STRUCTURE AND DYNAMICS OF THE VEGETATION IN WAPUSK NATIONAL PARK AND THE CAPE CHURCHILL WILDLIFE MANAGEMENT AREA OF MANITOBA: COMMUNITY AND LANDSCAPE SCALES

By

Ryan K. Brook

A thesis
Submitted to the Faculty of Graduate Studies
of the University of Manitoba
in partial fulfilment of the requirements for the degree of
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Declaration Regarding Contribution to Collaborative Research

The research in Chapter 4 of this thesis was conducted in collaboration with Dr. N.C. Kenkel at the University of Manitoba, Department of Botany. R.K. Brook's contribution was to collect all field data in the study area, interpret the analysis results and discuss the theory. Dr. Kenkel provided guidance on study design and statistical analysis. This chapter has been accepted for publication by the International Journal of Remote Sensing from R.K. Brook and N.C. Kenkel, titled: A multivariate approach to vegetation mapping of Manitoba's Hudson Bay Lowlands.

ABSTRACT

Vegetation in the Hudson Bay Lowlands of Manitoba exists as a complex mosaic of tundra and treed communities with a high level of complexity at all scales of consideration. The wide range of vegetation types reflects local variations in climate, geological history, permafrost, fire, wildlife grazing and human use. Characterizing the structure, floristic composition, distribution, and dynamics of the ecosystem is important in order to make informed management decisions. This study, in Wapusk National Park and the Cape Churchill Wildlife Management Area of northern Manitoba, uses a combination of satellite imagery (1996 Landsat-5 TM mosaic) and extensive ground data to characterize the vegetation of the region at both the landscape and community scales.

Field data on the floristic composition, moisture, and vegetation structure were collected at 600 sites throughout the study area between June and September of 1998-2000. From these data, sites with highly unique floristic characteristics, disturbance features, and unvegetated sites were recognized as distinct vegetation types and removed from further analysis. The remaining 272 sites, containing 114 species, were sorted into 6 major vegetation types using cluster analysis. Canonical correspondence analysis (CCA) was then used to examine the extent to which trends in the vegetation types reflect environmental variability. Environmental variables included soil moisture, latitude, distance to the Hudson Bay coast, surficial geology and nutrient availability. Results of the CCA indicate a strong relationship between the vegetation types and their relative position in relation to the Hudson Bay coast that also closely corresponds to wetness and nutrient availability. This indicates that, as the land rises out of Hudson Bay due to isostatic uplift, peat accumulates and lifts the ground surface above the water table, making the growing zone dryer and more nutrient poor. As the conditions change along this environmental gradient, plant communities change from fen types that are dominated by graminoids and mosses to bog types dominated by trees and/or lichens.

Vegetation types were examined in order to identify a set of 16 classes that were suitable for mapping at the landscape scale using satellite imagery. Principal component analysis (PCA) was used to examine the spectral properties of these classes and to identify outliers. Multiple discriminant analysis was then applied to determine the statistical separability of the vegetation classes in the satellite imagery. Finally, redundancy analysis was used to determine the amount of vegetation variance explained by the spectral reflectance data.

The Landsat 5-TM satellite image mosaic was then classified into 16 classes to develop a vegetation map for the entire study area. Unsupervised classifications were coupled with a layered, stratified approach for vegetation class separation. Results indicate that the map is of reasonable accuracy (97% overall) for most applications, though the accuracy of each class varied from 71-100%. The relationship between bog and fen classes is significantly influenced by distance from the Hudson Bay coast, with bog classes becoming increasingly common and fen classes declining in relative cover

with increasing distance. This application of the vegetation map illustrates the utility of a regional scale digital map in relating large-scale patterns of vegetation distribution to ecosystem-level processes.

The existing vegetation in the study area is primarily the result of a concurrent spatio-temporal gradient inland from the coast, reflecting ongoing isostatic emergence of the area from Hudson Bay along with climatic influences that are strongly influenced by air masses in Hudson Bay. The three major vegetation types are fen, bog and upland, with the fen and bog types being associated with organic soils and the upland types being associated with mineral soils. Vegetation is highly dynamic in the Hudson Bay Lowlands as a result of ongoing isostatic uplift and peat accumulation in conjunction with climatic variability, permafrost aggradation and degradation, fire and wildlife grazing. The highly complex interrelationships between these processes make prediction of future changes difficult at fine scales. However, at the landscape scale, vegetation can be expected to continue to become drier and more nutrient impoverished with increasing distance from the Hudson Bay coast.

The information presented from the approach developed here establishes a solid basis for understanding the structure and dynamics of the vegetation in the region. The approach developed to examine site level and landscape level structure and processes will serve to guide similar projects in the sub-arctic and arctic in Canada's national parks and elsewhere.

The information on landscape structure and dynamics from this thesis will form a baseline data set for the newly created Wapusk National Park against which future change can be measured. Descriptions of the vegetation communities, maps of the vegetation distribution at the landscape scale and interpretation of the dynamics at both plot and landscape scales, provides the basic knowledge necessary for making management planning decisions regarding land use and habitat sensitivity. The information can also be used to guide future research and monitoring efforts.

ACKNOWLEDGEMENTS

The three (plus!) years that went into this thesis represent one of the greatest challenges of my life, and writing these acknowledgements has reminded me of just how many people made significant contributions to my learning in general and my thesis in particular. I would like to send a general thank you to the multitude of people that helped me whenever I needed it, including those that I may neglect to mention by name.

I would like to acknowledge the guidance and support of Norm Kenkel for his unfailing enthusiasm for my project and willingness to make time to help me. I have learned so much from his sensible and rigorous approach to data collection and analysis. His many long hours of work helping with data crunching and writing are very much appreciated. The quality of this thesis is largely a result of his input throughout.

My advisory committee Helen Fast, Norm Kenkel, Doug Clark, and David Mosscrop have provided input at all stages of this project. Doug Clark acted as a valuable committee member, providing considerable insight in reviewing previous drafts of this thesis and also played a critical role in co-ordinating field logistics, making my time in Wapusk productive and safe.

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DEDICATION

This thesis is dedicated to my Mom and Dad for teaching me the greatest lesson in my life: "You don't quit when you are tired. You don't quit when you are bored. You quit when the job is done." These words were never spoken. They didn't have to be. They were demonstrated every single day. Thanks so much for making me into the person that had the confidence to even try to do this project and for giving me the patience to get it done.

"What would you do if you knew you could not fail?"

-Author Unknown

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CHAPTER 1: GENERAL INTRODUCTION

"In going northward, there is of course, a gradual diminution in size of the trees and the height of the forest, as well as in the number of species. Owing, however, to the fires which sweep over large tracts at different periods, it is seldom that one sees the full size to which the trees are capable of growing."
-R. Bell, geologist, on explorations in northern Manitoba, 1881.

INTRODUCTION

Vegetation exists as a complex mosaic across the landscape as a result of a wide variety of factors that influence plant growth, reproduction and dispersal. The plant communities present in a region are determined by broad-scale conditions such as climate, geology, soils and length of the growing season. At a finer scale, local characteristics of exposure, temperature, snow cover, soil texture, drainage, water chemistry, slope and aspect determine vegetation distribution and composition. The result of these various influences is a generally high variability in vegetation across the landscape. Concurrent with this level of spatial variability, vegetation is also highly dynamic. Successional processes and disturbance events such as fire, wildlife grazing and human activity can replace existing communities with a new floristic composition over time.

Ultimately, the communities present at a certain point in time are the result of all of the cumulative influences of the environmental factors and processes working together simultaneously. The arrangement or structural pattern of vegetation patches and the matrix that constitutes a landscape is a major determinant of functional flows and movements through the landscape, and of changes in its pattern and process over time

(Forman 1995). Understanding the vegetation that exists in an area can provide insights into the long-term nature of the environment and observations of change can be an important indicator of environmental modification. As a result, natural resource managers generally recognize the importance of vegetation in the management of global, regional, landscape and site level issues. Management decisions often require information about the structure, composition and distribution of vegetation across an entire region. Monitoring of progress toward management goals requires scientific evaluation of a range of factors that include vegetation. However, characterizing the vegetation over large areas is a significant challenge.

Until very recently, most of what was known about the structure and function of the earth's vegetation came from highly localized in situ studies. Aerial photography did not begin being commonly used until after World War II (Spurr 1948). The use of aerial photos for mapping extensive areas is labor intensive and the images must be obtained through specified flights that are very expensive and this often precludes regular updating. In contrast, satellite remote sensing provides an extensive data source for mapping vegetation at a variety of scales. A number of satellite platforms are available to satisfy a wide range of spatial and temporal resolution requirements. The digital nature of satellite data allows mapping to be a dynamic and flexible process so maps can be continually updated, as new information becomes available. Information can then be presented in an optimized format to satisfy the specific requirements of each question being asked.

The integration of geographic information systems (GIS) and global positioning system (GPS) technology with remote sensing, collectively provide powerful tools for mapping vegetation and characterizing landscape and plant community structure. This approach has also been used for a wide range of applications that use vegetation maps as input into analyses such as modelling habitat suitability, wildlife distribution and abundance, land use planning, habitat management, landscape sensitivity mapping, biodiversity monitoring, and studies of temporal change (Urban *et al.* 1987, Turner 1989, Aspinall and Veitch 1993, Lyon *et al.* 1998, Debinski *et al.* 1999). Wildlife habitat analysis has been the area of the greatest application for species such as caribou, *Rangifer tarandus* (Clifford 1979, Nenonen and Nieminen 1990, Colpaert *et al.* 1995), moose, *Alces alce*s (Oosenbrug *et al.* 1988), Muskox, *Ovibos moschatus* (Ferguson 1991, Pearce 1991), bison, *Bison bison athabascae* (Matthews 1991), wolves, *Canis lupus* (Mladenoff *et al.* 1995, Conway 1996), polar bears, *Ursus maritimus* (Clark 1996) and snow geese, *Anser caerulescens caerulescens* (Jano *et al.* 1998).

Satellite remote sensing provides ecologists and natural resource managers with a tool of tremendous potential value but only if they understand its capabilities and limitations. Use of this technology to support ecosystem based landscape management is still in its early developmental stages. Remote sensing provides information in the form of raw data that requires careful interpretation through rigorous statistical analysis of the image content in conjunction with field data collection. If the information is to be relevant, it must match the scale of management. The information must also be gathered in a rigorous and unbiased way and interpreted as objectively as possible. The information must be presented in a way that will provide useful insights to management policy and it must be communicated in a way that is understandable to the people who would use it. Remote sensing data provides the opportunity to extend the knowledge gained from intensive *in situ* ecological research to larger areas, at more frequent intervals and over a longer time series. If satellite imagery is carefully translated into meaningful information in an optimized manner, it can provide a very powerful tool for decision-making and research.

BACKGROUND

The vegetation in the Hudson Bay Lowlands in Manitoba forms a broad transition zone between the boreal forest to the south and the arctic tundra further north. Much of the early scientific studies of the vegetation focused on general descriptive work as well as species collection and identification (Bell 1881, Bell 1886, Beckett 1959, Ritchie 1956, Scoggan 1959). Generalized descriptions of plant communities were conducted by Shelford and Twomey (1941), McClure (1943), Ritchie (1957, 1960), Rewcastle (1983) and Johnson (1987). Related work in the Hudson Bay Lowlands in Ontario was done by Kershaw and Rouse 1973, Kershaw 1974, 1977, Sims *et al.* (1979, 1982), and Pala and Weischet (1982). More recently, considerable research on fine scale vegetation processes has been undertaken in the region. Studies have been mostly limited to small, intensely studied areas ($<100km^2$). Tree line dynamics have been examined by Scott *et al.* (1987) and Tews (2000). Salt marsh vegetation at La Perouse Bay has been described and studied by Jefferies (1979) and numerous other publications. However, few studies have examined ecosystem level structure or processes.

Ritchie (1962) initiated regional mapping of the plant communities at 1: 1,000,000 scale, using black and white air photo interpretation. Clark (1996) used LANDSAT TM satellite imagery to map the region at 1: 1,000,000 scale. While these studies have described the broad distribution of vegetation zones in the region with some success, no fine scale regional mapping of the vegetation had been completed prior to the current study. The previous maps also do not distinguish many of the unique communities that exist such as sedge meadows, beach ridges and coastal salt marsh. They also do not identify disturbance features such as burns.

ISSUE STATEMENT

The establishment of Wapusk National Park in 1996 has recently changed the management of the Hudson Bay Lowlands in Manitoba. WNP is under the jurisdiction of Parks Canada, a federal agency, but includes a co-management board that is comprised of federal, provincial, Fox Lake First Nation, York Factory First Nation and Town of Churchill representatives (Wapusk National Park 1998).

Development of a Park Management Plan is currently underway in a process that will develop a vision for the park for the future (Parks Canada 2001). Included in the development of the management plan is an Ecological Integrity Statement that defines the structural features and processes that make up Wapusk and develops objectives for protecting ecological integrity (M. Manseau pers. comm.1999, Parks Canada 2001). Concurrent with the management planning in Wapusk, the Cape Churchill Wildlife Management Area Management Plan is being revised by the Manitoba Department of Conservation (C. Elliot, pers. comm. 2000). Both federal and provincial governments, along with associated agencies and partners, must make a number of decisions that will significantly influence the future of the region. At the same time, operational decisions must be made that somehow fit into the broader goals and objectives. These decisions require a comprehensive understanding of the landscape at a wide range of scales.

Little work has been done to establish a consensus with management agencies and stakeholders regarding specific questions regarding vegetation in the region. A meeting of management agencies in 1996 after the establishment of Wapusk identified the following as research priorities in Wapusk and the Hudson Bay Lowland area: biodiversity monitoring programs; baseline data (particularly for inland, remote areas of the park); a detailed vegetation map of the Hudson Bay Lowland area; permafrost influences; fire history; and vegetation gradients (Churchill Northern Studies Centre 1996). Other questions that have been raised about the composition of the vegetation in the region at the landscape scale have often focused on the number and type of plant communities present, where they are located and how they are distributed across the landscape (personal communications with D. Clark 1999, M. Peniuk 1999, J. Leger 1999, E. Depatie 1999).

While vegetation is generally recognized as a primary ecosystem component, it has only been described in very general terms in Wapusk National Park and the Cape Churchill Wildlife Management Area (e.g. Ritchie 1957, 1960, Rewcastle 1983, and Johnson 1987). While much of the research conducted to date has been important for understanding some of the processes occurring (e.g. Bazely and Jefferies 1986, Scott *et al.* 1987, Hik and Jefferies 1990, Tews 2000), the patterns of the landscape have not been described in any detail. Basic ecological information regarding the composition, structure and distribution of the vegetation communities is lacking. Most of the vegetation research conducted to date has been done within less than 50 km of Churchill and so is not necessarily representative of the entire Hudson Bay Lowlands of Manitoba.

Ecological information characterizing landscape structure is required to develop a comprehensive knowledge regarding all ecosystem components to implement management of the area as a landscape (Thompson and Welsh 1993). The information required for decision-making must include descriptions of the pattern, size and thematic character of the vegetation at the landscape scale (Gluck *et al.* 1996), as well as the structure and species composition at the site level. In order to undertake ecosystem management, resource units and ecosystem boundaries must be also defined. A general understanding of the processes that influence vegetation dynamics at multiple scales is also essential.

The development of an accurate and useful vegetation map is a complex and labour intensive process. Landscape level vegetation mapping has not yet developed to the point where the resulting products are consistently of high utility and accuracy. Methods need to be developed to classify and map vegetation in a manner that ensures the map is of high utility. Landscape level data is essential so that scientific knowledge and management experience can be organized, communicated and applied effectively. Moreover, a clearer knowledge of vegetation at the site level provides an important basis for a better understanding of the processes and ecosystem relationships that exist and interact at broader landscape levels.

THESIS OBJECTIVES

This study provides baseline ecological information on the structure, composition, spatial distribution, and dynamics of the major vegetation communities within Wapusk National Park and the northern portion of the Cape Churchill Wildlife Management Area. A detailed examination of the relationship between the vegetation and the environmental conditions is also provided, along with comparisons between landscape and community scales.

Specific objectives were to:

1. delineate the major vegetation communities of the region using quantitative methods analyzing species cover, plant life form and environmental data.
2. summarize the floristic composition, community structure and vegetation-environment relationships associated with each community and identify vegetation dynamics based on the relationships observed.
3. construct a geocoded Landsat TM satellite image mosaic of the study area and examine the spectral reflectance values of the different vegetation communities in order to produce a set of mutually exclusive vegetation classes.
4. classify all Landsat TM pixels into one of the defined vegetation classes, perform an accuracy assessment of the classification using an independent data set, and use the vegetation map to infer landscape level vegetation dynamics.
5. provide suggestions on ways in which the plant community descriptions and the vegetation map can be integrated with existing data to support decision making in Wapusk National Park and the Cape Churchill Wildlife Management Area and identify future research needs.

STUDY AREA

The study area includes Wapusk National Park (11, 475 km^2) and the northern portion of the Cape Churchill Wildlife Management Area (CCWMA) (7, 295 km^2), which together encompass a large portion of the Hudson Bay Lowlands of Manitoba, Canada (Fig. 1.1). The area was chosen to include all of Wapusk at the request of Parks Canada and the northern portion of the CCWMA was included since it was covered by the existing satellite imagery and included a number of field camps which facilitated ground sampling.

The area is a flat, extensive coastal plain that forms a broad transition zone between continuous boreal forest to the south and arctic tundra to the north. The transition zone includes a dramatic change in both vegetation (Ritchie 1956, Scoggan 1959) and wildlife (Jehl and Smith 1970, Wrigley 1974).

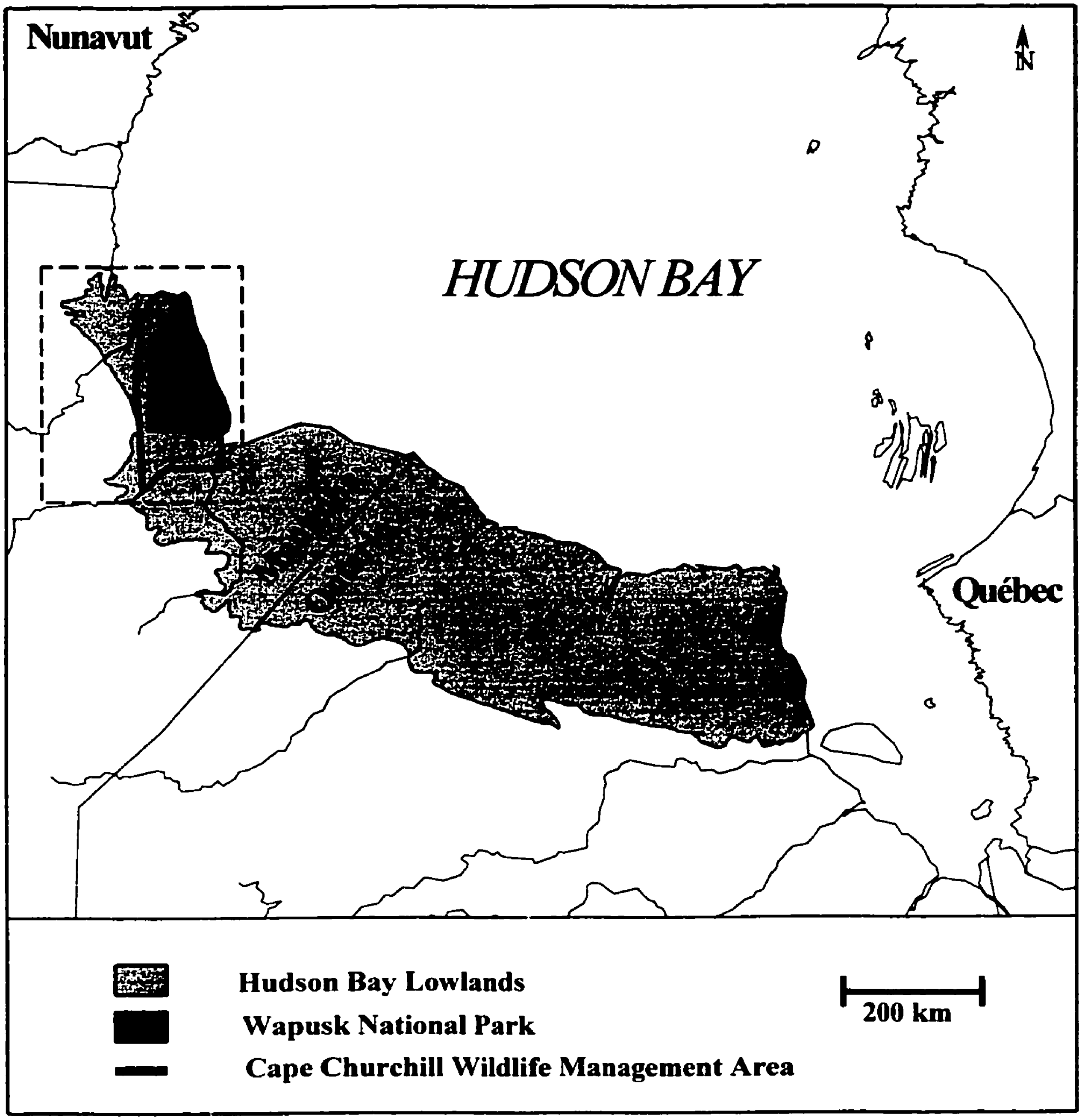


Figure 1.1. Wapusk National Park and the Cape Churchill Wildlife Management Area situated within the Hudson Bay Lowlands.

Hudson Bay has a dominant influence on the marine sub-arctic climate of the region (Rouse 1991), creating a northeast-southwest thermal gradient. This thermal gradient is summarized by comparing Churchill on the coast and Gillam, situated 150 km inland. Mean annual temperature is –7.3 °C at Churchill and –4.8 °C at Gillam (Dredge 1992). The July monthly average is 12°C at Churchill and 15°C at Gillam, while January monthly means are -28 °C at Churchill and -26°C at Gillam (Dredge and Nixon 1992). The average growing season varies from 100 to 143 days and the number of growing degree-days ranges from 500-1000 (Smith *et al.* 1998). Annual average precipitation is 400 mm at Churchill and 280 mm at Gillam. About half of the precipitation occurs as snow at both sites and prevailing winds are from the north-west during the entire year. The study area is underlain largely by continuous permafrost, except under lakes and streams. Permafrost is discontinuous only in the southern region (Dredge and Nixon 1992).

Geology of the region is described by Sjörs (1959) and Dredge and Nixon (1992). During the Pleistocene epoch the study area was covered by the Wisconsin glacier, which was 1.5-3 km thick. The weight of the glacier pushed down the Hudson Bay Lowlands region by approximately 600 vertical meters. Following deglaciation (approximately 7800 years ago), the entire study area was covered by the Tyrell Sea to a depth of 165 m above present sea level at Churchill and about 135 m above present sea level at Gillam. The land began emerging from the sea rapidly, immediately following deglaciation, rising approximately 5m per century due to isostatic uplift. The rate of emergence has since declined to a current rate of 0.5-1.0 m per century, which results in as much as 50-100 m of new shoreline per century (Dredge and Nixon 1992). Drainage in the study area is exceptionally poor as a result of a particularly gentle regional slope, low local relief, and general impermeability of the fine-grained, ice bonded substrate (Dredge and Nixon 1992).

Plant communities of the Hudson Bay Lowlands include uplands and extensive fen and bog complexes of highly variable tree cover intermixed with vast numbers of ponds and lakes (Ritchie 1956, Sims *et al.* 1982, Pala and Weischet 1982). The major tree species are *Picea mariana*, *Picea glauca,* and *Larix laricina*. Grazing by large herbivores, including geese and caribou, has significantly altered plant community structure in some areas (Campbell 1995, Ganter *et al.* 1996). Lightning induced fire is a frequent and recurring process on the landscape. Intensive human disturbance has occurred only in localized areas. The dynamic nature of the region, through the combined and often synergistic effects of climate, isostatic uplift, snow cover, drainage, grazing and fire create a complex mosaic of vegetation both spatially and temporally.

THESIS ORGANIZATION

The remaining body of this thesis is presented in six chapters, with each developed as an individual manuscript in the format suitable for publication in a refereed journal. A review of the literature on the forest-tundra was completed to examine the existing knowledge of the ecosystem as a whole, provide an information base to which this case study could be compared, place this study within the context of a much larger ecological region, and provide a theoretical context for the project (Chapter 2).

The process of developing a regional scale vegetation map using satellite imagery necessarily involves three general steps: (1) defining distinct vegetation communities based on ground sampling; (2) developing a set of vegetation classes that are separable in the satellite imagery; and (3) classifying all of the pixels in the satellite image into one of the defined classes and presenting this information in the form of a vegetation map. Chapter III delineates and describes the vegetation communities of the study area and describes the dynamics of the vegetation at the community scale. The structure and composition of 600 homogenous vegetation patches were described during field sampling conducted within Wapusk National Park and the Cape Churchill Wildlife Management Area between June and September, 1998-2000 (Fig. 1.2). These data were used in conjunction with statistical analysis tools to delineate the major vegetation communities that exist in the study area. Species and life form cover data were then summarized to describe each vegetation type.

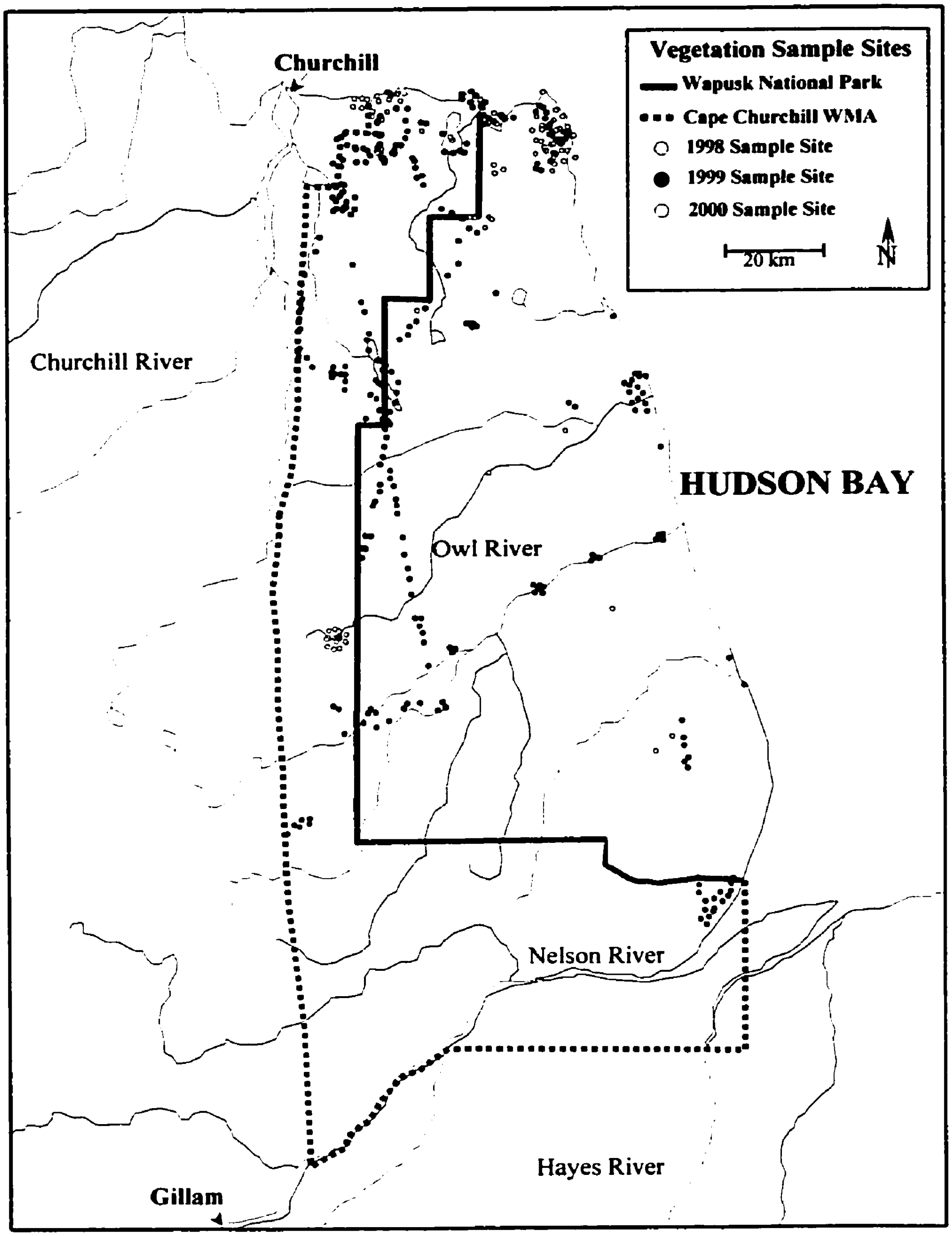


Figure 1.2. Field Sample Sites situated within Wapusk National Park and the Cape Churchill Wildlife Management Area.

Chapter IV translates the vegetation communities that exist on the ground into meaningful vegetation classes for satellite image classification. A multivariate analytical approach was developed to examine the spectral reflectance signatures of the different vegetation types in order to assess their relative separability in a LANDSAT TM satellite image mosaic. A version of this chapter has been submitted for publication to the International Journal of Remote Sensing by R.K. Brook and N.C. Kenkel, titled: A multivariate approach to vegetation mapping of Manitoba's Hudson Bay Lowlands.

Chapter V describes the development of the vegetation map itself and its application in examining vegetation dynamics. Once a set of 16 vegetation classes was developed that had distinct reflectance signatures, all pixels within the satellite image were classified into one of these classes. A quantitative assessment of accuracy was then completed on the classified satellite image using an independent data set.

Chapter VI summarizes the final conclusions developed from all aspects of this study and synthesizes the information learned. Chapter VII presents management recommendations based on the findings of the study. Suggestions are developed to support decision-making and research by integrating the plant community descriptions and vegetation map from Chapters 3 and 5 with existing data sets. Recommendations for future research are also provided.

CHAPTER 2: STRUCTURE AND DYNAMICS OF THE FOREST-TUNDRA ECOSYSTEM: A REVIEW

> *"Here in the northern forest we can see the direct effects of physical factors on organisms, we can unravel the simplified food web and examine the component food chains, we can see and experience directly the effects of seasonal changes in light. A number of ecological principles are put on display in graphic clarity. In the taiga, students of ecology can easily grasp the fundamental concepts of the science, as they are laid bare around them."*
>
> -W.O. Pruitt Jr., Wild Harmony

PREAMBLE

This chapter provides an overview of the structural components and the dynamic processes of the forest-tundra ecosystem in order to define the concepts that are discussed in subsequent chapters. The existing published literature that pertains to the forest-tundra ecosystem is synthesized. The applicability of this review in understanding the Hudson Bay Lowlands is then discussed.

INTRODUCTION

The forest-tundra ecosystem lies between the continuous closed canopy boreal forest to the south and the treeless arctic tundra to the north. It is characterized by a low but variable tree cover interspersed with areas of open tundra. The forest-tundra zone has been defined in a number of different ways in the literature including the Hemiarctique of Rousseau (1952), the Sylvotundra of Maini (1966), the lesotundra of Norin (1961) and the Woodland Tundra of Alexandrova (1970). Further south from the forest-tundra lies the Northern Boreal Open Woodland (Atkinson 1981), referred to in the literature as the Open Coniferous Forest (Hare and Ritchie 1972), the Subarctic Woodland (Rowe 1972) and the Lichen Woodland (Kershaw 1977). Even further south is the Main Boreal

Closed-crown Forest (Atkinson 1981), also known as the Closed Coniferous Forest (Hare and Ritchie 1972).

Larsen (1989) has defined the entire forest-tundra region as an ecotone between tundra and boreal forest because of the mosaic of tundra communities mixed with coniferous trees. However, field research by Ritchie (1959) in Northern Manitoba, by Kelsall *et al.* (1971) in the Northwest Territories and by Ducruc *et al.* (1976) in Northern Quebec suggest that the forest-tundra is characterized by the scarcity of trees rather than by the presence of arctic species. This may indicate that the forest-tundra is actually the northern fringe of the boreal forest, rather than a true transition zone to the Arctic (Atkinson 1981). At the continental scale, the forest-tundra is thought to have less plant species than either the northern tundra or the boreal forest to the south (Larsen 1980). Species found in the transition zone tend to be the ones that are also found in the arctic tundra and boreal forest. However, true tundra species and true forest species are less well adapted to the forest-tundra and only the species uniquely adapted to the forest-tundra conditions tend to survive.

A variety of criteria have been used to define the limits of the forest-tundra. Atkinson (1981) considers the northern boundary to be identical with the 'limits of continuous forest', while Hare and Ritchie (1972) equate the edge of the forest-tundra with the break point between closed crown boreal forest and open woodland. Other parameters have been suggested such as the stem density, height and growth forms of tree species (e.g.

Scott *et al.* 1987). A more quantifiable criterion is described by Timoney *et al.* (1992) where the northern and southern boundaries are at the 1000:1 and 1:1000 tree: upland tundra cover isolines respectively. However, Mackay (1969) points to the limited value of deliberating the different boundary criteria. Arguments over the definition of the forest-tundra ecosystem should be concluded with the remark by Larsen (1989): "..there will probable never be universal agreement as to where the Arctic begins and there will be a continual need for redefinition to suit individual requirements."

Payette (1983) describes the forest-tundra as being composed of two major sub-zones, the shrub sub-zone and the forest sub-zone. The forest sub-zone is dominated by coniferous forest with an open or closed canopy except on exposed slopes and uplands that are often composed of lichen-heath-dwarf birch communities. In the shrub sub-zone, lichen-heath-dwarf birch communities dominate the uplands and exposed slopes, with conifers being relegated to more sheltered areas.

CLASSIFICATION

The forest-tundra includes a wide range of vegetation types. Sorting these types into major groups has been done in a variety of ways by different agencies. Generally, there is agreement on the main difference between wetlands and uplands, as well as the breakdown of wetlands into four major groups.

A wetland is "land that is saturated with water long enough to promote wetland or aquatic processes as indicated by poorly drained soils, hydrophytic vegetation and various kinds of biological activity which are adapted to a wet environment" (National Wetlands Working Group 1988). Bogs and fens of the forest-tundra are often referred to as peatlands (also known as muskeg) because they sequester carbon through peat accumulation caused by primary production exceeding decomposition (Vitt *et al.* 1994).

A bog is a wetland with the water table at or near the ground surface (National Wetlands Working Group 1988). The surface is often raised above or level with the surrounding terrain, separating vegetation from the nutrients dissolved in the ground water. As a result, most nutrient input is from the atmosphere in the form of rain and snow (ombrotrophic). Bogs are generally considered to be the most nutrient poor wetlands. Thickness of the peat layer is normally greater than 40 cm (National Wetlands Working Group 1988). Ericaceous shrubs and *Sphagnum* mosses are often dominant. Trees and lichens may or may not be present.

Fens are wetlands with the water table at or just above the ground surface (National Wetlands Working Group 1988). The vegetation has access to nutrients in the ground water as it slowly moves through the soil, making it the richest wetland type. The peat layer is generally less than 40 cm thick. The vegetation consists mostly of sedges, mosses, shrubs, and in some cases trees.

Marshes are permanently flooded, intermittently exposed or seasonally flooded wetlands (National Wetlands Working Group 1988) that do not normally accumulate peat. Surface water levels may fluctuate seasonally. The growing substrate is mineral or well-decomposed non-peat organic material. The amount of nutrients available to vegetation in marshes is determined by the access of the vegetation to water. Alkaline marshes (dominated by calcium and bicarbonate) are dominated by *Carex* spp, *Scirpus* spp and *Typha* spp, while saline marshes (dominated by sodium and sulfate) generally have *Scirpus* spp. and *Salicornia* spp. (National Wetlands Working Group 1988).

Swamps are forested, wooded or shrubby non-peat forming wetlands. Peat accumulation is limited by the high decomposition rate. Standing or gently flowing water occurs in pools or channels (Racey *et al.* 1996). There may also be subsurface water flow. The water table may drop below the rooting zone of the vegetation, creating aerated conditions at the surface (Racey *et al.* 1996). The substrate is often composed of woody, well-decomposed soil or a mixture of mineral and organic material. Deciduous or coniferous trees or shrubs, graminoids, herbs and mosses dominate the vegetation cover.

Uplands with mineral sand and gravel soils are not normally influenced by wetland processes. These areas are typically elevated above the water table by geological features such as kames, eskers, and beach ridges. As a result, they are typically well drained, but may contain a wide range of plant communities, including spruce-lichen woodland (Raup 1930).

ENVIRONMENTAL FACTORS

Climate

The climate of the forest-tundra in Canada is closely associated with the High, Mid- and Low Subarctic ecoclimatic sub-provinces. Climate is characterized by short, warm summers and very cold winters (Zoltai *et al.* 1988). This is the area where the most frequent encounters occur between the arctic and temperate air masses (Dolgin 1970). To some extent, the forest-tundra region in Canada coincides with the average summer position of the Polar Front. The strong north/south temperature gradient produced at the average position of the Polar Front may account for the transition from an open tundra area on the cold side to open forested areas on the warmer side (Hare and Ritchie 1972). Since frontal activity is generally associated with precipitation events, it is not surprising that the forest-tundra normally receives more precipitation than the tundra (Scott 1995). A positive moisture balance throughout the forest-tundra may partially explain the extensive wetlands present.

Climatic factors that closely correlate with survival, growth and sexual reproduction of trees in the forest-tundra include mean summer position of the Arctic Front, mean annual net radiation, degree-days, and mean July air temperature (Bryson 1966, Hare and Ritchie 1972, Black 1977, Timoney *et al.* 1992). Relative to long-term climatic change, forest-tundra tree and tundra vegetation is in dynamic equilibrium with climate (Payette and Fillion 1985, Scott *et al.* 1987). The position of the tree line can thus be considered a reliable climate marker at different time scales (Payette and Lavoie 1994).

Nutrient Status

Nutrient levels are highly correlated with different vegetation communities in boreal regions (Sjörs 1950). Fens are high in minerals from surroundings soils, whereas bogs rely on a sparse supply of minerals from precipitation. The pH of the soil generally decreases as the organic content increases. Ground water in fens is circumneutral to slightly alkaline with a Ca content above 5 mg/kg and a Mg content above 2 mg/kg. These measures of nutrient status are much lower in bogs. In the surface peat, bogs have uniformly low amounts of Ca and Mg, but fens can have a wide range of these nutrients (Zoltai *et al.* 1988). As a fen develops into a bog, the supply of metallic cations (Ca^{2+}, Mg^{2}, Na^{+} and K^{+}) drops sharply. At the same time, as the organic content of the peat increases due to the slowing of the decomposition rate, the capacity of the soil to absorb and exchange cations increases. These changes lead to the domination by hydrogen ions in bogs and the pH falls sharply (Gorham 1967).

Jeglum (1971) allocated the pH values of moist peat into five fertility classes: very oligotrophic (pH 3.0-3.9), oligotrophic (pH 4.0-4.9), mesotrophic (pH 5.0-5.9), eutrophic (pH 6.0-6.9), and very eutrophic (pH >7.0). The nutrient status of the soil is controlled by a number of environmental factors, including surface and subsurface water movements, rate of peat accumulation, rate of decomposition, dissolved oxygen content, and atmospheric inputs (Damman 1990).

Black spruce has been shown to increase its growth rate when treated with nitrogen and phosphorus fertilizers (Cowles 1982). The growth of *Ledum groenlandicum* and to a lesser extent, *Betula glandulosa,* also responds to fertilization (Moore 1984). However, the addition of fertilizers to lichen results in little change in growth rate (Carstairs and Oechel 1978).

Hydrology

Hydrological processes strongly influence the chemical and physical properties of the ecosystem, particularly the oxygen availability and related chemistry such as nutrient availability, pH and toxicity (Mitsch and Gosselink 2000). Hydrology also transports sediments, nutrients and even toxic materials. Except in nutrient-poor bogs, ground water inputs are the main source of nutrients. The hydrology is also responsible for water outflows that remove biotic and abiotic material such as dissolved organic carbon, excessive salinity, toxins and excess sediments and detritus. When hydrologic conditions in wetlands change, even slightly, the biota may respond with massive changes in species composition and richness and in ecosystem productivity. Hydrology is likely the single most important determinant of wetland processes (Mitsch and Gosselink 2000).

Ombrogenous hydrological systems receive water from direct precipitation only and are isolated hydrologically from lateral inflow or upward seepage of water due to their position in the landscape (National Wetlands Working Group 1997). The ombrogenous systems are normally poor in nutrients, though they may be enriched in areas of high precipitation. Bogs are the only types of wetland found in ombrogenous hydrological systems.

Minerogenous hydrological systems are normally situated lower than adjacent mineral terrain, so that water and mineral elements are introduced by groundwater or littoral sources in addition to atmospheric sources (National Wetlands Working Group 1997). Fens and marshes are the wetlands generally found in minerogenous systems.

Soils

Soil conditions in the forest-tundra are significantly affected by permafrost and cryoturbation. As a result, Organic, Turbic and Static Cryosols are common with non-Cryosolic soils where permafrost has less affect on soils (Scott 1995). On well-drained slopes where permafrost is absent, Podzolic or Brunisolic associations frequently dominate. On lower slopes, poorer drainage encourages Gleysolic soils and where soils are almost permanently saturated, organic soils develop. Peat is fibric organic unconsolidated soil material consisting largely of partially decomposed plant material. Vegetation in an area is strong influenced by the soil characteristics. However, soil development is also significantly affected by vegetation composition. This synergistic feedback between vegetation and soil development is a critical process in the forest-tundra ecosystem.

Permafrost

Permafrost is defined as ground that remains continuously frozen for more than 2 consecutive years (Muller 1945). It generally occurs where the mean annual temperature is below 0°C (Zoltai 1971), which results in heat loss exceeding heat influx over the long term. Development of permafrost is significantly influenced by the insulating qualities of peat. Heat from the ground is lost quickly through frozen peat, but dry peat is a poor conductor during the summer, preventing the complete thawing of the frozen peat. Ultimately, the thickness of the permafrost and the active layer depends on the regional climate, local climate, vegetation cover, presence of organic matter, soil moisture and texture.

The active layer is the surface layer of soil and rock which lies above the permafrost and freezes in winter and thaws in summer. The term active layer is appropriate because most physical, chemical and biological activity takes place in this seasonally thawed layer. However, plants generally do not utilize the entire thawed area since most growth is concluded by late July, long before the full development of the active layer is reached.

The presence of permafrost can also change the structure of the landscape by raising the peatland above the surrounding terrain, forming palsas or peat plateaus, depending on the amount of water available for ice formation. Palsas are small, dome-shaped mounds of peat that are between 1 and 3m tall and normally have a diameter of less than 100 m (Sjörs 1961). Peat plateaus rise to approximately 1 m tall, but often cover several square kilometers (Zoltai and Tarnocai 1975). Both palsas and peat plateaus form from the same process of ice formation in the peat during winter (Brown 1968). As the previously water saturated surface layer becomes drained during permafrost accumulation, conditions for plant growth change dramatically, becoming dryer and less nutrient rich. The thickness of peat also influences the active layer. FitzGibbon (1981) found that the active layer in peat hummocks thawed 57 days later than in nearby fens.

Permafrost acts as a barrier to the drainage of water from rainfall and snowmelt, causing water to accumulate in some areas. Frozen ground can also act as a barrier to rooting and to the movement of nutrients out of the soil layer. In turn, vegetation cover can significantly affect permafrost distribution and depth, as well as the thickness of the active layer. Rouse (1984a, 1984b) showed that surficial forest soils near Churchill were significantly warmer than tundra soils and that the thaw period in the rooting zone is two months longer in forested areas.

Permafrosted regions are often separated into two major zones, the zone of continuous permafrost and the zone of discontinuous permafrost, based on the total area that is underlain by permafrost. The term "sporadic permafrost zone" has been used to refer to the southern fringe of the discontinuous zone, where small, scattered islands of permafrost are largely restricted to peatlands. The forest-tundra region is associated with both the continuous and discontinuous permafrost zones.

Snow

Snow is dominant on the forest-tundra landscape for six to ten months of the year

(Slaughter and Cook 1974). Distribution of snow is determined by topography, vegetation physiognomy and density, and wind patterns. Treed areas provide protection from the wind and so tend to collect soft snow that may become more than 2 meters thick. Open flat tundra areas are wind swept so snow is much harder and is normally less than 50 cm thick. In forest edges and low areas, hardness and thickness are dependent on exposure to wind (Brown Beckel 1957).

One of the greatest challenges for plants under snow cover for long periods is desiccation from the vapor transport from the ground to the snow surface (Slaughter and Cook 1971). The longer that the snows cover lasts, the worse the desiccation is. Blowing snow and ice crystals cause abrasion of exposed plant parts and can tear off growing parts. Collection of snow on tree branches provides insulation, but excessive loads may cause breakage. The thickness and duration of snow cover also regulates local drainage and water supply.

ECOSYSTEM METABOLISM

Productivity and Decomposition

Productivity in the forest-tundra is generally limited by the short growing season with relatively few degree-days above 0°C and the restricted availability of nitrogen, phosphorus, and water (Bliss 1986). Due to the complex mosaic structure of the forest-tundra, biomass and net primary productivity values vary considerably. Mature biomass generally ranges between 1.3-200 t/ha, while net primary productivity (NPP) is between 140-500 g/m^2/yr (Lieth 1975). NPP is an important measure of an ecosystem's success in harnessing insolation and soil nutrients to produce energy and increase tissue production.

In the forest-tundra, few nutrients generally enter the system through precipitation, release of minerals by soil weathering is very limited, and the nutrient flush from fires is small (Moore 1980, Dubreuil and Moore 1982). Much of the available nutrients in the soil are tied up in organic matter, which has been shown to decompose slowly, largely due to the low soil temperatures, low soil pH and the paucity of readily available nitrogen (Moore 1981).

The relatively low productivity of the forest-tundra places limits on nutrient availability; so litter decomposition and nutrient release are important processes. Low decomposition rates are attributable to the short growing season and sub-optimal substrate conditions (van Cleve et al. 1983). The limited decomposition in bogs is caused by the cold temperatures and an anaerobic, acidic substrate (Heilman 1968).

Productivity and decomposition are largely influenced by climate. The cold summer temperatures and long winters create difficult conditions for plant growth. If the rate of decomposition of plant material is slower than the rate of plant growth, peat accumulates. This process gradually raises the rooting zone above the water table, creating drier and more nutrient-deprived conditions. As a result, the productivity of nutrient-poor bogs is lower than that of nutrient rich fens, producing much different vegetation communities.

Vegetation

The forest-tundra exhibits considerable floristic and vegetational diversity at local, regional and continental scales (Bliss and Matveyeva 1992). There have been numerous studies of the forest-tundra vegetation across Canada (e.g. Hustich 1949a, 1949b, 1950, 1951, 1962, 1966, 1979, Larsen 1965, 1971, 1972, Maini 1966, Ritchie 1960a, b, c, 1962, 1984, Johnson 1981). However, the different objectives, definitions, methodology and interpretation make direct comparisons difficult among the different studies (Timoney *et al.* 1992). The most southern portion of the forest-tundra is dominated by treed vegetation and the most northern portion is dominated by open tundra, while the vast majority consists of patches of both types mixing at various scales.

Vegetation of the drier areas is dominated by a ground cover of lichens (*Cladina stellaris, Cladina mitis, Cladina rangiferina*) and shrubs (*Betula glandulosa, Dryas integrifolia*). Tree cover is characterized by sparse cover of white spruce (*Picea glauca*) and black spruce (*Picea mariana*) when present. Sedges (*Carex* spp., *Eriophorum* spp.) dominate wet fen areas with a larch (*Larix laricina*) tree cover if present. Wet bog areas are composed primarily of *Sphagnum* spp. moss (Vitt et al. 1994). Black spruce is the most common tree species in forested peatlands of North America, in association with leatherleaf (*Chamaedaphne clayculata*), Labrador tea (*Ledum* spp.), laurel (*Kalmia polifolia*), blueberry (*Vaccinium* spp.) and bog rosemary (*Andromeda polifolia*) (Vitt et al. 1994).

Plants with deep running roots such as paper birch (*Betula papyrifera*) and trembling aspen (*Populus tremuloides*) are normally limited to permafrost-free areas (Viereck 1983). Species with shallow roots, including white spruce, black spruce and balsam poplar are able to grow in areas with a shallow active layer above the permafrost (van Cleve et al. 1993).

Many plants are adapted to the low nutrient supply in the forest-tundra to enable them to accumulate and conserve nutrients. Carnivorous plants such as the sundew (*Drosera* spp.) and pitcher plant (*Sarracenia* spp.*)* derive nutrients from the decomposing insect bodies that they capture (Chapin and Pastor 1995). Some bog plants also carry out symbiotic nitrogen fixation such as bog myrtle (*Myrica gale*) which fixes atmospheric nitrogen (Mitsch and Gosselink 2000).

While the forest-tundra is defined as a zone of gradually decreasing tree cover, the causal factors related to the decline in trees are complex. The migration of treeline in response to global climate change (Jungerius 1969, Kearney and Luckman 1983) suggests that climate determines treeline and that trees are different from other vegetation in their ability to tolerate climate. The damaging effects of blowing ice (Klikoff 1965), strong winds (Hadley and Smith 1983) and the drying of plant parts above the snow pack (Sakai 1970) all relate to the inability of trees to survive in the harsh arctic climate.

Smaller statured plants are able to persist within the narrow zone of favorable conditions near the ground (Hadley and Smith 1987). To reach a large size, trees accumulate tissue over many growing seasons. The accumulating modular growth of trees brings some disadvantages such as a greater proportion of non-photosynthetic tissue, particularly if carbon fixation tends to be the limiting factor near treeline (Stevens and Fox 1991).

A wide range of environmental factors affect the composition, structure and patch size of vegetation communities including soil texture, slope, aspect, nutrient levels, moisture, snow thickness and duration, cryoturbation and the spatial scale of terrain variability (Zoltai and Pettapiece 1974, Ritchie 1986, Robinson *et al.* 1989). Plants in the forest-tundra have specialized adaptations to the harsh environmental conditions for growth and reproduction. Most forest-tundra plants are angiosperms, bryophytes and lichens (Brown 1963). The majority of angiosperms are herbaceous perennials or prostrate shrubs. Herbaceous perennials have four general growth forms, cushion (e.g. *Dryas integrifolia*), rosette (e.g. *Cerastium* spp.), leafy stemmed (e.g. *Arnica arctica*) and graminoid (e.g. *Carex aquatilis*) (Billings and Mooney 1968).

