

VEGETATION AND SOIL DISTURBANCES IN BOGS
TRAVERSED BY POWER LINE CORRIDORS IN MANITOBA

by

Borgthor Magnusson

A Thesis

Submitted to the Faculty of Graduate Studies

University of Manitoba

in Partial Fulfillment of the

Requirements for the Degree of

Doctor of Philosophy

Department of Botany

Winnipeg, Manitoba

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Í vorþeynum

Á meðan brimið þvær hin skreipu sker
og skýjaflotar sigla yfir lönd,
þá spyrja dægrin: Hvers vegna ertu hér,
hafrekið sprek á annarlegri strönd?

Það krækilyng sem eitt sinn óx við klett
og átti að vinum gamburmosa og stein,
er illa rætt og undarlega sett
hjá aldintré með þunga og frjóa grein.

Hinn rammi safi rennur frjáls í gegn
um rót er stóð í sinni moldu kyr,
en öðrum finnst sig vanta vaxtarmegn
þótt vorið fljúgi í lofti hraðan byr.

Drýpur af hússins upsum erlent regn,
ókunnir vindar kveina þar við dyr.

Jón Helgason

ABSTRACT

Effects of power line construction and vegetation management on bog vegetation and organic matter breakdown, were examined along two right-of-ways (ROWS) extending from 50-56 °N in Manitoba. Vegetation, peat and soil water were sampled and collected from the ROW and adjacent undisturbed forest at sixteen sites. Organic matter decomposition estimates were derived from measurements of seasonal soil respiration rates in the ROW and forest at three sites. Erosion of exposed peat was estimated in the ROW at three sites. Effects of the herbicides picloram and 2,4-D on growth of Sphagnum fuscum were studied at two sites; and in the laboratory the effects of the herbicides on respiration rate (CO₂ production) in S.fuscum peat cores were examined.

Decorana ordination and cluster analysis showed similar patterns in the vegetation. Major groupings were formed along moisture and disturbance gradients. Canonical correlation analysis revealed that variation in bare ground, peat humification and soil water chemistry correlated strongly with the vegetation pattern. Six (among them S.fuscum) of the fifty most abundant plant species were significantly reduced in abundance in the ROW compared to the forest, while only, Polytrichum strictum had significantly higher abun-

dance in the ROW than the forest community. Annual decomposition of soil organic matter was estimated to be of the order 350-670 g dwt m⁻² over the latitudinal range of the study area. It was not evident that disturbances in the ROW had affected rates of organic matter decomposition. Organic matter losses due to erosion of surface peat in the ROW were very low in comparison to decompositional losses. Applications of picloram on Sphagnum fuscum growing in hummocks were found to permanently destroy most of the shoots of the moss and suppress its regeneration for several years. Cyclic use of picloram in the type of bogs studied is considered to lead to a gradual destruction of the hummock habitat and its associated community. Picloram and 2,4-D applications caused initial stimulation of respiration rates in peat cores, which were minimal within 30 days of applications. The herbicides were not considered to cause long term changes in peat decomposition rates.

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GENERAL INTRODUCTION

In northern environments human populations have maintained a relatively low density due to the severity of the climate and consequently vegetation has evolved largely without human disturbances (Billings 1973). During the last two decades interest and engineering activity in northern Canada has increased considerably with the discoveries of important oil and gas deposits in the Arctic, and mining activities and hydroelectrical developments (Laing 1974). The impact of this recent invasion of human activity into northern regions indicates that northern vegetation and soils are fragile and easily disturbed. Peatlands in particular have been shown to be very susceptible to human disturbances (Bellamy et al. 1971; Hernandez 1973; Van Cleve 1977), because their soils have a relatively weak physical structure compared with upland soils and are easily disturbed by heavy machinery. In northern regions these soils are frequently underlain by permafrost which may be affected by vegetation removal and surficial disturbances (Bliss and Wein 1972; Billings 1973). Peat soils have also a relatively low nutrient content which, coupled with cool climate and short growing season in northern regions, makes plant recolonization of disturbed surfaces a slow process (Gore and Godfrey 1981).

Peatlands compose about one third of the total land area of Manitoba (Tarnocai 1978). The greatest concentrations of peat deposits occur in the northern part of the province (53 - 60° N) within the discontinuous permafrost zone. In the southern part of this zone (53 - 55° N), permafrost is generally restricted to peatlands where peat plateaus and palsas occur in abundance (Thie 1974).

Hydroelectrical development has expanded greatly during the last twenty years in Manitoba. The main development is on the Nelson River (56 - 57° N), whose potential has only been partly utilized thus allowing further development in the future (Weir 1983). Due to the northern location of the major hydro-electric generating stations, construction and maintenance of extensive transmission lines and right-of-ways is necessary for transportation of the generated electricity to the major users in the southern part of the province. Clearance and management of tree growth along the right-of-ways is essential so that a) it does not interfere with the lines or the conductance of the current, and b) the lines are readily accessible for maintenance and inspection (Niering 1958; MacLellan 1982). In forested areas, the construction of right-of-ways, normally consists of clear-cutting procedures, which tend to result in considerable environmental disturbances and affect lower growing vegetation (Egler 1975; Sims 1977). Vegetation management along power line right-of-ways in Manitoba is principally carried out

with herbicides, which commonly are applied with broadcast techniques (Sims 1977; MacLellan 1982). A common result of indiscriminant herbicide applications is that non-targeted and often desirable low-growing species, sensitive to the herbicides used, are greatly reduced or eliminated (Niering and Goodwin 1974; Egler 1975; Dreyer and Niering 1986).

During 1967 to 1970, Manitoba Hydro cleared a 900 km long, 120 m wide wooded area between Gillam and Winnipeg for the construction of a HVDC transmission line (MacLellan 1982). The northern part of the line traverses large expanses of peatlands. The initial line clearance and construction and subsequent line maintenance and vegetation control has had and will continue to have an ecological impact on extensive peatland plant communities and their underlying soils (Sims 1977).

In 1982 this research project was started to examine some of the effects of disturbances on peatlands, mainly along the HVDC power line between Gillam and Winnipeg. Earlier work (Sims 1977, Sims and Stewart 1981) and extensive ground surveys in 1982 led to the conclusion that in different peatland types, visual impact of various disturbances (vegetation clearance, vehicle traffic, herbicidal applications) was most severe in shrub-rich treed bogs (*sensu* Jeglum et al. 1974). This would include moss muskeg (Ritchie 1960b) or muskeg (Jeglum 1972) and wooded peat plateaus (Zoltai and Tarnocai 1971). This study was therefore mostly confined to

these habitats. The vegetation of the bogs in their natural stage is characterized by a conspicuous shrub layer of ericaceous shrubs, mostly Labrador-tea (Ledum groenlandicum) and an overstorey of stunted, open grown black spruce (Picea mariana), but tamarack (Larix laricina) may also be present. Sphagnum is dominant in the ground layer at the moister sites but lichens and feather mosses may gain dominance at the drier sites. In hollows where the water table is close to the surface or open pools persist, sedges are commonly found.

The main objective of the present study was to obtain a better understanding of how vegetation, peat degradation processes and environmental conditions in bogs are affected by disturbances associated with power line right-of-ways. Such knowledge is essential to evaluate existing management practices and forecast their long term impacts.

The study consisted of investigations into:

1. distribution and abundancies of understorey plants, and selected soil parameters at disturbed peatland sites;
2. soil respiration (CO₂ production), soil moisture, temperature and acrotelm thickness and organic matter breakdown; and erosion of exposed peat at disturbed peatland sites;

3. effects of the herbicides, picloram and 2,4-D on (a) growth of Sphagnum fuscum and, the associated plant community, and (b) decomposition rates of Sphagnum peat.

Chapter I

**VEGETATION AND SOIL FACTORS OF BOGS IN MANITOBA
DISTURBED BY POWER LINE RIGHT-OF-WAYS**

1.1 INTRODUCTION

The locations where peatland communities occur and how long they persist in a given landscape are largely determined by regional hydrological patterns and their frequency and direction of change. Differences in floristics of various natural peatland types within a geographical area are largely due to differences in mean level of their water tables, seasonal patterns of fluctuations, and in the rates of moisture and ionic supply (Sjörs 1950, 1963; Heinselman 1970; Jeglum 1971, 1972; Jeglum et al. 1974; Glaser et al. 1981; Jasieniuk and Johnson 1982; Karlin and Bliss 1984). Studies have shown (Spence 1964; cited in Tallis 1983) that wetland communities differ in their tolerance to variations of moisture and nutrient regime. Communities rooted above the mean water level are generally found to have a lesser tolerance range with respect to water level changes than communities rooted below the mean water level.

Of the different peatland types, bogs have the lowest mean water level (Jeglum et al. 1974). In the driest ombrotrophic bogs most of the plants grow above the watertable and without direct contact with it. Some bog types (e.g. peat palsas and plateaus) do not in general have free standing water. This implies, that bogs should have limited tolerance range to long term changes in mean water level and or moisture input and might therefore be considered as unstable systems. However of the different peatland types in conti-

mental areas of North America, black spruce muskegs and bogs are considered the most stable and widespread (Heinselman 1963, 1970; Jeglum 1973). These areas represent a climax or a terminal stage of the trends in peatlands toward decreasing levels of moisture and fertility regimes, taking place through paludification and or terrestriation processes (Tallis 1983). These communities are situated on the moisture gradient where the water supply is fairly constant and runoff fluctuations and lake level changes do not have direct effects (Jasieniuk and Johnson 1982). In addition bogs owe their stability to the underlying Sphagnum peat (Jeglum 1973) which is resistant to microbial breakdown and has a high waterholding capacity. These properties of peat allow a considerable groundwater storage and buffering of water level changes within these systems.

Although bogs have been considered as the most persistent among peatland types, it does not imply that their vegetation and soil are less sensitive to external disturbances than those of other peatland types. Bog vegetation is dependant upon the maintenance of a delicate and stable hydrological and nutrient regime which also plays a key role in controlling production/decomposition or accumulation processes in the bogs. Peat soil is a highly oxidizable substrate with a weak physical structure which makes it vulnerable to degrading agents (e.g. decomposers, water, wind, fire, machinery) when disturbed. The low nutrient content

and high acidity of bog peats restricts plant growth (Hemond 1980) and slows plant recolonization and stabilization processes on them when surficial disturbances occur.

In the extensive bog areas of central and northern Manitoba, construction and maintenance of power line right-of-ways has resulted in vegetation and substrate disturbances. A complete removal of vegetation cover has occurred in some areas, thus creating conditions for localized erosion and subsidence of organic soils (Sims 1977, Sims and Stewart 1981). The present study was undertaken to

1. examine the distribution and abundance of plant species in natural and disturbed bogs along power line corridors extending between the Nelson River and Winnipeg;
2. assess quantitatively the impact of surficial soil and plant disturbances on plant cover and soil factors and evaluate the natural recolonization of disturbed areas.

1.2 STUDY AREA

1.2.1 Location

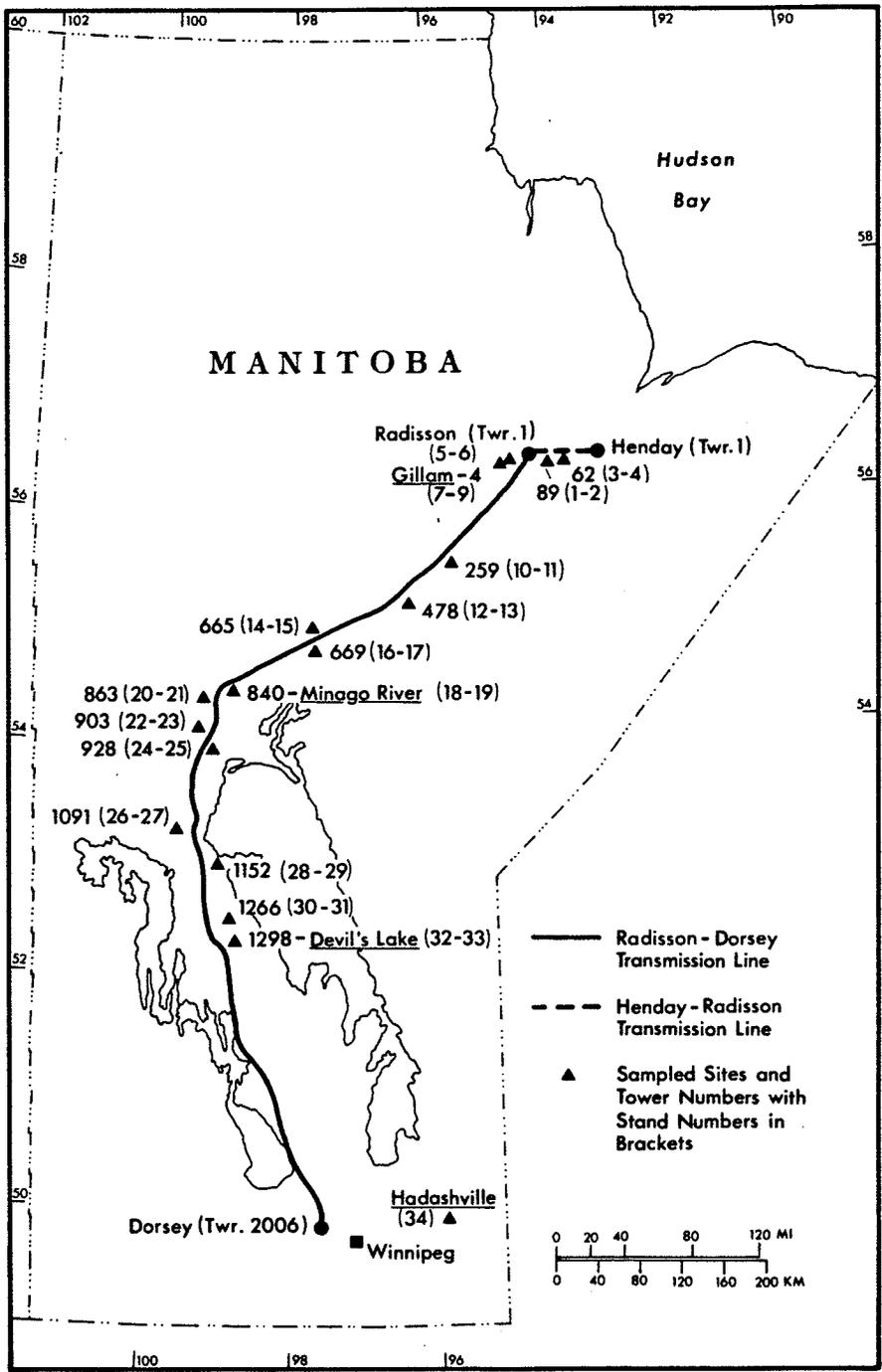
The peatlands sampled in this study are between 49° 45' and 56° 23' N latitude and 94° 25' and 99° 16' W longitude. Of the seventeen sites sampled 16 are north of 52° N and located along the electric transmission corridors extending between the Nelson River and Winnipeg, Manitoba (Fig 1).

1.2.2 Geology and topography

Within the extensive sampling area there are two major physiographic regions, the Precambrian Shield to the north and east and the Manitoba Plain to the west (Bostock 1970; Shaykewich and Weir, 1977). The Precambrian Shield is primarily composed of granitic and gneissic rocks (Teller 1984), and it has a relatively level, though hummocky topography, with a local relief ranging from 7 to 15 meters (Bostock 1970; Smith 1975). Rock outcrops are common and the bedrock is covered elsewhere by glacial drift composed mainly of granitic material (Weir 1983). The bedrock underlying the Manitoba Plain consists of limestone and dolomite of the Paleozoic era. The surficial mineral deposits are mainly lacustrine clays or modified till. Topography is flat to slightly undulating and local relief is generally less than 7 meters.

Figure 1.

Location of sites sampled along the transmission corridors, during 1983. For the major physiographic divisions and southern limit of permafrost in Manitoba, see Weir 1983.



The Manitoba Lowland receives the discharge of three major river systems which drain into the Hudson Bay via the Nelson River (Shaykewich and Weir 1977). Due to the flatness and undulating to hummocky topography much of these regions are poorly drained, which coupled with their cold and humid climate makes conditions excellent for the development of peatlands. In these regions Manitoba's greatest concentration of peatlands are found and from 25 to over 80 percent of the area is peat covered (Smith 1975).

1.2.3 Climate

Manitoba's continental climate is characterized by its large annual range of temperature over the 11 degrees of latitude (Weir 1983). The mean January temperature for Winnipeg is -19.3° C and -27.0° C for Gillam. The mean July temperature is 20.0° and 14.4° respectively (AES 1982). The average precipitation decreases in a south to north direction. The mean annual precipitation for Winnipeg is 51 cm and 42 cm for Gillam (AES 1982). According to the Thornthwaite method of climate classification 'average moisture deficit' in Manitoba decreases northward and towards the Hudson Bay. The deficit is estimated to be over 10 cm in the southwestern part of the province and less than 2.5 cm in the northeast (Smith 1975).

The study area north of 54° N is affected by permafrost. The permafrost is discontinuous in the southern section but

becomes widespread in the Gillam area (Zoltai and Tarnocai 1969). The permafrost is mostly confined to raised and some forested nonraised bogs, but is absent in wet peatlands and mineral soils (Brown 1968).

1.2.4 Vegetation

The study area falls within the Northern Coniferous Forest Zone, which is the most extensive of Manitoba's eight vegetation zones (Shay 1984). Upland vegetation is characterized by stands of White spruce (Picea glauca), Balsam fir (Abies balsamea) and Jack pine (Pinus banksiana) in varying proportions. Many stands also include Trembling aspen (Populus tremuloides) and White birch (Betula papyrifera). In the lower areas Black spruce (Picea mariana) is abundant and is co-dominant with tamarack (Larix laricina) in the wetter peatlands.

1.3 METHODS

1.3.1 Vegetation

Vegetation sampling was carried out in bogs along two electric transmission corridors (Figure 1). The Henday - Radisson line is a 42 km long DC-line with a SW-NE direction. It has a 30 meter wide right-of-way which was cleared of trees during the winter of 1977 - 1978 (Munro 1985, personal communication). The Radisson - Dorsey line is a 895 km long two-towered high voltage direct current (HVDC) transmission line. The right-of-way is approximately 120 m wide and was cleared of trees during 1967 - 1970, mostly in the wintertime (Sims 1977; MacLellan 1982).

An effort was made to disperse the sampling sites uniformly along the corridors to detect patterns of variation in species composition and the vegetation response to disturbance within and between sites on a latitudinal basis. However the final number and location of sites was determined by the geographical distribution of the bogs and their accessibility. On the Radisson - Dorsey line no suitable sampling sites were found on the southernmost 250 km part of the corridor, due to scarcity of peatlands in this region. One additional sampling site from a bog in southeastern Manitoba (Hadashville) was included in the sampling scheme to increase the latitudinal coverage of the study.

At each site along the power line corridors, vegetation was sampled separately within the disturbed right-of-way and in the adjacent forest. The undisturbed vegetation of the forest was assumed to be representative of the vegetation of the right-of-way areas prior to clearance. In the forest, sampling was carried out at least 15 m from the right-of-way to avoid possible edge effects at the forest right-of-way transition.

Vegetation was sampled in 34 stands (17 sites), each 10 x 10 m in size, which were located in homogenous areas, considered representative of the surrounding vegetation and soil conditions. Within each stand understorey vegetation (vascular plants, mosses and lichens) was sampled with 10 randomly placed quadrats 50 x 50 cm (0.25 m²), bare ground was also recorded. Cover was estimated visually from the cover classes of Braun-Blanquet (Goldsmith and Harrison 1976), and percentage cover calculated from cover class mid-points.

Nomenclature of vascular plants follows Scoggan (1978,1979), for lichens Hale and Culberson (1970) and Hale (1970), and for mosses Ireland et al. (1980).

1.3.2 Environmental factors

At each stand, depths to mineral soil, water table or permafrost were determined. Two samples of shallow ground

water were taken from each stand. Most of the samples were collected from shallow pits dug into the peat and allowed to fill with water. At the permafrost sites water was squeezed from the top 20 cm of the peat, since they generally did not have free water in them. All water samples were frozen within 8 h of sampling, stored and analyzed at a later date. In the laboratory, pH of the water samples was measured using a Fisher Accumet model 800 pH-meter, and conductivity was determined with a conductivity meter (Radiometer, Copenhagen). The conductivity results were standardized to 20° C and corrected by subtracting the conductivity contributed by hydrogen ions (Kcorr) (Sjörs 1950; Glaser et al. 1981; Jasieniuk and Johnson 1982). Water color, an indicator of dissolved organic matter content or stagnation (Jasieniuk and Johnson 1982; Glaser et al. 1981) was measured with a PYE CAM SP6 - 500 U.V. spectrophotometer as light absorbance at 320 nm. The samples were analyzed for Ca, Mg and K by atomic absorption spectrophotometry. NO₃ and PO₄ concentrations were determined by the Manitoba Provincial Soil Testing Laboratory with colorimetric analysis procedures.

From each stand two peat cores approximately 25 cm in depth were collected and stored at 4° C in the laboratory in plastic bags. At a later date the degree of humification of the peat (von Post scale) (von Post 1924, cited in Jeglum 1972) was determined from the top 10 cm of each core.

In the analysis of the environmental factors each stand was considered a homogenous environment and a mean value was used to represent each stand.

1.3.3 Data analysis

Species occurring in 3 stands or more of the 34 sampled stands were used in the data analysis. This reduced the species number from 102 to 50. Such data reductions have been found useful since they reduce distortion in multivariate analysis models and leave in the analysis species with the most continuous distribution which more closely satisfy statistical assumptions (Jasieniuk and Johnson 1982).

Prominence values (PV) for the 50 species were calculated by using the formula $PV=C F^{1/2}$ (Horton et al. 1979) where C = mean percentage cover and F = absolute frequency. These values were then used in the ordination, cluster analysis and analysis of variance calculations.

Stands were ordinated using the Detrended Correspondence Analysis (Decorana) technique (Hill 1979, Hill and Gauch 1980), which provides both stand and species ordination. Decorana is a modified version of the reciprocal averaging ordination technique (Hill 1973) that attempts to overcome its two main shortcomings, the arch-effect and the relative compression in the first axis ends (Gauch 1982). For large data sets the method has been found to give useful results

and allow ecologically meaningful interpretations (Pielou 1984), though its theoretical basis has been questioned (Austin 1985).

A cluster analysis using average linkage clustering (Pielou 1984) was computed using the BMDP1M program (Dixon and Brown 1979).

To investigate relationships between vegetation and environmental variables and to interpret the ordination axes the values for the environmental variables were plotted on the ordination diagrams. Correlation coefficients were calculated between stand positions on the first two Decorana axes and the environmental variables averages. Also, Canonical Correlation Analysis (CCA) (Gittins 1985) was performed, using stand scores from Decorana axes 1 and 2 as one variate set and the averaged values of the environmental variables (Ca, Mg, K, NO₃, PO₄, pH, conductivity, absorbance, bare ground and humification) for each stand as the other set. The analysis was run using the BMDP6M program (Dixon and Brown 1979).

To look more specifically into the effects of disturbance upon the right-of-way vegetation and environmental variables, univariate (ANOVA) multivariate MANOVA analysis of variance (Morrison 1976) tests were carried out. Stand 34 was now excluded from the analysis and the 33 right-of-way and forest stands separated and contrasted. A univariate

ANOVA was performed for the 50 most common species (occurring in 3 stands or more) and multivariate ANOVA using the species with the highest average PV scores in both the right-of-way and forest. In the model within-site variation was accounted for and separated from the disturbance (right-of-way) effects. As the data was not balanced (16 forest stands, 17 right-of-way stands) it had to be run under the General Linear Model (GLM) procedure of SAS (1979).

The same type of analysis was carried out on the environmental variables, using the averaged values from each stand.

1.4 STUDY SITES

Specific locations, sampling stand numbers, dates of sampling, and brief descriptions of each study site are provided below. The first two sites are on the Henday-Radisson line, and all the other sites are on the Radisson-Dorsey line with the exception of the Hadashville site. (Figure 1). The right-of-ways were cleared primarily by bulldozers, tractors and other heavy machinery. The tree shearing employed was a V-shaped blade mounted on the front of a caterpillar-type tractor. In the Radisson-Dorsey right-of-way, where tree density and height was considerably greater than in the Henday-Radisson right-of-way, fallen timber was piled and burnt. Clearing operations were mostly carried out during the winter-time on frozen ground (Sims 1977; Munro 1986, personal communications).

HENDAY 1 (Tower # 89-90) (Stand # 1 (ROW), 2 (forest)). 56° 23' N, 94° 36' W. June 29, 1983. On an extensive peat plateau, about 1 km east of the Radisson station. Peat thickness: not determined. Depth to permafrost: 25-30 cm.

ROW-stand is in the centre of the ROW, in an area highly disturbed by bulldozer activity when clearance occurred. Vegetation cover is greatly reduced. Tree species are absent.

"Forest"-stand is about 100 m north of ROW, where vegetation was burnt, over 20-30 (?) years ago. Tree species are absent. (Figure 2 a).

HENDAY 2 (Tower # 62-63) (Stand # 3 (ROW), 4 (forest)). 56° 17' N, 94° 25' W. July 2, 1983. On an extensive peat plateau, approximately 400 m north of highway. Peat thickness: not determined. Depth to permafrost: 25-30 cm.

ROW-stand highly disturbed by bulldozing of vegetation and surface peat, vegetation cover greatly reduced. Picea mariana scattered, (all < 1 m high, PV (prominence value) 2.4).

Forest-stand about 15 m south of ROW, with dense, stunted Picea mariana (all < 2 m high, PV 20.1).

GILLAM 1 (Tower # 4) (Stand # 5 (ROW), 6 (forest)). 56° 22' N, 94° 37' W. July 1 1983. On an access path into the right-of-way on the Radisson-Dorsey line, about 150 m east of tower 4. A peat covered basin sloping gently towards the Kettle River. Peat thickness: not determined in forest, 50-55 cm in ROW-stand. Depth to permafrost: 25-30 cm in forest, absent in ROW-stand. Watertable: 5 cm below surface in ROW-stand.

ROW-stand in centre of access path. Peat has mostly broken down, permafrost melted and the bog vegetation been replaced by sedges. Tree species are absent.

Forest-stand is in an area burnt over, 20-30 (?) years ago. Tree cover is sparse, Picea mariana (5 > 2 m high, PV 1.8) Larix laricina (3 > 2 m high, PV 2.8).

GILLAM 2 (Tower # 4-5) (Stand # 7 (ROW), 8 (forest) and 9 (ROW)). 56° 22' N, 94° 37' W. July 1 1983. On a peat plateau in a basin sloping gently towards the Kettle River. Peat thickness: not determined in stand 7 and 8; 10 - 30 cm in stand 9. Depth to permafrost: 25 - 30 cm in stands 7 and 8, thawed in stand 9.

Row-stand (# 7) is in western part of ROW, where the plateau surface has been heavily disturbed by bulldozer activity during clearance. Considerable recolonization has occurred in the stand. Picea mariana is scarce (8 < 50 cm).

Row-stand (# 9) is in the access path in the centre of the ROW, where the peat has been mostly broken down, permafrost has thawed, and the bog vegetation replaced by sedges. Tree species are absent.

Forest-stand, about 20 m west of ROW, is in an open canopy of Picea mariana (27 > 2 m high, PV 15.3) up to 4 m high. Depressions in stand with open water on top of permafrost.

JOY LAKE (Tower # 259-260) (Stand # 10 (ROW), 11 (forest)). 55° 43' N, 96° 02' W. June 30, 1983. On a peat plateau. Peat thickness: not determined. Depth to permafrost: 25 - 30 cm.

Row-stand is in the eastern part of the ROW, in an area highly disturbed by bulldozer activity, exposing peat, during clearance. Considerable recolonization has occurred. Picea mariana sparse (8 < 50 cm).

Forest-stand is about 15 meters to the east of ROW, with open canopy of Picea mariana (36 > 2 m, PV 22.3) up to about 6 m high. Larix laricina sparse (1 > 2 m). Depressions in stand with open water on top of permafrost.

BULGER LAKE (Tower # 478-479) (Stand # 12 (ROW), 13 (forest)). 55° 07' N, 97° 15' W. June 30, 1983. On a small peat plateau with a thin icelayer (could be penetrated). Peat thickness: 40-70 cm. Depth to ice: 30-35 cm. (Figure 3 a).

ROW-stand is in eastern part of ROW, which does not show signs of disturbance from clearance, except for scattered wood. Soil disturbance is minimal. Picea mariana is sparse (15 < 50 cm).

Forest-stand is about 20 m east of ROW in a fairly dense Picea mariana (45 > 2 m, PV 32.4) up to 5-6 m high.

JENPEG 1 (Tower # 665-666) (Stand # 14 (ROW), 15 (forest)). 54° 39' N, 98° 16' W. July 20, 1983. Sloping muskeg, in a peaty depression between mineral ridges less than 500 meters apart to the north and south. Bog vegetation confined to an area less than 1 ha in size. Peat thickness: 45 - 70 cm. Depth to water table: 0 - 30 cm.

ROW-stand is in western part of ROW, does not show signs of ground or vegetational disturbance from clearance. Picea mariana regeneration starting (30 < 50 cm, PV 2.2).

Forest stand is about 15 m west of ROW, with up to about 5 m high canopy of Picea mariana (18 > 2 m, PV 1.2) and Larix laricina (4 > 2 m, PV 19.1).

JENPEG 2 (Tower # 669-670) (Stand # 16 (ROW), 17 (forest)). 54° 39' N, 98° 17' W. July 19, 1983. Sloping muskeg in a small patch of bog vegetation in a peaty depression less than 1 ha in size. Peat thickness: 55-70 cm. Depth to water table: 35-40 cm.

ROW-stand in eastern part of ROW, vegetation and soil disturbances minimal at clearance. Some scattered wood. Picea mariana sparse (20 < 50 cm, PV 0.2).

Forest-stand is about 15 m east of ROW, with open canopy of Picea mariana (32 > 2 m, PV 16.4) up to about 5 m high.

MINAGO RIVER 1 (Tower # 840-841) (Stand # 18 (ROW), 19 (forest)). 54° 12' N, 99° 09' W. July 20, 1983. An extensive peat covered basin several square km in area. Peat depth: 40-60 cm. Depth to water table: 20-30 cm in ROW. Forest-stand had ice at 35 cm and no free water.

ROW-stand is in eastern part of ROW. Disturbance due to clearance minimal. Picea mariana regenerating (2 > 2 m, 15 < 2 m, PV 2.4). Larix laricina 1 < 2 m.

Forest-stand is about 15 m east of ROW, with a very dense canopy of Picea mariana (125 > 2 m, PV 56.5) up to about 3 m high.

MINAGO RIVER 2 (Tower # 863-864) (Stand # 20 (ROW), 21 (forest)). 54° 06' N, 99 ° 09' W. August 17, 1983. An extensive peat covered basin (same as Minago River 1). Peat thickness: 50-80 cm. Depth to water table: 20-60 cm.

ROW-stand is in centre of ROW. Disturbance due to clearance and management is minimal. The site is in an area where 2,4-D was applied in spots in 1973 (MacLellan 1982). It is however uncertain if the present stand area was treated with the herbicide. Picea mariana is sparse (10 < 70 cm).

Forest-stand is about 15 m east of the ROW, with an open canopy of Picea mariana (20 > 2 m, PV 18.3) up to 5 m high.

WILLIAM RIVER (Tower # 903-904) (Stand # 22 (ROW), 23 (forest)). 53° 55' N, 99° 13' W. August 17, 1983. Sloping muskeg in a peaty basin between William River and a mineral ridge ca 1.5 km south of the river. Peat depth: 45-80 cm. Depth to water table: 20-40 cm.

Row-stand is in western part of ROW. Disturbance due to clearance and management is minimal. 2,4-D application: see Minago River 2. Picea mariana considerable regeneration (38 < 2 m, PV 15.9).

Forest-stand is about 40 m west of ROW, with open canopy of Picea mariana (25 > 2 m, PV 6.9) and Larix laricina (1 > 2 m) up to about 5 m high.

LITTLE LIMESTONE LAKE (Tower # 928-929) (Stand # 24 (ROW), 25 (forest)). 53° 48' N, 99° 16' W. August 18, 1983. Sloping muskeg in a peaty basin between mineral ridges to the north (ca 1 km) and south (ca 3 km). Peat thickness: 90-105 cm. Depth to water table: 45-50 cm.

ROW-stand is in centre of ROW. Vegetation and soil disturbances minimal. 2,4-D application: see Minago River 2. Picea mariana sparse (19 < 2 m, PV 0.1).

Forest-stand is about 20 m east of ROW, with open canopy of Picea mariana (30 > 2 m, PV 11.8).

GRAND RAPIDS (Tower # 1091-1092) (Stand # 26 (ROW), 27 (forest)). 53° 08' N, 99° 16' W. August 18, 1983. In a peaty basin between the Saskatchewan River to the north (ca 1 km) and a mineral ridge to the south (ca 0.5 km). Peat thickness: 80-110 cm. Depth to water table: 45-50 cm.

ROW-stand is in eastern part of ROW. Area fairly disturbed, top peat churned and dry, numerous weedy species in stand. Bog vegetation greatly reduced. 2,4-D application: see Minago River 2. Picea mariana sparse (13 < 2 m, PV 3.1).

Forest-stand is about 20 m east of ROW, in open canopy of up to about 6 m high Picea mariana (20 > 2 m, PV 18.9) and Larix laricina (9 > 2 m, PV 4.5).

LONG POINT (Tower # 1152-1153) (Stand # 28 (ROW), 29 (forest)). 52° 55' N, 99° 10' W. July 21, 1983. Sloping muskeg in a peat filled basin, ca 0.5 km wide between mineral ridges to the north and south. Peat thickness: 130 cm. Depth to water table: 30-40 cm.

ROW-stand is in western part of ROW. Vegetation and soil disturbance due to clearance minimal. Tordon 10K was applied in the ROW in 1982, which has caused extensive damages of Sphagnum fuscum hummocks and associated vegetation, and exposed peat. Picea mariana regenerating prior to application (5 > 2, PV 2.8), trees beginning to die after the Tordon 10 K application.

Forest-stand is about 20 m west of ROW, with open canopy up to 6 m high of Picea mariana (21 > 2 m, PV 16.9) and Larix laricina (4 > 2 m, PV 3.1).

DEVIL'S LAKE 1 (Tower # 1266-1267) (Stand # 30 (ROW), 31 (forest)). 52° 29' N, 98° 53' W. August 16, 1983. Sloping muskeg in an extensive peat covered basin. Peat thickness: 50-55 cm. Depth to water table: 5-20 cm.

ROW-stand is in the eastern part of the ROW. Vegetation is highly disturbed, bog vegetation has mostly disappeared and been replaced by sedges and grasses. Peat hummocks have nearly degenerated. Tordon 101 was applied in the ROW in 1979. Picea mariana very sparse (1 < 50 cm).

Forest-stand is about 30 m east of ROW, with a dense canopy up to about 6 m high of Picea mariana (10 > 2 m, PV 46.4) and Larix laricina (6 > 2 m).

DEVIL'S LAKE 2 (Tower # 1298-1299) (Stand # 32 (ROW), 33 (forest)). 52° 21' N, 98° 53' W. August 15, 1983. A peat covered basin. Peat thickness: 80-90 cm. Depth to water table: 25 - 35 cm. (Figure 3 b).

ROW-stand is in the western part of the ROW. Vegetation and soil disturbance due to clearance minimal. Tordon 101 applied in ROW in 1979, has caused extensive damages of Sphagnum fuscum hummocks and associated vegetation, and exposed peat. Picea mariana regenerating (55 < 1 m, PV 7.4).

Forest-stand is about 20 m west of ROW, with open canopy up to 6 m high of Picea mariana (23 > 2 m, PV 8.3) and Larix laricina (2 > 2 m, PV 3.1).

HADASHVILLE (Stand # 34). 49° 45' N, 95° 56' W. July 7, 1983. An extensive, drained and raised bog, presently being mined for Sphagnum peat. Stand approximately 50 m east of a drainage ditch by peat mining area, ca 1 km south of the peat packing plant. Peat thickness: 3.5 m. Depth to water table: 45 - 55 cm. Open bog with scattered Picea mariana (3 > 2 m, PV 1.8) and Larix laricina (2 > 2 m). (Figure 2 b).

Figure 2.

Oblique aerial views of Henday 1 and Hadashville sites.

A) Henday 1 site, in the Gillam area, seen from south. The northernmost site in the study area. Arrows point at ROW (right) and undisturbed (left) peat plateaus, that represent the areas of vegetation sampling (stands 1 and 2) and soil respiration measurements (chapter 2). The raised peat plateaus have permafrost in them, but the lower fen areas between the plateaus are free of permafrost. (June 30, 1983).

B) Hadashville site, seen from north-east. The southernmost site in the study area. Peat harvesting field is to the right and peat packing plant on the lower left. Arrow points at the general study area of vegetation sampling (stand 34) soil respiration study (chapter 2) and herbicidal experiment (chapter 3). (May 20, 1982).



Figure 3. Bulger Lake and Devil's Lake 2 sites.

A) Oblique aerial view of Bulger Lake site (stands 12 and 13, twrs. 478-479), which was located on a peat plateau in southern part of the permafrost area. The right-of-way stand (12) was positioned in the centre of the gray, lichen-rich patch, and the forest stand (13) in the adjacent forest. Cladina mitis is the dominant understorey plant in both stands, but its prominence has been reduced in the right-of-way stand which has near 15% cover of bare ground. In the DCA-ordination, stand 12 has the lowest score on axis 1. (June 30, 1983).

B) Stand 33 (Devil's Lake 2, forest, twrs. 1298-1299). Example of an undisturbed and moist Sphagnum-rich southern forest stand on shallow peat. Stunted Picea mariana and scattered Larix laricina, with Sphagnum fuscum dominant in the understorey. Other species of relatively high abundance are Ledum groenlandicum, Empetrum nigrum, Vaccinium vitis-idaea, Sarracenia purpurea, Carex aquatilis and Juncus balticus. In the DCA-ordination the stand was among the southern forest stands which had the lowest scores on axis 2. (August 15, 1983).



1.5 RESULTS

1.5.1 Vegetation analysis

1.5.1.1 Ordination

Two-dimensional ordination of the 34 stands sampled is shown in Figure 4. With the exception of stands 5, 9 and 30 there is not a distinct separation between right-of-way and forest stands. This indicates that the disturbances have not caused major changes in species composition or abundances in the right-of-way community. The ordination however brings out certain patterns in the sampled vegetation and axis 1 and axis 2 can be interpreted as major gradients.

Axis 1, which accounts for 43% of the total variance accounted for by the first four axes, is principally a moisture gradient that also includes a disturbance factor (Figure 5 a-d). Along this axis stands are ordered from dry lichen-rich to wet sedge-rich with moist Sphagnum-dominated stands in the centre. The most outstanding change in vegetation along the axis is the decline in abundance of Cladina mitis and sudden increase in abundance of Carex aquatilis on the upper half of the axis. This becomes apparent when the prominence scores for the major plant species are plotted over the stand positions on the two-dimensional ordination graph (Figure 6 a-1).

Figure 4.

DCA ordination of all the stands sampled in the vegetation study. Forest stands are indicated by squares and right-of-way stands by diamonds. Stand numbers are shown within symbols.

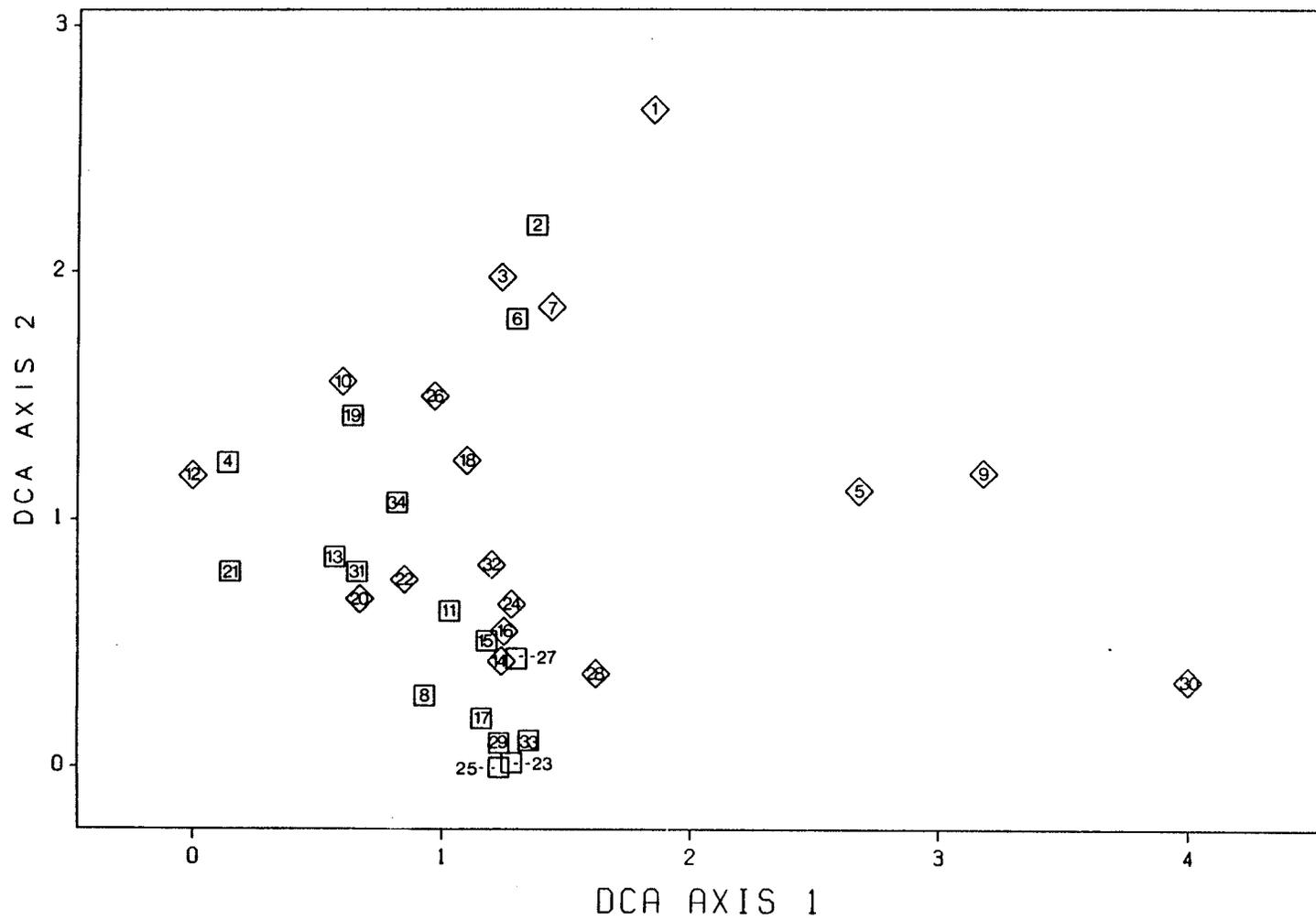


Figure 5.

Change in understorey vegetation along the moisture gradient.

The figure demonstrates the main change in understorey vegetation encountered along DCA axis 1, from dry lichen-rich stands, through moist Sphagnum-rich stands, to wet sedge-rich stands.

A) In stand 12 (ROW). A dry, raised stand on a peat plateau, without free water. Cladina mitis is the dominant plant (PV 35). Sphagnum fuscum has a very low abundance (PV < 1) in the stand. (June 30, 1983).

B) In stand 20 (ROW). Increasing moisture and abundance of S.fuscum (PV 26) and declining lichen cover, C. mitis is the dominant lichen (PV 15). (August 17, 1983).

C) In stand 23 (forest). A moist stand with high cover of S.fuscum (PV 58), lichens are nearly absent (PV < 1). (August 17, 1983).

D) In stand 9 (ROW). A disturbed wet sedge-rich stand where Carex aquatilis has gained dominance. Sphagnum and lichens are absent. (July 3, 1983).

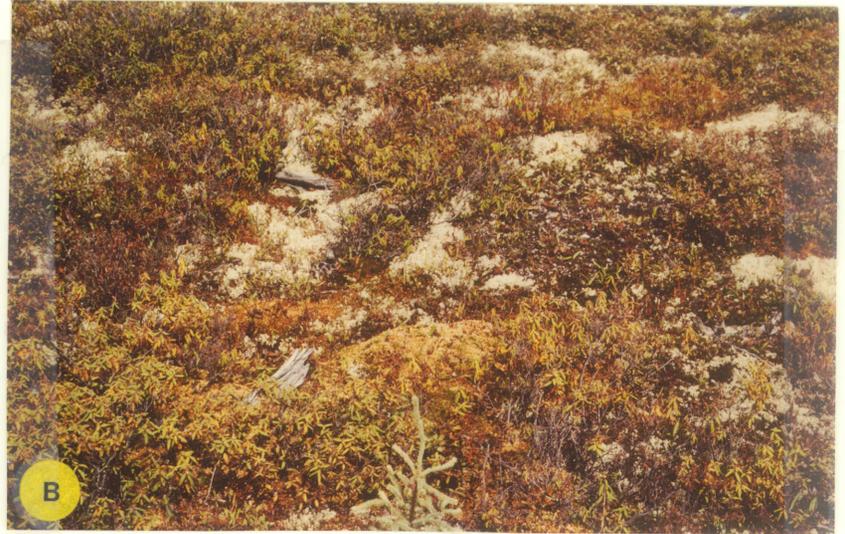
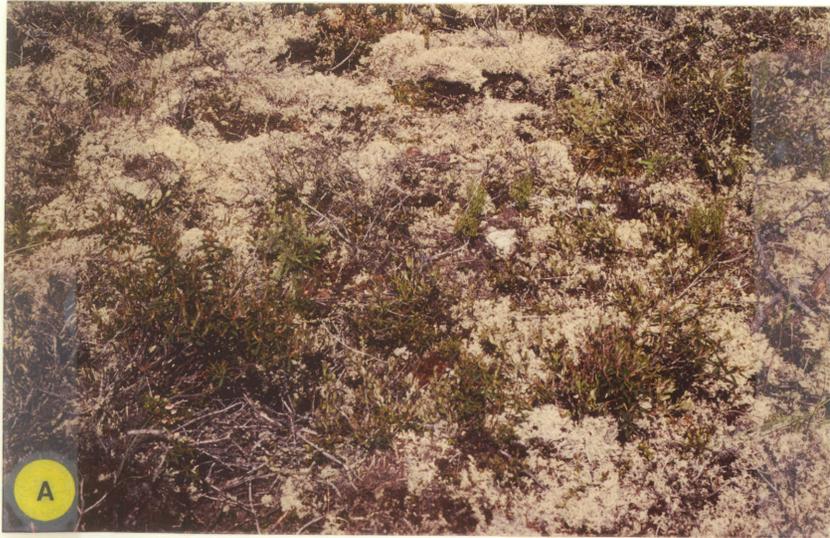


Figure 6. (Page 29-34)

Prominence values of twelve important species of the bog communities sampled, plotted over stand positions on the two-dimensional ordination. Stand positions are the same as in Figure 4.

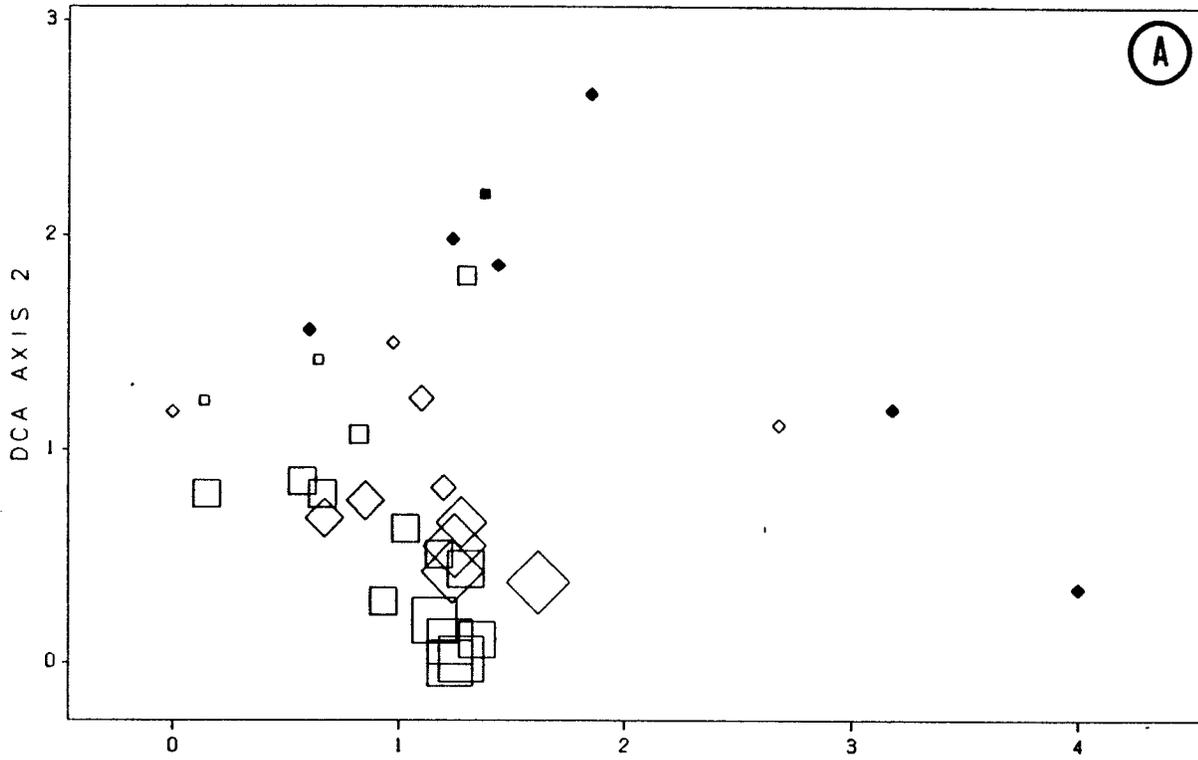
Forest stands are indicated by squares and right-of-way stands by diamonds. Square and diamond sizes indicate prominence of species in stands according to the following scale:

▪	◆	ABSENT
◻	◊	PV < 4.5,
◻	◊	PV 4.5 - 14.4,
◻	◊	PV 14.5 - 29.4,
◻	◊	PV 29.5 - 49.4,
◻	◊	PV > 49.5

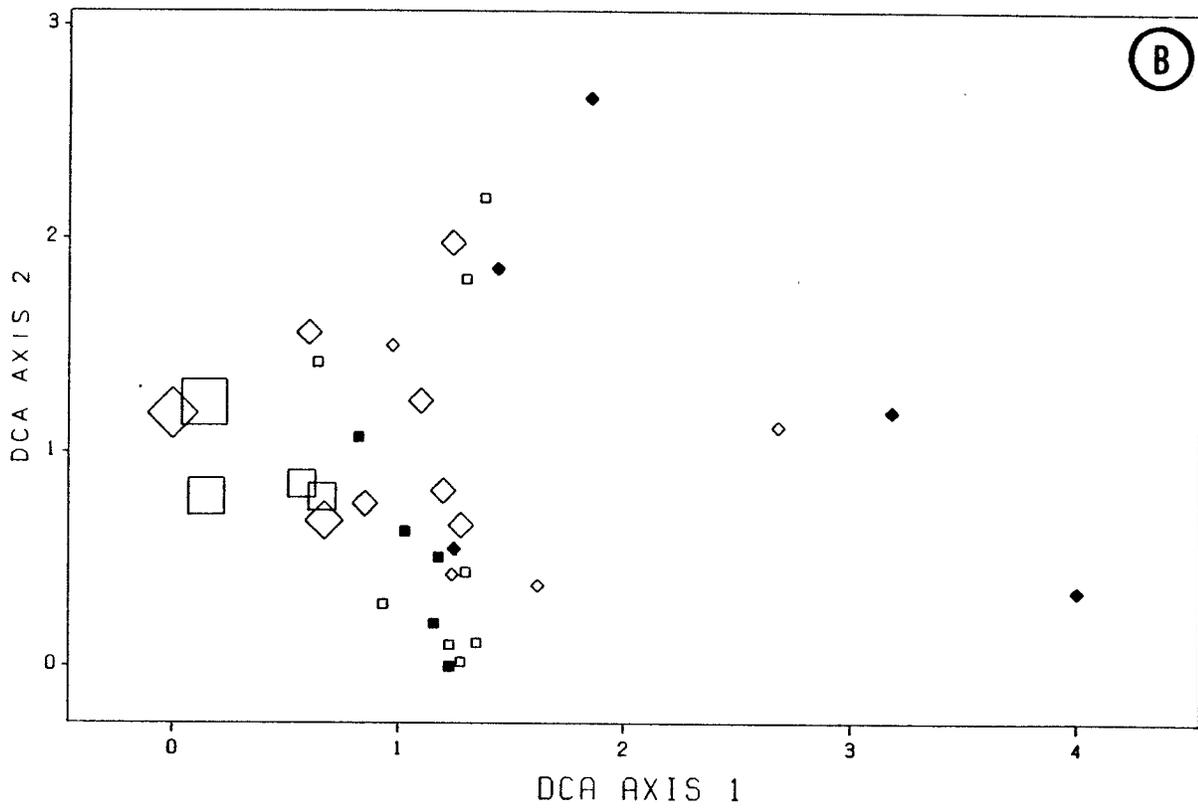
The species are:

- | | |
|-----------------------------------|---------------------------------|
| A) <u>Sphagnum fuscum</u> , | B) <u>Cladina mitis</u> |
| C) <u>Carex aquatilis</u> , | D) <u>Polytrichum strictum</u> |
| E) <u>Ledum groenlandicum</u> , | F) <u>Rubus chamaemorus</u> |
| G) <u>Vaccinium vitis-idaea</u> , | H) <u>Oxycoccus microcarpus</u> |
| I) <u>Drosera rotundifolia</u> , | J) <u>Cladina rangiferina</u> |
| K) <u>Pleurozium schreberi</u> , | L) <u>Ceratodon purpureus</u> |

