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A MORPHOLOGICAL INVESTIGATION INTO THE STRUCTURE OF MILLED PEAT IN STOCKPILES

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SUMMARY

Until recently, it had generally been perceived within the peat industry in Ireland that rewetting of milled peat was related to density, in that low density milled peat stockpiles are the most prone to rewetting and high density stockpiles are the least. Recent research has shown, however, that when examined volumetrically, this is not necessarily the case. In consideration of the importance that structure has on water movement and storage, a morphological investigation was conducted to see if structural images sampled *in situ* at the macroscale (> 100 μm) and microscale (> 100 μm) could be used to evaluate rewetting mechanisms. Quantification of pore and peat characteristics from impregnated blocks (by image analysis) and thin sections (by point counting) illustrated that selected milled peat types responded differently to the milling operations and while total porosity was constant, the nature of the porosity could be quite different. A two tier structure was identified, with stockpile surfaces being twice as porous as stockpile cores. In relation to rewetting, it was observed that the wettest milled peat type was related to the frequency of small pore sizes and that species composition affects the nature of water transmission and retention mechanisms.

Keywords: image analysis, micro-morphology, milled peat, mosses, pores.

INTRODUCTION

Milled peat is created by the scarification of a drained bog surface. The resultant particulate layer (10 – 50 mm thick) is atmospherically dried for up to 96 hours and then moved to stockpiles for storage. About 85% of production for electricity generation in Ireland is by the PECO system (Leinonen *et al.*, 1997) in which peat is stored in stockpiles (*ca.* 4 m high, 12 m wide and up to 1000 m long) distributed fairly evenly over a large bog area. Stockpiling is necessary to provide a winter fuel reserve since production is confined to the summer months by meteorological conditions. The value of milled peat as an energy source is determined by the water content with 45% (wet weight basis) being the optimum. Stockpiled milled peat is potentially exposed to precipitation for up to 18 months because artificial covering is not practicable for all stockpiles.

Several researchers (Aganin, 1973; Daly, 1987) have indicated that the rate of rewetting in milled peat stockpiles is closely related to the type of peat. Until recently, it had generally been perceived within the peat industry in Ireland that rewetting was related to density, in that low density milled peat stockpiles are the most prone to rewetting and that high density stockpiles are the least. Recent research (Holden & Ward, 1997) has shown, however, that when examined volumetrically this is not necessarily the case. Considering the mechanisms involved in rewetting, such as infiltration and storage, it might be expected that relationships exist with other factors including pore size distribution and species composition.

The structural architecture of a porous medium is an important factor in governing water movement. Research has shown that water flow is not always uniform owing to the occurrence of macropores that create by-pass channels (Beven & Germann, 1981; Bouma *et al.*, 1977). Previously, it was assumed that stockpile rewetting took place by wetting front advance from the surface, with wet layer depth related to initial water content and bulk density of the milled peat (Dykes *et al.*, 1997). Recent laboratory evidence (Holden & Ward, 1998) and field observation (Fig. 1), however, have indicated that by-pass flow readily occurs in milled peat. Droogers *et al.*, (1998) stated that by-pass flow was affected by the number, size, shape and distribution of macropores. There are many techniques currently available to examine porosity ranging from expensive scanning methods of computer assisted tomography (CT) (Daniel *et al.*, 1997) to physical laboratory determinations (Anderson & Bouma, 1977). A morphological approach using polished blocks (macro-scale) and thin sections (micro-scale) can be seen as a useful intermediate approach not least because it permits pore characteristics to be quantified using image analysis.

Thin sections of peat reveal detailed information regarding composition, fabric and structural arrangement (Lee, 1981). The composition of peat has important implications for water storage and movement associated with stockpile rewetting. *Sphagnum* spp. possess the ability to retain up to 20 times their own weight in water within cells/pores (Bord na Móna, 1997), therefore milled peat composition

Fig. 1: Finger storage associated with bypass flow in a field stockpile

could have considerable implication for rewetting and subsequent water storage. Landva & Pheeney (1980) examined sphagnum mosses and found that water contents in the region of 700 – 1000% (dry weight basis) were common.

This paper presents the results of a morphological investigation to examine and quantify the structural composition of stockpiled milled peat at a series of observation scales (metres to microns) and relates the findings to on-going research into the rewetting of milled peat stockpiles.

MATERIALS AND METHODS

Milled peat samples were collected from stockpiles at the Bord na Móna, Boora Works, County Offaly, Ireland (see location map in Holden & Ward, 1997) from two contrasting bogs located at Noggusboy and West Boora. Basic physical properties are summarised in Table 1 (after Holden & Ward, 1997).

Field Scale Investigation

Prior to sampling, a reconnaissance study of the bogs was conducted. Both raw bogs and milled peat stockpiles were sampled by spade, examined by hand lens and general wetness conditions and species composition recorded.

Macroscopic Scale Investigation

Following a method outlined by Mooney *et al.* (1998), resin impregnated blocks (orthogonal to the stockpile surface) were prepared *in situ*, from which digital images of milled peat were captured. It was important to preserve samples *in situ*, otherwise the pore structure would not be representative of field stockpile conditions. Image analysis was used to quantify pore and particle characteristics including number, shape, size and distribution. Pore size distribution was determined by chord analysis (Murphy *et al.*, 1977) and pore shape was calculated using circularity (Cox, 1927), where 1 is a perfect circle. Distribution of 'point to nearest neighbour distances' (Diggle, 1983) was carried out using a distance transform on an image, where the Euclidean distance to the nearest occurrence of an object of interest was measured. The scale of observation was limited by the pixel size, which in this instance was 100 μm, therefore results at this scale of observation relate to macro-porosity.

Table 1: Selected physical properties of Noggusboy and West Boora milled peat

Milled Peat Type	Relative Humification (Von Post)	Ash Content ($g\ g^{-1}$)	Poured Density ($mg\ m^{-3}$)	De Boodt Density (wet) ($mg\ m^{-3}$)	2 mm ratio†	Saturated Hydraulic Conductivity ($m\ s^{-1}$)	Porosity (%)
Noggusboy	2	0.03	0.28	0.23	4.48	5.12E-04	46
West Boora	4	0.10	0.47	0.35	1.03	3.31E-04	66

† Ratio of mass of particles >2 mm : <2 mm; a value >1 indicates a significant mass of large particles (after Holden & Ward, 1997).

Microscopic Scale Investigation

Thin sections (30 μm) were prepared using the acetone exchange by vapour method (FitzPatrick & Gudmunsson, 1978). Thin sections were examined using a polarising microscope at magnifications of up to x40, from which it was possible to make observations to 5 μm resolution. Thin sections were used to determine pore and particle characteristics and milled peat species composition (quantified by point counting). Milled peat species were classified as four components namely amorphous material, roots and other plant fragments, mosses and minerals.

Results & Discussion

Field Observation

From field examination it was possible to determine that Noggusboy (medium - low density) consisted of a considerable deposit (> 2 m depth) of humified, ombrotrophic peat of raised bog origin, rich in both *Eriophorum* spp. and *Calluna* sp. In comparison, West Boora (high density) consisted of a thin layer (150 mm) of acid fen material rich in *Sphagnum* spp., overlying an amorphous swamp fen peat deposit enriched with sedges and *Phragmites* sp. Both sites had been drained *ca.* 50 years ago by main drains, connected to a network of parallel field drains. Peat at Noggusboy bog was clearly distant from the mineral interface while West Boora bog had gone out of production and is now considered cutaway. The West Boora peat was very old (*ca.* 6000 years) and formed in fen conditions, whereas the Noggusboy peat was younger (*ca.* 3000 years) and formed by ridge-raised mire bog growth. Most interesting was the different responses of the two peats to the milling operations. Noggusboy milled peat had a much larger mean particle size (Table 1) than West Boora. This was because the milling process ripped peat into large, irregular shaped particles (typically > 5 mm) at Noggusboy in an action somewhat similar to tearing fabric. By contrast, West Boora milled peat was a fine-sized medium with many particles (< 1 mm) and plant species typically separating from the amorphous fraction suggesting a more hammer like action in response to the miller.

Stockpiles at both sites had surface wet layers and internal finger stores (Holden & Ward, 1999) at the time of sampling. Traditionally, it would have been assumed that Noggusboy milled peat would be wetter than West Boora, assuming uniform meteorological conditions (the bogs are less than 3 kilometres apart). Water contents were consistent, however, with those of Holden & Ward (1997) who recorded volumetric water contents at Noggusboy and West Boora of 0.31 $m^3\ m^{-3}$ and 0.47 $m^3\ m^{-3}$ respectively, following a sampling period at the same time of year. Field examination of water conditions was consistent with this result.

Macroscopic Observation

Quantified results from resin impregnated blocks are summarised in Table 2. Stockpile surfaces were typically twice as porous as stockpile cores, a feature that should influence the nature of the overall stockpile rewetting mechanisms. From field observations it was expected that there would be considerable differences in porosity between the milled peat types although the results illustrate that it is the nature of the porosity that varies as opposed to the total porosity. The number of pores per image show that West Boora consisted of many smaller pores whereas Noggusboy had fewer larger pores. Mean pore intercept length (MPIL - a measure of pore size) is the total number of pixels on a chord line that represents pores divided by the number of times the line intersects pore space. This indicated that the largest pores were found on stockpile surfaces, and that the larger particle size at Noggusboy was responsible for larger pores when compared to West Boora. The most porous areas (porosity > 60%) were found on the drier surfaces of the stockpile base where average particle size exceeded 10 mm. In contrast, images from the tops of stockpile surfaces had porosity values typically < 45%. This can be explained by the decreased frequency of particles > 2 mm in size towards the top of the stockpile as a result of slope processes. One factor that may influence this is the minimum detectable pore size of 100 mm whereby all pores characterised were macro-pores (defined as pores > 75 μm by Soil Science Society of America, 1997). The importance of pores < 100 μm (micro-pores) can be seen from the fact that the porosity derived by image analysis for the two milled peat types was very similar (Table 2) but, by laboratory determination (Table 1), West Boora was considerably more porous than Noggusboy.

Fig. 2 portrays a typical thresholded image (porosity = 39%) taken from the core of the Noggusboy stockpile. Fig. 3 shows the associated pore size distribution for Fig. 2 determined by the chord analysis. The graph shows that while there are a number of large pores (> 5 mm diameter), 46% of the total pores are < 1 mm in diameter. The physical arrangement observed is consistent with the theory suggested by Dexter (1993) and hypothesised for milled peat by Holden & Ward (1998), in which water flow through media with large particle sizes may occur as a series of flow paths meeting and combining to produce a funnelling effect.

The circularity index had mean values ranging from 0.83 to 0.86, which indicated a rounded pore shape in general although, from examination of Fig. 2, this does not appear to be the case. This may be explained by the large number of pores 1 and 2 pixels (100 – 200 μm) in size. These pores were so small that they were classified as circular (1) by image analysis. On this basis, it can be concluded that shape is best studied at smaller scales or by other means.

By determining 'point to nearest neighbour distances', it was possible to draw several conclusions about the structure of milled peat. The percentage of material within 0.1 mm of a pore indicated the relative pore surface area in the stockpiles. In both cases West Boora had a greater surface area that was associated with greater volumetric water content. This illustrates why volumetric water content is required to mechanistically study rewetting. West Boora milled peat had between 2 and 6% more milled peat volume nearer to pores (2 mm distance) than Noggusboy, which can be accounted for by the increased number of pores in the West Boora stockpile. This is consistent with the field observations of Holden & Ward (1997) where West Boora stored more water than Noggusboy, and with laboratory experimentation of Holden & Ward (1998) where Oughter milled peat (similar to West Boora) stored more water than Noggusboy. Stockpile surface milled peat had a greater proportion of its volume near to pores than in stockpile cores. It might be expected that core samples would have had a greater percentage of solid space within 0.1 – 2 mm of pore space than surface samples because of the greater number of pores but this was not the case. This can be explained by the fact that MPIL was greater in

Table 2: Mean pore characteristics derived by image analysis

Milled Peat	Number	Size		Shape	Distribution			
type and sample location	Number of pores per image	Porosity	Mean pore intercept length (mm)	Circularity	0.1 mm from a pore (%)	0.5 mm from a pore (%)	1 mm from a pore(%)	2 mm from a pore (%)
Noggusboy Internal	1272	0.30	0.85	0.83	20.67	65.92	86.00	96.59
West Boora Internal	1958	0.25	0.65	0.84	22.18	69.51	88.11	97.74
Noggusboy Surface	398	0.55	2.03	0.87	25.67	73.54	92.19	99.24
West Boora Surface	680	0.52	1.60	0.86	30.76	79.97	94.71	99.56

NB: Refers only to pores > 100 μm

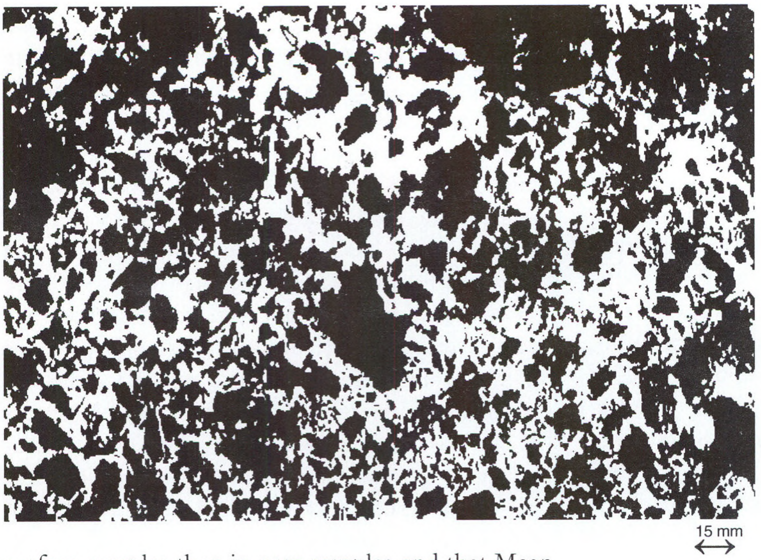

Fig. 2: A thresholded image acquired under UV light from the core of Noggusboy milled peat stockpile. Oriented orthogonal to the stockpile surface.

NB: Pore space is white, solid space is black.

surface samples than in core samples and that Mean Solid Intercept Length (MSIL – a measure of particle size) was typically 2.05 mm and 1.45 mm for cores and surfaces respectively. This was possibly because the minimum pixel size (100 μm) was too small to isolate all particles or simply by increased frequency of large particles within the core.

Microscopic Interpretation

From thin sections, it was possible to classify milled peat in terms of the nature of species composition and decomposition, as well as examine pores < 100 μm. The fragmentation and mixing of plant material and the associated pore space seen in thin section was a direct result of the milling process (Fig. 4). Species composition classification (Table 3) illustrated that both milled peats were comprised primarily of amorphous particles and loose root and stem plant material. At West Boora amorphous particles were dominant (~68%) whereas Noggusboy was less decomposed. The other components identified were minerals, which were negligible (< 1%), and mosses accounting for between 4 and 15% of the total composition. Thin sections aided the identification of species within milled peat, particularly mosses. This was important since percentage moss content has implications for rewetting owing to the water retention properties of their cellular structure (Fig. 5).

