underway, and soil organic matter is recovering. It is not surprising to see that 1-3 years post-burn, there are no differences between treatments and a high degree a variability in measures of inorganic N (Binkley and Hart 1989). The values measured for inorganic N were consistently between 1 and 3 mg/kg (Table 3.10).

Results from the ion-exchange resins (IER) at Lick Creek were interesting. No significant differences were detected between treatments, but there was evidence of
seasonal variation (Tables 3.10 and 3.13). The use of ion-exchange resins is attractive because it allows for in situ measurement of N mineralization which can be much more accurate in predicting N availability since N mineralization is so strongly affected by microenvironmental conditions. Binkley et al. (1986) found good correlations between N accumulated on resin bags and that measured with buried bags, a widely used and accepted method (Binkley and Hart 1989). Accumulations of ions on resin bags is dependent on N mineralization rate, ion form (NO\textsubscript{3}\textsuperscript{-} being more mobile than NH\textsubscript{4}\textsuperscript{+}), water movement to the bags, and competition with microbes and plants for the ions (Binkley and Hart 1989). Hart and Firestone (1989) found similar results when comparing IER bags, buried bags and the core IER methods, however seasonal patterns were not well correlated between the methods. This is likely due to the relative sensitivities of each of the methods to soil moisture.

The first summer at Lick Creek, amounts of adsorbed inorganic N were low, ranging from 0.2 to 0.4 ppm NH\textsubscript{4}\textsuperscript{+} and 0.2 to 0.9 ppm NO\textsubscript{3}\textsuperscript{-}. In the fall, values were equally low for both NH\textsubscript{4}\textsuperscript{+} and NO\textsubscript{3}\textsuperscript{-} (from 0.07 to 0.3 ppm). In the spring, as one might expect due to increased mineralization with the thawing of the soil (DeLuca et al. 1992), we measured larger concentrations of adsorbed ions, ranging from 1.0 to 1.7 ppm NH\textsubscript{4}\textsuperscript{+}, and 1.3 to 1.6 ppm NO\textsubscript{3}\textsuperscript{-}. The second summer, the values were undetectable in some cases, with the highest value for NH\textsubscript{4}\textsuperscript{+} at 0.11 ppm (Table 3.10). Nitrate values are not available for that summer. This was the summer that the soil was very dry at the time of sampling, so it follows that the limited water availability (assuming that it tended to be dry throughout the period that the bags were in place), there would be little transport of ions to the bags. Also, there is likely to be more competition from plants for available
ions during this part of the growing season. The higher concentration of ions observed in the spring at Lick Creek may be due to higher rates of mineralization common after a freeze-thaw event, (supported by more PMN, biomass N and microbial respiration on this date) or it could also be due to more moisture carrying more ions to the bag.

Unfortunately, measures of inorganic N tell us very little about the potential long-term productivity or sustainability of the N cycling at a site, as it is simply a measure of net mineralization at the time of sampling and is likely to be of substantial magnitude only after a dramatic disturbance (Binkley and Hart 1989). Furthermore, measures of net mineralization alone can be misleading in general because the dynamics of the soil system are so complex and turnover is so rapid, that we may miss important aspects of the cycle. For example, a measure of microbial biomass is a good indicator of N availability in that this fraction of SOM is subject to rapid turnover, as well as the driving force behind mineralization. But biomass alone does not give us an idea of the taxonomic make-up of the biomass, nor of the flux of N through the biomass, which is probably more important in regulation N availability in the field (Binkley and Hart 1989). For this reason, $^{15}$N technologies and procedures designed to measure gross N transformations can better elucidate N dynamics in forest soils.

