SOIL AND LANDSCAPE FACTORS INFLUENCING FOREST PRODUCTION FOLLOWING NATURAL AND HUMAN-INDUCED DISTURBANCE

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SUMMARY OF MAJOR FINDINGS

1) Our objectives were to i) develop a soil-landscape model for Mixedwood landscapes in the Prince Albert Model Forest to serve as a baseline for our impact assessment and ii) assess the impact of clear-cut forest harvest on the soil and landsurface.

2) Forty sites were analyzed under Model Forest-administered research: 11 clear-cut sites, 26 mature forested sites, 1 burned site; as well, one mature site and one clear-cut site were re-sampled subsequent to being burned in 1995.

3) The development of the baseline for mature forested sites involved an initial assessment of the ecological groupings of forest stands in the Model Forest region. Two distinct groups emerged from our analysis of the range of soil and landform properties. The Rich group generally had higher levels of properties related to organic carbon storage as well as higher levels of properties related to mobile base cations; the Poor group had correspondingly lower levels of these components.

4) The two groups showed a distinct placement in terms of the local slope system - the Rich sites were adjacent to either small wetlands dominated by Balsam Polar, adjacent to larger wetlands dominated by Organic soil/fen or bog conditions or where located in landform with concave across- and downslope curvature. The Poor group sites were more widely distributed throughout the study landscapes, and occupied the higher positions in the local slope system.

5) The distribution of the two groups did not show a strong association with soil texture. Specifically, the Jack Pine sites were associated with glacial till soils, not with sandy Brunisolic soils as is commonly assumed.

6) We believe the two groups result from the pattern of base-rich groundwater discharge in these landscapes. The Rich group is associated with the groundwater discharge sites, and the Low group with recharge sites.

7) Six distinct Ecosites emerged from this analysis for the dominantly Mineral soil Ecosections of the Prince Albert Model Forest: Rich/Balsam Poplar, Rich/Mixedwood, Rich/Black Spruce. Poor/Mixedwood, Poor/Black Spruce, and Poor/Jack Pine. The quantitative characterization of soil conditions in these Ecosites formed the baseline for subsequent comparisons.

8) In both former Mixedwood and former coniferous clear-cuts, the most severe effects of clear-cutting occurred in the biochemical indicators of soil quality (i.e., those related to soil organic matter levels).

9) High losses of soil organic carbon (23.5% loss, from 57.7 to 44.1 Mg ha^{-1}) occurred in medium-term (6 to 20 years of recovery) Mixedwood clear-cuts, losses of soil N were comparable (27.1% loss, from 1.99 to 1.45 Mg ha^{-1}). The cation exchange capacity decreased by about 20%. The LFH layer decreased by 28% (from 8.1 to 5.8 cm).

10) High losses of soil organic carbon (34.5% loss, from 55.9 to 36.9 Mg ha^{-1}) occurred in coniferous clear-cuts, along with comparable losses of soil N (39.4% loss) and soil S (31.9% loss) and a 46.8% decrease in the cation exchange capacity of the upper soil increment. The losses in these biochemical components was due, in part, to a almost 54% decrease in the thickness of the LFH layer from 6.9 to 3.5 cm.

11) The losses of soil organic carbon and soil N place the clear-cut sites outside of the range of natural variation associated with mature forested stands.

12) A clear-cut site and a mature Mixedwood site which had been sampled in 1994 were burned by wildfire in the spring of 1995. Losses of soil organic carbon and nitrogen were much higher in the clear-cut than the mature forested site which may indicate a loss or resilience in the clear-cut site.

13) Losses of mobile base cations (calcium and magnesium) were evident in both Mixedwood and coniferous clear-cuts; however the decreases were within the range of natural variability as measured at the mature sites.

14) No significant changes in the levels of exchangeable potassium or soluble organic phosphorus due to clear-cutting was observed, and soluble inorganic P increased by at the clear-cut sites.

15) The levels of mineral nitrogen (ammonium and nitrate) were extremely variable. Generally the cationic ammonium form was much higher than the anionic nitrate form, and leaching losses of mineral N are unlikely to be high from these landscapes.

16) Nitrogen dynamics were more intensively examined at three sites: clear-cut, burned, and mature Mixedwood.

17) The spatial pattern of mineral N was not related to large-scale topographic variations in these landscapes.

18) The ratio of ammonium:nitrate narrowed at the clear-cut site and the size of the microbial biomass increased, changes were also observed in the potential N and C mineralization and in net nitrification.

19) The observed changes indicate the clear-cutting has had a direct effect on N cycling in these areas, and that the effect of clear-cutting differs from the effect of fire.

20) The very high losses of components from the biochemical reservoir of these soils may have negative implications for seedling growth and development at these sites which may impair the long-term sustainability of forest production following clear-cutting.

TABLE OF CONTENTS

Chapter **Page**

TABLE OF CONTENTS continued

Chapter **Page**

LIST OF TABLES

Table **Page**

LIST OF FIGURES

Figure **Page**

1. INTRODUCTION

The effects of clear-cut harvest of forest products on soils and related ecosystem components have been widely studied. Although several common effects have been established in these studies, the specific effects of clear-cutting on soils of any given region have proved to be difficult to predict. The objectives of our research were to i) develop a soil-landscape model for Mid-Boreal Upland Ecoregion contained within the Prince Albert Model Forest, ii) to use this model to develop a baseline of soil conditions within mature Mixedwood and non-Mixedwood forests and iii) to assess the impact of clear-cut forest harvest on the soil and landsurface.

Clear-cutting affects the rates of most soil processes operating within forested landscapes. One immediate impact of clear-cutting on nutrient and chemical flux is the loss of nutrients through removal of the biomass. The potential losses through biomass removal differ depending on the harvest method, management of the slash after harvest, and the specific nutrients involved. At a site similar to those discussed in our research, Alban and Perala (1990) examined the proportion of total nutrient reserves removed in whole-tree harvest of aspen; nutrient removal due to harvest comprised 5% of nitrogen reserves, 2% of phosphorus reserves, and 11%, 8%, and 2% of potassium, calcium, and magnesium reserves, respectively. The percentage of the reserves lost through harvest differs depending on soil type (e.g., Gordon, 1983; Timmer *et al.*, 1983); nonetheless nutrient export through harvest is appreciable.

A second immediate impact is on the physical properties of the soil surface and subsoil; these impacts occur primarily due to the ground pressure exerted by mechanical harvest systems (Greacen and Sands, 1980). The physical disturbance commonly increases the bulls density and soil strength and reduces the infiltration capacity, all of which affect the partitioning of water at the soil-atmosphere interface. The physical impacts are primarily concentrated in heavily used areas within the harvest block (e.g., skidder trails and landings); the landscape-wide impact tends to be less significant (Hatchell *et al.*, 1970).

The loss of the forest cover (and possible disruption of the ground cover) changes both the microclimatic and hydrological contexts within which the soil processes are operating. The removal of the forest canopy (and an appreciable percentage of the ground cover) increases the radiation reaching the soil surface and hence soil temperatures rise after clear-cutting.

Probably the most significant impact of clear-cutting from a soil quality perspective, however, is on the hydrological processes operating at the site. In a review of clear-cutting impacts in a range of environments, Neary and Hornbeck (1994) found a median water yield increase of 48% in the first year after clear-cutting; this increase is largely attributable to the impact of biomass removal on reduced transpiration rates. This increase in water yield becomes apparent as either an increase in surface runoff (and possibly in related water erosion) or as increases in the recharge of groundwater; the partitioning between the two depends on the physical nature of the soils, the degree of disturbance associated with the clear-cutting, and the nature of the landscape itself (e.g., the slope gradient and continuity).

For coniferous forest stands in the Model Forest Region, Kachanoski and de Jong (1982) showed that the effects of clear-cutting on the hydrological cycle were most evident in the snow-melt period. They found that clear-cutting increased hillslope water yield fivefold during the snowmelt period in the year following clear-cutting. Some effects were, however, evident throughout the growing season - for example, drainage through the soil was twice as high in the clear-cut sites as in the mature forested sites. Hence the potential for both surface and sub-surface transfers of solutes and colloidal material is much higher after clear-cutting.

The increased soil temperatures, radiation levels, and soil moisture combine to increase the mineralization rate of organic materials in the soil; however increased additions and decomposition of slash and incorporation of the surface organic layer into the mineral soil during harvest may, in some cases, increase soil organic matter levels. In a recent review of the literature available, Johnson (1992) concluded that the changes in soil organic matter in temperate and boreal forests due to harvest are generally on the order of ± 10% of pre-harvest levels.

The increase in vertical and lateral water movement through the soil greatly increases the potential for leaching of soluble soil components, and these leaching losses have been identified as a major consequence of clear-cutting (Johnson, 1994). Higher mobility and losses of both cations (e.g., NH_4^+, K^+, and especially Ca^{2+}) and anions (e.g., NO_3^- and $H_2PO_4^-$) are commonly observed after clear-cutting (Krause and Ramlal, 1986; Johnson, 1994).

The increased leaching of nutrients is commonly believed to be highest in the first few years after clear-cutting (e.g., Gordon, 1983; Alban and Perala, 1990). When the groundcover and regenerating tree species are established, transpiration rates increase and the water available for leaching decreases; as well, increased ion uptake by plants decreases the ions available for leaching. Hence Alban and Perala (1990) argue that accelerated leaching of nutrients below the rooting zone is often a short-lived phenomenon.

Previous studies have identified a range of possible impacts of clear-cutting on soil functions: physical impacts of mechanical harvesting systems; disruption of chemical and nutrient cycles leading to accelerated loss of ions by biomass removal and changes in addition and decomposition of organic residues; and disruption of the partitioning of water within the ecosystem through changes in transpiration rates. The indicators chosen in our study were selected to assess the impact of these disturbances on soil functions in the Mixedwood forest of Central Saskatchewan.

The existing literature also provides a range of approaches which can be used to assess the effects of forest harvest on soil condition. Doran and Parkin (1994) discuss two of the most common methods for establishing reference conditions against which the observed levels of soil quality indicators can be compared. The first is to use the soil characteristics of an undisturbed soil to establish reference conditions; the second is to compare the observed characteristics with a range of conditions which have been established to maximize productivity and environmental performance. The latter approach presupposes a thorough understanding of the optimum functioning of the soil, and for the Mixedwood forest of central Saskatchewan this understanding is lacking; hence we compare our observed levels of soil indicators after clear-cutting to conditions which exist in analogous mature Mixedwood forests in the region. These reference conditions are then compared with the soil quality conditions observed in the disturbed ecosystem.

As discussed above, the literature suggests that the rate of change in soil quality conditions due to clear-cutting is greatest immediately after the harvest, and that a reduced rate of change occurs in the medium- and long-term. Two major assumptions underlie the application of the undisturbed/disturbed comparison approach (Dyck and Cole, 1994): that all of the ecosystems under study were identical at time zero and have not been selectively affected by biological factors since time zero; and that climate has not changed and is similar for all sites used in the comparison. The requirement for an identical starting point appears unlikely under field conditions, but the onus is on the researcher to demonstrate at least a strong similarity among the research sites.

2. THE STUDY REGION

The study sites are located within the Prince Albert Model Forest, which is composed of the northern two-thirds of Prince Albert National Park and an area of commercial forest land leased by Weyerhaeuser Canada adjacent to and east of the National Park. The Model Forest extends from 53° 40' N to 54° 20' N and from 105° 10' W to 106° 40' W.

The study region (Figure 2.1) falls within the Mid-Boreal Upland Ecoregion of the Boreal Plain Ecozone (Padbury and Acton, 1994). The upland areas of this ecozone are characterized by a sequence of steeply sloping, eroded escarpments, hilly glacial till plains, and level, plateau-like tops. The intervening areas between the uplands are relatively level, with large, sparsely treed peatlands common. This Ecozone accounts for the majority of Saskatchewan's merchantable timber. The climatic data for Waskesiu Lake is the only continuous source of climate information for the study region. The total annual precipitation (based on 1961-1990 records) is 456 mm, which includes average winter snowfalls of 147 cm. The annual water deficit is 180 mm (substantially less than the 332 mm deficit characteristic of the Aspen Parkland Ecozone to the south). The mean annual temperature range is high: from a mean July temperature of 16.3 °C to a mean January temperature of -18.9 °C. The frostfree period is between 70 to 80 days.

2.1 Ecodistrict Classification

The study region is divided into three major Ecodistricts (called Landscape Areas by Padbury and Acton, 1994): the Waskesiu Upland, the Montreal Lake Plain, and the Whiteswan Upland .

The boundary lines used to delineate these Ecodistricts were taken from Padbury and Acton (1994) (Figure 2.2). It should be noted that the boundary lines for the units differ from the original physiographic map created by Ellis and Clayton (1970).

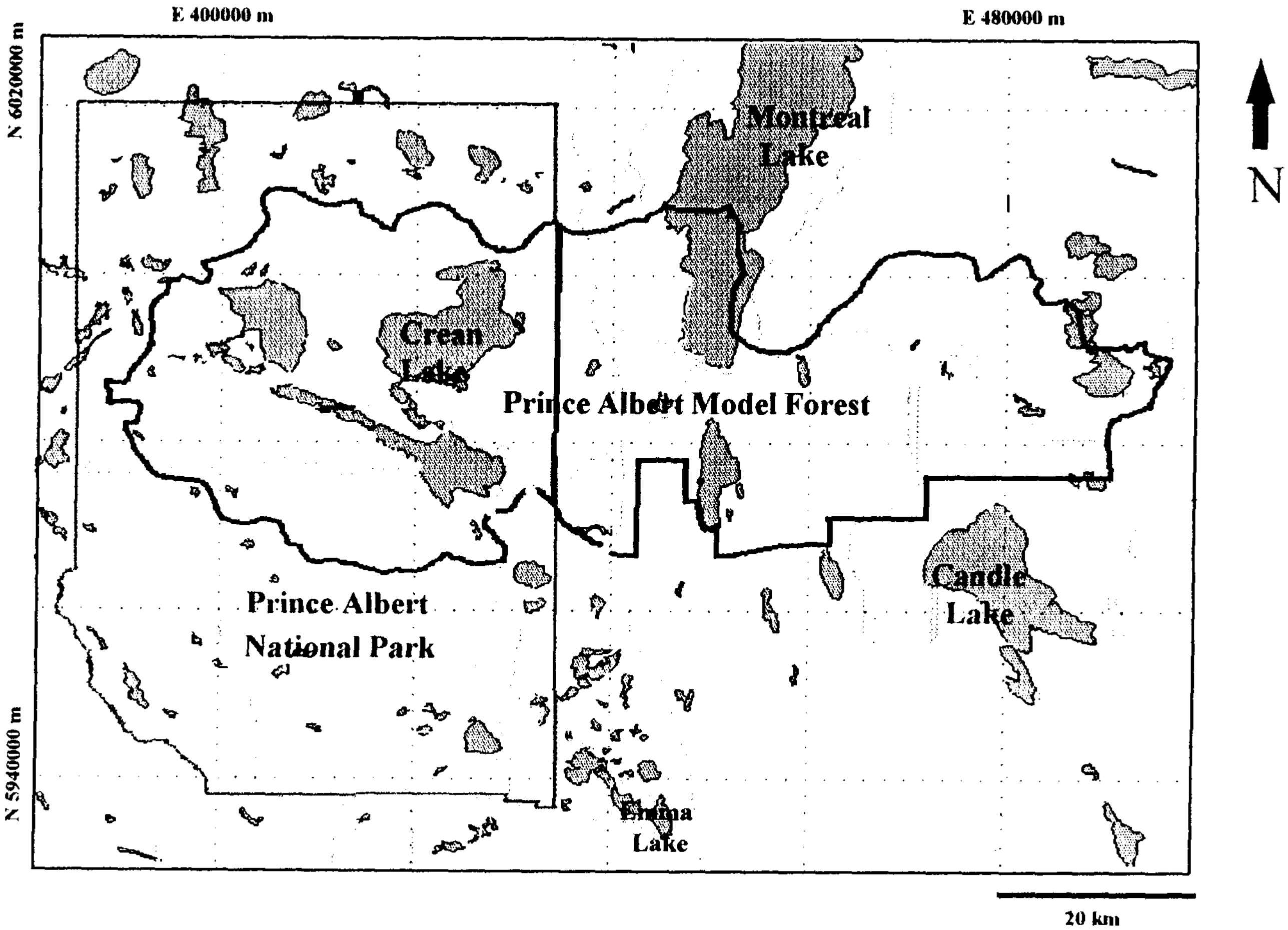
2.1.1 Waskesiu Upland Ecodistrict:

The Waskesiu Upland occupies the great majority of Prince Albert National Park in the study region; the exception is two intrusions of the Montreal Lake Plain along the Eastern Boundary of the Park. The Emma Lake Upland is presented as a separate Landscape Area in Padbury and Acton (1994); however, Ellis and Clayton (1970) viewed it as sub-division of the Waskesiu Upland. There is no distinctive physiographic separation between the Waskesiu and Emma Lake Uplands and they will be discussed as a single unit (i.e., the Waskesiu Upland) in this report.

The Waskesiu Upland ranges from elevation from approximately 520 m above sea level (masl) to a height of 720 m in the Waskesiu Hills. The segment of the Waskesiu Upland contains one escarpment system - the Waskesiu Escarpment occurs along the southern edge of Waskesiu Lake, and is a strongly dissected morainal escarpment with slopes ranging from 6 to 30% (Padbury *et al.*, 1978).

The landscapes of the Waskesiu Upland Ecodistrict are dominated by a series of knolls, mid-slopes, and depressions typical of morainal landscapes throughout Central and

Figure 2.1
Map of the Prince Albert Model forest and surrounding region. Map coordinates are eastings and northings for the Universal Transverse Mercator Grid, zone 13 U.



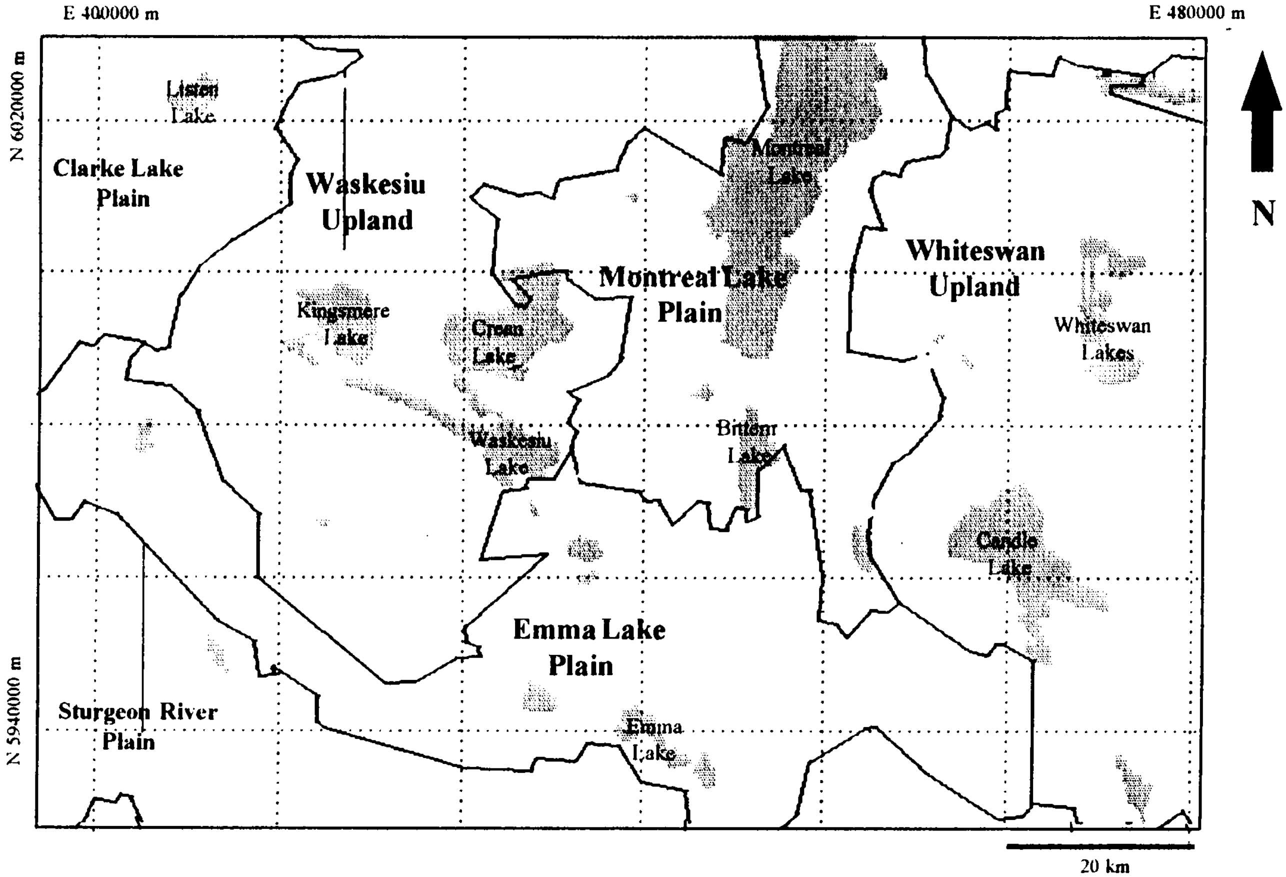


Figure 2.2
Map of the ecodistricts within the Prince Albert Model Forest. From Padbury and Acton (1994) . Map coordinates are UTM eastings and northings, zone 13U.