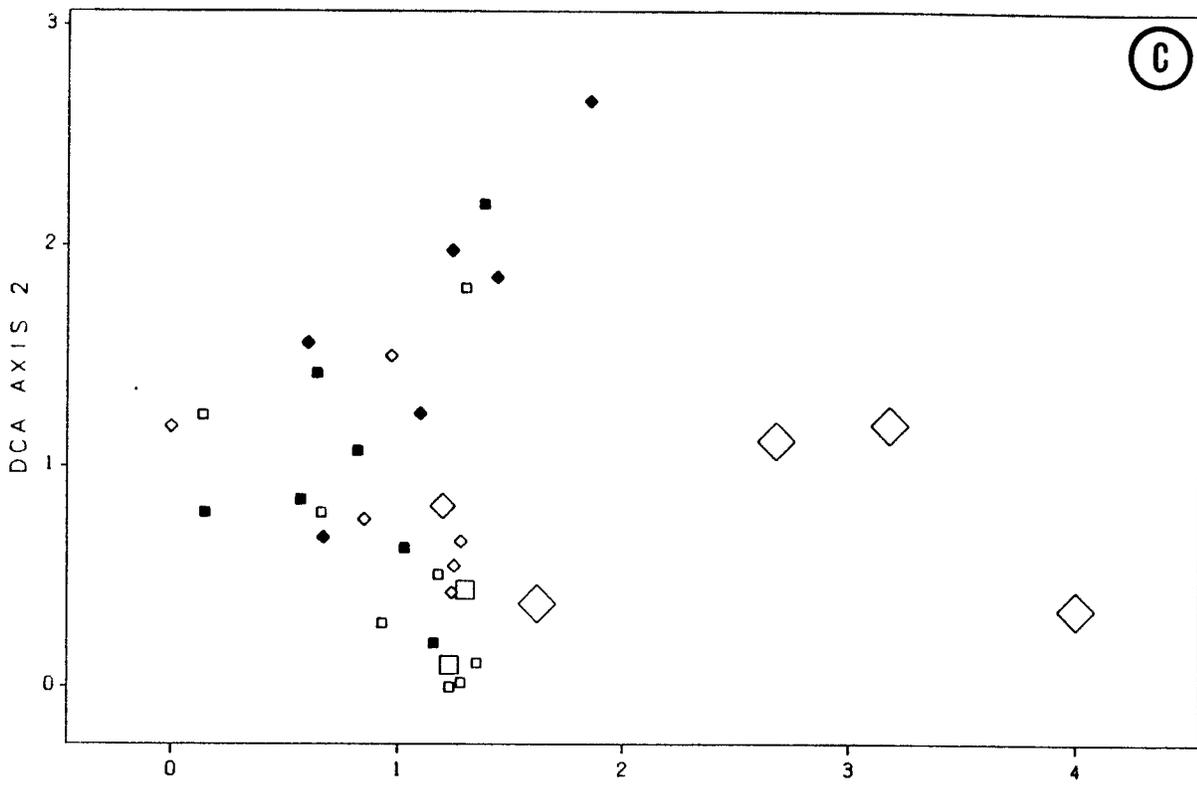
SPHAGNUM FUSCUM



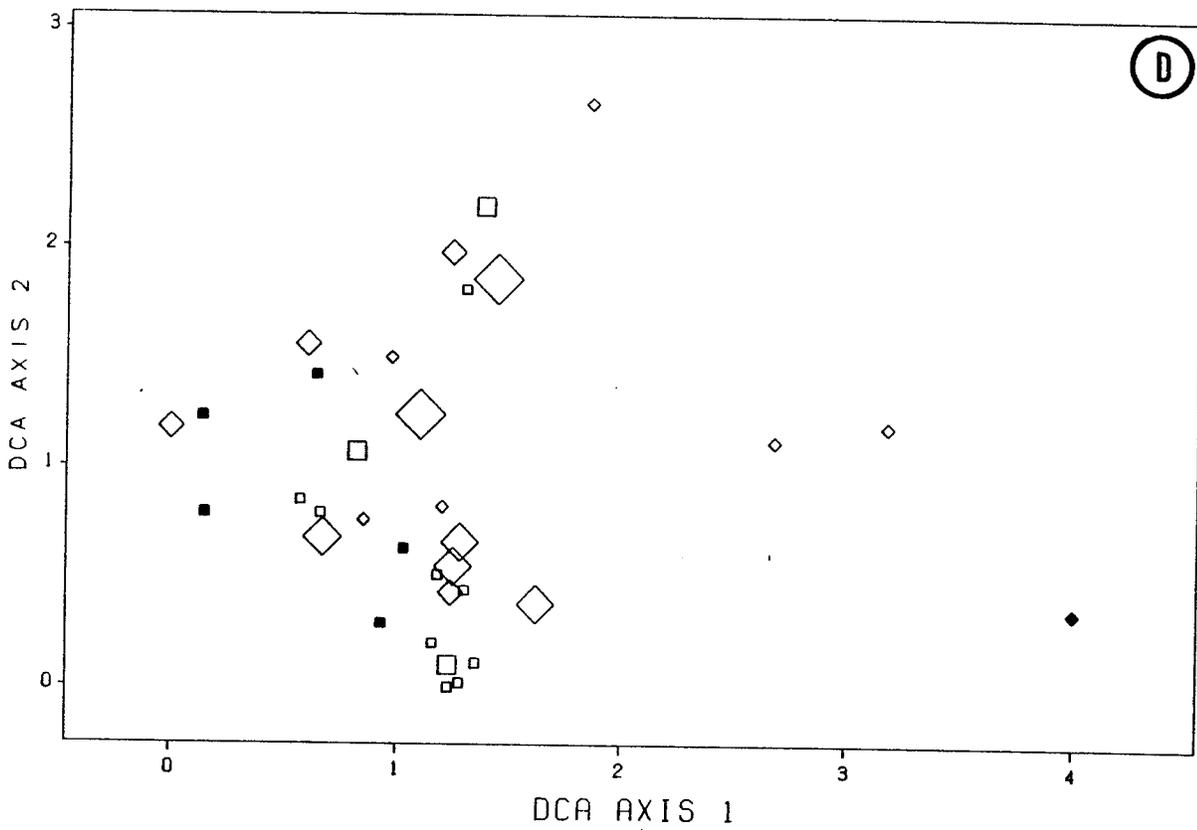
CLADINA MITIS



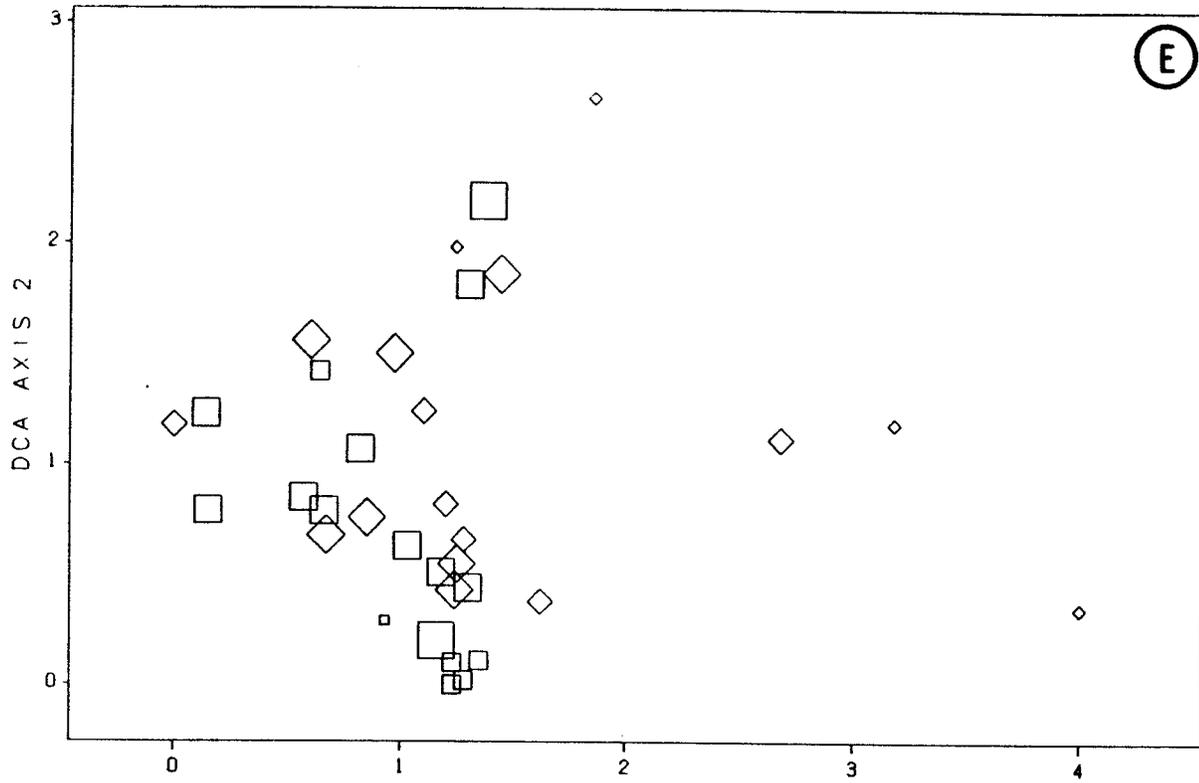
CAREX AQUATILIS



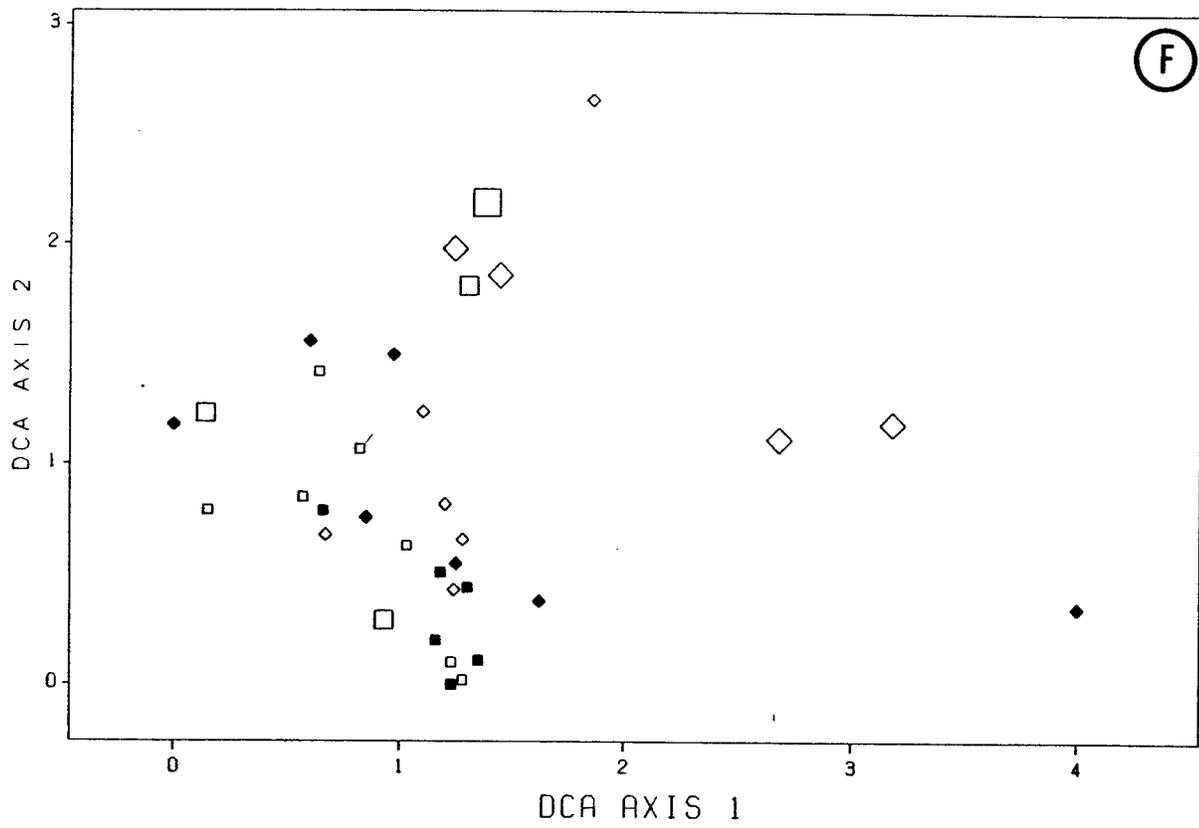
POLYTRICHUM STRICTUM



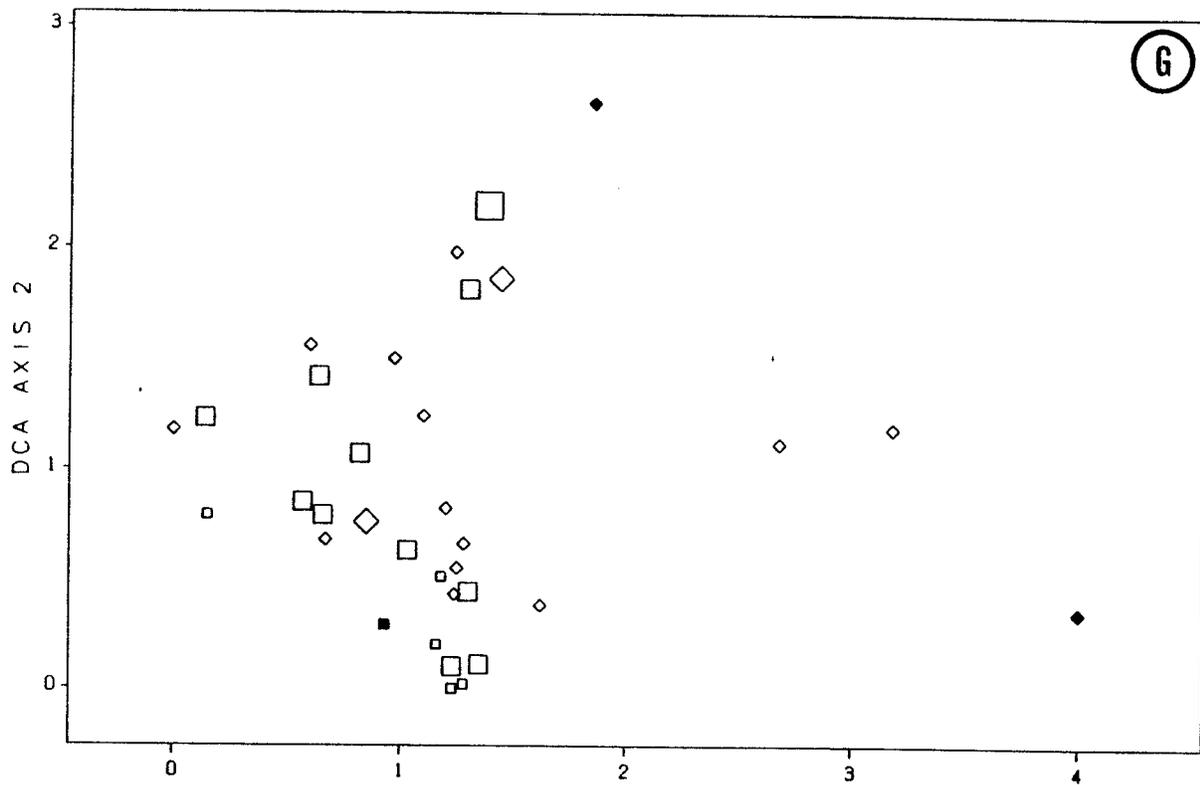
LEDUM GROENLANDICUM



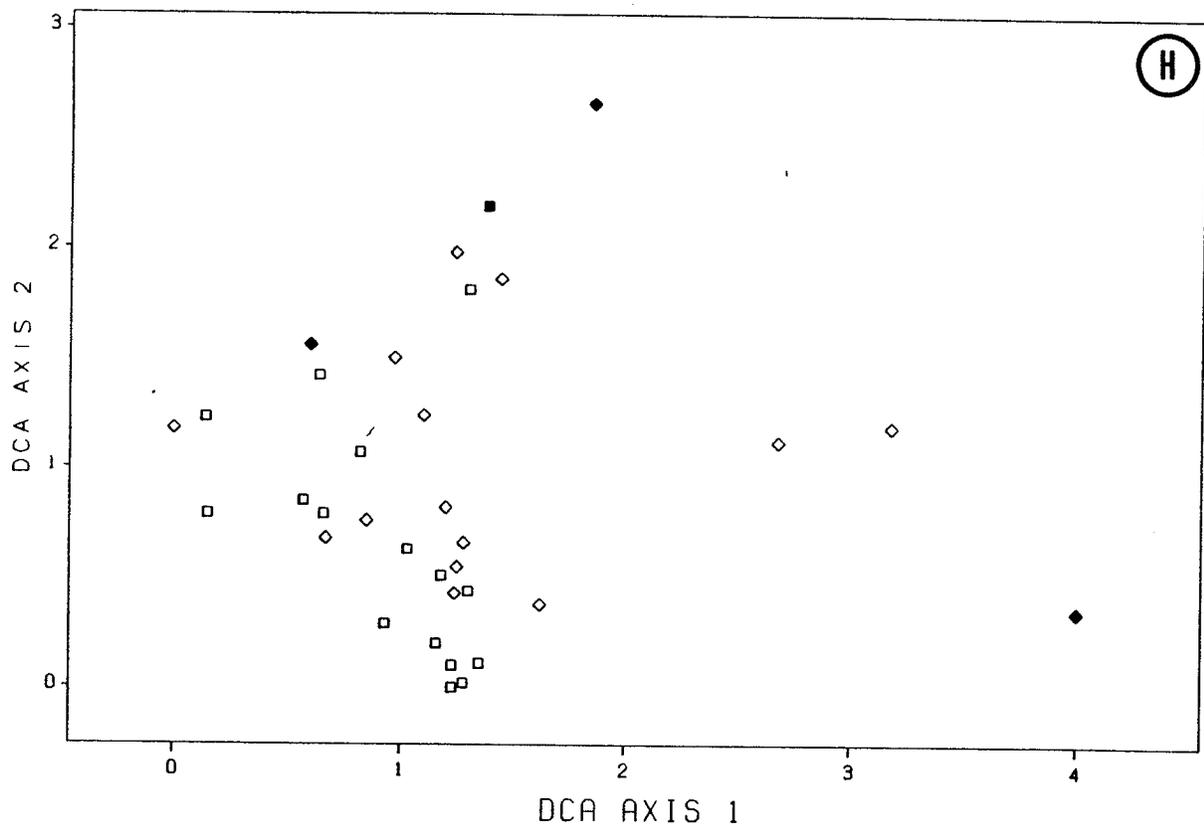
RUBUS CHAMAEMORUS



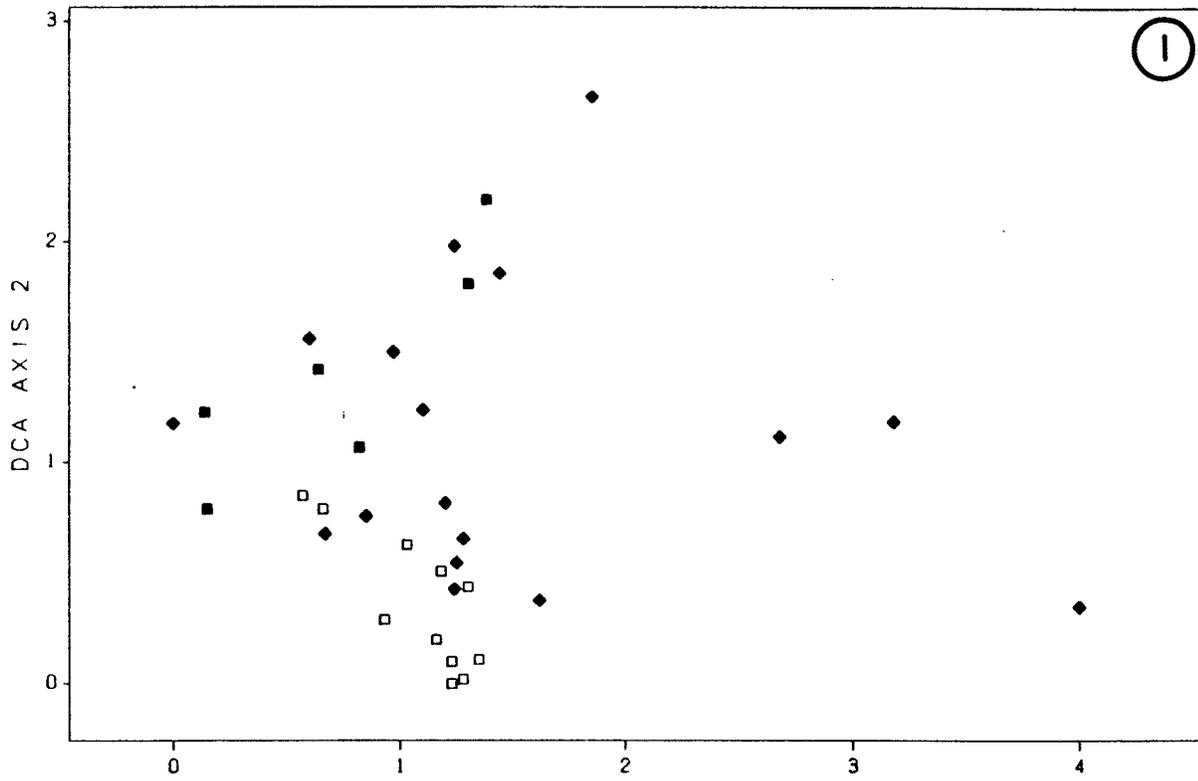
VACCINIUM VITIS-IDAEA



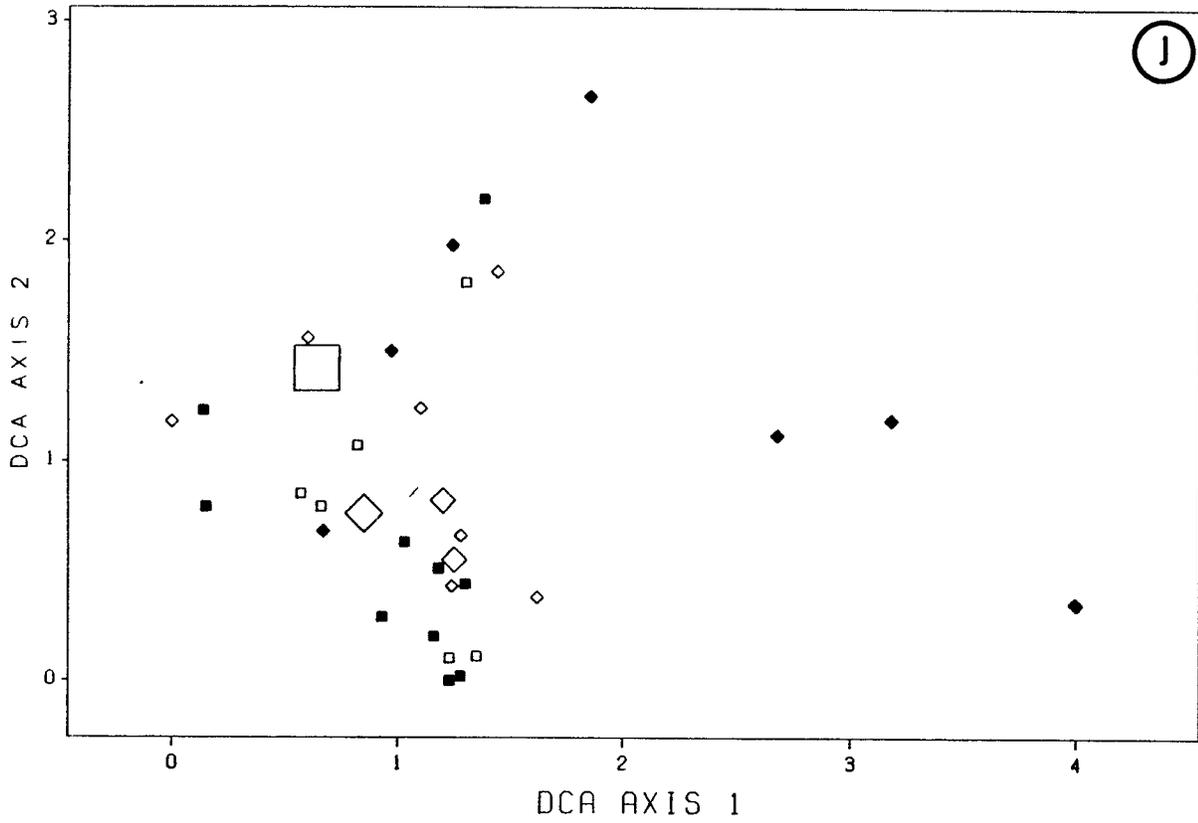
OXYCOCCUS MICROCARPUS



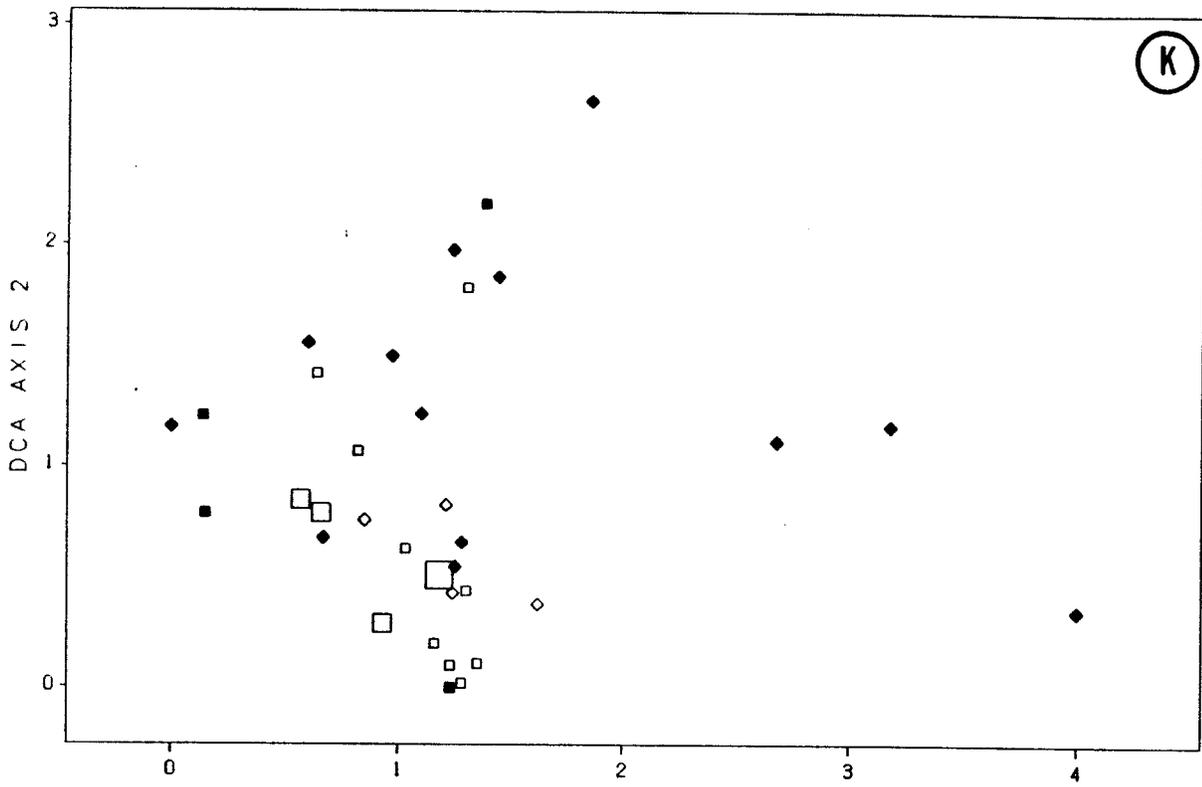
DROSERA ROTUNDIFOLIA



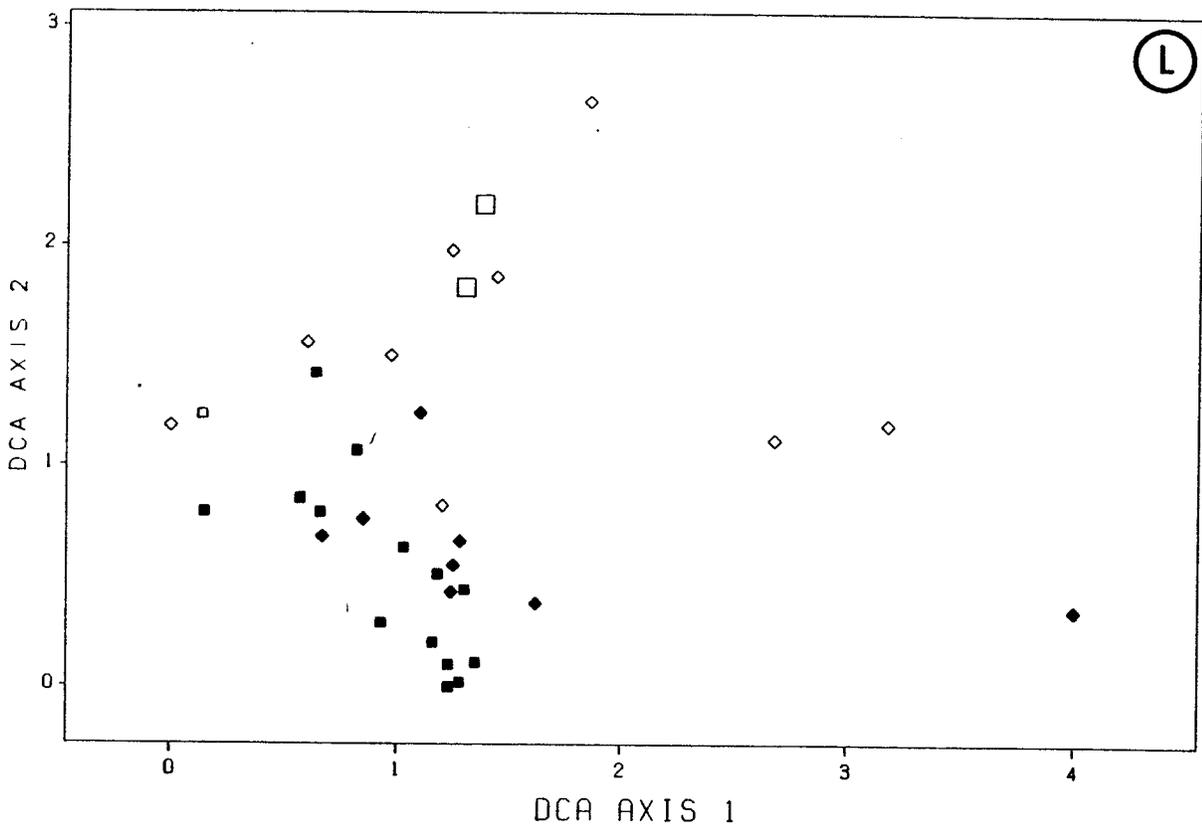
CLADINA RANGIFERINA



PLEUROZIUM SCHREBERI



CERATODON PURPUREUS



The outlying stands 5, 9 and 30 are all right-of-way stands of extreme disturbance. Stands 5 and 9 are from centres of vehicle tracks in which much of the natural bog vegetation and top peat has been removed, the permafrost has thawed and the ground left with open pools, where Carex aquatilis is the principal colonizer (Figure 6 b). The abundance of the original bog plants is low and the extent of bare ground is considerable. In these stands the moss Drepanocladus aduncus is found, which is common in the minerotrophic fens of the region (Jeglum 1972). In stand 5, Plagiomnium ellepticum, another moss species characteristic of wet ground (Crum 1983) is found. In these stands there are also an apparent increases in the abundances of Equisetum arvense, Calamagrostis spp., and Salix spp.. In Stand 30 there has been a dramatic decline in the bog vegetation and almost a complete removal of the most characteristic bog plants (ericoids, Sphagnum, lichens), (Figure 7 d). These bog plants have been replaced by sedges, grasses and other species more characteristic of fen vegetation, primarily Carex aquatilis and to a lesser extent Carex rostrata, Carex capillaris, Bromus ciliatus, Deschampsia caespitosa, and Muhlenbergia glomerata. The moss Tomenthypnum nitens dominates the ground layer of the stand, but it is very common in the minerotrophic fens of the region (Sjörs 1961, 1963; Sims 1977; Slack et al. 1980). Plagiomnium ellepticum is also found in the stand. Lichens are absent in stands 9 and 30, but stand 5 had Cladonia gracilis and C. cornuta in low

Figure 7. Right-of-way stands.

A) Stand 1 (Henday 1, twrs. 89-90), on a scraped peat plateau in the northernmost part of the study area. Example of a raised and dry, highly disturbed stand with extensive bare ground (85%) and low vegetation cover. Most prominent plant species are Rubus chamaemorus, Ledum groenlandicum, Polytrichum strictum and Ceratodon purpureus. In the DCA-ordination the stand had the highest score on Axis 2. (July 11, 1983).

B) Stand 10 (Joy Lake, twrs. 259-260), on a plateau. Example of a disturbed, raised and lichen-rich stand. Ledum groenlandicum is dominant in the shrub layer, but Polytrichum strictum and Cladina species are most prominent in the ground layer. Bare ground cover is near 15%. Sphagnum fuscum is absent. (June 30, 1983).

C) Stand 20, (Minago River 2, twrs. 863-864), from an extensive peat covered basin in the central part of the study area. Vegetation cover is complete and dominant plants are: Chamaedaphne calyculata, Ledum groenlandicum, Sphagnum fuscum, Polytrichum strictum and Cladina mitis. At the time of sampling Tordon 10K had recently been applied to the right-of-way and browning of some deciduous trees and shrubs was apparent. (August 17, 1983).

D) Stand 30, (Devil's Lake 1, twrs. 1266-1267), from southern part of study area. Example of a wet and extremely disturbed site. Characteristic bog plants have largely disappeared and been replaced mostly by sedges and grasses. Carex aquatilis is the dominant graminoid and Tomenthypnum nitens is abundant on the wet ground underneath the graminoids. In the DCA-ordination the stand had the highest score on axis 1. (August 16, 1983).



abundance on the most elevated and dry areas. The cause of disturbance in stand 30 is not clear, but the use of herbicides is suspected.

Axis 2 which accounts for 29% of the total variance can be interpreted as a disturbance gradient. The major change in vegetation along the axis is the decline in abundance of Sphagnum fuscum, which in the study was found to be the most abundant species overall (Figure 6 a). Also there is a consistent increase in bare ground along the axis (Figure 8). Stands 1, 2, 3, and 7, which have the highest scores on axis 2 are all from northern sites with permafrost. The right-of-way stands 1 (Figure 7 a), 3 and 7 represent sites where most of the vegetation and a thin layer of surface peat was scraped off during line clearance, leaving mostly exposed Sphagnum peat as substrate. The vegetation of the forest stands, 2 and 6, shows evidence of recovering after fire and like in the right-of-way stands plant cover is reduced. The vegetation is characterized by pioneer species which are able to colonize disturbed areas and increase in abundance within a short time after the disturbance. Examples of this are Ceratodon purpureus, Pohlia nutans, Polytrichum strictum, Cladonia gracilis, Ledum groenlandicum, Rubus chamaemorus and Vaccinium vitis-idaea (Jasieniuk and Johnson 1982).

The species ordination (Figure 9) reflects the same major trends as the stand ordination. On Axis 1 the species with the lowest scores are mostly lichens. Cladina mitis, Icma-

Figure 8.

Prominence values of bare ground, plotted over stand positions on the two-dimensional ordination.
(See Figure 4 & 6 for further details).

Scale:

□	◇	PV < 10.0
□	◇	PV 10.0 - 50.0
□	◇	PV > 50.0

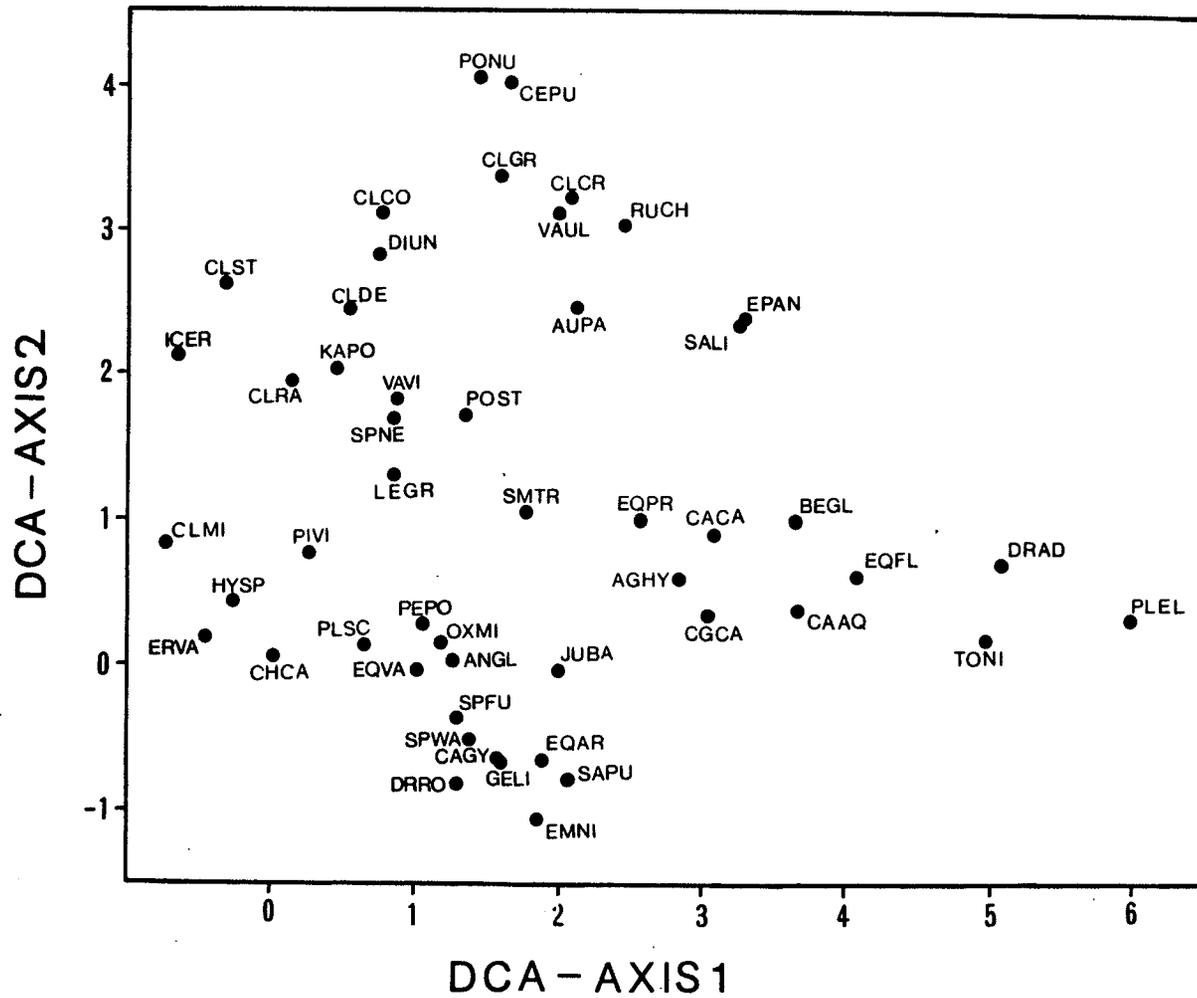
Figure 9.

Scatter figure for the 50 species occurring in 3 or more of the 34 stands sampled. The first axis proceeds from dry and raised lichen-rich stands (left), through moist Sphagnum-rich stands to wet and disturbed sedge-rich stands on the right. Axis 2 represents a change from relatively undisturbed Sphagnum-rich stands (below) to disturbed stands (above) characterized by exposed peat and the presence of early colonizers of disturbed ground.

The species names are:

AGHY= <u>Agrostis hyemalis</u>	ANGL= <u>Andromeda glaucophylla</u>
AUPA= <u>Aulocomnium palustre</u>	B EGL= <u>Betula glandulosa</u>
CAAQ= <u>Carex aquatilis</u>	CACA= <u>C. capillaris</u>
CAGY= <u>C. gynocrates</u>	CEPU= <u>Ceratodon purpureus</u>
CHCA= <u>Chamaedaphne calyculata</u>	CGCA= <u>Calamagrostis</u> spp.
CLCO= <u>Cladonia cornuta</u>	CLCR= <u>C. crista-tella</u>
CLDE= <u>C. deformis</u>	CLGR= <u>C. gracilis</u> ,
CLMI= <u>Cladina mitis</u>	CLST= <u>C. stellaris</u>
CLRA= <u>C. rangiferina</u>	DIUN= <u>Dicranum undulatum</u>
DRAD= <u>Drepanocladus aduncus</u>	DRRO= <u>Drosera rotundifolia</u>
EMNI= <u>Empetrum nigrum</u>	EPAN= <u>Epilobium angustifolium</u>
ERVA= <u>Eriophorum vaginatum</u>	EQAR= <u>Equisetum arvense</u>
EQFL= <u>E. fluviatile</u>	EQPR= <u>E. pratense</u>
EQVA= <u>E. variegatum</u>	GELI= <u>Geocaulon lividum</u>
HYSP= <u>Hylocomium splendens</u>	ICEC= <u>Icmadophila ericetorum</u>
JUBA= <u>Juncus balticus</u>	KAPO= <u>Kalmia polifolia</u>
LEGR= <u>Ledum groenlandicum</u>	OXMI= <u>Oxycoccus microcarpus</u>
PEPO= <u>Peltigera polydactyla</u>	PIVI= <u>Pinguicula villosa</u>
PLEL= <u>Plagiomnium ellepticum</u>	PLSC= <u>Pleurozium schreberi</u>
PONU= <u>Pohlia nutans</u>	POST= <u>Polytrichum strictum</u>
RUCH= <u>Rubus chamaemorus</u>	SALI= <u>Salix</u> spp.
SAPU= <u>Sarracenia purpurea</u>	SMTR= <u>Smilaciana trifolia</u>
SPFU= <u>Sphagnum fuscum</u>	SPNE= <u>S. nemoreum</u>
SPWA= <u>S. warnstorffii</u>	TONI= <u>Tomenthypnum nitens</u>
VAUL= <u>Vaccinium uliginosum</u>	VAVI= <u>V. vitis-idea</u>

SPECIES ORDINATION



dophila ericetorum, Cladina stellaris, Cladina rangiferina, and Cladonia deformis are among the 10 species with the lowest scores. The other species are Eriophorum vaginatum, Hylacomium splendens, Chamaedaphne calyculata, Pinguicula villosa and Kalmia polifolia. The mosses Plagiomnium ellepticum, Drepanocladus aduncus and Tomenthypnum nitens have the highest scores on the axis. Following the mosses are: Equisetum fluviatile, Betula glandulosa, Carex aquatilis, Epilobium angustifolium, Salix spp., Carex capillaris, and Calamagrostis spp.. On axis 2 the species with the lowest score are Empetrum nigrum, Drosera rotundifolia, Sarracenia purpurea, Equisetum arvense, Geocaulon lividum, Carex gynocrates, Sphagnum warnstorffii, S.fuscum, Equisetum variegatum and Juncus balticus, but the species with the highest scores on the axis are Pohlia nutans, Ceratodon purpureus, Cladonia gracilis, C.cristatella, Vaccinium uliginosum, C.cornuta, Rubus chamaemorus, Dicranum undulatum, Cladina stellaris and Aulacomnium palustre.

1.5.1.2 Cluster analysis

Four major clusters were formed at 30% level of similarity, but one stand did not join these clusters (Figure 10). When the clusters are superimposed on the stand ordination (Figure 11) it becomes apparent that the two techniques bring out the same major trends in the vegetation. No overlapping between clusters occurs and their patterns are not complicated.

Figure 10.

Cluster analysis. Average linkage clustering of all the stands sampled in the vegetation study. Clusters formed at 30% relative similarity. Stand numbers are shown below.

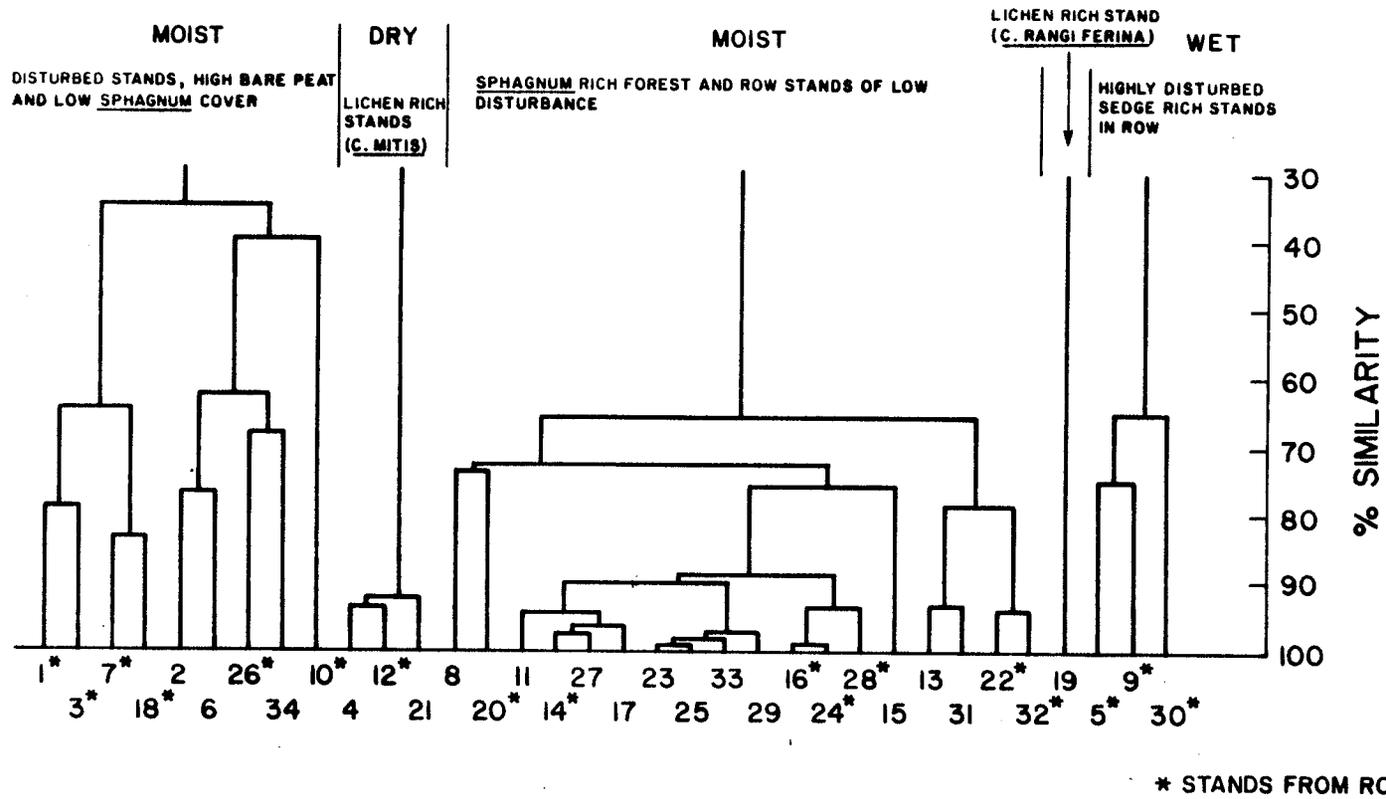
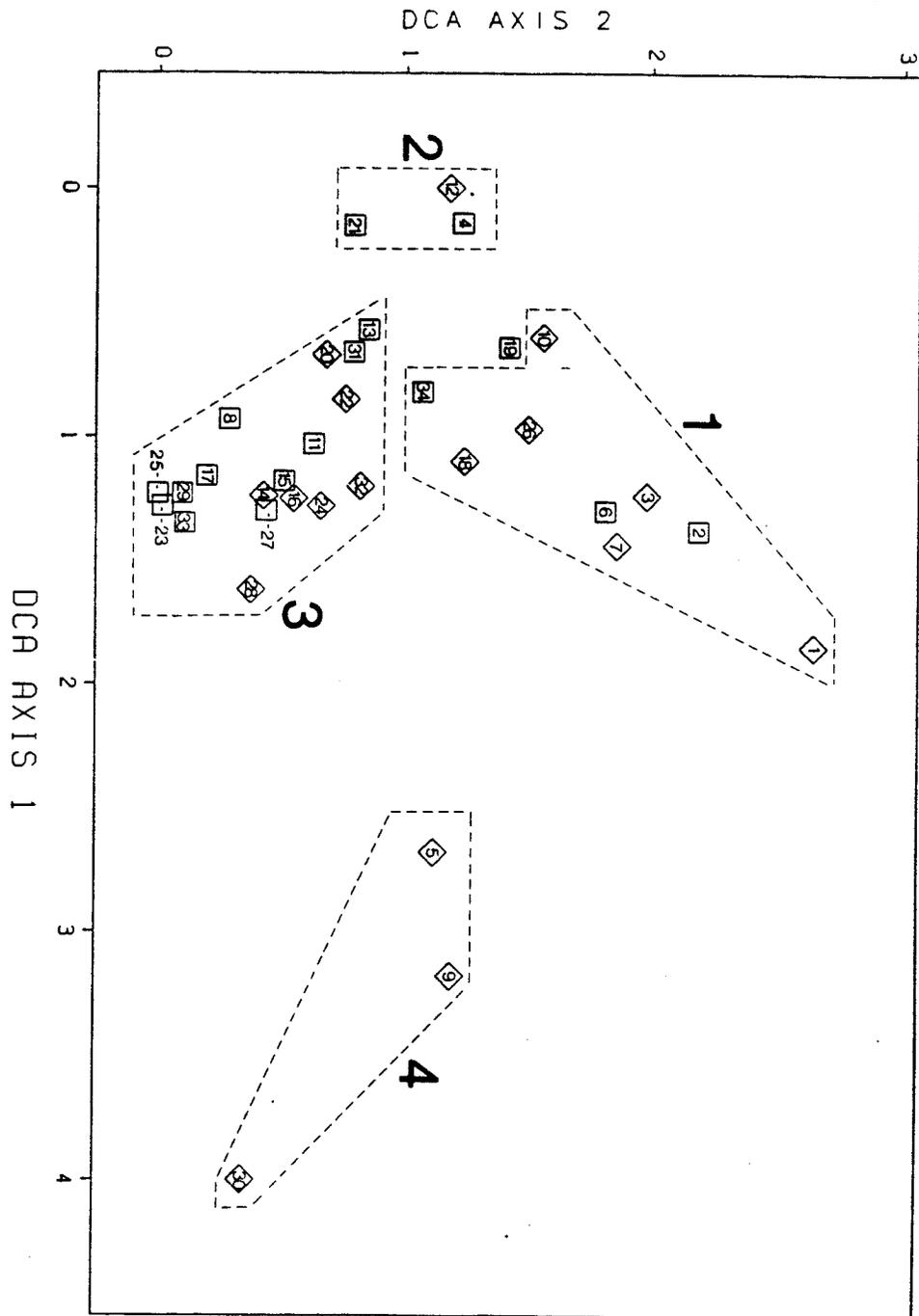


Figure 11.

Cluster analysis results compared with the stand ordination results obtained in DCA. The four main clusters formed are shown with dashed lines and are drawn on the two-dimensional ordination. Cluster numbers refer to the four main clusters shown in Figure 10.

- 1: Moist, disturbed stands with extensive bare peat and low Sphagnum cover.
- 2: Dry, lichen-rich stands.
- 3: Moist, Sphagnum-rich stands of low disturbance.
- 4: Highly disturbed, sedge-rich stands in ROW.



In group 1 there are nine stands, amongst them the northern and southernmost stands sampled. Six of the stands are right-of-way stands and three stands are forest stands. What all the stands have in common is the absence or a low abundance of Sphagnum fuscum and a relatively high abundance of Polytrichum strictum and bare ground. These stands are from disturbed sites where the disturbance has not led to considerable alteration in substrate composition or moisture levels. At first it may seem surprising to find the southernmost stand sampled (34) in this cluster, which is mostly made up of disturbed northern stands. Surficial disturbance has not occurred in the stand and its vegetation has not been disturbed directly. However the bog that the stand is from has been drained which may explain the presence of the stand in the cluster.

In group 2 there are only three stands, one right-of-way stand and two forest stands. The primary characteristic of these stands is the very high abundance of Cladina mitis in the understorey. Sphagnum fuscum is present in all the stands but it is not abundant probably because of their dry nature.

Group 3 is made up of eighteen or 53% of the stands sampled. Seven are right-of-way stands and eleven forest stands. These stands represent bog vegetation in a natural or relatively undisturbed state. The vegetation is characterized by the presence and high abundance of Sphagnum fus-

cum. With the exception of stands 13, 31 and 32 it is the dominant plant species in the stands of the cluster. There is little or no bare ground in the stands and pioneer species characteristic of disturbances are of low abundance or absent. Two common bog plants, which were strongly associated with Sphagnum-hummocks, Drosera rotundifolia (Figure 6 i) and Sarracenia purpurea are confined to stands within this cluster.

Group 4 is made up of the three wet and highly disturbed right-of-way stands discussed previously. Carex aquatilis is dominant in all the stands. Sphagnum fuscum is only present in one of the stands and in low abundance, while lichens are nearly absent.

Stand 19 did not join any of the clusters. According to the ordination results it is in vegetation composition most similar to the stands in cluster 1 (Figure 4) and might therefore be expected to join that cluster. What is exceptional about the stand is the extremely high abundance of Cladina rangiferina the dominant understorey plant. In no other stand does the species gain an overwhelming dominance, which may explain the stand's position in the cluster analysis.

1.5.2 Environmental gradients

1.5.2.1 Water chemistry

The means of the environmental factors determined for each stand vary considerably (Table 1). Their relation to each other and the two ordination axes are shown in Table 2. Correlations between the environmental factors and stand positions on the first two Decorana axes indicate that the axes represent complex gradients of intercorrelated factors. Axis 1 has positive and significant correlation with Ca, Mg, pH, conductivity and humification, and a significant negative correlation with NO_3 . Axis 2 has a positive and significant correlation with NO_3 , absorbance, bare ground and humification and a significant negative correlation with Mg and pH. Thus both axes are significantly correlated with Mg, NO_3 , pH and humification.

When the type of vegetation sampled is considered together with its relative homogeneity, the water chemistry values are high and the ranges are broad (Sjörs 1963, Heinselman 1970, Glaser et al. 1981; Vitt and Slack 1984; Vitt and Bayley 1984; Karlin and Bliss 1984). Based only on the water chemistry data (mainly pH, Ca and Mg concentrations) the peatlands which the stands represent would be classified as minerotrophic (Sjörs 1963; Karlin and Bliss 1984).

Conductivity is a measure of total conductance due to all ions minus the hydrogen ion (Sjörs 1950). It is strongly

TABLE 1

Environmental parameters from vegetation sampling stands.
DCA stand ordination scores are also given. Sampling sites
are separated by broken lines.

Stand no	DCA 1	DCA 2	Ca	Mg	K	NO ₃	PO ₄	Absorbance	pH	Conductivity	Bare ground	Humification ¹
1 *	185	266	35.7	2.2	32.0	0.42	0.32	1.62	4.3	277	85.5	3.0
2	138	219	12.3	2.3	5.7	0.21	0.15	1.03	5.0	72	15.9	3.5
3 *	124	198	17.6	2.7	9.4	0.47	0.22	1.36	4.2	166	61.7	5.0
4	14	123	15.0	2.0	11.3	0.30	0.19	0.46	4.5	122	3.6	4.0
5 *	268	112	28.5	3.1	2.1	0.24	0.74	0.98	6.4	101	23.1	6.0
6	130	181	14.6	1.7	2.7	0.21	0.61	0.85	6.1	72	2.8	5.0
7 *	144	186	17.7	2.4	2.8	0.27	0.17	1.06	4.5	65	6.9	2.5
8	93	29	3.2	0.4	0.6	0.26	0.15	0.42	5.0	32	2.2	2.0
9 *	318	119	38.1	8.4	2.2	0.21	0.18	0.81	6.7	220	12.7	7.0
10 *	60	156	19.3	1.3	5.3	0.45	0.42	1.33	4.3	75	16.0	3.0
11	103	63	3.5	0.6	1.9	0.32	0.15	0.45	4.9	31	0.0	3.0
12 *	0	118	14.9	1.6	5.1	0.53	0.76	1.44	4.1	73	15.0	3.5
13	57	85	21.7	3.9	3.5	0.28	0.31	0.61	5.9	73	0.8	2.5
14 *	124	43	16.4	1.7	1.1	0.26	0.20	0.65	5.9	42	3.0	2.0
15	118	51	14.9	5.1	2.1	0.19	0.22	0.51	7.0	108	0.0	2.5
16 *	125	55	14.4	8.0	3.4	0.39	0.22	0.88	7.2	151	0.5	2.5
17	116	20	19.5	7.1	3.5	0.35	0.61	0.70	7.1	132	0.0	1.0
18 *	110	124	25.6	6.3	0.8	0.34	0.31	0.55	6.8	157	6.9	3.5
19	64	142	10.6	3.6	2.6	0.31	0.18	1.44	5.8	50	0.8	4.5
20 *	67	68	31.8	2.2	0.4	0.29	0.15	1.96	4.5	50	0.0	4.5
21	15	79	12.4	1.7	0.8	0.28	0.14	1.17	4.7	52	0.0	3.5
22 *	85	76	39.8	19.7	0.9	0.29	0.15	0.69	7.4	317	0.0	5.0
23	128	2	34.9	9.9	1.6	0.26	0.16	0.53	7.1	209	0.2	1.0
24 *	128	66	29.0	12.6	1.0	0.27	0.15	1.08	7.2	169	0.0	4.0
25	123	0	45.2	20.4	2.4	0.22	0.21	0.62	7.8	309	0.0	2.0
26 *	97	150	55.8	21.7	1.7	0.42	0.36	0.60	7.6	364	0.8	3.5
27	130	44	41.6	16.9	0.6	0.27	0.23	0.54	7.3	267	0.0	1.0
28 *	162	38	26.4	12.0	1.0	0.26	0.15	0.45	8.3	183	0.1	1.0
29	123	10	26.5	13.5	0.7	0.21	0.16	0.45	8.1	179	0.6	2.0
30 *	402	35	47.8	31.4	0.4	0.22	0.16	0.57	7.7	427	0.0	6.0
31	66	79	42.5	23.5	1.8	0.28	0.20	0.44	7.6	342	0.8	4.0
32 *	120	82	31.4	19.4	0.7	0.26	0.17	0.17	7.7	256	13.3	3.0
33	135	11	38.4	28.1	1.6	0.30	0.18	0.27	7.6	353	1.3	2.5
34	82	107	7.6	2.1	4.2	0.42	0.28	2.50	4.1	54	3.7	2.5
\bar{X}			25.1	8.8	3.5	0.3	0.3	0.51	6.2	161.8	8.2	3.3
Lowest			3.2	0.4	1.6	0.2	0.1	0.17	4.1	31.0	0	1.0
Highest			55.8	31.4	32.0	0.5	0.8	2.50	8.3	427.0	85.5	7.0

¹ Units: Ca, Mg, K, NO₃, PO₄, = mg L⁻¹; Absorbance at 320 nm; Conductivity = umho cm⁻¹
Bare ground = FV; Humification = von Post scale. (K_{corr})

* Indicates right-of-way stands. N = 2 for environmental parameters, except
Bare ground, where N = 10.

TABLE 2

Correlations between environmental variables, and with DCA axis 1 and axis 2. N = 34.
Significant correlations are underlined, $\alpha = 0.05$.

	DCA Axis 1	DCA Axis 2	Ca	Mg	K	NO ₃	PO ₄	Absorbance	pH	Conduct- ivity	Humific.
Ca	<u>0.41</u>	-0.17									
Mg	<u>0.36</u>	<u>-0.44</u>	<u>0.81</u>								
K	0.00	<u>0.62</u>	-0.03	-0.29							
NO ₃	<u>-0.39</u>	<u>0.38</u>	-0.14	-0.24	<u>0.41</u>						
PO ₄	-0.03	0.24	-0.09	-0.26	0.14	<u>0.38</u>					
Absorbance	-0.16	<u>0.48</u>	-0.28	<u>-0.49</u>	<u>0.35</u>	<u>0.50</u>	0.20				
pH	<u>0.36</u>	<u>-0.59</u>	<u>0.62</u>	<u>0.78</u>	<u>-0.45</u>	<u>-0.48</u>	-0.17	<u>-0.67</u>			
Conductivity	<u>0.42</u>	-0.23	<u>0.89</u>	<u>0.93</u>	-0.02	-0.10	-0.19	<u>-0.42</u>	<u>0.68</u>		
Bare ground	0.17	<u>0.67</u>	0.01	-0.27	<u>0.84</u>	<u>0.44</u>	0.19	0.36	<u>-0.43</u>	0.01	
Humification	<u>0.39</u>	<u>0.41</u>	0.16	0.03	0.03	-0.05	0.13	0.22	-0.15	0.12	0.21

correlated with Ca (0.89) and Mg (0.93) concentrations, but correlations with K (-0.02), NO_3 (-0.10) and PO_4 (-0.19) are low and negative, indicating that the conductivity values are primarily a measure of Ca and Mg concentrations. pH shows a good correlation with conductivity (0.68) and is also significantly correlated with Ca (0.62) and Mg (0.78).

Water color, or absorbance is negatively and significantly correlated with pH (-0.67) and conductivity (0.42). Absorbance is most strongly correlated with NO_3 (0.50) of the individual nutrients.

The most distinctive trend in the water chemistry data is the difference between the northern permafrost stands, 1 - 13, and the southern frost free shallow peat stands, 22 - 33, on calcareous parent material. This becomes apparent when the scores for environmental factors are superimposed on the stand ordination diagram (Figure 12 a-g).