It is generally believed that in undisturbed forest ecosystems nutrients accumulate in both live and dead plant bodies where they are unavailable for cycling until, eventually, further growth could not occur (e.g. Harvey et al. 1989). White (1986, 1991) has proposed that the low mineralization rates observed in undisturbed ponderosa pine ecosystems of the southwest are a result of the allelopathic inhibition of nitrification by monoterpenoid compounds (the essential oil in the ponderosa pine needles) which are
volatilized by fire, thus freeing up the mineralization process after burning. In a mixed
conifer forest in California, it was initially reported that N availability was higher in a 10-
year-old, young-growth stand than in the 100-year-old, old-growth stand (Hart and
Firestone 1989). Later research at that site using $^{15}$N isotope dilution to measure gross
rates of N mineralization reported gross mineralization rates in the old-growth forest that
were 2-3 times higher than those in the young-growth forest (Davidson et al. 1992). In a
more recent study, Stark and Hart (1997) used a $^{15}$N isotope-dilution method to measure
gross rates of nitrification and microbial assimilation of NO$_3^-$ in 11 undisturbed forest
ecosystems in New Mexico and Oregon. They found that gross nitrification rates were
poorly predicted by net rates, and that the soil microbial communities had the ability to
assimilate most of the NO$_3^-$ produced (Stark and Hart 1997). These results contradict the
previous assumptions that nitrification rates are low in mature coniferous forests, and that
the N cycle somehow stagnates with the build-up of organic materials.

*Seasonal Changes*

Fall and spring sample dates have fewer differences between treatments and
overall higher magnitude in most measurements in most plots (Table 3.11). The expected
seasonal variation in nitrogen mineralization (i.e. lowest in the winter with leaching and
low microbial activity, a rise in spring with the thawing of soil and the commencement of
decomposition, decrease in summer due to consumption by plants, and increase in the fall
when growth ceases and residues decay) helps to explain this pattern. The control plot
still tended to have the highest values for all parameters in the fall with the exception of a
large amount of PMN in the more recently burned plot, and low and variable values for
NRN (Table 3.11). In the spring, the values for all measures were similar for all plots, although ARC was higher in the control plot as compared with treated plots, and the more recently burned plot had lower values for all parameters.

The ratios of PMN/total N were much higher in the fall as compared with both summers and spring, and may be accounted for by a larger microbial biomass present on this date (Tables 3.9 and 3.12). The biomass in the spring was also higher than the summers, but not as high as the fall, probably because the burst of mineralization after the thaw had begun, yet on the date we sampled, there was snow on the ground in the morning, indicating the temperature of the soil was probably still fluctuating substantially and that the peak in biomass usually seen in the spring had not yet occurred.

Conclusions

There is no question that logging activities and fire change the nature of the cycling of N at any given forest site (e.g. Bormann and Likens 1979, White 1986, Perry 1994). Whether and to what degree this translates to a greater or lesser N availability is a question relevant to intensively managed forests such as a plantations and tree farms, although it may be argued by some that it is a question relevant to the evaluation of the “health” of the forest ecosystem (e.g. Covington et. al 1997). The objective of this study was to assess differences in indices of N availability between treatment plots.

The answer to the question of how these treatments affect N availability is implicit in what we know about the factors that affect mineralization and immobilization (i.e. the quality and quantity of organic substrate, temperature, and moisture), and how these factors are altered by the treatments. It is also explicit in the results of research
aimed at quantifying and predicting this change in certain ecosystems. In the case of the Oregon studies under ponderosa pine in volcanic ash soils, productivity was decreased as a result of prescribed fire (e.g. Landsberg 1992) and vegetation removal (e.g. Busse et al. 1996). In the case of the Arizona studies under ponderosa pine in basaltic clay soils, we see large increases in available N (e.g. Covington and Sackett 1992), with moderate long-term reductions in growth following 1, 2, 8, and 10 year burning intervals, and improved growth with 4 and 6 year burning intervals (Peterson et al. 1994). Also from the Arizona studies, we see claims for improved ecosystem health (Covington et al. 1997). How these effects are expressed in the climate of western Montana on different soil types remains uncertain, yet similar claims for improved restoration of biodiversity and ecosystem processes have been made (e.g. Fiedler et al. 1992, Arno et al. 1995). The specificity of our sampling technique cannot capture the subtlety of the changes which when added together would give a truer representation of the changes. We know that the measured parameters vary with degree of disturbance (i.e. depth of physical disturbance and fire severity). An approach such as that used by Rab (1996) which accounts for the relative spatial extent of the ground disturbance may be more appropriate for estimating N availability on an areal basis. Our results are expressed on a gravimetric basis only, and provide a good first approximation of some long and short-term effects of selection harvest and prescribed underburning on N availability.