Southern Saskatchewan. The glacial sediments in this Ecodistrict range between 375 m (in the center of the Upland) to 200 m thick east of Waskesiu Lake. Christiansen (in Padbury *et al.*, 1978) argues that Waskesiu Lake and the Waskesiu Hills are an ice-thrust take/moraine sequence. As the ice-sheet advanced towards the upland, compressive flow at the margin of the ice sheet caused the slabbing-off of sediment at the point of greatest compressive flow; these slabs were then thrust onto the upland, creating the Waskesiu Escarpment and Hills; the depression remaining where the slabs had been excavated created Waskesiu Lake.

2.1.2 Montreal Lake Plain Ecodistrict:

The Montreal Lake Plain Ecodistrict lies between 485 and 530 mast and is bounded on the west by the Waskesiu Upland and on the east by the Whiteswan Upland Ecodistricts. The Plain overall is a gently undulating to rolling, stream-modified till plain with local glacio-fluvial features (Ellis and Clayton, 1970).

In the southern portion of the Ecodistrict (termed the Bittern Lake Plain by Ellis and Clayton, 1970), the till and glacio-fluvial features form elongate, north-south trending ridges of low-relief (generally 3 to 10 m). These ridges are separated by very gently undulating till plains often occupied by organic soils. Ellis and Clayton (1970) suggest that this area may have served as a meltwater channel during the glacial retreat from the area.

The area of the Plain west of Montreal Lake is a gently to undulating till plain which slopes from the Waskesiu and Thunder Hills upland towards Montreal Lake, and is bisected by a number of rivers such as the Waskesiu and the Maclennan rivers. The Montreal Lake Plain east of the lake is a relatively uniform, gently undulating till plain with very limited amounts of organic soils. The Model Forest boundaries largely exclude the portion of the Montreal Lake Plain east of Montreal Lake.

2.1.3 Whiteswan Upland Ecodistrict:

The Whiteswan Upland Ecodistrict within the Prince Albert Model Forest lies between 510 and 600 masl and has a very poorly defined boundary with the Montreal Lake Plain Ecodistrict to the west. The original boundary drawn by Ellis and Clayton (1970) generally followed the 1700 ft. contour line on the existing topographic maps; however, this contour line on subsequent maps departs greatly from the source they used and hence the location of the boundary is very problematic.

The Upland is a roughly undulating to moderately rolling morainic upland which lacks the characteristic N-S trending ridges of the Montreal Lake Plain. The medium-textured glacial till is frequently overlain by a mantle of 15 to 60 cm of moderately fine textured materials of unspecified origin (Head *et al.*, 1981). Overall the Whiteswan Upland within the Model Forest region is devoid of major topographic features, and a gradual decrease in topography occurs from the eastern portion to the contact with the Montreal Lake Plain.

3. MATERIALS AND METHODS

3.1 Study Sites

We have examined soil and landscape properties at a total of forty sites in the Model Forest Region. The Model Forest research program has involved intensive soil and landscape sampling at 31 clear-cut sites. Seven of these were in former Mixedwood landscapes including two samplings at site MF4, prior to harvest in 1993 and the year following harvest (1994). In the 1995 sampling program, four sites were sampled in former coniferous-dominated landscapes. Three burned sites were also examined: one that was burned in 1989, and two sites that had been sampled in 1993 were re-sampled after they were burned in the summer of 1995.

Our research under the Natural Resources Canada Ecological Land Classification program (administered by the Prince Albert Model Forest Association) was carried out at 13 mature Mixedwood sites in the summer of 1994 and at 12 coniferous dominated sites in the summer of 1995. As part of the 1995 sampling program a further two mature Mixedwood sites were sampled to complete a vegetation continuum at the sampling areas. This program involves a less intensive sampling program with only 9 sampling points in a 3 X 3 sampling grid. These research is designed to more fully characterize the distribution of ecological units in the Model Forest region and provides a baseline for our assessment of clear-cut impacts under the Model Forest initiative.

3.2 Sampling Design

The sampling design ensured sufficient sample numbers to make a landscape-scale assessment of the soil quality trends. Landscape-scale research requires that all structural components of the landscape are covered by the sampling framework; in our sites we chose 200 x 200 m study areas which contained several knoll-depression cycles for topographic survey and placement of the sampling grids.

At all sites, a square or rectangular grid sampling design was used; however the number of specific sampling points in the grid differed between the sites. In the summer of 1993, two intensive grids were laid out and sampled: a 13 x 13 grid with 7.5 m spacing between adjacent points within a mature Mixedwood landscape (ELC1) and a 7 x 7 grid with 15 m spacing in a recent clear-cut site (MF1). The results from the mature Mixedwood sites were analyzed using geostatistical techniques and landscape-classification approaches (Pennock *et al.*, 1987, 1994) and based on the results of these analysis a 3 x 3 grid design with 15 m spacing between successive points was adopted for subsequent sampling of the mature Mixedwood sites (designated as ELC sites in the subsequent discussion).

The range of disturbance conditions was much greater at the clear-cut sites (designated as MF sites) and we felt that high sample numbers were required to sample the range of disturbance conditions. Based on our analysis of the MF1 site, we adopted a standard 6 x 6 sampling design with 15 m spacing between successive points; in the case of site MF7 the placement of tree survival plots with the landscape necessitated the use of a 4 x 11 grid, although again with 15 m spacing.

A complete topographical survey was made at each study site and soil pits dug at each of the sampling points. At clear-cut sites that had been prepared for tree planting, the pits were located in the undisturbed strips between adjacent trenches and mounds so as to exclude site preparation effects on soil properties. In each pit the soil horizons were described and three samples (0-15, 15-30, and 30-45 cm) were taken using a core. The upper increment included both the surface organic layer and the mineral soil surface. No attempt was made to separate the surface organic layer and the mineral soil for sampling because a) at many sampling points in the clear-cut sites the surface organic layer was missing (which introduces major difficulties when comparing clear-cut and mature Mixedwood sites) and b) measurement errors associated with the generally thin surface organic layers can introduce major errors into subsequent analysis based on a mass per area basis

3.3 Laboratory Analysis

The physical properties selected for inclusion were soil bulk density, gravimetric moisture (for correcting ion concentrations), and particle size distribution. Bulk density and gravimetric moisture were calculated for all samples after oven-drying. Particle-size analysis using a modified pipette method (Indorante *et al.*, 1990) were calculated for selected sub-samples from each site. Three sample points were selected from each of the mature Mixedwood sites; ten sampling points were selected from each clear-cut site. To make the selection, each site was stratified by soil order, and then sampling points were randomly selected from each soil order proportional to their occurrence at the site.

The soil biochemical measurements were soil organic carbon (SOC), total nitrogen, soil ammonium and soil nitrate. Soil organic carbon was assessed using a LECO CR-12 Carbon Determinator (LECO Corp., St. Joseph, MI, U.S. A.). Total N was determined by mass spectrometry using a ANCA-MS system (Europa Scientific Ltd., Crewe, U.K.).

To minimize the possibility of nitrification occurring in the samples after sampling, 200 ml of a 2N KC1 solution was added in the field to 50 g soil samples from each sampled increment. After filtering, ammonium and nitrate levels were assessed using an Technicon Autoanalyzer II Single Channel Colorimeter (Technicon Instruments Corp., Tarrytown, NY). The ammonium and nitrate values were subsequently corrected to an air-dry soil basis.

Soil pH was measured on each sample using a 1:2 soil-to-water mixture and a digital pH meter. Total and inorganic bicarbonate-soluble P were determined spectrophotometrically (E. P. A., 1971). These soluble P fractions reflect the most labile P pool in the soils and hence were judged to be the most responsive to disturbance. Exchangeable cations were determined using the barium chloride-triethanolamine method (McKeague, 1978) followed by flame emission (Na, K) and atomic absorption (Mg, Ca) analysis on a Perkin-Elmer 3100 Atomic Absorption Spectrophotometer (Perkin-Elmer Corp., Norwalk, CT). Cation Exchange Capacity was subsequently determined by leaching the sample with ammonium acetate (McKeague, 1978) and using flame emission to measure barium concentration.

3.4 Statistical Analysis

Originally our intention was to stratify the clear-cut grids using a quantitative landform classification approach; however (as is discussed below) the stratification based on landform morphology was not necessary in these landscapes. Instead we viewed each individual site as a distinct, spatially independent replicate of the range of soil conditions present within the Model Forest Region, and the soil characteristics assessed for the individual sampling points within a site (i.e., 9 sampling points in mature sites and 35 to 49 points in clear-cut sites) were combined to give one value for that sampling site in subsequent statistical comparisons. Many of the sampling distributions for specific soil characteristics were highly skewed, and the median value for each study site was used as the measure of central tendency in subsequent analysis.

The initial stage of the analysis was to compare the levels of the soil properties at the mature sites against those for the clear-cut sites. The comparisons were made using a t-test following an assessment of the equality of the variances of the two groups using Levine's test (discussed in Snedecor and Cochran, 1989).

No a priori significance level was selected for this study; instead the significance level for each comparison is reported. This approach was used to address the concerns of authors such as Peterman (1990) concerning the importance of statistical power in conservation or environmental studies. Peterman (1990) argues that in conservation related research the practical consequences of a Type II error (i.e., of failing to detect a difference which did, in fact, occur in nature) may be much graver than the consequences of a Type I error (i.e., of detecting a difference which did not, in fact, occur in nature).

The probability of committing a Type 11 error is related to the a level chosen, the variability within the sampled groups, the number of replicates, and the effect size (or the actual difference in mean levels of the property in the two groups being compared). Generally in landscape-scale studies the variability within the sampled groups is high and the number of replicates is often low; hence an unquestioning acceptance of an 0.01 or 0.05 significance level may mean that even large effects size (i.e., large differences in group means) are undetectable.

The results of the statistical comparisons are further examined using boxplots of the range of values from the individual sampling sites. These plots are used in a manner analogous to the control charts discussed by Larson and Pierce (1994); basically the range of values for the mature Mixedwood sites provides a measure of the undisturbed or natural range of variability for the specific soil quality indicators. The range of values in the disturbed sites can then be compared to this natural range to assess the possible importance of the observed differences.

Boxplots (or box-and whisker diagrams) are a very useful tool to summarize both the central measure of a data set (the median) and the dispersion of values around the median. Boxplot construction begins by dividing the data set into four equally sized groups. Each of these are a quarter of the data set, and are formally called quartiles. The 50th percentile (the boundary between the second and third quartiles) is called the median, and is equal to the mean for normally distributed data sets. The measure of dispersion used is the interquartile range, which is range between the observation at the 25th percentile and the observation at the 75th percentile. Note that unlike the standard deviation, the interquartile range does not have to be symmetrical around the median.

Outliers (unusually high or low observations) in the data set are identified as those observations which are higher or lower than a pre-set threshold. Two thresholds are used: outliers, which are observations < or > 1.5X the interquartile range (IQR) away from the median, and extreme outliers, which are observations < or > than 3.0 X the IQR away from the median.

The boxplot allows us to graphically represent these summary statistics. First the median is represented by a line, and the 25th and 75th percentiles form the box around the median. The whiskers extend to the lowest and highest observations which are not outliers. Outliers are represented by an asterisk (*) and extreme outliers by a circle (o).

4. OVERVIEW OF THE SOILS AND PARENT MATERIALS IN THE STUDY REGION

Soils of three soil orders occur in the non-peatland soft landscapes in the Model Forest Region. The soils of the Luvisolic and Brunisolic Orders are associated with dominantly aerobic pedogenic environments and the soils of the Gleysolic Order with dominantly anaerobic environments.

Luvisolic soils in Saskatchewan are classified into the Gray Luvisolic great group in the Canadian System of Soil Classification. These soils typically have a LFH (or leaf litter layer) horizon, and a sandier Ae horizon which overlies a clay-rich Bt horizon. The Bt horizon typically overlies a C horizon with either primary or secondary carbonates, and the map units of the Luvisolic soils in the Model Forest Region are differentiated in part on the basis of the calcium carbonate content of the C horizon.

Brunisolic soils in the study region are primarily classified into the Eutric Brunisolic great group, which indicates that the dominant pH is greater than 5.5 and the A horizon is either absent or less than 10 cm thick. The Brunisolic soils are associated with high sand, glacio-fluvial sands in these landscapes. The lack of clay (which is inherent to the parent material) disallows the development of the diagnostic Bt horizon. Gleysolic soils are associated with landscape positions where water occupies the pore space of the profile for a sufficient length of time that anaerobic conditions are achieved. The reducing conditions associated with these anaerobic conditions lead to the mobilization and re-precipitation of iron, which causes the reddish-brown mottles and dull matrix colours characteristic of the Gleysolic Order.

The onset of anaerobic conditions can occur due to i) very slow internal drainage of water through the profile (which is normally associated with finer soil textures), ii) the accumulation of a greater volume of water on the soil surface than can be readily transmitted through the soil (associated with the redistribution of surface runoff or throughflow into low-lying areas), or iii) the discharge of groundwater from depth into the soil pedon (or seepage). The latter two causes of Gleysolic soil formation are associated with topographically influenced redistribution of water; the former cause is associated with the texture of the parent materials

The Gleysolic soils in our research sites appear to be dominantly associated with topographically influenced redistribution of water, not with finer textured parent materials. The basis for this statement can be illustrated by examining the textural range associated with the three major soil orders throughout the study area (Figure 4.1). The Brunisolic soils are dominated by sandy loam to loamy sand materials in the upper solum; the range for the Gleysolic and the Luvisolic soils are very similar and fall into the loam range of the textural triangle. Given that the loam range is not traditionally associated with drainage problems and because of the lack of a distinctively finer sediments in the Gleysolic range, it seems unlikely that textural differences are the cause of the occurrence of the Gleysolic soils. Nor do they appear to be associated with finer textured materials at depth. Samples were not taken beyond 45 cm, but our soil pit descriptions indicate no consistent evidence of the occurrence of impeding layers at depth associated with either of the groups.

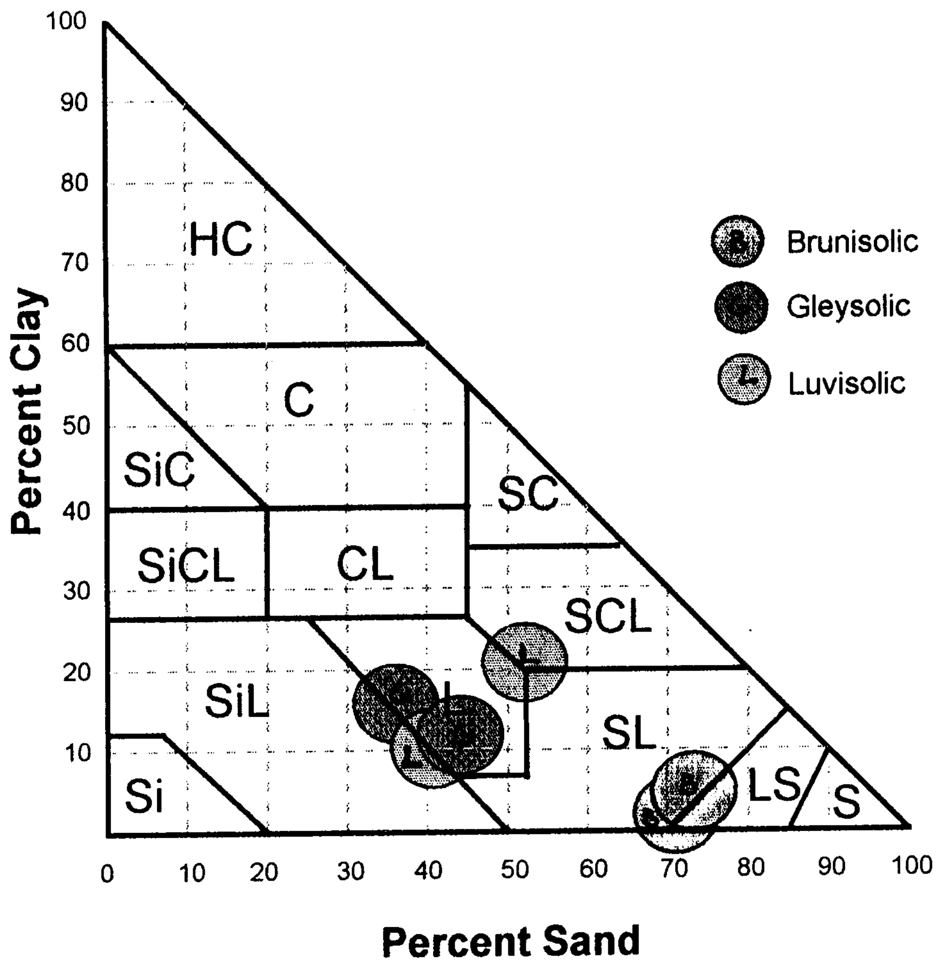


Figure 4.1
Textural triangle showing the median texture (±-5 %) at the 15 to 30 cm and 30 to 45 cm depths for each soil order.

The distribution of the dominantly aerobic soils, the Luvisolic and Brunisolic soils, shows a distinct regional pattern. This pattern occurs because of the strong parent material control on the occurrence of these two orders- the Luvisolic soils are associated with positions where glacial till occurs within the solum; the Brunisolic soils are associated with sampling points where the solum occurs entirely within glacio-fluvial sediments. In the Luvisolic soils, the surface of the glacial till is the top of the higher clay Bt horizon; where the parent materials are entirely glaciofluvial, there is insufficient clay in the profile for its eluviation from the Ae and illuviation in a Bt to be detectable.

In many of the study landscapes a silty-very fine sandy mantle is found overlying the till and the medium-coarse sand glacio-Fluvial parent materials. The mantle ranges from 5 to 90 cm in thickness. It commonly forms the Ae horizon of the Eluviated Eutric Brunisol soils, the Ae of the Orthic Gray Luvisolic soils, and the Ae/Bm horizons of the Brunisolic Gray Luvisolic soils. In some locations (especially in the central part of the study region), the surface mineral material has a very eolian/loess-like appearance (I. Corns, Pers. Comm.); however in other areas the stone content would argue against an eolian origin. Eolian caps are also common in forested soils in Minnesota (D. Grigal, pers. comm.) and stones are attributed to tree-throw and other forms of pedoturbation.

The occurrence of a specific aerobic soil order at a given site is therefore highly correlated to the occurrence of the parent materials - Luvisolic soils at sampling points where glacial till occurs within the solum and Brunisolic soils where glaciofluvial sediments occur. In the Waskesiu Upland (including the Emma Lake Upland), the landscapes are dominated by glacial till parent materials, and hence the research sites which occur in this area are dominated by Luvisolic soils and by smaller extents of Gleysolic soils in well-defined lower slope segments (Figure 4.2). If Brunisolic soils occur (e.g., ELC 1) they are confined to well-developed paleo-channel features in the landscape and hence are readily identifiable.

In the Montreal Lake Plain. and the sampled area of the Whiteswan Upland a much more complex mixing of glaciofluvial and till sediment occur. The study landscapes are dominated by Luvisolic soils in the aerobic positions, but significant inclusions of the sandy, glaciofluvial parent materials (and hence Brunisolic soils) occur Figure 4.3). The sandy, glacio-fluvial inclusions do not show any association with landform morphology as assessed using a quantitative landform classification system (discussed in Pennock *et al.*, 1987 and 1994). A strong landform morphology-soil distribution relationship had previously been shown to exist in this region by Donald *et al.* (1993); however their research site was a uniform till surface, and as Hairston and Grigal (1994) have shown, differences in the nature of the parent sediments can greatly alter the nature of the landform morphology- soil property relationship.

In summary, two basic landscape-scale soil distribution models were developed for the study region. The position of the Gleysolic soils is common to both - they occur in landscape positions where water concentration is high, due either to groundwater or hillslope hydrological processes. The occurrence of aerobic soils differs between the Ecodistricts- in the Waskesiu Upland the aerobic soils are dominantly Luvisolic soils on till parent materials; in the Montreal Lake Plain and Whiteswan upland a complex mixture of Brunisolic and Luvisolic soils occurs.