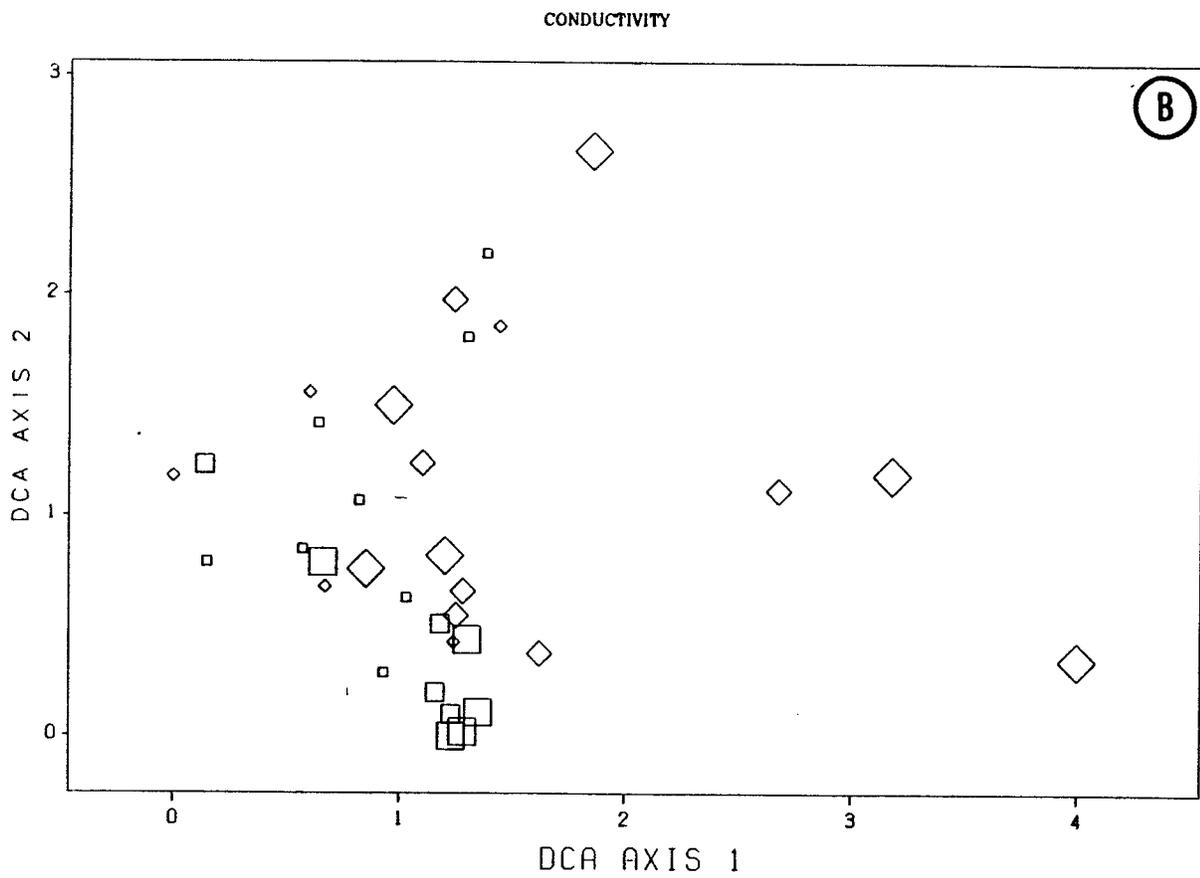
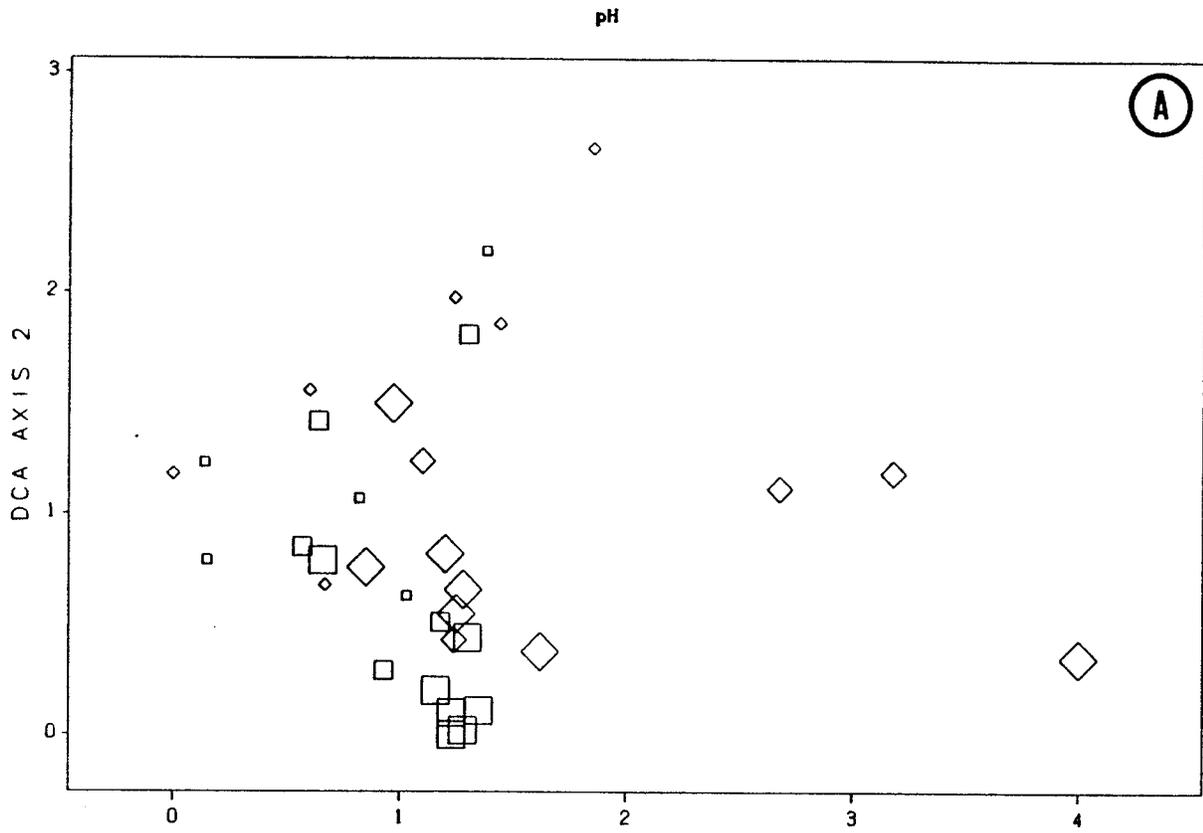
Figure 12. (Page 49-52)

Environmental parameters state classes plotted over stand positions on the two-dimensional ordination. Squares indicate forest and diamonds right-of-way stands. Stand positions are the same as in Figure 4.

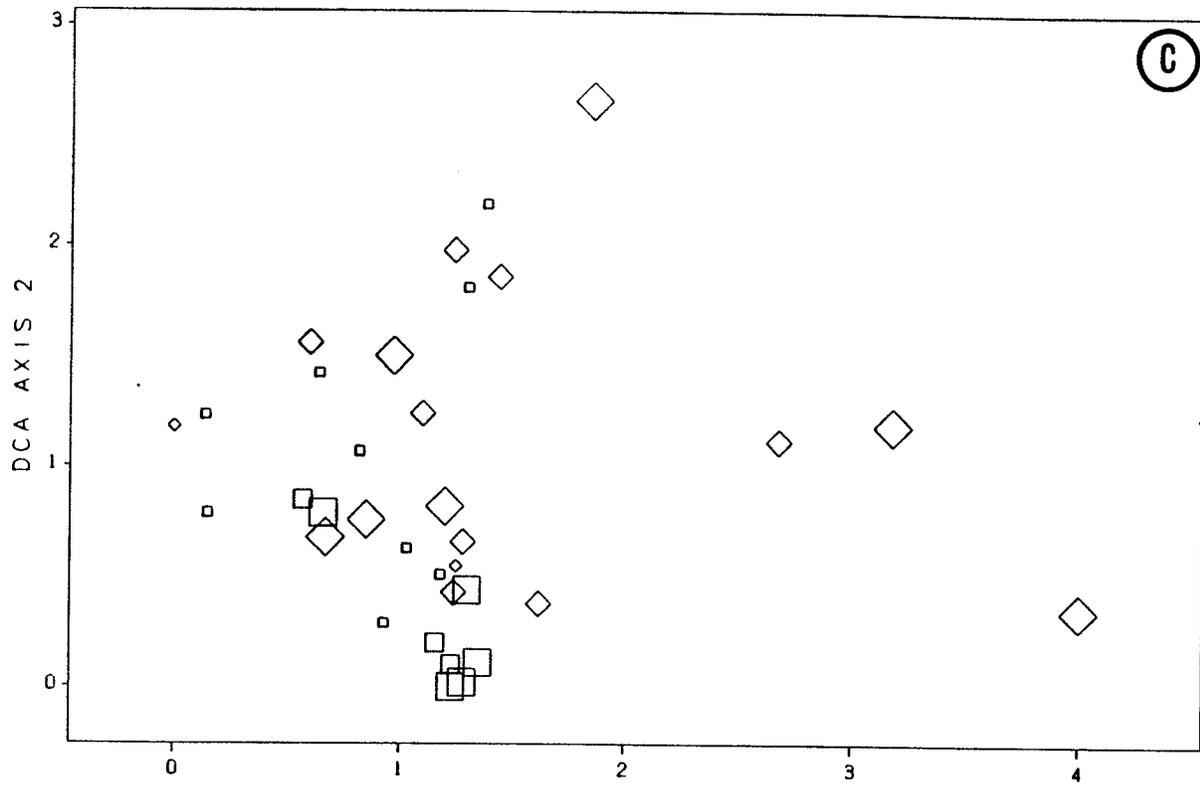
Each parameter has been divided into three classes which are indicated by the square and diamond sizes according to the following scale:

			
			
A) pH	< 5	5 - 7	> 7
B) Conductivity	< 100	100 - 200	> 200
C) Ca	< 15	15 - 30	> 30
D) Mg	< 5	5 - 10	> 10
E) K	< 2	2 - 5	> 5
F) NO ₃	< 0.25	0.25 - 0.35	> 0.35
G) Absorbance	< 0.5	0.5 - 1.0	> 1.0
H) Humification	< 3	3 - 6	> 6

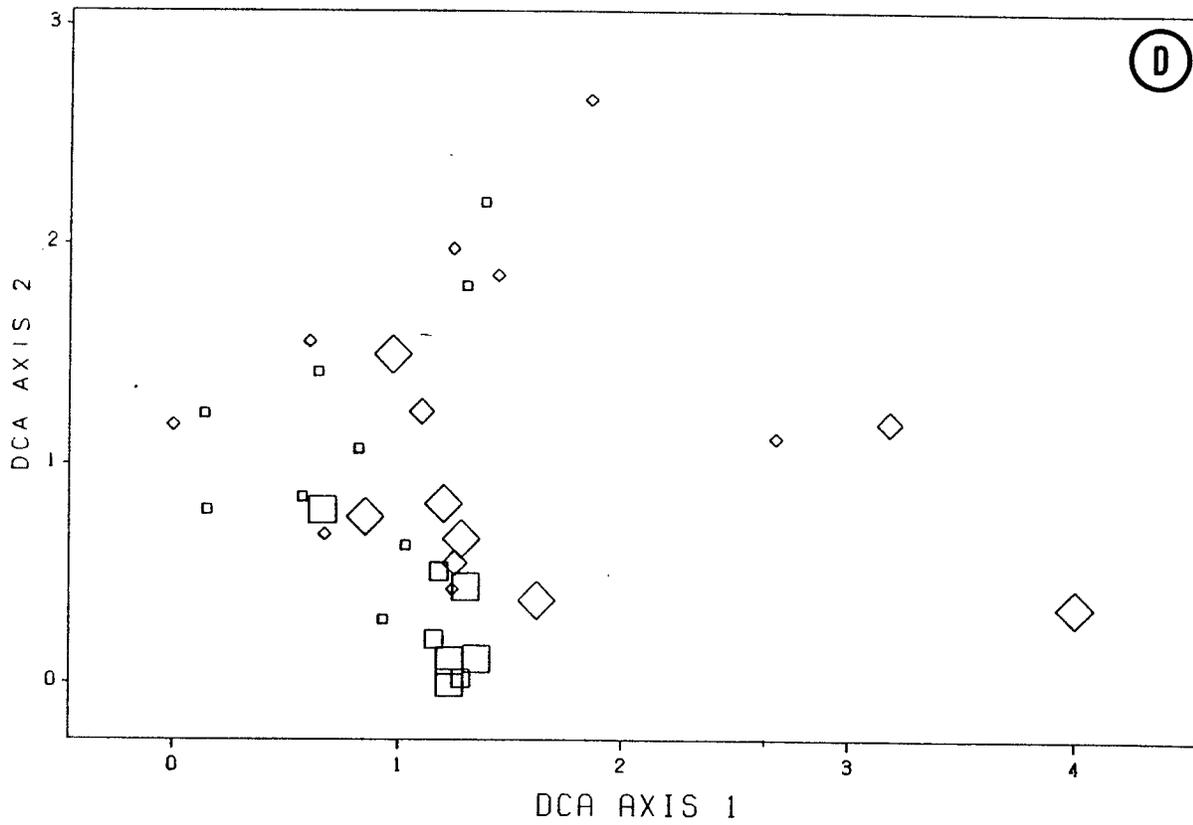
(Units are: Conductivity (Kcorr) = $\mu\text{mho cm}^{-1}$,
 Ca, Mg, K, NO₃ = mg L^{-1} ,
 Absorbance = at 320 nm,
 Humification = von Post.)



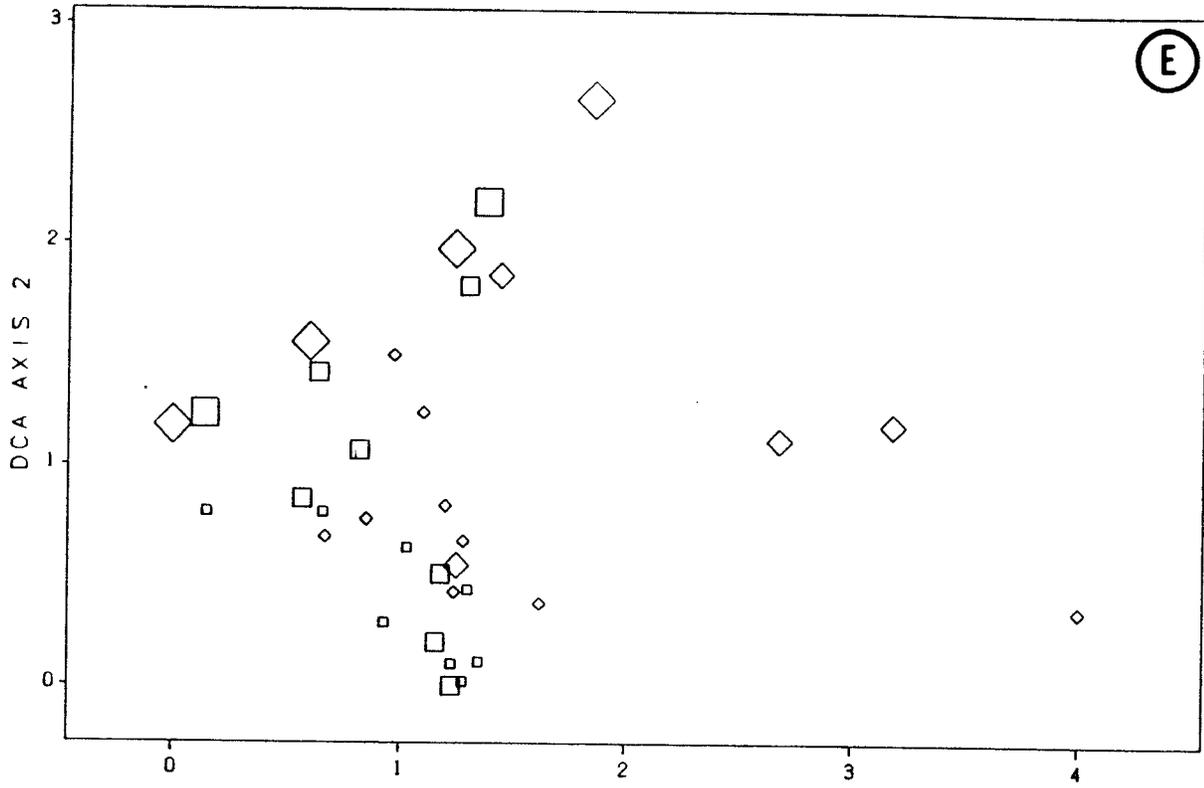
CALCIUM



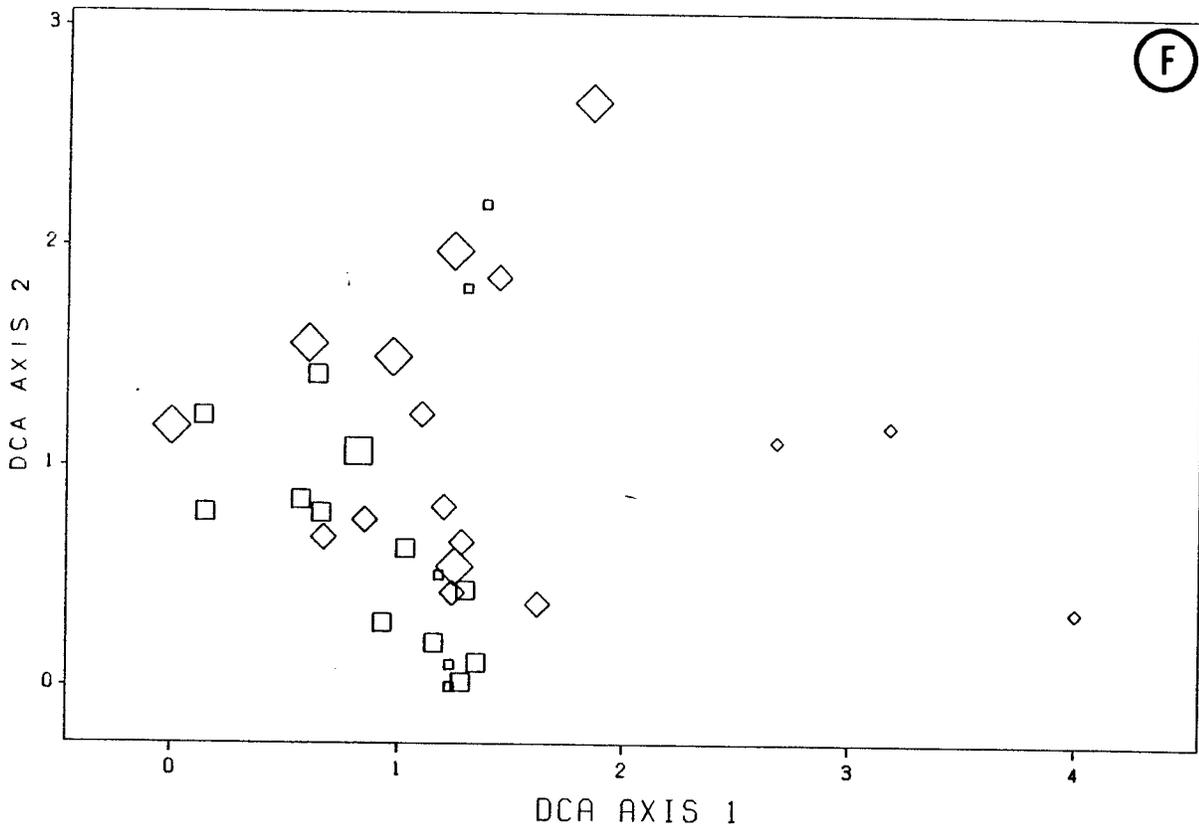
MAGNESIUM



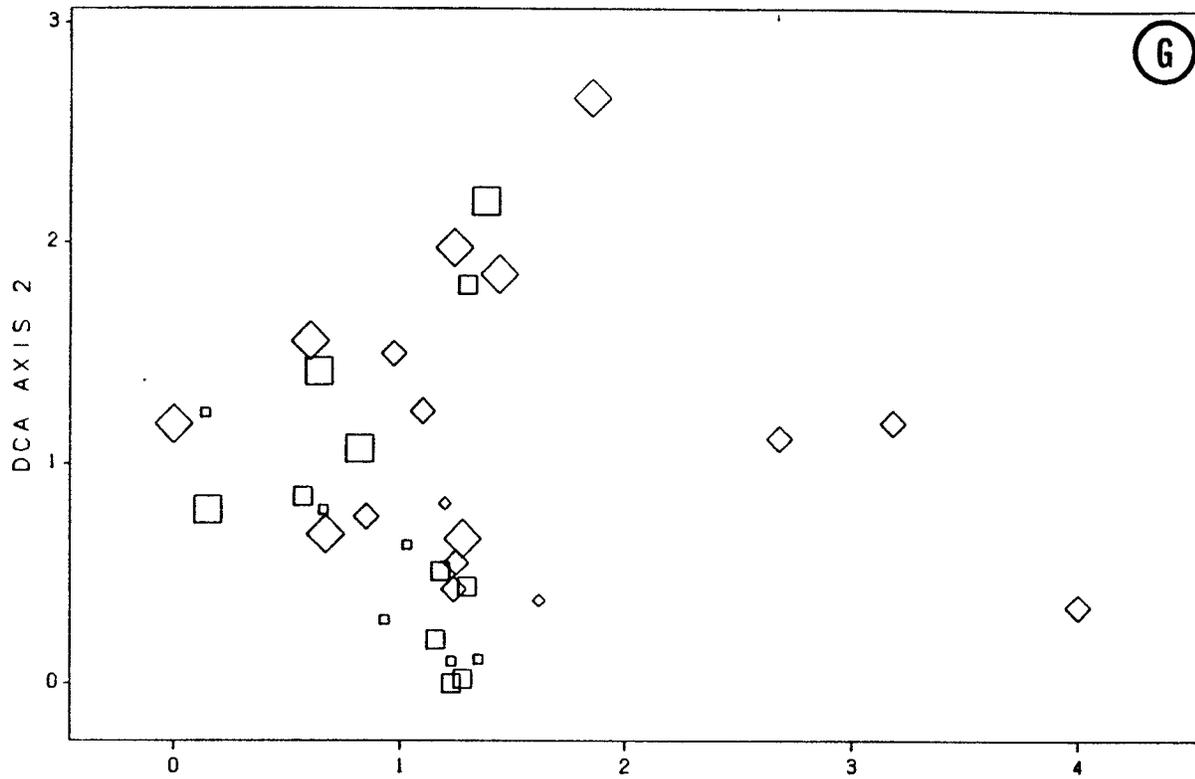
POTASSIUM



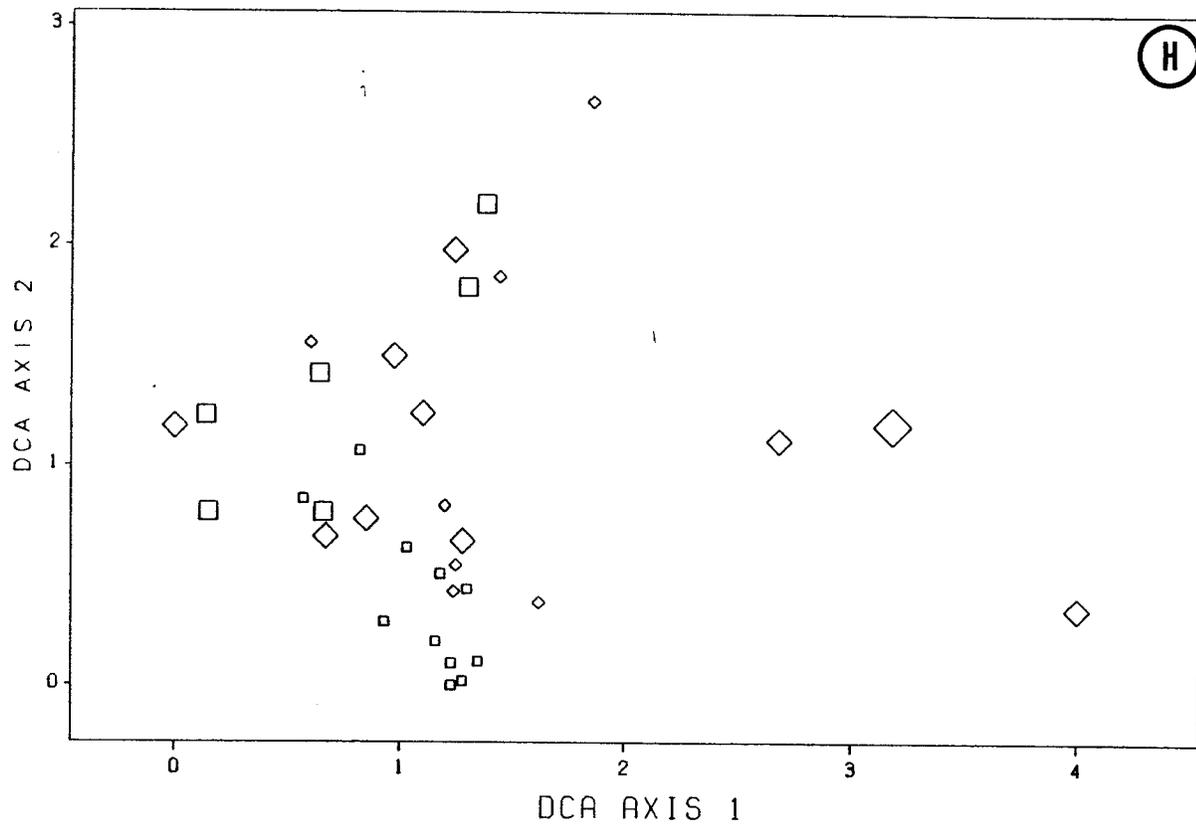
NITROGEN



ABSORBANCE



HUMIFICATION



Stands 1 - 13 are from the discontinuous permafrost zone and have all permafrost in them with the exception of the highly disturbed stands 5 and 9. The eleven stands with permafrost are characterized by (Table 1) relatively low pH (< 6.1 ($4.1 - 6.1$)), Ca-concentration (< 35.7 ($3.2 - 35.7$) mg L^{-1}), Mg-concentration (< 3.9 ($0.4 - 3.9$) mg L^{-1}) and conductivity (< 227 ($31 - 227$) umho cm^{-1}) and relatively high K-concentration (> 0.6 ($0.6 - 32.0$) mg L^{-1}) and PO_4 -concentration (> 0.21 ($0.21 - 0.53$) mg L^{-1}). The color of the water is relatively dark (Absorbance > 0.42 ($0.42 - 1.62$)). Glaser et al. (1981) report absorbance of $0.9 - 1.4$ in stagnant bog waters in the Red Lake Peatlands. On the basis of water-chemistry the stands would be classified as weakly to moderately minerotrophic.

In stands 5 and 9 which have free standing water in contact with mineral soil there is a rise in pH ($6.35, 6.70$), Ca ($28.5, 38.1$ mg L^{-1}) and Mg ($3.1, 8.4$ mg L^{-1}), conductivity is also relatively high ($101, 220$ umho cm^{-1}). K ($2.1, 2.2$ mg L^{-1}) and NO_3 ($0.24, 0.21$ mg L^{-1}) concentrations are low (Table 1).

Stands 22 - 33 are all south of the permafrost zone and characterized by relatively shallow peat, which in most stands is less than a meter in thickness (range ca $40 - 130$ cm), sitting on calcareous parent material. In the late

summer of 1983 the water tables in the stands were usually 35 - 50 cm (range ca 5 - 50 cm) beneath the hollows in the stands or ca 60 - 90 cm below the hummock tops. The water samples from these stands have a high pH (> 7.1 (7.1 - 8.3)), Ca (> 26.4 (26.4 - 55.8) mg L^{-1}), Mg (> 9.9 (9.9 - 31.4) mg L^{-1}) concentrations and conductivity (> 169 $\mu\text{mho cm}^{-1}$ (169 - 427)). K (< 2.4 (0.4 - 2.4) mg L^{-1}), NO_3 (< 0.42 (0.21 - 0.42) mg L^{-1}) and PO_4 (< 0.28 (0.15 - 0.28 mg L^{-1})) are relatively low (Table 1). The water is light in color and absorbance is, with one exception, less than 0.69 (0.17 - 0.69), and in the range reported for rapidly flowing fen waters (Glaser et al. 1981). The water color in stand 24 was relatively dark (absorbance 1.08) without other parameters being outstanding. The reason for this is not known. With this composition of water, stands 22 - 33 would be classified as strongly minerotrophic (Karlin and Bliss 1984).

Stands 14 - 21 occupy an intermediate position both in terms of latitude and water composition. These are stands without permafrost, but still within the discontinuous permafrost zone. The first four (14 - 17) are from the same area (Jenpeg 1 and 2) and only about 2 km apart. The area has a highly undulating topography and bog vegetation is confined to small patches a little upslope from the lowest parts of the depressions, where mineral-rich water from upland ridges probably seeps through. Peat thickness in the

stands is 45 - 70 cm and the water table is at or slightly below the surface (ca 5 cm) in the depresssions during late July. The composition of the water from these stands is intermediate but their vegetation bears a closer resemblance to southern than northern stands (Figure 4). The pH ranges from 5.9 - 7.2, Ca 14.4 - 19.5 mg L⁻¹, Mg 1.7 - 8.0 mg L⁻¹, conductivity 42 - 151 umho cm⁻¹. The color of the water from these stands is intermediate, absorbance ranges from 0.51 - 0.88 (Table 1).

Stands 18 - 21 although approximately 10 km apart are within the same extensive peat covered basin. They are not in immediate vicinity of mineral ridges and do not receive direct run-off from upland soils. Peat thickness in the stands is 30 - 60 cm in the two northern stands (18 and 19) and 50 - 80 cm in the southern stands (20 and 21) and the water table was 20 - 60 cm below the surface of the hollows. The water chemistry and vegetation of these four stands are more similar to the permafrost stands than those from the southern part of the study area. The pH in the stands ranges from 4.5 to 6.8, Ca 10.6 - 31.8 mg L⁻¹, Mg 1.7 - 6.3 mg L⁻¹, conductivity 50 - 157 umho cm⁻¹. The watercolor is fairly dark with three of the stands having absorbance over 1.17 (ranges are 0.55 - 1.96) (Table 1).

The water chemistry of the southernmost stand (34) does not have much in common with the strongly minerotrophic southern stands (22-33). The stand represents the only true

raised bog in the southern part of the sampling area, with a peat thickness of 3.5 meters. The water chemistry of the stand is similar to the least minerotrophic permafrost stands, pH 4.1, Ca 7.6 mg L⁻¹, Mg 2.1 mg L⁻¹ and conductivity 54 umho cm⁻¹ (Table 1). The water from the stand was the darkest of all stands sampled (absorbance 2.5) indicating highly stagnant conditions (Glaser et al. 1981).

1.5.2.2 Humification

The humification ratings of the top peat from the stands show variations which correlated reasonably with the vegetational variation encountered (Figure 12 h). With the exception of stands 5, 9 and 30 the top peat in the stands was primarily composed of Sphagnum-remains in a variable state of breakdown. The stands can be put into three main categories on the basis of degree of humification (Table 1). Firstly, there are stands with fresh and light colored, undecomposed or nearly undecomposed (von Post < 3.0) Sphagnum peat, which is mainly found in stands with high abundance of Sphagnum. Secondly, there are stands with darker colored peat of little to fair decomposition, but still with recognizable remains (von Post 3-5). This condition is more characteristic of the drier lichen-rich stands and northern moderately disturbed stands on permafrost with low Sphagnum-cover. Thirdly there are the highly disturbed and wet stands 5, 9 and 30 in which the top peat has been removed or

broken down. The peat in these stands is dark brown to black in color, fairly to strongly decomposed without recognizable structures, except for root fibres of living plants (von Post 6 - 7), which is more characteristic of fen rather than bog peats (Jeglum 1972).

1.5.2.3 Bare ground

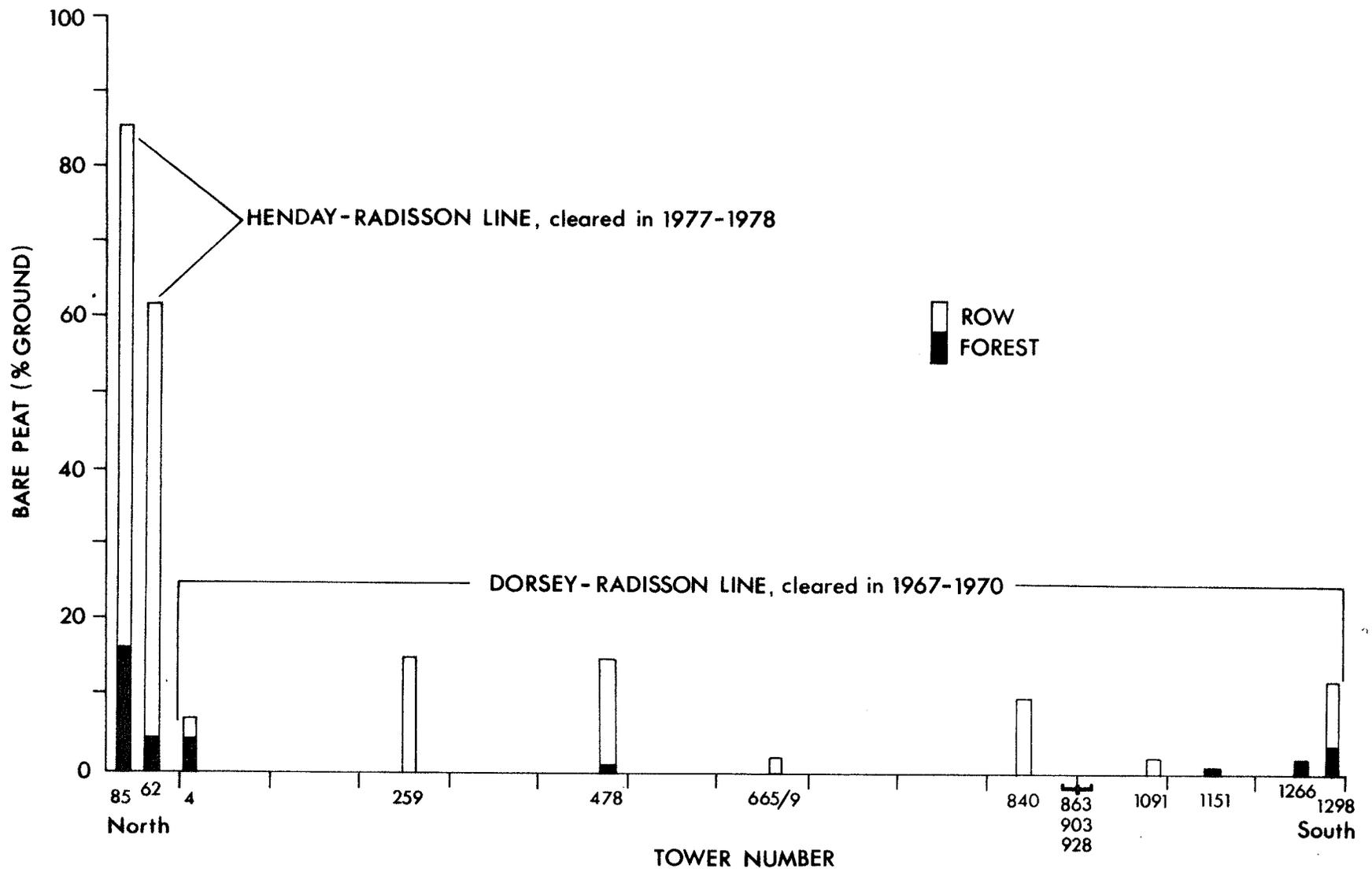
Bare ground was looked upon as an environmental variable since it gives an indication of the degree of disturbance that has occurred at a particular site and/or the state of recovery. Bare ground was largely confined to the northernmost right-of-way sites of the study area, where the initial clearance had apparently more severely affected the top peat and its vegetation cover than at the southern sites.

Along the Radisson-Dorsey line the cover of bare ground did not exceed 15%, and in most stands it was insignificant (Figure 13). In the southernmost right-of-way stand (32), bare ground was attributed to herbicidal application at the site in 1978, which had killed off the Sphagnum-hummocks and exposed the peat. In other cases reduced plant cover in the right-of-way stands was considered to be due to initial clearance for the line and construction activities.

The two right-of-way stands from the Henday-Radisson line had both over 60% bare ground, but at the time of clearance the vegetation and a thin layer of top peat was scraped off.

Figure 13.

Exposed peat (% cover, bare ground) at sites along the Radisson-Dorsey and Henday-Radisson transmission lines.



The sites were cleared in the winter of 1977-1978 and their vegetation has therefore had almost 10 years less to recover than in the Radisson-Dorsey right-of-way, cleared in 1967-1970. This may partly explain the difference in plant cover of these northern stands from the two right-of-ways.

1.5.3 Canonical correlation analysis (CCA)

The scatter diagrams combining the vegetational variation and changes in environmental variables (Figure 12 a-h) and the correlations of the environmental variables with the first two Decorana axes (Table 2) indicate a considerable interrelation between the two. However, these have been single comparisons incorporating only part of the available data. In the CCA, the overall linear relationships of the two data sets are investigated using simultaneously all variables that should help to explain the vegetation-environmental relations (Gittins 1985).

All environmental variables were included in the CCA despite strong correlations with other environmental variables. Matrix singularity was not encountered in the computation, indicating that no variable was a linear compound of another variable (Gauch and Wentworth 1976).

The canonical correlation coefficients and the significance of the two pairs of canonical variates formed in the analysis is shown in Table 3. The correlation coefficients

TABLE 3

Canonical correlation coefficients

Canonical correlation coefficients and their indices of goodness of fit with significance probabilities.

- (a) The canonical correlation coefficients (r_k) and percentage of total covariation ($r_k^2/\sum r_k^2$) accounted for by the the canonical roots r_k^2 .

k	r_k	r_k^2	$r_k^2/\sum r_k^2$
1	.8264	.6830	56%
2	.7379	.5445	44%

- (b) Significance of canonical correlation coefficients (by Barlett's transformation of Wilk's likelihood ratio Λ).

k	Roots	χ^2	df	P
1	1,2	51.28	20	0.0001
2	2	20.84	9	0.0134

(r) are relatively similar in size (0.83 and 0.74) and both are significant ($p < 0.05$). The null hypothesis of independence of the two data sets is rejected, a linear relationship exists between the vegetation and environmental variables. The two canonical variates account for 56% and 44% of the total covariation of the two sets. Table 4 and Table 5 present the correlation coefficients between the canonical variates and each variable of the two data sets. The first variate in the vegetation set (V_1) is very strongly correlated with Decorana axis 2 (0.91) with a positive correlation to axis 1 but much lower (0.38). The variate correlates strongly (0.83) with the first variate in the environmental set (U_1), which has the highest positive correlation with bare ground (0.84), K (0.70) and humification (0.65), and has the highest negative correlation with pH (-0.48) and Mg (-0.31).

The second variate in the vegetation set (V_2) correlates very strongly with Decorana axis 1 (0.92) and has a negative correlation with axis 2 (-0.41). It has a high correlation (0.74) with the second variate in the environmental set (U_2), which has the highest positive correlation with pH (0.76), Mg (0.67) and conductivity (0.64) and the most negative correlation with NO_3 (-0.68) and absorbance (-0.45).

The environmental variates, U_1 and U_2 , account for 50% (0.223 + 0.267) of the variance in the environmental variables. The redundancy index (Gittins 1985) for the vegeta-

TABLE 4

Canonical variates - vegetation set

Correlation coefficients between the canonical variates and the vegetation variables.

VARIABLE	V ₁	V ₂
Decorana axis 1	0.384	0.923
Decorana axis 2	0.912	-0.410
Variance extracted	0.490	0.510
Redundancy	0.335	0.278

TABLE 5

Canonical variates - environmental set

Correlation coefficients between the canonical variates and environmental variables.

VARIABLE	U ₁	U ₂
Ca	0.013	0.595
Mg	-0.312	0.667
K	0.696	-0.321
NO ₃	0.229	-0.681
PO ₄	0.254	-0.166
pH	-0.484	0.757
Conductivity	-0.048	0.636
Absorbance	0.454	-0.450
Bare ground	0.839	-0.136
Humification	0.654	0.269
Variance extracted	0.228	0.267
Redundancy	0.155	0.145

tion variates, V_1 and V_2 , shows that 61% ($0.335 + 0.278$) of the variance in the vegetation set is accounted for by the environmental variables used.

Table 6 and Table 7 show the multiple correlations of each variable in one data set with all variables in the other set. In the vegetation set (Table 6) only the second Decorana axis is significantly correlated ($p = 0.023$) with the environmental variables, which explain 66% of the variation in the axis. Correlation with the first Decorana axis is slightly lower and not significant ($p = 0.07$). The environmental variables explain 56% of the variation in that axis.

All the environmental variables (Table 7), with the exception of PO_4 , are significantly correlated ($P < 0.05$) with the vegetation variables. The highest multiple correlations are with bare ground (0.49), pH (0.47), K (0.39) and Mg (0.31).

The CCA has thus effectively described a strong relationship between the vegetation and the environmental variables, which have a stronger relation with the second Decorana axis than the first. The environmental variables whose variation has most strongly influenced or are related to the vegetation pattern encountered are: bare ground (disturbance), pH, K, humification and Mg.

TABLE 6

Correlations - vegetation variables vs. environment.

Squared multiple correlations of each variable in the vegetation set with all variables in the environmental set.

VARIABLE	R ²	F	Df	P
Decorana axis 1	0.5650	2.99	10 23	0.070
Decorana axis 2	0.6597	4.46	- -	0.023

TABLE 7

Correlations - environmental variables vs. vegetation.

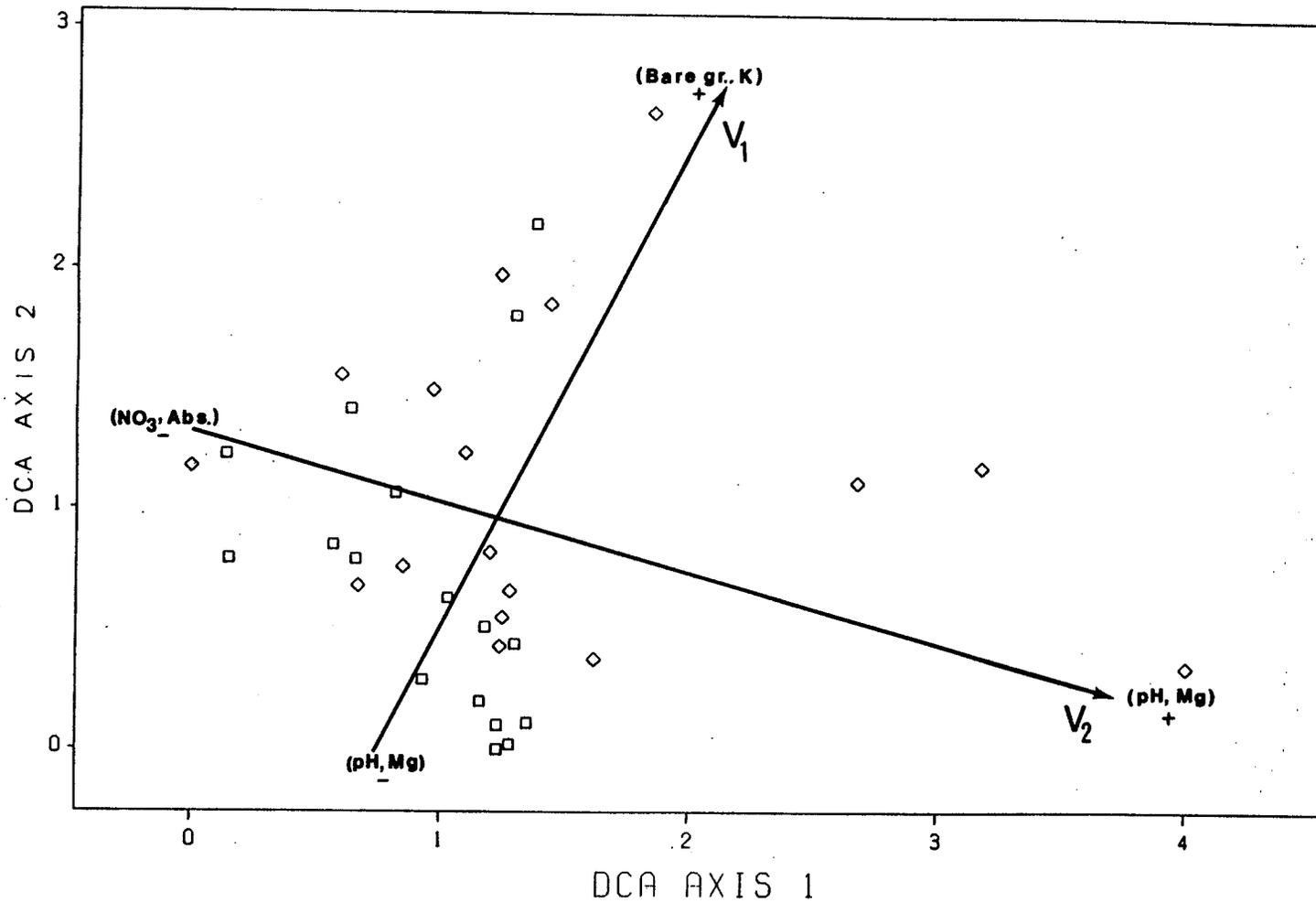
Squared multiple correlations of each variable in the environmental set with all variables in the vegetation set.

VARIABLE	R ²	F	Df	P
Ca	0.1930	3.71	2 31	0.036
Mg	0.3092	6.94	- -	0.003
K	0.3894	9.89	- -	0.000
NO ₃	0.2885	6.29	- -	0.005
PO ₄	0.0590	0.97	- -	0.390
pH	0.4721	13.86	- -	0.000
Conductivity	0.2221	4.43	- -	0.020
Absorbance	0.2514	5.20	- -	0.011
Bare ground	0.4904	14.92	- -	0.000
Humification	0.3312	7.68	- -	0.002

Plotting the canonical variable scores of each stand for the vegetation set (V_1 against V_2) and comparing it to the stand positions on the two dimensional Decorana diagram (Figure 4), shows that the two patterns are nearly identical after the axes have been reversed and tilted slightly. This can be seen in Figure 14, where the two variates have been drawn onto the Decorana stand diagram.

Figure 14.

Directions of vegetation variates (V_1 and V_2) extracted in the CCA, in the two-dimensional space created by DCA axis 1 and 2 in the stand ordination. The environmental variables showing the strongest positive and negative correlation with the vegetation pattern and influencing most the directions of V_1 and V_2 are shown in brackets.



1.5.4 Forest/Right-of-way contrasts

1.5.4.1 Vegetation

Fifty two plant species were found only in one or two stands and were not used in the analysis of the vegetation data. Of these 52 rare species, 38 were vascular plants, 6 mosses and 8 lichens. The division of these species between the right-of-way and forest stands shows no significant trends among the mosses or lichens. Of the vascular plants, 13 species are found both in the forest and right-of-way, 5 are only found in the forest and 20 only in the right-of-way. Five of these 20 right-of-way species are composites, four are grasses and three are sedges. The higher number of rare species in the right-of-way reflects a weak response of weedy or opportunistic species to the disturbances.

Of the fifty understorey species (occurring in 3 or more stands) used in the vegetation analysis, 41 were common to both the forest and right-of-way stands. The nine species confined to either forest or right-of-way are shown in Table 8. Most of these occurred in only 3 or 4 stands, had very low abundance, and were not of great significance in the forest or right-of-way communities. The complete absence of Drosera rotundifolia from the right-of-way stands, while occurring in eleven forest stands can hardly be caused by chance alone (Figure 6 i). The species grows primarily on living Sphagnum-hummocks. Although the abundance of Sphagnum has been significantly reduced in the right-of-way (PV

TABLE 8

Common species confined to forest or ROW stands.

Species found in forest only:	No. of stands	PV (\bar{X}) ¹
<u>Drosera rotundifolia</u>	11	0.4
<u>Pinguicula villosa</u>	3	0.01
<u>Sphagnum warnstorffii</u>	3	0.2
Species found in ROW only:	No. of stands	PV (\bar{X}) ²
<u>Carex capillaris</u>	4	0.01
<u>Agrostis borealis</u>	3	0.03
<u>Epilobium angustifolium</u>	4	0.1
<u>Drepanocladus aduncus</u>	3	1.2
<u>Cladonia cristatella</u>	4	0.1
<u>Peltigera polydactyla</u>	3	0.1

¹ \bar{X} = PV/16 (Average is calculated for the 16 stands sampled).
² \bar{X} = PV/17 (Average is calculated for the 17 stands sampled).

17) compared to the forest (PV 30) it is still in relatively high abundance in the right-of-way. This reduction can hardly by itself explain the absence of Drosera rotundifolia from the right-of-way. A partial explanation might be sought in the effects of canopy removal on microclimate; e.g. increased radiation and surface temperature, or moisture changes (Barry and Chorley 1976).

Pinguicula villosa is an arctic plant (Scoggan 1957) that grows in northern Manitoba mainly on moist forest soils and on organic soils. The study found Pinguicula villosa only in the three forest stands in the Gillam area, but the right-of-way stands in that area were highly disturbed and few original bog species remained within them.

Sphagnum warnstorffii was found in three forest stands which are strongly minerotrophic (pH > 7.0). These stands had a very high abundance of Sphagnum fuscum (PV > 50.0) that had been reduced by nearly 50% in the corresponding right-of-way stands. Sphagnum warnstorffii has been described as occurring rather consistently under minerotrophic conditions and as a codominant with S.fuscum in minerotrophically rich muskegs of western North America (Horton et al. 1979).

Only Drepanocladus aduncus, gains a relatively high abundance in the right-of-way compared to the other species listed in Table 8. This is due to its dominance (PV = 20)

in the ground layer of stand 9. The species is characteristic of fen vegetation (Jeglum 1972, Moss and Turner 1961; Slack et al. 1980) and has responded to the increased wetness and minerotrophy associated with the disturbance in the right-of-way. A similar, but much weaker response, is shown by Carex capillaris found in the wetter and more disturbed right-of-way stands.

Agrostis borealis and Epilobium angustifolium are both weedy species (Scoggan 1957; Looman and Best 1979) which appear in a few of the right-of-way stands but are insignificant. It appears that weedy species have not managed to gain a foothold in the right-of-way stands covered in this study. Their importance in the relatively bare northern stands is also extremely low.

Two lichens, Cladonia cristatella and Peltigera polydactyla are on the list of right-of-way species. Peltigera species and certain Cladonia species are known to achieve importance early in disturbed communities on organic soils (Black and Bliss 1978; Jasieniuk and Johnson 1982; Hale 1974). In most of the right-of-way stands disturbance has reduced the abundance of living Sphagnum and increased the bare organic substrate which is more favourable for lichen growth.

As described above, 41 species occurring in 3 or more stands were found both in the forest and right-of-way commu-

nity. Twenty three of these species are vascular plants, 11 are mosses and 7 are lichens. The majority of these species are insignificant and occur at relatively few sites in low abundance.

In the forest stands, Ledum groenlandicum (Table 9) is the single most important and characteristic species in the shrub layer occurring in every stand. Vaccinium vitis-idaea is subdominant. Other important species are Rubus chamaemorus, Chamaedaphne calyculata, Oxycoccus microcarpus, and Carex aquatilis. In the right-of-way community Ledum groenlandicum is still dominant in the field layer but considerably reduced in its prominence value. It is the only species present in every right-of-way stand (Figure 6 e).

A noteworthy change in the field layer of the right-of-way is the increase in Carex aquatilis, the second most important species among the vascular plants (Figure 6 c). The difference in PV values for the species is not significant between forest and right-of-way.

Like Ledum groenlandicum, the four other most important species in the field layer of the forest and have been reduced in prominence in the right-of-way. For Vaccinium vitis-idaea and Oxycoccus microcarpus the reduction is significant. Equisetum pratense and Salix spp. increase slightly in PV in the right-of-way, but not significantly.

TABLE 9

Forest/ROW vegetation - ANOVA

Important species¹ from vegetation sampling in forest and right-of-way in 1983.

FOREST/ROW ANOVA

SPECIES	FOREST (N=16)			ROW (N=17)			F	Pr > F
	PV \bar{x}	SE	N	PV \bar{x}	SE	N		
<u>Equisetum prat.</u>	1.14	(0.49)	8	1.98	(0.87)	10	1.25	0.28
<u>Carex aquatilis</u>	1.47	(0.53)	10	6.11	(2.37)	11	3.77	0.07
<u>Rubus chamaem.</u>	3.31	(1.82)	10	1.89	(0.75)	9	0.72	0.41
<u>Ledum groenland.</u>	18.62	(2.39)	16	11.89	(1.88)	17	3.56	0.08
<u>Chamaedaphne cal.</u>	2.83	(1.24)	10	1.30	(0.96)	4	0.98	0.34
<u>Oxycoccus micro.</u>	1.83	(0.30)	16	0.55	(0.19)	13	12.02	0.02*
<u>Vaccinium v.-id.</u>	6.79	(1.41)	15	2.49	(0.49)	15	7.03	0.02*
<u>Salix spp.</u>	0.15	(0.08)	8	2.38	(1.46)	6	3.21	0.09
<u>Sphagnum fuscum</u>	29.89	(5.39)	15	16.67	(6.24)	11	7.24	0.01*
<u>Pleurozium sch.</u>	3.27	(1.19)	12	0.21	(0.12)	4	7.09	0.02*
<u>Polytrichum str.</u>	1.49	(0.50)	12	11.81	(2.98)	15	8.37	0.01*
<u>Cladina rangifer.</u>	4.11	(3.93)	6	1.93	(1.01)	10	0.31	0.59
<u>Cladina mitis</u>	9.43	(4.67)	12	5.45	(2.19)	12	0.89	0.36

1 These species had the 10 highest average PV scores in forest and right-of-way. See appendix A, for complete species names.

* Significant at $\alpha = 0.05$

Two species not listed above were found to have significantly different PV scores between forest and right-of-way:
Drosera rotundifolia and Sphagnum nemoreum.

Sphagnum fuscum is the dominant plant in the ground layer of the forest, with the lichens Cladina mitis and C.rangiferina next important. In the right-of-way, S.fuscum has been significantly reduced to almost half the PV of the forest. The species is absent from six of the stands in the right-of-way compared to only one of the stands sampled in the forest (Figure 6 a). Cladina mitis (Figure 6 b), and C.rangiferina (Figure 6 j), have also been reduced considerably in PV values in the right-of-way, but not significantly. A significant reduction occurs however in the abundance of Pleurozium schreberi which is present in only four right-of-way stands. In the forest it was the second most important moss species and present in twelve stands (Figure 6 k). Heinselman (1963) correlated the abundance of P.schreberi and other feathermosses, in black spruce muskegs in Minnesota with tree density, being most abundant under dense canopies. The removal of black spruce led frequently to the drying up and eventual death of P. schreberi along with other feathermosses.

The species in the ground layer that has benefitted the most from the disturbance in the right-of-way is Polytrichum strictum (Figure 6 d). It is the most important colonizer of disturbed ground in the right-of-way stands, partly making up for the reduction in Sphagnum fuscum.

Table 9 lists those species with the 10 highest PV scores in the forest and right-of-way community. Of these thirteen

species, five show significant differences between the forest and right-of-way communities. Analysis of variance tests for the other 37 species occurring in 3 stands show only two other species having significant PV scores between forest and right-of-way, namely Drosera rotundifolia ($F=5.35$, $Pr > F = 0.03$) and Sphagnum nemoreum ($F=4.71$, $Pr > F = 0.05$). S.nemoreum had PV of 0.46 and 0.10 in the forest and right-of-way respectively.

In the univariate ANOVA statistics each species is treated independently of all other species occurring with it in the community. This is clearly a weakness, since the species interact in the community. A sudden disturbance will upset the existing balance between species and each species response to the disturbance will be partly shaped by the interaction to the responses of other species. One is therefore left to question the interpretation of these univariate statistics. Is it possible to conclude, that since there are significant differences in PV scores of 9 species of the 50 that there are real differences in vegetation composition of the two communities? Of these 9 species only two show considerable changes in PV scores.

In a multivariate ANOVA (MANOVA) it is possible to compare the forest and right-of-way using species assemblages and thus better address the problem. Unfortunately, it is impossible to carry out a MANOVA test incorporating all 50 species (variables) since the number of observations has to

be equal or greater than the number of variables (Morrison 1976). It was therefore decided to use in the MANOVA test the most prominent species in the two communities since it can be argued that they characterize the communities. Also due to their abundance they were presumably more accurately quantified in the field than the rarer species and are present in a greater number of stands which should statistically give more meaningful results.

A MANOVA was first carried out using the 13 species in Table 9 which have the ten highest PV scores in the forest and right-of-way communities. For the hypothesis of no overall forest/right-of-way effects the results gave:

$$F(13,4) = 5.44; \text{ Pr} > F = 0.0573$$

Therefore for $\alpha = 0.05$, the composition of the two communities is not quite significantly different with respect to these thirteen species. When Kalmia polifolia, Ceratodon purpureus and Tomenthypnum nitens, are added into the analysis which then is based on the 16 species with the highest overall PV scores, the results are:

$$F(16,1) = 21.97; \text{ Pr} > F = 0.1662$$

The univariate ANOVA calculations for these three additional species for forest/right-of-way effects was insignificant ($F=1.57$, $\text{Pr} > F = 0.23$; $F=1.80$, $\text{Pr} > F = 0.20$; $F=0.37$, $\text{Pr} > F = 0.55$), and their addition into the MANOVA gave more in-

significant differences. This indicates that if further addition of species was possible, (e.g. the inclusion of the other 34 species from the data set) it would not show any significant differences in the composition of the two communities.

Working with only the ten most common species based on PV scores and leaving out Equisetum pratense, Oxycoccus microcarpus and Salix spp. from Table 9 resulted in:

$$F(10,7) = 7.55, \quad Pr > F = 0.0068$$

or significant differences between the forest and right-of-way with respect to these species in the MANOVA. The outcome therefore depends largely upon the number of species used in the MANOVA analysis.

1.5.4.2 Environmental parameters

When the environmental parameters and their relation to the ordination results were looked at earlier it was evident that the disturbance in the right-of-way had caused some changes. However, there was a wide range in the means from both the forest and right-of-way stands and an overlap in values between the two. Table 10 lists each environmental variable, their means in the forest and right-of-way and the results of the univariate ANOVA.

TABLE 10

Forest/ROW environmental parameters - ANOVA

Environmental variables from vegetation sampling stands sampled in 1983.

FOREST/ROW ANALYSIS OF VARIANCE

VARIABLE	FOREST		ROW		F	Pr > F
	\bar{x}	SE	\bar{x}	SE		
Ca	22.30	(3.52)	28.84	(2.89)	4.53	0.05
Mg	8.79	(2.24)	9.22	(2.18)	0.16	0.70
K	2.71	(0.66)	4.14	(1.83)	0.83	0.38
NO ₃	0.27	(0.01)	0.33	(0.02)	10.44	0.01*
PO ₄	0.24	(0.04)	0.28	(0.05)	1.26	0.28
pH	6.33	(0.31)	6.14	(0.37)	1.63	0.22
Conductivity	150.19	(28.14)	179.00	(27.30)	1.79	0.20
Absorbance	0.66	(0.08)	0.95	(0.11)	6.86	0.02*
Bare ground	1.81	(0.98)	14.44	(5.76)	5.80	0.03*
Humification	2.75	(0.31)	3.82	(0.38)	6.74	0.02*

* Significant at $\alpha = 0.05$

Units: Ca, Mg, K, NO₃, PO₄ = mg L⁻¹

Conductivity (Kcorr) = umho cm⁻¹

Absorbance = 320 nm

Bare ground = PV

Humification = von Post

In the water samples variables (Ca - absorbance) there is an increase of each in the right-of-way with the exception of pH, which decreases slightly. In general the waters of the right-of-way can be interpreted as slightly more minerotrophic, and darker in color than those from the undisturbed forest. The univariate ANOVA results showed that only two of the water variables were significantly higher in the right-of-way than in the forest, namely NO_3 concentration and absorbance, which are presumably interrelated. Ca concentration is almost significantly higher at $p < 0.05$, but other water variables have considerably lower F - values. A MANOVA test using the eight water variables gave the following results:

$$F(8,8) = 1.72, \quad \text{Pr} > F = 0.2295$$

The chemical composition of the peat waters from the forest and right-of-way is therefore not significantly different when these eight variables are all considered together.

The two other environmental variables, bare ground and humification were significantly different between the forest and right-of-way according to the univariate ANOVA. More bare ground and humified top peat was noted in the right-of-way. A MANOVA test using these two variables with the eight water variables gave:

$$F(10,7) = 2.47, \quad \text{Pr} > F = 0.1217$$

and therefore not a significant difference between the forest and right-of-way environment based on these ten variables.

1.6 DISCUSSION

1.6.1 Environment

Numerous studies of vegetation and environmental factors in peatlands show that moisture and fertility regimes are the key factors influencing floristics, vegetation types and successional trends (e.g. Sjörs 1950, 1963; Heinselman 1970; Jeglum 1971, 1972, 1973; Jeglum et al. 1974; Schwintzer 1981; Jasieniuk and Johnson 1982; Ruuhijärvi 1983; Karlin and Bliss 1984). Both of these regimes represent a series of interrelated factors and may be described as "complex gradients" (Whittaker 1967, 1975; Jeglum 1973).