The low growth form of the plants allows them to utilize the heat provided at the soil-air interface (Bliss 1960). Pigmentation with red anthocyanin traps incoming solar radiation and retains heat for growth, photosynthesis and reproduction. The pigment also allows many species (e.g. *Arctostaphylos rubra*, *Vaccinium* spp.) to collect outgoing infrared radiation under snow banks, allowing them to begin photosynthesizing before the snow is completely melted in the spring (Savile 1972).

Most forest-tundra plants exhibit rapid shoot growth early in the spring by utilizing carbohydrates and lipids from the previous season stored in roots, rhizomes and bulbs (Billings and Mooney 1968). Pollination is conducted by a wide range of insects and by wind during a short period of two to three weeks (Bliss 1960). Flowers are brightly colored blue, pink and purple and provide a rich nectar source to attract insects (Savile 1972). Seed production occurs only opportunistically, depending on temperature during the growing season. The seeds are capable of remaining dormant or frozen for a number of years until suitable germinating conditions exist (Bliss 1960). Seed dispersal is mostly by wind or by animals consuming the fruits around the seeds. However, seedling establishment is normally rare and occurs slowly. In many cases, sexual reproduction is replaced by vegetative means.

Evergreen leaves in the harsh forest-tundra environment allow the plants to function during the short periods of warmth at the beginning and end of winter (Daubenmire 1978). A firm waxy cuticle provides a barrier to evaporation and protection from abrasion (Bliss 1960). Hairs on the leaves also minimize evaporation losses (Bannister 1976). High concentrations of soluble carbohydrates provide antifreeze protection (Billings and Mooney 1968).

DISTURBANCE

Understanding the spatial and temporal variation that exists in plant communities in an ecosystem requires at least a basic knowledge of the disturbance regime that exists. Disturbances are events that alter the physical environment or ecosystem, community and substrate availability (White and Pickett 1985). Disturbance can change the species composition, abundance and distribution of plant communities. The extent, frequency, and intensity of disturbances, along with the synergistic and antagonistic effects that different disturbance regimes have, can all affect landscape pattern (White and Pickett 1985).

Disturbances disrupt the balance between the vegetation, soil, and permafrost that developed during the stable condition. The destruction of the stable vegetation may result in drastic changes in the land surface and an entirely new assemblage of plants may colonize the area.

Herbivory

Grazing activities can significantly alter vegetation productivity and community composition (Jameson 1963, Henry and Gunn 1991, Jefferies *et al.* 1994, Manseau *et al.* 1996, Crête and Doucet 1998). The effects of herbivory are potentially large and long-lasting (Pastor *et al.* 1993). Herbivores change the structure, biomass, productivity and species composition of vegetation (McInnes *et al.* 1992). Herbivores may demonstrate preferences for specific plant species and even individual plants within a species

(Belovsky 1981).

While grazing in some ecosystems (e.g. Africa) has been shown to actually increase the primary production (McNaughton 1979), the forest-tundra ecosystem is sensitive to grazing due to its low overall net primary productivity. Plant species differ in their response to grazing and herbivores are more or less selective in their choice of forage. As a result, grazing and physical disturbance by vertebrates in the forest-tundra can significantly alter ecosystem structure and function and initiate long-term community and landscape level changes. Vertebrate herbivores in northern regions are typically generalist in their use of plant species.

The dominant herbivores in the forest-tundra include: caribou (*Rangifer tarandus*), moose (*Alces alces*), muskox (*Ovibos moschatus*), hare (*Lepus* spp.), beaver (*Castor canadensis*), red squirrel (*Tamiasciurus hudsonicus*), lemming (e.g. collared lemming (*Dicrostonyx groenlandicus*), lesser snow goose (*Anser caerulescens caerulescens*) and Canada goose (*Branta canadensis)* (Bryant and Kuropat 1981, Nault *et al.* 1991, McInnes *et al.* 1992). These different species have adapted specifically to their foraging niche.

Caribou typically feed extensively rather than intensively and exhibit an important seasonal change in diet. During the summer, high protein plants such as graminoids and shrubs form the majority of the diet. They then move into the forest-tundra during the snow season and rely primarily on terrestrial lichens. Caribou grazing intensity can dramatically alter plant community composition (Pegau 1968). A study of Caribou in Quebec by Crête *et al.* (1990) determined that caribou consume 0.5-0.9% of the available lichen biomass annually.

Moose are generalist herbivores, feeding on a wide range of trees, shrubs and herbs (Belovsky 1981, McInnes *et al.* 1992), including aspen, birch and alder (Kistchinski 1974, Krefting 1974). They can consume 3,000 to 6,000 kg of dry vegetation per year (Pastor *et al.* 1993). Moose break the stems of moderately large saplings and tall shrubs to feed on the crown twigs (Telfer and Cairns 1978).

Beavers consume early-successional deciduous species such as trembling aspen (*Populus tremuloides*), balsam poplar (*Populus balsamifera*) and willow (*Salix* spp.) (Packard 1942, Northcott 1971). When these preferred species are not available, birch is selected (Jenkins 1975). Conifers are rarely eaten by beavers (Northcott 1971). Dam creation floods surrounding areas, kills nearby trees, and can adversely affect downstream hydrology (Naiman et al. 1988). Beaver populations fluctuate in response to disturbance, food supply, disease and predation. As scatterhoarders, squirrels play an important role in seed dispersal in the forest-tundra (Stapanian and Smith 1984).

Snow geese *(Anser caerulescens caerulescens)* and Canada geese (*Branta canadensis*) are migratory waterfowl that return annually to the forest-tundra to nest and raise their young. In areas where geese are the dominant herbivores, grazing may have a positive or negative influence on plant production, depending on its intensity and on the plant parts that are eaten (Cargill and Jefferies 1984, Jefferies 1988, Kerbes et al. 1990). Intensive grazing or root grubbing generally leads to decreased plant production and habitat degradation (Kerbes et al. 1990, Iacobelli and Jefferies 1991). However, moderate grazing on above ground vegetation can have a positive effect on plant production (Smith and Odum 1981, Jefferies 1988), but this is not always the case (Gauthier *et al.* 1995).

While vertebrate herbivory has been relatively well studied, much less is know of the influence of invertebrates on the forest-tundra vegetation. Studies by MacLean and Jensen (1985) and MacLean (1981) suggest that insect herbivory does not have a major effect on the species composition of plant communities. However, the overall influence of insects on plant communities remains unresolved (Bonan and Shugart 1989). Defoliating insects may cause considerable alteration to ecosystem function without a major physical disturbance. Outbreaks of cankerworm (*Alsophila pometaria*), spruce budworm (*Choristoneura tumiferana*), mountain pine beetle (*Dendroctonus ponderosae*), tent caterpillar (*Malacosoma disstria*), and gypsy moth (*Portheria dispar*) can alter vegetation production, nutrient cycling and ground water chemistry (Dyer 1986). The overall importance of insects in the forest-tundra requires further study.

Grazing on sub-arctic vegetation generally occurs on three major forms: graminoid, shrub and lichen. The recovery rate of these groups after herbivory is very different.

Lichens tend to recover very slowly over a period of 20 years or more (Klein 1987). Shrubs generally recover much more quickly, though Crête and Doucet (1998) showed that dwarf birch did not recover well after grazing by caribou. However, the effects of grazing may be highly variable due to differences in frequency and intensity of the grazing. The role of herbivores in creating "grazing lawns" is gaining acceptance in some habitats. In these "grazing lawns", moderate levels of herbivory can increase plant productivity by facilitating nitrogen cycling (Jefferies 1988, Raillard 1992). Sedge meadows composed primarily of graminoids have been shown to have increased net production in response to moderate muskox grazing (Henry and Svoboda 1989).

Permafrost and Periglacial Activity

The process of permafrost development can be reversed through natural and catastrophic events. Cracks in the peat surface may form from extensive accumulation of peat or drying of the peat surface, exposing the core to accelerating thawing (Zoltai 1993). Other disturbances such as fire, windthrow and human activity can tear open the peat surface and initiate thermal degradation. When the frozen core is thawed, the ground surface subsides down to the surrounding fen. This collapse process can occur rapidly, within several years (Thie 1974). As a result, the frozen peatlands are highly dynamic systems that tend to reach an equilibrium state under the existing environmental conditions (Zoltai and Tarnocai 1975).

Seasonal frost heaving of peat plateaus and palsas, along with the random degradation of ice bodies in the soil cause considerable disturbance to vegetation (Brown 1970). Ice-rich frozen soils cool in the fall, causing contraction and cracking at the surface. One of the most obvious manifestations of this soil agitation is in the varidirectional tilting of spruce trees which causes the phenomenon known as "drunken forest" (Benninghoff 1952, French and Gilbert 1982). In cases of severe cryoturbation, the trees may fall over and die.

Fire

Fire forms a major disturbance regime in the forest-tundra and is largely responsible for the vegetation mosaic that exists (Wein, 1976, Viereck 1983, Auclair 1983, Payette et al. 1989). The ecological role of fire is significant in the forest-tundra where tree species are growing at the limit of their range and disturbances can cause major changes in vegetation patterns (Payette and Gagnon 1985, Payette *et al.* 1989). The spatial and temporal pattern of fire is important at the landscape scale because it determines the age distribution and successional status of the vegetation. The net result is a landscape of vegetation patches at different stages of regeneration.

The majority of fire research has been focused to date on quantifying the amount of burned ground per unit time (Niklasson and Granstrom 2000). Various terms have been used to describe this burn rate, including "average fire interval", "fire cycle" (time to burn the equivalent of the study area [Reed *et al.* 1998]), or "fire frequency at point scale" (percentage annual burn [Johnson 1992]). Estimates of fire intervals may be estimated from fire intervals in scarred trees (Engelmark 1984) or from the distribution of age classes across the landscape (Johnson and Larsen 1991).

Fires in the forest-tundra tend to vary widely in both intensity and frequency (Henselman 1973). Fires are less frequent than in the boreal forest and many forest-tundra fires spread in from the boreal forest (Timoney and Wein 1991). Fire is generally rare or absent on the arctic tundra (Wein 1976).

The immediate effect of fire is the killing of the vegetation. Trees, shrubs, mosses and lichens are killed and partially consumed by the flames. However, the long-term affects of the fire are variable. The insulating effect of the ground cover may be diminished and within two seasons the active layer may become 50%-100% deeper than before the fire (Zoltai and Pettapiece 1973). When the insulating organic layer is destroyed, the permafrost table may be lowered, resulting in subsequent subsidence of the ground. After a fire, the insulating effect of the vegetation layer is lost and the black soil surface absorbs significantly more solar radiation, causing the active layer to become thicker (Wein and Bliss 1973). Thie (1974) found that fire can induce rapid degradation of permafrost in a localized area, but generally wildfires do not initiate widespread thawing of the permafrost. Jaseniuk and Johnson (1982) considered fire to have no effect on habitat condition or stand composition.

The quantity of fuel and moisture level are key components in determining fire susceptibility. The intensity of a fire is a physical measure of the force of the fire. High intensity fires tend to completely consume the vegetation, while low intensity fires tend to leave unburned and partially burned areas. In the forest-tundra, fires generally do not destroy the peat layer to any significant depth. Burn intensity appears to be significantly influenced by the amount of tree cover in the subarctic, with fires being more intense when there is a greater frequency of trees (Rowe *et al.* 1975). Open tundra will burn, but generally with a low intensity and only over small areas (Wein 1976). Recovery time for the vegetation in burned areas is related to the intensity of the burn, with more intense burns taking longer to recover. In turn, burn frequency for a patch is largely determined by recovery time, as early recovery stages are less susceptible to fire.

In the forest-tundra, fires are carried by continuous, fine-textured ground vegetation rather than by tree crowns (Wein 1976). Willow-birch thickets and even sedge meadows will burn (Wein 1976). However, plants with high ash contents such as graminoids are less flammable than resinous shrubs (Auclair 1983). Lakes larger than 15 km^2 seem to act as effective fire breaks (Timoney 1988).

Depending on the intensity of the fire, the regeneration processes after a fire may take up to hundreds of years. *Sphagnum* mosses begin to invade the area and the ground lichens (*Cladina* and *Cetraria*) will be established on the hummocks. As the organic layer begins to develop, the permafrost table rises to its normal level and the active layer becomes thinner. In the moister regions, a different process occurs. Fire kills the trees and most of the ground vegetation, but the sedge and cottongrass tussocks survive and thrive in the absence of competition.

The vegetation that grows in a site after fire can come from three sources: vegetative reproduction (sprouting), viable seeds buried in the soil, or invasion by propagules (Johnson 1981). Many of the species that are common early after a burn reproduce vegetatively such as *Ledum groenlandicum*, *Vaccinium uliginosum*, *V. vitis-idaea*, *Alnus crispa* and *Betula glandulosa*. In contrast, black spruce reproduces exclusively by seed after fire and grows slowly. It uses layering to maintain and increase its density only in older stands (Johnson 1981).

Plants that produce abundant, lightweight seeds or spores (e.g. *Epilobium* spp., *Corydalis* spp. *Carex*, spp., *Calamagrostis* spp, and *Polytrichum* spp.) invade the areas newly opened by fire (Johnson 1981). These species grow, mature, flower and fruit very rapidly. Within 5-10 years after a fire, these colonizing species are greatly reduced in abundance or die out completely. In general, this group is intolerant of crowding and is only present for a short period after a fire. The seed bank appears to be relatively unimportant in the forest-tundra in post-fire regeneration (Johnson 1975).

Lichens begin growing in burned areas through thallus fragments, sporedia and apothecia. *Cladonia* spp. (e.g. *Cladonia cornuta*, *C. coccifera*) are typically early arrivals through dispersal of soredia, while *Cladina* spp. (e.g. *C. mitis* and *C. rangiferina*) tend to be much slower in arriving as they disperse through fragmentation.

ANTHROPOGENIC DISTURBANCE

Direct human impacts on the forest-tundra zone have been limited by access in the past. The trees of the region are often of insufficient size to be of value for forestry development, while peat extraction has occurred in some areas. Human activity can disturb the delicate balance that maintains permafrost. Most artificially induced changes in the permafrost are a result of disturbance to the surface layer of the soil and vegetation. Removal of the vegetation layer exposes darker material, such as peat, which absorbs solar energy and promotes deep thawing of the permafrost.

Management of herbivore and carnivore populations through hunting, trapping and predator control programs influence herbivory processes. The loss of natural predator populations could increase grazing pressure on some plant species and communities.

In areas where fire suppression occurs, the natural vegetation mosaic may be altered (Bergeron and Dansereau 1993, Glenn-Lewin *et al.* 1992). Dramatic changes may result, since many forest-tundra species are adapted to a relatively frequent catastrophic fire interval. These fire adaptations may prove maladaptive under a fire suppression scenario (Kenkel et al. 1997).

The most severe effects of human activity in the forest-tundra are likely to be from what are known as the "Big Three" of global human stressors: climate change, acid deposition and increased exposure to ultraviolet radiation caused by stratospheric ozone depletion (Schindler 1998). These influences continue to increase in their long-term impacts on a variety of ecosystem processes (Schindler 1988, Minns *et al.* 1990, Bothwell *et al.* 1994, Vinebrook and Leavitt 1996). These three stressors have cumulative and perhaps synergistic effects since they interact in significant and complex ways (Schindler *et al.* 1996, Yan *et al.* 1996). Without meaningful action to reduce the anthropogenic inputs to the forest-tundra, there is the potential for dramatic changes to water quality, plant communities and animal populations in this century. Considerable research is needed in order to understand and mitigate the influences of climate change, acid deposition and increased exposure to ultraviolet radiation

VEGETATION DYNAMICS

The forest-tundra is a dynamic ecosystem, subject to both rapid and long-term changes. The plant communities of the forest-tundra are constantly changing as a result of small environmental shifts, large events such as the draining of a thaw lake, or patterns of succession across the landscape (Bliss and Peterson 1992). The forest-tundra zone has not remained in a stable position over time, but has varied with the fluctuations in climate and will continue to do so (Hustich 1958, Lavoie and Payette 1994). However, the dynamics of this ecosystem remain poorly understood.

The concept of succession involves the replacement of plant species in a general sequence. This concept has an extensive history that was first developed in the 19th century (Cowles 1899, Clements 1916). Succession was then adapted and extended to include ecosystem properties such as productivity, respiration and diversity (Odum (1969). Classical succession theory involves three fundamental concepts: (1) vegetation occurs in recognizable and characteristic communities; (2) communities change over time through biological processes (i.e. change is autogenic); and (3) changes are linear and directed toward a mature, stable climax community (Odum 1971). Although classical succession has been the dominant paradigm for many years, it continues to be a controversial topic in the literature (Finnegan 1984).

Gleason (1917) developed an alternative "individualistic" hypothesis to explain the distribution of plant species. This theory developed into the continuum concept (Whittaker 1967) which suggest that the distribution of a species is determined by its response to its environment (i.e. change is allogenic). Since each species responds differently to its environment, no species will occupy exactly the same zone. The result is a continuum of overlapping species, each responding to subtly different environmental factors. According to the continuum concept, no communities exist in the true Clementsian sense and although ecosystems do change, there is little evidence that this is directed or that it produces a particular climax. Recently, the classical succession model of Clements has been largely rejected in favor of the individualistic approach of Gleason (Johnson 1979, Cook 1996). However, the complexity of successional processes may preclude a consensus from ever being reached.

Disturbance is now generally recognized as an integral component of vegetation dynamics (Kenkel et al. 1997). Where fire has been excluded for 150 years or more, a dense growth of lichens tends to accumulate, particularly *Cetratria nivalis, Cladina stellaris* and *Stereocaulon paschale*. (Strang 1973, Kershaw 1977). Only recently have attempts been made to model the relationships between vegetation and the environmental factors that drive this dynamic system (e.g. Timoney 1988, Timoney *et al.* 1993). The successional changes that grazing has on vegetation is poorly understood in the forest-tundra.

The dynamics of the forest-tundra can be explained through a number of general processes, including terrestrialization and paludification. Terrestrialization is a process whereby peat formation begins through the development of minerotrophic vegetation (Vitt *et al.* 1994). When climatic conditions are appropriate, autogenic processes such as peat accumulation, acidification and oligotrophication produce conditions suitable for bog development (Vitt and Kuhry 1992). As peat accumulates above the water table, the vegetation becomes increasingly isolated from the ground water nutrient supply, becoming more and more nutrient poor (Kratz and DeWitt 1986). This process also isolates the vegetation from groundwater and nutrient renewal.

Bogs can exceed the boundaries of a basin and encroach onto formerly dry land by a process known as paludification. This process is initiated by climatic change, geomorphological change, beaver dams, or the natural advancement of a peatland. Often, the lower layers of peat compress and become impermeable, causing a perched water table near the surface of what was formerly the mineral soil. This process causes wet and acid conditions that kill or stunt trees and allow only ombrotrophic bog species to exist. In some situations, the progression from forest to bog can take place in only a few generations of trees (Heilmann 1968).

Two approaches have generally been used to identify successional sequences in peatlands: spatial-temporal conversion and stratigraphy (Jasieniuk and Johnson 1982). In the spatial-temporal conversion approach, peat is assumed to accumulate over time and induce environmental changes favoring the selection of successively less hydrophytic communities. The spatial arrangement of vegetation communities around open water was considered to reflect the temporal succession sequence. This approach involved describing the vegetation around an open pool and ordering their successional development according to their spatial position (Transeau 1903, Clements 1916, Gates 1942). Though these hydroseres were popular in North American literature, long-term studies of vegetation changes around some of these bog pools indicated that the spatial-temporal approach does not accurately describe peatland dynamics. Water level fluctuations disrupt this hydroseral succession model (Schwintzer 1978).

The stratigraphic approach uses macrofossils, pollen and peat types to interpret the vegetation history and identify peatland succession (Conway 1948). Later analytical approaches were more sophisticated and less deterministic (e.g. Heinselman 1963, Janssen 1968) revealing complex transitions between communities (Walker 1970). Succession was also revealed to be a stochastic process that is continuous and heavily influenced by local and regional conditions (Heinselman 1963).

THE HUDSON BAY LOWLANDS

While the literature review above provides a general overview of the forest-tundra ecosystem, little work has been done to critically compare the structure and dynamics of the Hudson Bay Lowlands with other forest-tundra regions. The extensive work by Timoney (1988) only included the northwestern portion of the study area that was used for this thesis, but did identify some important differences between the Hudson Bay Lowlands and the forest-tundra of continental Western Canada. The Hudson Bay Lowlands is composed of a much higher percentage of open water and wetland and a much lower percentage of upland tundra, treed vegetation, and burned tree vegetation (Timoney 1988). Drainage in the study area is exceptionally poor as a result of a particularly gentle regional slope, low local relief, and the impermeability of the fine-grained, ice bonded substrate (Dredge and Nixon 1992). Permafrost is discontinuous only in the southern part of the study area. Human activity is relatively low in the Lowlands of Manitoba, with few roads and only a few settlements. Large hydroelectric developments have significantly influenced both the Churchill and Nelson Rivers.

Many plant species (e.g. *Picea mariana, Picea glauca, Larix laricina, Ledum groenlandicum, Sphagnum* spp., and *Cladina* spp.) and vegetation communities (e.g. lichen/conifer bog and sedge fen) are similar to those dominating other forest-tundra regions (Ritchie 1956, 1957, 1963, Timoney 1988). Wildlife species in the Lowlands of Manitoba include relatively high caribou numbers (Elliot 1998), few beavers and moose (Dubois 2001), extremely high snow goose populations (Kerbes *et al.* 1990), and relatively high Canada goose populations (Rusch et al. 1996).

MANAGEMENT IMPLICATIONS

The complex structural and dynamic nature of the forest-tundra vegetation provides considerable challenges to understanding and managing the ecosystem. The literature review points to several issues that are particularly important to management.

While the number of ecosystem components (i.e. species and communities) may be relatively low compared with other ecosystems, the arrangement of the components and the processes that drive them are tremendously complex and diverse. The ecology of plant species in the forest-tundra is reasonably well understood but the large-scale ecological processes are not. Landscape level analysis is critical for examining structure and dynamics of the forest-tundra. Ecosystem structure must be considered at a broad scale.

Natural and anthropogenic disturbances are important influences on ecosystem structure and dynamics and are often recurrent and highly unpredictable. While wildlife grazing is important at small scales, fire is currently the single most influential disturbance regime in the forest-tundra at the landscape scale. However, the influences of human activity on ecosystem process can not be discounted, particularly in relation to global climate change, acid rain and ozone depletion. Disturbances can alter vegetation communities or completely destroy them, at least temporarily. Differential responses to disturbances, year-to-year effects of weather, wildlife grazing and land-use introduce an element of stochasticity and unpredictability in vegetation dynamics. The synergistic and antagonistic influences of disturbances on each other further complicate the interrelationships that exist.

Ecosystem processes are intertwined in a highly complex order, such that every process influences other ecosystem processes. For example, permafrost is generally recognized as being influenced by a wide range of factors from climate and human activity, to fire, slope, aspect, hydrology, soil, and vegetation cover. Human activities and management prescriptions may have wide spread and complex influences on the ecosystem as a whole.

The forest tundra is undergoing constant change at site, regional and continental scales; however, the causes and manifestations of this change are only partially understood. Successional theory is continuing to evolve, as new information is becoming available. Early successional models viewed plant communities as stages in a cycle that was self-perpetuated toward a 'climax' community. More recently, succession is viewed as a process that involves differential species responses to environmental and autogenic processes that are significantly influenced by disturbance. More work is needed to evaluate the similarities and differences between different areas of the forest-tundra and the processes that drive short term and long term changes.

Ecological differences between the Hudson Bay Lowlands and other forest-tundra regions suggest that unique approaches are needed for research and management in the Lowlands. The largely saturated conditions, high amount of open water, dominance of wetlands, and relatively low fire frequency indicates that studies of other forest-tundra regions may not always be directly applicable in the Hudson Bay Lowlands. Research conducted in other regions will need to be critically evaluated to determine its applicability to the unique conditions found along Hudson Bay.

CHAPTER 3: VEGETATION STRUCTURE AND DYNAMICS IN THE HUDSON BAY LOWLANDS OF MANITOBA

"I do dimly perceive that whilst everything around me is every-changing, ever-dying, there is underlying all that change a living power that is changeless, that holds all together, that creates, dissolves, and re-creates."

-M.K. Gandhi

PREAMBLE

This chapter presents a quantitative characterization of the structure and dynamics of the vegetation communities that currently occur in the Hudson Bay Lowlands in Manitoba. Defining the vegetation associations and examining the environmental variables influencing them are accomplished using multivariate methods for classification and ordination. Prior to this study, vegetation associations in the region have been delineated subjectively and no sampling has done through most of the inland areas. This chapter presents the first quantitative determination of vegetation associations in the study area.

INTRODUCTION

The Hudson Bay Lowlands form one of the most extensive peatlands in the world, covering over 370,000 km^2 in Ontario and Manitoba and representing 3.5% of the landmass of Canada (Ecological Stratification Working Group 1995). This complex peatland mosaic forms a broad transition zone between the boreal forest to the south and the arctic tundra to the north. Shelford and Twomey (1941) first alluded to successional processes in the Lowlands ecoregion, stating that "there is a clear convergence ... to a climax in which lichens play an important part". In the Hudson Bay Lowlands, fen communities dominate geologically younger sites near the Hudson Bay coast while lichen-heath plateau bogs and treed lichen-heath bogs predominate on the oldest sites further inland (Ritchie 1960, Riley 1982). A number of researchers have hypothesized a fen to bog successional sequence in subarctic peatland ecosystems (Zoltai et al. 1988). This sequence is driven by a terrestrialization process that occurs as peat accumulates above the water table. As peat accumulates the vegetation becomes increasingly isolated from the ground water, resulting in increasingly acidic and oligotrophic conditions that promote bog development (Kratz and DeWitt 1986, Vitt and Kuhry 1992). Terrestrialization in the Lowlands ecoregion reflects the continuous raising of land out of Hudson Bay resulting from post-glacial isostatic uplift (Webber et al. 1970). As a result of these terrestrialization processes, vegetation zonation in the Lowlands ecoregion reflects a successional continuum from younger sites near the coast to older sites further inland (Ritchie 1960).

Vegetation dynamics in peatlands is an on-going process driven by a wide range of local and regional events (Heinselman 1970). A number of environmental factors are known to affect the floristic composition and physiognomy of northern wetlands, including pH, nutrient level, water table level and fluctuations, drainage, nature of the substrate (organic or mineral), surficial geology, climate, and groundwater movement (Jeglum 1971, 1973, Jeglum et al. 1974, Stanek et al. 1977, Jaseniuk and Johnson 1982, Robinson et al. 1989, Vitt and Chee 1990, Timoney et al. 1993). Many of these factors are correlated, however, and overall they affect of the ability of plants to sequester nutrients. It is therefore difficult to obtain meaningful measures of overall nutrient status in northern wetlands (Gorham 1956, Sjors 1963, Jeglum 1973, Jaseniuk and Johnson 1982). An alternative method for estimating nutrient availability is through the use of indicator species, under the assumption that the species present are indicator 'phytometers' (Sjörs 1961, Jeglum 1971, Glaser 1983).

The majority of vegetation research in the Hudson Bay Lowlands is descriptive in nature, and few quantitative floristic and environmental data are available for Manitoba. This largely reflects the inaccessibility and spatial extent of the ecoregion. Early landscape-scale studies were based on very limited ground survey data and normally included descriptions of habitat-based plant communities and often inferred probable successional processes (e.g. Shelford and Twomey 1941, McClure 1943, Ritchie 1957, 1960, 1962, Sjörs 1959). Ritchie (1962) described fourteen vegetation types for northern Manitoba, nine of which occur in the Hudson Bay Lowlands: heath, lichen heath, sedge meadow, shrub and scrub forest, open spruce forest with lichen-shrub, open spruce forest with moss-shrub, closed spruce forest, and larch forest. Along the coast of Hudson Bay, coastal rock-lichen, salt marsh, and beach ridge-tundra heath communities also occur (Jefferies *et al.* 1979, Rewcastle 1983).

Few studies have examined the vegetation dynamics of the forest-tundra ecosystem, particularly in the Hudson Bay Lowlands of Manitoba. Some quantitative studies have been done in the Hudson Bay Lowlands in Ontario by Kershaw and Rouse (1973), Neal and Kershaw 1973, Kershaw (1974), Pierce and Kershaw (1976), Sims *et al.* (1979, 1982), Riley (1982), Pala and Boissonneau (1982), Pala and Weischet (1982), but the similarity between the Lowlands in Manitoba and Ontario remain untested. Few quantitative investigations have been done along inland to coast gradients to compare the sites that are differently aged due to ongoing isostatic uplift. Kershaw (1974) examined the hypothesis proposed by Ritchie (1960) of the developmental sequence of fen on younger sites and bog on older sites through ordination methods, predicting that the sequence should be reflected in the ordination results. Peat accumulation favours the development of successively less hydrophytic communities over time. By contrast, uplands with mineral sand and gravel soils that begin as kames or newly emerged beaches, support much different processes and eventually develop into spruce-lichen woodland on old beach terraces, remaining relatively dry at all stages (Raup 1930). Disturbance events such as fire can significantly influence vegetation dynamics over the short term (<200 years), but habitat conditions such as nutrients and water level determine the dynamics over longer periods (>200 years) (Jaseniuk and Johnson 1982)

The objectives of the chapter are to: (1) characterize and describe the composition and structure of recurrent vegetation types from the study area; (2) delineate species ecological groups (Bergeron and Bouchard 1984) and examine the interrelationships between the vegetation types and these groups; (3) interpret the vegetation types in relation to selected site factors and environmental gradients; (4) summarize the major biotic and abiotic factors and processes determining vegetation dynamics; and (5) infer successional pathways and assess their significance in site-to-site differences in species composition.

The approach used in this chapter is somewhat different from other gradient analysis studies of the temporal or spatial dimension. The vegetation of the Hudson Bay Lowlands exists on a concurrent spatio-temporal gradient inland from the coast due to on ongoing isostatic uplift (Ritchie 1962, Webber *et al.* 1970). Examination of the floristic variability that exists spatially in relation the distance from Hudson Bay can provide valuable insights into the processes that drive vegetation change over time. Multivariate analysis provides a powerful approach to examining these highly complex gradients.

METHODS

Vegetation Data Collection

Field data were collected throughout the study area between June and September of 1998-2000. Sample sites were accessed by helicopter, canoe, train, and on foot, with additional sites near the Churchill townsite accessed by road. Because of the large size of the area (20,000 km^2), limited number of field camps and expensive helicopter time, a cluster sampling approach was used. Researchers were flown to selected locations throughout the study area. At each location, sample plots were selected based on a stratified random sampling program. Sites were deemed acceptable for sampling if they fit the following a priori criteria: relative continuity of the vegetation cover, no obvious evidence of human disturbance, and consistent substrate and topography throughout.

At the centre of each site, percent cover values for all plant species were estimated within a 10 x 10m plot. Vegetation was further subsampled with the plot using two randomly placed 2 x 2m quadrats. In addition, a height profile was measured of the vegetation across a 50m transect that bisected the plot. The nature and degree of variability of the surrounding vegetation and disturbance features were also noted. The majority of plants were identified in the field, though some were collected for identification in the laboratory. Approximately 350 voucher specimens were placed into the University of Manitoba herbarium (WIN). Species nomenclature follows Porsild and Cody (1980) for vascular plants, Scott (1990) for bryophytes and Thomson (1984) for the lichens.

The initial data set included 600 sites located throughout the study area. Coastal salt marsh samples (n=52) were removed because the salt-adapted communities shared few or no species with the inland communities and including them would provide the trivial result of separating coastal and inland areas, which previous analyses R.K. Brook 2000, unpublished data) have shown to be highly distinct. These salt marshes have been described extensively by Jefferies et al. (1979), form less than 1% of the study area (R.K. Brook 2000, unpublished data), and function much differently than inland areas. Coastal beach ridge sites (n=69) dominated with *Dryas integrifolia* form a highly unique vegetation type (composed often of only 1-4 species) were also removed since the low number of species present and the variable vegetation cover (30-100%) make this type highly unique. Recent (within the last 5 years) and regenerating (within the last 80 years) burned areas (n=42) were also removed from the ordination analysis in order to avoid confusion between the influences of fire and the influence from other environmental considerations. Finally, unvegetated sites (<30% vegetation cover) (n=165) were removed. While all of these sites are not included in the gradient analysis, their structure and role in vegetation dynamics is discussed.

Environmental Data Collection

The precise location of each site was determined using a hand held Global Positioning System (GPS), by taking two location fixes per second for 30-45 minutes and then averaging them over time. The spatial error associated with these points was normally less than 10 m when compared with differentially corrected GPS (R.K. Brook 2000, unpublished data). All of the sample site locations collected by GPS were converted to vector points and used to extract associated ancillary data using a geographical information system (GIS) (SPANS, PCI Geomatics 1999). Using vector coverage of the Hudson Bay coast, the minimum distance to the coast was calculated as an estimate of the relative age of each site, since sites further inland generally emerged from Hudson Bay earlier than those closer to the coast. The relative north-south position of each site was also determined by using the UTM northing value.

Since sampling was conducted over the course of the entire summer, a general index of moisture was estimated (1 = xeric site with completely dry substrate, 2 = substrate moist but not saturated, 3 = substrate saturated with no standing water, 4 = substrate saturated with 2-20% standing water, 5 = substrate saturated with >20% standing water). The percent ground cover of open water and unvegetated substrate was also estimated within each plot.

Surficial geology data were compiled from a digital geology map of the region (Dredge and Nixon 1992). Of the 18 types defined by Dredge and Nixon (1992) for northwestern Manitoba, only 8 were associated with sample sites in the study area and of these, only four were common. As some of the geology types were of similar structural composition (e.g. sandy material), they were aggregated to form 4 major surficial types: (1) Sand, composed of kame and esker sands, beach ridge complexes, alluvial deposits and littoral and sublittoral sand; (2) Marine, composed of stony marine pelite; (3) Silt, composed of intertidal silt; and (4) Offlap, composed of undifferentiated marine offlap deposits.

Ecological indicator species values (Persson 1981) were used to develop an estimate of nutrient availability for each plot. Each of the 61 most commonly occurring species (frequency greater than 6) in the 272 plots was assigned to one of the five nutrient status categories suggested by Jeglum (1971): (1) very oligotrophic; (2) oligotrophic; (3) mesotrophic, (4) eutrophic; (5) very eutrophic. Determination of the species indicator values were based on previous studies, in particular Jeglum (1971), Jeglum *et al.* (1974), Bergeron and Bouchard (1984), Vitt and Slack (1984), Vitt and Chee (1990), Racey *et al.* (1996), Harris et al. (1996), and Ringius and Sims (1997). The nutrient status of each plot was calculated as a percent cover weighted value for all species within the plot, following Kenkel (1987).

Cluster analysis

Cluster analysis (Ward's method, Legendre and Legendre 1998), based on a chord distance matrix (Orloci 1967) of log transformed species cover data, was performed to classify plots into vegetation types (after Orloci and Stanek 1979) and species into ecological groups (after Bergeron and Bouchard 1984) (HIERCLUS, Podani 1994). Cluster analysis was based on 272 plots containing all 114 species into 6 major vegetation associations and used the 61 most commonly occurring species (frequency greater than 6) to delineate 7 species ecological groups (HIERCLUS, Podani 1994). Measured vegetation composition and physiognomy, environmental variables, surficial geology (from Dredge and Nixon 1992), and derived nutrient index data were summarized for each vegetation type.

Concentration Analysis

Concentration analysis (Feoli and Orloci 1979) was used to examine relationships within and between the vegetation types and ecological species groups determined by the cluster analysis. The method begins by ordering the data matrix by vegetation association and species group. The number of occurrences of species within a given group in each of the vegetation associations was then computed. The resulting r x c contingency table (where r = no. of species and c = no. of vegetation associations) is input to a correspondence analysis (Hill 1974) after adjustments to equal block size (Feoli and Orloci 1979). This approach provides a simultaneous ordination of species groups and vegetation associations through a partitioning of the total contingency chi-squared.

Canonical Correspondence Analysis

Canonical correspondence analysis (CCA) was used to examine the extent to which trends in the vegetation types reflect environmental variability. This method summarizes and dquantifies the relationship between two datasets (in this case environment and vegetation), and determines variable redundancy and cross-correlations. In CCA, the CA ordination axes are constrained by the environmental data using a multiple regression approach. Environmental variables include wetness index, UTM northing, distance to Hudson Bay coast, surficial geology and nutrient index. The relative positions of the six vegetation types were then examined along each environmental gradient.

Successional Dynamics

Identification of successional dynamics is based on the results of the CCA of the environmental gradients associated with each of the six vegetation types. Positions of each of the six vegetation types within the CCA were interpreted with respect to the influence of the distance from the coast (interpreted as a measure of site age), moisture status and nutrient status. Dynamics of the six vegetation types were then predicted based on these vegetation-environment relationships. Additional information from the literature (e.g. Gorham 1956, Ritchie 1957, 1959, 1960a, 1960b, 1960c, Ingram 1967, Jeglum 1971, 1973, Jeglum et al. 1974, Heinselman 1970, Kenkel 1987, Zoltai et al. 1988, Vitt and Chee 1990, Vitt and Kuhry 1992) was used to validate the predicted successional sequence.

RESULTS

Classification of Plots

Six vegetation types (labelled I-VI) have been recognized based on the cluster analysis dendrogram of the 272 plots (Fig. 3.1). Each vegetation type is named for its dominant plant life form (highest mean percent cover), physiognomic characteristics (e.g. bog, fen) and relative nutrient status.

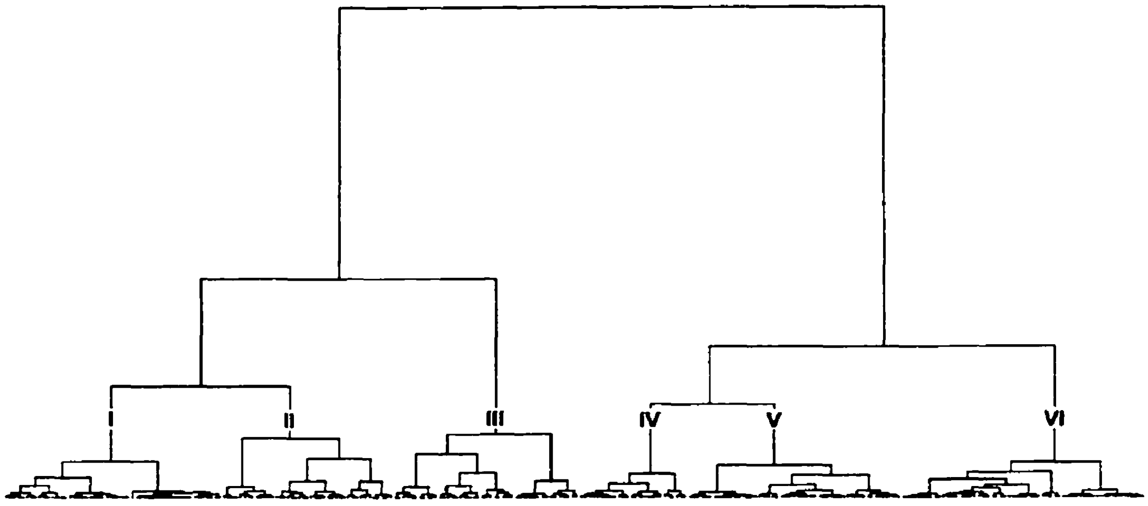


Figure 3.1. Sum of squares dendrogram, based on chord distance of the 272 plots. Six vegetation types (I-VI) are indicated.

Vegetation Type Descriptions

The following overview of the 6 major vegetation types describes the floristic composition, physiognomic structure and vegetation-environment relationships. Table 3.1 provides a characterization of the floristic and physiognomic features of the six types in detail along with their associated environmental parameters. Tables 3.2-3.7 provide the species compositions for vegetation types I-VI, respectively. A complete list of identified species is presented in Appendix I.

The first dichotomy of the cluster analysis of the plots separates fen vegetation types from bog types. Types I, II, and III are fen wetlands. They all are found exclusively on organic soils that are saturated to moist with relatively high nutrient levels. Types V and VI are bogs that occur on mesic to dry organic soils and have relatively low nutrient levels. Type IV is found on well-drained sand and gravel mineral soil that is relatively dry supporting vegetation that shares some commonality with the two bog types.

Table 3.1. Environmental, physiognomic and floristic characterization of the six vegetation types recognized in Fig.3.1.

	I	II	III	IV	V	VI
	Sedge Meadow Fen	**Sedge Shrub Fen**	**Shrub Treed Fen**	**Spruce Heath Treed Upland**	**Spruce Heath Treed Bog**	**Lichen Heath Plateau Bog**
Number of Sites Sampled	52	42	44	25	52	57
Nature of Surficial Geology	marine	silt, marine, offlap	marine	sand	offlap	offlap
Nature of Substrate	peat	peat	peat	sand and gravel	peat	peat
Mean Distance From Coast (km)	6.60	23.15	25.58	20.10	65.30	40.90
Mean Nutrient index (max=5)	3.77	3.51	2.99	1.94	1.45	1.24
Nutrient Status	eutrophic	mesotrophic-eutrophic	mesotrophic	oligotrophic	very oligotrophic	very oligotrophic
Wetness (max=4)	2.94	3.29	3.05	2.04	2.12	2.07
% open water	0.58	6.19	1.72	-	0.02	0.35
Tree Height (mean)	-	1.71	4.18	9.91	5.27	0.33
Species Richness	32	51	48	32	37	35
Mean Tree Cover	-	3.60	10.51	28.04	23.85	1.30
Mean Tall Shrub Cover	2.06	4.31	27.98	11.12	1.17	0.00
Mean Low Shrub Cover	12.35	15.40	17.40	26.40	28.10	26.32
Mean Herb Cover	55.19	49.29	20.05	3.16	4.00	4.86
Mean Moss Cover	21.17	12.45	19.12	5.44	11.10	5.11
Mean Lichen Cover	0.60	1.86	0.98	26.12	32.33	62.28
Dominant Trees	-	*Larix laricina*	***Larix laricina***	***Picea glauca***	*Picea mariana*	-
	-	*Picea mariana*	*Picea mariana*	*Larix laricina*	*Larix laricina*	-
Dominant Tall Shrub (>0.5m)	*Betula glandulosa*	*Betula glandulosa*	*Salix planifolia*	*Betula glandulosa*	*Betula glandulosa*	-
Dominant Low Shrubs (<0.5m)	*Salix arctophila*	*Salix arctophila*	*Vaccinium uliginosum*	*Ledum groenlandicum*	***Ledum groenlandicum***	***Ledum decumbens***
	Salix reticulata	***Dryas integrifolia***	*Salix reticulata*	*Empetrum nigrum*	*Ledum decumbens*	*Empetrum nigrum*
Dominant Herbs	-	*Equisetum sp.*	*Equisetum sp.*	*Rubus chamaemorus*	*Rubus chamaemorus*	*Rubus chamomerus*
	-	***Menyanthes trifoliata***	***Smilicina trifolia***	-	-	-
Dominant Graminoids	*Carex aquatilis*	*Carex aquatilis*	*Carex aquatilis*	-	-	*Carex capillaris*
	Scirpus caespitosus	*Scirpus caespitosus*	*Carex capillaris*	-	-	-
Dominant Lichens	-	-	-	*Cladina stellaris*	*Cladina stellaris*	*Cladina stellaris*
	-	-	-	*Cladina rangiferina*	*Cladina rangiferina*	*Cetraria nivalis*
Dominant Moss	***Scorpidium sp.***	***Drepanocladus sp.***	*Sphagnum sp.*	*Sphagnum sp.*	*Sphagnum fuscum*	*Sphagnum sp.*

Type I (Sedge Meadow Fen, n=52) is a nutrient rich very wet fen with no trees and few shrub covered hummocks. Graminoids are the dominant life form (40-60%), particularly *Carex aquatilis* and *Scirpus caespitosus*, with mosses also being important (20-40%), particularly *Scorpidium* spp. Shrubs such as *Betula glandulosa*, *Salix arctophila*, and *S. reticulata* comprise the shrub layer (0-30%). Herbs are generally rare (<1%). The relatively homogenous, saturated nature of this type results in low species diversity. The open physiognomic structure limits snow accumulation in the winter. Water table is at or above the soil surface during most of the growing season, though there is generally little to no open water within this type. Marine clay, till and limestone deposits make the water alkaline.

Two variants of this type are recognized, shrub and open. The open variant has a high nutrient status, high cover of *Scirpus caespitosus*, and minimal shrub cover (0-10%). The shrub variant has a lower nutrient status, higher cover of *Carex capillaris* and significant low shrub cover (20-40%), (*Salix arctophila* and *Betula glandulosa*).

Similar types have been referred to as "sedge meadow" (Kershaw 1974), "graminoid open fen" Sims et al. (1982) and "sedge tundra" (Campbell 1995).

Table 3.2. Vegetation Type I (Sedge Meadow Fen). Frequency and mean cover for all species present.

Life Form	Species	Mean Cover (%)	Frequency
Tree	*Larix laricina*	0.02	0.02
Shrub	*Salix arctophila*	5.98	0.62
	Betula glandulosa	4.48	0.54
	Vaccinium uliginosum	0.48	0.13
	Rhododendron lapponicum	0.35	0.17
	Empetrum nigrum	0.29	0.12
	Myrica gale	0.27	0.10
	Arctostaphylos sp.	0.17	0.17
	Salix candida	0.15	0.15
	Salix lanata	0.08	0.08
	Salix brachycarpa	0.04	0.04
	Vaccinium vitis- idaea	0.04	0.04
	Ledum decumbens	0.04	0.04
	Dryas integrifolia	0.02	0.02
Graminoid	*Carex aquatilis*	38.08	1.00
	Scirpus caespitosus	12.50	0.50
	Carex capillaris	3.87	0.23
	Salix reticulata	1.85	0.25
	Salix pedicularis	0.98	0.08
	Andromeda polifolia	0.62	0.27
	Carex saxatilis	0.10	0.10
Herb	*Pedicularis sudetica*	0.19	0.19
	Bartsia alpina	0.13	0.13
	Potentilla palustris	0.13	0.13
	Petasites sagitatus	0.08	0.08
	Pinguilicula vulgaris	0.06	0.06
Moss	*Scorpidium sp.*	20.58	0.94
	Dicranum sp.	0.58	0.02
	Drepanocladus sp.	0.04	0.04
Lichen	*Cetraria nivalis*	0.60	0.06
	Cladina rangiferina	0.02	0.02
	Alectoria ochroleucra	0.02	0.02

Type II (Sedge Shrub Fen, n=42) are fens dominated by graminoids (10-50%), particularly *Carex aquatilis* and *Scirpus caespitosus*. The moss layer is variable (0-30%), but is primarily *Drepanocladus* spp. Herbs include *Menyanthes trifoliata* and *Equisetum* spp. Scattered stunted *Larix laricina* and/or *Picea mariana* (<2m tall) may be present at less than 10% cover, but are absent in many plots. Microtopography includes low hummocks with weakly to moderately developed hollows and may also include raised "strings" perpendicular to the water flow, separated by semi-permanent pools (flarks). The hummocks are dominated by low shrubs such as *Betula glandulosa*, *Dryas integrifolia*, *Arctostaphylos rubra*, *Rhododendron lapponicum*, *Vaccinium uliginosum* and mosses, particularly *Tomenthypnum nitens*. This type contains the highest number of species due to the wide variety of microhabitats from wet to dry and the relatively high degree of minerotrophy. Trees are almost exclusively found clumped on hummocks. Shrubs are abundant with moderately rich herb, graminoid and moss layers usually present. Water table is at or below the soil surface for most of the growing season.

Three variants of this type are recognized, hummock (IIa), wet (IIc), and shrub (IIb). The wet variant (IIc) has a high nutrient status and is much wetter, often with large patches of open water (10-40%). It is treeless with a high moss cover that is mostly *Drepanocladus spp*. The shrub variant (IIb) is also dominated by graminoid vegetation but includes a much higher cover of *Scirpus caespitosus*. It wetter than the hummock variant, is treeless, and has a higher nutrient status. The hummock variant (IIa) has relatively low nutrient status and primarily composed of graminoid vegetation with low, treeless hummocks dominated by low shrubs such as *Betula glandulosa*, *Dryas integrifolia*, *Arctostaphylos rubra*, *Rhododendron lapponicum*, *Vaccinium uliginosum* and mosses, particularly *Tomenthypnum nitens*. Lichen such as *Cetraria nivalis* occurs at low cover (<5%). Tall and medium shrubs are rare or absent. Soil is wet in the low areas but is quite dry on the raised hummocks. Hummocks generally have a sparse tree cover of *Larix laricina* and *Picea mariana*.

Similar types have been referred to as "low shrub open fen" Sims et al. (1982) and "low shrub fen" (Jeglum 1973).

Table 3.3. Vegetation Type II (Sedge Shrub Fen). Frequency and mean cover for all species present.

Life Form	Species	Mean Cover (%)	Frequency
Tree	*Larix laricina*	1.93	0.19
	Picea mariana	1.19	0.10
	Picea glauca	0.48	0.02
Shrub	*Betula glandulosa*	5.40	0.55
	Salix arctophila	3.90	0.57
	Dryas integrifolia	2.43	0.21
	Arctostaphylos sp.	1.93	0.43
	Rhododendron lapponicum	1.50	0.40
	Salix reticulata	1.50	0.19
	Vaccinium uliginosum	1.48	0.40
	Salix brachycarpa	0.76	0.07
	Empetrum nigrum	0.62	0.19
	Andromeda polifolia	0.48	0.26
	Sheperdia canadensis	0.48	0.02
	Salix candida	0.36	0.14
	Salix lanata	0.31	0.10
	Ledum decumbens	0.14	0.14
	Salix planifolia	0.05	0.05
	Ledum groenlandicum	0.05	0.05
	Vaccinium vitis- idaea	0.05	0.05
	Kalmia polifolia	0.02	0.02
	Salix pedicellaris	0.02	0.02
	Oxycoccus microcarpa	0.02	0.02
Graminoid	*Carex aquatilis*	24.55	0.90
	Scirpus caespitosus	11.95	0.50
	Carex capillaris	3.86	0.24
	Carex vaginata	1.67	0.10
	Carex saxatilis	0.26	0.05
	Eriophorum angustifolium	0.07	0.07
	Triglochin maritima	0.05	0.05
Herb	*Equisetum sp.*	1.50	0.17
	Menthanthes trifolia	1.19	0.12
	Potentilla palustris	0.81	0.12
	Pedicularis sudetica	0.31	0.10
	Bartsia alpina	0.24	0.24
	Pinguilicula vulgaris	0.12	0.12
	Petasites sagitatus	0.07	0.07
	Smilicina trifolia	0.02	0.02
	Epilobium angusitifolium	0.02	0.02
Moss	*Drepanocladus sp.*	8.81	0.40
	Tomenthypnum nitens	1.98	0.21
	Scorpidium sp.	0.74	0.10
	Sphagnum fuscum	0.02	0.02
Lichen	*Cetraria nivalis*	0.88	0.21
	Cladina stellaris	0.50	0.07
	Cladina rangiferina	0.31	0.10
	Stereocaulon alpinum	0.24	0.02
	Cladina mitis	0.05	0.05
	Cetraria cucculatta	0.02	0.02
	Bryocaulon divergens	0.02	0.02
	Evernia mesomorpha	0.02	0.02

Type III (Shrub Treed Fen, n=44) include fens with scattered tamarack (generally 1-6m tall) at more than 10% cover. Tall, medium and low shrubs are common in most sites. Water table is at or below the soil surface for most of growing season. Microtopography includes low hummocks with weakly to moderately developed hollows. Strings and flarks are less common. Trees are often clumped on higher hummocks. Microtopography generally includes intermediate to high hummocks with hollows, though flat microtopography dominated by tall shrubs or a flat lawn of *Sphagnum angustifolium* may also occur.