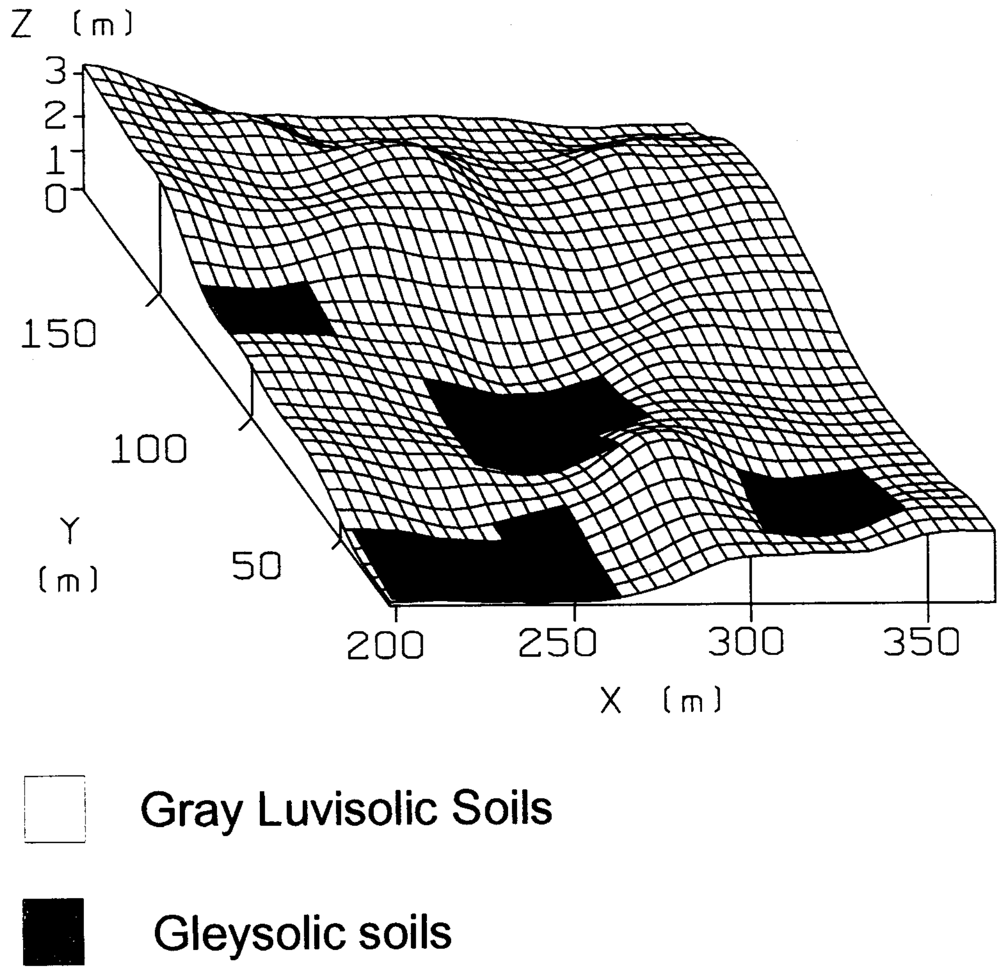
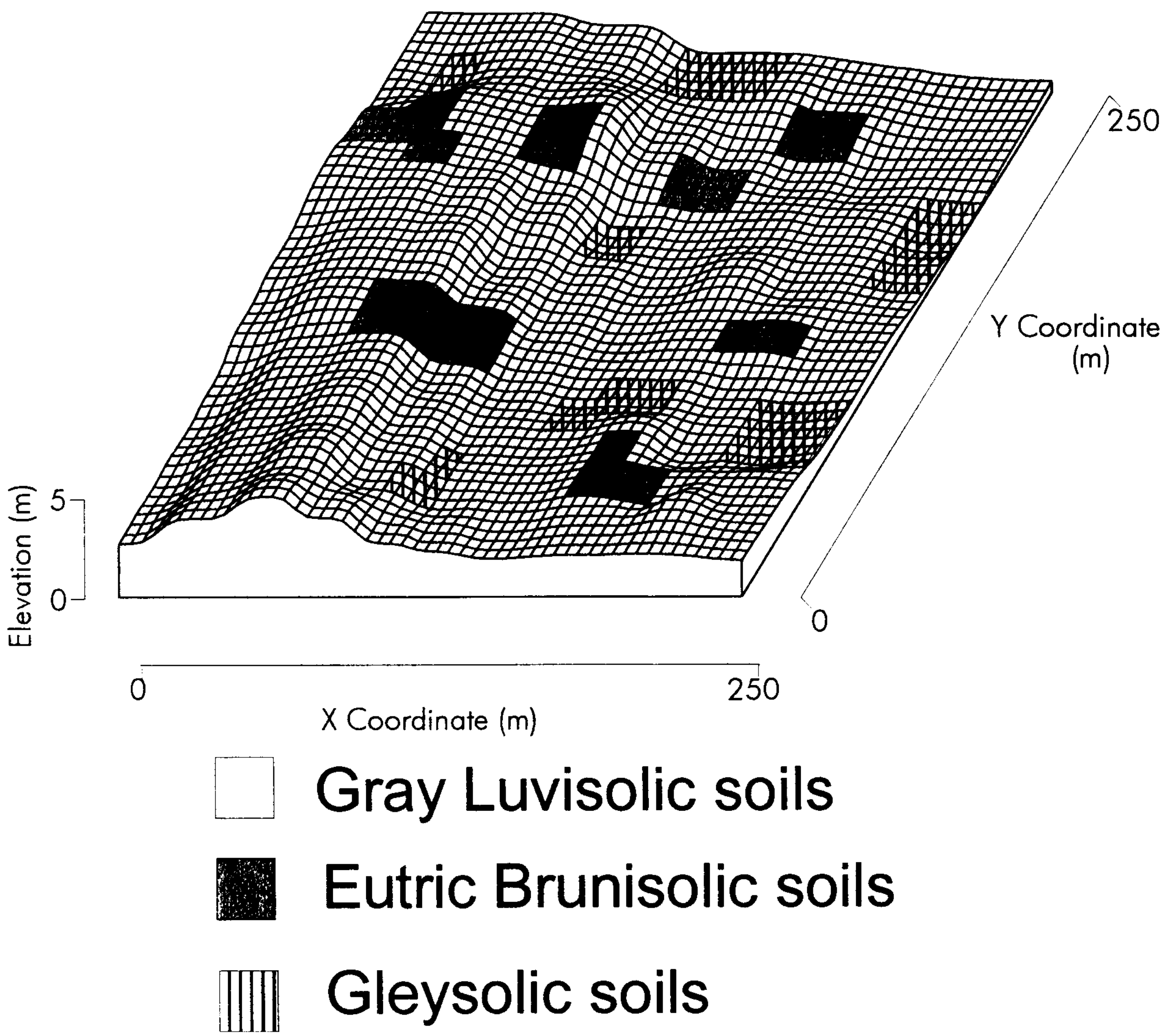


Figure 4.2:
Wakesiu Hills Soil Distribution Model


250
Y Coordinate (m)
0
Elevation (m)
5
0
0
250
X Coordinate (m)
Gray Luvisolic soils
Eutric Brunisolic soils
Gleysolic soils

5. DEVELOPMENT OF A BASELINE FOR SOIL QUALITY CONDITIONS IN MODEL FOREST LANDSCAPES

5.1 Baseline Site Selection

The specific sampling sites for baseline development were selected in an effort both to represent the range of natural site conditions found across the study region and to sample from each of the three Ecodistricts discussed above. In order to reduce variations due to stand age it was decided to select only those sites classified as mature stands (i.e., those greater than 60 years in age).

Potential field sites were selected using the 1:12,500 Forest Inventory maps produced by the Forestry Branch, Saskatchewan Environment and Resource Management. After field inspection of the potential sites a specific location within the stand was selected for sampling.

The sites sampled were chosen in accordance with the moisture-nutrient grid developed for the study region by Beckingham *et al.* (1995) (Figure 5.1). The shaded portions of the grid indicate the sites associated with dominantly organic Soils which were excluded from the sampling program. Sites which were included in this study are identified as having a dominant tree over of either Jack Pine, Trembling Aspen, Balsam Poplar, or Mixedwood stands of Trembling Aspen and White Spruce (although some Jack Pine occurred in the latter group).

The majority of Mixedwood sites (ELC 1 through 14) were selected to range from the topographic high to the low point at each site, although this was commonly only an elevation difference of 3 to 5 m (Table 5.1). Sites ELC 21 through 25 formed a toposequence from a major topographic high (ELC 22) through to the Balsam Poplar dominated site ELC 25 in a depressional area. The total elevation difference from ELC 22 to ELC 25 was approximately 10 in. Three other groups of sites in distinct toposequences were also sampled: ELC 28 and 29, ELC 31, 32, and 33, and ELC 33, 34, 35. In all of these cases the difference in elevation from high to low was only on the order of 2 to 3 m.

At sites ELC 41 through 45, soil pits were dug and the vegetation and soils described, but no samples were taken. These sites were used to extend the sampling program into areas of the Model Forest which were under-sampled in the course of the study.

5.2 Development of Baseline and Grouping of Research Sites

The data analysis involved three distinct steps. In the first, the results of the data set as a whole were examined for evidence of groupings of data or patterns using the tools of Exploratory Data Analysis (EDA) such as the boxplot. Then these potential groups were examined more rigorously using the standard statistical tools of Confirmatory Data Analysis such as the t-test. Finally, the groups which emerge from this analysis are interpreted in terms of the possible landscape controls on their occurrence, and the spatial implications of the groups are determined. In this way we believe a quantitative, ecologically relevant classification of the data can be achieved.

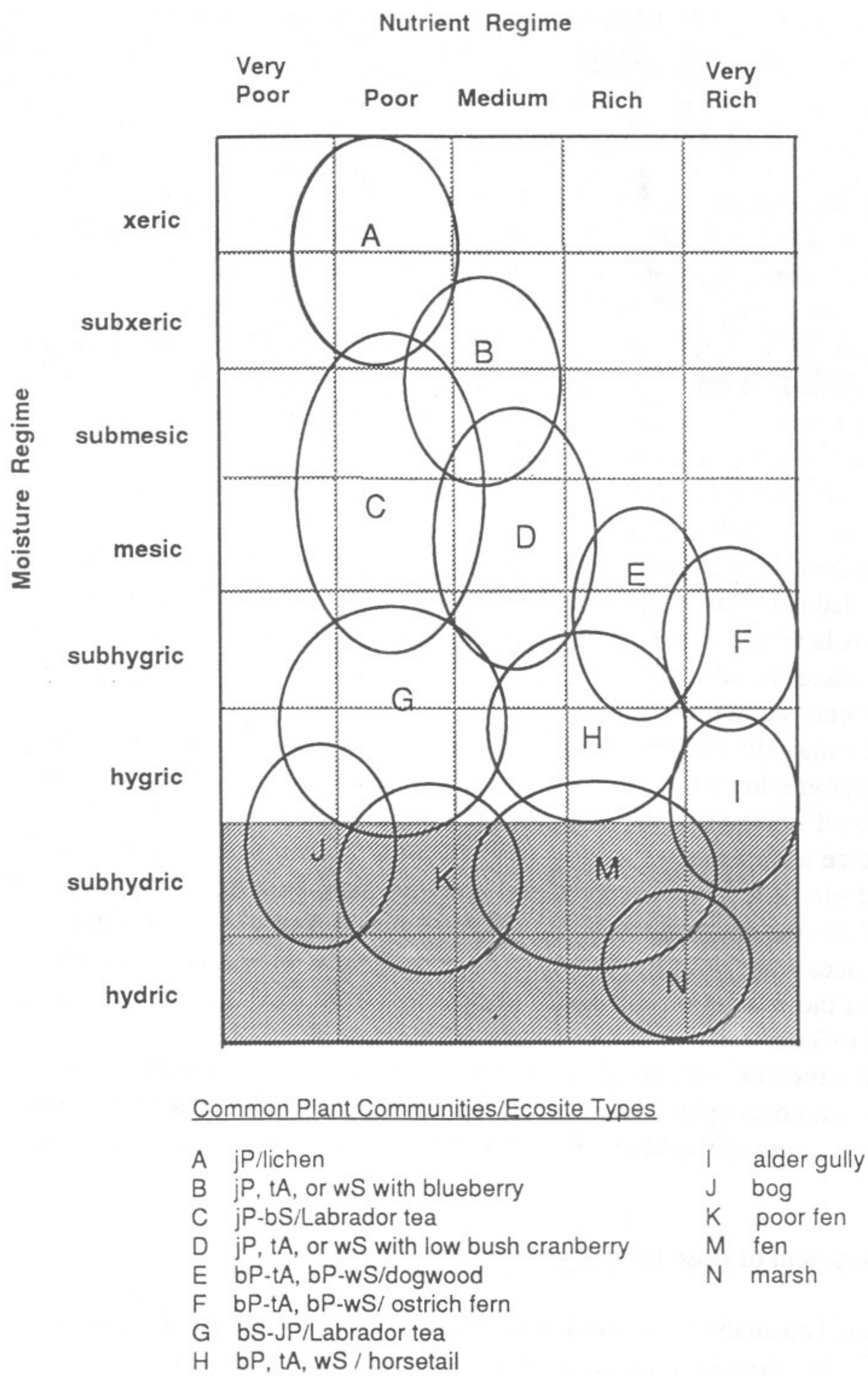


Figure 5.1 Moisture-nutrient grid for site conditions in the Prince Albert Model Forest (from Beckingham *et al.*, 1995). The gray tones identify sites dominated by organic soils which are excluded from this study.

Table 5.1:
Listing of sites, UTM coordinates (UTM zone 13), and dominant tree species.

Site	Easting	Northing	Tree Species
ELC 2	415250	5980700	MW[1]
ELC 4	420100	5977100	MW
ELC 5	427600	5971800	MW
ELC 6	441270	5975050	MW
ELC 7	445300	5950600	MW
ELC 8	441600	5981600	MW
ELC 9	471580	5987298	MW
ELC 10	468900	5989450	MW
ELC 11	469900	5988750	MW
ELC 12	484250	5977100	MW
ELC 13	439250	5967300	MW
ELC 14	449300	5932400	MW
Toposequence 1: ELC 21 to 25 Waskesiu Upland			
ELC 21	441549	5960951	bS
ELC 22	441477	5960935	tA (MW)
ELC 23	441208	5960723	jP
ELC 24	441177	5960774	bS
ELC 25	441131	5960778	bP
ELC 26	440923	5964885	jP-bS
Toposequence 2: ELC 28, 29 Montreal Lake Plain			
ELC 28	431713	5974127	JP
ELC 29	431751	5974136	bS
Toposequence 3: ELC 31, 32. 33 Montreal Lake Plain			
ELC 31	456890	5989457	JP

Table 3 cont.

Site	Easting	Northing	Tree Species
ELC 32	456905	5989311	bS
ELC 33	456944	5989236	bP
Toposequence 4: ELC 34, 35. 36 Montreal Lake Plain			
ELC 34	463209	5969419	bS
ELC 35	453272	5969408	jP
ELC 36	463314	5969354	MW
ELC 41[2]	483947	5986157	jP-bS
ELC 42	480580	5989576	jP
ELC 43	421571	5981448	MW
ELC 44	438060	5000211	JP
ELC 45	457957	5978210	JP-bS

1. Key to species: bS-Black Spruce, tA-Trembling Aspen, MW-Mixed Wood, jP-Jack Pine, bP-Balsam Poplar.

2. At sites 41 through 45 only a preliminary field investigation was conducted. to examine vegetation/soil/landform relationships, no laboratory analysis was done for these particular sites.

5.2.1 Grouping of Ecosites

The EDA of the data set indicated that at the broadest level there were two distinct groups of sites. The first group had high levels of soil organic carbon (SOC), soil nitrogen, and soil sulphur, thicker LFH horizons and hence lower bulk densities, higher pH at all depths, and higher amounts of exchangeable calcium, magnesium, and potassium. The second group had lower levels of all these components.

The process of delineating these groups can be illustrated with the boxplots for SOC by site (Figure 5.2), Two groups are generally apparent: Sites ELC 1, 5, 6, 7, 8, 9, 10, 12, 13, 21, 22, 23, 26, 28, 31, and 35 have median values below 75 Mg ha^{-1} yr^{-1} and have a small degree of variability (i.e., the box and whiskers are confined to a narrow range of values). Sites ELC 2, 4, 14, 24, 25, 29, 32, 33, 34, 36 have median levels above 75 Mg ha^{-1} yr^{-1}, and generally have much higher variability associated with them.

The same basic grouping is apparent for many of the remaining properties. For example, the same grouping of sites is apparent in the boxplots for exchangeable calcium to 45 cm - the first group has median levels above 10.0 Mg ha^{-1} to 45 cm; the second group has median levels below this (Figure 5.3). For the pH of the 30- to 45-cm increment, a similar pattern emerges - the first group has pH values close to or above 7.0; of the second group all but ELC 26 have values below 7.0 (Figure 5.4).

Because of the generally higher levels of the soil properties in the first group of sites it was termed the Rich group; the second group was termed the Poor group. Note, however, that sites in the Xeric portion of Beckingham's grid were not sampled as part of this study, and hence the Rich/Poor gradient in our study in smaller than that of the Mixedwood Region as a whole.

5.2.2 Testing of the Ecosite Groupings

The purpose of the EDA is to suggest broad groupings of sites into those that show a similar pattern in the assessed properties. In the Confirmatory Data Analysis, these preliminary groupings are subjected to more rigorous statistical testing to determine if, in fact, a distinct range of properties is associated with them. To accomplish this, the median values for each of the properties from each site was calculated (the medians are used in preference to the means because of the high skewness inherent to many of the variables on a site basis). Hence in this analysis there are 16 median values for sites classified into the Poor group for each property, and 10 median values for sites in the Rich group. These are then analyzed using a simple t-test for to test the null hypothesis that there are no significant differences between the mean values for the properties (i.e., the mean values for the medians of each site in the group) between the two groups (Table 5.2),

The results of the simple difference testing bear out the potential groupings derived from the EDA. Of the biochemical indicators, SOC and soil N show a three-fold difference among groups; the soil S and LFH thickness are approximately twice as high in the Rich group. Soil pH is consistently higher at all three depths in the Rich group, and the difference increases from 0.5 pH units at the surface to almost 2 pH units in the 30 to 45-cm increment. Of the exchangeable cations, sodium, calcium, and magnesium all show substantially higher levels in the Rich group; the latter two show a three-fold increase in the Rich group.

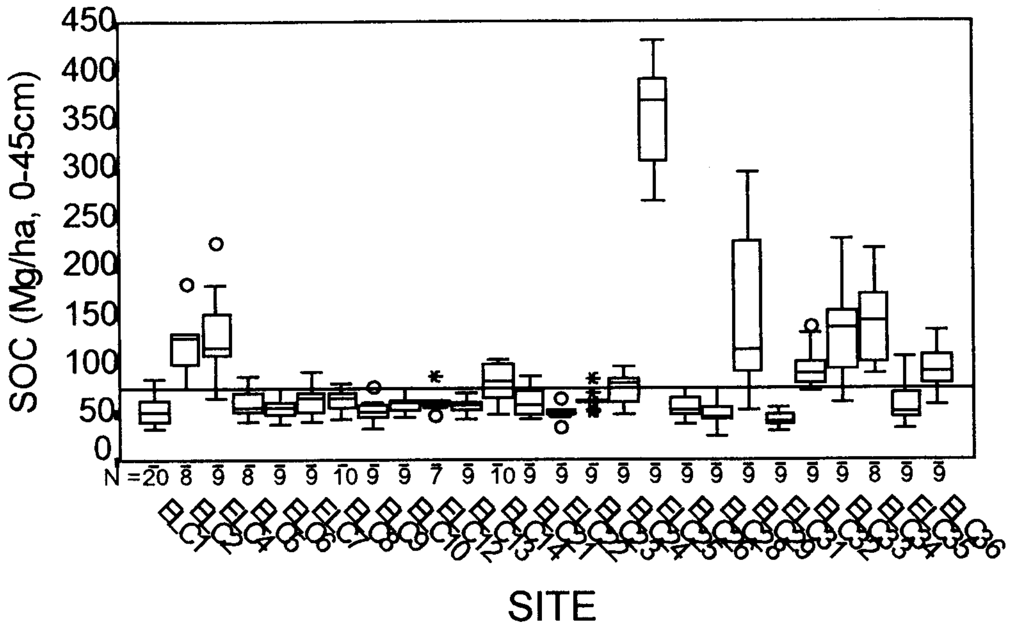


Figure 5.2:
Boxplots of the soil organic carbon values associated with the mature forested sites. The horizontal line approximately delineates the boundary between the Rich and Poor group of ecosites.

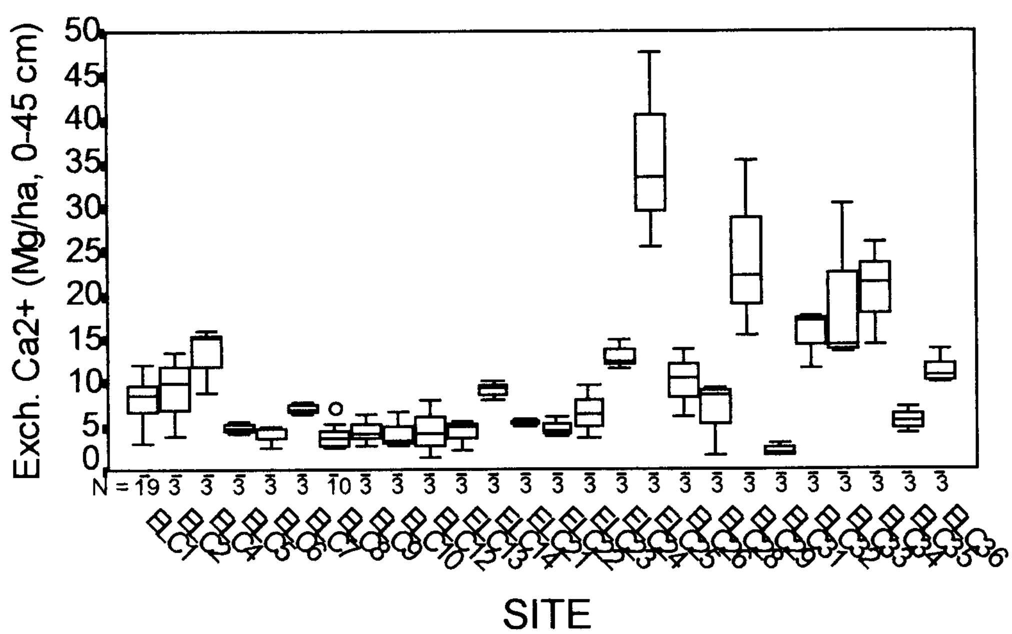


Figure 5.3:
Boxplots of the exchangeable calcium values for the mature forested sites.