The division of peatlands into ombrotrophic and minerotrophic types on the basis of source of mineral input has become well established in the literature (Sjörs 1963; Heinselman 1970; Moore and Bellamy 1974; Schwintzer 1981; Karlin and Bliss 1984). The chemical characteristics of waters from ombrotrophic and minerotrophic peatland are fairly well known as well as the ecological amplitudes of peatland plants and changes in floristics which occur along nutrient and moisture gradients (Jeglum 1971; Ruuhijärvi 1983). This indicates that the ecological gradients in peatlands are gradual and no sharp limit exists between ombrotrophic and minerotrophic vegetation (Parkkarinen 1976; Jeglum et al. 1974).

Compared to other wetland types bogs are at the dry and nutrient deficient end of the "complex gradients". Peat accumulation in bogs tends to isolate the vegetation from the underlying mineral soil and to restrict an influx of mineral-rich water from outside. The bog surface becomes raised and most of the bog plants root or grow above the water table.

In ombrotrophic bogs (e.g. raised and plateau bogs) the isolation is complete and their vegetation is solely supplied with water and minerals through precipitation and dustfall (Schwintzer 1981). In some bogs the isolation is not complete (e.g. muskegs on unilateral slopes) and influence of telluric waters can be seen in relatively high concentrations of minerals and presence of plant species indicative of minerotrophy (Sjörs 1963; Jeglum 1973).

The most commonly used chemical indicators of minerotrophy in peatland waters are pH and Ca concentration. For waters to be ombrotrophic they should in general have pH of less than 4.5 and Ca concentration of less than 2 mg L⁻¹, while minerotrophic waters have higher pH and Ca concentration (Karlin and Bliss 1984).

None of the stands sampled in the present study falls into the truly ombrotrophic category on the basis of water chemistry. However many of the stands from the northern part of the study area come from elevated peat plateaus

where the vegetation does not have contact with telluric water. Vegetation of peat plateaus and other peat landforms with permafrost, elevated above the water table of surrounding fens is generally considered ombrotrophic (Sjörs 1963; Horton et al. 1979).

Sampling of soil water from peat plateaus is difficult since they seldom have free water on them, except where it is found in small isolated pockets which may gather runoff water from surrounding and higher parts. Due to this, water sampling is frequently not carried out in these landforms if they occur in peatlands under study, as water samples cannot be obtained in the same manner as from landforms which have free water (Horton et al. 1979). Substrate chemistry is then often the alternative used to compare nutrient status of different sites (Jasieniuk and Johnson 1982). However direct comparisons of sites still remains difficult if the measurements are done under standard moisture conditions. These conditions do not take into account differences in field moisture conditions which affect ion activity and concentrations in the peat solution that are particularly important in the drier peats (Jasieniuk and Johnson 1982). Two of the stands (8 and 11) from peat plateaus had free standing water in depressions from which the water samples were collected. The water chemistry of these samples differs considerably from that of the other samples from peat plateaus, and is closest to the ombrotrophic category. The

waters from these two stands have by far the lowest Ca, Mg and K concentrations, conductivity and absorbance (Table 1). pH is however not the lowest, but it is in the low range. The water samples from the other plateau stands, which were obtained by squeezing water from surface peat had very high mineral concentration, especially Ca ($15-35 \text{ mg L}^{-1}$), Mg ($1.3-3.9 \text{ mg L}^{-1}$) and K ($2.8-32.0 \text{ mg L}^{-1}$), considering that the sites from which they came are not fed by telluric water. The water obtained from the peat that way is held in it against gravity by capillary forces (Puustjarvii and Robertson 1975). It occupies smaller spaces in the peat (Clymo and Hayward 1982), is in closer contact with the organic matter and contains higher amounts of solutes than the free water occupying larger spaces in the peat of the moister sites.

In spite of the high mineral concentrations and conductivity most of the water samples from these dry stands had very low pH values (< 4.5) and were in the ombrotrophic range. A characteristic of these acid samples is that they have a very high absorbance, usually over 1.0. This indicates that in these waters low pH is associated with a high dissolved organic matter content, e.g. organic acids. Gorham (1957) states that high acidity of most ombrogenous bog waters in Britain is mostly due to free sulfuric acid, and an abundance of colloidal organic acids in the peat water. Gorham (1957) describes conditions of exceptionally high

levels of both acidity and metal cations in bog waters which can occur in extremely dry weather. This is due to oxidation of hydrogen sulfide to acid and an exchange of hydrogen ions for cations adsorbed on the peat. Hemond (1980) working in an ombrotrophic bog in Massachusetts found that the high acidity of the bog waters was dominantly influenced by the presence of weak organic acids, while the effects of cation exchange on increasing mineral acidity of the bog waters were modest by comparison.

These findings indicate that the high acidity of mineral-rich peat waters from the dry plateau stands could be maintained by their high content of dissolved organic acids. Table 2 shows that for the water chemistry data set, there is a high, negative correlation between pH and absorbance (-0.67), which is of a similar magnitude to the correlation between pH and conductivity (0.68). Glaser et al. (1981) describe similar relationships in surface water from the Red Lake Peatland, where pH is more strongly associated with absorbance than conductivity.

The apparent shift in water chemistry associated with the change from raised peat landforms in the permafrost zone to the sloping muskegs of southern areas with shallow peat sitting on calcareous parent material must partly be due to the differences in water sampling methods, since an abrupt change in vegetation composition is not associated with the change in water chemistry. In the more moist southern

stands with abundant free water, the samples were usually obtained from depressions with standing water or shallow pits dug in low parts of the stands. The light colored and strongly minerotrophic waters (pH 7.1-8.3, Ca 26.4-55.8 mg L⁻¹, absorbance 0.17-1.08) obtained in these stands (Table 1) reflect flowing waters in close contact with the calcareous parent material. This probably does not describe the conditions prevailing in the upper part of the peat profile, (e.g. in the upper level of the hummocks).

Studies have shown that steep chemical and floristic gradients can exist between hollows and hummocks, especially if the parent material is calcareous or the peatland is fed by mineral-rich water (Bellamy and Rieley 1967; Jeglum 1972, 1973; Karlin and Bliss 1984).

1.6.2 Vegetation

The bog vegetation studied in this work and its relative homogeneity shows considerable similarity to that described in other studies in Manitoba (Ritchie 1956, 1960a, 1960b, 1960c; Zoltai and Tarnocai 1971; Reader and Stewart 1971; Sims and Stewart 1981), Alberta (Horton et al. 1970; Karlin and Bliss 1984), Saskatchewan (Jeglum 1972, 1973), Ontario (Sjörs 1963; Vitt and Bayley 1984), Northwest Territories (Jasieniuk and Johnson 1982) and eastern Canada (Dansereau and Segadas-Vianna 1952; Forster 1984). Likewise studies of bog vegetation in the north central USA, e.g. Minnesota

(Heinselman 1963, 1970; Glaser et al. 1981; Vitt and Slack 1984) and Michigan (Schwintzer 1981) show that the northern U.S. areas are strongly similar to the southern Canadian bogs.

The stands of this study are distributed along a strong climatological gradient. A distance of over 500 km separates the northern Gillam area (stands 1-9) from the Devils Lake stands (30-33), and the Hadashville stand (34) is approximately 720 km south of Gillam. The direct effects of climate on the vegetational variation encountered were not pronounced since the more important understorey species sampled (Table 9) are found throughout the study area. Stand 8, for example, is the only forest stand from the Gillam area that does not show evidence of fire-history and disturbance. The stand is positioned close to the southern forest stands on the ordination diagram (Figure 4) and is in a cluster (Figure 10) with stands from southern and central parts of the study area.

Comparison with the literature suggests that the bog vegetation sampled is from the dry end of the moisture gradient in bogs. The bogs show the strongest resemblance with the vegetation of moss and lichen muskegs, palsa and string bogs from northern Manitoba (Ritchie 1960a,b), and the vegetation of bog and muskeg from Candle Lake, Central Saskatchewan (Jeglum 1971,1972).

The dominance of Ledum groenlandicum in the shrub-layer of most of the stands is one indicator of their dryness. In central and western Canada, L.groenlandicum replaces Chamaedaphne calyculata, which usually dominates in the shrub-layer of bogs in eastern regions of Canada (Dansereau and Segadas-Vianna 1952; Jeglum et al. 1974) where precipitation is higher. The Ledum-bogs represent a late stage in bog development (Segadas-Vianna 1955).

Another indicator of the stand dryness is high lichen cover. In about one-third of the stands, lichens had higher prominence than Sphagnum-mosses (mainly S.fuscum). All of these stands are from the central and northern part of the study area and situated on raised peat plateaus or other sites that topographically are not favorable for the moisture input necessary for active Sphagnum-growth. As previously described, the lichen-rich stands had partly decomposed Sphagnum peat (von Post 3-5) in their top layers (Table 1). This is a drier substrate with lower hydraulic conductivity, and more favorable for lichen growth than fresh Sphagnum peats (Jasieniuk and Johnson 1982).

By comparison, the moister Sphagnum-rich stands with active Sphagnum-growth, were mainly found in non-raised situations, either at the base of upland slopes or on unilateral slopes between upland and fen or swamp on the lower edges. Topographically the stands are similar to those described by Jasieniuk and Johnson (1982) and Jeglum (1972). Under these

conditions moisture input is higher than in the lichen-rich stands. These Sphagnum-rich stands were most common in the southern part of the study area south of the permafrost zone and its associated peat plateaus and palsas. Two northern stands with permafrost (8 and 11) had relatively high Sphagnum fuscum covers but low lichen presence. Both stands had free standing water in depressions and occupied a relatively low position on the peat plateaus where they occurred.

The trend in the undisturbed or relatively undisturbed vegetation from dry lichen-rich stand to moist Sphagnum-rich stands is reflected fairly well in the ordination results. These results indicate that the change in vegetation composition is gradual but with some discontinuity at the dry end of the gradient (Figure 4). This is also brought out in the cluster analysis, where stands 4, 12 and 21 join and form a cluster (Figure 10).

A few of the less important species used in the vegetation analysis were found only in northern or southern stands. In most instances these limits to distribution were associated with localized physical conditions (e.g. disturbance, minerotrophy) within the spectrum of the sampled stands. Three species were limited in geographical distribution within the study area. Vaccinium uliginosum and Pinquicula villosa are both species with northerly distribution in Manitoba (Scoggan 1957) and in the study they were only found in stands in the Gillam area. Sarracenia purpurea was found in only four southern stands, which agrees

with the known southern distribution of the species in Manitoba (Scoggan 1957).

Three moss species Pohlia nutans, Ceratodon purpureus and Drepanocladus aduncus, were confined to northern stands (1-13), except that C.purpureus was found in one southern stand (26). The presence of these species in the northern stands is associated with the disturbance and habitat change in some of these stands, but all the species have a wide distribution which extends far outside the study area (Crum 1983). Pohlia nutans and Ceratodon purpureus are both known colonizers of bare and dry peat surfaces, but in the study these conditions were mostly confined to the raised peat plateaus. Stand 26, where Ceratodon purpureus also occurred, was relatively dry with vegetation composition similar to the dry and disturbed northern stands. On the other hand Drepanocladus aduncus occurred in the very wet and disturbed stands of the northern area, which had standing water in them and fen-like vegetation. D.aduncus is common to the sedge-rich calcareous fens of western Canada (Jeglum 1972; Crum 1983).

Although the vegetation of the southern stands is dominated by plants characteristic of ombrotrophic bogs, they contain a few species which are commonly found under minerotrophic conditions. Four species, Tomenthypnum nitens, Juncus balticus, Andromeda glaucophylla and Equisetum fluviale, are all confined to the strongly minerotrophic

southern stands (22-33), but other workers have found them in bogs associated with minerotrophic conditions (Jeglum 1973; Heinselman 1970; Sjörs 1963). Tomenthypnum nitens was also found outside this area in stand 15, which in terms of environmental conditions and vegetation was very similar to the southern stands. Several other species indicative of minerotrophic influences (Sjörs 1963; Heinselman 1970; Jeglum 1973) were found throughout the study area and not just confined to the strongly minerotrophic southern stands. Examples of these are: Equisetum pratense, E. arvense, Calamagrostis spp., Carex aquatilis, Betula glandulosa and Larix laricina.

1.6.3 Vegetation/Environment

In the data analysis the vegetation-environmental relations are explored in a stepwise manner. Firstly the Deco-rana axes were interpreted directly from the stand positions and species-abundances distribution diagrams. These suggested that axis 1 represented a moisture gradient and axis 2 a disturbance (bare ground) gradient (Figure 4).

The next step was to combine the variation in the environmental variables and the stand positions in scatter diagrams for each variable, which showed strong trends and obvious relationships between vegetational and environmental variation (Figure 12). However, the direction of change in the environmental parameters did not in general correspond

with the two axes directions, the closest visual correspondence being between bare ground and axis 2 (Figure 8).

No variable could be used in the analysis as a direct indicator of moisture regime, since depth of the water table was not available for all stands. None of the variables shows a clear change in the direction of axis 1.

The correlations between the Decorana axes and the environmental parameters explained the relationships further, but the environmental parameters show in general higher correlations with axis 2 than axis 1 (Table 2). Axis 1 has the highest correlation with conductivity (0.42) and a slightly lower but significant correlation with Ca, Mg, NO_3 , pH and humification, indicating that as well as being a moisture gradient it also shows a considerable relationship with nutrient-regime and represents a "complex gradient". The change along the axis is from dry ombrotrophic to wet and minerotrophic conditions.

Axis 2 has the highest correlation with bare ground (0.67), which confirms the disturbance aspect associated with the axis. However, the axis is also significantly correlated with Mg, K, NO_3 , absorbance, pH and humification and clearly does not represent a separable gradient.

In the CCA it was attempted to relate quantitatively the variation in the vegetation as presented in the Decorana results, to the variation in the environmental parameters.

However, the question arises about the nature of the vegetation variables and whether the CCA brought forward anything significant in these relationships that was not obvious from the direct interpretation of the stand positions, species-abundances distributions on the ordination diagrams, the scatter diagrams and the correlations relating these positions and the environmental parameters.

Various workers have found CCA ineffective and questioned its assumptions of linear relations between environmental variables and vegetation. (Gauch and Wentworth 1976; Gauch 1982; Baker 1983). However, CCA has been found useful as an exploratory tool to summarize and detect the presence of vegetation - environmental relationships and in determining their directions (Gittins 1979; Baker 1983), as was found in the present work. The results of the CCA revealed quite strong relationships between the vegetation and environmental parameters and more than half (61%) of the vegetation variation could be explained by the variation in the environmental variables. This is high compared to some other studies using CCA to investigate plant-environment relationships (Huntley and Birks 1979; Eldred and Maun 1982). A strong relationship was also indicated by the simpler methods. However, it was not quantified and summarized with a single value as in the CCA. The canonical variates for the vegetation set (V_1 and V_2) show the directions of the two main complex environmental gradients within the vegeta-

tion data set (Figure 14), which was not revealed as clearly by using other methods.

1.6.4 Forest/right-of-way contrasts

In the bogs studied, different types and intensities of disturbance occurred. The effects on vegetation, soil parameters and recovery patterns seem to have been determined by the degree of surficial disturbances and how it interfered with the key parameters of water and ionic supply.

Within the right-of-way the disturbances at all the sampled sites are due to the initial line clearance with tree removal and construction activities, plus continuing traffic of heavy equipment used for line maintenance and vegetation management. Due to the northern location and slow regeneration of trees at most of the bog sites they had in 1983 not required vegetation management since line construction occurred. At the southernmost right-of-way sites (Long Point, Devil's Lake 1 and 2), herbicides have been applied in the last few years to control tree growth, but many common bog plants are very sensitive to herbicides (see chapter 3) which can cause considerable changes in vegetation of the bogs.

1.6.4.1 Environment

The impact of clearance and construction activities on the ground vegetation and soil varied along the length of the transmission corridors. Tree removal probably had similar effects at most sites, but severe disturbance of ground

vegetation and soil was mostly confined to the peat plateaus in the Gillam area which had been scraped and left bare in places.

The effects of forest stands on microclimate are complex and quantitative changes associated with canopy removal are difficult to measure. Also, the effects may vary with tree type, height and density, climatic and geographical conditions. Canopy removal will lead to modifications of energy transfer, air flow, the humidity and thermal environment (Barry and Chorley 1976).

Decomposition rates in bogs are largely determined by peat temperature. Temperature changes of even one or two Celcius degrees may alter carbon fixation/release ratios (Silvola and Hanski 1979). A temperature increase may shift peat forming conditions to net breakdown of peat. Brown (1976) studied the effects of canopy removal on peat temperature regime in a black spruce bog in Minnesota. The canopy removal did not affect peat temperature down to 1 m depth and alterations in microbial activity and decomposition processes due to temperature changes were not anticipated. In the soil respiration study (see chapter 2) it was not evident that canopy removal had caused consistent differences in peat temperatures between the forest and right-of-way plots.

In the northern bogs with permafrost, canopy removal may affect snow accumulation and the thickness of the unfrozen peat layer in the winter. Zoltai and Tarnocai (1971) found that on a palsa complex in the southern fringe of the permafrost region in Manitoba, the thickness of the active layer was least under the densest black spruce stands, increased with the decreasing tree density and was greatest in the openings. In the present study measurements of depth of the active layer indicated that there were no differences between the forest and right-of-way stands. However in the soil respiration study (see chapter 2) average depth of the layer was found to be slightly greater in the forest than in the right-of-way. In September of 1976, Sims and Stewart (1981) working in the area of the Gillam 2 site of the present study, found the same depth of active layer in the forest and right-of-way and concluded that the removal of trees and vegetation cover had not affected the frozen core in the 7 years since the disturbance occurred. Studies of the effects of forest fires on wooded peat palsas and plateaus in the Hudson Bay Lowland and in the southern part of the discontinuous permafrost zone in Manitoba have shown that the burning of the tree canopy and understorey vegetation appears to have little influence on the permafrost or rate of its collapse (Brown 1968; Thie 1974).

One of the effects of canopy removal is a reduction in losses of water by evapotranspiration. This may lead to the

rise in the water level and result in stimulation of Sphagnum growth and paludification in bogs (Jeglum et al. 1974; Tallis 1983). As described above, Sphagnum prominence was significantly lower in the right-of-way than in the forest. If the moisture regime is more favorable for Sphagnum growth in the right-of-way, then other factors must have affected Sphagnum.

Another factor associated with canopy removal and right-of-way construction is soil compaction, from the use of heavy equipment. Compaction of the top 20 - 40 cm of the surface peat, through which most of horizontal water movement occurs, may result in a reduction in hydraulic conductivity and a permanent rise in the water table (Boelter and Close 1974, cited in Grigal 1983). In a study of the impact of right-of-way construction on soil bulk density in the Red Lake Peatland, Grigal (1983) found that soil bulk densities were 5% and significantly higher in the right-of-way than in the adjacent forest when data from different peatland soils for 2 years following construction were combined. These differences were considered small and not likely to cause any restriction of water flow across the peatland. At low shrub and treed bog sites significant differences in bulk density were not found between forest and right-of-way samples for data for 1 and 2 years following construction. In the sampling areas disturbances had not caused changes such as deep rutting or other soil displacement. The construc-

tion of the transmission line was mostly done in the winter-time to minimize disturbance. This study from Minnesota and its results have considerable relevance to the present study as the general intensity of disturbance on the soil caused by vegetation clearance and construction activities appears to have been similar in the two regions.

The most intense disturbance of bog soils in the present study was found in areas frequently traversed by heavy vehicles and equipment, e.g. access paths and centre path in the right-of-way where stands 5 and 9 (Figure 5 d) are examples, but these stands had deep ruts. Permafrost had thawed and the water table was close to the surface, pH and conductivity of water samples were higher than of the water samples from the adjacent forest stands. This indicates contact of the water with mineral soil (Table 1). The effects of vehicle perturbation upon soil-plant system on tundra and half-bog soils near Barrow, Alaska have been studied (Gersper and Challinor 1975; Challinor and Gersper 1975). The study was carried out six years after the disturbance of the tundra surface by several years of infrequent passage by tracked vehicles. The soils within the tracks were found to have higher bulk densities and temperatures, deeper active layer during summer and lower moisture percentage than the undisturbed soil. The depressions formed by the tracked vehicles resulted in nutrient inflow into them enriching their nutrient reservoir, therefore the warmer track

soils contained higher amounts of exchangeable bases and were less acid than the undisturbed soils. Nutrient content and pH of the soil solution was also higher in the track soils. In the enriched environment of the tracks plant size and production was enhanced. Changes in floristic composition also occurred. Although these disturbances described from Alaska are less severe than those found in vehicle tracks in the right-of-way of stands 5 and 9, the changes in the soil-plant environment associated with them are probably somewhat comparable. This can be seen in nutrient enrichment, altered plant composition and enhanced growth in the stands.

The type of disturbance encountered in stands 5 and 9 was not common in the right-of-way at the sampled sites and it represents an extreme which in the total right-of-way area is of relatively low proportion. The disturbances in the northern right-of-way (e.g. stands 1, 3 and 7) where the surface of the peat plateaus had been scraped (Figure 7 a), removing the ground vegetation and thin layer of top peat in places were considered the most widespread, severe disturbances of vegetation in these locations. However in terms of habitat change they can not be looked upon as severe. Only a thin layer of top peat had been removed leaving underneath a substrate of the same or similar chemical and physical properties as that removed. These disturbances did not appear to interfere with moisture or nutrient regimes nor did they affect permafrost of these sites.

At most of the southern right-of-way sites severe disturbances of ground vegetation and soil did not seem to have occurred during canopy removal and line construction and damages from traffic were small. At one southern right-of-way site, Devil's Lake 1, stand 30, extreme disturbance had occurred and there was almost a complete change in vegetational composition with a sedge fen replacing bog vegetation (Figure 7 d). The reason for this change is not clear. The herbicide Tordon 101 (picloram + 2,4-D) was applied in the area in 1979 which is the only application of herbicides at the site since line construction (MacLellan 1982). Sphagnum fuscum and the low ericoid shrubs are very sensitive to picloram applications (see chapter 3). However the application in 1979 can hardly have caused the change in only 4 years, but at the same time the Devil's Lake 2 site (stand 32) was treated with the same herbicide without such a dramatic change in vegetation. The possibility remains that the site (30) was treated with herbicides earlier without the application being documented, or that heavy ground disturbance occurred at the site during line construction and canopy removal.

The MANOVA analysis showed no significant difference in water chemistry or all environmental parameters between forest and right-of-way. The individual water chemistry parameters, with the exception of pH, showed enrichment, with NO_3 and absorbance being significantly higher in the right-of-

way. The right-of-way stands had also significantly more bare ground and humified top peat than the forest stands (Table 10).

In the ANOVA and MANOVA analyses no segregation of sites was made on the basis of intensity of disturbance or location. However, it is evident (Table 1) that the northern sites show some differences in environmental parameters between forest and right-of-way. Some of these contrasts between the forest and right-of-way, (e.g. the Gillam 2 and Joy Lake site), are attributable to differences in type of water sampling.

The number of samples from each stand was very low so the variation in the environmental parameters within a stand was probably not covered as well as in the vegetation sampling scheme. However bearing these shortcomings in mind the results of the environmental parameters indicate that in spite of a slight enrichment in terms of water chemistry of the overall right-of-way environment studied the disturbances have not caused distinct habitat changes.

1.6.4.2 Vegetation

In the ANOVA and MANOVA analyses of the vegetation, as for the environmental variables, sites were not categorized on the basis of location or degree of disturbance. Separation was only made between forest and right-of-way. Consid-

erable differences were found, with five of the common species showing significant difference in abundance. The results of the MANOVA were dependent on the number of species included in the analysis, but when the ten most common species were used significant differences in vegetation composition were found between forest and right-of-way. The outcome of the analysis has no doubt been shaped considerably by the few extremely disturbed sites where the contrasts in vegetation were greatest. At the majority of the sites vegetation change due to or following disturbance were not considerable as shown in the results of the ordination and cluster analysis. Clearly the change in vegetational composition and species abundances has been determined by the type and degree of disturbance that has occurred at each site.

On the basis of vegetational change the right-of-way stands can be put into three categories. Firstly there are the relatively undisturbed stands with little or no bare ground, hardly any noticeable changes in species composition, but some in abundances. Secondly, there are the stands where surficial disturbances occurred exposing bare ground but without significant alterations in substrate type or moisture condition. In these stands considerable reduction in species number and abundances occurred which has partly been made up for by colonizing species. Thirdly, there are the stands where the substrate has been highly

disturbed and the moisture and nutrient regime has been altered. In these stands species composition and physiognomy has been greatly altered.

Relatively undisturbed right-of-way stands:

In an area disturbed by logging or canopy removal the regeneration will be dependent on the intensity of the cut, to what extent it affects underground plant parts, the survival of buried seeds, and on the import of disseminules to the area (Moore and Wein 1977). In a study of plant succession following logging and burning in the western cascades of Oregon, Dyrness (1973) showed that in areas where the ground vegetation remained undisturbed by the clear cut little alterations in the vegetation followed. Thus, if disturbances do not destroy large expanses of vegetation or disrupt the substrate, regeneration can be from existing roots and rhizomes. The small exposed patches are colonized from the sites and they are of insufficient area and separated by too long distances to support large amount of colonizing species (Hernandez 1973).

In the right-of-way stands (14, 16, 20, 22, 24, 28, 32) which were classified into cluster 3 in the cluster analysis (Figure 10) and in stand 18, relatively little vegetational change had occurred from the corresponding forest stands (Figure 6 d).

Stand 20, 22 and 24 are in an area where herbicides were applied in the right-of-way in 1973. 2,4,-D was mainly used, but 2,4,5-T was used sparingly. The herbicides were applied only in spots (MacLellan 1982) so these stands may not have been treated. At least the vegetation does not show any long term adverse effects.

Tordon 101 was applied in the right-of-way in the area of stand 32 in 1979. Sphagnum fuscum plants had been killed, reducing the species prominence nearly 75% from that of the forest (47.2 to 11.4) and exposing bare peat on the hummocks. In the right-of-way PV for bare ground was recorded 13.3 but only 1.3 in the forest. Oxycoccus microcarpus, Vaccinium vitis-idaea, Empetrum nigrum and Sarracenia purpurea were all reduced in prominence in the right-of-way compared to the forest while Ledum groenlandicum had similar prominence in the right-of-way and forest. Carex aquatilis and Polytrichum strictum had higher prominence in the right-of-way than in the forest, but both the species are resistant to the herbicide (see Chapter 3) and seem to benefit from the disturbance. The species composition in the right-of-way stand compared to that of the forest has not changed to any degree, the shift is only in species abundances.

The herbicide Tordon 10K (picloram) was applied in the right-of-way in the area of stand 28 in July of 1982. When vegetation sampling was carried out in the stand a year later some doubt remained about whether some of the plants in

the stand should be recorded as alive or left out entirely as they looked dead or dying. For this reason the prominence of Sphagnum fuscum, Ledum groenlandicum, Kalmia polifolia, Oxycoccus microcarpus, Vaccinium vitis-idaea and Saracenia purpurea has probably been overestimated, but these species showed all symptoms of having been affected by the herbicide. (In a visit to the site in 1984 it was evident that nearly 100% of the Sphagnum fuscum was dead, and at that time would have been recorded as bare ground. The ericoids however showed some signs of recovery). As in right-of-way stand 32, Carex aquatilis and Polytrichum strictum had increased in prominence in the stand, and overall changes were in species abundances rather than composition.

In these ten right-of-way stands where the disturbances were not considered high the main trends in the vegetation change from the forest were these: Ledum groenlandicum showed a decline in one stand, increased in three and was of similar abundance in right-of-way and forest at four sites. Vaccinium vitis-idaea and Oxycoccus microcarpus declined in more than half of the right-of-way stands but were also found to increase in a few stands or be of similar abundance in forest and right-of-way. Polytrichum strictum was the only species which showed a change in the same direction in every right-of-way stand but the species had higher prominence in all right-of-way stands than in the corresponding forest stands.

In their study of the effects of fire and habitat on peatland vegetation in the Northwest Territories, Jasieniuk and Johnson (1982) found that Polytrichum strictum increased in abundance within a short interval after fire and peaked after 10-20 years but declined in abundance in older stands. Ledum groenlandicum and Vaccinium vitis-idaea were also found to have excellent regenerative ability and recover shortly after fire, while Sphagnum fuscum tended to recover more slowly. Habitat changes did not in general result from the fires and vegetation changes were in species abundances rather than stand composition. In stands where the vegetation was only partly burned it recovered rapidly to nearly predisturbance composition.

Black and Bliss (1978) also studied the recovery sequence of black spruce forest after fire in the Northwest Territories. They found very little change in species composition of vascular plants with increasing time after fire, but Polytrichum juniperinum was one of the early colonizer of burnt stands, which predominated during the first 15-20 years. Suffling and Smith (1979) reported a rapid increase in P. juniperinum and P. commune following applications of Tordon 101 on vegetation in a power line right-of-way in Ontario.

Disturbed right-of-way stands with relatively unaltered substrate conditions:

The vegetation in right-of-way stands no 1, 3, 7, 10, 12 and 26 had obviously been altered considerably by disturbances without severely affecting the substrate conditions. In stands 1 (Figure 7 a) and 3 which had been cleared in the winter of 1977-1978, less than half of the surface was covered with plants. Stands 7, 10 (Figure 7 b) and 12, which were cleared of vegetation about 10 years earlier, also showed signs of having been scraped, but vegetation cover was however over 80%.

The removal of the top peat has more severe effects on shallowly rooted plants than those with deep root systems. The shrubs most common in the study, (i.e. Ledum groenlandicum, Vaccinium vitis-idaea and Oxycoccus microcarpus), are all shallowly rooted, (Karlin and Bliss 1984) having their roots and rhizomes located within 5-10 cm of the surface (Hernandez 1973). Scraping of the peat will therefore remove the roots and rhizomes of these species so they cannot resprout. The viability of seeds in bog soils is very low and seedling emergence is mainly from the top few centimeters of the undisturbed soil (Moore and Wein 1977), which will be removed by blading and further hinder rapid regeneration of these surfaces.

In the bare stands 1 and 3, Rubus chamaemorus had the highest prominence (PV 1.7 and 6.2) of the vascular plants, but the species has a deep root system (Karlin and Bliss 1984) and is known to recover rapidly after surficial distur-

bances (Bellamy et al. 1971). In stand 1, Ledum groenlandicum is the only other recorded vascular plant but its prominence is extremely low (0.1). In stand 3 other vascular plants in decreasing order of prominence are Ledum groenlandicum, Vaccinium vitis-idaea, Salix spp., Betula glandulosa, Epilobium angustifolium and Vaccinium uliginosum.

Polytrichum strictum appears in few patches of the two stands and also in lower abundance are Ceratodon purpureus and Pohlia nutans which are common early colonizers of disturbed peat surfaces (Black and Bliss 1978; Jasieniuk and Johnson 1982). Lichen establishment in the stands has been slower than of the mosses, but Cladonia gracilis is found in low abundance in both stands. Jasieniuk and Johnson (1982) describe the species as recovering rapidly and acquiring high abundance after fire. In stand 1, Cladonia cristatella was also recorded and Cladina mitis in stand 3. Stands 7, 10 and 12 with a longer recovery time and where the disturbances probably were not as severe, lichen cover and species number are considerably higher. The species Cladina stellaris, Cladonia gracilis, C.deformis, C.cornuta and Icmadophila ericetorum are found in all three stands, Cladina mitis and C.rangiferina in stands 10 and 12 and Cladonia cristatella in stand 7. Sphagnum fuscum has been replaced and is only present in low abundance (PV < 1) in stand 12. Polytrichum strictum has instead become the dominant moss in all three stands. Ledum groenlandicum has a high prominence

in the stands and in stand 7 and 10 it is of comparable abundance to what it is in the forest stands. Other low shrubs are also well established in the stands, e.g. Kalmia polifolia (stands 7 and 12), Chamaedaphne calyculata (stands 10 and 12) and Vaccinium vitis-idaea (in all three stands).

At the Grand Rapids site the vegetation in the right-of-way, stand 26, was very different from that of the forest stand, 27. Twenty-four of the common species used in the vegetation analysis were recorded at the site but only eight were shared by the two stands. On the site it did not appear that the right-of-way stand had been scraped or heavily disturbed by traffic and the cause of the vegetation change was not clear. The stand is however in an area which was treated with herbicides (2,4-D + 2,4,5-T) in 1973 and it may well be that the stand area was sprayed, although this cannot be confirmed. The peat surface in the stand was unusually hard and dry and the top peat more humified than in the forest. Sphagnum fuscum was nearly absent from the right-of-way (PV < 1), but in the forest it had high prominence (PV 45.3). Contrary to the trend in most other disturbed right-of-way stands Polytrichum strictum did not increase in abundance and its prominence was less than in the undisturbed forest. Three other mosses were recorded in the stand and in low abundance, Ceratodon purpureus, Dicranum undulatum and Bryum pallescens. The vascular plants (Equisetum pratense, Carex aquatilis, Ledum groenlandicum,

Oxycoccus microcarpus, Vaccinium vitis-idaea, Geocaulon lividum) shared by two stands, had a considerably lower prominence in the right-of-way, but that was partly made up for by a high abundance of Kalmia polifolia, which was not recorded in the forest. No lichens were found in the forest, where the substrate was mostly living Sphagnum, but on the dry and dead Sphagnum peat substrate in the right-of-way stand lichens had become established in low abundance, with Cladonia gracilis being most prominent. Other lichens species in the stand were Cladina mitis, Cladonia cristatella, C. cornuta and Peltigera polydactyla. An unusually high number of graminoids and other opportunistic species was found in the right-of-way. The abundance of individual species was low but together they gave the stand a distinct appearance. Among these species were: Agrostis borealis, Agropyron repens, Bromus ciliatus, Muhlenbergia glomerata, Potentilla fructicosa, Erigeron acris, Solidago spp. and Epilobium angustifolium.

Highly disturbed stands with altered substrate conditions:

In the three right-of-way stands, 5, 9 and 30, surficial disturbance is the greatest and a definite habitat change has occurred at the sites with a consequent change in vegetation. Some of these changes have been described above and they will not be repeated here. What the three stands have in common is an increase in wetness. This has led to a dramatic increase in Carex aquatilis now the dominant plant in

all three of them, but the species was only found in low abundance in the corresponding forest stands (Figure 6 c). Throughout this study it was recorded at the majority of sites in low abundance, being mainly absent from northern plateau stands with absence of standing water.

Carex aquatilis is a deep rooted species which can grow both in hollow and hummock communities in peatlands, though with lower cover in the higher locations (Jeglum 1971; Karlin and Bliss 1984). It has been found to be an important colonizer of wet sites in disturbed northern peatland and tundra communities (Bellamy et al. 1971; Hernandez 1973; Challinor and Gersper 1975; Van Cleve 1977). It's roots appear to withstand severe surficial disturbances and it has an ability to respond quickly and take advantage of the improved nutrient conditions and reduced competition for resources by other species following a disturbance. It's rhizomes readily invade these disturbed areas of increased wetness.

Stand 30 is fairly homogenous in terms of substrate condition and vegetation. There has been a complete alteration of vegetation within the entire stand area and all characteristic bog plants have disappeared. In the northern stands (5 and 9) however the substrate disturbance has not occurred in the entire stand areas. It is in the ruts themselves where Carex aquatilis and associated fen species are dominant. Outside these ruts, patches with the bog vegeta-

tion are found and in stand 5 most of the original bog species still remain within the stand. The stand ordination results demonstrate this difference in vegetation change in these three stands quite well with stand 5 departing least from the other stands, and stand 30 showing the greatest departure (Figure 4).

1.7 SUMMARY AND CONCLUSIONS

Analysis of the vegetation data with Decorana-ordination and cluster analysis showed similar trends in the vegetation forming major groupings of: dry lichen-rich stands, moist Sphagnum-rich stands, disturbed and wet sedge-rich stands, and moist and disturbed stands with high abundance of bare ground.

In terms of vegetation change Decorana axis 1 was interpreted as a moisture gradient with dry lichen-rich stands having the lowest and wet sedge-rich stands the highest scores on the axis. Decorana axis 2 was interpreted as a disturbance (bare ground) gradient with relatively undisturbed Sphagnum-rich stands having the lowest and disturbed, high bare ground stands the highest scores on the axis.

The most distinctive trend in the environmental parameters was the difference between the northern raised permafrost stands and the southern frost free and non-raised shallow peat stands underlain by calcareous parent material. The raised bog stands were characterized by relatively low pH (4.1-6.1), Ca concentration (3.2-35.7 mg L⁻¹), conductivity (31-227 umho cm⁻¹), dark color (absorbance 0.42-1.62) of water samples, and their top peat was generally slightly to fairly humified (von Post 3-5). The water samples from the southern stands had relatively high pH (7.1-8.3), Ca concentration (26.4-55.8 mg L⁻¹), conductivity (169-427 umho cm

-1), light water color (absorbance 0.17-0.69) and their peat was generally light colored and mostly undecomposed (von Post 1-3).

The CCA results showed a significant relationship to exist between the vegetation (Decorana axis 1 and axis 2 stand scores) and environmental parameters. The environmental variables explained 61% of the variance in the vegetation data set. Squared multiple correlations gave significant correlations between all environmental parameters, except PO_4 concentration, and the vegetation data set. The highest correlations were with bare ground (0.49) and pH (0.47).

Of the fifty species used in the vegetation analysis seven showed significant differences in prominence values in the univariate ANOVA between forest and right-of-way. Sphagnum fuscum, S.nemoreum, Pleurozium schreberi, Oxycoccus microcarpus, Vaccinium vitis-idaea and Drosera rotundifolia were all significantly ($p < 0.05$) reduced in abundance in the right-of-way compared to the forest with the last species not being recorded in the right-of-way. Only Polytrichum juniperinum had increased significantly in abundance in the right-of-way compared to the forest according to the ANOVA. The MANOVA did not show significant differences in vegetational composition between forest and right-of-way when the sixteen most common species were used, but differences were clearly significant when the analysis was based on the ten most common species.

Vegetation differences between the right-of-way and forest reflected altered species abundances rather than composition. Disturbed areas were generally not invaded by weeds or fugitive species. Polytrichum strictum was the main colonizer of disturbed peat surfaces with relatively unaltered moisture conditions, while Carex aquatilis became the dominant in disturbed stands of increased wetness.

The MANOVA for the environmental parameters did not show significant differences between the forest and right-of-way. In the univariate ANOVA bare ground and humification of the top peat was significantly higher in the right-of-way. The water chemistry parameters indicated an enrichment of the right-of-way with higher Ca, Mg, K, NO₃ and PO₄ concentrations, conductivity and absorbance measured in the right-of-way than in the forest. Of these only NO₃ concentration and absorbance were significantly higher in the right-of-way, in the univariate ANOVA. A MANOVA of the water chemistry variables did not give significant differences between forest and right-of-way.

Canopy removal or scraping of surface peat in northern permafrost stands did not affect permafrost in the stands. Thawing of permafrost on peat plateaus had however occurred in stands sampled in vehicle tracks, which are regularly used for navigation, resulting in increased wetness and improved nutrient conditions and replacement of bog vegetation with fen-type vegetation. The total impact of such disturbances in the right-of-way area is thought to be minimal.

Extent of bare ground was low in the Radisson-Dorsey right-of-way, but all the right-of-way stands had over 85% vegetation cover. The two right-of-way stands from the more recently cleared Henday-Radisson right-of-way had over 60% bare ground. Signs of severe peat erosion were not encountered in stands with bare ground. Steps to promote revegetation of bare surfaces are not considered necessary as disturbances do not in general lead to erosion in the right-of-way. The native vegetation gradually recolonizes the disturbed surfaces, with complete recolonization taking less than 30 years.

Recent introduction of the herbicide picloram into the southern bogs in the Radisson-Dorsey right-of-way to control tree regeneration, indicates that many important understory bog plants, e.g. Sphagnum fuscum and the low ericoid shrubs, are very sensitive to the herbicide. Herbicidal applications may have the potential to have more severe long term effects on the bog communities than the initial vegetation clearance practices.

Chapter II

**SOIL RESPIRATION AND DECOMPOSITION IN BOGS
DISTURBED BY POWER LINE CORRIDORS AND DRAINAGE**

2.1 INTRODUCTION

In peatlands primary production is in excess of respiration and decomposition and the surplus production accumulates as peat (Moore 1975; Clymo 1983; Bradbury and Grace 1983). The wet, relatively cool, acidic and nutrient-deficient environment of peatlands retards the breakdown of organic matter.

Although the accumulation rate of organic matter in bogs is higher than in other peat-forming ecosystems, the accumulation is only a minor fraction (10-20%) of the net annual primary production (Reader and Stewart 1972; Sjörs 1981). In bogs there is a relatively small difference between yearly primary production and decomposition; hence peat formation depends on the maintenance of a delicate balance in biotic and environmental factors.

Stratigraphic records have shown that the rate of peat accumulation in bogs differs markedly from one time period to another and that peat formation may be interrupted by periods of net breakdown of peat or erosion. These changes can be hastened by changes in climate, local drainage, vegetation and through fire (Durno 1961; Nichols 1967; Heinselman 1970).

Human disturbances such as clearance by physical or chemical means cause vegetational and/or environmental changes in bogs and thus may greatly affect production and decay

processes in them. In central and northern Manitoba extensive bog areas have recently been cleared of trees during the construction of hydroelectrical transmission corridors. Since the transmission corridors will be used well into the foreseeable future continued management of vegetation in the cleared areas (right-of-way) will be necessary. These developments are therefore likely to cause long term biotic and environmental changes in the peatlands traversed by the transmission lines.

Past studies of production by bog or peatland vegetation along transmission lines in central and northern Manitoba have shown that net annual production within the right-of-way is generally reduced in comparison to that of the adjacent undisturbed vegetation (Sims 1977; Sims and Stewart 1981). No documentation exists of how soil decomposition processes in the disturbed peatlands of this region are affected by canopy removal or herbicidal applications. Such information is essential to quantify the long term impact of disturbances on peat accumulation processes.

Work by Svensson et al. (1975) and Clymo and Reddaway (1971, cited in Clymo 1983) has shown that in raised dry bogs, carbon dioxide (CO_2) is the principal end product of decomposition processes, while methane (CH_4) constitutes a low and insignificant proportion of the total carbon release.

In an attempt to assess how right-of-way disturbances interfere with soil decomposition processes and peat accumulation in bogs, soil respiration (CO_2 production) was measured in bogs near Devil's Lake, Minago River and Gillam. The bogs are traversed by the Radisson-Dorsey and Henday-Radisson transmission corridors. At each site soil respiration, soil temperature, acrotelm depth and soil moisture were measured in the right-of-way and in the adjacent undisturbed forest.

The same type of measurements as described above were also carried out at a peat mining area near Hadashville in southeastern Manitoba. The impact of the drainage at the Hadashville site on soil respiration activity and environmental factors was compared with the general impact of the right-of-way disturbances at the other sites.

2.2 LITERATURE REVIEW

2.2.1 Soil respiration: the concept and methods

The rate of CO₂ evolution from soil surfaces has been commonly used to measure soil respiration (Kucera and Kirkham 1971). Soil respiration has been defined as "the sum total of all soil metabolic functions in which CO₂ is produced" (Singh and Gupta 1977).

The main sources of CO₂ evolving from the soils of most terrestrial ecosystems are microbial, microfaunal and root respiration. The microbial and microfaunal respiration together represent decomposition in the soil by soil heterotrophs or decomposers. The decomposers receive reduced forms of carbon in plant and animal residues and oxidize the carbonaceous materials to the lowest energy state of carbon, which is CO₂ (Wildung et al. 1975).

Soil respiration as defined here does not reflect anaerobic soil metabolic activity leading to the production of CH₄, which may be an important source of carbon evolving from the soil surface in certain wetland types (Svensson et al. 1975, Clymo 1983).

It has proven difficult to obtain accurate estimates of the contribution of root systems to the total soil respiration, and estimates range from approximately 5 to 70% for forest ecosystems (Chapman 1979).

Studies of soil respiration fulfill a variety of objectives, namely a) to measure and compare soil metabolism in different ecosystems or in different successional stages within the same ecosystem, b) to study litter decomposition and breakdown of soil organic matter and to estimate litter production, c) to examine the effects of environmental factors on soil respiration, d) to investigate the effects of disturbances (e.g. pollutants, drainage of organic soils) on soil metabolic processes. (Makarov 1960; Chapman 1971; Anderson 1973; Singh and Gupta 1977; Belkovskiy and Reshetnik 1981; Prescott and Parkinson 1985).

The most frequently used methods for measuring soil respiration in the field are alkali absorption and infrared gas analysis (IRGA). The relative accuracy of these two methods has been debated, but both have associated difficulties in creating natural conditions for experimentation. The chambers used to cover the soil and trap CO₂ are usually pressed into the soil which causes disturbance from the cut edge and probably a temporary increase in respiration rate. The chambers may also alter microclimatic conditions which can affect the diffusion gradient and evolution rate of CO₂ from the soil. (Singh and Gupta 1977; Chapman 1979; Davis 1980; Gupta and Singh 1981; Poole and Miller 1982).

Most comparisons of the IRGA and the absorption method indicate that the latter may underestimate CO₂ production. Witkamp (1969) found absorption values 20% of IRGA values.

Kucera and Kirkham (1971) reported absorption values 61% of IRGA values, while Edwards and Sollins (1973) found that respiration measurements with the absorption method were 63% at 20° C and 90% at 12° C of IRGA values. In other studies similar CO₂ releases have been obtained using both methods (Ino and Monsi 1969, cited in Singh and Gupta 1977; Chapman 1979).

The efficiency of CO₂ absorbance is dependent on the volume and concentration of the alkali used and the ratio of the area isolated to the absorbing area. Minderman and Vulto (1973) using 60 ml of alkali, observed a linear increase in the rate of CO₂ absorption with increasing alkali concentration up to 0.25N, with little or no increase in amount of CO₂ absorbed with higher alkali concentration. In their study of the effect of alkali concentration, volume and absorption area on the measurement of soil respiration in a tropical sward, Gupta and Singh (1977) recorded the largest values of soil respiration over a 24 hour period using 50 ml of 0.25N and 0.5N alkali solutions. As there was no significant difference between the maximum values obtained with these solutions they concluded that 50 ml of 0.25N alkali solution was optimal for absorbing the CO₂ evolved from the soil in the habitat they were working. They also found that increasing the absorption area beyond 20% of the total surface area enclosed had no significant effect on the measured soil respiration rate when suitable concentrations of alkali

were used and that at least 35% of the alkali remained unused after the absorption.

Although the alkali absorption method may underestimate CO₂ release it has been found well suited for measuring soil respiration due to the relatively simple apparatus used and low cost, which enable extensive replication of measurements in a variety of habitats (Coleman 1973; Singh and Gupta 1977). The results obtained by the method may therefore not give absolute values but rather be a relative measure of the intensity of soil respiration. Such data can however be useful in comparing respiration under different conditions (Reiners 1968).

2.2.2 Soil respiration rates

Data on soil respiration rates from boreal forests and bog communities are scarce, but numerous studies of soil respiration have been carried out in the temperate forest zone and in the arctic tundra.

In a wet tundra meadow at Barrow, Alaska (71° N), where the soil starts to thaw after snowmelt in mid June, soil respiration increased from 75-125 to 150-300 mg CO₂ m⁻² hr⁻¹ in the last week of July after which the rates declined with falling atmospheric and soil temperatures (Peterson and Billings 1975; Billings et al. 1977).

Poole and Miller (1982) measured CO₂ release from lichen heath, low shrub tundra and tussock tundra in north-central Alaska (65-69° N) in July and August. Of green vegetation only shrubs over 5-10 cm were excluded from the respiration chambers. At all the sites CO₂ evolution decreased from July to late August. Values range from 90-260 mg CO₂ m⁻² hr⁻¹ in July to 10-80 mg CO₂ m⁻² hr⁻¹ in late August.

Svensson *et al.* (1975) determined soil respiration rate at different sites in Norwegian alpine communities (60° N). The seasonal averages ranged from 100-680 mg CO₂ m⁻² hr⁻¹, being lowest in lichen heath and highest in birch forest.

In a study of soil respiration in pine stands on the Kola Peninsula (77° N), USSR, Repnevskaya (1967) found soil respiration rates to rise from 50-150 mg CO₂ m⁻² hr⁻¹ in May to 150-300 mg CO₂ m⁻² hr⁻¹ in July. Respiration rates declined from August to October with falling air temperatures.

Anderson (1973) monitored CO₂ evolution from two deciduous woodland soils in southern England (52° N). Soil respiration was measurable throughout the year and the respiration rates ranged from about 100 to 550 mg CO₂ m⁻² hr⁻¹, with the highest evolution rates being recorded between mid-July and mid-September. The soil respiration rates determined by Chapman (1979) in Calluna heathlands in southern England over a period of one year appear rather high by comparison, but they ranged from about 70 to 1000 mg CO₂ m⁻² hr⁻¹.

Maximum soil respiration rates determined in different forest types in Germany ranged from 220 to 430 mg CO₂ m⁻² hr⁻¹ (Walter and Haber 1957, cited in Singh and Gupta 1977).

In a recent study, Prescott and Parkinson (1985) determined soil respiration in pine stands affected by sulphur pollution in west-central Alberta (54° N). The area is in the transition zone between the boreal and subalpine forest regions. Highest respiration rates, 160-180 mg CO₂ m⁻² hr⁻¹, were reported for July and August.

In a study of CO₂ evolution from the floor of three forest types in southeastern Minnesota (45° N), Reiners (1968) estimated soil respiration rates up to 1000 mg CO₂ m⁻² hr⁻¹ in an oak forest and marginal fen forest and rates up to 700 mg CO₂ m⁻² hr⁻¹ in a cedar swamp. In general the CO₂ evolution rates were found to parallel soil temperatures. The rates were highest in June and July. Soil respiration was still measureable in mid December (150-200 mg CO₂ m⁻² hr⁻¹) but December was characterized by frost and thaw periods. By mid-January the soils were frozen and covered with snow and they remained frozen until mid-March. CO₂ evolution rates were at or near zero when determined in early January after freezing of the soil and they were assumed to be negligible until thawing occurred in March. A few days after thawing respiration was detectable again and had risen above 50 mg CO₂ m⁻² hr⁻¹.

Garret and Cox (1973) measured soil respiration rates in a deciduous forest in Missouri (39° N) over a year. Respiration continued throughout the year. The maximum summer rates were 1200 mg CO₂ m⁻² hr⁻¹ and the maximum winter rates 180 mg CO₂ m⁻² hr⁻¹. These rates coincided with highest and lowest yearly soil temperatures.

2.3 STUDY SITES

Gillam 56° 22' N, 94° 37' W.

Field measurements of soil respiration were conducted on peat plateaus on the Radisson-Dorsey line between towers 4 and 5 (Gillam 2), and the Henday-Radisson line between towers 89 and 90 (Henday 1).

At the Gillam study site measurements in the undisturbed forest were carried out in a black spruce stand approximately 20 m to the west of the right-of-way. Vegetation covered over 95% of the peat surface with Ledum groenlandicum, Chamaedaphne calyculata, Rubus chamaemorus, Sphagnum fuscum and Cladina mitis as the dominant understory plants. Vascular plants occupied between 60 and 70% of the peat surface (Figure 15 a). In the cleared right-of-way respiration was measured on a scraped and level part of a plateau near the centre of the right-of-way, where bare ground was over 75%. Polytrichum strictum was dominant in the study plot and covered approximately 20% of the peat surface. Also present were few scattered plants of Rubus chamaemorus, Ledum groenlandicum and Epilobium angustifolium (Figure 15 b).

The Henday site consisted of two separated peat plateaus approximately 50 meters apart (Figure 2 a). Both the plateaus were of similar height above the surrounding fen (1-1.5 m) and of comparable structure. The undisturbed pla-

teau had negligible tree cover, but dead trunks and charred remains of trees indicated that the site had burned and not recovered. Vegetation cover on the plateau was near 85% of the peat surface with Ledum groenlandicum, Vaccinium vitis-idaea as the dominant plants. Vascular plant cover was over 75%.

The right-of-way site was located on a scraped and level plateau with about 80% bare ground. Within the measurement area Polytrichum strictum and Ceratodon purpureus were dominant with nearly 20% cover, but it also had scattered plants of Rubus chamaemorus.

At both of these study sites, relatively undecomposed Sphagnum peat (von Post 1-2) was common in the upper part of the peat profile, becoming darker and more humified and mixed with sedge-remains underneath (Appendix B). Cores drilled through the ice at both sites showed peat thickness of 70-110 cm. The peat was underlain by fine textured clay.

Respiration was measured three times at each site between June 29, and September 18, 1983.

Minago River 54° 12' N, 99° 09' W

The site is on the Radisson-Dorsey line between towers 840 and 841 (Minago River 1).

The undisturbed forest site consisted of a black spruce stand approximately 30 meters to the west of the right-of-

way. Understorey vegetation cover was complete and the most characteristic plants were Ledum groenlandicum, Chamaedaphne calyculata, Vaccinium vitis-idaea, Sphagnum fuscum and Cladina mitis. Vascular plant cover was estimated at 60%.

The right-of-way study site was located approximately 20 meters from the forest in the western part of the right-of-way. Vegetation cover was near 50% of the peat surface. The dominant plants were Ledum groenlandicum, Vaccinium vitis-idaea, Polytrichum strictum and Cladonia spp. Vascular plant cover was estimated near 30%.

The upper part of the peat profile consisted of light brown to brown Sphagnum peat (von Post 1-4) with slightly more humified peat in the forest. Peat thickness varied from 55-70 cm, and silty clay underlay the peat.

At the Minago River site respiration was monitored four times between June 9 and October 13, 1983.

Devil's Lake 52° 21' N, 98° 53' W

The site is located between towers 1298 and 1299 on the Radisson-Dorsey line (Devil's Lake 2).

The forested site was in a black spruce stand about 30 meters to the east of the right-of-way. Vegetation covered the ground completely in the study plot. The dominant understorey plants were Ledum groenlandicum, Empetrum nigrum, Juniperus horizontalis, Sphagnum fuscum, Pleurozium schreberi and Cladina mitis, with vascular plant cover near 70%.

The right-of-way site was located 30 meters east from the forest, where bare ground was near 25% by cover. The dominant plants were Ledum groenlandicum, Vaccinium vitis-idaea, Carex aquatilis, Sphagnum fuscum, Polytrichum strictum and Cladina mitis. Vascular plant cover was estimated at 30% (Figure 15 c).

Sphagnum peat dominated the upper part of the peat profile, and was of slight to medium humification (von Post 1-5) with the fresh peat on top. Peat thickness was measured 40-70 cm in the forest but 100-115 cm in the right-of-way. In both parts a coarse mineral aggregate (glacial drift) was found underneath the peat.

At the Devil's Lake site respiration was monitored between May 10 and October 13 in 1983 on seven occasions and between May 23 and October 15 in 1984.

The herbicide Tordon 101 (picloram + 2,4-D) had been applied in the right-of-way at the site during the summer of 1979, and the bare ground in the right-of-way was attributed to the application, which had partly killed off Sphagnum fuscum hummocks. The right-of-way was again treated with a herbicide in early June of 1984. Tordon 10K (picloram) was at that time applied at the site.

Hadashville 49° 45'N, 95° 56' W

The site is in an extensive raised bog which has been drained and is being harvested commercially by Fison Western Peat for Sphagnum horticultural peat. Respiration was measured under three different conditions: in a relatively undisturbed part of the bog (bog plot); by a drainage ditch (drained plot); and in a completely unvegetated area (bare plot). These conditions were all found in the same general area east of the peat harvesting field (Figure 2 b). A drainage ditch divides the peat harvesting field from the study area. Water levels in the ditch during autumn are about 180-220 cm below the general ground surface of the study area.

Bog plot: Respiration was measured in an open part of the bog, with scattered trees of black spruce (Picea mariana), tamarack (Larix laricina) and jack pine (Pinus banksiana). Vegetation cover was nearly complete, with Ledum groenlandicum, Chamaedaphne calyculata and Kalmia polifolia forming a dense shrub layer. In the ground layer Sphagnum fuscum, Polytrichum strictum and Aulacomnium palustre were most prominent (Figure 15 d). The site was ca 40 m east of a drainage ditch by the peat harvesting field.

Drained plot: This site was adjacent to or 2 meters east of the drainage ditch (about 40 meters west of the bog plot), where the water table was very low. The bog vegetation had mostly been replaced by a sparse cover of weeds, the most prominent being Agrostis spp., Cirsium spp., Epilo-

bium spp., Sonchus spp. and Salix spp. Of the bog plants only Vaccinium vitis-idaea and Chamaedaphne calyculata remained at the site. Bare ground was near 30%.

Bare plot: The site is almost 100 meters north of the bog plot and about 50 meters east of the drainage ditch in a path cleared of all vegetation. There were no roots in the soil and all CO₂ production was therefore attributable to decomposition.

Over 1 meter of Sphagnum peat is found in the peat profile at the site, which is relatively undecomposed at the surface (von Post 1-3). Total peat thickness is near 3 meters with silty clay underneath, mixed with shell fragments.

Respiration was measured in the bog and drained plots between May 30 and October 22 in 1983 on seven occasions, and between May 10 and October 19 in 1984 on nine occasions. In the bare plot respiration was only measured in 1984, between May 30 and October 19, on eight occasions.

Figure 15.