It appears that at E/L ranch, the effects on that type of soil in that type of ecosystem are similar to those in Oregon, where a decrease in productivity has been observed following disturbances and vegetation removal, and that excess vegetation or organic matter removal (via fire and logging) may result in a decline in potential site
productivity, such as that seen in the low productivity, N poor sites in Oregon. Cochran and Hopkins (1991) found better growth without disturbance, and concluded that disturbance decreased site productivity. While differences at E/L Ranch were not always significant due to high variability and low sample size, they were consistent enough to suggest the need for further evaluation.

In typical, undisturbed western-montane forest soils, this organic horizon is 2-10 cm of litter and 0-5 cm duff or humus with varying amounts of decayed wood (Page-Dumroese et al. 1991). It has been shown that total SOM content generally mirrors site productivity and that the least productive sites in this region with the shallowest organic matter deposits are Ponderosa pine stands (Page-Dumroese et al. 1991).

One of the few researchers who has studied nutrient cycling in forests of western Montana is Stark (e.g. 1979, 1980a, 1980b, 1982). Her focus was on the concept of the “biological life of a soil” and on identifying “chemically fragile soils”, where excessive disturbance may destroy so much of one of the available nutrients, that the soil goes into “nutrient shock” and is unable to support trees for hundreds or thousands of years (Stark 1978). In North America, susceptible soils are extremely young or occur on nutrient deficient parent materials, such as thin soils on talus slopes, or over permafrost, or on poor granites (Stark 1978). This concept deals only with those plant nutrients that are derived from rock, and can be studied using mass balance approaches measuring changes in the chemistry of soil water below the rooting zone as a means of evaluating nutrient losses, and precipitation, throughfall, and pollen rain for inputs (Stark 1978). She is careful to point out that nitrogen losses and additions must be studied in a different manner, and that biological fixation of N is the potentially limiting process for that
element and can temporarily or permanently restrict plant growth at any time (Stark 1978).

In applying these concepts in the evaluation of two Douglas-fir/western larch forests in western Montana subjected to different types of harvest and levels of slash removal (1979, 1980a, 1980b), she comes to similar conclusions regarding sustainable harvest practices as the studies which focus on N as a limiting nutrient. That is, that retention of organic matter is important in sustaining productivity on the immature soils of this region. However, a first glance may be misleading to those who go no further than the abstract, in that she states that “the length of time that the soil can chemically support trees under identical treatments on a 70-year rotation [is] in excess of 40,000 years” (Stark 1980a). She concludes that nutrient shock is unlikely to occur with the removal of woody components only (risk of nutrient shock is greater if more than wood and bark is removed from the site), and is careful to point out that, to be ecologically balanced, the forest must have “normal” components of trees, shrubs, herbs, and microorganisms, not just wood (Stark 1980a).

