DEFORESTATION IN CENTRAL SASKATCHEWAN: EFFECTS ON LANDSCAPE STRUCTURE AND ECOSYSTEM CARBON DENSITIES

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ABSTRACT

Deforestation is recognized as a serious global problem that contributes to biodiversity loss, soil degradation and atmospheric change. This thesis is an investigation of deforestation in central Saskatchewan. The purposes are: to quantify the extent and rates of deforestation and associated changes in spatial structure for multi-jurisdictional boreal landscapes; and to estimate the magnitude of carbon losses associated with agriculture-induced deforestation at sites within one of these landscapes.

Deforestation was analyzed using topographic map chronosequences for two 460 000 ha landscapes in central Saskatchewan. An estimated 16 400 ha was deforested between 1963 and 1990 within the Waskesiu Hills landscape (53° 45' N, 106° 15' W) and 371 000 ha was deforested between 1957 and 1990 within the Red Deer River landscape (52° 45' N, 103° 00' W). Federal and provincial legislation establishing publicly owned parks and forests served to inhibit deforestation within portions of these landscapes. On agricultural lands within the two landscapes, where private holdings dominate and forests are not protected under federal or provincial law, deforestation occurred at rates exceeding 1.2 % yr^{-1} over the time periods examined even though human populations declined. Within the two study areas, extant forests that are unprotected by legislation remain vulnerable to deforestation.

Spatial structure was analyzed for portions of these two landscapes using landscape metrics. A positive correlation between largest patch size index and proportion of land area wooded was evident for both 1975/76 ($r^2 = 0.99$, $p < 0.01$) and 1990 ($r^2 = 0.99$, $p < 0.05$). Since past deforestation disproportionately reduced the sizes of the largest wooded patches, future reforestation efforts should be aimed at expanding large patches. Reforestation with large patches contiguous to protected forests may initiate a reversal of the process of fragmentation that has impaired forest wildlife and ecosystem processes.

Vegetation carbon densities were compared at six forest sites, six pasture sites and six cultivated sites on hummocky glacial till landforms across three townships within the Waskesiu Hills landscape. Medians for aboveground biomass (60 Mg C ha^{-1} for forests, 1 Mg C ha^{-1} for pastures and 4 Mg C ha^{-1} for cultivated sites) were significantly different ($p < 0.15$). Including estimated losses of coarse roots, deforestation and subsequent agricultural land use led to losses of approximately 70 Mg C ha^{-1} for vegetation.

Statistically significant losses of soil organic carbon were not detected between the forest sites and the agricultural sites. The experimental design accounted for land use and topographic landform effects, but these were small (< 20 Mg C ha^{-1}) relative to the inherent range of variation in soil organic carbon within the study area (> 50 Mg C ha^{-1} for natural forest sites). Across all sites regardless of land use, there was a positive correlation ($r_S = 0.76$, $p < 0.01$) between soil organic carbon and the proportional frequency of groundwater influenced soils (Gleysols plus Woody Calcareous Chernozems).

Terrestrial carbon losses due to deforestation within the Waskesiu Hills and Red Deer River landscapes prior to 1990 were crudely estimated as approximately 100 Gg C yr^{-1}. Carbon releases due to deforestation across the Boreal Plain Ecozone of Saskatchewan prior to 1990 were estimated to be an order of magnitude greater, and may have exceeded direct carbon emissions resulting from fossil fuel burning by the provincial agricultural production sector. Reestablishing natural forests on 2 % of agricultural lands within the Boreal Plain Ecozone might offset a substantial portion of the direct carbon emissions from agricultural fossil fuel burning in Saskatchewan.

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DEDICATION

This thesis is dedicated to my loving wife and life-partner Colleen Watson, for contributing her time and energy to digitize maps, drive to study sites, collect vegetation (amidst the bison, bears and snakes), dry and weigh samples, sieve sand, enter data, and perform many other vital tasks over the past five years; all the while generously providing love, friendship, encouragement and emotional support to her husband, parents, children and grandchild.

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LIST OF ABBREVIATIONS

C: carbon

COHE: degree of coherence

dbh: diameter at breast height

IGI: index of groundwater influence

LPS: largest patch size

LPSI: largest patch size index

MPS: mean patch size

MPSI: mean patch size index

MSIZ: effective mesh size

NP: number of patches

NTS: National Topographic System

P:A: perimeter-to-area ratio

PD: patch density

SOC: soil organic carbon

"Wood may be regarded as merely a by-product of trees. Their greatest value is probably their beneficial effect upon life, health, climate, soil, rainfall and streams" (Baker* 1944, p. 244).

1. INTRODUCTION

In pre-agricultural times forests and woodlands are thought to have covered approximately 6000 Mha, or almost half of the Earth's land surface (FAO 2001). Trees provide direct material benefit to human society in the form of a renewable supply of energy, building materials, food, paper and other products. In addition, forests serve many critical functions that support all life in the biosphere. Forests are the largest global reservoirs of terrestrial species and genetic diversity (Raven 1988). Forest vegetation and soils play important roles in biogeochemical cycles, particularly global fluxes of water and carbon. Forest ecosystems greatly influence energy flows because tree cover lowers the Earth's albedo and increases photosynthetic conversion of light to chemical energy (Woodwell 2001).

Global forest cover was reduced to 3 869 Mha, or 30 % of the Earth's land surface, by the turn of the millenium (FAO 2001). Global deforestation has resulted from over-harvesting, fuelwood collection, pollution and other factors, but its primary cause has been agricultural expansion by way of burning and clearing to create pastures and croplands (Williams 1989a). Global deforestation reduces the potential domestic harvest and commercial production of forest products, but more importantly, sacrifices irreplaceable environmental services. The perpetual decline in the Earth's forest cover is now recognized as one of the most serious global problems presently facing humanity (Myers 1996, WCFSD 1999).

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* Richard St. Barbe Baker, one of the first students to enroll at the University of Saskatchewan, was one of the twentieth century's foremost advocates of international action to stop deforestation and restore global tree cover.

The loss of forest cover has often been treated as an inevitable consequence of advancing technology and cultural development. Thus it has received relatively little historic scrutiny (Williams 1990). In local contemporary parlance, the word "deforestation" is often avoided in favour of less pejorative phrases such as "conversion to agriculture" (SERM 1993; p. 149) and "land clearing" (SERM 1995a; p. 2.32).

1.1 Definitions

The term "deforestation" is rarely defined in the scientific literature. A recent unpublished review by Lund (2000) classifies existing definitions into those that define deforestation as: (i) a change in land cover; (ii) a change in land use; or (iii) a change in land cover and use. Change in land cover without a change in land use (e.g. the encroachment of non-forested wetlands by trees or the expansion of wetlands due to beaver activity) or change in land use without a change in land cover (e.g. transfer of public forest land from a forestry agency to a non-forestry agency without removal of tree cover) is only of minor interest from a conservation point of view. Deforestation is of greatest concern when it involves change to both land use and land cover. An applicable definition is "the removal of a forest stand where the land is put to a non-forest use" (Helms 1998, p.44).

Similarly, the definitions of reforestation that are most applicable to this research effort are those that include restoration of land cover and a change in land use. An example of such a definition is to "establish forest on land that has been under other land use for some prior period of time" (Noble et al. 2000, p. 69). Reforestation occurs on land that supported tree cover in recent times, whereas afforestation occurs on land that has not supported forest in historical time (Noble et al. 2000).

1.2 Global Deforestation

Loss of biodiversity is one of the most important ecological implications of deforestation (FAO 1995). Agricultural expansion in forest biomes causes direct habitat loss for forest-dwelling species, resulting in species extinctions and losses of genetic diversity (Ledig 1992). In addition to losses in forest area, both biota and ecosystem processes are affected by fragmentation of natural habitats (Harris 1984, Wilcove et al. 1986, Andren and Angelstam 1988, Andren 1997). Forest fragmentation reduces ecosystem diversity and habitat heterogeneity, creates abrupt forest edges adjacent to areas inhospitable to forest biota, reduces native biotic diversity and increases opportunities for invasions of exotic species (Barnes et al. 1998).

Other environmental effects of deforestation include soil degradation, alteration of atmospheric composition and climatic change (FAO 1995). Past deforestation and associated agricultural land-use activities, cultivation in particular, reduced soil organic matter (Buringh 1984, Schlesinger 1984, Mann 1986, Davidson and Ackerman 1993) although recent agricultural practices may have resulted in a partial recovery of soil carbon stocks (Buyanovsky and Wagner 1998). Deforestation also reduced aboveground terrestrial carbon stocks by diminishing vegetation stature and decreasing standing woody biomass (Houghton et al. 1983, Houghton and Skole 1990). After fossil fuel burning, land-use change is the next largest source of global carbon emissions. Approximately 87% of global emissions from land-use change over the past 150 years were from forest regions and the dominant cause of these emissions was agricultural expansion (Houghton 1999). Over the 1980s, net emissions from changes in land use contributed 2.0 ($\pm$ 0.8) Pg C to the atmosphere annually. This compares in magnitude to 5.5 ($\pm$0.5) Pg C emitted annually from fossil fuel burning and a net atmospheric increase of 3.3 ($\pm$0.2) Pg C after accounting for terrestrial and oceanic sinks (Houghton 1999).

Rapid and widespread deforestation in tropical regions has gained global attention over recent decades (Myers 1980, FAO 1993). However, the greatest cumulative loss in forest area has occurred in temperate rather than tropical forests (Repetto 1988). Approximately 250 Mha of forested land in temperate and boreal biomes were converted to agriculture between 1850 and 1990 (Woodwell 2001).

1.3 Deforestation in Western Canada

Peak agriculture-related deforestation in eastern North America occurred a century ago. In this and many other temperate regions, rural depopulation, farmland abandonment and expansion of tree plantations have led to increases in the extent of forest cover (Williams 1989b, Rudel 1998, FAO 2001). Land-use change in western Canada runs counter to this pattern. Agriculture is a more recent phenomenon in the prairie provinces (Ramankutty and Foley 1999), and conversion of boreal forests to annual crops and perennial grasses may not yet have subsided.

Deforestation in Saskatchewan and adjacent provinces evaded public concern until the very recent past (e.g. Brady 2000, Global Forest Watch Canada 2000) for a variety of reasons. Short, small diameter boreal trees in western Canada do not have the aesthetic appeal of taller, more majestic trees from temperate and tropical environments. Boreal forests have lower productivity and biomass than other Canadian forest types and thus lower commercial value. Boreal regions are low in species richness for vascular plants, mammals, butterflies and other taxa relative to adjacent temperate zones (Ricketts et al. 1999). In addition, boreal regions within the continental interior are entirely devoid of endemic species for major terrestrial and aquatic taxa with the lone exception of the whooping crane (Ricketts et al. 1999, Abell et al. 2000).

Boreal forests represent a very small fraction of global species richness. In terms of global biogeochemistry, however, boreal forests play a far more substantial role. Considered together, boreal vegetation and soils constitute one of the largest terrestrial carbon pools within the biosphere. Boreal ecosystems (vegetation plus top 1 m of soils) contain 559 Pg C of the global terrestrial organic C pool of 2477 Pg C (Bolin and Sukamir 2000). The Canadian boreal forest contains approximately 200 Pg C (Apps et al. 1993). Transfers of carbon from boreal forests to the atmosphere contribute to changes in atmospheric composition and climate.

Within western Canada, few data are available on deforestation or its environmental effects. Despite the fact that Canada has 10 percent of the world's forests, the United Nations reported that Canada was unable to provide official data on the change in forest and woodland area over recent decades (FAO 1995, ECE and FAO

2000). Contemporary deforestation and its ecological effects are not documented in recent state of the environment and state of the forest reports (e.g. SERM 1995b, NRCan 2000).

1.4 Investigating Deforestation in Central Saskatchewan

Investigations at broad national or provincial scales would require the commitment of substantial financial, technical and human resources. With fewer resources, however, it was feasible to mount an investigation of deforestation at a finer spatial scale. Landscape-scale and site-scale studies such as those described herein, provide one of few current scientific insights into problems related to deforestation within Canada.

1.4.1 Purposes

Recent deforestation within southern boreal regions of Western Canada has served as the impetus for this research effort. My thesis research, which focussed on landscapes and sites in central Saskatchewan, had four main purposes:

(i) to quantify extents and rates of deforestation and to examine differences in wooded area changes among various land uses,

(ii) to quantify changes in landscape spatial structure associated with deforestation,

(iii) to quantify the spatial distribution of organic carbon within forest sites and deforested sites, and

(iv) to estimate differences in organic carbon densities between forest sites and deforested sites.

Landscape-scale research related to purposes (i) and (ii) relied on descriptive analyses of secondary data, where hypotheses are constructs representing expected findings. For site-scale investigations related to purposes (iii) and (iv) primary data were collected for statistical evaluation of hypotheses tested using comparative mensurative experiments (Hurlbert 1984).

1.4.2 Extents and Rates of Deforestation

Estimates of the historic extent and rate of deforestation are necessary to evaluate the management importance of land-use change in southern boreal regions of western Canada. Numerous spatially explicit data sources on recent and historic land cover could have been analyzed to address this data gap. Although satellites can provide relatively recent data on land cover at appropriate scales, this technology was not in place prior to the 1970s. In addition, multi-decade analyses of land cover change are complicated by the alterations in spectral bandwidths and spatial resolution that occur with satellite replacement over time. Forest inventory data are available for longer periods than satellite data, but are fragmented by jurisdiction and exhibit varying classification schemes, spatial scales and temporal frequencies.

The data I chose to analyze the extent and rates of boreal deforestation were a sequence of maps produced through Canada's National Topographic System. These maps provide no information on forest composition, but they have several other advantages. First, these maps capture land cover synchronously over multi-jurisdictional landscapes that include private lands as well as federal and provincial Crown lands. Second, the maps are repeated over a multi-decade time frame using a uniform methodology. Third, map scale at 1:50 000 is consistent across jurisdictions and over time.

In Chapter 2, I provide a quantitative analysis of this topographic map data and report the extents and rates of change for wooded area over three recent decades across two 0.45 Mha landscapes in central Saskatchewan. Human population and road network changes were also quantified for the purpose of comparing the interactions among population, road length and forest area in this boreal region to that reported elsewhere. The research approach was to investigate temporal changes in wooded area and road length for these two landscapes of fixed spatial location and boundaries. Spatial comparisons among protected area, commercial forest and agricultural land within each landscape were completed to evaluate the influence of land use on deforestation.

1.4.3 Effects of Deforestation and Reforestation on Landscape Spatial Structure

In addition to quantifying the areal extent and rates of deforestation for boreal landscapes, there is a need to examine the ecological effects of deforestation. The transformation of natural landscapes to cultivated landscapes through the process of deforestation often results in changes to landscape spatial structure (Forman and Godron 1986, Forman 1995). Many of the changes to landscape structure are ecologically deleterious. For example, patches of native vegetation decrease in size, increase in number, decrease in connectivity, increase in edge length and exhibit altered microclimatic conditions (Forman and Godron 1986, Forman 1995, Chen et al. 1999). These changes can reduce biodiversity of native species, particularly fauna that are difficult to conserve such as habitat interior specialists, and lead to invasions of exotic species (Forman and Godron 1986, Forman 1995, Barnes et al. 1998). Changes in landscape spatial structure also alter ecosystem processes such as dispersal, nutrient cycling and thermodynamics (Forman and Godron 1986, Forman 1995).

Increasing the area of boreal forests has the potential to provide numerous benefits because forests serve many important ecological functions. Some of these benefits, increases in the breeding habitat for forest-interior songbirds for example, will depend not just on the magnitude of area reforestated but also on the spatial configuration of reforested areas. Evaluation tools are required to determine whether a specific approach to reforestation will reverse the deforestation-induced changes to landscape structure.

Monitoring of landscape spatial structure using patch metrics and fragmentation indices to detect changes in forest conditions has been suggested by Baskent and Jordon (1995), Haines-Young and Chopping (1996) and Noss (1999). The spatial effects on forest landscape structure of deforestation have not been investigated for boreal regions. A quantitative analysis of the observed effects of deforestation on landscape spatial structure in boreal landscapes will provide baseline data against which the potential effects of reforestation can be compared.

The utility of landscape metrics for quantifying landscape spatial structure was evaluated for a subset of the study areas used in Chapter 2. In Chapter 3, I use landscape metrics to quantify the effects of boreal deforestation on landscape spatial structure. I examine temporal changes for landscape metrics for each landscape, and variation in landscape metrics across a spatial gradient in the proportion of land area wooded for two time periods.

In the final portion of Chapter 3, I briefly move beyond the retrospective analyses that are central to the overall thesis research effort. I describe a simulation of two different strategies for potential future reforestation. The first approach was the addition of small reforestation patches dispersed throughout agricultural portions of each landscape, as might occur with an economic incentive program to encourage voluntary reforestation by landowners. The second approach was the addition of a single large patch contiguous to protected forests, as might occur with a planned reforestation program with land acquisition by governments or private conservation organizations. The effects of both approaches on forest landscape spatial structure were evaluated to determine which best mirrors the historic baseline data for deforestation. Ecological restoration of forest landscapes will require that changes in landscape metrics observed during deforestation be reversed during reforestation.

1.4.4 Effects of Deforestation on Organic Carbon at Sites

In Canada, deforestation accounts for an estimated 1.7 to 2.5 % of Canada's total CO_2 emissions. This proportion is small in global terms but still significant relative to Canada's emission reduction target (Robinson et al. 1999).

Organic carbon stocks or densities in extant forest and nearby agricultural fields have been compared for non-boreal areas of North America such as southern Ontario (Ellert and Gregorich 1996) and Minnesota (Johnston et al. 1996, Homann and Grigal 1996). Only limited field data based on relatively few sites are available for boreal regions of Saskatchewan (Ellert and Bettany 1995). This latter study did not include estimation of organic carbon densities in vegetation.

Changes in organic carbon density are easier to quantify at a site scale (several hectares) rather than a landscape scale (thousands of hectares). Landscape-scale data such as air photos, maps or satellite images are not sufficient to estimate ecosystem carbon stocks because of the difficulty of remotely sensing the magnitude of soil organic carbon.

Results from field studies of vegetation and soil carbon at forest and agricultural sites in central Saskatchewan are reported in Chapters 4 to 6. The purpose of these investigations was to estimate the magnitude of organic carbon losses associated with deforestation. Although cultivation is the dominant agricultural land-use in this region, I included pastures to determine whether carbon densities differed between grazing and annual cropping systems. In order to make the comparisons among land uses more valid, I restricted site selection to a portion of one of the landscapes examined in Chapter 2. Sampling was limited to sites of close geographic proximity within the southernmost portion of Prince Albert National Park or the northernmost portion of the Rural Municipality of Shellbrook. In addition, study site selection was restricted to soil map units classed as dominantly Dark Gray Chernozems and/or Dark Gray Luvisols on hummocky glacial till landforms. These were among the most common soil taxa and landforms that occurred across the study area.

In 2000, I intensively sampled one forest, one pasture and one cultivated site to assess the horizontal and vertical distribution of soil organic carbon within each site type. The first use of this data was to compare the relative carbon content of aboveground and belowground ecosystem components at each 2.25 ha site. In Chapter 4, I report the carbon densities for living and dead aboveground vegetation in several size strata and soil organic carbon for several depth increments at a forest site, a pasture site and a cultivated site. I also estimate carbon densities for unsampled ecosystem components and make recommendations for efficient allocation of future sampling effort.

The data from sampling during 2000 were also used to evaluate whether topographic landform position aids in the description of the spatial pattern for soil organic carbon storage within study sites. An understanding of the relationships between topography on carbon density was required to select future sampling methods. In Chapter 5, I report the distribution of organic carbon at shoulders, levels and footslopes at the three sites. The research approach was to compare sites at different spatial locations with the hope that any land-use effects on the relationship between landform position and soil properties would be exhibited as differences among the forest site, pasture site and cultivated site.

Direct estimation of the magnitude of organic carbon densities in forests, pastures and cultivated ecosystems required replication for each land use type. The results from intensive sampling at single sites aided the development of strategies for more extensive sampling at multiple sites in 2001. Fifteen additional 0.4 ha sites were sampled to estimate organic carbon densities for vegetation and soil strata. In Chapter 6, I report ecosystem carbon densities for the six forest sites, six pasture sites and six cultivated sites sampled over both the intensive and extensive sampling phases. Antecedent data for carbon densities on each sampling site were not available, which precluded direct examination of temporal effects of land-use change at specific spatial locations. The research approach for Chapter 6 used space as an analogue for time by comparing present conditions at the locations of agricultural sites to present conditions at forests, assuming that historic conditions at the former would have been similar to present conditions at the latter. The hope was that temporal land-use effects on organic carbon densities would be discernable as spatial differences in carbon densities among forest, pasture and cultivated characteristic treatments.

1.4.5 Research Synthesis

In Chapter 7, the research results reported in the previous five chapters are synthesized. I estimate the order of magnitude of vegetation carbon losses due to historic deforestation and potential reforestation in central Saskatchewan landscapes, recommend future research to better evaluate the effects of deforestation and reforestation on soil organic carbon, and propose management actions aimed at sequestering carbon and perpetuating forest biodiversity.

2. EXTENTS AND RATES OF DEFORESTATION IN CENTRAL SASKATCHEWAN LANDSCAPES

2.1 Literature Review

Agriculture is a more recent phenomenon in western Canada than in many other temperate regions. In France, Denmark, and New England, for example, agricultural land area peaked in the 1800s and forest area increased over the past century due to farmland abandonment and reforestation (Williams 1989b, Mather 1990). In much of western Canada, agricultural settlement was barely underway by 1900, and development of croplands and pastures persisted for the next 90 years (Ramankutty and Foley 1999). Deforestation, primarily accomplished by clearing and burning of woody vegetation, occurred with agricultural settlement of southern boreal regions.

Deforestation in western Canada has received little scientific scrutiny. No national long-term monitoring system exists to estimate changes in the total area of Canadian forests (Robinson et al. 1999). Provincial governments have jurisdiction over most forestlands in western Canada, but have not undertaken systematic efforts to study deforestation. A brief reference to the loss of 0.6 Mha of forest within the Mixedwood section between 1953 and 1975 (Kabzems et al. 1986) is one of few official estimates of agriculture-related deforestation for Saskatchewan.

Given the lack of government data, there is a need to determine the areal extent of changes in forest cover within representative landscapes in Canada. Although data on agricultural land are sometimes used to estimate changes in forest cover, a better approach is to use land cover data (Robinson et al. 1999).

Several factors that influence deforestation and reforestation have been described at the global scale. In developing countries, human population density or growth rates are determining factors for the rate of deforestation (Allen and Barnes 1985, FAO 1995, Pahari and Murai 1999). Road construction in forests leads to planned and spontaneous migrations and expansion of agriculture (Williams 1989a). Legislation aimed at forest protection may inhibit deforestation in some jurisdictions. In developed countries, a transition from deforestation to reforestation generally occurs with industrialization as the population shifts from rural to urban areas (Mather 1990, Rudel 1998). The influences of these factors on deforestation in western Canada have not been evaluated.

2.2 Objectives, Research Design and Hypotheses

The objectives for Chapter 2 are (i) to quantify changes in wooded area (including forests, woodlands, and shrublands), road length and population over three recent decades for two landscapes in central Saskatchewan, and (ii) to quantify differences in wooded area and road length among land use classes within each of these landscapes. The research design employed included both temporal comparisons for each landscape and spatial comparisons among land use classes within each landscape.

Hypotheses pertinent to the first objective were that total wooded area would decrease, total road length would increase, and total population would decrease, monotonically within both landscapes. Hypotheses relative to the second objective were that the absolute magnitude of expected negative rates of change for wooded area and expected positive rates of change for road length would be inversely related to the degree of legal protection for forests within each land use class. More specifically, the rates of change for wooded area would be closest to zero within protected area, intermediate within commercial forest, and furthest below zero within agricultural land, and the rates of change for road length would be closest to zero within protected area, intermediate in commercial forest and furthest above zero within agricultural land.

2.3 Study Areas Descriptions

Two study areas or landscapes were selected in central Saskatchewan, Canada (Figure 2.1). Each landscape consists of five contiguous 1:50,000 map sheets from Canada's National Topographic System (NTS). Each map sheets spans 30' longitude and 15' latitude. Map sheets were selected based on three criteria: each map had to cover two or more legal jurisdictions, adjacent maps of the same edition had to be derived from air photos taken during the same or an immediately following year, and each map had to be wholly contained within the Boreal Plain Ecozone (Padbury and Acton 1994).

The 4570 km^2 Waskesiu Hills landscape (NTS map sheets 73 G/9, G/10, G/15, G/16 and H/12) is centered at approximately 53°45' N and 106°15' W. Physiography is dominated by hummocky and moderately rolling glacial till plains. Gray Luvisolic and Dark Gray Chernozemic soils are dominant (Acton et al. 1998). Water bodies account for approximately 8 percent of the surface area.

The Waskesiu Hills landscape encompasses portions of Prince Albert National Park, administered by the Government of Canada, and the Northern Provincial Forest, administered by the Government of Saskatchewan. Outside of these two large jurisdictions there is an aggregation of smaller holdings of cultivated, grazed and natural lands under a mix of private and public ownership (e.g. private farms, Crown agricultural land, community pastures, Montreal Lake Indian Reserve). The landscape was classified into three land use classes: protected area (Prince Albert National Park), (ii) commercial forest (Northern Provincial Forest), and (iii) agricultural land (all remaining lands).

The 4692 km^2 Red Deer River landscape (NTS map sheets 63 D/11, D/12, D/14, D/15 and D/16) is centered at approximately 52°45'N and 103°00'W. Physiography is dominated by undulating glacial till and glaciolacustrine plains. Dark Gray Chernozemic and Gray Luvisolic soils are dominant (Acton et al. 1998). Water bodies account for approximately 3 percent of the surface area.

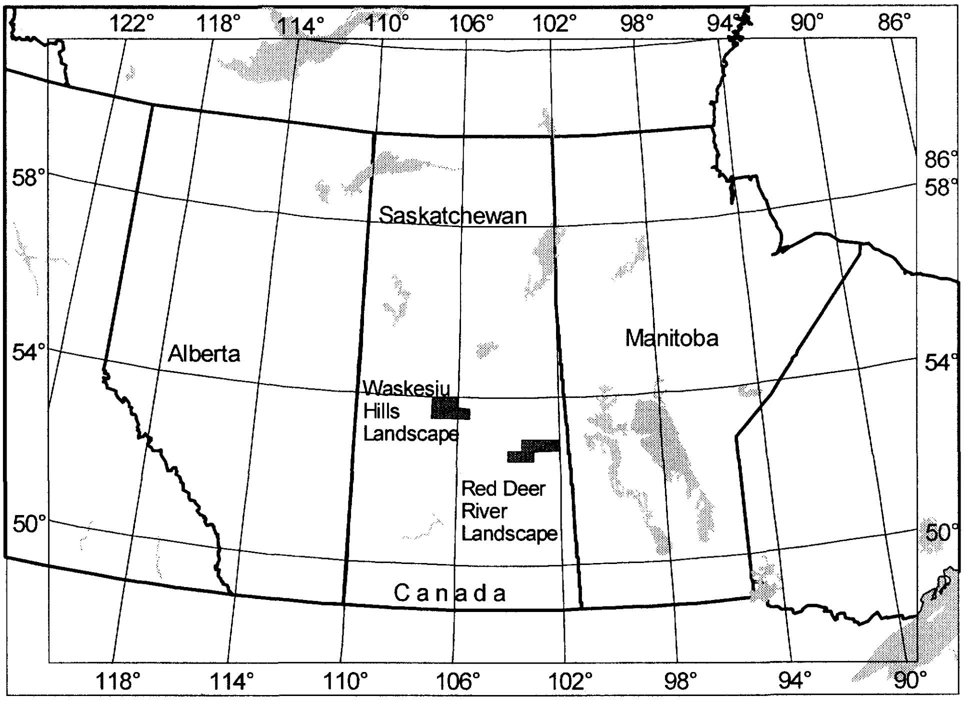


Figure 2.1 Location of the study landscapes in Saskatchewan.

The Red Deer River landscape includes portions of the Northern and Porcupine Provincial Forests, administered by the Government of Saskatchewan. Outside of these areas there is a mix of smaller holdings similar to that described for the Waskesiu Hills landscape, except that First Nation Reserves are not present. This landscape was also classified into the three land use classes: protected area (Greenwater Provincial Park), commercial forest (Provincial Forests excluding Greenwater Provincial Park), and agricultural land (all remaining lands).

Both landscapes have sub-humid cool continental climates with mean annual temperatures of approximately 0°C and mean annual precipitation of 400 to 500 mm (Kabzems et al. 1986, Acton et al. 1998).

Boreal forest was the dominant land cover for both landscapes a century ago (DNR 1955, Rowe 1959, Weir et al. 2000), but major portions were cleared for agriculture over the past century (Kabzems et al. 1986, Weir and Johnson 1998). Dominant tree species in the Boreal Plain Ecozone include trembling aspen (*Populus tremuloides* Michx.), white spruce (*Picea glauca* (Moench) Voss), black spruce (*Picea mariana* (Mill.) B.S.P.), and jack pine (*Pinus banksiana* Lamb), with lesser amounts of white birch (*Betula papyrifera* Marsh.), balsam poplar (*Populus balsamifera* L.), balsam fir (*Abies balsamea* (L.) Mill), and tamarack (*Larix laricina* (Du Roi) Koch) (Acton et al. 1998). Dominant tall shrub genera include willows (*Salix*) and alders (*Alnus*).

In both landscapes, the protected areas provide the greatest degree of legal protection for native vegetation. In protected areas and commercial forests, the clearing of wooded land for agriculture is prohibited by legislation. On the remaining portions of these landscapes, forests are afforded no legal protection under federal or provincial law.

2.4 Materials and Methods

NTS maps were used as the source of land cover data. The National Topographic System is consistent across all legal jurisdictions and is periodically updated over time. Green areas on these maps are wooded, defined as land at least 35 % covered by trees or shrubs with a minimum height of 2 m (NRCan 1996).

Measuring deforestation requires identification of both land cover change and land use change, but it is easier to track the former than the latter. Land cover changes appear clearly on the air photos from which topographic maps are created. Changes in wooded area reflect primarily changes in land cover. Fortunately, the NTS maps do not reflect ephemeral forest harvesting and regeneration activities that are not included in the definitions of deforestation or reforestation. However, the maps do depict other changes to wooded area not relevant to deforestation or reforestation. An example is the movement of forest-graminoid ecotones around wetlands or grasslands. Thus, change in wooded area serves as a proxy measure for deforestation (decrease in wooded area) and reforestation (increase in wooded area).

Three editions of the NTS maps have been published. Waskesiu Hills wooded area is estimated for 1963, 1976 and 1990 and Red Deer River wooded area is estimated for 1957, 1975 and 1990. Wooded area data were assumed to represent the year air photos were taken, rather than the year the maps were published. Comparing sequential maps allows estimation of changes in total wooded area but does not provide any information on changes in species composition, age structure, canopy height or other vegetation attributes.

Area data from the topographic maps were purchased if commercially available or digitized manually. All area features on the maps were classified into three nominal categories: wooded area (green areas on the maps such as forests, shrublands and treed muskeg), open area (white areas on the maps such as cultivated land, grasslands, graminoid marshes and gravel pits) and surface water (blue areas on the maps including lakes, ponds, intermittent water bodies, flooded land and those rivers represented as area features).

Only entities depicted on the NTS maps as area features were used to quantify wooded area. Estimates of total wooded area do not account for entities within wooded areas depicted as lines because they are not represented on the maps in an area-proportional manner. Wooded patches contain both mapped linear entities (e.g. roads, railways, streams and transmission lines) and unmapped linear entities (e.g. survey lines, small power lines, and narrow linear disturbances not associated with any particular infrastructure) that are too narrow to be accurately represented on the maps.

Similarly, estimates of wooded area do not account for entities within wooded patches represented as points. Although buildings, dugouts, campgrounds and other features occupy area on the ground, their individual areas are too small to be accurately represented on the maps. Using area features as mapped, while disregarding line and point overlays, was selected as the most reproducible approach to estimate wooded area from these maps.

It was feasible to measure the lengths of roads because of their Euclidean geometry. Road and railway data were purchased or manually digitized depending on availability. Total road lengths reported include all roads, except those mapped as residential streets, cart tracks or trails, plus railways. When a road or railway segment formed the boundary of two jurisdictions, half of its length was assigned to each land use class.

Digital wooded area and road data for each of the map editions were compiled into a common geographic information system. Areas and lengths were calculated using ArcView® software (ESRI 1996).

A correspondence check (e.g., Miller et al. (1998)) was performed to assess the agreement for wooded area, open and water classes between the analog/digital map data and the original air photographs. For two of the five map sheets in each landscape, 50 random points were selected on the edition 1 maps, avoiding locations within 50 m of the boundary between any two area features. At each random location, the 1963 air photo (Waskesiu Hills landscape) or the 1957 air photo (Red Deer River landscape) was inspected, and the location was classed as wooded area, open area or surface water. The class of each point was read off the analog and digital maps and compared to the air photo class. This process was repeated for the edition 3 maps and the 1990 air photos. Correspondence tables were produced comparing the air photo and map data.

Since the topographic maps do not contain population data, census reports for 1956, 1961, 1976 and 1991 were used. Census tracts correspond to the boundaries of municipalities rather than the boundaries of the topographic maps. Population estimates for each landscape include the entire population of all municipalities that overlap that landscape in whole or in part.

Mean annual rates of change (r) in % yr^{-1} were calculated for wooded area, road length and population using a formula adapted from Dirzo and Garcia (1992):

$$r = -[1 - (f/i)^{1/n}] \cdot 100 \quad (2.1)$$

where: f is the final wooded area, road length or population,

i is the initial wooded area, road length or population, and

n is the number of years between the initial and final estimates.

Positive values of r indicate expansion of wooded area, road network length or population, and negative values of r indicate contraction of wooded area, road network length or population.

2.5 Results

2.5.1 Wooded Area, Road Length and Population

In the Waskesiu Hills landscape, the total area mapped as wooded declined during both the 1963 to 1976 period and the 1976 to 1990 period (Table 2.1). The total loss of wooded area was 164 km^2. The annual rate of change for wooded area was -0.19 % yr^{-1} over the entire 27-year period (-0.24 % yr^{-1} from 1963 to 1976, and -0.15 % yr^{-1} from 1976 to 1990).

Table 2.1: Total wooded area and road length for the Waskesiu Hills landscape in 1963, 1976, and 1990, by land use class.

	Wooded Area (km^2)			Road Length (km)		
	1963	1976	1990	1963	1976	1990
Agricultural Land	781	620	553	893	1032	997
Commercial Forest	543	552	558	99	125	161
Protected Area	1924	1977	1973	276	244	206
Totals	3248	3149	3084	1269	1401	1364

In the Waskesiu Hills landscape, road length increased by 132 km from 1963 to 1976 and then decreased by 37 km from 1976 to 1990 (Table 2.1). The mean annual rate of change for road length was +0.27 % yr^{-1} for the 27-year period (+0.77 % yr^{-1} between 1957 and 1975, and -0.19 % yr^{-1} between 1975 and 1990).

In the Red Deer River landscape, the total area mapped as wooded declined during both the 1957 to 1975 period and the 1975 to 1990 period (Table 2.2). The total loss of wooded area was 371 km^2. The mean annual rate of change for wooded area over the 33-year period was -0.43 % yr^{-1} (-0.29 % yr^{-1} between 1957 and 1975, and -0.61 % yr^{-1} between 1975 and 1990).

Table 2.2: Total wooded area and road length for the Red Deer River landscape in 1957, 1975, and 1990, by land use class.

	Wooded Area (km^2)			Road Length (km)		
	1957	1975	1990	1957	1975	1990
Agricultural Land	1388	1154	929	1707	1996	2159
Commercial Forest	1226	1321	1316	108	133	159
Protected Area	154	155	153	21	25	25
Totals	2769	2630	2398	1836	2154	2343

In the Red Deer River landscape, road length increased by 318 km from 1957 to 1975 and by 189 km from 1975 to 1990 (Table 2.2). Roads segments that actually dissected wooded areas patches decreased in total length from 339 km in 1957 to 326 km in 1990. The mean annual rate of change for road length was +0.74 % yr^{-1} for the Red Deer River landscape over the 33-year period (+0.89 % yr^{-1} between 1957 and 1975, and +0.56 % yr^{-1} between 1975 and 1990).

The accuracy assessment for the wooded areas delineated on topographic maps showed 100% correspondence between analog and digital maps, and 99% correspondence between the air photos and the analog/digital maps (Table 2.3). Two of the four cases of non-correspondence between reference photos and map data were for points classified as open land on the photos, but mapped as water. In both cases, the points were located in depressions that could have been seasonally flooded. These types of errors do not affect the accuracy of wooded area estimates. The other two cases of non-correspondence occurred where narrow linear disturbances visible on the air photo were not represented on the maps. The two points that occurred on these linear features were classed as open on the photo, but were located within areas shown as wooded on the maps.

Table 2.3: Correspondence check for nominally classified random points from two editions of NTS maps within the Waskesiu Hills and Red Deer River landscapes.

	Analog/Digital Map Classification		
Air Photo Classification	Wooded Area (number of points)	Open Area (number of points)	Surface Water (number of points)
1963 Waskesiu Hills (73 G/10 and G/15)			
Wooded Area	66	0	0
Open Area	2	26	0
Surface Water	0	0	6
1990 Waskesiu Hills (73 G/10 and G/15)			
Wooded Area	57	0	0
Open Area	0	36	1
Surface Water	0	0	6
1957 Red Deer River (63 D/11 and 63 D/14)			
Wooded Area	60	0	0
Open Area	0	38	0
Surface Water	0	0	2
1990 Red Deer River (63 D/11 and 63 D/14)			
Wooded Area	35	0	0
Open Area	0	60	1
Surface Water	0	0	4

Rural municipalities, villages and the national park which occur wholly or partially within the Waskesiu Hills landscape decreased in estimated total population throughout the time period (Table 2.4). The mean annual rate of change for the total population was -1.40 % yr^{-1} between 1961 and 1976, -0. 41 % yr^{-1} between 1976 and 1991, and -0.89 % yr^{-1} over the entire period.