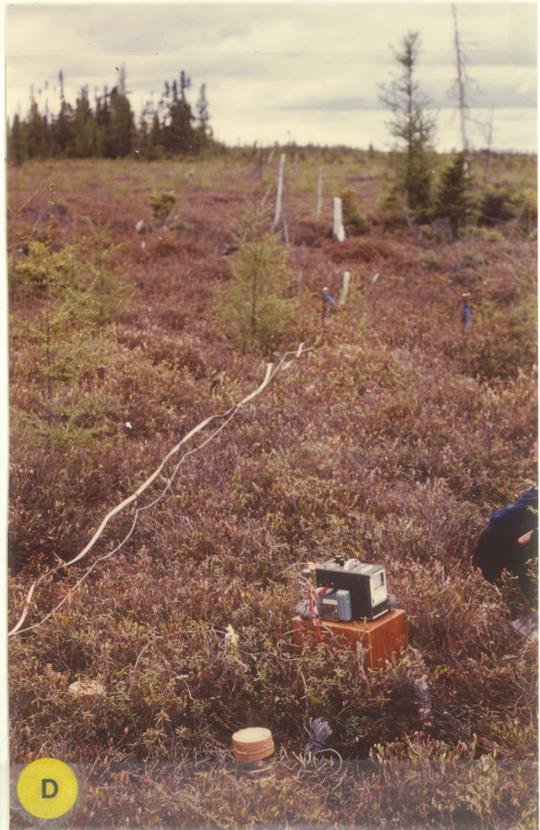
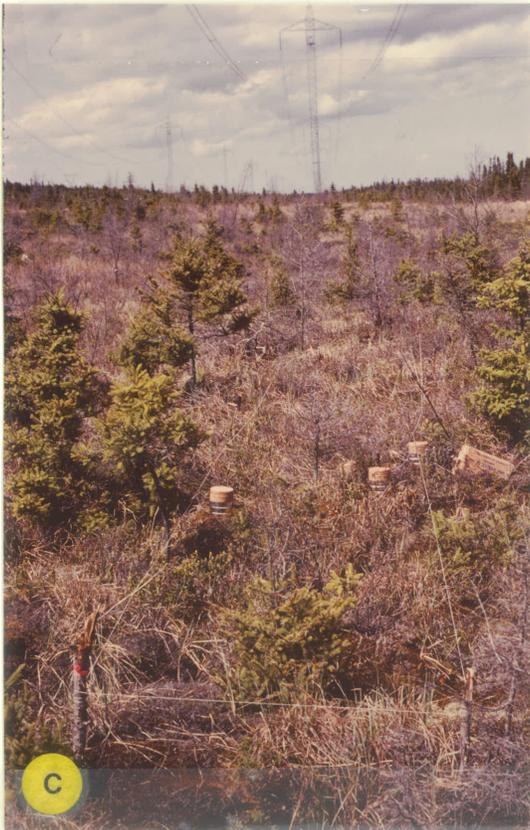
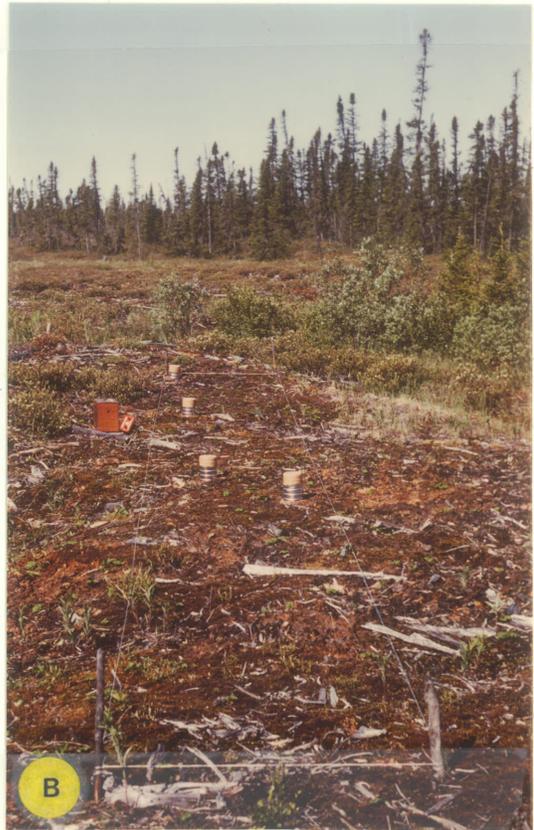
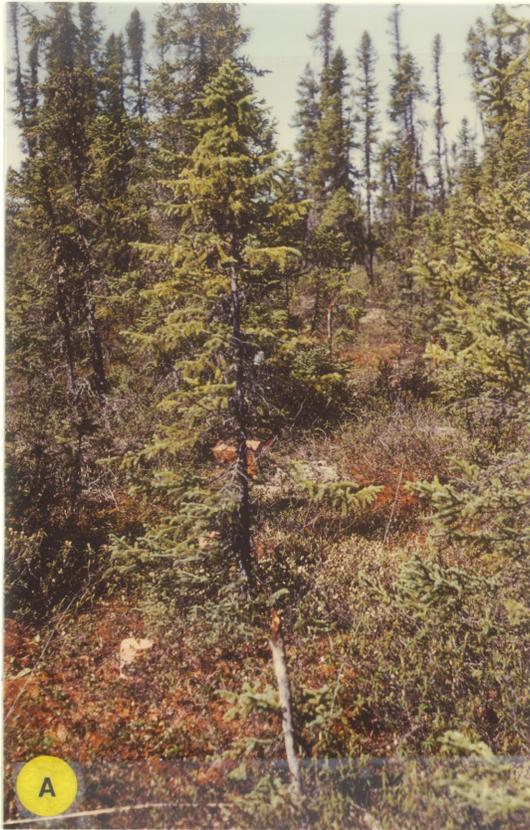
Soil respiration study plots.

A) Gillam 2, forest plot (1 x 10 m) seen from north east. (July 1, 1983).

B) Gillam 2, ROW plot seen from north. Cylinders used for soil respiration measurements are in place. (July 1, 1983).

C) Devil's Lake, ROW plot seen from south. (June 8, 1983).

D) Hadashville, bog plot seen from south. Apparatus on wooden box is Grant thermometer used for recording soil temperature. (May 30, 1983).



2.4 MATERIALS AND METHODS

2.4.1 Soil respiration rate

The permanent study plots were set up in homogenous areas considered representative of the sites. The size of each plot was 10 x 1 m.

Soil respiration was determined by using an alkali absorption method (Gupta and Singh 1977). Open ended metal cylinders 10.6 cm dia x 17 cm ht were inserted 5 cm into the peat, which was first cleared of all green vegetation. Four cylinders were used in each plot, which were only left in the ground while measurements were carried out. New random positions within the plots were chosen at the beginning of each measurement. A wide-mouth glass jar (6.0 cm dia x 8.5 cm ht) containing 50 ml of 0.25N KOH solution was placed into the cylinders for absorption of CO₂. A wire was placed inside the cylinders which kept the jars about 2 cm above the soil surface so they did not interfere with CO₂ evolution from the soil. The cylinders were sealed with airtight styrofoam caps and left in the ground for 24 h. (In a few instances the cylinders were left in the ground from 18 - 25 hours). Four cylinders open at one end, and of the same volume as above the ground in the active cylinders, were left sealed in the field with jars containing the same amount of KOH, to serve as controls.

CO₂ absorption was analyzed by titrating the unneutralized KOH with diluted standardized HCl (approx 0.25N) after precipitation of the carbonate with excessive (15 ml) 3N BaCl and using phenolphthalein as indicator (Strotzky 1965).

The following formula (Strotzky 1965) was used to calculate the amount of CO₂ evolved:

$$\text{mg CO}_2 \text{ or C} = (B - V) N E$$

where: V = volume of acid required to titrate the KOH in the active cylinders to the end point; B = volume of acid to titrate the KOH in the control cylinders to the end point; N = Normality of HCl; E = equivalent weight, E = 22 if the data are expressed as CO₂, but E = 6 if expressed in terms of C. (Therefore: mg C = mg CO₂ 6/22).

The volume and concentration of alkali and size of absorption area used in the present study were based on experience from studies described in the literature (Minderman and Vulto 1973; Gupta and Singh 1977). Due to the extensiveness of the study area and variety of plot types it was not considered practical to carry out tests to check the efficiency of the method used. However before regular sampling started in 1983, a trial run was done at the Hadashville site to test the apparatus. The respiration rates determined in the test were compared to rates given in other studies of soil respiration, which indicated that the method produced results within the expected range.

The absorption area (28.3 cm^{-2}) used was 32% of the total surface area (88.3 cm^{-2}) enclosed and the capacity of the 50 ml of 0.25N KOH was to absorb 275 mg of CO_2 ., which is equivalent to $1274.2 \text{ mg CO}_2 \text{ m}^{-2} \text{ hr}^{-1}$ for a 24 hour period.

2.4.2 Soil temperature, moisture and acrotelm

Soil temperature measurements were taken in three locations at 10 cm depth within each study plot. A Grant thermometer was used to record the temperature every hour from three thermistors inserted within 10 cm of the respiration cylinders. Temperature was only recorded while the cylinders were in the field.

Moisture content of surface peat (0-10 cm) sampled underneath the respiration cylinders at the termination of each measurement, was determined gravimetrically after drying samples at 80° C . Moisture content was calculated as % wet wt.

Depth to water table was monitored in soil pits dug within 2 meters of the respiration plots. Permanent markers were placed in the pits at the surface level from which the water level was measured at the termination of each respiration measurement. At Minago River, Devil's Lake and Hadashville bog, where the ground was uneven due to hummocky topography, three holes were dug by each study plot. Only one hole was dug by the Hadashville drained and bare plots since

they had level topography. Ice was found in the ground at all sites early in the season and at the Gillam sites permafrost was present. Depth to ice or permafrost was measured with a metal probe underneath each respiration cylinder when measurements were terminated.

The term acrotelm has been suggested for the upper layer of peatland soils, which is above the low water table level (Ingram 1978). The use of the term will be expanded here to include the active layer in peatlands with permafrost in them, since both the aerobic and active layers are the zones of active decomposition. In this text the layer of peat soil above the water table or ice at any given time will be referred to as acrotelm.

2.4.3 Statistical techniques

To investigate the influence of soil temperature, moisture and depth of acrotelm on soil respiration, correlations of the environmental variables with soil respiration rate were calculated and regression equations (Steel and Torrie 1980) constructed using two years data. The data were run under the BMDP-package (Dixon and Brown 1979). In the regression equations soil respiration (R) was considered as the dependent variable with temperature (T), acrotelm (A) and moisture (M) as the principal independent variables.

A two-way ANOVA (or GLM, for unbalanced data) was run under SAS (1979) and means were compared with Duncan's multiple range test, to contrast soil respiration rates of forest and right-of-way plots from the same sites and the bog, drained and bare plots at the Hadashville site. Where sufficient data were available the same type of analyses were performed using the environmental variables.

2.4.4 Carbon-budget

To get an estimate of the amount of carbon given off during the periods from earliest to last measurements in the season, the area under the curves showing changes in respiration rate with time was integrated.

No attempts were made to measure respiration rate or any of the environmental parameters at the sites in winter or early spring (November - April). It is therefore not known if conditions in the soil permitted some respiration activity to continue throughout the year. The field data from 1983 and 1984 indicate that in both seasons soil respiration had started prior to the day of earliest measurements and would continue for some time in the fall after the last field measurements. In an attempt to estimate the yearly carbon release at the sites the assumption was made that soil respiration is of insignificant level during the winter months but will start in the spring when the daily mean temperature rises above the 0° and continue until the daily

mean temperature falls again below 0° C. Adjustments were made for the days in the spring and fall when the daily mean fluctuated around the 0° level and had not permanently risen above or fallen below it. These estimated days of thaw and freeze-up when respiration was set at zero were connected by straight lines with respiration rate of earliest and latest field measurements and the area under the curves was integrated as before. Daily meteorological records (Environment Canada, Atmospheric Environmental Service) from the nearest meteorological stations were checked for each site to arrive at the approximate dates.

For Devil's Lake the reference stations were the Grand Rapids Hydro station ($53^{\circ} 09' N$, $99^{\circ} 17' W$), approximately 100 km north of the study site, and the Ashern station ($51^{\circ} 10' N$, $98^{\circ} 20' W$), which is 130 km SSE of the site.

For Minago River records from the Grand Rapids Hydro station, 110 km south of the study site, were used again. Records from Thompson ($55^{\circ} 45' N$, $97^{\circ} 47' W$), 190 km to the NNE of the site were also checked.

For the sites in the Gillam area, records from the Gillam Airport station, approximately 4 km to the west of the study sites, were checked.

For Hadashville records from stations at Elma ($49^{\circ} 53' N$, $95^{\circ} 54' W$) and Pinawa ($50^{\circ} 11' N$, $96^{\circ} 03'$) were checked. The Elma station is about 11 km NNE of the study site but

Pinawa is about 42 km NNE of the site. The Pinawa records were checked as the Elma records were not complete.

At Gillam soil respiration was not measured from early July to September which made it difficult to establish a carbon-budget for the sites there. The maximum respiration rates from the southern sites as well as the environmental parameters and their change with latitude were used as a basis to estimate the respiration rates at Gillam in late summer. The meteorological records for Gillam in 1983 showed that the daily mean air temperature started to decline by the middle of August. It was assumed that soil respiration would be at highest level at that time and would decrease with falling air temperature. Because of the assumptions involved, a carbon-budget was only compiled for the Gillam 2 site.

2.5 RESULTS

2.5.1 Soil respiration and environmental parameters

Results are presented in Figure 16 to Figure 19 and summarized in Table 11 and Table 12. (See also Appendix D).

2.5.1.1 Devil's Lake

Soil respiration rate. Soil respiration was measured for nearly five months (May-October) during 1983 and 1984. The average soil respiration rate ranged from 148.7 to 595.0 mg CO₂ m⁻² hr⁻¹. The changes in respiration rates with the season were fairly similar in the two years (Figure 16). In 1983 the measurements were started two weeks earlier than in 1984 and the sampling frequency was higher. Therefore the data from 1983 is assumed to give a more representative picture of the real seasonal changes than the 1984 data, which also is obscured by the herbicidal application at the site during 1984.

Comparison of the peaks in respiration rates for both the forest and right-of-way plots reveals that the maximum rates were considerably higher in 1984 than 1983, the differences being 39% for the forest and 30% for the right-of-way. The highest respiration occurred following the herbicidal application in June of 1984, which is believed to have caused the rates to become abnormally high.

TABLE 11

Soil respiration - summary table

Ranges in soil respiration ($\text{mg CO}_2 \text{ m}^{-2} \text{ hr}^{-1}$), soil temperature ($^{\circ} \text{C}$ at 10 cm), acrotelm thickness (cm), soil moisture (% wet wt, 0 - 10 cm) measured at four bog sites in Manitoba.

SITE	YEAR	RESPIRATION	TEMPERATURE	ACROTELM	MOISTURE
Gillam	'83	153.1-246.9	3.9-10.0	21.7- 52.5	72.3-88.4
Minago R	'83	171.3-405.0	3.6-17.4	19.7- 57.0	77.5-87.8
Devils L	'83-4	148.7-595.0	1.5-20.1	11.4- 56.3	79.4-90.0
Hadashv. ¹ :					
- bog	'83-4	138.4-529.0	3.4-20.8	20.0-107.0	74.4-86.4
-drained	'83-4	252.2-945.2	5.3-21.0	44.0-155.0	45.3-76.1
-bare	'84	112.6-596.6	3.7-18.6	15.0- 66.0	81.5-86.4

¹ Due to great differences in environmental conditions between the the bog and drained plots at the Hadashville site and the shorter measurement period in the bare plot, the data are given separately for each plot.

TABLE 12

Soil respiration - within site contrasts

Average seasonal soil respiration rate ($\text{mg CO}_2 \text{ m}^{-2} \text{ hr}^{-1}$), soil temperature ($^{\circ} \text{C}$), acrotelm thickness (cm) and soil moisture (% wet wt) measured within each study plot at four bog sites in Manitoba.

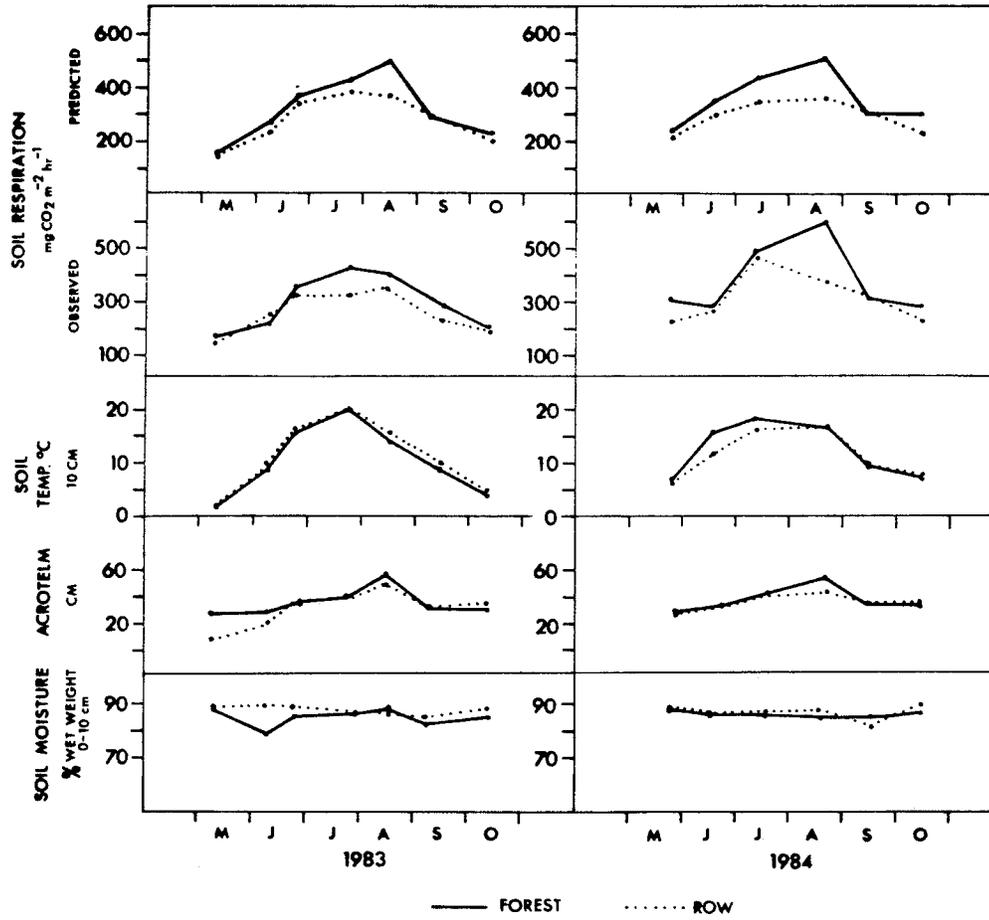
<u>DEVIL'S LAKE</u>	1983 (N=7)			1984 (N=6)		
	FOR	ROW	ROW%FOR	FOR	ROW	ROW%FOR
Respiration	301	266 *	88	377	318 *	84
Temperature	10.7	11.4 *		12.7	11.3 *	
Acrotelm	36.3	32.4 *		38.3	35.9 *	
Moisture	85.1	87.8 *		86.5	87.6	
<u>MINAGO RIVER</u>	1983 (N=4)					
	FOR	ROW	ROW%FOR			
Respiration	275	271	99			
Temperature	11.9	11.8				
Acrotelm	44.5	42.8				
Moisture	84.2	81.3				
<u>GILLAM</u> 1983	GILLAM 2 (N=3)			HENDAY 1 (N=3)		
	FOR	ROW	ROW%FOR	FOR	ROW	ROW%FOR
Respiration	220	198	90	202	188	93
Temperature	7.7	8.2		4.5	6.5 *	
Acrotelm	37.1	32.2 *		33.0	32.6	
Moisture	86.7	82.8 *		79.7	83.9 *	
<u>HADASHVILLE</u>	1983 (N=7)			1984 (N=9)		
	BOG	DRAINED	DRA%BOG	BOG	DRAINED	DRA%BOG
Respiration	300	651 *	217	314	635 *	202
Temperature	10.6	14.2 *		10.3	12.7 *	
Acrotelm	60.4	147.9 N.D.		66.1	132.2 N.D.	
Moisture	82.2	68.1 *		79.9	62.3 *	

* Difference is significant at $\alpha = 0.05$.

ROW%FOR: ROW respiration calculated as % of Forest respiration.
DRA%BOG: Respiration in drained plot calculated as % of respiration in bog plot.

Figure 16. Soil respiration at Devil's Lake site.

Seasonal trends in soil respiration rate in forest and right-of-way at Devil's Lake site in relation to soil temperature, acrotelm thickness and soil moisture, during the 1983 and 1984 seasons.



Soil respiration rate was higher in the forest plot than in the right-of-way (Figure 16). During 1983 the rate in the forest was higher in six of the seven measurements. The average respiration rate in the forest and right-of-way was 300.8 and 265.9 mg CO₂ m⁻² hr⁻¹ respectively. The respiration rate in the right-of-way was therefore 88% of that in the forest, but it ranged from 78 to 106% throughout the sampling period. The average respiration rates during 1984 in the forest and right-of-way were 377.2 and 319.2 mg CO₂ m⁻² hr⁻¹ respectively, and the right-of-way respiration rate averaged 84% of the forest rate (range 62 to 106%).

The ANOVA test showed that the respiration rate in the forest was significantly higher than in the right-of-way during both years. (For 1983: F=14.12; Pr > F=0.0005; Df=1. For 1984: F=13.94; Pr > F=0.0007; Df=1). Changes in respiration rates with time were also significant (For 1983: F=17.57; Pr > F=0.0001; Df=6. For 1984: F=28.90; Pr > F=0.0001; Df=5). For the 1983 data the plot*time effect was not significant (F=1.65; Pr > F=0.1587; Df=6) which means that changes in respiration rate in the two plots were relatively synchronous as the season progressed (Figure 16). For the 1984 data the plot*time effect was significant (F=5.05; Pr > F=0.0013; Df=5) which most likely is due to the Tordon 10K application and the difference in time of it's action in the right-of-way and forest.

Environmental parameters. The environmental conditions and seasonal changes in the forest and right-of-way plots are very similar with respect to soil temperature, acrotelm and soil moisture (Figure 16).

On the five days in 1983 for which there are separate records of soil temperature for the forest and right-of-way there are very little differences between the values for the two plots. A GLM test did give significant differences between forest and right-of-way based on the five days data ($F=2.27$; $Pr > F=0.15$; $Df=1$), but the average soil temperature was $11.4^{\circ} C$ in the right-of-way and $10.7^{\circ} C$ in the forest. The temperature in the right-of-way plot tended to be marginally higher than in the forest plot, but the largest difference was only $1.5^{\circ} C$. (Due to the similarity in temperature values between the two plots, the temperature of the right-of-way plot was also as used in regression calculations to represent the forest plot on the two dates from which temperature data is missing).

The temperature trends were fairly similar during 1984, but separate records are available for the forest and right-of-way plots for each day of respiration measurements. The temperature in the forest plot was higher than in the right-of-way plot early in the season, the greatest difference being $4.0^{\circ} C$ during the June measurement. The GLM test gave significant differences in soil temperature between the forest and right-of-way ($F=10.9$; $Pr > F=0.0035$; $Df=1$). The av-

erage temperature in the forest was 12.7° C, but 11.3° C in the right-of-way, (Table 12).

Ice was encountered in both the forest and right-of-way plots when the respiration was measured during May of 1983. Ice still persisted in the right-of-way plot during June but it had almost disappeared in the forest plot. The acrotelm is therefore given as depth to the ice in May for both plots and for the right-of-way plot in June, but otherwise as depth to the water table (Figure 16). After the ice had thawed in the right-of-way there was very little difference in thickness of the acrotelm between the forest and right-of-way plot. The GLM-analysis gave significant differences between the forest and right-of-way ($F=14.9$; $Pr > F=0.0006$; $Df=1$). The average for the forest was 36.3 cm and 32.4 cm for the right-of-way. The largest difference was of 5 cm in August. In 1984 ice had thawed in the plots when the site was first visited in May. The water level in the forest which tended to be lower than in the right-of-way was with the exception of the August measurement never more than 4 cm below the right-of-way level. The GLM analysis again gave significant differences between forest and right-of-way ($F=5.80$; $Pr > F=0.0241$; $Df=1$). The averages were 38.3 cm for the forest and 35.9 cm for the right-of-way.

Soil moisture was the environmental variable which showed the least seasonal changes. During 1983 the moisture in the forest ranged from 79.4 to 88.6% with an average value of

85.1%. In the right-of-way the moisture level was slightly higher, ranging from 86.6 to 90% and averaging 87.8%. The GLM test showed that the differences were significant ($F=11.59$; $Pr > F=0.0016$; $Df=1$). During 1984 the moisture in the forest ranged from 84.6 to 88.0%, averaging 86.5%, but in the right-of-way the values were 83.6 to 89.6% and the average 87.6%. The GLM test did not give a significant difference between forest and right-of-way ($F=2.09$; $Pr > F=0.1571$; $Df=1$).

2.5.1.2 Minago River

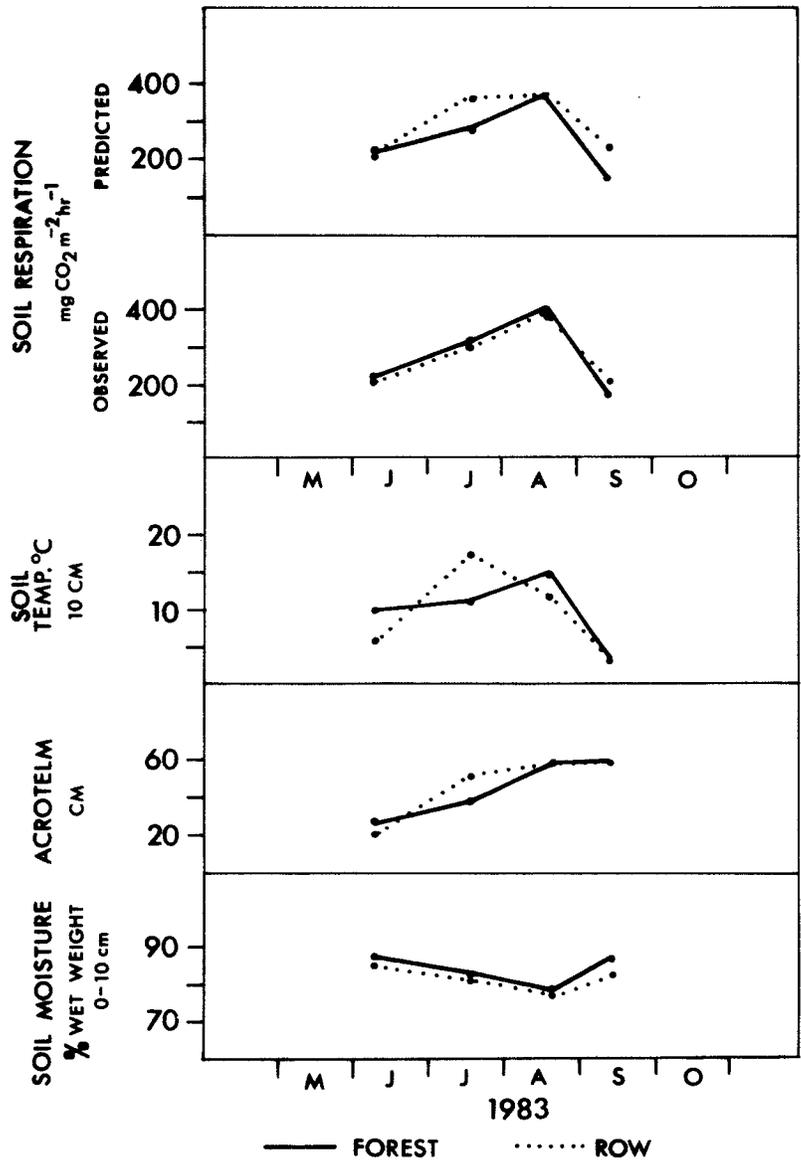
Soil respiration rate. Soil respiration was measured in the field on four occasions over a four-month period (June - October) during 1983. Average respiration rates ranged from 171.3 - 405.1 $\text{mg CO}_2 \text{ m}^{-2} \text{ hr}^{-1}$. The changes in respiration rate with the season are shown in Figure 17. Due to the infrequency of measurements the curves have to be considered as approximations.

On three of the four sampling dates the respiration rate was slightly higher in the forest than the right-of-way. The average respiration rate in the forest and right-of-way was 274.8 and 270.9 $\text{mg CO}_2 \text{ m}^{-2} \text{ hr}^{-1}$ respectively. The respiration rate in the right-of-way plot therefore averaged 99% of that of the forest rate, with a range of 90 to 118%.

The ANOVA test contrasting the forest and right-of-way did not show significant differences in respiration rate

Figure 17. Soil respiration at Minago River site.

Seasonal trends in soil respiration rate in forest and right-of-way at Minago River site in relation to soil temperature, acrotelm thickness and soil moisture during the 1983 season.



($F=0.00$; $Pr > F=0.9518$; $Df=1$). Changes in respiration rate with time were significant ($F=42.84$; $Pr > F=0.0001$; $Df=3$), and changes in respiration rate in the forest and right-of-way plots were synchronous in time and the plot*time effect was not significant ($F=0.65$; $Pr > F=0.59$; $Df=3$) (Figure 17).

Environmental parameters. The trends in the environmental parameters are shown in Figure 17. Soil temperature was not significantly different between the forest and right-of-way ($F=0.00$; $Pr > F=0.9558$; $Df=1$), with average temperature values of 11.9° C in the forest and 11.8° C in the right-of-way.

Ice was in the ground during early June in the forest and right-of-way but it had thawed by mid-July and the acrotelm was measured as depth to water table. During August and October the water levels in both forest and right-of-way had receded below the peat and into the underlying clay. The acrotelm was then given as depth to the peat/clay transition. A GLM test did not give significant differences between the forest and right-of-way ($F=0.43$; $Pr > F=0.5208$; $Df=1$), but the averages were 42.8 cm for the forest and 44.5 cm for the right-of-way.

The soil moisture in the forest plot was higher than in the right-of-way plot. The average soil moisture in the forest was 84.2% (range: 78.8-87.8%) but 81.3% in the right-of-way (range: 77.5- 85.1%).

2.5.1.3 Gillam

Soil respiration and environmental parameters. The soil respiration data from Gillam was limited to three measurements done at each site, two of which were within the same week.

At the Gillam 2 site respiration was measured on July 1 and 2, and September 18. The respiration rate on the two consecutive days was of the same order or 246.9 and 242.9 mg CO₂ m⁻² hr⁻¹ in the forest and 216.3 and 224.9 mg CO₂ m⁻² hr⁻¹ in the right-of-way. Soil temperature on those days was 8.7 and 9.4° C in the forest and 7.4 and 9.7° C in the right-of-way. Depth to permafrost was 33.5 and 28.2 cm in the forest and 24.0 and 25.2 cm in the right-of-way; moisture content was 86.7 and 85.2% in the forest and 81.7 and 79.4% in the right-of-way. On September 18 the respiration rate was considerably lower or 171.5 and 153.1 mg CO₂ m⁻² hr⁻¹ in the forest and right-of-way respectively. Soil temperature was then 4.9 and 7.6° C; depth to permafrost 49.5 and 47.3 cm; and moisture 88.4 and 87.4%, in the forest and right-of-way respectively. Soil temperature was not significantly different between forest and right-of-way (F=0.38; Pr > F=0.5507; Df=1) but the averages were 7.7° in the forest and 8.2° in the right-of-way. The acrotelm was however significantly deeper in the forest than the right-of-way (F=4.92; Pr > F=0.0396; Df=1). The averages were 37.1 cm in the forest and 32.2 cm in the right-of-way.

Moisture was significantly higher in the forest than the right-of-way ($F=8.88$; $Pr > F=0.0080$; $Df=1$) and the averages were 86.7% in the forest and 82.8% in the right-of-way.

At the Henday 1 site respiration was measured on June 29, July 3 and September 16. On the first two days the respiration rate was 211.8 and 216.8 $\text{mg CO}_2 \text{ m}^{-2} \text{ hr}^{-1}$ in the "forest" and 183.2 and 208.8 $\text{mg CO}_2 \text{ m}^{-2} \text{ hr}^{-1}$ in the right-of-way. Soil temperature is not available from the forest for the earlier date but it was 5°C on the later date. In the right-of-way soil temperatures were 10.0 and 6.7°C during the two days respectively. Depth to permafrost was 21.7 and 24.7 cm in the forest and 22.7 and 24.2 cm in the right-of-way, while moisture was 78.3 and 78.4% in the forest and 82.2 and 82.4% in the right-of-way. On September 16, the respiration rate had fallen to 176.5 and 172.6 $\text{mg CO}_2 \text{ m}^{-2} \text{ hr}^{-1}$ in the forest and right-of-way while temperature, depth to permafrost and moisture in the forest and right-of-way were 3.9 and 6.3°C , 52.5 and 50.8 cm and 82.5 and 87.3%. ANOVA gave significant differences ($F=144.12$; $Pr > F=0.0001$; $Df= 1$) between forest and right-of-way with respect to temperature on the two dates records were available, and the averages were 4.2°C in the forest and 6.5° in the right-of-way. Difference in depth of acrotelm was not significant ($F=0.23$; $Pr > F=0.6357$; $Df=1$) with the averages 33.0 cm for the forest and 32.6 cm for the right-of-way. Difference in soil moisture was however significant ($F=5.08$; $Pr > F=$

0.370; Df=1), the averages were 79.7% in the forest and 83.9% in the right-of-way.

At both sites the respiration was higher in the forest than in the right-of-way. The averages were 220.5 and 198.1 mg CO₂ m⁻² hr⁻¹ in the forest and right-of-way respectively, at the Gillam 2 site. The respiration in the right-of-way averaged 90% of the forest rate ranging from 88 to 93%. An ANOVA test did not give significant difference between forest and right-of-way with respect to respiration rate (F=3.22; Pr > F=0.09; Df=1), the the number of observations was limited to three. The respiration rates were however significant with respect to time (F=14.18; Pr > F=0.0002; Df=2), but the plot*time effect was not significant (F=0.11; Pr > F=0.90; Df=2).

At the Henday 1 site the average respiration rates were 201.7 and 188.2 mg CO₂ m⁻² hr⁻¹ in the forest and right-of-way. Respiration in the right-of-way averaged 93% of the forest rate (range 86 to 98%). There was no significant difference in respiration rate between forest and right-of-way (F=1.25; Pr > F=0.28; Df=1) and neither with respect to time (F=0.81; Pr > F=0.46; Df=2) nor plot*time (F=1.47; Pr > F=0.26; Df=2)

2.5.1.4 Hadashville

Soil respiration rate: At the Hadashville site field measurements of soil respiration covered nearly five months during both 1983 and 1984 (May - October) (Figure 18 and 19).

In the bog plot, soil respiration ranged from 138.4 to 529.0 mg CO₂ m⁻² hr⁻¹, 252.2 - 945.2 mg CO₂ m⁻¹ hr⁻¹ in the drained plot and 112.6 - 596.6 mg CO₂ m⁻² hr⁻¹ in the bare plot (Table 11).

The soil respiration rate in the drained plot was always considerably higher than in the relatively undisturbed bog plot (Figure 18). In 1983 the respiration rate in the drained plot averaged 217% (range 168-274%) of the rate in the bog plot. The average respiration rate was 300.0 and 651.2 mg CO₂ m⁻² hr⁻¹ in the bog and drained plots respectively (Table 12). During 1984 the respiration rate in the drained plot was on the average 202% (range 142-369%) of that in the bog plot. The average respiration rates were 314.3 and 635.7 mg CO₂ m⁻² hr⁻¹ respectively.

The ANOVA test gave significantly higher respiration rates in the drained plot than in the bog plot in both years (For 1983: F=266.1; Pr > F=0.0001; Df=1. For 1984: F=232.2; Pr > F= 0.0001; Df=1). The respiration rates were also significantly different with respect to time in both years (For 1983: F=48.4; Pr > F=0.00021; Df=6. For 1984: F=18.7; Pr > F=0.0001; Df=8) and the plot*time effect was also significant in both years (For 1983: F=7.8; Pr > F=0.0001; Df=6. For 1984: F=6.3; Pr > F=0.0001; Df=8).

The respiration rate in the bare plot at Hadashville averaged 111% (range 62-178%) of the rate in the bog plot.

Figure 18. Soil respiration at Hadashville site, bog and drained plots.

Seasonal trends in soil respiration rate in bog plot and drained plot at Hadashville site in relation to soil temperature, acrotelm thickness and soil moisture, during the 1983 and 1984 seasons.

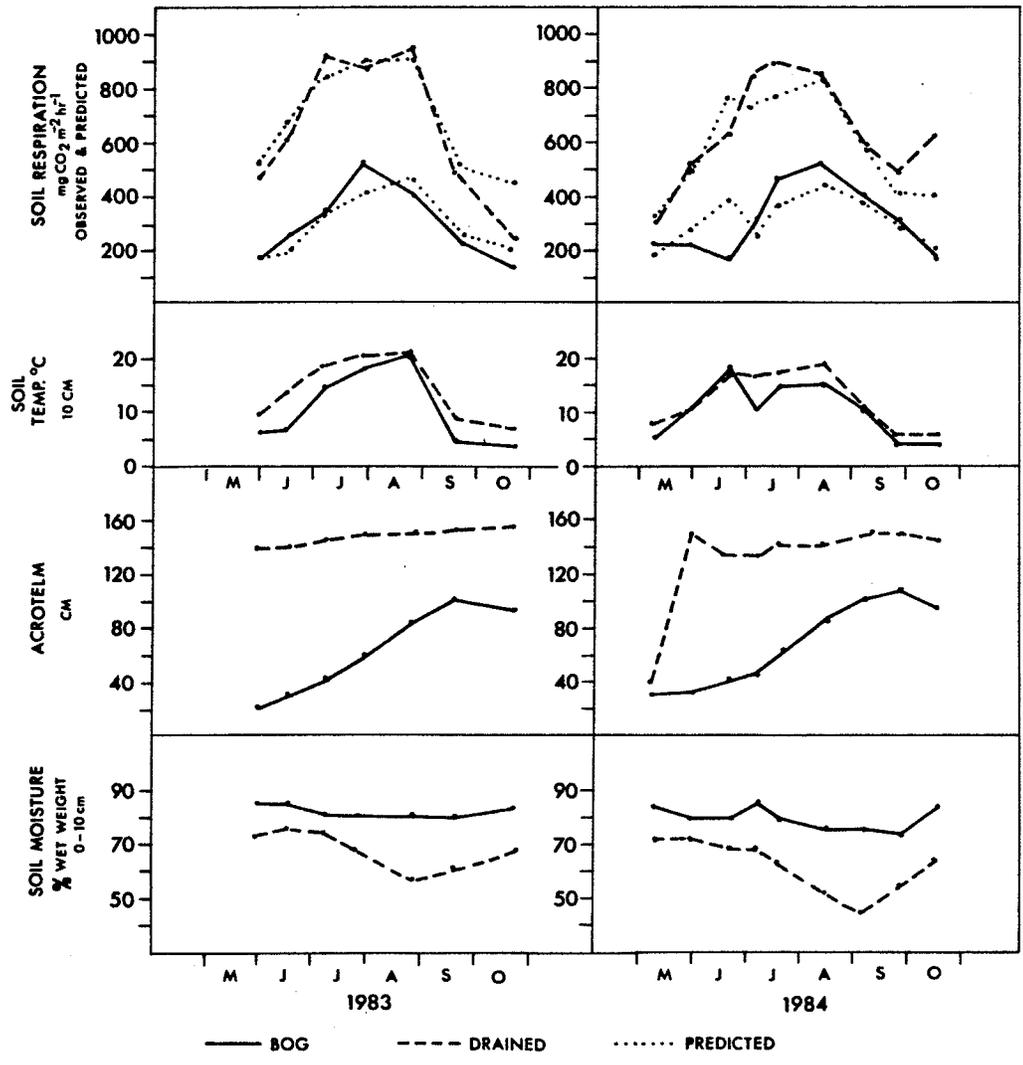
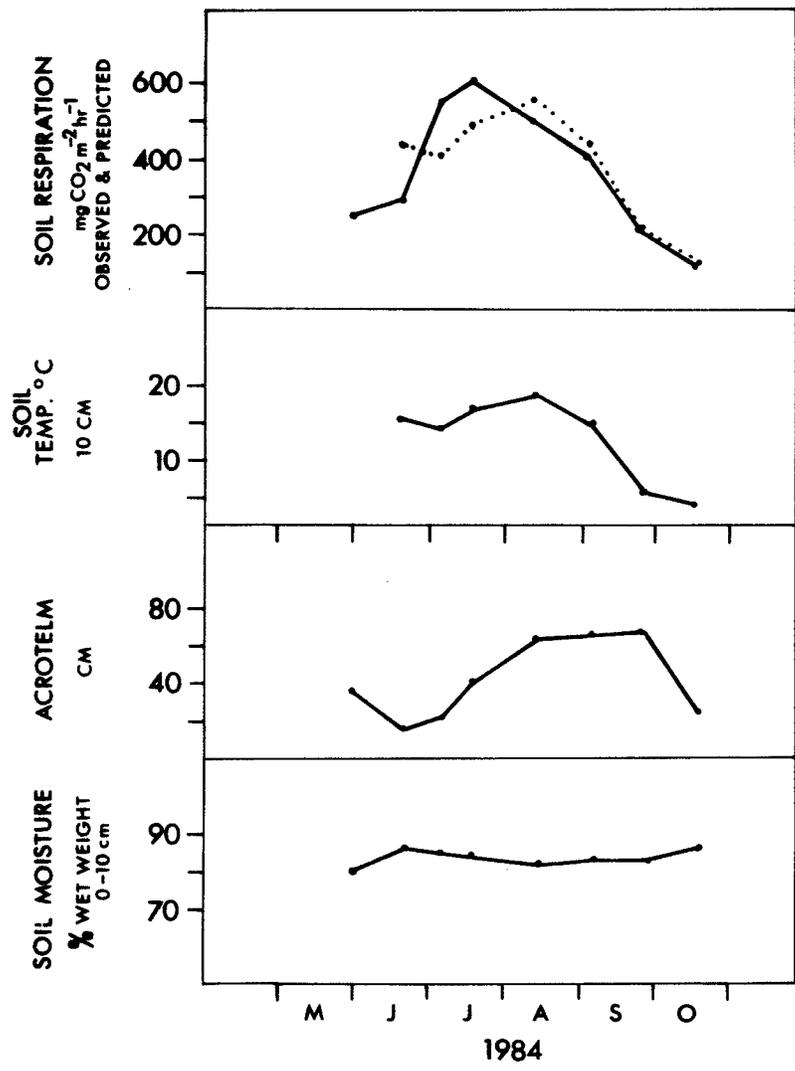


Figure 19.

Soil respiration at Hadashville site, bare plot.

Seasonal trends in soil respiration rate in bare plot at Hadashville site in relation to soil temperature, acrotelm thickness and soil moisture, during the 1984 season.

———— Measured
..... Predicted



The average respiration rates were 339.9 and 376.4 mg CO₂ m⁻² hr⁻¹ in the bog and bare plots respectively. The rate in the bare plot was higher in the first four measurements (May 30, to July 19), but was lower in the measurements later in the summer. The ANOVA test did not give significant differences in respiration rate between the plots for the entire season (F=3.81; PR > F=0.06; Df=1). The respiration rates were however significant with respect to time (F=21.94; Pr > F=0.0001; Df=7) and the plot*time effects was also significant (F=4.36; Pr > F=0.0008; Df=7).

Environmental parameters. The seasonal changes in soil temperature, acrotelm and soil moisture in the study plots at the Hadashville site are shown in Figure 18 and Figure 19.

Soil temperature was always higher in the drained plot than in the bog plot in 1983. An ANOVA test showed that the temperature was significantly higher than in the bog plot (F=101.98; Pr > F=0.0001; Df=1), and the averages were 14.2° C and 10.6° C.

In 1984 there was a less difference in temperature between the two plots than in 1983. The temperature was again significantly higher in the drained plot than in the bog plot (F=56.25; Pr > F=0.0001; Df=1), and the averages were 12.7° C and 10.3° C (Table 12).

The decline in soil temperature in early July was due to heavy rainfall at the site in June of 1984 which brought the water table up above the ground ice. Contact of groundwater with ice has presumably caused cooling of the groundwater and the soil. In the bare plot it was first possible to measure soil temperature on June 21. With the exception of the first and last measurement the soil temperature in the bare plot is higher than in the bog plot, the difference being from 1.7 to 3.5° C.

Ice was present in the bog plot at the end of May 1983 and it persisted into July, thawing by July 27. No ice was found in the drained plot during 1983. The water table was very low in the plot throughout the season and changed relatively little.

During 1984 there was ice in the bog plot in May and some ice on June 21. By that time the water table had risen above the ice due to heavy rains in June. The acrotelm is therefore given as depth to ice in May but as depth to water table from June 21. There was ice in the drained plot on the first day of measurements in 1984 but it had thawed completely on May 30.

When respiration was first measured in the bare plot during 1984 (May 30) ice was present. On June 21, some ice was encountered in parts of the plot, but the water level had risen above the ice.

Soil moisture in the bog plot during 1983 ranged from 79.2 to 85.8% with average of 82.2%. In the drained plot the soil moisture was considerably lower than in the bog plot and showed greater seasonal changes. The average soil moisture in the plot during 1983 was 68.1%. Difference in moisture level was significant between the plots ($F=166.54$; $Pr > F=0.0001$). During 1984 the soil moisture was on the average lower than in 1983. In the bog plot the average soil moisture for the season was 79.9%, (range 74.4 to 85.4%). The average soil moisture in the plot for the season was 62.3% (range 45.3 to 72.4%). Difference in moisture level was again highly significant ($F=420.0$; $Pr > F=0.0001$; $Df=1$). In the bare plot soil moisture showed relatively little seasonal trends in 1984 and the moisture level was high throughout the season. The moisture ranged from 79.9 to 86.4%, the average being 83.5%.

2.5.2 Soil respiration vs. environmental parameters

To assess the effects of soil temperature, acrotelm and soil moisture on soil respiration rate the simple correlations were evaluated and where sufficient data was available simple and multiple linear regression equations were calculated and evaluated (Table 13 and 14).

Of the three environmental variables, soil temperature has generally the strongest and most often significant correlation with soil respiration rate. With the exception of

TABLE 13

Soil respiration - environmental variable correlations

Correlations (r) of soil respiration rate with environmental variables. Variation (%) in soil respiration rate explained by single variables in simple linear regression equations is given in brackets.

SITE, PLOT	N	TEMP.	ACROTELM	MOISTURE
Devils L. For	13	0.79* (62.5)	0.83* (68.4)	0.07 (0.5)
" " ROW	13	0.83* (68.4)	0.74* (55.0)	-0.40 (15.6)
" " Tot.	26	0.78* (61.0)	0.78* (60.7)	-0.18 (3.3)
Minago R. For	4	0.93 (N.D.)	0.24 (N.D.)	-0.96* (N.D.)
" " ROW	4	0.68 (N.D.)	0.52 (N.D.)	-0.87 (N.D.)
" " Tot.	8	0.77 (59.5)	0.38 (14.5)	-0.83* (68.3)
Gillam Total	12	0.45 (N.D.)	-0.68* (N.D.)	-0.32 (N.D.)
Hadashv. Bog	16	0.66* (43.9)	0.21 (4.3)	-0.51* (25.9)
" Drain.	16	0.85* (72.3)	0.31 (9.8)	-0.10 (1.0)
" Bare	8	0.82* (68.0)	0.05 (0.2)	-0.25 (6.1)
" Total	40	0.69* (46.9)	0.62* (38.5)	-0.60* (35.6)
All data comb.	86	0.67* (45.1)	0.70* (48.4)	-0.63* (39.2)

* r is significant at $\alpha = 0.05$

TABLE 14

Statistical analysis of multiple linear regression models relating soil respiration (R) to soil temperature (T), acrotelm thickness (A), and soil moisture (M) at four peatland sites in Manitoba.

SITE	N	REGRESSION EQUATION ^a	R ²	Stand. err. of estim.	Stand. part. reg. coeff. ^b		
					T	A	M
<u>Devil's Lake:</u>							
- forest	13 (1)	R = -57.22 + 9.33 T + 6.91 A + 0.35 M	0.806 ^{***}	60.55	0.45	0.54	0.01
- forest	13 (2)	R = -28.14 + 9.29 T + 6.94 A	0.806 ^{***}	57.45	<u>0.45</u>	<u>0.55</u>	
- ROW	13 (3)	R = 953.71 + 10.05 T + 1.35 A - 9.34 M	0.739 ^{***}	49.70	<u>0.66</u>	<u>0.16</u>	-0.18
- for + ROW	26 (4)	R = 301.47 + 8.61 T + 4.91 A - 3.01 M	0.723 ^{***}	58.19	<u>0.46</u>	<u>0.45</u>	-0.07
<u>Minago River:</u>							
- for + ROW	8 (5)	R = 1444.36 + 7.89 T - 0.38 A - 14.88 M	0.841 [*]	47.10	0.45	-0.07	-0.63
- for + ROW	8 (6)	R = 1331.56 + 8.16 T	0.838 [*]	42.47	<u>0.47</u>		<u>-0.58</u>
<u>Hadashville:</u>							
- bog	16 (7)	R = 710.02 + 14.10 T + 0.84 A - 7.45 M	0.567 [*]	94.83	0.64	0.20	-0.21
- drained	16 (8)	R = 96.50 + 32.41 T + 1.34 A - 1.05 M	0.754 ^{***}	120.71	<u>0.82</u>	0.16	-0.04
- drained	16 (9)	R = 14.80 + 32.36 T + 1.44 A	0.752 ^{***}	116.39	<u>0.82</u>	0.17	
- bare	8 (10)	R = 9571.80 + 20.80 T - 8.83 A - 108.27 M	0.757 ^{NS}	128.46	0.64	-1.07	-1.13
- bare	8 (11)	R = 593.67 + 26.33 T	0.685 ^{NS}	126.60	<u>0.81</u>		-0.24
- bog + dr. + bare	40 (12)	R = 211.01 + 25.19 T + 2.06 A - 3.23 M	0.768 ^{***}	118.26	<u>0.61</u>	<u>0.43</u>	-0.14
<u>Dev. L.+ Min.R.</u>							
<u>Hadashv.+Gill.</u>	86 (13)	R = 274.84 + 19.84 + 2.05 A - 3.22 M	0.777 ^{***}	94.47	<u>0.54</u>	<u>0.46</u>	-0.15

^aR = Respiration (mg CO₂ m⁻² hr⁻¹); T = Soil temperature at 10 cm (°C); A = Acrotelm thickness (cm); M = Moisture (% w wt).

^bUnderlined standard partial regression coefficients are significant at $\alpha = 0.05$.

* Significant at $\alpha = 0.05$, *** significant at $\alpha = 0.001$, NS = not significant (for R²)

the Gillam data the correlation (r) ranges from 0.66 - 0.85. By itself soil temperature explains 43.9 - 72.3% of the variation in soil respiration rate within single plots. For the combined data, both within sites and for all the data, soil temperature is also significantly correlated with soil respiration rate ($r = 0.67 - 0.78$) and it explains 45.1 - 61.0% of the variation in soil respiration rate. Acrotelm thickness shows generally positive correlation with soil respiration rate. For single plot data the correlation is only significant at the Devil's Lake site ($r = 0.74 - 0.83$) where changes in acrotelm thickness explain 55.0 - 68.4% of the variation in soil respiration rate. For the Hadashville plots the correlation ($r = 0.05 - 0.31$) is low and insignificant. For the combined Devil's Lake data, Hadashville data and the combined data from all the sites the correlations are significant ($r = 0.62 - 0.83$) and 30.5 - 60.7% of the variation in soil respiration rate is explained by variation in acrotelm thickness.

Soil moisture was generally inversely related to soil respiration rate but the correlations are low and insignificant for most single plot data. The relationship is only significant at the Minago River forest ($r = -0.96$) and Hadashville bog plot ($r = -0.51$) where moisture explains 92.2% and 25.9% of the variation in soil respiration rate respectively. In the combined Minago River data moisture shows high and significant correlation with soil respiration rate

and explains 68.2% of the variation. In the combined Haddashville data and the combined data from all the sites, moisture is significantly correlated with soil respiration rate ($r = -0.60$ - -0.63) and explains 35.6 - 39.2% of the variation in it.

In the Devil's Lake data multiple linear regressions employing all three environmental variables accounted for 80.6% of the variation in soil respiration rate in the forest and 73.9% in the right-of-way (72.3% in the combined data), (Table 14). The standard partial regression coefficients show that temperature and acrotelm are important in the forest in explaining variability in soil respiration rate, and both contribute significantly to the equation (1) while moisture has a very low standard partial correlation coefficient and insignificant. A second equation (2) was calculated for the forest data using only temperature and acrotelm, which accounted for the same amount of variation and had a lower standard error. The equation was used to predict the respiration rate in the forest (Figure 16).

In the right-of-way at the Devil's Lake, soil temperature was three to four times more important than acrotelm and moisture in explaining variability in soil respiration rate, and in the regression equation (3) only the temperature coefficient is significant. The equation was used to predict the respiration rate in the right-of-way (Figure 16). The equation accounts for 5% more variation in soil respira-

tion rate than soil temperature does alone in a simple linear regression.

The predicted peaks in soil respiration at Devil's Lake are similar for the same plot in the two years. In the forest they are 501.8 and 503.7 mg CO₂ m⁻² hr⁻¹ respectively, but in the right-of-way 384.8 and 358.3 mg CO₂ m⁻² hr⁻¹. It is of interest to note that the largest differences between predicted and measured respiration rates are found in measurements following the herbicidal application in June of 1984. This was during July in the right-of-way when the measured respiration rate was 464.8 vs predicted 345.5, and in August in the forest when the measured rate was 595.5 vs predicted of 503.7.

In the Minago River data, multiple linear regression equations were only calculated for the combined data due to the low number of observations at the site. An equation employing all three environmental variables accounted for 84.1% of the variation in soil respiration rate (Table 14). According to the standard partial regression coefficients temperature and moisture are both important in explaining variability in soil respiration rate while acrotelm is of very low importance and shows negative relationship with soil respiration rate in equation 5. A second regression equation (6) was calculated without the acrotelm which explains almost the same amount of variation in soil respiration rate with a lower standard error of estimate. The

equation was used to predict the respiration rate presented in Figure 17.

For the Hadashville data, a multiple linear regression using all three environmental variables accounted for 56.7% of the variation in soil respiration rate in the bog plot, 75.4% in the drained plot and 75.8% in the bare plot (76.8% in the combined data) (Table 14). The standard partial regression coefficients show that within the plots temperature is three to four times more important than acrotelm or moisture in explaining variability in soil respiration rate at the site. For the drained plot data the moisture variable was taken out of the equation (8) and respiration rate was predicted from soil temperature and acrotelm only (Figure 18) by using equation 9.

In the bare plot data from Hadashville (equation 10) acrotelm showed negative relationship with soil respiration rate. A second equation (11) was calculated using only soil temperature and moisture. The equation was used to predict soil respiration rate presented in Figure 19.

When the Hadashville data was combined, the importance of the acrotelm in the equation (12) was considerably higher than of moisture, indicating that the acrotelm is more important than moisture in explaining differences in respiration rates between plots. The same trend was noted when the data from all the sites were combined (equation 13).

2.5.3 Carbon-budget

Results of the estimated carbon release for each site are presented in Table 15.

The highest carbon release, $741 - 762 \text{ g C m}^{-2} \text{ yr}^{-1}$ was estimated from the drained plot at Hadashville and the lowest, $185 \text{ g C m}^{-2} \text{ yr}^{-1}$ from the Gillam right-of-way plot. During 1983 when respiration was measured at all the study sites the carbon release ranges from $206 - 362 \text{ g C m}^{-2} \text{ yr}^{-1}$ in the undisturbed bogs. The differences between sites reflect shortening of season and decrease of seasonal soil respiration rates with increasing latitude.

There are relatively little differences in length of season between 1983 and 1984, for the Devil's Lake and Hadashville sites. The estimated carbon release is however 19 to 25% higher in 1984 than 1983 for the Devil's Lake plots and Hadashville bog plot, which is due to higher average seasonal soil respiration rates in these plots in 1984.

At Gillam the season length in 1983 is estimated as 166 days, but at Hadashville as 221 and 219 days in 1983 and 1984 respectively. For the 30 year normal monthly temperature means (Figure 20) the season length would be estimated as 160 days (May 6 - October 12) at Gillam, 198 days (April 15 - October 28) at Grand Rapids and 208 days (April 9 - November 3) for Pinawa.

TABLE 15

Estimated seasonal carbon release at study sites

Estimated time of spring-thaw, freeze-up and seasonal length (days) at the study sites and estimated carbon release ($\text{g C m}^{-2} \text{ yr}^{-1}$) for each study plot.

SITE,	PLOT	YEAR	THAW	FREEZE-UP	SEASON LENGTH	CARBON RELEASE ¹
Gillam 2 ²	For	1983	May 16	Oct 28	166	206 (146 + 60)
"	ROW	"	" "	" "	"	185 (131 + 54)
Minago R.	For	1983	May 1	Nov 11	191	277 (243 + 34)
"	ROW	"	" "	" "	"	308 (280 + 43)
Devil's L.	For	1983	Apr 19	Nov 9	204	362 (320 + 42)
"	"	"	" "	" "	"	308 (280 + 28)
"	For	1984	Mar 27	Oct 24	212	451 (386 + 65)
"	ROW	"	" "	" "	"	366 (324 + 42)
Hadashv.	Bog	1983	Apr 13	Nov 9	221	329 (288 + 41)
"	Drain.	"	" "	" "	"	762 (659 + 103)
"	Bog	1984	Mar 23	Oct "	219	394 (354 + 40)
"	Drain.	"	" "	" "	"	741 (687 + 54)
"	Bare	"	" "	" "	"	407 (350 + 57)

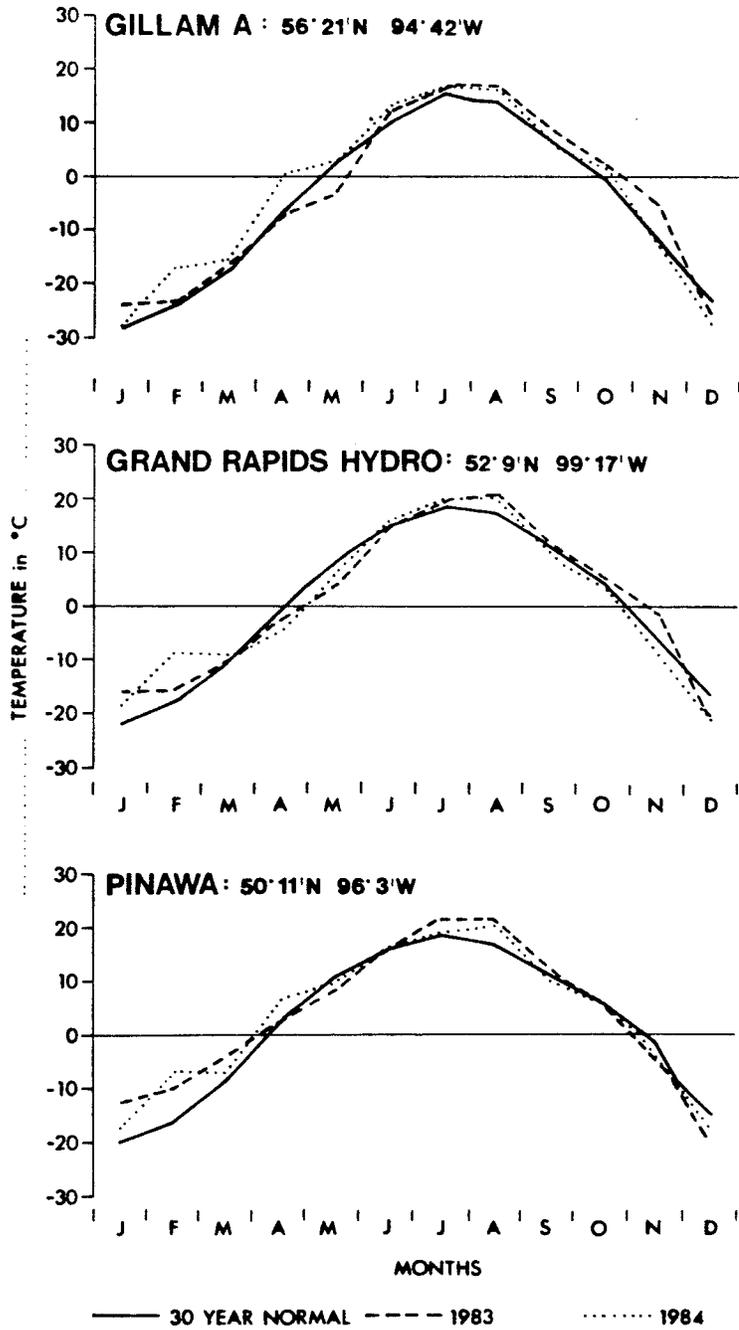
¹ The values in brackets represent firstly the estimated carbon release during the time from first to last measurement of soil respiration in the field and secondly the addition for the part of the season in spring and fall not covered in the fieldwork.