Three variants of this type are recognized, wet *Sphagnum*, hummocky, and tall shrub. Though all types have the same general nutrient status, they vary widely in ground wetness. The hummock type is wet in low areas, but quite dry on the intervening hummocks. Graminoid cover is mostly *Carex aquatilis* with some *Eriophorum angustifolium.* Hummocks have well-developed treed cover (10-30%) of *L. laricina and P. mariana*, sparse lichens, *Cladina mitis* and *C. rangiferina* and *Sphagnum fuscum* moss cover. *Salix planifolia* is the dominant shrub. The *Sphagnum* variant is found on saturated soils with a flat lawn of *Sphagnum angustifolium* interspersed with *Equisetum* spp. and *Eriophorum angustifolium.* Herb cover includes *Smilicina trifolia* and *Carex capillaris*. Tree cover is exclusively *L. laricina* (10-30%). The tall shrub variant is on relatively dry soil and is dominated by tall and medium shrubs (70-100%), including *Myrica gale, Salix planifolia, S. candida, S. brachycarpa* and *S. lanata* with little or no coniferous trees.

Similar types have been referred to as “mixed swamp forest” (Hustich 1949), “tamarack swamp” (Hustich 1957), “larch forest” (Ritchie 1962) “tamarack fen” (Brokx 1965), “wooded rich fen” (Sjörs 1963), and “treed fen” (Sims et al. 1982, Pala and Weischet 1982).

Table 3.4. Vegetation Type III (Shrub Treed Fen). Frequency and mean cover for all species present.

Life Form	Species	Mean Cover (%)	Frequency
Tree	*Larix laricina*	6.00	0.48
	Picea mariana	4.34	0.27
	Picea glauca	0.05	0.05
Shrub	*Salix planifolia*	12.14	0.57
	Betula glandulosa	11.48	0.70
	Myrica gale	4.34	0.25
	Vaccinium uliginosum	3.57	0.39
	Salix reticulata	1.84	0.16
	Salix candida	1.66	0.18
	Kalmia polifolia	1.45	0.23
	Empetrum nigrum	1.30	0.27
	Ledum groenlandicum	1.30	0.23
	Salix pedicellaris	1.20	0.18
	Salix arctophila	1.18	0.11
	Salix brachycarpa	0.91	0.07
	Salix lanata	0.73	0.11
	Rhododendron lapponicum	0.45	0.05
	Oxycoccus microcarpa	0.30	0.30
	Arctostaphylos sp.	0.27	0.07
	Dryas integrifolia	0.25	0.05
	Andromeda polifolia	0.11	0.11
	Vaccinium vitis- idaea	0.07	0.07
	Ledum decumbens	0.05	0.05
Graminoid	*Carex aquatilis*	7.00	0.50
	Carex capillaris	3.41	0.11
	Eriophorum angustifolium	1.57	0.30
	Scirpus caespitosus	0.02	0.02
	Triglochin maritima	0.11	0.11
Herb	*Equisetum sp.*	4.27	0.43
	Menthanthes trifolia	1.55	0.30
	Smilicina trifolia	2.36	0.30
	Potentilla palustris	0.20	0.20
	Rubus chamaemorus	0.57	0.16
	Petasites sagitatus	0.14	0.14
	Epilobium angusitifolium	0.02	0.02
Moss	*Sphagnum sp.*	10.98	0.55
	Sphagnum fuscum	2.98	0.18
	Drepanocladus sp.	1.89	0.23
	Dicranum sp.	1.64	0.14
	Tomenthypnum nitens	0.93	0.09
	Scorpidium sp.	0.45	0.05
	Pleurozium schreberi	0.23	0.02
Lichen	*Bryocaulon divergens*	0.75	0.14
	Cladina mitis	0.45	0.02
	Cladina rangiferina	0.25	0.05
	Evernia mesomorpha	0.14	0.14
	Cladina stellaris	0.05	0.05
	Peltigera apthosa	0.05	0.05

Type IV (Spruce Heath Treed Upland, n=25) occurs on kame and beach sand or gravel mineral substrate on a thin organic soil. Overstory is dominated by tall (5-15m, well-developed *Picea glauca* and *Larix laricina*. Shrub cover includes tall shrubs (e.g. *Betula glandulosa)* and low shrubs (e.g. *Ledum groenlandicum*). Ground cover is dominated by lichen, particularly *Cladina stellaris*. Water table is below the soil surface for most or all of the growing season and the ground is normally well drained. The dry, nutrient poor condition results in low species diversity.

Two variants of this type are recognized, shrub and lichen. Both variants have similar tree canopies and moss cover. The shrub variant is dominated by tall and medium shrubs (10-60%), including *Betula glandulosa*, *Salix planifolia* and low shrubs (10-40%) including *Empetrum nigrum* and *Vaccinium uliginosum*. The lichen variant has a sparse or non-existent tall and medium shrub cover and a low shrub cover (10-40%) that includes *Ledum groenlandicum* and *Empetrum nigrum*. Lichens are dominant (20-60%), particularly, *Cladina stellaris*, *C. rangiferina* and *C. mitis*. Similar types have been referred to as "white spruce forest" (Campbell 1995).

Table 3.5. Vegetation Type IV. (Spruce Heath Treed Upland). Frequency and mean cover for all species present.

Life Form	Species	Mean Cover (%)	Frequency
Tree	*Picea glauca*	20.88	0.96
	Larix laricina	7.12	0.76
	Picea mariana	0.44	0.08
Shrub	*Ledum groenlandicum*	15.52	0.88
	Empetrum nigrum	11.40	0.92
	Betula glandulosa	3.72	0.28
	Vaccinium uliginosum	2.92	0.68
	Salix planifolia	2.28	0.48
	Arctostaphylos sp.	1.20	0.48
	Vaccinium vitis- idaea	1.04	0.68
	Sheperdia canadensis	0.60	0.24
	Ledum decumbens	0.24	0.08
	Dryas integrifolia	0.08	0.08
	Salix reticulata	0.08	0.08
Graminoid	*Carex aquatilis*	0.40	0.04
	Scirpus caespitosus	0.04	0.04
Herb	*Rubus chamaemorus*	1.60	0.12
	Geocaulon lividum	0.64	0.28
	Epilobium angusitifolium	0.24	0.24
	Equisetum sp.	0.08	0.08
Moss	*Sphagnum sp.*	3.28	0.24
	Hylocomium splendens	1.32	0.24
	Pleurozium schreberi	0.48	0.12
	Dicranum sp.	0.48	0.12
	Tomenthypnum nitens	0.08	0.08
Lichen	*Cladina stellaris*	17.00	0.96
	Cladina rangiferina	6.00	0.84
	Cladina mitis	2.88	0.28
	Cetraria nivalis	0.52	0.16
	Cladonia sp.	0.44	0.40
	Peltigera apthosa	0.32	0.32
	Stereocaulon alpinum	0.04	0.04

Type V (Spruce-Heath Treed Bog, n=52) is a treed wetland with black spruce. Tree size (1-10m tall) and cover (10-40%) is highly variable. Understory is dominated by ericaceous shrubs, particularly *Ledum groenlandicum*. Minerotrophic indicator species are absent. Ground cover consists primarily of *Sphagnum* spp., particularly *Sphagnum fuscum* and lichens such as *Cladina* stellaris and *Cetraria nivalis*. Water table is below the soil surface for most of the growing season.

Two variants of Type V include lichen and moss. The moss variant has a moss layer (20-30%) dominated by *Sphagnum fuscum* and a relatively low lichen cover (10-30%), while the lichen variant has a lower moss cover (0-20%) and a high lichen cover (20-60%).

Similar types have been referred to as "open spruce forest with lichen-shrub" and the "open spruce forest with moss-shrub" types identified by Ritchie (1962) or "black spruce muskeg" (Ritchie 1956), "treed bog" (Sims et al. 1982, Pala and Weischet 1982) and "black spruce palsa" (Campbell 1995).

Table 3.6. Vegetation Type V (Spruce Heath Treed Bog). Frequency and mean cover for all species present.

Life Form	Species	Mean Cover (%)	Frequency
Tree	*Picea mariana*	22.50	1.00
	Larix laricina	1.37	0.33
	Picea glauca	0.23	0.06
Shrub	*Ledum groenlandicum*	19.27	0.98
	Ledum decumbens	7.08	0.83
	Empetrum nigrum	1.58	0.54
	Vaccinium vitis- idaea	1.10	0.75
	Betula glandulosa	1.08	0.17
	Vaccinium uliginosum	1.02	0.50
	Salix planifolia	0.27	0.10
	Oxycoccus microcarpa	0.23	0.23
	Andromeda polifolia	0.08	0.08
	Salix arctophila	0.06	0.06
	Kalmia polifolia	0.06	0.06
	Salix pedicellaris	0.02	0.02
Graminoid	*Carex vaginata*	0.04	0.04
	Carex aquatilis	0.02	0.02
	Carex capillaris	0.02	0.02
Herb	*Rubus chamaemorus*	4.04	0.85
	Equisetum sp.	0.04	0.04
	Smilicina trifolia	0.02	0.02
	Epilobium angusitifolium	0.02	0.02
Moss	*Sphagnum fuscum*	5.87	0.44
	Sphagnum sp.	4.35	0.35
	Pleurozium schreberi	0.77	0.06
	Dicranum sp.	0.21	0.04
	Hylocomium splendens	0.02	0.02
Lichen	*Cladina stellaris*	19.13	0.98
	Cladina rangiferina	9.73	1.00
	Cladina mitis	2.94	0.63
	Cetraria nivalis	1.62	0.38
	Cladonia sp.	0.85	0.63
	Bryocaulon divergens	0.21	0.04
	Peltigera apthosa	0.04	0.04
	Stereocaulon alpinum	0.02	0.02
	Cetraria cucculatta	0.02	0.02
	Evernia mesomorpha	0.02	0.02

Type VI (Lichen-Heath Plateau Bog, n=57) includes bogs with a well-developed peat layer that includes significant permafrost. Scattered, stunted black spruce (<2m tall) may be present at <10% cover. Ericaceous shrubs (e.g. *Ledum decumbens*) often occur at high cover. Lichens such as *Cladina stellaris* and *Cetraria nivalis* generally form over 50% of the ground cover. Water table is below the soil surface for most or all of the growing season. The raised areas no longer have a water budget, which prevents the growth of most peat-forming species. Polygonal surface patterns are common with permafrost within 50 cm of the surface.

No floristic variants are recognized for this type, as there is generally little trended variability in the floristic data. However, variation in the level of polygonal cracking and melt pond development results in a wide range of water cover within this type. Similar types have been referred to as "lichen heath" (Ritchie 1962) and "hummocky lichen tundra" (Campbell 1995).

Table 3.7. Vegetation Type VI (Lichen Heath Plateau Bog). Frequency and mean cover for all species present.

Life Form	Species	Mean Cover (%)	Frequency
Tree	*Picea mariana*	0.95	0.12
	Larix laricina	0.35	0.04
Shrub	*Ledum decumbens*	15.65	1.00
	Empetrum nigrum	7.07	0.84
	Vaccinium uliginosum	1.74	0.32
	Vaccinium vitis- idaea	1.40	0.77
	Arctostaphylos sp.	0.54	0.07
	Rhododendron lapponicum	0.54	0.07
	Betula glandulosa	0.30	0.14
	Dryas integrifolia	0.21	0.05
	Salix arctophila	0.18	0.02
	Andromeda polifolia	0.12	0.12
	Kalmia polifolia	0.11	0.11
	Ledum groenlandicum	0.09	0.09
	Salix reticulata	0.02	0.02
Graminoid	*Carex capillaris*	1.44	0.14
	Carex vaginata	0.23	0.07
	Scirpus caespitosus	0.19	0.04
	Eriophorum angustifolium	0.02	0.02
Herb	*Rubus chamaemorus*	2.75	0.84
	Menthanthes trifolia	0.18	0.02
	Potentilla palustris	0.02	0.02
Moss	*Sphagnum sp.*	2.98	0.37
	Sphagnum fuscum	2.12	0.33
	Tomenthypnum nitens	0.05	0.05
	Dicranum sp.	0.04	0.04
Lichen	*Cladina stellaris*	23.51	0.91
	Cetraria nivalis	13.39	1.00
	Cladina mitis	11.75	0.98
	Cladina rangiferina	8.86	0.72
	Alectoria ochroleucra	2.65	0.58
	Cladonia sp.	1.35	0.37
	Cetraria cucculatta	1.28	0.18
	Bryocaulon divergens	0.70	0.39
	Stereocaulon alpinum	0.28	0.12

Species Ecological Groups

Seven species ecological groups have been recognized based on the cluster analysis dendrogram of the 61 most common species (Fig. 3.2). The 61 species in the 7 groups is given in Table 1. The species groups are briefly described as follows. Group A is comprised of 7 species that generally occur in nutrient rich saturated areas on well-decomposed organic soils. Group B contains 10 species characteristic of elevated hummocks situated in fen wetlands. Soils are poorly decomposed, oligotrophic and acidic peat and are generally raised 0.1-1.0 m above the water table. Group C includes 10 species that are found in moderately nutrient rich and moderately decomposed peat soils with low saturated hollows and slightly elevated hummocks. Group D consists of 10 species that occur on moderately to poorly decomposed peat soils with low saturated hollows and elevated hummocks. Group E contains 7 species associated with elevated and well-drained mineral soils. Group F is comprised of 7 species on poorly decomposed, oligotrophic and acidic peat. Group G includes 10 species also found on poorly decomposed, oligotrophic and acidic peat with a dry peat surface.

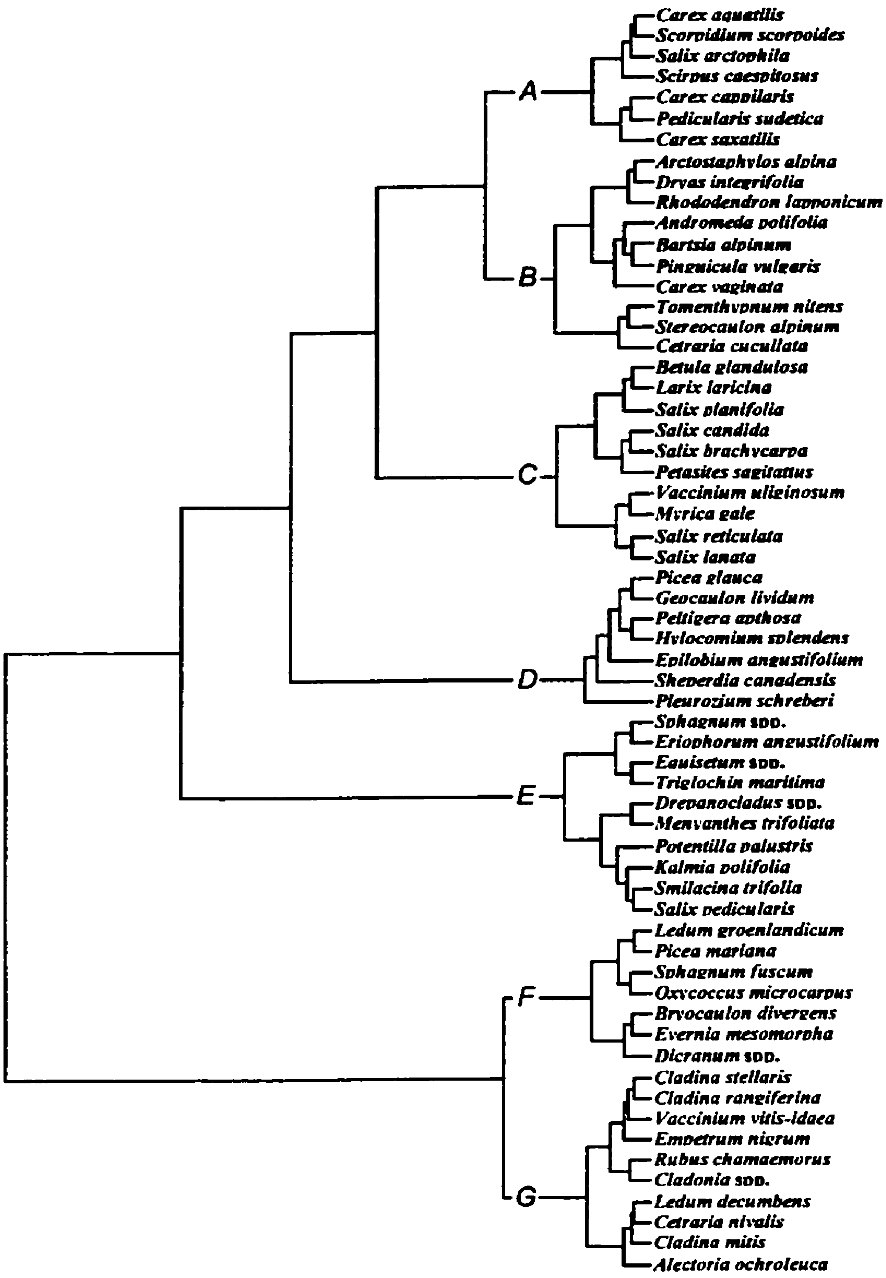


Figure 3.2. Sum of squares dendrogram, based on chord distance of the 61 most common plant species (frequency greater than 6). The seven ecological species groups (A-G) are indicated.

Concentration Analysis

Vegetation types and species groups show a high level of association in the concentration analysis (Fig. 3.3). The cumulative eigenvalues for both axes I and II is 76%, indicating that these two axes represent a large portion of the variation that exists. The first axis represents a wetness-nutrient gradient, with wetness and nutrients increasing from left to right. The second axis appears to represent a woodyness gradient, with increasing woodyness from bottom to top.

Type I shows a strong affinity to Group A, with both being found in wet, nutrient rich conditions. Type II is closely associated with Group B but is also near to Group A indicating the similarities in nutrient rich and saturated to mesic conditions. Type III shows a strong affinity to Group C and also to Group D as a result of the combination of saturated and relatively dry hummocks that are interspersed throughout. Type IV is not strongly associated with any one particular group, but is weakly associated with Groups C, D, E, and F as the unique mineral soils with an organic layer on top create well drained oligotrophic conditions. Type V is closest to Group G, also due to the poorly decomposed, oligotrophic and acidic peat conditions. It is also near Group F that is likewise found on decomposed, oligotrophic and acidic peat conditions. Type VI is near Group G, reflecting poorly decomposed, oligotrophic and acidic peat.

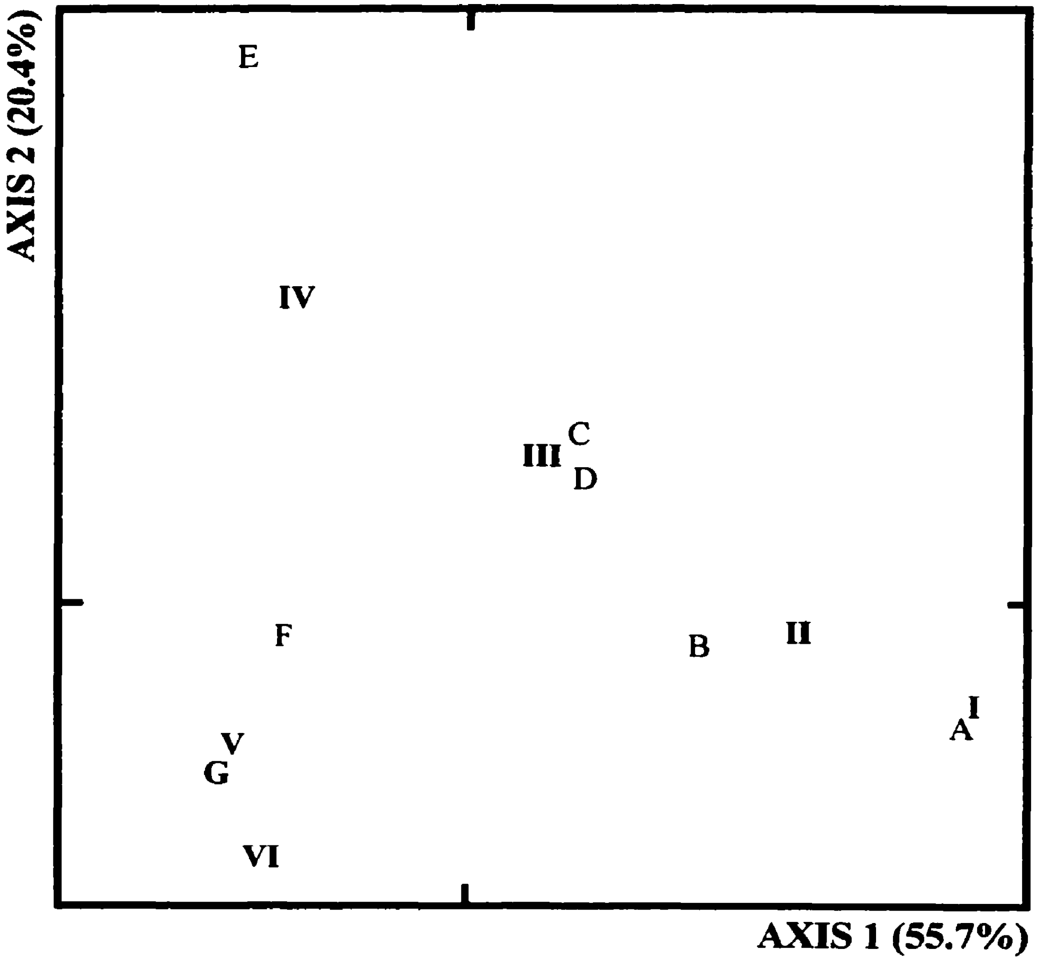


Figure 3.3. Concentration analysis ordination of the six vegetation types (I-VI) and seven ecological species groups (A-G).

Ordination of Plots (CCA)

In the two-dimensional ordination of 272 plots and 114 species (Fig. 3.4), the vegetation types are identified (see Table 3.1) to aid interpretation. The redundancy value was 16%. The variable "nutrients" shows the highest correlation with axis I, indicating a strong trend of increasing nutrients from left to right. Soil "wetness" shows a similar trend along axis I of increasing wetness from left to right. The variable "distance" from the coast is also highly correlated with the first axis, but increasing from right to left. So axis I is a wetness-nutrient gradient that is associated with distance from the coast. Vegetation types I, II, and III are closest to the Hudson Bay, are wetter and have higher nutrient levels, while those sites farther from the coast (types IV, V, VI) are drier and more oligotrophic. Along this gradient, the vegetation types are generally ordered from right to left in the following order: I, II, III, IV, V, VI.

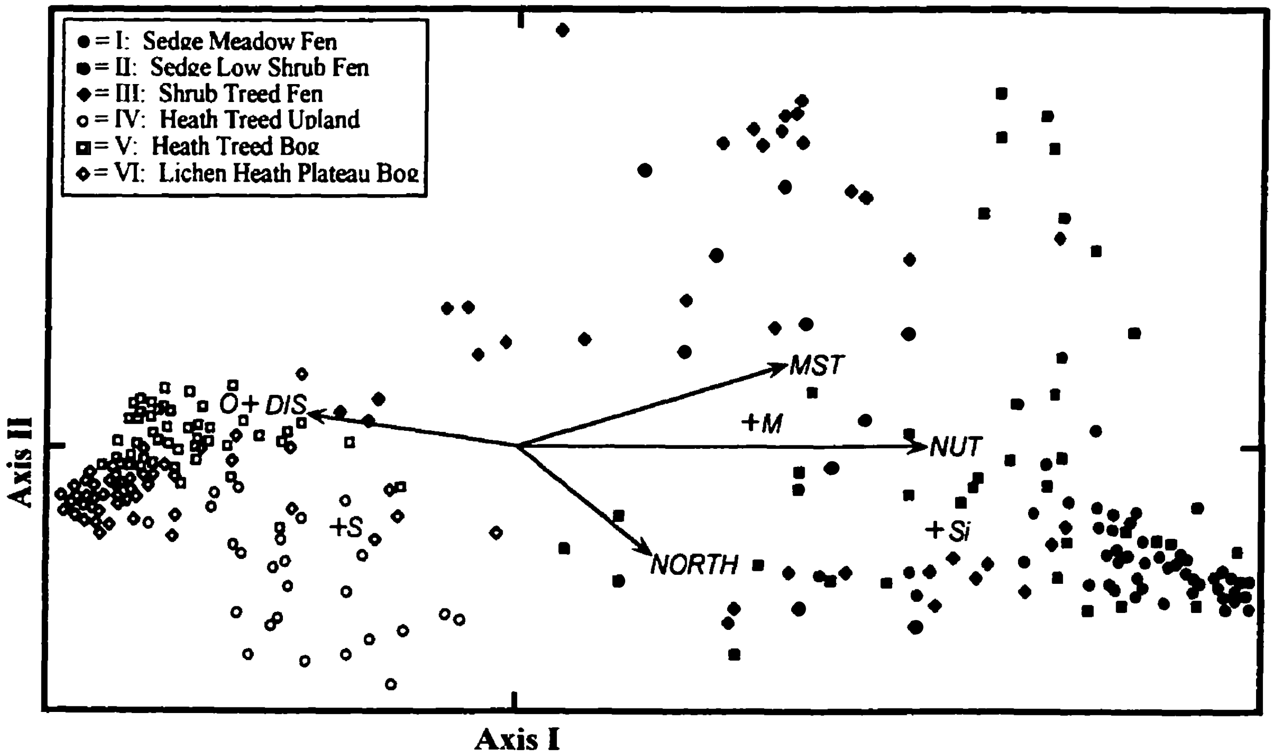


Figure 3.4. Canonical correspondence analysis biplot of 272 plots and five environmental variables (distance to coast (DIS); latitude (NORTH); moisture (MST); nutrients (NUT); and surfical geology: sand (S); marine (M); silt (Si); and Offlap (O) (axes I and II).

DYNAMICS OF PEATLAND VEGETATION

A synoptic model of the successional trajectories is presented for vegetation types on organic soil (Fig. 3.5). There is a general progression in the vegetation from Type I to Type II to Type III of increasing distance from the Hudson Bay coast, indicating an increase in age across these types. This corresponds to an increasing thickness in the peat layer and decreasing wetness with a corresponding decline in nutrient status.

There is also a distinct difference in the vegetation types with respect to tree cover, with a treed and a treeless form that becomes apparent early in the successional process. While the wet Type I does not have trees growing on it, the slightly drier Type II with its sparse, low hummocks provide conditions where trees could grow. However, only some sites have trees, while others do not. Type III is generally treed (>10% cover).

The progression from Type I to Type III represents a transition of the vegetation from a very wet, nutrient rich fen vegetation dominated system (Type I) to a slightly less wet, less nutrient rich system (Type II) that is dominated by fen vegetation, but also contains some bog adapted species (e.g. *Dryas integrifolia*) and then to a system that is dryer, much less nutrient rich and while still dominated by fen vegetation, contains a range of bog adapted species (e.g. *P. mariana, Sphagnum* spp.). Shrub cover increases, on average, as the sites become dryer from Type I (14%) to Type II (20%) to Type III (45%).

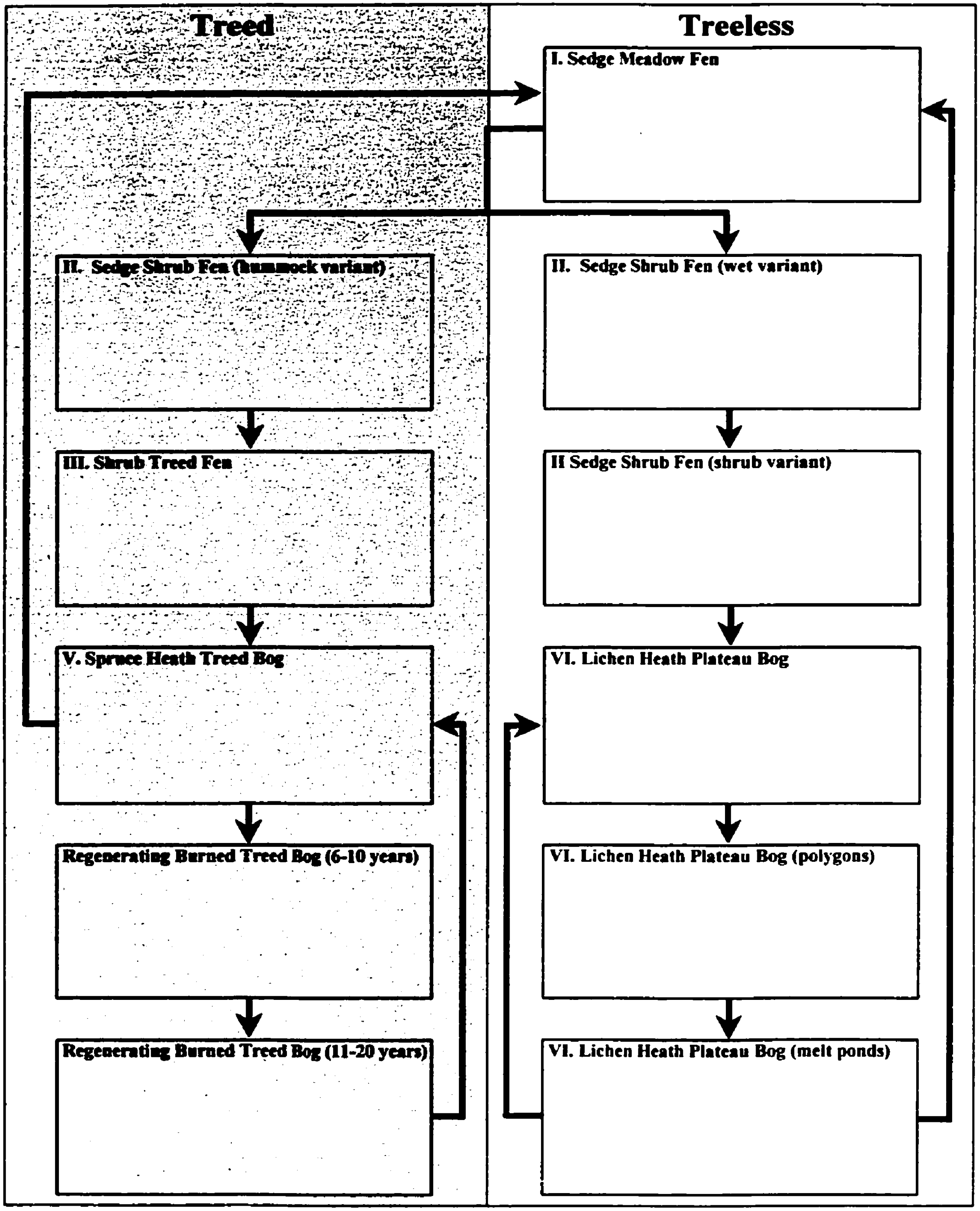


Figure 3.5. A synoptic model of vegetation dynamics for the study area based on interpretation of the CCA analysis and related literature.

Type V (Spruce-Heath Treed Bog) represents the next step in the progression from fen to bog. It is found on sites that are generally further inland than the fen types and that have accumulated enough peat to raise the growing surface above the water table and away from the nutrient supply.

Type VI (Lichen-Heath Plateau Bog) occurs under very similar soil conditions as type V, but this type does not develop tree cover. The treeless type (Type VI) develops from the raised hummocks (Type IV). The absence of trees is a result of climatic factors, including the damaging effects of blowing ice (Klikoff 1965), strong winds (Hadley and Smith 1983) and the drying of plant parts above the snow pack (Sakai 1970).

The overall trend in the vegetation is from a wet fen that dries with a shift from *Scorpidium* in saturated soils to *Drepanocladus* in slightly drier condition as peat continues to accumulate. Additional height is gained through the expansion of water within the peat. Small hummocks eventually coalesce to form larger hummocks. Ridges and strings form where a number of coalescing hummocks occur perpendicular to the slope. Significant peat accumulation occurs once mesophytic mosses such as *Tomenthypnum nitens* and *Sphagnum* spp. colonize the drying peat mounds. *Andromeda polifolia*, *Dryas integrifolia* and *Vaccinium uliginosum* then develop on the hummocks. Once the peat raises the growing zone completely above the water table, the pH becomes highly acidic and nutrients become scarce. A peat plateau is initially less than 30 cm high with a *Sphagnum fuscum* layer and *Picea mariana* tree cover. Over time, peat continues to accumulate and the insulation of peat and *Sphagnum* facilitates ice lens development. As the growing zone is further removed from the water table, lichens such as *Cladina* spp. accumulate over the *S. fuscum* cap. A "mature" peat plateau is up to 2 m above the water table and may be several kilometres in extent. As thickness of the peat layer increases and pH decreases, *Sphagnum warnstorfii* and *S. fuscum* develop, along with *Andromeda polifolia*, *Dryas integrifolia* and *Vaccinium uliginosum*.

DYNAMICS OF UPLAND VEGETATION

While Type IV has similar floristic composition to Types V and VI, it is found exclusively on well-drained mineral sand and gravel soils and so operates under different successional influences. The low, wet conditions on which the organic substrates of the wetland types are generally located, along with the highly oxidizable nature of the peat substrate makes them more subject to changes in their physical properties than most upland mineral soil sites, which are more physically and compositionally stable for long periods.

Mineral soils are initially unvegetated elevated beach and kame mineral deposits that are colonized by *Dryas integrifolia* and *Saxifraga tricuspidata*. As organic soil accumulates, lichens such as *Alectoria ochroleucra* begin to develop. *Picea glauca* then slowly colonizes the site, developing into White Spruce-Heath Treed Upland.

CATASTROPHIC DISTURBANCE (FIRE)

Fire rarely burns fen vegetation types (Types I-III) due to the wetness and open water patches, only occasionally burning the graminoid vegetation. Fire is not a major factor in influencing the dynamics of these types, except through indirect effects such as nutrient cycling in the ecosystem.

The Spruce Heath Treed Upland (Type IV) undergoes a somewhat unique burn and regeneration process due to its mineral, rather than organic, substrate. This type is highly susceptible to fire with its open-canopied *Picea glauca*, erect, woody, shrub and often-dry lichen cover. In the first five years after a burn, it is dominated by fugitive herb species such as *Epilobium angustifolium*, *Corydalis sempervirens* and *Calamogrostis neglecta*, along with low shrubs such as *Ledum groenlandicum* and *Betula glandulosa.* Partially burned standing dead trees, exposed sand and gravel soil are common.

Spruce-Heath Treed Bog (Type V) is highly susceptible to fire. Fire generally kills the surface vegetation, but is not intense enough to destroy the peat to any depth. After a fire, standing dead and partially burned trees are common on exposed peat soil. Vegetation dynamics in burned areas follows a relatively straightforward recovery sequence to approximately the pre-disturbance composition. However, not all species recover at the same rate, so species with wide habitat tolerances and strong regenerative abilities from rhizomes, rhizoids and gemmae tend to dominate early after a fire, such as *Ledum decumbens* and *Rubus chamaemorus* common (Fig. 3.6). *Sphagnum fuscum* is also common (Fig. 3.6) since it reproduces vegetatively (Lane 1977) and so can invade burned areas. Slower growing species such as lichens (e.g. *Cladina mitis*) and trees (e.g. *Picea mariana*) are very rare in the first 10 years, but increase in abundance over time. Fugitive herb species such as *Epilobium angustifolium, Corydalis sempervirens,* and *Calamogrostis neglecta* that occur on mineral soils immediatley following a fire, are rare or absent on organic soils. The lower nutrient levels in the organic types appear to mitigate against this reproductive strategy (Jaseniuk and Johnson 1982). The high level of nutrients required to avoid crowding and ensure reproduction is likely the limiting factor. The vegetation in regenerating areas is typically a dense cover of low and medium shrubs and small *P. mariana* of varying tree height, depending on the age of the burn and soil conditions. Forest fires generally do not initiate widespread thawing of the permafrost (Thie 1974).

Lichen-Heath Plateau Bog (Type VI) does not burn frequently or extensively, despite being relatively dry, due to non-conducive fuel geometry and small microtopographical fire barriers, and the low levels of fuel (Wein 1976). Fire that originates in treed areas may spread out onto the open lichen areas, but rarely travel far (pers. obs., 1999).

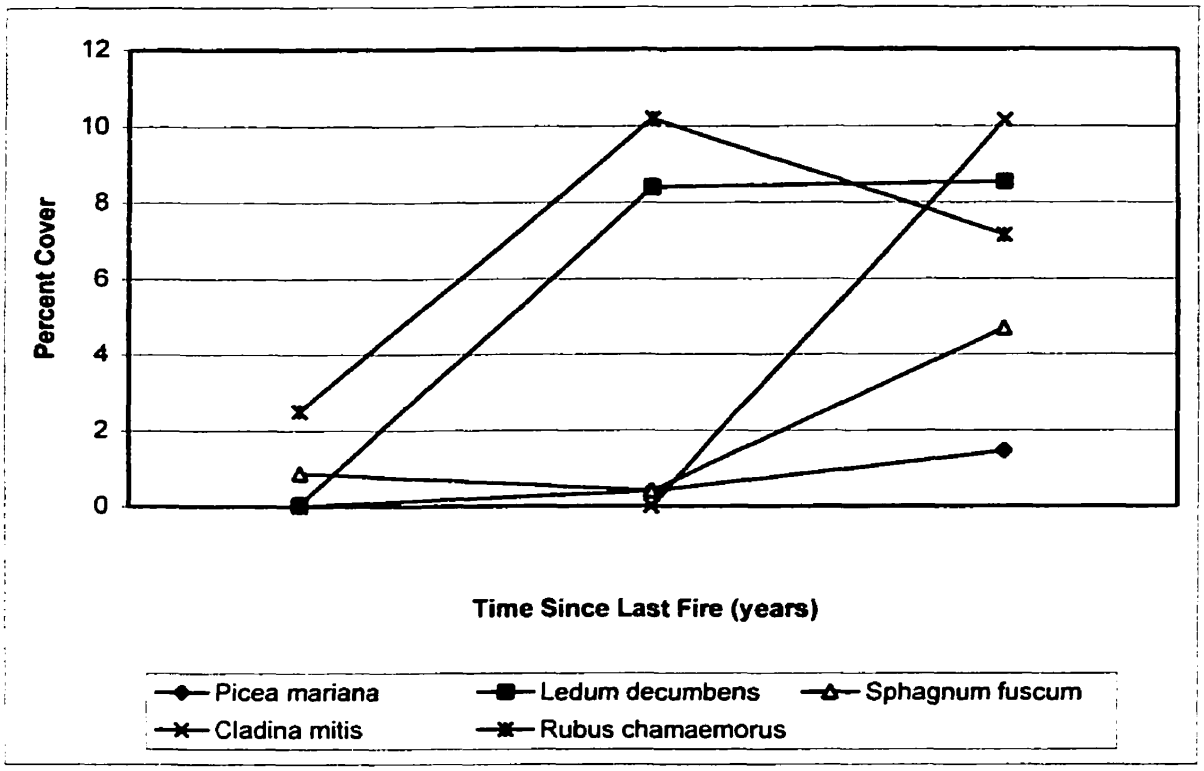


Figure 3.6. Percent cover of selected representative tree (*Picea mariana*), shrub (*Ledum decumbens*), herb (*Rubus chamaemorus*), moss (*Sphagnum fuscum*), and lichen (*Cladina mitis*) at different stages in the regeneration process following fire.

CATASTROPHIC DISTURBANCE (THERMOKARST PROCESSES)

Vegetation Types V and VI have relatively thick peat deposits that reach a maximum of approximately 2m (Dredge 1979) and are generally found on palsas and peat plateaus with well developed permafrost near the ground surface. However, there is a delicate balance between the development and degradation of permafrost in these features. As peat accumulates over time, cracks in the peat surface may form from extensive accumulation of peat or drying of the peat surface, exposing the core to accelerating thawing (Zoltai 1993). Other disturbances such as fire, windthrow, and human activity can also tear open the peat surface or impede soil drainage and initiate thermal degradation (Dredge and Nixon 1979). When the frozen core is thawed, the ground surface subsides, often down to the surrounding fen. This collapse process can occur rapidly, within several years and produce thermokarst depressions (Thie 1974). Dionne (1984) documented a cyclic progression of development and collapse of peat plateaus and palsas.

Type V (Lichen-Heath Plateau Bog) appears to be prone to significant permafrost degradation in response to the formation of ice wedge polygons that can facilitate melting by opening up the peat surface. The overlying peat is often ruptured as it accumulates past a certain threshold thickness (Dredge and Nixon 1979). Melting occurs where ice wedge troughs intersect, with the ground slumping in to form thermokarst ponds. Once the ponds are formed, the water acts as a radiation trap, accelerating the melting process and causing the ponds to grow laterally (Dredge and Nixon 1979). These ponds can then aggregate into larger ponds and lakes due to ice ramming and wave battering. This permafrost aggradation and water body forming process can continue extensively or it can be stopped at any point if equilibrium is reached between the rate of winter freezing and the depth of summer thawing. Thawing may also be halted by altered drainage that channels the water out of the lakes (Dredge and Nixon 1979). Once drained, the pond bottom may accumulate permafrost and terrestrial vegetation can colonize the newly exposed substrate.

Type VI (Spruce-Heath Treed Bog) can undergo significant, localized melting of permafrost to form collapse scars. The collapse scars are most easily recognized by the standing dead trees that were killed by the excessive moisture in the newly created fens. If the collapse scar occurs entirely within the peat plateau, the frozen peat around the collapse may prevent water movement, creating more oligotrophic conditions. Under these circumstances, wet bog vegetation would dominate, including *Sphagnum* spp., *Drosera* spp. that are typical of nutrient poor conditions. Tree cover may provide some cooling influence on the soil (Rouse 1984), lessening the frequency and extent of melting.

DISCUSSION

On organic soils, saturated open fens slowly accumulate peat over time that causes a concomitant development of permafrost due to the insulating qualities of the peat (Jasieniuk and Johnson 1982). The combination of peat accumulation and ice lense development in the soil results in the development of hummocks that are generally elevated 5-20 cm above the water table. These slightly elevated hummocks are drier and more acidic creating conditions more suitable for bog vegetation. The bog vegetation continues the process of peat accumulation, further raising the hummocks above the water table, while the entire plot continues to accumulate peat and become drier. In more sheltered areas this generally means an increase in tree cover as well, though exposed sites may remain treeless due to the damaging effects of windblown ice and snow. In response to the changing wetness and nutrient availability, plant species composition changes from rich fen species such as *Carex aqu*atilis and *Scorpidium spp.* to bog species such as *Ledum groenlandicum* and *Cladina stellaris*. Peat is thickest (approximately 4 m) in areas that were elevated above Hudson Bay more than 6000 years ago and thinnest in recently emerged areas near the coast (Dredge and Nixon 1992).

There is a high degree of similarity between species ecological groups and vegetation types within the study area, suggesting that these groups are the result of unique arrangement of environmental factors associated with each type. The vegetation in the study area shows some similarity to that described in other studies in the Hudson Bay Lowlands (Ritchie 1956, 1957, 1959, 1960a, 1960b, 1960c, 1962, Sims *et al.* 1982, and

Campbell 1995). The general progression of fen to bog found in my study is similar to that observed in Hudson Bay Lowlands (by Ritchie 1957, 1960, Kershaw 1974) and other northern regions in general (Zoltai et al. 1988). However, the quantitative approach that I have employed presents more clearly defined plant communities based on objective analysis. Predictions of vegetation dynamics are based on examination of complex ecological gradients.

Some patterns in the dynamics of vegetation groups have emerged from the analysis of plots, species and environmental factors. Peat accumulation related changes in wetness and nutrient availability appear to drive the development of multidirectional, irregular cycles of vegetation dynamics. While the processes driving these dynamics may be reasonably well understood, they are unpredictable at the site level. Recent stratigraphic studies have indicated that changes in peatland vegetation is far from deterministic, but rather are largely influenced by variable influences including water level fluctuation, drainage, nutrient inputs and peat accumulation rates (Sjors1963, Ingram 1967, Schwintzer 1978).

Tree ground cover and tree height within plots is generally variable with respect to their distance from the Hudson Bay coast. However, tree cover and tree height are strongly related. Subarctic conifers generally control snow distribution (Marr 1977, Daly 1984). In conifer stands with little snow accumulation, stem anomalies such as asymmetric stems and reiterated stem leaders are common above the snow line (Payette et al. 1985) indicating that microclimate is exerting a strong limiting force on tree growth and survival (Payette et al. 1986, Arsenault and Payette 1992). Sites with higher tree densities accumulate more snow in winter and collectively produce microclimates less hostile to tree growth. The highly variable snow cover that exists in the study area (Campbell 1995) may, to some degree, explain the variation in tree cover observed. The harsh climatic conditions may also influence conifer seed quality and growth, since seeds from stunted individuals are generally of low quality (Black and Bliss 1990). The relatively high tree cover in the southwest corner of the study area may possibly be explained by the diminished severity of the climate.

Lichen-Heath Plateau Bog (Type V) is elevated and dry, but is generally treeless and can be found up to 80 km from Hudson Bay. Tree establishment may be inhibited on these sites by a continuous lichen mat that inhibits seedling establishment (Cowles 1982, Morneau and Payette 1989). Fire also plays a role in the long-term dynamics that influence the degree of tree cover. Treed areas that burn, may regenerate as treeless lichen dominated plateaus, likely due to climatic fluctuations (Larsen 1965, Payette and Gagnon 1985, Arsenault and Payette 1992).

As peat accumulation occurs on the older sites and more bog vegetation types develop, these areas may become more fire susceptible due to the increasing fuel and generally drier conditions. As the relative cover of bog vegetation increases, movement of fire across the landscape may be facilitated by an increasing connectivity of the dry bog vegetation types. As a result, the potential for fire movement across the peatlands may generally increase with distance from the Hudson Bay coast. At the site level, fire is initiated by stochastic lightning strikes and its movement is determined by complex topographical, wetness, fuel and climate variables and remains unpredictable (Payette et al. 1989, Timoney and Wein 1991).

Since permafrost is more developed and influential on bog vegetation types, catastrophic disturbance from thermokarst processes appears to only occur in vegetation types V and VI. The processes that influence the structural changes in the permafrost are well understood, but remain highly unpredictable due to the complex variety of factors that drive them, including disturbance, vegetation cover, peat accumulation, drainage, climate, and vegetation cover.

The results of the application of indicator species values (Persson 1981) supports their utility in assessing the nutrient status of vegetation plots in the forest-tundra. This information provides an indication of overall nutrient status across the entire growing season rather than individual field measurements at a single point in time that do not reflect the dynamic nature of nutrients in the ecosystem. The successful application of indicator species in boreal and sub-arctic environments (e.g. Sjörs 1963, Wells 1981, Sims et al. 1982, Vitt and Slack 1984, Kenkel 1987) further supports their utility in exploratory data analysis for examining vegetation-environmental relationships (Persson 1981). Further research to determine more accurate measures of the nutrient status for individual species would increase the resolution of this approach.

CONCLUSION

The classification reported in this chapter is hierarchical and is based on a rigorously defined quantitative methodology. The existing vegetation in the study area is primarily the result of a concurrent spatio-temporal gradient inland from the coast, reflecting ongoing isostatic emergence of the area from Hudson Bay along with climatic influences that are strongly influenced by air masses in Hudson Bay. The three major vegetation types are fen, bog and upland, with the fen and bog types being associated with organic soils and the upland type being associated with mineral soils.

The results of this chapter achieve the first two objectives of this thesis to delineate the major vegetation communities of the region using quantitative methods analyzing species cover, plant life form and environmental data; and to summarize the floristic composition, community structure and vegetation-environment relationships associated with each community and identify vegetation dynamics based on the relationships observed. In the study area, I suggest that the predominant environmental gradient acting on the vegetation is moisture-nutrients which is ultimately influenced by peat thickness, which tends to increase with increasing distance from the Hudson Bay coast. The successional process may also be constrained to some degree by climatic influences on tree growth.

At the landscape scale, general trends in the vegetation may be predictable. Distance from the Hudson Bay coast is a general indicator of the time since the land emerged from Hudson Bay which is an estimate of time available for peat accumulation. However, irregularity in topography and rate of isostatic uplift create considerable fine-scale anomalies in emergence of the land from the water. The ubiquitous distribution of millions of lakes and ponds throughout the region further confounds this relationship. Despite this complexity, I predict that a gradient exists at the landscape scale with an increasing dominance of bog vegetation types and decreasing fen types with increasing distance from the Hudson Bay Coast. I also predict that fire extent in the peatlands will generally increase with distance from the Hudson Bay coast due to increased dominance and connectivity of bog vegetation types.

MANAGEMENT IMPLICATIONS

Vegetation is highly dynamic in the Hudson Bay Lowlands as a result of ongoing isostatic uplift and peat accumulation in conjunction with climatic variability, permafrost aggradation and degradation, fire and wildlife grazing. The complex interrelationships between these processes make prediction of future changes difficult at fine scales. At the landscape scale vegetation is expected to continue to become dryer and more nutrient impoverished with increasing distance from Hudson Bay and so more fire susceptible. More work is needed to sample the vegetation throughout the study area in order to further identify and describe variants and discover new communities that may exist.

CHAPTER 4: A MULTIVARIATE APPROACH TO VEGETATION CLASS IDENTIFICATION

"It is enough to work on the assumption that all of the details matter in the end, in some unknown but vital way."

-E.O. Wilson, Biophila.

PREAMBLE

This chapter presents a method developed that uses ground information and spectral data from multiple Landsat-5 TM bands to develop an iterative strategy for the classification of highly complex sub-arctic vegetation. The vegetation communities identified in Chapter 3 are translated into meaningful vegetation classes for satellite image classification. Multivariate analytical methods were developed to examine the spectral reflectance signatures of the different vegetation types in order to assess their relative separability in the satellite imagery.