Rural municipalities, towns and villages occurring wholly or partially within the Red Deer River landscape, also decreased in estimated total population (Table 2.5). The average annual rate of change for the total population was -0.95 % yr^{-1} between 1956 and 1976, -1.54 % yr^{-1} between 1976 and 1991, and -1.19 % yr^{-1} over the entire period.

Table 2.4: Census population estimates for municipalities occurring within the Waskesiu Hills landscape, 1961, 1976 and 1990.

	1961* (number of people)	1976† (number of people)	1990‡ (number of people)
RM§ of Big River	828	939	848
RM of Canwood and Village of Debden	3,729	2,653	2,227
RM of Shellbrook	2,322	1,805	1,834
RM of Paddockwood, RM of Lakeland, and Village of Christopher Lake	2,046	1,948	2,023
Prince Albert National Park	109	177	170
Totals	9,034	7,522	7,102

* DBS (1963).
† Statistics Canada (1977).
‡ Statistics Canada (1992a).
§ RM = Rural Municipality.

Table 2.5: Census population estimates for municipalities occurring within the Red Deer River landscape, 1956, 1976 and 1990.

	1956* (number of people)	1976† (number of people)	1990‡ (number of people)
RM§ of Barrier Valley	1,376	831	702
RM of Bjorkdale, Villages of Bjorkdale and Mistatim	2,772	1,897	1,618
RM of Porcupine, Villages of Porcupine Plain and Carraganna	3,733	2,979	2,171
RM of Hudson Bay and Town of Hudson Bay	3,513	4,540	3,750
RM of Tisdale	2,336	1,342	1,163
RM of Arborfield	1,250	790	548
Totals	13,962	12,379	9,959

* DBS (1963).
† Statistics Canada (1977).
‡ Statistics Canada (1992a).
§ RM = Rural Municipality

Within both landscapes, wooded area declined monotonically as hypothesized. Road length increased during both time periods in the Red Deer River landscape as hypothesized. Within the Waskesiu Hills landscape, however, road length increased prior to 1976 and decreased after 1976. Population decreased monotonically within both landscapes as hypothesized.

2.5.2 Differences Among Land Use Classes

Within the Waskesiu Hills landscape between 1963 and 1990, wooded area in the protected area and commercial forest increased by 49 and 15 km^2, respectively (Table 2.1). Over the same period, 224 km^2 of wooded area were lost on agricultural land. The mean annual rate of change in wooded area on agricultural land was -1.27 % yr^{-1} over the 27-year period. Rates of change in the other land use classes were much smaller (+0.10 % yr^{-1} over the entire period for both the protected area and the commercial forest). Rates of change varied between the two time periods (Figure 2.2).

Between 1963 and 1990, road length increases of 104 km on agricultural land and 62 km in commercial forest were partially offset by a decrease in road length of 70 km in the protected area (Table 2.1). Mean annual rates of change for road length over the 27-year period were +1.82 % yr^{-1} in the commercial forest, +0.41 % yr^{-1} in agricultural areas and -1.08 % yr^{-1} in the protected area. Rates of change were similar for the two time periods for the commercial forest and protected area, but reversed from positive to negative on agricultural land (Figure 2.2).

Wooded lands within the protected area and commercial forest of the Red Deer River landscape were relatively stable or increased between 1957, 1976 and 1990 (Table 2.2). However 459 km^2 of wooded area were lost on agricultural land over the entire period. The mean annual rate of change in wooded area on agricultural land was negative and much greater in magnitude (-1.21 % yr^{-1} between 1957 and 1990) than the rates for the protected area and commercial forest (-0.02 % yr^{-1} and +0.22 % yr^{-1}, respectively).

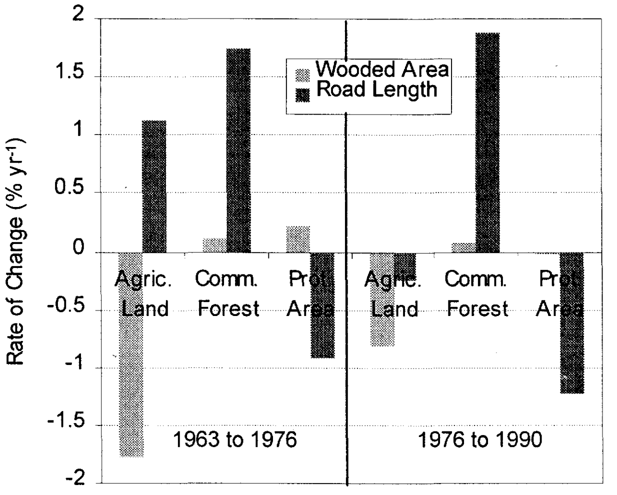


Figure 2.2: Rates of change for wooded area and road length in the Waskesiu Hills landscape, by land use class and time period.

There were differences between the first and second time periods. The greatest rate of increase in wooded area in the commercial forest occurred between 1957 and 1975, and the greatest rate of decline in wooded area in agricultural land occurred between 1975 and 1990 (Figure 2.3).

Road length in both agricultural land and commercial forest within the Red Deer River landscape followed the overall pattern of steady road expansion. In the protected area, road length increased between 1957 and 1975 but then remained stable to 1990 (Table 2.2). Over the 33-year period, the highest mean annual rate of change for road length occurred in the commercial forest (+1.18 % yr^{-1}) and the lowest rate occurred in the protected area (+0.58 % yr^{-1}). The rate for agricultural land was +0.71 % yr^{-1}. Mean annual rates of change diminished between the 1957 to 1975 period and 1975 to 1990 period for agricultural land and the protected area, but not for commercial forest (Figure 2.3).

Negative rates of change for wooded area were greatest in magnitude for agricultural land within both landscapes. As hypothesized, the largest rates of loss occurred in portions of the landscape in which forests were afforded no legal protection. Expected negative rates of change for wooded area within commercial forests and protected areas were not consistently observed. For both landscapes, the largest positive rates of change for road length occurred within commercial forests rather than within agricultural land as hypothesized. Road length within protected area decreased within the Waskesiu Hills landscape (Prince Albert National Park), but increased within the Red Deer River landscape (Greenwater Provincial Park).

2.6 Discussion

2.6.1 Wooded Area, Road Length and Population

Most analyses suggest that forest area in boreal and temperate biomes are stable or increasing in area (Williams 1989a, Mather and Sdasyuk 1991, FAO 1995, Potter 1999, ECE and FAO 2000, FAO 2001). While this may be correct at the global scale,

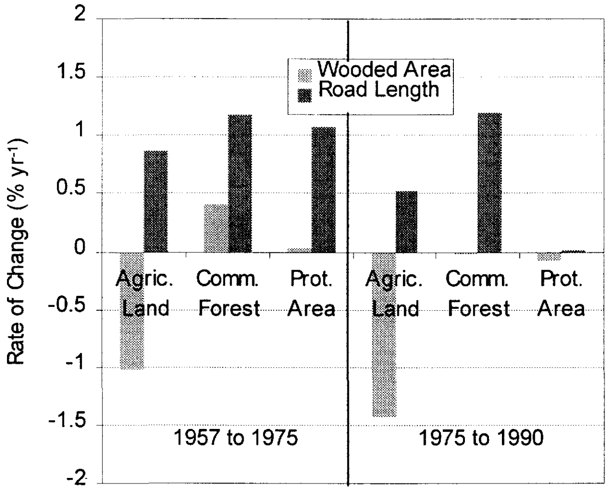


Figure 2.3: Rates of change for wooded area and road length in the Red Deer River landscape, by land use class and time period.

there are regions within these biomes where boreal forests have exhibited deforestation over recent decades.

The quantitative results for these two boreal landscapes can be compared to numerous studies that document recent rates of change for forest and woodland area in temperate landscapes. The loss rates for wooded area for central Saskatchewan landscapes prior to 1975/76 (-0.24 and -0.29 % yr^{-1}) were much lower than the -2.1 to -4.9 % yr^{-1} loss rates for forests calculated from map data for periods ending 1967 to 1975 for Louisiana, Arkansas, and Mississippi (Sternitzke 1976, Kress et al. 1996). The mean annual rates of change for the most recent time periods examined for the Waskesiu Hills landscape (-0.15 % yr^{-1}) is within the range of rates (-0.11 % yr^{-1} to -0.19 % yr^{-1}) calculated from forest cover data reported by Zipperer (1993) for three agricultural/urban landscapes in Maryland. The most recent mean annual rates of change for the Red Deer River landscape (-0.61 % yr^{-1}) was closer to the rates reported by Zheng et al. (1997) for a multi-jurisdictional landscape in China/North Korea (-0.73 % yr^{-1} between 1972 and 1988).

The deforestation observed in the two Saskatchewan landscapes contrasted the expansion of forests reported over similar periods for agricultural landscapes in Ohio (Medley et al. 1995), New York (Smith et al. 1993) and Quebec (Pan et al. 1999), and for a forestry/agricultural landscapes in southern Finland (Löfman and Kouki 2001).

For the first half of the twentieth century, deforestation in the Waskesiu Hills and Red Deer River landscapes occurred concomitantly with an influx of settlers and the development of transportation infrastructure, a pattern similar to that reported for non-boreal regions (Allen and Barnes 1985, Williams 1989a). Over recent decades, however, wooded area decreases occurred simultaneously with population declines.

2.6.2 Differences Among Land Use Classes

Analysis of topographic map data for multi-jurisdictional boreal landscapes in central Saskatchewan showed that wooded area was relatively stable within commercial forests and parks, but declined in agriculture zones where forests were afforded no legal protection.

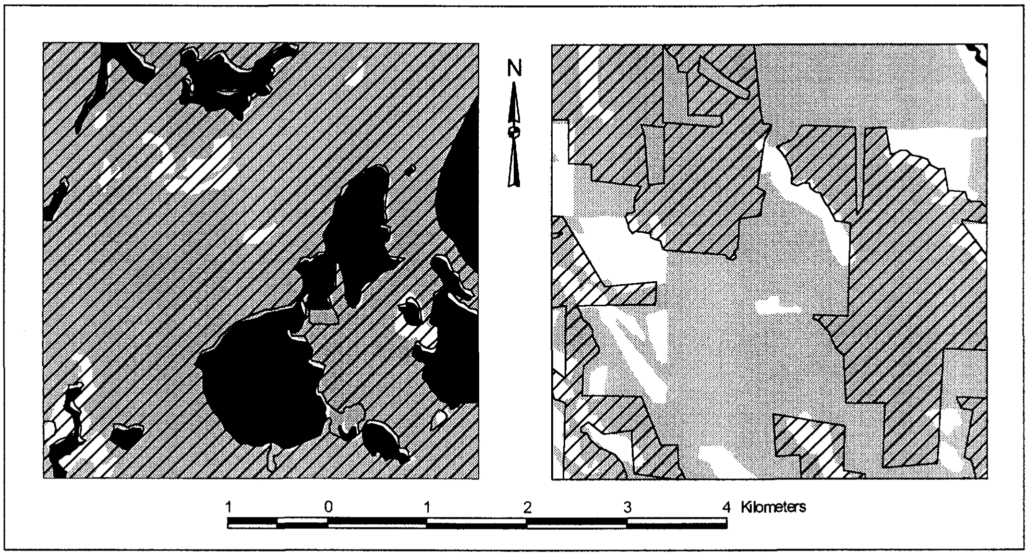
A qualitative difference in the spatial pattern of expansions and contractions of wooded area was observed. In the commercial forest, areas wooded in 1990 but open in 1963 were irregularly shaped and appeared to be associated with wetlands (Figure 2.4). This observed increase in wooded area was likely caused by encroachment of woody vegetation due to succession or decreasing water levels. These increases in wooded area do not represent reforestation.

Areas wooded in 1963 but open in 1990 within agricultural land often exhibited linear boundaries (Figure 2.4). It is likely that land-use conversion caused the straight perimeters for these areas. The negative rates of change observed for wooded area outside of protected forests represents deforestation because land cover changes were, for the most, part accompanied by the introduction of a new land use (i.e., agriculture).

For land use classes within the Waskesiu Hills and Red Deer River landscapes there was no relationship between the rates of change for wooded area and the rates of change for road length (Figures 2.2 and 2.3). In both protected areas, the rates of change for wooded area were close to zero even though road length increased in Greenwater Provincial Park and decreased in Prince Albert National Park. In commercial forests, where the greatest increases in road length were observed, there was no evidence that road expansion led to deforestation beyond the road allowances. Rates of change for wooded area were negative on agricultural land whether road length increased or decreased.

The highest rate of road network expansion occurred in commercial forests. The road grid utilized for agricultural development was almost completed in both landscapes at the start of the study period, whereas the all-weather logging roads used for commercial forestry were initiated and developed during the period under examination.

These two boreal landscapes with declining human populations exhibited rates of increase for road length that were less than 1 % yr^{-1}. In comparison, Theobald et al. (1996) reported rates of increase for road length ranging from 5 to 12 % yr^{-1} for a Colorado study area with an increasing human population.



1963 Open Area

1963 Wooded Area

1990 Wooded Area

Surface Water

Figure 2.4 Little Nesslin Lake (left) and Ladder Valley (right) are portions of the Waskesiu Hills landscape located within the commercial forest and agriculture land use classes respectively. Irregularly shaped white areas with cross-hatching (left) are primarily wetlands mapped as open in 1963 and wooded in 1990. Gray areas with linear boundaries and no cross-hatching (right) were mapped as wooded in 1963 but were cleared for crops or pasture prior to 1990.

Since few citizens are resident within protected areas or commercial forests in central Saskatchewan, the population declines occurred almost exclusively in the same agricultural portions of Waskesiu Hills and Red Deer River landscapes where the largest decreases in wooded area were observed. This indicates that population pressure is not a major determining factor in the deforestation process as observed at this temporal and spatial scale. Studies documenting changes over only a few decades might not be long enough to detect the transition toward reforestation that generally occurs following declines in rural population with urbanization and industrialization (Rudel 1998). The two processes may also have occurred independently in these landscapes. Deforestation on agricultural land may be triggered by economic factors unrelated to population size, such as agricultural subsidy programs and commodity prices. As the size of both farm machinery and average land holdings increases, agricultural land use can intensify during periods of population decline.

No changes in land use regulations have occurred to prevent future deforestation in agricultural portions of these boreal landscapes. Within the two study areas, extant wooded lands that are unprotected by legislation remain vulnerable to deforestation.

2.6.3 Deforestation within the Boreal Plain Ecozone

In the Boreal Plain Ecozone, where deforestation occurred as a result of agricultural expansion, analyzing NTS maps was a cost-effective way to estimate changes in forest area. The correspondence check showed that NTS maps represent wooded areas with a high degree of accuracy. Thus, changes reported for wooded area on the topographic maps can be taken to represent real changes on the landscape.

Inaccuracies identified with the methodology were related to linear disturbances not mapped as area features. The estimated losses in wooded area across the two landscapes might be conservative because the methodology does not account for losses due to roads, transmission lines, buildings and other features not represented on topographic maps in an area-proportional manner. However, this error is estimated to be very small. Open lands visible on air photos but mapped as wooded accounted for only 0.5 % of the random points used for the correspondence check. For the Red Deer River landscape, estimating a fixed 30 m wide right of way for all road segments that dissected wooded patches would result in adjustments of less than 0.5 % in the estimated total wooded area (-10.2 km^2 in 1957 and -9.8 km^2 in 1990). This minor adjustment of the initial and final areas would cause no alteration of the estimated rate of change for wooded area between the two dates.

The methodology, although effective for evaluating changes in wooded area prior to 1990, can be easily repeated only when new map editions are produced. More recent air photos, if available at an appropriate scale, could be used to update wooded area data, but this would be labour-intensive.

If landscapes could have been randomly selected from the Boreal Plain Ecozone additional sampling would have allowed inference of the regional rate of deforestation. However, NTS data does not lend itself to sampling over specific time periods because different map sheets within a region are updated in different years. New editions have not been released for over 25 years for portions of the Boreal Plain Ecozone in Saskatchewan.

Although no statistical inferences can be made from this data, the two landscapes can be considered replicates. The same basic changes (decreased wooded area primarily within agricultural land, increased road length primarily in commercial forests, and decreased population) were observed in both replicates. It is likely this pattern was repeated in other southern boreal regions within western Canada. In particular, wooded area losses are probably widespread within those portions of the Boreal Plain Ecozone where forests lack legal protection.

2.7 Chapter Summary

Deforestation was investigated through analysis of topographic map chronosequences for two multi-jurisdictional landscapes in central Saskatchewan. Estimated rates of deforestation within the Waskesiu Hills and Red Deer River landscapes for the three decades prior to 1990 ranged from 0.15 to 0.61 % yr^{-1}. Federal and provincial legislation establishing publicly owned parks and forests served to inhibit deforestation within legally protected portions of these landscapes. In Prince Albert

National Park, strict legal protection also served to prevent the proliferation of roads. In provincial forests, road networks expanded but deforestation beyond the road allowances was not observed. On agricultural land, where private holdings dominate and forests lack legal protection, deforestation occurred continuously throughout the periods examined even though human populations declined. Within the two study areas, wooded lands that are unprotected by legislation remain vulnerable to deforestation.

3. EFFECTS OF DEFORESTATION AND REFORESTATION ON SPATIAL STRUCTURE IN CENTRAL SASKATCHEWAN LANDSCAPES

3.1 Literature Review

From an ecological conservation perspective, the primary effect of deforestation is the direct reduction in area of native woody vegetation. Several important biological, chemical and physical attributes and processes of forest ecosystems were described in Chapter 1. Many of these structural elements or functions are lost or altered when tree cover is removed and non-forest land use is initiated. Two examples of phenomena associated with deforestation include the extirpation of forest-dependent species (Ledig 1992), and changes to thermodynamics (increased albedo, sensible heat flux and conductive heat flux, and decreased latent heat flux (Lewis 1998, Brovkin et al. 1999, Fuller 2001)).

The ecological effects of deforestation are not directly proportional to the areal extent of deforestation. A secondary effect of deforestation and other human land transformation processes is the fragmentation of remaining native vegetation into smaller patches (Forman 1995). For the purposes of this study, fragmentation is defined as the "breaking up of habitat or cover type into smaller, disconnected parcels" (Turner et al. 2001, p. 3).

Small natural vegetation patches have lower faunal species richness than large natural vegetation patches and include few rare, habitat-interior, upper trophic level or large-home-range species (Forman and Godron 1986, Forman 1995). Small forest patches with greater edge length are also more susceptible to forces originating in surrounding agroecosystems through processes such as exotic species invasion and advection (Saunders et al. 1991, Forman 1995, Fuller 2001).

The severity of the ecological effects caused by deforestation increases as forest fragmentation increases. Thus the ecological impact of deforestation is proportional not simply to the area of forest cover lost, but rather to the area of forest cover loss combined with the degree of fragmentation of forest remnants.

3.1.1 Landscape Metrics and Analysis of Landscape Change

Advances in geographic information system software over recent decades have facilitated increased quantitative analysis of landscape-scale data such as maps and satellite images (Turner et al. 2001). Landscape metrics serve as important tools used by landscape ecologists to study the spatial structure of landscapes. Examples of common landscape metrics include the number of patches, mean patch size and perimeter-to-area ratio (Table 3.1, Figure 3.1). If the three landscapes in Figure 3.1 were entirely forested at some previous time, each has since been 75% deforested. As the degree of fragmentation of the residual forest area increases from (a) to (c), number of patches increases, mean patch size decreases and perimeter-to-area ratio increases.

Changes for the number of patches, mean patch size, and perimeter-to-area ratio with deforestation may be counter-intuitive relative to the trends above. Processes common to deforestation such as patch perforation, patch shrinkage and patch attrition occur simultaneously with fragmentation and cause different responses in specific landscape metrics (Forman 1995). In a landscape with a range of forest patch sizes, deforestation-induced attrition and perforation can cause the number of patches to decrease, mean patch size to increase and perimeter-to-area ratio to decrease (Figure 3.2).

Many basic landscape metrics trend in one direction in early stages of land transformation and trend in the opposite direction in later stages (Forman 1995, Trani and Giles 1999). This complicates the interpretation of landscape change data and makes it difficult to establish management targets for restoration of landscape spatial structure.

Table 3.1: Computational formulas for selected landscape metrics.

Landscape Metric	Computational Formula
number of patches (NP)	NP = number of wooded patches > 2500 m^2 in area
patch density (PD)	$PD = NP / A_L$ where A_L is the total land area on the map sheet (total area minus water area)
perimeter-to-area ratio (P:A)	$P:A = \Sigma p_i / \Sigma a_i$ for $i = 1$ to NP where p_i is the perimeter of wooded patch i, including portion of the perimeter caused by the map extent boundary, and a_i is the area of wooded patch i
mean patch size (MPS)	$MPS = \Sigma a_i / NP$ for $i = 1$ to NP
mean patch size index (MPSI)	$MPSI = MPS / A_L$
largest patch size (LPS)	LPS = area of largest wooded patch on the map sheet
largest patch size index (LPSI)	$LPSI = LPS / A_L$
degree of coherence (COHE) (Jaeger 2000)	$COHE = \Sigma (a_i / A_L)^2$ for $i = 1$ to NP
effective mesh size (MSIZ) (Jaeger 2000)	$MSIZ = (\Sigma a_i^2) / A_L$ for $i = 1$ to NP

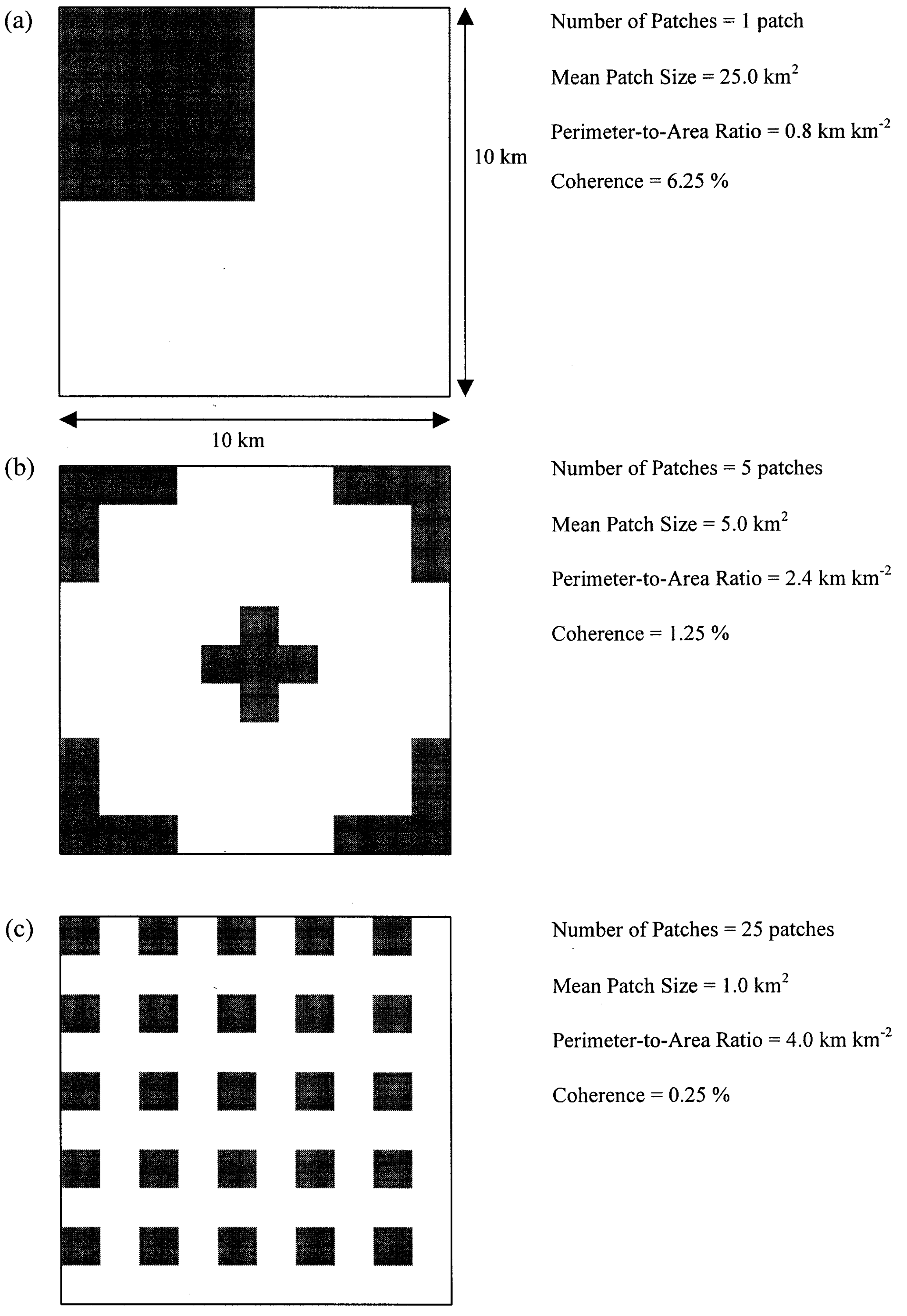
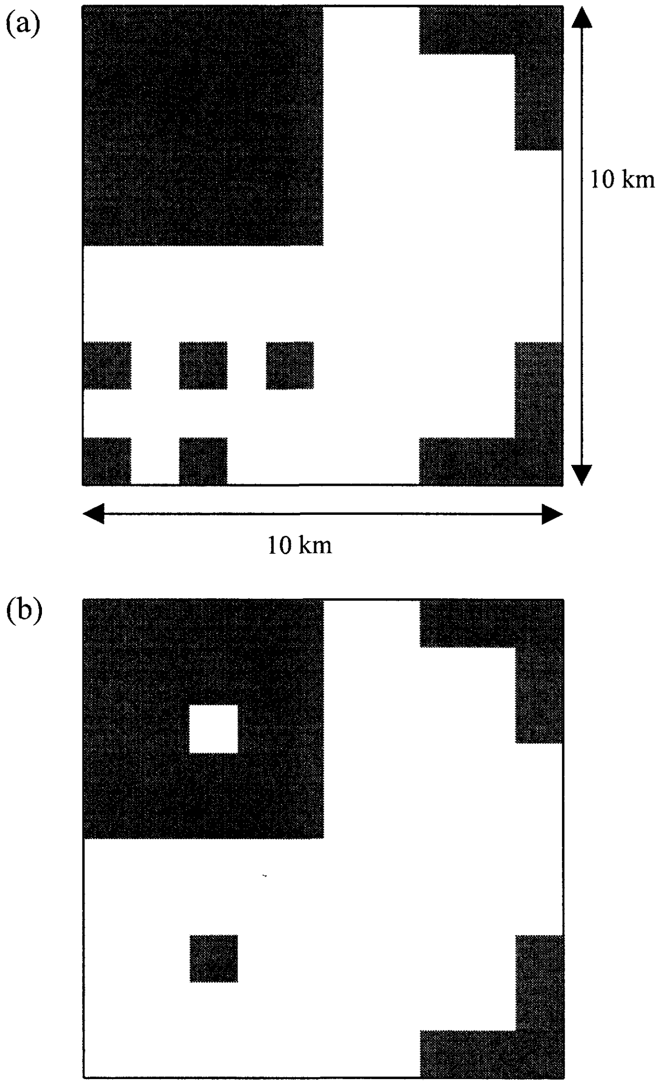


Figure 3.1: Landscape metrics for three 100-km2 landscapes that are 75% deforested (white areas) with varying degrees of fragmentation of forest remnants (gray areas). Adapted from Franklin and Forman (1987) and Turner et al. (2001).



Number of Patches = 8 patches

Mean Patch Size = 5.0 km^2

Perimeter-to-Area Ratio = 1.6 km km^{-2}

Coherence = 6.8 %

Number of Patches = 4 patches

Mean Patch Size = 8.75 km^2

Perimeter-to-Area Ratio = 1.58 km km^{-2}

Coherence = 6.27 %

Figure 3.2: Landscape metrics for two 100-km2 landscapes. Deforested area (white) increases from 60 % in (a) to 65 % in (b). Losses of forest from (a) to (b) occur due to perforation and attrition of forest patches (gray).

Jaeger (2000) proposed several landscape metrics designed to exhibit monotonic changes with land transformation processes such as patch perforation, patch fragmentation, patch shrinkage and patch attrition. These metrics are derived from the cumulative area distribution function for all habitat patches in a landscape and are proportional to the sum of squared patch sizes. The first metric, degree of coherence, represents the probability that any two random points on a binary map (e.g. habitat/non-habitat) are contained within the same habitat patch. In a simple landscape with 25 % habitat and 75 % non-habitat, the probability of two random points occurring within the same habitat patch is dependent on the number of habitat patches (e.g. is lower in more fragmented landscapes (Figure 3.1)). Unlike the other landscape metrics described above, coherence exhibits the same behaviour (i.e., decreases) not only with patch fragmentation, but also with patch perforation, patch attrition and patch shrinkage (Figure 3.2).

Effective mesh size, the second metric proposed by Jaeger (2000), is area-proportionately additive* and thus suitable for comparison between landscapes of different areas (Jaeger 2000). Effective mesh size is expressed in units of area (e.g. km^2). It is the product of a probability (degree of coherence) and the total study area size. Its maximum value is equal to the size of the study area in a landscape that is entirely unfragmented habitat. Its minimum value is zero in landscapes with no remaining habitat, and it tends toward zero in landscapes with highly dissected habitat.

Further research is required to describe the behaviour of specific landscape metrics with land transformation and to evaluate the sensitivity of these landscape metrics to landscape changes of ecological or management significance (Turner et al. 2001).

Despite incomplete knowledge on the behaviour and sensitivity of specific metrics, monitoring of landscape spatial structure using patch metrics and fragmentation indices to detect temporal changes or spatial differences in forest conditions has been recommended by Baskent and Jordon (1995), Haines-Young and Chopping (1996) and Noss (1999). Analyses using landscape metrics have provided valuable insights into the

* An area-proportionally additive measure is independent of landscape size. It can be calculated for a combination of two landscapes in the same way that concentration is determined for two combined liquids (Jaeger 2000).

ecological effects of human activities, particularly logging, in forest regions (e.g. Mladenoff et al. 1993, Spies et al. 1994, Wallin et al. 1994). These frequently cited studies illustrate three basic approaches to landscape ecology research.

Spies et al. (1994) compared spatial landscape structure within a single landscape for several time periods to determine the landscape-scale temporal effects of forestry activities. This is the retrospective research approach that I used in Chapter 1. It requires a chronosequence of data for study landscapes.

To study a similar problem, Mladenoff et al. (1993) used an alternative research approach. They compared the spatial landscape structure within a single time period for two landscapes at different locations (one unaltered landscape and one managed forest landscape). This type of research uses space as an analog for time and requires the assumption that all landscapes were similar prior to the anthropogenic activities under study. This research approach is advantageous in many situations where antecedent data for a study area is unavailable or of an insufficient quality.

The approaches of Spies et al. (1994) and Mladenoff et al. (1993) quantify the effects of past and present human activities on forest landscapes but provide no insight into future landscape conditions. Due to the difficulty of conducting manipulative experiments at the landscape scale, simulations are the preferred approach to investigate future landscape changes (Turner et al. 2001). To investigate the effects of future logging, Wallin et al. (1994) compared future landscape spatial structure with simulation of dispersed cutovers and aggregated cutovers.

These three approaches to landscape ecology research (temporal comparisons, spatial comparisons and simulations) can be used individually or in combination.

3.1.2 Landscape Metrics and the Study of Deforestation and Reforestation

Numerous researchers have documented the temporal changes in landscape structure with forest harvesting in temperate and boreal regions (Ripple et al. 1991, Spies et al. 1994, Tian et al. 1995, Reed et al. 1996, Miller et al. 1998, Sachs et al. 1998, Tinker et al. 1998, Cushman and Wallin 2000). These studies provide few insights into the fragmentation effects of deforestation because the areas they defined as disturbed, open or non-forest represented forest regeneration patches, not deforestation patches.

Quantification of deforestation-induced temporal changes to the spatial structure of landscapes has been reported for hardwood forests in eastern portions of the United States (Zipperer 1993, Kress et al. 1996, Fuller 2001) and for montane forests in Asia (Zheng et al. 1997). Kress et al. (1996), Zheng et al. (1997) and Fuller (2001) reported that the number of patches increases with deforestation. Perimeter-to-area ratio calculated from the data of Kress et al. (1996) increased with deforestation. Kress et al. (1996) and Zheng et al. (1997) reported decreases in mean patch size with deforestation, but Fuller (2001) reported that mean patch size exhibited an inconsistent response with deforestation, increasing in some watersheds and decreasing in other watersheds. Kress et al. (1996) and Zheng et al. (1997) reported decreases in largest patch size with deforestation. Similar data for deforestation induced changes to landscape spatial structure in boreal regions have not been reported.

Spatial comparisons of several landscapes in different stages of deforestation or conversion to agriculture are common for temperate portions of the United States (Krummel et al. 1987, O'Neill et al. 1988, Zipperer et al. 1990, Knick and Rotenberry 1997, Miller et al. 1997, Trani and Giles 1999, Wickham et al. 1999). Trani and Giles (1999) compared contiguous forest and fragmented forest landscapes in Virginia, and reported that fragmented landscapes had higher forest patch densities, lower mean patch sizes, and higher forest edge length. Wickham et al. (1999) reported that largest forest patch size decreased at a much more rapid rate than the proportion of anthropogenic land cover increased across a series of watersheds in the Eastern United States. Most of the other studies used alternative landscape measures and did not report data for the basic landscape metrics described above. Spatial comparisons of landscape structure for boreal regions have not been reported.

Reforestation and afforestation for purposes of mitigating the climatic effects of carbon dioxide emissions from fossil fuel burning have been investigated as options for sequestering organic carbon in terrestrial ecosystems of western Canada (Izaurralde et al. 1997, Guy and Benowicz 1998, Peterson et al. 1999). Many researchers have simulated the spatial effects of logging within forested landscapes (Franklin and

Forman 1987, Li et al. 1993, Wallin et al. 1994, Baskent 1999), but there are few studies on the spatial effects of simulated reforestation within landscapes that include agriculture (Dunn et al. 1991).

3.2 Objectives, Research Design and Hypotheses

This investigation examines spatial changes for wooded area within multi-jurisdictional boreal landscapes that include agricultural land use. The objectives for Chapter 3 are (i) to evaluate the applicability of a variety of landscape metrics to the analysis of deforestation and reforestation within boreal landscapes, (ii) to document the effects of historic deforestation on spatial structure in central Saskatchewan landscapes, and (iii) to explore the effects of potential reforestation on spatial structure for these same landscapes.

I quantified patch area and perimeter metrics for wooded area across four landscapes that exhibited deforestation over approximately three decades prior to 1990. The four landscapes in central Saskatchewan were selected to represent a gradient in the proportion wooded. Each of the four landscapes included lands legally protected from deforestation (national park or provincial forest) and lands not legally protected from deforestation (privately and publicly held agricultural and other lands). Analysis included temporal comparisons within each landscape and spatial comparisons between the landscapes within each time period.

I used simulations to contrast the spatial effects of two reforestation strategies: dispersed and contiguous. The first strategy entailed the addition of numerous small reforestation patches dispersed throughout agricultural portions of the landscape, as might occur with an economic incentive program to encourage voluntary reforestation by landowners. The second strategy involved the addition of a single large reforestation patch contiguous to protected forests, as might occur with a planned reforestation program with land acquisition by governments or private conservation organizations.

The first objective of Chapter 3 was accomplished by calculating a set of landscape metrics (Table 3.1) for a three dates for each of the four landscapes. Each metric was plotted over time for each landscape (temporal comparisons) or plotted across the proportion-wooded gradient for each time period (spatial comparisons). Whether each metric consistently exhibited monotonic changes with deforestation was then determined.

Several hypotheses were generated with respect to the effects of deforestation on landscape spatial structure (second objective of Chapter 3).

1. Within each landscape over time, the number of patches would increase due to patch fragmentation in landscapes with more than 50% of land area wooded, and decrease due to patch attrition in landscapes with less than 50% of land area wooded, as predicted by Gustafson and Parker (1992) and Forman (1995).
2. Within each landscape over time, (a) perimeter-to-area ratio would increase, (b) mean patch size would decrease, (c) largest patch size would decrease and (d) degree of coherence would decrease, due to fragmentation, shrinkage and attrition of forest patches based on predictions of Forman and Godron (1986), Forman (1995) and Jaeger (2000).
3. For the spatial comparisons within time periods, patch density would be greatest in the landscape closest to 50 % deforested.
4. For the spatial comparisons within time periods, (a) perimeter-to-area ratio would be inversely related to the proportion of land area wooded, (b) mean patch size index would be positively related to the proportion of land area wooded, (c) largest patch size index would be positively related to the proportion of land area wooded, and (d) effective mesh size would be positively related to the proportion of land area wooded.

To evaluate whether fragmentation actually compounded the effect of direct forest area losses, I adapted the spatial comparison method of Wickam et al. (1999) for temporal data. The observed change in landscape metrics for temporal deforestation was compared with the expected change if deforestation occurred only through patch shrinkage. One final hypothesis was generated regarding the historic deforestation data.

5. For (a) mean patch size, (b) largest patch size, and (c) the degree of coherence, the magnitude of the observed change for temporal deforestation would exceed that which would be expected if each wooded patch was reduced in size at the same proportion that wooded area was lost in the landscape as a whole.