² The respiration rate at the Gillam 2 site was assumed 350 and 315 $\text{mg CO}_2 \text{ m}^{-2} \text{ hr}^{-1}$ in the forest and right-of-way on August 15.

Figure 20.

A comparison of monthly temperature means ($^{\circ}$ C) for three meteorological stations in northern, central and southern part of the study area.

(c/o Environment Canada, AES, Winnipeg, Manitoba).



2.6 DISCUSSION

2.6.1 Soil respiration and environmental parameters

The respiration rates and their seasonal changes measured in the relatively undisturbed bogs agree well with rates given by other workers. The maximum respiration rates range from about 250 mg CO₂ m⁻² hr⁻¹ at Gillam to about 500 mg CO₂ m⁻² hr⁻¹ at the Hadashville site. These rates fall in between rates of soil respiration determined in alpine and arctic communities (Repnevskaya 1967; Svensson *et al.* 1975; Peterson and Billings 1975; Billings *et al.* 1977; Poole and Miller 1982; Prescott and Parkinson 1985) and southern temperate forests communities (Reiners 1968; Garrett and Cox 1973; Edwards and Sollins 1973; Edwards 1975), and they are of a similar order as soil respiration rates determined in lowland communities of comparable latitudes in Europe (Anderson 1973; Singh and Gupta 1977).

With the respiration rates determined in the study the amount of alkali left unused after absorption was equal to or greater than 53% of the initial amount for all sites, except for the very high mid-summer rates in the drained plot at the Hadashville site. There, the amount of alkali left unused after absorption was 26 - 41% in the July and August measurements and reduction in absorption efficiency may have occurred at the highest rates (Gupta and Singh 1977).

Standard errors of the respiration rate estimates were generally low in spite of few replicates within plots. The mean standard error was 8.2%, but for individual estimates the standard errors ranged from 0.8 - 19.3%.

Soil respiration or decomposition is a complex process which is regulated by a host of variables (Witkamp 1966b; Flanagan and Veum 1974; Edwards 1975; Wildung et al. 1975; Singh and Gupta 1977; Swift et al. 1979). The variables most often mentioned in the literature are temperature, moisture, substrate quality (e.g. nutrient status, lignin content, pH) and quantity (e.g. thickness of organic layer), soil density and soil biota community structure. These variables have complex interdependent effects which may differ temporally within or between sites. It is therefore difficult, especially under field conditions, to determine the controlling effects of single variables on soil respiration rates (Garret and Cox 1973; Edwards 1975). In spite of this complexity, the prevailing climatic environment and substrate quality are generally considered as the most important factors influencing decay rates (Meetenmeyer 1978; Boddy 1983).

In this study soil respiration rates were related to changes in parameters which all are related to climate, but all the sites were assumed strongly similar in bog habitat and peat substrate type.

The correlations and regression equations showed that of the three environmental variables measured in this study soil temperature is the single most important variable controlling the seasonal trends in respiration rates within individual plots. However when the data are combined soil temperature tends to explain less variation in soil respiration rate than within single plots indicating increasing importance of other variables in explaining differences between plots.

The dominant regulatory effects of soil temperature on soil respiration rates over other environmental parameters has been reported in numerous studies from temperate to arctic areas, where moisture is generally not limiting (e.g. Douglas and Tedrow 1959; Witkamp 1966a; Reiners 1968; Anderson 1978; Chapman 1979; Belkovskiy and Reshetnik 1981; Garrett and Cox 1973).

Carbon dioxide evolution rates have frequently been found to increase exponentially with rising temperature, and log-linear models have better described the relationship between CO₂ evolution rates and temperature than linear models (Douglas and Tedrow 1959; Reiners 1968; Kucera and Kirkham 1971; Flanagan and Veum 1974; Chapman 1979; Silvola and Heikkinen 1979). In the present data, plots of CO₂ rates against soil temperature did not reveal curvilinear relationships, neither for within site data nor for combined data, and linear model was generally found to describe the

relationships better than a log-linear model. This may be because the soil temperature which was measured at 10 cm did not adequately represent the seasonal temperature trends for the entire acrotelm. Most of the studies which describe the exponential relationships are from soils with relatively thin organic matter layers of fixed depths, where it should be easier to obtain representative measurements of temperature within the organic matter.

In peatlands the layer of the organic soil above the water table or ice is generally considered as the critical stratum for decomposition and the source of CO₂ evolving from these soils (Makarov 1960; Clymo 1978, 1983; Lindholm and Markkula 1984). Increased depth of the acrotelm has been associated with increasing CO₂ production in organic soils (Neller 1944; Volk 1973; Peterson et al. 1984) as was found in this study where the acrotelm had generally a positive correlation with soil respiration rate. The relationship was however only significant for single plots in the Devil's Lake data, indicating that the variable does not govern seasonal trends within plots. Where sufficient data is available, e.g. at Hadashville and for the combined data from all the sites, the correlations and regression equations show that acrotelm is of importance in explaining between plots or sites differences.

The relationship between moisture and decomposition rates in organic matter have been shown to be complex and vary

with changes in temperature (Douglas and Tedrow 1959; Bunnell and Tait 1974). Inverse, direct and curvilinear relationships have been described between soil moisture content and soil respiration rates (Chapman 1979) depending on the prevailing moisture levels.

Moisture may limit soil respiration in two ways, i.e. in the low moisture content region where the soil biota requires water for metabolism, and in the high moisture content region where decomposition and root activity may become limited due to lack of aeration (Bunnell et al. 1977; Boddy 1983). In bogs low moisture content in the top layers should normally not limit soil respiration rate. Limits are more likely to be caused by high moisture content and waterlogged conditions. The negative relationship found between soil moisture and respiration rate in this study indicates that in the bogs high soil moisture exerts some limiting influence on soil respiration rate. Correlations between soil respiration rate and moisture are mostly low and insignificant and the variable is of low importance in the regression equations, except in the Minago River data. The effects of soil moisture on soil respiration rate are definitely less pronounced than thickness of the acrotelm.

With the exception of the drained plot at Hadashville the soil moisture content in the top 10 cm of the plots was generally of the order 75 - 90% and seasonal changes within plots were in the range of 5 - 10%. At this high moisture

content the observed changes in moisture content are likely to have affected respiration in the top soil. In organic soils, moisture content of the peat will normally increase with increasing depth towards the water table (Makarov 1960; Volk 1973; Ingram 1983) and seasonal fluctuations in moisture content will decrease with increasing soil depth. The significance of moisture changes in affecting soil respiration will therefore decrease with increasing soil depth. The changes in soil moisture content measured in the top 10 cm of the plots may not have been representative of the changes occurring in the whole soil profiles above the water table, which were considerably thicker than 10 cm.

In general the multiple linear regression equations have good predictive capability and describe adequately the responses of the soil biota to changes in soil temperature, acrotelm and soil moisture. The equations did not account for 19 - 43 % of the variation in soil respiration in single plot data sets and 16 - 28% of the variation in combined data sets. A more detailed measurement of soil temperature and determinations of soil moisture content from greater soil depths might have resulted in improved predictions of the regression equations. It is likely, that in order to improve the predictions it would have been necessary to study factors which have been found to influence respiration rates in other works but were not taken into account in the study; e.g. precipitation (Gupta and Singh 1981), substrate quality (Wynn-Williams 1980), root biomass (Chapman 1979).

Precipitation was extremely high in the Hadashville area during June of 1984, which is likely the cause of the suppression in soil respiration activity in the bog and bare plots at the Hadashville site in late June (Figure 18). The regression equations predicted considerably higher respiration rates than were observed indicating that the suppression of the respiration rates could not be accounted for by changes in the three environmental variables monitored. The high discrepancy between measured and predicted respiration rate in the Hadashville drained plot in October of 1984 was believed related to precipitation. Unexpectedly the observed respiration rate increased considerably from the September rate, while the regression predicts continuing declining trend in respiration rate. On the day of the measurement heavy precipitation fell at the site after a dry August and September. The moisture content in the surface layers of the plot had become very low in August and September and may have reached the level where it started to limit respiration rates (Waksman and Purvis 1932, cited in Kavanagh and Herlihy 1975).

Sudden and extreme changes in temperature and or moisture, e.g. freezing and thawing, wetting and drying, have been found to be able to cause a burst of decomposers activity, which cannot be accounted for by the proportional changes in temperature and moisture (Douglas and Tedrow 1959; Witkamp 1969; Bunnell et al. 1975). Such peaks in

respiration activity are usually not long lasting but they are attributed to a sudden release of large amounts of soluble materials which are easily decomposed. The sudden rewetting of the surface layers at the Hadashville bog plot in October of 1984 may have caused the observed respiration rate to be considerably higher than what could be accounted for by the environmental variables in the regression equation.

2.6.2 Effects of disturbances on the soil environment, respiration rates and decomposition

The disturbances observed in this study are of two types. Firstly, are the right-of-way disturbances at the Devil's Lake, Minago River and Gillam sites, where there was a permanent removal of the tree canopy and disturbances due to line construction and maintenance. Disturbances due to vegetation management differ between the sites. At Devil's Lake herbicides have been used to control regeneration of trees in the right-of-way, but not at Minago River and Gillam. Secondly, are the disturbances caused by the drainage at the Hadashville site which were studied to serve as a basis for comparison and aiding in assessing the extent of the right-of-way disturbances on soil respiration.

2.6.2.1 Right-of-way disturbances: general

When the soil respiration rates in the forest and right-of-way plots were compared it was evident that the rates were most often higher in the forest plots. At all the sites the average respiration rates are higher in the forest than in the right-of-way (Table 12), the average respiration rates in the right-of-way ranging from 84 to 99% of the forest rates. The difference is greater at the Devil's Lake site and only significant there, but at that site the number of observations is substantially higher than at the Minago River and Gillam sites. Can these differences in measured soil respiration rates between forest and right-of-way be explained in terms of differences in the environmental variables studied?

The only variable which shows a consistent difference between forest and right-of-way when the average values are compared is the acrotelm (Table 12), whose average thickness is always greater in the forest than in the right-of-way, the difference being from 0.4 to 4.9 cm. At the Devil's Lake and Gillam 2 site the difference is significant.

The generally shallower acrotelm in the right-of-way might be caused by depressed evapotranspiration due to the canopy removal which may have led to a rise in water level. (Jeglum et al. 1974; Tallis 1983).

A greater thickness in acrotelm in the forest than right-of-way should explain some of the observed differences in respiration rates. The other two variables, soil temperature and moisture as measured in the study do not show as consistent differences between forest and right-of-way (Table 12). The differences are significant in some cases, but the averages for each variable are either higher in the forest or the right-of-way and there is no distinct trend. At the Devil's Lake site though the average moisture is lower in the forest than right-of-way in both years which contributes to higher respiration rates in the forest.

The averages of the environmental variables for forest and right-of-way from the Devil's Lake site were used to predict respiration rates in separate multiple linear regression equations for the forest and right-of-way (equations 1 and 3 in Table 14) for 1983 and 1984. In all four cases higher respiration rates were predicted in the forest and the respiration rate in the right-of-way was 92 - 94% of the forest rates. Thus between forest and right-of-way the degree of difference in environmental parameters would generally give rise to smaller differences in respiration rates than those measured. This indicates that the higher respiration rates measured in the forest and right-of-way at the Devil's Lake site can partly but not entirely be accounted for by differences in three environmental parameters monitored.

At the Minago River site average soil respiration rate is nearly the same in the forest and right-of-way (Table 12). The acrotelm is deeper in the forest but it shows a low correlation with soil respiration (Table 13) and is not of value in the multiple linear regression (equation 5, Table 14). Moisture shows very high correlation with respiration rate in both forest and right-of-way. Average soil temperature is almost the same in forest and right-of-way. The multiple linear regression equation (equation 6 in Table 14) predicts generally higher respiration rates in the right-of-way than forest based on the averages in environmental parameters in forest and right-of-way, indicating that environmental conditions in terms of these variables are generally better for soil respiration in the right-of-way than forest.

A regression model is not available for the Gillam sites and reference has to be made to the the other sites. At the Gillam 2 site slightly higher average soil temperature and significantly lower soil moisture in the right-of-way than the forest plot should approximately balance the effects of significantly deeper acrotelm in the forest (Table 12). At the site environmental conditions in the forest do not appear to favour higher respiration rates there than in the right-of-way.

At the Henday 1 site soil temperature is significantly higher in the right-of-way while moisture is significantly lower in the forest. These variables should approximately

balance each other's effects in causing differences in soil respiration rates between forest and right-of-way. Differences in acrotelm at the site are low and insignificant. In the multiple linear regression for the combined data from all the sites (equation 13 in Table 14) slightly higher (3 - 6%) respiration rates were predicted in the right-of-way than in the forest for the Gillam sites.

Thus it appears that the inflated respiration rates measured in the forest compared to the right-of-way can only partly or not be accounted for by differences in the three environmental variables measured. Other factors not measured, which differ between forest and right-of-way, are likely to have contributed to the measured differences.

The most probable explanation is that the consistently higher average respiration rates in the forest is due to greater root biomass and activity in the forest than the right-of-way. Root biomass was not determined in this study but contrasts in vegetation between forest and right-of-way plots and observations in the field support the view that root biomass was generally greater in the forest than right-of-way.

At the two sites in the Gillam area, bare ground was in the range of 70 - 80% in the right-of-way plots and surrounding areas. The vegetation cover within the plots was mainly from mosses. Of vascular plants, only scattered

plants of Ledum groenlandicum and Rubus chamaemorus were within the plots. Roots were not encountered underneath the respiration cylinders when soil respiration was measured and contribution by roots to total soil respiration within these plots is assumed to be negligible. The forest plots on the other hand had nearly complete vegetation cover with conspicuous shrub layers. The Gillam 2 plot was in a stand of black spruce but tree cover was low at the Henday 1 site. In these plots roots were frequently encountered underneath the respiration cylinders.

If the difference in measured average soil respiration at the Gillam sites is due to differences in root respiration, it means that in the forest plots near 10% of the total soil respiration at the Gillam 2 site and near 7% at the Henday 1 site, is due to root respiration. Therefore 90 -93% of the soil respiration is due to decomposition. If the slightly higher average respiration rate predicted in the right-of-way than in the forest at the sites is taken into consideration then the contribution may be somewhat higher than 7 - 10%.

The lack of trees at the Henday 1 site might explain the smaller difference in measured respiration between forest and right-of-way at that site than at the Gillam 2 site. Sims and Stewart (1981) working at the Gillam 2 site found that in the forested bog, black spruce made up about 30% of the above ground biomass of vascular plants. This might give an indication of the relative below ground biomass.

At the Minago River and Devil's Lake sites the right-of-way plots were not as devoid of vascular plants as at the Gillam sites, and root respiration must have contributed to the total soil respiration. The vascular plants were however much reduced in cover in the right-of-way plots compared to the forest. In 1975, Sims (1977) estimated aerial biomass and primary production at four peatland sites (including the Devil's Lake and Gillam 2 sites) along the Radisson-Dorsey transmission line. Biomass in the right-of-way was compared to biomass in the adjacent undisturbed forest. Biomass of vascular plants was found to have been reduced over 90% at all sites, which was mostly due to the removal of the tree cover.

At Minago River the measured average soil respiration rate in the right-of-way was 1% less than in the forest, and at Devil's Lake it was 12 - 16% less. Comparison of the environmental conditions in forest and right-of-way at the sites indicated that respiration rates should have been generally higher in the right-of-way than forest at Minago River. At Devil's Lake differences in the environmental parameters can hardly have given rise to all the observed differences in soil respiration. Higher contribution of root respiration to total soil respiration in the forest as opposed to the right-of-way can explain some of these differences. Considering the reduction in biomass in the right-of-way compared to the forest in 1975 and slow tree

regeneration in the right-of-way, it is highly unlikely that root biomass and activity in the right-of-way had by 1983 reached more than one fourth of that of the forest. One fourth is probably an overestimate, especially for the Devil's Lake site which was treated with herbicides in 1979.

The relatively small differences in observed soil respiration rates between forest and right-of-way at all the sites indicate that root respiration in the undisturbed bog communities is a relatively small proportion of the total soil respiration and in the range of 10 - 15%, while 85 - 90% can be attributed to decomposition processes.

The relatively high soil respiration rates measured in the bare plot at the Hadashville site compared to the bog plot also support the view that most of the soil respiration at the sites is due to decomposition and that root respiration is a relatively small proportion. In the bare plot the soil respiration was entirely due to decomposition of peat as the soil did not contain any live roots. The average respiration rate in the bare plot was higher than in the bog plot, but differences were not significant.

Estimates of contribution of root respiration to total soil respiration vary considerably. This variation partly reflects the differences in methodology and difficulties in estimating root respiration, but also is due to differences in plant communities and soil types. Phillipson et al.

(1975) attributed only 3 - 4% of soil respiration to roots in a beechwood in Britain, while Chapman (1979) estimated root respiration as 70% of total soil respiration in a Calluna heathland in Britain. For forest communities most estimates of root respiration as percentage of total soil respiration fall in the range of 20 - 50% (Anderson 1973; Singh and Gupta 1977; Chapman 1979). The value of 10 - 15% estimated in the present work is therefore in the low range. However most of these studies of soil respiration and root respiration are from forests on mineral soils and with relatively thin organic mats. Considering the dominantly organic nature of bog soils, the generally low primary production of the vegetation and the high contribution of mosses (Reader 1978; Bradbury and Grace 1983), then the relative contribution of roots to total soil respiration may be expected to be less in bogs than in more productive forest types with relatively thin litter and organic matter layers.

2.6.2.2 Right-of-way disturbances: herbicides

When soil respiration was measured at the Devil's Lake site in 1983 four years had passed since the herbicide Tordon 101 (picloram + 2,4-D) was applied in the right-of-way at the site to control tree regeneration. The general pattern of CO₂ evolution rates in the forest and right-of-way at the site in 1983 and comparison with soil respiration at the other sites in 1983 does not show that the herbicidal

application had caused major and long term changes in soil respiration activity in the right-of-way (Figure 16). Studies investigating the effects of herbicides on soil microorganisms and respiration have not revealed that herbicides have any long-term harmful effects on microbial numbers, composition or activities when used at field application rates (Johnen and Drew 1977; Wainwright 1978).

There are on the other hand indications that herbicidal applications may temporarily affect soil respiration or decomposition rates which may be attributed to direct toxic effects of herbicides on the litter and soil biota (Hendrix and Parmelee 1985) and indirect effects resulting from altered chemical composition of substrates (Gottschalk and Shure 1979) or altered microclimatic conditions (Suffling and Smith 1979).

Tordon 10K (picloram) was applied in the right-of-way at the Devil's Lake site around June 10, 1984. When soil respiration was measured at the site on July 11, the herbicide had recently dissolved and the vegetation in the right-of-way had apparently been affected. On that day the respiration rate was the highest ($465 \text{ mg CO}_2 \text{ m}^{-2} \text{ hr}^{-1}$) measured in the right-of-way plot during the two years. On the next visit to the site, on August 20, the rate had fallen and was at a similar level as at a comparable time in the previous year (Figure 16).

It was noted on August 20, that the herbicide had been carried with groundwater into the forest to the east of the right-of-way and downslope from it. Vegetation in the forest, including that of the respiration plot, as far as 50 meters from the right-of-way had apparently been affected by the herbicide. Measurements of the soil respiration rate gave exceptionally high results ($595 \text{ mg CO}_2 \text{ m}^{-2} \text{ hr}^{-1}$) or the highest rates measured in the plot in the two years. In the next measurement, September 15, the respiration rate in the forest had fallen to a level similar to September of 1983 (Figure 16).

Since the periods between measurements were relatively long, the exact intensity and length of high respiration activity following the herbicidal application may not have been accurately revealed.

Although these results can not be interpreted as a proof of a direct link between the herbicidal application at the site and the high respiration rates encountered, they are considered to indicate a trend in that direction.

In laboratory experiments where Tordon 10K was applied at field rates to cores of Sphagnum peat and respiration was measured from 1 to 90 days following the application, the respiration rates were found to be considerably inflated in the treated cores 10 - 30 days following application but close to control levels after that (see chapter 4).

The sudden increase in respiration following herbicidal application at Devil's Lake and in the laboratory experiment, indicate that the herbicide had direct effects on the soil biota. Although some of the respiration increase may have resulted from microbial utilization of the herbicide (Wainwright 1978) it is likely that the herbicide has caused a damage of root and microbial cells and a sudden release of soluble substrates which were quickly metabolized by microbes unaffected by the application.

2.6.2.3 Carbon-release and decomposition

The annual estimated carbon release of $206 - 451 \text{ g C m}^{-2}$ from the relatively undisturbed bogs is high compared to other available estimates of carbon releases from bogs.

Svensson et al. (1975) estimated annual CO_2 -carbon loss from an elevated ombrotrophic tundra bog in northern Sweden as $50 \text{ g C m}^{-2} \text{ yr}^{-1}$. At Moor House, England the efflux of CO_2 from a bog surface calculated as carbon was $31 - 54 \text{ g C m}^{-2} \text{ yr}^{-1}$ (Clymo 1983). The higher estimates of the present work are to some extent due to environmental differences, e.g. cooler summer climate at the Swedish site (Svensson et al. 1975) than in the present study area, and a very high water table (0 - 30 cm) in the Moor House bog (Heal and Smith 1978).

Estimates of carbon release from temperate forest soils are generally higher than the estimates of this study, which show stronger resemblance to results from work in alpine and boreal plant communities. Reiners (1968) estimated 12 month carbon release in swamp, fen and oak forest in Minnesota from 700 - 800 g C m⁻² yr⁻¹. In different alpine plant communities in Norway, Svensson et al. (1975) estimated yearly carbon release to range from about 100 - 700 g C m⁻² yr⁻¹. Haavas and Mäenpää (1972) estimated carbon release from the floor of a spruce forest in Finland near 440 g C m⁻² yr⁻¹.

Schlesinger (1977) has examined longitudinal trends in values of estimates of yearly soil carbon release and compiled data from studies of soil respiration in different forest and woodland communities between 1 and 77° N. He calculated a linear regression (C-evolution = -24.2 (Lat) + 1721.5; R = 0.60; F = 30.05) relating annual carbon release and latitude. For the latitudes of the present study the equation predicts carbon release of 360 - 520 g C m⁻² yr⁻¹, which is slightly higher than the estimated releases for the sites of this study. However the regression is mostly based on studies from forests on mineral soils which tend to be more productive than bog forests. Lower annual carbon release values than predicted by the equation are therefore to be expected for bog sites.

The carbon budgets for each site can be used to estimate the amount of organic matter degraded in the soil by decom-

posers. At the forested sites root respiration is assumed to be 15% of total soil respiration rate, 4% in the right-of-way at Devil's Lake and Minago River, negligible in right-of-way at Gillam and in the bare plot at Hadashville, and 10% in the treeless bog and drained plots at Hadashville. Conversion of carbon to dry matter is calculated on the assumption that carbon content of the organic material being broken down is 50% of the dry matter (Flanagan and Veum 1974; Silvola and Hanski 1979). The results are presented in Table 16.

The estimated organic matter decomposition tends to decrease with increasing latitude as a result of shortening of the season and lower soil respiration rates. In 1983 when respiration was measured at all four sites the estimated yearly breakdown at the Gillam site, $350 \text{ g O.M. dwt m}^{-2} \text{ yr}^{-1}$, is about 60% of what is estimated for the southern Devil's Lake and Hadashville bog plots (616 and $592 \text{ g O.M. dwt m}^{-2} \text{ yr}^{-1}$ respectively).

There are few direct estimates of total organic matter breakdown in undisturbed peatland soils based on soil respiration. If the estimates of carbon release from the bogs in northern Sweden and Moor House, England quoted above, are converted to organic matter they are in the range of $60 - 110 \text{ g dwt m}^{-2} \text{ yr}^{-1}$. These figures may be influenced by root respiration and overestimate organic matter breakdown. Douglas and Tedrow (1959) estimated organic matter breakdown

TABLE 16

Carbon-release, root respiration, and O.M. decomposition

Estimated seasonal carbon release (g C m^{-2}), root respiration (g C m^{-2}) organic matter decomposition (g C m^{-2} and g O.M. m^{-2}) at four peatland sites in Manitoba.

SITE, PLOT	YEAR	SEASON LENGTH DAYS	CARBON RELEASE	ROOT RESP.	DECOMPOSITION	
					C	O.M.
Gillam 2, For	1983	166	206	31	175	350
" ROW	"	"	185	0	185	370
Min.Riv. For	1983	191	277	42	235	470
" ROW	"	"	282	11	271	542
Dev.Lake For	1983	204	362	54	308	616
" ROW	"	"	308	12	296	592
" For	1984	212	451	68	383	766
" ROW	"	"	366	15	351	702
Hadashv. Bog	1983	221	329	33	296	592
" Drained	"	"	762	76	686	1372
" Bog	1984	219	394	39	355	710
" Drained	"	"	741	74	667	1334
" Bare	"	"	407	0	407	814

in a half-bog soil at Barrow, Alaska as $190 \text{ g dwt m}^{-2} \text{ yr}^{-1}$, while decomposition estimates based on soil respiration in tundra communities in north central Alaska by Poole and Miller (1982) would be equivalent to near $45 - 70 \text{ g dwt m}^{-2} \text{ yr}^{-1}$.

Davis (1980) has estimated organic matter loss due to decomposition in two Antarctic moss bank communities. At a site with peat thickness of 15 cm and growing season of 181 days the weight loss due to decomposition was estimated as $288 \text{ g dwt m}^{-2} \text{ yr}^{-1}$, while at a site with a 24 cm thick peat layer and a growing season of 151 days the estimated organic matter breakdown was $571 \text{ g dwt m}^{-2} \text{ yr}^{-1}$.

Silvola and Hanski (1979) simulated carbon accumulation in a raised Empetrum nigrum/Sphagnum fuscum bog in Finland on the basis of laboratory measurements of CO_2 exchange (Silvola and Heikkinen 1979). Peat respiration was estimated as equivalent to breakdown of 248 g dwt m^{-2} organic matter in a seven months season. The stimulation did not include decomposition of peat deeper than 17 cm. Summer temperatures in the top peat reported in the study were considerably lower than those at the sites of the present study. The respiration rate of the peat in the Finish study showed relatively little seasonal changes having a maximum of about $120 \text{ mg CO}_2 \text{ m}^{-2} \text{ hr}^{-1}$ and averaging less than $100 \text{ mg CO}_2 \text{ m}^{-2} \text{ hr}^{-1}$ for the season. These rates are 2 - 3 times lower than the average soil respiration rates determined in

the undrained bogs of the present study. Considering this the estimated organic matter breakdown at different sites in the present study do not seem unrealistically high.

The estimates of length of season for each site were made on the assumption that soil respiration activity is insignificant while daily mean temperatures stayed below 0° C. This may have introduced errors into the estimated seasonal carbon release and organic matter breakdown for each site. The rate of soil thawing in the spring and freezing in the fall probably differed between sites due to climatic and soil differences. It is for example likely that at the Gillingham sites where there is permafrost in the ground and air temperatures generally coldest in the study area (Figure 20), that cooling and freezing of the soil in the fall will be more rapid than at the southern sites.

Reports of continued measurements of soil temperatures indicate that in central Canada, conditions for soil respiration should generally be unfavorable during the winter, or from the end of November to the beginning of April (Williams 1968; Reimer and Shaykewich 1980).

In southeastern Minnesota, Reiners (1968) in a study of soil respiration could not detect any soil respiration activity from mid January to mid March while soils remained frozen but respiration activity started again when the top soil began to thaw in early spring. Similar are the find-

ings of Svensson et al. (1975) and Repnevskaya (1967) who found soil respiration activity to start in the spring immediately after melting of snow. In studies of respiration in peat moss banks in the Antarctic no respiration was detected in frozen peat (Wynn-Williams 1980), but soil respiration started immediately after thawing of the top peat. Season length at the study sites was estimated as the number of days in each month with temperature of 0° C or more (Davis 1980).

There is no conclusive evidence that the right-of-way disturbances have affected decomposition processes in the soils of the right-of-way. The estimated amounts of organic matter broken down in the forest and right-of-way soils are fairly similar. At Devil's Lake where most data on soil respiration was sampled, 4 - 8% higher organic matter breakdown is estimated in the forest than in the right-of-way in the two years of study. At Minago River 15% higher organic matter breakdown is estimated in the right-of-way than forest in 1983 and at the Gillam 5% higher organic matter breakdown is estimated in the right-of-way than forest. However due to the low sampling intensity at these two sites and the assumptions made in the calculations at all the sites the real meaning of these differences has to be questioned.

While decomposition processes in the bog soils in the right-of-way have not been greatly affected by disturbances,

it does not imply that peat accumulation in bogs within the right-of-way has not been affected. Due to the canopy removal and vegetation disturbances in the right-of-way area and continued management practises the biomass and primary production is considerably reduced from that in the undisturbed forest. This will lead to a reduction in litter input to the soil in the right-of-way and possibly an imbalance in the production:decomposition ratio if the production of litter entering the soil is less than the amount of organic matter broken down every year. This condition is likely to prevail while the transmission corridor is in use, which could be several decades or even exceed a century. A prolonged imbalance in production:decomposition ratio of the bogs would cause a net breakdown of the organic matter reserve in the peat soils, but the rate of that breakdown would be largely determined by the relative reduction in production.

Sims (1977) estimated yearly above-ground primary production at the Devil's Lake site in 1975 as 527 g dwt m⁻² in the forest and 171 g dwt m⁻² in the right-of-way. At the Gillam 2 site the estimated primary production was 344 and 61 g dwt m⁻² in the forest and right-of-way respectively. Below-ground primary production was not estimated in the study. Estimates of below-ground production in bogs from other studies show great variation and have to be taken with caution due to a lack of standard methods to determine be-

low-ground production. In the majority of studies below-ground production has been estimated as less but not lower than 20% of above-ground production, while few studies estimate below-ground production considerably higher than above-ground production (Reader 1978; Bradbury and Grace 1983).

The production values from Sims (1977) can be used as a tentative comparison to contrast available estimates of production at the Devil's Lake and Gillam 2 sites, which may give an indication of how the actual production:decomposition ratio within the right-of-way bog communities have been affected by disturbances. To estimate total primary production, below-ground production was assumed to be equal to above-ground production. The results are presented in Table 17.

The amount of litter added to the soil each year will generally be less than the annual net production since some soluble materials will be translocated from senescing leaves into branches and stems, and then some decomposition loss will occur from aerial parts before they are added to the soil surface. Reader and Stewart (1972) estimated that 90% of the net primary production of a bog in southeastern Manitoba would be added to the soil each year.

In general the comparisons indicate that with the same reduction in primary production as encountered by Sims (1977) in bogs within the right-of-way, the yearly decompo-

TABLE 17

Primary production - O.M. decomposition estimates

Estimates of total primary production and soil organic matter decomposition in forest and right-of-way at Devil's Lake and Gillam 2 sites (g O.M. dwt m⁻² yr⁻¹).

SITE	PRIMARY PRODUCTION ¹	SOIL ORGANIC MATTER DECOMPOSITION
Devil's Lake:		
forest	1054	616 - 766
ROW	342	592 - 702
Gillam 2:		
forest	688	350
ROW	122	370

¹ Primary production estimates are based on work by Sims (1977) and the assumption that below-ground production is equal to above-ground production.

sition of organic matter in the right-of-way will greatly exceed the annual net primary production and addition of new organic matter to the soil. A net loss of 250 - 350 g dry matter $\text{m}^{-2} \text{yr}^{-1}$ would be predicted from these comparisons. With peat thickness of 110 cm in the right-of-way at the Devil's Lake site and 90 cm at the Gillam 2 site and assumed peat bulk density of 0.06 cm^{-3} for the type of peat encountered (See chapter 4; Silvola and Hanski 1979; Zoltai and Johnston 1985) the dry organic matter reserve in the right-of-way at these sites would be estimated as 66 kg m^{-2} at the Devil's Lake site and 54 kg m^{-2} at the Gillam site. A net yearly organic matter loss of 250 - 350 g dry matter m^{-2} would therefore reduce the soil organic matter reserve by ca. 0.5% per year.

The organic matter breakdown in the drained plot at the Hadashville site is about twice that in the bog plot (Table 16). As the drained plot was positioned very close to the drainage ditch where the water table is lowest in the bog, the organic matter breakdown represents an extreme. With increasing distance from the ditch the water level in the bog rises and the effects of drainage on organic matter breakdown will decrease.

Drainage of peatlands has universally been found to reverse the conditions which permit the accumulation of organic soils and lead to a breakdown of these soils. Drainage is followed by subsidence of the soils which is largely

cause by increased biological oxidation of the soil organic matter (Broadbent 1960; Browder and Volk 1978; Hutchinson 1980).

2.7 SUMMARY AND CONCLUSIONS

The soil respiration rates determined in the study at Gillam, Minago River and Devil's Lake ranged from 148.7 - 595.5 mg CO₂ m⁻² hr⁻¹, while at Hadashville respiration rates ranged from 112.6 to 945.2 mg CO₂ m⁻² hr⁻¹.

At Gillam, Minago River and Devil's Lake the average depth of the acrotelm was less in the right-of-way than in the adjacent forest. The difference was significant at the Devil's Lake and Gillam 2 sites. Soil temperature and moisture did not show consistent differences between forest and right-of-way when all the sites were considered. At these sites the average seasonal soil respiration rate was higher in the forest than right-of-way. The respiration rate in the right-of-way averaged 84 - 99% of the forest rates.

At Hadashville lowering of the water table in the disturbed part of the bog had caused a significant rise in soil temperature and a significant lowering of soil moisture compared to the relatively undisturbed part of the bog. The average respiration rate in the disturbed part of the bog was over twice that in the relatively undisturbed part.

Of the three environmental variables measured only soil temperature showed generally high and significant correlation with soil respiration rate. It explained 43.9 - 72.3% of the observed variation in soil respiration rate within single plots.

Acrotelm was only significantly correlated with soil respiration rate at the Devil's Lake site, where the variable explained 55 - 68.4% of the observed variation in soil respiration within single plots. In plots at other sites correlation was low and acrotelm explained less than 15% of the observed variation in soil respiration rate.

Soil moisture was generally negatively correlated with soil respiration. Correlation was only high and significant in the Minago River and Hadashville bog plots, where the variable explained up to 92% of the variation in soil respiration rate. In other plots correlation of the variable with soil respiration was generally low and the variable explained less than 16% of the variation in soil respiration rate.

Multiple linear regression models relating soil respiration rates to changes in soil temperature, acrotelm and soil moisture explained 56.7 - 80.6% of the observed variation in soil respiration rates for single plots. For combined data from plots within sites the models explained 72.3 - 84.1% of the variation in soil respiration rate. When the data from all sites was combined 77.7% of the variation in soil respiration was accounted for by a multiple linear regression model.

In the regression models soil temperature was most important in explaining variability in soil respiration rate and

contributed significantly to the models. The importance of acrotelm and soil moisture in the models was overall considerably less than that of soil temperature. In models where data from more than one plot or sites was combined the contribution of the acrotelm to explaining variation tended to increase from single plot models indicating that the variable is important in explaining differences in respiration rates between plots or sites. The same trend was not observed for soil moisture.

Comparisons of environmental conditions in forest and right-of-way at the Gillam, Minago River and Devil's Lake sites indicated that respiration rates in the forest in comparison to the right-of-way rates were generally higher than could be accounted for by differences in the three environmental variables measured. Higher root respiration in the forest than right-of-way was considered to attribute to these differences. In the undisturbed forest root activity was estimated to give rise to 10 - 15% of total soil respiration while 85 - 90% was attributed to decomposition processes.

It is not evident from the study that decomposition processes in the soil in the right-of-way at the Gillam, Minago River and Devil's Lake sites have been altered due to disturbances from line construction, canopy removal and continued vegetation management practices.

Exceptionally high soil respiration rates were measured at the Devil's Lake site in 1984 following an application of the herbicide picloram in the right-of-way in June, but the respiration rates were back to expected seasonal levels in the fall.

At Hadashville drainage has resulted in approximately doubling of rate of decomposition of soil organic matter in the disturbed part of the bog studied compared to the relatively undisturbed part.

The estimated length of seasonal soil respiration activity varied from 166 days at the Gillam sites to 220 days at the Hadashville site.

The annual estimated breakdown of soil organic matter for Gillam 2 site was 350 and 370 g O.M. dwt m⁻² for the forest and right-of-way; at Minago River 470 and 542 g O.M. dwt m⁻² for the forest and right-of-way; at Devil's Lake 616 - 766 and 592 - 702 g O.M. dwt m⁻² for the forest and right-of-way; at Hadashville 592 - 710 and 1334 - 1372 g O.M. m⁻² for the bog and drained plots.

Comparison of available estimates of net annual primary production of bog vegetation within the right-of-way and the annual estimates of soil organic matter breakdown indicate that due to the reduced primary production in the right-of-way annual breakdown exceeds the amount of new litter added to the soil each year.

Chapter III

**EFFECTS OF PICLORAM AND 2,4-D ON SPHAGNUM FUSCUM
HUMMOCKS AND ASSOCIATED VEGETATION**

3.1 INTRODUCTION

Sphagnum mosses are important components of most bog communities of North Temperate and Boreal zones where they are abundant in surface vegetation and due to their slow decomposition rates contribute significantly to peat formation (Clymo and Hayward 1982). They impede drainage, create an acid habitat and control development and successional trends in bogs (Crum 1983; Luken et al. 1985).

The Sphagnum flora of a bog is usually made up of several species which are distributed along moisture, water chemistry, pH and light gradients (Vitt and Slack 1975, 1984; Andrus et al. 1983). Some Sphagnum mosses are restricted to wet hollows, some are found growing on lower margins of hummocks formed by still other Sphagnum species (Andrus et al. 1983).

S.fuscum is the dominant component of hummocks in open and treed shrub rich bogs of southeastern, central and northern Manitoba (Ritchie 1956, 1960a,b; Zoltai and Tarnocai 1971; Reader 1971; Sims 1976, 1977) as in bogs of other parts of central Canada (Jeglum 1972; Jeglum et al. 1974; Horton et al. 1979; Jasieniuk and Johnson 1982). It is also abundant in bogs of southeastern Canada, the northern United States, Alaska, Scandinavia and western Siberia (Heinselman 1963; Horton et al. 1979; Luken et al. 1985). According to Horton et al. (1979) S.fuscum may be the most

common and abundant component of circumboreal-subarctic peatlands. It generally builds the most extensive and highest hummocks formed by any Sphagnum species, and represents the dry end of the aquatic to upland sphagna gradient.

Sphagnum hummocks are of particular importance in bogs because they create a raised relatively dry substrate which enables other plant species to become established. The hummocks thus develop a characteristic flora which in Manitoba and central Canada commonly consist of Picea mariana, Ledum groenlandicum, Kalmia polifolia, Chamaedaphne calyculata, Vaccinium vitis-idaea, Oxycoccus microcarpus, Rubus chamaemorus, Polytrichum strictum, Pleurozium schreberi, and Cladina lichens (Ritchie 1956, 1960a; Zoltai and Tarnocai 1971; Reader 1971; Jeglum 1972; Sims 1977; Horton et al. 1979).

The hummock or raised communities of bogs are generally considered among the most stable of peatland vegetation types (Jeglum 1973) and not susceptible to surficial disturbances which do not disrupt the underlying substrate. In burned over areas S.fuscum has been shown to regenerate in a few decades and invade burned-out patches resulting in restoration of the pre-fire living Sphagnum substrate with its characteristic community (Jasieniuk and Johnson 1982).

In recent years herbicides, mainly picloram (4-amino-3,5,6-trichloropicolinic acid) and 2,4-D (2,4-dichlorophenoxyacetic acid), have been applied to bog

vegetation in central Manitoba (MacLellan 1982) as a part of a program to control tree regeneration along electric transmission right-of-ways. Herbicides have adverse effects on trees and ericaceous shrubs in peatlands, but Sphagnum species and other mosses have generally been considered resistant to them (Dana 1967; Sims 1976; Suffling 1975; Suffling and Smith 1979).

During field work at the Devil's Lake site in the summer of 1982 it was noted that the living Sphagnum fuscum of most hummocks in the right-of-way had partly or completely disappeared and left extensive patches of underlying S.fuscum peat (Figure 21 a). The herbicide Tordon 101 (picloram + 2,4-D) had been applied in the right-of-way during the summer of 1979 (MacLellan 1982). The observed damage suggested that the application may have caused the disappearance of living S.fuscum from the hummocks.

The primary purpose of this study was to determine the effect of picloram and 2,4-D applications on growth of S.fuscum in the field. Attention was also paid to other prominent species of the hummock community.

Figure 21.

Effects of herbicides on Sphagnum fuscum hummocks at Devil's Lake site.

A) A damaged hummock in ROW, photographed on June 6, 1982. Tordon 101 was applied in the ROW during the summer of 1979. A small patch of the moss has survived on the centre top of the hummock. The hummock top is distanced furthest from the ground-water, which probably was the source of herbicide (picloram) to the hummock. (Size of quadrat on hummock top is 25 x 25 cm).

B) A part of a healthy (untreated) hummock in the forest. Plants seen associated with S.fuscum are Sarracenia purpurea, Empetrum nigrum, Vaccinium vitis-idaea, Ledum groenlandicum, Drosera rotundifolia and Polytrichum strictum. (September 10, 1983).

C) Part of a hummock treated with Tordon 101 on June 8, 1983, and photographed on September 10, 1983. Note deformation, and fading of natural color of Sphagnum shoots. Associated plants have been affected.



3.2 STUDY SITES

Hadashville

A site in the same general area of the vegetation sampling (Chapter 1) and soil respiration (Chapter 2) studies was chosen. A lowlying area and open part of the bog, about 100 m to the east of the peat harvesting field was selected where extensive patches of Sphagnum fuscum form low (< 20 cm high) hummocks (Figure 22 a). The moss Polytrichum strictum is commonly found growing in the hummocks which have low and open shrub layers formed by Kalmia polifolia, Chamaedaphne calyculata, Andromeda glaucophylla, Ledum groenlandicum, Vaccinium vitis-idaea and Oxycoccus microcarpus. The water table in early spring is approximately 30 cm below the hummock surface and recedes down to 50 - 60 cm in the fall.

Devil's Lake

At Devil's Lake field work was carried out within the forest to the east of the right-of-way in the same general area and vegetation as described in Chapters 1 (Devil's Lake 2) and 2. In the forest S.fuscum forms conspicuous hummocks which can cover several square meters areas and may rise 30 - 50 cm above the surrounding hollows. Other species of common occurrence on the hummocks of the site are Polytrichum strictum, Ledum groenlandicum, Vaccinium vitis-idaea, Oxycoccus microcarpus, Empetrum nigrum and Sarracenia purpurea (Figure 21 b). The water table in early spring is 20 - 30

Figure 22.

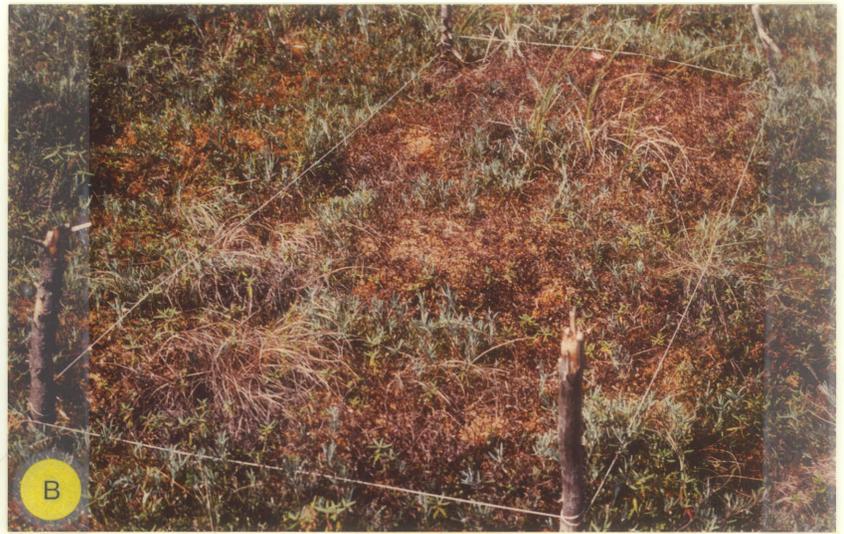
Experimental plots at Hadashville site, photographed August 25, 1983. Herbicides were applied on May 31, 1983.

A) Plots (1 x 2 m in size) viewed from south. Plots from left to right, are: Control, 2,4-D, Tordon 10K and Tordon 101.

B) Plot treated with 2,4-D. Natural color of tops of Sphagnum fuscum has faded and ericoid cover has been reduced. Living plants of Andromeda glaucophylla and Ledum groenlandicum are apparent within the plot.

C) Plot treated with Tordon 101. Near complete kill of Sphagnum fuscum and ericoids has occurred. Carex aquatilis is the only vascular plant unaffected.

D) Plot treated with Tordon 10K. Effects similar as in the plot treated with Tordon 101, except that a few shoots of ericoids remain within the plot.



cm below the surfaces of hummocks and recedes down to about 50 cm in the fall.

3.3 MATERIALS AND METHODS

3.3.1 Experimental design and herbicidal applications

At each study site four plots 1 x 2 meters in size were laid out on hummocks with near complete cover of living S.fuscum. Hummocks similar in vegetation and structure were subjectively chosen for study.

At the Hadashville site the four plots each separated by a distance of 2 meters were laid out in an area of 14 x 1 meters running in a N - S direction (Figure 22 a).

At Devil's Lake where the Sphagnum mat was not as continuous and the plots were more dispersed and located more than 5 meters from one another.

A random selection of plots was made at each site for control and applications of Tordon 10K (picloram), Tordon 101 (picloram + 2,4-D) and 2,4-D.

The herbicides were applied at rates generally used by Manitoba Hydro line maintenance crews for brush control in similar habitats.

Tordon 10K (potassium salt containing 0.1 g picloram per g of product) which is in a solid pellet form, was applied at a rate of 8.96 g to a 2 m² plot (44.8 kg of Tordon 10K per ha, which is equivalent to 0.45 g picloram per m²).

Tordon 101 mixture (contains 62.3 g picloram and 249.2 g 2,4-D per L of product) 10 ml was diluted in 1000 ml of water and 374 ml of the solution sprayed on a plot of 2 m² (18.7 L of Tordon 101 mixture per ha, which is equivalent to 0.12 g picloram and 0.47 g 2,4-D per m²).

2,4-D mixture (amine 80) 4.2 ml was diluted in 500 ml of water and 180 ml of the solution sprayed on a plot of 2 m² (7.6 L of 2,4-D mixture per ha, which is equivalent to 0.38 g 2,4-D per m²).

The liquid herbicides were applied using a one gallon tank sprayer (Continental, Model 050).

At Hadashville the herbicides were applied on May 31, 1983, and at Devils Lake on June 8, 1983.

3.3.2 Chlorophyll determinations

Samples of Sphagnum fuscum were collected from the plots in the 1983 and 1984 seasons. On each sampling date, 4 samples (ca. 8 cm dia x 10 cm long) were removed from random locations within each plot. To minimize disturbance the holes left in the plots after removal of samples were filled in with fresh samples of Sphagnum peat.

Samples were brought moist to the laboratory in plastic bags and stored at room temperature overnight. For chlorophyll extractions, S.fuscum shoots were cleaned and the top 3 cm cut off for extraction (Karunen and Salin 1982). Shoot

samples of 6 - 8 g fresh weight were cut into small segments and mixed. Each sample was then divided into three subsamples and their fresh weight determined. Two of the subsamples were dried at 80° C for 24 hours to determine their dry wt and moisture content. The third subsample (2 - 3 g fresh wt) was ground in a mortar and extracted with 80% acetone (Arnon 1949; Bruinsma 1963; Ferguson et al. 1978). When homogenized the sample was filtered into a 100 ml volumetric flask. The first filtrate was washed into the mortar and homogenization was repeated. The sample was then refiltered to make a total extract volume of 100 ml.

In the extractions done during May - July of 1983 the volumetric flasks were not covered and the chlorophyll extracts were exposed to light during extractions. In the extractions done during August of 1983 or later the flasks were covered with aluminium foil to prevent chlorophyll breakdown by light (Bruinsma 1963).

The concentrations of chlorophyll a and b were determined by measuring in a 1 cm cell the absorbancy (A) of the chlorophyll extracts with a Bausch and Lomb 20 Spectrophotometer at 645, 652 and 663 mu. Concentrations in milligrams per liter were calculated from the following equations (Bruinsma 1963):

$$Ca + b = 20.2 A_{645} + 8.0 A_{663}$$

$$Ca + b = 27.8 A_{652}$$

The average of the two values was taken and the results expressed in mg g^{-1} dry wt of sample.

3.3.3 Sphagnum elongation.

Elongation of S.fuscum plants was measured in the Hadashville experimental plots during the 1984 season. On June 21, 10 cranked wires (Clymo 1970) were installed at 20 cm intervals along the long axis center of each plot. The cranked wire is ca. 17 cm long made of stainless steel. One end of the wire, about 10 cm long is driven vertically into the moss carpet until the horizontal section (1 cm long) is level with the capitula. The free end of the wire, which is of exactly known length projects into the air and the growth of the Sphagnum plants is measured as the difference between the total length of the free end and the length still above the moss surface at any given time. The wires were left in the plots until September 26, when elongation was measured.

3.3.4 Water chemistry.

At the end of the 1984 season, four water samples were collected in random locations within each plot at Hadashville. Water was squeezed from top 10 cm into clean plastic bottles and frozen within 4 hours.

Similar water sampling was carried out at Long Point between towers 1152 and 1152 in the Radisson-Dorsey right-of-

way. The site was treated with Tordon 10K during July of 1982 (Munro 1985, personal communication) which caused extensive damage to S.fuscum hummocks. Water samples were collected from damaged hummocks in the right-of-way and from undisturbed hummocks in the adjacent forest on October 16, 1984.

Later, pH of the water samples was measured with a mini digital pH meter, model 55 (Macalaster Bicknell Comp.) and conductivity determined using a YSI M33 S-C-T meter. The conductivity was measured at 20° C and the values were corrected (Kcorr) by subtracting the conductivity contributed by hydrogen ions (Sjörs 1950).

3.3.5 Statistical analysis.

Comparisons involving treatments only were tested with one-way ANOVA, while comparisons involving both treatments and time were tested with two-way ANOVA. Duncan's multiple range test was used to test difference between treatment means for significance. All data was run under SAS (1979).

3.4 RESULTS

Performance of Sphagnum fuscum was assessed in terms of chlorophyll content (Table 18 and Table 19, Figure 23 and Figure 24) at both sites and in terms of extension in length and water chemistry (Figure 25) at the Hadashville site.

Unlike the 1984 data, chlorophyll content of shoots from the control plots at both sites in 1983 showed considerable differences between sampling dates (Table 18 and Table 19). These differences between the two years are believed to be caused by greater inconsistency in laboratory procedures during 1983. To adjust for this discrepancy the chlorophyll content of the samples from the treated plots was calculated as percentage of the control values (Figure 23 and Figure 24).

The natural color of the plants sprayed with Tordon 101 and 2,4-D faded within a week of spraying. The chlorophyll content of the plants treated with Tordon 101 was significantly reduced from control values on the first sampling date at both sites or to 43 - 60%. In the plots treated with 2,4-D the chlorophyll content was only significantly reduced at the Hadashville site, to 66% of control values.

The solid Tordon 10K herbicide did not have an immediate effect, consequently color loss and chlorophyll reduction did not occur until 3 - 4 weeks after application.

TABLE 18

Herbicidal experiment at Hadashville site

The effect of herbicides on chlorophyll content (mg g^{-1} dwt) of *S.fuscum* (top 3 cm) in experimental plots. Herbicides were applied on May 31, 1983. Standard errors are shown in brackets. Treatment means with the same letter within date do not differ significantly at $p < 0.05$. (N=4).

DATE	CONTROL	TORDON 10K	TORDON 101	2,4-D
3. 6.1983	0.82 a (0.07)	0.76 a (0.04)	0.49 b (0.04)	0.54 b (0.03)
28. 7.1983	0.53 a (0.04)	0.30 bc (0.04)	0.26 c (0.02)	0.38 b (0.01)
25. 8.1983	0.62 a (0.06)	0.40 b (0.09)	0.35 b (0.02)	0.36 b (0.02)
22. 9.1983	0.74 a (0.08)	0.31 b (0.04)	0.28 b (0.03)	0.38 b (0.06)
21.10.1983	0.68 a (0.05)	0.32 b (0.07)	0.26 b (0.02)	0.33 b (0.05)
23. 6.1984	0.88 a (0.05)	0.51 b (0.03)	0.47 b (0.01)	0.61 b (0.08)
16. 8.1984	0.96 a (0.11)	0.65 b (0.02)	0.58 b (0.03)	0.63 b (0.07)
26. 9.1984	0.88 a (0.01)	0.44 c (0.04)	0.62 b (0.09)	0.72 ab (0.04)
1983 total	0.68 a (0.03)	0.42 b (0.05)	0.33 c (0.02)	0.40 b (0.02)
1984 total	0.90 a (0.04)	0.53 b (0.03)	0.55 b (0.04)	0.65 c (0.04)
1983-1984 total	0.76 a (0.03)	0.46 bc (0.03)	0.41 c (0.03)	0.49 b (0.03)

TABLE 19

Herbicidal experiment at Devil's Lake site

The effects of herbicides on chlorophyll content (mg g^{-1} dwt) of S.fuscum (top 3 cm) in experimental plots. Herbicides were applied on June 8, 1983. Standard errors are shown in brackets. Treatment means with the same letter within date do not differ significantly at $p < 0.05$. (N=4).

DATE	CONTROL	TORDON 10K	TORDON 101	2,4-D
10. 6.1983	0.45 a (0.03)	0.35 a (0.02)	0.19 b (0.02)	0.42 a (0.08)
26. 6.1983	0.64 a (0.13)	0.46 ab (0.12)	0.33 b (0.03)	0.34 b (0.02)
10. 9.1983	0.85 a (0.13)	0.44 b (0.09)	0.28 b (0.03)	0.40 b (0.03)
19. 6.1984	0.89 a (0.05)	0.57 b (0.06)	0.61 b (0.07)	0.59 b (0.04)
21. 8.1984	0.86 a (0.09)	0.78 a (0.08)	0.64 a (0.06)	0.74 a (0.05)
16. 9.1984	0.86 a (0.05)	0.50 b (0.03)	0.52 b (0.01)	0.52 b (0.08)
1983 total	0.64 a (0.08)	0.41 b (0.05)	0.27 c (0.02)	0.39 b (0.03)
1984 total	0.87 a (0.04)	0.62 b (0.03)	0.59 b (0.04)	0.62 b (0.04)
1983-1984 total	0.76 a (0.05)	0.451bc (0.04)	0.413c (0.04)	0.450b (0.03)

Figure 23.

Herbicidal experiment at Hadashville site.
Chlorophyll content of top 3 cm of Sphagnum fuscum
from plots treated with herbicides on May 31, 1983.

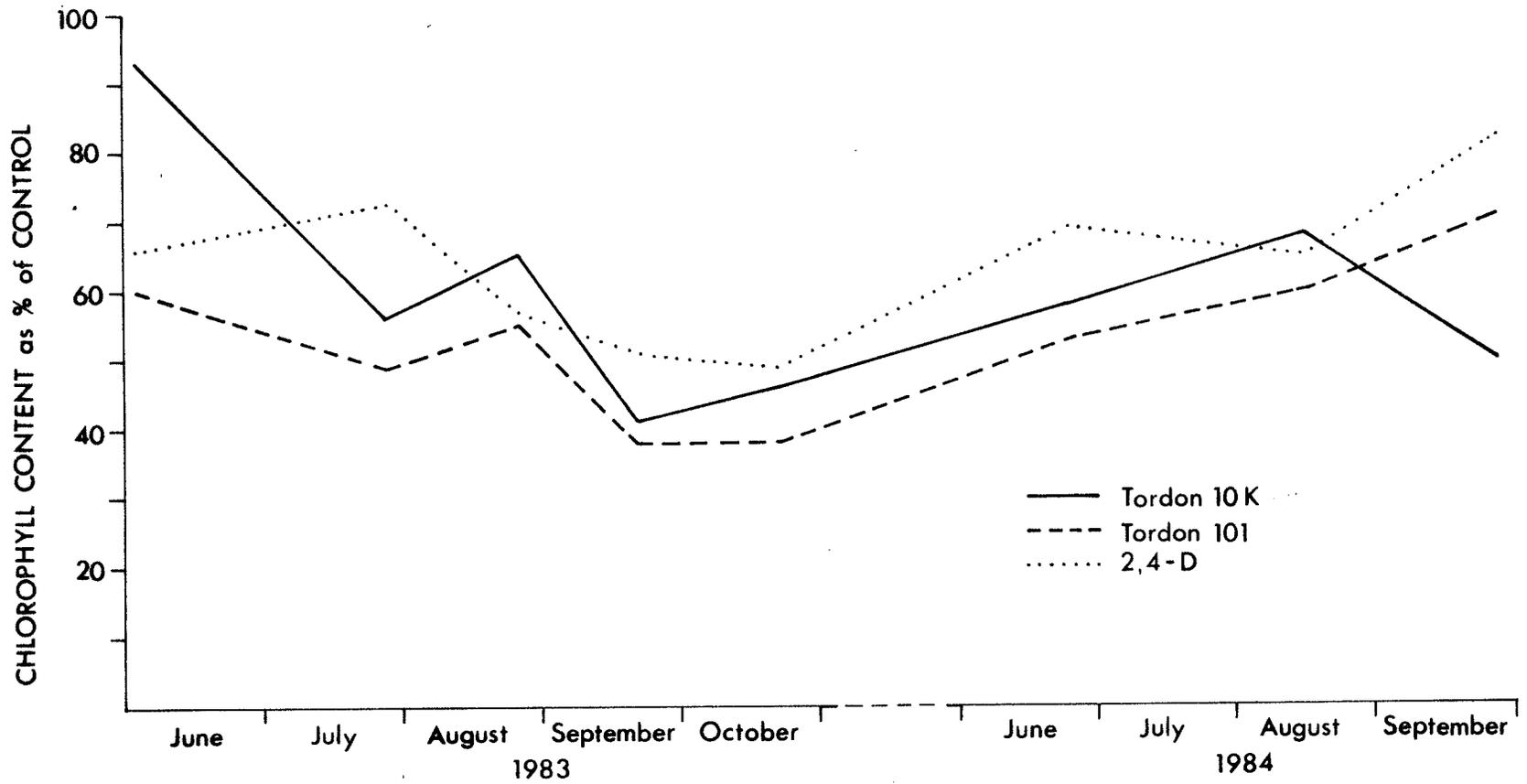


Figure 24.

Herbicidal experiment at Devil's Lake site.
Chlorophyll content of top 3 cm of Sphagnum fuscum
from plots treated with herbicides on June 8, 1983.