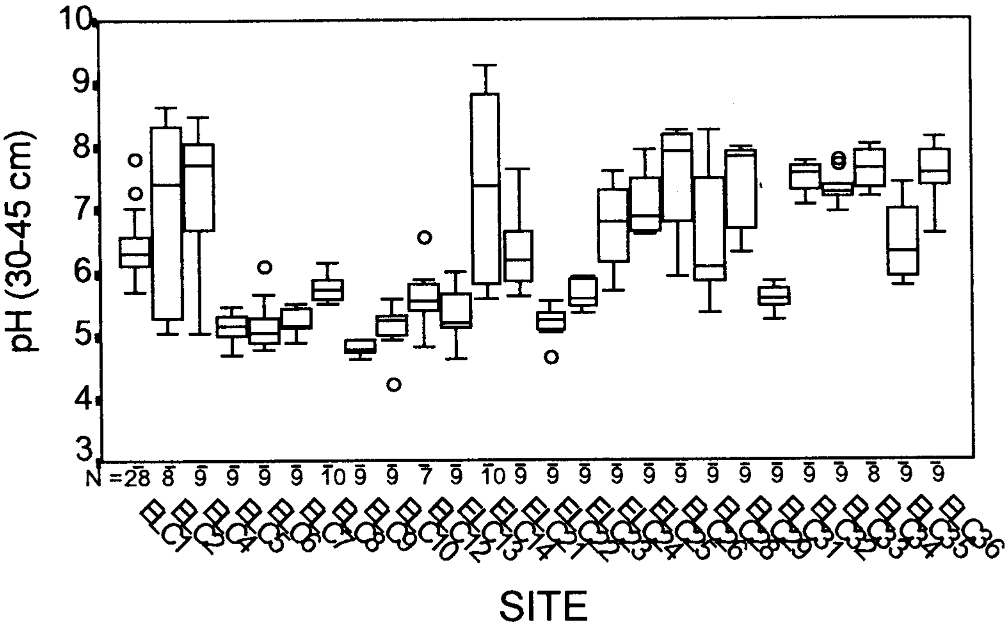


Figure 5.4:
Boxplots of the pH values for the 30 to 45 cm increment.

Table 5.2:
Mean values. standard deviations (in brackets) and results of the t-test.
For selected properties after grouping into Rich and Poor classes.

	Rich Group	Poor Group	t- test Significance Level
Soil Organic Carbon	141.3	54.8	.001
($Mg\ ha^{-1}$, to 45 cm)	(88.6)	(6.4)	
Soil Nitrogen	4455	1684	.000
($kg\ ha^{-1}$, 0 to 15 cm)	(2836)	(490)	
Soil Sulphur	343.7	165.2	.000
($kg\ ha^{-1}$, 0 to 45 cm)	(67.4)	(53.4)	
LFH Thickness	14.7	7.4	.001
(cm)	(6.78)	(2.19)	
pH (0 to 15 cm)	6.15	5.03	.000
	(0.69)	(0.63)	
pH (15 to 30 cm)	7.04	5.43	.000
	(0.68)	(0.77)	
pH (30 to 45 cm)	7.41	5.53	.000
	(0.36)	(0.78)	
Soil NH^{4+}	23.6	12.6	.000
($kg\ ha^{-1}$, 0 to 45 cm)	(7.7)	(4.7)	
Soil $N0_3^-$	.09	.09	.499
($kg\ ha^{-1}$, 0 to 45 cm)	(.18)	(.22)	
Exchangeable Na^+	97.6	55.7	.030
($kg\ ha^{-1}$, 0 to 45 cm)	(29.3)	(24.1)	
Exchangeable K^+	601.5	534.1	.479
($kg\ ha^{-1}$, 0 to 45 cm)	(192.7)	(160.1)	
Exchangeable Ca^{2+}	15027	5027	.000
($kg\ ha^{-1}$, 0 to 45 cm)	(7414)	(2201)	
Exchangeable Mg^{2+}	3380	1042	.000
($kg\ ha^{-1}$, 0 to 45 cm)	(1983)	(481)	
Soluble Inorganic Phosphorus	13.3	14.4	.718
($kg\ ha^{-1}$, 0 to 15 cm)	(8.79)	(8.3)	
Soluble Organic Phosphorus	15.0	20.9	.028
($kg\ ha^{-1}$, 0 to 15 cm)	(7.74)	(6.4)	

The levels of the plant-available nutrients depart somewhat from the trends discussed above. Soil ammonium levels are substantially higher than the nitrate levels, and are higher in the Rich sites; however, the nitrate levels are very low in both groups. Soluble inorganic P levels show no difference between the two groups, and soluble organic P levels are somewhat higher in the Poor group. Hence it is difficult to characterize the Rich group as a nutrient-rich group; although several soil components show appreciable increases, the primary soil nutrients do not follow the general trend.

5.3 Possible Landform/Groundwater Controls on the Occurrence of the Two Groups

Our explanation of the occurrence of the two groups is that the Rich group occurs at sites where the discharge of base-rich groundwater occurs; the Poor sites occur where there is no interaction between the soil column and groundwater (i.e., in recharge areas).

The Rich group sites primarily occur in landscape positions where a high potential for surface soil/groundwater interaction exist. The first group of Rich sites (ELC 2, 24, 29, 32, 34) is associated with positions adjacent to bog and fen areas or to the Balsam Poplar-dominated sites (ELC 25 and ELC 33, both of which exhibit Rich group conditions). The origin of the second group of sites (ELC 4, 14, and 36) is less clear - they are associated with lower slope positions which exhibit strongly developed concave plan and profile curvature, but are not dominated by Gleysolic soils or other features suggesting direct groundwater interactions. The higher levels of properties at these sites may result from root uptake of base materials from a groundwater table at depth in the profile (analogous to the uptake of solutes by willows in Prairie environments), but no direct observations were made on this in our study.

The Poor sites are associated with landform positions where no direct interaction with the groundwater occurs - those positions dominated by either recharge conditions or with relatively rapid transmission of water laterally through the soil.

The importance of groundwater/soil interactions as an ecological factor has emerged over the past decade and Richardson *et al.* (1992) have summarized the importance of these processes in Prairie environments. The validity of these concepts to the forest land has not been widely recognized; however the physical processes that govern groundwater movement operate consistently in different environments.

This explanation is consistent with the differences in observed site characteristics. The higher levels of the biochemical soil properties in the Rich group are associated with the influence of higher soil moisture on organic matter cycling - the wetter conditions can lead to a) higher biomass productivity and b) reduced decomposition due to anaerobic conditions. As well, calcium-organic matter complexes are resistant to microbial decomposition, and hence the higher exchangeable calcium levels in these positions may protect the soil biochemical components from microbial degradation.

The higher levels of exchangeable calcium and magnesium at sites of the Rich group are associated with the discharge of base-rich groundwater and the accumulation of solutes at the point of discharge. Ions which are more less mobile in the soil system (i.e., potassium, phosphorus) are not leached to and moved by the groundwater system, and hence are not higher in these areas.

The mineral N levels (especially the nitrate levels) show a different response to the groundwater discharge control. Ammonium levels are higher in the Rich group; however, the cationic ammonium complexes are not mobile in the groundwater system. There is no difference observed for the nitrate levels, although the anionic nitrate is the most likely to be leached out of the soil and discharged in the groundwater system. Hence it does not appear that the N dynamics at the sites is coupled to the base-rich groundwater discharge system in these landscapes.

The importance of base-rich groundwater discharge for soil formation has been recognized by several groups working in the region where the study sites are located. For example, Rural Municipality 520 (Paddockwood) located immediately south of the Model Forest boundary contains a high percentage of Paddockwood Association soils, which are Gleyed Dark Gray soils that are highly carbonated (i.e., they show high secondary enrichment of calcium carbonate due to groundwater discharge) (Staff, Sask. Soil Survey, 1993).

5.4 Field Criteria for Placement of Sites Into the Rich and Poor Groups

Clearly if the two groups delineated above are to be useful for field classification of units there must be suitable field criteria developed to allow grouping of sites into one of the two groups. A simple delineation of sites into the two groups can be made on the basis of the median pH of the 30- to 45-cm depth and can be further refined by measuring the median thickness of the LFH horizon (Figure 5.4). Samples for pH can either be assessed in the field or readily stored for subsequent analysis without concerns about deterioration of the sample.

The sites which fall below the line shown are classified as Poor groups sites; the sites above the line are classified as Rich group sites.

5.5 The Association Between the Rich and Poor Groups and other Landscape Components

The two groups developed above are the major soil and landscape divisions of the non-Organic soils in the research area, and as such should form the one of the criterion for the Ecosite level of classification, Before such a split can be made, it must be demonstrated that these two groups have some basic association with other ecologically relevant properties, and that they have a predictable occurrence within the landscape. The former allows us to develop an ecological explanation for the two groups; the latter allows us to predict the occurrence of the groups in the landscape and hence use them in a later mapping phase of the Ecological Land Classification.

5.5.1 Association with Vegetation

In the first stage of analysis, we examined the association between the two groups and the dominant tree species in the canopy of each site (Table 5.3). The dominant tree species are readily determined from the existing Forest Cover maps or from a basic field survey, and hence are appropriate for use at the Ecosite scale of mapping.

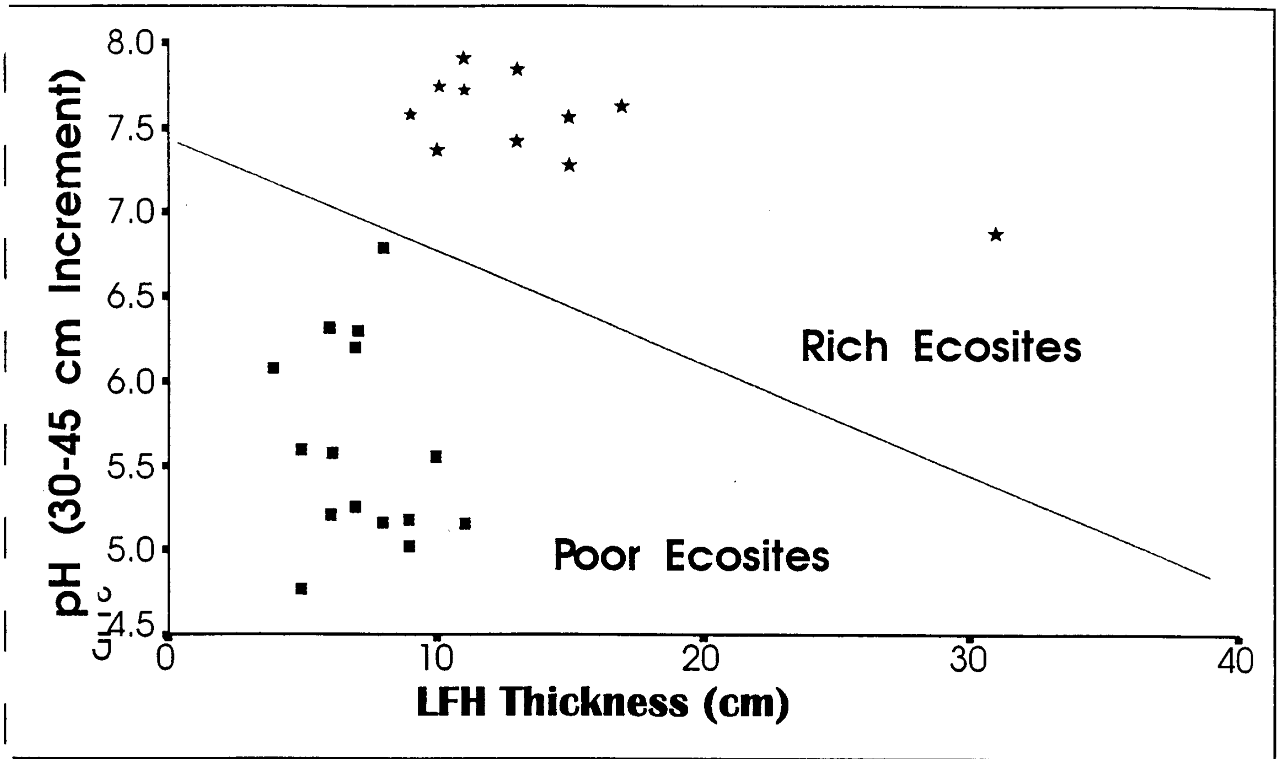


Figure 5.5:
LFH - pH of the 30-45 cm increment relationship for field placement of research sites.

Table 5.3:
The association between the dominant tree species in the canopy of each site and the Rich and Poor groups.

	Mixedwood	Dominant Black Spruce	Tree Species in the Jack Pine	Canopy Balsam Poplar	Jack Pine/Black Spruce
Number of Sites					
Rich	4	4	0	2	0
Poor	10	1	4	0	1

The association between the dominantly Jack Pine and dominantly Balsam Poplar sites and the two groups is clear - the Jack Pine sites are confined to the Poor group, and the Balsam Poplar to the Rich. The Mixedwood and Black Spruce sites span the two groups, although clearly the Mixedwood sites dominantly occur in the Poor group, and the Black Spruce sites in the Rich group.

5.5.2 Association with Soil Distribution:

Our purpose in this section is to determine if a distinctive grouping of soil units are associated with the Rich and Poor groups (Table 5.4). The Balsam Poplar sites do show a clear association - all of the individual soil pits in these sites were classified into the Gleysolic Order. The soils of the Black Spruce sites were either Luvisolic or Gleysolic; the latter are confined to the Rich group, whereas the Luvisolic soils occurred in both the Rich and Poor groups.

The Jack Pine sites (including the mixed Jack Pine/Black Spruce site) are associated with both Luvisolic (78%) and Brunisolic soils (22%) - hence the correlation between Jack Pine and coarse-textured Brunisolic soils which is often assumed in the Boreal Forest was not true for this study region. The greatest range of soils occurs in the Mixedwood sites - Luvisolic, Brunsolic, and Gleysolic soils occur in both the Rich and Poor groups. Hence there is no clear relationship between soils and vegetation for the Mixedwood stands.

A second soil-related factor that is commonly associated with ecological differences in the Mixedwood forest is soil texture. However the soils from both groups fall into a range from loam to sandy loam, which is not a textural range traditionally associate with drainage problems. Hence the texture of the first 45 cm of the soil does not appear to be an important control on the distribution of the two groups.

Table 5.4:
The association between the soil Orders at each sampling point and the Rich and Poor groups.

Dominant Tree Species	Landscape Group	Luvisolic	Brunisolic	Gleysolic
Mixedwood	Rich	21	11	4
	Poor	74	26	1
Black Spruce	Rich	10	0	26
	Poor	9	0	0
Jack Pine	Rich	0	0	0
	Poor	27	9	0
Balsam Poplar	Rich	0	0	17
	Poor	0	0	0
Jack Pine/ Black Spruce	Rich	0	0	0
	Poor	8	1	0

5.6 Characteristics of Specific Ecosites

The stratification of the research sites into specific Ecosites can be done on the basis of dominant trees) in the canopy and the Rich and Poor groups developed above. The specific Ecosites are denoted using a combination of the Group (Rich or Poor) and the Dominant tree species) in the canopy.

The Rich/Balsam Poplar and Poor/Jack Pine Ecosites form the two endpoints of a nutrient/base status continuum (Table 5.5). The Rich/Balsam Poplar sites are dominated by Gleysolic soils (Table 5.4), and have the highest levels of SOC, soil N, soil S, the thickest LFH horizons, highest ammonium and nitrate levels, the highest C.E.C., and the highest levels of exchangeable cations and soluble P fractions of any of the groups. The high quality of the organic material in this site is also reflected in the C:N ratio of the 0 to 15-cm increment; the 15.2:1 ratio is very similar to the levels found in grassland soils well to the south.

The Poor/Jack Pine sites occupy the other end of the spectrum. No Gleysolic soils were observed at the Poor/Jack Pine sites (Table 5.4). The biochemical soil properties (SOC, N, S, LFH, C.E.C., C:N ratio, NH_4^+, NO_3^-) are generally the lowest here of any of the groups, and the levels of soluble and exchangeable components are also low. The only exception are the exchangeable K^+ levels - these are consistently higher in the sites in the Poor group than in the Rich group. The C:N ratio is the highest observed (39:1), which has major importance for the likely direction of N cycling at these sites - the higher the C:N ratio, the more likely it is that any available N in the system will be immobilized rather than mineralized into more plant-available fractions.

The Black Spruce-dominated sites show a very different suite of soil conditions in the two Ecosites. The majority of Black Spruce sites occur in the Rich group (four out of five), - the paucity of samples in the Poor/Black Spruce class should be borne in mind when examining Table 5.5. The majority of the Rich/Black Spruce Ecosites (ELC 29, 32, and 34) are dominated by Gleysolic soils (Table 5.4) and the only Poor/Black Spruce Ecosite is dominated by Luvisolic soils; however, ELC 24 is classified as a Rich site on the basis of most soil properties and yet does not show evidence of prolonged water saturation in the farm of Gleysolic soils.

With the exception of exchangeable K^+, every measured property is higher in Rich/Black Spruce Ecosite than the Poor/Black Spruce Ecosite. Overall, however, the richness of the Rich/Black Spruce Ecosite is considerably lower than that of the Rich/Balsam Poplar Ecosite - for example, the C:N ratio is a relatively high 31:1 in the Black Spruce/Rich site (and in fact differs little from the Black Spruce/Poor Ecosite at 33.5:1).

The Poor/Black Spruce Ecosite is more similar to the Poor/Jack Pine Ecosites than to the Poor/Mixedwood Ecosites - the lower surface pH values and lower C:N ratios of the Poor/Black Spruce site suggest that a similar set of fertility constraints exists here as in the Poor/Jack Pine sites (unsurprising given the common coniferous vegetation at both).

The differences between the Rich and Poor variants of Black Spruce sites may be reflected in the understory vegetation. The Rich/Black Spruce sites consistently have *Equisetum* sp. associated with them; the understory at the only sampled Poor/Black Spruce site was notable for a paucity of species generally, with Bunchberry occurring as the only major understory species

The Mixedwood sites in both Ecosites are again very distinct from each other in terms of their properties, The distinction in terms of soils orders is not as clear as at the Rich/Black Spruce sites- only at site ELC 2 are an appreciable percentage of the soil sampling points classified as Gleysolic. At all the remaining Mixedwood sites, both Rich and Poor, a varying percentage of Luvisolic and Brunisolic soils occur.

The remaining soil properties do, however, show a clear difference between the Rich and Poor Mixedwood groups (Table 5.5). The values of all properties except exchangeable K^+ and the soluble P fractions are higher at the Rich Ecosites; most of the properties related to the organic fraction (SOC, C.E.C., LFH thickness) are approximately doubled at the Rich sites. Note, however, that the C:N ratio is actually lower in the Poor/Mixedwood Ecosites.

On the basis of our preliminary vegetation assessment there appears to be no consistent understory species which can be used to discriminate between the Rich and Poor groups in the Mixedwood stands. A range of species occurred in the shrub layer and shrub layer at all the Mixedwood stands, and generalizations about the association between any particular species and site conditions is difficult. A more methodical vegetation survey might be able to identify consistent indicator species.