INTRODUCTION

Remotely sensed data from satellites have been used for thematic land cover mapping at a wide range of spatial scales and for numerous applications (Cihlar 2000). Such data have most commonly been interpreted as land cover classes that reflect the combined characteristics of numerous landscape components. Continental and global scale projects have typically produced generalized landscape level maps that identify broadly-defined land cover classes (e.g. 'agricultural', 'grassland', 'boreal forest' or 'barren land') that are separable without having to resort to detailed ground-cover information. However, such broadly defined map classes are often of limited utility in addressing landscape-scale ecological questions. In response to this limitation, finer-scale maps have been produced in which land cover is interpreted in terms of identifiable ground cover components (Roughgarden et al. 1991). For example, vegetation maps interpret the classes as plant communities based on floristic composition and structure. Many vegetation mapping projects have used a standard land cover mapping approach, applying a single supervised or unsupervised classification directly to the entire image and subsequently performing an accuracy assessment. In many cases, such maps had low accuracy and therefore could not be used for their intended purpose (Townshend 1992).

The reasons for the unfulfilled potential of satellite remote sensing in vegetation mapping are numerous. A primary limitation of satellite sensor data is that they provide only a measure of the amount and type of reflected and emitted electromagnetic energy. Spectral information is affected by a number of factors, including soil moisture, substrate, structural configuration, topography and atmospheric effects, as well as the amount, vigour, productivity, structure, and floristic composition of vegetation (Richardson and Wiegland 1977, Vogelmann and Moss 1993). The complexity of ecosystem structure results in a continuum of vegetation cover and a corresponding continuum in multi-spectral space (Richards 1993). Subdivision of this multi-spectral continuum into meaningful vegetation classes is a major challenge that requires careful consideration.

Classification is essentially a modelling procedure (*sensu* Jeffers 1982) whereby a complex continuum of land cover information is translated into ecologically meaningful classes. Vegetation mapping is an important practical exercise, since a level of generalization is required if ecosystem complexity is to be represented in a meaningful and readily interpretable way. In the initial verification of a classification model, reflected spectral radiation data are related to ground cover information. Specifically, field-collected calibration data are examined to establish the feasibility of using satellite imagery to separate known ground cover classes (Cihlar 2000). The developed classification paradigm is then applied to the entire satellite image to produce a vegetation map. This is followed by model validation, in which map accuracy is assessed using an independently derived ground-survey data set. All phases of model development, including verification and validation, are highly dependent on the quality and quantity of ground-survey data. Detailed site-level information on vegetation composition and structure is required to objectively define meaningful ground cover classes. Ground survey data also provide essential insights into factors determining spectral reflectance, and are required for classification accuracy assessment.

Classification is necessarily a subjective process that involves carefully considered sequential decisions. The quality and utility of a vegetation classification is determined by the analyst's skill, judgement and familiarity with the study area (Foody 1999). Scale of application is an essential consideration, since the concept of spatial heterogeneity is a scale-dependent descriptor of the inter-relatedness of land cover components across the landscape (O'Neill *et al.* 1988). Each mapping project has unique challenges that will affect map accuracy. Decisions must be made regarding the sampling design used to collect ground cover data, including the number and distribution of sample sites. The ground cover data must then be analyzed and the sites allocated to mutually exclusive vegetation classes based on floristic composition (Legendre and Legendre 1998). Following this, the defined vegetation classes must be translated into spectrally separable classes. This is a critical step, since there is no guarantee that floristically distinct land cover classes will have distinct spectral signatures. Indeed, an accurate map requires that each land cover class has unique and characteristic spectral properties over at least part of the measured electromagnetic spectrum. Classes lacking unique spectral signatures must either be amalgamated (e.g. Matthews 1991) or separated using ancillary data (e.g. Nilsen *et al.* 1999).

A number of studies have examined the relationship between spectral reflectance and broad-scale land cover classes (e.g. Perry and Lautenschlager 1984, Hope et al. 1993, Korobov and Railyan 1993, Cihlar 2000). However, this relationship has not been as well studied for finer-scale vegetation classes, particularly those based on floristic composition and structure. Determining the relationship between spectral reflectance and vegetation composition is particularly important in northern ecoregions, where vegetation mapping presents unique logistic and theoretical challenges. Much of the arctic and sub-arctic is inaccessible, making the collection of ground-cover data difficult and expensive. Arctic ecosystems are spatially and floristically heterogeneous and vegetation may be sparse or non-existent in many areas (Ferguson 1991, Nilsen et al. 1999). Arctic vegetation is characterized by low shrubs, graminoids, bryophytes and lichens (Bliss et al. 1973). Lichens, which have unique spectral reflectance properties (Petzold and Goward 1988), often form the dominant ground cover. A number of approaches have been used to classify and map northern vegetation in the Hudson Bay Lowlands (Ritchie 1962, Pala and Boissoneau 1982) and other northern regions (Thompson et al. 1980, Horn 1981, Matthews 1983, Shasby and Carneggie 1986, Ferguson 1991, Matthews 1991, Pearce 1991, Morrison 1997, Nilsen et al. 1999), with varying degrees of success.

Bivariate scatterplots (comparing two-band combinations of spectral data) are commonly used to examine class separability, and to determine the ideal band combinations for classification purposes (e.g. Ferguson 1991, Muller et al. 1999). While this approach is useful for comparing band pairs, vegetation classification generally involves the simultaneous use of three or more spectral bands. Multivariate analysis is an optimal strategy for simultaneously comparing the spectral reflectances of numerous satellite bands (Richards 1993). In multivariate analysis, interrelationships among the original variables (e.g. multiple spectral bands) are summarized as a reduced set of derived variables (Legendre and Legendre 1998). Multivariate analysis thus provides a powerful strategy for data verification, such as comparing the separability of vegetation classes and examining the utility of different band combinations in characterizing classes.

The objective of this chapter is to demonstrate the utility of multivariate methods in developing and verifying a vegetation mapping approach based on groundcover data and remotely sensed spectral information. Emphasis is placed on the examination of vegetation class separability in order to maximize accuracy of the final map product. The multivariate techniques considered include cluster analysis, principal component analysis (PCA), correspondence analysis (CA), multiple discriminant analysis (MDA), and redundancy analysis (RDA).

LANDSAT TM SATELLITE IMAGERY

LANDSAT TM is part of an American series of Earth observation satellite that have been continuously archiving data since 1972. LANDSAT 5 was launched in 1984, producing images of the earth's surface that are 185 x 170 km in size with a nominal ground resolution of 30 m (Lillesand and Kiefer 1994). This satellite measures reflected electromagnetic energy in seven wavelengths (Quattrochi and Pelletier 1990). Band 1 (0.45-0.52 um; blue) is designated for water body penetration, soil and vegetation discrimination and forest type mapping (Lillesand and Kiefer 1994). It is useful for discriminating between coniferous and deciduous vegetation. Band 2 (0.52-0.60 um; green) operates in the chlorophyll absorption region and is best for detecting roads, bare soils and vegetation types. Band 3 (0.63-0.69 um; red) is designed for chlorophyll absorption detection important for vegetation discrimination. Band 4 (0.76-0.90 um; near-infrared) is used to estimate biomass and can determine vegetation types. It is also useful for delineating water bodies and for soil moisture discrimination. Band 5 (1.55-1.75 um; mid-infrared) is generally considered to be the best single band overall, providing a good contrast between different types of vegetation (Lillesand and Kiefer 1994). It also has excellent atmospheric and haze penetration and may be used to differentiate between snow and clouds. Band 6 (10.4-12.5 um; thermal infrared) responds to thermal radiation emitted by the target, providing the ability to measure plant heat stress, biomass burning, and soil moisture. Band 7 (2.08-2.35 um; mid-infrared) is useful for interpreting vegetation cover and soil moisture, as well as discriminating mineral and rock types.

The reflectance of light from vegetation on the ground is determined by a combination of factors including morphology, leaf geometry, plant physiology, plant chemistry, soil type, solar angle, climatic conditions (Barrett and Curtis 1992), water levels and plant phenology (Lillesand and Kiefer 1994). Plant structure, soil background and the surface condition of the plant's reproductive and vegetative parts have a particularly significant influence on spectral reflectance.

METHODS

Analytical Approach

The collection and analysis of vegetation cover and spectral reflectance data, and classification of the satellite image, followed an iterative approach that included a number of decision steps (Fig. 4.1). The methodology focuses on making careful decisions based on multivariate data analysis, considerations of map utility, and the inherent limitations of satellite imagery. Adaptive learning is an essential component of the procedure, with each decision having an important impact on subsequent decisions in the mapping process.

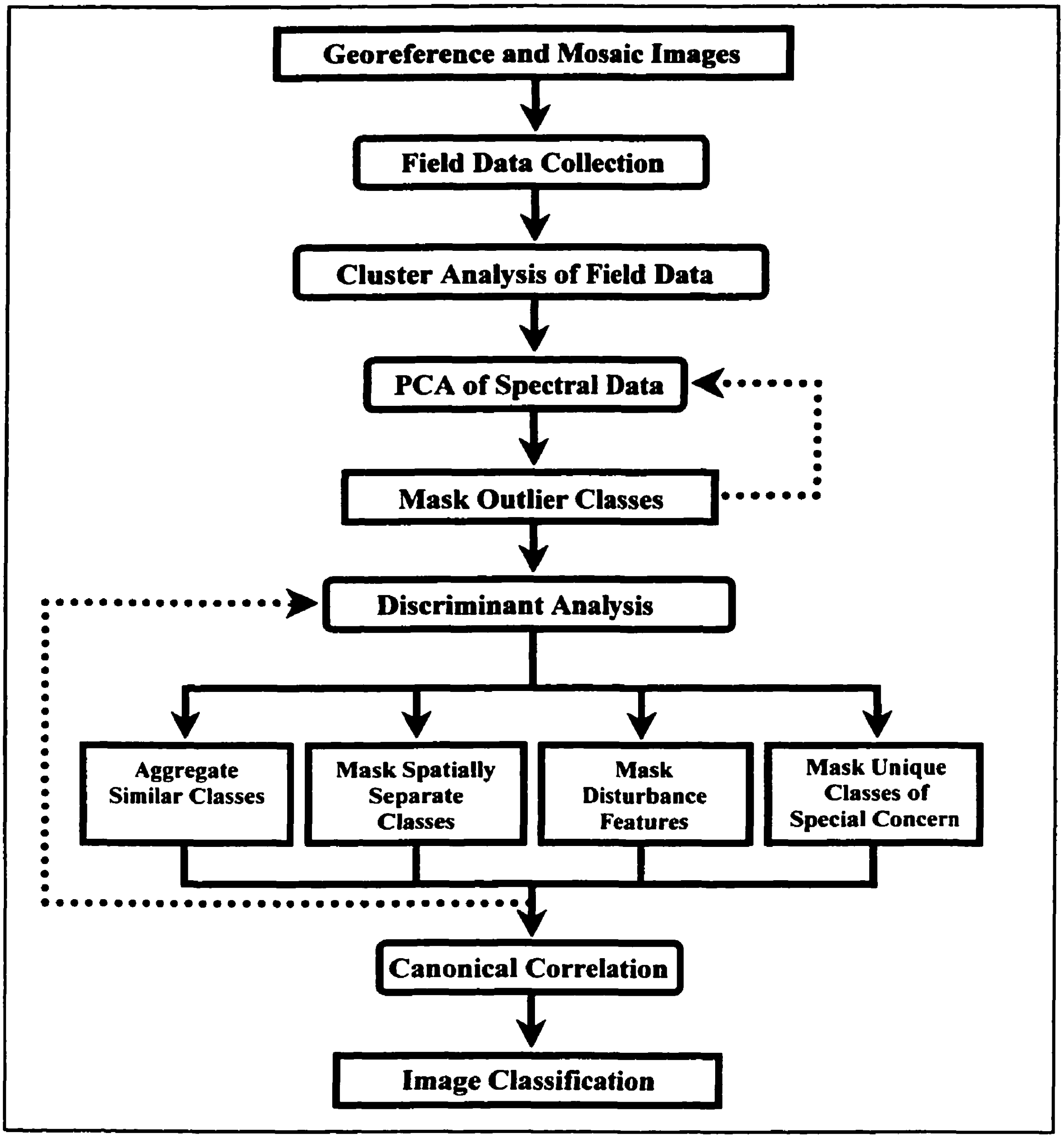


Figure 4.1. Flow model diagram of a generalized approach to satellite image classification using multivariate data analysis. Square boxes are steps involving satellite image manipulation and classification, while rounded boxes are data analysis steps. Dashed arrows depict iterative steps.

Data Collection

Two full Landsat TM images covering the study area, taken on 27 July 1996 were acquired. These were the most recent cloud-free TM scenes available for the study area during the peak of the growing season (early July to mid-August). All non-thermal bands were included. The thermal band (TM6) was not used due to its low spatial resolution (120 m). In order to remove geometric distortion in the Landsat TM imagery, each scene was geocoded to a Universal Transverse Mercator (UTM) grid referenced to the North American Datum of 1927 (NAD 27) zone 15 projection. A total of 15 ground control points (GCP's) were collected for each image (GCPWORKS, PCI Geomatics 1998) from 1:50,000 NAD 27 National Topographic System (NTS) paper maps that are the most accurate spatial data currently available for the region. GCP selection was based on the ability to find a corresponding pixel within the image and the maps accurately. Three GCP's were selected that were common to both images in order to ensure that when mosaicked together, the fit would be precise. Image to image GCP collection, first order transformation and nearest neighbour re-sampling of uncorrected imagery was performed. Image identifiable points were selected and matched to map co-ordinates. The green, red and near-infrared bands of the Landsat TM image displayed as a false composite image were used for point identification because it produced maximum contrast between land, vegetation and water features. The resulting Root Mean Square (RMS) error for the imagery was 0.96 pixel (28.8m), 0.63 pixel (18.9m) (x, y) for the north image and 0.57 pixel (17.1m), 0.87 pixel (26.1m) (x, y) for the south image. Once the two scenes were georeferenced, they were mosaicked together to produce a single composite image. This geocoded image was then subset to size it to the bounds of the study area. A mask was digitized over the ice in Hudson Bay and it was cut out of the image to remove its confusing effect on the classification process.

Field data were collected throughout the study area from June to September of 1998-2000. Four hundred sites that were visually homogenous at a minimum scale of 50 x 50 m were sampled. At the centre of each site, percent cover values for each plant species and unvegetated substrate were estimated within a 10 x 10 m plot. Each plot was then sub-sampled using two randomly placed 2 x 2 m plots. An additional 200 sites were independently sampled for classification accuracy assessment. The location of each site was used to obtain Landsat-5 TM spectral reflectance values (TM bands 1-5 and 7) for all sites, which allowed direct comparison to the ground cover information to the thematic data provided by the satellite image.

Initial Class Allocation

Cluster analysis (Ward's method, Legendre and Legendre 1998), based on a chord distance matrix of the ground-cover data, was performed to delineate major vegetation classes (HIERCLUS, Podani 1994) (Chapter 3). This procedure provided an objective analysis of natural vegetation groupings, independent of the spectral data. The six major vegetation types described (Chapter 3) were then carefully examined to determine their utility in landscape level vegetation mapping. The subtypes of these six classes were also considered as potential vegetation classes. Decisions regarding the delineation of the preliminary set of classes to be used were based on the objective of mapping the major vegetation types in the region that are important in ecological research and management. As a result, some physiognomic or productivity variants of the major vegetation types were selected as distinct classes if they represented common and important vegetation units. Type I (Sedge Meadow Fen) was recognized as being highly variable in terms of overall productivity with a rich variant (mean Normalized Difference Vegetation Index (NDVI) = 0.61) that occurred mostly inland and a relatively poor variant (mean NDVI = 0.33), though species composition was similar overall. Type II (Sedge Shrub Fen) is recognized as a single vegetation class. Type III (Shrub Treed Fen) includes three floristic and physiognomically distinct variants that are defined as unique vegetation classes: Sphagnum Larch Fen (Larch trees with *Sphagnum* spp. ground cover), Sedge-Larch Treed Fen (Larch trees with *Carex* spp. ground cover), and Willow-Birch Shrub Fen (tall shrub dominant).

Type IV (White Spruce-Heath Treed Upland) and Type V (Black Spruce-Heath Treed Bog) are both dominated by spruce tree cover with a heath shrub, moss and lichen understory. Cluster analysis (Chapter 3) indicates that they have very similar floristic composition. However, Type IV is a relatively rare vegetation class that is limited primarily to inland raised beaches and kames. As a result of the anticipated spectral overlap between Type IV and Type V, it was included with Type V as a single Lichen-Spruce Bog class. Type VI (Lichen Heath Plateau Bog) is recognized as a vegetation class, Lichen Heath Plateau Bog. A variant of type VI is created through the mixing of the raised lichen-dominated plateaux with open water in associated melt ponds, which intersperse at a wide range of scales (Lichen Melt Pond Bog). Two additional vegetation classes were identified, based on analysis of all sites, that identified Graminoid-Willow Salt Marsh and Dryas-Heath Upland as unique communities.

Two disturbance classes were recognized: recent burns, occurring within the last five years (1991-1996) and regenerating burns (occurring 6-80 years ago). All burns identified in the 1996 image were visually compared with a 1991 Landsat TM image and each burn was classified as occurring pre-1991 (regenerating burn) or post-1991 (recent burn). Three unvegetated classes (<20% vegetation cover) were also recognized based on prominent physical characteristics: these were water, unvegetated ridges, and unvegetated shorelines. In total, seventeen classes were recognized (Table 4.1).

Digital numbers (DN) of the six Landsat TM bands were obtained for each sample site to form a verification data set. These data were refined to exclude outliers that clearly fell outside of the expected range of homogenous sites. This step was performed to ensure that the training data set contained no mixed pixels (Arai 1992). For illustration purposes, a representative sample of 15 pixels from each of the 17 classes was chosen to represent inter-class spectral variability.

Identification of Outlier Classes

The spectral data (17 classes, each with 15 sites, and six Landsat bands) were analyzed using principal component analysis, based on a correlation matrix (ORDIN, Podani 1994). PCA is a multivariate ordination method that considers inter-correlations of the spectral data to produce an optimized and simplified representation of the underlying data structure. PCA is particularly well suited to the identification of outlier groups in multidimensional spectral band space. In this analysis, sites are represented as scores on the two major component axes, while the six spectral bands are shown as ordination biplot scores (Gabriel 1971). Classes appearing as prominent outliers in the two-dimensional ordination space were identified and removed. Classes were considered 'outliers' when the cluster of 15 sites defining the class was clearly separated from the remaining sites. After one or more outlier classes were removed, PCA was run again on the reduced data set. This process was repeated until no strong outlier groups remained.

Table 4.1. Land cover and vegetation classes of the study area with the average digital number from Landsat TM imagery (s.d.); *n* = 15 for each class.

	Landsat TM Digital Number (DN)					
Class	**TM1**	**TM2**	**TM3**	**TM4**	**TM5**	**TM7**
1. Sphagnum Larch Fen	55 (1.4)	24 (0.7)	22 (1.1)	75 (2.2)	49 (1.5)	16 (1.4)
2. Sedge Rich Fen	55 (1.4)	23 (1.0)	20 (1.1)	81 (3.6)	59 (2.2)	18 (1.3)
3. Willow Birch Shrub	53 (0.7)	21 (0.5)	18 (0.6)	64 (2.4)	58 (3.0)	19 (1.8)
4. Sedge Larch Fen	55 (2.1)	21 (0.8)	20 (1.3)	46 (2.0)	50 (2.8)	18 (1.2)
5. Sphagnum Spruce Bog	52 (1.3)	22 (0.5)	18 (0.5)	47 (2.1)	39 (1.7)	13 (0.7)
6. Graminoid Willow Salt Marsh	65 (4.6)	31 (3.9)	34 (7.2)	64 (7.8)	70 (6.3)	25 (2.6)
7. Willow Sedge Poor Fen	57 (2.1)	25 (1.6)	24 (1.6)	62 (4.8)	60 (9.5)	21 (3.2)
8. Lichen Spruce Bog	63 (2.7)	28 (1.3)	29 (2.2)	63 (4.6)	68 (3.5)	25 (2.0)
9. Sedge Bulrush Poor Fen	59 (1.9)	24 (1.1)	26 (1.1)	51 (2.1)	69 (3.2)	26 (1.6)
10. Lichen Melt Pond Bog	61 (2.5)	25 (0.9)	27 (1.8)	45 (3.6)	72 (4.4)	29 (2.5)
11. Lichen Peat Plateau Bog	73 (1.3)	33 (0.9)	39 (1.4)	59 (1.6)	90 (1.3)	35 (1.1)
12. *Dryas* Heath Upland	70 (10.2)	31 (6.9)	35 (9.3)	55 (4.9)	99 (10.3)	42 (8.3)
13. Regenerating Burn	56 (1.8)	23 (0.9)	23 (1.8)	42 (3.5)	70 (1.7)	30 (2.4)
14. Recent Burn	53 (0.7)	20 (0.5)	21 (1.0)	24 (1.7)	38 (1.9)	24 (1.9)
15. Unvegetated Ridge	122 (12.6)	68 (8.5)	89 (11.8)	86 (11.0)	150 (19.5)	83 (9.9)
16. Unvegetated Shoreline	87 (6.1)	45 (4.8)	56 (8.4)	53 (6.7)	54 (4.7)	23 (2.6)
17. Water	48 (2.8)	17 (4.1)	14 (3.1)	7 (0.7)	4 (1.6)	3 (1.1)

Based on results of the iterative PCA, three major divisions of the data were identified: vegetated, unvegetated, and water. An unsupervised 203-class isodata classification was run on the entire image mosaic (IMAGEWORKS, PCI Geomatics 1998), and the resulting classes assigned to one of these three types. From the unsupervised classification, the unvegetated group was separated into either the unvegetated ridge class (high reflectance in TM bands) or unvegetated shoreline class (low reflectance in TM bands). The water, unvegetated ridge and unvegetated shoreline classes were then converted to individual bitmap masks in order to remove them from further image classification.

The boundaries of known recent burns were digitized based on visual interpretation to create a bitmap mask over the recently disturbed areas (IMAGEWORKS, PCI Geomatics 1998). An unsupervised 28-class isodata classification was then run on the image, and all classes within the spectral range of recent burns were aggregated to form a single recent burn class. This area was then converted into a bitmap mask and removed from further classification.

Discriminating Vegetation Classes

Once the main outlier groups were removed from the spectral data set, the separability of individual vegetation classes was assessed using multiple discriminant analysis (MDA) based on spectral bands 3,4,5 and 7 (ORDIN, Podani 1994). MDA maximizes the ratio of the between-to within-groups variance, and graphically displays interclass variation in a low-dimensional ordination space (Legendre and Legendre 1998). MDA thus provides valuable insight into the relative separability of vegetation classes. In this analysis, the vegetation classes are represented as 95% confidence ellipsoids, while the spectral bands are shown as biplot scores (Gabriel 1971).

MDA was used to determine whether the floristically-based vegetation classes overlapped in spectral band space. Overlapping classes were then carefully examined to determine the nature of their spectral similarity. For each class, a decision was made to do one of the following: (1) Retain the class, while acknowledging that overlap with another class will compromise the accuracy of the final classification; (2) Aggregate two overlapping classes, if it can be established that they are sufficiently similar floristically and if doing so does not compromise the utility of the final map; (3) Mask a class, if it is a disturbance feature that crosses over vegetation communities and is spatially distinct; (4) Mask a class, if it is floristically distinct and can be spatially separated from its overlapping class; (5) Mask a class, if it contains features that are considered unique and important to the final map and might be lost during classification and filtering.

Verifying the Model

Once overlapping classes and strong outliers are removed, more subtle relationships between spectral reflectance and vegetation cover on the ground can be examined. Canonical analysis is the appropriate model to examine the correlation between two multivariate data sets (Gittins 1985). I used a canonical model known as redundancy analysis (RDA) to examine the relationship between spectral reflectance (TM bands 3,4,5 and 7) and vegetation ground cover. RDA is an extension of multiple regression for modelling multivariate response data (Legendre and Legendre 1998). The method constrains the vegetation data such that ordination vectors are linear combinations of spectral reflectance values. Floristic data was obtained from 10 sites in each of the 9 vegetation classes. A total of 55 plant species and 3 unvegetated variables (water, organic soil, and plant litter) were included. These data were first processed using correspondence analysis (CA) ordination (ORDIN, Podani 1994). This step was necessary to linearize the data and to reduce its dimensionality (Green 1993). RDA was then performed on the 90 sites, using the scores on the first four CA ordination axes as the response variables and the four spectral reflectance bands as the explanatory variables.

RESULTS

Principal Component Analysis

PCA ordination of all 17 classes (Table 4.1) indicated two strong outliers, water and unvegetated ridge classes (Fig. 4.2a). The water class is characterized by low reflectance in all bands, while spectral band reflectances for the unvegetated ridge class are uniformly high. A residual PCA (water and unvegetated ridge classes removed) revealed an additional outlier class, unvegetated shoreline (Fig. 4.2b). This class is characterized by very high reflectance in the visible spectrum (bands 1-3). Following removal of the unvegetated shoreline class, the recent burn class proved to be an outlier with low reflectance in all bands (Fig. 4.2c). PCA of the remaining 13 classes suggested no additional strong outliers (Fig. 4.2d).

Each step in the iterative process summarized above revealed various features of the spectral data. As outlier unvegetated groups were removed, band 4 (near infrared) became increasingly important in discerning the vegetated classes. Bands 1-3 (visible) were proximate in all cases, indicating a high degree of multicollinearity. Bands 5 and 7 are also correlated with the visible bands, but band 4 contains unique information not carried by the other bands.

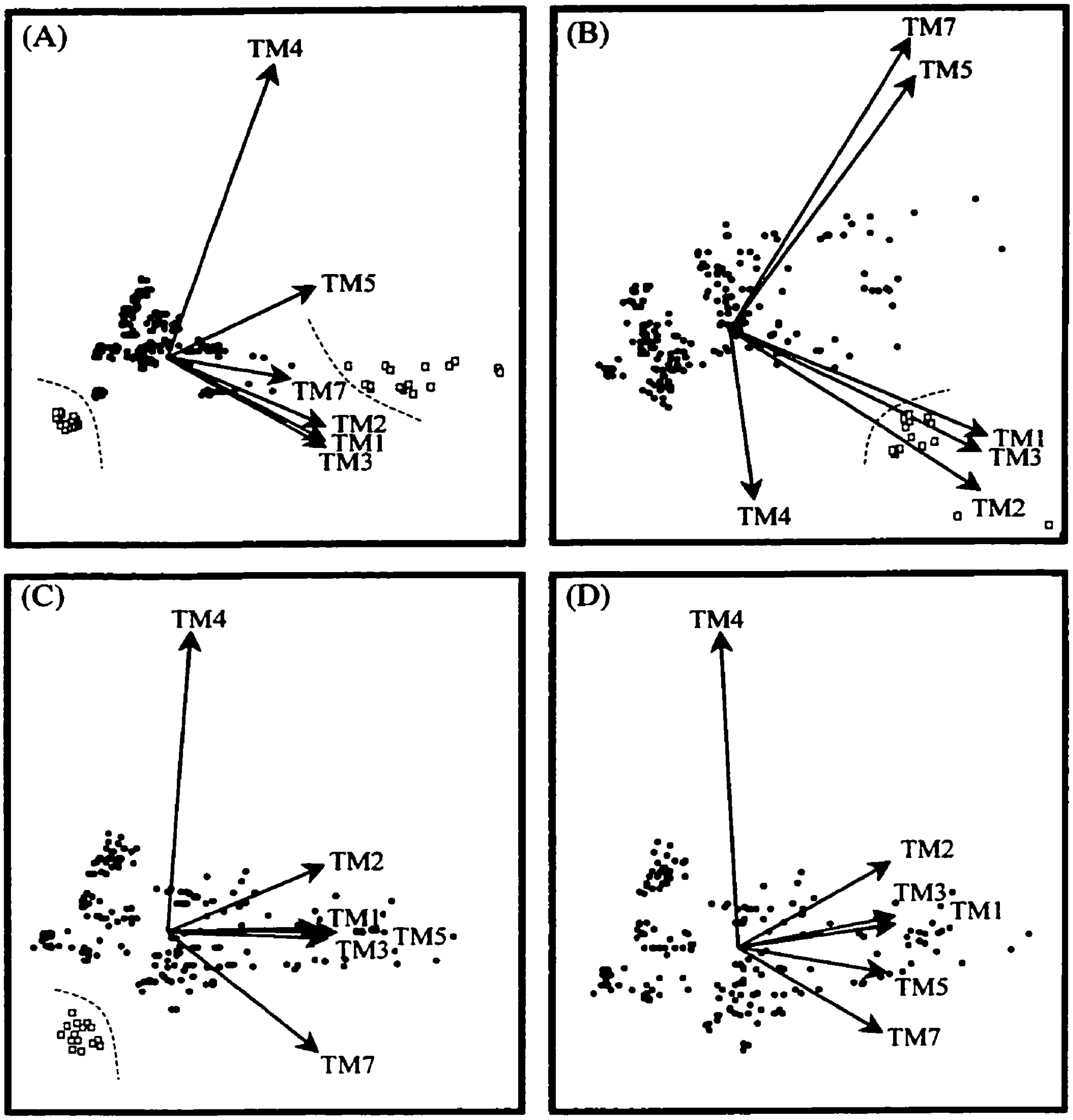


Figure 4.2. Principal component analysis of spectral reflectance variables, each point representing data for a single site where field data and corresponding digital numbers of Landsat TM bands were collected. Sites are identified as outliers (open boxes) if all 15 sites in the class are strongly separated from the other sites in the data set (•). Outlier classes are indicated by a dashed line. Biplots of variables are shown as arrows from the ordination centroid. **A.** PCA of 17 classes, outlier class on the left of the diagram is water, outlier class on the right is unvegetated ridge (axis I = 81.05%, axis II = 12.145). **B.** PCA of 15 classes (water and unvegetated ridge removed), outlier class is unvegetated shoreline (axis I = 59.02%, axis II = 21.83%). **C.** PCA of 14 classes; (unvegetated shoreline removed), outlier class is recent burn (axis I = 72.06%, axis II = 19.29%). **D.** PCA of 13 classes (recent burn removed), no outlier classes are present (axis I = 72.81, axis II = 18.59%).

Multiple Discriminant Analysis

An MDA of the 13 classes revealed varying degrees of interclass separation (Fig. 4.3a). Four of the classes were subsequently masked or combined with other classes:

1. Classes 3 (Willow Birch Shrub Fen) and 7 (Willow Sedge Poor Fen) show considerable overlap in spectral space. As they cannot be separated spectrally and are floristically similar, they were combined to form a single class.

2. Classes 6 (Graminoid Willow Salt Marsh) and 8 (Lichen Spruce Bog) also show considerable overlap in spectral space, but they are floristically very different and are spatially distinct (salt marsh occurs exclusively along the coast, while Lichen Spruce Bog is found farther inland). To distinguish the salt marsh class on the vegetation map, a 15 km wide strip along the shore of Hudson Bay was digitized to produce a bitmap mask. Unvegetated areas of the mask were subtracted and an unsupervised 28-class isodata classification was run to spectrally separate salt marsh from other vegetation classes.

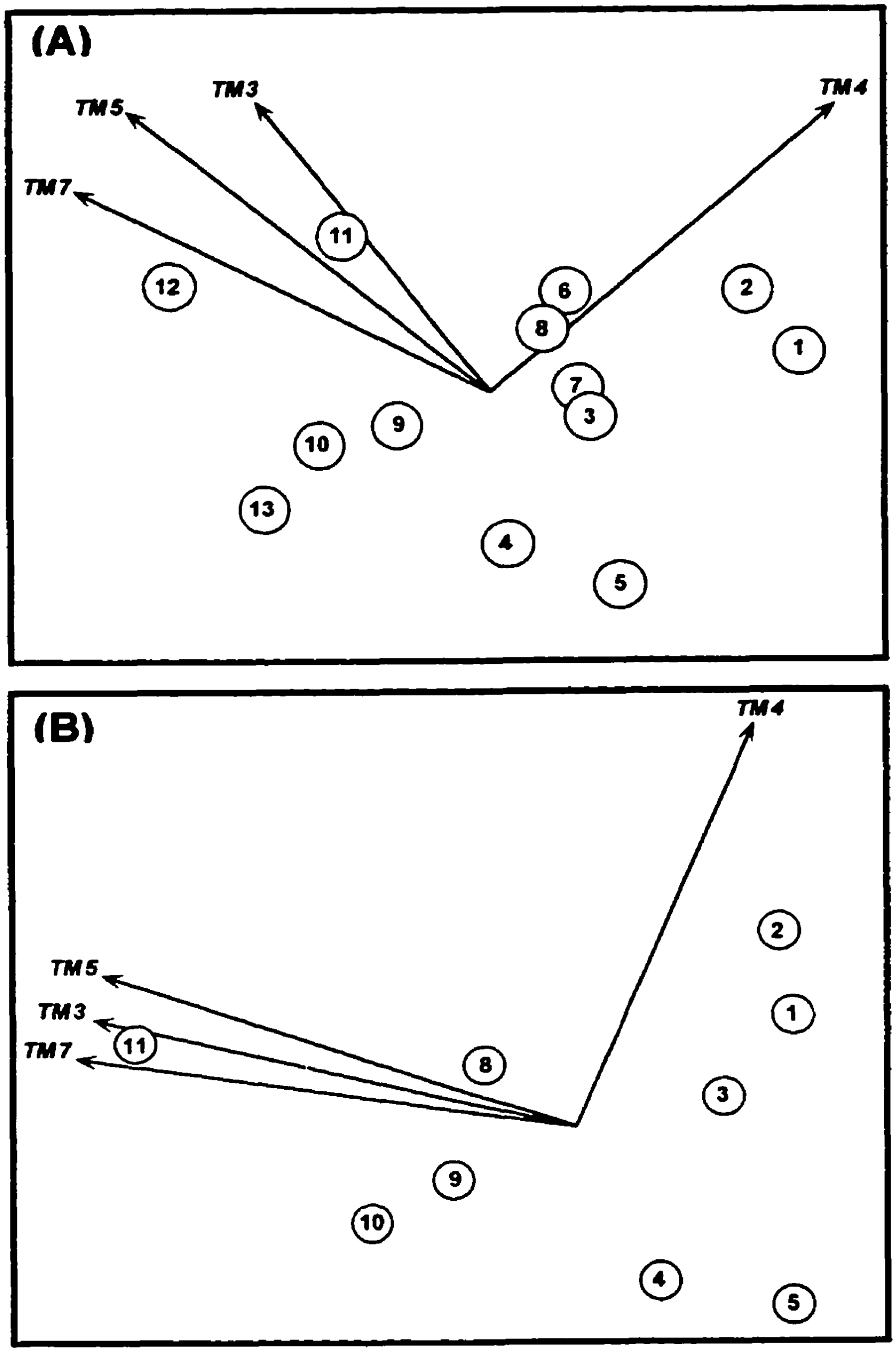


Figure 4.3. Multiple discriminant analysis of spectral reflectance variables of vegetation classes. 95% confidence ellipses for means are shown; symbols as in table 1. Biplots of variables are shown as arrows from the ordination centroid. A. MDA of 13 vegetation classes (Wilk's Λ, = 0.0010, $p < 0.001$). B. MDA of 9 vegetation classes with classes 6, 7, 12, 13 removed (Wilk's Λ, = 0.0002 , $p < 0.001$).

3. Class 13 (Regenerating Burn) proved to be spectrally distinct, but it represents a highly variable disturbance feature that spans several vegetation classes. Regenerating burn boundaries were therefore hand-digitized to create a bitmap, and an unsupervised 28-class isodata classification was run under the mask. All classes falling within the spectral range of the ground-sampled regenerating burns were aggregated into a single regenerating burn class, converted to a bitmap mask, and removed from further classification.

4. Class 12 (*Dryas* heath) occurs on long, narrow former beach ridges that parallel the Hudson Bay coastline. While limited in spatial extent, this vegetation class represents a distinct and ecologically/culturally significant landscape feature. To ensure that these small, elongated regions were retained on the final map, a mask was created for this spectrally distinct class using the same procedure applied to regenerating burns.

A residual MDA (excluding classes 6, 7, 12 and 13) reveals strong separation of the 9 remaining classes (Fig. 4.3b). The lichen peat plateau (class 11) has the highest reflectance in bands 3, 5 and 7. The most productive sites (classes 1-3) have highest reflectance in band 4, but low reflectance in bands 3, 5 and 7.

Redundancy Analysis

Redundancy analysis indicates that 43% of variance in the CA ordination space can be explained by variation in spectral reflectance (bands 3, 4, 5 and 7). This suggests a reasonably strong relationship between floristic composition and Landsat spectral reflectance, but not surprisingly a large amount of the variation in floristic composition remains unaccounted for. This occurs because spectral reflectance is largely a function of the structural, rather than floristic, properties of vegetation. The RDA ordination (Fig. 4.4) indicates that vegetation classes dominated by lichen (classes 8, 10 and 11) are characterized by high reflectance in bands 3, 5 and 7, but low reflectance in band 4. More productive wet fen and shrub habitats (classes 1, 2 and 3) show highest reflectance in band 4, but low reflectance in bands 3, 5 and 7.

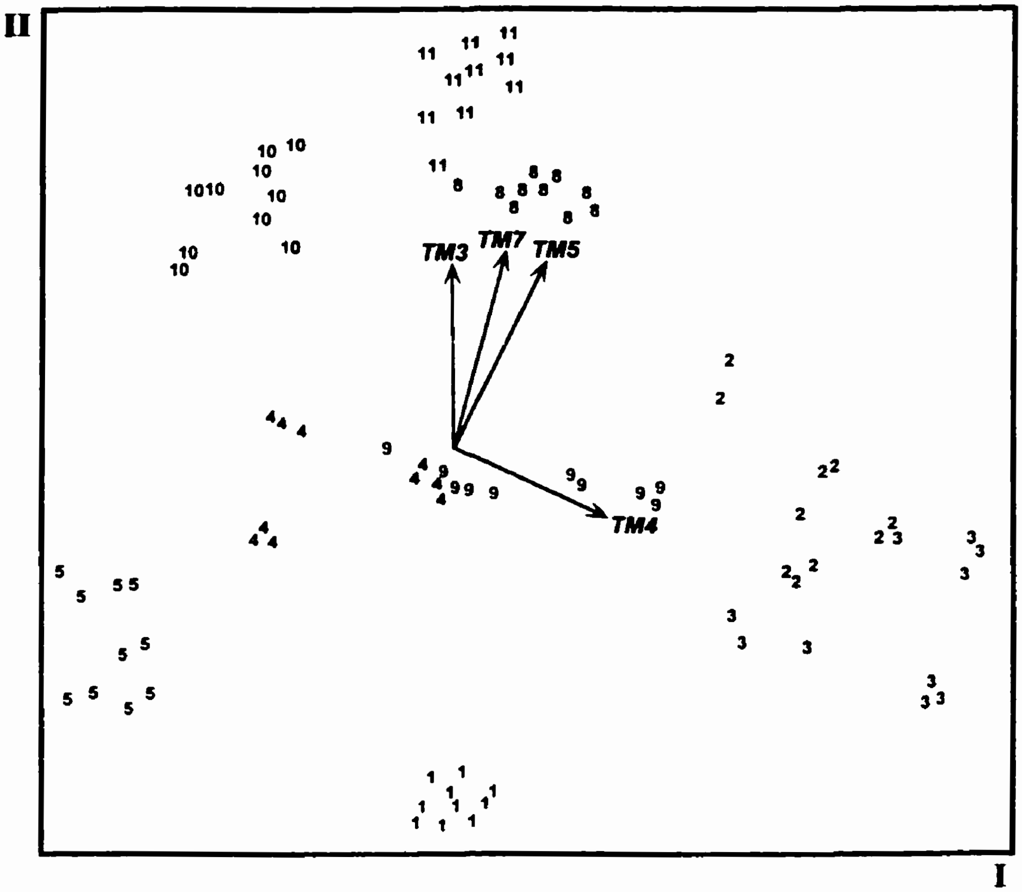


Figure 4.4. Redundancy analysis relating nine vegetation classes (numbers, see table 1 for codes) and four spectral TM bands (TM 3-5, and 7, displayed as biplots). The two ordination axes I and II account for 77.7% of the vegetation-spectral band relation. Total redundancy is 43%.

DISCUSSION

Coarse-scale mapping projects have often downplayed the importance of collecting ground-cover data in the field. Indeed, such data are often only used to label classes derived from an unsupervised classification of spectral information (Cihlar 2000). For finer-scale vegetation mapping projects, my results indicate that detailed and extensive field data is essential to the accurate interpretation of satellite imagery. I suggest that image classification (whether using a supervised or unsupervised approach) should occur only after vegetation classes have been identified and characterized using appropriate statistical analyses of ground cover field data. Once this has been done, the spectral information associated with each vegetation class must be examined to ensure that the classes are spectrally distinct. The mapping of ecologically meaningful vegetation classes must begin with a classification of ground vegetation, not a classification of satellite spectral reflectance data (e.g. Thompson et al. 1980, Nilsen et al. 1999).

The use of ground data to evaluate spectral information provides critical insights into optimal band combinations for classification purposes, and identifies the degree of separability of different classes. It is generally recognized that not all Landsat TM bands are required for classification purposes (Richards 1993). Band selection should be based on the analysis of correlations among bands, and on the ability of different band combinations to resolve the classes of interest. In the initial analyses, PCA was performed using six spectral bands (bands 1-5, and 7) as variables. Including the three highly collinear visible (photosynthetic) bands 1-3 places strong emphasis on distinguishing vegetated from unvegetated classes. Band 1 is considered particularly effective in separating vegetation from barren ground (Lillesand and Kiefer 1994). After the unvegetated outliers had been recognized and removed, bands 1 and 2 were excluded from subsequent multivariate analyses (MDA, RDA) in order to reduce variable multicollinearity. When discriminating vegetation types, it is generally recommended that at least one band from each of the visible, near-infrared and mid-infrared regions be included (Beaubien 1994). The combination of bands 3 (visible red), 4 (near infra-red), and 5 (mid-infrared) has been widely used (e.g. Benson and De Gloria 1985, Horler and Ahern 1986, Moore and Bauer 1990, Beaubien 1994), although band 7 (mid-infrared) is sometimes substituted for band 5. Multivariate biplots (Gabriel 1971) effectively summarize the correlations among bands and trends in the spectral reflectance properties of vegetation classes, thus allowing the analyst to select bands that are most appropriate for a particular application.

The initial 17 land cover/vegetation classes (Table 4.1) displayed some overlap in spectral space. A classification based directly on these 17 groups would therefore have resulted in an inaccurate and misleading vegetation map. The reasons for overlapping spectral signatures varied. Classes 3 and 7 (Willow Birch Shrub Fen and Willow Sedge Poor Fen) are floristically similar, and are distinguished mainly by higher shrub cover in the Willow Birch Shrub Fen class. These two classes were amalgamated since this relatively subtle difference in vegetation could not be resolved by the TM spectral reflectance data. In contrast, the spectral overlap between classes 6 and 8 (Graminoid Willow Salt Marsh, and Lichen Spruce Bog) illustrates how very different vegetation classes can have nearly identical spectral signatures. The salt marsh class is characterized by small patches (1-20 m^2) of halophytic vegetation (graminoids and low shrubs) interspersed with highly reflective unvegetated clay, while Lichen Spruce Bogs are completely vegetated and characterized by a sparse cover of coniferous trees and a highly reflective ground layer dominated by lichens. These two classes are structurally somewhat similar (patches of highly absorptive vegetation on a highly reflective matrix), but they share no species in common and are functionally very different. They cannot be distinguished using spectral reflectance information alone, and so must be mapped using ancillary information.

In mapping natural vegetation, it is important to recognize that difficulties in characterizing thematic classes are not merely related to a mixed pixel problem. Indeed, very different combinations of vegetation cover and substrate conditions can sometimes produce nearly identical reflectance signatures. This phenomenon can only be identified through extensive collection of field data and a careful examination of its spectral properties. A number of approaches are available to separate distinct vegetation classes showing overlap in spectral space. It is difficult to recommend a single approach, however, since each mapping exercise presents unique problems and challenges. Ancillary information from digital elevation models, aerial photographs, substrate data and so forth have long been used to improve the separation of classes in vegetation mapping exercises (e.g. Nilsen et al. 1999, Walker 1999). Alternatively, multi-temporal satellite imagery can be used to separate classes that vary spectrally through time, for example over a growing season (Fuller and Parsell 1990, Lunetta and Balogh 1999). Whatever method is used, it is of course necessary to first identify and characterize the overlapping classes.

I feel that the production of an effective and optimized vegetation map is best achieved using an adaptive learning process that involves careful examination of the relationship between the ground cover and spectral data. The approach advocated here is labour intensive and involves numerous iterative steps. However, I believe that it provides a highly robust and flexible approach to thematic mapping, since the analyst is able to make informed choices at all stages of the decision-making process. The graphical approach that I employ provides an uncomplicated and intuitive method for displaying very complex multivariate relationships between ground cover and spectral reflectance data. Presenting multivariate analysis results only in tabular form, while informative, is rarely enlightening.

CONCLUSION

A model is a simplification of a natural system that retains the important or usable information while removing 'noise' that would otherwise obscure important trends. The development of accurate and functional landscape-scale vegetation/land cover maps requires a rigorous modelling approach that includes both validation and verification. A graphically-oriented multivariate approach can assist the modelling process by allowing the analyst to consider multiple satellite bands simultaneously, and to identify overlap in spectral reflectance values prior to classification. This is a critical process, since mutually exclusive classes must be defined in order to unambiguously classify pixels in a satellite image.

The results of this chapter achieve the third objective of this thesis; to construct a geocoded Landsat TM satellite image mosaic of the study area and examine the spectral reflectance values of the different vegetation communities in order to produce a set of mutually exclusive vegetation classes.

MANAGEMENT IMPLICATIONS

The complex methodology used in developing the vegetation classes and the overlap that exists in the spectral reflectance requires that caution be used in mapping vegetation in the region and interpreting spectral reflectance values in satellite imagery. This is particularly important for using satellite images to detect change and examine productivity over time since these measurements are based on spectral values.

CHAPTER 5: DEVELOPMENT OF A LANDSCAPE LEVEL VEGETATION MAP FOR WAPUSK NATIONAL PARK

"...the general zonation of vegetation probably reflects accurately the sequence of colonization. Accordingly, one might interpret the prevalence of large tracts of fen on younger sites and its scarcity further inland as an indication of the expected sequence of fen to bog, with the earliest stages being marsh and scrub."

-J.C. Ritchie, on the vegetation of the Hudson Bay Lowlands in Manitoba, 1960.

PREAMBLE

In this chapter, the vegetation classes identified in chapter 4 are mapped using the Landsat-5 TM satellite image mosaic, along with extensive field data and a stratified, layered classification approach. This highly complex landscape demonstrates the methods needed to effectively develop a landscape level vegetation map.

INTRODUCTION

Ecosystems by their very nature are structurally complex, making them inherently difficult to characterize. However, ecological information describing landscape structure is essential to develop a comprehensive database that allows management decisions to be made effectively (Trotter 1991). The information required includes descriptions of the pattern, size and thematic character of ecosystem components. Mapping this information accurately at an appropriate scale within a reasonable time frame is a significant challenge.

Satellite remote sensing has emerged as an important mapping tool in the last decade due to its provision of regular, repeated coverage of large areas over many years at a reasonable cost. Digital imagery has become the dominant format, as it is amenable to computer manipulation. Satellite imagery provides detailed information about the composition of the landscape. However, in order to produce a useful regional scale map product, this high level of detail across a complex continuum must be simplified into a group of discrete classes. Classification is the process by which the continuum is sorted into meaningful groups or classes. This involves the transformation of raw data into meaningful information.

The vast majority of thematic mapping projects using satellite imagery have produced highly generalized land cover classes (Townshend 1992) (e.g. 'wetland, 'grassland', 'open forest' or 'urban') that reflect the major components of the landscape. However, such broadly defined map classes are often of limited utility in addressing landscape-scale ecological questions (Defries and Belward 2000). In response to this limitation, finer-scale maps have been produced in which land cover is interpreted in terms of identifiable ground cover components (Roughgarden *et al.* 1991). For example, vegetation maps interpret the classes as plant communities based on floristic composition and structure (Kenk *et al.* 1988).

Standard image classification techniques include supervised and unsupervised classification (Lillesand and Kiefer 1994). In both of these approaches, a pixel is allocated into the class to which it is most spectrally similar, based solely on the spectral signature of that pixel. The decision for the classification of the pixel is made independently of its position in the image or the class allocation of nearby pixels. Despite the extensive use of this procedure, with a variable amount of success, it generally leads to at least some misclassifications because the spectral signature of a single pixel may be influenced by a wide range of factors that can cause confusion in its class identity.

A variety of algorithms can be used to group individual pixels into discrete classes. The cover types can be user defined with the number of different classes being defined by the classification software. Unsupervised classification uses statistical clustering techniques to combine individual pixels into classes according to their degree of reflectance similarity in each spectral band. This process generates a unique set of spectral classes that must then be assigned to represent a particular vegetation type (Wilkie and Finn 1996). In contrast, supervised classification allows the researcher to use existing knowledge of the area to locate areas within the imagery that are known to belong to particular vegetation types (Wilkie and Finn 1996). Using the pixels within the known areas, the spectral characteristics of each type is then calculated. These areas are "training sites" to be used for the classification of the entire image. The classification software can then assign all other pixels to the vegetation class that most closely matches the spectral characteristics of the training sites.

Both supervised and unsupervised methods are generally applied using all available information in a single set of decision rules to simultaneously allocate all pixels into the class to which they are spectrally most similar. The underlying assumption of these methods is that each pixel can be assigned to a given class (Townshend 1992). However, this is often a false assumption due to the high variability of landscapes that produces a correspondingly complex range of spectral reflectance values across all satellite bands. One of the primary limitations of satellite data is that it provides only a measure of the amount and type of reflected and emitted electromagnetic energy.