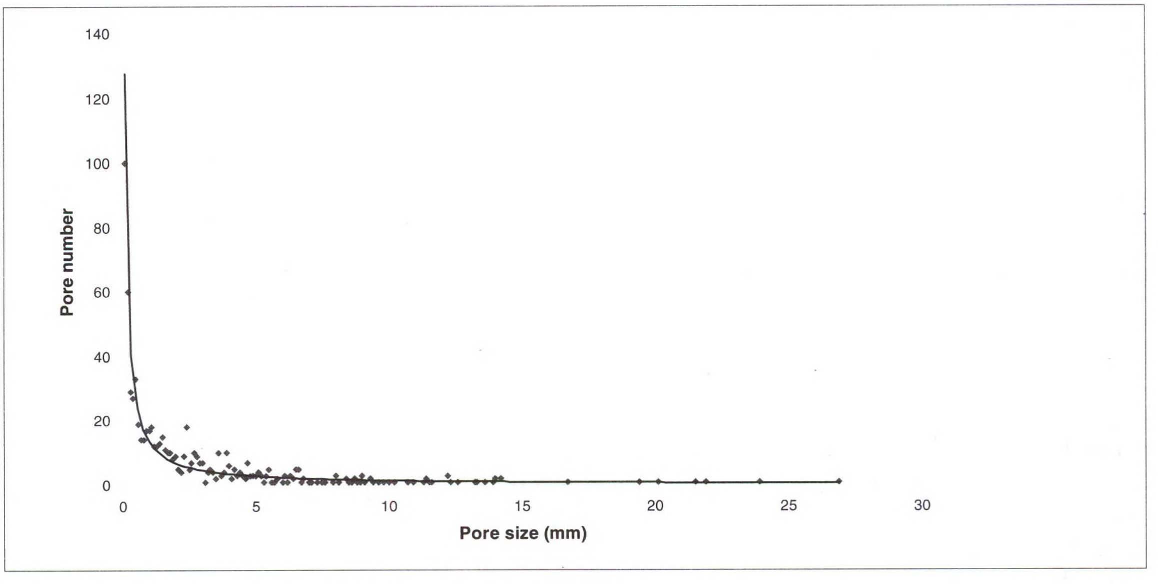

Fig. 3: Pore size distribution graph of Fig. 2 obtained by chord analysis.

Rewetting Interpretation

Holden & Ward (1997) concluded that West Boora milled peat stored more water than Noggusboy milled peat and therefore had a greater propensity to rewet. Following this they suggested, after examination of water movement in small scale laboratory stockpiles of milled peat, that a reduced resistance to flow was more significant than a change in potential gradient and a fine particulate layer would act as a barrier to infiltration. Pore characteristics at all three scales of observation would support this, in terms of mean numbers and sizes of pores at West Boora by comparison with Noggusboy. Significant differences (using one way ANOVA, $F = 404.55$ and $F_{crit} = 3.904$) were recorded by contrasting pore characteristics from stockpile cores with stockpile surfaces. The reduced porosity and associated characteristics between cores and surfaces can be used to account for the differences in volumetric water contents described at West Boora by Holden & Ward (1997). The same trend between cores and surfaces in terms of pore characteristics was identified at Noggusboy although Holden & Ward (1997) found that for volumetric water content variability was considerably reduced from that documented at West Boora. One possible reason for this could be evaporative losses from stockpile surfaces. Leinonen & Erkkilä (1994) found that the highest rates of drying from milled peat particles were associated with those between 10 – 20 mm diameter. Considering that stockpile surfaces tend to have an accumulation of larger sized particles (Mooney *et al.*, 1997), it can be speculated that the physical sorting of material may contribute to differences in core and surface volumetric water contents. When comparing these data to the industrial perception, the role of milled peat density in dictating relative wet weight water content should be considered. It is evident that for the same volume of stored water a high density milled peat will have a lower volumetric water content than a medium density milled peat.

It is generally accepted that low density milled peat stores more water than medium to high density milled peat. This can be accounted for by the water retention properties of moss species, which can occupy > 70% of total composition of low density milled peat and its relative density. Conclusions cannot be made about rewetting with regard to moss content on the basis of Table 3 since both milled peat types had little moss (less than 16%), with the wettest milled peat having the least moss. From Fig. 6, however, the effect that species composition has on water transmission is clear to see. Using methylene blue as a dye tracer, it was observed that moss (A) acted as a sponge on contact with water, absorbing readily whereas with amorphous particles (B) only the point of contact became wet. Examination of milled peat composition in thin section allows the extent of decomposition to be evaluated easily. From this it can be concluded that more rewetting occurred in the most decomposed milled peat type, suggesting that pore size distribution is more significant in relation to the propensity to rewet than the nature of species composition when the moss content is low.

CONCLUSION

It can be concluded that micro-morphology is a most useful tool for the examination of stockpiled milled peat since it provides an insight into the structural arrangement not possible by other means. Impregnated blocks and thin section illustrated that the selected milled peat types responded differently to the milling operations. By quantifying porosity and associated characteristics by image analysis, it was noted that while total porosity values were very similar, the nature of the porosities were quite different. The smaller particle and pore size distributions at West Boora can account for the greater volumetric water contents recorded previously compared to Noggusboy that appeared to be contrary to perceived expectation. It is also clear that species com-

Table 3: Mean species composition of the selected milled peats derived by point counting data

Milled Peat type	Amorphous material (%)	Roots and other plant fragments (%)	Mosses (%)	Mineral (%)
Noggusboy	56	28	15	1
West Boora	68	27	4	1

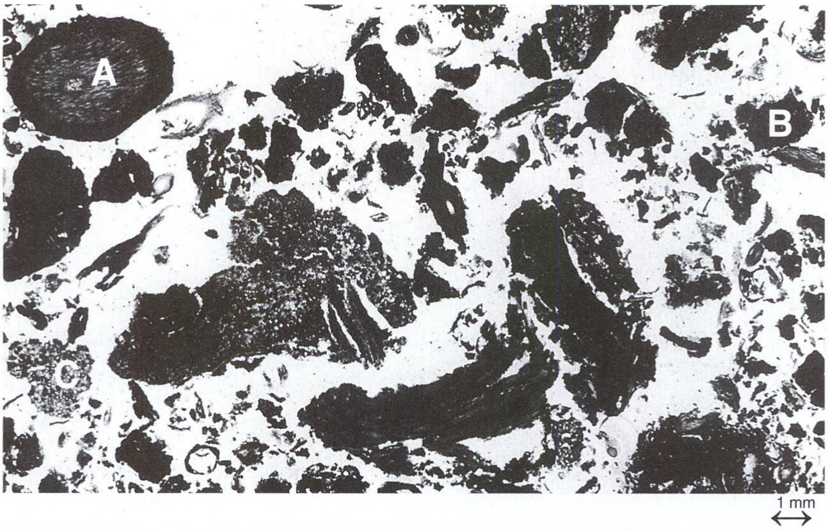

Fig. 4: Thin section photomicrograph of a horizontal section of Noggusboy milled peat showing the porous nature and species composition taken under plane polarised light. The section contains amorphous particles (A), root cross sections (B) and moss (C).

Fig. 5: A photomicrograph under plane polarised light of a Sphagnum moss fragment exhibiting a cellular internal structure from Noggusboy milled peat stockpile.

Fig. 6: Photomicrograph of stained Sphagnum moss fragments (A) and amorphous particles (B) to illustrate the difference in the nature of the staining process.

position and associated decomposition affects rewetting in terms of water retention and transmission. Whilst it should be considered that analysis of more milled peat types is required to assess fully the role structure has to play in rewetting, it can be seen that by visualising the nature of porous media, it is possible to interpret laboratory derived data regarding rewetting mechanisms.

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ATMOSPHERIC AND SPECTRAL CORRECTION OF LANDSAT TM DATA TO ESTIMATE WETLAND SURFACE ALBEDO: A CASE STUDY OF KUSHIRO MIRE, HOKKAIDO, JAPAN

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SUMMARY

In order to estimate the vegetative surface albedo from satellite data accurately, a method of atmospheric and spectral correction was developed, tested for reliability and then applied to analyze albedo distribution in Kushiro Mire, Japan. The vegetative surface albedos of the Kushiro Mire obtained using this method were then compared with observed data. Results show that the satellite inferred albedos agree well with the diurnal mean of ground observed albedos (3% systematic error). Based on estimated results, an albedo distribution in Kushiro Mire was analyzed. Results clearly indicate a surface albedo distribution pattern characteristic of the vegetation types found in the mire. The estimated albedo of *Sphagnum* spp. was smallest (0.188), followed by *Alnus* forest (0.193), *Carex* spp. (0.212) and *Phragmites* sp. (0.223).

Keywords: albedo, Landsat TM, satellite imagery, atmospheric and spectral correction, Kushiro Mire.

INTRODUCTION

While wetlands are considered to provide habitat for a significant number and variety of wild animals and plants, they are also a major source of methane gas (from anaerobic decomposition). Yamagata *et al.*, (1996) have reported that 10-40% of total methane gas is released from wetlands. Because methane gas is one of the strongest greenhouse gases (its contribution to global warming is almost comparable to that of carbon dioxide), it is important to develop accurate simulation models for monitoring micro- and local climates in wetlands.

Early studies on the importance of surface albedo for climate modelling culminated in an extensive review of the issue by Henderson-Sellers & Wilson (1983). These studies provided the fundamental linkage between surface albedos of wetlands and climate. Since then, wetland albedos have been the focus of more extensive research. For example, vegetative surface albedo has been found to be an important factor controlling the heat and water balance in wetlands (Takahashi & Nakamura, 1993). More recent research shows that vegetative surface albedo is strongly affected by groundwater level and vegetation type (Zhao *et al.*, 1997). One objective of this research has been to monitor the distribution of vegetative surface albedo over time and space with greater precision and accuracy.

Before satellite observation became possible, the accuracy of applying traditional locally based physiological models to global scales was limited. (Typically, representative values of albedos obtained at specific sites within an area were used to estimate a water and heat balance for an entire area). In addition, in a large wetland, it is very difficult to obtain representative data for the whole wetland through field observations alone owing to the complexity, vastness, and limited access ability of the wetland. By using satellite remote-sensing techniques, however, albedo can be determined at the pixel level over an entire wetland. This allows more accurate investigation of some important earth science problems.

While satellite remote sensing allows continuous data collection for large areas, the accuracy of some estimated parameters (e.g. albedo, surface temperature, and Leaf Area Index) is limited by the geometric and radiometric errors inherent in satellite data. Therefore, before these parameters can be incorporated effectively into water and heat balance models, corrections must

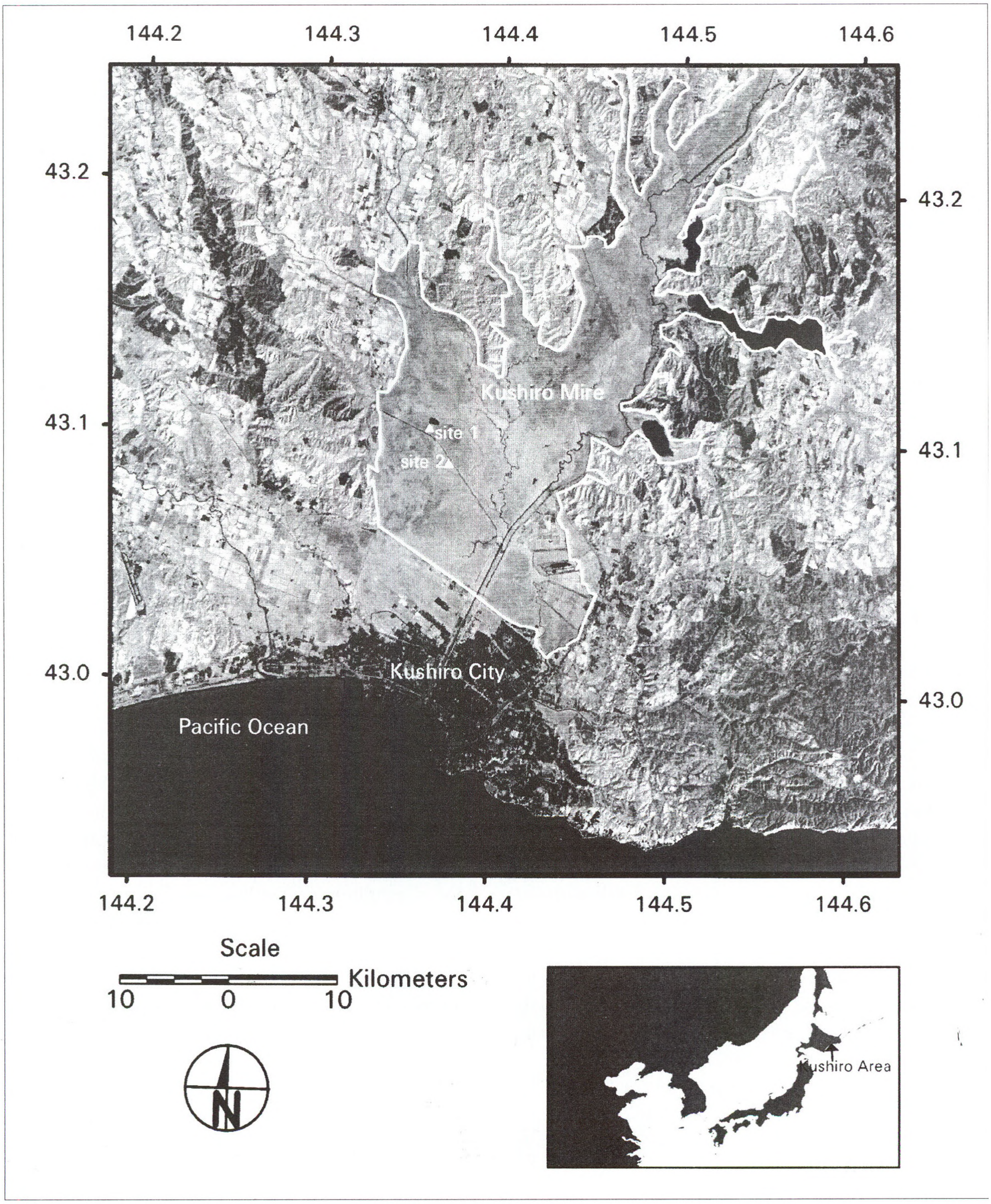

Fig. 1: Landsat TM image of Kushiro Mire and reference map of Japan (Red: band 5, Green: band 4, Blue: band 3; Aug. 26, 1996; provided by National Space Development Agency of Japan).

be made. The purpose of this study was to develop a new method to more accurately estimate surface albedo with Landsat TM data and to apply these estimates to analysis of albedo distribution in Kushiro Mire, Hokkaido, Japan.

REGIONAL DESCRIPTION

Kushiro Mire faces the Pacific Ocean on the eastern side of Hokkaido Island, Japan (Fig. 1). The mire is otherwise surrounded by steeply sloped hills and marine terraces that rise to a maximum height of 140 metres. The topography of Kushiro Mire is characterized as low and flat with a mean altitude below 10 metres above sea level.

Peat covers 80% of the mire's total surface area of 29,000 hectares, while the remaining, primarily coastal portions, consist of sand dunes. Of the peat-covered portion, only 1.4% is transitional or high mire peat, with the rest being low mire peat. Vegetation in the low mire is primarily made up of communities of *Phragmites*, *Alnus-Phragmites* and *Alnus-Carex*. Alternatively, high mire vegetation is dominated by *Sphagnum* with some *Phragmites* communities interspersed as a result of existing inorganic soil and mineral nutrients from the Kushiro River. There are no notable representative vegetation communities in the transitional area of Kushiro Mire.

METHODS

Daily surface albedo parameters are utilized in water and heat balance models. Although satellite data provide a very rich information source for determining surface albedos, a series of important corrections are required before these data can used[1] reliably and accurately.

Fig. 2 provides an overview of the methods. Satellite data are first split into two sets of bands – visible and infrared. Then planetary albedos (A_{p1} and A_{p2}), which are observable from satellite directly, are converted to surface albedos (A_{g1} and A_{g2}) using atmospheric correction (surface albedo is not directly observable from satellite data because of atmospheric interference). Explanation of the relationship between planetary albedo and surface albedo is provided in Appendix 1. The two resulting filtered surface albedos (A_{g1} and A_{g2}) are then spectrally corrected with unfiltered instantaneous surface albedo (A_g). Finally the corrected instantaneous surface albedo (when satellite passes over the area) is converted to daily surface albedo.