Table 5.5:
Descriptive statistics (median and interquartile range of individual sites) for selected soil properties of the Ecosites of the Rich and Poor Groups

Soil Property	Rich Group			Poor Group		
	Dominant Tree			Dominant Tree		
	Mixed-wood	Black Spruce	Balsam Poplar	Mixed-wood	Black Spruce	Jack Pine
SOC (Mg ha^{-1}, 0-45 cm)	100.2 (36.5)	127.2 (54.5)	196.4 (90.1)	56.7 (12.5)	58.6 (15.2)	49.2 (16.2)
Soil N (kg ha^{-1}, 0-15 cm)	3727 (2402)	2874 (891)	7171 (3080)	2268 (860)	1299 (313)	1090 (443)
Soil S (kg ha^{-1}, 0-15 cm)	327 (130)	237 (86.5)	547 (246)	162 (74.9)	158 (46.1)	145 (39.6)
LFH Thickness (cm)	10.9 (3.8)	16.3 (6.1)	23.0 (13.5)	6.8 (2.7)	6.3 (0.6)	5.9 (2.5)
C:N Ratio (0-15 cm)	22.6 (7.6)	31.4 (5.3)	15.2 (2.9)	18.7 (5.3)	33.5 (4.6)	39.1 (11.7)
Ammonium (kg ha^{-1}, 0-45 cm)	24.9 (13.9)	20.6 (8.5)	38 .0 (15.0)	14.6 (11.8)	21.1 (15.2)	10.2 (12.9)
Nitrate (kg ha^{-1},0-45 cm)	.51 (1.20)	.02 (.05)	.52 (.94)	.47 (.89)	.17 (.24)	.00 (.02)
C.E.C. ($cmol_c$ kg^{-1}, 0-45 cm)	67.6 (46,6)	140.2 (71.5)	183.9 (100.5)	30.2 (8.4)	28.7 (2.51)	28.7 (11.7)
pH (0-15 cm)	5.71 (.86)	6.36 (.71)	6.76 (.23)	5.22 (.52)	4.43 (.34)	4.78 (.62)
pH (15-30 cm)	6.57 (1.53)	7.17 (.76)	7.04 (41)	5.31 (.64)	6.05 (.80)	5.77 (.82)
pH (30-45 cm)	7.25 (1.24)	7.31 (.61)	7.23 (.44)	5.56 (.65)	6.35 (.66)	6.04 (.73)
Exchangeable K^+ (kg ha^{-1}, 0-45 cm)	277 (97)	285 (70)	359 (252)	704 (225)	499 (197)	502 (226)
Exchangeable Mg^{2+} (kg ha^{-1}, 0-45 cm)	2071 (1092)	3828 (1982)	4239 (2772)	1231 (796)	1617 (336)	1073 (928)
Exchangeable Ca^{2+} Mg ha^{-1}, 0-45 cm)	11.3 (3.7)	18.4 (7.0)	26.3 (14.0)	5.0 (2.6)	5.5 (.43)	4.8 (3.0)
Solub. Inorg. P (kg ha^{-1}, 0-15 cm)	9.6 (2.39)	11.9 (5.7)	30.5 (22.4)	14.5 (8.3)	2.6 (.36)	14.2 (15.1)
Solub. Org. P (kg ha^{-1}, 0-45 cm)	13.4 (3.2)	14.1 (5.7)	43.8 (52.4)	17.7 (7.61)	13.4 (9.1)	10.0 (4.1)

5.7 Summary

The objective of this component of the research was to characterize the range of Ecosites in the major Ecosections of the Wapawekka Hills, Montreal Lake Plain, and Waskesiu Hills Ecodistricts of Central Saskatchewan. This characterization was undertaken to a) provide a baseline of ecological conditions against which the impact of clear-cut forest harvest practices could be assessed and b) provide the basis for an Ecological Land Classification of this region. The research focused on the non-Organic soil portions of the region - that is, the areas not classified as dominantly Organic soils on the available Soil Surveys.

Two distinct groups emerged from the analysis of the range of soil and landform properties. The Rich group generally had higher levels of properties related to organic carbon storage (e.g. soil organic carbon, LFH thickness, soil N and S, cation exchange capacity) as well as higher levels of properties related to mobile base cations (exchangeable calcium and magnesium, base saturation, pH); the Poor group had correspondingly lower levels of these components. A simple field test was developed based primarily on the pH of the 30 to 45 cm depth to allow placement of research sites into the Rich or Poor groups in the field.

The two groups generally showed a distinct placement in terms of the local slope system - the Rich sites were adjacent to either small wetlands dominated by Balsam Poplar, or were adjacent to larger wetlands dominated by Organic soil/fen or bog conditions. This position of sites, coupled with the higher levels of exchangeable cations, supports a groundwater discharge origin for the Rich group. The Poor group sites were more widely distributed throughout the study landscapes, and generally occupied the higher positions in the local slope system (those positions commonly associated with recharge conditions). It is very important to note that this elevation difference was often only on the order of 2 to 5 m over 100m - a very slight slope difference can result in very different characteristics in these landscapes.

The distribution of the two groups did not show a strong association with soil texture. The parent materials throughout the study region were dominated by loam to sandy loam materials, and the texture of the Rich group sites did not differ measurably from the basic suite of parent materials found throughout the area. Hence the occurrence of the two groups in the study region is apparently more related to landform position than to differences in parent materials.

Two vegetation groups showed were uniquely associated with the two groups- Jack Pine-dominated stands only occurred in the Poor group and Balsam Poplar-dominated stands only occurred in the Rich group. The majority of Black Spruce sites dominantly showed characteristics associated with the Rich group, but one Black Spruce site was clearly associated with the Poor group. The Rich Black Spruce sites consistently had a more diverse group of understory species with *Equisetum* sp. present, whereas the sole Poor/Black Spruce site had a limited number of understudy species and no *Equisetum* present.

Like the Black Spruce sites, the Mixedwood-dominated sites occurred in both the Rich and Poor groups. Unlike the Black Spruce sites, we could not identify consistent understory indicators for discriminating between the Rich and Poor Mixedwood sites, and use of the simple field test developed in the report would be essential at these sites.

Overall, six distinct Ecosites emerged from this analysis for the dominantly Mineral soil Ecosections of the Prince Albert Model Forest: Rich/Balsam Poplar, Rich/Mixedwood, Rich/Black Spruce, Poor/Mixedwood, Poor/Black Spruce, and Poor/Jack Pine. The order in which these sites are presented also represents a simple drainage/nutrient continuum for the sites, from the richest (Rich/Balsam Poplar) to the poorest (Poor/Jack Pine). Each has a distinctive range of properties associated with them which can be used to develop a quantitative Ecological Land Classification for the region; equally the range of properties for each ecosite forms the baseline against which the impact of disturbance in these region can be assessed.

6. EFFECT OF CLEAR-CUTTING ON SOIL QUALITY CONDITIONS IN MIXEDWOOD LANDSCAPES

6.1 Research Sites

The literature available on clear-cut impacts indicates that at least two distinct stages occur: in the period immediately after clear-cutting, increases in SOC and related biochemical fractions in the soil may occur and high leaching loss of ions occurs; in the medium- to long term, a ± 10% change in SOC has been observed and the leaching rate of ions decreases. To examine these two stages in our study, we divided the clear-cut sites into two groups: sites with 1 and 3 years of recovery after harvest (sites MF 4 and AF 1), and sites with 5 to 20 years of recovery (sites MF 5, MF 6, MF 7, MF 8) (Table 6.1). The former group represents a short-term response; the latter group a medium-term response. Clear-cut forestry was not practised in the study area prior to 20 years ago.

Table 6.1:
Locations and year of harvest for the Mixedwood clear-cut sites.
All sites are in UTM zone 13 (NAD27 datum)

Site	UTM Coordinates		Years of regeneration at time of sampling
MF1	440400 E.	5981750 N.	3
MF4	484100E.	5977100 N.	1 (summer following winter cut)
MF5	441290 E.	5975050 N.	11
MF6	469900 E.	5988750 N.	19
MF7	472656 E.	5988842 N.	9
MF8	470000 E.	5982500 N.	15

Site MF4 was sampled prior to cutting in 1994, and was sampled again the summer following harvest. Site MF7 was sampled in 1994 and was burned in the spring of 1995 in the "Monday" fire. It was subsequently re-sampled in the fall of 1995.

None of the clear-cut sites had levels of soil quality indicators that would suggest that they were from the Rich/Mixedwood group of ecosites discussed in the previous section. Hence for these Mixedwood clear-cut sites, our baseline values are drawn from the Poor/Mixedwood groups of sites as documented in Table 5.5.

6.2 Physical Indicators of Soil Quality

The potential for physical disruption due to mechanized harvest is very large; however major impacts due to mechanized traffic are largely confined to roads, landings and skidder trails. The large grids in the clear-cut sites include a proportion (5 to 11%) of these disturbed areas but, overall, the bulk density values measured in our study are dominated by areas without the high level of physical disturbance.

The bulk density levels of the three depth increments do not differ significantly between the mature Mixedwood sites and the two groups of clear-cut sites (Table 6.2).

Hence overall neither significant surface nor sub-soil increases in bulk density were observed in our study, and major physical impacts within the rooting zone are unlikely on a landscape-scale. This does not, however, preclude the possibility that major physical disruption occurred at specific, concentrated positions in the landscape.

Table 6.2:
Comparison of bulk density levels in mature Mixedwood, and clear-cut sites with <5 years recovery and 6 to 20 years recovery after harvest.

	Mature Mixedwood	Clear-Cut < 5 Years		Clear-Cut 6-20 years	
Depth	Bulk Density g cm^3			Bulk Density g cm^3	
	Mean (S.D.)	Mean (S.D.)	Significance Level[a]	Mean (S.D.)	Significance Level[b]
0-15 cm	0.96 (0.16)	0.95 (0.01)	.78	1.03 (0.23)	.56
15-30 cm	1.67 (0.06)	1.80 (.11)	.32	1.65 (0.11)	.76
30-45 cm	1.74 (0.16)	1.85 (0.16)	.42	1.75 (0.12)	.96

a. Significance level of t-test of mature Mixedwood vs. Clear-cut sites with < 5 years recovery after harvest.
b. Significance level of t-test of mature Mixedwood vs. Clear-cut sites with 6 to 20 years recovery after harvest.

6.3 Biochemical Indicators of Soil Quality

Substantial changes in the storage of SOC, soil nitrogen, and the thickness of the LFH horizon occurred due to clear-cutting (Table 6.3). In the short-term, increases in SOC and the thickness of the LFH horizon were observed; however, the C:N ratio also increased in this period, indicating that the organic materials may be of lower quality.

In the medium-term, substantial, landscape-wide losses of SOC, soil nitrogen, and LFH thickness were observed. Overall losses of 23% of SDC, 27% of soil nitrogen, and 28% of the original LFH thickness were observed (Table 6.3). Nor can this loss of LFH be attributed to formation of a more compact, denser LFH horizon: no major increase in the bulk density of the 0 to 15 cm increment was observed.

Table 6.3:
Changes in the biochemical indicators of soil quality in mature Mixedwood and clear-cut sites with <5 years recovery and 6 to 20 years recovery after harvest.

	Mature Mixedwood	Clear-Cut < 5 Years		Clear-Cut 6-20 years	
Depth	**Mean (S.D.)**	**Mean (S.D.)**	**Significance Level[a]**	**Mean (S.D.)**	**Significance Level[b]**
		Soil Organic Carbon (Mg ha^{-1})			
0-45 cm	57.7 (5.6)	62.4 (0.4)	.04	44.1 (5.4)	.02
		Total Soil Nitrogen (Mg ha^{-1})			
0-15 cm	1.99 (0.16)	2.06 (0.18)	.74	1.45 (0.10)	.00
		C: N Ratio			
0-15 cm	21.5 (3.0)	24.4 (1.1)	.10	23.4 (3.6)	.91
		LFH Thickness (cm)			
	8.1 (2.0)	9.3 (1.0)	.35	5.8 (1.7)	.07

a. Significance level of t-test of mature Mixedwood vs. Clear-cut sites with < 5 years recovery after harvest,
b. Significance level of t-test of mature Mixedwood vs. Clear-cut sites with 6 to 20 years recovery after harvest.

The levels of microbial- and plant-available N fractions (soil ammonium and nitrate) show extremely high variability, with coefficients of variation in the 35% to 175% range. Moreover, the great temporal range associated with these very dynamic fractions precludes comparisons between sites which have been sampled in different periods of the growing season. It is noteworthy, however, that by far the highest levels of mineral N are found in the soil ammonium pool rather than the nitrate pool: median values of 10.4, 15.4 and 14.6 kg ha^{-1} of NH_4^+ -N were determined for the mature Mixedwood, short-term clear-cuts, and medium-term clear-cuts, respectively, versus only 0.16, 0.80, and 0.05 kg ha^{-1} of NO_3^- -N in the same sequence of sites. Hence the mineral N in these soils largely occurs as the more readily retained cationic form rather than the more mobile anionic form.

6.4 Chemical Indicators of Soil Quality

The changes in chemical and nutrient indicators of soil quality were examined for both the 0 to 15 cm increment only and 0 to 45 cm depth. The former indicates the conditions within the rooting zone for the early stages of establishment of the tree species; the upper 45 cm of the soil as a whole may indicate the longer-term limits on tree establishment as the rooting zone expands in the first 20 years of growth after harvest.

Overall no substantive decreases in pH, soluble phosphorus, exchangeable cations, C.E.C., or base saturation were observed in the first two years after clear-cutting in the 0 to 15 cm increment (Table 6.4). Major losses did, however, occur in the medium-term relative to the mature Mixedwood sites: exchangeable calcium and magnesium drops by around 30%; C.E.C. and base saturation decrease by about 20%; soil pH experiences a decrease from about 5 to 4.7 and soluble organic phosphorus levels decrease by 15%. All of these losses point to sustained losses of soluble ions from the soil during the 5 to 20 year period of regeneration. These changes may be largely due to the loss of SOM from the medium-term sites - for example the organic matter is an important source of charge sites in forested soils.

Relative losses of ions in the upper 45 cm of the soil are lower than those observed for the 4 to 15 cm increment only (Table 6.5). For example, losses of exchangeable calcium in the upper 45 cm total 1160 kg ha^{-1}; 589 kg ha^{-1} of this loss occurs in the 0 to 15 cm increment. Only the 14.5% decrease in base saturation observed in the medium-term has a significance level below 0.20; this loss represents the cumulative loss of base cations from the soil.

Table 6.4:
Changes in chemical indicators of soil quality (0 to 15 cm increment) in mature Mixedwood, and clear-cut sites with <5 years recovery and 6 to 20 years recovery after harvest.

	Mature Mixedwood	Clear-Cut < 5 Years		Clear-Cut 6-20 years	
Depth	Mean (S.D.)	Mean (S.D.)	Significance Level[a]	Mean (S.D.)	Significance Level[b]
		Cation Exchange Capacity (cmol kg^{-1})			
0-15 cm	13.9 (3.7)	14.3 (0.6)	.68	11.2 (l.9)	.11
		Base Saturation (%)			
0-15 cm	59.8 (16.0)	59.7 (2.9)	.99	46.3 (13.3)	.16
		Exchangeable Calcium (Mg ha^{-1})			
0-15 cm	1.73 (0.53)	2.00 (0.15)	.24	1.15 (0.408)	.06
		Exchangeable Magnesium (kg ha^{-1})			
0-15 cm	180.2 (52.3)	154.8 (10.0)	.21	127.4 (34.1)	.06
		Exchangeable Potassium (kg ha^{-1})			
0-15 cm	188.6 (25.5)	202.8 (56.7)	.78	193.8 (46.6)	.85
		Soluble Inorganic Phosphorus (kg ha^{-1})			
0-15 cm	13.3 (6.6)	12.0 (1.3)	.34	11.8 (8.9)	.40
		Soluble Organic Phosphorus (kg $ha^{-}1$)			
0-15 cm	25.0 (6.4)	23.4 (l.34)	.48	21.2 (1.64)	.03
		pH			
0-15 cm	5.03 (43)	5.21 (.36)	.61	4.74 (.16)	.11

a. Significance level of t-test of mature Mixedwood vs. Clear-cut sites with < 5 years recovery after harvest.
b. Significance level of t-test of mature Mixedwood Vs. Clear-cut sites with 6 to 20 years recovery after harvest.

Table 6.5:
Changes in the chemical indicators of soil quality (0 to 45 cm increment) in mature Mixedwood, and clear-cut sites with <5 years recovery and 6 to 20 years recovery after harvest.

	Mature Mixedwood	Clear-Cut < 5 Years		Clear-Cut 6-20 years	
Depth	**Mean (S.D.)**	**Mean (S.D.)**	**Significance Level[a]**	**Depth**	**Significance Level[b]**
		Cation Exchange Capacity (cmol kg^{-1})			
0-45 cm	28.6 (5.2)	28.3 (5.7)	.94	23.6 (4.8)	.28
		Base Saturation (%)			
0-45 cm	65.4 (13.5)	69.1 (4.43)	.79	51.5 (6.6)	.11
		Exchangeable Calcium (kg ha^{-1})			
0-45 cm	5129 (1637)	5438 (1423)	.71	3969 (1405)	.30
		Exchangeable Magnesium (kg ha^{-1})			
0-45 cm	1022 (469)	832 (63)	.31	750 (218)	.25
		Exchangeable Potassium (kg ha^{-1})			
0-45 cm	620 (175)	640 (108)	.89	536 (93)	.22
		pH			
15-30 cm	5.09 (.45)	5.35 (.78)	.72	5.04 (.11)	.74
30-45 cm	5.35 (.45)	5.67 (.54)	.55	5.29 (.06)	.68

a. Significance level of t-test of mature Mixedwood vs. Clear-cut sites with < 5 years recovery after harvest.
b. Significance level of t-test of mature Mixedwood vs. Clear-cut sites with 6 to 20 years recovery after harvest.

7. EFFECTS OF CLEAR-CUTTING ON SOIL QUALITY CONDITIONS IN CONIFEROUS LANDSCAPES

7.1 Research Sites

Four former coniferous clear-cuts were sampled and analyzed in the 1995-1996 field season (Table 7.1). All of the sites had previously been dominated by Jack Pine-Black Spruce. One of the sites (ELC 12) was dominated by Gleysolic soils, and generally had a suite of soil properties which indicated placement into the Black Spruce/Rich Ecosite range, whereas the other three sites had conditions indicative of placement into the Poor group of ecosites. Hence to develop the baseline for the three clear-cut sites, all of the coniferous sites in the Poor group were grouped together, and average conditions were calculated for this group. Because of the small number of sites we did not attempt to split them into different age categories of recovery (as with the Mixedwood clear-cut sites). As well, we have not at this time attempted to compare the results for the single Rich clear-cut (ELC 12) to the remainder of the RicH/Black Spruce mature stands.

Table 7.1:
Site locations and years of regeneration following harvest for the coniferous clear-cut sites.

Site	UTM Coordinates		Years of regeneration at time of sampling
MF11	440400 E.	5981750 N.	5
MF12	484100 E.	5977100 N.	4
MF13	441290 E.	5975050 N.	4
MF6	469900 E.	5988750 N.	17

7.2 Physical Indicators of Soil Quality

The bulk density of both the surface and the 30-45 cm increment is higher in the clear-cut sites than in the comparable mature stands (Table 7.2). The increase in the surface increment is attributable to the loss of LFH horizon encountered in these soils (see below). The increase in the 30-45 cm increment is less readily explained - indeed the decrease in bulk density from the 15-30 cm increment to the 30-45 cm increment is somewhat unusual itself.

7.3 Biochemical Indicators of Soil Quality

A 30 to 50% decrease in all of the biochemical indicators of soil quality due to clear-cutting in the former coniferous landscapes is apparent from the data (Table 7.3). This can be attributed to, in part, the 3.4 cm (48.9%) decrease in the median LFH thickness at the clear-cut sites; because the LFH horizons were initially thinner at these sites than at comparable Mixedwood sites, the relative decrease is much higher at these sites. Put in other terms, the thinner the LFH horizon at the mature sites, the more sensitive the sites are to losses in LFH materials due to clear-cut effects.

The decreases in SOC, total soil N, and soil sulphur surpass the values for loss for the Mixedwood clear-cuts; for example the decrease in SOC storage in the Mixedwood landscapes due to clear-cutting was only 23.5%. Moreover the higher initial levels of the biochemical indicators at the Mixedwood sites makes them less sensitive to loss - for example the total soil N store at the mature Mixedwood sites is 1.99 Mg ha^{-1} of N, whereas the initial levels at the coniferous sites are only 1.31 Mg ha^{-1}. The loss of N in both cases is comparable (about 0.5 Mg ha^{-1}), but the relative loss is much higher in the coniferous sites.

The decrease in the SOC storage also leads to a decrease in the cation exchange capacity of the upper increment (which is largely due to reactive organic materials) comparable in magnitude to the loss of the other biochemical indicators. The only indicator of biochemical processes which does not alter is the C to N ratio (which is an approximate measure of the quality of the organic residue) - no statistically distinct decrease occurs. The C:N ratio for both sites is well in excess of 25:1, indicating that decomposition of SOM may lead to immobilization of mineral N from the soil pool, although the threshold at which immobilization occurs depends on the microbial community present at the site (D. Grigal, pers. comm.).

7.4 Chemical Indicators of Soil Quality

As in the Mixedwood clear-cuts, the highest losses occur in the exchangeable calcium stores in the soil - in all a very large 3.1 Mg ha^{-1} loss associated with the clearcut coniferous sites (Table 7.4). Comparable losses of magnesium occur, but the very high variability of exchangeable magnesium levels in these soils complicates the comparison between the grouped sites.

The 66.0% increase in soluble inorganic soil phosphorus is the sole positive change observed in the data set. No change occurred in the levels of soluble organic phosphorus in the soils of the clear-cut sites, and no significant change was observed in the levels of potassium available in the upper 45 cm of the soil. The high retention of potassium in these soils was also observed in the Mixedwood clear-cut sites.

The mean pH values of the lower soil increments are statistically significant, but the observed values in the clear-cut sites (5.4 and 5.7) are not low enough to pose any known problems in terms of mobilization of potentially toxic elements in these landscapes.

Table 7.2:
Comparison of bulk density levels in mature coniferous and clear-cut coniferous sites.