The final set of hypotheses is related to the third objective of Chapter 3. Dispersed and contiguous reforestation strategies were expected to have different spatial effects as demonstrated by Wallin et al. (1994) for dispersed and aggregated cutovers. If one of the aims of reforestation is ecological restoration of forest landscapes, then the best strategy is that which leads to the greatest reversal of the changes in spatial structure observed with historic deforestation.

6. For each landscape metric, the direction of temporal change occurring with the contiguous reforestation strategy would be opposite to the direction of temporal change observed for historic deforestation.
7. For each landscape metric, the temporal change occurring with the contiguous reforestation strategy would be larger in magnitude than the temporal change occurring with the dispersed reforestation strategy.

3.3 Study Area Descriptions

Of the 10 NTS maps used in Chapter 2, four were selected for detailed analysis of landscape spatial structure: two from the Waskesiu Hills area and two from the Red Deer River area (Figure 3.3). Each of the four selected landscapes spans 15' latitude and 30' longitude. Name, location, area data and the proportion protected as national park or provincial forest, is provided for each landscape in Table 3.2. The four landscapes span a gradient in the proportion of land area wooded (Figure 3.4).

Table 3.2: NTS map name and number, latitude range, longitude range, land area, total area, and protected forest, for the study landscapes in central Saskatchewan.

Map Sheet Name	Map Sheet Number	Latitude Range	Longitude Range	Land Area (km^2)	Total Area (km^2)	Protected Forest (%)
Bjorkdale	63 D/12	52° 30' to 52° 45'	103° 30' to 104° 00'	905	941	7
Debden	73 G/10	53° 30' to 53° 45'	106° 00' to 106° 30'	834	916	24
Halkett Lake	73 G/9	53° 30' to 53° 45'	106° 30' to 107° 00'	875	916	68
Mistatim	63 D/14	52° 45' to 53° 00'	103° 00' to 103° 30'	897	936	37

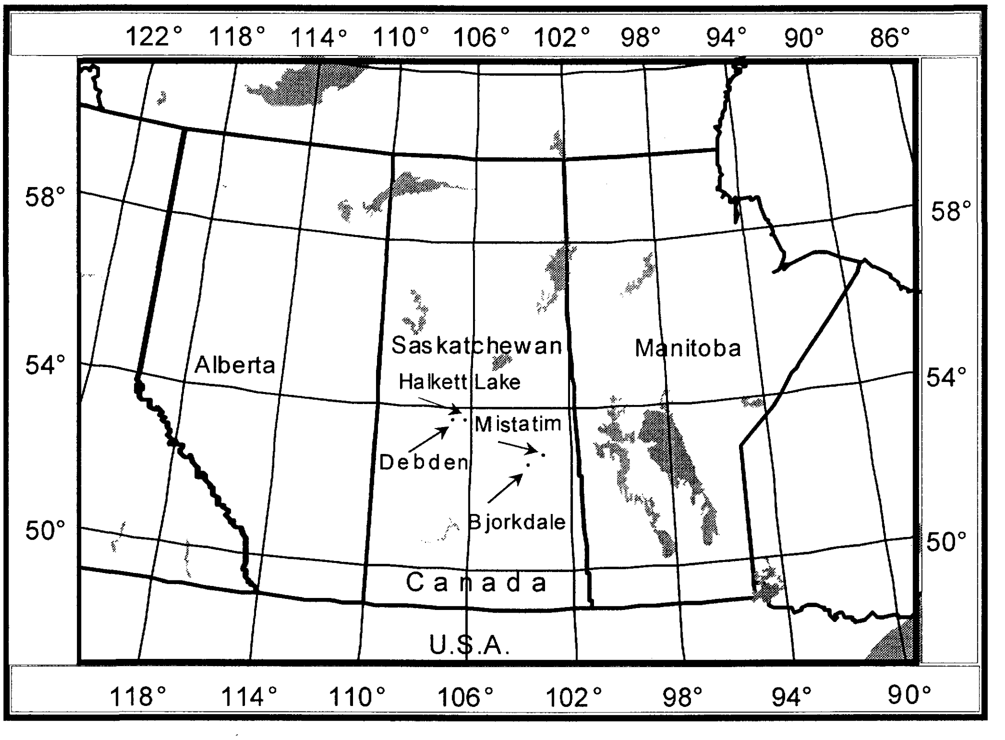


Figure 3.3: Location of the Halkett Lake, Mistatim, Debden and Bjorkdale landscapes in central Saskatchewan.

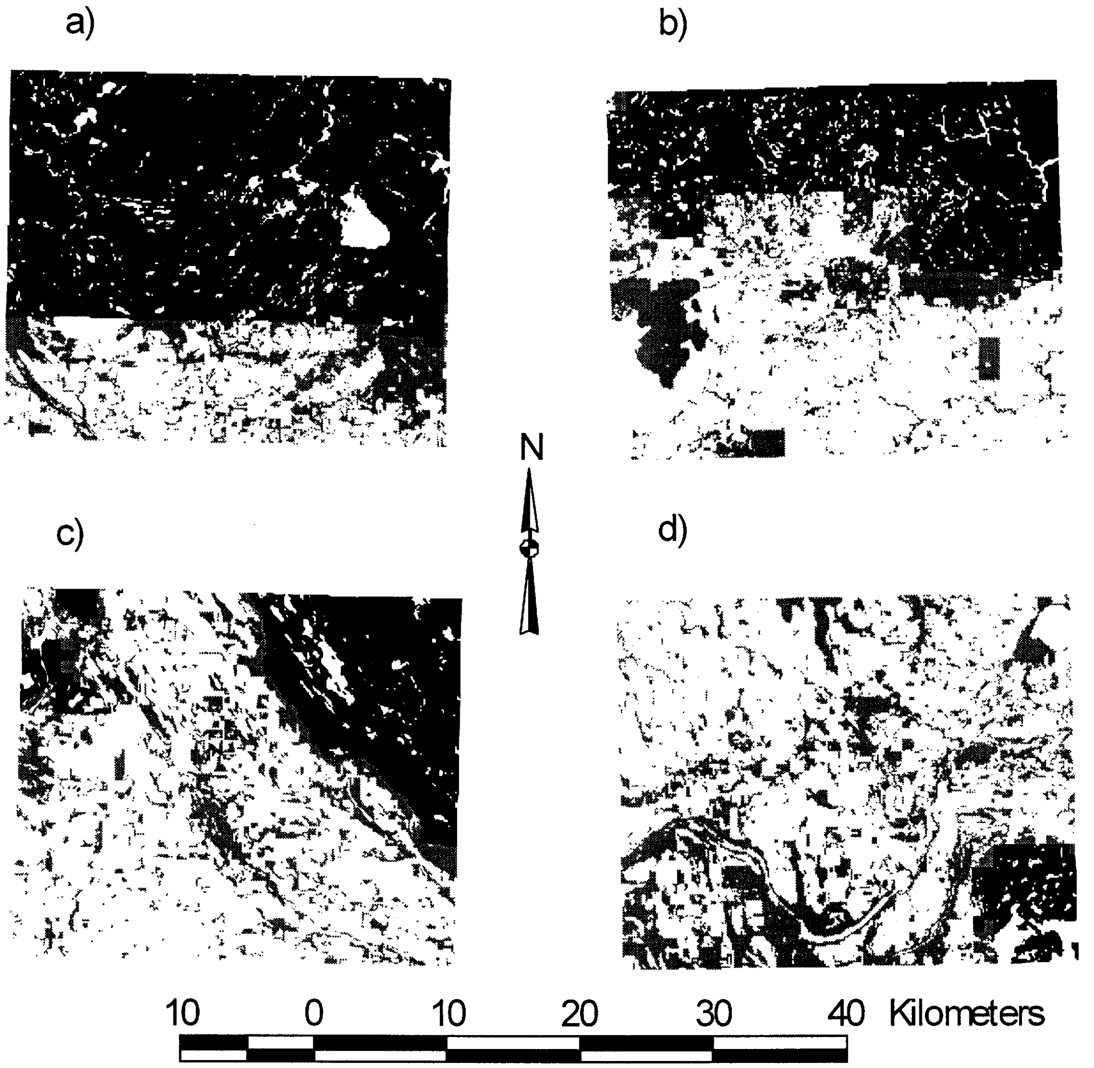


Figure 3.4: A descending gradient in proportion of landscape area protected from land-use change from (a) Halkett Lake, to (b) Mistatim, (c) Debden and (d) Bjorkdale. Legally protected areas mapped as wooded in 1990 (black) and other areas mapped as wooded in 1990 (gray).

On the Halkett Lake and Debden landscapes, dominant landforms are hummocky and moderately rolling glacial till plains, and the most prevalent soils are Gray Luvisols and Dark Gray Chernozems (Acton et al. 1998). These landscapes encompass portions of Prince Albert National Park, administered by the Government of Canada, and the Northern Provincial Forest, administered by the Government of Saskatchewan. Outside of these jurisdictions there is an aggregation of smaller holdings of cultivated, grazed and natural lands under a mix of private and public ownership (e.g. private farms, Crown agricultural land, Indian Reserves, and community pastures).

On the Mistatim and Bjorkdale landscapes, dominant landforms are undulating glacial till and glaciolacustrine plains and most frequently occurring soils are Dark Gray Chernozems and Gray Luvisols (Acton et al. 1998). The Mistatim landscape includes a portion of the Northern Provincial Forest and the Bjorkdale landscape includes a portion of Greenwater Provincial Park, which is also part of the provincial forest system. Outside of these areas is a mix of smaller holdings similar to that described for the Halkett Lake and Debden landscapes.

Clearing of forests and shrublands for agricultural use is prohibited by legislation in the national park and provincial forest. On the remaining lands, forests are afforded no legal protection under federal or provincial law.

Each landscape has a sub-humid cool continental climate with a mean annual temperature of approximately 0°C and mean annual precipitation of 400 to 500 mm (Kabzems et al. 1986, Acton et al. 1998). Boreal forest was the dominant land cover for these landscapes a century ago (DNR 1955, Rowe 1959), but major portions were cleared for agriculture over the past century (Kabzems et al. 1986, Weir and Johnson 1998). Dominant tree species in the Boreal Plain Ecozone include trembling aspen, white spruce, black spruce, and jack pine. Lesser amounts of white birch, balsam poplar, balsam fir, and tamarack are also present (Acton et al. 1998). Dominant tall shrub genera include willows and alders.

3.4 Materials and Methods

3.4.1 Source Data

Maps from Canada's National Topographic System were used as the source of land cover data. Green areas on these maps are wooded, defined as land at least 35% covered by trees or shrubs with a minimum height of 2 m (NRCan 1996). The chronosequence of map data allowed temporal analysis of landscape spatial structure between edition 1 (1957, 1958 or 1963), edition 2 (1975 or 1976) and edition 3 (1990). Wooded area data represent the year that air photos were taken, rather than the year that the maps were published.

Digitization and accuracy assessment were described in Chapter 2. Areas and perimeters for digital wooded area data were calculated using ArcView® software (ESRI 1996). Grain size for the analysis was 2500 m^2 (1 mm by 1 mm on the 1:50,000 analog map).

Analyses of landscape spatial structure were restricted to quantification of mapped area features (i.e. wooded areas, open areas, and water bodies). Those features represented as lines (e.g. roads and transmission lines) or points (e.g. buildings and campgrounds) were not considered because they were not depicted on the maps in an area-proportional manner. As a result, the patch area and perimeter metrics do not account for dissection of wooded polygons by line features and perforation of wooded polygons by point features. Only methods suitable for vector polygon data were utilized.

3.4.2 Selection of Landscape Metrics

Of the plethora of landscape metrics available for application (McGarigal and Marks 1995, Haines-Young and Chopping 1996, Gustafson 1998), only a select few were calculated. Only metrics suitable for binary maps with vector polygons were considered. I restricted the analyses primarily to metrics common to a number of temperate deforestation studies (e.g. mean patch size), or indices based on these metrics.

The exceptions were degree of coherence and effective mesh size (Jaeger 2000), two recently proposed metrics that were previously untested with real landscape data.

Landscape metrics analyzed for temporal changes within each landscape (hypotheses 1 and 2) included number of patches, perimeter-to-area ratio, mean patch size, largest patch size, and degree of coherence (Table 3.2). Perimeters of wooded patches included those created by the map extent boundaries. Patch sizes were truncated by map extent.

The degree of coherence (Jaeger 2000) is influenced by the existence of lakes and rivers that cause geogenic fragmentation* within the landscape (Jaeger 2000). To better quantify the effects of anthropogenic fragmentation related to deforestation, which can not occur in water bodies, the degree of coherence was calculated using the total land area of the map sheet rather than the total area of the map sheet (land plus water). Changes over time in the area of water within each landscape were deemed negligible.

Related metrics were used for time-specific spatial landscape comparisons (hypotheses 3 and 4). These included patch density, perimeter-to-area ratio, mean patch size index, largest patch size index and effective mesh size (Table 3.2). For each landscape and each time period, patch number, mean patch size and largest patch size were converted to densities or indices through division by the total land area. This was done to account for variation in total land area between the four map sheets. Perimeter-to-area ratio is less affected by differences in land area than the other metrics and was unadjusted for the spatial comparisons. Effective mesh size was designed to be suitable for comparison between landscapes of different areas (Jaeger 2000). For similar reasons as described for degree of coherence, I calculated effective mesh size using total land area in the denominator rather than the total study area size (land plus water).

Mean patch size, largest patch size and degree of coherence were used to test hypothesis 5 for each landscape comparison from time 1 to time 2. The observed changes for these three metrics were plotted against the expected changes if deforestation occurred through patch shrinkage alone. Expected changes were calculated by reducing the area of each patch for time 1 by the same percentage that

* Geogenic fragmentation of forests is caused by natural barriers such as water bodies or wetlands (Jaeger 2000).

wooded area was reduced across the entire landscape. For example, if a 10% loss of wooded area was observed for a landscape from time 1 to time 2, universal patch size shrinkage by 10% would cause a 10% decrease in mean patch size and largest patch size ($MPS_2 = MPS_1 \times 90\ \%$, $LPS_2 = LPS_1 \times 90\ \%$) and a 19% reduction in degree of coherence ($COHE_2 = COHE_1 \times (90\ \%)^2$).

3.4.3 Simulations

On each landscape, separate simulations were completed for the dispersed and contiguous reforestation strategies. The 1990 wooded area data was used as the basis for each simulation. A 10-year reforestation target of 2% of non-forest lands in agricultural regions was used for all simulations. Converting 2% of non-wooded areas to forest over 10 years was considered an ambitious objective, given that there is no historical precedent for conversion of agricultural land to forests in this region.

For the dispersed strategy, a random pattern of small reforestation patches was introduced. Each patch was 32 ha (one-eighth mile by one-quarter mile). This size was chosen because it was a convenient fraction (one-half) of the basic unit of land ownership, the quarter section, and it was a small proportion (less than 10%) of average farm size in this region of Saskatchewan (Statistics Canada 1992b). Each 32 ha rectangular patch was randomly located using a random numbers table and the UTM grid coordinates for each map. Each patch was oriented north to south and aligned with the nearest quarter-section land boundary to minimize the potential impact of reforested areas on farming operations on adjacent open lands.

For the contiguous strategy, a single patch equivalent in area to the sum of all dispersed patches was appended to the protected forest portion of the landscape at a random location along its boundary. The number of patches, perimeter-to-area ratio, mean patch size, largest patch sizes and degree of coherence were calculated for both simulations for each landscape (hypotheses 6 and 7).

3.5 Results

3.5.1 Evaluation of Selected Landscape Metrics

Wooded area declined within each of the four landscapes between the dates of the first, second and third map editions (Figure 3.5(a)). Changes for number of patches, perimeter-to-area ratio, mean patch size, largest patch size, and degree of coherence with progressive deforestation are shown in panels (b) to (f) of Figure 3.5.

Number of patches, perimeter-to-area ratio, and mean patch size did not exhibit monotonic change over time consistently across all landscapes. All three metrics exhibited a positive slope during one time period and a negative slope in another time period for at least two of the four landscapes. Net changes for number of patches and perimeter-to-area ratio over the entire period of examination included both increases and decreases for different landscapes. Largest patch size and degree of coherence exhibited monotonic change with deforestation. Values for both metrics decreased within each landscape for each time interval.

Proportion of land area wooded varied from 43 to 79 % in 1975/1976 and from 34 to 77 % in 1990 (Figure 3.6(a)). The values for patch density, perimeter-to-area ratio, mean patch size index, largest patch size index and effective mesh size across this gradient are shown in panels (b) to (f) of Figure 3.6. Patch density and mean patch size index are affected by mapping anomalies related to the smallest wooded patches. Monotonic relationships with proportion of land area wooded were observed for patch density and mean patch size index in only one of the time intervals. Monotonic relationships were observed (negative correlations for 1975/76, and 1990) for perimeter-to area ratio ($r^2 = 0.96$, $p < 0.05$, $r^2 = 0.83$, $p < 0.10$), largest patch size index ($r^2 = 0.99$, $p < 0.01$, $r^2 = 0.99$, $p < 0.01$), and effective mesh size ($r^2 = 0.99$, $p < 0.01$, $r^2 = 0.97$, $p < 0.05$) across the gradient in proportion of land area wooded.

Considering both the temporal and spatial comparisons, only metrics proportional to the size of the largest patch (i.e., largest patch size and largest patch size index) or to the sum of squared patch sizes (i.e., degree of coherence and effective mesh size) consistently exhibited monotonic changes with deforestation.

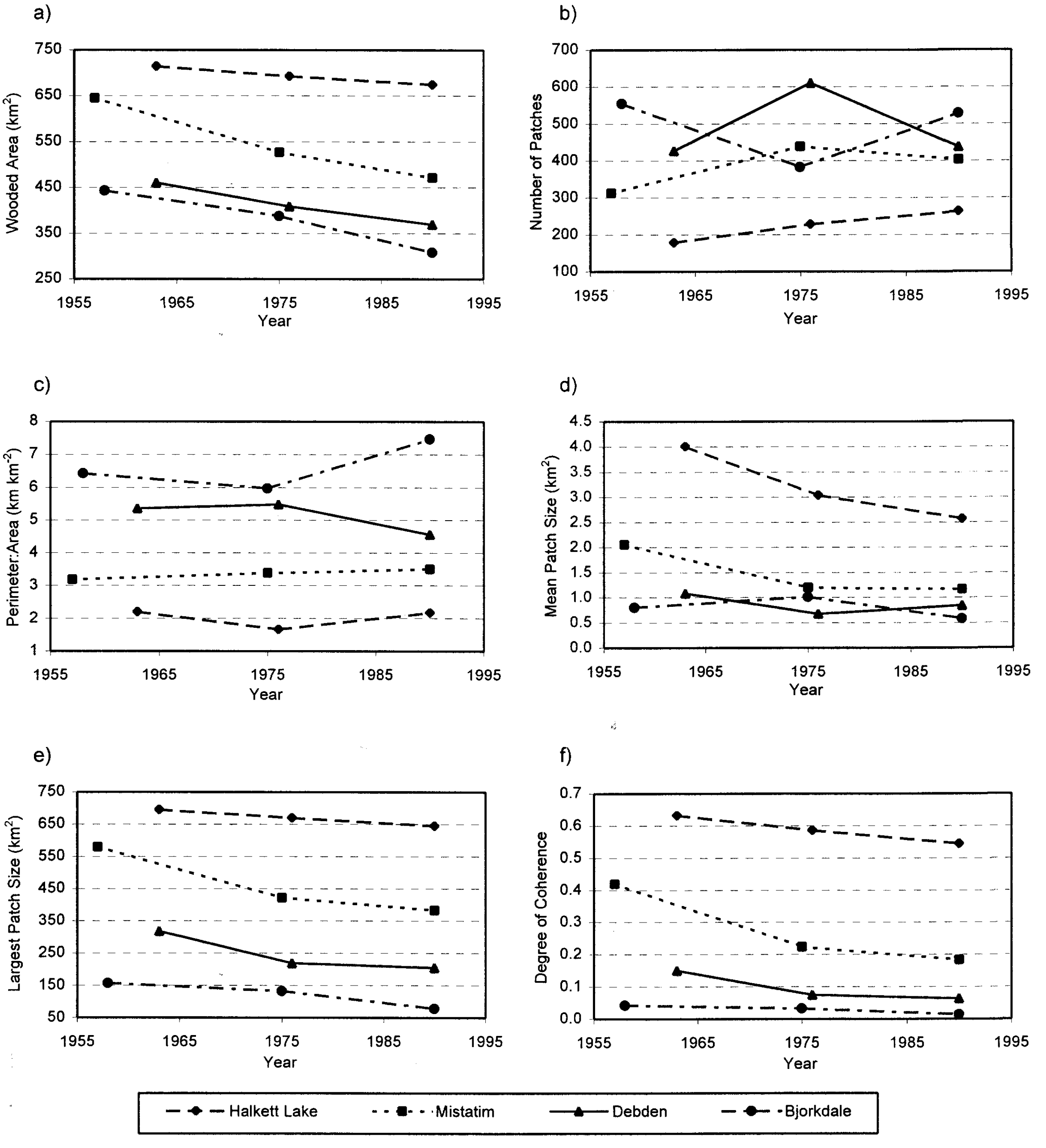


Figure 3.5: Temporal changes for (a) wooded area, (b) number of patches, (c) perimeter-to-area ratio, (d) mean patch size, (e) largest patch size and (f) degree of coherence within the Halkett Lake, Mistatim, Debden and Bjorkdale landscapes.

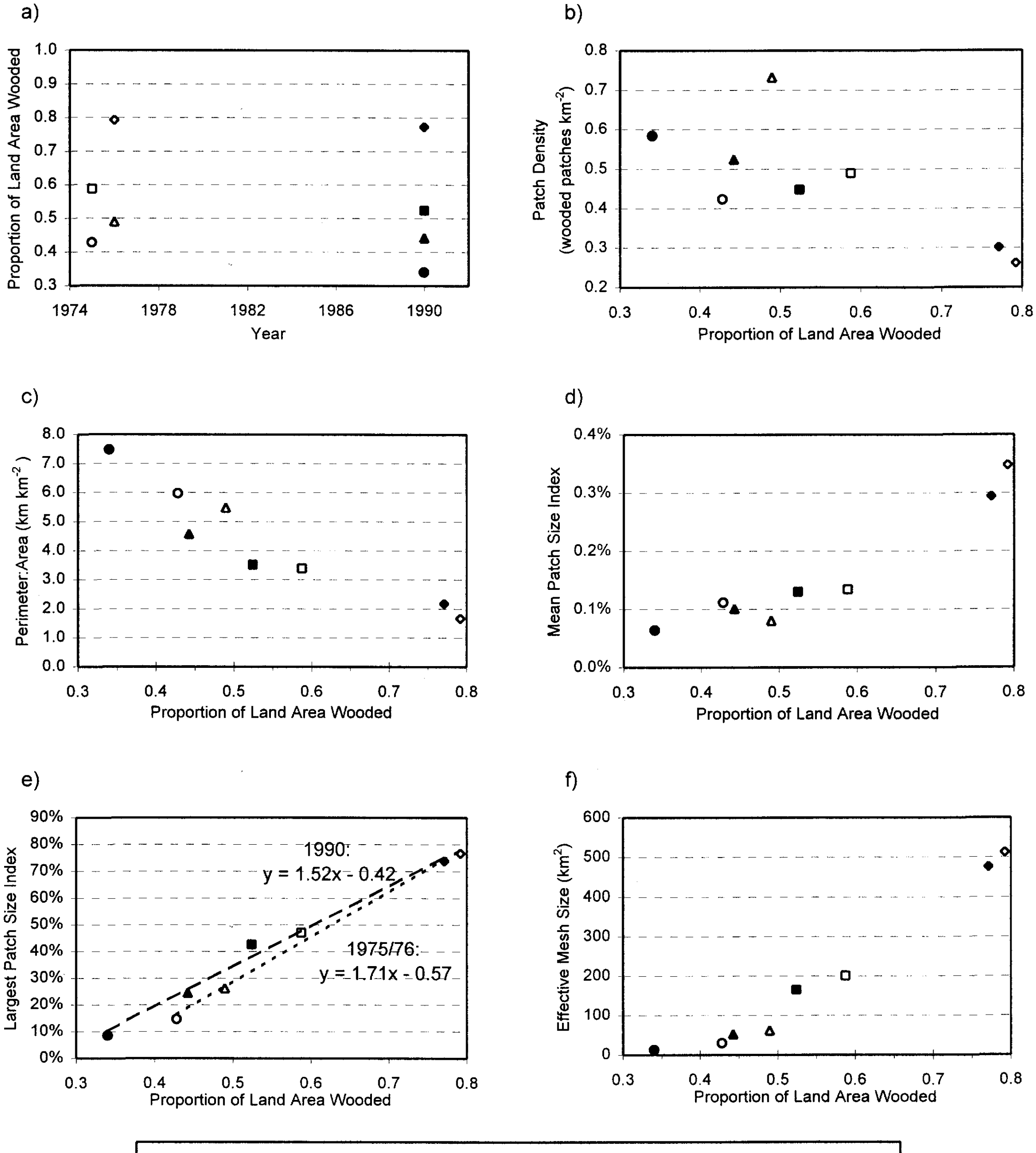


Figure 3.6: Spatial comparisons of (a) proportion of land area wooded, (b) patch density, (c) perimeter-to-area ratio, (d) mean patch size index, (e) largest patch size index and (f) effective mesh size in 1975/76 (open symbols) and in 1990 (closed symbols) for the Halkett Lake, Mistatim, Debden and Bjorkdale landscapes. Lines in (e) represent the principal axes of the observed relationships between largest patch size index and proportion of land area wooded.

3.5.2 Changes in Landscape Spatial Structure with Deforestation

Comparisons of temporal changes for landscape metrics with progressive deforestation within each landscape provided results for the first two hypotheses investigated. Wooded area decreased in each landscape during both time periods (Figure 3.5(a)). The greatest loss occurred in the Mistatim landscape between 1957 and 1975 (118 km^2 decrease) and the smallest loss occurred in Halkett Lake between 1976 and 1990 (18 km^2 decrease).

Changes in the number of patches within a landscape over one time increment ranged from an increase of 186 patches to a decrease of 173 patches. Changes for number of patches did not generally conform to the hypothesis. For example, the number of patches was hypothesized to decline due to patch attrition in Bjorkdale, the landscape with the least wooded area. However, the number of patches decreased prior to 1975 and increased after 1975 Figure 3.5(b).

Changes in perimeter-to-area ratio ranged from an increase of 1.5 km km^{-2} to a decrease of 0.9 km km^{-2}. Hypothesized increases for perimeter-to-area ratio were not generally observed (Figure 3.5c).

Changes in mean patch size ranged from an increase of 0.17 km^2 to a decrease of 0.97 km^2. The pattern for mean patch size mirrored the pattern observed for the number of patches. Mean patch size decreased through both time periods in the Halkett Lake landscape as hypothesized, but in the remaining landscapes mean patch size decreased during one of the time periods, and either increased or remained unchanged during the other time period (Figure 3.5d). Mean patch size declined with large losses of wooded area (Mistatim 1957 to 1975 and Bjorkdale 1975 to 1990), but showed variable responses with smaller losses of wooded area.

Decreases in largest patch size ranged from 25 to 158 km^2. In each of the four landscapes, largest patch size exhibited a temporal pattern of decline as hypothesized (Figure 3.5e). The greatest absolute decline in largest patch size coincided with the landscape and time period where the greatest absolute decline in wooded area was observed (Mistatim between 1957 and 1975).

The degree of coherence decreased over each time period within each landscape as hypothesized. Changes in coherence ranged from 0.01 to 0.20 (Figure 3.5f). The greatest decrease in coherence was associated with the greatest loss of wooded area (1957 to 1975 Mistatim). In each landscape, the temporal patterns observed for coherence were very similar to those observed for the size of the largest patch. The magnitudes of the decrease for coherence were small in Bjorkdale, a landscape dominated by non-forest.

Comparisons among the four landscapes provided results for the third and fourth hypotheses. The arrangement of the four landscapes in a descending scale for the proportion of land area remaining wooded was Halkett Lake, Mistatim, Debden, and then Bjorkdale for both the 1975/76 and 1990 time periods (Figure 3.6a).

Patch density for the four landscapes ranged from 0.26 to 0.73 patches km^{-2} in 1976 and from 0.30 to 0.58 patches km^{-2} in 1990. Debden, the landscape closest to the mid-point of the deforestation process in 1975/76, exhibited the greatest patch density as hypothesized (Figure 3.6b). For 1990, patch densities did not peak for landscapes close to 50% wooded.

Across the four landscapes, perimeter-to-area ratio ranged from 1.7 to 6.0 km km^{-2} in 1975/76 and from 2.2 to 7.5 km km^{-2} in 1990. For both time periods, perimeter-to-area ratio varied across the gradient from Halkett Lake to Bjorkdale and was consistently greater in landscapes with lower proportions of land area wooded (Figure 3.6c).

Mean patch size index ranged from 0.08 to 0.35% of the total land area in 1975/76 and from 0.06% to 0.29% of the total land area in 1990. The hypothesis was that mean patch size index would be lower in landscapes with lower proportion of remaining wooded area. For 1975/76 data, mean patch size index decreased across most of the gradient of decreasing proportion of wooded area (from Halkett Lake to Mistatim to Debden) but the minimum mean patch size index did not occur in Bjorkdale (Figure 3.6d). For 1990 data, mean patch size index was greatest in Halkett Lake and least in Bjorkdale as hypothesized.

Largest patch size index across the four landscapes ranged from 15 to 79% of the total land area in 1975/76 and from 9 to 74% of the total land area in 1990. For both time periods, largest patch size index was lower in landscapes with lower proportions of land area remaining wooded as hypothesized (Figure 3.6e).

Effective mesh size was minimum in Bjorkdale (29.7 km^2 in 1975 and 12.6 km^2 in 1990) and maximum in Halkett Lake (512 km^2 in 1976 and 476 km^2 in 1990). For both time periods the observed pattern was consistent with that hypothesized: mesh size was lower in landscapes with lower proportions of land area remaining wooded (Figure 3.6f).

From these temporal and spatial comparisons, reduction in the size of the largest wooded patch emerged as one of the most consistent effects of deforestation on landscape spatial structure. A linear relationship between largest patch size index and proportion of land area wooded was not hypothesized, but was evident across the four landscapes for both 1975/76 (Figure 3.6(e)).

The slope of this linear relationship can be used to evaluate the degree to which fragmentation compounds the effects of wooded area losses. If deforestation occurred without fragmentation, then the slope* would be expected to be one. In the absence of fragmentation, if the proportion of land area wooded decreased by half, then the largest patch size index would decrease by half. In 1990, the proportion of land area wooded for Bjorkdale (34 %) was half that for Halkett Lake (77 %), but the largest patch size index for Bjorkdale (0.09 %) was only one-eighth that for Halkett Lake (74 %). Slopes were greater than unity for both time periods (Figure 3.6(e)), indicating that deforestation had disproportionately reduced the sizes of the largest patches by fragmenting them.

The fragmentation effects of deforestation were also evaluated by comparing observed changes for landscape metrics to expected changes for landscape metrics with patch shrinkage (hypothesis 5). The two time intervals and four landscapes allowed 8 temporal comparisons. In 5 cases the observed decrease of mean patch size was greater than the expected change (Figure 3.7a). In two cases (not shown in Figure 3.7a) mean patch size increased even though wooded area decreased.

* the principal axis (Sokal and Rohlf 1981) was used rather than a linear regression because of less restrictive assumptions.

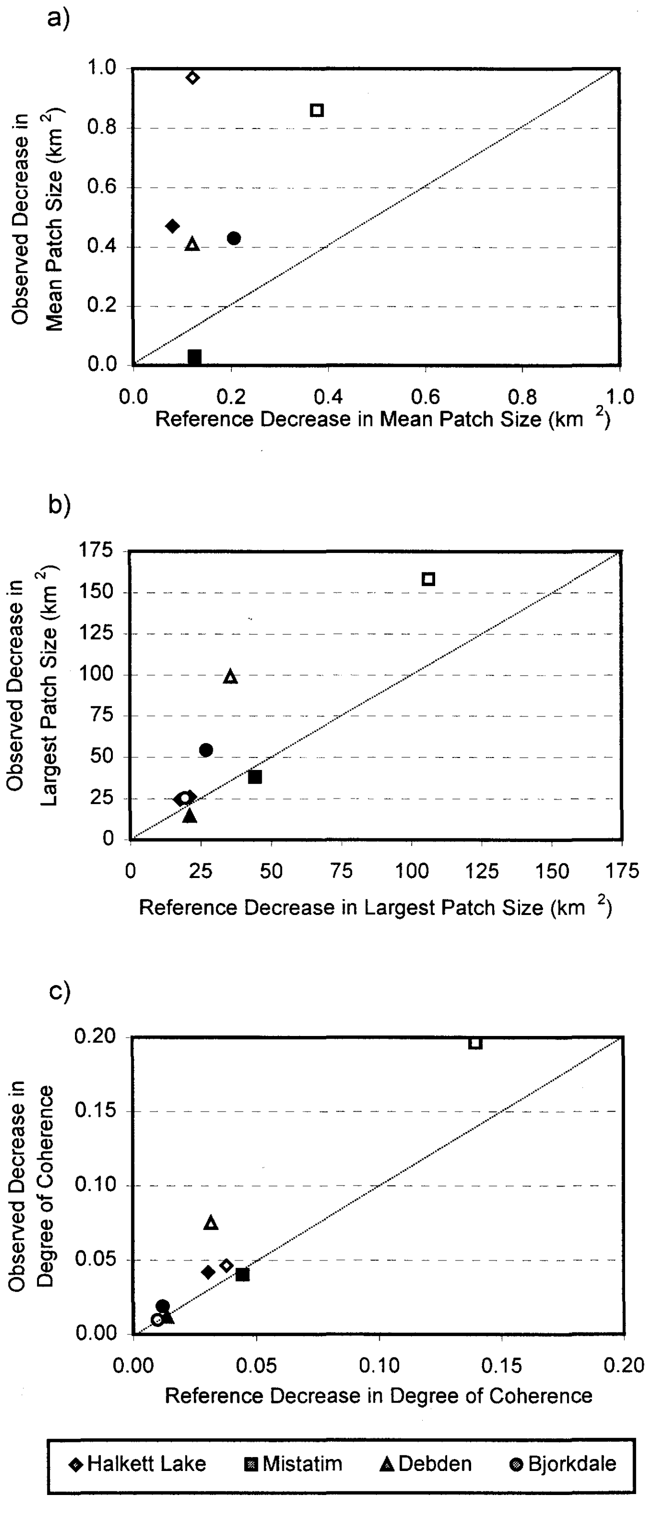


Figure 3.7: Observed decreases in (a) mean patch size, (b) largest patch size, and (c) degree of coherence in relation to reference decreases for patch shrinkage proportional to wooded area losses for the Halkett Lake, Mistatim, Debden and Bjorkdale landscapes. Open and closed symbols represent changes prior to 1975/76 and 1990, respectively.

For 6 of 8 comparisons, the observed decrease in largest patch size was larger than the expected decrease for universal patch shrinkage (Figure 3.7b). In 5 of 8 cases, the observed decrease in coherence was greater than the reference change for universal patch shrinkage (Figure 3.7c). In the majority of these temporal comparisons, patch sizes and the sum of squared patch sizes decreased at a rate greater than that expected for patch shrinkage, indicating that patch fragmentation was occurring.

3.5.3 Changes in Landscape Spatial Structure with Reforestation

The 2% reforestation target for non-wooded lands resulted in simulated increases in wooded area of 4.0 km^2 in Halkett Lake, 8.0 km^2 in Mistatim, 9.5 km^2 in Debden and 12 km^2 in Bjorkdale by 2011. Figure 3.8 shows a portion of the Halkett Lake landscape with both dispersed and contiguous patches from the two simulations.

For all four landscapes, the number of patches increased with the dispersed small patch simulation (Table 3.3). In Halkett Lake, Mistatim, and Debden, perimeter-to-area ratio increased, and mean patch size decreased. In Bjorkdale, the most fragmented landscape, the addition of the dispersed patches decreased perimeter-to-area ratio and increased mean patch size. For the dispersed patch simulation, largest patch size and coherence were unchanged for all landscapes but Mistatim where a small increase occurred.

For the contiguous block simulation, decreases in the number of patches and perimeter-to-area ratio, and increases in mean patch size, largest patch size and coherence were observed for all landscapes (Table 3.3). The magnitude of the increases for largest patch size and coherence were smallest in Bjorkdale even though the simulated reforestation block was the largest for this landscape. The protected portion of the Bjorkdale landscape, to which the reforestation block was appended, was not the largest wooded patch in the landscape in 1990 (Figure 3.4).