ATMOSPHERIC CORRECTION

General Description

Based on previously defined atmospheric parameters (Lacis & Hansen, 1974), Chen & Ohring (1984) derived a linear relationship between clear-sky planetary albedo and surface albedo. Researchers have since encountered three problems when using their model on satellite data. Firstly, the parameters used in the model are applicable only to unfiltered solar radiation. Because satellite sensors provide data with filtered solar radiation, it is necessary to convert these parameters to the filtered range of satellite bands (Nakagawa & Ooi, 1992).

Secondly, original parameterization suggests significant effects of water vapour absorption for wavelengths greater than 0.7 μm, and of ozone absorption and Rayleigh scattering for wavelengths less than 0.7 μm. In Chen & Ohring's model, however, wavelength based corrections for water vapour absorption, ozone absorption, and Rayleigh scattering were not considered. Because it is well known that optical effects of the atmosphere in visible and infrared bands are different (Koepke, 1989), these wavelength regions must be separated before determining atmospheric effects. The filtered range of Landsat TM bands is 0.45-2.35 μm (band 1-5 and 7), and includes both visible and infrared bands. To correct for wavelength effects, the filtered range of Landsat TM bands were separated into two parts: wavelengths shorter than 0.7 μm (bands 1, 2, and 3) and wavelengths equal to or longer than 0.7 μm (bands 4, 5, and 7). These were named Spectral Region 1 and Spectral Region 2, respectively (Fig. 2). The atmospheric correction formulas are different for the two parts.

Thirdly, in Chen & Ohring's model, Mie scattering (aerosol) was not considered. Therefore, in this study, the effects of Mie scattering are corrected by subtracting path radiance from the total radiance

[1] *To estimate surface albedo from satellite data accurately, it is necessary to correct for geometric, atmospheric, spectral and topographic effects. Because the data used in this study (obtained from RESTEC – Remote Sensing Technical Center) were previously geometrically-corrected, no further geometric correction was needed. In addition, there is no need to correct for topographic effects in the data because Kushiro Mire is flat.*

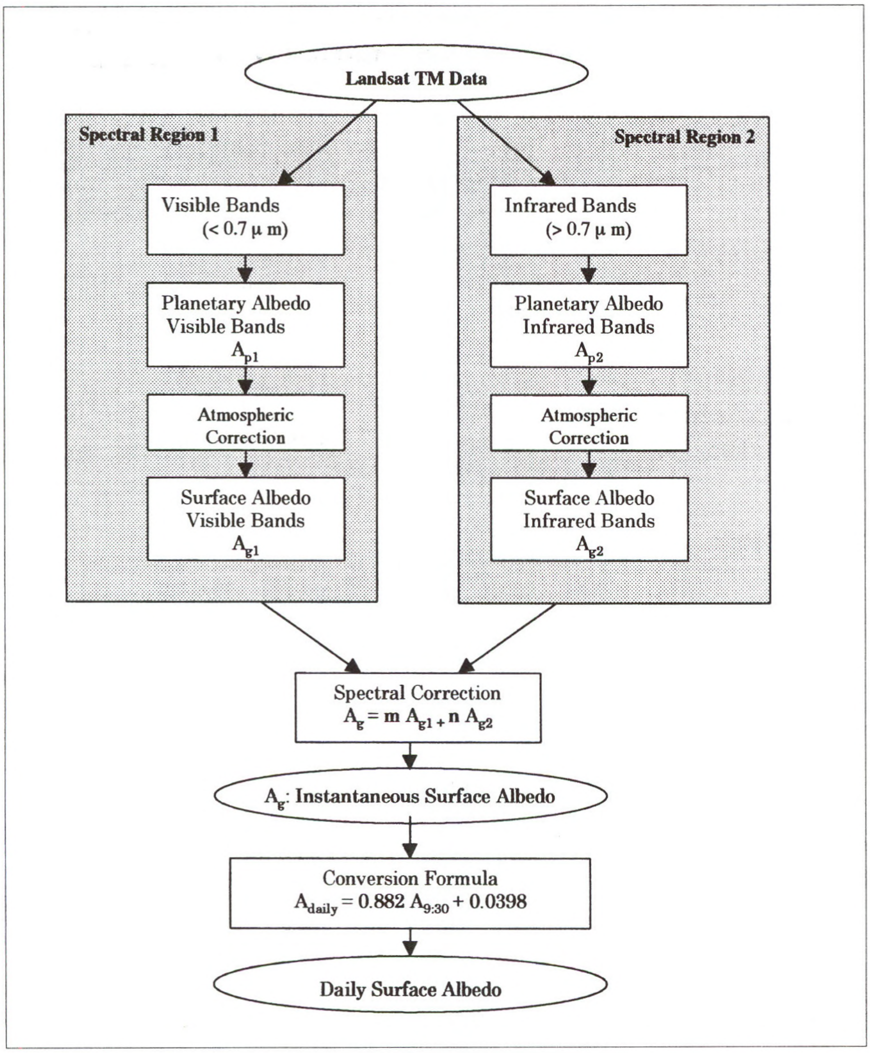

Fig. 2: Method of daily surface albedo estimation from Landsat TM data.

that reached the sensor. A detailed discussion of the atmospheric correction conducted in this study follows.

Detailed Method for Atmospheric Correction

The filtered planetary albedos (A_{p1} and A_{p2}) were obtained for Spectral Regions 1 and 2, respectively, using the following formulae:

Equation 1

$$A_{p1} = \frac{\{1+0.0167\sin[2\pi(J-93.5)/365]\}^2 \times \pi \sum (L_{S_i} - L_{P_i})}{138.2 \times 0.2716\mu}$$

(i=1, 2, 3)

Equation 2

$$A_{p2} = \frac{\{1+0.0167\sin[2\pi(J-93.5)/365]\}^2 \times \pi \sum L_{S_i}}{138.2 \times 0.1441\mu}$$

(i=4, 5, 7)

where μ is the sine of the solar elevation, i is the band number, J is the Julian day, L_s is the total radiance that reached sensor, and L_p is the path radiance resulting from Rayleigh scattering and Mie scattering (aerosol). The path radiance L_p decreases as wavelength increases and, for wavelengths longer than 0.7μm, L_p tends to zero[2]. The sum of the terms in the brackets in both

[2] *Path radiance: Because of scattering alone, radiation can reach the sensor from adjacent pixels and also via diffuse scattering of the incoming radiation that is actually scattering towards the sensor by the atmospheric constituents before it reaches the ground. Path radiance (Equations 1 and 2) was estimated from field measurements using a spectroradiometer and Landsat TM data.*

equations provides the necessary correction for the distance between the sun and earth (Gurney & Hall, 1983). Finally, 138.2w/m$_2$ is the total solar constant, and 0.2716 and 0.1441 are ratios of filtered solar constant to total solar constant in spectral regions 1 and 2, respectively (Thekaekara & Drummond, 1971).

Nakagawa & Ooi (1992) initially modified Chen & Ohring's (1984) general model in order to obtain surface albedo from planetary albedo using Landsat MSS data. Much the same method was applied to Landsat TM data in this study. The additional distinction between spectral regions 1 and 2 was made. As a result Nakagawa's equations deriving filtered surface albedo from filtered planetary albedo were modified in the following two equations.

Equation 3

$$A_{p1} = 1 - A_{O3}(x) - R_a[A_{O3}(x^*) - A_{O3}(x)] - \frac{1 - R_r - A_{O3}(x)}{(1 - R_r^* A_{g1})} +$$
$$\{\frac{1 - R_r - A_{O3}(x)}{(1 - R_r^* A_{g1})} + \frac{(1 - R_a)(1 - R_a^*)}{1 - R_a^* A_{g1}} \times [A_{O3}(x^*) - A_{O3}(x)]\} A_{g1}$$

Equation 4

$$A_{p2} = [1 - A_w(y^*)] A_{g2}$$

where A_{O3} is ozone absorptivity, A_w is water vapor absorptivity, R_r is the Rayleigh albedo of the total atmosphere for the incoming solar radiation flux, R_r^* is the total atmosphere for the radiation from below, R_a is the Rayleigh albedo of the atmosphere underlying the ozone layer for the incoming solar radiation flux, and R_a^* is the spherical albedo of the atmosphere underlying the ozone layer for radiation from below. Additionally, x^* and y^* are the respective total path lengths of ozone and water vapor traversed by the reflected radiation in reaching the top of the atmosphere. x and y are the total atmospheric ozone and water vapor path lengths traversed by the direct solar beam in reaching the earth's surface, respectively (Nakagawa & Ooi, 1992). These equations define the relationship between the filtered planetary albedo for Spectral Regions 1 (A_{p1}) and 2 (A_{p2}) and the filtered surface albedos for Spectral Region 1 (A_{g1}) and Spectral Region 2 (A_{g2}).

Based on equations (3) and (4), filtered planetary albedos were first simulated using five types of data: filtered surface albedo, surface air temperature, surface atmospheric pressure, surface water vapour pressure and total ozone. Using actual climatic data from Kushiro Mire on August 26, 1996, the numerical relationship between filtered planetary and filtered surface albedos[3] (as defined by equations 3 and 4) is shown below in equations 5 and 6:

Equation 5

$$A_{g1} = 1.0872 A_{p1} - 0.0327$$

Equation 6

$$A_{g2} = 0.9891 A_{p2}$$

In the above example, the parameters of surface air temperature, surface atmospheric pressures, surface water vapor pressures, and total ozone are assumed to be constant for the entire Kushiro Mire region.

Spectral Correction

General Description

One additional technical problem occurs when surface albedo is estimated from satellite data. Satellite sensors provide only filtered albedo for a narrow spectral region. Therefore, filtered albedo must first be converted to broad band albedo. However, only broad wavelength albedo is useful for modelling. In order to make this conversion, a method for determining the specific spectral correction weights for both spectral regions is utilized.

To determine the spectral correction weights, short wavelength, long wavelength and total wavelength albedos were measured for the seven major types of vegetation on Kushiro Mire. Spectral conversion coefficients were then obtained for each type of vegetation using weighted averages. As a final step, the

[3] *The temperature (T_o), atmospheric pressure (p_o) and water vapor pressure (e_o) are easily obtained from conventional meteorological observations. Solar elevation can be obtained from header information of satellite data. Although, total ozone amount (u) is only observable at major meteorological stations, it has been shown that total ozone amount is not sensitive to atmospheric coefficients. Therefore, total ozone amount values from the nearest meteorological station can be used. An example of the relationship between planetary and surface albedos estimated by equations (3) and (4) under the climatic conditions that existed on August 26, 1996 at Kushiro Mire; i.e., T_o = 289.4 K, e_o = 13.8 hPa, p_o = 1012.3 hPa, u = 0.321 atm-cm, and solar elevation = 49°.*

weighted average coefficients for each spectral region were averaged to obtain final spectral correction regression coefficients for both spectral regions. To make the conversions in this section, it was assumed that the albedos for spectral regions 1 and 2 (A_{g1} and A_{g2}) were approximately equal to the short and long wavelength albedos ($A_{<0.7\mu m}$ and $A_{\geq 0.7\mu m}$), respectively.

Detailed Spectral Correction Method

Broad wavelength albedo can be estimated using a weighted average of short and long wavelength albedos. In an ideal state this relationship is expressed as:

Equation 7

$$A_{broad} = m\, A_{<0.7\mu m} + n\, A_{\geq 0.7\mu m}$$

Equation 8

$$m + n = 1.0$$

where $A_{<0.7\mu m}$ is short wavelength albedo ($< 0.7\mu m$), $A_{\geq 0.7\mu m}$ is long wavelength albedo ($\geq 0.7\mu m$), A_{broad} is broad wavelength albedo, and m and n are weights. If $A_{<0.7\mu m}$, $A_{\geq 0.7\mu m}$ and A_{broad} are known, it is possible to solve for m and n. In an ideal state, the sum of m and n is 1.0 representing the total energy of the whole wave region.

To obtain $A_{<0.7\mu m}$, $A_{\geq 0.7\mu m}$ and A_{broad}, field observations using a LI-1800 portable spectroradiometer (with wavelength range of 300 – 1100 nm and scan interval of 1 nm) were carried out on August 26-27, 1996. Using this instrument, the spectral reflective and incident radiance of seven types of vegetation were measured.

Albedo is theoretically defined as the ratio of reflective radiance to incident radiance integrated across the whole wave region. However, because the scan interval of the LI-1800 portable spectroradiometer is very small (1 nm), it was possible to use the sum of the radiance per nanometre as a practical and accurate substitute for integration[4]. Therefore, $A_{<0.7\mu m}$, $A_{\geq 0.7\mu m}$ and A_{broad} can be rewritten as:

Equation 9

$$A_{<0.7\mu m} = \frac{\sum_{i=1}^{400} R_i}{\sum_{i=1}^{400} I_i}$$

[4] $A = \frac{\int R}{\int I} \approx \frac{\sum Ri}{\sum Ii}$

Equation 10

$$A_{\geq 0.7\mu m} = \frac{\sum_{i=401}^{800} R_i}{\sum_{i=401}^{800} I_i}$$

Equation 11

$$A_{broad} = \frac{\sum_{i=1}^{800} R_i}{\sum_{i=1}^{800} I_i}$$

where R_i is the measured reflected radiance, I_i is the measured incident radiance, and i is the instrument band number. Results of calculated albedos using equations 9, 10 and 11 for all seven types of vegetation are shown in Table 1.

While the objective is to convert filtered albedo to broad band albedo, the spectral range of the LI-1800 is limited owing to the spectral characteristics of its sensor. Since the wavelength range of the LI-1800 is 300-1100nm equation 7 and 8 are converted to reflect the limitation of the instrument's sensor. The resulting conversion is shown in equation 12 and 13. (Note that the sum of the weights m and n changes from 1.0 to 0.7441. This is because the LI-1800 instrument measures only 74.41% of the total energy in the entire wave region).

Equation 12

$$A_{total} = m\, A_{300\text{-}700} + n\, A_{700\text{-}1100}$$

Equation 13

$$m + n = 0.7441$$

Resulting weighted average coefficients m and n for all seven types of vegetation are shown in Table 2. The mean values of m and n for 7 types of vegetation were used to estimate the coefficients of spectral correction.

As mentioned above, to use Landsat TM data it was assumed that the albedo of Spectral Region 1 is equal to the albedo of the short wavelength region (<0.7 µm) and that the albedo of Spectral Region 2 is equal to the albedo of the long wavelength region ($\geq 0.7\mu m$). Thus, as a final step, equations 12 and 13 were modified in the following way:

Equation 14

$$A_g = m'\, A_{g1} + n'\, A_{g2}$$

Equation 15

$$m' + n' = 1,$$

Table 1. Results of measurement by the LI-COR portable spectroradiometer

Object	Albedo 0.3µm< λ <0.7µm	Albedo 0.7µm< λ <1.1µm	Albedo 0.3µm< λ <1.1µm
Carex	0,044	0,344	0,165
Sphagnum (wet)	0,032	0,392	0,178
vegetation	0,017	0,112	0,055
Sphagnum (red)	0,053	0,396	0,188
Sphagnum (dry)	0,076	0,439	0,219
Phragmites	0,046	0,445	0,204
Carex	0,052	0,447	0,208

Table 2. Weighted average coefficients for spectral correction

Object	m 0.3µm<λ<0.7µm	n 0.7µm<λ<1.1µm
Carex	0,302	0,442
Sphagnum (wet)	0,317	0,427
vegetation	0,302	0,442
Sphagnum (red)	0,310	0,435
Sphagnum (dry)	0,296	0,449
Phragmites	0,320	0,424
Carex	0,315	0,429
average	0,309	0,435

and,

Equation 16

$$m' = m / 0.7441$$

Equation 17

$$n' = n / 0.7441$$

where m' is the spectral correction weight for surface albedo in Spectral Region 1, and n' is the spectral correction weight for surface albedo in Spectral Region 2.