Depth	Mature Non-Mixedwood Sites	Clear-Cut Non-Mixedwood Sites	Significance Level[a]
	Bulk Density (g cm^{-3})		
0-15 cm	1.12	1.30	0.09
	(0.12)	(0.15)	
15-30 cm	1.63	1.70	.22
	(.09)	(.05)	
30-45 cm	1.54	1.73	.06
	(0.14)	(0.08)	

a: Significance level of t-test of mature coniferous vs. Clear-cut sites

Table 7.3:
Summary of means and standard deviations (in brackets) for biochemical soil properties in mature coniferous sites compared to clear-cut coniferous sites;

Depth	Mature Non-Mixedwood Sites	Clear-Cut Non-Mixedwood Sites	Significance Level[a]	Percentage Change
	SOC (Mg ha^{-1})			
0-45 cm	55.9	36.9	.04	-34.5
	(12.8)	(4.1)		
	Soil N (Mg ha^{-1})			
0-15 cm	1.31	0.80	.05	-39.4
	(0.46)	(0.23)		
	Soil S (kg ha^{-1})			
0-15 cm	134.3	92.4	.05	-31.2
	(32.9)	(10.4)		
	LFH Thickness (cm)			
	6.9	3.5	.01	-48.9
	(2.3)	(0.5)		
	C:N Ratio			
0-15 cm	37.2	32.4	.86	
	(10.1)	(3.4)		

a: Significance level of t-test of mature coniferous vs. Clear-cut sites

Table 7.4:
Comparison of levels of chemical indicators of soil quality in mature coniferous and clear-cut coniferous landscapes.

Depth	Mature Non-Mixedwood Sites	Clear-Cut Non-Mixedwood Sites	Significance Level	Percentage Change
	Solub. Inorg. P (kg ha^{-1})			
0-15 cm	13.1a	21.7b	.17	+66.0
	(9.5)	(3.2)		
	Solub. Org. P (kg ha^{-1})			
0-45 cm	15.1a	15.5a	.90	
	(4.4)	(3.9)		
	pH			
0-15 cm	5.09a	4.81 a	.42	
	(0.86)	(0.08)		
15-30 cm	6.01a	5.39b	.25	
	(0.81)	(0.17)		
30-45 cm	6.34a	5.67b	.07	
	(0.80)	(0.18)		
	Exchangeable K^+ (kg ha^{-1})			
0-45 cm	552a	560a	.94	
	(173)	(70.7)		
	Exchangeable Mg^{2+} (kg ha^{-1})			
0-45 cm	1978a	855a	.44	
	(2287)	(309)		
	Exchangeable Ca^{2+} (Mg ha^{-1})			
0-45 cm	7.3a	4.2b	.09	-42.3
	(3.5)	(1.5)		
	C.E.C. ($cmol_c$ kg^{-1})			
0-45 cm	33.3a	20.8a	.29	
	(18.1)	(4.1)		
	C.E.C. ($cmol_c$ kg^{-1})			
0-15 cm	14.32a	7.55b	.08	-46.8
	(4.68)	(1.29)		

8. ECOLOGICAL SIGNIFICANCE OF OBSERVED SOIL QUALITY CHANGES IN MIXEDWOOD AND CONIFEROUS CLEAR-CUTS

8.1 Comparison of Observed Changes to the Natural Range of Variability in Mature Forest Stands

The crude statistical assessment of changes in soil quality indicators can be placed in a larger context by examining the observed changes relative to the range of values in the individual mature forest stands. This was done by comparing the boxplots for individual clear-cut sites with those from the individual mature forest landscapes.

The relevance of this comparison can be shown using two examples. For soil organic carbon (Figure 8.1 a, b), the median levels for the clear-cut are clearly below the range defined by the highest and lowest median levels in the mature sites; hence the clear-cut levels are clearly outside the natural or ecological range defined by the mature Mixedwood site. Similar relationships are also evident for soil N (Figures 8.2 a, b). For other soil properties such as soil pH, a significant difference in levels between the mature and clear-cut sites was observed using the t-test comparisons; however, the medians for the clear-cut sites (and, for the most part, the dispersion of values around the medians) is within the range defined by the mature sites. In these cases, the impact of clear-cutting has not moved the clear-cut sites outside of the natural range of variability associated with the mature sites, and hence the ecological significance of the change in soil quality is probably minor.

Of the statistically-defined changes in soil quality determined in the previous section (e.g., those with significance levels between 0.00 and 0.20), only soil nitrogen and SOC in the medium-term, clear-cut Mixedwood sites and the coniferous clear-cuts are well outside of the range defined by the mature sites; the remainder in either the 0 to 15 cm increment or the upper 45 cm as a whole (LFH thickness, soil pH, soluble organic phosphorus, base cation, base saturation, and cation exchange capacity) are within the range of mature sites as depicted by the boxplot comparisons. Hence overall the majority of the changes observed using the statistical comparison are within the range defined by the natural conditions in the study region.

8.2 Evaluation of Observed Changes

Two broad categories of soil quality indicators have been effected by clear-cutting in the Model Forest region. The first is the biochemical indicators of soil quality (i.e., those related to the organic matter levels in the soil). These show the most severe levels of decrease due to clear-cutting, and as discussed in section 8.1, the degree of loss places the clear-cut sites outside of the natural range of variability for comparable landscapes.

The second type of loss is the decrease in exchangeable cations, specifically calcium and magnesium. These losses are less severe than the losses of the biochemical stores in these soils, and do not appear to place the clear-cut sites outside of the range of natural variation. In the both Mixedwood and coniferous landscapes comparable and high losses of calcium were observed.

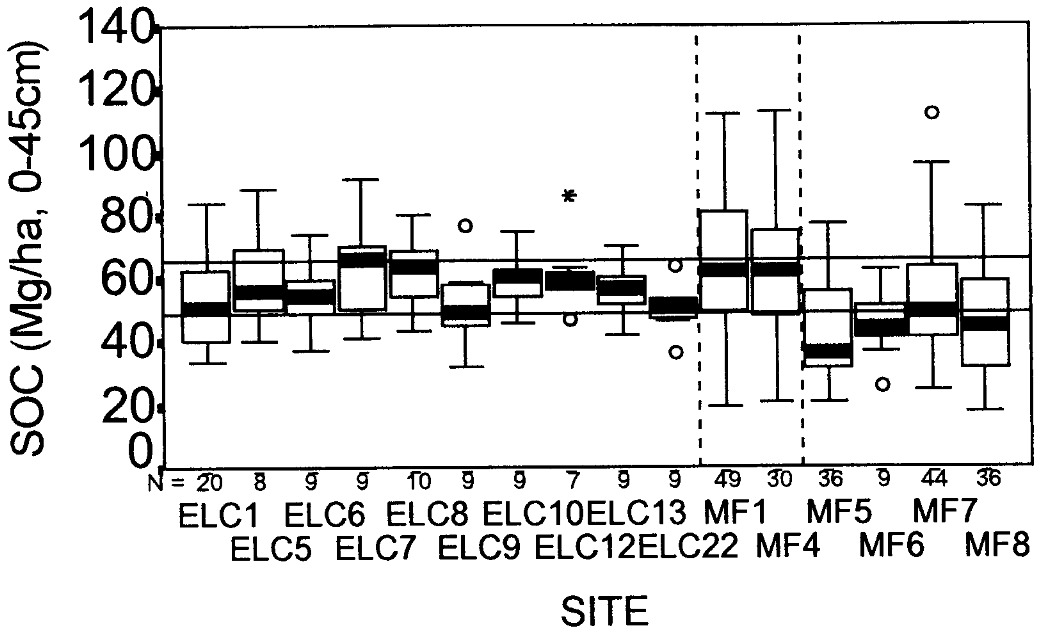


Figure 8.1 a:
Boxplots of soil organic carbon for mature Mixedwood (ELC) and Mixedwood clear-cut (MF) sites. The horizontal lines delineate the highest and lowest median values associated with the mature Mixedwood sites; vertical lines separate the mature, short-term clearcut and medium-term clearcut sites.

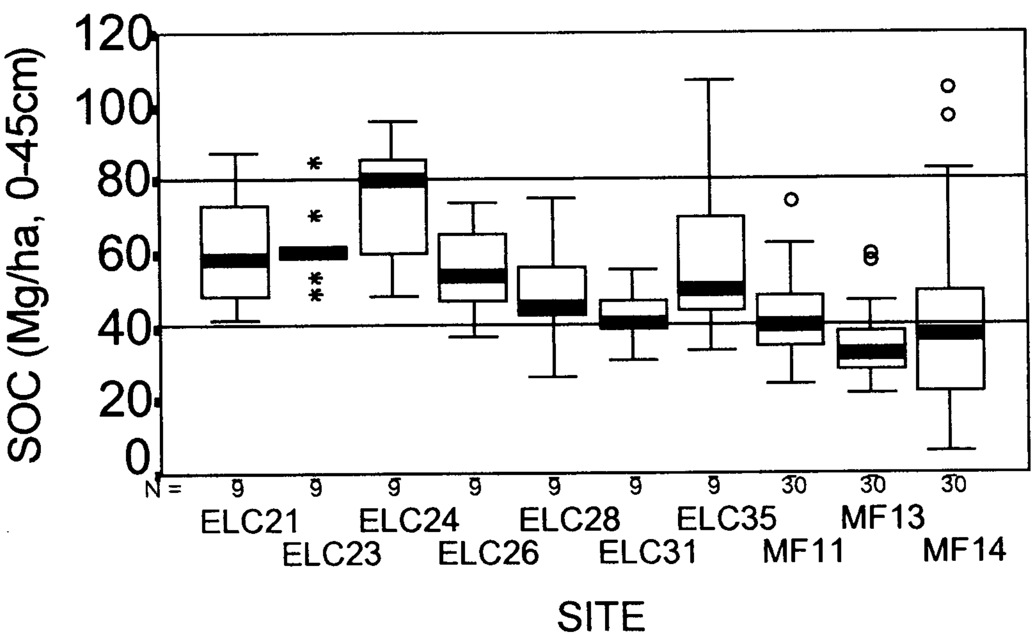


Figure 8.1 b:
Boxplots of soil organic carbon results for mature coniferous (ELC) and coniferous clear-cut (MF) sites. The horizontal lines delineate the highest and lowest median values associated with the mature coniferous sites.

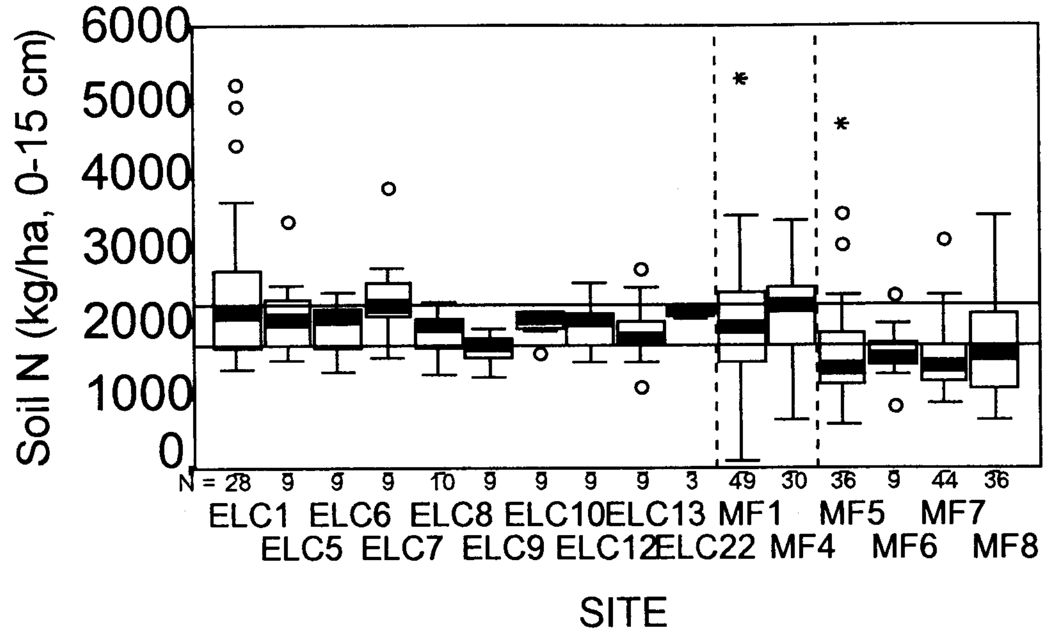


Figure 8.2a:
Boxplots of total soil N in the U-15 cm increment for mature Mixedwood (ELC) and Mixedwood clear-cut (MF) sites. The horizontal lines delineate the highest and lowest median values associated with the mature Mixedwood sites; vertical lines separate the mature, short-term clearcut and medium-term clearcut sites.

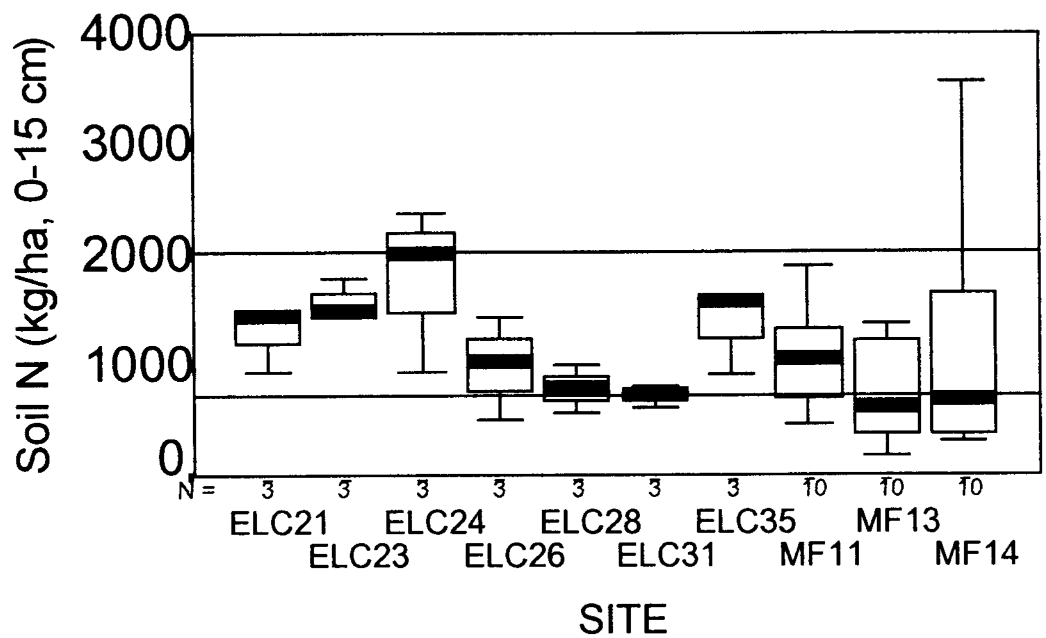


Figure 8.2b:
Boxplots of total soil N in the Q-15 cm increment for mature coniferous and coniferous clear-cut sites. The horizontal lines delineate the highest and lowest median values associated with the mature coniferous sites.

In Mixedwood clear-cuts, these losses may be due to a) leaching of the cations from the surface layer or b) uptake of the cations by the vegetation, In his PAMF-sponsored research, Dr. Van Rees has demonstrated very high levels of strontium uptake from the 0- to 15-cm increment by aspen roots at the clear-cut sites, and strontium has a very similar chemical behavior to calcium. The high base uptake by aspen has also been widely noted by other researchers. Hence the possibility of high base cation uptake by the regenerating aspen may provide a mechanism for the retention of these cations in the Mixedwood clear-cut sites.

In the non-Mixedwood sites, however, the lack of Aspen growth (and the paucity of regenerating understory species generally) greatly decreases the possibility for cation uptake, and hence the possibility for overall loss is much greater. We observed poor recovery by the understory species at the four coniferous clear-cuts, and the growth of the naturally regenerating or planted conifer species was limited. Hence we suggest that the losses in the coniferous stands may be of greater long-term significance than in the Mixedwood stands.

The losses of carbon and nitrogen differ from the losses of soluble soil components in terms of their off site effects. Although the losses of, for example, exchangeable calcium are very large from soils of the clear-cut sites, a high potential for uptake of these solutes from the soil by the regenerating vegetation exists (Alban, 1982) and this uptake may account for a substantial proportion of the observed losses. Moreover any solutes which were leached from the clear-cut sites simply enhance hydrological redistribution of solutes in these landscapes: the occurrence of base-rich discharge sites such as those discussed above and by Donald *et al.* (1993) indicates that the migration of high loads of solutes within the groundwater system is a wide-spread hydrological phenomenon in this region.

The losses of C and N have greater potential off-site effects because of their possible relevance for the greenhouse gas complex, The decrease in LFH thickness and the corresponding loss of C and N could be due to either i) physical removal of the organic layer from the soil surface during harvest and site preparation and/or accelerated erosion in the years subsequent to harvest or ii) in situ changes in carbon and nitrogen dynamics. We observed no major evidence for water erosion at the sampled sites, but the possibility of physical removal of LFH material during harvest or subsequent site preparation cannot be precluded.

The in situ losses of carbon and nitrogen occur due to a combination of decreased input of fresh organic detritus after harvest and increased decomposition of the existing organic material. Using data from a subset of the clear-cut sites presented in this paper, Pennock and van Kessel (1995) argued that the gaseous losses of CO_2 to the atmosphere from organic matter decomposition in clear-cuts is, however, probably greatly outweighed by the CO_2 losses from soil organic matter of the agricultural land immediately south of the forest boundary in Saskatchewan. Research on N_2O emissions from forested land 100 km south of the study region has also shown that N_2O emissions from NO_3-poor forested soils are very low (Corre *et al.*, 1995) relative to those from fertilized agricultural land. Hence, although the magnitude of C and N emissions to the atmosphere due to clear-cutting in this region should be more rigorously assessed, the contributions of the agricultural region to the south of probably of greater significance for the global climatic system.

In summary, the impact of clear-cutting on the indicators of soil quality discussed above has been substantial. The landscape-scale changes in soil bulk density do not indicate that any wide-spread alteration in the partitioning of water at the soil surface has occurred. Nor does the ability of these soils to act as an environmental filter appear to have been greatly altered: the cation exchange capacity levels are relatively unaffected, and the minor decreases in soil pH do not place it in a range where increased mobility of potentially toxic ions such as aluminum will occur.

From the perspective of sustained forest productivity in these areas, the potential impact of clear-cutting on the soil as a medium for plant growth and biological activity are perhaps the greatest concern. Substantial losses of nutrients and base cations from the upper 45 cm of the soil were observed, and the relative losses from the soil outstrip those reported for biomass export of nutrients in the initial section of this paper. The losses of nutrients and soluble bases do not, however, place the clear-cut sites outside of the natural range of conditions observed at the mature Mixedwood sites and hence the ecological consequences of the observed changes are unlikely to be major.

The SOC and total soil N levels at the medium-term Mixedwood clear-cut sites and the coniferous clear-cuts are clearly outside the range of natural conditions for these stands. The potential exists for further regeneration of SOM levels in the clearcut stands as the mature forest canopy is established - however the relative ecological impact of the losses on the development of the second-growth forest if such regeneration does not occur may be severe and warrants further study in this region.

9. EFFECTS OF FIRE ON SOIL QUALITY CONDITIONS IN A MATURE MIXEDWOOD AND A MIXEDWOOD CLEAR-CUT

The major fires which burned in the Model Forest in 1995 presented an opportunity to measure the effects of fire on soil quality conditions. Two sites had been sampled in 1994 prior to the burn a mature Mixedwood site (ELC9) and a medium-term clear-cut site (MF7). These were both burned as part of the Monday Fire in spring, 1995. We were able to locate the original sampling points at both sites, and we re-sampled both sites in late summer, 1995 after the burn. Samples were only taken from the 0-15 increment at both sites - we wanted to reduce the surface disturbance such that we can continue to sample these sites into the future.

The results of the comparison of pre- and post-burn conditions generally reflect the trends which are evident in published literature studies on the effect of fire. The levels of exchangeable bases and inorganic P increase after the fire, and the pH levels at both sites rose by almost 1 pH unit (i.e., a 10-fold decrease in H^+ activity) (Table 9.1). At both sites there is also a significant decrease in SOC levels, as well as decreases in soil N and large (but non statistically significant) differences of S at the MF7 site.

Although the two sites show similar types of responses to fire, the magnitude of the response to fire differs greatly between the two sites. Generally the decreases in the soil biochemical stores are much greater at MF7 site - for example 33.7% of the original SOC is lost from MF7 versus only 14% at the mature site (Table 9.1); the decrease in soil N is slight at the mature site but is higher (and statistically significant) at the clear-cut site. The increases in exchangeable ions is also higher at the clear-cut sites - the increase in calcium, magnesium, and phosphorus are all in the 30 to 70% range. These two observations are not, of course, unrelated - presumably the higher the consumption of the soil organic matter in the fire, the higher is the release of ions into the ash.

Clearly the higher losses of SOM from the clear-cut site are a concern insofar as these sites have already lost considerable amounts of SOM due to changes in the soil environment following clear-cutting. The higher losses in the clear-cut may indicate a decrease in the resilience of these sites - a decrease in their ability to resist further disturbance, whether from fire or other sources. If so this decrease in resilience would be a major source of concern in terms of the effects of clear-cutting in these landscapes. We hope to continue to monitor these sites to document the soil quality changes in the short-term.

Table 9.1:
Pre-Burn and post-burn levels of selected soil quality indicators at site ELC 9 (mature Mixedwood) and MF7 (Mixedwood clear-cut).