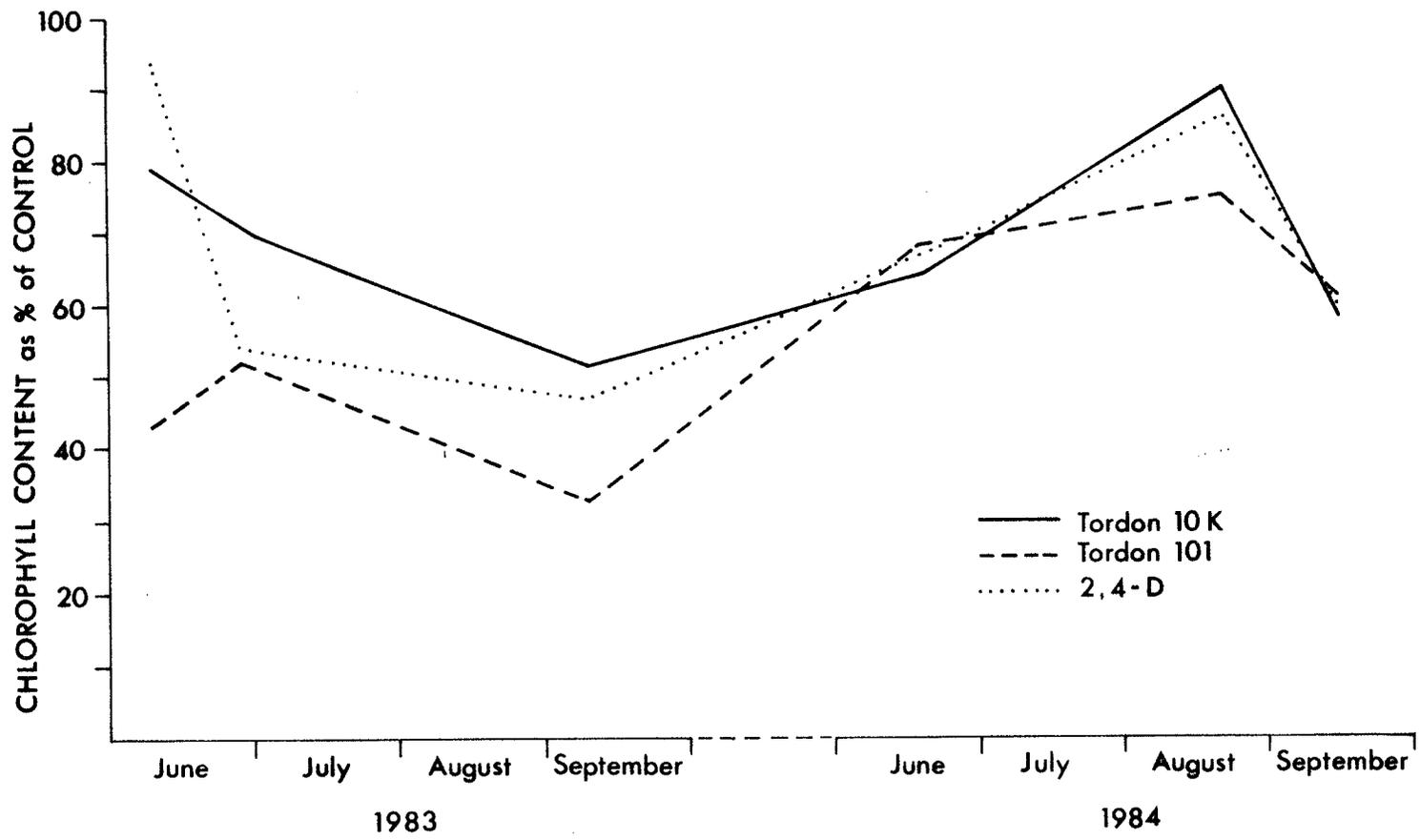
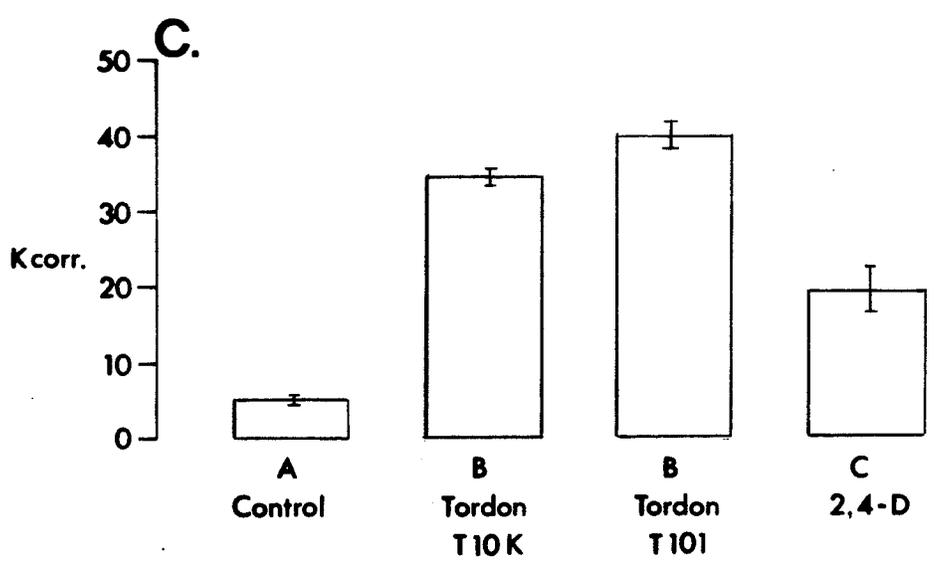
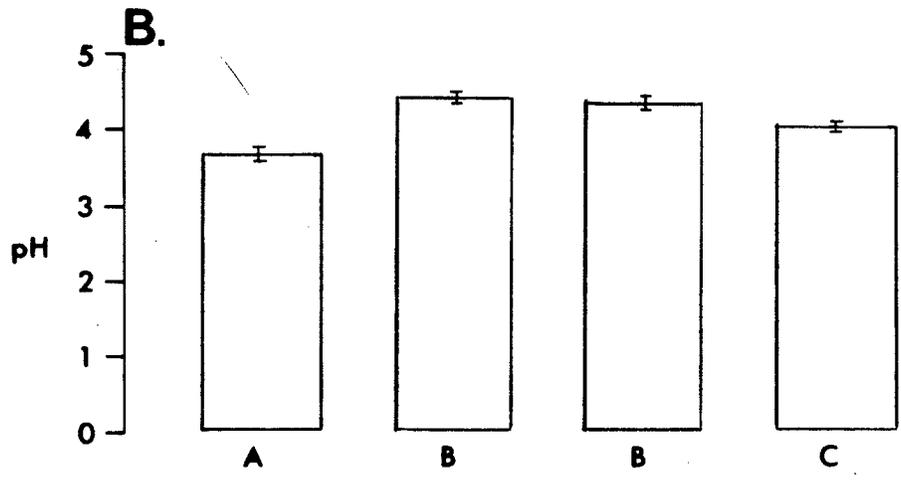
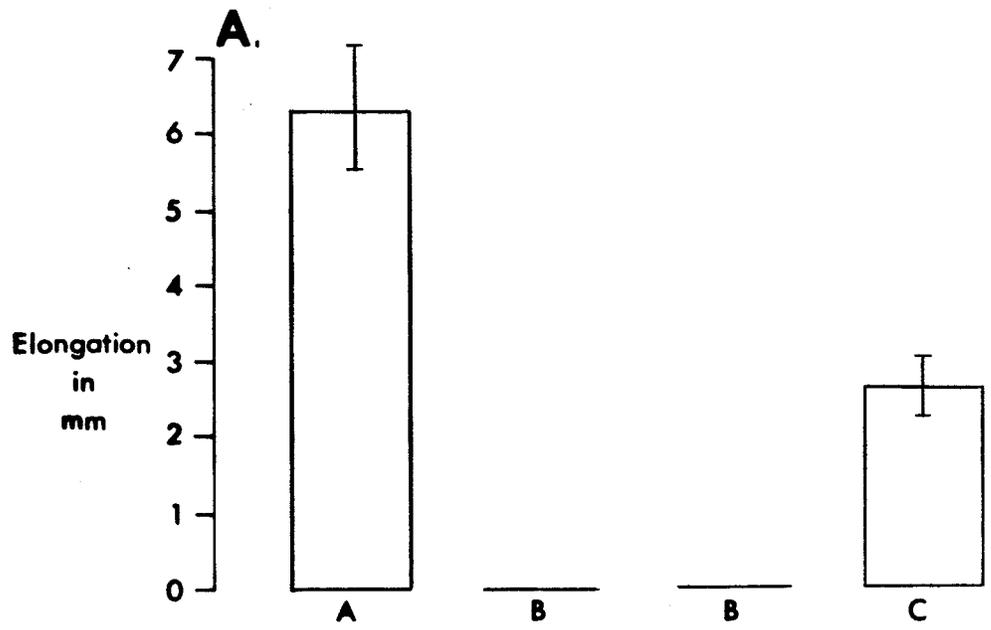


Figure 25.

Herbicidal experiment at Hadashville site.
Effects of herbicides on Sphagnum fuscum growth.
Herbicides were applied on May 31, 1983. Means with
the same letter do not differ significantly at
 $\alpha = 0.05$.

- A) Elongation of shoots between June 21, and September
26, 1984.
- B - C) pH and conductivity ($K_{corr} \text{ umho cm}^{-1}$) of water
samples from top 10 cm of shoots collected on
October 18, 1984.



In all treated plots at both sites, the chlorophyll content of shoots shows a declining trend during the 1983 season. At the end of the season it has been reduced to 33 - 52% of control values. The average seasonal chlorophyll content in the plots treated with Tordon 101 is significantly lower than in the plots treated with Tordon 101 or 2,4-D.

The appearance of the tops of S.fuscum plants in the herbicidal plots in the fall of 1983 indicated that at least the top 1 - 2 cm of the shoots had died following the herbicidal applications. The tops of the shoots became bent and frequently withered (Figure 21 c). The color of the capitula and top branches turned to light-gray and some of the shoots lost their capitula. Damaged shoot-tops did not regain their former appearance. It was noted in the samples collected between August and October of 1983 in the plot treated with only 2,4-D at Hadashville that the S.fuscum plants showed signs of recovery. In a few cases new green shoots (1 per stem), originating from the old stem 2 - 3 cm below the damaged top appeared (Figure 26 b). The new shoots did not grow much in 1983. They were generally shorter than 5 mm in length, had a diameter of 1 - 2 mm and did not extend up to the surface of the Sphagnum mat.

The declining trend in chlorophyll content of S.fuscum shoots from the treated plots in 1983 was reversed in the 1984 season, when the chlorophyll content increased again.

At Hadashville the chlorophyll content of shoots from the plots treated with Tordon 10K and Tordon 101 rose to near 70% of control levels, but for all sampling dates is significantly lower than in the control plot. In the plot treated with 2,4-D the chlorophyll content was significantly lower than in the control plot on the first two sampling dates. On the last sampling date the chlorophyll content had reached 82% (Figure 23) of the control level, and was then not significantly different (Table 18).

The average seasonal chlorophyll content of shoots from the plot treated with 2,4-D was significantly higher than values from Tordon 10K and Tordon 101 treatments, but it was significantly lower than the the control (Table 18).

At Devil's Lake the chlorophyll content of shoots from plots treated with the herbicides rises in 1984 up to 75 - 90% of control levels (Figure 24). In the treated plots the chlorophyll contents are significantly lower than in the control plot on the first sampling date, but on the second there is no significant difference between the two plots (Table 19).

On the last sampling date the chlorophyll content of shoots from the treated plots decreases from the previous sampling date while that in the control plot remains the same. Again there is a significant difference between the control and treated plots. This consistent decline in chlorophyll content in the shoots from the treated plots was

most likely caused by the Tordon 10K which was applied in the adjacent right-of-way in June, 1984. The herbicide was later apparently carried into the study area. In September the vegetation within and around the study plots showed signs of having been affected by the herbicide. The vegetation within the control plot was also affected, but no decline in chlorophyll content of S.fuscum shoots was detected.

The seasonal average chlorophyll content of shoots from the treated plots at the Devil's Lake site during 1984 was significantly lower than in the control plots. Differences between the treated plots were not significant (Table 19).

In 1984 at both study sites the Sphagnum fuscum treated with only 2,4-D showed a considerable recovery (Figure 26 b). Over 90% of the damaged shoots grew new tops with capitula and had by the end of the 1984 season, nearly regained their former appearance.

During 1984 some new growth, comparable to that observed in 1983 on the plants from the 2,4-D plot at Hadashville, was noticed on plants from both sites collected in the plots treated with Tordon 10K and Tordon 101. However new growth was observed on very few plants (< 10%) at the end of the 1984 growing season.

At both sites the damaged tops of S.fuscum in the plots treated with Tordon 10K and Tordon 101, became a substrate

Figure 26.

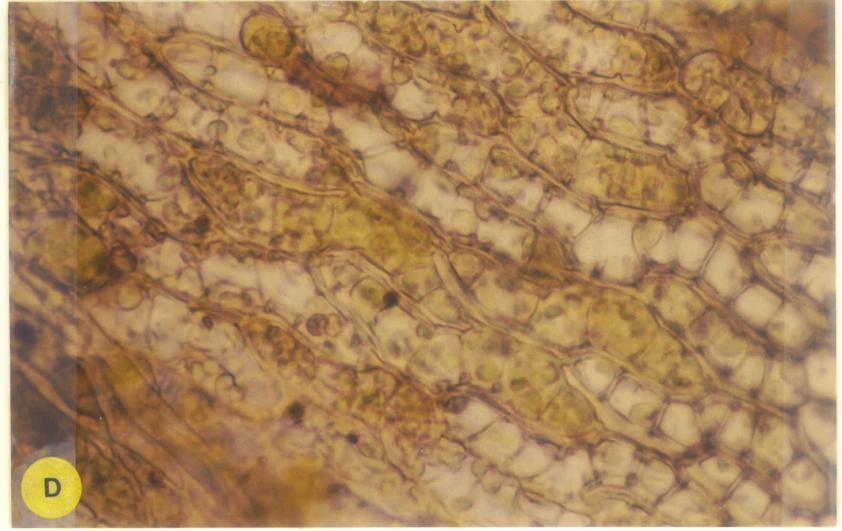
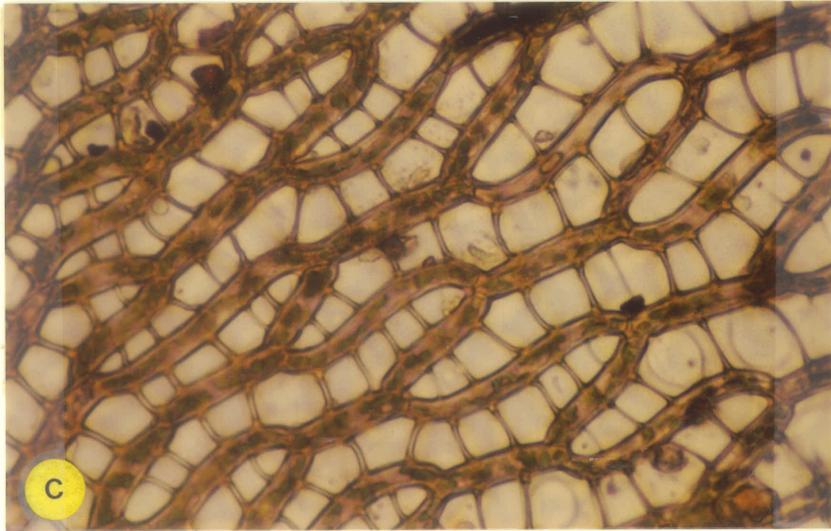
Sphagnum fuscum from experimental plots at Hadashville site.

A) A top part of a healthy plant from the untreated plot. The shoot has a normal growing tip (capitulum). x 5. (Collected, September 26, 1984).

B) A top part of a plant from a plot treated with 2,4-D on May 31, 1983. The growing shoots of the plants were damaged following the herbicidal application, but new growing shoots formed from the stem below the damaged portion. On the photograph the damaged shoot is on the upper left and the new growing shoot to the right. x 5. (Collected, June 22, 1984).

C) Enlargement (x 510) of a branch leaf from a top shoot of a plant growing in the untreated plot. Chloroplasts can be seen within the narrow chlorophyllose cells, while hyaline cells appear relatively clear. (Collected, September 26, 1984).

D) Enlargement (x 510) of a branch leaf from a damaged top of a plant collected in the plot treated with Tordon 10K on May 31, 1983. The plants treated with Tordon 10K and Tordon 101 did not form new growing shoots, but became contaminated with coccoid green and blue-green algae (here seen within the hyaline cells). The algae were most likely the main source of chlorophyll in the shoots during the 1984 season. (Collected, September 26, 1984).



for growth of coccooid green and blue-green algae during 1984 (Figure 26 d). Microscopic observations of leaves from damaged shoots revealed that the algae were growing on the shoots and also inside the hyaline cells. Plants from the control and 2,4-D treated plots were also checked for algal growth, but none was found (Figure 26 a). The increase in chlorophyll content of shoots from the plots treated with Tordon 10K and Tordon 101 observed in 1984 is attributed to the algal growth on the damaged plants as very few plants acquired any new growth during the 1984 season.

The recovery of the plants in the plots treated with 2,4-D relative to the plants in the plots treated with Tordon 10K and Tordon 101 is further demonstrated in the results of the measurements of elongation and water chemistry of S.fuscum at the Hadashville site in 1984 (Figure 25). In the control plot an average elongation of 6.3 mm was observed from June 21 to September 26, while the elongation of the plants in the 2,4-D treated plot was 2.6 mm. In the Tordon 10K and Tordon 101 treated plots no measureable elongation of shoots occurred.

The water from shoots in the control plot had the lowest average pH (3.66) and conductivity (Kcorr = 5.0). In the plot treated with 2,4-D, pH (3.95) and conductivity (Kcorr = 18.8) were significantly higher than in the control plot, but however significantly lower than in the plots treated with Tordon 10K (pH = 4.34, Kcorr = 34.0) and Tordon 101 (pH = 4.30, Kcorr = 39.3)

The analysis of the water samples collected at the Long Point site gave similar results (Figure 27). The average pH (4.07) and conductivity (Kcorr = 31.4) of samples collected from undisturbed Sphagnum fuscum hummocks in the forest were significantly lower than pH (4.61) and conductivity (Kcorr = 112.8) of water from plants collected in hummocks in the right-of-way damaged by the Tordon 10K application during 1982.

On a visit to the Hadashville site at the end of the third season (October 1985) it was observed that the plants treated with picloram (Tordon 10K and Tordon 101) had not recovered any further than was observed in the previous year and over 95% of the plants were permanently destroyed. The plants treated with 2,4-D had generally the same appearance and cover as the plants in the control plot and seemed to have recovered completely from the application (Figure 28).

Figure 27.

pH and conductivity (Kcorr $\mu\text{mho cm}^{-1}$) of water samples collected from Sphagnum fuscum hummocks at Long Point site on October 16, 1984. Tordon 10K was applied in the right-of-way at the site during July of 1982.

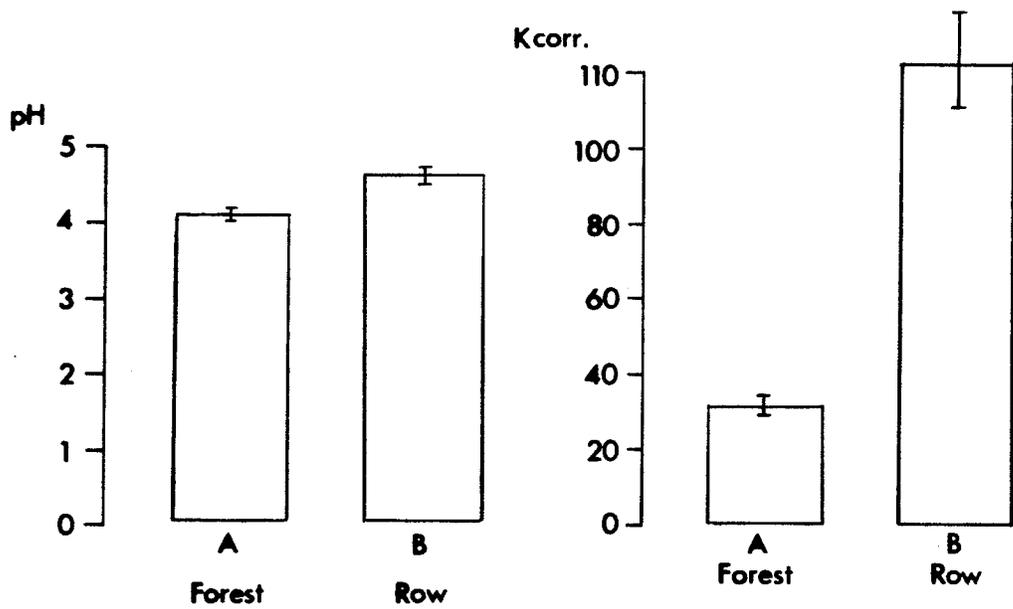


Figure 28. Moss layer in experimental plots at Hadashville site.

Samples of moss layer were collected from experimental plots at Hadashville on October 24, 1985 and photographed in the laboratory.

A) Control plot. Sphagnum fuscum forms a continuous layer in the plot, which contains scattered plants of Polytrichum strictum.

B) Plot treated with 2,4-D on May 31, 1983. Sphagnum fuscum has recovered and formed new growing shoots. Abundance of Polytrichum strictum has increased slightly.

C) Plot treated with Tordon 10K on May 31, 1983. S.fuscum has completely disappeared and Polytrichum strictum has gained dominance in the moss layer.

(The same change occurred in the plot treated with Tordon 101).



3.5 DISCUSSION

3.5.1 Sphagnum growth

This study has shown that S.fuscum is sensitive to picloram and 2,4-D applications, and that at least the picloram applications may cause irreversible damages of these non-targeted plants. In view of the results, it is likely that the disappearance of living S.fuscum from hummocks in the right-of-way at Devil's Lake observed in 1982, was caused by the picloram ingredient of Tordon 101 applied at the site in 1979. The sensitivity to picloram is also supported by the observation in the right-of-way at the Long Point site, where the application of Tordon 10 K during the summer of 1982 was followed by extensive disappearance of living S.fuscum. Due to the difference in concentrations of picloram and 2,4-D between treatments, it can however not be ruled out that the irreversible damages of plants treated with Tordon 101, were partly caused by the 2,4-D ingredient. In the Tordon 101 treatment 2,4-D concentration (0.47 g m^{-2}) is higher than in the 2,4-D treatment (0.38 g m^{-2}), and then the picloram concentration (0.12 g m^{-2}) is considerably lower than in the Tordon 10K (0.45 g m^{-2}) treatment. To answer this it would have been necessary to carry out further tests with different concentrations.

The similarity in initial effects of the herbicides on but later apparent differences can be explained by the known behaviour of picloram and 2,4-D. Picloram has been classi-

fied as an auxin type of herbicide similar to 2,4-D. The distortions of growing stems and leaves in plants treated with picloram are like those produced by 2,4-D. Both herbicides are toxic to broadleaved plants while their effects on graminaceous plants are moderate. In dicotyledonous plants 2,4-D and picloram have been found to inhibit photosynthesis and cause loss of chlorophyll, and interfere with nucleic acid metabolism and protein synthesis (Sharma and Born 1972; NRCC 1974; Loos 1975; Foy 1976). Picloram is a more effective herbicide than 2,4-D, because of its mobility and resistance to breakdown within plants and soils. Half-lives of 2,4-D disappearance in agricultural soils are generally less than 7 days (Ou 1984) and it should not persist in the soil for more than few weeks following application (Smith 1982). On the other hand picloram is resistant to breakdown within plants and by soil microorganisms (NRCC 1974; Loos 1975). In soils the degradation of picloram is microbiological in nature (Meikle et al. 1973) and reported estimated half-lives of picloram disappearance vary from 2 weeks to 19 months depending on conditions (NRCC 1974). In Canadian studies 32 - 69% of picloram applications have been estimated remaining in soils after one year, and 10 - 14% after two years (Smith 1982). Sphagnum moss from a bog in Wisconsin treated with picloram carried toxic residues for three growing seasons after application (Dana 1967). In contrast 2,4-D exerted no, or very temporary, effect and no evidence of the herbicide was found when tests were made a year after application.

In the environment of the S.fuscum hummocks the phytotoxicity of picloram and 2,4-D is likely to have been affected differently by the low pH level. Corbin et al. (1971) found that in highly organic soils the phytotoxicity of 2,4-D decreased significantly with decreasing soil pH from 7.5 to 4.3. Maximum phytotoxicity for 2,4-D was observed at pH 6.5. Increasing acidity had no significant effect on phytotoxicity of picloram but there was a trend to increasing toxicity with lower pH levels.

Although the growing apex of the S.fuscum plants treated with 2,4-D only was destroyed as in the plots treated with picloram, the herbicide, at the concentration applied, apparently did not damage or persist long enough to suppress the lower parts of the plants which were able to produce new shoots.

The individual leaves of Sphagnum are known to die when they fall into the dense shade created by the growth of branches above them. The axillary buds of stems, although remaining inactive in the shade, do not die. If the growing apex of shoots is destroyed, then the axillary buds, as deep as 10 cm below the surface, may grow into new lateral shoots. Lateral buds have been shown to tolerate at least an 18 month long period of darkness at 2° C (Clymo and Hayward 1982).

The failure of the plants treated with picloram to recover from the applications during the study may have been caused by slow breakdown of the herbicide and persistence of residues in the peat. Thus preventing regeneration from the axillary buds below the destroyed shoot tops. Observations of Sphagnum hummocks at the Devil's Lake site 4 - 5 years after application of Tordon 101 indicate that S.fuscum is not capable of regenerating from old shoots within damaged patches. It is therefore likely that if the axillary buds are not destroyed shortly after the herbicidal application, the long phytotoxic persistence of picloram will eventually outweigh the ability of the plants to survive without the photosynthetic apices.

The chlorophyll content (approx. $0.9 \text{ mg g}^{-1} \text{ dwt}$) of the top 3 cm S.fuscum shoots from the control plots in this work was low compared to other reports of chlorophyll content in Sphagnum shoots. This low chlorophyll content probably reflects a relatively low seasonal elongation of Sphagnum shoots at the sites compared to findings of other studies (Pakarinen 1978; Karunen and Salin 1982; Grigal 1985), and a concentration of the chlorophyll in the top 1 - 2 cm of the shoots.

At both study sites, the effects of Tordon 101 on chlorophyll reduction were more immediate and stronger than of 2,4-D or Tordon 10K, i.e. the mixture of the two herbicides caused greater damage than when they were applied separate-

ly. At the end of the 1983 season chlorophyll content of the plants treated with Tordon 101 was 33 - 38% of control levels, while that of the plants treated with 2,4-D or Tordon 10K was 47 - 49% and 46 - 52% of control levels respectively. Although the differences between treatments were not significant they may indicate that the effects of the two herbicides were synergistic. From studies of vascular plants it has been reported that the effects of the two herbicides when applied together can be additive (NRCC 1974; Foy 1976). It must again be kept in mind that the concentration of picloran and 2,4-D varied for the different treatments, which makes generalizations about synergistic effects questionable.

In 1983 the S.fuscum shoots were not checked for algal growth and it is possible that some algal chlorophyll was in the extracts from the late season. A number of blue-green and green algal species are known to be associated with Sphagnum species (Clymo and Hayward 1982; Brown 1982). The algae can be found both on the surface of the plants and inside the hyaline cells of the leaves. Algal associations and activity is particularly common on Sphagnum growing in flushed and minerotrophic habitats, but may be absent or rare in hummock species.

In this work no algal growth was observed during 1984 on the growing shoot tops from the control and 2,4-D treatment plots. It has been suggested that the lack of blue-green

algae at the apex of Sphagnum is caused by unfavourable substrate conditions, e.g. rapidly dividing and growing tissues, production of antimicrobial substances, and also by the high acidity in hummock communities (Brown 1982).

With the damage and lack of regeneration of the growing apex in the shoots treated with picloram the conditions inhibiting algal growth on the plants are reversed as can be seen on the material sampled in 1984. The plants are no longer able to create new cation exchange sites necessary to maintain a highly acid environment around the shoots (Clymo 1967; Clymo and Hayward 1982). Reduced cation exchange capacity and probable release of nutrients from destroyed and decomposing shoots causes a significant increase in the pH and mineral content (Kcorr) of the water in and around the shoots. The algae are able to take advantage of these changed conditions and colonize the shoots in the plots treated with picloram (Tordon 10K and Tordon 101). The return of growth during 1984 of the plants treated with 2,4-D only brings substrate conditions in the top shoots back towards those in the untreated plants and algal colonization of the shoots does not occur.

Natural populations of blue-green algae and green algae in soils were little affected by picloram, 2,4-D and picloram-2,4-D mixtures at field application rates, over an 18-month period (Arvik et al. 1971).

Knowledge of the effects of herbicides on mosses is very limited. Numerous moss species have however been shown to be sensitive to atmospheric pollution, which can give rise to structural damages and inhibit essential metabolic functions of the plants (Inglis and Hill 1974; Simola 1977; Ferguson et al. 1978). In Britain there is evidence of that the recent disappearance of Sphagnum from vast areas in the southern Pennines has been caused by sulphur pollutants (Ferguson et al., Clymo and Hayward 1982).

Little or no attention has been paid to mosses in plant communities, which have been treated with herbicides (see e.g. Tschirley 1969; Westing 1971; Malone 1972; Tomkins and Grant 1974; Norris et al. 1982; Nickerson and Thibodeau 1984). This lack of information is most likely due to the low abundance of mosses in plant communities where herbicides are used, but may also reflect a resistance of mosses to many herbicides. Then the input of herbicides to plant communities is not continuous as in the case of many air pollutants and sensitive species may therefore be set back only temporarily following applications of herbicides.

Various herbicides, including picloram and 2,4-D, have been tested in a peat bog in Wisconsin to reduce encroachment of woody species and facilitate production of Sphagnum magellanicum which is harvested in the area (Dana 1967). The moss was tolerant of all treatments applied (among them applications of picloram (4.4 kg per ha = 0.44 g m⁻²), but

woody species were effectively controlled by the picloram applications.

Mosses were not adversely affected by an application of Tordon 101 (9.4 L Tordon 101 mixture per ha = 0.06 g picloram and 0.23 g 2,4-D m⁻²) in a power line right-of-way in Ontario (Suffling 1975; Suffling and Smith 1979), but picloram residues were found in living mosses. Polytrichum commune and P. juniperinum increased in abundance following the herbicidal application. Standing crop of other mosses, including Sphagnum capillaceum (= S. nemoreum) and S. palustre was not affected by the herbicidal application. The application rate of Tordon 101 was however only half of that in the present study.

The sensitivity of S. fuscum to picloram found in this study, but tolerance of S. magellanicum (Dana 1967), S. capillaceum and S. palustre (Suffling 1975) reported in other studies may be due to: a) differences in concentrations of the herbicides applied, b) physiological differences between the species, c) be caused by habitat and environmental differences affecting the behaviour of the herbicide.

S. fuscum and S. nemoreum are both hummock species and are typically found at the dry end of the moisture gradient in bogs (Horton et al. 1979). S. magellanicum has more of an intermediate position and occupies hummock sites and lawns (Andrus et al. 1983; Crum 1983). S. palustre is a species

which occurs under still wetter bog conditions, and it is not found in the continental areas of Western Canada (Horton *et al.* 1979; Ireland *et al.* 1980). With the change of Sphagnum species along the environmental moisture gradient there are associated changes in structural characteristics (Hayward and Clymo 1982) and biochemical composition (Clymo and Hayward 1982) of the species to which particular plant functions, e.g. cation exchange capacity and water uptake, have been correlated. In the case of cation exchange capacity it has been difficult to determine if differences in efficiency between species are chiefly caused by physiological differences or by differences in external physico-chemical factors (Aulio 1982). The same may apply to the tolerance of different Sphagnum species to herbicides.

Contrary to the findings of Suffling (1975), S.nemoreum was quite sensitive to picloram in a laboratory test (Magnusson and Stewart, unpublished data) and the species was affected by the herbicide in the same manner as S.fuscum which was grown and treated identically. This indicates that tolerance of Sphagnum species to picloram may be affected by environmental conditions not yet promulgated.

The field conditions affecting Sphagnum tolerance to picloram, which may have differed the most between this study and the Wisconsin (Dana 1967) and Ontario (Suffling 1975) studies is the moisture regime. As picloram is highly soluble in water its direction of movement in the soil is gov-

erned by the mass flow of water which serves as the carrier (NRCC 1974). The activity of picloram is affected by soil moisture content, with decreasing moisture content the bioactivity increases as the concentration of the herbicide in the soil-water phase increases (Grover 1970). At Devil's Lake and Hadashville the water table in the spring was 20 - 30 cm below the hummock tops and receded 30 - 40 cm during the course of the summer. In the hummocks, net movement of water has therefore been upward which may have prevented leaching and caused concentration of the herbicide in the tops of the plants. Picloram has been shown to move readily upwards in soil columns under subirrigated conditions (Grover 1973).

At the Wisconsin site, where water table was slightly above the bog surface in the spring and 5 - 20 cm below the surface in late summer (Dana 1967), S. magellanicum is under wetter conditions than S. fuscum at Devil's Lake and Hadashville. It can be assumed that at least S. palustre was growing under relatively wet conditions at the Ontario site. It is uncertain if S. capillaceum was growing under comparable moisture conditions as S. fuscum at the present site, but it may be pointed out that annual precipitation, 787 mm (Suffling et al. 1974), in the Ontario study area is higher than annual precipitation, 460 - 560 mm (Smith 1975), in southeastern and central Manitoba. It is therefore possible that the moisture regime of S. fuscum hummocks at Devil's Lake and

Hadashville has led to that picloram concentrations in the living shoots reached higher levels there than under the wetter conditions of the other sites, and contributed to that damages noted for S.fuscum. However further studies are needed to determine the importance of herbicide concentrations, species physiological differences and environment in affecting their responses to picloram.

3.5.2 Effects of herbicides on the hummock plant community

The herbicides caused changes in the vegetation on the hummocks at Devil's Lake and Hadashville. The low woody shrubs were affected by 2,4-D in the first year, but with the exception of Kalmia polifolia, suffered less than 50% kill (Figure 22 b). In the second year the plants recovered to 70 - 80% of their original abundance. All Kalmia polifolia was killed following the 2,4-D application, but the plants began to recover in the second year and by fall had 20% of their original abundance. By the end of the third season at Hadashville it was apparent that the herbicide would not cause long term reduction in abundance of any species. Dana (1967) found 2,4-D to exert no or very temporary effects on woody shrubs in a Sphagnum bog.

The picloram applications had more dramatic effects than the 2,4-D applications on the low woody shrubs at both sites. In the plots treated with Tordon 101, complete kill of all woody shrubs occurred in the first season and over

90% in the plots treated with Tordon 10K (Figure 22 c and d). By the second season the species were estimated to have been reduced by near 95% in abundance. A few living shoots of Kalmia polifolia, Chamaedaphne calyculata, Andromeda glaucophylla, Vaccinium vitis-idaea, Oxycoccus microcarpus and Ledum groenlandicum were found in plots treated with picloram. No further recovery of the shrubs was noticeable after the third season at Hadashville.

Other peatland species which were noted to be damaged by picloram applications either in the study plots or in right-of-way areas were: Picea mariana, Larix laricina, Betula glandulosa, Salix spp., Pinus banksiana, Thuja occidentalis, Juniperus horizontalis, Empetrum nigrum, Sarracenia purpurea and Geocaulon lividum. The sensitivity of numerous peatland plants to picloram observed in this study is consistent with the knowledge of the herbicide as being highly toxic to a variety of woody and herbaceous broadleaf species (Dana 1967; NRCC 1974; Suffling 1975).

One member of the hummock community Polytrichum strictum increased in abundance following the herbicidal applications. Browning of the top leaves of the moss appeared both in the plots treated with 2,4-D and picloram for a few weeks after the applications but the plants were not killed and regained normal appearance later in the first season. Cover of P. strictum in the hummocks prior to application of herbicides was of 5 - 10%. During the first season the species

did not appear to increase in abundance, but a notable increase occurred in all treated plots during the second season. At Hadashville, where changes were monitored more closely than at Devil's Lake, the species increased its cover to approximately 15% in the plot treated with 2,4-D and to over 20% in the plots treated with picloram. At the end of the third season at Hadashville, the species had increased its cover to over 80% in the plots treated with picloram, but in the plot treated with 2,4-D, S.fuscum had recovered but Polytrichum had not increased in abundance from the previous season (Figure 28 c). A similar proliferation of Polytrichum mosses following application of Tordon 101 was reported by Suffling (1975) and Suffling and Smith (1979) who attributed the increase to the removal of the dominant shrub above the mosses. The success of P.strictum in the plots treated with picloram where S.fuscum was nearly wiped out, but failure to increase in abundance in the plot treated with 2,4-D after recovery of S.fuscum, indicates that the moss was stimulated by changes in substrate conditions, e.g. increased substrate stability or nutrient release, rather than by removal of the shrub canopy which was sparse before the application.

Polytrichum strictum is an opportunistic species which has a wide distribution in bogs (Andrus et al. 1983). It has low abundance in actively growing hummocks but may become dominant on old, relatively dry or disturbed hummocks (Jasieniuk and Johnson 1982; Crum 1983)

Picloram has been used as a brush killer in the Radisson-Dorsey right-of-way since 1976. The Tordon 101 formulation was used until 1980, but since 1981 only Tordon 10K has been applied (MacLellan 1982; Munro 1985, personal communication). By 1985 picloram had been applied at least once on the southern two thirds of the right-of-way (Tower # 670 - 2006) excluding farmland. Herbicide control tends to be maintained on a five to seven year cycle depending on the effectivity of the applications (Sims 1977).

Picloram is not as effective for controlling tree growth in bogs as in some upland communities, since black spruce is relatively tolerant of the herbicide and is able to recover vegetatively following applications (Suffling 1975). The application of Tordon 101 in the right-of-way in the Devil's Lake area during 1979 was for example not very effective and application was repeated during the summer of 1984 with Tordon 10K.

The use of picloram as a primary brush killer in the Radisson-Dorsey right-of-way and other power line right-of-ways in Manitoba is likely to be continued in the years to come and will conceivably expand into more northern areas where bog vegetation is abundant. Continued and repeated applications of the herbicide at bog sites are likely to result in reduction or disappearance of living S.fuscum from the bogs and gradual destruction of the hummock habitat and its associated plant community. The moss requires longer periods to

recover (Jasieniuk and Johnson 1982) than the average periods between applications. Without S.fuscum, the primary peat building plant and with reduced abundance of low wood shrubs, the production of decay-resistant litter will be reduced (Reader and Stewart 1972; Suffling and Smith 1979). Peat forming conditions are therefore likely to be changed resulting in net breakdown of peat in the hummocks, which will start to degenerate.

A long term change would lead to the reduction in relative proportion of hummock areas in the right-of-way and subsequent increase in wetter hollow or lawn areas, with fen type vegetation (Jeglum et al. 1974) dominated by sedges and grasses, resistant to picloram. The graminoids would not only be favoured by the increase of moist habitat but also by the elimination of susceptible species with their competitive effects (Tomkins and Grant 1977). Studies from both upland and wetland communities have demonstrated that picloram applications stimulate growth and proliferation of graminoids within the season of application and can induce pronounced long term effects on increased graminoid abundancies in treated areas (Tomkins and Grant 1977; Suffling and Smith 1977; MacLellan 1982; see also Chap.1)

The exposure of Sphagnum peat and reduction in shrub cover following broadcast picloram applications may favour regeneration of Picea mariana in the right-of-way. Bare Sphagnum surfaces are known to provide good sites for germination

and seedling establishment (Jeglum et al. 1974). In Ledum-type bogs, dense shrub cover can suppress tree regeneration (Jeglum et al. 1974) and reduction of shrub cover may therefore open the way for reestablishment of trees. The potential use of shrubs as a stable cover in vegetation management to resist tree invasion, as an alternative to repeated broadcast applications of herbicides, has been demonstrated (Niering and Egler 1975; Niering and Goodwin 1974; Egler 1975; Dreyer and Niering 1986). More attention should be paid to these alternatives in future considerations of vegetation management of bogs in the right-of-way.

The notion that repeated picloram applications can adversely affect and possibly destroy the S.fuscum hummock habitat and its associated plant community in bogs is certainly not in line with the impression of bogs as stable habitats with a plant community capable of recovering relatively quickly after catastrophic events such as fire (Jasieniuk and Johnson 1982) and right-of-way construction and maintenance, including herbicidal use (Nickerson and Thibodeau 1984). The potential of picloram applications to cause more adverse effects in bogs in Manitoba than elsewhere is because the key plant species, Sphagnum fuscum, is eliminated. The species is an integral part of and essential in the making and maintenance of the hummock or raised habitat of the bogs.

3.6 SUMMARY AND CONCLUSIONS

Herbicidal applications damaged the growing apex and induced a significant reduction in chlorophyll content of S.fuscum at both sites. The liquid herbicides, Tordon 101 and 2,4-D had more immediate effects than the solid herbicide Tordon 10K. At both sites Tordon 101 caused significantly greater seasonal reduction in chlorophyll content than 2,4-D (singly) and Tordon 10K.

During the second season chlorophyll content of shoots increased in all the treated plots and reached 71 - 75%, 81 - 82% and 68 - 90% of control level for Tordon 101, 2,4-D and Tordon 10K respectively. The plants treated with 2,4-D recovered during the season after the herbicidal application and it was estimated that over 90% of the shoots formed new growing apices. The increase in chlorophyll content of plants treated with 2,4-D (singly) was therefore attributed to new growth. The plants treated with Tordon 101 and Tordon 10K showed negligible recovery, but the increase in chlorophyll was attributed to growth of algae on the damaged shoots.

Elongation of S.fuscum shoots over a 3 month period at Hadashville was 6.3 mm for untreated shoots and 2.6 mm for the shoots treated with 2,4-D (singly). The plants treated with Tordon 101 and Tordon 10K did not elongate.

Both pH and conductivity of water sampled from the plants showed the same trend: Control < 2,4-D < Tordon 101 =Tordon 10K. Differences between samples from control and treated plants were significant, and between samples from plants treated with 2,4-D and the two Tordon herbicides, which did not differ amongst themselves.

At Hadashville at the end of the third season a visual estimate indicated that plants treated with 2,4-D had completely recovered from the application, but the plants treated with the Tordon herbicides showed no significant recovery and over 95% were considered destroyed.

Low woody shrubs within the treated plots were very sensitive to the Tordon herbicides which killed most of their overgrowth and suppressed regeneration during the three seasons. 2,4-D exerted considerably less damage to the plants and they resumed growth and recovered well in the second season. In plots treated with Tordon herbicides the moss Polytrichum strictum increased in abundance very significantly and had over the three growing seasons increased its cover from less than 10% to over 80% at the Hadashville site.

It is concluded that under the field conditions of the present study sites, picloram applications on Sphagnum fuscum hummocks will permanently kill most of the shoots of the moss and suppress its regeneration for several years. Most

of the vascular plants of the hummock community are also very sensitive to the herbicide. Cyclic use of picloram in this type of bogs will lead to gradual destruction of the S.fuscum habitat and community. This will favour expansion of fen conditions and vegetation in the bogs.

Chapter IV

**EFFECTS OF RIGHT-OF-WAY CONSTRUCTION AND
MAINTENANCE ON DEGRADATION OF SURFACE PEAT IN
THREE BOGS**

4.1 INTRODUCTION

The removal of plant cover and disturbances of surficial soil tend to have more serious consequences in peatlands than in upland communities, as organic soils have a relatively weak physical structure and are made up a biologically oxidizable substrate. The nutrient reserves of organic soils are frequently poor and consequently vegetation restoration on disturbed peat surfaces can be slow (Gore and Godfrey 1981).

The vegetation canopy provides a protective cover for the soil. Vegetation removal is therefore likely to be followed by an increase in soil erosion since it upsets the existing balance between the erosive forces (e.g. rain, wind, slope) and the vegetation and soil conditions that have adjusted to them (Imeson, 1971). Due to their weak physical structure, organic soils are not resistant to erosive forces and vegetation removal can result in substantial losses from exposed peat layers (Bower 1962; Tallis 1973; Parent et al. 1982).

With the removal of plant cover, the microclimatic conditions in the upper part of the soil are altered (Van Cleve 1977). The net radiation reaching the soil surface and heat flow into the ground increase. This may cause soil temperatures to rise, increase depth of thaw in frozen soils, and alter moisture conditions (Bliss and Wein 1972; Haag and Bliss 1974; Gersper and Challinor 1975). These changes may

in turn increase decomposition rates in the surface layers of soils (Suffling and Smith 1979).

A recent construction of power line right-of-ways and their subsequent management in northern and central Manitoba has resulted in localized removal of all vegetation and exposure of bare peat in some bog areas. Most of the initial vegetation removal was caused by mechanical disturbance (e.g. scraping of hummocky terrain, vehicle ruts) and is often most severe on raised peat plateaus and palsas of northern permafrost regions (Sims 1977). The disturbed patches remain unvegetated for several years, which leaves them exposed to erosive and decomposing agents. Concern has been expressed that recolonization by natural vegetation may not occur on the disturbed patches and that deterioration of their exposed top peat will continue (Sims 1977; Sims and Stewart 1981).

Vegetation along power line right-of-ways in Manitoba is primarily managed with herbicides. In right-of-way bogs treated with herbicides (Tordon 101 and Tordon 10K) in recent years, it has been found that Sphagnum fuscum, commonly dominant in the ground vegetation and an important hummock and peat former, is greatly reduced in abundance following herbicidal applications resulting in exposure of bare peat (see Chapters 1 and 3). The herbicides may therefore induce erosion of hummocks and alter decomposition rates in the affected peat as they can exert effects on soil microbial ac-

tivity (Grossbard and Davis 1976; Gottschalk and Shure 1979; Hendrix and Parmelee 1985).

The aim of the present study was to measure the rate of degradation (erosion and decomposition) of exposed surface peat in bogs disturbed by power line right-of-ways in central and northern Manitoba, and to predict the long term impact of the disturbances on organic soil degradation.

4.2 STUDY SITES

Field work was carried out in the Radisson-Dorsey right-of-way at the Devil's Lake, Minago River and Gillam (2) sites, in the same general area as described in Chapter 2. At the Devil's Lake site exposure of bare peat was caused by the application of the herbicide Tordon 101 in the right-of-way during the summer of 1979, but at the other sites bare ground was attributed to the initial vegetation clearance and line construction activities, during 1967 - 1970.

4.3 MATERIALS AND METHODS

4.3.1 Peat erosion

Measurements of erosion were performed by using cranked wires (Clymo 1970) following a procedure similar to that of Imeson (1971). The wires were pushed 10 cm into the peat, down to their horizontal part, which served as a reference level from which erosion was measured. If erosion occurred it was measured, to the nearest mm, as the distance below the horizontal part of the wire to the eroded peat surface (Figure 29 a).

At each site five or more representative patches of exposed peat, ranging in size from less than a square meter to several square meters were selected. Within each patch five wires were driven into the ground at randomly chosen points, at the beginning of the season. Measurements of erosion were made at the end of the season.

Figure 29.

Peat erosion measurement, and litterbags in
in the field.

A) Measurement of peat erosion carried out in
the field. Erosion represents the distance
between the peat surface and the horizontal
part of the cranked wire. (September 25, 1982).

B) Litterbags with peat samples in the field
at the Minago River site. (June 24, 1982).



Bulk densities of top peat (0 - 10 cm) were determined by dividing oven-dry (80° C) weights of 4.7 cm thick peat core by their volume. The samples were collected with a sharp edged peat corer within each patch. From the bulk densities, peat erosion was converted to organic matter losses. Total organic matter loss within the right-of-way at each site was estimated by multiplying the average loss determined in the patches by estimated percentages of bare ground determined in the vegetation sampling at the sites (see Chapter 1).

4.3.2 Decomposition of surface peat

Decomposition in exposed surface peat was studied using the litter bag technique, according to procedures outlined by Suffling and Smith (1974). At each site exposed peat was sampled from the surface down to a depth of 10 - 15 cm in disturbed patches. At the Devil's Lake and Gillam sites the sampled material consisted of light-colored fibric Sphagnum peat nearly undecomposed (von Post 1-2). The material from the Minago River site was dark-brown in color and more decomposed (von Post 4-5) with a fine granular particle size. It had few recognizable plant remains, except wood fragments. In the laboratory 10 - 20 g (dwt) peat samples were put into 12 x 20 cm preweighed (dwt) nylon bags (mesh size 1 mm), which were folded and closed with a staple. The litterbags were then put into preweighed (oven-dry) paperbags,

and the samples were dried to constant weight at 80° C. The samples were brought back to the field in paper bags, where the litterbags were taken out and placed into the surface of disturbed ground patches, with one side exposed (Figure 29 b). The empty paper bags were brought back to the laboratory and redried at 80° C for 24 hours to determine the weight of spillage from the litterbags (Suffling and Smith 1974). On harvesting dates five randomly selected litterbags were collected and placed immediately into preweighed (oven-dry) paper bags. The samples were brought to the laboratory and dried to a constant weight at 80° C. Percentage dry weight loss of peat samples in the field was determined after accounting for weight of spillage, empty litterbags, paper bags and staples.

4.3.3 Effects of herbicides on peat respiration rate

Laboratory tests were conducted on selected peat samples in order to investigate the effects of the herbicides Tordon 101, Tordon 10K and 2,4-D on respiration rates (CO₂ production). Laboratory tests are generally considered more rigorous than field tests (Johnen and Drew 1977). Respiration correlates well with other activities in the soil (e.g. carbon, phosphorous and nitrogen cycling) and respiration studies are considered to reveal reasonably well the response of the soil microbiota to herbicides (Wainwright 1978; Greaves and Malkomes 1980).

4.3.3.1 Peat sampling

Peat samples were obtained in the Hadashville bog on May 30, 1984 adjacent to the herbicide experimental plots (see Chapter 3) from a uniform patch with nearly 100% cover of Sphagnum fuscum.

Samples (20-25 cm long) of S.fuscum plants and peat complex were removed from the bog with a sharp metallic cylinder (10.6 cm dia x 17 cm ht). The cores thus obtained were made up of fresh S.fuscum peat forming a continuum with the living plants at the top.

The samples, 28 in all, were transferred to the laboratory in plastic bags, where the living tops (2 - 3 cm) of Sphagnum were removed and the cores cut to a standard length of 17 cm. The peat cores were then put into 1.8 L glass jars (internal dia 11 cm in the lower 16 cm, dia 7.4 cm at the top, total ht 21.3 cm) and deionized water was added to keep them moist. The water level in the jars was kept at 15 cm from the peat surface when respiration measurements were carried out. Every week the water level was checked and water added to compensate for evaporation loss.

The samples were randomly divided into seven groups of four and transferred to a growth chamber where they were stored at 20° C in darkness during the experimental period.

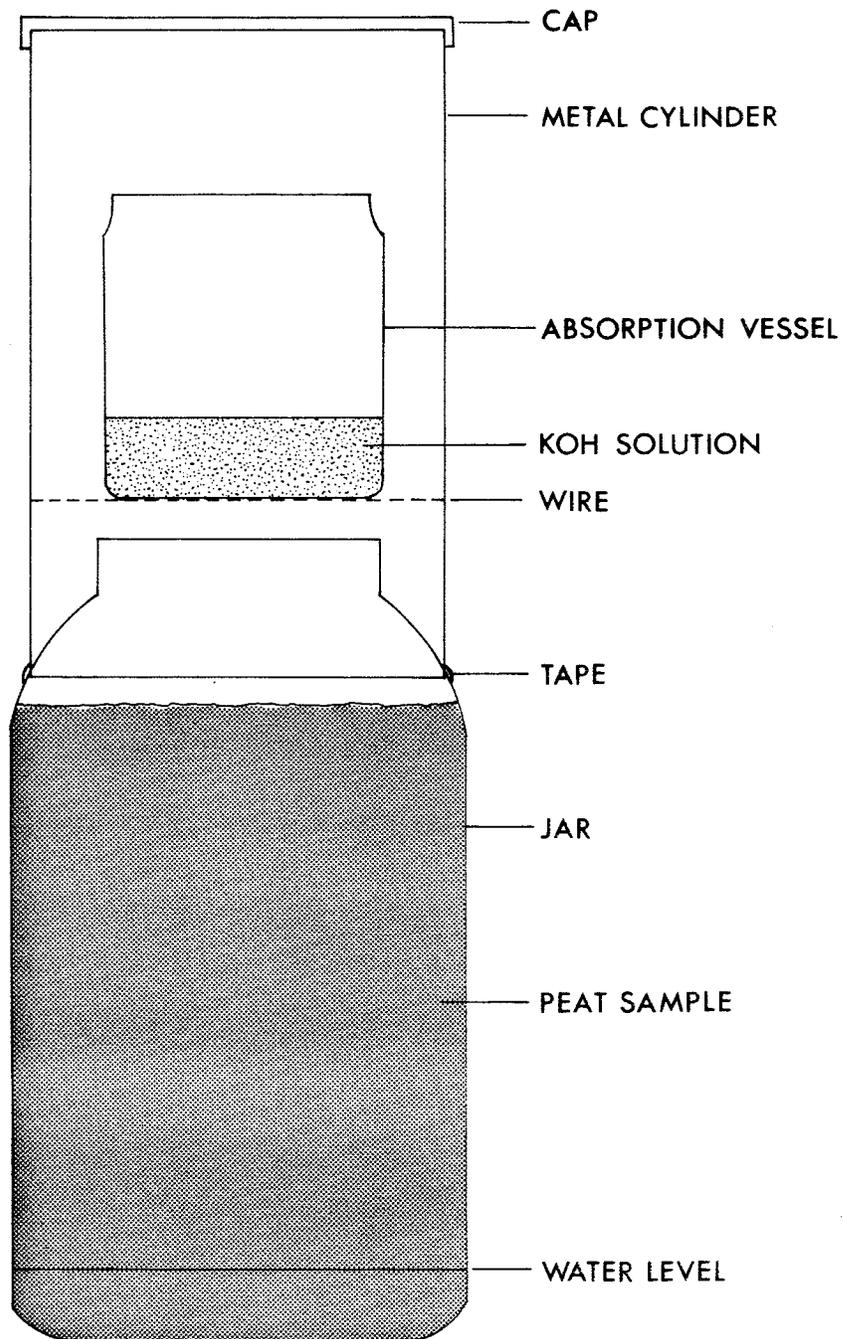
4.3.3.2 Peat respiration rate

Alkali absorption method comparable to the field method described in Chapter 2 was used to measure respiration in the peat samples. Open-ended cylinders (10.6 cm dia x 17 cm ht) were mounted on the the glass jars containing the peat samples (Figure 30) and fastened and sealed to the jars with a 48 mm wide tape (Highland Tape "371", 3M Canada). For absorption of CO₂, 25 ml of 0.25N KOH solution was used, and samples were incubated in the growth chamber for 12 hours. CO₂ absorption was analyzed by titration (Strotzky 1965) and results were converted to mg CO₂ m⁻² hr⁻¹. The CO₂ respiration rates were used to calculate cumulative carbon release and organic matter breakdown in the samples based on the same assumptions as described in the methods section of Chapter 2.

Preliminary studies were used to verify the validity of the measurement procedures used. In one series of measurements respiration was measured in peat cores from the Haddashville site of the same type as in this experiment, over a temperature range of 5 - 30° C. The average respiration rates determined for each temperature level (N=4) were 35.6, 79.8, 97.9, 168.8, 250.2 and 338.9 mg CO₂ m⁻² hr⁻¹ for 5, 10, 16, 20, 25 and 30° C respectively. These respiration rates are very similar to the respiration rates measured over the same temperature range in Sphagnum fuscum peat cores with water level at 15 cm by Silvola and Heikkinen

Figure 30.

Schematic drawing of apparatus used to measure
peat respiration rate in the laboratory.
Scale 1:2.



(1979) who used infra-red gas analysis to measure CO₂ exchange.

Peat respiration was first measured in the samples after they had been left in the growth chamber for five days, and before any herbicides were applied to them (Day 0). The herbicides were added to the samples at the termination of the first 12 hour respiration measurement run and then respiration rates were monitored from 2 to 90 days after application. A minimum duration of respiration tests for determining effects of microbial activity should be in the range of 2 to 6 weeks (Johnen and Drew 1977).

4.3.3.3 Herbicidal applications

The same formulations of herbicides were added to the samples (4 replicates) on a surface area basis as described for the field experiment in Chapter 3, but in addition each herbicide was applied at 10-fold the normal application rate.

Tordon 10K pellets were ground to powder and 0.040 and 0.403 g of the herbicide dispersed over the peat samples for the normal (0.45 g picloram m⁻²) and 10-fold (4.5 g picloram m⁻²) application rates respectively.

Diluted solutions of Tordon 101, 1.68 ml and 16.80 ml were poured over the surface of the samples for the normal (0.12 g picloram and 0.47 g 2,4-D m⁻²) and 10-fold 1.2 g pi-

cloram and 4.7 g 2,4-D m⁻²) application rates, and of the 2,4-D diluted solution 0.81 and 8.1 ml (0.38 and 3.8 g 2,4-D m⁻²) were measured and added to the samples. Four samples were not treated with herbicides and served as controls.

4.3.4 Statistical analysis

For the litterbag data, linear regressions were calculated for each site relating % mass loss and time and assuming that all the litter was remaining at time 0. For slowly decomposing litter linear functions have been found to fit data well, although the underlying assumptions may be hard to justify biologically (Wieder and Lang 1982). To investigate if mass loss rates were significantly different between sites, analysis of variance of regression coefficients over groups was performed, using non-fixed intercept method (Wieder and Lang 1982). The data were run under the BMDP-package (Dixon and Brown 1979).

The respiration data were analyzed with one-way ANOVA for comparisons involving treatments only and with two-way ANOVA for comparisons involving treatments and time. Treatment means were contrasted with Duncan's multiple range test. The data were run under SAS (1979).

4.4 RESULTS AND DISCUSSION

4.4.1 Peat erosion

Average erosion ranged from 0.52 mm at Gillam (1983) to 4.56 mm at Minago River (1983) (Table 20). As the measurement periods did not span the entire season at any of the sites, it may be assumed that the measurements underestimate seasonal erosion. The periods were shortest at Gillam, where only about half the season was covered each year. The measurement period at Devil's Lake in 1983 was relatively longest and covered about four fifths of the season.

The estimated rates of erosion in the three bogs are low compared to some estimates of erosion of peaty soil in other regions. In Britain, erosion of peaty soil in upland Calluna heathlands affected by fire ranged from 0.74 mm yr⁻¹ in areas with vegetation cover of 40 - 100% to 45.3 mm yr⁻¹ in areas without vegetation (Imeson 1971). In a four year study of subsidence of reclaimed organic soils in Quebec, subsidence was estimated 9.9 mm yr⁻¹ in field sown with oats. In unprotected fields without vegetation, the subsidence rate was 45.3 mm yr⁻¹. Erosion was considered as the main factor attributing to higher subsidence rates in the unprotected fields (Parent et al. 1982). For cultivated organic soils in Michigan wind erodibility index of up to 2200 g m⁻² (22 metric tons/ha) has been reported (Robertson et al. 1978; cited in Parent et al. 1982) which is over ten times higher than the highest organic matter losses (214.3 g

TABLE 20

Estimated erosion of peat in right-of-way at three bog sites during 1982 and 1983.