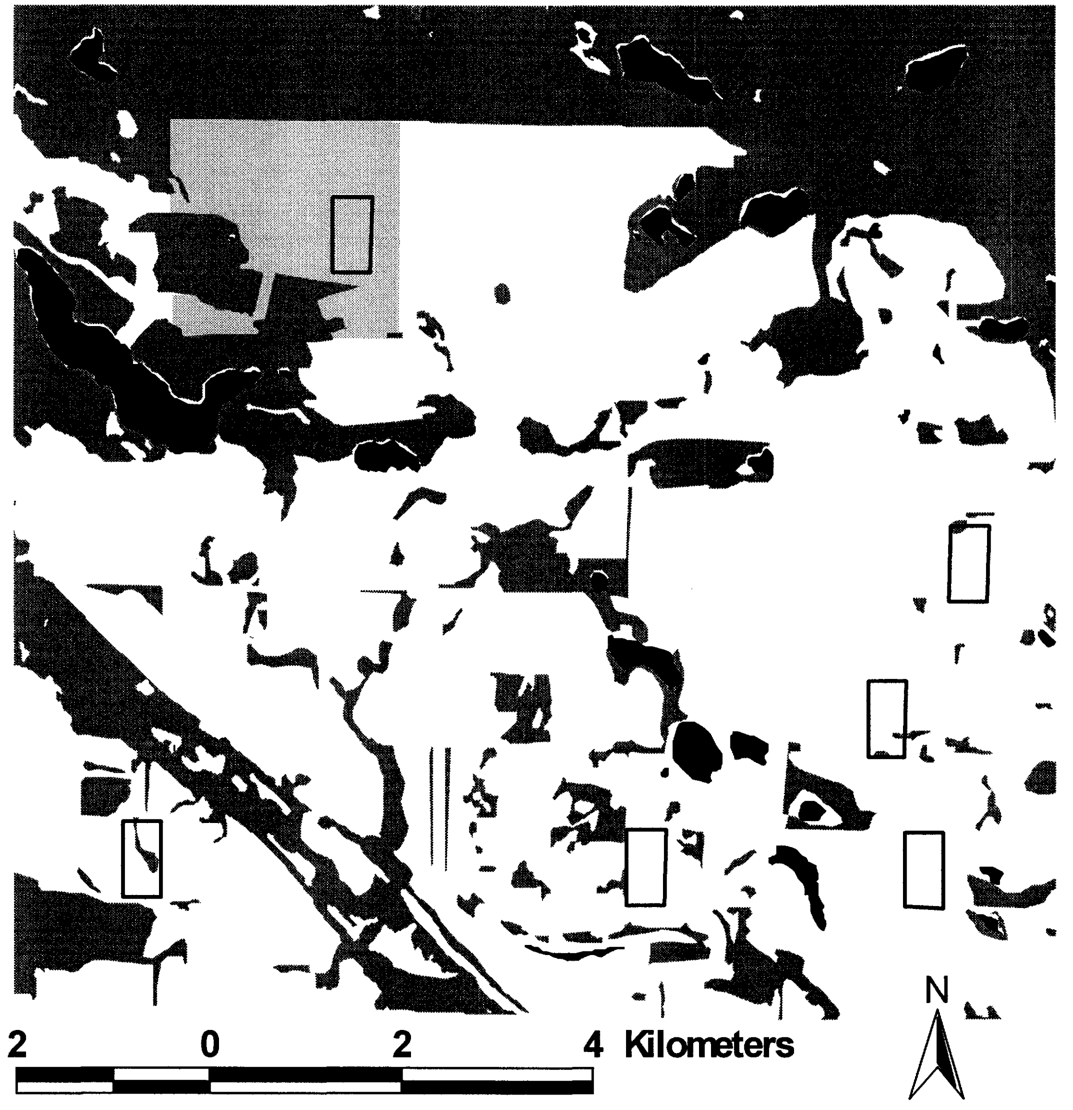


Figure 3.8: Southeast portion of the Halkett Lake landscape showing areas wooded in 1990 (dark gray), with 6 of 14 randomly dispersed small reforestation patches (rectangles with black outline), one large reforestation block (light gray) contiguous to Prince Albert National Park (top), and areas of water shown (black).

Table 3.3: Metric data for actual 1990 landscapes and simulated 2011 landscapes with dispersed and contiguous reforestation strategies, for Halkett Lake, Mistatim, Debden and Bjorkdale, Saskatchewan.

Landscape	Year	Number of Patches	Perimeter-to-Area Ratio (km km^{-2})	Mean Patch Size (km^2)	Largest Patch Size (km^2)	Degree of Coherence
Halkett Lake	1990	263	2.17	2.57	645	0.544
	2011 Dispersed	265	2.19	2.56	645	0.544
	2011 Contiguous	259	2.14	2.62	653	0.557
Mistatim	1990	403	3.50	1.17	384	0.184
	2011 Dispersed	423	3.55	1.13	385	0.186
	2011 Contiguous	393	3.43	1.22	393	0.193
Debden	1990	437	4.56	0.84	204	0.063
	2011 Dispersed	454	4.62	0.83	204	0.063
	2011 Contiguous	428	4.42	0.88	216	0.070
Bjorkdale	1990	528	7.47	0.58	78	0.014
	2011 Dispersed	538	7.42	0.59	78	0.014
	2011 Contiguous	520	7.17	0.61	79	0.016

The spatial effects were different for the two reforestation strategies. Consistent decreases for number of patches and perimeter-to-area ratio, and consistent increases for mean patch size, largest patch size and coherence occurred with the contiguous reforestation strategy. Consistent with hypothesis 6, the directions of the changes for largest patch size and coherence were opposite for contiguous reforestation and historic deforestation. The dispersed strategy generally had no effect on largest patch size and coherence. The changes for landscape metrics were smaller in magnitude for the dispersed strategy relative to the contiguous strategy, confirming hypothesis 7.

3.6 Discussion

3.6.1 Evaluation of Selected Landscape Metrics

Quantifying changes in landscape spatial structure with progressive deforestation and forest fragmentation is important to facilitate future landscape management and monitoring actions. Efforts to stabilize landscapes or reverse historical trends will require baseline measures of landscape structure. In this study, landscape metrics proportional to the size of the largest patch or the sum of squared patch sizes proved more useful for this purpose than number of patches, mean patch size, perimeter-to-area ratio and related indices or densities.

Metrics based on the size of the largest patch showed consistent responses with deforestation in this and other studies. Temporal comparisons revealed a consistent negative relationship between the size of the largest patch and deforestation in this study, similar to the results of Kress et al. (1996) and Zheng et al. (1997) for temperate North American and Asian landscapes respectively. Largest-patch metrics are simple and useful measures for monitoring deforestation.

Degree of coherence and effective mesh size are proportional to the sum of squared patch sizes and are greatly influenced by large patches within the landscape. These two metrics also exhibited lower values in landscapes with less wooded area in this study. The advantage of coherence and mesh size over largest patch metrics is that they are mathematical functions of all patch sizes, rather than being dependent upon the size of only one patch. Degree of coherence and effective mesh size, although less intuitive than largest patch size, may also prove to be a useful tool for landscape management and monitoring.

Number of patches and patch density varied unpredictably with deforestation in this study where landscapes were 30% to 80% wooded. These metrics do increase in the earliest stages of forest loss and fragmentation, (e.g. Zheng et al. 1997) and decrease in the latest stages of deforestation (Zipperer et al. 1990, Gustafson and Parker 1992), but may behave erratically in between these periods.

Mean patch size and its related index are a function of the reciprocal of the number of patches and thus exhibit similarly unpredictable behavior. The largest temporal decreases in wooded area in this study resulted in decreases in mean patch size as was observed by Burgess and Sharpe (1981), Kress et al. (1996) and Zheng et al. (1997). However, mean patch size was variable over time with more moderate declines in wooded area in this study. Similar variability over time for mean patch size was reported for landscapes with increasing forest area in Wisconsin (Sharpe et al. 1987) and Quebec (Pan et al. 1999).

Mean patch size and number of patches, whether expressed directly or as an index or density, may be greatly affected by small patches that together represent only a minor fraction of total wooded area. In landscapes dominated by one large forest patch, these are not the best metrics to characterize temporal changes or spatial differences in landscape structure.

Perimeter-to-area ratio showed inconsistent effects with deforestation between the temporal and spatial comparisons within this study. A clear negative relationship between perimeter-to-area ratio and proportion of landscape wooded was evident for the spatial comparisons of the 1975/76 and 1990 data, consistent with the findings of Burgess and Sharpe (1981) for a Wisconsin landscape. However, this relationship was not consistently evident with temporal data for individual landscapes in this study. Given the variations in response, perimeter to area ratio may not be a useful metric for monitoring landscape structure or establishing landscape management objectives.

3.6.2 Changes in Landscape Spatial Structure with Deforestation

The results of this study provide evidence that fragmentation compounded the negative effects of absolute declines in wooded area on the spatial structure of these landscapes. Temporal comparisons revealed that fragmentation associated with deforestation caused decreases in largest patch size that exceeded what would occur through patch shrinkage proportional to the wooded area loss across the landscape. This finding was corroborated in the spatial comparisons, which revealed a negative relationship between largest patch size index and proportion of land area wooded. For these landscapes, which are between 30% and 80% wooded, the slope of the linear relationship between largest patch size index and proportion of land remaining wooded indicated that deforestation had disproportionately large impacts on size of the largest wooded patch. Wickham et al. (1999) reported similar findings for landscapes with less than 70% anthropogenic cover in the mid-Atlantic region of the eastern United States.

Boreal deforestation associated with agricultural land use transforms previously contiguous forests into a series of forest fragments. Several examples from boreal regions illustrate that remnant forest patches differ ecologically from contiguous forests in terms of ecosystem structure, composition and process.

Hobson and Bayne (2000) studied avian communities in remnant forest/agricultural matrix landscapes in the southern boreal mixedwood region of western Canada. The richness of forest-interior bird species was less than expected in comparison with contiguous forest landscapes. Forest-interior long-distance migrants such as Swainson's Thrush (*Catharus ustulatus*), Black-throated Green Warbler (*Dendroica virens*) and Bay-breasted Warbler (*Dendroica castanea*) were virtually absent from fragmented landscapes, yet were common in contiguous forest. In comparison to birds in contiguous forests, birds in forest fragments exhibit lower daily nest survival and higher cowbird parasitism. Kurki et al. (2000) reported a negative correlation between breeding success for Black Grouse (*Tetrao tetrix*) and Capercaillie (*Tetrao urogallus*) and fragmentation of boreal forests by farmland in Finland. Generalist predators, which occur in great abundance in boreal forests fragmented by agriculture, may reduce the breeding success of forest birds (Bayne and Hobson 1997, Bayne and Hobson 2000, Kurki et al. 2000).

In a study of fire-frequency in the boreal mixedwood region of western Canada, Weir et al. (2000) reported that forest clearance in a region of agriculture settlement reduced the probability of wildfire spread and exerted an influence on fire frequency extending tens of kilometers into an adjacent contiguous forest region.

These examples demonstrate the importance of contiguous forest cover for the conservation of forest wildlife and perpetuation of ecosystem processes in boreal landscapes. Real landscapes can rarely be categorized into only two mutually exclusive groups: contiguous forest and fragmented forest (Trani and Giles 1999). However, if the largest wooded patch increases in size, a landscape moves toward the contiguous end of the contiguous-fragmented continuum. Thus, the size of the largest patch serves as a valuable indicator of the state of landscape fragmentation.

3.6.3 Changes in Landscape Spatial Structure with Reforestation

Reforestation of agricultural land, proposed to offset carbon emissions from fossil fuel burning, would increase total wooded area within these landscapes. Whether such reforestation reverses the process of fragmentation associated with historic deforestation and contributes to ecological restoration depends, in part, on the spatial distribution of reforested areas. Simulated reforestation with the dispersed approach resulted in little change in the size of the largest patch and the degree of coherence for all landscapes. Adding additional small wooded patches in these landscapes stalled but did not reverse the trajectory of change in landscape spatial structure.

The contiguous approach led to increases in largest patch size and coherence for all landscapes. Expansion of large wooded patches did initiate reversals of the trends observed with historic deforestation and fragmentation. The magnitudes of these reversals were small relative to the magnitude of the changes observed with deforestation over preceding decades. This difference in magnitude reflects the 2% reforestation target over a 10 year time period, which compares to the observed losses of wooded area that ranged from 3 to 21% over the 14 to 15 years prior to 1990.

The simulation presented, with randomly located reforestation patches of fixed dimensions, represents a simplification of any actual reforestation program. Nonetheless, if reforestation proceeds in these landscapes with dispersed small patches, it is unlikely that the trajectory of degeneration in landscape spatial structure can be reversed. Management measures applied in a diffuse manner, such as subsidies to landowners for voluntary conversion of agricultural fields to forests, might further increase the number of small patches and result in no change to the degree of forest fragmentation. Proliferation of small patches provides little management benefit (Forman 1995).

More geographically strategic reforestation actions designed to expand the size of larger patches could begin to reverse the fragmentation that has occurred in these landscapes due to historic deforestation. Examples of such measures include increasing the size of provincial forests or protected areas. Public land agencies have successfully maintained wooded cover (see Chapter 2) and native biodiversity within this region over recent decades. Alternatively, funding could be provided for private conservation organizations to acquire and reforest areas adjacent to public forest lands or other large forest patches. Reforestation that expands the largest forest patches could provide supplementary benefits beyond carbon sequestration by reversing the process of fragmentation that has impaired forest wildlife and ecosystem processes. Only vigilant planning will provide society with the benefits derived from large patches of native vegetation (Forman 1995).

3.7 Chapter Summary

Landscape spatial structure was analyzed for several boreal landscapes in central Saskatchewan that exhibited persistent deforestation over three decades prior to 1990. Landscape metrics were utilized for temporal and spatial comparisons of wooded area data from digital topographic map data. Largest patch size and degree of coherence exhibited consistent temporal declines as deforestation proceeded in each landscape. Two related metrics, largest patch size index and effective mesh size, decreased as the proportion of land area wooded declined when the four landscapes were compared within fixed time periods. Temporal and spatial analyses yielded evidence that fragmentation exacerbated the effects of absolute wooded area losses. Deforestation disproportionately reduced the sizes of the largest patches, a finding with important management implications. Efforts to reverse deforestation in these landscapes would contribute to ecological restoration if priority is placed on expanding large patches.

Simulated reforestation with dispersed small patches resulted in little change in fragmentation as measured with landscape metrics. Simulated reforestation with single large patches contiguous to protected forests initiated a reversal of the historic trends for largest patch size and degree of coherence.

Increasing the size of provincial forests or protected areas, or restoring forest cover adjacent to them, would reverse the process of fragmentation that has impaired forest wildlife and ecosystem processes.

4. CARBON DENSITIES FOR ECOSYSTEM COMPONENTS WITH IMPLICATIONS FOR SAMPLING EFFORT

4.1 Literature Review

Organic carbon is distributed among various ecosystem components: living and non-living, aboveground and belowground, ephemeral and persistent. From the perspective of sampling efficiency, the relative carbon content of various ecosystem components should play an important role in the allocation of sampling effort.

Assessments of terrestrial carbon stocks or carbon densities frequently focus on live aboveground tree biomass (Botkin and Simpson 1990, Isaev et al. 1995) or soil organic carbon to a fixed depth (Homann and Grigal 1996, Pennock and van Kessel 1997b). For southern boreal deciduous or mixedwood regions of Saskatchewan, there are relatively few estimates of ecosystem carbon density that include live and dead overstory and understory vegetation as well as several depth increments for soils.

Ellert and Bettany (1995) reported soil organic carbon for aspen forests, pastures and cultivated sites in the Gray Soil Zone. Soil organic carbon to 18 cm depth ranged from 45 to 56 Mg C ha^{-1} for all sites. Vegetation carbon was not estimated. Gower et al. (1997) estimated aboveground carbon densities at the BOREAS Old Aspen site in the southern portion of Prince Albert National Park. Carbon densities for live and standing dead trees were 83 and 10 Mg C ha^{-1}, respectively. Downed trees were not sampled. Understory carbon density was less than 1 Mg C ha^{-1}. Carbon density for the LFH layer was 19 Mg C ha^{-1}. Gower et al. (1997) reported unresolved uncertainty regarding two widely varying carbon density estimates for mineral soils to 70 cm depth at the site: 36 Mg C ha^{-1} and 90 Mg C ha^{-1}.

Halliwell and Apps (1997a, 1997b, and 1997c) estimated carbon densities for the LFH layer, soil horizons and live trees at a BOREAS auxiliary site along the southern edge of Prince Albert National Park. Biomass density for live standing trees was estimated as 93 Mg ha^{-1}. Carbon density was not measured for standing dead trees or understory vegetation. Estimates for woody detritus on the ground ranged from 3 to 8 Mg C ha^{-1}. Soil organic carbon was estimated as 82 Mg C ha^{-1} for 8 cm LFH plus 40 cm mineral soil (A and B horizons).

Pennock and van Kessel (1997b) reported soil organic carbon to 45 cm for forest sites and agricultural sites in central Saskatchewan. Carbon density was 57 Mg C ha^{-1} for mature mixedwood forest sites in Gray Luvisolic landscapes. Soil carbon density was 75 and 116 Mg C ha^{-1}, respectively, for a cultivated site and forested pasture site in the Black Soil Zone. Aboveground biomass carbon was not sampled at any of the sites.

Sulistiyowati (1998) reported mean biomass densities of 135 Mg ha^{-1} for live trees and 52 Mg ha^{-1} for dead trees at mixedwood stands in the Prince Albert Model Forest. Soil organic carbon was not measured.

4.2 Objectives

Quantifying the carbon density in both live and dead vegetation as well as soils will aid efforts to quantify organic carbon stocks for forest and agricultural ecosystems within boreal regions. One of the aims of field sampling in 2001 was to assess the vertical distribution of organic carbon within forest and agricultural ecosystems. The objectives for Chapter 4 are (i) to estimate the carbon densities for ecosystem components within a forest site, a pasture site, and a cultivated site, and (ii) to develop a sampling strategy for efficient estimation of ecosystem carbon densities for these three land use types.

From August to October 2000, I intensively sampled one forest site, one pasture site and one cultivated site within the Waskesiu Hills landscape described in Chapter 2. In Chapter 4, I report the carbon densities for various ecosystem components and recommend which of these should be included in more extensive sampling scheduled for a second field season.

4.3 Site Selection and Site Descriptions

Three sites were selected for sampling within Township 53, Range 3, West of 3rd Meridian. The Sugar Creek forest site (53° 35' 03" N, 106° 23' 10" W) was vegetated by trembling aspen and balsam poplar that regenerated naturally following a forest fire in 1940 (Weir 1996). This site is within Prince Albert National Park, and has been protected from logging, wildfire and domestic animal grazing since stand establishment.

The Cookson pasture site (53° 34' 57" N, 106° 23' 48" W) is within the Cookson Community Pasture, which was established in 1949 (Rump and Harper 1980). Examination of air photos revealed that the site was cleared between 1963 and 1968. After clearing it was seeded to non-native species such as smooth brome (*Bromus inermis* Leyss.), alfalfa (*Medicago sativa* L.), and Kentucky bluegrass (*Poa pratensis* L.).

The Larsen cultivated site (53° 32' 28" N, 106° 23' 13" W) is located on private land 36 km north of Shellbrook, Saskatchewan. It was cleared approximately seven decades ago and has been under continuous cultivation for production of small grains and oilseeds. The Larsen cultivated site was seeded to canola (*Brassica napus* L.) in autumn 1999 and was harvested in 2000 prior to the sampling date.

Land use and present vegetation differed among the three sites, but the sampling locations were selected based on soil inventory maps to minimize differences in soil taxa, texture, parent materials and landform morphology. The soils at all study sites were mapped as a mixture of Waitville Orthic Gray Luvisols, Waitville Dark Gray Luvisols and Whitewood Orthic Dark Gray Chernozems with loam or sandy loam textures on hummocky glacial till landforms (Padbury et al. 1978, Saskatchewan Land Resource Centre 1997). The three study sites were within close geographic proximity to minimize differences in historic vegetation, climate or other environmental factors.

4.4 Materials and Methods

The overall sampling approach was to stratify the ecosystem into discrete components and to estimate the carbon densities of each. A 2.3 ha area was selected for systematic sampling at each site. Systematic grid sampling was used to ensure comparability to the results of previous research in the Gray Soil Zone and in the Black Soil Zone of Saskatchewan (Pennock and van Kessel 1997b). A square grid with 49 sampling points (7 x 7, Figure 4.1) was used at each site. The 25 m spacing provided a degree of independence between adjacent sample points. For each of the 49 points, samples or measurements were collected to estimate carbon densities within several soil and vegetation strata.

4.4.1 Vegetation Sampling and Analysis

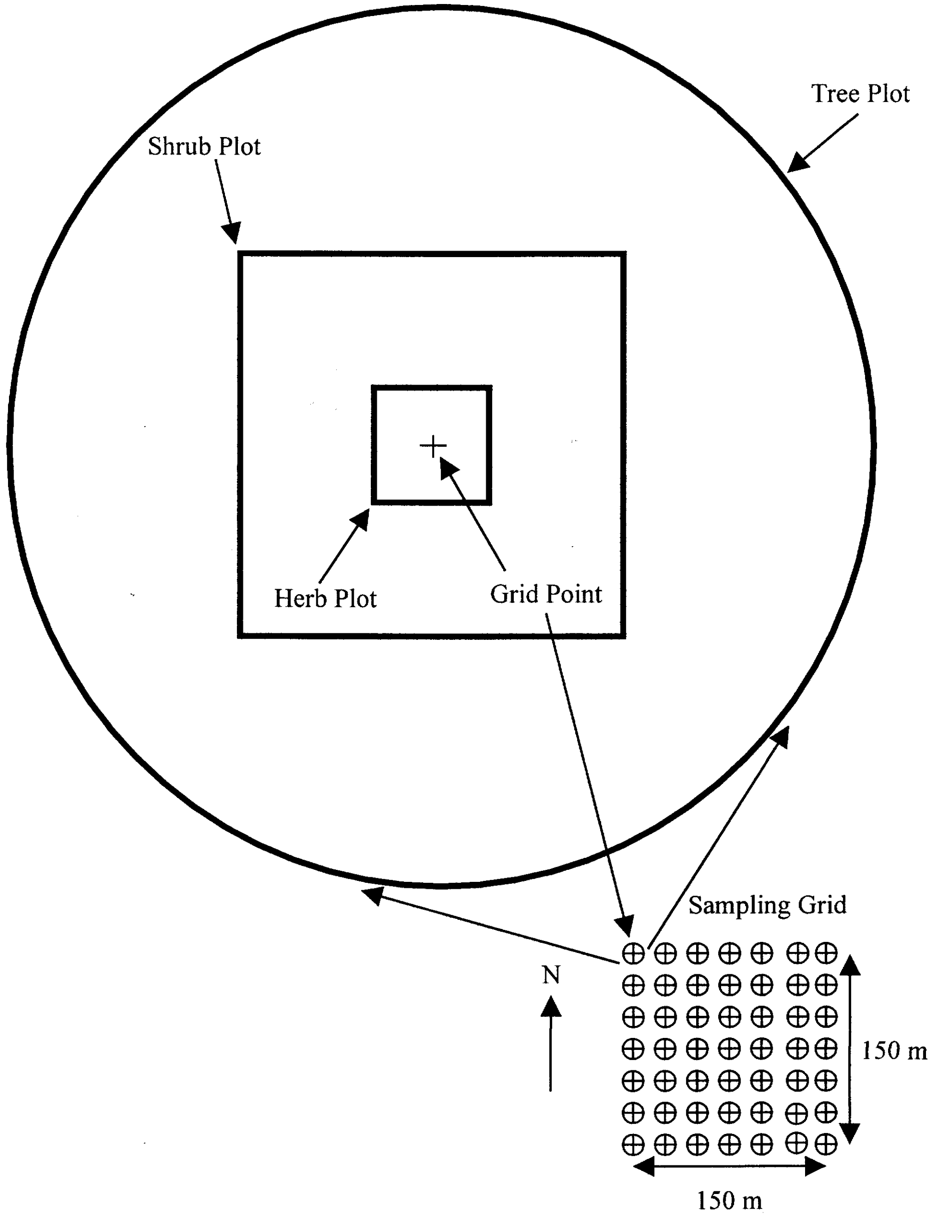
The sampling strategy differed for live and dead aboveground vegetation biomass at the forest site. Live aboveground biomass was divided into height strata. All stems rooted within the plot were included. Dead aboveground biomass was divided into diameter strata. Stems that were rooted in the plot and standing vertically (±30°) were included in their entirety. The portion of down detritus lying within the plot boundaries was also included. Fine root biomass was sampled with soil organic carbon. Because sampling equipment used in the forest was unable to penetrate structural roots (diameter > 1 cm), coarse roots were not sampled.

At the forest site, carbon content was estimated for six aboveground vegetation strata: (i) live vegetation ≤ 0.5m in height; (ii) live vegetation between 0.5 and 4.0 m in height; (iii) live vegetation ≥ 4.0 m in height; (iv) detritus ≤ 2 cm diameter; (v) detritus between 2 cm and 10 cm diameter; and (vi) detritus ≥ 10 cm diameter.

A nested plot design was used for live vegetation sampling at the forest site (Figure 4.1). At each sampling grid point, one 0.25 m^2 square quadrat (herb plot) was clipped to collect biomass of stems less than or equal to 0.5 m in height, and one 4.0 m^2 square quadrat (shrub plot) was clipped to collect biomass of stems greater than 0.5 m in height but less than 4 m in height. Biomass samples were weighed wet and subsampled if necessary. Biomass samples or subsamples were oven dried to constant weight at 60 °C.

At the same grid point, a 100-m^2 circular plot (tree plot) was used to census stems greater than or equal to 4.0 m in height. For each live stem, diameter at breast

Figure 4.1 Systematic grid of 49 sampling points (bottom) with exploded view of nested plots used at each sample point (top) at the Sugar Creek forest site. Square quadrats for herb and shrub strata (0.25 m^2 and 4.0 m^2 respectively), and the circular plot for tree strata (100 m^2) are not shown to scale.



height (1.3 m) and height were measured using a diameter tape and Haga altimeter. Live tree strata biomass was estimated using species-specific diameter-height-biomass relationships (Evert 1985).

Live vegetation biomass density was estimated as the sum of herb strata, shrub strata and tree strata biomass (dry-weight) after all were converted to Mg ha^{-1} units. Biomass-to-carbon conversion factors ranging from 0.45 to 0.5 have been used for boreal or boreal transition regions (e.g. Botkin and Simpson 1990, Johnston et al. 1996, Alexeyev et al. 1995). I estimated aboveground live vegetation C density as the product of biomass density and the 0.5 conversion factor used by Alexeyev et al. (1995), Krankina and Harmon (1995) and Gower et al. (1997).

Oven-dry biomass of woody detritus was measured for the 0- to 2-cm and 2- to 10-cm diameter strata within the herb and shrub plots respectively by collecting dead woody material and drying it to a constant weight at 60°C. The oven-dry biomass for the largest diameter strata was estimated by measuring volumes and converting to biomass using wood density estimates. Volumes of standing dead trees were estimated from dbh and height measurements using the shape of a cylinder for the portion of the tree at or below 1.3 m in height and the shape of a cone for the portion of the tree above breast height:

$$V = \pi r^2 h \text{ for } h \leq 1.3 \text{ m} \tag{4.1}$$

or

$$V = [\pi r^2 (1.3 \text{ m})] + [\pi r^2 (h - 1.3 \text{ m})/3] \text{ for } h > 1.3 \text{ m} \tag{4.2}$$

where V = volume (m^3),

r = 0.5 dbh (m), and

h = height (m).

A density of 0.36 Mg m^{-3} was used to convert volume to oven-dry mass. This density was obtained by taking the mean of reported densities for trembling aspen (Gonzolez 1990) and reducing it by 10 %, a correction factor used by Alexeyev et al. (1995) to take into account decay. This density estimate is close to the 0.382 Mg m^{-3} value reported for the largest size class (5- to 7-cm diameter) of dead and downed *Populus* round-wood in Alberta and the Northwest Territories (Nalder et al. 1997). It also falls within the range of densities for woody detritus estimated by Halliwell et al. (1995) and Harmon et al. (2000) for species of Canadian and Russian boreal forests respectively.

Krankina et al. (1999) reported a carbon fraction very close to 50% for oven-dry woody detritus for boreal species in Russia. Total woody detritus carbon was estimated as 50 % of the sum of oven-dry biomass estimates for the three diameter strata after all were converted to Mg ha^{-1} units.

The six strata for aboveground vegetation were collapsed into a single stratum at the pasture and cultivated field due to the lack of woody material and the lower stature of vegetation. At these agricultural sites, one 0.5-m^2 square quadrat (herb plot) was placed at each grid point. All aboveground vegetation (live or dead) that was rooted within the quadrat was clipped and then dried at 60°C to a constant weight. Aboveground vegetation carbon was estimated as 50% of oven-dry biomass.

4.4.2 Soil Sampling and Analysis

At all three sites, soil organic carbon was stratified into three strata: (i) 0- to 15-cm depth; (ii) 15- to 30-cm depth; and (iii) 30- to 45-cm depth. These samples included fine roots.

For the forest site, a manual fixed volume corer (4.0 cm radius) was used to extract soil samples. For the agricultural sites, sampling was carried out using a truck-mounted hydraulic punch (2.75 cm radius). In each case, samples of known volume were taken from the three depth increments. The 0- to 15-cm depth increment for the forest site included the LFH layer.

Each soil sample was air-dried and weighed. Bulk density was calculated from the air-dry mass for each 15-cm depth increments of soil that was collected in the field using the manual or truck-mounted corers. Soil samples (including LFH) were ground to pass through a 2 mm sieve. SOC were determined on the soil samples using a LECO CR-12 Carbon Determinator (LECO, St. Joseph MO). Mass fraction for organic carbon was determined by combustion at 840 °C, a method described by Wang and Anderson (1998). Soil organic carbon was estimated as:

$$SOC_a = CF_a \cdot BD_a \cdot K \qquad (4.3)$$

where SOC_a is soil organic carbon for depth increment a (Mg C ha^{-1}),

CF_a is the organic carbon fraction for depth increment a (proportion C),

BD_a is the bulk density of the sample from depth increment a (g cm^{-3}),

K is a constant = [1 500 m^3 ha^{-1}][1 000 000 cm^3 m^{-3}][0.000 001 Mg g^{-1}]

The first term in the formula for the constant represents the volume of a 15-cm increment of soil across 1.0 ha of land.

4.5 Results and Discussion

4.5.1 Carbon Densities

Carbon densities for replicates within strata were not normally distributed, precluding the use of means and standard errors. Carbon content for strata are reported as median values with interquartile ranges in parentheses.

The carbon density for the forest site was 179 (52) Mg C ha^{-1} within aboveground biomass and soil to 45 cm depth (Table 4.1). This estimate can be compared to two other aspen stands in the southern portion of Prince Albert National Park: 158 Mg C ha^{-1} (aboveground biomass and soil to 70 cm depth) at the BOREAS old aspen site (Gower et al. 1997) and 135 Mg C ha^{-1} (overstory and soil to 61 cm depth) at the AIM-13 BOREAS auxiliary site (Halliwell and Apps 1997a, 1997b and 1997c).

The carbon density for live trees at the forest site was 53 Mg C ha^{-1}, which compares to 93 Mg C ha^{-1} and 47 Mg C ha^{-1} at the BOREAS old aspen and auxiliary sites respectively (Gower et al. 1997, Halliwell and Apps 1997b). The carbon content of live vegetation below 4 m in height was less than 1 Mg ha^{-1}, similar to the finding of Gower et al. (1997) at the BOREAS old aspen site.

Total woody detrital mass was 8 (9) Mg C ha^{-1}. The greatest magnitude of woody detritus occurred within the largest diameter class (Table 4.1). Gower et al.

(1997) reported 9.6 Mg C ha^{-1} for standing dead trees at the BOREAS old aspen site. Halliwell and Apps (1997c) reported approximately 6 Mg C ha^{-1} for down woody detritus at the BOREAS auxiliary site. Sulistiyowati (1998) reported carbon densities of 13 to 95 Mg C ha^{-1} for mixedwood stands in the Prince Albert Model Forest.

Table 4.1 Carbon density for ecosystem components at Sugar Creek forest site.

Strata	Median Carbon Density (Mg C ha^{-1})	Interquartile Range (Mg C ha^{-1})
Live Vegetation $\geq$ 4.0 m height	52.7	35.2
0.5 m height < Live Vegetation < 4.0 m height	0.5	0.7
Live Vegetation $\leq$ 0.5 m height	0.1	0.1
Woody Detritus $\geq$ 10 cm diameter	3.2	4.7
2 cm diameter < Woody Detritus < 10 cm diameter	2.0	4.4
Woody Detritus $\leq$ 2 cm diameter	0.8	1.0
Soil 0- to 15-cm depth increment	72.1	24.9
Soil 15- to 30-cm depth increment	24.7	23.5
Soil 30- to 45-cm depth increment	10.9	13.3

SOC at the forest site was concentrated in the 0- to 15-cm depth increment, which included the duff layer. The 72 (25) Mg C ha^{-1} observed for this increment compares to estimates of 19 Mg C ha^{-1} and 38.5 Mg C ha^{-1} for duff alone at the BOREAS old aspen and auxiliary sites respectively (Gower et al. 1997, Halliwell and Apps 1997c). Carbon density declines rapidly with depth. The 30- to 45-cm depth increment contained only 6 % of the total estimated organic carbon at the site.

The median carbon density for the pasture site was 103 (47) Mg C ha^{-1}. Carbon was concentrated in the soil, and decreased rapidly with depth (Table 4.2). The estimated soil organic carbon to 45 cm was 101 Mg C ha^{-1}, which was within the range of values reported for a forested pasture on glacial till in the Black Soil Zone (Pennock and van Kessel 1997b) and a cleared pasture in the Gray Soil Zone (Ellert and Bettany 1995). Aboveground vegetation accounted for less than 2% of the carbon measured at the Cookson Pasture site even though it was not grazed during the growing season prior to sampling in September.

Table 4.2 Carbon density for ecosystem components at the Cookson pasture site.

Strata	Median Carbon Density ($Mg\ C\ ha^{-1}$)	Interquartile Range ($Mg\ C\ ha^{-1}$)
Aboveground Vegetation	1.3	0.6
Soil 0- to 15-cm depth increment	67.7	23.7
Soil 15- to 30-cm depth increment	22.8	16.3
Soil 30- to 45-cm depth increment	7.5	6.7

The median carbon content for the Larsen cultivated site was 41 (20) $Mg\ C\ ha^{-1}$ to 45 cm (Table 4.3). This value is lower than the 75 $Mg\ C\ ha^{-1}$ to 45 cm reported for a cultivated site in the Black Soil Zone (Pennock and van Kessel 1997b) and the 45 to 47 $Mg\ C\ ha^{-1}$ to 18 cm values reported for the Gray Soil Zone (Ellert and Bettany 1995). Organic carbon at the Larsen cultivated site was concentrated in the 0- to 15-cm depth increment of soil. The carbon density of aboveground crop residue sampled after harvest at the cultivated site was less than 2% of the total estimated density of organic carbon.

Table 4.3 Carbon density for ecosystem components at the Larsen cultivated site.

Strata	Median Carbon Density ($Mg\ C\ ha^{-1}$)	Interquartile Range ($Mg\ C\ ha^{-1}$)
Aboveground Vegetation (Crop Residue)	0.8	0.3
Soil 0- to 15-cm depth increment	28.8	14.9
Soil 15- to 30-cm depth increment	6.2	5.3
Soil 30- to 45-cm depth increment	4.5	1.7

4.5.2 Implications for Sampling

It was as time-consuming to sample the lesser live vegetation and detritus within the herb and shrub plots as it was to sample the tree plots. Because the carbon densities for the lesser vegetation strata were 2 $Mg\ C\ ha^{-1}$ or less, neglecting these smaller plots would have resulted in very little change to the aboveground carbon density estimated for the forest site.

It is questionable whether it is worth sampling woody detritus within the tree plots given that the median value for carbon density was only 3 Mg C ha^{-1}. Continued sampling is recommended because this value was low relative to other woody detritus estimates for aspen or mixedwood stands in the same region (Gower et al. 1997, Halliwell and Apps 1997c, Sulistiyowati 1998). Given that the tree plots will already be in place to estimate live tree biomass, it requires little additional effort to continue sampling woody detritus greater than 10 cm in diameter.

It took the same amount of time to measure small trees and large trees. Fifty percent of the 1050 trees measured had a dbh of less than 10 cm. These small diameter trees accounted for less than 5% of estimated live tree biomass (approximately 2.6 Mg C ha^{-1}). Using a 10-cm dbh criterion rather than a 4 m height criterion for live trees would result in very little error in estimating total live tree biomass and considerable sampling time would be saved.

The low carbon density of the vegetation strata in cultivated and pasture ecosystems and the ephemeral nature of this biomass are probable reasons why aboveground plant carbon is ignored in many assessments of carbon densities at agricultural sites. Despite the low carbon density, continued sampling of aboveground carbon could be justified to facilitate comparisons of aboveground carbon density estimates among forests, pastures and cultivated sites. Sampling prior to harvest would provide a better estimate of maximum vegetation carbon within cultivated sites.

4.5.3 Estimates for Unsampled Ecosystem Components

Although an attempt was made to estimate carbon for as many ecosystem components as possible, some components were omitted. These include coarse root carbon, soil organic carbon at depths greater 45 cm, and organic carbon within mobile organisms such as vertebrate wildlife and insects.

Given the biomass estimate of 105 Mg ha^{-1} for the forest site, coarse root biomass (including belowground portions of stumps) can be estimated as 23 Mg ha^{-1} and 20 Mg ha^{-1} using the regression relationships provided by Kurz et al. (1996) and Cairns et al. (1997) respectively. These values are close to other field measurements or estimates: 19 Mg ha^{-1} for stump and root biomass at a trembling aspen stand with 98 Mg ha^{-1} aboveground biomass in northern Wisconsin (Ruark and Bockheim 1987); 20 Mg ha^{-1} for roots greater than 5 mm in diameter for a 50 year old pure aspen stand in Alberta (Peterson and Peterson 1992); and 21 Mg ha^{-1} for roots greater than 5 mm in diameter at the BOREAS old aspen site in Saskatchewan (Steele et al. 1997). Using a 0.5 conversion factor to carbon density, these estimates of coarse root biomass represent at least 10 Mg C ha^{-1}. Although coarse root carbon density is greater in magnitude than carbon densities for other strata recommended for sampling, attempting to measure it directly would require massive increases in sampling effort. In addition, the environmental impact of the sampling would be unacceptable within a national park.

Coarse roots were found only at the forest site. Neglecting to measure this ecosystem component will serve to underestimate the magnitude of the difference in carbon density between forest ecosystems and agricultural ecosystems.