ELC9			MF7		
Pre-Burn	**Post-Burn**	**Significance Levels[a]**	**Pre-Burn**	**Post-Burn**	**Significance Level**
Soil Organic Carbon (Mg ha^{-1}, 0-15 cm increment)					
35.2	30.1	.16	40.7	27.0	.000
(6.7)	(7.9)		(15.5)	(8.0)	
Soil N (kg ha^{-1}, 0-15 cm increment)					
1650	1550	.52	1471	1216	.03
(203)	(400)		(430)	(460)	
Soil S (kg ha^{-1}, 0-15 cm)					
126	131	.79	282	128	.28
(28)	(34)		(415)	(60)	
Exch. Calcium (Mg ha^{-1}, 0-15 cm)					
1.01	1.17	.47	.70	1.2	.002
(.15)	(.35)		(.12)	(.60)	
Exch. Magnesium (kg ha^{-1}, 0-15 cm)					
109	132	.26	98	131	.12
(24)	(38)		(37)	(83)	
Exch. Potassium (kg ha^{-1}, 0-15 cm)					
117	228	.03	177	204	.27
(19)	(52)		(50)	(87)	
pH (0-15 cm increment)					
4.59	5.41	.000	4.59	5.59	.000
(.19)	(.34)		(.33)	(.73)	
Soluble Inorganic Phosphorus (kg ha^{-1}, 0-15 cm)					
15.9	23.2	.37	32.1	50.4	.06
(12.7)	(11.6)		(18.3)	(34.4)	

a: Significance level for t-test of pre- and post-burn properties

10. LANDSCAPE-SCALE VARIABILITY OF NITROGEN MINERALIZATION IN FOREST SOILS. (Prepared by Dr. F. Walley)

10.1 Summary

Our understanding of the controls on N-cycling and availability in forest soils following disturbance is limited. A comparative study was conducted to examine the spatial distribution of soil N in forest soils and assess the impact of site disturbance. Sampling grids were established within a 120 by 120 m representative area at a native, a burned, and a clear-cut site. A three-dimensional classification of landscape form was used to stratify each landscape into distinct landform elements. The spatial distribution of inorganic-N was not related to landform element, irrespective of site disturbance, indicating an absence of topographic control at the scale studied. However, a narrowing of the NH^{4+}/NO_3^- ratio and an increase in the size of the microbial biomass at MF1 (clear-cut site) compared to MF3 (mature Mixedwood site) suggests that N-cycling and microbial dynamics were influenced by site disturbance. An assessment of potential N and C mineralization and of net nitrification in the forest floor and surface mineral horizons similarly indicated that site disturbance had a direct impact on N-cycling processes in these forest soils.

10.2 Introduction

Available N typically is low in boreal forest soils, implying a tight coupling between consumptive and productive N-cycling processes. An uncoupling of N mineralization and immobilization due to pedogenic or anthropogenic factors is likely to alter the soil N status, and undesirable N losses from the system can occur.

Disturbances are often associated with nutrient losses from an ecosystem. Clearcut logging may increase losses of ions due to leaching (Vitousek *et al.* 1982) whereas nutrient removal associated with fire may be due to volatilization and aeolian particle loss (Raison *et al.* 1985). Soil heating due to fire may cause an increase in the NH^{4+} concentrations in the short term (Prieto-Fernandez *et al.* 1993) and lead to accelerated mineralization of the remnant, pyromorphic soil organic matter in the long term (Rapp 1990). High nitrification in burned forest soils has been observed (Kutiel and Naveh 1987) which may lead to a higher potential for nutrient losses due to leaching.

Topography is known to play a critical role in modifying both the microclimate and the hydrological conditions within a landscape (Rowe 1984). These modifications within a landscape can, in turn, influence or control the type and intensity of soil processes (Pennock *et al.* 1994). Distinct spatial patterns of soil moisture, redox potential, bulk density, N mineralization, denitrification and respiration have been observed in agricultural landscapes (Schimel *et al.* 1991, Goovaerts and Chiang 1993, Pennock *et al.* 1992, Van Kessel *et al.* 1993).

Little is known, however, regarding the influence of topography on the spatial distribution of soil parameters in forest landscapes. Microbial response to topographic control on moisture and nutrient redistribution within a forest landscape potentially could alter nutrient cycling processes and, consequently, influence the potential for nutrient losses. Unlike arable landscapes, however, the presence of trees in forest landscapes represent distinct sinks which are likely to farther modify microclimatic and hydrological conditions, and may result in considerable microscale variability (Binkley and Hart 1989). Moreover, removal of significant sinks due to disturbance events including clear-cut logging and burning is likely to have a significant impact on the subsequent type and intensity of soil processes within a landscape.

This study was conducted to identify landscape-scale patterns of soil N and soil N-cycling in forest soils and to investigate the influence of burning and clear-cut logging on subsequent N redistribution.

10.3 Materials and Methods

Three sites, representing three levels of disturbance (i.e., native (MF3), burned (MF2), and clear-cut (MF1)) and occurring on a similar geomorphic surface, were located within or associated with the Prince Albert Model Forest in Saskatchewan, Canada. MF3 has a mature mixed-wood forest assemblage. Vegetation and understory corresponded to a *Populus tremuloides-Picea glauca/Alnus/Cornus-Aralia/Linnaea* assemblage. MF2 has a former mixed-wood assemblage that had undergone an uncontrolled burn in 1989. MF1 was a former mixed-wood assemblage that had been cleared, trenched and planted to white spruce (*Picea glauca*) in 1989. Regeneration of trembling aspen (*Populus tremuloides*) was significant. Sites were characterized by 2-15% slopes with a maximum relief of 11-m.

In July, 1993, a representative 120 by 120-m area was selected at each site and a regularly spaced grid was laid out on the surface. Sites were surveyed and landscapes were stratified into distinct landform element complexes (Pennock *et al.*, 1994). Soils were characterized and sampled in 15-cm increments to a depth of 45-cm at each sampling point. The upper 15-cm increment included the LFH horizon at both the native and the burned sites. Because of site disturbance at MF1, a LFH horizon was not always discernible. Soil inorganic-N, pH, total C (C_{tot}) and total soil N (N_{tot}) were determined.

Ten footslope and ten shoulder complexes were randomly selected from each of the three sites, and sampled to a depth of 15-cm for subsequent determination of potential N mineralization and nitrification during an 8-wk aerobic laboratory incubation. Subsamples of mineral soil (20 g dry weight) and LFH material (5 g dry weight) were incubated in the dark at 20°C and 90% relative humidity in a controlled environment chamber. Production of $C0_2$ was measured for 24 h after each incubation period and NO_3^- -N and NH_4^+ -N was determined colorimetrically using a Technicon AutoAnalyzer (Technicon Industrial Methods 1978). The fumigation-extraction method (Vance *et al.* 1987) was used to estimate microbial biomass C (MB-C) and N (MB-N) at the onset of the incubation experiment.

The quantitatively defined landform element complexes were the units used to characterize spatial variability within the study sites. Differences between sample points grouped by landform element complexes were assessed using Kruskal-Wallis one-way analysis of variance by ranks corrected for ties (Siegal and Castellan 1988). A multiple comparison extension of the Kruskal-Wallis method was used to identify significant differences associated with landform element complexes at the three sites at the P=0.20 level.

10.4 Results and Discussion

Several soil characteristics, including pH, bulk density, C_{tot}, N_{tot} and C_{tot}/N_{tot} ratio differed markedly between sites (Table 10.1). In particular, C_{tot} and the C_{tot}/N_{tot} ratio were highest at MF1, which represents the greatest degree of site disturbance. Increased levels of C_{tot} may reflect inputs related to physical incorporation of logging slash and surface vegetation, and inputs related to post-disturbance plant succession. Differences also may reflect, in part, inherent soil characteristics although sites occurred on a similar geomorphic surface and shared a similar range of soils.

Table 10.1:
Median values (mean rank[a]) of soil characteristics (0-15 cm) at the study sites.

Forest Type	pH	Bulk density (g cm^{-3})	Moisture (%)	C_{tot} (%)	N_{tot} (%)	C_{tot} to N_{tot} ratio
MF3 (n=49)	5.5 (74a[b])	1.2 (77a)	26 (78)	2.1 (58b)	0.12(70a)	18 (46b)
MF2 (n=36)	5.7 (74a)	1.2 (72a)	24 (59)	1.8 (57b)	0.09 (56b)	20 (61b)
MF1 (n=49)	5.5 (57b)	1.0 (55b)	25 (62)	3.9 (85a)	0.15 (72a)	25 (92a)

[a] Mean rank = 1/n S (rank of individual sampling sites).
[b] Within columns, mean ranks followed by the same letter are not significantly different (multiple comparison extension of the Kruskal-Wallis *H*-test).

Although clearly defined foot-slope centered patterns of N accumulation within a toposequence have been observed (Schimel *et al.* 1991), the distribution of inorganic N and total N accumulation was random at all sites (data not shown), suggesting that large-scale topography had no detectable control on the distribution of N in these forest landscapes.

Lack of topographic control over distribution of inorganic-N likely was related to the dominant form of inorganic N . The relatively immobile cationic NH_4^+ -N form dominated at each site whereas levels of the mobile anionic NO_3^- -N form remained very low, irrespective of site disturbance (Table 10.2). Although the process of nitrification is favoured in arable soils because nitrifying organisms obtain their energy from this process, NH_4^+ typically dominates in forest soils (Vitousek *et al.* 1982, Prieto-Fernandez *et al.* 1993, Olson and Reiners 1983).

Various mechanisms that may limit net nitrification in forest soils have been suggested including allelochemic inhibition of nitrification, competition among organisms for limited nutrients, low initial populations of autotrophic nitrifying organisms, and edaphic conditions unfavorable to autotrophic nitrifiers, including low pH (Vitousek *et al.* 1982). It is unlikely that nitrification in these forest soils was restricted by low pH alone. The pH of the three soils (ranging from 5.5 to 5.7) were marginally below the range of pH 5.8-8.5 that is commonly recognized as being conducive to the growth of autotrophic ammonium oxidizers (Watson *et al.* 1989) (Table 10.1). Moreover, there are now numerous reports of active nitrification in forest soils of pH less than 5 (Pennington and Ellis 1993).

Table 10.2:
Median (mean rank[a]) of the inorganic-N in the upper soil profile (0-15 cm) at the study sites.

Forest Type	NH_4^+ -N (mg m^{-2})	NO_3^- -N (mg m^{-2},)	NH_4^+ to NO_3^- ratio
MF3 (n=49)	920 (74 a[b])	28 (47 b)	25 (76 a)
MF2 (n=36)	989 (75 a)	54 (74 a)	22 (62 a)
MF1 (n=49)	625 (55 b)	63 (82 a)	9 (43 b)

[a] Mean rank = 1/n S (rank of individual sampling sites).
[b] Within columns, mean ranks followed by the same letter are not significantly different (multiple comparison extension of the Kruskal-Wallis H-test).

Because NO_3^- is favoured for plant uptake, it is possible that the vegetation at the MF3 provided a strong NO_3^- sink which served to limit the detection of available soil NO_3^-. It follows that the removal of the sink would result in an increase in NO_3^- levels. Indeed, although NO_3^- levels were significantly higher at both the MF1 and MF2, the actual levels of NO_3^- remained low at all sites (Table 10.2). A narrowed NH_4^+ / NO_3^- ratio at MF1 suggests that N-cycling processes may have changed in response to disturbance.

The microbial biomass is considered a relatively labile fraction of the soil organic matter and, thus, is an important factor governing the availability of plant nutrients. The microbial biomass C and N in these forest soils was similar to values reported by others (Diaz-Ravina *et al.* 1993, Martikainen and Palojärvi 1990) (Table 10.3).

The MB-C was only significantly higher in soils from shoulder positions at MF2 (Table 10. 3). With this exception, MB-C and MB-N did not differ significantly between landform element complexes within sites. A non-parametric index of variability ((interquartile range/median) x 100)) indicated a high degree of variability for both MB-C (37%) and MB-N (41%). Thus, detection of differences in the microbial biomass between landform element complexes may have been limited by a high degree of variability unrelated to site topography.

Although significant differences in MB-C within sites were limited, differences between sites were apparent (Table 10.3). The MB-C was significantly higher in soils from the shoulder positions at MF 1 than at MF3. Restricted MB-C at MF3 may have been due to nutrient limitations resulting from sequestration of nutrients in standing biomass. No significant differences in MB-N between sites were detected.

Table 10.3:
Median values (mean rank[a]) of MB-C and MB-N, and relationships with C_{tot} and N_{tot}.

Forest type (n=10)		MB-C ($g\ m^{-2}$)	MB-N ($g\ m^{-2}$)	MB-C to MB-N ratio	MB-C to C_{tot} ratio (%)	MB-N to N_{tot} ratio (%)
MF3	Shoulder	59 (23 bcd[b])	10 (29)	5.5 (22 b)	1.8 (24)	5.8 (22 b)
	Footslope	56 (14d)	10 (19)	6.0 (26 b)	2.0 (26)	6.1 (24 ab)
MF2	Shoulder	79 (40 ab)	13 (37)	6.3 (29 b)	2.0 (33)	7.5 (36 ab)
	Footslope	59 (20 cd)	11 (30)	5.4 (20 b)	2.0 (35)	6.2 (31 ab)
MF1	Shoulder	95 (49 a)	12 (32)	9.0 (51 a)	2.4 (33)	6.1 (29 ab)
	Footslope	83 (37 abc)	14 (36)	6.4 (34ab)	2.2 (32)	10.3 (38 a)

[a] Mean rank = 1/n S (rank of individual sampling sites).
[b] Within columns, mean ranks followed by the same letter are not significantly different (multiple comparison extension of the Kruskal-Wallis H-test).

The widening of the MB-C/MB-N ratio, particularly in shoulder positions at MF1, suggests a shift in the microbial biomass species composition (Table 10.3). According to data presented by Anderson and Domsch (1984), the mean C/N ratio of 10 species of soil bacteria was 5.61 whereas the mean C/N ratio of 14 species of soil fungi was 8.26. These values suggest that soils with a high C/N ratio at MF1 were likely dominated by a decomposer microbial population characterized by fungi.

Cumulative C mineralization was unaffected by landscape position or level of site disturbance, irrespective of significant differences in soil C/N ratios at the three sites (Table 10.4). Relatively high cumulative net N mineralization in shoulder positions at MF3 likely reflects a relative abundance of readily available C and N resources. It is possible that C availability at MF1 was limited by the form of organic C, i.e., refractory versus readily available.

Total nitrogen mineralization may have been limited in the soils from footslopes of disturbed sites as compared to soils from shoulder positions at MF3 by a reduction in the quality or quantity of readily available C and/or nutrient reserves prior to incubation (Table 10.4). Many reports have suggested that rates of mineralization, and particularly nitrification, increase after major disturbance events including clear-cut logging (Matson and Vitousek 1981, Vitousek and Andariese 1986) and fires (White 1986). Typically these reports suggest that disturbance increases mineralization rates for a relatively short period of time which is followed by a longer period of recovery during which net mineralization decreases.

Table 10.4.
Median values (mean rank[a]) and cumulative C and N mineralization, and relationships with MB-C and MB-N.

Forest type (n=10)		Cumulative C Mineralization (g CO_2 -C m^{-2})	Net N Mineralization (g N m^{-2})	C mineralized to MB-C ratio ($mgCd^{-1}\ g^{-1}\ C_{mic}$)	N mineralized to MB-N ratio ($mgNd^{-1}\ g^{-1}\ N_{mic}$)
MF 1	Shoulder	562 (31)	7.8 (45 a[b])	175 (37 ab)	13.6 (45 a)
	Footslope	828 (34)	4.7 (34 ab)	214 (40 a)	9.5 (39 ab)
MF 2	Shoulder	573 (23)	5.7 (35 ab)	126 (21 bc)	9.4 (31 abc)
	Footslope	610 (28)	3.8 (22 b)	160 (35 ab)	5.9 (22 bc)
MF 1	Shoulder	419 (23)	4.9 (31 ab)	84 (15 c)	7.2 (29 abc)
	Footslope	733 (33)	2.3 (18 b)	141 (30 abc)	4.1 (18 c)

[a] Mean rank = 1/n S (rank of individual sampling sites).
[b] Within columns, mean ranks followed by the same letter are not significantly different (multiple comparison extension of the Kruskal-Wallis H-test).

An explanation for reduced mineralization in disturbed sites may rest in the activity of the microbial biomass. A reduction in the quality of the C substrate or some other nutrient limitation may have limited microbial activity. Losses of nutrients via leaching following clear-cut logging (Vitousek *et al.* 1982) and volatilization and aeolian loss during burning (Raison *et al.* 1985) have been reported. 1t also has been suggested that fire could reduce the most labile organic fractions leaving the recalcitrant fractions, which would reduce the ability of the ecosystem to supply inorganic N through organic N mineralization (White 1986). Prieto-Fernandez (1993) similarly observed that the potentially mineralizable N decreased, whereas the rate of N mineralization increased in soils following wildfire, and suggested that this combination could lead to a rapid depletion of the labile organic N.

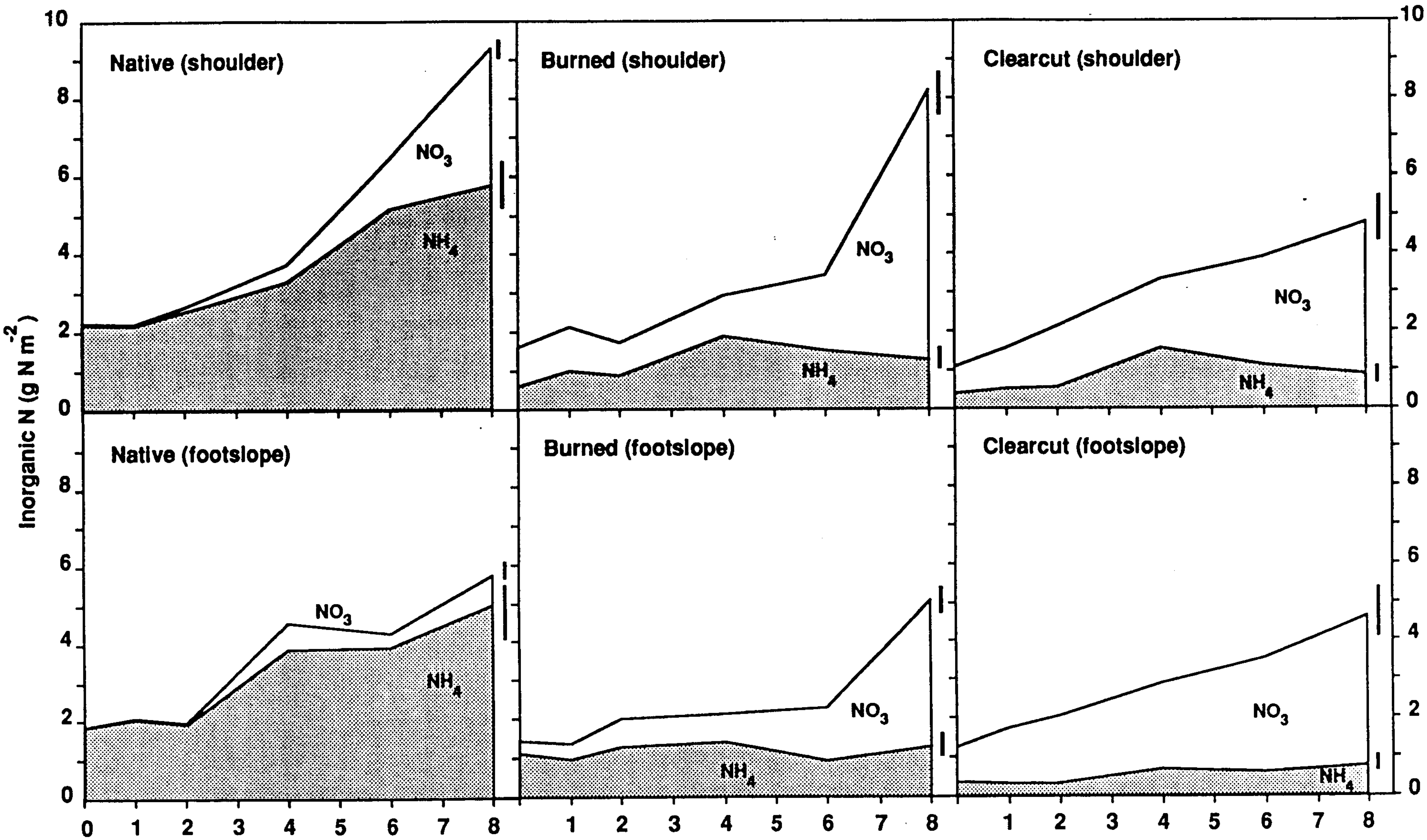
In general, values of the rate of C and N turnover per unit of microbial biomass were higher at MF3 (Table 10.4). A change in the rates of C and N turnover through the microbial biomass may be indicative of a shift in the bacterial-fungal ratio (Anderson and Domsch 1993). Alternatively, Anderson and Domsch (1993) suggested that an increased rate of C turnover per unit microbial biomass also can be an indicator of community stress because organisms divert energy from growth and production to maintenance. It follows that soils from MF3 would be expected to have the lowest, rather than the highest, rates of C and N turnover per unit of microbial biomass as compared to disturbed sites. However, the microbial biomass was measured at the initiation of the laboratory incubation and sample collection, preparation, and incubation likely stimulated and contributed to a rapid microbial colonization of soils. In, particular, fast- growing, opportunistic organisms may have flourished in the presence of readily available C, subsequent to the determination of microbial biomass. Thus the more immediate disturbance (i.e., sample collection), likely overshadowed any effects of previous site history.