Based on the above formulation and using averaged coefficients from Table 2, the coefficients of the regression formula for spectral correction are given below:

Equation 18

$$A_g = 0.415\ A_{g1} + 0.585\ A_{g2}$$

CALCULATION OF DAILY ALBEDO FROM LANDSAT TM DATA

Daily surface albedo parameters for use in water and heat balance models have traditionally been developed based on field observation. While traditional field observation provides a limited set of data points, satellite readings provide pixel level data with better spectral, spatial and radiometric characteristics. Nevertheless, Landsat TM data allows only instantaneous readings at one point in time during any day. Therefore, to fully utilize Landsat TM data to estimate daily surface albedo for water and heat balance models, instantaneous surface albedos must be converted to daily surface albedos.

Landsat TM satellite has a repeat cycle of 16 days in which image data are acquired nominally at 9:30 a.m. local time in a near polar, sun synchronous orbit. Consequently, instantaneous albedo at 9:30 a.m. can only be estimated from Landsat TM data on days when the satellite passes over the area.

As a final step in this method, an experimental formula for estimating daily albedo from Landsat TM instantaneous data was developed. Over a four-year period, from June 23, 1993 to June 20, 1997, reflective radiance and incident radiance were measured using two pairs of actinometers at ten-minute intervals in high and low mires. A total of 1673 albedos at 9:30 a.m. and daily albedos were obtained at the two sites. The total variation explained (r^2) was 0.8697. The regression equation is written below.

Equation 19

$$A_{daily} = 0.882\ A_{9:30} + 0.0398$$

This equation shows that a 1.0 increase in instantaneous surface albedo is associated with a 0.882 increase in daily surface albedo (standard error of the regression equation is 0.08). Thus, instantaneous albedo at 9:30 from Landsat TM data was estimated first, and then daily albedo was obtained using the above equation.

RESULTS

Reliability

To ensure reliability and accuracy of this method, albedos estimated from Landsat TM data were compared to albedos measured in the field by an albdeo meter. The vegetative surface albedo in Kushiro Mire was estimated from seven scenes of Landsat TM data, which were received on July 1, 1993; May 17, 1994; July 13, 1994; August 5, 1994; June 14, 1995; August 17, 1995; and August 26, 1996. Field observations were carried out at two sites at the same time as the overpass of the Landsat satellite. Reflective radiance and incident radiance were measured by using two pairs of actinometers at ten-minute intervals, and the data were converted to surface albedo in the laboratory. Results of the comparison (Fig. 3) show that the estimated albedo closely approximates the observed albedo. The average error of estimated surface albedo by this method was found to be about 3%.

Validation: Distribution of Vegetative Surface Albedo in Kushiro Mire

To provide some indication of the general validity of this method two applications of the method were conducted. Firstly, distributions of vegetative surface albedo in Kushiro Mire were compared for two different times of the year. Secondly, albedo distribution based on elevation was analyzed.

Previously, Oguma & Yamagata (1997) developed a vegetation map for Kushiro Mire using Landsat TM multi-temporal data. The accuracy of their vegetation classification was shown to be 99.7%. Based on this vegetation map, vegetation type was matched with estimated daily surface albedo from satellite data at the pixel level in Kushiro Mire for July 13, 1994 and August 26, 1996. Table 3 shows the resulting average albedo, maximum albedo, minimum albedo and the number of pixels of major vegetation type in Kushiro Mire. Upon comparison, there is a clear relationship between the distribution patterns of the albedo and vegetation.

Data for July 13,1994 indicate average surface albedos to be 0.172 for *Sphagnum*, 0.191 for *Alnus*, 0.207 for *Phragmites* and 0.218 for *Carex*. These surface albedo distributions show that the albedo of *Sphagnum* was the smallest, followed by *Alnus* forest, *Phragmites*, and *Carex*. Data obtained for 26 August 1996 show, however, that the albedo of *Sphagnum* was the smallest (0.188), followed by *Alnus* forest (0.193), *Carex* (0.212) and then *Phragmites* (0.223). The primary difference between these two dates is the order of *Carex* and *Phragmites* albedos.

The reason that the albedo for *Phragmites* is larger than *Carex* in August, while the reverse is true for July is explained by the different temporal growth-patterns of the two plants. *Carex* begins to grow in late May, reaches a maximum size in early August, and

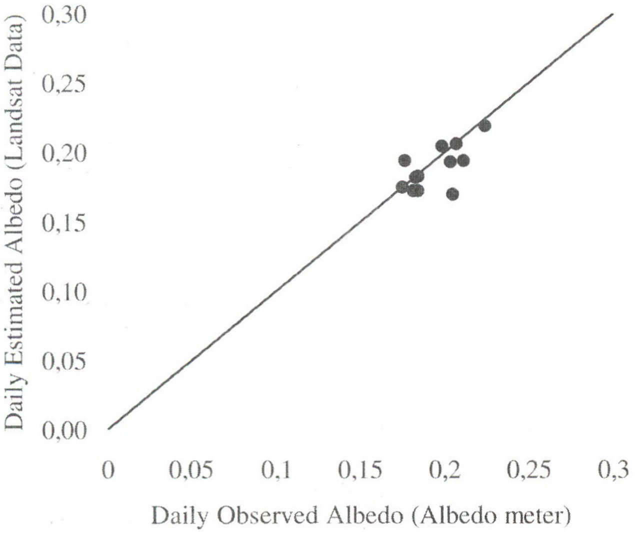

Fig. 3: Comparison of estimated and observed daily albedos.

Table 3. Distribution of vegetative surface albedo in Kushiro Mire (July 13, 1994 and August 26, 1996, with LANDSAT TM data, row-path 106-30)

Classification	Albedo			
	AVR	MAX	MIN	NUM
1994				
Sphagnum	0,172	0,185	0,157	5800
Alnus	0,191	0,219	0,167	106000
Alnus ·*Phragmites*	0,205	0,231	0,170	36600
Phragmites	0,207	0,229	0,185	24900
Phragmites ·*Carex*	0,215	0,235	0,189	34800
Carex	0,218	0,249	0,187	97000
pool	0,137	0,174	0,067	2500
water	0,020	0,083	0,012	8700
1996				
Sphagnum	0,188	0,196	0,185	5800
Alnus	0,193	0,222	0,175	106000
Alnus ·*Phragmites*	0,211	0,227	0,189	36600
Phragmites	0,223	0,228	0,217	24900
Phragmites ·*Carex*	0,220	0,229	0,187	34800
Carex	0,212	0,217	0,206	97000
pool	0,141	0,185	0,071	2500
water	0,032	0,058	0,002	8700

then begins to wither. *Phragmites* begins to grow in late June, reaches a maximum size in late August, and then dies back (Yamagata *et al.*, 1996). These results seem to provide some evidence that the method developed in this paper accurately estimates daily surface albedo.

The second application of the method compares albedo distributions in high and low mires. It was found that the vegetative surface albedo in high mire (where *Sphagnum* dominates) is lower than in low mire (where *Phragmites*, *Carex*, and *Alnus* dominate).

Comparisons of these findings from both applications with field observations from Kushiro Mire confirm the general results and tend to validate the method.

CONCLUSION

This paper develops a new method of atmospheric and spectral correction for estimating vegetative surface albedo using Landsat TM data. In this method, five input parameters (temperature, atmospheric pressure, water vapor pressure, total ozone amount and solar elevation) are needed. An experimental formula to estimate daily albedo from instantaneous Landsat TM data was also developed.

To test reliability, the estimated surface albedo was compared with field measured albedo. The average error was found to be about 3%. To provide some indication of the validity and accuracy of this method, this was used to estimate the distribution of surface albedo in Kushiro Mire. Two primary results were found. Firstly, the distribution patterns of surface albedo at two different times of the year, correspond with known vegetation growth patterns. Secondly, vegetative surface albedo in high mire is lower than that in low mire. These results also agree with actual field observations. The reliability and accuracy of estimated surface albedo from Landsat TM data using this method suggests the possibility for further application.

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Appendix 1:

In this diagram (Fig. 4), observable planetary albedo (A_p, top) is separated from surface albedo (A_g, bottom) by the atmosphere. Both albedos were calculated as reflectance (R) divided by incidence (I). The method described in this paper develops a way of accurately and reliably accounting for atmospheric effects such that surface albedo can be estimated from planetary albedo.

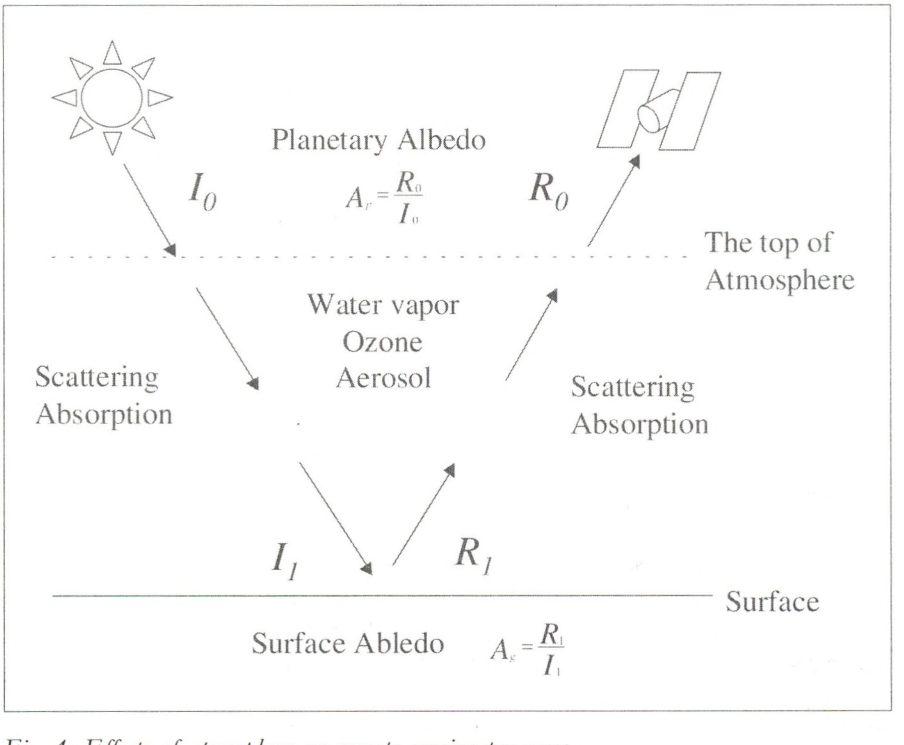

Fig. 4: Effects of atmosphere on remote sensing processes.

STREAMFLOW, WATER QUALITY AND SUBSTRATE CONDITIONS OF A DRAINED BLACK SPRUCE PEATLAND, QUÉBEC, CANADA.

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SUMMARY

A comprehensive study was initiated in a black spruce (*Picea mariana* (Mill.) B.S.P.) peatland of eastern Québec, Canada, to evaluate the effects of drainage on substrate conditions, tree growth, streamflow and water quality. Considering the drawdown due to drainage during periods of highest and lowest water table as well as average seasonal values, the 20- and 30-m spacing showed lower water table than the 40-, 50- and 60-m spacing, which were efficient at midpoint between the ditches only during dry periods. Water table lowering was substantial in the first 5 m from the ditches and minimal beyond 15 m. Peat moisture content was reduced by 20-29% and maximum summer soil temperatures increased by 1.5 to 3.5°C at 10-cm depth, while rates of decomposition were accelerated mainly at the 30-cm depth. Drainage increased nutrient content of the soil water solution proportionally to ditch closeness. Base flow was increased by at least 0.7 L s^{-1}, while total runoff was increased by 0.9 L s^{-1} which represents a 25% increase due to drainage. The concentration of suspended sediments significantly increased only during ditching and the following four weeks. Streamflow temperature fluctuations were increased after drainage, by lowering of the average weekly minimum and increasing the average weekly maximum, by 2 and 7°C, respectively. The treatment increased electric conductivity of surface water, which was clearly related to increases in mineral N (NH_4 + NO_x), Ca, Mg, Na and S concentrations.

Keywords: forest drainage, water table, soil water, streamflow, water quality, nutrient leaching.

INTRODUCTION

Canada's vast areas of peatlands represent a great potential for forestry (Hillman, 1987). In the Province of Québec, peatlands cover 6.3% (2.7 million ha) of the productive forestlands, south of the 50th parallel (Bolghari, 1986). Peatlands are principally concentrated in lowlands of some regions such as Abitibi-Témiscamingue, Lake Saint-John and St. Lawrence valley, where they represent a large potential for wood production. Drainage for peatland forestry has been practised in the Province on 4000 ha y^{-1} in the 1980's increasing to 5000 ha y^{-1} in 1998.

Because of poor substrate conditions, the fibre productivity of forested peatlands is generally low. Numerous studies, principally conducted in Fennoscandia, have demonstrated that tree growth can be improved by forest drainage (Paavilainen & Päivänen, 1995). Generally, growth increase following drainage is not immediate because tree resources are initially allocated to the development of the root system and leaf area. The length of time to obtain a growth increase varies with species and site conditions. Changes created in the growing substrate can be observed early after drainage and are indicative of the potential for stemwood growth response.

Lowering of the water table by drainage is inversely proportional to ditch spacing (Braekke, 1983; Belleau *et al.*, 1992; Prévost *et al.*, 1997). This lowering decreases the water content in the top layer and allows a better aeration for the root physiological processes (Kramer, 1987). Water table depth, however, is not necessarily a good indicator of the moisture content of the overlying soil (Munro, 1984; Rothwell *et al.*, 1996) and consequently of its aeration. The high water holding capacity of peat and the associated capillary rise as well as the surface subsidence following drainage suggest that aeration improvement may be small under the rainy conditions that prevail in the Province of Québec. Near-surface soil temperature rise and accelerated organic matter decomposition are other expected positive effects of drainage that have to be verified under the ecological conditions that operate there.

Besides improving soil conditions for tree growth, drainage may cause many environmental changes in the peatland ecosystem (Sheehy, 1993; Paavilainen & Päivänen, 1995). Summer low flows and total runoff are usually increased during a few years but decreases can occur owing to enhanced vegetation growth (Lundin & Bergquist, 1990). Contradictory results were reported on the effects of drainage on peak flows. The combination of factors such as drainage efficiency, proportion of the basin drained, soil type, vegetation and weather conditions has resulted in peak flow increases or decreases following drainage (Lundin, 1988; Berry *et al.*, 1995; Paavilainen & Päivänen, 1995). The overall effects remain difficult to predict indicating a need for more regional information. The quality of surface water may also be affected by drainage. The Fennoscandian literature reports increased concentrations of N, P, K, Hg and suspended sediments, and variable effects on leaching of Fe and Al (Paavilainen & Päivänen, 1995). Wetlands are very sensitive to disturbance and the environmental impacts of drainage must be considered as well as growth response.

In 1990, an experiment was set up to study the effects of forest drainage on hydrological processes, substrate conditions, water quantity and quality, and black spruce (*Picea mariana* (Mill.) B.S.P.) growth, which have received very little attention in eastern Canada. The first 5-year results are summarized in this paper.

MATERIAL AND METHODS

Site Description and Experimental Design

The study was conducted in a pure black spruce peatland located 25 km east of Rivière-du-Loup, Québec, Canada (47° 49' N, 69° 15' W). The climate of the region is cold and humid, with mean annual air temperature of 2.3°C and precipitation 1015 mm. The frost-free period extends from 1st June to 15th September (Ferland & Gagnon, 1967) and the potential evapotranspiration is evaluated at 493 mm (MEFQ, 1991).