In the typically N-limited forest soils of the Inland West, it is thought that light to moderate burning (in the absence of harvest, or as site preparation) without severe disturbance or losses of organic matter, should not have deleterious effects on productivity, although the need for more specific information regarding fire effects on specific soil components such as litter, humus, decayed wood and older residues is noted (Harvey et al. 1989). The E/L, Lick Creek study, with its focus on PMN and the active microbial fraction of SOM provides evidence of short and long-term decreases in these parameters with the potential for decreased productivity.
The research presented is inconclusive and incomplete if one considers the multitude of different combinations of variables and how they interact to bring about conditions on the landscape. Managers must try to use the information on the different effects of fire and logging on different soil types, in different ecosystems, under different burning conditions to try to optimize nutrient gains, minimize nutrient losses and achieve the desired above ground objectives in intensively managed forests (fuel reduction, thinning of brush, improved understory vegetation, thinning of saplings and seedling, and controlling stand destiny).
Appendix A

### Table 1.2. Trees per acre by diameter class midpoint, Lick Creek.

<table>
<thead>
<tr>
<th>DBH</th>
<th>4</th>
<th>8</th>
<th>12</th>
<th>16</th>
<th>20</th>
<th>24</th>
<th>28</th>
<th>32</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Contol</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>before harvest</td>
<td>48</td>
<td>41</td>
<td>30</td>
<td>16</td>
<td>4</td>
<td>3</td>
<td>0</td>
<td>1</td>
</tr>
<tr>
<td>after harvest</td>
<td>15</td>
<td>18</td>
<td>15</td>
<td>12</td>
<td>5</td>
<td>1</td>
<td>2</td>
<td>0</td>
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<tr>
<td><strong>Cut</strong> before harvest</td>
<td>43</td>
<td>35</td>
<td>29</td>
<td>16</td>
<td>7</td>
<td>4</td>
<td>2</td>
<td>0</td>
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<tr>
<td>after harvest</td>
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<td>18</td>
<td>15</td>
<td>12</td>
<td>5</td>
<td>1</td>
<td>2</td>
<td>0</td>
</tr>
<tr>
<td><strong>Cut+Burn</strong> before harvest</td>
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<td>46</td>
<td>48</td>
<td>23</td>
<td>8</td>
<td>4</td>
<td>2</td>
<td>0</td>
</tr>
<tr>
<td>after harvest</td>
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<td>15</td>
<td>18</td>
<td>14</td>
<td>3</td>
<td>4</td>
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<td>0</td>
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### Table 1.3. Inventory of slash (tops and small trees) left on site, Lick Creek.

<table>
<thead>
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<th>Diameter (in)</th>
<th>Cut + Burn</th>
<th>Cut Only</th>
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<tbody>
<tr>
<td>2 Whole</td>
<td>11</td>
<td>12</td>
</tr>
<tr>
<td>3 trees</td>
<td>4</td>
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</tr>
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<td>4</td>
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<td>5</td>
<td>3</td>
<td>4</td>
</tr>
<tr>
<td>6</td>
<td>9</td>
<td>5</td>
</tr>
<tr>
<td>7 Tops</td>
<td>6</td>
<td>4</td>
</tr>
<tr>
<td>8 Only</td>
<td>7</td>
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<td>9</td>
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### Table 1.4. Fuel moisture conditions, Lick Creek.

<table>
<thead>
<tr>
<th>Component</th>
<th>Moisture Condition</th>
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<td>Litter</td>
<td>12%</td>
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<tr>
<td>Lower duff</td>
<td>30%</td>
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<tr>
<td>Slash tops</td>
<td>75%</td>
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<tr>
<td>Old slash</td>
<td>48%</td>
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### Table 1.5. Fuel reduction, Lick Creek burn.

<table>
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<tr>
<th>Size</th>
<th>Pre-burn Tons/Acre</th>
<th>Post-burn Tons/Acre</th>
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<tbody>
<tr>
<td>Fine fuels</td>
<td>4.7</td>
<td>1</td>
</tr>
<tr>
<td>1-3&quot;</td>
<td>2.7</td>
<td>1.6</td>
</tr>
<tr>
<td>3&quot;+</td>
<td>1.9</td>
<td>0.1</td>
</tr>
<tr>
<td>total</td>
<td>9.3</td>
<td>2.7</td>
</tr>
</tbody>
</table>
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