Patterns of NO_3^- and NH_4^+ accumulation differed markedly between soils from the three sites (Fig. 10.1). Although NH_4^+ accumulation continued to increase in soils from MF3, NO_3^- accumulation remained limited. In contrast, NH_4^+ accumulation in soils from both the MF1 and MF2s remained limited whereas NO_3^- accumulated rapidly, particularly in soils from MF1. Vitousek and co-workers (1982) reported that soil nitrogen was mineralized rapidly to NH_4^+ during a laboratory incubation in both forest floor and mineral soils; however, abundant NO_3^- was produced only in the mineral soil. These observations lead Olson and Reiners (1983) subsequently to suggest that disturbance of soils with a layered system of N-cycling typical of forest soils, as by clearcut logging, could result in substantial NO_3^- losses in the hydrological flux out of the system. In our study, the incorporation of the underlaying mineral soil into the forest floor material during clear-cut and subsequent planting operations may have facilitated the establishment of rapid nitrification, thereby limiting the accumulation of NH_4^+. The presence of strong vegetative sinks, including the profuse regeneration of aspen at MF1, may have limited NO_3^- accumulation in situ, whereas rapid NO_3^- accumulation was detected in vitro.

Lags in nitrification, particularly evident in soils from both the burned and native sites, have been observed by others (Vitousek *et al.* 1982, Hart *et al.* 1994). Vitousek *et al.* (1982) suggested that lags of a few weeks or months in NO_3^- production behind the appearance of elevated NH_4^+ could occur because NO_3^- is not produced or any that is produced is lost rapidly through immobilization or denitrification. Ammonium, however, is preferred by heterotrophs as a nitrogen source (Jones and Richards 1977) and thus it is unlikely that NO_3^- was preferentially immobilized over NH_4^+. Vitousek *et al.* (1982) concluded that lags in nitrification were probably caused by either low initial populations of nitrifying bacteria or the allelochemic inhibition of nitrification. In a subsequent study, Vitousek and Matson (1985) concluded that mechanisms by which nitrification is delayed may vary between sites.

Differences between NO_3^- and NH_4^+ accumulation between shoulder and footslope positions within sites generally were not significant (Fig. 10.1). Lensi *et al.* (1991) reported that the potential for nitrification in the A1 soil horizon of an acid forest soil showed a high degree of variability at a 1 m scale along a 10-m transect. Noting that Robertson (1982) reported a variability of the same order of magnitude between different forest soils, they suggested that the differences observed at a 10-m scale at the same site were probably similar to those observed between different sites. Thus, microscale variability in N-cycling processes may limit the detection of possible macroscale patterns of N-cycling related to topographic controls within these forest landscapes.

Figure 10.1.
Accumulation of inorganic-N during a 55-d aerobic laboratory incubation in soils from footslope and shoulder positions of the native, burned and clear-cut sites. Within each graph, the upper line represents total inorganic-N (NO_3^- plus NH_4^+-N) whereas the lower line represents the accumulation of NH_4^+-N. The error bars represent the median absolute deviation from the median (MAD).



10.5 Conclusions

The results of this study emphasize the conservative nature of undisturbed forest ecosystems for nitrogen. Even during an extended incubation period under conditions favorable for nitrification, NO_3^- production remained limited in soil from MF3. Reduced levels of NO_3^- consequently limits the potential for N loss via leaching and/or denitrification.

Several measured soil parameters were significantly altered in response to site disturbance. Generally, the effects of clear-cut logging were more pronounced than the effects of burning. Few soil characteristics, however, were significantly affected by landscape position within sites, suggesting an absence of topographic control, irrespective of site disturbance, at the scale studied. Failure to detect evidence of topographic controls on the distribution of inorganic and total N suggests that topographically controlled hydrologic processes had a limited impact on N-cycling in these forest soils. Alternatively, microscale variation in soil characteristics may have limited detection of macroscale differences.

Soils from a native site had the greatest potential for N mineralization and subsequent loss. Reduced levels of potential net N mineralization in soils from footslope positions within a previously burned or clear-cut site suggest that N-cycling processes may have been limited by a reduction in the quantity of readily available C and/or nutrient reserves incurred as a consequence of disturbance prior to incubation. Disturbance may have altered the microbial biomass in favour of fast-growing, opportunistic organisms, resulting in a subsequent depiction of C and nutrient reserves.

11. LIST OF PRESENTATIONS AND PUBLICATIONS RESULTING FROM MODEL FOREST RESEARCH

Thesis

J.T. Braidek, 1996. Development of a landscape-scale soil distribution model for the Mixedwood region of Saskatchewan. M. Sc. Thesis, Dept. of Soil Science, University of Saskatchewan, Sept. 1996.

Publications

Walley, F.L., C. van Kessel, and D.J. Pennock. 1995. Landscape-scale variability of nitrogen mineralization in forest soils. Soil Biology and Biochemistry 28: 383-391.

Pennock, D.J. and C. van Kessel. 1996. Impact of agriculture and forestry on landscape-scale soil organic carbon storage in Saskatchewan. Canadian Journal of Soil Science (in press).

Pennock, D.J. and C. van Kessel. 1996. Clear-cut forest harvest effects on soil quality indicators in the Mixedwood Forest of Saskatchewan. Geoderma (in press).

Pennock, D.J., J. Braidek, R. Anderson, and K. Elliott. 1996. Characterization of the major ecosections in the Wapawekka Hills and Waskesiu Hills Ecodistricts of Central Saskatchewan. Report submitted to Forestry Canada, 48 pp.

Presentations

Pennock, D.J., C. van Kessel, and F. Walley. 1994. Landscape-scale variability of soil properties in the Mixedwood forest of Saskatchewan. Oral presentation made at Canadian Society of Soil Science Annual Meeting, July 10-13, 1994, Regina, Saskatchewan. Abstract: pg.6 of Abstracts of Technical Papers and Posters.

Walley, F., C. van Kessel, and D.J. Pennock. 1994. Landscape-scale variability of nitrogen mineralization and nitrification in undisturbed and disturbed forest soils in Saskatchewan, Canada. Poster presentation made at International Boreal Forest Research Association Conference, September 25-28, 1994, Saskatoon, Saskatchewan. Abstract: pg. 91 of IBRFA '94 Conference Abstracts.

Pennock, D.J., C. van Kessel, and F. Walley. 1994, Large-scale variability of soil properties in the mixedwood forest of Saskatchewan, Canada. Oral presentation made at international Boreal Forest Research Association Conference, September 25-28, 1994, Saskatoon, Saskatchewan. Abstract: pg. 64 of IBRFA '94 Conference Abstracts.

Walley, F., C. van Kessel, and D.J. Pennock. 1994. Spatial variability of nitrogen mineralization and nitrification in forest soils of Saskatchewan, Canada. Oral presentation made at Soil Science Society of America Annual Meetings, Seattle Washington, November 13-18, 1994. Abstract: pg. 382 of Agronomy Abstracts, 1994 Annual Meetings of Soil Science Society of America.

Pennock, D.J. and C. van Kessel. 1995. Impact of Agriculture and Forestry on Soil Organic Carbon Storage in Saskatchewan. Oral Presentation to Soils and Crops Workshop 1995, Saskatoon Saskatchewan, February 23, 1995.

Pennock, D.J. and C. van Kessel. 1995. Impact of Agriculture and Forestry on Soil Organic Carbon Storage in Saskatchewan. Proceedings, Soils and Crops Workshop 1995, Saskatoon, Saskatchewan.

Pennock, D.J. and C. van Kessel. 1995. Impact of Agriculture and Forestry on Soil Organic Carbon Storage in Saskatchewan. Oral presentation to Canadian Society of Soil Science Annual Meeting, Laval, July, 1995.

Pennock, D.J. 1996. Clear-cut forest harvest impacts on the soils of the Mixedwood Forest in Saskatchewan. Oral Presentation to the Canadian Society of Soil Science Annual Meeting, Lethbridge, July 1996.

Braidek, J.T. and D.J. Pennock. 1996. Development of a landscape-scale soil distribution model for the Mixedwood region of Saskatchewan. Oral Presentation to the Canadian Society of Soil Science Annual Meeting, Lethbridge, July 1996.

Personnel Involved with the Model Forest Research

Graduate and Post-Graduate Researchers:

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Field Assistants:

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J. Nitschelm, Summer Assistant (4 months)
A. Frick, Summer Assistant (4 months)
S. Exner, Summer Assistant and Undergraduate Thesis research (12 months)
S. Blechinger, Summer Assistant (4 months)

12. REFERENCES CITED

Acton, D.F. and Gregorich, L.J.(editors), 1995. The Health of Our Soils. Toward sustainable agriculture in Canada. Centre for Land and Biological Resources Research, Agriculture and Agri-Food Canada. Publication 1906/E, Ottawa, Canada.

Alban, D.H., 1982. Effects of nutrient accumulation by aspen, spruce, and pine on soil properties. Soil Sci. Soc. Am. J. 46: 853-861.

Alban, D.H. and Perala, D.A., 1990. Impact of Aspen timber harvesting on soils. In: Gessel, S.P., Lacate, D.S., Weetman, G.F., and Powers, R.F. (eds.) Sustained Productivity of Forest Soils. Proc. 7th North Amer. Forest Soils Conf., Univ. of British Columbia, Vancouver, B.C., pp. 377-391.

Anderson, D.W. and Ellis, J.G., 1976. The soils of the Provincial forest reserves in the Prince Albert Map Area 73H Saskatchewan. Sask. Inst. of Pedology Publ. SF3, Saskatoon, Saskatchewan, Canada.

Anderson, J.P.E. and Domsch K.H. (1980) Quantities of plant nutrients in the microbial biomass of selected soils. Soil Science 130, 211-216.

Binkley D. and Hart S.C., 1989 The components of nitrogen availability assessments in forest soils. Advances in Soil Science 10, 57-112

Beckingham, J.D., D.G. Nielsen, and V.A. Futoransky. 1995. Field Guide to the MidBoreal Ecoregion of Saskatchewan. Draft 2. Geographic Dynamics Corporation, Edmonton, Alberta.

Corre, M.D., van Kessel, C. and Pennock, D.J., 1996 (in press). Landscape-scale patterns and seasonal flucuations of N2O emission in a semi-arid region, Saskatchewan, Canada. Soil Sci. Soc. Am. J. (in press).

Diaz-Ravina, M. Acea M.J. and Carballas T. 1993 Microbial biomass and its contribution to nutrient concentrations in forest soils. Soil Biology and Biochemistry 25, 25-31.

Doran, J.W. and Parkin, T.B., 1994. Defining and assessing soil quality. In: Doran, J.W., Coleman, D.C., Bexdicek, D.F., and Stewart, B.A. (eds.). Defining Soil Quality for a Sustainable Environment. SSSA Special Publ. Number 35. Soil Sci. Soc. of Am. Inc. and Am. Soc. of Agron. Inc., Madison, Wisconsin, pp. 3-21.

Donald, R.G., Anderson, D.W., and Stewart, J.W.B., 1993. The distribution of selected soil properties in relation to landscape morphology in forested Gray Luvisolic soils. Can. J. Soil Sci. 73: 165-172.

Dyck, W.J. and Cole, D.W., 1994. Strategies for determining consequences of harvesting and associated practises on long-terra. productivity. In: Dyck, W.J., Cole, D.W., and Comerford, N.B. (eds.). Impacts of forest harvesting on long-term site productivity. Chapman and Hall, London, pp. 13-40.

Ellis, J.G. and Clayton, J.S. 1970. The Physiographic Divisions of the Northern Provincial Forest in Saskatchewan. Sask. Inst. of Pedology Publ. SP3.

Environmental Protection Agency, 1971. Methods of chemical analysis for water and Wastes. E.P.A., Cincinnati, Ohio.

Gordon, A.G., 1983. Nutrient cycling dynamics in differing spruce and mixedwood ecosystems in Ontario and the effects of nutrient removals through harvesting. In: Wien, R.W., Riewe, R.R., and Methuen, I.R., (eds.) Resources and Dynamics of the Boreal Zone. Proc. of Assoc. of Canadian Univ. for Northern Studies, Thunder Bay, Ontario, August, 1982, pp. 97-118.

Greacen, E.L. and Sands, R., 1984. Compaction of forest soils: a review. Aust. J. Soil Res. 18: 163-189.

Goovaerts P. and Chiang C.N. 1993 Temporal persistance of spatial pattens for mineralizable nitrogen and selected soil properties. Soil Science Society of America Journal 57,372-381.

Hairston, A.B. and Grigal, D.F., 1994. Topographic variation in soil water and nitrogen for two forested landforms in Minnesota, USA. Geoderma 64: 125-138.

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

Hatchell, G.E., Ralston, C.W., and Foil, R.R., 1970. Soil disturbances in logging. J. For. 69: 772-775.

Head, W.K., Anderson, D.W., and Ellis, J.G., 1981. The soils of the Wapawekka Map Area 73-1 Saskatchewan. Saskatchewan Inst. of Pedology Pub. SF5, Saskatoon, Saskatchewan, Canada.

Indorante, S.J., Follmer, L.R., Hammer, R.D., and Koenig, P.G., 1990. Particle-size analysis by a modified pipette procedure. Soil Sci. Soc. Am. J. 54:560-563.

Johnson, D.W., 1992. Effects of forest management on soil carbon storage. Water, Air and Soil Pollut. 64: 83-120.

Johnson, D.W., 1994. Reasons for concern over impacts of harvesting. In: Dyck, W.J., Cole, D.W., and Comerford, N.B. (eds.). Impacts of forest harvesting on long-term site productivity. Chapman and Hall, London, pp. 1-12.

Jones J.M. and Richards B.N. 1977 Effect of reforestation on turnover of ^{15}N-labelled nitrate and ammonium in relation to changes in soil microflora. Soil Biology and Biochemistry 9, 383-392.

Kabzems, A., Kosowan, A.L., and Harris, W.C., 1986. Mixedwood section in an ecological perspective. Technical Bulletin No. 8. (2nd ed.), Saskatchewan Parks and Renewable Resources, Forestry Division, Prince Albert, Saskatchewan, Canada.

Kachanoski, R.G. and de Jong, E. 1982. Comparison of the soil water cycle in clear-cut and forested sites. J. Environ. Qual. 11: 545-549.

Krause, H.H., and Ramlal, D., 1987. In situ nutrient extraction by resin from forested, clear-cut and site-prepared soil. Can. J. Soil Sci. 67: 943-952.

Kutiel P. and Naveh Z. 1987 The effect of fire on nutrients in a pine forest soil. Plant Soil 104, 269-274.

Larson, W.E. and Pierce, F.J., 1994. The dynamics of soil quality as a measure of sustainable management. In: Doran, J.W., Coleman, D.C., Bexdicek, D.F., and Stewart, B.A. (eds.). Defining Soil Quality for a Sustainable Environment. SSSA Special Publication Number 35. Soil Sci. Soc. of America Inc. and Am. Soc, of Agron. Inc., Madison, Wisconsin, pp. 37-51.

Lensi R., Lescure C., Clays-Josserand A. and Gourbiere F. 1991 Spatial distribution of nitrification and denitrification in an acid forest soil. Forest Ecology and Management 44, 29-40.

Martikainen P.J. and Palojärvi A. 1990. Evaluation of the fumigation-extraction method for the determination of microbial C and N in a range of forest soils. Soil Biology and Biochemistry 22, 797-802.

Matron P.A. and Vitousek P.M. 1981. Nitrification potentials following clearcutting in the Hoosier National Forest, Indiana. Forest Science 27, 781-791.

McKeague, J.A. (ed.), 1978. Manual on soil sampling and methods of analysis (2nd ed.). Canadian Society of Soil Science, Ottawa, Ontario.

Neary, D. G. and Hornbeck, J. W., 1994. Impacts of harvesting and associated practices on off-site environmental quality. In: Dyck, W.J., Cole, D.W., and Comerford, N.B. (eds.). Impacts of forest harvesting on long-term site productivity. Chapman and Hall, London, pp. 81-118.

Olson R.K. and Reiners W.A. 1983. Nitrification in subalpine balsam fir soils: tests for inhibitory factors. Soil Biology and Biochemistry 15, 413-418.

Padbury, G.A., and Acton, D.F., 1994. Ecoregions of Saskatchewan. Center for Land and Biol. Resources Res., Res. Branch, Agriculture and Agri-Food Canada, Ottawa.

Padbury, G.A., W.K. Head, and W.E. Souster. 1978. Biophysical Resource Inventory of the Prince Albert National Park, Saskatchewan. Sask. Inst. of Pedology Publ. 5185.

Pennington P.I. and Ellis R.C. 1993. Autotrophic and heterotrophic nitrification in acidic forest and native grassland soils. Soil Biology and Biochemistry 10, 1399-1408.

Pennock, D.J. and van Kessel, C., 1996 (in press). Effect of agriculture and of clear-cut forest harvest on landscape-scale soil organic carbon storage in Saskatchewan. Can. J. Soil Sci.

Pennock, D.J., Zebarth, B.J., and de Jong, E., 1987. Landform classification and soil distribution in hummocky terrain, Saskatchewan, Canada. Geoderma 40: 297-315

Pennock, D.J., Anderson, D. W., and de Jong, E., 1994. Landscape-scale changes in indicators of soil quality due to cultivation in Saskatchewan, Canada. Geoderma 64: 1-19.

Pennock D.J., van Kessel C, Farrell R.E. and Sutherland R.A. 1992. Landscape-scale variations in denitrification. Soil Science Society of America Journal 56, 770-775.

Peterman, R.M., 1990. Statistical power analysis can improve fisheries research and management. Can. J. Fish. Aquat. Sci. 47: 2-15.

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

Raison R.J., Khanna P.K. and Woods P.V. 1985. Transfer of elements to the atmosphere during low intensity prescribed fires in three Australian subalpine eucalypt forests. Canadian Journal of Forest Research 15, 657-664.

Rapp, M. (1990) Nitrogen status and mineralization in natural and disturbed Mediterranean forests and coppices. Plant Soil 128, 21-30.

Robertson G.P. 1982, Nitrification in forested ecosystems. Philosophical Transactions of the Royal Society of London Bulletin 296, 445-457.

Richardson, J.L., Wilding, L.P., and Daniels, R.B. 1992. Recharge and discharge of groundwater in aquic conditions illustrated with flownet analysis. Geoderma 53: 65-78.

Rowe J.S. 1984. Forestland classification: limitations of the use of vegetation. In: Forestland Classification; Experiences, Problems, Perspectives. (J.G. Bockheim, Ed.), pp. 132-148. Department of Soil Science, Madison, Wl.

Schimel D., Kittel T.G.F., Knapp A.K., Seastedt T.R., Parton W.J. and Brown V.B. 1991. Physiological interactions along resource gradients in a tallgrass prairie. Ecology 72, 672-684.

Schimel J.P., Firestone M.K., and Killham K.S. 1984 Identification of heterotrophic nitrification in a Sierras forest soil. Applied and Environmental Microbiology 48, 802-806.

Siegal S. and Castellan Jr. N.J. 1988. Nonparametric Statistics for the Behavioral Sciences. McGraw-Hill Book Company, Toronto.

Snedecor, G.W. and Cochran, W.G., 1989. Statistical Methods (Eighth Edition). Iowa State University Press, Ames, Iowa.

Technicon Industrial Systems. 1978. Ammonium in water and seawater. Industrial Method 154-71 W/B. Technicon Industrial Systems, Tarrytown, N.Y.

Timmer, V.R., Savinsky, H.M., and Marek, G.T., 1983. Impact of intensive harvesting on nutrient budgets of boreal forest stands. In: Wien, R.W., Riewe, R.R., and Methuen, I.R., (eds.) Resources and Dynamics of the Boreal Zone. Proceed. of Assoc. of Canadian Univ. for Northern Stud., Thunder Bay, Ontario, August, 1982, pp. 131-147.

Van Kessel C., Pennock D.J. and Farrell R.E. 1993. Seasonal variations in denitrification and N20 evolution at the landscape-scale. Soil Science Society of America Journal 57, 988-995.

Vance E.D., Brookes P.C. and Jenkinson D.S. 1987. An extraction method for measuring soil microbial biomass C. Soil Biology and Biochemistry 19, 703-707. Vitousek P.M. and Andariese S.W. 1986. Microbial transformations of labelled nitrogen in a clear-cut pine plantation. Oecologia 68, 601-605.

Vitousek P.M. and Matson P.M. 1985. Causes of delayed nitrate production in two Indiana forests. Forest Science 31, 122-131.

Vitousek P.M., Gosz J.R., Grier C.C., Melillo J.M. and Refiners W.A. 1982. A comparative analysis of potential nitrification and nitrate mobility in forest ecosystems. Ecological Monographs 52, 155-177.

White C.S. 1986. Effects of prescribed fire on rates of decomposition and nitrogen mineralization in a ponderosa pine ecosystem. Biology and Fertility of Soils 2, 87-95.