The study was set up as a paired watersheds experiment (Hewlett, 1982), using two small (20 and 18 ha) headwater basins (Fig. 1). The forested peatland, surrounded by upland areas, covers 12 ha (drained: 8 ha; control: 4 ha) and supports a 70 years old black spruce stand with an average of 4900 stems ha^{-1}, 80% of which had diameter at breast height (dbh) less than 10 cm. Mean dbh and height of the main canopy were 13 cm and 10 m, respectively. *Ledum groenlandicum* Oeder, *Kalmia angustifolia* L. and *Vaccinium myrtilloides* Michx. were the most common shrubs and *Sphagnum capillifolium* (Ehrh.) Hedw., *S. fuscum* (Schimp.) Klinggr., *S. russowii* Warnst., *S. magellanicum* Brid., *Pleurozium schreberi* BSG Mitt. and *Dicranum* spp. the dominant mosses. Peat bulk densities (0.072 to 0.080 mg m^{-3}), C:N ratios (around 70) and saturated hydraulic conductivity (5.7 x 10^{-6} to 12.6 x 10^{-6} m s^{-1}) indicated a weakly decomposed peat material throughout the 110-cm profiles (Prévost *et al.*, 1997). Peat characteristics and vegetation are indicative of ombrotrophic conditions (bog) and the site corresponds to the fibric to mesic organic soil stands (S14) of the *Ledum* operational group (OG11) in the forest ecosystem classification for the Ontario clay belt (Jones *et al.*, 1983).

The paired watersheds were calibrated from June 1990 to late September 1991 when digging was carried out with a Caterpillar excavator equipped with a U-shaped bucket. The ditches, 1 m deep and 50 cm wide at the bottom with side slopes of 1:1 delimited one 20-, 30-, 40- and 50-m wide spacing. Two 60-m spacings were established on both sides of a forest road bordered by old ditches that were without outlet before digging of the main ditch.

Data Collection

Water table levels were measured in 5-cm diameter perforated plastic pipes inserted 120 cm into the peat at different distances from the ditches. Depth of water table was measured weekly with an electrical buzzer probe, from June to October 1990-1995. All water table measurements were referenced to the peat surface. From June 1991, the water table level was also recorded at the midpoint of the 40-m spacing and at the control site with Leopold-Stevens water-level recorders.

Peat water content and temperature were measured at 10-, 20-, 30- and 40-cm depths, in profiles located at 5, 10, 15 and 20 m from the center of the drainage ditch in the 40-m spacing and at one location in the control site. Peat water content was estimated with the time domain reflectometry (TDR) described by Topp *et al.* (1980) and calibrated in pe

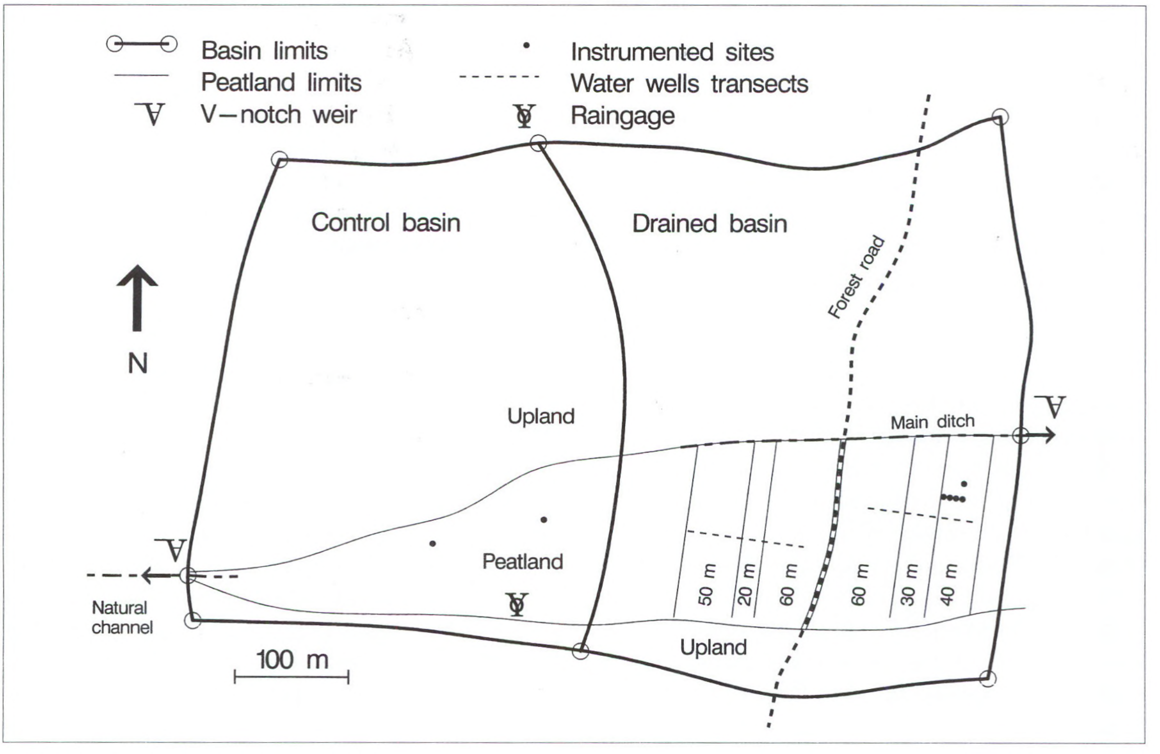


Fig. 1: Experimental area and design.

by Pepin *et al.* (1992). TDR measurements were taken weekly with a cable tester (Tektronix, model 1502, Beaverton, Oregon, USA) from June to October 1991-1995 in the drained site and 1993-1995 in the control site. Peat temperature was monitored with thermistor probes (107B) connected to dataloggers, from June 1991 to October 1995 in the control site and at 10 and 20 m from the ditch in the drained site. Complete temperature records were also available at 5- and 15-m distances from June 1993.

Rates of decomposition were estimated for a 3-year period by loss of mass of three different materials (*Sphagnum* peat, cellulose and wood sticks) that were buried in 1-mm mesh polyester bags (10 x 10 cm). Three weeks after ditching in 1991, pits (30-cm depth x 30 cm x 180 cm) were dug with a chain saw near each soil moisture and temperature sampling profile. Five sample bags of each material were inserted at the 10- and 30-cm depths within each 60-cm long section. The sections provided for three subsequent bag recoveries in mid-October of 1992, 1993 and 1994.

Streamflow from both drained and control basins was measured with thin-plated 45°-V-notch weirs. Water level was monitored on a 5-min basis with Leopold-Stevens water-level recorders connected to dataloggers. Water level data (H, m) were transformed into streamflow (Q, L s^{-1}), using the relation $Q = 570.6 H^{2.5}$. Precipitation was recorded with a tipping bucket rain gauge, in a 25-m diameter forest opening in the control site. Standard meteorological data were also available from a nearby station (Saint-Clément, 47° 55' N, 69° 06' W).

Water sampling was carried out from June to October, weekly from 1990 to 1994 and biweekly in 1995 and 1996. Water was collected from streamflow and from the unsaturated and upper phreatic zones. Soil water samples were collected at 20- and 40-cm depths, at 1.5, 5, and 15 m from the center of each ditch in the 40-m spacing and at two locations in the control site. These samples were collected with pressure-vacuum soil water samplers (Soil Moisture, model 1920) into 250-ml polyethylene bottles. Samples of the same volume were also collected from the upper phreatic zone by inserting a hand pump into adjacent water wells. Before sampling, wells were purged of old water. Surface water was sampled in 500-ml polyethylene bottles at the streamflow gauging station of each basin. These bottles were rinsed twice beforehand in the water to be collected. All water samples were carried in coolers to the laboratory within four hours. Surface water samples were also collected during spring snowmelts of 1992 to 1996, and every four to twelve hours during two rainfall events that produced peak flows (July 1992 and June 1993). The surface water temperature was measured with U-shaped minimum-maximum thermometers (Weather Measure, Model E811). For a complete description of the study site, experimental design and sampling procedure, the reader is referred to Prévost *et al.*, (1997; 1999).

Chemical Analyses

Water samples were first analyzed for pH and electric conductivity, and then filtered (0.45 mm porosity) and kept frozen until the end of each field season, when they were analyzed for mineral N by colorimetry and nutrient concentrations by an inductively coupled plasma (ICP) spectrometer. Surface water samples for 1996 were also analyzed to determine total N (colorimetry) and total dissolved C (conductivimetry) after oxidation of organic substances with $K_2S_2O_8$ under U.V. radiation. Concentrations of SO_4-S of these samples were also obtained by ionic chromatography.

Data Analysis

A graphical analysis was used to compare water table profiles between spacings. The effect of spacing was determined by comparing water table fluctuations at midpoint between ditches with the control water table, before and after drainage (Braekke, 1983). Linear regression was used to establish the relation between the water table depth at the center of each spacing of the drained site (dependent variable) and at the control site (independent variable) for the calibration period. The change in water table depth after drainage was estimated by comparing the actual water table level with the predicted value from the regression. Stepwise regression was used to select a response model relating the water table depth of both sites for the post-drainage period (Freund *et al.*, 1986). Average annual peat water content and temperature were submitted to analysis of variance considering years as replications in time. Mass losses were subjected separately, for each collection year and material, to an analysis of variance as a completely randomised design with 5 pseudo-replicates. When the effect of treatment was significant (a = 0.05), the Waller-Duncan *K*-ratio *T* test was used to separate the means. Daily streamflow and surface water characteristics of the drained basin were related to those of the control one, for the calibration period and the treatment period. The effect of drainage was estimated by testing the heterogeneity of slopes and difference in intercepts of the linear regressions, using the sampling period as treatment groups in a covariance analysis on the pooled data (Freund *et al.*, 1986). In all cases, the Bartlett's test was used to verify the homogeneity of variances among sites.

Results and Discussion

Water Table Levels

Water table levels at the control site and at midpoint of all spacings were highly correlated during the calibration period (Fig. 2 and Table 1A). The relationships were linear and the regression coefficients (0.82 to 1.15) were all very significant ($p < 0.001$). As a whole, the control site explained around 90% of water table fluctuations in the site to be drained. For all spacings, the post-drainage relationships with the control site were clearly distinct from the pre-drainage ones. The relationships were also highly significant but the coefficients of determination were lower (R^2 = 0.49 to 0.70), indicating that water table fluctuations were then influenced by the ditch network in all spacings. The stepwise regression procedure first identified a significant linear effect for all spacings with regression coefficients ranging from 1.59 to 2.29, which represent slope increases of 89 to 109% compared to pre-drainage relationships. Furthermore, a third order term was found significant to adjust the post-drainage models in the lower water table range (Table 1C) yielding models with R^2 between 0.64 and 0.83. The selected model ($D = b_0 + b_1C + b_2C^3$) was found to produce a better data adjustment than any non-linear relation.

The general departure between pre- and post-drainage data was inversely proportional to ditch spacing and strongly dependent on the actual water table level (Fig. 2). During periods of near surface water table, the lowering effect of drainage decreased with spacing increase owing to the lag in evacuating excess water from midpoint to the ditches. Hence, lowering was substantial only in the 20-, 30- and 40-m spacings during periods of high water input. During periods of low rainfall and when the water table was below the 20-cm depth in the control site, a significant water table lowering occurred at midpoint of all spacings. On these occasions, the water table lowered two to three times more rapidly at the drained site than at the control

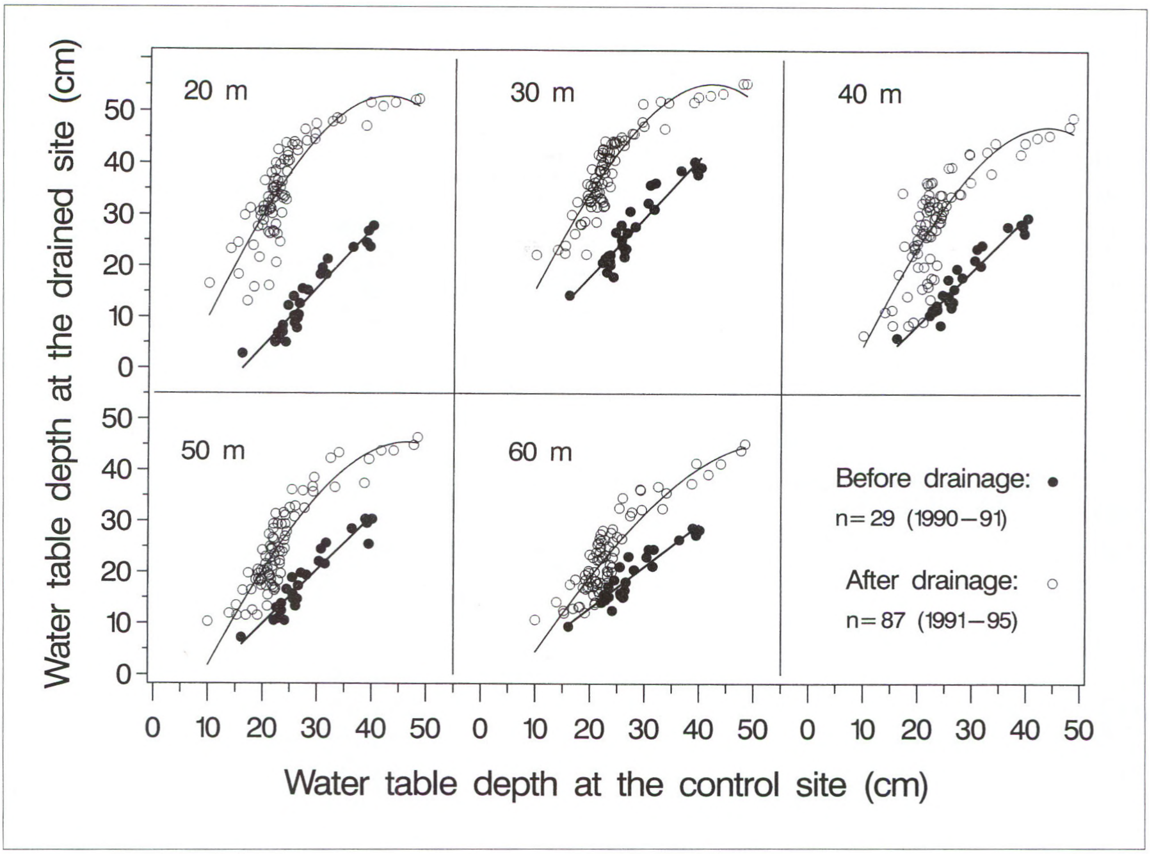


Fig. 2: Regression curves between water table level in the control site and in the drained site at midpoint of the 20-, 30-, 40-, 50- and 60-m ditch spacings, before (dot) and after drainage (circle).

The effect of drainage was greatest for the water levels ranging between 30 and 40 cm at the control site. During extended periods without rainfall, when the water table fluctuated below the 40-cm depth at the control, the drainage, at the exception of the 60-m spacing, did not have any effect as indicated by the plateau between 45 and 55-cm depth (Fig. 2). The coefficients of the third order term decreased with spacing (Table 1C, b_2 coefficient), indicating that this plateau was reached later in the 50- and principally the 60-m spacings, where ditches continued to evacuate excess water from midpoint.

The 4-year average water table lowering estimated from the regressions were 26, 17, 16, 11 and 7 cm at midpoint of the 20-, 30-, 40-, 50- and 60-m spacings, respectively (Table 1B). These results are similar to those of Belleau *et al.*, (1992) who observed a midpoint lowering of 25 to 6 cm for 20-, 40- and 60-m spacings in a black spruce wild holly bog of the Saint-Lawrence lowlands. Berry (1988) reported water table lowering, among three operational groups (Jones *et al.*, 1983) near Cochrane in Ontario, ranging from 13 to 48 cm for 25- to 35-m spacings and from 1 to 10 cm for 40- to 60-m spacings. The water table lowering in an open-canopy fen of Alberta was about 20 cm at 15 m from the ditch in a 40-m spacing, during dry summers of 1984 and 1985 receiving respectively 78 and 59% of the 63-year average rainfall (350 mm). For spacings of 30 to 60 m in a treed fen in Alberta, Hillman (1992) reported a much greater lowering, the averages across the profiles range from 56 to 79 cm. Hillman's site received an average rainfall of 235 mm from May to October, which is low compared to the 481 mm of the present study (June to October). It is possible that rainfall was in part responsible for the difference between sites. This strong response to drainage, however, is partly due to the inclusion of greater water table lowering near the ditch and to a 5-fold hydraulic conductivity (4.7×10^{-5} m s^{-1}) compared to the Québec site (9.4×10^{-6} m s^{-1}) at a water depth of 30 cm. In the present study, peat had a moderate hydraulic conductivity (Päivänen, 1973) ranging from 7.5×10^{-6} to 1.3×10^{-5} m s^{-1} in the 30-70 cm layer and initially this appeared

Table 1. Regression statistics between water table depth at midpoint between ditches in the drained site (D) and in the control site (C), before (1990-91, n=29) and after ditching (1991-95, n=87).