Carbon deep within the soil profile was not sampled at either the forest or agricultural sites. Jobbágy and Jackson (2000) estimate that 75% of SOC within the first 100 cm of boreal soils is contained within the first 40 cm. Furthermore, the rate of organic carbon dynamics decreases as depth increases (Anderson 1995), and thus the effects of recent deforestation on carbon densities would be expected to be most evident within the top 45 cm.

Mobile organisms were also not sampled. Wiegert and Owen (1971) estimated that herbivores exploit only 2.5% of aboveground primary production in mature deciduous forests. Because of low ecological efficiencies between trophic levels in forest ecosystems, the magnitude of organic carbon within heterotrophic organisms would be minor in relation to the magnitude of organic carbon within autotrophic organisms.

In conclusion, measuring live and dead trees greater than 10 cm in diameter and sampling soils to a depth of 45 cm will provide reasonable estimates of organic carbon density for mature trembling aspen stands in central Saskatchewan. Biomass carbon density for live vegetation less than 4 m in height and woody detritus less than 10 cm in diameter were small enough in magnitude (2.0 Mg C ha^{-1} or less) that these ecosystem components can be ignored in future sampling. Sampling aboveground biomass and soils to a depth of 45 cm should provide equivalent estimates of carbon density for agricultural ecosystems in the same region. These methodologies facilitate comparisons between forest and agricultural ecosystems, but estimated differences in organic carbon densities should be corrected by approximately 10 Mg C ha^{-1} to account for forest coarse root biomass.

4.6 Chapter Summary

The ecosystem components with the highest organic carbon densities at the Sugar Creek forest site (in descending order) were the 0- to 15-cm depth increment of soils, live vegetation greater than 4 m in height, the 15- to 30-cm depth increment of soils, the 30- to 45-cm depth increment of soils, and woody detritus greater than 10 cm in diameter. The carbon densities for lesser live and dead vegetation were small enough in magnitude that they could be ignored in future sampling. Reasonable estimates of organic carbon density at similar sites could be acquired by sampling only live aboveground vegetation greater than 10 cm dbh, dead aboveground vegetation greater than 10cm in diameter, and soil organic carbon to 45 cm depth. Coarse root biomass can not be easily sampled, but can be estimated using regression relationships with aboveground biomass.

Organic carbon at the agricultural sites was concentrated in soils and declined with depth. Although the carbon density of aboveground vegetation was minimal, continued sampling at replicate sites will facilitate comparisons of aboveground vegetation among treatment groups (forest sites, pasture sites and cultivated sites).

5. RELATIONSHIPS BETWEEN TOPOGRAPHIC LANDFORM POSITION AND SOIL ORGANIC CARBON AT A FOREST SITE, A PASTURE SITE AND A CULTIVATED SITE IN CENTRAL SASKATCHEWAN

5.1 Literature Review

Boreal forest and grassland biomes will play an important role in future carbon dynamics between terrestrial ecosystems and the atmosphere (see, for example, Paustian et al. 1997, Sellers et al. 1997). Estimating the magnitude of current organic carbon stocks at broad spatial scales and monitoring changes in these stocks over time is difficult because carbon stores are sensitive to both broad-scale climatic and vegetation patterns (Whittaker and Likens 1975, Eswaran et al. 1993) and local factors such as soil texture (Bhatti and Apps 2000), topography (Schimel et al. 1985, Pennock and van Kessel 1997b), natural disturbances and land use (Burke et al. 1989, Kurz et al. 1995).

For the grassland biome, Pennock and Frick (2001) recommend that extrapolation of research results from finer to broader spatial scales will only be reliable if topographic variation in SOC is taken into consideration. Investigations of site-scale spatial pattern of SOC have been aided by a spatial soil-landform model that describes a replicable association between landform morphology and soil types of the Black soil zone of Saskatchewan (Pennock et al. 1994).

In the Black soil zone, native vegetation is described as aspen parkland; open grasslands interspersed with trembling aspen groves (Mitchell et al. 1944, Rowe 1959). Within this region, Pennock et al. (1994) described a pattern of Regosolic soils and thin Chernozemic soils on shoulders, Orthic Chernozems on low catchment footslopes, and Gleysolic soils on high catchment footslopes and depressional levels. Distinctive pedogenic regimes on different topographic positions occur because of differences in the rates of processes that control soil properties including the magnitude of SOC storage (Pennock and van Kessel 1997b).

In the Gray soil zone of central Saskatchewan, native vegetation is boreal mixedwood forest including coniferous and deciduous species such as white spruce and trembling aspen. Xiao (1987) reported minimal differences in SOC estimates among upper, mid and lower slope positions in a trembling aspen forest with Orthic Gray Luvisolic soils near Meeting Lake, Saskatchewan. In a landscape under white spruce, balsam fir and trembling aspen vegetation in Prince Albert National Park, Donald et al. (1993) described a relationship between soil properties and landform morphology similar to that observed by Pennock et al. (1994) in the Black soil zone. Orthic Gray Luvisols, with LFH layers less than 10 cm deep and well-leached Ae horizons generally greater than 45 cm in thickness, occurred on well drained uplands. Humic Luvic Gleysols, with litter layers greater than 15 cm deep and Ae horizons less than 40 cm in thickness, occurred in imperfectly drained lower slopes. Organic soils occurred on poorly to very poorly drained positions of the landscape where the water table remained within 1 m of the soil surface for most of the growing season. Pennock and van Kessel (1997a, 1997b) studied more than ten replicate landscapes under trembling aspen and white spruce vegetation within the Prince Albert Model Forest and failed to confirm the pattern observed by Donald et al. (1993). Gleysols were found in topographic positions with large local catchments, but a mix of Luvisols and Brunisols occurred on upper landform positions. Luvisols and Brunisols with different textures were highly interspersed in upland positions due to the complex mixture of glacial sediments at the study landscapes. These texture differences and soil orders did not show any association with landform morphology in non-depressional positions (Pennock and van Kessel 1997a).

A clear relationship between landform morphology and soil properties has been repeatedly described within landscapes of the Black soil zone, but observed in only one of numerous studies of soils in the Gray soil zone. No data on topography-SOC relationships are available for landscapes between the Black and Gray soil zones, which are dominated by Dark Gray Chernozemic and Dark Gray Luvisolic soils.

5.2 Objectives, Research Design and Hypotheses

The purpose of this chapter is to evaluate whether topographic landform position aids in the description of the spatial pattern for SOC storage and in the evaluation of the magnitude of carbon storage for a forest site, a pasture site and a cultivated site located within the Dark Gray Soil Zone.

The study used a comparative mensurative experimental design (Hurlbert 1984). The basic sampling approach was to compare soils among topographic positions within a forest site, a pasture site and a cultivated field. Detection of land-use effects on the relationship between topographic position and SOC requires the assumption that these relationships were the same at all sites prior to the initiation of agricultural activities.

The null hypothesis investigated was that no statistically significant differences in organic carbon storage would be observed across a range of topographic positions within sites.

5.3 Site Selection and Site Descriptions

Three study sites (Table 5.1) were selected within the Boreal Transition Ecoregion (Acton et al. 1998) of Saskatchewan, Canada. Boreal mixedwood species including trembling aspen, balsam poplar, white spruce and white birch, dominated upland sites in the region prior to agricultural development (Mitchell et al. 1950, Rowe 1959), but the latter two species were reduced in abundance between the 19th and the 20th century due to logging and frequent settlement fires (Mitchell et al. 1950, Weir and Johnson 1998). The study sites are near the northern extent of the Mixedwood-Parkland Ecodistrict (Kabzems et al. 1986) or Boreal Transition Ecoregion (Padbury and Acton 1994) and could have also included areas of aspen groves and grassland vegetation. The mean January, July and annual temperatures in the Boreal Transition Ecoregion are -20°C, 17°C, and 0°C respectively. Mean annual precipitation is 45 cm (Acton et al. 1998).

Table 5.1: Description of the Sugar Creek forest site, the Cookson pasture site and the Larsen cultivated site.

Study Site	Latitude and Longitude	Ownership and Land Use History	Soil Map Descriptors
Sugar Creek Forest	53° 35' 03" N 106° 23' 10" W	Protected from logging, grazing and cultivation by Parks Canada since the 1920s.	Orthic Gray Luvisolic and Dark Gray Luvisolic soils, hummocky glacial till with 9% maximum slopes[1]
Cookson Pasture	53° 34' 57" N 106° 23' 48" W	Managed as a community pasture by Saskatchewan Agriculture and Food since it was cleared in the 1960s.	Orthic Gray Luvisolic and Dark Gray Luvisolic soils, sandy loam to loam, hummocky with 10% maximum slopes.[2]
Larsen Cultivated	53° 32' 28" N 106° 20' 13" W	Cultivated by private farmers since it was cleared in the 1930s.	Orthic Dark Gray Chernozemic soils intermixed with Orthic Gray Luvisolic and Dark Gray Luvisolic soils, loam, hummocky with 10% maximum slopes.[2]

1. Padbury et al. 1978.
2. Saskatchewan Land Resource Centre 1997.

The Sugar Creek forest site, on the southern edge of Prince Albert National Park, is vegetated by a stand of trembling aspen and balsam poplar that regenerated naturally following a forest fire in 1940 (Weir 1996). The site has been protected from exploitation since stand establishment.

The Cookson pasture site is located within a community pasture. Dominant cover consists of smooth brome (*Bromus inermis* Leyss.), alfalfa (*Medicago sativa* L.), and Kentucky blue grass (*Poa pratensis* L.).

The Larsen cultivated site is located on private land 36 km north of Shellbrook, Saskatchewan. It was cleared approximately seven decades ago and since that time has been under continuous cultivation for production of small grains and oilseeds. The site was seeded to canola (*Brassica napus* L.) in the spring of 2000 and was harvested prior to the sampling date.

Although these three sites are currently characterized by differences in vegetation and land use, they were selected on the basis of similarities in ecological conditions. Each site occurred within a small geographic area (a single township) so as to minimize differences in historic climate and vegetation. Each site was situated within areas described as a mixture of Orthic Gray Luvisols, Dark Gray Luvisols and

Orthic Dark Gray Chernozems on local soil inventory maps (Table 5.1). Soil textures were mapped as loam or sandy loam (Padbury et al. 1978, Saskatchewan Land Resource Centre 1997). Landforms at all of the study sites were mapped as hummocky and the range in local elevation within sites was approximately 5 m.

5.4 Materials and Methods

A 2.3 ha site was selected at each location and sampled using a square grid sampling design with 49 sampling points (7 x 7 with 25 m spacing as described in Chapter 4).

At each sampling grid point, soil horizons were described and classified using the Canadian System of Soil Classification (Agriculture Canada Expert Committee on Soil Survey 1987). Soils were hand-textured in the field. For the forest site, a fixed volume corer was used to extract soil samples. For the agricultural sites, sampling was carried out using a truck-mounted hydraulic punch. In each case, samples of known volume were taken from three depth increments (0- to 15-cm, 15- to 30-cm and 30- to 45-cm. The 0- to 15-cm depth increment in the forest included the LFH layer. SOC was reported in Mg C ha^{-1} to 45 cm depth.

SOC was also reported as elemental mass in equivalent soil mass (Ellert and Bettany 1995). Use of elemental mass in equivalent soil mass has been recommended over elemental mass to a fixed depth when sampling includes sites of contrasting bulk densities (Ellert and Bettany 1995, Ellert and Gregorich 1996). The forest site has an LFH layer of low bulk density that is absent at the pasture and cultivated sites. Reporting elemental mass in equivalent soil mass minimizes the confounding effect of variation in the mass of soil samples either within or among the sites. An equivalent soil mass of 3500 Mg ha^{-1} was used as in the comparison of forest and agricultural sites by Ellert and Gregorich (1996). This soil mass allowed inclusion of carbon within the entire 0-15 cm increment and all or most of the 15- to 30-cm increment for all sample points, as well as a portion of the 30- to 45-cm increment for some forest and pasture sample points.

At each site, the topography of the sampling grid and the immediate surroundings was surveyed using a Total Station survey system. The topographical surveys were used to derive a Digital Elevation Model (DEM) of the elevation surface at each site. Point data from the Total Station survey were brought into Surfer (Golden Software Inc., Golden Colorado). A DEM with 10 m X 10 m cells was created by interpolation using kriging with the default linear variogram. Using the methodology of Pennock et al. (1987), a series of slope morphological and positional attributes were calculated for each 10 m by 10 m cell of the DEM, and these attributes were used to classify each cell into one of seven landform elements: slope segments with a defined range of slope curvature and gradient.

Specific dispersal areas were calculated for each grid cell using the DEMON model (Costa-Cabral and Burges 1994). Each cell was also evaluated as to whether it was depressional. The same threshold as used by Bedard-Haughn (2001), a specific dispersal area of less than 2.0 m^2 m^{-1}, distinguished depressional cells.

Because plan curvature was less pronounced in these study sites than in hummocky landscapes with more distinct circular knob and kettle landforms, the seven landform elements were combined with depressional status and collapsed into three topographic position classes. Gridpoints within DEM cells that were depressional or had concave profile curvature were grouped into the footslope class. The level class was comprised of all cells classed as linear (the absolute value of profile curvature was less than 0.1 $^{\circ}$ m^{-1}). Cells with convex profile curvature were grouped into the shoulder class. Each sample point was assigned the topographic position class of the DEM cell in which it was located.

Bulk density and SOC for soil samples was determined by the methods described in Chapter 4. Sand fraction was estimated by wet sieving a 25 g air-dried subsample through a 53 micron (No. 270) sieve. The sand retained on the sieve was then dried, weighed and expressed as a percentage of the air-dry sub-sample weight.

The distributions of bulk density, sand fraction and SOC were summarized using medians and interquartile ranges. Distribution-free statistical techniques were used due to skewness and low sample sizes. Differences were tested for statistical significance

(α= 0.10) using a multiple comparison extension of the Kruskal-Wallis test (Siegel and Castellan 1988). SYSTAT 8.0 (SPSS 1998) was used for statistical analysis.

5.5 Results

5.5.1 Characterization of Study Sites

The forest and pasture sites included one or more wetlands or former wetlands within the sampling grid (Figure 5.1). The cultivated site contained a draw that drained to a former wetland located beyond the sampling grid. Although mapped as hummocky, the study sites included relatively linear slopes and lacked the prominent circular knob and kettle forms often found in other hummocky glacial till deposits. Levels occurred most frequently and shoulders occurred least frequently at all sites. Relative to the other two sites, the forest had a more even distribution of the three topographic position classes (Table 5.2).

Table 5.2: Topographic position class frequencies within the Sugar Creek forest site, the Cookson pasture site and the Larsen cultivated site.

	Topographic Position Class		
Study Site	Footslopes (sampling point frequency)	Levels (sampling point frequency)	Shoulders (sampling point frequency)
Forest (n=49)	18	21	10
Pasture (n=49)	11	32	6
Cultivated (n=49)	14	31	4

The forest site had a relatively even frequency of Chernozemic, Gleysolic and Luvisolic soils within the sampling grid (Table 5.3). Soil profiles at two forest sampling points were disturbed due to animal activity or tree-fall and could not be classified. Chernozemic soils occurred at two-thirds of the sampling points at the pasture site with Gleysolic and Luvisolic soils occurring at a similar frequency at the remaining sampling points. At the cultivated site, Chernozemic soils occurred at 69 % of sampling points while Luvisolic soils occurred at 18% of sampling points. Brunisolic soils occurred at 4% and 12% of the sampling points within the forest and cultivated sites, respectively.

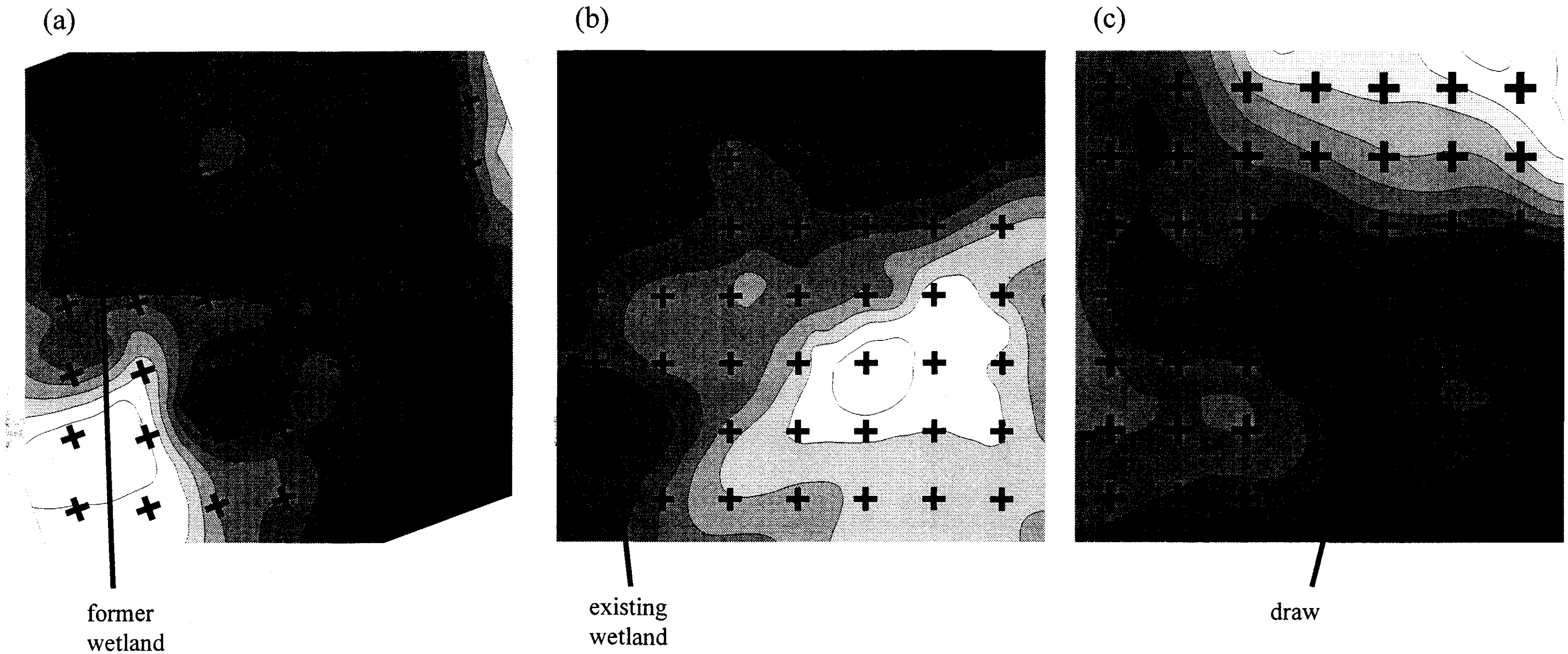


Figure 5.1: Sampling grid (crosses) and 0.5 m contours (lighter shades at higher elevations) for (a) the Sugar Creek forest site, (b) the Cookson pasture site and (c) the Larsen cultivated site. Sampling points are 25 m apart and the grids are oriented north-south (↕) and east-west (↔).

Table 5.3: Soil Order frequencies within the Sugar Creek forest site, the Cookson pasture site and the Larsen cultivated site.

	Soil Order			
Study Site	Brunisolic (sampling point frequency)	Chernozemic (sampling point frequency)	Gleysolic (sampling point frequency)	Luvisolic (sampling point frequency)
Forest (n=47)	2	16	17	12
Pasture (n=49)	0	30	9	10
Cultivated (n=49)	6	34	0	9

Texture was not distinctly sandier at sample points classed as Brunisols than at sampling points classed as Luvisols or Chernozems. They were classified as Brunisolic soils only because they lacked a Chernozemic A horizon or a Luvisolic Bt horizon. The Brunisolic Soil Order is the default in the Canadian System of Soil Classification (Agriculture Canada Expert Committee on Soil Survey 1987).

Surface soil horizons were sandy loam to silty loam at the forest site, loam to silty loam at the pasture site and sandy loam at the cultivated site. The ranges for sand fraction for 0- to 45-cm depths overlapped across the three sites, and median sand fraction differed by only 14% (Table 5.4). The sediments were not well sorted; however sand lenses of 10 to 40 cm thickness occurred in B horizons of 5, 9 and 15 sampling points in forest, pasture and cultivated sites respectively. Data for the three depth increments show little variation in sand fractions with depth (Figure 5.2). The extreme value (datum greater than 3 times the inter-quartile range outside of the 25th or 75th percentile) for the 0- to 15-cm depth at the forest site (circle in panel (a) of Figure 5.2) was a grid point with organic matter deeper than 15 cm in depth.

Median bulk density for the 0- to 45-cm depth was between 1.2 and 1.5 g cm^{-3} for the three sites (Table 5.4). Samples at the forest grid points included the LFH horizon, resulting in lower bulk densities for the 0- to 15-cm increment (Figure 5.3). Bulk densities were very similar at all sites for depth increments between 15 and 45 cm. The extreme values for the 0- to 15-cm depth at the pasture site (circles in panel (b) of Figure 5.3) were samples taken in a wetland that included a thick organic mat.

Table 5.4: Median values for sand fraction, bulk density and SOC (with inter-quartile range in parenthesis) within the Sugar Creek forest site, the Cookson pasture site and the Larsen cultivated site.

Study Site	Sand Fraction (% for 0- to 45-cm depth)	Bulk Density ($g\ cm^{-3}$ for 0- to 45-cm depth)	SOC ($Mg\ C\ ha^{-1}$ to 45 cm depth)	SOC ($Mg\ C\ ha^{-1}$ in equivalent soil mass*)
Forest	54 (11)	1.2 (0.19)	113 (48)	103 (33)
Pasture	49 (13)	1.4 (0.12)	101 (47)	89 (39)
Cultivated	63 (13)	1.5 (0.08)	40 (20)	34 (17)

Sample size is 49
* equivalent mass = 3500 Mg ha^{-1}

SOC values, for both the 0- to 45-cm depth and equivalent soil mass, were highly variable within each site. The magnitude of the inter-quartile range was approximately one-third to one-half of the magnitude of the medians (Table 5.4). SOC diminished rapidly with depth for all sites (Figure 5.4). Median SOC for the deepest third of the sample (30- to 45-cm) contained 9.7%, 7.4% and 11.1% of the median SOC for the entire sample (0- to 45-cm) for the forest, pasture and cultivated sites respectively. Positively skewed distributions for SOC in the 15- to 30-cm and 30- to 45-cm depth increments at the cultivated site resulted in several extreme values (circles in panel (c) of Figure 5.4). These high SOC sample points were almost exclusively footslopes and depressions (see next section below).

5.5.2 Differences Among Topographic Positions Within Sites

The only soil order strongly associated with specific topographic positions at these sites was the Gleysolic order (Table 5.5). At the forest and pasture sites, Gleysols occurred almost exclusively on footslopes and levels where depth to groundwater is minimal. Brunisols did not occur on footslopes. Both Chernozems and Luvisols were observed at all topographic positions at all sites.

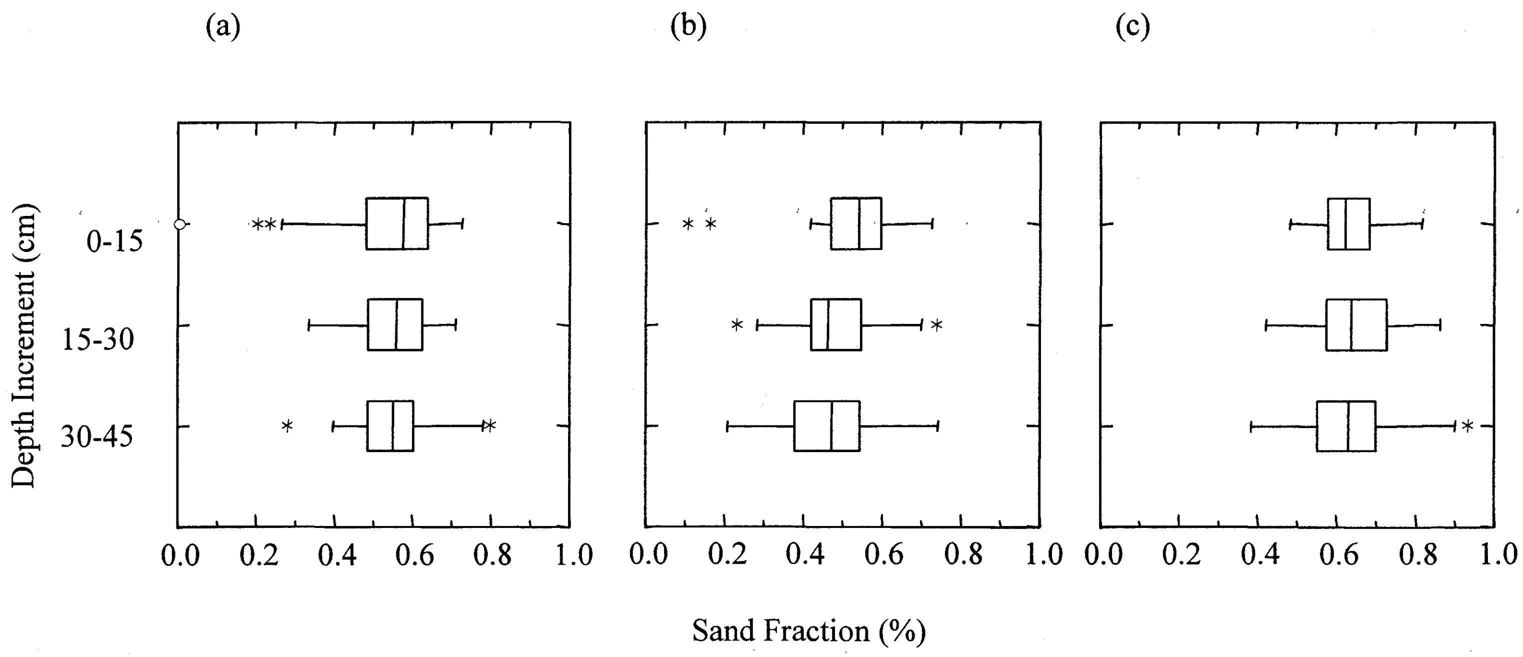


Figure 5. 2: Boxplots of sand fraction by depth increment for (a) the Sugar Creek forest site, (b) the Cookson pasture site and (c) the Larsen cultivated site.

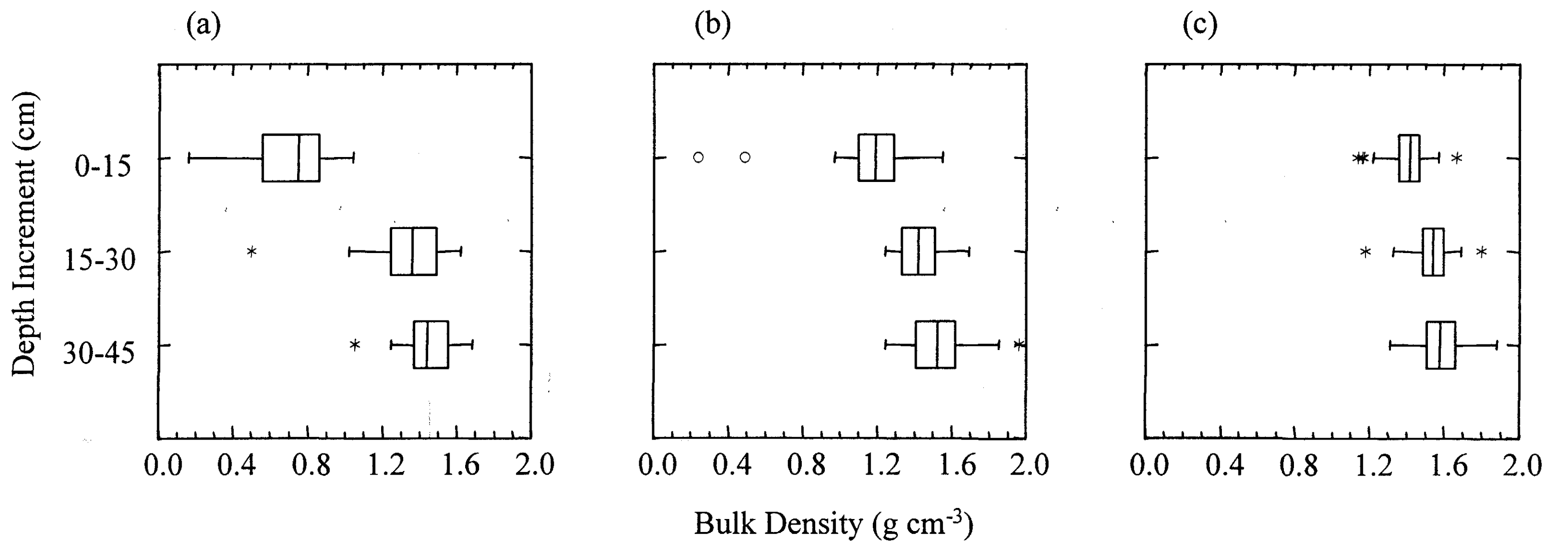


Figure5.3: Boxplots of bulk density by depth increment for (a) the Sugar Creek forest site, (b) the Cookson pasture site and (c) the Larsen cultivated site.

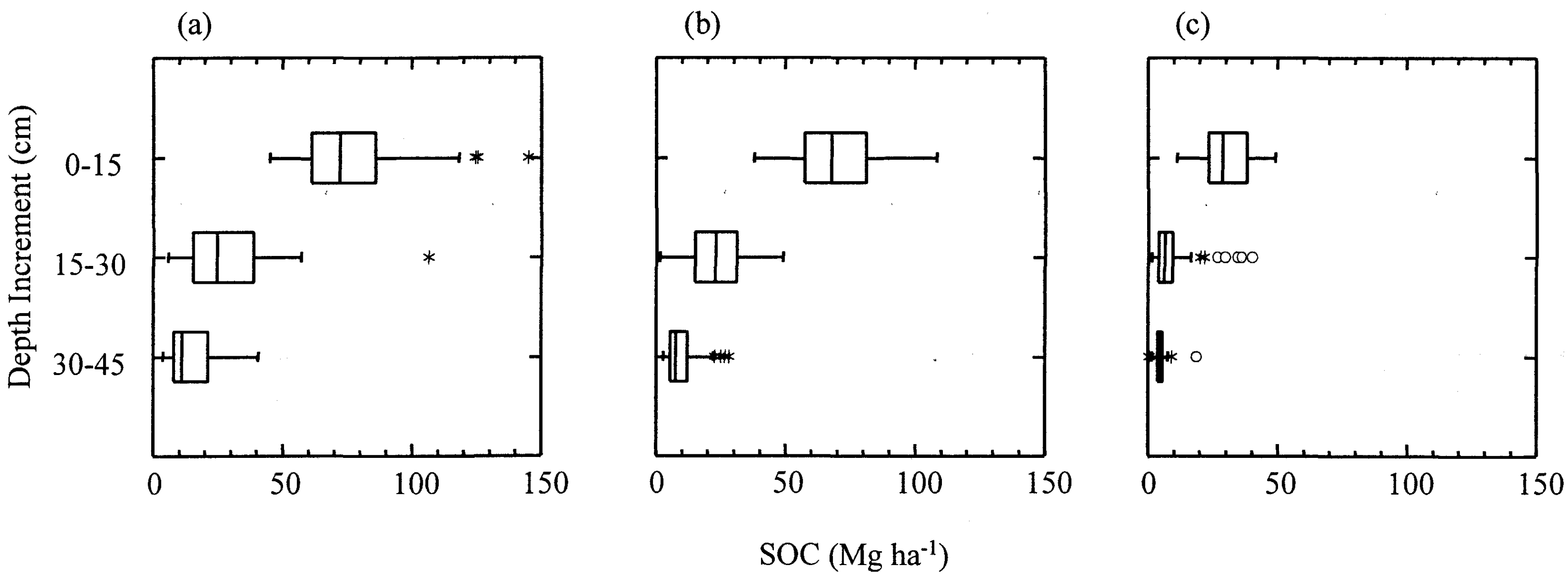


Figure 5.4: Boxplots of soil organic carbon by depth increment for (a) the Sugar Creek forest site, (b) the Cookson pasture site and (c) the Larsen cultivated site.

Table 5.5: Soil Order frequencies by topographic position classes for the Sugar Creek forest site, the Cookson pasture site and the Larsen cultivated site.

Study Site topographic position class	Soil Order Brunisolic (sampling point frequency and (%))	Chernozemic (sampling point frequency and (%))	Gleysolic (sampling point frequency and (%))	Luvisolic (sampling point frequency and (%))
Forest				
footslopes	0 (0)*	4 (9)	11 (23)	2 (4)
levels	1 (2)	8 (17)	6 (13)	5 (11)
shoulders	1 (2)	4 (9)	0 (0)	5 (11)
Pasture				
footslopes	0 (0)	5 (10)	3 (6)	3 (6)
levels	0 (0)	23 (47)	5 (10)	4 (8)
shoulders	0 (0)	2 (4)	1 (2)	3 (6)
Cultivated				
footslopes	0 (0)	13 (27)	0 (0)	1 (2)
levels	5 (10)	20 (41)	0 (0)	6 (12)
shoulders	1 (2)	1 (2)	0 (0)	2 (4)

* % of all soils at each site

For median sand fraction for the 0- to 45-cm depth, differences between shoulders and other topographic positions were 10% or less in magnitude and were not significant (p>0.10, Table 5.6). Differences in median sand fraction between footslopes and levels were small in magnitude. Median bulk densities for the 0- to 45-cm increment differed by less than 0.1 g cm^{-3} across topographic positions within sites (Table 5.6).

Table 5.6: Median values for sand fraction and bulk density (with inter-quartile range in parenthesis) at footslopes, levels and shoulders within the Sugar Creek forest site, the Cookson pasture site and the Larsen cultivated site.

Study Site	Topographic Position Class Footslopes	Levels	Shoulders
		Sand Fraction (%)	
Forest	51a (10)	56b (10)	59ab (10)
Pasture	49a (5)	47a (15)	51a (8)
Cultivated	62a (15)	63a (14)	73a (9)
		Bulk Density (g cm^{-3})	
Forest	1.2 (0.2)	1.2 (0.2)	1.2 (0.1)
Pasture	1.4 (0.2)	1.4 (0.1)	1.4 (0.2)
Cultivated	1.5 (0.1)	1.5 (0.1)	1.5 (0.1)

Sample sizes correspond to frequencies given in Table 5.2
Values in the same row with a different letter are significantly different (p<0.10)

At the forest site, median values for SOC to 45 cm varied by less than 6 Mg C ha^{-1} among topographic positions and differences were not significant ($p>0.10$, Table 5.7). At the pasture site, median SOC to 45 cm at shoulders was 13 and 20 Mg C ha^{-1} less than median SOC to 45 cm at footslopes and levels respectively, but the differences were not significant ($p>0.10$). Differences for median SOC in equivalent soil mass were also not significant among topographic positions for the forest and pasture site.

Table 5.7: Median values for SOC (with inter-quartile range in parenthesis) at footslopes, levels and shoulders within the Sugar Creek forest site, the Cookson pasture site and the Larsen cultivated site.

	Topographic Position Class		
Study Site	Footslopes SOC	Levels SOC	Shoulders SOC
		SOC (Mg C ha^{-1} to 45 cm depth)	
Forest	117a (56)	113a (50)	111a (21)
Pasture	104a (33)	111a (51)	91a (25)
Cultivated	47a (37)	38b (19)	20c (6)
		SOC (Mg C ha^{-1} in equivalent soil mass)*	
Forest	108a (44)	102a (49)	92a (19)
Pasture	89a (21)	95a (44)	80a (20)
Cultivated	40a (23)	31b (18)	17c (8)

Sample sizes correspond to frequencies given in Table 5.2
Values in the same row with a different letter are significantly different ($p<0.10$)
* Equivalent soil mass = 3500 Mg ha^{-1}

At the cultivated site, median values for SOC to 45 cm for footslopes were 9 Mg C ha^{-1} greater than levels, and median values for levels were 18 Mg C ha^{-1} greater than shoulders, with all differences significant ($p<0.10$). Median values for SOC in equivalent soil mass showed similar differences (Table 5.7). Differences among topographic positions were greatest at the 0- to 15-cm depth (Figure 5.5).

The vertical distribution of carbon between the 15- to 30-cm and 30- to 45-cm depth increments showed greater variation among topographic positions at the cultivated site in comparison to the pasture site and the forest site. At the cultivated site, many of the footslopes also had much higher carbon in the 15- to 30-cm increment than the 30- to 45-cm increment, whereas shoulders had similar carbon in these two depth increments (Figure 5.5).