This spectral information is influenced by a wide variety of factors including soil moisture, substrate, structural configuration, topography and atmospheric effects, as well as the amount, vigour, productivity, structure, and floristic composition of vegetation (Richardson and Wiegland 1977, Vogelmann and Moss 1993). This high variability inevitably leads to challenges in allocating pixels into unique classes, particularly pixels that contain more than one vegetation community. If the proportion of these mixed pixels is sufficiently large, map accuracy may be significantly reduced. While these errors occur to some degree in most classifications, they may represent a small portion of the entire image and have little effect on map utility. However, even small errors in mapping can result in low confidence in the final map product if they are particularly conspicuous. For example, a large saturated wetland may have some pixels misclassified as burned forest due to confusion between the pixels since both classes typically have low reflectance values (Chapter 4). This misclassification may be only a minor influence in the quantitative accuracy assessment. But to someone unfamiliar with the concepts of remote sensing, this misclassification could appear to be a critical error that seriously undermines their confidence in the map product. If vegetation maps are to be useful information tools, they must provide reasonable overall accuracy, but also garner the confidence of those who would use them. A variety of methods have been developed to increase classification accuracy (Townshend 1992, Kenk *et al.* 1998, Cihlar 2000).

Layered classification is a hierarchical process that uses a series of classifications, with separate class(es) being masked out of the imagery in successive iterative steps. The advantage of this approach is that it optimizes the separation of individual classes (Lauver and Whistler 1993, Kartikeyan *et al.* 1998). The analyst can select the optimum combination of spectral bands that maximally separates each class (Kenk *et al.* 1988, Boresjö 1989). The importance of an optimal combination of bands is recognized in Chapter 4. Layered classification also increases the resolving power of each successive classification by removing entire classes with all of their inherent variation, making the subtle differences between the more similar classes more easily separable.

During the classification process, stratification of the study area may also be performed in order to limit class distribution with pre-defined boundaries (Hutchinson 1982, Gurney and Townshend 1983, Mason *et al.* 1988, Joria and Jorgenson 1996). This procedure allows ancillary data to be used to define spatial limits for certain classes that would otherwise be problematic (Kenk *et al.* 1988, White *et al.* 1995). Ancillary data can include soils, geology, vegetation and topography (Hutchinson 1982, Frank and Isard

1986, Cibula and Nyquist 1987). In each step, only the classes that are likely to occur under each mask are allocated in each unsupervised classification. This allows the number of misclassifications between classes to be reduced, if they occur under different masks (Boresjö Bronge 1999). However, stratification produces discrete boundaries, which may result in inconsistencies across boundaries, so caution is required.

Robust approaches to classification are particularly important in the Hudson Bay Lowlands of Manitoba due to its spatial and floristic heterogeneity. The complex mosaic of vegetation in the region provides an ideal case study for considering the challenges associated with producing a landscape level vegetation map.

Ritchie (1960) suggested that the vegetation of the Hudson Bay Lowlands of Manitoba showed zonation, with fen vegetation dominating near the Hudson Bay coast and bog vegetation dominating further inland, based on qualitative observations. This trend was explained by isostatic uplift, which slowly raises the land out of the marine waters, so those sites further inland were older and had more time to accumulate peat. However, the map developed by Ritchie (1962) lacked the detail necessary to quantitatively test this relationship. Field measurements at the site level provide quantitative support for this theory (Chapter 3, Sims 1982), but prior to this study, it remained untested at the landscape scale.

This project was initiated to develop a vegetation map database that can function as a flexible tool that can be adapted to fit the needs of local people, researchers and management agencies. This requires a repeatable mapping strategy that produces a product that is both accurate and flexible to be used for a range of applications. An application of the map is presented to show the utility of a regional scale digital map in relating large-scale patterns of vegetation distribution to the ecosystem-level process of peat accumulation- mediated succession from fen to bog.

The objectives of this chapter were to (1) classify all Landsat TM pixels in the study area into one of the defined vegetation classes; (2) describe the composition and structure of the vegetation associated with each class; (3) perform an accuracy assessment of the classification using an independent data set; and (4) use the map to test the hypothesis that the relative proportion of bog vegetation at the landscape scale will increase and the relative proportion of fen vegetation will decrease with increasing distance from the Hudson Bay coast.

METHODS

Field Data

Field data were used to develop an accurate training data set, describe the composition and structure of the vegetation associated with each class and compile an independent data set of sampled vegetation communities to be used exclusively for assessing the accuracy of the final classification.

In addition to the ground plots sampled, helicopter surveys were done in August 1999 to augment the accuracy assessment data set. During these flights, 1100 visually homogenous patches that were 50 x 50 m or larger were located. The helicopter then hovered 30 m above the centre of the patch or landed while estimates were made of the wetness, cover of major plant species and vegetation structure.

All of the sites sampled were converted into vector points to overlay on the TM imagery. This provided a direct association between the spectral reflectance of each pixel and its vegetation cover, substrate composition, and wetness. Of the 600 ground points and 1100 helicopter points, 300 ground points were used to classify the Landsat TM mosaic. The remaining 300 ground points, along with all 1100 helicopter sites were used for accuracy assessment. All 600 ground sites were used to delineate the vegetation communities and describe their composition and structure using aspatial analyses (Chapter 3).

Image Classification

The 16 vegetation classes delineated in Chapter 4 using multivariate analysis are used for the classification. Layered classification was used so that an individual classification was run to separate each vegetation class in the satellite image (see below for more details) and mask it out of future iterative classification steps. In cases where there were problems of class confusion that could be resolved using differences in spatial location of the classes, masks digitized. When classifying the TM image, the analyst could then selectively exclude classes in those masked areas that were known to be absent on the ground, but were causing confusion in the classification. After all classifications were completed, the individual masks were merged together into a single classification so that every single pixel was allocated into a vegetation class, with no unclassified pixels remaining. The resulting map was colour-coded and a legend was added in preparation for hard copy output.

Initial arbitrary cluster allocation (Isodata) unsupervised classification was used for all classifications (IMAGEWORKS, PCI Geomatics 1998). This is an iterative technique that initially seeds a specified number of cluster centroids. The Euclidean distance between each pixel and each cluster centroid is calculated and the pixel assigned to a class. A new set of class mean vectors is then calculated from the results of the previous calculation and the pixels are reassigned to the new cluster vector. This process is continued in an iterative fashion and any clusters having excessively large standard deviations are split into smaller clusters. Clusters that are too close together in multidimensional space are merged into a single cluster. Clusters with too few pixels in it are discarded. This process continues until there is little change in class assignment between iterations or the maximum number of iterations is reached.

Classification Stage 1 (Unvegetated Classes)

Analysis of spectral signatures indicated that the visible bands (1-3) in the TM image were particularly useful for separating unvegetated classes (Chapter 4), so TM bands 1-5,7 were used in separating the unvegetated classes. All other classifications used bands 2, 3, 4, 5, 7. The entire Landsat TM image mosaic was initially run through a 255-class isodata unsupervised classification. All of the resulting 203 classes were identified as being either water or non-water. All water classes were aggregated into a single water class and converted into a bitmap mask. All areas of the TM image under the mask were removed from further classification. Next, a 255-class isodata unsupervised classification was run on the image outside of the water mask. All of the resulting 209 classes were identified as being either unvegetated ridge, unvegetated shoreline or "other" and aggregated into single bitmap masks for each vegetation class. These unvegetated classes were then masked out of future classifications.

Classification Stage 2 (Low Vegetation Cover Classes)

The boundaries of known recent burns were digitized using a visual interpretation to create a bitmap mask over the recently disturbed areas (IMAGEWORKS, PCI Geomatics 1998). An unsupervised 38-class isodata unsupervised classification was then run on the image, and all classes within the spectral range of recent burns were aggregated to form a single recent burn class. This was converted into a bitmap mask and removed from further classification. This same procedure was performed using the boundaries of identified regenerating burns.

Class XI (*Dryas* Heath Upland) occurs in close proximity to Hudson Bay on long, narrow former beach ridges that parallel the existing coastline. While limited in spatial extent, this vegetation class represents a distinct and ecologically/culturally significant landscape feature. To ensure that these small, elongated regions were retained on the final map, a mask was created for this spectrally distinct class using the same digitizing procedure applied to the boundaries of regenerating burns.

Classes VI (Graminoid Willow Salt Marsh) and VIII (Lichen Spruce Bog) showed considerable overlap in spectral space but are known to be separated spatially (salt marsh occurs exclusively along the coast, while lichen/spruce bog is found farther inland). To distinguish the salt marsh class on the vegetation map, a 15 km wide strip along the shore of Hudson Bay was hand-digitized and masked. Unvegetated areas of the mask were subtracted and an unsupervised 28-class isodata classification was run to spectrally separate salt marsh from other vegetation classes.

Classification Stage 3 (Treeless Lichen Dominated Classes)

Previous classifications indicated that class IX (Lichen Melt Pond Bog) showed some confusion with class VIII, resulting in some class IX pixels occurring in the northern half of the study area, near Cape Churchill where this class is known to be non-existent. This incongruity would lower classification accuracy and could potentially influence user confidence in the final map. To resolve this issue, a mask was digitized along a 20 km strip of coastline at Cape Churchill to exclude class IX from this area. A 205-class isodata unsupervised classification was then run on the remaining TM image, creating 114 classes, which were then aggregated and separated as a bitmap mask. A 205-class isodata unsupervised classification was then run on the remaining TM image to separate Class X (Lichen Peat Plateau Bog).

Classification Stage 4 (High Vegetation Cover/Productivity Classes)

The remaining TM image was run through a 255-class isodata unsupervised classification and each of the resulting classes was sorted into one of the remaining classes (I, II, III, IV, V, VII, VIII) based on their reflectance signatures.

Floristic Composition and Structure

Measured vegetation composition was summarized for each vegetation class as mean, standard deviation, minimum and maximum percent cover values for all species and life forms.

Accuracy Assessment

Accuracy of the 16-class vegetation map was calculated using ground verified assessment sites that were independent of the training sites. The vegetation map was compared to an independent ground collected data set of 1400 sites, on a site by site basis to develop an error matrix in order to estimate map accuracy. Agreement and disagreement between the map and verification data set is summarized by three simple descriptive statistics. Accuracy is presented as the percent agreement between the field and image based classifications.

A conventional confusion matrix is used to identify errors of omission and commission, following Story and Congalton (1986). Error matrix characteristics are discussed elsewhere in the literature (Aronoff 1982a, 1982b, Story and Congalton 1986). The omission accuracy (producer's accuracy) is defined as the total number of correct pixels in a specific class divided by the total number of pixels sampled in that class. An error of omission occurs when a pixel is not assigned to its appropriate class. Commission accuracy (user's accuracy) is defined as the total number of correct pixels in a class divided by the total number of pixels that were actually classified in that category. An error of commission occurs when a pixel is assigned to a class to which it does not belong. Overall accuracy provides a measure of the average classification accuracy, represented by the total correct (sum of the major diagonal in the error matrix) divided by the total number of pixels in the error matrix.

Post-Classification Processing

Image post-processing includes the application of a modal filter to the entire classification. Modal filtering is a process applied to the per-pixel classification, allocating single pixels with diverging class identity into the majority nearby class in order to produce a more map like product. Modal filters have been shown to increase classification accuracy (Gurney and Townshend 1983). However, filtering changes the number and distribution of classified pixels. The type of filtering applied is determined by the intended use of the map. Applications requiring a more generalized map will use a high degree of filtering. In order to satisfy the various needs of a diverse group of end-uses, a number of maps were produced by running a modal filter on the final classification. The filter was run separately on three different versions of the vegetation map at a 3x3 pixel, 5x5 pixel and 7x7 pixel scale to produce three representations of the vegetation cover at various levels of generalization.

Wetland Map

In order to examine changes in vegetation at the landscape scale in relation to the distance from the Hudson Bay coast, a map of bog and fen vegetation was constructed by aggregating all fen classes into a single class and all bog classes into a single class. The composition of the landscape was then determined using ten 45 km transects across the bog-fen vegetation map. The start point location of each transect was determined using a stratified random design. A vector of the Hudson Bay coastline was stratified into five equal 62 km sections and points were randomly placed along the vector. Transects were run perpendicular to the coastline in order to sample along a "distance from coast" transect. Each transect was sampled at five kilometre intervals to collect the vegetation class identity of all pixels within each 30 x 30 pixel (900 x 900 m^2) block for a total of 10 samples per transect. The relative frequencies of the bog and the fen types were then compared with the distance from the coast. This information was then used to test the prediction that the relative proportion at the landscape scale of bog vegetation will increase and the relative proportion of fen vegetation will decrease with increasing distance from the Hudson Bay coast (Chapter 3).

Lichen Cover Map

In order to represent the vegetation data in an alternative format for different applications, a lichen cover map was constructed by allocating each vegetation class from the initial map into one of three classes based on its average lichen ground cover determined by field sampling. The lichen cover classes were: 0-20%, 21-40%, and 41-90%. The accuracy of the lichen cover map in predicting the percent cover of lichens on the ground was tested using the percent cover of all lichens within all of the accuracy assessment sites used for the vegetation map.

RESULTS

Image Classification

The resulting 16-class vegetation map is presented in Fig. 5.1. The relative abundance of the different classes and the area that they cover is summarized in Table 5.1. Relative proportions of the different classes (Fig. 5.2) indicates that the study area is dominated by three classes (Sedge Bulrush Poor Fen, Lichen Melt Pond Bog and Lichen Spruce Bog), that collectively make up over 50% of the total area. Physical descriptions of the classes include floristic composition summary data (Tables 5.2-5.17) as well as aerial and ground photos and a distribution map for each class (Figures 5.3-5.18). The error matrix for the 16-class vegetation map is presented in Table. 5.2. Overall, the map was 97% accurate, though this varied among classes, ranging from 88% to 100% accuracy.

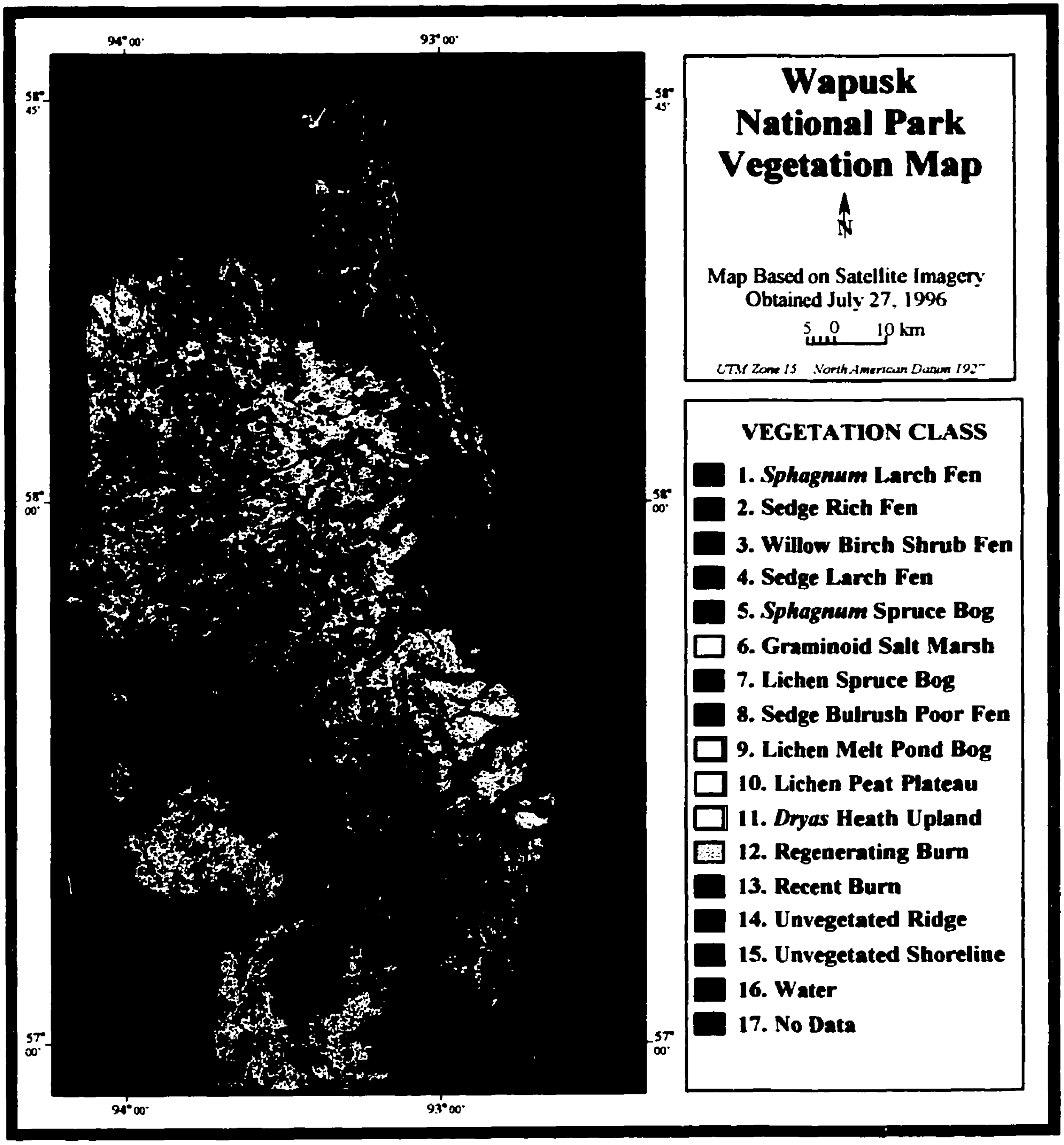


Figure 5.1. Vegetation map of the study area depicting 16 vegetation classes, based on Landsat TM satellite imagery.

Table 5.1 Accuracy assessment error matrix for the 16-class vegetation map.

	Independent Ground Reference Data																
Vegetation Map	I	II	III	IV	V	VI	VII	VIII	IX	X	XI	XII	XIII	XIV	XV	XVI	**User's Accuracy (%)**
I	28	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	**97**
II	0	51	0	0	0	1	0	0	0	0	0	0	0	0	0	0	**98**
III	0	2	17	2	0	2	0	1	0	0	0	0	0	0	0	0	**71**
IV	0	0	0	73	0	0	1	0	0	0	0	0	0	0	0	0	**99**
V	0	0	0	4	12	0	0	0	0	0	0	0	0	0	0	0	**75**
VI	0	0	0	0	0	36	0	0	0	0	0	0	0	0	0	0	**100**
VII	0	0	0	1	0	0	71	1	1	1	0	0	0	0	0	0	**95**
VIII	0	1	0	2	0	0	4	45	4	0	0	0	0	0	0	0	**80**
IX	0	0	0	0	0	0	0	5	123	0	0	0	0	0	0	0	**96**
X	0	0	0	0	0	0	2	0	1	69	0	0	0	0	0	0	**96**
XI	0	0	0	0	0	0	0	0	0	0	127	0	0	0	0	0	**100**
XII	0	0	0	0	0	0	0	0	0	0	0	107	0	0	0	0	**100**
XIII	0	0	0	0	0	0	0	0	0	0	0	0	34	0	0	0	**100**
XIV	0	0	0	0	0	0	0	0	0	0	0	0	0	23	0	0	**100**
XV	0	0	0	0	0	0	0	0	0	0	0	0	0	0	87	0	**100**
XVI	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	200	**100**
Producer's Accuracy (%)	**100**	**94**	**100**	**89**	**92**	**92**	**91**	**95**	**95**	**99**	**100**	**100**	**100**	**100**	**100**	**100**	**97%**
# Sites	28	54	17	84	13	39	78	52	129	70	127	107	34	23	87	200	

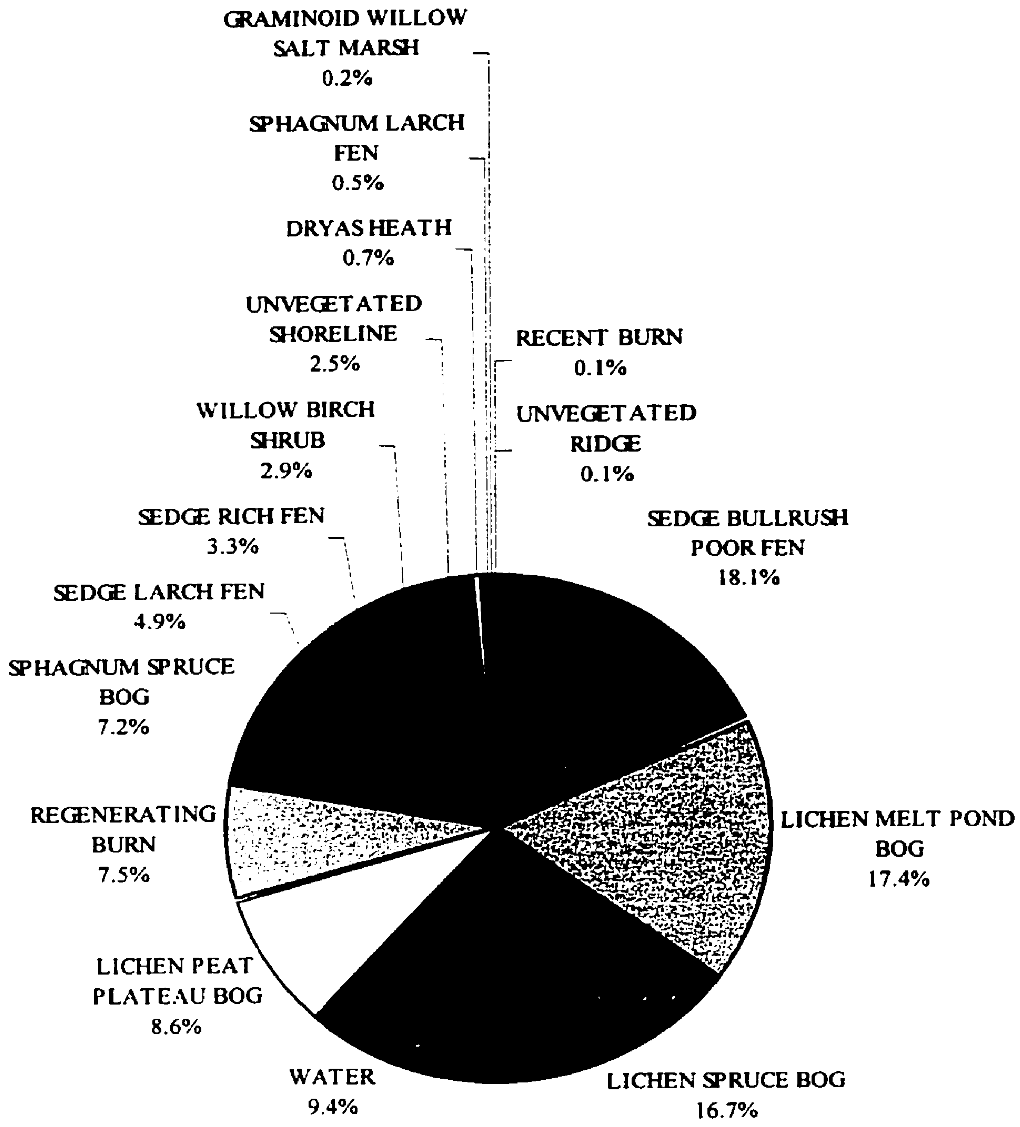


Figure 5.2. Relative cover of the 16 vegetation classes presented in Fig. 5.1.

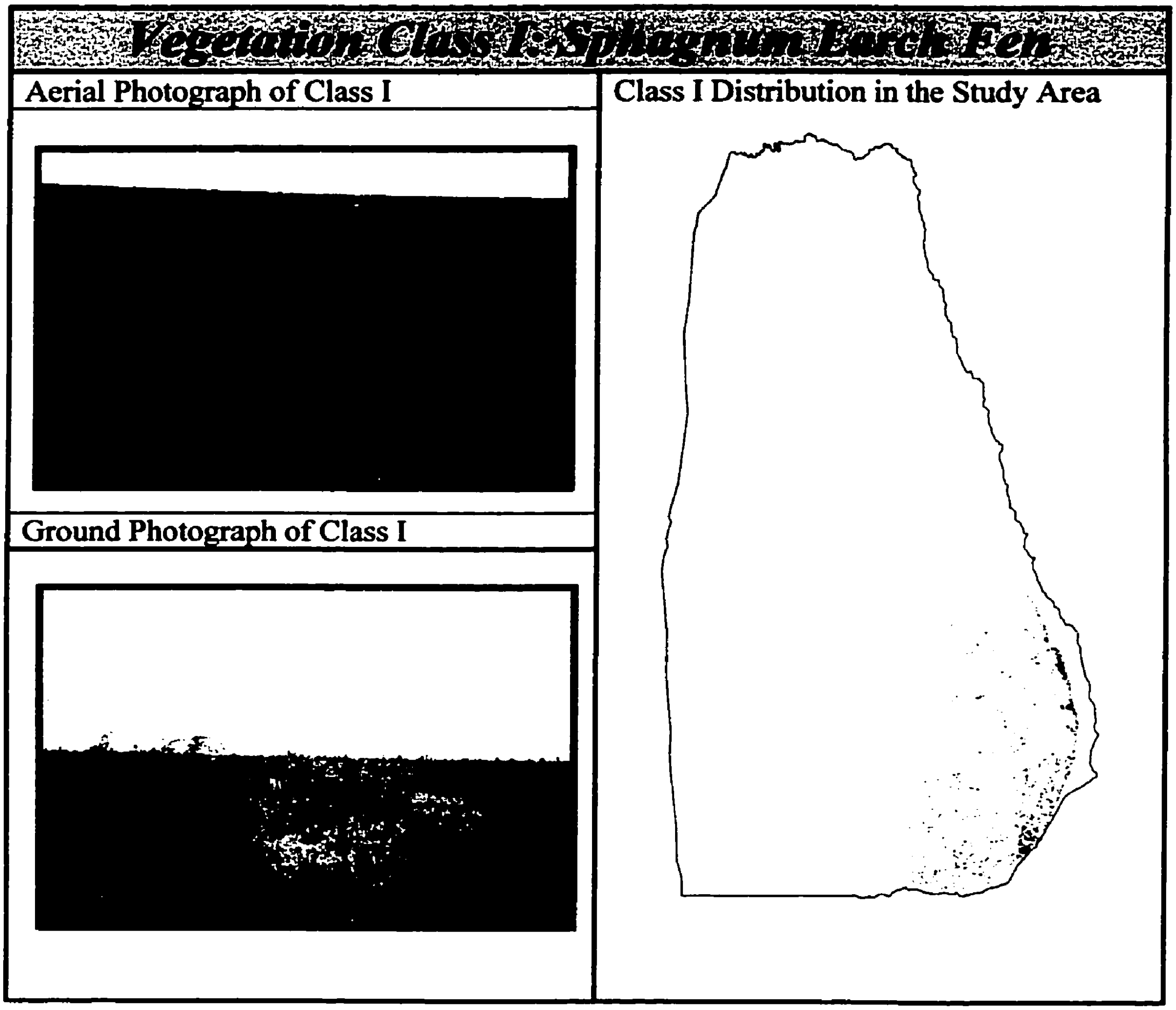


Figure 5.3. Photographs and distribution map of vegetation class I (*Sphagnum* Larch Fen).

Table 5.2. Species composition of Class I (Sphagnum Larch Fen), based on 8 sample sites.

	Average	Std Dev	Max	Min
TREE				
Larix laricina	17.5	4.6	20.0	10.0
SHRUB				
Betula glandulosa	18.8	3.5	20.0	10.0
Kalmia polifolia	6.6	4.7	10.0	1.0
Ledum groenlandicum	0.1	0.4	1.0	0.0
Salix pedicellaris	6.5	4.8	10.0	0.0
HERB				
Carex aquatilis	2.0	3.3	10.0	0.0
Eriophorum angustifolium	1.0	0.0	1.0	1.0
Habernaria hyperborea	0.3	0.5	1.0	0.0
Menyanthes trifoliata	6.8	7.0	20.0	1.0
Oxycoccus microcarpus	0.9	0.4	1.0	0.0
Petasites sagittatus	0.3	0.5	1.0	0.0
Potentilla palustris	0.9	0.4	1.0	0.0
Smilacina trifolia	9.0	6.0	20.0	1.0
Triglochin maritimum	0.3	0.5	1.0	0.0
Equisetum variegatum	8.0	8.4	20.0	1.0
Triglochin maritimum	0.3	0.5	1.0	0.0
MOSS				
Sphagnum sp.	18.8	8.3	30.0	10.0

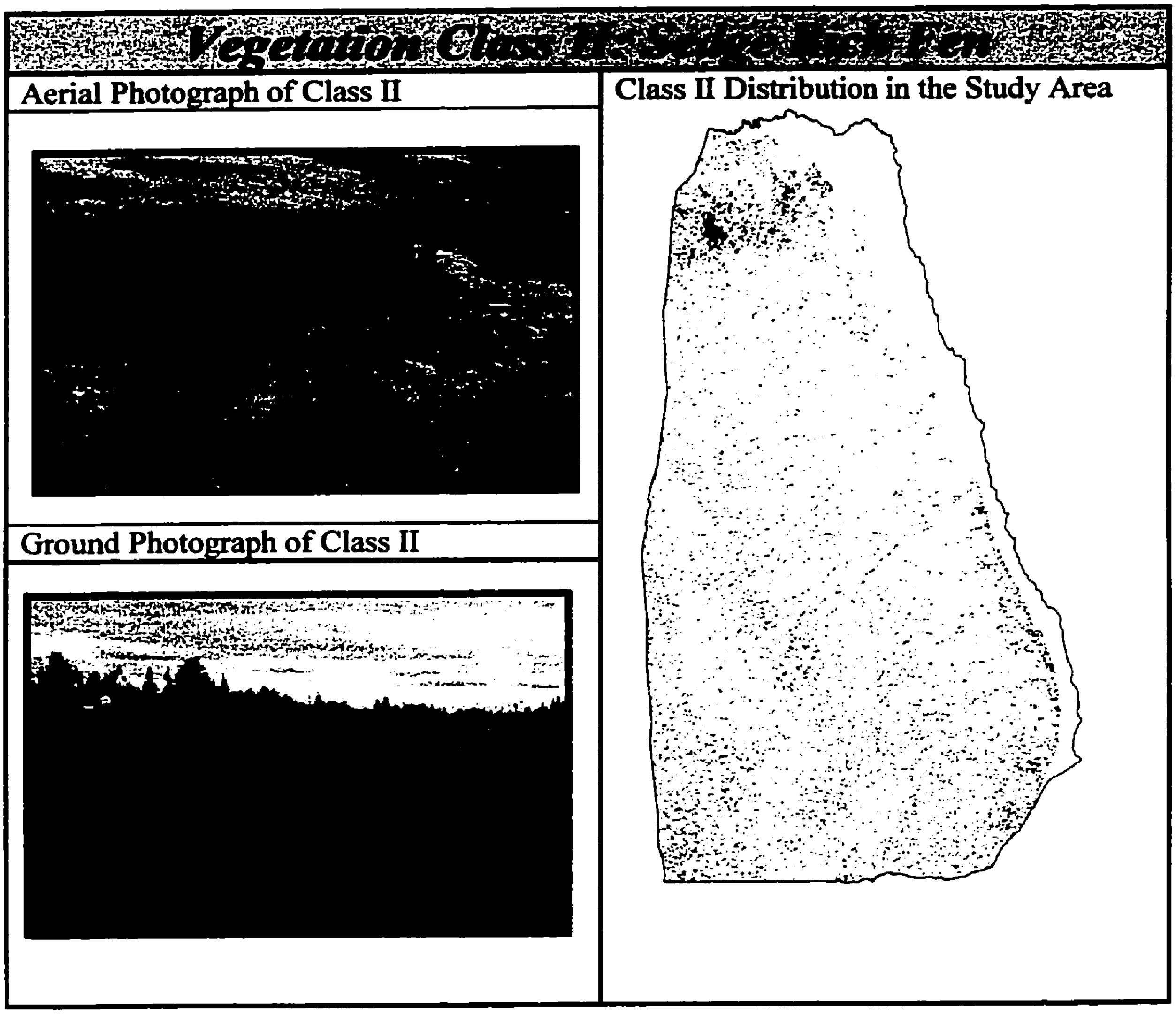


Figure 5.4. Photographs and distribution map of vegetation class II (Sedge Rich Fen).

Table 5.3. Species composition of Class II (Sedge Rich Fen), based on 14 sample sites.

	Average	Std Dev	Max	Min
TREE				
Larix laricina	0.1	0.3	1.0	0.0
SHRUB				
Andromeda polifolia	0.1	0.4	1.0	0.0
Betula glandulosa	1.0	2.6	10.0	0.0
Chamaedaphne calyculata	0.1	0.4	1.0	0.0
Kalmia polifolia	0.1	0.3	1.0	0.0
Salix candida	0.1	0.3	1.0	0.0
Salix planifolia	0.1	0.4	1.0	0.0
Vaccinium uliginosum	0.1	0.3	1.0	0.0
HERB				
Carex aquatilis	20.7	19.4	50.0	0.0
Carex capillaris	6.5	13.3	40.0	0.0
Carex capitata	0.1	0.3	1.0	0.0
Carex sp.	12.1	18.1	50.0	0.0
Drosera anglica	0.9	2.7	10.0	0.0
Drosera rotundifolia	0.1	0.3	1.0	0.0
Eriophorum angustifolium	1.6	3.6	10.0	0.0
Eriophorum vaginatum	0.1	0.3	1.0	0.0
Menyanthes trifoliata	3.7	4.9	10.0	0.0
Pedicularis sudetica	0.7	2.7	10.0	0.0
Potentilla anserina	1.4	5.3	20.0	0.0
Potentilla egedei	0.1	0.4	1.0	0.0
Potentilla palustris	2.3	8.0	30.0	0.0
Ranunculus aquatilis	0.7	2.7	10.0	0.0
Rubus chamaemorus	0.1	0.3	1.0	0.0
Scirpus caespitosus	1.4	5.3	20.0	0.0
Scirpus hudsonianus	0.7	2.7	10.0	0.0
Triglochin maritimum	0.1	0.4	1.0	0.0
MOSS				
Thuidium sp.	2.1	5.8	20.0	0.0
Scorpidium sp.	4.3	9.4	30.0	0.0
Drepanocladus sp.	0.7	2.7	10.0	0.0
Sphagnum sp.	9.3	12.1	30.0	0.0
Tomenthypnum nitens	2.1	5.8	20.0	0.0
NON-VASCULAR				
Equisetum sp.	7.9	10.5	30.0	0.0

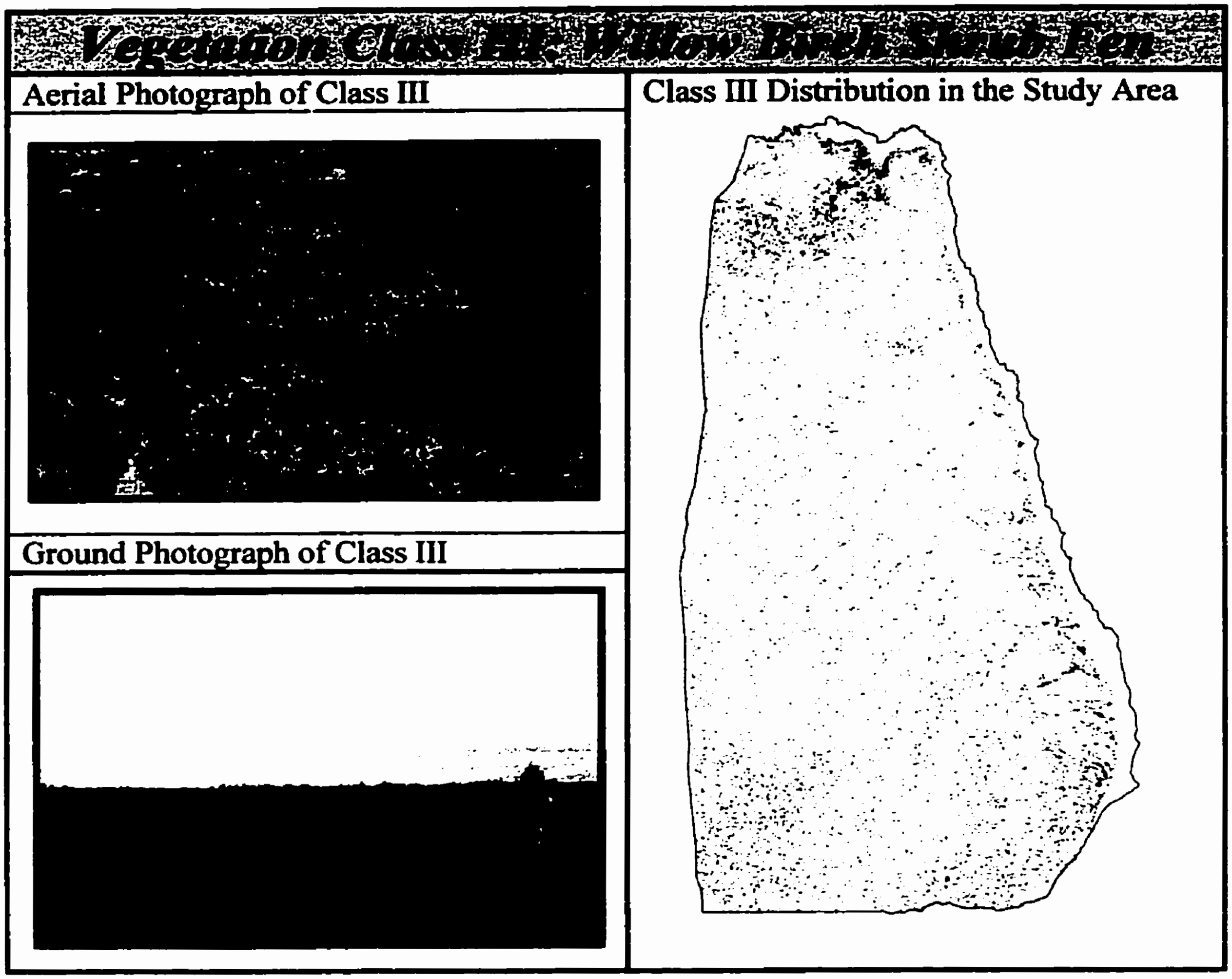


Figure 5.5. Photographs and distribution map of vegetation class III (Willow Birch Shrub Fen).

Table 5.4. Species composition of Class III (Willow Birch Shrub Fen), based on 15 sample sites.

	Average	Std Dev	Max	Min
TREE				
Picea glauca	0.1	0.3	1.0	0.0
SHRUB				
Andromeda polifolia	0.1	0.3	1.0	0.0
Betula glandulosa	17.3	14.2	40.0	0.0
Dryas integrifolia	0.1	0.3	1.0	0.0
Empetrum nigrum	1.9	4.0	10.0	0.0
Myrica gale	13.6	11.2	30.0	0.0
Rhododendron lapponicum	0.9	3.0	10.0	0.0
Salix brachycarpa	0.9	3.0	10.0	0.0
Salix candida	3.6	9.2	30.0	0.0
Salix lanata	1.1	3.0	10.0	0.0
Salix planifolia	17.3	29.0	90.0	0.0
Salix reticulata	5.5	8.1	20.0	0.0
Salix sp.	18.2	25.6	70.0	0.0
Vaccinium uliginosum	7.3	7.9	20.0	0.0
Viburnum edule	2.7	9.0	30.0	0.0
HERB				
Carex aquatilis	1.0	3.0	10.0	0.0
Carex sp.	1.8	4.0	10.0	0.0
Hedysarum mackenzii	0.1	0.3	1.0	0.0
Petasites sagittatus	0.1	0.3	1.0	0.0
Pyrola rotundifolia	0.2	0.4	1.0	0.0
Rubus aucalis	0.1	0.3	1.0	0.0
Rubus chamaemorus	0.1	0.3	1.0	0.0
NON-VASCULAR				
Equisetum sp.	1.8	4.0	10.0	0.0

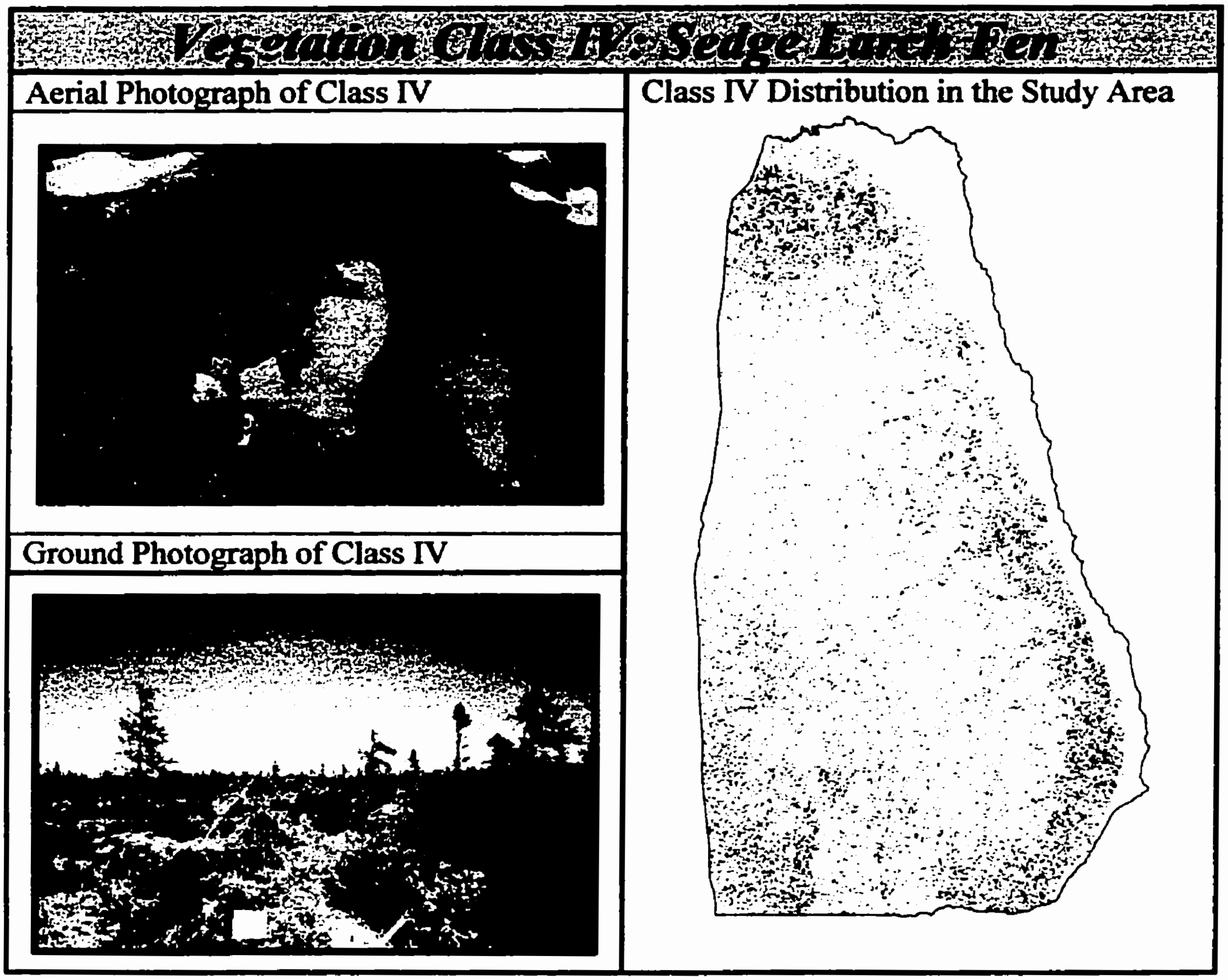
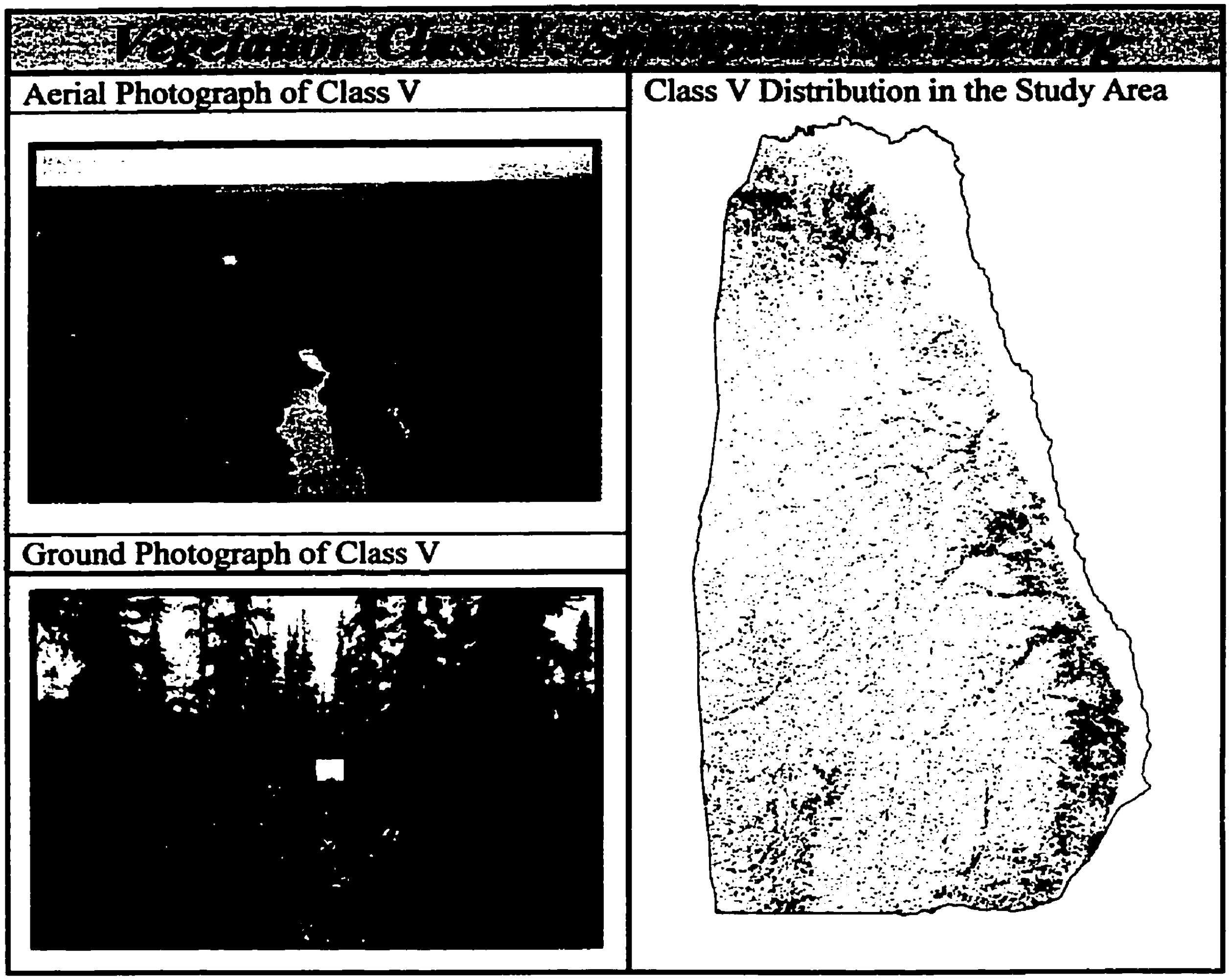


Figure 5.6. Photographs and distribution map of vegetation class IV (Sedge Larch Fen).

Table 5.5. Species composition of Class IV (Sedge Larch Fen), based on 9 sample sites.

	Average	Std Dev	Max	Min
TREE				
Larix laricina	11.1	3.3	20.0	10.0
Picea glauca	0.1	0.3	1.0	0.0
Picea mariana	6.7	7.1	20.0	0.0
SHRUB				
Andromeda polifolia	0.6	0.5	1.0	0.0
Arctostaphylos sp.	0.7	0.5	1.0	0.0
Betula glandulosa	13.3	5.0	20.0	10.0
Dryas integrifolia	0.1	0.3	1.0	0.0
Empetrum nigrum	1.7	3.2	10.0	0.0
Kalmia polifolia	0.1	0.3	1.0	0.0
Ledum decumbens	0.4	0.5	1.0	0.0
Ledum groenlandicum	0.3	0.5	1.0	0.0
Rhododendron lapponicum	1.6	3.2	10.0	0.0
Salix arctophila	0.9	0.3	1.0	0.0
Salix candida	0.4	0.5	1.0	0.0
Salix pedicellaris	0.1	0.3	1.0	0.0
Salix planifolia	1.4	3.2	10.0	0.0
Vaccinium uliginosum	1.8	3.1	10.0	0.0
Vaccinium vitis-idaea	0.1	0.3	1.0	0.0
HERB				
Bartsia alpina	0.4	0.5	1.0	0.0
Carex aquatilis	15.7	15.0	50.0	1.0
Carex saxatilis	0.1	0.3	1.0	0.0
Carex sp.	4.4	7.3	20.0	0.0
Oxycoccus microcarpus	0.2	0.4	1.0	0.0
Parnassia palustris	0.2	0.4	1.0	0.0
Petasites sagittatus	0.1	0.3	1.0	0.0
Pinguicula vulgaris	0.1	0.3	1.0	0.0
Potentilla palustris	0.1	0.3	1.0	0.0
Rubus aucalis	0.1	0.3	1.0	0.0
Rubus chamaemorus	0.2	0.4	1.0	0.0
Scirpus caespitosus	20.0	12.2	30.0	0.0
Smilacina trifolia	0.2	0.4	1.0	0.0
Triglochin maritimum	0.2	0.4	1.0	0.0
Triglochin palustris	0.1	0.3	1.0	0.0
MOSS				
Drepanocladus sp.	5.6	8.8	20.0	0.0
Sphagnum sp.	2.3	6.6	20.0	0.0
Sphagum fuscum	0.2	0.4	1.0	0.0
Tomenthypnum nitens	4.6	10.1	30.0	0.0
LICHEN				
Bryocaulon divergens	0.1	0.3	1.0	0.0
Cetraria cucullata	0.1	0.3	1.0	0.0
Cetraria nivalis	0.3	0.5	1.0	0.0
Cladina mitis	0.1	0.3	1.0	0.0
Cladina rangiferina	0.2	0.4	1.0	0.0
Cladina stellaris	1.2	3.3	10.0	0.0
Evernia mesomorpha	0.1	0.3	1.0	0.0
NON-VASCULAR				
Equisetum sp.	0.2	0.4	1.0	0.0



Aerial Photograph of Class V

Ground Photograph of Class V

Class V Distribution in the Study Area

Figure 5.7. Photographs and distribution map of vegetation class V (*Sphagnum* Spruce Bog).

Table 5.6. Species composition of Class V (*Sphagnum* Spruce Bog), based on 13 sample sites.