Ditch Spacing (m)	(A) Pre-drainage relationship: $D = b_0 + b_1C$					(B) Post-drainage average depths (cm)		
	R^2	b_0	p	b_1	p	Measured	Predicted from (A)	Δ Measured-Predicted
20	0.92	-18.73	< 0.001	1.15	< 0.001	34.8	8.4	26.4
30	0.89	-4.90	0.044	1.15	< 0.001	38.8	22.2	16.6
40	0.91	-12.58	< 0.001	1.06	< 0.001	28.5	12.4	16.1
50	0.90	-11.11	< 0.001	1.05	< 0.001	24.9	13.7	11.1
60	0.87	-3.39	0.067	0.82	< 0.001	23.2	16.0	7.2

Ditch Spacing (m)	(C) Post-drainage relationship : $D = b_0 + b_1C + b_2C^3$						
	R^2	b_0	p	b_1	p	b_2	p
20	0.72	-12.27	0.010	2.29	< 0.001	-4.2×10^{-4}	< 0.001
30	0.83	-5.6	0.082	2.17	< 0.001	-4.1×10^{-4}	< 0.001
40	0.64	-17.41	0.003	2.22	< 0.001	-3.9×10^{-4}	< 0.001
50	0.82	-18.44	< 0.001	2.06	< 0.001	-3.2×10^{-4}	< 0.001
60	0.79	-11.21	0.003	1.59	< 0.001	-1.8×10^{-4}	0.006

to be acceptable for drainage purposes. It is possible that the subsidence observed along all ditches has created zones of very low hydraulic conductivity that may have impaired the effectiveness of the drainage network. One year after treatment, the measured peat surface subsidence averaged 20 cm at 1.5-m distance from the ditches, 4 cm at 10-m and 3 cm at 20-m distances (Prévost *et al.*, 1997). No pre-drainage level measurements were made at 3- and 5-m distances, but the subsidence was estimated to be about 10 cm by linear interpolation. For a 26-month period, Hillman (1992) observed an average of 11.2 cm peat subsidence in the treed fen. Even though the latter author observed no consistent relation between subsidence and distance from the ditch, the phenomenon appeared related to ditch spacing. In the present study, no clear relation was found between subsidence and ditch spacing.

In terms of average water table lowering at midpoint between ditches, the 20-m spacing gave better results than the 30- and 40-m spacings that were roughly equivalent (Table 1B). It is not clear why the 30- and 40-m spacings gave similar results in the present study. No confounding difference between the two spacings was detected in peat hydrodynamic properties or forest composition. When considering the extreme values at the center of the spacings and the average lowering across the profile between two ditches, the 20- and 30-m spacings differed from the 40- 50- and 60-m spacings. The 30-m spacing lowered the minimum and maximum depths (by 14 and 17 cm, respectively) as much as the 20-m spacing (16 cm for both extreme values). Lowering of extreme water levels in the 40-m spacing was closer to that of the 50- and 60-m spacings (7 to 11 cm). The most important lowering occurred in the first 3 m of all spacings, in which the average depth was 55 cm at 1.5 m and 44 cm at 3 m. The 1.5-m distance wells showed, however, that the water table almost never reached the bottom level of the ditches (1 m) during this period. The average water table levels were intermediate at 5- (38 cm), 10- (35 cm) and 15-m (35 cm) distances and remained 10 cm closer to the peat surface at 20- (26 cm) 25- (25 cm) and 30-m (26 cm) distances from ditches. As a result, the 20- and 30-m spacings generally showed lower water table across the profiles than the 40-, 50-, and 60-m spacings, where the influence of ditches decreased rapidly beyond the first 10-15 m.

Of the post-drainage seasons, the 1992 period received more rainfall (3.9 mm d^{-1}) than the 1993, 1994 and 1995 periods (2.3, 3.1 and 3.3 mm d^{-1}, respectively), which resulted in higher water table lowering in these latter years. During the rainy season, water table levels were occasionally observed within 10 cm of the peat surface as close as 3 m from ditches in all spacings. The 1995 season, characterized by extended dry periods, received rainfall of only 1.8 mm d^{-1} until mid-October (Fig. 3) and showed the most important lowering at midpoint of each spacing.

Peat Water Content

The average seasonal water content at the control site followed the precipitation with its highest and lowest values obtained in 1992 and 1995, respectively (Fig. 3). Peat water content at the 20- and 30-cm depths remained always near saturation in both sites before drainage and at the control site after drainage. The near saturation water content was almost always maintained when the water table dropped to 40-cm depth indicating the importance of the capillary fringe. In the absence of drainage, the aeration (porosity minus water content) exceeded the threshold value of 10% (Paavilainen & Päivänen, 1995) only at the 10-cm depth and its value varied between locations (5, 10, 15 and 20 m) in response to differences in microtopography.

After drainage, the very low water contents (20 to 40%) observed in early June (Fig. 3) were artificially caused by frozen peat, since the TDR technique measures only the liquid water content. These values were excluded from the data for the statistical analysis. Drainage with the 40-m spacing significantly decreased peat water content at 10-cm depth at all distances from the ditch ($p < 0.001$). The relative decrease appeared slightly more pronounced at 5- and 10-m (24 and 29%, respectively) than at 15- and 20-m distances (21 and 20%). The average volumetric water contents were 32, 44, 62 and 44%, respectively at 5, 10, 15 and 20 m from the ditch, compared to 77% in the control site. These values were in line with water table profiles.

At all distances from the ditch, water content was also decreased at the 20-cm depth ($p < 0.001$), but the effect was most important at 5 m from the ditch where the 4-year post-drainage water content averaged 60%. The average water contents of 83, 86 and 81%, respectively at 10-, 15- and 20-m from the ditch, left less than 10% of the peat volume for aeration. At the 30-cm depth, the water content was significantly reduced only at 5 m from the ditch where the 4-year average was 75% compared to 88 to 90% at greater distances.

Peat Temperature and Decomposition Rate

Drainage did not change peat temperature at the 30-cm depth. At 10 cm, the treatment amplified the annual cycle of temperature. Drainage significantly increased the 4-year average of summer maximum temperature, by 3.5°C at 5-m and 1.5°C at 10-, 15- and 20-m distances from the ditch (Table 2). Increased maximum temperatures of the same order were also observed in an open fen of north central Alberta (Lieffers & Rothwell, 1987). In this study, the average summer temperature was not affected by drainage except at 5 m from the ditch where a gain of 1°C was obtained. No pre-drainage site comparison was available for autumn, winter and spring peat temperatures but the post-drainage monitoring suggested that, compared to the control, the drained site cooled earlier every autumn and remained 1 to 2°C cooler during every winter. Peat in the drained site quickly reached the temperature of the control site during the spring or warmed earlier as in 1993. In contrast, Lieffers & Rothwell (1987) reported that the drained peat warmed above 0°C 5-6 days later than the undrained substrate. In the present study, the 10-cm depth spring water content was usually lower in the drained site than in the control, where there was relative thermal stability. The annual temperature lag in the control site would be related to the thermal capacity of the wetter surface peat layer.

Three years after drainage (1991-94, Table 2), losses of mass were highest with cellulose, intermediate with wood sticks and lowest with *Sphagnum*. For each of the three collection years, a strong interaction ($p < 0.001$) between burial location (5, 10, 15 and 20 m from the ditch and control site) and depth was found for wood sticks and cellulose, making the interpretation of the results more complex. *Sphagnum* peat decomposition did not show any interaction ($p = 0.605$ to 0.991) and was clearly related to drainage. Even though *Sphagnum* decomposition of the first two years was faster at 10 cm than at 30 cm, the 3-year losses of mass did not differ with burying depth ($p = 0.572$). For both depths, loss of mass at the control site (10% at 10 cm, 9% at 30 cm) differed from the drained site only at 5 m from the ditch (15 and 18%). At 30-cm depth, decomposition tended to be higher at 10 m from the ditch (13%) than at the

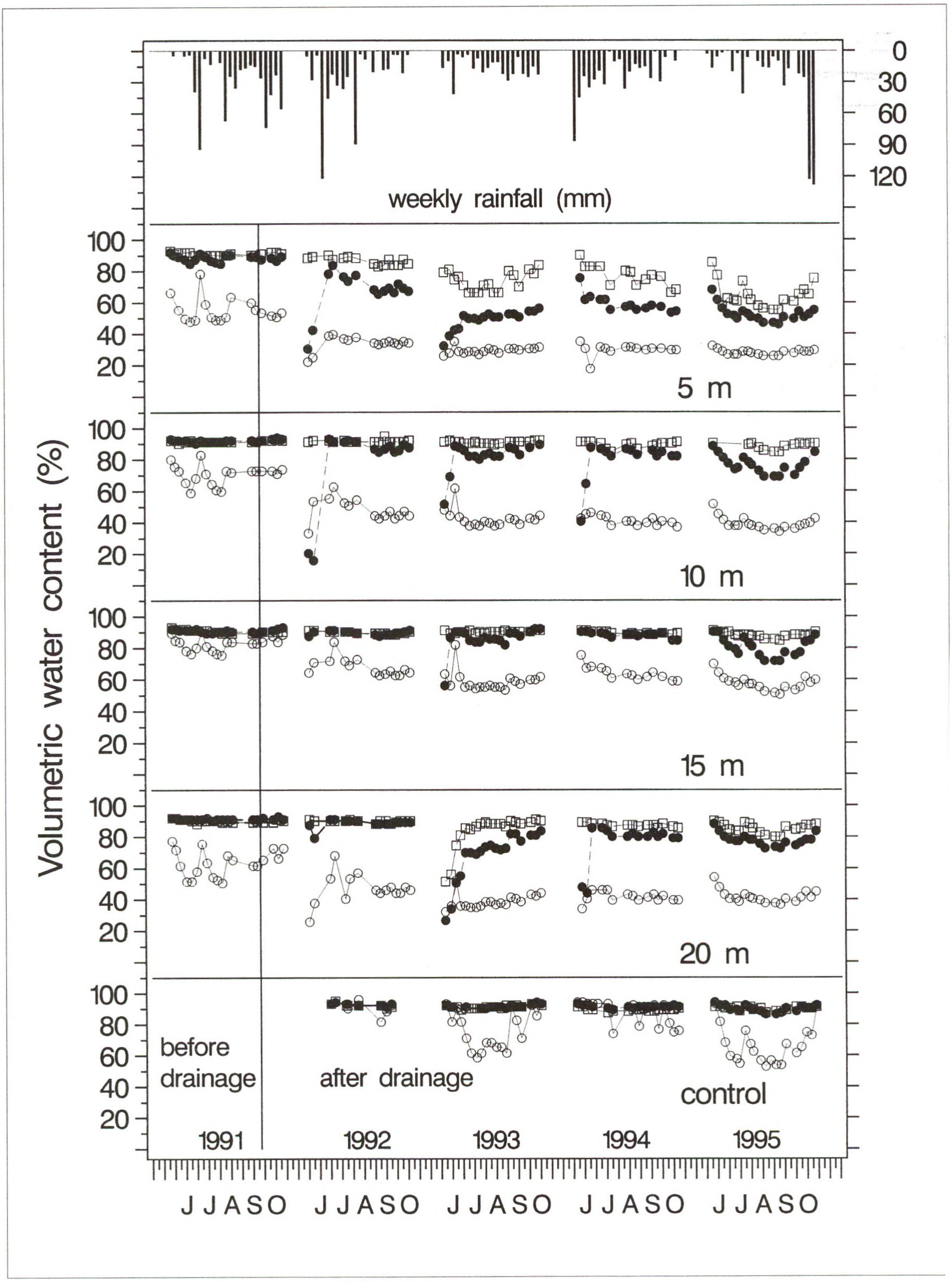


Fig. 3: Peat volumetric water content (%) at 10- (circle), 20- (dot) and 30-cm (square) depths at the control site and at 5-, 10-, 15- and 20-m distances from the ditch in the 40-m spacing of the drained site, before (1991) and after drainage (1991-95).

Table 2. Peat temperature (°C) at 10 cm under the base of the living mosses before (B, June to October 1991) and after (A, November 1991 to October 1995) drainage, and 1-year (1991-92) 2-year (1991-93) and 3-year (1991-94) post-drainage decomposition rates (% mass loss) of sphagnum peat, wood sticks and cellulose related to distance from the drainage ditch in the 40-m spacing.

		Distance from the ditch (m)				
	Period	5	10	15	20	Control
Peat temperature at 10-cm depth						
Average minimum	B	-	-	-	-	-
	A	-1.3 ab	-1.1 ab	-1.8 b	-1.9 b	-0.2 a
Average maximum	B	-	14.6	-	13.8	14.2
	A	17.1 a	15.0 b	15.4 b	14.8 b	13.6 c
Summer average	B	-	10.0	-	9.1	9.8
(June to Oct.)	A	10.7 a	9.5 b	9.2 b	9.3 b	9.5 b
Winter average	B	-	-	-	2.4	1.6
(Nov. to May)	A	-0.4 b	-0.2 b	-0.6 b	-0.4 b	0.8 a
Decomposition at 10-cm depth						
Sphagnum peat	1991-92	10.6 a	10.5 a	7.7 b	7.2 b	6.1 b
	1991-93	12.3 a	10.9 ab	9.6 ab	8.4 b	8.9 b
	1991-94	14.6 a	12.8 ab	12.3 ab	13.5 ab	10.0 b
Wood sticks	1991-92	8.8 b	14.3 a	7.4 b	8.0 b	8.7 b
	1991-93	14.8 b	23.0 a	14.4 b	12.1 b	19.4 a
	1991-94	22.5 a	24.8 a	17.5 a	16.8 a	24.6 a
Cellulose	1991-92	26.5 b	16.2 c	13.1 c	11.3 c	37.4 a
	1991-93	34.5 cd	7.6 bc	28.8 d	65.2 a	56.3 ab
	1991-94	76.0 b	49.1 c	73.3 b	93.7 a	67.1 b
Decomposition at 30-cm depth						
Sphagnum peat	1991-92	5.2 a	5.5 a	4.0 a	4.0 a	2.7 a
	1991-93	8.9 a	6.4 a	5.2 a	4.1 a	4.9 a
	1991-94	17.7 a	13.6 ab	8.5 b	8.4 b	9.4 b
Wood sticks	1991-92	13.5 a	10.4 b	7.7 c	8.0 c	7.8 c
	1991-93	23.5 a	18.1 b	14.2 b	16.2 b	7.7 c
	1991-94	29.4 a	32.0 a	29.5 a	20.4 b	6.9 c
Cellulose	1991-92	8.1 a	5.3 ab	3.2 bc	1.2 c	0.6 c
	1991-93	16.6 a	5.1 b	18.5 a	2.8 b	14.0 a
	1991-94	25.3 b	53.9 a	54.6 a	3.2 c	26.0 b

Note : For a given post-drainage period and material type, means followed by the same letter do not differ significantly (Waller-Duncan K-ratio T-test, a = 0.05).

control (9%), but remained unaffected by drainage beyond this distance. Decomposition was clearly accelerated by drainage only within a 10-m distance from the ditch. The first year losses for *Sphagnum* peat (6.1 to 10.6%) at 10-cm depth were similar to those obtained by Bartsch & Moore (1985) for natural peatland fens near Schefferville, Québec (6.4 to 10.8%) but slightly lower than the losses of around 20% reported by Lieffers (1988).

Drainage did not affect decomposition of wood sticks at 10-cm depth (Table 2). At 30-cm depth, decomposition at all distances from the ditch differed from the control site after two years. The 3-year losses of mass were finally highest at 5, 10 and 15 m from the ditch (29 to 32%), intermediate at 20 m (20%) and minimal at the control site (7%) indicating an effect of drainage and distance from the ditch.