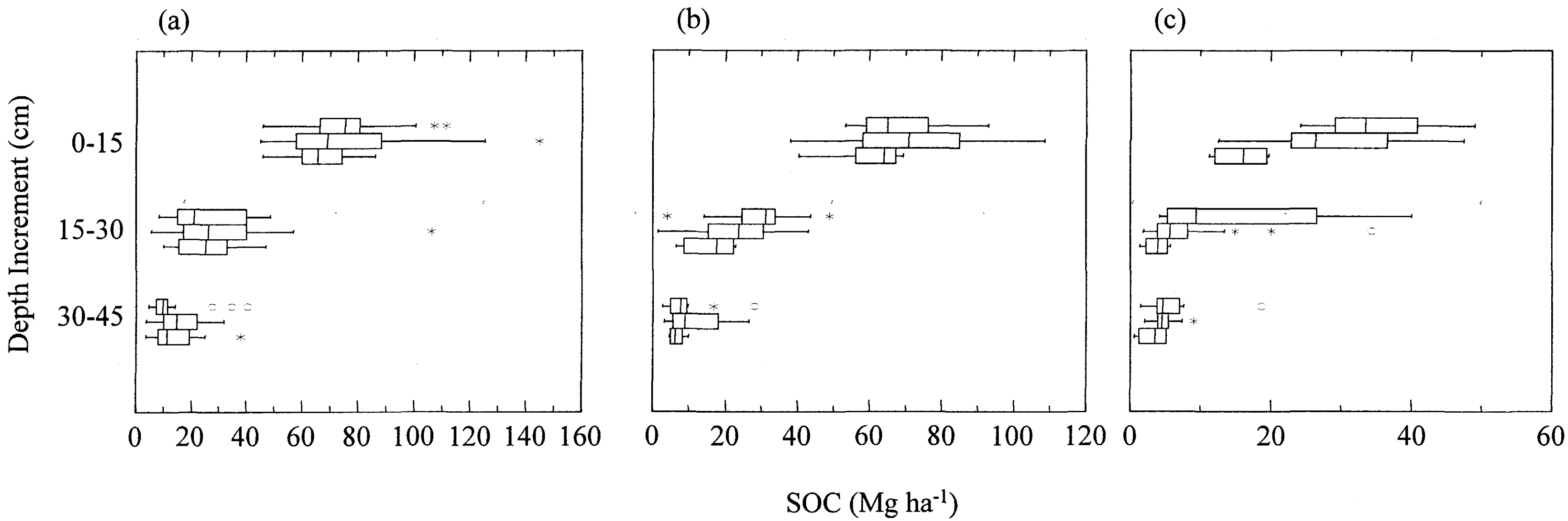


Figure 5.5: Boxplots of soil organic carbon by topographic position class and depth increment for (a) the Sugar Creek forest site, (b) the Cookson pasture site and (c) the Larsen cultivated site. The three boxplots at each depth represent footslopes (top), levels (middle) and shoulders (bottom). Note the variation in scales on the x-axes.

The pattern at the cultivated site of high SOC at footslopes and depressions is clearly evident in the first east-west row in the foreground and the north-south column at the right in Figure 5.6. The three gridpoints with lowest SOC values (SOC≤21 Mg C ha^{-1}) occurred at the most prominent shoulder positions. No such pattern exists at the forest site. As is evident in the second east-west row from the bottom, SOC can be either high (155 Mg C ha^{-1}) or low (83 Mg C ha^{-1}) in footslopes or depressions (Figure 5.7). As is apparent from the second north-south column from the right, SOC can be either high (128 Mg C ha^{-1}) or low (84 Mg C ha^{-1}) at shoulders.

5.5.3 Differences Among Soil Orders Within Sites

Differences among soil orders were also examined to evaluate the role of pedogenic factors in determining SOC at grid points within sites.

At the forest site, median SOC differed by only 11 Mg C ha^{-1} for the 0- to 45-cm depth and only 13 Mg C ha^{-1} for equivalent mass depth among the four soil orders (Table 5.8). Median SOC for soil orders in the pasture differed by 25 Mg C ha^{-1} for the 0- to 45-cm depth and 21 Mg C ha^{-1} for equivalent mass depth. The differences in median SOC levels for soil orders within forest and pasture were not significant ($p>0.10$). Gleysols at these two sites had much higher variability in SOC than the other soil orders, with inter-quartile ranges greater than 50 Mg C ha^{-1}.

At the cultivated site median SOC for Chernozems were 19 Mg C ha^{-1} greater for 0 to 45 cm depth and 14 Mg C ha^{-1} greater for equivalent mass depth than Brunisols, a difference that was highly significant ($p<0.05$). Differences between median SOC between Chernozems and Luvisols and between Luvisols and Brunisols were not significant ($p>0.10$).

At the forest and pasture sites, carbon content decreased with depth for both Chernozems and Luvisols (Figure 5.8). At the cultivated site, a similar pattern was observed for Chernozems, but carbon content for the 15- to 30-cm and the 30- to 45-cm depth increments were similar for Luvisols.

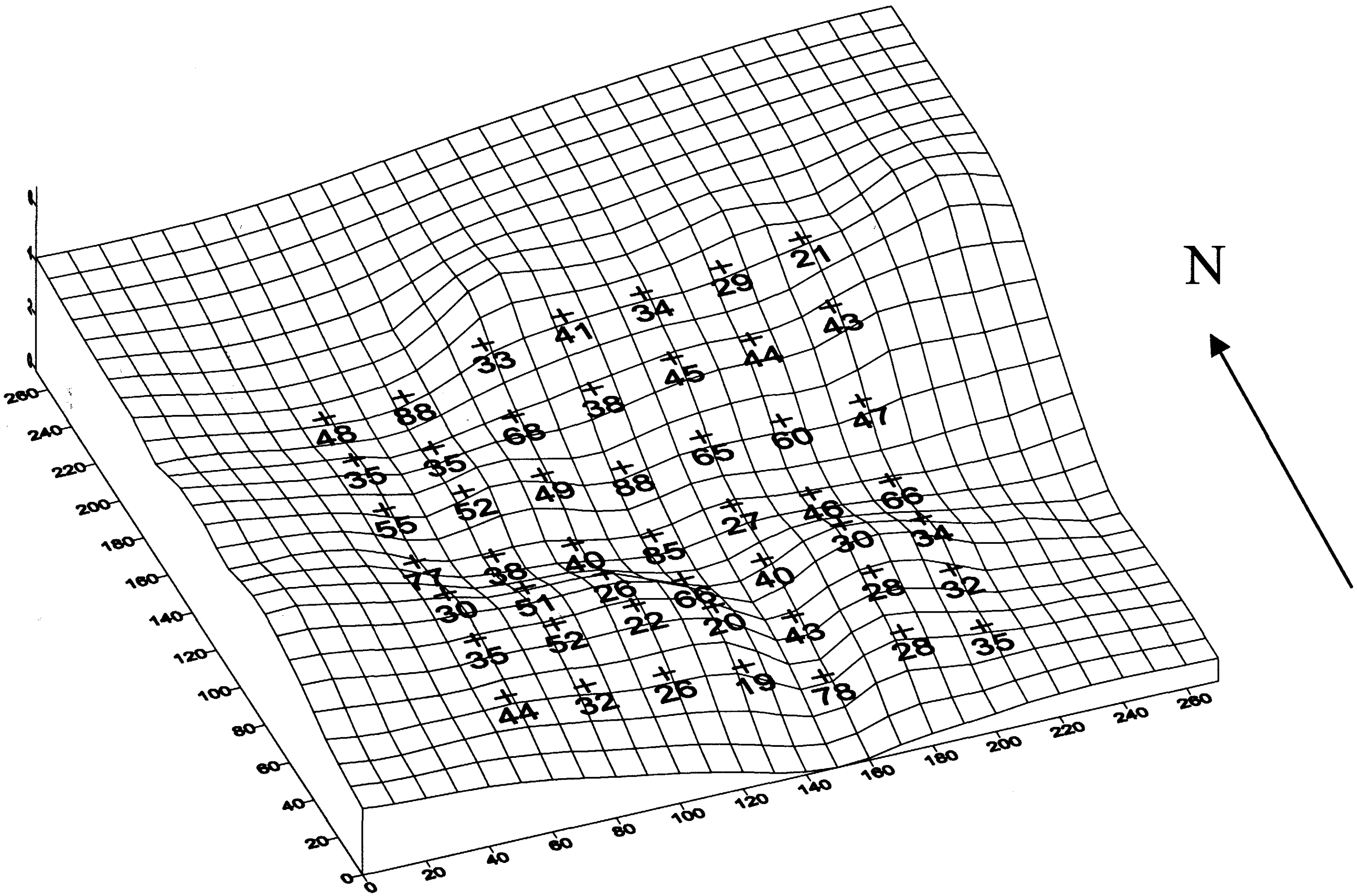


Figure 5.6: Spatial pattern of SOC at the Larsen cultivated site. Vertical topography exaggerated by a factor of ten. SOC at each sample point reported in MgC ha^{-1}.

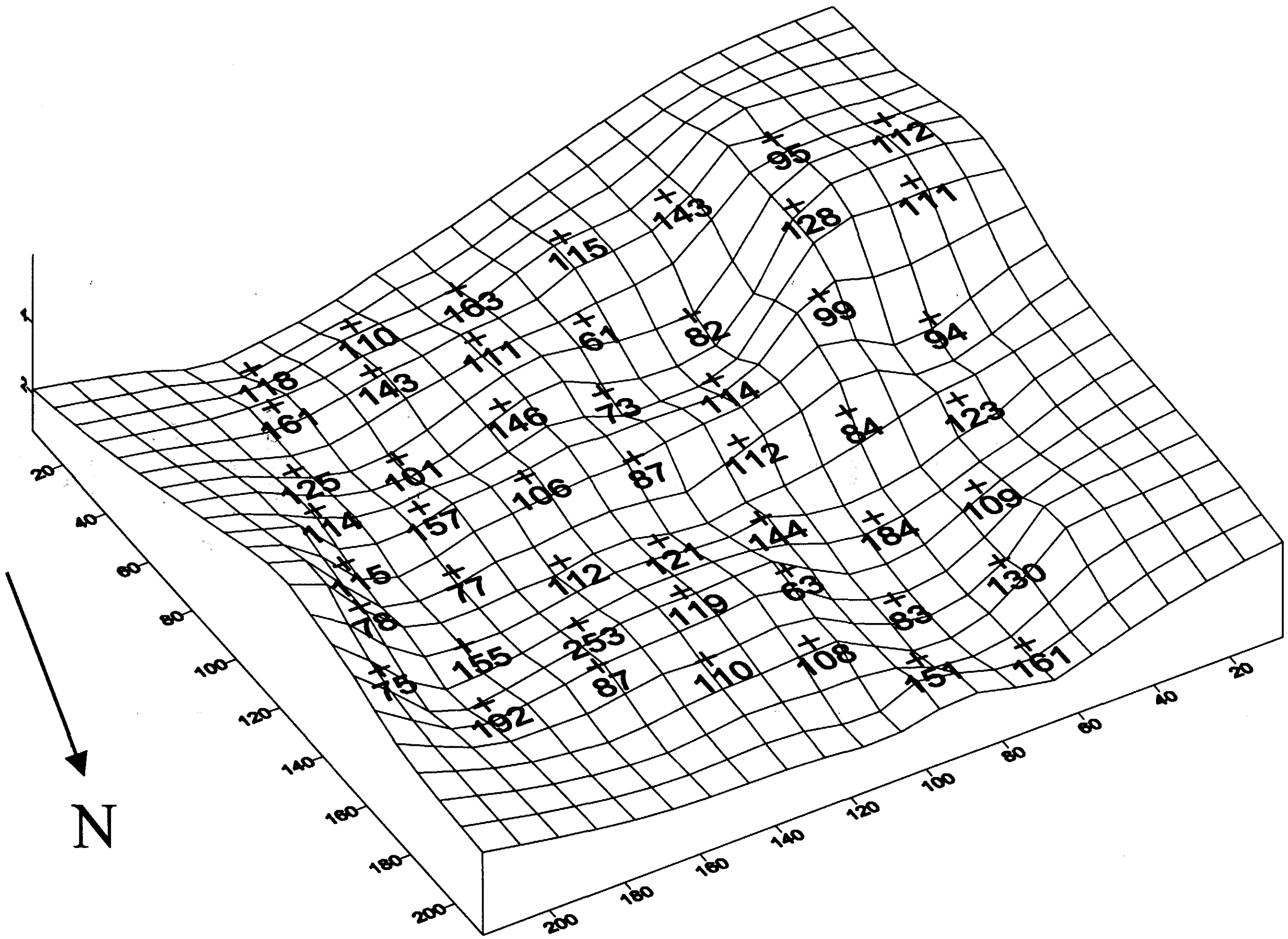


Figure 5.7: Spatial pattern of SOC at the Sugar Creek forest site. Vertical topography exaggerated by a factor of ten. SOC at each sample point reported in MgC ha^{-1}.

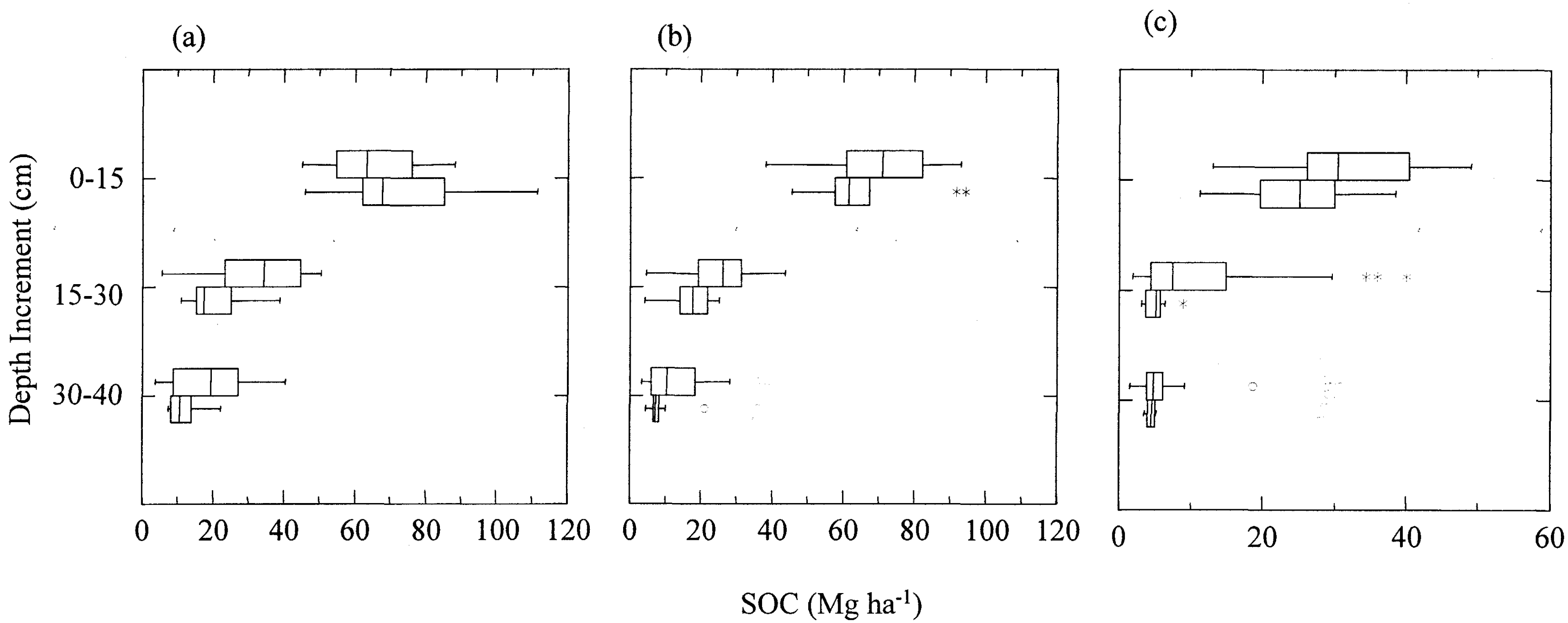


Figure 5.8: Boxplots of soil organic carbon by soil order and depth increment for (a) the Sugar Creek forest site, (b) the Cookson pasture site and (c) the Larsen cultivated site. The two boxplots at each depth represent Chernozems (top) and Luvisols (bottom).

Table 5.8: Median values for SOC (with inter-quartile range in parenthesis) at sample points with Brunisolic, Chernozemic, Gleysolic and Luvisolic Soil Orders within the Sugar Creek forest site, the Cookson pasture site and the Larsen cultivated site.

	Soil Order			
Study Site	Brunisolic SOC	Chernozemic SOC	Gleysolic SOC	Luvisolic SOC
		(Mg C ha^{-1} to 45 cm depth)		
Forest	104a (NA)	114a (36)	115a (71)	109a (29)
Pasture		113a (36)	101a (66)	88a (18)
Cultivated	27a (6)	46b (31)		35c (8)
		(Mg C ha^{-1} in equivalent soil mass)*		
Forest	96a (NA)	104a (29)	106a (65)	93a (32)
Pasture		97a (29)	87a (53)	76a (15)
Cultivated	23a (3)	37b (24)		27c (10)

Sample sizes correspond to frequencies given in Table 5.5
NA - not available due to small sample size
Values in the same row with a different letter are significantly different ($p<0.10$)
* Equivalent mass = 3500 Mg ha^{-1}

5.4 Discussion

A major assumption of the comparative mensurative experiment is that all 3 sites were the same prior to the imposition of land-use differences. In an effort to satisfy this assumption, the forest site, the pasture site and the cultivated site were carefully selected based on strong similarities for the major soil-forming factors (Jenny 1941) including climate, organisms, relief, parent materials and time.

The sites were located within 5 km of each other to ensure similarity in local climate and the duration of time available for pedogenesis (time since deglaciation). This close geographic proximity would also serve to minimize possible differences in historic vegetation. The presettlement vegetation for the township encompassing the three sites and the eight adjacent townships has been mapped as closed canopy mixedwood boreal forest (Weir et al. 2000). However, the mix of Luvisolic and Chernozemic soils common to all of the sites may indicate both grassland and forest vegetation occupied the region at various times in the past.

Local soil inventory maps were used to select sites with similar parent materials. The forest, pasture and cultivated sites were all located within map polygons labelled as glacial till landforms with similar textures and dominant soil taxa. Sand fraction data demonstrated that the sites were indeed similar in terms of texture.

Sites were also selected from map polygons labelled as hummocky with maximum slopes of 9% to 10%. Topographic survey data was used to show that local elevation differences within each of the 2.3 ha sites were approximately 5 m.

These otherwise similar sites exhibited different patterns for the spatial distribution of SOC. The null hypothesis investigated was that no statistically significant differences in organic carbon storage would be observed across a range of topographic positions within each site. No differences were observed in SOC across topographic position classes at the forest site. With agricultural land use however, a topographic pattern emerged. Shoulders had marginally lower SOC (but not statistically significant) at the pasture site. The null hypothesis was rejected for the cultivated site because shoulders had significantly lower SOC than levels and footslopes, and levels had significantly lower SOC than footslopes. Further results presented in Chapter 6 corroborate the presence of a difference in SOC between shoulders and footslopes at cultivated sites, and the absence of a difference for forest sites and pasture sites.

Comparative mensurative experiments are not designed to provide evidence of the underlying processes that cause observed differences in spatial patterns for soil properties (Pennock and Corre 2001). However, the similarities between the forest and the cultivated site for the basic soil-forming factors (climate, organisms, parent materials, relief and time) suggest that another factor may be responsible for the difference observed for SOC-topographic position relations.

Other studies have demonstrated that cultivation redistributes soil, including carbon, from convexities to concavities (see review by Govers et al. (1999)). The emergence of differences in SOC among topographic position classes in the cultivated site is entirely consistent with the process of tillage erosion and translocation. Extremely low SOC at the 15- to 30-cm depth increment at cultivated shoulders relative to other landform position classes suggests that the soil profile may have been truncated through tillage.

SOC did not vary with soil orders at the forest site. Carbon storage varied by 15 Mg C ha^{-1} or more among Soil Orders at the cultivated site but this is likely due to the relationship between topographic positions and soil orders. Tillage redistribution of carbon may also explain this observed difference between the forest and cultivated sites. At the cultivated site, Chernozems are the most frequently occurring soil order at footslopes and depressions and would be most likely to gain carbon due to cultivation. Luvisols are the most frequent occurring soil at shoulder positions and would be most likely to lose carbon due to cultivation. The similarity of SOC between the 15- to 30-cm and 30- to 45-cm depth increments in cultivated Luvisols suggests that these soil profiles may have been truncated through tillage.

Although care was taken to find three sites of similar topographic relief within a narrow range of mapped parent materials, landform morphology, texture and dominant soil taxa, differences in soil properties among the sites were greater than could be explained by land-use change alone.

The Sugar Creek forested site with Dark Gray soils investigated in this study was estimated to have 113 Mg C ha^{-1} to 45 cm, which is at the upper range of values reported for aspen or mixedwood forests in adjacent soil zones (Table 5.9).

Organic carbon at the Cookson pasture site was estimated to be 89 Mg C ha^{-1} for an equivalent soil mass of 3500 Mg ha^{-1} which compares to an estimate of total carbon of 93 Mg C ha^{-1} for an equivalent soil mass of 3431 Mg ha^{-1} for an unfertilized hayed site at Canwood, Saskatchewan dominated by smooth brome, Kentucky bluegrass, and rough hair grass (*Agrostis scabra* Willd) on Dark Gray Chernozemic soil (Nyborg et al. 1999).

Table 5.9: Reported SOC for forested sites in the Black and Gray soil zones in Saskatchewan.

Location	Site Description	SOC to Fixed Depth Mg ha^{-1}	Fixed Depth (cm)	Reference:
Lanigan, SK.	Aspen forest in Black soil zone	116	45	Pennock and van Kessel (1997b)
Fish Lake Trailhead, Prince Albert National Park, SK	Orthic Gray Luvisols on till moraine	82	48	Halliwell and Apps (1997c)
Meeting Lake, SK	aspen forests on Orthic Gray Luvisols on upper, mid and lower slope	62.5, 62.3 and 63.4	45, 50 and 53 cm respectively	Xiao (1987)
Prince Albert Model Forest	mixedwood forests in Gray soil zone means for sites influenced by base-rich groundwater and sites not influenced by base-rich groundwater, respectively	109 and 57	45 and 45	Pennock and van Kessel (1997a)
Star City, SK	aspen forest sites on Orthic Gray Luvisols	54 and 56	18 cm of mineral soil plus the LFH layer	Ellert and Bettany (1995)

The Larsen cultivated site investigated in this study was estimated to have 40 Mg C ha^{-1} to 45 cm, a value which is low compared to reports for the surrounding region: 75 Mg C ha^{-1} to 45 cm for a site cultivated for 80 years in the Black soil zone near Lanigan, Saskatchewan (Pennock and van Kessel 1997b); 87 and 88 Mg C ha^{-1} to 60 cm for a wheat (*Triticum aestivum* L. cv. Neepawa) and canola field respectively in the Black soil zone near Hepburn, Saskatchewan (Pennock and Corre 2001); and 45 and 47 Mg C ha^{-1} to 18 cm for two sites in a wheat-fallow rotation in the Gray soil zone near Star City, Saskatchewan (Ellert and Bettany 1995).

The Sugar Creek forest site and the Cookson pasture site investigated seemed relatively comparable as soil taxa and SOC levels were similar. The cultivated site exhibited less diverse soil taxa and less than half of the SOC present at the forest and pasture sites. Mann (1986) and Davidson and Ackerman (1993) estimated losses of 20% to 30% of SOC often occurs after cultivation of previously untilled soils. For Saskatchewan, Anderson (1995) estimated that such losses average 20 Mg C ha^{-1}. It is unlikely that cultivation could account for the difference of over 60 Mg C ha^{-1} between the cultivated field and the other two sites.

The lack of Gleysolic soils at even the lowest topographic positions at the cultivated site indicates that surface soils are relatively unaffected by groundwater interactions. Pennock and van Kessel (1997b) reported differences of greater than 50 Mg C ha^{-1} between sites with and without groundwater influence in undisturbed mixedwood forests within the Gray soil zone. The forest sites with elevated SOC storage were associated with greater potential for groundwater interactions (Pennock 1997). The high variability of carbon content in Gleysolic soils under both Sugar Creek forest site and the Cookson pasture site further suggests a strong influence of hydrology on SOC. Data presented in Chapter 6 provides evidence that the influence of groundwater on SOC is not restricted to Gleysolic soils.

A direct comparison of the magnitude of SOC densities among forest, pasture and cultivated sites in the Dark Gray soil zone will require a sampling design with replication for each land use type. High variation was observed within subsamples at each site. For SOC to 45 cm, interquartile ranges expressed as a proportion of the median were 0.42, 0.47 and 0.50 for the forest, pasture and cultivated site respectively. Given the similarities for this measure of variation within subsamples, an equal number of sample replicates is recommended for future sampling of forests, pastures and cultivated sites in the region.

Surveying the topography and developing digital elevation models for each sampling site was a time-consuming exercise. Although topographic position explained some of the variation in SOC exhibited at the cultivated site, broad topographic position classes such as shoulders, footslopes and levels can be readily identified visually in the field. For replicate forest, pasture and cultivated sites, use of detailed topographic surveys and digital elevation models is not recommended.

5.7 Chapter Summary

One forest site, one pasture site and one cultivated site were intensively sampled to compare SOC among topographic positions within sites. At the Sugar Creek forest site, no differences were detected among SOC values for shoulders, levels and footslopes. Results from the pasture site were inconclusive, with a non-significant reduction in SOC to 45 cm of 13 to 20 Mg C ha^{-1} observed at shoulders. At the cultivated site SOC to 45 cm at footslopes were significantly greater than levels, and SOC to 45 cm at shoulders were significantly greater than levels, with differences in magnitude between medians ranging from 9 to18 Mg C ha^{-1}. Differences among topographic positions were greatest at the 0- to 15-cm depth. The differences observed between the forest site (no SOC-topographic position relationship) and cultivated site (evident SOC-topographic position relationship) is attributed to land use, or more specifically, cultivation. Differences between the sites for other major soil forming factors were minimized through site selection.

The forest site and the pasture site had median SOC values in excess of 100 Mg C ha^{-1}. The cultivated site had a median SOC of 41 Mg C ha^{-1}. It is unlikely that cultivation alone could be responsible for differences in SOC of this magnitude. Despite efforts to use soil inventory maps to select equivalent sites in all aspects but land use, inherent differences between the cultivated site and the other two sites were observed. The forest site and the pasture site exhibited a diverse array of soil orders including Gleysols, which indicate strong groundwater interactions affecting pedogenesis. The cultivated site exhibited only upland soil taxa. The degree of groundwater interactions may be an important factor controlling SOC in these landscapes.

6. ORGANIC CARBON IN FORESTS, PASTURES AND CULTIVATED SITES

6.1 Literature Review

Deforestation results in losses of terrestrial ecosystem organic carbon densities in several ways. Removal of trees following clearing transfers carbon off a site. Oxidation (through burning or decomposition) of aboveground dead biomass left on site following clearing results in transfers of carbon to the atmosphere (Houghton et al. 1983). Increases in land surface temperatures following removal of overstory vegetation causes an increase in soil temperature, which controls rates of mineralization of soil organic matter by soil biota (Mann 1986, Baldock and Nelson 2000). Deforestation is followed by a decline in the production of litter (Schlesinger 1984) and a reduction in the woody proportion of litter. Increases in erosion and leaching may also cause net off-site transfers of organic carbon.

Numerous estimates are available for vegetation and/or soil carbon densities for the central Saskatchewan region. These were reviewed previously in Chapter 4. Fewer estimates of the changes in carbon densities associated with deforestation are available for this or similar regions.

6.1.1 Deforestation Effects on Vegetation Carbon

Houghton (1999) estimated the carbon density of North American boreal forest vegetation as 90 Mg C ha^{-1} and the carbon density of crops as 5 Mg C ha^{-1}. The difference between the two estimates (85 Mg C ha^{-1}) represents the decrease in carbon densities estimated to occur with agriculture-induced deforestation.

Johnston et al. (1996) reported differences of 90 to 100 Mg C ha^{-1} in vegetation carbon densities of upland forest sites and upland non-forest sites (primarily abandoned agricultural fields) for east-central Minnesota.

6.1.2 Cultivation Effects on Soil Organic Carbon

Pennock and Van Kessel (1997b) performed spatial comparisons to estimate the effects on soil organic carbon storage of agriculture in the Black Soil Zone (south of the Waskesiu Hills Landscape) and forestry in the Gray Soil Zone (north of the Waskesiu Hills Landscape). They recommended that future research on carbon storage in these regions of Saskatchewan should focus on soil carbon losses associated with agriculture (40 Mg C ha^{-1} for glacial till landscape) as these were greater than those associated with forestry (8 Mg C ha^{-1}).

For the Black soil zone, Anderson (1995) used paired native and cultivated soil profiles to estimate a 14 Mg C ha^{-1} reduction in organic carbon loss due to cultivation. McGill et al. (1988), estimate losses of approximately 25 Mg C ha^{-1} in the A and B horizons due to cultivation in the Dark Gray Soil Zone in Alberta.

Houghton (1999) estimated losses of 51 Mg C ha^{-1} in the top 1 m of soil for North American boreal areas converted to agriculture.

Several studies compare soil organic carbon in temperate forest regions of North America. Using paired site comparisons, Ellert and Gregorich (1996) reported that forest sites had 37 Mg C ha^{-1} (in equivalent soil mass of 3500 Mg ha^{-1}) greater SOC than cultivated sites in southern Ontario. Carbon densities for a site in east-central Minnesota were summarized by Johnston et al. (1996). The carbon densities they reported indicate that forests had approximately 15 Mg C ha^{-1} (O horizon and mineral soil to 25 cm depth) greater soil organic carbon than upland non-forest. For the same site, Homann and Grigal (1996) report differences of approximately 10 to 20 Mg C ha^{-1} (O horizon and mineral soil to 50 cm depth) from terminal points of forest-field transects.

6.2 Objectives, Research Design and Hypotheses

In this chapter I report on the results of field investigations of organic carbon densities for replicate forest sites, pasture sites and cultivated sites in a small portion of the Waskesiu Hills landscape. The objectives of the investigation were (i) to estimate the magnitude of organic carbon losses associated with agriculture-induced deforestation at sites in central Saskatchewan and (ii) to partition these losses among specific ecosystem components such as above-ground vegetation and soils to a fixed depth.

The ideal way to evaluate changes in carbon densities due to deforestation would be a manipulative experiment involving measurement of carbon densities at sites prior to and following the initiation of agriculture land use treatments. Unfortunately no antecedent data on carbon densities exists for pastures and cultivated fields within this landscape. An alternative approach is to use space as an analogue for time in a comparative mensurative experimental research design (Hurlbert 1984). The main assumption in this comparison is that mean carbon densities for the forested sites, for the pasture sites, and for the cultivated sites were the same prior to agricultural land use.

Three hypotheses were investigated. For organic carbon in aboveground vegetation I hypothesized:

$$C_{forests} >> C_{cultivated} > C_{pasture}.$$

Forests were expected to have the greatest aboveground carbon because of their perennial woody tissue and taller stature. Peak biomass C at cultivated sites was expected to exceed peak biomass C at pastures because of fertilization and lack of grazing at the cultivated sites. For soil organic carbon I hypothesized:

$$C_{forests} = C_{pasture} > C_{cultivated}.$$

Cultivated sites were expected to have the lowest SOC due to tillage-induced losses (see reviews by Mann (1986), Davidson and Ackerman (1993)). The relationship between SOC and topographic position classes was also investigated to determine whether replication could confirm the pattern reported in Chapter 5.

For estimated total ecosystem organic carbon (aboveground vegetation plus soil) I hypothesized:

$C_{forests} > C_{pasture} > C_{cultivated}$.

Forests would be expected to have the greatest density of organic carbon because of greater aboveground vegetation C. Cultivated sites would be expected to have lowest carbon densities due to tillage-induced SOC losses.

6.3 Site Selection and Site Descriptions

An area approximately three townships in size, located within the Waskesiu Hills landscape of central Saskatchewan (see Chapter 2), was selected for extensive sampling of carbon densities. This area (which I will refer to as the Cookson study area) includes Township 53, Ranges 1 (west half), 2, 3, and 4 (east half), West of 3rd Meridian.

Sites were selected within three characteristic treatment groups: forests, pastures and cultivated sites. Presettlement vegetation in this landscape prior to agricultural settlement was mixedwood boreal forest (Weir et al. 2000). Pastures and cultivated fields occupy areas of former boreal forest.

One site within each group was sampled in August and September of 2000. Sites F0, P0, and C0 correspond to the Sugar Creek forest site, the Cookson pasture site and the Larsen cultivated site described in Chapters 4 and 5). Five additional replicates within each group (F1 to F5, P1 to P5, C1 to C5) were selected for sampling from June to August 2001. This resulted in a total of six replicates within each land use type. The locations of sites and their ownership status are listed in Table 6.1.

The six forest sites are located in Prince Albert National Park and have no history of cultivation or grazing by domestic animals. Forest sites were last burned over in 1929 (F2), 1940 (F0, F3, F4, F5) and 1947 (F1). All of the sites have been protected from logging for over half a century.

Table 6.1: Location and ownership of forest sites, pasture sites and cultivated sites within the Cookson study area.

Characteristic Treatment: Site Number	Latitude, Longitude	Ownership
Forests:		
F0	53° 35' 03" N, 106° 23' 10" W	Parks Canada
F1	53° 35' 19" N, 106° 08' 04" W	Parks Canada
F2	53° 35' 38" N, 106° 16' 08" W	Parks Canada
F3	53° 35' 02" N, 106° 23' 03" W	Parks Canada
F4	53° 35' 02" N, 106° 24' 38" W	Parks Canada
F5	53° 36' 10" N, 106° 29' 32" W	Parks Canada
Pastures:		
P0	53° 34' 57" N, 106° 23' 48" W	Saskatchewan Agriculture and Food
P1	53° 33' 36" N, 106° 22' 21" W	Saskatchewan Agriculture and Food
P2	53° 33' 37" N, 106° 22' 10" W	Saskatchewan Agriculture and Food
P3	53° 34' 56" N, 106° 23' 14" W	Saskatchewan Agriculture and Food
P4	53° 33' 32" N, 106° 20' 25" W	Saskatchewan Agriculture and Food
P5	53° 33' 14" N, 106° 19' 32" W	Private Landowner
Cultivated Sites		
C0	53° 32' 28" N, 106° 20' 13" W	Private Landowner
C1	53° 33' 07" N, 106° 19' 39" W	Private Landowner
C2	53° 32' 26" N, 106° 21' 12" W	Private Landowner
C3	53° 32' 27" N, 106° 19' 18" W	Private Landowner
C4	53° 34' 10" N, 106° 07' 06" W	Private Landowner
C5	53° 34' 59" N, 106° 12' 23" W	Private Landowner

Five of the six pasture sites were located on public land in the Cookson Community Pasture. Two of these five sites (P1 and P2) were cleared prior to 1963 and the remainder (P0, P3 and P4) were cleared between 1963 and 1968. Private long-term pastures on hummocky glacial till deposits were difficult to find in this study area because cultivation of annual crops is the dominant form of agricultural production in the region. A single private pasture site (P5) was grazed for over 60 years prior to sampling in 2001.

All of the cultivated sites were cleared prior to 1963, some several decades earlier. These sites have been under continuous cultivation for production of small grains and oilseeds for several decades. All of the landowners practice conventional tillage and add nitrogen fertilizer.

Land use and present vegetation differed among the three groups of sites, but the sampling locations were selected to minimize differences in soil taxa, textures, parent materials and landform morphology based on soil inventory maps. The soils at all study sites were mapped as a mixture of Waitville Orthic Gray Luvisols, Waitville Dark Gray Luvisols and Whitewood Orthic Dark Gray Chernozems with loam or sandy loam textures on hummocky glacial till landforms (Padbury et al. 1978, Saskatchewan Land Resource Centre 1997). The eighteen study sites were in close geographic proximity. All were located within a three-township area in order to minimize differences in historic vegetation, climate or other environmental factors.

6.4 Materials and Methods

The intensive approach for sites sampled in 2000 (F0, P0, and C0) was described in Chapter 4 and Chapter 5. The extensive phase of sampling completed for replicate forest, pasture and cultivated sites in 2001 involved a smaller sampling area (0.4 ha), a reduced number of ecosystem components sampled, and a reduced number of subsamples (gridpoints) per site. All 2001 sites (F1 to F5, P1 to P5, and C1 to C5) were sampled using a 3 x 4 rectangular grid with 25 m spacing, oriented with its long axis oriented in a north-south direction.

Sampling for vegetation occurred at every other grid point. Six of the 100 m^2 tree plots described in Chapter 4 were used at each site. Sampling to estimate live and dead tree biomass proceeded as described for tree plots in Chapter 4, but standing trees less than 10 cm dbh were not measured. Biomass was converted to carbon for live and dead trees as described in Chapter 4. Estimated total aboveground vegetation carbon for forests does not include live and dead vegetation less than 10 cm in diameter.

Six 0.25 m^2 herb plots as described in Chapter 4 were used to sample vegetation carbon at pastures and cultivated sites. Similar to the tree plots at forest sites, every other soil sampling grid point was sampled for herbaceous plant biomass. Each sample included both live and dead vegetation. Sample collection and processing, as well as conversion to C densities, was as described for the herb strata at pastures and cultivated sites sampled in 2000 (Chapter 4). For the canola field harvested at site P0 prior to the sampling date, peak biomass was estimated as 4.0 Mg ha^{-1} based on 14 years of yield data from Melfort (Nuttall et al. 1992). A vegetation carbon density of 2 Mg C ha^{-1} was assumed for site P0 rather than the crop residue carbon density reported in Chapter 4.

Soil sampling, lab analysis and C density calculations were identical to the methods described in Chapters 4 and 5 except that samples were taken at 12 rather than 49 gridpoints per site. No topographic surveys were completed, but each sampling grid point was visually classified based on the surrounding topography into one of the three topographic position classes described in Chapter 5: shoulders (convex profile curvature), levels (including linear slopes but excluding depressions) and footslopes (concave profile curvature or depressional levels). Clay content analysis was performed on a subset of the soil samples using the pipette method (Gee and Bauder 1986).

Non-parametric statistics were used due to low sample sizes and problems with skewness. The multiple-comparison extension of the Kruskal-Wallis test (Siegel and Castellan 1988) was used to determine differences among forest sites, pasture sites, and cultivated sites. I used a significance level of 0.15 due to low sample sizes, but report the probability with greater accuracy were possible. Spearman rank correlation was used because data were not normally distributed. SYSTAT 8.0 (SPSS 1998) was used for statistical analysis.

6.5 Results

6.5.1 Vegetation Carbon Densities

Trembling aspen was the dominant tree species at all forest sites. Each of the pastures was vegetated by a mix of introduced species including smooth brome, alfalfa and Kentucky bluegrass. Wheat was grown at the five cultivated sites sampled in 2001. At the cultivated site sampled in 2000, canola residue was sampled because the crop was harvested prior to the sampling date.