	Average	**Std Dev**	**Max**	**Min**
TREE				
Larix laricina	4.9	6.4	20.0	0.0
Picea glauca	1.5	5.5	20.0	0.0
Picea mariana	18.5	9.9	30.0	0.0
SHRUB				
Andromeda polifolia	0.1	0.3	1.0	0.0
Arctostaphylos sp.	0.2	0.4	1.0	0.0
Betula glandulosa	1.8	3.7	10.0	0.0
Empetrum nigrum	2.0	3.6	10.0	0.0
Kalmia polifolia	0.1	0.3	1.0	0.0
Ledum decumbens	1.8	3.7	10.0	0.0
Ledum groenlandicum	11.0	9.3	20.0	0.0
Ribes hudsonianum	0.1	0.3	1.0	0.0
Ribes oxyacanthoides	0.1	0.3	1.0	0.0
Salix arctophila	0.1	0.3	1.0	0.0
Salix candida	0.8	2.8	10.0	0.0
Salix planifolia	6.9	9.5	30.0	0.0
Vaccinium uliginosum	2.6	4.2	10.0	0.0
Vaccinium vitis-idaea	0.6	0.5	1.0	0.0
HERB				
Carex aquatilis	0.8	2.8	10.0	0.0
Drosera rotundifolia	0.1	0.3	1.0	0.0
Eriophorum vaginatum	0.1	0.3	1.0	0.0
Habernaria hyperborea	0.1	0.3	1.0	0.0
Oxycoccus microcarpus	0.5	0.5	1.0	0.0
Petasites sagittatus	0.1	0.3	1.0	0.0
Potentilla palustris	0.1	0.3	1.0	0.0
Ranunculus arbortivis	0.1	0.3	1.0	0.0
Ranunculus pedatifidus	0.1	0.3	1.0	0.0
Rubus chamaemorus	7.8	8.2	20.0	0.0
Smilacina trifolia	2.4	4.4	10.0	0.0
MOSS				
Sphagnum sp.	13.1	14.4	40.0	0.0
Sphagum fuscum	13.8	13.3	30.0	0.0
LICHEN				
Cladina mitis	0.2	0.4	1.0	0.0
Cladina rangiferina	1.2	2.7	10.0	0.0
Cladina stellaris	1.8	3.7	10.0	0.0
Cladonia sp.	0.2	0.4	1.0	0.0
Cladonia chlorophorea	0.1	0.3	1.0	0.0
Cladonia scubruiscula	0.2	0.4	1.0	0.0
Peltigera apthosa	0.2	0.4	1.0	0.0
Evernia mesomorpha	0.5	0.5	1.0	0.0
NON-VASCULAR				
Equisetum arvense	0.8	2.8	10.0	0.0
Equisetum sp.	0.9	2.8	10.0	0.0

Vegetation Class VI: Graminoid Willow Salt Marsh

Aerial Photograph of Class VI

Ground Photograph of Class VI

Class VI Distribution in the Study Area

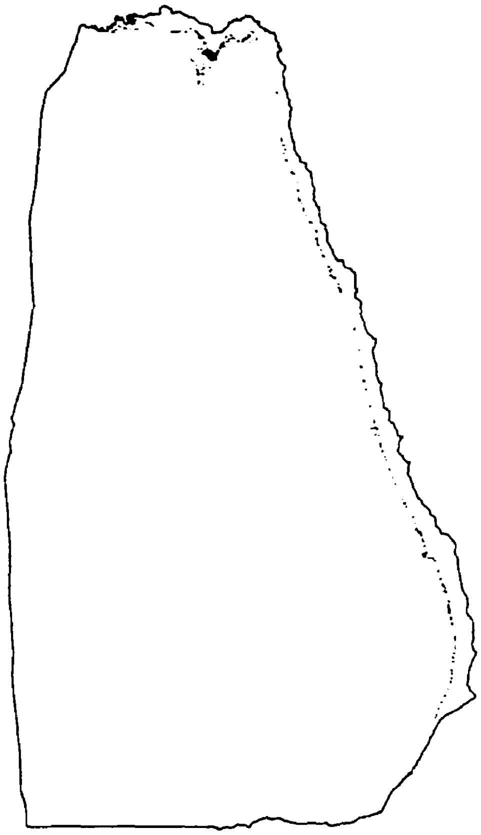


Figure 5.8. Photographs and distribution map of vegetation class VI (Graminoid Willow Salt Marsh).

Table 5.7. Species composition of Class VI (Graminoid Willow Salt Marsh), based on 52 sample sites.

	Average	Std Dev	Max	Min
SHRUB				
Betula glandulosa	2.4	6.1	30.0	0.0
Empetrum nigrum	0.2	1.4	10.0	0.0
Myrica gale	1.2	4.7	20.0	0.0
Salix brachycarpa	7.0	10.0	30.0	0.0
Salix candida	4.2	10.3	50.0	0.0
Salix lanata	0.0	0.2	1.0	0.0
Salix myrtillifolia	3.5	9.7	40.0	0.0
Salix pedicellaris	0.2	1.4	10.0	0.0
Salix planifolia	3.9	8.6	40.0	0.0
Salix reticulata	0.1	0.2	1.0	0.0
HERB				
Achillea nigrescens	0.1	0.3	1.0	0.0
Atriplex glabriuscula	0.3	1.4	10.0	0.0
Carex aquatilis	1.5	7.0	40.0	0.0
Carex subspathacea	4.2	14.4	70.0	0.0
Castilleja raupii	0.1	0.3	1.0	0.0
Elymus arenarius	0.1	0.3	1.0	0.0
Euphrasia arctica	0.0	0.2	1.0	0.0
Festuca sp.	3.8	7.3	30.0	0.0
Hippurus tetraphylla	0.0	0.1	1.0	0.0
Hippurus vulgaris	2.6	10.3	60.0	0.0
Honkenya peploides	5.8	20.6	90.0	0.0
Lomatogonium rotatum	0.0	0.2	1.0	0.0
Matricaria ambigua	0.0	0.2	1.0	0.0
Parnassia palustris	0.2	0.4	1.0	0.0
Petasites sagittatus	0.4	1.4	10.0	0.0
Plantago maritima	0.8	4.2	30.0	0.0
Poa sp.	0.0	0.1	1.0	0.0
Potentilla egedei	0.8	3.1	20.0	0.0
Potentilla palustris	0.2	1.4	10.0	0.0
Puccinellia sp.	5.3	8.9	30.0	0.0
Pyrola grandiflora	0.0	0.2	1.0	0.0
Ranunculus cymbalaria	0.6	2.4	10.0	0.0
Ranunculus pedatifidus	0.0	0.1	1.0	0.0
Ranunculus purshii	0.0	0.1	1.0	0.0
Rhinanthus borealis	0.4	1.4	10.0	0.0
Rubus aucalis	0.0	0.1	1.0	0.0
Rumex occidentalis	1.1	5.5	40.0	0.0
Salicornia borealis	10.5	18.8	60.0	0.0
Senecio congestus	0.5	1.4	10.0	0.0
Senecio pauperculus	0.1	0.3	1.0	0.0
Stellaria logipipes	0.0	0.1	1.0	0.0
LICHEN				
Cetraria nivalis	0.0	0.1	1.0	0.0

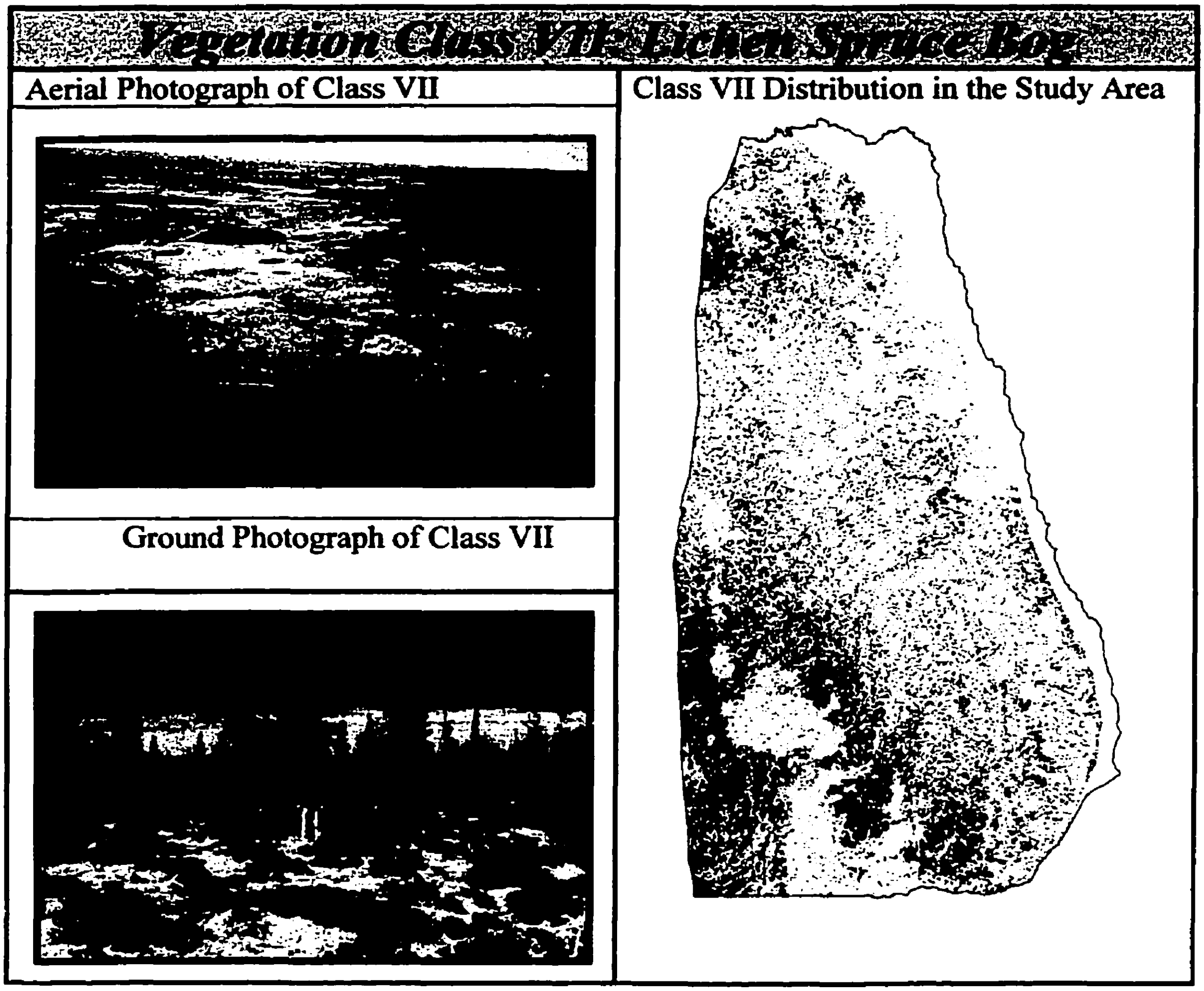


Figure 5.9. Photographs and distribution map of vegetation class VII (Lichen Spruce Bog).

Table 5.8. Species composition of Class VII (Lichen spruce bog), based on 49 sample sites.

	Average	Std Dev	Max	Min
TREE				
Larix laricina	2.9	4.8	20.0	0.0
Picea glauca	4.6	9.1	30.0	0.0
Picea mariana	16.1	11.3	40.0	0.0
SHRUB				
Andromeda polifolia	0.0	0.2	1.0	0.0
Arctostaphylos sp.	0.3	1.4	10.0	0.0
Betula glandulosa	0.8	3.2	20.0	0.0
Chamaedaphne calyculata	0.0	0.1	1.0	0.0
Dryas integrifolia	0.0	0.2	1.0	0.0
Empetrum nigrum	3.3	5.2	20.0	0.0
Juniperus communis	0.2	1.4	10.0	0.0
Kalmia polifolia	0.1	0.2	1.0	0.0
Ledum decumbens	7.6	8.9	30.0	0.0
Ledum groenlandicum	16.1	9.0	30.0	0.0
Ribes oxyacanthoides	0.0	0.1	1.0	0.0
Salix arctophila	0.0	0.2	1.0	0.0
Salix myrtillifolia	0.0	0.1	1.0	0.0
Salix pedicellaris	0.0	0.1	1.0	0.0
Salix planifolia	0.2	0.4	1.0	0.0
Salix reticulata	0.0	0.2	1.0	0.0
Salix sp.	0.0	0.1	1.0	0.0
Sheperdia canadensis	0.1	0.2	1.0	0.0
Vaccinium uliginosum	1.6	3.6	20.0	0.0
Vaccinium vitis-idaea	0.9	1.4	10.0	0.0
HERB				
Carex aquatilis	0.2	1.4	10.0	0.0
Carex sp.	0.1	0.2	1.0	0.0
Epilobium angustifolium	0.1	0.3	1.0	0.0
Geocaulon lividum	0.3	1.4	10.0	0.0
Oxycoccus microcarpus	0.1	0.3	1.0	0.0
Rubus chamaemorus	1.4	2.6	10.0	0.0
Scirpus caespitosus	0.0	0.1	1.0	0.0
Smilacina trifolia	0.0	0.1	1.0	0.0
Solidago multiradiata	0.0	0.1	1.0	0.0
MOSS				
Sphagnum sp.	1.4	3.9	20.0	0.0
Sphagum fuscum	2.3	5.1	20.0	0.0
LICHEN				
Cetraria cucullata	0.0	0.1	1.0	0.0
Cetraria nivalis	2.2	4.3	20.0	0.0
Cladina mitis	5.7	7.4	30.0	0.0
Cladina rangiferina	10.8	7.2	30.0	0.0
Cladina stellaris	25.1	10.8	60.0	10.0
Cladonia sp.	0.1	0.4	1.0	0.0
Cladonia borealis	0.1	0.3	1.0	0.0
Cladonia chlorophorea	0.1	0.3	1.0	0.0
Cladonia scubruiscula	0.2	0.4	1.0	0.0
Stereocaulon alpinum	0.0	0.1	1.0	0.0
Peltigera apthosa	0.1	0.2	1.0	0.0
NON-VASCULAR				
Lycopodium sp.	0.0	0.1	1.0	0.0

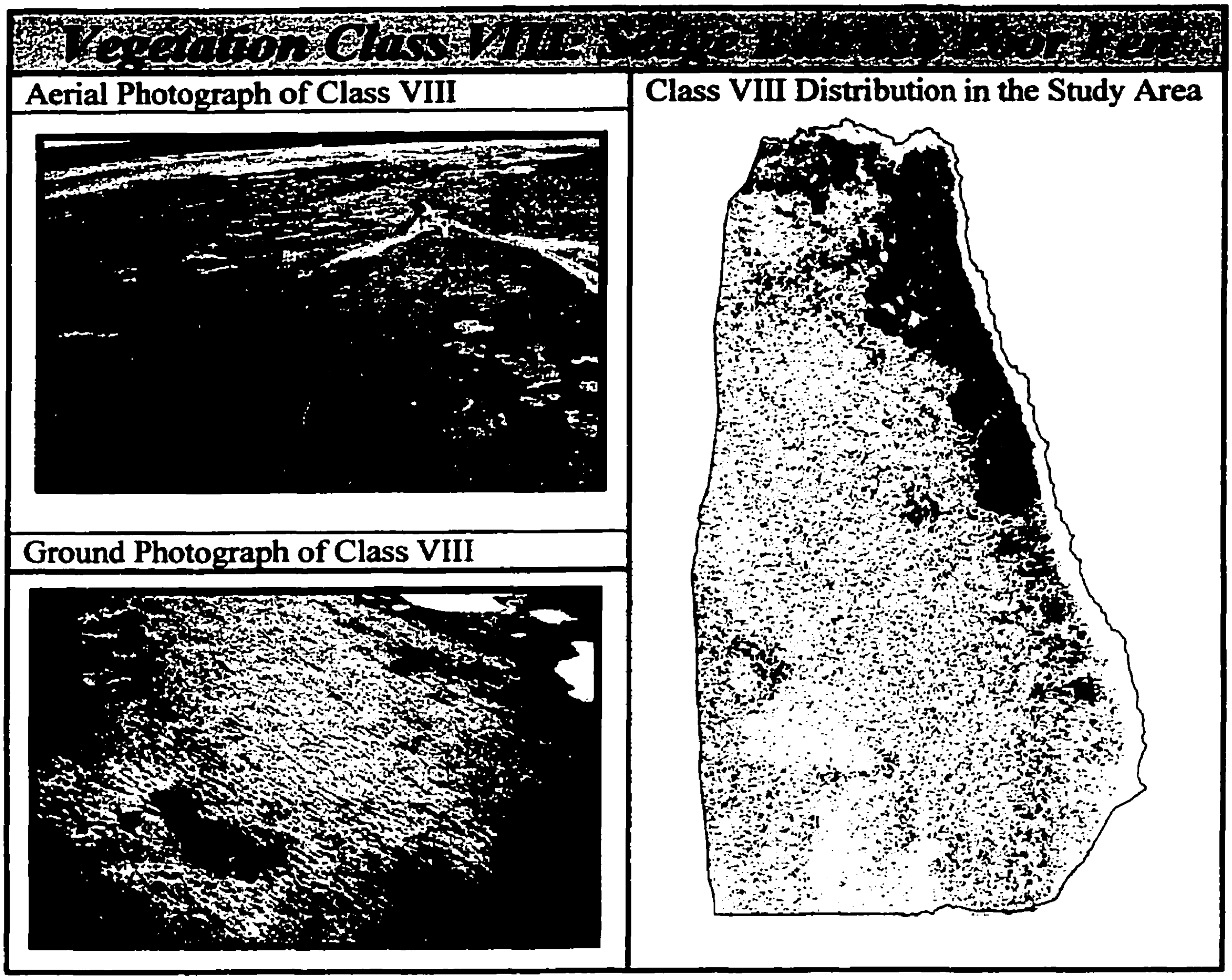
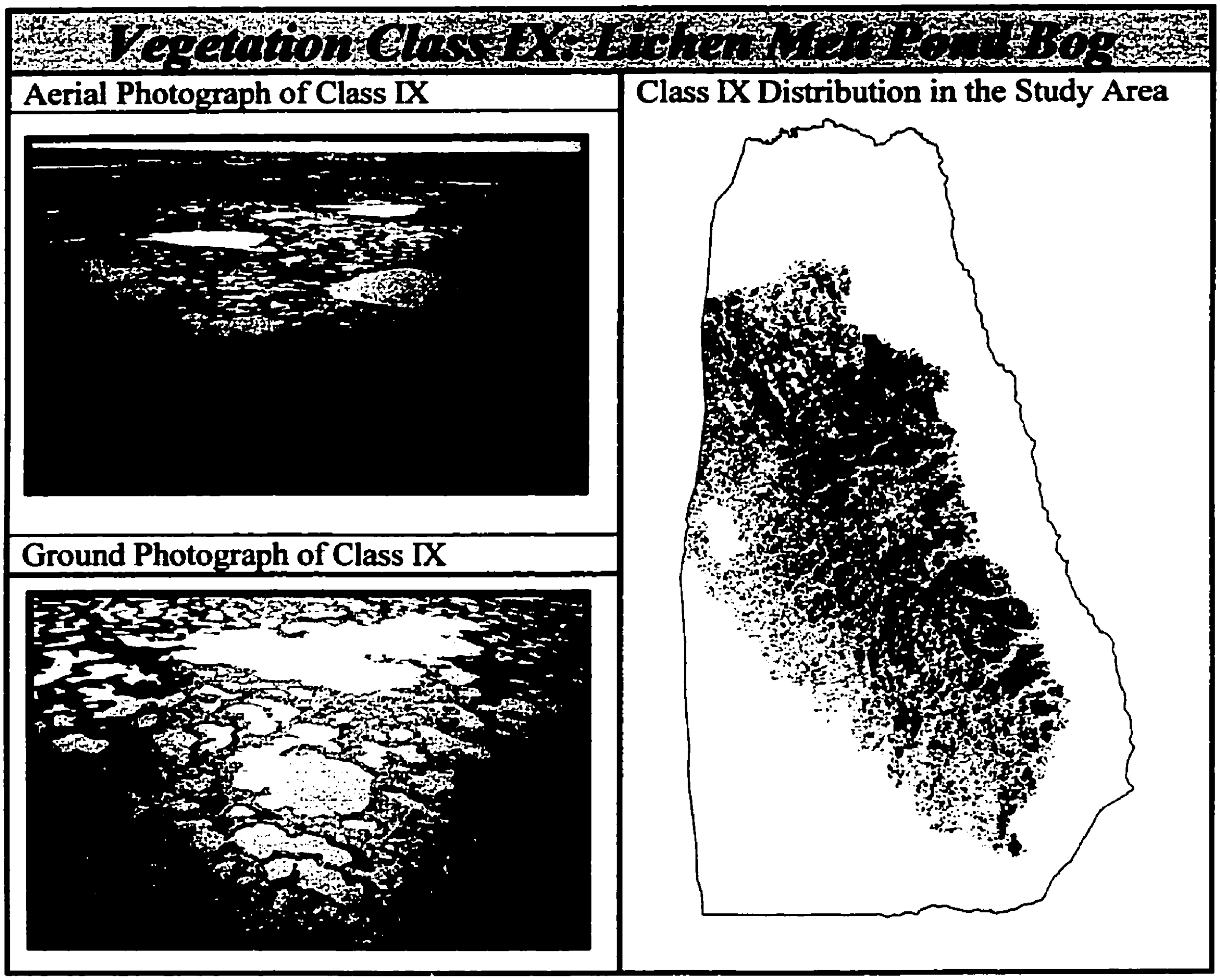


Figure 5.10. Photographs and distribution map of vegetation class VIII (Sedge Bulrush Poor Fen).

Table 5.9. Species composition of Class VIII (Sedge bulrush poor fen), based on 46 sample sites.

	Average	Std Dev	Max	Min
TREE				
Larix laricina	0.0	0.2	1.0	0.0
SHRUB				
Andromeda polifolia	0.9	2.5	10.0	0.0
Arctostaphylos sp.	0.4	1.5	10.0	0.0
Betula glandulosa	3.3	5.8	30.0	0.0
Dryas integrifolia	0.3	1.5	10.0	0.0
Empetrum nigrum	0.0	0.2	1.0	0.0
Ledum decumbens	0.0	0.1	1.0	0.0
Myrica gale	0.0	0.2	1.0	0.0
Rhododendron lapponicum	0.6	2.1	10.0	0.0
Salix arctophila	4.4	6.4	20.0	0.0
Salix brachycarpa	0.0	0.1	1.0	0.0
Salix candida	0.1	0.2	1.0	0.0
Salix lanata	0.1	0.3	1.0	0.0
Salix pedicellaris	0.0	0.1	1.0	0.0
Salix reticulata	1.4	3.4	10.0	0.0
Salix sp.	0.5	2.9	20.0	0.0
Vaccinium uliginosum	0.4	1.5	10.0	0.0
Vaccinium vitis-idaea	0.0	0.1	1.0	0.0
HERB				
Bartsia alpina	0.1	0.3	1.0	0.0
Carex aquatilis	33.3	18.5	70.0	0.0
Carex bicolor	0.2	1.5	10.0	0.0
Carex capillaris	3.9	8.5	30.0	0.0
Carex capitata	0.0	0.1	1.0	0.0
Carex limosa	0.0	0.2	1.0	0.0
Carex saxatilis	0.1	0.2	1.0	0.0
Carex scirpoidea	0.9	4.1	20.0	0.0
Carex vaginata	0.9	4.1	20.0	0.0
Carex sp.	1.1	6.0	40.0	0.0
Epilobium angustifolium	0.0	0.1	1.0	0.0
Eriophorum vaginatum	0.0	0.1	1.0	0.0
Pedicularis sudetica	0.2	0.4	1.0	0.0
Petasites sagittatus	0.0	0.1	1.0	0.0
Pinguicula vulgaris	0.1	0.3	1.0	0.0
Poa sp.	0.0	0.1	1.0	0.0
Potentilla palustris	0.1	0.3	1.0	0.0
Ranunculus pedatifidus	0.0	0.1	1.0	0.0
Saxifraga hirculus	0.0	0.1	1.0	0.0
Scirpus caespitosus	17.0	17.0	60.0	0.0
MOSS				
Brown Moss	1.5	6.0	30.0	0.0
Drepanocladus	1.5	6.3	30.0	0.0
Scorpidium sp.	0.0	0.1	1.0	0.0
Sphagnum sp.	11.5	13.7	50.0	0.0
Tomenthypnum nitens	0.0	0.2	1.0	0.0
LICHEN				
Alectoria ochroleucra	0.0	0.1	1.0	0.0
Cetraria nivalis	0.1	0.2	1.0	0.0


Vegetation Class IX: Lichen Melt Pond Bog

Aerial Photograph of Class IX

Ground Photograph of Class IX

Class IX Distribution in the Study Area

Figure 5.11. Photographs and distribution map of vegetation class IX (Lichen Melt Pond Bog).

Table 5.10. Species composition of Class IX (Lichen Melt Pond Bog), based on 3 sample sites.

	Average	Std Dev	Max	Min
SHRUB				
Andromeda polifolia	0.7	0.6	1.0	0.0
Betula glandulosa	0.3	0.6	1.0	0.0
Empetrum nigrum	10.0	0.0	10.0	10.0
Ledum decumbens	10.0	0.0	10.0	10.0
Vaccinium vitis-idaea	0.3	0.6	1.0	0.0
HERB				
Carex capillaris	13.3	11.5	20.0	0.0
Drosera rotundifolia	0.3	0.6	1.0	0.0
Menyanthes trifoliata	3.3	5.8	10.0	0.0
Potentilla palustris	0.3	0.6	1.0	0.0
Rubus chamaemorus	1.0	0.0	1.0	1.0
Scirpus hudsonianus	0.3	0.6	1.0	0.0
MOSS				
Sphagnum sp.	13.3	11.5	20.0	0.0
Sphagum fuscum	6.7	11.5	20.0	0.0
LICHEN				
Alectoria ochroleucra	0.7	0.6	1.0	0.0
Bryocaulon divergens	1.0	0.0	1.0	1.0
Cetraria nivalis	7.0	5.2	10.0	1.0
Cladina mitis	10.0	0.0	10.0	10.0
Cladina rangiferina	4.0	5.2	10.0	1.0
Cladina stellaris	16.7	5.8	20.0	10.0

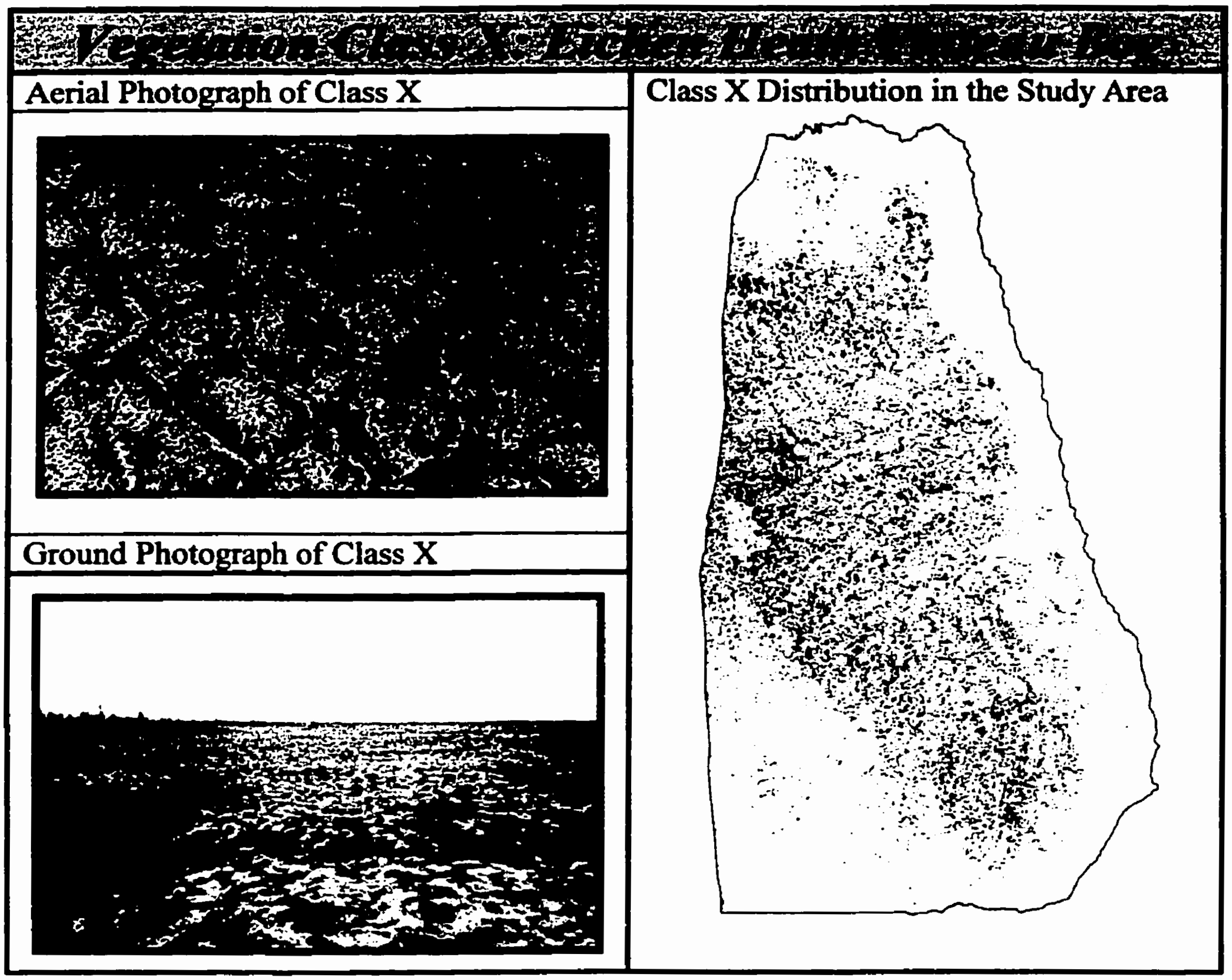


Figure 5.12. Photographs and distribution map of vegetation class X (Lichen Heath Plateau Bog).

Table 5.11. Species composition of Class X (Lichen Peat Plateau Bog), based on 52 sample sites.

	Average	**Std Dev**	**Max**	**Min**
TREE				
Picea mariana	0.0	0.2	1.0	0.0
SHRUB				
Andromeda polifolia	0.1	0.3	1.0	0.0
Arctostaphylos sp.	0.6	2.4	10.0	0.0
Betula glandulosa	0.2	1.4	10.0	0.0
Dryas integrifolia	0.4	2.0	10.0	0.0
Empetrum nigrum	7.6	5.8	20.0	0.0
Kalmia polifolia	0.1	0.2	1.0	0.0
Ledum decumbens	14.6	7.0	30.0	1.0
Rhododendron lapponicum	0.8	2.8	10.0	0.0
Salix arctophila	0.0	0.1	1.0	0.0
Salix lanata	0.0	0.1	1.0	0.0
Salix reticulata	0.2	1.4	10.0	0.0
Salix sp.	0.2	1.4	10.0	0.0
Vaccinium uliginosum	1.8	3.7	10.0	0.0
Vaccinium vitis-idaea	1.7	2.9	10.0	0.0
HERB				
Carex vaginata	0.2	1.4	10.0	0.0
Carex sp.	1.1	3.0	10.0	0.0
Eriophorum angustifolium	0.0	0.1	1.0	0.0
Eriophorum vaginatum	0.0	0.2	1.0	0.0
Rubus chamaemorus	3.0	4.5	20.0	0.0
Scirpus caespitosus	0.2	1.4	10.0	0.0
MOSS				
Sphagnum sp.	2.7	5.9	30.0	0.0
Sphagum fuscum	2.0	4.8	20.0	0.0
Tomenthypnum nitens	0.0	0.2	1.0	0.0
LICHEN				
Alectoria ochroleucra	3.0	4.7	20.0	0.0
Bryocaulon divergens	0.8	2.0	10.0	0.0
Cetraria cuculata	1.5	3.5	10.0	0.0
Cetraria nivalis	14.5	6.1	30.0	10.0
Cetraria laevigata	0.1	0.2	1.0	0.0
Cladina mitis	11.2	8.7	30.0	0.0
Cladina rangiferina	8.4	8.0	30.0	0.0
Cladina stellaris	23.5	13.6	50.0	0.0
Cladonia sp.	1.9	4.2	20.0	0.0
Cladonia borealis	0.1	0.3	1.0	0.0
Stereocaulon alpinum	0.3	1.4	10.0	0.0

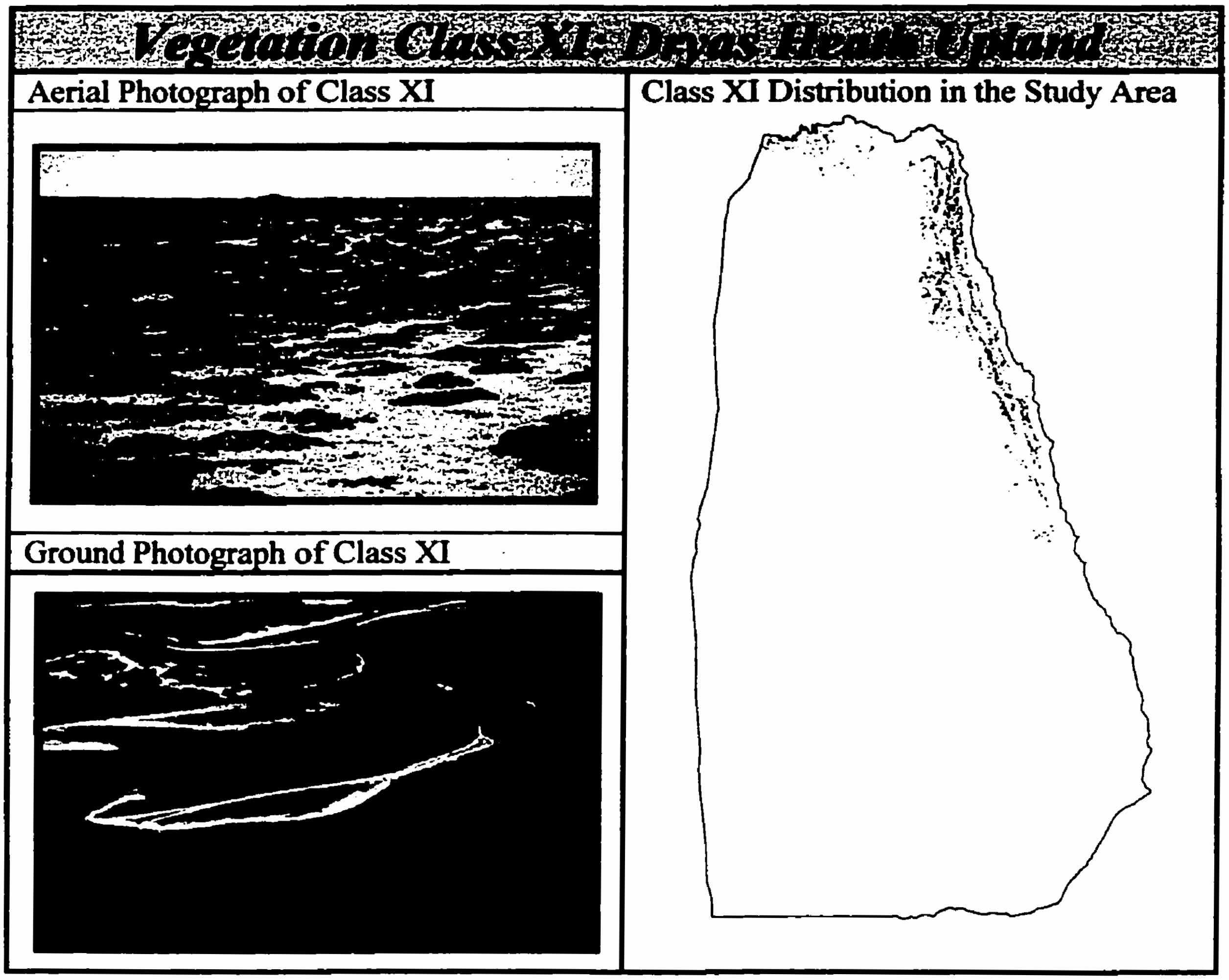


Figure 5.13. Photographs and distribution map of vegetation class XI (Dryas Heath Upland).

Table 5.12. Species composition of Class XI (Dryas Heath), based on 69 sample sites.

	Average	Std Dev	Max	Min
SHRUB				
Andromeda polifolia	0.2	1.2	10.0	0.0
Arctostaphylos sp.	4.4	8.5	60.0	0.0
Betula glandulosa	0.3	2.4	20.0	0.0
Dryas integrifolia	37.7	15.3	70.0	0.0
Empetrum nigrum	0.4	2.4	20.0	0.0
Rhododendron lapponicum	5.6	9.5	60.0	0.0
Salix arctophila	0.0	0.1	1.0	0.0
Salix brachycarpa	0.8	6.1	50.0	0.0
Salix candida	0.0	0.1	1.0	0.0
Salix lanata	0.0	0.2	1.0	0.0
Salix myrtillifolia	0.0	0.1	1.0	0.0
Salix pedicellaris	0.3	1.7	10.0	0.0
Salix reticulata	0.4	2.4	20.0	0.0
Salix vestita	0.0	0.1	1.0	0.0
Sheperdia canadensis	1.0	2.6	10.0	0.0
Vaccinium uliginosum	0.9	3.1	20.0	0.0
Vaccinium vitis-idaea	0.0	0.2	1.0	0.0
HERB				
Achillea nigrescens	0.0	0.1	1.0	0.0
Astragalus alpinus	0.0	0.2	1.0	0.0
Bartsia alpina	0.0	0.2	1.0	0.0
Carex aquatilis	0.4	2.7	20.0	0.0
Carex capillaris	0.4	3.6	30.0	0.0
Carex capitata	0.0	0.1	1.0	0.0
Carex glacialis	1.0	3.3	20.0	0.0
Carex rupestris	1.1	4.3	20.0	0.0
Elymus arenarius	0.4	2.4	20.0	0.0
Hedysarum mackenzii	0.0	0.1	1.0	0.0
Lesquerella arctica	0.0	0.1	1.0	0.0
Oxytropis campestris	0.0	0.1	1.0	0.0
Pinguicula vulgaris	0.0	0.2	1.0	0.0
Poa sp.	0.0	0.2	1.0	0.0
Polygonum viviparum	0.0	0.1	1.0	0.0
Potentilla palustris	0.0	0.1	1.0	0.0
Pyrola grandiflora	0.0	0.1	1.0	0.0
Rubus aucalis	0.0	0.1	1.0	0.0
Saxifraga oppositifolia	0.5	1.7	10.0	0.0
Saxifraga tricuspidata	0.5	1.7	10.0	0.0
Scirpus caespitosus	0.0	0.1	1.0	0.0
Tofieldia pusilla	0.0	0.1	1.0	0.0
LICHEN				
Alectoria ochroleucra	0.2	1.2	10.0	0.0
Bryocaulon divergens	0.1	0.3	1.0	0.0
Cetraria cuculata	0.2	1.2	10.0	0.0
Cetraria nivalis	3.3	6.8	30.0	0.0
Cetraria laevigata	0.3	1.7	10.0	0.0
Cladina mitis	0.3	1.7	10.0	0.0
Cladina rangiferina	0.0	0.1	1.0	0.0
Cladina stellaris	0.2	1.2	10.0	0.0
Stereocaulon alpinum	0.3	2.4	20.0	0.0

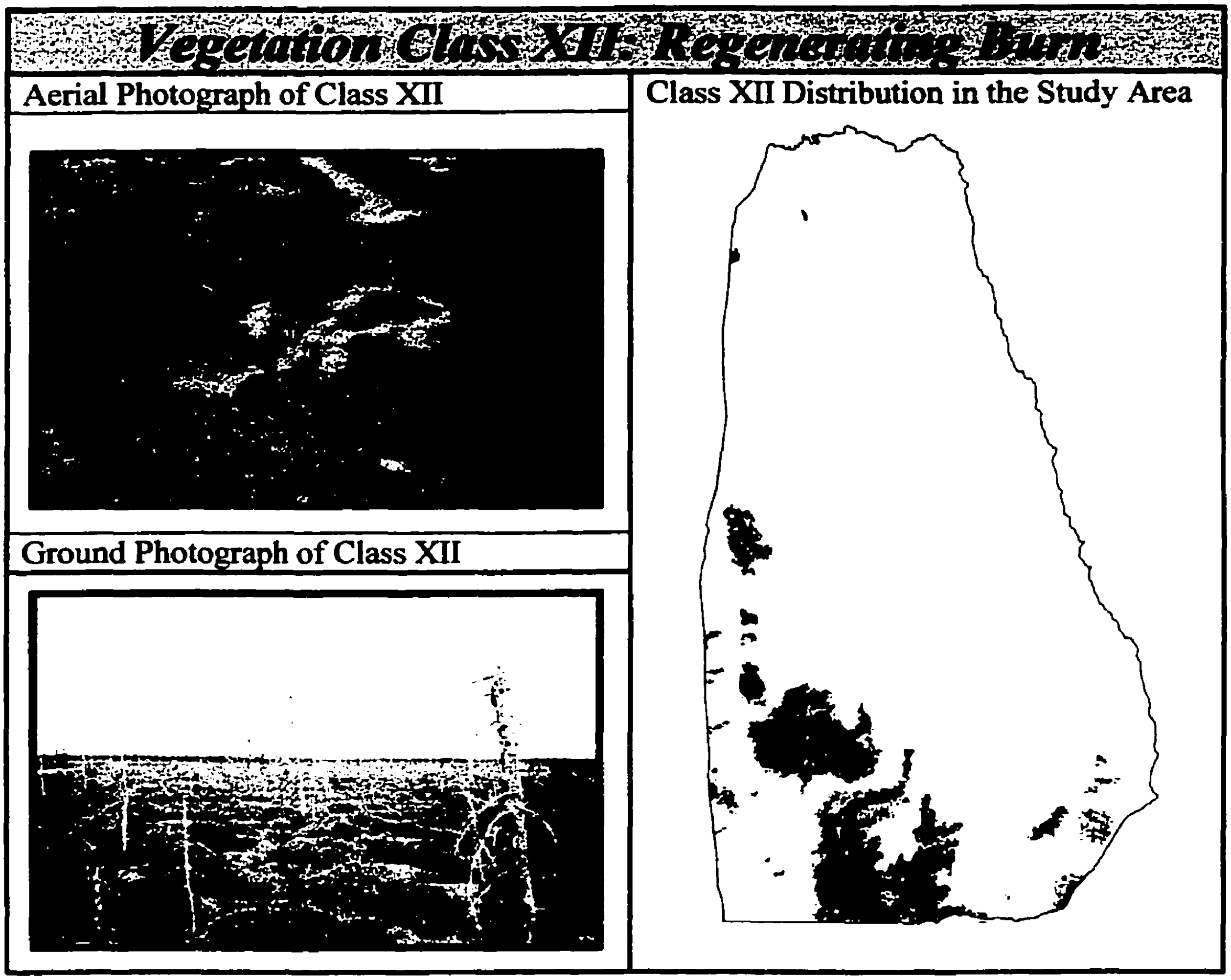


Figure 5.14. Photographs and distribution map of vegetation class XII (Regenerating Burn).

Table 5.13. Species composition of Class XII (Regenerating Burn), based on 18 sample sites.

	Average	Std Dev	Max	Min
TREE				
Larix laricina	0.1	0.2	1.0	0.0
Picea glauca	0.1	0.3	1.0	0.0
Picea mariana	1.2	2.3	10.0	0.0
Populus tremuloides	0.1	0.2	1.0	0.0
SHRUB				
Andromeda polifolia	0.1	0.3	1.0	0.0
Betula glandulosa	5.0	13.4	50.0	0.0
Chamaedaphne calyculata	0.1	0.2	1.0	0.0
Empetrum nigrum	0.7	2.3	10.0	0.0
Kalmia polifolia	0.1	0.3	1.0	0.0
Ledum decumbens	8.5	13.3	40.0	0.0
Ledum groenlandicum	21.7	11.9	40.0	0.0
Ribes hudsonianum	0.1	0.2	1.0	0.0
Rosa acicularis	0.6	2.4	10.0	0.0
Salix arctophila	0.6	2.4	10.0	0.0
Salix lanata	0.1	0.2	1.0	0.0
Salix maccalianna	0.1	0.2	1.0	0.0
Sheperdia canadensis	0.6	2.4	10.0	0.0
Vaccinium uliginosum	2.8	4.0	10.0	0.0
Vaccinium vitis-idaea	2.3	4.9	20.0	0.0
Viburnum edule	0.1	0.3	1.0	0.0
HERB				
Carex aquatilis	0.6	2.4	10.0	0.0
Epilobium angustifolium	1.3	3.2	10.0	0.0
Oxycoccus microcarpus	0.2	0.4	1.0	0.0
Potentilla fruticosa	0.1	0.2	1.0	0.0
Rubus chamaemorus	8.0	7.9	20.0	0.0
Solidago multiradiata	0.1	0.2	1.0	0.0
MOSS				
Sphagnum sp.	1.7	5.1	20.0	0.0
Sphagum fuscum	3.5	5.9	20.0	0.0
LICHEN				
Cetraria nivalis	0.1	0.2	1.0	0.0
Cladina mitis	7.3	8.2	30.0	0.0
Cladina rangiferina	0.1	0.3	1.0	0.0
Cladina stellaris	0.2	0.4	1.0	0.0
Cladonia sp.	2.8	5.7	20.0	0.0
Cladonia borealis	0.2	0.4	1.0	0.0
Cladonia chlorophorea	4.0	6.0	20.0	0.0
Cladonia scubruiscula	0.2	0.4	1.0	0.0
Stereocaulon alpinum	0.7	2.4	10.0	0.0

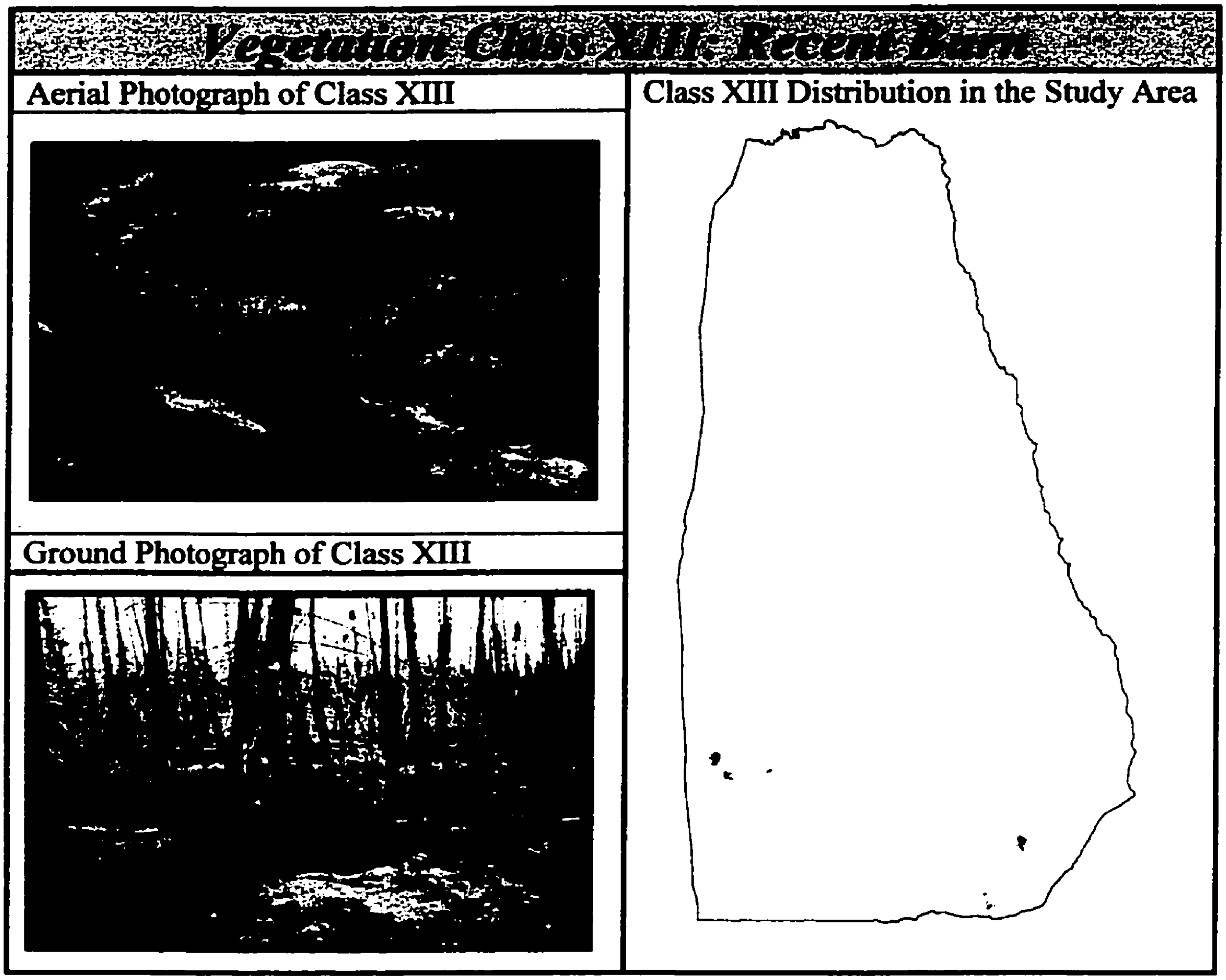


Figure 5.15. Photographs and distribution map of vegetation class XIII (Recent Burn).

Table 5.14. Species composition of Class XIII (Recent Burn), based on 24 sample sites.