Cellulose decomposition differed statistically between burying locations for both 10- and 30-cm depths (Table 2). Results were highly variable, however, and clear tendencies were not observed. At 10 cm, the 3-year loss of mass was highest at 20 m, intermediate at 5 m, 15 m and control site and lowest at 10 m. At 30 cm, decomposition was highest at 10 and 15 m, intermediate at 5 m and control site and lowest at 20 m. From these results, no clear relation can be established with distance from the ditch, water table fluctuations, peat moisture or temperature regimes of the burying locations. As a whole, the first year results were much lower than the 85 and 90% losses obtained by Lieffers (1988) for cellulose. They were comparable to those of 15 to 41% for peatlands in the temperate climate of the British Isles (Heal *et al.*, 1978) and higher than the 5.6 to 6.6% found in subarctic peatlands (Bartsch & Moore, 1985).

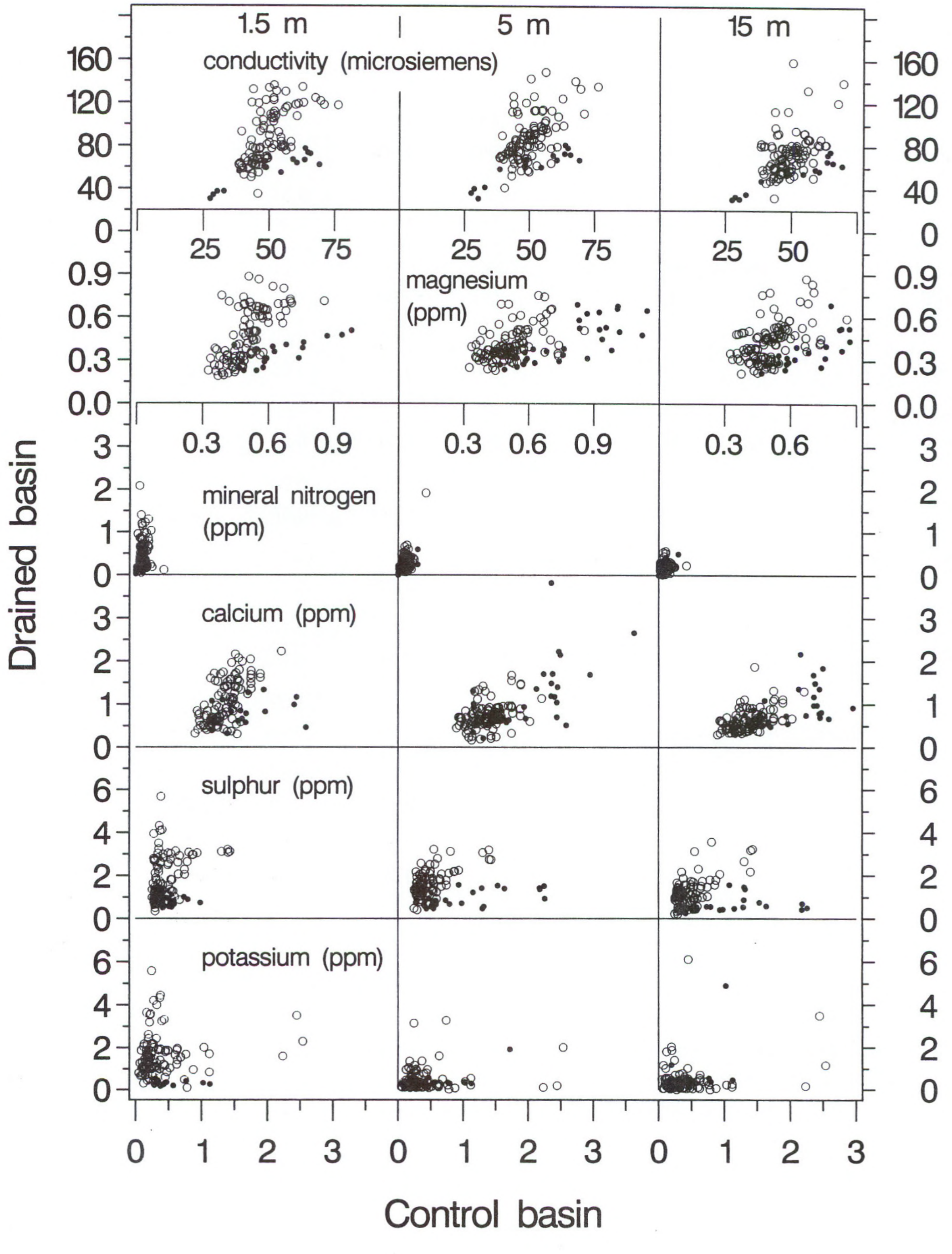
The environmental factors controlling the decomposition rates are known but their effects were confounded in the field study. As noted earlier by Lieffers (1988) for an Alberta fen, it is possible that low moisture contents at 10-cm depth had countered the temperature effect close to the ditch. Furthermore, significant water content decreases were observed in this layer at all burying locations. Hence, the lack of departure in decomposition rates at this depth could be attributed to the similarities in moisture conditions between sites. Even though burying locations were selected with care to avoid hummocks and hollows, results may also reflect the strong influence of terrain micro relief on loss of mass in surface peat (Farrish & Grigal, 1985).

Soil Water Quality

Drainage clearly increased available nutrients in peat compared to the calibration period (Fig. 4). As a whole, this was reflected by marked increases of electric conductivity of soil water at all distances from the ditches. The drainage effect was generally proportional to ditch closeness for S and Mg contents. The predominance of SO_4 and the improved aeration near the ditches suggested that enhanced aerobic conditions and sulphur oxidation were mainly responsible for the S release. Increases of N (NH_4 + NO_x), K and Ca contents were observed mainly within 5 m of the ditches. Nutrient increases, also associated with slight pH decreases, were related to depth, with concentrations being generally higher at the 20-cm than at the 40-cm depth (not shown). These results are in line with improvements of substrate conditions (warming, aeration and decomposition) that were observed principally in the top 30-cm and near the ditches where water table drawdown was greatest. Compared to the control, concentrations of ions in the soil water solution were generally higher in the drained site where aerobic mineralization likely occurred under more favourable conditions. In the Ontario Clay Belt, concentrations of ions were generally higher in the groundwater of the drained sites, 10 years after treatment (Berry *et al.*, 1995). In the present study, no effect on tree growth was yet observed five years after drainage, but no observation has been made to assess root development. Dang & Lieffers (1989) reported a 3- to 5-year null increase period for black spruce following drainage and, considering improved conditions, we can suppose that growth increase will be observed soon in the future.

Surface Water Flow and Quality

Daily streamflows of the paired basins were highly correlated for both the calibration and treatment periods (Table 3 and Fig. 5). During the 130 days of calibration, maximum and mean daily streamflows were naturally higher at the basin to be drained (23.4 and 2.1 L s^{-1}, respectively) than at the control (20.5 and 1.5 L s^{-1}). Only three values exceeded 10 L s^{-1} on the control site compared to seven values on the site to be drained. The post-drainage period was characterized by higher flows with ten mean daily values exceeding 20 L s^{-1} on the control basin (Fig. 5). However, the relationship between the paired basins did not change much after ditching. Slopes of the pre- (1.28) and post-drainage (1.37) linear relationships do not statistically differ (p = 0.247, Table 3), which indicates that streamflow fluctuations of both basins remained similarly related after drainage. Intercepts of the pre- and post-drainage regression were found to be different (p = 0.004) and the difference (0.80 – 0.11 L s^{-1}) may be interpreted as an increase of base flow around 0.7 L s^{-1}. The total daily streamflow estimated from the regressions indicate an increase of 0.93 L s^{-1} following drainage representing a 25% increase. This result confirmed field observations on the drained site, where the ditch network sustained base flow during low rainfall summers of 1993 and 1995, while the control site channel was dried during a few weeks. The increased base flow also correlates well with the water table lowering observed at the drained site. Increases of summer low flows were also observed in Sweden, (Lundin, 1988) and northern Ontario (Berry *et al.*, 1995



ig. 4: Scatter plot illustrating quality parameters of the soil water solution at 1.5, 5 and 15 m from the centre of e ditches in the 40-m spacing and at the control site before (dot) and after ditching (circle). Each point is the mean four samples (two locations × two depths, 20 and 40 cm).

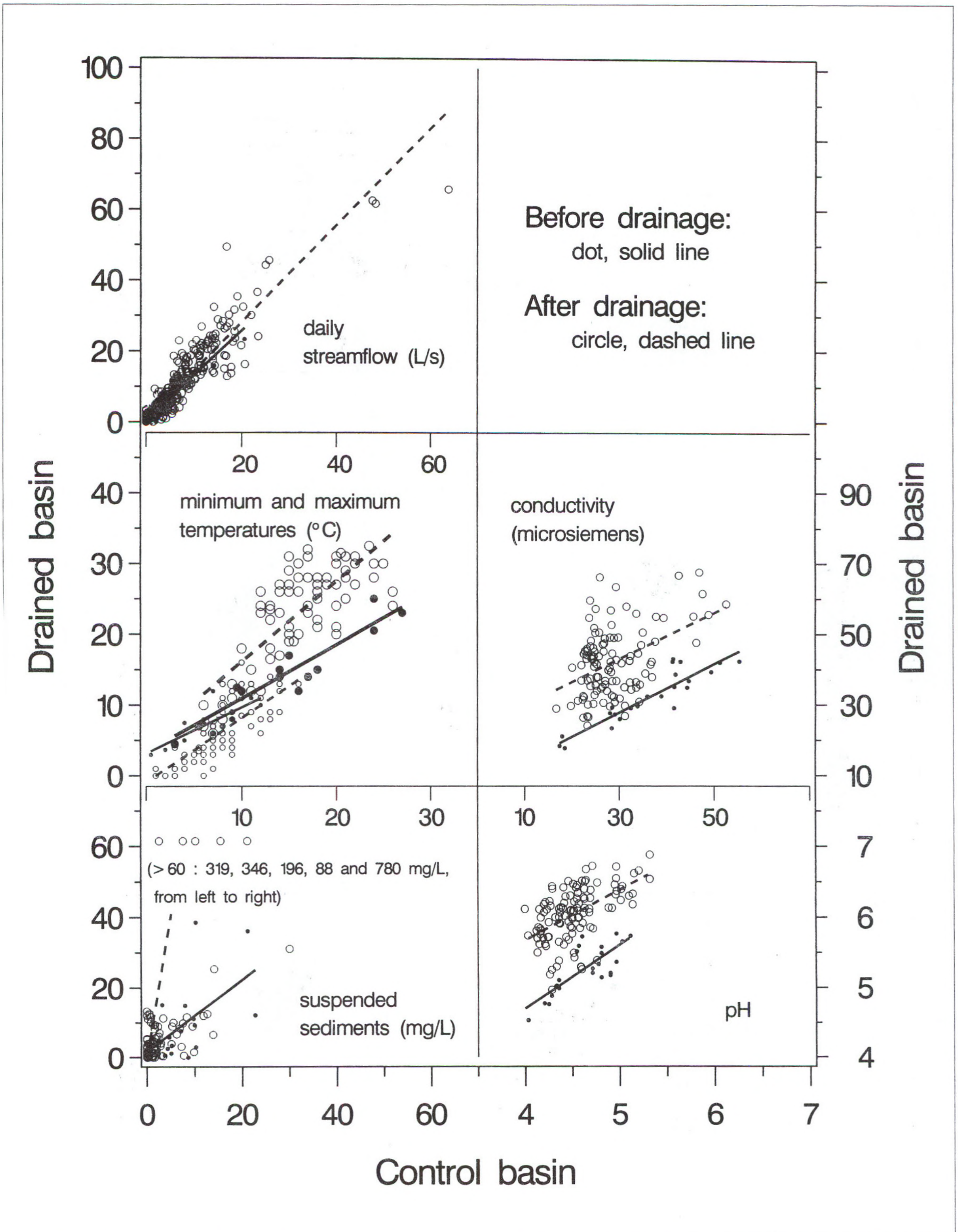
Table 3. Regression analyses of flow characteristics from the drained (D) and control (C) basins ($D = b_0 + b_1 C$), before (B) and after (A) drainage.

Parameter	Period	N	R^2	b_0	p	Difference in intercepts (p)	b_1	p	Heterogeneity of slopes (p)
Daily	B	130	0.94	0.11	0.242		1.28	<0.001	
streamflow	A	949	0.90	0.80	<0.001	0.004	1.37	<0.001	0.247
Minimal	B	24	0.67	3.2	0.004		0.65	<0.001	
temperature	A	112	0.61	-1.2	0.100	<0.001	0.93	<0.001	0.177
Maximal	B	24	0.88	3.4	0.027		0.76	<0.001	
temperature	A	112	0.62	4.8	0.007	<0.001	1.14	<0.001	0.048
Specific	B	24	0.83	7.2	0.009		0.69	<0.001	
conductivity	A	112	0.20	23.5	<0.001	<0.001	0.65	<0.001	0.856
Suspended	B	24	0.36	1.8	0.477		1.02	0.002	
sediments	A	112	0.20	1.3	0.872	0.040	8.29	<0.001	0.012
pH	B	24	0.62	1.1	0.124		0.90	0.001	
	A	112	0.27	3.0	<0.001	<0.001	0.68	<0.001	0.356
Mineral N	B	24	0.38	0.03	0.073		0.25	0.002	
	A	112	0.01	0.19	<0.001	0.007	0.18	0.201	0.781
K	B	24	0.11	0.18	0.074		0.74	0.104	
	A	112	0.07	0.34	<0.001	0.445	0.10	0.003	0.126
Ca	B	24	0.81	2.43	<0.001		1.02	<0.001	
	A	112	0.44	3.05	<0.001	<0.001	1.22	<0.001	0.272
Mg	B	24	0.82	0.256	<0.001		0.67	<0.001	
	A	112	0.46	0.269	<0.001	<0.001	1.18	<0.001	0.003
Na	B	24	0.73	0.39	0.001		0.71	<0.001	
	A	112	0.15	0.87	<0.001	<0.001	0.68	<0.001	0.943
S	B	24	0.71	0.43	0.037		0.78	<0.001	
	A	112	0.29	0.80	<0.001	<0.001	1.05	<0.001	0.149

Our data indicate that drainage did not change peak flows, but a lack of measured high flows during the calibration period prevents any clear conclusion.

Surface water was analyzed for 24 and 112 sampling dates, respectively, during the calibration and treatment periods. The pre-drainage correlation between basins was high for pH, temperature, electric conductivity and concentrations of Mg, Ca, Na and S, while suspended sediments and mineral N and K concentrations were weakly related between basins (Table 3 and Figs 5 and 6). The first major effect of drainage was an increase of suspended sediments, easily visible during ditching and the few following weeks. Pre-drainage levels were exceeded by 100 to 200 times and the concentrations (88 to 780 mg L^{-1}) wer occasionally above the 10-25 mg L^{-1} usually considere very good for aquatic life (EIFAC, 1965; Nisbett & Verneaux, 1970; Gouin, 1977). These levels have y to be defined by the Québec Ministry c Environment and Wildlife (MEFQ, 1993). Th temporary high inputs of sediments could diminis aquatic productivity if deposited in a short strea section.