Carbon densities for live trees at the forest sites ranged from 35 Mg C ha^{-1} to 81 Mg C ha^{-1}. Carbon densities for dead trees in the forest group were much smaller in magnitude and equivalent to those within live crops for the cultivated group (Table 6.2). The general pattern observed was consistent with the first hypothesis; vegetation carbon was greatest for forests and least in pastures. Median live plus dead aboveground vegetation C for the forest group was 56 Mg C ha^{-1} greater than for the cultivated group ($p<0.15$) and 60 Mg C ha^{-1} greater than the pasture group ($p<0.05$). Median aboveground vegetation C for pastures was lower than that in cultivated sites ($p < 0.15$).

Table 6.2 Aboveground vegetation carbon densities (medians with interquartile ranges in parentheses) at forest sites, pasture sites and cultivated sites within the Cookson study area.

Characteristic Treatment	Live Vegetation C (Mg C ha^{-1})		Dead Vegetation C (Mg C ha^{-1})		Total Vegetation C (Mg C ha^{-1})	
Forests (n=6)	54a	(42)	4	(3)	60a	(46)
Pastures (n=6)	1b	(<1)	*		1b	(<1)
Cultivated (n=6)	4c	(1)	*		4b	(2)

Medians in the same column with a different letter are significantly different ($p<0.15$).
* Dead vegetation was included with live vegetation samples.

6.5.2 Soil Carbon Densities

The forested sites exhibited a diverse array of soil taxa. Luvisols, Gleysols, Chernozems and Brunisols were each the dominant soil order at one or more individual forest sites (Table 6.3). Luvisolic and Chernozemic soils were dominant at all pasture sites. Gleysols occurred at only two pastures and Brunisols were entirely absent. Chernozems were the most frequently occurring Soil Order at 5 of 6 cultivated sites. Luvisols and Brunisols were widespread, but Gleysols occurred at only one site within the cultivated group. In summary, forested sites had the greatest diversity in soil orders, cultivated sites had the highest incidence of Chernozemic soils and both types of agricultural sites had lower incidence of Gleysolic (wet) soils.

Table 6.3 Frequencies for Soil Orders at forest sites, pasture sites and cultivated sites within the Cookson study area.

	Soil Order			
Characteristic	Brunisolic	Chernozemic	Gleysolic	Luvisolic
Treatment: Site Number	Frequency (%)	Frequency (%)	Frequency (%)	Frequency (%)
Forest Sites:				
F0 (n=47)	2 (4)	16 (34)	17 (36)	12 (26)
F1 (n=12)	2 (17)	1 (8)	2 (17)	7 (58)
F2 (n=12)	2 (17)	5 (42)	1 (8)	4 (33)
F3 (n=12)	3 (25)	3 (25)	0 (0)	6 (50)
F4 (n=12)	6 (50)	1 (8)	0 (0)	5 (42)
F5 (n=12)	0 (0)	0 (0)	1 (8)	11 (92)
Pasture Sites				
P0 (n=49)	0 (0)	30 (61)	9 (18)	10 (20)
P1 (n=12)	0 (0)	5 (42)	2 (17)	5 (42)
P2 (n=12)	0 (0)	0 (0)	0 (0)	12 (100)
P3 (n=12)	0 (0)	1 (8)	0 (0)	11 (92)
P4 (n=12)	0 (0)	2 (17)	0 (0)	10 (83)
P5 (n=12)	0 (0)	5 (42)	0 (0)	7 (58)
Cultivated Sites				
C0 (n=49)	6 (12)	34 (69)	0 (0)	9 (18)
C1 (n=12)	1 (8)	6 (50)	0 (0)	5 (42)
C2 (n=12)	2 (17)	7 (58)	1 (8)	2 (17)
C3 (n=12)	0 (0)	8 (67)	0 (0)	4 (33)
C4 (n=12)	4 (33)	5 (42)	0 (0)	3 (25)
C5 (n=12)	1 (8)	3 (25)	0 (0)	8 (67)

Soil taxa at sites within each treatment group exhibited varying degrees of influence of groundwater on soils. Woody Calcareous soils* (Mitchell et al. 1950) of the Chernozemic Soil Order occur near recharge depressions and are associated with lateral groundwater movement, capillary rise, and biocycling of calcium by poplar (*Populus*) species (Fuller et al. 1999). Soils of the Gleysolic Soil Order occur where groundwater is, seasonally at least, near the surface of the soil (Agriculture Canada Expert Committee on Soil Survey 1987). Within each of the treatment groups there was at least one site (F0, P0, and C3) where these two soil types occurred at over one third of sampling gridpoints (Table 6.4). Within each group there was also at least one site (e.g. F4, P2, and C0) where these two soil types did not occur at any sampling

* Woody calcareous soils, classified as Rego Black Chernozems by Fuller et al. (1999), may in this case be Rego Dark Gray Chernozems under the Canadian System of Soil Classification.

gridpoints. Based on the proportional frequency of occurrence of these groundwater influenced soil taxa, I derived an index of groundwater influence:

$$IGI_x = (f_{CA.DG} + f_{GLEYSOLS}) / n$$

where: IGI_x is the Index of Groundwater Influence at site x,

$f_{CA.DG}$ is the frequency of Woody Calcareous Chernozems at site x,

$f_{GLEYSOLS}$ is the frequency of Gleysols at site x, and

n is the number of sampling gridpoints at site x.

The index of groundwater influence is also shown in Table 6.4.

Table 6.4: Frequency of Woody Calcareous Chernozems and Gleysols, the number of sampling points, and the Index of Groundwater Influence at forest sites, pasture sites and cultivated sites within the Cookson study area.

Characteristic Treatment: Site Number	Frequency of Woody Calcareous Chernozems (number of gridpoints)	Frequency of Gleysols (number of gridpoints)	Total Number of Sampling Gridpoints (number of gridpoints)	Index of Groundwater Influence (proportion)
Forest:				
F0	9	17	47	0.55
F1	0	2	12	0.17
F2	0	1	12	0.08
F3	3	0	12	0.25
F4	0	0	12	0.00
F5	0	1	12	0.08
Pasture:				
P0	8	9	49	0.35
P1	0	2	12	0.17
P2	0	0	12	0.00
P3	0	0	12	0.00
P4	0	0	12	0.00
P5	0	0	12	0.00
Cultivated:				
C0	0	0	49	0.00
C1	4	0	12	0.33
C2	2	1	12	0.25
C3	5	0	12	0.42
C4	0	0	12	0.00
C5	1	0	12	0.08

Soil textures were not well sorted, with textures ranging from sandy loam to loam. Sand fraction ranged from 44% to 60% for the forest group, from 49% to 70% for the pasture group and from 54% to 67% for cultivated group. Median sand fraction for forests was significantly less than that for cultivated sites ($p>0.15$, Table 6.5) but this difference was small in magnitude (9%). Sand fraction exhibited greatest variation within the pasture group, and differences in sand fraction between pastures and other land uses were not significant ($p>0.15$). Pebbles were common at all sites with the exception of site F2, where they occurred at less than one quarter of the sampling points. Clay fraction ranged from 16% to 19% at a subset of three forest sites and from 12% to 17% at a subset of three cultivated sites.

Table 6.5: Median values for sand fraction, bulk density and SOC (with inter-quartile range in parenthesis) at forest sites, pasture sites and cultivated sites within the Cookson study area.

Treatment Group	Sand Fraction (% for 0- to 45-cm depth)	Bulk Density (g cm^{-3} for 0- to 45-cm depth)	SOC (Mg C ha^{-1} to 45 cm depth)	SOC (Mg C ha^{-1} in equivalent soil mass*)
Forest (n=6)	55a (3)	1.2a (0.11)	83a (28)	76a (18)
Pasture (n=6)	63ab (13)	1.5b (0.11)	61a (34)	53a (28)
Cultivated (n=6)	64b (7)	1.5b (0.08)	83a (39)	66a (35)

Medians in the same column with a different letter are significantly different ($p<0.15$).
* equivalent mass = 3500 Mg ha^{-1}

Median values for SOC to 45 cm depth were the same for forest and cultivated groups and 22 Mg C ha^{-1} lower for the pasture group (Table 6.5), but differences were not statistically significant ($p=0.46$). Results for soil organic carbon in equivalent soil mass were very similar to results for soil organic carbon to 45 cm depth. SOC in an equivalent soil mass of 3500 Mg ha^{-1} ranged from 58 to 103 Mg C ha^{-1} for forests, from 40 to 89 Mg C ha^{-1} for pastures and from 34 to 107 Mg C ha^{-1} for cultivated sites. The forest group median was 23 Mg C ha^{-1} greater than the pasture group median and 10 Mg C ha^{-1} greater the cultivated group median (Table 6.5), but differences were not significant ($p>0.25$). Whether using units of elemental mass to fixed depth or elemental mass in equivalent soil mass, comparisons of soil organic carbon among treatment groups produced a consistent result: failure to reject the null hypothesis. The expected reduction in SOC associated with cultivation was not detected.

Soil organic carbon exhibited high variation at sites within treatment groups and at grid points within sites (Appendix A). Site medians for SOC to 45 cm ranged from 63 Mg C ha^{-1} to 114 Mg C ha^{-1} at forests and from 47 Mg C ha^{-1} to 101 Mg C ha^{-1} for pastures (Table 6.6). For the cultivated group, the maximum site median for SOC to 45 cm (113 Mg C ha^{-1}) was more than three times the minimum site median (41 Mg C ha^{-1}). Interquartile ranges expressed as a proportion of the median were 0.33, 0.56 and 0.47 for forests, pastures and cultivated sites, respectively.

Table 6.6: Median aboveground vegetation C and SOC at forest sites, pasture sites and cultivated sites within the Cookson study area.

Characteristic *Treatment*: Site Number	Aboveground Vegetation C (Mg C ha^{-1})	SOC to 45 cm depth (Mg C ha^{-1})	SOC in equivalent soil mass (Mg C ha^{-1})
Forest Sites:			
F0	58	114	103
F1	36	97	85
F2	87	84	78
F3	41	82	73
F4	63	69	66
F5	90	63	58
Pasture Sites			
P0	1	101	89
P1	<1	50	46
P2	<1	47	40
P3	1	84	74
P4	1	61	53
P5	1	60	52
Cultivated Sites			
C0	2	41	34
C1	3	88	68
C2	4	80	64
C3	4	133	107
C4	4	49	34
C5	4	85	69

The relationship between SOC and clay content, and SOC and groundwater influence, was explored in order to attempt to explain the great variation in SOC among sites within groups.

No relationship is evident in the scatterplot of SOC to 45 cm and clay fraction for the 0- to 45-cm depth for a subset of forest and cultivated sites that spanned the full range of observed carbon densities (Figure 6.1). The difference in SOC to 45 cm depth was 50 Mg C ha^{-1} for two forests (F0 and F5) with almost identical clay fractions (between 16 and 17%). Site C3 had 81 Mg C ha^{-1} greater SOC to 45 cm than site C4 but had similar clay fractions (15 and 17%, respectively).

A relationship was evident between SOC to 45 cm and the Index of Groundwater Influence (Figure 6.2). The Spearman rank correlation coefficient (r_S) between SOC to 45 cm and Index of Groundwater Influence was 0.76, which was highly significant ($p < 0.01$).

Soils of the Luvisolic Order occurred at all study sites. Luvisolic soils are unlikely to be affected by groundwater because they are characterized by downward movement of water through the soil profile. The median SOC to 45 cm for Luvisolic soils at each site was calculated and group medians were statistically compared (Table 6.7). The group median for forests was 18 Mg C ha^{-1} greater than pastures, and 9 Mg C ha^{-1} greater than cultivated sites, but variation within treatments was high, and differences were not statistically significant ($p>0.25$).

Table 6.7: Median values (with inter-quartile range in parenthesis) for SOC to 45 cm for Luvisolic soils at forest sites, pasture sites and cultivated sites within the Cookson study area.

Treatment Group	SOC (Mg C ha^{-1} to 45 cm depth)
Forest (n=6)	79a (26)
Pasture (n=6)	61a (40)
Cultivated (n=6)	70a (31)

Medians in the same column with a different letter are significantly different ($p<0.15$).

Median SOC for Luvisolic soils at sites was also positively correlated with the index of groundwater influence ($r_S = 0.62$, $p<0.01$). This provides evidence that groundwater effects on SOC are not localized at the scale of a few meters (i.e. individual sampling gridpoints) but rather extend throughout the sites.

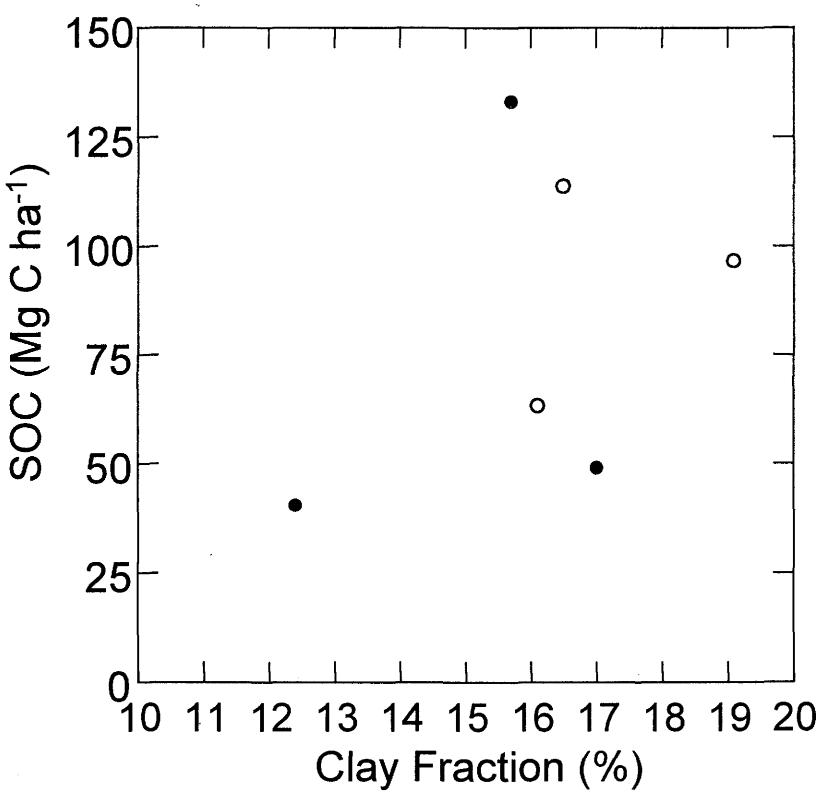


Figure 6.1: Scatterplot of site median SOC and clay fraction for a subset of forest sites (open circles) and cultivated sites (closed circles) that spanned the full range of observed carbon densities.

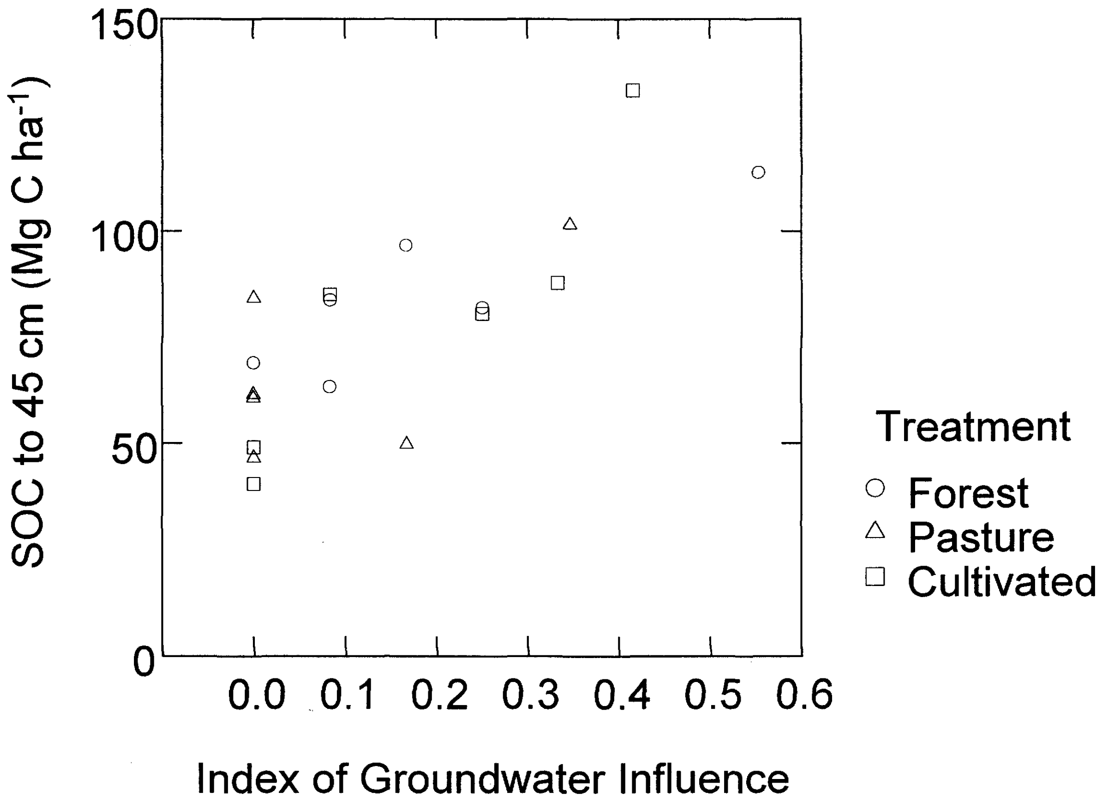


Figure 6.2: SOC to 45 cm plotted against index of groundwater influence (the proportion of sampling gridpoints at a site with Woody Calcareous or Gleysolic soils).

SOC residuals (differences between the median value for a topographic position class at a site and the overall site median) for shoulders, levels and footslopes were not significantly different within the forest and pasture groups (Table 6.8). In the cultivated group, SOC at shoulders were 17 to 19 Mg C ha^{-1} less than the site medians and were significantly different from SOC at footslopes ($p<0.05$).

Table 6.8 Residual organic carbon (medians with interquartile ranges in parentheses) for topographic position classes at forest sites, pasture sites and cultivated sites within the Cookson study area.

	Topographic Position Classes					
Characteristic Treatment	Shoulders (Mg C ha^{-1})		Levels (Mg C ha^{-1})		Footslopes (Mg C ha^{-1})	
Forests (n=6,6,6)	+2a	(17)	-3a	(19)	+1a	(9)
Pastures (n=5,6,6)	+2a	(18)	-2a	(12)	+1a	(2)
Cultivated (n=5,6,6)	-17a	(2.5)	-3ab	(14)	+13b	(30)

Medians in the same row with a different letter are significantly different ($p<0.15$).

6.5.3 Total Ecosystem Carbon Densities

Median organic C density for aboveground vegetation plus soil to 45 cm was 158 (29) Mg C ha^{-1} for the forest group, which was significantly different ($p<0.05$) from 63(26) Mg C ha^{-1} for the pasture group and the 81(41) Mg C ha^{-1} for cultivated group (values reported are medians with interquartile ranges in parentheses). Eleven of the 12 agricultural sites had lower organic C density than the forest site with the least carbon (Figure 6.3). The single agricultural site with a carbon density within the same range as the forest group was site C3, which had the highest SOC observed across the data set. Organic carbon densities for the cultivated group were highly variable among sites, and difference between the cultivated group and pasture group was not significant ($p>0.30$).

The third hypothesis was supported in part since forest ecosystems had the greatest density of organic carbon. However the higher carbon density expected for pastures over cultivated sites was not observed.

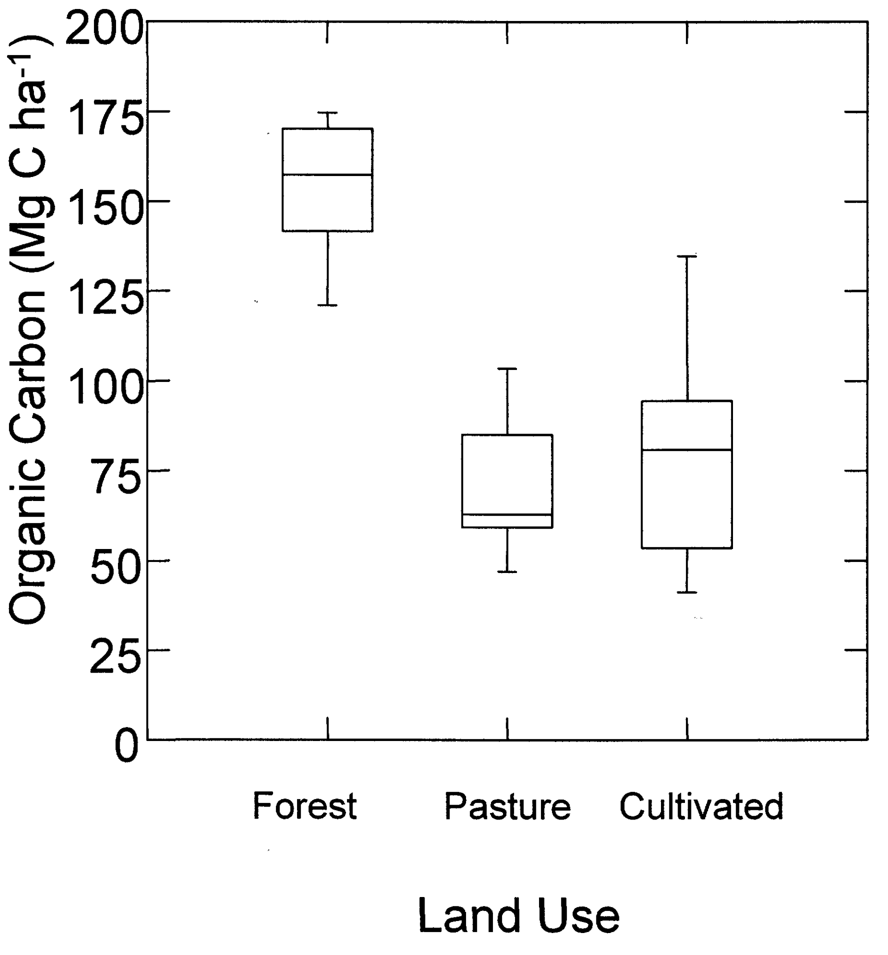


Figure 6.3: Boxplots of total estimated ecosystem carbon for forest sites, pasture sites and cultivated sites.

6.6 Discussion

A basic assumption for this comparative mensurative experiment is that the three groups of sites were similar prior to the imposition of land use differences. Site selection was used to minimize ecological differences among the three treatment groups. All sites were located in a relatively small geographic area (3 townships) to minimize differences in historic climate and vegetation. Soil maps were used to restrict sampling sites to a limited range of landforms and dominant soil taxa.

Unfortunately, 18 sites across three townships can never be invariant in terms of desired biophysical characteristics. There were inherent differences among the 18 sites selected for this experiment. The purpose of replication of forests, pastures and cultivated fields was to sample the range of initial conditions across the landscape within each treatment group. Although the sites within the three treatment groups exhibited limited variation in properties such as median sand fraction and dominant soil order, no gross differences among treatment groups were evident, thus allowing statistical comparisons of carbon densities in forested and deforested groups.

A difference of 56 to 60 Mg C ha^{-1} was observed between forest and agricultural carbon densities for vegetation. These differences are underestimates because only trees greater than 10 cm in diameter were sampled. The results from Chapter 3 suggest that smaller diameter live and dead aboveground vegetation may account for approximately 3 Mg C ha^{-1}. More importantly, belowground vegetation biomass was not estimated. Although fine roots were included in the soil samples and reported as soil organic carbon, structural coarse root carbon was not quantified. As discussed in Chapter 3, accounting for coarse roots would likely increase vegetation carbon densities for forests by at least 10 Mg C ha^{-1}. Adding these two unsampled pools, losses of vegetation carbon for the pasture and cultivated groups as compared to forests would be 73 Mg C ha^{-1} and 69 Mg C ha^{-1} respectively.

Soil organic carbon densities were larger than vegetation carbon densities for all treatment groups. Statistical differences in soil organic carbon among treatment groups were not detected. Despite high sample numbers for this type of research design, spatial variation was a confounding factor that prevented assessment of the temporal effects of land-use change on soil organic carbon storage. The high spatial variation within all three treatment groups suggests there may be a separate factor controlling soil organic carbon that is not accounted for in the research design.

In Chapter 4, I suggested that deforestation and associated land use (i.e. cultivation) might not be entirely responsible for the differences in SOC to 45 cm of over 60 Mg C ha^{-1} between the cultivated site (C0) and the forest and pasture sites (F0 and P0). There may have been inherent differences among these sites that would have been evident prior to deforestation. Replication for each treatment group provided further insight into the inherent variation in carbon density among sites. For example, both the minimum and maximum site medians for SOC to 45 cm depth (41 Mg C ha^{-1} and 133 Mg C ha^{-1}) for the 18 sites sampled occurred at sites within the cultivated group. Some factor other than land use and the site properties controlled by site selection must be responsible for this 92 Mg C ha^{-1} range in observed carbon densities.

Comparative mensurative research designs are not intended to uncover the causal factors responsible for differences observed among treatments or among replicates within treatments (Pennock and Corre 2001). However, the data can be explored to look for hypotheses that could be confirmed by other forms of experimentation.

Soil texture did not explain the wide variation in SOC within treatment groups. For example, the C3 site had 81 Mg C ha^{-1} greater SOC to 45 cm than the C4 site but had similar clay fractions (15% clay for C3 and 17% clay for C4). This is consistent with the assertion by Anderson (1995) that clay content is not an important determining factor for organic carbon content in cold northern soils.

Variation in the degree of groundwater influences may be responsible for much of the inherent differences among sites within treatments. The index of groundwater influence was positively correlated with SOC to 45 cm across all eighteen sites. This positive correlation held even for Luvisols, a Soil Order that would not be affected by groundwater saturation or upward groundwater movement. The groundwater factor was unaccounted for in the research design because the presence or absence of strong groundwater influence at a site was not predictable based on soil inventory maps or site reconnaissance based on vegetation.

Braidek (1996) described rich and poor sites in the Prince Albert Model Forest, with rich sites exhibiting greater SOC. Gleysols were the most frequently occurring Soil Order at all rich sites, while either Brunisols or Luvisols were the most frequently occurring Soil Order at the poor sites. Unlike the Prince Albert Model Forest sites, the sampling sites in the Cookson study area could not be divided into two groups. Rather, there appeared to be a continuum between the high carbon and low carbon sites for most treatment groups (Figure 6.2).

Reviews by Mann (1986) and Davidson and Ackerman (1993) suggest that losses due to cultivation are often on the order of 20% to 30% of initial SOC stocks. Assuming an average initial carbon density of 83 Mg C ha^{-1} (mean for the six forest sites), losses of 17 to 25 Mg C ha^{-1} might be expected following cultivation. This is similar to the observed losses for a variety of soils in Saskatchewan (Anderson 1995) and Alberta (McGill et al. 1988). However, the observed variation in initial carbon densities was much larger, 50 Mg C ha^{-1} for the six forest sites sampled within the Cookson study area. Given the large spatial variation in soil organic carbon across this study area, perhaps related to the degree of groundwater influences, relatively small temporal losses of SOC due to land use change will be difficult to detect with research designs using space as an analog for time.

Studies using paired forest and agricultural sites (e.g. Ellert and Gregorich 1996) might be more effective at quantifying deforestation losses. However, forests and agricultural lands are rarely configured in a manner that facilitates ideal paired site comparisons.

Differences in total ecosystem organic carbon were detected between forests and agricultural sites. Vegetation carbon rather than soil organic carbon accounted for most of these differences. The magnitude of the losses in vegetation carbon estimated in this study (approximately 70 Mg C ha^{-1}) was substantially larger than cultivation-induced losses in SOC reported for sites in the Black Soil Zone (Anderson 1995, Pennock and van Kessel 1997b) and the Gray Soil Zone (Ellert and Bettany 1995) of Saskatchewan.

Sampling mature forests (greater than 60 years of age) may have biased the comparison of vegetation carbon between forest and agricultural sites. However, in the southern portion of Prince Albert National Park, stands younger than 60 years are extremely rare (Weir 1996).

Based on field data collected during this experiment, deforestation in central Saskatchewan resulted in losses of vegetation carbon that exceeded 55 Mg C ha^{-1}. The effects of deforestation on soil organic carbon could not be detected, perhaps due to the confounding effect of groundwater that was unaccounted for in the research design.

6.7 Chapter Summary

Vegetation carbon densities were compared at six forest sites, six pasture sites and six cultivated sites on hummocky glacial till landforms in central Saskatchewan. Deforestation and subsequent agricultural land use led to losses of approximately 70 Mg C ha^{-1} in aboveground vegetation at pastures and cultivated sites relative to forest sites.

Statistically significant losses of soil organic carbon due to cultivation were not detected between the forest sites and the cultivated sites. Detection of a land use effect on soil organic carbon was confounded by groundwater influences. Across all sites regardless of land use, there was a positive correlation between soil organic carbon storage and the proportional frequency of groundwater influenced soils (Gleysols plus Woody Calcareous Chernozems).

Differences in total ecosystem organic carbon densities between the forest group and the agricultural group reflected differences in aboveground vegetation carbon.

7. SYNTHESIS

Global boreal and temperate forest area has been described as stable or increasing over recent decades (Williams 1989a, Mather and Sdasyuk 1991, FAO 1995, FAO 2001). However, slow but persistent deforestation has occurred in some temperate and boreal landscapes of the hemisphere throughout this period (Zipperer et al. 1990, Zipperer 1993, Zheng et al. 1997). Deforestation in western Canada has, until very recently, received little scientific scrutiny. This research effort contributes scientific data on the phenomenon of deforestation in central Saskatchewan and provides management considerations regarding potential reforestation activities in the region.

The quantitative results presented herein on the extent and ecological effects of deforestation can be combined with other recent research results to evaluate the significance of this land-use change from a management perspective.

The extrapolations from sites to landscapes, or from landscapes to regions included in the sections below are intended only for the purpose of evaluating the order of magnitude of effect size. Extrapolations should not be interpreted as quantitative estimates of actual effects.

7.1 Extent of Deforestation in Landscapes of the Boreal Plain Ecozone

Deforestation on agricultural land within the Waskesiu Hills and the Red Deer River landscapes led to losses of 16 400 to 37 100 ha of wooded area, respectively, over recent decades. These two landscapes account for approximately 5 % of the Boreal Plain Ecozone of Saskatchewan. Although no statistical inferences can be made from the two landscapes to the rest of the ecozone, it is likely that wooded area losses for the Boreal Plain were in the order of magnitude of several hundred thousand ha in the three decades prior to 1990.

Robinson et al. (1999) reviewed recent estimates of the total forest area converted to agriculture on an annual basis for Canada. National deforestation rates due to agricultural land conversion, estimated from agricultural land statistics, ranged from 12 000 to 98 000 ha yr^{-1} for a five to ten year period preceding 1991. Data from Chapter 2 suggest that combined losses of wooded area from the Waskesiu Hills and Red Deer River landscapes were approximately 2 000 ha yr^{-1} over the 1975/76 to 1990 period. Deforestation within the Boreal Plain Ecozone of Saskatchewan, which would be estimated as an order of magnitude larger, was of significance at a national scale.

7.2 Deforestation Effects on Vegetation Carbon

Robinson et al. (1999) reviewed estimates of annual emissions from agriculture-related deforestation. Terrestrial carbon loss estimates ranged from 330 to 5200 Gg C yr^{-1} prior to 1991.

The estimated difference in vegetation carbon density between forest and agricultural sites reported in Chapter 6, applies to boreal deciduous forests growing on hummocky glacial till landforms in a three-township area near Cookson, Saskatchewan. The magnitude of differences in vegetation carbon between forests and agricultural lands across the Waskesiu Hills or Red Deer River landscapes would likely vary by geographic location, parent materials, landforms, time since disturbance and dominant lifeform. Equivalent data on carbon density differences for all of these cases do not exist. As a result, the estimate of 70 Mg C ha^{-1} (see Chapter 6) will be used as the baseline to estimate the temporal mean difference in vegetation carbon density between forest sites and agriculture sites across these two landscapes.

Boreal forests exhibit increased biomass with age for the first century following disturbance (Bhatti et al. 2001, Sulistiyowati 1998). Field sampling for this study was carried out only at mature stands, which results in an overestimation of the carbon density differences between forest sites and agricultural sites. The estimate of 70 Mg C ha^{-1} may represent the upper bound for the mean difference between carbon densities for forest and agricultural vegetation. The baseline estimate should be adjusted downward if it is to be applied at the landscape-scale where stand age varies due to disturbances.

The lower bound would be the difference between forest vegetation carbon density soon after a disturbance (e.g. fire, logging) and agricultural vegetation carbon density. Disturbances rarely reduce forest biomass carbon density to zero. Live vegetation carbon and dead vegetation carbon mirror each other to a great degree (Dewar 1991, Sulistiyowati 1998). A fire will kill a live tree but volatize only a portion of its biomass carbon. Amiro et al. (2001) estimated fuel consumption for the Boreal Plain Ecozone as 24 Mg ha^{-1} (12 Mg C ha^{-1}). The unburned portion of live biomass is transferred to the dead biomass pool (Bhatti et al. 2001), which decays slowly in boreal environments (Sulistiyowati 1998).

Halliwell and Apps (1997c) estimated woody detrital carbon densities of 21 and 41 Mg C ha^{-1} at two recently burned aspen sites, and 9 Mg C ha^{-1} at a recently logged site in central Saskatchewan. Sulistiyowati (1998) studied aspen-white spruce mixedwood stands within the Prince Albert Model Forest. Mean woody debris carbon densities for three burned sites and five logged sites sampled within 6 years of disturbance were 53 and 23 Mg C ha^{-1}, respectively. Mean carbon density for dead biomass for the 11 recently disturbed sites sampled by Halliwell and Apps (1997c) and Sulistiyowati (1998) was 31 Mg C ha^{-1}.

Recently disturbed sites would also contain live vegetation biomass. This live biomass likely has a carbon density roughly equivalent to that estimated for pasture sites and cultivated sites in Chapter 6 (approximately 1 to 4 Mg C ha^{-1}). Recently disturbed forests would also contain live and dead coarse root biomass. Thus, 30 Mg C ha^{-1} would be a conservative estimate of the lower bound for the mean difference in vegetation carbon density between forest and agricultural sites.

The mean differences in vegetation carbon density between forest sites and agricultural sites will fluctuate between this lower bound (30 Mg C ha^{-1}) and upper bound (70 Mg C ha^{-1}) as the frequency of disturbance events vary over time. For purposes of estimating the order of magnitude of carbon losses associated with deforestation in these landscapes, I will assume that the temporal mean over a multi-decade period would be at the midpoint of these two established bounds, that is, 50 Mg C ha^{-1}.

Changes in vegetation carbon stocks for the Waskesiu Hills landscape between 1976 and 1990 can be approximated as the product of the wooded area loss estimate reported in Chapter 2 (6 500 ha) and the estimated carbon density difference between forest and agricultural sites (50 Mg C ha^{-1}). A loss in the range of 330 Gg C occurred over 14 years. Applying the same calculations for the Red Deer River landscape, results in a loss of approximately 1 200 Gg C (23 200 ha X 50 Mg C ha^{-1}) over 15 years.

Annualized loss rates for the 14 to 15 year period prior to 1990 can be estimated as 23 and 77 Gg C yr^{-1} for the Waskesiu Hills and Red Deer River landscapes, respectively, or 100 Gg C yr^{-1} for the two landscapes combined. Estimated annual carbon losses for the Boreal Plain Ecozone would be expected to be an order of magnitude greater. In comparison to the estimated national vegetation carbon losses due to agriculture-induced deforestation prior to 1990 described above (327 to 5181 Gg C yr^{-1}), the carbon losses for this region were of significance at a national scale.

As a further comparison, fossil fuel burning by agricultural producers in Saskatchewan was estimated to release 719 Gg C yr^{-1} in 1990 (AAFC 1999). This is several times larger than annual carbon losses for the Waskesiu Hills and Red Deer River landscapes combined. Annual carbon losses due to deforestation across the entire Boreal Plain Ecozone of Saskatchewan would perhaps be of similar magnitude to annual carbon emissions from fossil fuel burning by agricultural producers.

7.3 Deforestation Effects on Soil Organic Carbon

Robinson et al. (1999) also reviewed estimates of the soil organic carbon losses associated with deforestation. These losses, ranging 409 to 682 Gg C yr^{-1}, were larger than those estimated for vegetation carbon. Although soil organic carbon densities exceeded vegetation carbon densities in the Cookson study area (see chapter 4), high spatial variation in soil organic carbon inhibited the ability to detect significant land use effects using a comparative mensurative research design. As a result I was unable to combine the field and map data to evaluate the management significance of soil organic carbon losses in these landscapes.

The experimental design for this study accounted for two potential causes of observed spatial variation in soil organic carbon densities: land use and topographic landform position. The expected magnitude of cultivation-induced losses was approximately 20 Mg C ha^{-1} (see Discussion in Chapters 5 and 6). The observed magnitude of topographic landform position effects was a difference of 23 Mg C ha^{-1} (Chapters 5) and 30 Mg C ha^{-1} (Chapter 6) at cultivated sites only. Both of these effects were smaller in magnitude than the observed variation in soil organic carbon attributed to differences in the degree of groundwater effects (from 50 Mg C ha^{-1} across the six forest sites to 82 Mg C ha^{-1} across six cultivated sites). Groundwater effects may be the cause of high spatial variation in soil organic carbon densities observed within the Cookson study area, but further investigation is required to identify the process mechanisms responsible.