	Average	Std Dev	Max	Min
SHRUB				
Betula glandulosa	2.5	6.8	30.0	0.0
Kalmia polifolia	0.0	0.2	1.0	0.0
Ledum decumbens	0.0	0.2	1.0	0.0
Ledum groenlandicum	2.6	6.0	20.0	0.0
Ribes hudsonianum	0.0	0.2	1.0	0.0
Ribes oxyacanthoides	0.0	0.2	1.0	0.0
Salix maccalianna	0.9	2.8	10.0	0.0
Salix planifolia	1.3	3.4	10.0	0.0
Salix sp.	0.9	2.8	10.0	0.0
Vaccinium uliginosum	0.0	0.2	1.0	0.0
Vaccinium vitis-idaea	0.1	0.3	1.0	0.0
HERB				
Calamagrostis neglecta	0.5	2.0	10.0	0.0
Corydalis sempervirens	1.0	2.8	10.0	0.0
Epilobium angustifolium	6.3	7.7	20.0	0.0
Oxycoccus microcarpus	0.0	0.2	1.0	0.0
Rubus chamaemorus	2.5	6.1	20.0	0.0
Smilacina trifolia	0.1	0.3	1.0	0.0
Solidago multiradiata	0.8	2.8	10.0	0.0
MOSS				
Sphagum fuscum	0.9	4.1	20.0	0.0

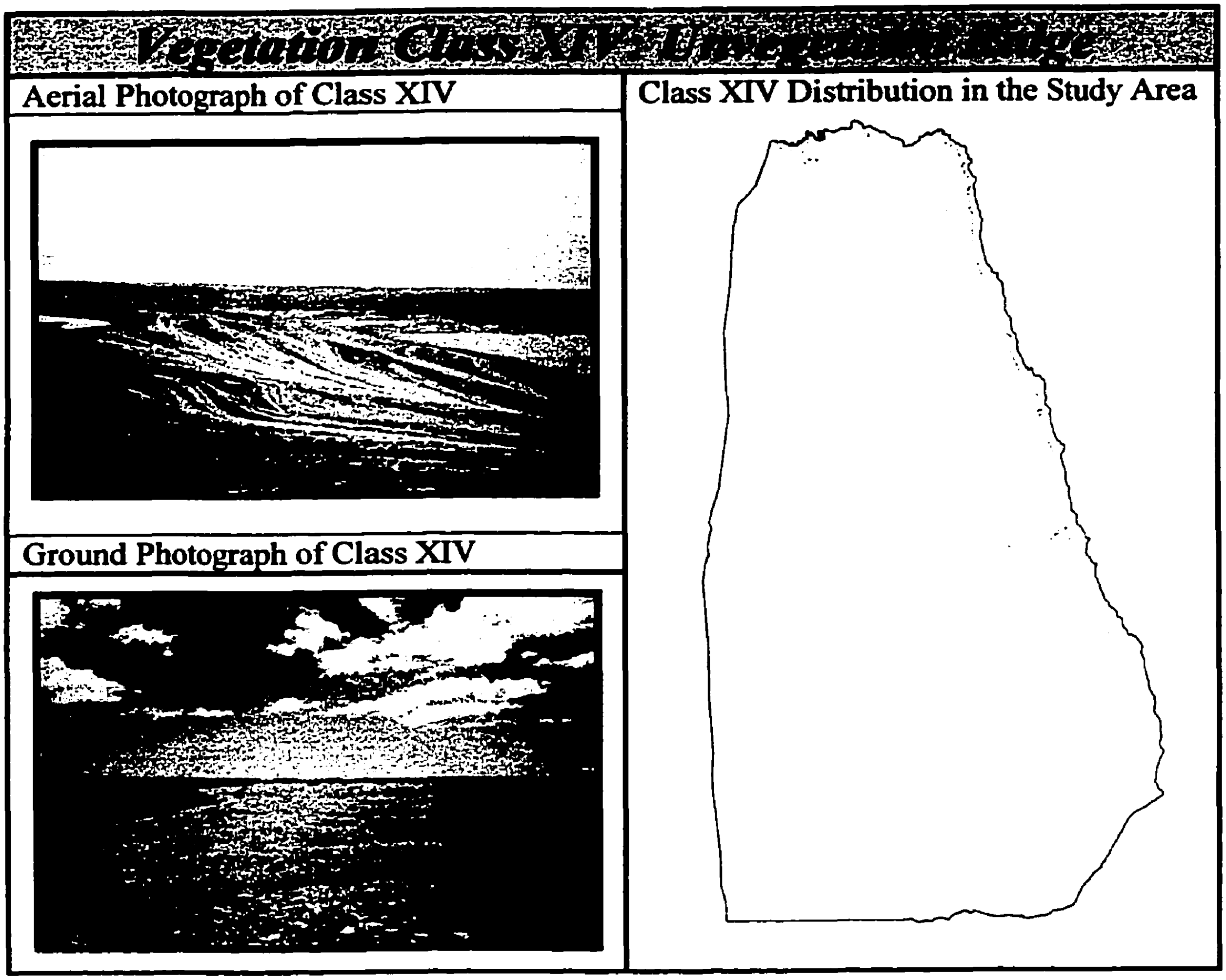


Figure 5.16. Photographs and distribution map of vegetation class XIV (Unvegetated Ridge).

Table 5.15. Species composition of Class XIV (Unvegetated Ridge), based on 28 sample sites.

	Average	Std Dev	Max	Min
SHRUB				
Arctostaphylos sp.	0.2	0.4	1.0	0.0
Betula glandulosa	0.4	1.9	10.0	0.0
Dryas integrifolia	13.2	12.8	30.0	0.0
Empetrum nigrum	0.0	0.2	1.0	0.0
Rhododendron lapponicum	0.0	0.2	1.0	0.0
Salix candida	0.0	0.2	1.0	0.0
Salix lanata	0.0	0.2	1.0	0.0
Salix planifolia	0.0	0.2	1.0	0.0
Sheperdia canadensis	0.5	1.9	10.0	0.0
HERB				
Androsace septrionalis	0.1	0.3	1.0	0.0
Carex glacialis	0.2	0.4	1.0	0.0
Carex rupestris	0.0	0.2	1.0	0.0
Elymus arenarius	0.1	0.4	1.0	0.0
Epilobium angustifolium	0.0	0.2	1.0	0.0
Honkenya peploides	0.1	0.4	1.0	0.0
Pedicularis sudedica	0.0	0.2	1.0	0.0
Saxifraga oppositifolia	0.1	0.3	1.0	0.0
Saxifraga tricuspidata	0.5	1.9	10.0	0.0
LICHEN				
Cetraria nivalis	0.4	0.5	1.0	0.0

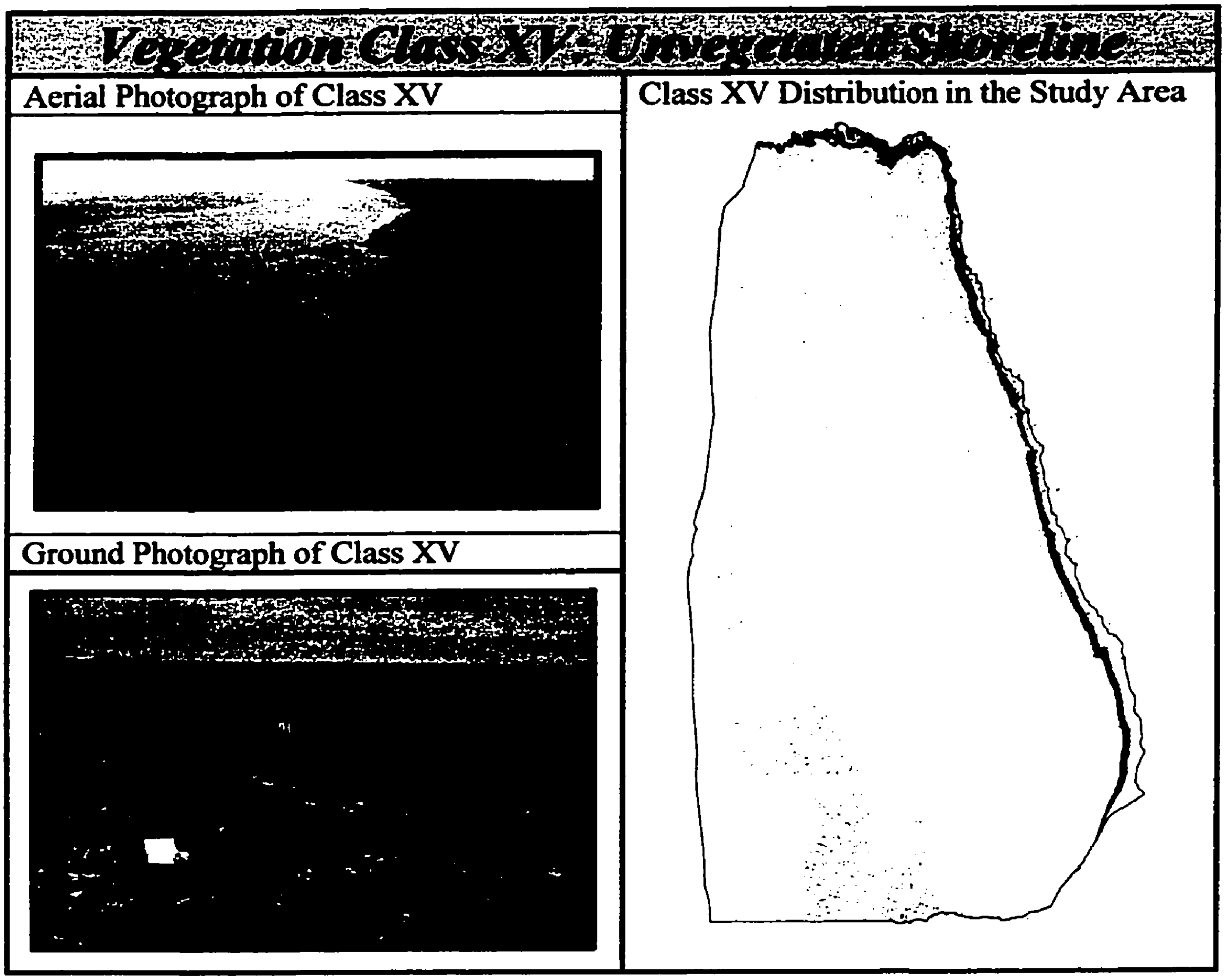


Figure 5.17. Photographs and distribution map of vegetation class XV (Unvegetated Shoreline).

Table 5.16. Species composition of Class XV (Unvegetated Shoreline), based on 55 sample sites.

	Average	Std Dev	Max	Min
HERB				
Carex palacea	0.1	0.4	1.0	0.0
Carex subspathacea	1.4	3.8	10.0	0.0
Hippurus vulgaris	3.0	7.5	20.0	0.0
Puccinellia sp.	3.3	4.6	10.0	0.0
Senecio congestus	0.3	0.5	1.0	0.0

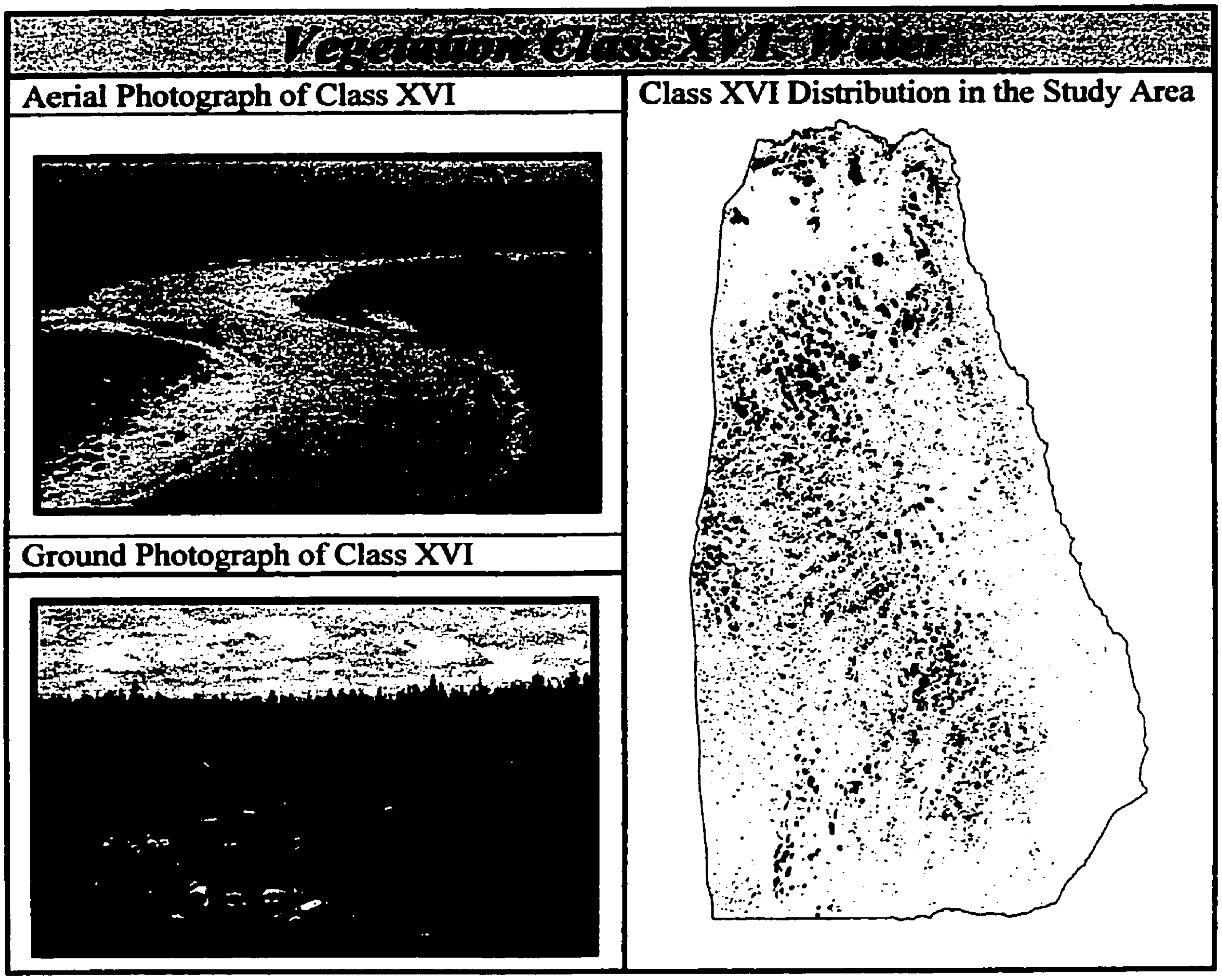


Figure 5.17. Photographs and distribution map of vegetation class XVI (Water).

Wetland Map

The wetland map indicates the distribution of the two major wetland types in the study area; bog and fen (Fig. 5.18). This map shows a dominance of fen vegetation along the Hudson Bay coast and the predominance of bog vegetation inland. Transect data from the 30 x 30 pixel blocks indicates a gradient of vegetation inland from the coast that represents increasing relative frequency of bog vegetation (Fig. 5.19) and a corresponding decreasing relative frequency of fen vegetation.

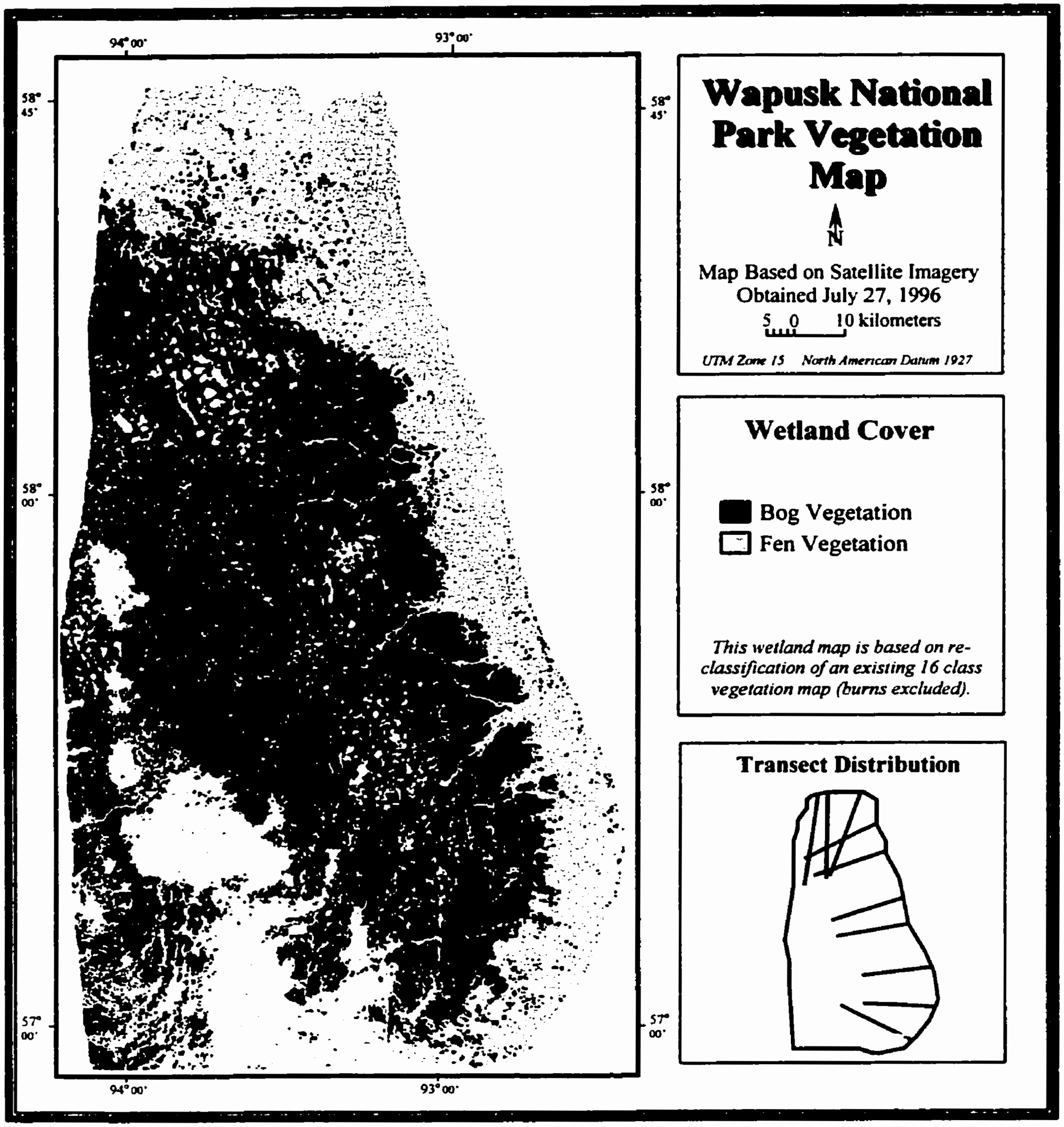


Figure 5.19. Wetland vegetation map of the study area depicting the relative distribution of bog and fen vegetation.

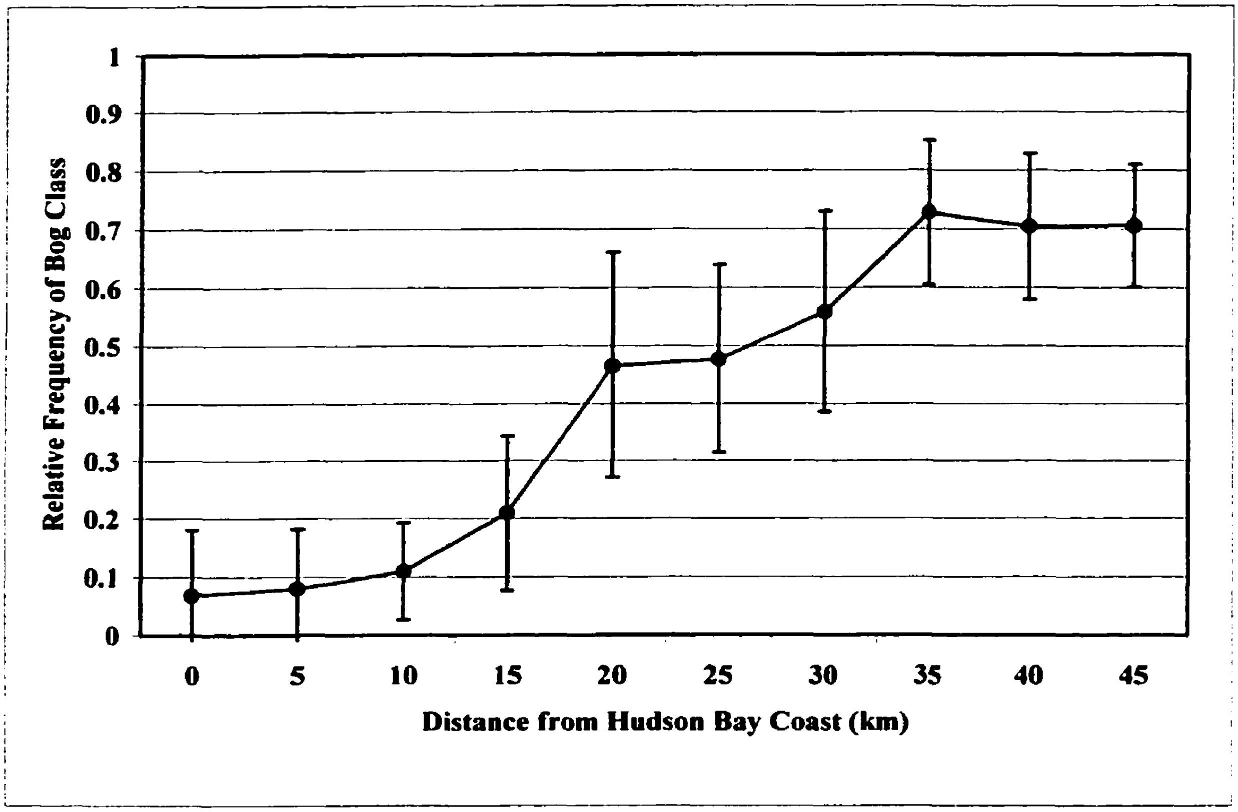


Figure 5.20. Mean relative frequency (± SE) of the bog vegetation class for ten 900 x 900m plots situated in each of ten 45 km transects on the wetland vegetation map running perpendicular to the Hudson Bay coast.

Lichen Cover Map

The lichen cover map for the study area is presented in Fig. 5.20. Overall, the map was 97% accurate, though this varied among classes, ranging from 92% to 99% (Table 5.19).

Burn Distribution

The fires in the study area are quite complex with evidence of overlapping, multiple-age burns. The extent of fires in the region in the current map indicates that fires are much more extensive in areas farthest from the Hudson Bay Coast in the southwest corner of the study area.

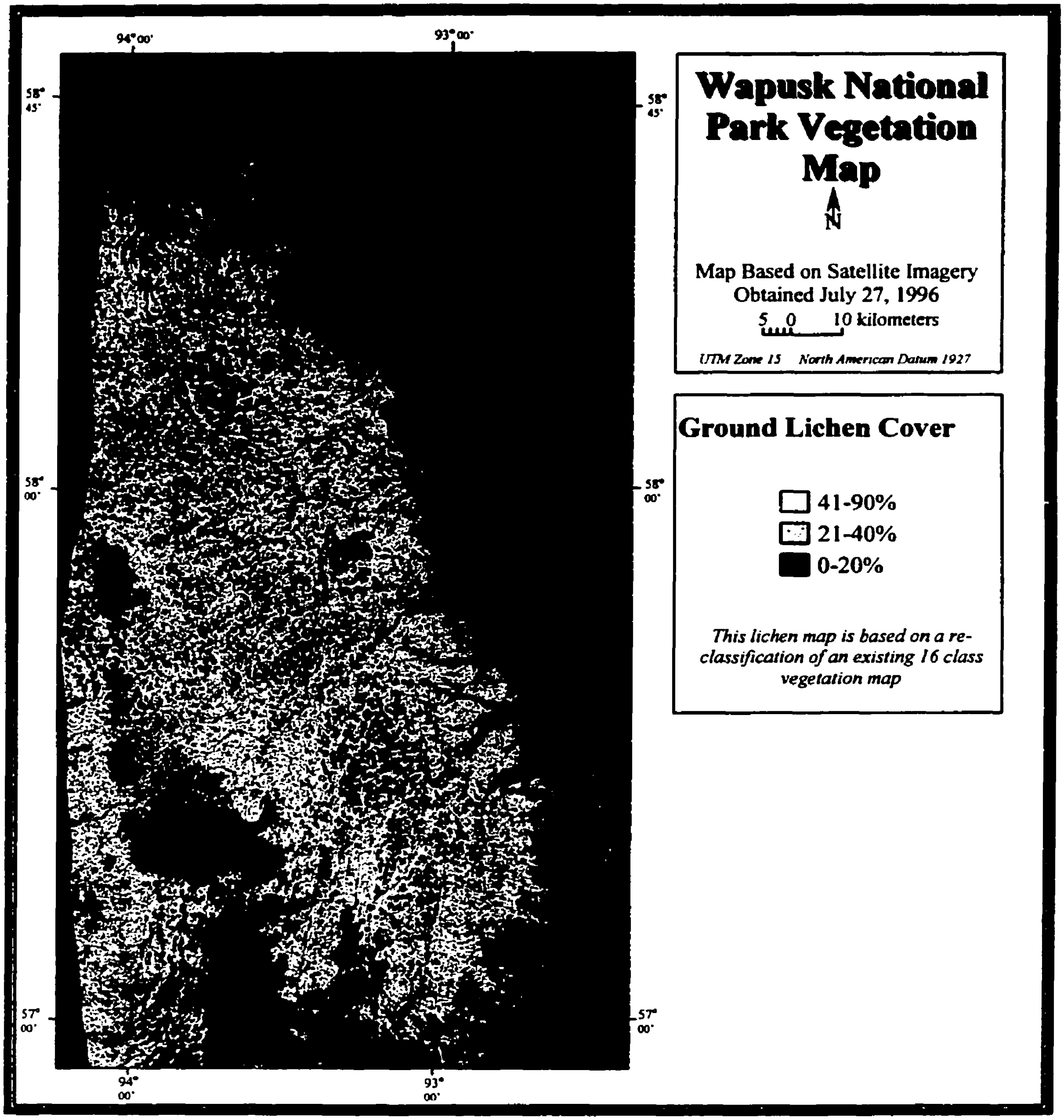


Figure 5.21. Lichen cover map of the study area depicting the relative distribution of lichens.

Table 5.17 Accuracy assessment error matrix for the 3-class lichen cover map, based on 717 ground sample sites.

	GROUND REFERENCE DATA			
Lichen Map Class	1	2	3	**users accuracy (%)**
1	434	3	2	**99**
2	1	119	5	**95**
3	4	7	141	**93**
producers accuracy (%)	**99**	**92**	**95**	**97**

DISCUSSION

The accuracy of the 16-class vegetation map is consistently higher than 70% with the majority of classes having greater than 90% accuracy. These descriptive statistics provide an easily understood measure of the correspondence between the classified map and the ground data. Classes with low vegetation growth forms and little productivity (X-XVI) had consistently higher accuracy than those classes with taller and/or more productive vegetation (I-IX), likely due to the higher variability introduced by the highly variable physiognomic structure and productivity which may result in spectral overlap. The two unvegetated classes (Unvegetated Ridge and Unvegetated Shoreline) are 100 % accurate due to their highly unique spectral signatures (Chapter 4). The low vegetation cover, distinctive substrate, and level of wetness allow them to be separated relatively easily in the satellite imagery. Class XIV (Unvegetated Ridge) is composed primarily of dry exposed gravel and sand deposits, while Class XV (Unvegetated Shoreline) is composed primarily of saturated clay and mud. The water class (XVI) is also 100 % accurate due to its highly unique spectral reflectance (Chapter 4). Overall, the most important factor that appears to cause misclassification errors is the small-scale character of the landscape, which creates vast numbers of mixed signatures.

Similar overall accuracy levels have been obtained in the arctic and sub-arctic by Matthews (1991) (91%), Ferguson (1991) (88%), Morrison (1997) (93%). Other researchers have achieved much lower overall accuracy (Joria and Jorgenson 1996)

(48%), (Felix and Binney 1989) (38%). However overall accuracy is only a very general indication of the correctness of the map because it can be significantly increased by high values in easily separated classes such as water and is often only based on homogenous pixels.

The utility of a thematic map is largely dependent on its accuracy. Accuracy assessment is normally determined by comparing the map with an independent data set. While this approach does provide valuable summary of the overall quality of the classification it does not always recognize problems associated with mixed pixels. In the map, classification of homogenous pixels compares quite favourably with ground sampled homogenous pixels. However, this does not examine the effectiveness of the classification in dealing with mixed pixels. It is generally assumed that a mixed pixel will be allocated into the class to which it is most floristically similar, i.e. the class to which it is most spectrally similar. However, this is not always the case. Mixed pixels may have unique values distinct from either of the classes present within the pixel. Accuracy is determined by a number of factors, including class allocation, landscape heterogeneity and spatial scale.

The differences in scale and methodologies of previous mapping efforts preclude anything more than a general comparison. The map from this project and those produced by Ritchie (1962) and Clark (1996) all show a predominance of fen vegetation along the coast, particularly near Cape Churchill, with the majority of the central area (locally known as the "little barrens") dominated by treeless, lichen dominated vegetation types. Tree cover is generally variable in all maps throughout the study area, but is relatively continuous in the southwest corner. The Land Cover Map of Canada, based on AVHRR satellite imagery (1.1 x 1.1km spatial resolution) (Cihlar *et al.* 1999) indicates that the southwest corner of the study area is at the edge of the continuous "needle leaf" forest. This map also indicates an open area dominated by lichens and low shrubs in the "little barrens". The map from the current project provides considerably more detail and greater accuracy than any previous effort for this region.

Ritchie (1960) was the first to point out the zonation of vegetation that occurs in the region with a dominance of fen vegetation on younger sites near the coast that becomes much scarcer on older sites further inland. However, the map by Ritchie (1962) is much too coarse to examine the potential gradient that may exist in the vegetation following the distance to the Hudson Bay coast. The wetland map produced from this project provides a high level of spatial resolution that affords the opportunity to conduct the first quantitative test of this theory at the landscape scale. The results indicate that a gradient does exist in the vegetation, representing increasingly older sites in areas further from Hudson Bay in response to isostatic uplift. This fen to bog sequence represents a terrestrialization process that involves peat accumulation, acidification and oligotrophication which produces conditions suitable for bog development (Chapter 3, Vitt and Kuhry 1992).

The application of the wetland map provides a case study of how the map can be applied in landscape level analyses. By aggregating the classes in the 16-class vegetation map and focusing on a single characteristic (bog vs. fen), generalizations can be made about the processes functioning at the landscape scale. The wetland cover map clearly indicates a trend across the landscape that is not easily discernible using site level data sets. A robust succession model should consider spatial scaling, particularly changes in vegetation pattern at the landscape level because perceptions of successional dynamics are necessarily scale-dependent. A 'pure' stand at the scale of a research plot is part of a heterogeneous landscape mosaic when viewed at coarser scales (Frelich and Reich 1995). As a result, landscape scale analysis using regional data can provide insight into the successional patterns occurring at broad scales and may provide a means of coupling ground based measurements with landscape level mapping efforts.

The lichen map indicates a general trend of very low lichen cover along the Hudson Bay coast increasing with distance from the coast, with maximum lichen cover occurring on the inland sites on the west side of the study area. This trend lends further support for the terrestrialization theory because lichens are an excellent indicator of site conditions, as they are found almost exclusively in dry, nutrient poor conditions. The trend of increasing lichen density is somewhat confounded by wildfire effects which repeatedly destroy lichen cover on these inland sites (see the vegetation map to identify burned areas). An understanding of the landscape level distribution of lichens could be particularly important in some applications, such as a study of caribou winter habitat.

While the current vegetation map provides only a snapshot in time, some inferences can be made about the role of fire in the ecosystem. All fires are highly patchy in their structure and intensity of the fire. The total area covered by recent burns occurring between 1991-1996 (11.5 km^2) is a relatively very small proportion of the entire study area (0.1%). The regenerating burn class includes a much larger area (1309 km^2) and covers a significant portion of the study area (7.5%). This is at least partly due to the longer time frame (1960-1990) that this class encompasses. While there are many small burns throughout the study area, the presence of two very large burns indicates that fires can be extensive. Fire is initiated by stochastic lightning strikes and its movement is determined by complex topographical, wetness, fuel and climate variables and remains relatively unpredictable (Payette *et al.* 1989, Timoney and Wein 1991). However, wet fen types have a very low probability of igniting and sustaining a fire compared with drier bog types. Connectivity of the drier sites is an important factor in determining fire spread. Areas further inland with a greater abundance of bog vegetation are more likely to facilitate the movement of fire across the landscape. This is likely why the large fires that can be seen in the vegetation map are limited to the southwest corner of the study area where there is more continuous tree cover and a much lower frequency of saturated fens.

The spatial pattern of fire is an ecologically significant variable, as it influences the grain of the landscape and thus factors such as dispersal distances for colonizing organisms (Moloney and Levin 1996). At the same time, the structure of the landscape is influential on fire spread. As peat accumulation occurs on the older sites with more bog-associated vegetation, fire may become more likely to occur due to increasing fuel and generally drier conditions. Movement of the fire across the landscape may be facilitated by an increasing connectivity of the dry bog vegetation types.

Other studies of fires in the forest-tundra have indicated that fires are variably in their frequency, extent and intensity. Horn (1981) measured a 2.7% burn area over a 4-year period in Nunavut from 1977-1980. Miller (1976) calculated an annual burn rate of 0.17% during a 12-year period from 1955-1967 in northern Manitoba and Nunavut. Scotter (1964) estimated 0.87% land area burned annually during the period 1940-1955 in north central Saskatchewan. The variability is largely related to natural variations in topography, weather, vegetation and fuel conditions, as well as landscape pattern and vegetation cover.

CONCLUSION

The results of this chapter achieve the fourth objective of this thesis; to classify all Landsat TM pixels into one of the defined vegetation classes, perform an accuracy assessment of the classification using an independent data set, and use the vegetation map to infer landscape level vegetation dynamics.

MANAGEMENT IMPLICATIONS

Vegetation mapping using Landsat-5 TM satellite imagery can be done with a high level of accuracy if methods are developed to carefully separate the variable spectral signatures. This project resulted in a seamless digital vegetation map database for WNP and much of the surrounding CCMA area. The vegetation map characterizes the study area as a much more complex mosaic of vegetation than previous mapping efforts. The map shows that the entire study area is extremely heterogeneous at all scales. The vegetation classes are interspersed throughout, however certain general trends can be seen that reflect ongoing isostatic emergence and vegetation succession patterns.

The growing emphasis that scientists and managers are placing on landscape level mapping and monitoring requires the capability to build datasets that are reliable, consistent, flexible and repeatable. The methods presented here provide consistent data across a large area that includes multiple jurisdictions; providing increased capabilities in landscape ecological research and management.

The final vegetation map product is a flexible management tool that can be used for a wide range of applications. Use of the map within a Geographic Information System (GIS) allows the information to be presented in new ways and at different spatial scales Maps have already been produced at 1:250,000 and 1:125,000 scale and can be printed at any scale for any portion of the study area as needed by individual researchers such as an area within a 10 km radius of a field camp, a canoe route, or helicopter survey route. The flexibility of the map allows aggregations of classes to be produced if a certain level of accuracy is required, allowing tradeoffs to be easily made between map accuracy and the number of classes used. Choices can also be made regarding the level of generalization of the map product that can include the classified image or versions that have been run through a modal filter at the 3x3 pixel, 5x5 pixel, or 7x7 pixel scale. More generalized maps are produced by the filtering process.

Since classification of satellite imagery is a statistical procedure that is based on the mean and standard deviation in the signatures of each vegetation class, using different sizes of satellite imagery that cover different areas will likely produce somewhat different classification results. Therefore even identical classification methodologies will produce different results if the study area is changed. Brook *et al.* (2001) mapped an area adjacent to the existing vegetation map using the same TM image mosaic and classification methodology. However, the resulting classification was not entirely consistent with the existing map. These results indicate that while classification of satellite imagery can be done with considerable accuracy, the allocation of many of the mixed pixels that exist is somewhat tenuous and can be changed to other classes. This has important implications for monitoring changes in the landscape over time, since even small changes in spectral reflectance may result in pixels being allocated to new classes.

Accuracy of the final map is an essential consideration if the map is going to be used for any application. It is important to recognize that accuracy assessment methods provide a general indication of the correspondence between the map and ground data. However, methods used to assess accuracy may be biased. Pixel-by-pixel estimates of accuracy are typically optimistic when ground sampling occurs in homogenous blocks (Hammond and Verbyla 1995). An alternative method is to compare the classified map with a reference grid derived from some other source of information such as aerial photography. However, this approach will lead to conservative estimates of accuracy (Verbyla and Hammond 1995). Users of the map should recognize that errors can occur in classifying all classes.

The vegetation map from the present study provides regional scale coverage, but was not developed to answer fine scale (<30m scale) questions regarding vegetation distribution. While large homogenous areas are typically of high accuracy, individual pixels may be much more variable in their accuracy due to mixed pixel effects, variable spectral signatures, and limited spatial resolution of the TM imagery. Areas with specific management concerns should be mapped at finer scales using colour infrared aerial photography or satellite imagery with a high spatial resolution (1-10m). Frequent updating of the map will also be required, particularly in areas that are burned.

CHAPTER 6: SUMMARY AND SYNTHESIS

"The landscape conveys an impression of absolute permanence. It is not hostile. It is simply there - untouched, silent and complete. It is very lonely, yet the absence of all human traces gives you the feeling you understand this land and can take your place in it."

-Edmund Carpenter

PREAMBLE

This chapter provides a short summary and synthesis of the information from the previous chapters.

SUMMARY AND SYNTHESIS

1. The vegetation in the region is highly complex at all scales of analysis, making it inherently difficult to characterize and map. However, management decisions require information about vegetation structure, floristic composition, distribution, and dynamics. In order to be useful, this information should be both accurate and representative of the region as a whole.

2. Six major recurrent vegetation types exist in the study area as a result of complex environmental factors including site moisture, soil and geological conditions, fire history, peat thickness, and nutrient availability. There is considerable variation within each of these types that reflect fine scale differences in these environmental variables. Other vegetation types are present in the study area, including coastal salt marsh and *Dryas* heath upland types, which have highly

unique floristic composition. Burn vegetation types exist along a temporal scale as regeneration occurs over time. Considerable work is needed to further describe these types and identify the many sub-types that make up each community.

3. While the climate is recognized as the predominant determining factor on vegetation distribution in the region, isostatic uplift is the next most dominant agent influencing vegetation distribution and dynamics. Through the process of isostatic uplift, the entire study area has slowly risen out of Hudson Bay over the last 6000 years. This process has continuously exposed new land so those sites further inland have had more time to develop vegetation cover and accumulate peat. Peat accumulation slowly raises the growing surface above the ground water, limiting access to moisture and nutrients. As this terrestrialization process occurs, at the community scale, wet adapted vegetation dominated by sedges and mosses becomes colonized by dry adapted vegetation, including lichens, trees and ericaceous shrubs. The effects of terrestrialization can also be observed at the landscape scale, where there is a shift in the relative abundance of communities. Fen types dominate along the coast and become much scarcer inland. Bog types become increasingly dominant with increasing distance from the Hudson Bay coast. Lichen cover generally increases with distance from the coast, in response to the drier, more nutrient poor conditions. Tree cover is highly variable, likely due to small-scale differences in wind intensity and snow cover, but is largely continuous in the southwest corner of the study area.

4. A map of the distribution of burned areas, along with an understanding of landscape scale vegetation dynamics in the study area allows some inferences to be made about the role of fire in the ecosystem. Fire may be more likely to ignite in areas further inland due to the drier conditions and higher density of fuel, including trees, lichens and ericaceous shrubs, which are all highly flammable. Fire may also be more likely to move across the landscape on inland sites where the increased abundance of bog vegetation may facilitate fire movement. The process of fire regeneration does not generally appear to be successional, as plant communities do not typically replace others over the regeneration process. More typically, regeneration involves re-growth of species that existed prior to the fire. However, if permafrost degrades significantly, the growing surface may slump down to the water level. If the local ground water is nutrient rich, the vegetation may then revert to fen types.

CHAPTER 7: RECOMMENDATIONS FOR VEGETATION RESEARCH AND MANAGEMENT IN THE HUDSON BAY LOWLANDS OF MANITOBA

"There is increasing realization that simply designating an area is not enough to protect it. Parks must be established and managed in new ways if they are to protect Canada's wild ecosystems in the long term."
-Parks Canada Web Site (www.parkscanada.gc.ca)

PREAMBLE

This chapter provides an overview of the existing information available for vegetation management in the Hudson Bay Lowlands of Manitoba. Suggestions are presented in which the vegetation community information presented in Chapter 3 and the vegetation map presented in Chapter 5 can be integrated with this existing information as an information tool for make decisions regarding multi-scale and multi-jurisdictional environmental management issues.

INTRODUCTION

The Hudson Bay Lowlands of Manitoba form a complex mosaic of biotic and abiotic components. For researchers and managers interested in protecting this region and making informed decisions about its future use, monitoring and managing change will require considerable information regarding the structure, composition and distribution of vegetation across the entire ecosystem. Most current natural resource management programs generally acknowledge that detailed and accurate spatial data on vegetation distribution is key to understanding and detecting changes in ecosystem function at the local and landscape scales. These data are also critical for correlating and extrapolating findings from field studies to larger areas in order to answer broader scale vegetation questions. If it is to be useful, research must generate reliable information and insights about environmental systems.

Field data have been collected in various locations in the Hudson Bay Lowlands of Manitoba, but the vast majority of this work has been within less than 50 km of the town of Churchill. Regional scale mapping efforts are particularly expensive and time consuming and so have been limited to surficial geology (Dredge and Nixon 1992) and vegetation (Ritchie 1962, Clark 1996). As a result, relatively few maps exist and available maps are often dated or inconsistent in terms of completeness, accuracy, and spatial referencing.

Information is needed for a variety of applications that fit into three broad groups: research support, ecosystem monitoring, and management support. Research support includes information and co-ordination that would facilitate research in the region. Ecosystem monitoring includes a broad range of factors that require monitoring over the long term in order to measure ecological integrity. Management support includes information that may be directly applicable to decision making in the region.

RESEARCH SUPPORT

One of the fundamental barriers to conducting scientific research in the region is the lack of co-ordinated information. This lack of co-ordination has limited the accumulation of data sets into a useable database. It can also result in the duplication of efforts by various agencies. A framework is needed to link work done by government, academia, non-profit and for-profit sectors. Development of a comprehensive understanding of vegetation will require a supporting database that compiles existing spatial and aspatial information in both digital and hard copy formats. However, the challenges associated with developing a database of this magnitude are significant. Researchers in the region are from a wide range of universities from Canada, the United States and around the world (Churchill Northern Studies Centre Website, http://www.cancom.net/~cnsc). Investigators working at different times of year and in different areas may not even be aware of other projects being conducted. Available information regarding existing research would benefit all researchers and facilitate information sharing and collaboration.

Literature Database and Review

Research in the Hudson Bay Lowlands of Manitoba has been undertaken by a wide range of provincial and federal agencies as well as many different universities (e.g. McClure 1943, Ritchie 1957, 1960, 1962, Sjörs 1959, Jefferies *et al.* 1979, Johnson 1987, Scott *et al.* 1987, Rouse 1998), however, many of the studies are in the form of unpublished theses (Campbell 1995, Clark 1996, Tews 2000) or are only found in the grey literature (Brown-Beckel 1954, Ritchie 1962, Teillet 1983, Rewcastle 1983, Dredge and Nixon 1992). As a result, much of the existing knowledge is not readily available to those who might use it. A preliminary literature database has been compiled by the Churchill Northern Studies Centre in Churchill, but it is not complete, up to date or in a format that is compatible with modern software. It is not generally available and does not provide a means of obtaining the literature itself, which is often difficult to find.

A literature database should be compiled in a commonly used searchable format such as Microsoft Access, EndNote, Procite, or HTML and updated on a regular basis, as new information becomes available. Efforts should be made to include all unpublished reports and grey literature as well. A comprehensive literature database would provide a powerful information tool and would minimize repetition in research projects. To be useful, the database should be available to anyone who requires it on compact disc or through the Internet. A library of these works is also needed to provide access, particularly the documents that are not commonly available. This library could be at the Churchill Northern Studies Centre in Churchill or at the University of Manitoba. In the future, federal (e.g. Wapusk National Park) and provincial (e.g. Cape Churchill Wildlife Management Area) research permits should require that all papers produced (published and unpublished) be provided and stored in the literature database in digital (e.g. Adobe Acrobat .pdf) and hard copy format. This is currently being done in some national parks such as Riding Mountain.

Work is also needed to review the existing literature in order to develop a comprehensive understanding that provides a framework to which each individual study can be associated with. This would provide a broad context into which each study could be placed and would help identify information gaps for future research. There is extensive literature on many of the processes in the region (see Chapter 2 for a representative sample), such as fire and permafrost that may be applicable. However, careful analysis is needed to identify the applicability of each study, given differences in this region such as climate, geology, soils, and wildlife.

Base Map

A common digital base map representing major landscape features such as water bodies and roads at a fine scale (e.g. 1:50,000) is needed to reduce errors associated with mismatching that can often be a serious problem in spatial data sets with different accuracy and various projections. A common system of standards should also be adopted for the navigational units, projection, and ellipsoid used to ensure data sets can be integrated effectively. A mosaic exists of Landsat 5-TM imagery at 30m spatial resolution (Chapter 4). This is currently the most recent, accurate and consistent base map available for the region. It also includes a seamless water layer for the region at 30m spatial resolution.

The 1:250,000 scale map created from the digital NTS map sheets (Wilson 1998) is the most complete vector data for coastline, roads, rail lines, hydro, coastline, water and digital elevation model for large area applications. These layers are useful for regional level mapping and analysis. However, the accuracy of these layers is on the order of 100-200 m, so their use in fine scale application should be done with caution. This map is also not seamless, particularly the water layer, due to differences in the standards of the original map sheets which were developed between the 1940's-1980's.

An extensive survey should be conducted using a differential GPS unit to collect ground control points that can be unambiguously identified both on the ground and in remote sensing imagery. This will provide a highly accurate common geographical reference system that all spatial data can be referenced to. A regional mosaic of panchromatic Landsat 7-TM imagery at 10m spatial resolution could make a more useful base map. The Landsat 7-TM images (http:landsat.gsfc.nasa.gov) are advertised to have excellent geometric integrity (not yet been proven) and could likely be rectified to a standard map co-ordinate system with an RMS error of less than 1 pixel (<10m). This would provide a high quality base map usable for a wide range of applications at multiple scales.

A digital elevation model should also be produced at a fine scale (e.g. 1:50,000; 7.5 min) scale in order to provide insights into the role of drainage and terrain in vegetation distribution, growth and succession. Quality elevation data is critical for a wide range of management question and is an important data gap for the region.

All GIS data should be provided in a relational database structure to store and maintain the attributes associated with the various data sources (attribute is a GIS term for the characteristics of a spatial element). Attributes should be stored in a commonly used and widely applicable database format, such as Microsoft Access. Metadata for all of these layers should also be stored in a common and accessible format (e.g. Naughten *et al.* 2000).

Common Plant Species Database

Plant species have been collected intensively in localized areas, particularly near Churchill (e.g. Tyrrell 1898, Preble 1902, Stormer 1933, Beckett 1945, Brown-Beckel 1954, Ritchie 1956, Beckett 1959, Scoggan 1959, Johnson 1987, Scott 1990) and to a lesser degree at York Factory (Scoggan 1951, Punter 1972). A regional flora has not yet been produced. Unfortunately, many of the species collections to date have generalized locations (e.g. "near Churchill") associated with them, that preclude them from being integrated into an accurate spatial database.

Ground collected data on species locations are essential for mapping species distributions, describing vegetation communities, identifying biodiversity "hotspots", and identifying rare, threatened and endangered species, along with their specific locations (Riley 1982). Species collection information should be compiled from voucher materials currently in the various herbaria in Canada and the U.S. These data should be aggregated into a single database that includes accurate geographic locations of each species sampled. Future plant collections should include an accurate GPS location along with detailed environmental data. Some caution is needed in interpreting the existing species lists, since the apparent rarity of some species in the region or even the entire province and country may be a consequence of inadequate search effort. One of the most pertinent questions related to plant species is which species are particularly sensitive to climate change (Gorham 1994).

An analysis should be conducted of the numbers and distribution of non-native species in the region. Scott (1996) identified 107 species of non-native species in the Churchill area. The influence of human activity remains unknown, particularly near areas of potential non-native seed inputs such as the Port of Churchill.

The vegetation map can be used to guide future plant collecting by identifying areas most likely to contain the species of interest. Studies of non-native species could use the vegetation map in conjunction with GIS layers of human activity such as roads, gravel pits, railways, hydro lines, field camps and other buildings to identify disturbed areas.

Common Plot and Classification Database

A variety of organizations such as the Manitoba Conservation Data Centre maintain local databases of vegetation plot samples. However, there is currently little plot data for the region available. A regional plot database should include complete descriptions and accurate location information for all plots sampled in the region, including plant species, vegetation height, elevation, hydrology, nutrients, soils (nutrient levels and selected physical, chemical and biological properties), pH, and geology. Descriptions and identification criteria should be developed for all plant community types, including rare types. Fine scale classifications for regional scale analyses should also be integrated with larger scale initiatives such as the Canadian Vegetation Classification System (National Vegetation Working Group. 1990) and the Canadian Wetland Classification System (National Wetlands Working Group 1997). For plots sampled by different researchers to be comparable, a standardized sampling methodology is needed (Oliver and Beattie 1993). Additional information should be included such as measurements of biomass and photographs of each site from the air and on the ground. If the sites are permanently marked, this information can be used for long-term monitoring, community descriptions, and for validating satellite-based monitoring and research.

Field Guide

A field guide to the vegetation communities and plant species distributions should be produced for the region (e.g. Harris *et al.* 1996, Racey *et al.* 1996, Beckingham and Archibald 1996, Beckingham *et al.* 1996) in a format that is suitable for a range of users including scientists, wardens, conservation officers, and tourists. Hu (1999) provides an example of how a database of vegetation data can be presented using descriptive text, scanned images and sound in a multimedia format.

Data Accessibility

Data accessibility is a particularly challenging issue due to the difficulties associated with providing access for a large and diverse audience to a database that potentially includes vast quantities of information. Currently, data sharing is done on a case by case basis as individual agencies and individuals form partnerships and agreements.

All people conducting research in the region should provide metadata on their project as part of the federal (e.g. Wapusk National Park) and provincial (e.g. Cape Churchill Wildlife Management Area) permitting process that includes, at a minimum, the specific location(s) of their research, type(s) of data collected, contact information and expected outcomes (e.g. thesis, paper, report, map) (Table 7.1). This information should be compiled into an Internet accessible meta-database to provide researchers immediate access to contact information for other researchers. ASTIS (the Arctic Science and Technology Information System) is an ideal model that contains 46,000 records describing publications and research projects about northern Canada that is searchable on the Internet (http://www.aina.ucalgary.ca/astis). Actual datasets could also be provided through web-based online linkages, as copyright permits. However, prior to data being distributed, a written agreement will be required to protect the data owner and ensure proper use.

Table 7.1. Research project meta-data form for the proposed researcher database for the Hudson Bay Lowlands of Manitoba.