SITE	PERIOD	EROSION			BULK DENSITY			O.M. LOSS g m ⁻² dwt	BARE GR. %	TOTAL EROSION g m ⁻²
		mm			g cm ⁻³	dwt				
		N	\bar{x}	s.e.	N	\bar{x}	s.e.			
Devil's Lake	Jun 6 - Sep 26, 1982	30	0.63	(0.16)	44	0.061	(0.003)	38.4	13	5.0
" "	May 5 - Oct 12, 1983	39	2.97	(0.61)				181.7		23.6
Minago River	Jun 5 - Sep 25, 1982	25	2.24	(0.65)	16	0.047	(0.002)	105.3	10	10.5
" "	Jun 10 - Oct 12, 1983	18	4.56	(1.21)				214.3		21.4
Gillam	Jul 9 - Sep 18, 1982	30	0.63	(0.22)	12	0.051	(0.003)	32.1	7	2.3
"	Jul 1 - Sep 18, 1983	25	0.52	(0.20)				26.5		1.9

m⁻² at Minago River in 1983) in disturbed ground patches in the present study (Table 20).

In both years erosion was highest at the Minago River site, probably because the surface peat in the disturbed patches was more humified, drier and had smaller particle sizes than the surface peat at Devil's Lake and Gillam.

Although the erosion rates probably underestimate the seasonal erosion at the three sites, they reflect a generally low erodibility of exposed peat in disturbed areas within the right-of-way. The combination of conditions necessary to stimulate extensive peat erosion do not seem to exist in the bogs studied.

Average annual precipitation is relatively low, 420 - 470 mm, being lowest in the northern regions. About 70% of the precipitation falls as rain (Environment Canada, AES). During the measurement periods at Gillam in 1982 and 1983, 158.4 mm and 250.8 mm of rainfall were recorded at the Gillam Airport meteorological station, and the greatest rainfall intensity over a 24 hour period was 38.8 mm. In 1983, 330.1 mm of rain was recorded at the Grand Rapids Hydro meteorological station between May 5 and October 12, which was the measurement period for the Devil's Lake site. The greatest rainfall in 24 hours during that period was 31.4 mm. (A complete precipitation record is not available for the 1982 season). As the water table in the type of bogs

studied is relatively low or does not exist (peat palsas or plateaus) and the peat in the upper layers is Sphagnum rich and has a high hydraulic conductivity (Boelter 1965), even the most extreme rainstorm events in the area are not likely to result in surface runoff of excess rainwater (Tallis 1973). Conditions for surface runoff are more likely to exist during snowmelt in spring when the top peat is still frozen and limited vertical or horizontal percolation of water can occur (Tallis 1973). However the topography of the bogs is relatively flat and any slopes are gentle and short, e.g. hummock to hollow or palsa-edge to fen, so runoff events will rarely cause much erosion of peat.

In the British uplands, where extensive erosion of blanket peat has occurred and is still occurring, the climate and topography is very different from that of the present study area. The peat blankets may cover hill slopes of up to 20° ; the yearly precipitation is relatively high, from 950 to over 1350 mm; snow cover may be scanty and intermittent during the winter and snow-lie is very short compared to this study area; also freeze-thaw cycles are common during the winter which contributes to peat breakdown (Imeson 1971; Tallis 1973). Some of the high altitude peat blankets are very exposed (Tallis 1965) which intensifies the action of wind, rain and frost on the bog surfaces.

Another feature contributing to the generally low erosion of exposed peat in the right-of-way is peat structure. The

peat in the upper part of the profiles is mostly fibric Sphagnum peat of low humification. The particles are intertwined and of large size and therefore relatively resistant to erosive forces of rain and wind. The moisture condition in the top peat does not appear to change in the exposed patches compared to the vegetated, and drying out of surface peat generally does not occur (see Chapter 2). Dried out peat particles become more friable and are more easily displaced than when moist.

Difference in structure of surface peat may be the main reason for the considerably higher erosion rates measured on unvegetated fields in Quebec by Parent et al. (1982) than in this study. The surface peat layer of the fields was described as amorphous peat with humigranic microstructure. It was light and pulverized at the surface which makes it highly susceptible to erosion.

At the three bog sites, bare ground in the right-of-way was estimated in 1983 to cover 7 - 13% (Table 20). Assuming that the estimated average erosion rates are representative for disturbed patches in the right-of-way area at all three sites, then the total organic matter loss due to erosion over the measurement periods ranges from 1.9 - 23.6 g m⁻² (Table 20). It is of interest to compare the estimated erosional losses of organic matter with the estimated breakdown of soil organic matter in decomposition. This is possible for the 1983 season when both erosion and soil respiration

(Chapter 2) were measured at the three sites. The results show that total degradation of peat through erosion was extremely low compared to what was broken down in decomposition, or only 0.5 - 4.0% of the decomposition (Table 21). It must be kept in mind that the erosion losses are not estimated for the entire season in contrast to decomposition, and erosion is probably somewhat underestimated. However it is concluded that erosion rate of exposed peat in bogs in the right-of-way is very low. Erosion contributes very little to soil deterioration and can hardly be considered to hamper vegetation recolonization in disturbed areas.

TABLE 21

Erosion and decomposition of organic matter in right-of-way

Comparison of estimated total erosional and decomposition losses of organic matter (g dwt m⁻²) in right-of-way at three bog sites in 1983.

SITE	EROSION	DECOMPOSITION	ERO. % DECO. ¹
Devil's Lake	24	592	4.0
Minago River	21	542	3.9
Gillam	2	370	0.5

¹ Erosion calculated as % of decomposition.

4.4.2 Decomposition of surface peat

The results of the litterbag study are presented in Table 22 and Fig 31. A few litterbags were destroyed or lost due to animal activity at the Devil's Lake and Minago River sites.

The weight losses are very low at all sites and range from 2.88 to 4.70% yr⁻¹. The relationship between weight loss and time were significant ($p < 0.05$), but decomposition rates between sites were not significantly different. Clymo (1965) reports similar weight loss, 3.0% yr⁻¹, of Sphagnum papillosum peat collected at 60 cm depth from a bog in Britain and placed on the bog surface in litterbags. Losses were much higher from fresh S.papillosum litter, or 25.8% yr⁻¹, from the capitulum (0 - 2 cm) and 8.7% yr⁻¹ from the mature section (2 - 6 cm) of the plants. Decreasing rate of mass loss with increasing age of litter were considered related to lower amounts of easily metabolized organic compounds (Clymo 1965). That view is in agreement with the findings of Karunen and Salin (1982) who reported pronounced decreases in lipid compounds concentrations of S.fuscum shoots with increasing depth.

Other reports of decomposition of fresh Sphagnum litter have shown very low decomposition rates compared with other litter types and in the range of 0 - 12% loss in mass per year (Reader 1971; Reader and Stewart 1972; Heal et al.

TABLE 22

Weight loss of peat enclosed in litterbags

Weight loss of exposed peat in litterbags at three bog sites in Manitoba.

SITE	HARVESTING DATE	NUMBER OF DAYS IN FIELD	% INITIAL MASS REMAINING		
			\bar{x}	(s.e.)	N
Devil's L.	Jul 21, 1982	26	99.29	(1.05)	5
	Aug 20, 1982	56	99.59	(0.52)	5
	Sep 26, 1982	93	99.25	(1.27)	4
	Jun 8, 1983	348	97.67	(0.80)	5
	Oct 13, 1983	474	92.88	(6.24)	2
Minago R.	Jul 20, 1982	26	99.63	(0.74)	5
	Aug 21, 1982	58	98.40	(0.65)	5
	Sep 25, 1982	93	96.63	(0.64)	5
	Jun 10, 1983	351	97.38	(3.33)	4
	JUL 20, 1983	391	96.18	(2.21)	5
	Aug 18, 1983	420	97.55	(0.42)	3
	Oct 13, 1983	473	96.98	(4.19)	5
Gillam	Sep 18, 1982	37	98.08	(0.59)	5
	Jul 1, 1983	321	95.66	(0.68)	5
	Sep 17, 1983	399	95.42	(1.35)	5

Figure 31.

Weight loss of exposed peat enclosed in litterbags
at three bog sites in Manitoba.

1978; Clymo 1983; Bartsch and Moore 1985). The decomposition rates of the exposed peat estimated in this study agree well with these reports considering that the peat is composed of more senescent litter.

The litterbag technique has been used successfully to study decomposition in a variety of habitats (Chamie and Richardson 1978). The method should give reasonable estimates of decay rates provided consideration is given to effects of mesh size, handling and spillage, heating and drying, root growth into bags, on measured decomposition rates (Clymo 1965; Suffling and Smith 1974; Davis 1980).

The 1 mm mesh size used in this work should not have excluded any important decomposers from entering the bags, but microorganisms are considered to be mainly responsible for the loss of organic matter for dead Sphagnum or peat in bogs (Clymo 1965 ; 1983).

Without accounting for spillage of litter from the bags during handling and transport, the decomposition rates would have been estimated considerably higher. The weight of spillage left in the paperbags after the litterbags had been placed out in the field was similar at Devils Lake and Gillingam or averaging 1.5% and 1.3% of the litterweight respectively. Spillage was not weighed separately upon harvesting. If similar losses occurred then as when the bags were brought out to the field, then spillage would have amounted

to similar or even slightly higher weight losses than decomposition did in one year. Suffling and Smith (1974) have pointed out the importance of accounting for spillage in litterbag studies in order to increase the precision of decomposition estimates. They found decomposition corrected for spillage to be significantly lower than uncorrected rates for litter enclosed over one year in 1 mm and 0.5 mm mesh-size litterbags.

In this study decomposition rates of exposed peat were low at all sites, and there is no indication that the herbicidal application at Devil's Lake in 1979 has had any long term effects on decomposition rates of exposed peat. As decomposition rates of peat underneath a vegetation canopy were not studied in this work, it is not known if the vegetation removal has affected decomposition rates of surface peat. However the low decomposition rates indicate that disturbances resulting in exposure of peat in the bogs in the right-of-way will generally not create conditions under which rapid breakdown of organic soils occurs.

4.4.3 Effects of herbicides on peat respiration rates

The response of peat microorganisms to the different herbicides and concentrations during the 90 day experiment is shown in Table 23 and Figure 32.

Peat respiration in the seven groups of samples was similar before the herbicides were added. On day 0 the rates ranged from 151.23 - 174.03 mg CO₂ m⁻² hr⁻¹ and differences were not significant between any group averages.

In the samples treated with the liquid formulation herbicides, 2,4-D and Tordon 101, there were immediate increases in respiration rate after the herbicidal applications, and the highest respiration rates during the 90 days of study were measured 2 days after the applications.

In the cores treated with 2,4-D the average respiration rates were 235.47 and 333.88 mg CO₂ m⁻² hr⁻¹ for normal and tenfold rates respectively 2 days after application. After that the rates decreased and were near and not significantly different from the control rate when measured 30 to 90 days after application. The respiration rate in the cores treated with normal application rate was only significantly higher than the control rate 10 days after the application. Differences in respiration rates in the cores treated with the two different application rates were only significant on day 2. The average respiration rates of the treated cores for the entire study period were both significantly higher

TABLE 23

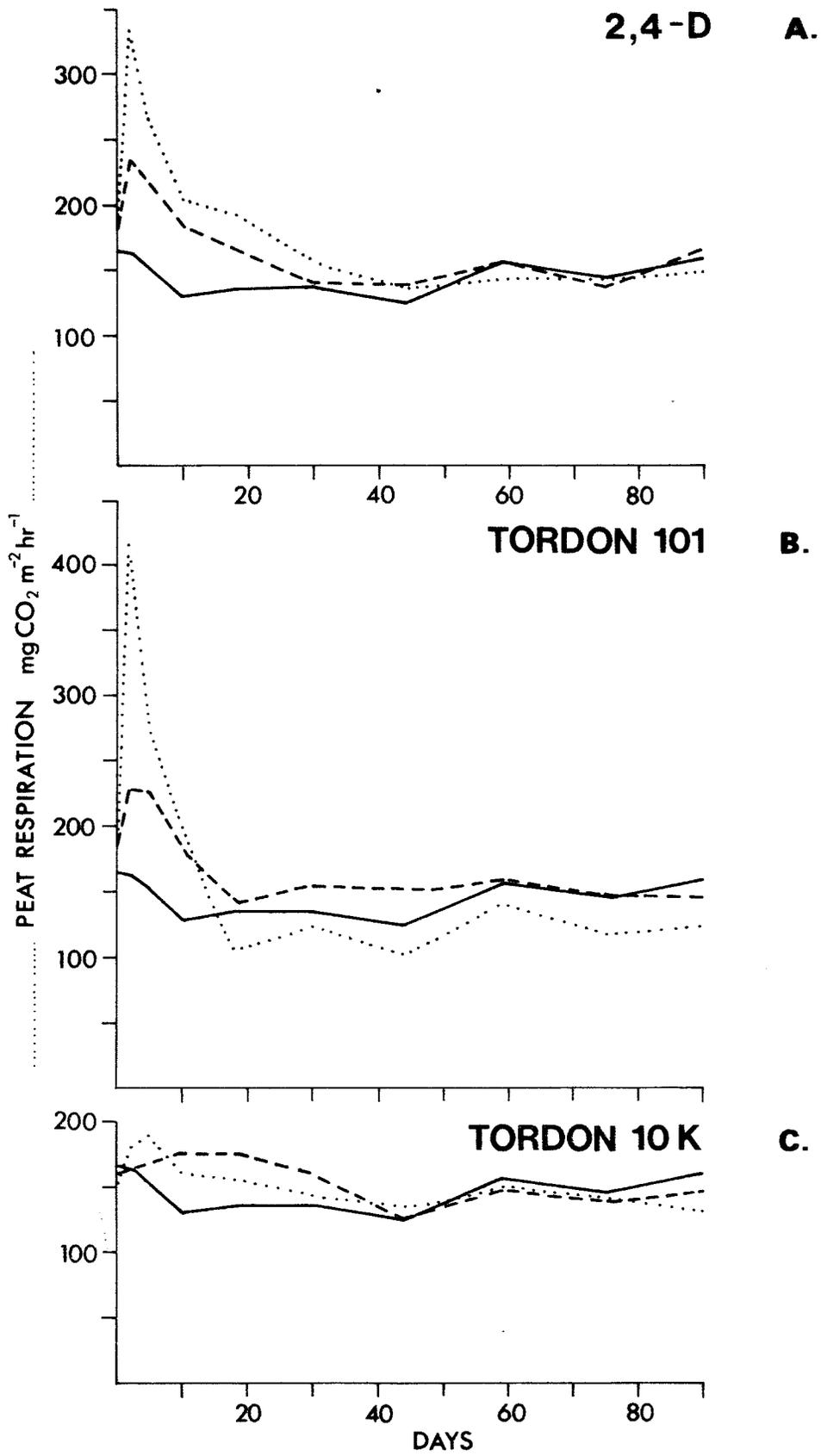
The effect of herbicides on respiration rate ($\text{mg CO}_2 \text{ m}^{-2} \text{ hr}^{-1}$) in Sphagnum fuscum peat cores measured at 20°C and 15 cm waterlevel in the laboratory. Standard error is given underneath means. Means with the same letter within sampling time do not differ significantly at $p = 0.05$. $N = 4$.

Days fr. applic.	CONTROL		2,4-D		TORDON 10 K		TORDON 101	
	normal	x 10	normal	x 10	normal	x 10	normal	x 10
0	166.80 ^a 12.36	171.25 ^a 15.68	165.69 ^a 14.16	159.29 ^a 10.70	151.23 ^a 12.59	174.03 ^a 4.17	158.18 ^a 14.87	
2	162.35 ^a 15.08	235.47 ^a 18.07	333.88 ^b 29.14	164.02 ^a 16.90	177.36 ^a 16.92	230.74 ^a 3.32	414.78 ^c 41.92	
5	152.90 ^a 12.53	220.45 ^{abc} 10.27	265.77 ^{bc} 14.80	169.02 ^{ab} 39.67	188.20 ^{abc} 21.68	227.12 ^{abc} 6.43	280.50 ^c 31.25	
10	128.44 ^a 11.89	186.26 ^{bc} 14.19	207.11 ^c 15.93	174.86 ^{bc} 7.03	160.13 ^{ab} 15.41	180.70 ^{bc} 4.82	201.27 ^{bc} 22.27	
18	135.39 ^{ab} 7.44	164.53 ^{bc} 10.98	193.49 ^c 21.71	173.47 ^{bc} 17.56	155.12 ^{abc} 15.01	140.95 ^{abc} 26.49	104.53 ^a 8.93	
30	135.39 ^a 8.26	140.95 ^a 11.94	156.79 ^a 16.95	160.13 ^a 16.98	142.34 ^a 9.73	156.52 ^a 5.57	123.31 ^a 11.22	
44	124.82 ^{ab} 12.14	139.84 ^{ab} 18.85	137.61 ^{ab} 11.54	125.38 ^{ab} 14.88	133.72 ^{ab} 16.35	150.40 ^a 7.84	101.47 ^b 5.06	
59	156.09 ^a 10.65	156.24 ^a 13.57	142.06 ^a 7.76	150.95 ^a 18.94	147.62 ^a 12.50	158.74 ^a 3.70	139.00 ^a 15.79	
75	144.56 ^a 8.77	137.89 ^a 13.21	144.56 ^a 8.21	139.28 ^a 17.69	141.22 ^a 13.41	146.23 ^a 6.43	117.32 ^a 12.09	
90	158.18 ^{ab} 9.08	164.85 ^a 6.74	149.56 ^{ab} 7.44	145.67 ^{ab} 21.69	130.10 ^{ab} 9.20	145.12 ^{ab} 10.24	123.16 ^b 15.72	
0 - 90	146.49 ^a 3.81	171.78 ^b 6.37	189.65 ^c 10.74	156.21 ^a 6.04	152.71 ^a 4.95	171.05 ^b 5.76	176.42 ^{bc} 16.24	

Figure 32.

Effect of herbicides on peat respiration rate in Sphagnum fuscum peat cores. Measured in dark at 20° C.

———— Control
- - - - - Normal field application rate
..... Tenfold field application rate



than in the control cores and the cores applied with tenfold rates had significantly higher rates than the cores treated with normal rates. A similar stimulation in CO₂ release from mineral soils treated with 2,4-D at 50 and 500 ppm was reported by Ou et al. (1978) from a laboratory study. After an initial peak in CO₂ evolution the rates had fallen to control levels 20 - 30 days after the applications.

In the cores treated with Tordon 101 the respiration rates were 230.74 and 414.78 mg CO₂ m⁻² hr⁻¹ when measured 2 days after the application. Eighteen days after the application the respiration rates of the treated samples had decreased to near control levels. In cores treated with normal application rates the respiration rate was only significantly higher than the control rates on day 10, but in the cores treated with tenfold rates the respiration rates were significantly higher from 2 - 10 days after application. From day 18 - 90 the respiration rate in the cores treated with tenfold rates was repressed and consistently lower than in the control and normal application rate cores, and the average respiration rate in the cores treated with with tenfold rates was, for that period, significantly lower ($p < 0.05$). The respiration rate for the tenfold rate at day 2, which was the highest measured for any treatment, may reflect the higher concentration of 2,4-D in the Tordon 101 treatment than in the tenfold 2,4-D treatment.

In the samples treated with Tordon 10K there was a relatively small increase in respiration rates compared to the samples treated with the Tordon 101 and 2,4-D mixtures. In the samples treated with normal rates of Tordon 10K, highest respiration rates, $174.86 \text{ mg CO}_2 \text{ m}^{-2} \text{ hr}^{-1}$ were measured on day 10, which was the only day when the respiration rate in the cores was significantly higher than in the control cores. After that the rates declined and were near control levels on day 44. In the cores treated with tenfold rates of Tordon 10K the highest respiration rates, $188.20 \text{ mg CO}_2 \text{ m}^{-2} \text{ hr}^{-1}$ were measured on day 5. On day 30 the rates were down to control levels. On the different measurement dates there was never a significant difference between the respiration rates of the treated and control cores. The average respiration rates of the samples treated with Tordon 10K were not significantly higher than of the control samples for the 90 day study period.

The generally mild effects of picloram on peat respiration rate are in accordance with other reports of CO_2 production in soils treated with the herbicide. Goring et al. (1967) monitored CO_2 evolution from six different soil types treated with picloram concentrations up to 1000 ppm and incubated in the laboratory over a nine month period. The herbicide was not found to have significant effects on CO_2 production from the soils compared to untreated soils, even at the highest dosages. Grover (1972) did not find carbon

dioxide evolution nor oxygen uptake of Regina heavy clay soil to be affected by picloram applications of up to 100 ppm. An enhancement of oxygen uptake was observed with concentrations of 100 to 300 ppm. The measurements were only continued for two weeks which is not sufficient time to determine whether long term changes occurred. Normal application rates of picloram were not found to have any significant effect on CO₂ production in sandy loam soil samples incubated for 8 weeks (Schreven et al. 1970). CO₂ release for the 8 week period for samples treated with tenfold application rates was however 8% and significantly higher than from untreated samples.

The 90 day duration of the experiment appears to have been sufficient to reveal the effects of the herbicides on microbial activity in the peat, since the respiration rate in the treated samples had stabilized and returned to near control levels from 20 - 44 days after application of the herbicide depending on treatment (Johnen and Drew 1977).

One of the major criticisms of respiration measurements is that they indicate the overall effects of the herbicides on respiration of stimulated and unstimulated microbes, but do not reveal how different taxonomic groups are affected. Thus only the net effect is measured, which could be the result of e.g. large stimulation of a particular group of soil organisms and a partly counterbalancing inhibition of another (Johnen and Drew 1977; Greaves and Malkomes 1980).

This sets limits to the interpretation of respiration data, but more specific tests are generally necessary to reveal how different groups of organisms are affected by pesticides.

The main objective of this study was to assess whether the herbicidal applications would exert long term effects on peat respiration rates and consequently rates of organic matter breakdown. Herbicides have been shown to accelerate decomposition of plant litter but frequently indirect effects of the herbicides, e.g. changes in microclimate or substrate quality, have been suggested as explanations (Suffling and Smith 1979; Gottschalk and Shure 1979; Hendrix and Parmelee 1985).

The stimulation of respiration rates from all the treatments compared to the control rates, were in all cases reversible and of relatively short duration. In no instance was respiration rate significantly higher in treated cores than control cores after 18 days had passed from the applications, but stimulation trends could be followed longer. At normal application rates none of the herbicides caused permanent changes in respiration rate, and neither did the tenfold application rates of 2,4-D or Tordon 10K. There were however signs of that the tenfold rates of Tordon 101 were causing a significant inhibition of respiration rate during the period of 20 - 90 days following application. The carbon release from the samples treated with herbicides

was not significantly higher than from the control cores for the 90 day period for any of the treatments (Table 24). The increases in carbon release for the normal and tenfold application rates were 10.6% and 16.5% for the 2,4-D, 11.3% and -1% for Tordon 101 and 6.0% and 2.7% for Tordon 10K respectively.

The immediate stimulation in respiration rates and similarity in overall response in the samples treated with 2,4-D and Tordon 101 applications might indicate that it is the 2,4-D component that exerts these effects, since picloram alone (Tordon 10K) does not cause a comparable rise in respiration rate. However it cannot be ruled out that differences in inert ingredients have contributed to differences in respiration responses. Besides the herbicidal components the 2,4-D and Tordon 101 mixtures contain additional carbon and nitrogen sources which Tordon 10K does not have (NRCC 1974; Ou et al. 1978).

The cumulative carbon release for the different treatments over the 90 day period shows that for most of the treatments, e.g. all the normal application rates, the increase in carbon release from control values is far greater than the estimated amount of carbon added in the applications (Table 24). Therefore the observed increases in CO₂ production can to a very small extent be explained by breakdown of the herbicides or organic impurities of the mixtures. The increases are mostly due to increased breakdown of peat.

TABLE 24

Carbon release from untreated and treated peat cores

The effects of herbicides on carbon release (g m^{-2}) in S. fuscum peat cores.

TREATMENT	CARBON RELEASE		T - C ¹	MAXIMUM ESTIMATED CARBON INPUT IN HERBICIDAL APPLIC.*
	\bar{x}	(S.E.)		
Control	83.71	(4.0)		
2,4-D	92.64	(6.98)	8.93	0.4
2,4-D 10x	97.81	(6.78)	14.10	4.0
Tordon 10K	89.07	(8.43)	5.36	0.2
Tordon 10K 10x	86.28	(6.69)	2.57	2.0
Tordon 101	93.43	(3.14)	9.72	0.7
Tordon 101 10x	82.78	(6.83)	-0.93	7.0

* For 2,4-D and Tordon 101: Calculated from weight of undiluted mixtures (g m^{-2}) and assuming carbon content not exceeding 40% by weight.
For Tordon 10K: Calculated from weight of active ingredient and assuming carbon content not exceeding 35% by weight.

¹ T - C = Carbon release of control samples is subtracted from that of samples treated with herbicides.

In terms of organic matter breakdown the cumulative carbon release from the control samples is equivalent to that 167 g O.M. dwt m⁻² (assuming 50% carbon content by dwt of peat) have been broken down over the 90 day period. In the cores treated with herbicides the estimated increase in breakdown of peat, g dwt m⁻², is in the range of 17 - 18 g and 20 - 28 g for the normal and tenfold 2,4-D applications respectively; 10 - 11 g and 1 - 5 g for the Tordon 10K applications and 18 - 19 and -2 - -16 g for the Tordon 101 applications, depending on how much of the carbon added to the samples in the herbicidal applications was released as CO₂ over the 90 day period.

Microorganisms are considered to be principally responsible for the degradation of both 2,4-D and picloram (Meikle et al. 1973; Loos 1975). The degradation of 2,4-D in soils has been found to proceed at a relatively fast rate (Ou et al. 1978; Cullimore 1981; Ou 1984) and the herbicide usually disappears from soils in few weeks. Ou et al. (1978) using labelled carbon found that at low concentrations in silty clay loam soil samples, 2,4-D was nearly completely degraded to its final oxidation products of CO₂, H₂O and Cl⁻ within 40 days of incubation. It is therefore probable that in the present study appreciable amounts of 2,4-D were degraded within the study period. In contrast to the rapid breakdown of 2,4-D, picloram degradation in soils progresses at much slower rates (NRCC 1974; Loos 1975). Youngson et

al. (1967) reported that during an 89 day incubation period of three different soil types, 0.4 - 30.0% of picloram was decomposed and oxidized to CO₂, depending on application rate. Similarly, Guenzi and Beard (1976) found very low (0.01 - 3.36%) decomposition rates of picloram in different soil types incubated at different temperatures for 20 days. The contribution of picloram degradation in the study to CO₂ production may therefore be expected to have been generally less than that of 2,4-D degradation.

It has not been shown that 2,4-D or picloram, when used at recommended rates, have permanent effects on the activity or functions of soil microorganisms. 2,4-D has been extensively used for nearly 40 years and studies of long term repeated applications or high concentrations of the herbicide have not revealed presence or accumulation of harmful 2,4-D metabolites in soils (Ou et al. 1978; Smith 1982; Ou 1984). In spite of generally higher toxicity of plants and greater persistence in soils, picloram applications do not appear to have significant effects on microbial quantities and composition or interfere with essential nutrient cycling processes (Goring et al. 1967; Schreven et al. 1970 ; Grover 1972; NRCC 1974; Foy 1976).

In laboratory studies soils are removed from field conditions and it is the potential activity that is being measured rather than the actual field biological activity (Smith 1982). In the field fluctuations in temperature and mois-

ture and local differences in soil properties may modify responses of soil microorganisms to herbicides (Johnen and Drew 1977). Direct interpretation of results obtained under laboratory conditions onto field conditions is therefore not possible and results have to be viewed as indicating certain trends rather than revealing absolute relationships.

In this work all three herbicidal applications were found to temporarily stimulate respiration in peat, but no long term changes in peat respiration were observed for normal field application rates. Over the 90 day incubation period increase in organic matter decomposition in the samples treated with normal application rates was in the range of 10 - 18 g dwt m⁻². These increases are low when compared to the estimated annual breakdown of organic matter, 350 - 700 g dwt m⁻², in bog soils in the Dorsey-Radisson right-of-way (see Chapter 2). The results indicate that normal field applications of 2,4-D, Tordon 101 and Tordon 10K to bog soils will generally not exert long term direct effects on decomposers activity or significantly alter decomposition rates.

4.5 SUMMARY AND CONCLUSIONS

Estimated erosion of exposed peat during the measurement periods was in the range of 0.52 - 4.56 mm, being highest at the Minago River site, where the disturbed surface peat had finer particle sizes. The erosion measured during 1983 was estimated to be equivalent to the loss of 23.6, 21.4 and 1.9 g O.M. dwt m⁻² in the right-of-way at Devil's Lake, Minago River and Gillam sites respectively. These organic matter losses are 0.5 - 4.0% of the estimated breakdown of organic matter due to decomposition in the peat soils at the sites during the 1983 season.

Peat enclosed in litterbags lost 2.88 - 4.70% of the original dwt in a year. Highest average losses occurred at the Gillam site. Losses were significantly related with time at all three sites, but did not differ significantly between sites.

All the herbicidal applications caused initial stimulation of respiration rate in peat cores. The stimulatory effects of the 2,4-D and Tordon 101 applications were considerably greater than of Tordon 10K. The highest respiration rates measured were 235.57 and 333.88, 230.74 and 414.78, 174.86 and 188.20 mg CO₂ m⁻² hr⁻¹ in samples treated with normal and tenfold rates of 2,4-D, Tordon 101 and Tordon 10K respectively, while respiration rate in untreated samples ranged from 128.44 - 162.35 mg CO₂ m⁻² hr⁻¹. The respira-

tion rate in the treated cores had declined to near control levels from 20 - 44 days of applications and there was no sign of long term stimulation of respiration rates. Cumulative CO₂ production over the 90 day study period was not significantly higher in the treated than untreated samples. It was estimated that the CO₂ production in the untreated samples was equivalent to the breakdown of 167 g O.M. dwt m⁻² over the 90 days. In samples treated with normal application rates the increase was almost entirely due to increase in the breakdown of peat which was estimated as 17, 10 and 18 g O.M. dwt m⁻² for the 2,4-D, Tordon 10K and Tordon 101 applications respectively.

It is concluded that for exposed peat in bogs in the right-of-way:

1. erosion is generally low
2. decomposition proceeds at a slow rate
3. the herbicidal applications do not cause long term changes in decomposition rates.

Mechanical or chemical disturbances in the general right-of-way area do not lead to considerable increase in degradation of organic soils beyond that occurring through natural processes.

GENERAL CONCLUSIONS

The overall objective of this work was to investigate how vegetation, peat degradation and environmental conditions of bogs in central and northern Manitoba are affected by disturbances associated with power line right-of-ways. The investigations undertaken to meet the objective and their findings have been described in the four chapters of this thesis.

In the first chapter it was shown that in general the vegetation variation encountered reflected differences in moisture and nutrient regimes. Analysis of the vegetation data revealed that vegetation change in the ROW stands from the relatively undisturbed forest stands reflected altered species abundances rather than changes in composition. Sphagnum fuscum was the most abundant understorey species both in the forest and ROW community, but its abundance was almost 50% less in the ROW. The ROW community had generally lower abundance of ericoids and significantly higher extent of bare ground. Polytrichum strictum was the only species which had significantly higher abundance in the ROW than the forest community. It was the main colonizer of disturbed peat surfaces with relatively unaltered moisture conditions. Water chemistry parameters indicated a slight enrichment of

the ROW environment, but overall differences were not significant between the ROW and forest. Permafrost in northern ROW stands had generally not been affected by canopy removal, but thawing had occurred in places on peat plateaus where access paths cut across them. Frequent vehicle traffic results in disruption of the bog vegetation cover and the underlying peat, leading to a thawing of permafrost and increased wetness and nutrient concentrations in the paths. These highly disturbed stands are primarily colonized by Carex aquatilis, but the area in the ROW where this type of disturbance occurs is small. Steps to promote revegetation in areas where vegetation cover has been greatly reduced are not considered necessary as disturbances are generally not followed by excessive deterioration of exposed peat. The disturbed surfaces are gradually recolonized by the native vegetation.

The second chapter demonstrated that disturbances appear to have had insignificant effects on annual decomposition of organic matter in peat soils in the ROW. Soil respiration rates were higher in the undisturbed forest than the ROW, but differences were attributed to differences in root respiration rates, since environmental conditions influencing respiration rates were similar in the ROW and forest. It was suggested that due to the low primary production in the ROW annual breakdown of organic matter in the peat soil there exceeds the amount of new litter added to the soil each year.

The most significant contribution of chapter 3 was the demonstration of the sensitivity of Sphagnum fuscum to picloram applications. Cyclic use of the herbicide to control tree regeneration at bog sites in the ROW is seen as leading to the gradual disappearance of the moss and destruction of the hummock habitat and its associated community. It is predicted that this will lead to an expansion of fen vegetation and conditions in the affected bogs. There is a need for further research in this field to determine the long term effects of the herbicide on the bog plant community. Consideration should be given to alternate methods of managing vegetation at bog sites.

The last chapter revealed that degradation of exposed surface peat in disturbed areas is slow and appears to have little impact on the habitats studied. Herbicidal applications caused a short term increase in peat respiration rates, but they are not considered to have long term impact on peat decomposition processes.

The study has therefore added to our knowledge of vegetation and peat decomposition processes in disturbed bogs of Manitoba. It is hoped that this knowledge will aid in future management of bog vegetation and minimize the impact of disturbances.

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Appendix A
PLANT SPECIES LIST

Nomenclature and arrangement of vascular plants is according to Scoggan (1978,1979) while that of bryophytes follows Ireland et al. (1980), and for lichens, Hale and Culberson (1970) except for Cladina stellaris which follows Hale (1979).

(Stand numbers labelled with r are right-of-way stands, while unlabelled stand numbers represent forest stands)

Vascular plants (continued)	Tower no.	85	62	4	4	259	478	665	669	840	863	903	928	1091	1152	1266	1298																		
	Stand no.	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25	26	27	28	29	30	31	32	33	34

LENTIBULARIACEAE

Pinguicula villosa L.

X X X

CAPRIFOLIACEAE

Linnaea borealis L.

X

COMPOSITAE

Erigeron acris var. elatus (Hook.) Cronq.

X X

Erigeron spp.

X

Aster spp.

X

Achillea millefolium L.

X

Solidago spp.

X

Senecio aureus L.

X

Appendix B

SOIL RESPIRATION STUDY - PEAT PROFILES

Descriptions of peat profiles at sites of soil respiration measurements.

DEVIL'S LAKE

Forest, profile 1:

0 - 8 cm. Light brown, nearly undecomposed (von Post 1-2) Sphagnum peat.

8 - 23 cm. Dark brown, weakly to moderately decomposed (von Post 4-5) Sphagnum peat mixed with Carex remains and wood fragments.

23 - 43 cm. Black amorphous peat very strongly to almost completely decomposed (von Post 8-9).

43 - cm. Rock (boulders).

Forest, profile 2:

0 - 15 cm. A mixture of brown nearly undecomposed (von Post 1-2) Pleurozium and Cladina material.

15 - 73 cm. Brown to dark brown Sphagnum peat with Carex increasing with increasing depth. Wood fragments throughout. Weakly to moderately decomposed (von Post 4-5).

73- cm. Rock (boulders).

Right-of-way

0 - 20 cm. Light brown, mostly undecomposed (von Post 1-2) Sphagnum peat.

20 - 80 cm. Dark brown, fibrous Carex peat with Spha-
num mixed in in upper part. Weakly to moderately decomposed
(von Post 4-5).

80 - 115 cm. Black, nearly amorphous peat, very strong-
ly to almost completely decomposed (von Post 8-9).

115- cm. Rock (boulders).

MINAGO RIVER

Forest:

0 - 47 cm. Light brown, nearly undecomposed (von Post 1-2) Sphagnum peat with living root material.

33 - 48 cm. Brown, weakly to moderately decomposed (von Post 4-5) Sphagnum peat with some wood fragments.

47 - 66 cm. Black, strongly decomposed (von Post 8), nearly amorphous peat.

66- cm. Light brownish gray granular clay.

Right-of-way:

0 - 34 cm. Light brown, nearly undecomposed (von Post 1-2) Sphagnum peat with some wood fragments.

34 - 53 cm. Dark brown to black, strongly decomposed (von Post 8) amorphous peat.

53- cm. Light brownish gray granular clay.

GILLAM 2

Forest:

0 - 43 cm. Light brown, mostly undecomposed (von Post 1-2) Sphagnum peat.

43 - 52 cm. Brown, weakly decomposed (von Post 3-4) Sphagnum peat.

52- cm. Permafrost

(Ice auger was used to bore through frozen peat and clay was found at 75 cm.)

Right-of-way:

0 - 25 cm. Light brown, mostly undecomposed (von Post 1-2) Sphagnum peat.

25 - 37 cm. Dark brown woody Sphagnum peat mixed with Carex remains, weakly to moderately decomposed (von Post 4).

37 - 47 cm. Dark brown to black fibrous woody peat, strongly decomposed (von Post 7).

47- cm. Permafrost.

(Clay was found at 70 cm underneath frozen peat).

HENDAY 1

Forest:

0 - 10 cm. Light brown, undecomposed (von Post 1-2)
Sphagnum peat mixed with living roots.

10 - 27 cm. Brown, moderately decomposed (von Post 6)
Carex peat.

27 - 52 cm. Black, strongly decomposed (von Post 7 -8)
and mostly amorphous peat.

52- cm. Permafrost.

Right-of-way:

0 - 26 cm. Light brown, undecomposed (von Post 1-2)
Sphagnum peat.

26 - 38 cm. Brown, weakly decomposed (von Post 3-4)
mixture of Sphagnum and Carex peat.

38 - 56 cm. Brown to black, moderately to strongly de-
composed (von Post 5-7) woody Carex peat.

56- cm. Permafrost.

HADASHVILLE (bog):

0 - 25 cm. Light brown undecomposed (von Post 1-2)

Sphagnum peat with wood fragments.

25 - 34 cm. Brown, weakly decomposed (von Post 3)

Sphagnum peat mixed with wood fragments.

34 - 40 cm. Light brown, undecomposed (von Post 1-2)

Sphagnum peat, with wood fragments.

40 - 48 cm. Dark brown, weakly to moderately decomposed

(von Post 4-5) Sphagnum/Carex peat.

48 - 57 cm. Light brown, very weakly decomposed (von

Post 2-3) Sphagnum peat, some Carex remains.

57 - 105 cm. Dark brown, moderately decomposed (von

Post 5) Sphagnum peat with Carex remains mixed in.

105- cm. Watertable.

(A metal probe was used to determine total peat thickness, which was found to be 310 cm. Dark grey clay mixed with shell fragments as underneath peat.)

Appendix C

VEGETATION DATA SET FOR ORDINATION

Data set for DECORANA ordination. The first number in each row of the data indicates the stand. The rest of each row contains couplets comprised of a species and a measure of its relative abundance (PV x 10) in the community. The abbreviated species names, and stand numbers are included at the end of the data deck.

VEGETATION SAMPLING 1983 - DECORANA ORDINATION

(I2,5(I3,F7.2))

1	16	17.43	19	0.82	31	8.76	36	15.49	37	2.19
1	44	0.98	45	0.32						
2	12	0.32	16	288.00	19	376.00	20	2.19	23	15.18
2	25	218.20	29	10.13	31	127.01	32	64.40	33	85.00
2	36	59.77	37	8.05	42	4.93	44	8.84	46	7.50
2	47	15.48	48	0.95						
3	13	4.74	16	61.98	18	4.74	19	38.49	24	4.47
3	25	19.09	31	12.39	32	3.83	36	71.00	37	7.07
3	42	47.00	44	1.79	50	17.44				
4	10	0.95	12	19.09	13	4.74	16	87.85	19	242.00
4	20	1.79	21	21.50	23	11.50	24	8.05	25	134.00
4	26	0.32	28	12.02	31	3.83	32	13.91	37	0.95
4	42	618.00	44	2.19	47	1.26				
5	1	0.32	3	11.31	4	15.18	7	3.83	10	283.66
5	12	4.93	13	4.74	16	99.15	18	8.05	19	55.77
5	20	0.32	23	7.59	24	3.83	25	19.09	28	34.65
5	31	0.95	32	2.68	33	17.57	36	21.69	38	0.95
5	39	0.32	42	0.95	44	1.79	50	8.05		
6	1	0.45	3	2.68	7	0.95	10	36.06	11	0.32
6	16	47.69	17	0.95	19	278.00	20	0.32	23	7.59
6	24	15.18	25	66.00	26	0.32	28	66.41	29	19.92
6	31	107.33	32	6.32	33	8.52	35	2.19	36	18.07
6	40	0.32	41	0.95	42	3.83	44	116.30	46	17.53
6	47	12.39	48	0.95						
7	3	9.19	12	42.00	13	0.32	16	68.78	19	243.50
7	20	4.93	23	0.95	24	4.93	25	77.81	31	1.10

7	32	2.19	36	392.00	41	0.45	43	11.71	44	57.87
7	45	8.05	46	23.00	47	12.39	48	0.95	50	182.46
8	10	11.50	12	2.68	15	1.10	16	53.03	17	4.74
8	19	36.06	20	19.68	21	171.71	23	41.14	26	1.10
8	28	257.17	29	2.68	32	2.68	35	90.92	42	7.16
8	44	0.95	46	2.68	47	0.32	48	0.95		
9	1	0.95	3	82.54	6	1.26	7	19.09	10	232.00
9	12	4.93	13	32.53	16	47.69	18	1.79	19	19.61
9	23	0.95	24	0.32	25	18.07	31	4.93	33	2.68
9	38	202.70	50	182.46						
10	9	0.32	19	152.15	21	6.00	25	2.70	31	4.93
10	32	1.40	36	77.50	41	27.00	42	48.38	43	72.00
10	44	4.43	46	18.50	47	3.56	48	0.95		
11	15	0.45	16	2.85	19	146.50	20	4.43	23	7.59
11	25	62.50	28	175.19	32	1.57	35	18.34		
12	3	0.95	10	1.57	12	2.68	19	82.54	20	11.50
12	21	48.38	23	3.56	25	32.26	28	2.68	31	6.01
12	32	0.32	36	78.23	41	5.30	42	351.00	43	15.18
12	44	13.94	46	30.00	47	10.29	48	8.84		
13	1	13.39	3	12.01	12	0.32	15	4.43	16	0.32
13	19	172.00	20	3.56	21	0.95	23	20.87	25	137.00
13	28	206.66	29	12.02	32	30.60	35	50.35	36	23.24
13	40	4.74	41	2.68	42	234.33	44	0.95	46	0.95
13	47	3.56	50	4.74						
14	3	59.77	7	5.66	10	4.93	16	2.68	19	242.00
14	20	1.79	21	6.32	23	25.61	25	32.20	28	561.00
14	29	10.41	32	0.95	33	0.95	35	8.05	36	102.00
14	41	0.32	42	1.79	44	2.68	45	0.32	46	1.79

14	47	2.68	49	11.50	50	5.69				
15	3	17.20	7	2.68	10	4.93	11	4.93	15	4.95
15	19	172.00	23	25.00	25	25.94	28	209.17	29	2.68
15	30	19.92	32	2.68	33	37.79	34	29.09	35	155.62
15	36	2.68	46	0.95	50	4.74				
16	1	0.32	3	9.19	6	2.53	7	9.19	10	13.91
16	12	2.68	19	173.00	20	3.16	23	4.93	25	7.59
16	28	557.00	29	4.93	36	290.00	41	46.48	44	1.26
16	46	2.68	50	0.32						
17	4	87.01	12	4.74	15	4.65	19	311.00	23	25.61
17	25	37.00	28	680.21	29	19.92	33	0.95	35	2.19
17	36	0.95	39	4.74	42	0.32	44	0.95		
18	3	125.23	16	13.42	19	136.00	20	6.32	23	23.72
18	25	42.00	28	83.18	32	5.48	36	336.00	41	13.94
18	42	131.48	43	3.83	46	0.45	47	0.32	48	0.95
18	50	4.74								
19	3	60.42	16	4.93	19	138.51	21	30.36	23	2.68
19	25	59.77	28	21.36	32	1.79	35	43.63	41	630.00
19	42	8.05	43	0.95	44	0.32	46	0.32	47	1.79
20	9	2.68	12	21.47	13	0.32	16	5.66	19	149.00
20	20	43.64	21	160.10	23	1.79	25	30.00	28	261.84
20	32	0.45	36	150.84	42	146.10	44	0.32	46	0.45
21	9	0.32	12	6.32	16	37.65	19	160.33	20	1.55
21	21	59.77	23	3.83	25	25.61	28	151.05	29	0.95
21	32	2.74	36	6.32	42	441.85				
22	1	0.32	3	1.10	4	7.53	10	23.72	11	2.68
22	19	194.48	20	4.43	23	6.69	25	54.00	28	256.02
22	32	3.83	33	0.95	34	0.95	35	0.32	36	41.14

22	41	162.67	42	93.71	44	0.32	50	1.79		
23	1	0.45	3	2.68	4	38.46	10	5.66	11	1.79
23	12	0.32	15	5.66	16	1.79	19	126.00	20	0.32
23	21	1.79	23	2.80	25	42.00	28	584.00	30	12.02
23	32	0.32	34	42.17	35	29.52	36	3.16	42	2.68
24	1	0.45	3	17.57	4	2.68	5	0.32	7	19.09
24	10	10.07	11	2.19	16	3.83	19	125.00	21	0.95
24	22	2.19	23	0.95	25	11.50	28	378.52	29	0.95
24	33	43.27	34	27.83	36	192.58	41	2.68	42	52.49
24	44	1.79	46	3.83	49	0.32				
25	3	42.00	10	1.79	11	1.79	12	12.39	15	6.64
25	19	125.00	21	48.38	22	1.79	23	30.00	25	2.68
25	28	707.00	30	4.74	33	1.79	34	0.95	36	16.99
25	50	11.50								
26	3	19.68	6	0.95	8	0.95	10	13.91	13	0.95
26	18	0.95	19	183.00	20	115.41	23	4.43	25	21.47
26	27	0.32	28	2.68	31	0.95	32	0.95	36	9.20
26	42	2.19	44	7.25	45	0.32	47	0.95	49	2.67
27	3	43.64	10	51.49	15	1.10	19	292.19	21	4.93
27	23	31.83	25	53.67	27	4.43	28	453.00	33	1.79
27	35	3.83	36	32.53						
28	10	182.15	14	6.32	19	102.00	20	0.95	23	10.61
28	25	19.68	28	582.00	32	0.95	33	19.02	35	13.42
28	36	193.53	41	2.68	42	0.95	44	4.74	46	6.32
28	47	0.95								
29	2	1.79	10	62.61	12	18.97	13	0.95	14	6.32
29	15	28.00	16	4.74	19	48.38	20	4.43	21	114.00
29	22	7.59	23	30.00	25	50.35	27	12.39	28	630.00

29 29 0.95 32 8.05 33 0.95 34 0.95 35 3.83
 29 36 59.77 41 23.00 42 2.68
 30 2 7.53 5 0.32 10 207.00 12 2.12 34 161.28
 30 39 15.56
 31 3 12.39 8 0.45 10 16.99 15 0.32 19 242.00
 31 23 15.18 25 101.00 27 1.26 28 157.24 29 4.74
 31 32 3.83 33 8.05 34 0.95 35 112.70 36 4.93
 31 41 0.32 42 182.46 44 0.32 46 0.32 50 0.32
 32 1 0.32 2 7.59 4 0.45 5 0.95 8 7.59
 32 10 66.00 13 0.45 14 8.76 17 5.48 19 124.28
 32 20 7.16 22 4.74 23 1.79 25 35.98 27 0.32
 32 28 114.64 32 1.79 33 7.16 34 0.95 35 14.14
 32 36 36.06 40 0.32 41 67.39 42 50.91 46 2.19
 32 50 0.95
 33 1 6.64 2 0.45 4 2.68 8 9.19 10 42.93
 33 11 15.49 13 4.74 14 46.02 15 3.54 17 184.00
 33 19 114.00 20 2.12 22 10.61 23 21.47 25 71.15
 33 27 2.19 28 472.44 32 0.95 34 4.74 35 10.41
 33 36 10.61 41 0.95 42 2.68 46 4.93 50 2.19
 34 12 0.95 19 292.00 20 30.00 21 116.69 22 13.94
 34 23 0.32 25 59.77 28 56.89 33 22.77 35 0.32
 34 36 68.94 41 0.32 45 0.32 46 1.79

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EQUI VAREQUI FLUEQUI PRAEQUI ARVCARE CAP
 AGRO BORCALA CANJUNC BALERIO VAGCARE AQU
 CARE GYNSMIL TRIBETU GLASARR PURDROS ROT
 RUBU CHAEMPE NIGEPIL ANGLEDU GROKALM POL
 CHAM CALANDR GLAOXYC MICVACC ULIVACC VIT

PING VILGEOC LIVSPHA FUSSHPA NEMSPHA WAR
CERA PURDICR UNDAULO PALTOME NITPLEU SCH
POLY STRPOHL NUTDREP ADUPLAG ELLHYLO SPL
CLAD RANCLAD MITCLAD STECLAD GRACLAD CRI
CLAD DEFCLAD CORICMA ERIPELT POLSALIX SP
STAND1 STAND2 STAND3 STAND4 STAND5
STAND6 STAND7 STAND8 STAND9 STAND10
STAND11 STAND12 STAND13 STAND14 STAND15
STAND16 STAND17 STAND18 STAND19 STAND20
STAND21 STAND22 STAND23 STAND24 STAND25
STAND26 STAND27 STAND28 STAND29 STAND30
STAND31 STAND32 STAND33 STAND34

Appendix D
SOIL RESPIRATION STUDY - DATA

PLOT = Study plot within site.

DA-M = date, month.

RESPIR = Soil respiration rate ($\text{mg CO}_2 \text{ m}^{-2} \text{ hr}^{-1}$)

MOIST = Soil moisture content (% wet wt, 0 - 10 cm)

ACRO = Acrotelm thickness (cm)

TEMP = Soil temperature (°C, 10 cm)

PLOT DA-M RESPIR MOIST ACRO TEMP

FOR	1005	172.33	88.40	24.4	1.5
ROW	1005	148.73	89.13	11.4	1.5
FOR	0806	221.01	79.42	30.0	9.8
ROW	0806	257.43	89.95	22.5	9.2
FOR	2506	357.79	85.66	36.0	15.7
ROW	2506	325.54	88.69	37.0	16.6
FOR	2507	428.68	86.36	39.3	20.1
ROW	2507	333.97	88.41	40.7	20.1
FOR	1508	411.44	88.58	56.3	15.0
ROW	1508	356.40	86.61	51.3	15.9
FOR	1009	314.33	82.63	32.0	10.0
ROW	1009	244.08	87.96	34.3	11.5
FOR	1210	200.44	85.79	31.7	3.8
ROW	1210	195.44	89.24	35.0	4.8

DEVIL'S LAKE

1983 DATA

FOR	2305	304.41	88.00	29.0	7.2
ROW	2305	234.08	89.33	28.3	5.8
FOR	1706	281.06	86.41	33.0	15.9
ROW	1706	272.72	87.52	32.0	11.9
FOR	1107	484.28	86.49	41.3	18.4
ROW	1107	464.82	88.31	40.7	16.1
FOR	2008	595.48	84.58	54.7	16.4
ROW	2008	370.58	88.23	43.7	16.9
FOR	1509	315.25	85.27	35.7	9.5
ROW	1509	333.05	83.61	36.0	9.4

DEVIL'S LAKE

1984 DATA

FOR	1410	282.73	87.26	37.0	7.6
ROW	1410	234.08	89.56	33.7	7.6

FOR	0906	221.57	87.82	26.3	9.9
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ROW	0906	200.16	85.13	19.7	5.9
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FOR	1907	301.07	83.16	38.0	11.0
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ROW	1907	291.07	81.17	52.3	17.4
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FOR	1708	405.05	78.83	57.0	14.7
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MINAGO RIVER

ROW	1708	390.31	77.45	57.0	12.2
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1983 DATA

FOR	1210	171.25	87.15	57.0	3.6
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ROW	1210	202.11	81.32	57.0	3.6
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FOR	0107	246.86	86.66	33.5	8.7
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ROW	0107	216.29	81.72	24.0	7.4
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FOR	0207	242.97	85.22	28.2	9.4
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GILLAM 2

ROW	0207	224.90	79.41	25.2	9.7
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1983 DATA

FOR	1809	171.53	88.35	49.5	4.9
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ROW	1809	153.13	87.36	47.3	7.6
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FOR	2906	211.84	72.25	21.7	10.0
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ROW	2906	183.20	82.15	22.7	10.0
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FOR	0307	216.84	78.41	24.7	5.0
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HENDAY 1

ROW	0307	208.78	82.39	24.2	6.7
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1983 DATA

FOR	1609	176.53	82.47	52.5	3.9
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ROW	1609	172.64	87.25	50.8	6.3
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DRAIN	3005	469.45	74.63	140.0	9.5
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BOG	3005	171.62	85.84	20.0	6.5
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DRAIN	1606	615.76	76.14	140.0	14.0
BOG	1606	253.53	84.55	28.5	6.8
DRAIN	0607	916.66	74.38	145.0	18.9
BOG	0607	353.06	81.31	39.5	14.4
DRAIN	2707	889.60	66.91	150.0	20.6
BOG	2707	529.04	80.45	57.0	17.9
DRAIN	2508	945.20	57.74	150.0	21.0
BOG	2508	418.95	80.54	83.0	20.8
DRAIN	2209	469.27	60.40	155.0	8.5
BOG	2209	235.47	79.18	101.7	4.3
DRAIN	2110	252.15	66.73	155.0	7.0
BOG	2110	138.44	83.70	93.3	3.4

HADASHVILLE

1983 DATA

DRAIN	1005	318.03	72.35	44.0	7.9
BOG	1005	224.04	84.30	29.3	5.1
DRAIN	3005	517.92	72.34	150.0	10.9
BOG	3005	224.90	79.82	32.5	10.7
DRAIN	2106	653.30	69.11	134.0	17.6
BOG	2106	176.81	79.03	39.0	17.9
DRAIN	0507	862.36	68.41	136.0	16.4
BOG	0507	311.36	85.39	47.0	10.5
DRAIN	1907	898.22	63.34	143.0	17.4
BOG	1907	472.04	79.98	58.7	14.7
DRAIN	1508	749.21	50.48	142.0	18.9
BOG	1508	518.75	75.57	86.7	15.4
DRAIN	509	597.98	45.28	147.0	11.6
BOG	509	412.55	76.33	100.7	11.1
DRAIN	2709	495.95	54.36	149.0	5.7

HADASHVILLE

1984 DATA

BOG	2709	310.53	74.42	107.0	4.0
DRAIN	1810	628.01	65.39	145.0	5.3
BOG	1810	177.37	84.34	94.0	4.0
BARE	3005	240.75	79.85	34.7	
BARE	2106	288.75	86.35	15.0	15.5
BARE	0507	555.73	85.45	21.0	14.0
BARE	1907	596.59	83.45	39.0	16.5
BARE	1508	484.00	81.54	63.0	18.6
BARE	0509	403.10	82.89	64.0	14.6
BARE	2709	194.05	82.15	66.0	5.7
BARE	1810	112.59	85.90	24.0	3.7

Appendix E
SUGGESTIONS TO MANITOBA HYDRO

Bog communities - ROW management: Future considerations.

This research has shown that the disturbances associated with power line right-of-ways in bogs in Manitoba can have considerable impact on the non-targeted plant species and the underlying organic soil. The severity of these disturbances, and to what extent they should be a matter of concern, is debatable. The following points are brought forward here to emphasize some of the main concerns and to suggest how further research may aid in improving existing management practices.

This study has identified three main types of disturbances in bogs along the Radisson-Dorsey and Henday-Radisson transmission lines:

(1) Construction and clearance:

The impact of these disturbances was greatest in the northern part of the study area, near Gillam. During clearance, vegetation and a thin layer of surface peat was scraped off the peat plateaus, leaving extensive, exposed peat surfaces. These bare areas remain mostly unvegetated to date and it may take several decades before the natural vegetation has colonized the right-of-way. Had more care been taken during clearance, the negative effects of these disturbances could have been minimized.

(2) Traffic:

The impact of traffic was also most noticeable on the perennially frozen peat plateaus in the Gillam area. Where access paths lie across the plateaus the resulting surface disturbance frequently leads to degradation of peat, alteration of drainage and melting of permafrost. The paths become water-saturated and often have open pools of water. The natural bog vegetation is replaced by fen-type vegetation with sedges as dominants. Due to the increase in wetness and melting of permafrost the access paths become very difficult to navigate on even with tracked vehicles. This type of disturbance is difficult to prevent, but it may be possible to decrease its impact by directing traffic onto alternative routes.

(3) Herbicides:

This study has identified the great sensitivity of non-targeted bog vegetation to the herbicides Tordon 10K and Tordon 101, about which little was known previously. Also it was shown that it is the picloram ingredient that causes the damages.

Of greatest concern is the disappearance of Sphagnum fuscum from hummocks and reduction in cover of ericoid shrubs following the applications. Repeated use of these herbicides at the same bog sites is likely to lead to a gradual replacement of the bog vegetation with fen-type vegetation

dominated by sedges and to a lesser extent by grasses. The maintenance of the natural bog vegetation and habitat in the right-of-way is desirable for several reasons:

1. Good management practices should aim at causing minimal disturbances to non-targeted species by trying to maintain the right-of-way area as natural as possible.
2. Conditions for tree growth (Black spruce and Tamarack) are generally poor in the bogs in their natural state. Reduction in shrub-cover, exposure of Sphagnum-peat and improved nutrient conditions resulting from picloram applications, may in the long-run improve conditions for tree establishment and growth at the bog sites.
3. The degeneration of Sphagnum-hummocks after picloram application will gradually give rise to wetter conditions in the treated bog areas, which will conceivably make the use of vehicles and equipment more difficult on unfrozen ground.

The long-term effects of the continued use of Tordon herbicides to control tree growth in bogs on the Radisson-Dorsey right-of-way are considered ecologically undesirable. There is a questionable need to introduce the picloram herbicides into the northern areas (Jenpeg to Gillam) where the Black spruce growth is slow and the canopy is generally open and does not interfere with the lines. At the southern bog

sites (Devil's Lake to Jenpeg), alternative means of vegetation management should be considered if economically feasible. The initial mechanical clearance operations have demonstrated that trees can be removed from the bogs in winter without causing considerable disturbances to ground vegetation or soils. Handcutting of trees in bog areas might be another alternative. The feasibility of these alternatives will of course depend on how costly they are compared with current management by herbicides.

Possible future research.

Further research is needed to answer some of the questions this study has raised. Among these are:

(a) As the bog areas in in the Radisson-Dorsey right-of-way have all been treated with Tordon herbicides recently (1979 or later), the long term effects of the Tordon applications are relatively little known. It would be of interest to investigate further the fate of Sphagnum fuscum and the associated shrub community, general changes in species composition and environmental changes in the treated bogs.

(b) Do existing management practices in the bogs improve conditions for tree reestablishment and growth?

(c) The role of ericoid shrubs (e.g. Ledum, Chamaedaphne, Kalmia and Vaccinium) in the bogs as competitors of Black

spruce and Tamarack should be investigated. Is it possible through management practices to facilitate growth of the shrubs and deter tree invasion?

(d) Are other effective herbicides, with less side effects, available to control Black spruce growth in bogs?