Shallow depths to groundwater occurring within the Cookson study area are observed throughout the region from Debden to Nipawin, Saskatchewan (Glenn Padbury, Agriculture Canada, personal communication). Within this region, attempting to estimate soil organic carbon stocks at the landscape scale without accounting for groundwater effects may lead to large errors.

Assessment of the impact of deforestation and associated agricultural land uses on soil organic carbon will require long-term manipulative experiments or other research methodologies free from the confounding effects of spatial variability in soil organic carbon densities.

7.4 Future Deforestation and Reforestation

Losses of forest area of the magnitude observed prior to 1990 cannot be sustained indefinitely because the land available to be deforested is limited. Agriculture-induced deforestation has continued to occur after 1990 (personal observation), but most lands of high agricultural capability within these regions have already been converted to cultivated fields or pastures. The areal extent of deforestation may decline over time, but chronic deforestation may persist into future decades in the absence of measures to encourage forest retention by public land managers and private landowners.

If deforestation and associated terrestrial vegetation carbon losses occurring within the Boreal Plain Ecozone of Saskatchewan prior to 1990 were of significance at the national scale, then the potential for future reforestation and carbon sequestration within this region will also be important at the national scale.

Reforestation of agricultural lands has been investigated as a possible means to counteract increases in carbon dioxide concentrations in the atmosphere (Sedjo 1989, Vitousek 1991, Winjum et al. 1992, Freedman and Keith 1996) although the approach has its limitations (Jarvis 1989). Establishing forests on agricultural lands in western Canada has been evaluated for its potential to increase carbon stocks (Peterson et al. 1999, Samson et al. 1999, van Kooten et al. 1999).

Reforestation of agricultural lands will sequester carbon during the transient period of initial forest biomass growth. However the benefits also exist at the equilibrium state. Future carbon densities at reforested, former agricultural sites will exceed current carbon densities of the agricultural sites to the degree that the temporal mean vegetation carbon density of the new forests exceeds that of pasture sites or cultivated sites. As discussed above, this difference between natural forest sites and agricultural sites is approximately 50 Mg C ha^{-1}.

7.5 Potential Effects of Reforestation on Vegetation Carbon Stocks

A modest reforestation target, such as converting 2 % of open lands to natural forest cover over a 10 year period (see Chapter 3), would result in reforestation of 22 400 and 43 200 ha for the Waskesiu Hills and Red Deer River landscapes, respectively. Over the following decades, sequestration on the order of 1 100 and 2 200 Gg C would occur in the two landscapes, respectively. This requires that reforested sites revert to carbon densities observed in extant forest sites at present, and that chronic losses of extant forests do not continue. This increase in forest vegetation carbon would be several-fold larger than the losses due to deforestation over the 1975/76 to 1990 period. If it took 100 years to fully achieve the 50 Mg C ha^{-1} difference in mean carbon densities between reforested sites and agricultural sites, the rate of C accumulation for these two landscapes combined would be 33 Gg C yr^{-1}. Applying a similar reforestation target across the Boreal Plain Ecozone would have an effect an order of magnitude larger. Carbon accumulation as a result of reforestation of a small portion of this region could offset a substantial portion of agriculture-related fossil fuel burning, which was estimated as 719 Gg C yr^{-1} in 1990 (AAFC 1999).

Changes in forest area in the Boreal Plain Ecozone will be reflected on remote sensing imagery such as air photos, allowing estimation of changes in vegetation carbon stocks. However, as reported by Johnston et al. (1996), uncertainties around estimates of carbon stocks changes will be far greater in magnitude than uncertainties around estimates of carbon stocks. Changes in soil organic carbon stocks with reforestation will be very difficult to estimate at the landscape scale due to the high spatial variability of soil organic carbon, and the inability to remotely sense soil organic carbon densities.

The *a priori* assumption that only marginal agricultural land should be considered for reforestation is anachronistic in this ecologically enlightened era (Gord Miller, University of Toronto, personal communication). In order to fully realize the potential of reforestation to mitigate the effects of fossil fuel burning, forests must be seen as a valid competing use of land presently being cultivated or grazed.

7.6 Other Benefits of Reforestation

One of the strongest forces behind reforestation proposals at the present time is carbon sequestration. Increasing forest area in any spatial configuration may serve this narrow purpose.

Historic deforestation caused by agricultural expansion in central Saskatchewan changed the spatial structure of boreal landscapes (see Chapter 3). The disproportionate rate at which deforestation reduced the size of the largest wooded patches in these landscapes is a finding with important management implications. Efforts to prevent or reverse deforestation within these landscapes should place priority on protecting or expanding large patches. Large forest patches are important for protecting aquatic ecosystems, habitat-interior species, large-home-range species, and natural disturbance regimes (Forman 1995, see discussion in Chapter 3). Expanding the size of contiguous natural forest patches will conserve forest wildlife and perpetuate ecosystem processes, thereby providing supplementary benefits beyond carbon.

Based on the simulation presented in Chapter 3, it is highly unlikely that landscape spatial structure will be restored through a spatially *ad hoc* approach to reforestation. A geographically strategic reforestation program could begin to reverse the deforestation-induced changes to landscape structure. Only vigilant planning will provide society with the benefits derived from large patches of native vegetation (Forman 1995).

7.7 Role of Governments and Publicly-Owned Forest and Park Lands

Federal and provincial legislation establishing publicly owned parks and forests served to inhibit deforestation within legally protected portions of the Waskesiu Hills and Red Deer River landscapes in the three decades prior to 1990 (see Chapter 2). By preventing deforestation, protected forests also inhibited the degree of fragmentation of wooded areas within these landscapes (see Chapter 3). In Prince Albert National Park, strict legal protection also served to prevent the proliferation of roads that dissect otherwise contiguous forest areas.

Agricultural expansion in the Boreal Plain Ecozone occurred primarily through the sale of public lands. These sales convert forest trees from a public resource to private property (Gord Miller, University of Toronto, personal communication). If a serious effort at carbon sequestration for broad national public benefit is under contemplation in this region, it is well worth investigating the option of expanding the public forest land base in order to recover forest trees and the carbon they contain as a public asset.

Plantation forestry for economic profit on private land does have a role in efforts to sequester carbon in Canadian agroecosystems. However, economic factors are not yet driving widespread conversion of agricultural land that is cropped every year, to forest land that is cropped every few decades. Ratification of the Kyoto Protocol (UNFCCC 1997) by Canada might lead to economic instruments such as carbon credits that provide economic incentives for private land reforestation. Would such economic incentives be lucrative enough to entice a sizable number of agricultural producers into the forestry business? Would parallel economic mechanisms (carbon debits) be instituted as a disincentive for private land deforestation? At present there are no answers to these questions. The prospects of sequestering provincially significant stocks of carbon through voluntary reforestation actions by landowners may be limited given the recent history of private land deforestation in Saskatchewan.

Ending deforestation and increasing reforestation at provincial and national scales will require bold action by governments. Public initiatives, such as Crown acquisitions of agricultural land for the establishment of new forest-carbon reserves, might be more effective in sequestering carbon than cost-neutral, market-oriented approaches.

> "The world has filled ... land and other resources are more and more intensively contested. Compromises that could be reached in an earlier time are no longer possible. In these circumstances a new set of considerations enters the public realm, one that has more to do with what will work in a biophysical sense than what will work in an economic or political sense. The issue turns to landscapes and how to keep them integral and functioning in support of the entire human endeavor. The marketplace does not serve that function; government must" (Woodwell* 2001, p. xvii).

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9. APPENDIX A: Soil Sample Data

Site Numb	Site Numbers (see Chapter 6)
GridLett	Grid Point Column (A to C is west to east)
GridNum	Grid Point Row (1 to 4 is north to south)
SBGPCode	Soil SubGroup Code*
BD015	Bulk Density for the 0 to 15 cm depth increment
BD1530	Bulk Density for the 15 to 30 cm depth increment
BD3045	Bulk Density for the 30 to 45 cm depth increment
OC015	Organic Carbon Fraction for the 0 to 15 cm depth increment
OC1530	Organic Carbon Fraction for the 15 to 30 cm depth increment
OC3045	Organic Carbon Fraction for the 30 to 45 cm depth increment

* Agriculture Canada Expert Committee on Soil Survey, 1987. The Canadian System of Soil Classification, 2nd Ed. Agriculture Canada Publication 1646, Ottawa, ON.

Site Numb	GridLet	GridNum	SBGPCode	BD015	BD1530	BD3045	OC015	OC1530	OC3045
				g cm-3	g cm-3	g cm-3	%	%	%
C0	A	1	O.DG	1.48	1.18	1.66	1.351	0.476	0.222
C0	A	2	D.GL	1.37	1.60	1.47	1.179	0.220	0.229
C0	A	3	D.GL	1.37	1.56	1.48	0.954	0.244	0.225
C0	A	4	O.DG	1.16	1.46	1.36	2.814	1.214	0.070
C0	A	5	O.DG	1.35	1.45	1.52	1.796	0.439	0.396
C0	A	6	O.DG	1.48	1.56	1.43	1.091	0.225	0.238
C0	A	7	O.DG	1.43	1.58	1.60	1.603	0.263	0.291
C0	B	1	O.EB	1.38	1.69	1.67	0.898	0.252	0.276
C0	B	2	O.DG	1.17	1.56	1.88	1.594	0.856	0.148
C0	B	3	O.DG	1.40	1.61	1.57	1.867	0.263	0.224
C0	B	4	GLD.GL	1.46	1.51	1.80	1.365	0.226	0.127
C0	B	5	D.GL	1.34	1.57	1.50	1.921	0.378	0.197
C0	B	6	O.DG	1.14	1.37	1.72	1.584	0.217	0.125
C0	B	7	O.DG	1.29	1.33	1.45	2.108	2.015	0.347
C0	C	1	O.DG	1.55	1.62	1.44	0.699	0.165	0.269
C0	C	2	D.GL	1.47	1.57	1.58	0.571	0.197	0.206
C0	C	3	O.EB	1.43	1.80	1.60	0.784	0.182	0.186
C0	C	4	O.DG	1.38	1.51	1.63	1.405	0.332	0.155
C0	C	5	O.DG	1.39	1.53	1.70	1.936	0.215	0.156
C0	C	6	O.DG	1.39	1.59	1.58	2.273	0.688	0.192
C0	C	7	D.GL	1.47	1.43	1.57	1.142	0.165	0.168
C0	D	1	D.GL	1.49	1.46	1.73	0.501	0.141	0.196
C0	D	2	O.DG	1.67	1.63	1.61	0.522	0.197	0.081
C0	D	3	O.DG	1.36	1.41	1.63	1.879	1.014	0.248
C0	D	4	O.DG	1.44	1.61	1.31	1.916	1.485	0.373
C0	D	5	O.DG	1.36	1.49	1.41	2.271	1.536	0.344
C0	D	6	D.GL	1.45	1.51	1.51	1.239	0.282	0.197
C0	D	7	O.DG	1.22	1.61	1.66	1.434	0.490	0.110
C0	E	1	O.DG	1.50	1.53	1.41	1.346	1.288	0.882
C0	E	2	O.DG	1.39	1.61	1.53	1.382	0.378	0.203
C0	E	3	O.DG	1.43	1.65	1.64	1.596	0.139	0.080

Site Numb	GridLet	GridNum	SBGPCode	BD015	BD1530	BD3045	OC015	OC1530	OC3045
C0	E	4	E.EB	1.49	1.54	1.67	0.938	0.136	0.113
C0	E	5	O.DG	1.33	1.49	1.52	2.351	0.598	0.222
C0	E	6	O.DG	1.47	1.48	1.47	1.393	0.331	0.320
C0	E	7	O.DG	1.36	1.62	1.56	1.297	0.182	0.141
C0	F	1	O.EB	1.55	1.52	1.62	0.685	0.347	0.162
C0	F	2	O.DG	1.57	1.45	1.75	0.821	0.124	0.211
C0	F	3	O.DG	1.44	1.49	1.72	1.188	0.084	0.100
C0	F	4	O.DG	1.38	1.45	1.70	1.566	0.442	0.157
C0	F	5	O.DG	1.43	1.57	1.65	2.213	0.335	0.202
C0	F	6	D.GL	1.28	1.46	1.79	1.902	0.166	0.140
C0	F	7	O.DG	1.44	1.53	1.58	1.037	0.159	0.106
C0	G	1	O.DG	1.42	1.62	1.51	1.105	0.258	0.208
C0	G	2	O.DG	1.55	1.59	1.70	1.007	0.176	0.181
C0	G	3	O.DG	1.41	1.60	1.61	1.241	0.160	0.175
C0	G	4	O.DG	1.27	1.56	1.52	2.304	0.638	0.306
C0	G	5	O.DG	1.34	1.56	1.58	1.911	0.177	0.178
C0	G	6	O.EB	1.51	1.52	1.46	1.445	0.248	0.200
C0	G	7	O.EB	1.49	1.51	1.66	0.855	0.060	0.020
C1	A	1	CA.DG	1.45	1.50	1.28	3.694	1.314	0.938
C1	A	2	CA.DG	1.45	1.61	1.22	3.367	1.225	1.226
C1	A	3	E.EB	1.39	1.61	1.36	2.603	0.901	0.331
C1	A	4	D.GL	1.41	1.41	1.77	1.463	0.242	0.287
C1	B	1	CA.DG	1.47	1.43	1.50	2.405	0.834	0.557
C1	B	2	D.GL	1.31	1.54	1.58	5.024	1.082	0.443
C1	B	3	D.GL	1.48	1.62	1.49	2.610	0.174	0.362
C1	B	4	D.GL	1.57	1.96	1.34	2.966	0.787	0.151
C1	C	1	O.DG	1.61	1.55	1.58	2.648	0.298	0.807
C1	C	2	CA.DG	1.48	1.65	1.33	2.123	1.715	1.159
C1	C	3	O.DG	1.68	1.62	1.56	2.040	0.218	0.273
C1	C	4	O.GL	1.60	1.73	1.56	1.844	0.221	0.266
C2	A	1	D.GL	1.54	1.56	1.44	2.904	2.150	0.383
C2	A	2	O.DG	1.45	1.49	1.42	3.833	1.999	0.697
C2	A	3	HU.LG	1.44	1.69	1.61	3.199	1.973	0.197
C2	A	4	E.EB	1.75	1.48	1.42	0.875	0.393	0.410
C2	B	1	O.EB	1.40	1.58	1.41	1.623	0.209	0.147
C2	B	2	D.GL	1.54	1.63	1.68	2.441	0.419	0.121
C2	B	3	O.DG	1.45	1.87	1.38	2.611	0.590	0.014
C2	B	4	GL.DG	1.49	1.69	1.61	3.061	0.972	0.056
C2	C	1	CA.DG	1.43	1.35	1.16	1.741	1.343	1.215
C2	C	2	O.DG	1.55	1.70	1.67	1.558	0.291	0.176
C2	C	3	CA.DG	1.55	1.59	1.66	1.731	0.209	0.201
C2	C	4	GL.DG	1.46	1.36	1.44	2.893	4.327	2.148
C3	A	1	CA.DG	1.42	1.57	1.46	4.177	2.072	1.157
C3	A	2	O.DG	1.58	1.31	1.64	3.978	1.325	0.707
C3	A	3	O.DG	1.57	1.38	1.47	3.093	1.523	0.480
C3	A	4	O.DG	1.86	1.63	1.37	3.505	0.730	0.261
C3	B	1	CA.DG	1.45	1.42	1.49	4.088	2.720	1.410
C3	B	2	D.GL	1.42	1.32	1.48	3.911	3.285	1.076

Site Numb	GridLet	GridNum	SBGPCode	BD015	BD1530	BD3045	OC015	OC1530	OC3045
C3	B	3	CA.DG	1.48	1.23	1.32	3.956	3.127	2.218
C3	B	4	D.GL	1.37	1.75	1.52	3.513	1.915	0.158
C3	C	1	GLD.GL	1.48	1.64	1.76	4.933	2.414	0.239
C3	C	2	CA.DG	1.36	1.62	1.31	3.344	1.412	0.463
C3	C	3	CA.DG	1.51	1.44	1.39	2.930	1.393	0.573
C3	C	4	D.GL	1.72	1.81	1.49	2.588	0.116	0.178
C4	A	1	O.EB	1.97	1.49	1.35	0.652	0.424	0.481
C4	A	2	D.GL	1.28	1.46	1.27	2.344	0.302	0.844
C4	A	3	E.EB	1.49	1.58	1.28	1.112	0.387	0.541
C4	A	4	O.DG	1.81	1.39	1.38	2.255	1.926	0.836
C4	B	1	O.DG	1.56	1.68	1.45	0.987	0.200	0.114
C4	B	2	O.EB	2.00	2.16	1.49	1.440	0.139	0.000
C4	B	3	D.GL	1.68	1.59	1.46	2.103	0.881	0.249
C4	B	4	O.DG	1.75	1.43	1.48	1.209	0.396	1.027
C4	C	1	O.DG	1.59	1.68	1.84	1.205	0.223	0.549
C4	C	2	O.EB	1.86	1.40	1.93	0.788	0.211	0.108
C4	C	3	O.DG	1.56	1.61	1.62	1.602	1.510	1.277
C4	C	4	D.GL	1.67	1.45	1.63	0.749	0.113	0.201
C5	A	1	O.GL	1.40	1.64	1.41	1.963	0.404	0.296
C5	A	2	D.GL	1.44	1.56	1.60	2.353	0.347	0.220
C5	A	3	D.GL	1.40	1.54	1.54	4.280	1.172	0.406
C5	A	4	D.GL	1.32	1.61	1.64	5.186	0.583	0.183
C5	B	1	D.GL	1.87	1.41	1.28	1.102	0.257	0.287
C5	B	2	D.GL	1.66	1.66	1.42	1.884	0.226	0.496
C5	B	3	O.EB	1.54	1.69	1.40	2.099	0.720	0.782
C5	B	4	O.DG	1.04	1.67	1.47	8.692	1.762	0.141
C5	C	1	D.GL	1.58	1.41	1.49	2.410	0.250	0.349
C5	C	2	CA.DG	1.33	1.42	1.44	4.570	1.253	0.359
C5	C	3	GLD.GL	1.31	1.39	1.49	3.583	0.402	0.268
C5	C	4	O.DG	1.45	1.65	1.44	3.220	0.431	0.294
F0	A	1	D.GL	0.60	1.43	1.48	9.341	0.808	0.491
F0	A	2	O.GL	0.48	1.31	1.39	11.860	0.882	0.381
F0	A	3	GL.GL	0.93	1.21	1.59	4.791	0.954	0.417
F0	A	4	O.GL	0.52	1.23	1.43	12.440	0.774	0.513
F0	A	5	O.DG	0.55	1.36	1.28	9.409	1.078	0.521
F0	A	6	HU.LG	0.38	1.15	1.24	18.870	0.872	0.416
F0	A	7	O.LG	0.32	1.16	1.39	25.930	1.573	0.372
F0	B	1	E.EB	0.77	1.62	1.64	7.078	0.410	0.143
F0	B	2	O.GL	0.43	1.28	1.59	11.480	2.027	0.631
F0	B	3	O.GL	0.77	1.43	1.41	5.713	1.215	0.354
F0	B	4	O.GL	0.89	1.56	1.28	4.511	0.660	0.450
F0	B	5	O.LG	0.31	1.55	1.51	31.000	1.375	0.324
F0	B	6	O.LG	0.97	1.27	1.32	4.513	0.468	0.428
F0	B	7	DIST	0.61	1.06	1.05	7.809	2.803	2.206
F0	C	1	O.GL	0.54	1.42	1.50	13.720	1.128	0.327
F0	C	2	O.LG	0.69	1.02	1.49	5.979	0.915	0.293
F0	C	3	O.LG	0.86	1.58	1.59	6.235	1.125	0.296
F0	C	4	CA.DG	0.67	1.30	1.41	7.181	1.557	0.423

Site Numb	GridLet	GridNum	SBGPCode	BD015	BD1530	BD3045	OC015	OC1530	OC3045
F0	C	5	CA.DG	0.82	1.30	1.39	6.210	2.057	1.327
F0	C	6	GL.DG	0.88	1.50	1.55	3.439	0.591	0.159
F0	C	7	O.GL	0.86	1.31	1.37	4.971	1.479	0.713
F0	D	1	O.DG	0.82	1.52	1.45	5.336	1.084	1.151
F0	D	2	O.DG	0.95	1.59	1.63	3.650	0.234	0.161
F0	D	3	O.LG	0.88	1.58	1.51	3.472	0.556	0.624
F0	D	4	HU.LG	0.56	1.43	1.30	7.951	0.697	0.278
F0	D	5	O.G	0.57	1.45	1.53	9.387	1.395	0.435
F0	D	6	DIST	0.73	1.33	1.37	6.021	1.309	1.323
F0	D	7	CA.DG	0.95	1.19	1.34	3.503	1.975	1.285
F0	E	1	CA.DG	0.75	1.24	1.31	7.799	2.648	1.341
F0	E	2	D.GL	0.86	1.46	1.40	5.362	0.891	1.052
F0	E	3	CA.DG	0.76	1.29	1.37	5.397	2.423	1.843
F0	E	4	CA.DG	0.93	1.41	1.51	3.243	1.952	0.851
F0	E	5	O.LG	0.54	1.49	1.42	10.625	0.638	0.545
F0	E	6	HU.LG	0.16	0.50	1.27	51.320	14.160	1.144
F0	E	7	O.DG	0.94	1.40	1.55	4.102	0.927	0.401
F0	F	1	O.DG	1.04	1.47	1.62	3.723	1.500	0.791
F0	F	2	CA.DG	1.03	1.33	1.49	3.946	2.534	1.420
F0	F	3	HU.LG	0.86	1.58	1.53	5.826	0.661	0.440
F0	F	4	HU.LG	0.64	1.27	1.33	9.183	2.112	1.441
F0	F	5	O.LG	0.58	1.37	1.26	5.532	0.877	0.602
F0	F	6	HU.LG	0.50	1.28	1.44	13.500	2.252	0.511
F0	F	7	HU.LG	0.42	1.18	1.57	18.950	3.227	0.714
F0	G	1	CA.DG	0.83	1.52	1.68	6.061	1.468	0.345
F0	G	2	O.DG	0.64	1.22	1.43	7.519	2.657	1.876
F0	G	3	CA.DG	1.02	1.51	1.64	5.116	1.875	0.183
F0	G	4	O.EB	0.59	1.17	1.40	7.790	1.320	1.014
F0	G	5	HU.LG	0.64	1.59	1.59	10.190	0.351	0.399
F0	G	6	D.GL	0.80	1.43	1.56	4.723	0.507	0.441
F0	G	7	D.GL	0.78	1.17	1.45	3.948	0.873	0.608
F1	A	1	O.GL	0.78	1.14	1.34	3.562	0.789	0.726
F1	A	2	D.GL	0.92	1.55	1.64	4.054	0.848	0.896
F1	A	3	E.EB	0.81	1.24	1.35	8.528	2.166	0.906
F1	A	4	O.DG	0.83	1.02	1.25	7.013	2.738	1.312
F1	B	1	D.GL	0.63	1.21	1.83	6.372	1.453	0.670
F1	B	2	D.GL	0.77	1.27	1.14	4.054	1.097	1.638
F1	B	3	E.EB	0.50	1.22	1.32	13.130	1.142	1.623
F1	B	4	O.GL	0.81	1.38	1.21	3.861	0.699	0.377
F1	C	1	D.GL	0.35	0.83	1.62	14.800	5.181	0.508
F1	C	2	O.LG	0.68	1.21	1.24	7.572	0.600	0.231
F1	C	3	O.LG	0.95	1.35	1.49	3.908	0.555	0.456
F1	C	4	O.GL	0.99	1.48	1.45	4.502	0.636	0.547
F2	A	1	O.DG	0.62	1.37	1.38	7.636	0.706	0.246
F2	A	2	E.EB	0.93	1.38	1.55	3.092	0.834	0.304
F2	A	3	O.DG	1.02	1.26	1.46	2.736	0.448	0.391
F2	A	4	HU.LG	0.88	1.40	1.36	4.910	0.519	0.170
F2	B	1	O.GL	0.52	1.35	1.51	7.198	0.609	0.413

Site Numb	GridLet	GridNum	SBGPCode	BD015	BD1530	BD3045	OC015	OC1530	OC3045
F2	B	2	D.GL	0.91	1.38	1.50	5.364	0.328	0.431
F2	B	3	O.DG	1.05	1.31	1.62	3.980	0.933	0.290
F2	B	4	O.EB	1.23	1.69	1.33	2.970	0.356	0.437
F2	C	1	D.GL	0.83	1.38	1.27	4.002	0.372	0.212
F2	C	2	O.DG	0.94	1.50	1.24	6.637	0.791	0.842
F2	C	3	O.DG	0.98	1.42	1.41	5.218	1.159	0.255
F2	C	4	D.GL	1.07	1.41	1.54	4.459	0.708	0.203
F3	A	1	E.EB	0.68	1.32	1.67	8.760	0.628	0.317
F3	A	2	CA.DG	0.96	1.59	1.57	4.197	0.825	1.118
F3	A	3	CA.DG	0.54	1.05	1.54	7.648	3.503	1.879
F3	A	4	CA.DG	0.73	1.08	1.32	7.498	4.470	2.554
F3	B	1	O.GL	0.93	1.41	1.49	3.313	0.412	0.118
F3	B	2	O.GL	0.89	1.34	1.66	4.742	0.236	0.419
F3	B	3	O.GL	0.86	1.41	1.62	3.158	0.855	0.312
F3	B	4	E.EB	0.70	1.24	1.46	5.667	1.069	0.284
F3	C	1	O.GL	0.98	1.54	1.73	3.378	0.411	0.187
F3	C	2	O.GL	0.67	1.30	1.55	5.949	0.345	0.346
F3	C	3	E.EB	0.82	1.42	1.35	6.296	0.816	0.376
F3	C	4	O.GL	0.77	1.30	1.44	3.849	0.869	0.542
F4	A	1	O.EB	0.95	1.34	1.37	2.505	0.587	0.370
F4	A	2	E.EB	0.59	1.28	1.61	7.149	0.929	0.264
F4	A	3	D.GL	0.65	1.33	1.44	5.406	0.403	0.131
F4	A	4	O.GL	0.99	1.47	1.57	2.970	0.324	0.344
F4	B	1	O.EB	1.02	1.34	1.45	2.955	0.612	0.180
F4	B	2	O.EB	0.89	1.28	1.34	3.792	0.255	0.158
F4	B	3	O.DG	0.72	1.33	1.41	5.701	0.632	0.377
F4	B	4	D.GL	0.69	1.27	1.50	6.397	1.060	0.197
F4	C	1	O.GL	0.79	1.24	1.20	4.793	0.699	0.255
F4	C	2	E.EB	0.93	1.27	1.62	3.458	0.271	0.180
F4	C	3	D.GL	0.55	1.26	1.69	7.541	0.672	0.342
F4	C	4	E.EB	0.71	1.14	1.19	7.630	1.719	0.678
F5	A	1	O.GL	0.96	1.18	1.25	2.142	0.479	0.377
F5	A	2	O.GL	0.94	1.31	1.69	2.699	0.252	0.222
F5	A	3	O.GL	0.74	1.59	0.15	5.171	0.397	0.207
F5	A	4	O.GL	0.76	1.54	1.76	6.143	0.660	0.289
F5	B	1	O.GL	0.95	1.73	1.32	3.060	0.211	0.491
F5	B	2	O.GL	0.78	1.22	1.37	4.239	0.999	0.203
F5	B	3	O.LG	0.67	1.49	1.61	5.476	0.593	0.371
F5	B	4	O.GL	1.08	1.64	1.72	3.270	0.493	0.180
F5	C	1	O.GL	0.88	1.33	1.70	2.476	0.147	0.176
F5	C	2	O.GL	0.92	1.24	1.48	3.923	0.548	0.266
F5	C	3	O.GL	0.91	1.54	1.66	3.419	0.318	0.224
F5	C	4	GL.GL	0.91	1.42	1.99	2.644	0.468	0.249
P0	A	1	R.HG	0.24	1.24	1.53	30.680	2.073	0.243
P0	A	2	HU.LG	0.49	1.28	1.27	10.570	2.548	0.217
P0	A	3	O.HG	1.06	1.53	1.56	5.494	1.371	0.514
P0	A	4	CA.DG	1.36	1.54	1.59	2.817	0.729	0.297
P0	A	5	O.DG	1.10	1.42	1.62	5.070	1.316	0.794

Site Numb	GridLet	GridNum	SBGPCode	BD015	BD1530	BD3045	OC015	OC1530	OC3045
P0	A	6	O.DG	1.06	1.43	1.51	4.927	1.054	0.668
P0	A	7	O.DG	1.19	1.44	1.50	2.751	0.888	0.462
P0	B	1	CA.DG	1.33	1.28	1.36	3.552	2.246	0.515
P0	B	2	R.HG	1.12	1.67	1.85	3.870	1.315	0.093
P0	B	3	D.GL	1.26	1.36	1.72	3.481	0.998	0.314
P0	B	4	D.GL	1.39	1.30	1.38	2.989	1.147	0.481
P0	B	5	O.DG	1.23	1.40	1.41	4.678	1.223	1.063
P0	B	6	D.GL	1.08	1.33	1.57	5.842	1.016	0.318
P0	B	7	O.DG	1.18	1.59	1.51	3.538	0.916	0.452
P0	C	1	CA.DG	1.24	1.39	1.42	3.298	1.027	1.243
P0	C	2	O.DG	1.16	1.35	1.43	3.883	1.303	0.817
P0	C	3	D.GL	1.23	1.42	1.59	3.073	1.030	0.319
P0	C	4	CA.DG	1.18	1.31	1.39	4.012	1.589	1.346
P0	C	5	D.GL	1.24	1.34	1.65	3.213	0.695	0.279
P0	C	6	D.GL	1.13	1.45	1.41	5.391	1.152	0.996
P0	C	7	O.DG	1.13	1.29	1.47	4.678	1.058	0.487
P0	D	1	O.DG	0.97	1.30	1.37	5.775	2.209	1.062
P0	D	2	O.DG	1.30	1.48	1.48	2.674	0.519	0.234
P0	D	3	O.DG	1.26	1.40	1.64	3.085	0.645	0.238
P0	D	4	O.DG	1.31	1.37	1.32	3.816	1.347	0.846
P0	D	5	O.DG	0.97	1.29	1.47	5.141	2.227	1.121
P0	D	6	O.DG	1.08	1.44	1.52	5.072	1.274	0.807
P0	D	7	O.DG	1.10	1.29	1.33	5.309	1.716	0.565
P0	E	1	CA.DG	1.13	1.55	1.53	3.139	1.473	0.428
P0	E	2	O.DG	1.40	1.62	1.42	1.812	0.186	0.151
P0	E	3	O.DG	1.19	1.33	1.41	3.875	1.146	0.261
P0	E	4	D.GL	1.27	1.43	1.96	3.185	0.185	0.182
P0	E	5	O.DG	1.17	1.41	1.56	5.112	1.346	0.313
P0	E	6	CA.DG	1.07	1.44	1.63	5.087	1.449	0.162
P0	E	7	O.DG	1.36	1.43	1.56	3.479	0.779	0.255
P0	F	1	O.HG	1.28	1.33	1.32	3.174	1.678	0.460
P0	F	2	O.DG	1.17	1.51	1.73	3.463	0.384	0.188
P0	F	3	O.HG	1.35	1.42	1.52	2.785	0.222	0.154
P0	F	4	O.DG	1.33	1.59	1.55	2.814	0.367	0.207
P0	F	5	D.GL	1.29	1.55	1.54	2.964	0.608	0.190
P0	F	6	CA.DG	1.09	1.52	1.75	5.696	1.913	0.261
P0	F	7	O.DG	1.11	1.48	1.64	4.186	1.149	0.168
P0	G	1	CA.DG	1.00	1.27	1.24	5.733	1.569	1.058
P0	G	2	D.GL	1.39	1.44	1.30	2.181	0.323	0.350
P0	G	3	O.HG	1.44	1.58	1.76	1.872	0.275	0.170
P0	G	4	O.HG	1.55	1.69	1.85	2.243	0.057	0.168
P0	G	5	D.GL	1.12	1.27	1.49	3.992	0.779	0.295
P0	G	6	O.DG	1.20	1.53	1.85	4.300	1.199	0.325
P0	G	7	O.HG	1.27	1.32	1.52	2.973	1.661	0.241
P1	A	1	O.DG	1.63	1.64	1.55	1.357	0.189	0.166
P1	A	2	O.DG	1.47	1.50	1.38	2.002	0.230	0.049
P1	A	3	O.GL	1.51	1.51	1.24	1.070	0.213	0.160
P1	A	4	O.DG	1.47	1.39	1.72	2.225	1.465	0.101

Site Numb	GridLet	GridNum	SBGPCode	BD015	BD1530	BD3045	OC015	OC1530	OC3045
P1	B	1	D.GL	1.48	1.71	1.49	1.871	0.135	0.095
P1	B	2	HU.LG	1.43	1.64	1.66	1.109	0.443	0.054
P1	B	3	O.LG	1.24	1.85	1.43	2.646	0.145	0.208
P1	B	4	O.DG	1.57	1.29	1.51	1.355	1.683	0.198
P1	C	1	D.GL	1.50	1.53	1.57	1.434	0.095	0.097
P1	C	2	O.DG	1.29	1.70	1.71	2.313	0.371	0.124
P1	C	3	D.GL	1.32	1.53	1.46	1.684	0.327	0.055
P1	C	4	D.GL	1.52	1.30	1.69	1.413	1.428	0.169
P2	A	1	D.GL	1.39	1.57	1.57	2.231	0.147	0.172
P2	A	2	D.GL	1.14	1.42	1.50	3.329	0.492	0.130
P2	A	3	D.GL	1.40	1.63	1.52	1.829	0.152	0.219
P2	A	4	D.GL	1.43	1.60	1.57	1.458	0.140	0.160
P2	B	1	D.GL	1.47	1.58	1.80	2.201	0.195	0.130
P2	B	2	D.GL	1.48	1.55	1.59	1.108	0.082	0.120
P2	B	3	D.GL	1.34	1.68	1.36	1.442	0.201	0.097
P2	B	4	D.GL	1.25	1.57	1.64	2.385	0.105	0.216
P2	C	1	D.GL	1.53	1.47	1.62	1.512	0.143	0.193
P2	C	2	D.GL	1.45	1.43	1.68	1.980	0.079	0.128
P2	C	3	D.GL	1.31	1.59	1.47	1.828	0.142	0.213
P2	C	4	D.GL	1.58	1.48	1.78	0.836	0.272	0.324
P3	A	1	D.GL	0.95	1.27	1.45	4.306	1.573	0.230
P3	A	2	D.GL	1.33	1.45	1.44	2.511	0.193	0.243
P3	A	3	D.GL	1.43	1.93	1.74	2.522	0.222	0.128
P3	A	4	D.GL	1.41	1.81	1.48	2.786	0.499	0.190
P3	B	1	D.GL	1.00	1.82	1.34	4.924	0.580	0.405
P3	B	2	D.GL	1.20	1.62	1.34	3.631	0.130	0.229
P3	B	3	D.GL	1.43	1.84	1.80	3.591	0.436	0.182
P3	B	4	D.GL	1.67	1.63	1.93	2.759	0.468	0.213
P3	C	1	D.GL	1.13	1.41	1.71	4.006	0.337	0.259
P3	C	2	O.DG	1.26	2.01	1.83	2.357	0.269	0.303
P3	C	3	D.GL	1.57	1.59	1.92	3.696	0.605	0.200
P3	C	4	D.GL	1.49	1.86	2.04	3.949	0.596	0.165
P4	A	1	D.GL	1.17	1.31	1.87	4.093	1.067	0.221
P4	A	2	D.GL	1.34	1.83	1.65	2.566	0.126	0.186
P4	A	3	D.GL	1.11	1.39	1.55	3.297	0.497	0.287
P4	A	4	O.GL	1.44	1.49	1.64	1.451	0.333	0.187
P4	B	1	D.GL	1.31	1.68	1.40	1.523	0.185	0.214
P4	B	2	O.GL	1.14	1.39	1.86	2.590	0.620	0.214
P4	B	3	O.DG	0.98	1.47	1.87	6.037	0.367	0.499
P4	B	4	D.GL	1.12	1.63	1.60	2.454	0.589	0.448
P4	C	1	D.GL	1.13	2.01	1.28	2.221	0.272	0.235
P4	C	2	D.GL	1.28	1.44	1.67	2.217	0.322	0.252
P4	C	3	D.GL	1.16	1.72	1.45	2.866	0.207	0.218
P4	C	4	O.DG	1.04	1.49	1.61	4.387	0.613	0.378
P5	A	1	D.GL	1.52	1.60	1.49	2.205	0.326	0.178
P5	A	2	D.GL	1.37	1.64	1.46	2.419	0.342	0.361
P5	A	3	D.GL	1.50	1.49	1.65	2.246	0.199	0.231
P5	A	4	O.DG	1.43	1.30	1.23	3.054	0.924	0.698

Site Numb	GridLet	GridNum	SBGPCode	BD015	BD1530	BD3045	OC015	OC1530	OC3045
P5	B	1	D.GL	1.41	1.38	1.71	1.608	0.120	0.218
P5	B	2	D.GL	1.44	1.52	1.41	1.994	0.210	0.207
P5	B	3	O.DG	1.40	1.54	1.26	2.725	0.524	0.591
P5	B	4	O.DG	1.55	1.56	1.73	1.160	0.137	0.046
P5	C	1	D.GL	1.44	1.42	1.50	2.640	0.744	0.127
P5	C	2	O.DG	1.44	1.19	1.43	1.652	0.831	0.343
P5	C	3	D.GL	1.39	1.61	1.63	2.288	0.110	0.234
P5	C	4	O.DG	1.58	1.50	1.73	1.109	0.111	0.045