PROJECT TITLE

Structure and Dynamics of the Vegetation in Wapusk National Park and the Cape Churchill Wildlife Management Area of Manitoba: Community and Landscape Scales

CONTACT INFORMATION

Principal Investigator:	Ryan K. Brook
Other Investigator(s):	Evan S. Richardson
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Alternate Contact Person:	Tom Naughten, Data Management Specialist, Parks Canada Western Canada Service Center, Winnipeg, Manitoba thomas_naughten@pch.gc.ca

PROJECT DESCRIPTION

Study Site(s):	throughout Wapusk and the CCWMA, more intensive sampling near Churchill Northern Studies Center, Twin Lakes, La Perouse Bay, Cape Churchill, Broad River (Spence's Cabin), along Owl River, Port Nelson, Wat'chee Lodge
Objectives:	to delineate the major plant communities in the region; to develop a landscape level vegetation map; to examine broad scale patterns in vegetation dynamics
Start Date:	May 1998
Completion Date:	May 2001

DATA COLLECTED

Plant Species Identifications	✓
Plant Species Locations	✓
Species Cover Estimates	✓
Site moisture	✓
Site nutrients	
Site pH	
Soil information	
Tree age	
Biomass measurements	

EXPECTED PROJECT OUTPUTS

Written Report	✓
Research Poster	✓
GIS data layers	✓
GPS site locations	✓
Herbarium Voucher specimens	✓

Field Camps

Field research in the Hudson Bay Lowlands of Manitoba during the summer months requires a camp that provides strong protection from polar bears. The current scarcity of suitable camps limits the areas where fieldwork can be conducted. Several camps are needed in the central and southern areas of Wapusk National Park and the Cape Churchill Wildlife Management Area.

Remote Sensing Archive

Satellite imagery offers high repeatability and coverage of large areas at a relatively low cost. A wide range of remote sensing products exists to answer a correspondingly broad range of questions. Parks Canada currently receives ten-day cloud free composites of the Advanced Very High Resolution Radiometer satellite (AVHRR) for the area and has been compiling the data since 1998 (T. Naughten, pers. comm. 2001). Landsat 5-TM coverage is owned by Parks Canada for July 27, 1996 and June 28, 1991.

In order to provide information for a range of problems, all cloud-free recent and historic satellite imagery should be archived in order to compile a continuous temporal and spatial coverage of the region. Landsat -TM data is available for the region, with cloud free images sometimes being obtained several times per summer. McCanny *et al.* (1995) suggested that Landsat TM imagery be obtained and analyzed every three years for Manitoba national parks. Full coverage of this data should be purchased at least every five years and added to the remote sensing archive, but ideally every year due to the significant changes caused by fire. An existing archive of satellite imagery would provide opportunities for research and monitoring.

Historical aerial photography should be obtained from sources such as the National Air Photo Library (http://airphotos.nrcan.gc.ca). The high-resolution military photos used by Ritchie (1962) for his regional mapping project should be digitized and archived in order to ensure that this important data source is protected. Aerial photos are a complementary data source to satellite imagery, as they often provide greater detail and longer historical coverage than satellites. Information such as cultural land use, tree height and density, landforms, and patterned ground may also be available. While complete coverage of the region by aerial photos may be cost prohibitive, specific monitoring sites or areas of concern should be regularly photographed to complement the vegetation map. Accumulation of spatial data from a wide range of sources that include remote sensing and GIS provides opportunities for landscape level analysis of multiple factors (Gong 1995).

ECOSYSTEM MONITORING

Monitoring the ecosystem is an essential part of management that allows feedback on the results of the decisions made. It also provides the potential to detect changes and respond to them, if required. The range of natural variation in an ecosystem is scale dependent, both temporally and spatially, so measurements must be made across multiple scales in order to adequately monitor it (Noss 1990). Considerable labour and expense has been devoted to producing a scientifically-based ecological monitoring program for Canada's national parks (e.g. McCanny and Henry 1995, McCanny *et al.* 1995, Timoney 1997) and other protected areas (e.g. Herman *et al.* 1995).

Landscape Level Vegetation Map

One of the most important applications of the vegetation map created in Chapter 5 will be in providing a baseline of conditions for future comparison and detection of change. The imagery for the map was obtained in 1996, the year Wapusk National Park was established. The influence of park management decisions may be observed in the changes in the landscape over time.

Change Detection

Detecting change at the landscape scale can be done using a number of different approaches. Visual interpretation and comparison of the vegetation map with other maps can be done successfully to detect change. The advantage of this approach is that it allows the analyst to use expert knowledge of the region to interpret the meaning of the changes that are observed. The disadvantage is that the process is slow, labour intensive, costly (Wilkie and Finn 1996) and is only able to identify differences at a coarse scale. The ability to detect change is also limited by the quality of the maps used. In contrast, computer-assisted methods are able to detect changes in landscapes very quickly, over large areas and so are more commonly used (Jensen *et al.* 1995). Satellite images provide spectral values that can be compared to produce a quantitative measure of change.

Change can be detected using either the satellite imagery or an existing vegetation map. The results from these two approaches can produce much different results since the vegetation map is a generalization of the satellite imagery. Comparison between two vegetation maps simply involves a pixel-by-pixel comparison to produce a change matrix that presents the numbers and types of changes between pixels. Detecting changes between satellite images is more complex, because the data is continuous rather than discrete (e.g. Jano *et al.* 1998). The greatest challenge in this approach is defining the threshold at which the spectral data is assumed to have changed.

Regardless of what method is used for detecting change, it is essential that the factors that are responsible for the change be clearly identified. It must be recognized that differences between images may be the result of "noise", which includes differences attributable not to the change of interest but to imperfect geographical matches between images; spectral differences caused by atmosphere and sun angle; and environmental variation such as soil moisture, temperature, and phenology.

The spectral and spatial limitations of Landsat TM satellite imagery (Chapter 4) limit the ability to detect change. Change detection should be carried out as a scientific experiment rather than a simple comparison to examine what changes have occurred.

Imagery should be selected during the time of year when the features of interest exhibit the greatest spectral contrast on the landscape. Image selection should also include environmental considerations since selecting images from the same day but different years does not ensure that they are representative of similar conditions. The acquisition date for the imagery should be based on the same number of growing degree-days, if phenology is to be compared from year to year.

Prior to change detection, the satellite images must be spatially coregistered. Wilkie and Finn (1996) suggest that the displacement between a pixel in the two images should not exceed half of the spatial resolution of the satellite image (e.g. 15m for Landsat TM). This level of correspondence between the two images may be difficult to achieve due to slight changes in the attitude of the satellite (yaw, pitch, and roll) and errors introduced during georeferencing.

Change in water distribution and abundance in the region using the existing 1996 and 1991 TM imagery provides an interesting example of the challenges and opportunities in using satellite imagery for change detection. Since the objective in this case is to examine water, the imagery should be obtained between late June and late August to ensure that there is no ice in the imagery to confound the analysis. The first step in the analysis should be georeferencing the 1991 Landsat TM image to the existing 1996 TM mosaic. While the RMS error in the 1996 is approximately one pixel (30m), that error is less important than ensuring that the two images are properly aligned together. A classification of the water should then be done in the 1991 image using the exact same methodology defined in Chapters 4 and 5. Masks for the 1991 water layer and 1996 water layer are then compared using a change detection algorithm and a map is produced of three classes: water in both 1991 and 1999, water only 1991, water only in 1999. However, while this change detection procedure is relatively simple, the interpretation of the results is not. The causes of change in water abundance and distribution may be related to permafrost activity, drainage, climate fluctuations, and winter snow deposition, which there may not be existing data for, to examine the differences.

Landscape Level Indicators

There are a variety of metrics that quantify landscape pattern through space and time (Noss 1990). These metrics can then be correlated with specific aspects of ecosystem function (Lyon *et al.* 1998). Changes in spatial metrics may be used as indicators of changes in the ecological integrity of the landscape. Long-term monitoring of landscape metrics should be a key component of monitoring in the region. Once a vegetation map is stored within a GIS, it is a straightforward process to collect a wide range of metrics by using software such as FragStats (McGarigal and Marks 1994).

A wide range of metrics have been applied to landscape monitoring and assessment (Riitters *et al.* 1995). Landscape metrics may be useful for detecting broad scale changes in productivity, succession, or habitat fragmentation that are difficult or impossible to detect at the site level. One of the most basic measures of ecological integrity is the frequency distribution of patch sizes of natural vegetation (O'Neill *et al.* 1997). Changes in vegetation cover may indicate important landscape level changes. However, while the comparisons between successive vegetation maps may be simple, the interpretation is not. Given the complex and sometimes overlapping spectral signatures of different vegetation classes (Chapter 4), along with the complicated methodology involved in developing a vegetation map (Chapter 5), caution should be used in performing change detection analyses. Changes in spectral signatures in the satellite imagery may be due to natural variation in wetness and vegetation phenology or data collection parameters such as sun angle, atmospheric affects, and correction algorithms.

Beyond simple cover change algorithms, much of the influence of landscape pattern on ecological processes is due to the spatial configuration of patches (Franklin and Forman 1987). Landscape metrics such as patch size, patch shape, contagion/dispersion, connectivity, and fractal dimension provide a way of quantifying and comparing differences in landscape structure. However, the resulting metrics are highly influenced by the data used to determine them. It is essential to recognize that different satellite image platforms such as Landsat, SPOT, and AVHRR or different classification methodologies may produce divergent landscape metrics for the same area. As a result, landscape metrics may not be comparable when different methods are used. Landscape diversity (measure of richness and distribution of patch types throughout the landscape) is largely determined by the number of classes included. For example, the two class wetland map (Chapter 5) would have much lower landscape diversity than the 16-class vegetation map due to differences in the number of classes selected by the analyst.

Vegetation productivity is another landscape level indicator of ecological integrity. Both the level and timing of productivity may provide insights into regional factors that influence plant growth such as climate. The onset of the growing season in the spring and the decline in productivity in the fall have important implication for herbivores. Migratory grazing animals such as snow geese, Canada geese and caribou respond to temporal changes in plant productivity and are influenced by variations in plant growth.

The Normalized Difference Vegetation Index (NDVI) is a measure collected from satellite imagery that is used to characterize vegetation productivity over large areas (regional to global) and can be used to infer changes in the imagery related to phenology. This approach has been used extensively around the world and is currently being collected for the region by Parks Canada using 10-day cloud free composite AVHRR imagery (T. Naughten, pers. comm. 2001). However, further research is needed to validate and verify this approach for the unique characteristics of the region. The coarse spatial resolution of AVHRR (1.1 km) means that all pixels are composed of diverse and often unrelated cover types. The results may be confounded by complex mixtures of water, lichen, trees, and sedges within a single pixel that each have very unique reflectance values. The same cover type can also show a variable phenology across the region due to the climate gradient created by Hudson Bay. The vegetation map can be an important tool in examining the factors that influence NDVI and can be used to develop appropriate correction models (O'Brien and Kenkel 2001).

Multivariate analysis of indicators (Riitters *et al.* 1995) show that many of them are highly inter-correlated. Research is needed to find a small number of statistically independent metrics that are appropriate for the region. Indicators need to be relatively easily and reliably measured; have a defined reference level with a variance; account for catastrophic changes; provide for early detection of change (Woodley 1993); represent a broad range of ecological functions over a variety of geographical and temporal scales; represent a range of successional stages; and be measurable using currently available data. Work will also be needed to test the sensitivity of the indicators to measurement and classification errors before they can be considered to be reliable measures of change. Landscape level indicators should be used in conjunction with extensive fieldwork in order to validate their utility.

Site Level Indicators

While landscape level indicators are important for monitoring large-scale changes, they have been criticized for being only able to detect statistically significant changes that have already become a serious problem (for example see Jano *et al.* 1998). More sensitive indicators are needed to complement landscape metrics, identifying change earlier and at finer scales.

Individual species may be important indicators of environmental conditions (Sjörs 1961, Jeglum 1971, Glaser 1983, Ringius and Sims 1997). An indicator species is generally selected that has meaningful characteristics such as presence or absence, population density, dispersion, and reproductive success. These characteristics are then used as an index of ecological attributes that are too difficult, inconvenient or expensive to measure otherwise.

Indicator species may include dominant and ubiquitous species such as *Picea mariana, Cladina stellaris and Carex aquatilis*; less common species such as *Drosera rotundifolia, Elymus arenerius,* and *Salicornia borealis* that are closely associated with specific environmental conditions; or exotic species such as *Bromus inermis, Hordeum jubatum,* and *Taraxacum officinale*. However, the use of indicator species has received considerable criticism (Block *et al.* 1987).

Fire

Fire is a predominant disturbance feature on the landscape. It has widespread influences on permafrost, nutrient cycling, vegetation growth and long-term successional processes. Long term monitoring of the frequency, extent and intensity of fires should be done using a combination of satellite imagery (AVHRR and Landsat), historical aerial photography, and local knowledge. Monitoring of permafrost and vegetation should also be conducted in burns over time, as they regenerate to better understand the processes that occur.

Vegetation Environmental Data

A wide range of environmental factors are important influences on floristic composition and physiognomy of vegetation communities. Some of the most influential factors include pH, nutrient level, water table level and fluctuations, drainage, nature of the substrate (organic or mineral), surficial geology, snow, climate, as well as the movement of groundwater. Many of these factors are highly inter-correlated and influence the ability of plants to sequester nutrients from the ground. However, very little work has been done to measure these variables. Measurements should be made of the chemical components of ground water and soil including pH, C, N, P, S, Ca, Mg, Na, K, Fe, Mn, and Al at numerous sites throughout the region. These measurements should then be compared with the approach in Chapter 3 that uses indicator species to determine nutrient levels.

Existing surficial geology maps (Klassen 1986, Dredge and Nixon 1992) collectively cover the region, but the two maps are based on different methodologies and so can not be joined seamlessly. The map by Dredge and Nixon (1992) is available digitally at 1:500,000 scale (M. Manseau, pers. comm. 2000) and could be digitized from the original 1:60,000 and 1:250,000 maps for finer resolution (L. Dredge, pers. comm. 1999). The two geology maps could be mosaicked by going back to the original maps and re-interpreting them as a single unit (L. Dredge, pers. comm. 1999). A map of the distribution of sand and gravel is also available for the region in hard copy format

(1:250,000 scale) (Dredge and Nixon 1980).

Permafrost temperature, distribution, depth and thickness of active layer should also be measured. A regional scale map should be created of permafrost distribution and active layer thickness. Spatial and temporal changes in permafrost may be examined through analysis of melt ponds, using aerial photography and satellite imagery.

Wildlife Management

A vegetation map is a key input into a wide range of wildlife habitat analyses (e.g. Palmeirim 1985, Agee *et al.* 1989, Brooks 1990). The foundation of almost any wildlife management plan is the ability to determine the habitats that are critical to the survival of the species and identify the distribution and quality of the habitat with accuracy. This requires data on the movements and abundance of the wildlife and good quality habitat information. Landscape level analysis of wildlife habitat has been conducted in the Hudson Bay Lowlands of Manitoba for snow geese (Jano *et al.* 1998, Gadallah 2001) and polar bears (Clark 1996).

Vegetation maps have successfully been used as a primary layer in analyzing wildlife habitat for examining habitat structure (Benoit 1996); landscape level habitat use; (Pearson 1993); scale of habitat selection (Naugle *et al.* 1999); delineation of critical habitat (Butler *et al.* 1995); and monitoring habitat changes (Williams and Lyon 1991, Kemka *et al.* 1992, Jano 1998, Prasad and Tiwari 1998). GIS models can be developed for predicting animal abundance based on land cover data and habitat preference (Miller *et al.* 1989, Walker 1990, Yonzon *et al.* 1991).

Spatial analysis for wildlife habitat assessment is becoming more sophisticated as new approaches are being developed to integrate GIS technology with spatial databases (e.g. Walker 1990, Aspinall 1991, Pereira and Itama 1991). A range of analytical approaches exist including stepwise multiple regression (Campbell 1983), logistic regression (Pereira and Itami 1991, Mace *et al.* 1999), discriminant function analysis (Haworth and Thompson 1990), canonical correspondence analysis (Hill 1991), and habitat suitability index (HSI) modelling. A novel approach proposed by Aspinal and Veitch (1993) is to use wildlife survey data to drive the classification of the satellite imagery in order to produce a habitat map from the perspective of the animals. Additional data such as climate (Walker 1990) and elevation (Gagliuso 1990) can be integrated with the vegetation map to further examine habitat use. However, the methods need to be evaluated and modified for each application due to differences in the study area, scale of analysis, species of interest and data available for the species and habitat.

MANAGEMENT SUPPORT

Vegetation Map

A variety of landscape level mapping projects are currently being conducted in proximity to this area in northern Manitoba and Nunavut (Chapter 5, M. Campbell pers. comm. 1999, A. Jano pers. comm. 1999, A. Didiuk pers. comm. 2000). These works should be synthesized into a larger regional-scale mapping network that allows the development of regional ecosystem management initiatives by providing quality spatial data at a known level of accuracy.

Landscape Sensitivity

Arctic and sub-arctic landscapes are generally highly sensitive to human disturbance, particularly those characterized by fine-grained sediments with a high ice content, since disturbance leads to permafrost melting (Babb and Bliss 1974, MacKay 1970, Walker and Walker 1991, Kevan *et al.* 1995). Mapping landscape sensitivity for large regions has been done using GIS analysis of data such as vegetation, soils, surficial geology, topography, and human activity (Walker and Walker 1991, Tarnocai and Veldhuis 1998, Wilson *et al.* 1999). Additional data may be required for landscape wetness and permafrost distribution. Satellite imagery may be able to provide information on environmental wetness across the landscape. The Kauth-Thomas Tasseled Cap Transformation (Wilkie and Finn 1996) can be applied to Landsat TM data to produce a three dimensional output that defines planes of brightness, greenness, and wetness. However, since the Kauth-Thomas transformation was created as an agricultural crop investigation tool, its utility in mapping sub-arctic vegetation will require further analysis. Further analysis should include transformations of vegetated areas only, rather than the entire scene, which has such a wide range of variation within it (Chapter 4).

Landscape sensitivity mapping should be done for the Hudson Bay Lowlands of Manitoba in order to provide information for managing increasing human activity, including the growing tourism industry. The intensive damage that has been done to the Gordon Point area exemplifies the sensitivity of the region to disturbance. Landscape level mapping will identify areas that are potentially sensitive in order to guide field level monitoring and the production of larger scale maps to monitor disturbance.

Local Knowledge

Long-term success of any natural resource conservation effort will depend on public education and support. The challenge will be to provide the information in a manner suitable for use by a wide range of people. Information from the database can be shared with the community to involve local people in the process. Maps could be produced for trappers and hunters, bird watchers, and could be used in ecotourism operations. Local knowledge should be collected regarding the distribution and health of vegetation across the landscape. Local people often perceive and use the landscape in much different ways than scientists study it. As a result, collection of local knowledge may help make management decisions more relevant. Information needs to be presented in a non-technical format that is easily understandable to all people. Many people from Churchill and Gillam have become frustrated by the lack of clear information regarding research and management planning (personal observation, 1998 and 1999).

CLIMATE CHANGE

Climate change is likely one of the most important issues facing the region for the future. The complexity of the issue and the high level of uncertainty surrounding it, creates significant management challenges and information needs. The predominance of peatlands (~ 76% of the study area) in the Hudson Bay Lowlands of Manitoba suggests that this region may be particularly altered by climate warming since climatic parameters are the dominant factors influencing the development of wetlands (Glaser and Janssens 1986, Vitt *et al.* 1996).

Permafrost peatlands may experience dramatic vegetation changes during the projected 4-8°C climate warming over the next century (Boer 1992, Zoltai 1993, Myneni *et al.* 1997, Rowntree 1997). Harris (1987) has pointed out that an increase in the mean annual temperature of only 2°C, much less than is predicted for northern landscapes in current global warming models (Post 1990), will shift the southern boundary of permafrost far to the north. This would have a profound effect in the Hudson Bay

Lowlands of Manitoba, which is at the southern edge of the continuous permafrost zone.

Scott and Suffling (2000) have predicted that future temperature change in Wapusk National Park will be variable from season to season, but will generally increase from 2-5°C in spring, 0-5 °C in summer, 2-4 °C in fall, and 4-8 °C in winter. They also predict that precipitation will change from -1% to +22% in spring, +6% to +39% in summer, +2% to +16% in fall, and +4% to +30% in winter. In contrast, Boer *et al.* (1992) predicted a decrease in precipitation over the next century in northern Manitoba. The effects of climate change can not be anticipated with any degree of certainty unless precipitation levels can be predicted with some accuracy. Precipitation, particularly snow, influences a wide range of processes, including the annual thermal budget of the soil. While peatlands may slowly migrate north from its current limit, this expansion will likely be outpaced by peatland degradation at the southern end of its range (Gorham 1991). Monitoring of the extent of peatlands is required, particularly at the southern edge of the area.

Vegetation responses to climate change will be complex in permafrosted landscapes (Halsey *et al.* 1995, Camill and Clark 1998). The species composition of permafrost peatlands is simultaneously controlled by local factors such as water table depth and nutrients (Chapter 3, Camill 1999) and landscape level factors such as regional differences in pH, permafrost (Camill 1999), and rainfall (Halsey *et al.* 1997). It is likely that rapid climate warming will alter all or most of these factors.

Permafrost thaw will also alter vegetation structure and community composition (Zoltai 1993). Assuming that this region does not become too arid for peat accumulation (Vitt *et al.* 1994, Halsey *et al.* 1998), declining water tables will likely result in changes in species composition. Drier conditions could select for more xeric species (Chapin *et al.* 1995). Any of these changes will impact ecosystem function (Hobbie 1996, Camill 1999). Movement of species into peatlands is unlikely due to acidic, nutrient poor conditions.

Melting of permafrost will most likely dramatically alter the structure of the peatlands and landscape pattern, resulting in thermokarst erosion and lowered water tables in some area and flooded thaw lakes in others (Billings 1987, Gorham 1991). Zoltai and Wein (1990) suggested that permafrost melting might shift black spruce/*Sphagnum*/lichen bogs on permafrost back to the wetter fen conditions from which they originated.

As the climate continues to warm, fire is expected to become more important in Canada (Flannigan and Van Wagner 1991, Anonymous 1993), particularly if the frequency of drought increases (Clark 1989). While Scott and Suffling (2000) suggest that global change will result in fires in Wapusk burning down to the water table to form new ponds and marshes, there is little data to support this argument. Satellite monitoring of burn distribution and frequency in comparison with climate variables such as precipitation and temperature could provide a means of examining large-scale patterns in fire over decades.

The multi-scale influences of global warming will require sophisticated approaches to integrate and model site, landscape, regional, national and global datasets (e.g. Anonymous 1993). A long-term monitoring program for permafrost and climate in northern Canada has been suggested by Etkin *et al.* (1988). Future work is needed to examine the effects of increased temperature and decreased water levels on peat accumulation rates (*sensu* Billings *et al.* 1982).

Paleocology provides the opportunity to establish trajectories of change over the past 200-300 years. Mosses are particularly useful since they often form the major peat-forming vegetation (Janssens 1983, 1988). This information can provide insights into the processes that have determined current vegetation patterns and may allow predictions to be made of climate warming effects.

Understanding climate change and making management decisions regarding how to deal with the resulting effects will require a long-term, multiscale, multidisciplinary analysis of a wide range of factors. It will also require careful consideration of uncertainty that will continue to be a major influence in this process due to its complexity.

CONCLUSION

This chapter provides some perspectives on how remote sensing imagery, GIS and ground sample data can be integrated into a powerful set of tools for research and decision support. The information provided can be used for monitoring environmental quality, identifying research priorities and providing background information to future research projects. Integration of existing data can help alleviate the difficulties of answering complex multi-scale questions in a remote and poorly understood ecosystem. The pitfalls associated with "throwing money" at the problems should be recognized (see Schindler 1992).

Considerable work needs to be done to refine the research needs and priorities for the region. It should be an iterative process that involves input from a range of management agencies, researchers, and local people. If the suggested database is to be an effective research and management tool, it must be accessible to a wide range of users. It must also be continuously updated, as new information becomes available.

EPILOGUE

Go my son, burn your books. Buy yourself stout shoes. Get away to the mountains, the deserts, and the deepest recesses of the earth. In this way and no other, will you gain a true knowledge of things and of their properties.

-Peter Serverinus, 1571

N. Scott Momaday said that "once in his life a man ought to concentrate his mind upon the remembered earth. He ought to give himself up to a particular landscape in his experience; to look at it from as many angles as he can, to wonder upon it, to dwell upon it." Indeed, this has been my experience with the Hudson Bay Lowlands. I have learned much from its teaching, looking at it for long hours through satellite imagery and aerial photos. But my real learning has come from travelling through the region by train, plane, helicopter, canoe, snowmobile, and particularly on foot. The amazing level of diversity and complexity that I have seen has been a tremendous challenge to study. I hope that this thesis provides another piece to the complex puzzle that is the Hudson Bay Lowlands ecosystem.

Whatever evaluation we finally make of a stretch of land....no matter how profound or accurate, we will find it inadequate. The land retains an identity of its own, still deeper and more subtle than we know.

Barry Lopez, Arctic Dreams

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Appendix 1. A list of all vascular and non-vascular plant species encountered in the study area during the 1998-2000 field seasons.

Form	Genus	Species	Authority	Common Name
Tree	*Larix*	*laricina*	(Du Roi) K.Koch	Eastern Larch: Tamarack
	Picea	*glauca*	(Moench) Voss	White Spruce
	Picea	*mariana*	(P. Mill.) B.S.P.	Black Spruce
	Populus	*balsamifera*	L.	Balsam Poplar: Black Poplar
	Populus	*tremuloides*	Michx.	Trembling Aspen
Shrub	*Andromeda*	*polifolia*	L.	Bog-Rosemary
	Arctastophylos	*alpina*	(L.) Spreng.	Alpine Bearberry
	Arctastophylos	*rubra*	(Rehd. and Wils.) Fern.	Red Bearberry
	Betula	*glandulosa*	Michx.	Dwarf Birch: Scrub Birch
	Chamaedaphne	*calyculata*	L.	Leatherleaf
	Dryas	*integrifolia*	M.Vahl.	Arctic Avens
	Empetrum	*nigrum*	L.	Black Crowberry: Curlewberry
	Juniperus	*communis*	L.	Juniper:Dwarf Juniper
	Kalmia	*polifolia*	Wang.	Bog Laurel
	Ledum	*decumbens*	(Ait.) Lodd.	Dwarf Labrador Tea
	Ledum	*groenlandicum*	Oeder	Labrador Tea
	Loiseleuria	*procumbens*	(L.) Desv.	Alpine Azalea: Trailing Azalea
	Myrica	*gale*	L.	Sweet Gale: Bog-Myrtle
	Rhododendron	*lapponicum*	(L.) Wahlenb.	Lapland Rose-Bay
	Ribes	*glandulosum*	Grauer	Skunk Currant
	Ribes	*hudsonianum*	Richards	Northern Black Currant
	Ribes	*lacustre*	(Pers.) Poir.	Swamp Gooseberry
	Ribes	*oxyacanthoides*	L.	Northern Gooseberry
	Ribes	*triste*	Pall.	Swamp Red Currant
	Rosa	*acicularis*	Lindl. s. lat.	Prickly Rose
	Salix	*arctophila*	Cockerell	Trailing Willow
	Salix	*brachycarpa*	Nutt.	Short-Capsuled Willow
	Salix	*candida*	Flugge	Hoary Willow: Silver Willow
	Salix	*glauca*	L.	Blue-Green Willow
	Salix	*lanata*	(Fern. and Wieg.) Hult.	Lime Willow
	Salix	*maccalianna*	Rowlee	-
	Salix	*myrtillifolia*	Anderss.	Myrtle-Leaved Willow
	Salix	*pedicellaris*	Pursh	Bog Willow
	Salix	*planifolia*	Pursh	Flat-Leaved Willow
	Salix	*reticulata*	L.	Snow Willow
	Salix	*vestita*	Pursh	Rock Willow
	Sheperdia	*canadensis*	(L.) Nutt.	Soapberry: Canada Buffaloberry
	Vaccinium	*uliginosum*	L.	Alpine Bilberry: Arctic Blueberry
	Vaccinium	*vitis-idaea*	L.	Dry-Ground Cranberry
	Viburnum	*edule*	(Michx.) Raf.	Low Bush Cranberry

Form	**Genus**	**Species**	**Authority**	**Common Name**
Graminoid	*Calamagrostis*	*neglecta*	(Ehrh.) Gaertn.	Narrow Reed Grass
	Carex	*aquatilis*	Wahlenb.	Water Sedge
	Carex	*bicolor*	Bellardi	-
	Carex	*capillaris*	L.	Hair Sedge: Hair-Like Sedge
	Carex	*glacialis*	Mack.	Glacier Sedge
	Carex	*limosa*	Wahlenb.(Sm.)	Mud Sedge:Scant Sedge
	Carex	*palacea*	Wahl.	-
	Carex	*rotundata*	Wahl.	-
	Carex	*rariflora*	(Wahlenb.) Sm.	-
	Carex	*rupestris*	All.	-
	Carex	*saxatilis*	L.	Rocky-Ground Sedge
	Carex	*scirpoidea*	Michx.	Rush-Like Sedge
	Carex	*subspathacea*	Wormskj.	-
	Carex	*vaginata*	Tausch	-
	Eleocharis	*palustris*	(L.) R.and S.	Creeping Spike-Rush
	Elymus	*arenarius*	L.	Sea Lime Grass
	Eriophorum	*angustifolium*	Honck.	Tall Cotton-Grass
	Eriophorum	*vaginatum*	L.	Sheathed Cotton-Grass
	Festuca	*ovina*	Schultes	Alpine Fescue
	Festuca	*rubra*	L.	-
	Festuca	*sp.*	L.	Fescue Grass
	Juncus	*balticus*	Willd.	Baltic Rush
	Poa	*sp.*	L.	Meadow Grass
	Puccenellia	*phryganodes*	(Trin.) Scribn. & Merr.	Alkali Grass
	Scirpus	*caespitosus*	L.	Tufted Bulrush: Deer-Grass
	Scirpus	*hudsonianus*	L.	Alpine Cotton-Grass
	Triglochin	*maritima*	L.	Seaside Arrow-Grass

Form	Genus	Species	Authority	Common Name
Herb	*Achillea*	*nigrescens*	(E. Mey.) Rydb.	Yarrow: Milfoil
	Androsace	*septrionalis*	L.	Pygmyflower: Rock Jasmine
	Anemone	*multifida*	Poir	Cut-Leaved Anemone
	Armeria	*maritima*	(Mill.) Willd.	Thrift
	Arnica	*alpina*	(L.) Olin	Alpine Arnica
	Astragalus	*alpinus*	L.	Alpine Milk-Vetch
	Atriplex	*glabriuscula*	Edmon.	Smooth Orache
	Atriplex	*patula*	L.	Orache:Spearscale
	Bartsia	*alpina*	L.	Velvet Bells: Alpine Bartsia
	Caltha	*palustris*	L.	Yellow Marsh-Marigold
	Castilleja	*raupii*	Pennell	Purple Paintbrush: Indian Paintbrush
	Chenopodium	*capitatum*	(L.) Asch.	Strawberry Blight
	Chrysanthemum	*arcticum*	L.	Arctic Daisy
	Corydalis	*sempervirens*	(L.) Pers.	Pink Corydalis
	Cypripedum	*passerinum*	Richards.	Northern Lady Slipper
	Draba	*glabella*	(Richards.) Gelert	Sand-Dwelling Rock Cress
	Drosera	*anglica*	Huds.	Oblong-Leaved Sundew
	Drosera	*rotundifolia*	L.	Round-Leaved Sundew
	Epilobium	*angustifolium*	L.	Fireweed: Great Willowherb
	Epilobium	*latifolium*	L.	Broad-Leaved Fireweed: Willowherb
	Equisetum	*arvense*	L.	Common Horsetail
	Equisetum	*sp.*	L.	Horsetail:Scouring Rush
	Equisetum	*variegatum*	Schleich.	Variegated Scouring Rush
	Euphrasia	*arctica*	Lange	Northern Eyebright
	Fragaria	*vesca*	L.	Woodland Strawberry
	Gentiana	*propinqua*	Richards.	Arctic Gentian
	Geocaulon	*lividum*	(Richards.) Fern.	Northern Comandra
	Habenaria	*hypoborea*	(L.) R.Br.	Green-Flowered Bog Orchid
	Hedysarum	*mackenzii*	Richards.	Northern Hedysarum
	Heracleum	*lanatum*	Michx.	Cow-Parsnip
	Hippurus	*tetraphylla*	L.f.	Four-Leaved Mare's Tail
	Hippurus	*vulgaris*	L.	Common Mare's Tail
	Honckenya	*peploides*	(L.) Ehrh.	Sea-Purslane: Seabeach Sandwort
	Lesquerella	*arctica*	S. Wats.	Northern Bladderpod
	Linum	*lewisii*	Pursh	Lewis' Wild Flax
	Lomatogonium	*rotatum*	(L.) Fries	Star Gentian
	Matricaria	*ambigua*	(Ledeb.) Kryl.	Sea-Shore Chamomile
	Melandrium	*affine*	J. Vahl.	Arctic Bladder-Campion
	Mentha	*arvensis*	L.	Wild Mint
	Menyanthes	*trifoliata*	L.	Buck-Bean: Bog-Bean
	Mertensia	*maritima*	(L.) S.F. Gray	Seaside Lungwort: Oysterwort
	Mertensia	*paniculata*	(Ait.) G. Don	Tall Lungwort
	Moneses	*uniflora*	(L.) Gray	One-Flowered Wintergreen

Form	Genus	Species	Authority	Common Name
Herb	*Lycopodium*	*annotinum*	L.	Stiff Club Moss
	Lycopodium	*sp.*	L.	Club Moss
	Oxycoccus	*microcarpus*	Turcz.	Swamp Cranberry: Bog Cranberry
	Oxytropis	*campestris*	(L.) DC.	Late Yellow Locoweed
	Parnassia	*palustris*	L.	Northern Grass-Of-Parnassus: Bog Star
	Pedicularis	*flammea*	L.	Flame-Colored Lousewort
	Pedicularis	*sudetica*	Willd.	Purple Rattle
	Petasites	*sagittatus*	(Banks) A. Gray	Arrow-Leaved Colt's Foot: Butterbur
	Pinguicula	*vulgaris*	L.	Common Bladderwort: Bog-Violet
	Plantago	*maritima*	L.	Seaside Plantain
	Polygonum	*viviparum*	L.	Alpine Bistort
	Potentilla	*anserina*	L.	Silverweed
	Potentilla	*egedii*	Wormskj.	Egede's Cinquefoil: Pacific Silverweed
	Potentilla	*fruticosa*	L.	Shrubby Cinquefoil
	Potentilla	*palustris*	(L.) Scop.	Marsh Cinquefoil
	Potentilla	*norvegica*	L.	Rough Cinquefoil
	Primula	*egaliksensis*	Wormskj.	Greenland Primula
	Pyrola	*grandiflora*	Radius	Large-Flowered Wintergreen
	Pyrola	*rotundifolia*	Radius	Smooth Wintergreen
	Ranunculus	*arbortivus*	L.	Small-Flowered Buttercup
	Ranunculus	*aquatilis*	L.	Large-Leaved Watercrowfoot
	Ranunculus	*cymbalaria*	Pursh	Shore Buttercup
	Ranunculus	*pedatifidus*	Sm.	Northern Buttercup
	Ranunculus	*lapponicus*	L.	Lapland Buttercup
	Ranunculus	*purshii*	Richards.	Small Yellow Watercrowfoot
	Rhinanthus	*borealis*	Chab.	Yellow rattle
	Rubus	*acaulis*	Michx.	Stemless Raspberry: Dewberry
	Rubus	*chamaemorus*	L.	Cloudberry: Bake-Apple Berry
	Rumex	*occidentalis*	S. Wats.	Western Dock
	Salicornia	*borealis*	Wolff & Jefferies	Northern Samphire
	Saxifraga	*aizoides*	L.	Yellow Mountain Saxifrage
	Saxifraga	*caespitosa*	L.	Tufted Saxifrage
	Saxifraga	*hirculus*	L. s. lat.	Yellow Marsh Saxifrage
	Saxifraga	*oppositifolia*	L.	Purple Saxifrage
	Saxifraga	*tricuspidata*	Rottb.	Three-Toothed Saxifrage: Prickly Saxifrage
	Senecio	*congestus*	(R.Br.) DC.	Marsh Ragwort: Mastodon Flower
	Senecio	*pauperculus*	Michx.	Balsam Groundsel
	Smilicina	*trifolia*	(L.) Desf.	Three-Leaved Solomon's Seal
	Solidago	*multiradiata*	Ait.	Alpine Goldenrod
	Tofieldia	*pusilla*	(Michx.) Pers.	Bog Asphodel
	Thalictrum	*sparsiflorum*	Turcz.	Flat-Fruited Meadow Rue
	Thalictrum	*venulosum*	Trel.	Meadow-Rue
	Triglochin	*maritima*	L.	Seaside Arrow-Grass
	Triglochin	*palustris*	L.	Marsh Arrow-Grass

Form	**Genus**	**Species**	**Authority**	**Common Name**
Lichens	*Alectoria*	*ochroleuca*	(Hoffm.) Mass.	-
	Bryocaulon	*divergens*	(Ach.) Karnef.	-
	Cetraria	*cucullata*	(Bell.) Ach.	Curled Snow Lichen
	Cetraria	*nivalis*	(L.) Ach.	Flattened Snow Lichen
	Cetraria	*laevigata*	Rass	-
	Cladina	*mitis*	(Sandst.) Hustich	Yellow Reindeer Lichen
	Cladina	*rangiferina*	(L.) Nyl	Grey Reindeer Lichen
	Cladina	*stellaris*	(Opiz) Brodo	Star Reindeer Lichen
	Cladonia	*borealis*	des Abb.	Red Pixie Cup
	Evernia	*mesomorpha*	Nyl.	Spruce Moss
	Peltigera	*aphthosa*	(L.) Willd.	Freckle Pelt
	Rhizocarpon	*geographicum*	(L.) DC.	Green Map Lichen
	Stereocaulon	*alpinum*	Laur.	Coral Lichen
	Usnea	*hirta*	-	Old Man's Beard
	Xanthoria	*elegans*	(Link.) Th. Fr.	Elegant Orange Lichen
Bryophyte	*Aulocomnium*	*turgidum*	(Wahlenb.) Schwaegr.	-
	Amblisteygium	*sp.*	-	-
	Calliergon	*sp.*	-	Water Moss
	Cillerium	*sp.*	-	-
	Campyllium	*sp.*	-	-
	Dicranum	*sp.*	-	-
	Drepanocladus	*sp.*	-	-
	Funaria	*sp.*	-	-
	Hypnum	*sp.*	-	-
	Hylocomium	*splendens*	(Hewd.) B.S.G.	-
	Lepidozia	*sp.*	-	-
	Meesia	*triquetra*	(Richt.) Angstr.	-
	Pleurozium	*schreberi*	(Brid.) Mitt	-
	Polytrichum	*piliferum*	Hedw.	-
	Polytrichum	*sp.*	-	-
	Ptilidium	*ciliare*	-	-
	Scorpidium	*sp.*	-	-
	Sphagnum	*fuscum*	(Schimp.) Klinggr.	Common Brown Sphagnum
	Sphagnum	*sp.*	-	Peat Moss
	Tomenthypnum	*nitens*	(Hewd.) Loeske	Golden Moss
	Thuidium	*abietinum*	(Hedw.) B.S.G.	Wiry Fern Moss
	Tillidium	*sp.*	-	-

APPENDIX II: CAREY LAKE BURN TRANSECT

Permafrost depth was measured at 100 sites along a transect through the Carey Lake Burn on August 8, 1999. The fire went through the area in the first week of August and was still burning nearby during the sampling. Depth to permafrost was measured using a metal rod inserted to maximum depth into the ground. The transect began at UTM 15, 471229 E 6459586 N NAD 27, approximately 50 m outside of the burn and continued 50 m past the end of the burn, ending at 471516 E 6460414 N. A complex mosaic of burned and unburned vegetation was included in the transect.

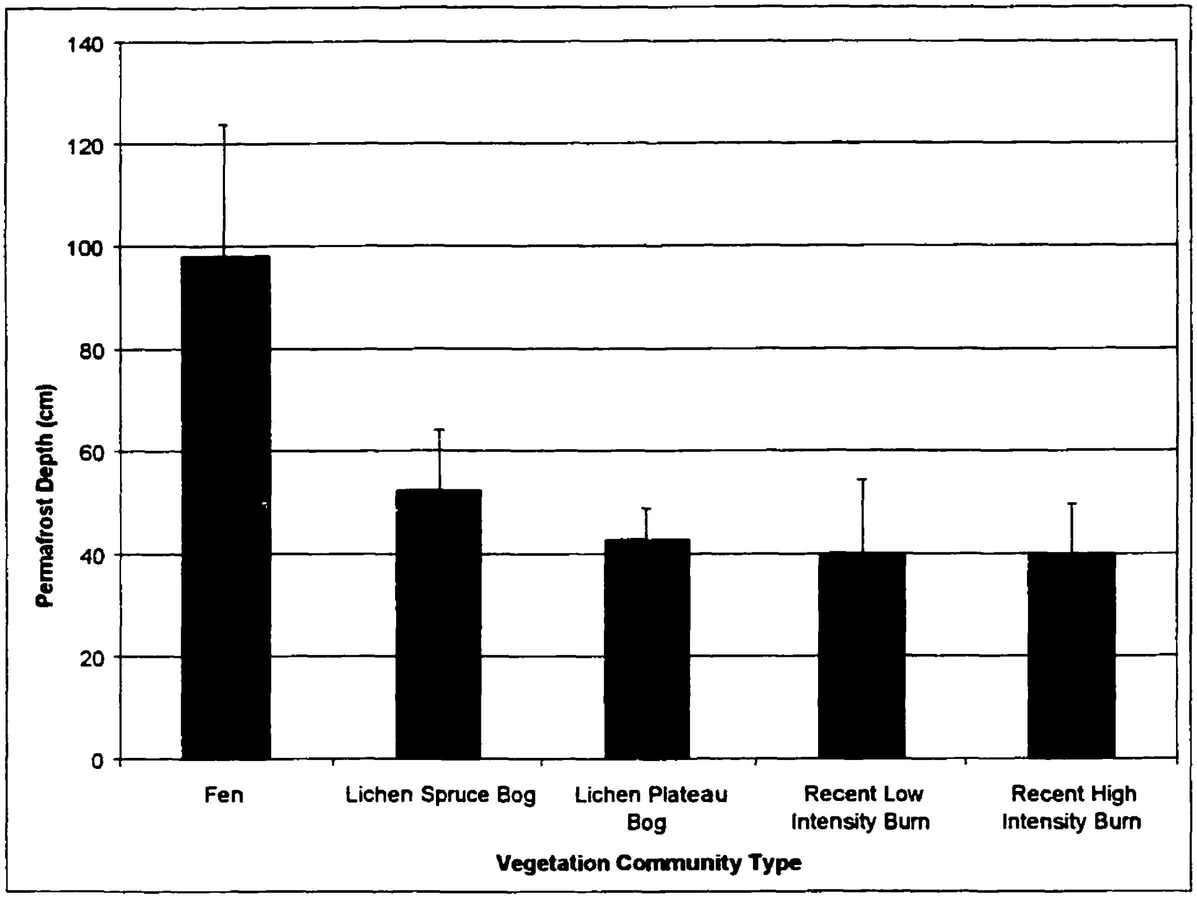

Figure A1. Permafrost depth (mean ± standard deviation) for the different unburned plant community types and high intensity and low intensity burned areas).

Table A2. Measurements of vegetation cover and permafrost depth for 100 sites at Carey Lake, following a fire.

Site	Vegetation Type	Tree(m)	Ground Veg(cm)	Shrub(cm)	Permafrost Depth(cm)
1	Lichen Spruce Bog	3.5	10.0	49.0	59.5
2	Lichen Spruce Bog	4.0	6.5	48.0	59.5
3	Lichen Spruce Bog	2.7	13.5	0.0	49.0
4	Lichen Spruce Bog	1.0	9.0	15.5	64.5
5	Lichen Spruce Bog	0.8	0.0	21.0	70.5
6	Lichen Spruce Bog	0.0	4.5	10.0	62.0
7	Lichen Spruce Bog	0.4	3.0	17.0	63.5
8	Lichen Spruce Bog	1.3	4.0	16.5	45.0
9	Lichen Peat Plateau Bog	0.0	4.0	0.0	45.5
10	Lichen Peat Plateau Bog	0.0	2.5	0.0	51.0
11	Lichen Peat Plateau Bog	0.0	2.4	11.0	43.5
12	Lichen Peat Plateau Bog	0.0	2.5	16.0	33.0
13	Lichen Peat Plateau Bog	0.0	6.0	0.0	41.0
14	Lichen Peat Plateau Bog	0.0	4.0	0.0	41.0
15	Lichen Peat Plateau Bog	1.5	4.4	0.0	40.0
16	Lichen Peat Plateau Bog	0.0	2.5	0.0	49.0
17	Recent Burn	0.0	0.0	0.0	33.5
18	Recent Burn	0.0	0.0	0.0	34.0
19	Recent Burn	0.0	0.0	0.0	41.0
20	Recent Burn	0.0	0.0	0.0	35.5
21	Recent Burn	0.0	0.0	0.0	39.5
22	Recent Burn	0.0	0.0	0.0	32.0
23	Recent Burn	0.0	0.0	0.0	35.5
24	mixed burn/lichen spruce bog	2.0	5.0	15.5	31.0
25	Lichen Spruce Bog	1.8	4.0	11.5	51.0
26	mixed burn/dead	1.8	6.0	0.0	68.5
27	Recent Burn	0.0	0.0	0.0	34.5
28	mixed burn/dead	0.0	0.5	0.0	54.0
29	mixed burn/dead	0.0	0.2	0.0	43.0
30	Recent Burn	0.0	0.0	0.0	39.0
31	Recent Burn	0.0	0.0	0.0	35.0
32	mixed burn/dead	0.0	0.3	0.0	38.0
33	Recent Burn	0.0	0.0	0.0	31.5
34	Recent Burn	0.0	0.0	0.0	42.0
35	Recent Burn	0.0	0.0	0.0	33.5
36	mixed burn/dead	2.8	0.0	0.0	34.0
37	mixed burn/dead	0.0	0.5	0.0	30.0
38	Recent Burn	0.0	0.0	0.0	35.0
39	mixed burn/dead	4.1	0.0	0.0	35.5
40	mixed burn/dead	0.0	0.5	0.0	36.0
41	mixed burn/dead	2.9	1.0	0.0	35.5
42	mixed burn/dead	3.2	7.5	32.0	42.0
43	mixed burn/dead	2.3	0.0	15.5	47.0
44	Recent Burn	0.0	0.0	0.0	60.0
45	mixed burn/dead	2.1	0.0	0.0	33.5

Appendix 11: Carey Lake Burn Transect.

Site	Vegetation Type	Tree(m)	Ground Veg(cm)	Shrub(cm)	Permafrost Depth(cm)
46	mixed burn/dead	2.4	0.0	0.0	34.5
47	mixed burn/dead	2.3	0.0	0.0	41.0
48	mixed burn/dead	2.6	0.0	0.0	45.0
49	mixed burn/dead	1.9	0.0	0.0	41.0
50	Recent Burn	0.0	0.0	0.0	55.0
51	mixed burn/dead	0.0	1.1	0.0	44.0
52	Recent Burn	0.0	0.0	0.0	53.5
53	mixed burn/dead	0.0	0.9	0.0	65.0
54	Recent Burn	0.0	0.0	0.0	55.0
55	Recent Burn	0.0	0.0	0.0	42.5
56	mixed burn/dead	0.0	0.0	0.0	31.5
57	mixed burn/dead	3.4	0.0	0.0	30.5
58	mixed burn/dead	4.5	0.0	0.0	29.0
59	mixed burn/dead	2.6	0.0	0.0	30.0
60	Recent Burn	0.0	0.0	0.0	24.0
61	Recent Burn	0.0	0.0	0.0	13.0
62	Recent Burn	0.0	0.0	0.0	12.5
63	Recent Burn	0.0	0.0	0.0	22.0
64	Recent Burn	0.0	0.0	0.0	21.0
65	Recent Burn	0.0	0.0	0.0	17.0
66	mixed burn/dead	1.2	0.0	0.0	13.5
67	mixed burn/dead	1.8	0.0	0.0	18.5
68	mixed burn/dead	1.3	0.0	0.0	5.5
69	mixed burn/dead	4.2	4.5	39.0	41.0
70	Recent Burn	0.0	0.0	0.0	39.0
71	mixed burn/dead	3.6	0.0	0.0	35.5
72	mixed burn/dead	2.0	5.5	19.0	47.0
73	mixed burn/dead	1.2	0.0	0.0	40.5
74	Lichen Spruce Bog	2.5	7.5	41.0	45.0
75	mixed burn/dead	3.1	0.0	26.0	26.5
76	Recent Burn	1.6	0.0	0.0	28.5
77	mixed burn/dead	2.2	0.0	0.0	28.0
78	mixed burn/dead	2.4	0.0	0.0	27.5
79	Lichen Peat Plateau Bog	0.0	0.0	0.0	50.0
80	mixed burn/lichen spruce bog	1.1	0.0	0.0	34.0
81	Recent Burn	0.0	0.0	0.0	41.5
82	Recent Burn	0.0	0.0	0.0	47.0
83	mixed burn/dead	0.8	0.0	0.0	26.0
84	Recent Burn	0.0	0.0	0.0	48.0
85	Recent Burn	0.0	0.0	0.0	35.0
86	Poor Fen	0.0	33.5	0.0	107.0
87	Poor Fen	0.0	49.0	0.0	140.0
88	Poor Fen	0.0	36.0	0.0	112.0
89	Lichen Spruce Bog	1.7	2.0	0.0	33.5
90	Lichen Peat Plateau Bog	0.0	3.0	13.0	38.5
91	Lichen Spruce Bog	1.0	1.5	35.0	40.0
92	Lichen Spruce Bog	0.6	3.0	12.0	37.0
93	Lichen Peat Plateau Bog	0.0	3.0	9.5	36.5
94	Lichen Peat Plateau Bog	0.7	42.0	0.0	61.0

Site	Vegetation Type	Tree(m)	Ground Veg(cm)	Shrub(cm)	Permafrost Depth(cm)
95	Lichen Peat Plateau Bog	0.0	6	12.5	42.0
96	Lichen Peat Plateau Bog	0.0	5.0	9.5	38.0
97	Lichen Peat Plateau Bog	0.0	5.5	12.0	42.5
98	Lichen Peat Plateau Bog	0.0	3.5	8.0	37.0
99	Lichen Peat Plateau Bog	0.0	5.5	7.0	40.0
100	Lichen Peat Plateau Bog	0.0	3.0	8.5	43.0