The treatment increased temperature fluctuatio of surface water, through a 2°C lowering of the ave age weekly minimum and a 7°C increase of the avera weekly maximum (Fig. 5). Heterogeneity of slopes f maximum temperature confirms that warming



'ig. 5: *Relationships between streamflow, and temperature, electric conductivity, suspended sediments and pH of ırface water at the outlet of the drained and control basins before (dot, solid line) and after ditching (circle, dashed ne). For water temperature, small dots and circles represent weekly minimal temperature, and large dots and circles 'present maximal temperature.*

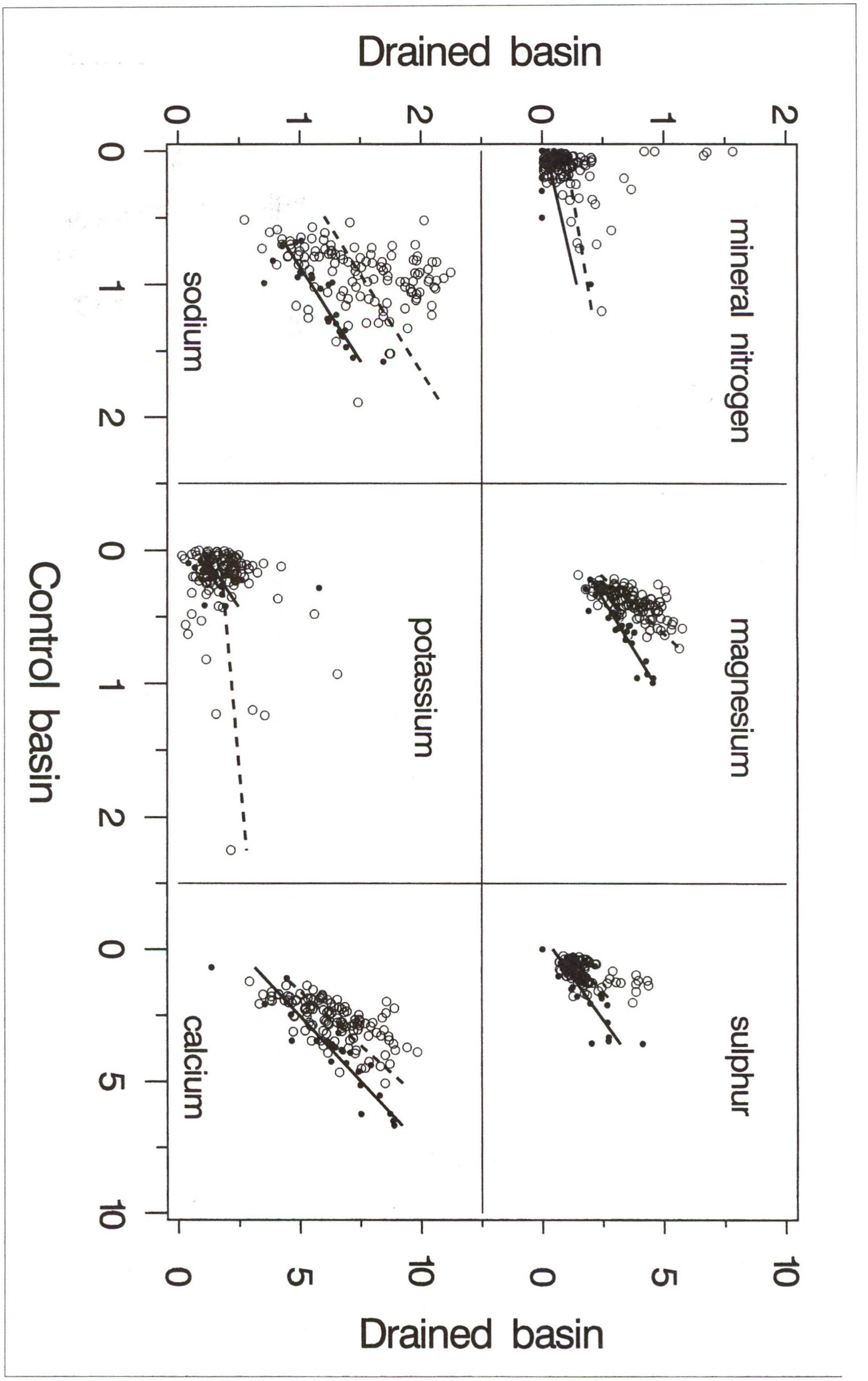


Fig. 6: Relationships between nutrient content (ppm) of surface water at the outlet of the drained and control basins before (dot, solid line) and after ditching (circle, dashed line).

surface water was more pronounced following drainage (Table 3). Furthermore, the difference in intercepts for the weekly minimum indicates that water temperature was significantly lower for the whole range of observed data covering the May-October period. Soil temperature changes, water exposure to solar radiation and night losses of long wave radiation in the open ditch system are identified as being mostly responsible for these changes. Often reaching 25 °C or more in the drained basin, water temperature exceeded the preferred temperature range for brook trout (MEFQ, 1997).

It is usually assumed that forest drainage leads to the acidification of surface water, particularly in nutrient poor peatlands with a thick peat cover (Paavilainen & Päivänen, 1995). In this study, a pH increase of about 1 unit was observed and was still noticeable after five years (Fig. 5). Prior to drainage, water from the basin to be drained had pH values naturally higher (5.1) than the control (4.3) and this difference was attributed to runoff from uplands surrounding the peatland to be drained. Hence, the most probable explanation for the pH rise is that the main ditch increased the volume of runoff routed directly from the upland (as observed during rainfall) to the outlet of the drained site. Similarly, Lundin (1988) noted that the changes in chemical composition of runoff (increases of HCO_3, K, Ca, Cl, Al and C) were affected by the lagg-ditches penetrating the mineral soil beneath the peat. Paavilainen & Päivänen (1995) attributed pH increases of several studies to ditch interception of more neutral groundwater after drainage.

The pH increase of the present study did not act much on surface water electric conductivity, because nutrient leaching from the drained site strongly counterbalanced the decrease in H^+ concentrations. As a whole, drainage increased electric conductivity by 54% through clear increases in mineral N (320%), Ca (27%), Mg (44%), Na (50%) and S (40%), while concentrations of K were not affected (Figs. 5 and 6). Slopes of pre- and post-drainage relationships do not differ (Table 3), except for Mg which appeared to be less affected by drainage during periods of low concentrations. All intercepts were significantly different between periods, which indicates that nutrient increases occurred over the entire concentration range. Sallantaus (1992) found that losses of Ca, Mg and K in water discharging from drained mires occurred soon after treatment. In the present study, Mg and Ca were leached more easily than K, which was presumably retained more effectively by the vegetation, as was also observed by Laiho & Laine (1992). Leaching of N occurred mainly during ditching and the two following years, but other parameters were still above their pre-drainage levels five years after treatment. Finally, no effect related to drainage was found for Zn, Fe and Al contents.

CONCLUSIONS

In this drainage experiment, water table lowering was greatest in the first 5-m distance from the ditches and very small beyond 15 m from the ditches. Moisture content was significantly reduced at the 10-cm depth but the effects on peat temperature and rate of decomposition were limited. Nutrient availability was mainly enhanced within 5 m of the ditches and at the 20-cm depth. The improvement of substrate conditions was less pronounced than expected, particularly for the 40- to 60-m spacings. Considering the first 5-year water table lowering, the 20- and 30-m spacings would be recommended, although it may not be economical. However, visual observations at old drained sites have indicated that drainage efficiency may increase with time, with enhanced vegetation growth starting near the ditches and progressing inward. Hence, the 40-, 50- and 60-m spacings may significantly improve growth over the long-term. Given the shallow root system of black spruce in peatlands, it is foreseen that improved substrate conditions will initially promote root development, principally near the ditches. Response to the new conditions will have to be monitored by analysing root development and tree growth at various distances from the ditches. Also, drainage significantly increased base flow and leaching of nutrients. Five years after treatment, most water quality parameters had not returned to their pre-drainage levels and monitoring of ion leaching should be continued.

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FOREST RESOURCES OF FINNISH PEATLANDS IN 1951-1994

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SUMMARY

This paper examines the changes of the main inventoried characteristics of peatland forests in Finland from the beginning of the 1950s to the inventory results of the 8th National Forest Inventory and its updating, during the years 1989-94. Finland's land area is 30.46 million hectares, of which 86% is classified as forestry land. Since the 1950s, extensive forest improvement programs have been practised in Finland. The drainage of peatlands, suitable for wood production, has been a part of these programs. Of the present peatland area (8.9 million hectares) 53% has been drained. The share at the beginning of the 1950s was 9% of the peatland area of 9.7 million hectares. The volume of growing stock of trees in the area classified as peatland at the beginning of the 1950s has increased from 252 10^6 m^3 to 449 10^6 m^3 and the annual increment of the growing stock from 9.9 10^6 m^3 to 20.3 10^6 m^3, i.e. by 10.4 10^6 m^3.

The annual increment of the growing stock volume of the Finnish forests since the beginning of the 1950s has increased from 55.2 10^6 m^3 to 75.4 10^6 m^3. Peatland forestry has been responsible for over one half of the increase of the annual increment and it has had a significant affect on the increase of the volume of the growing stock and the subsequent future cutting possibilities.

Keywords: forest inventory, peatland drainage, volume of growing stock, growth increment.

INTRODUCTION

National Forest Inventories (NFI) have measured the Finnish forests eight times. The first inventory was carried out between 1921 and 1924 and the eighth one between 1986 and 1994. In 1994, the oldest part of the data was updated by re-measuring 38% of the field plots in southern Finland. The ninth inventory began in 1996. In the current inventory, about 150 characteristics are being measured or assessed in the field. These characteristics include information concerning the soil, site fertility, the structure and amount of the growing stock, tree growth, damage, accomplished and required sylvicultural as well as cutting measures and indicators of the biological diversity of the forests.

The land area of Finland is 30.46 million hectares of which 26.28 million hectares are forestry land. This includes 20.03 million hectares of forestland (i.e. forests with a mean annual productivity of at least 1 $m^3ha^{-1}a^{-1}$ over the rotation), 2.99 million hectares of other wooded land (with a productivity of at least 0.1 but less than 1 m^3 $ha^{-1}a^{-1}$), and 3.13 million hectares of wasteland. A site is defined as peatland (or mire) if the organic layer is peat (minimum thickness is not required) or if at least 75% of the ground vegetation consists of mire plant species. The area of sites classified in the inventory as peatland is 8.93 million hectares, which is 33.9% of the forestry land area.

An extensive and long-term forest amelioration program was begun in Finland in the 1950s. Peatland drainage was a part of this program. The purpose of drainage activities is to lower the water level so as to ensure sufficient aeration and to make the sites better for wood production (Paavilainen & Päivänen, 1995). Lowered water level increases rooting depth and the availability of nutrients. The growing stock and its annual increment on these sites has increased considerably, but the nature of the peatland has also changed over wide areas owing to these operations (Vasander, 1996).

From the contemporary wood production point of view, large-scale thinning cuttings, as well as complementary ditching or cleaning of ditches, are needed in peatland forests. The role of peatland forestry concerning future cutting possibilities depends largely on how these operations will be implemented.

This paper presents the estimates of basic forest resource characteristics for Finnish peatland forests and their changes from the beginning of the 1950s.

MATERIALS AND METHODS

Inventory Data

This paper is based on the results of the 3rd (1951-53), 5th (1964-70), 6th (1971-76), 7th (1977-84) and 8th (1986-94) National Forest Inventories and on the updating of the 8th inventory in southern Finland (1989-94 and called 8+th inventory in this paper) (Table 1) (Ilvessalo, 1956; Kuusela & Salminen, 1991; Tomppo & Henttonen, 1996). Linewise survey sampling was utilised in the 3rd inventory, the total length of survey lines was 25,000 km with the total number of field plots amounting to 30,809. Detached L-shaped tracts were employed instead of continuous lines in the 5th inventory (Kuusela & Salminen, 1969) and systematic clusterwise sampling has been employed since the 6th inventory (Tomppo *et al.*, 1997).

The parameters and method of the 8th inventory are described briefly in this paper. A more comprehensive description is presented by Tomppo *et al.* (1997). The variables measured in the field can be grouped into stand-level characteristics, describing the stands intersecting a field plot, and tree measurements on the field plots.

Stand level data can be classified into the following groups: 1) general data of the field plot cluster, 2) plot identification data, 3) site data (e.g. land class, main site fertility type, mixture of site fertility types, specification of mire type, type of soil, quality and thickness of organic layer, accomplished and proposed drainage), 4) crown layer information (e.g. species of layers, development class, stand establishment, dominant tree species, species mixture, number of stems, quality, mean diameter/height, age, damage syndrome, time of origin, cause and seriousness of damage), and 5) stand level information (e.g. damage, lichen survey, stand quality, accomplished sylvicultural and cutting measures and time, proposed measures and time, basal area).

Peatland sites can be classified in many different ways. The three basic classification criteria considered in this paper, and applied in Finnish forestry are: (1) main site class (*spruce mires, pine mires, treeless bogs and fens*), (2) drainage stage (*undrained peatland, newly ditched peatland, transforming peatland and transformed peatland*) and (3) site fertility class (6 classes). The understorey vegetation of newly (or recently) drained sites still closely resembles that of undrained mire. Transforming drained mires are sites where the succession of the understorey vegetation has gone further and the tree stand has increased its growth. Transformed drained mires represent the final stage of succession. Understorey vegetation has developed into a relatively stable form and resembles more the vegetation of mineral soils. Tree growth corresponds to that of mineral soils (Paavilainen & Päivänen 1995). Note that the classification of a site into these classes is independent of the classification into categories of forestland, other wooded land and waste land (see Introduction).

Tree level data are measured at two levels of intensity, at the tally tree level and the sample tree level. The variables, (1) co-ordinates (only on permanent plots), (2) tree species, (3) diameter, (4) timber assortment class and its precision, and (5) crown layer, are measured for tally trees. The additional variables measured for sample trees (each nth tally tree, where n is 7 - 8, depending on the area) are: (1) upper diameter (at the height of 6 m, on every 9th cluster), (2) bark thickness, (3) height increment, (4) diameter increment, at the height of 1.3 m, (5) age at 1.3 m and age - age at 1.3 m, (6) damage syndrome, the cause and seriousness, as well as the time of origin of the damage, (7) defoliation, and (8) the lengths, timber quality classes and reasons for possible deterioration of each part of the stem.

The number of measured field plots on forestry land in the 8th inventory was 70,500, of which 24,000 were on peatlands. Altogether, about 0.5 million trees were measured.

Estimation Methods

Area estimates are products of the known total land areas and the ratios of the numbers of field plot centres of the strata of interest. The volume estimates are derived as follows:

1) Volume estimates of sample trees (m^3 ha^{-1}) were derived utilising sample tree measurements and taper curve models.
2) Sample tree volumes were generalised for tally trees using a nearest neighbour method.
3) Mean volumes of the stratum of interest were computed using single tree volumes of the stratum and the number of field plot centres of the stratum. The total volume estimator is the product of the mean volume estimator and the area estimator (Kuusela & Salminen, 1969; Tomppo *et al.*, 1997).

The increment of the growing stock is measured as the mean of increments of the inventory year and four years preceding the inventory, (as the mean of

Table 1: Development of peatland drainage, 1951-1994.

THE ENTIRE COUNTRY

Inventory	Period	Undrained	% of peatland area Newly ditched peatland	Trans-forming peatland	Transformed peatland	Total peatland area (1000 ha)
NFI3	(1951-53)	90.6	2.8	5.2	1.4	9742
NFI5	(1964-70)	68.2	18.8	9.2	3.8	9779
NFI6	(1971-76)	56.4	9.5	18.3	5.8	9337
NFI7	(1977-84)	50.4	13.5	28.7	7.4	9019
NFI8	(1986-94)	47.6	9.7	32.5	10.2	8924
NFI8+	(1989-94)	47.3	11.8	30.4	10.5	8927

SOUTH FINLAND

Inventory	Period	Undrained	% of peatland area Newly ditched peatland	Trans-forming peatland	Transformed peatland	Total peatland area (1000 ha)
VMI3	(1951-53)	82.3	4.4	9.9	3.4	3846
VMI5	(1964-68)	52.6	23.0	16.8	7.6	3958
VMI6	(1971-74)	36.1	25.1	28.0	10.8	3630
VMI7	(1977-82)	29.3	15.5	40.0	15.2	3448
VMI8	(1986-92)	25.1	12.4	42.3	20.2	3437
VMI8+	(1989-94)	24.3	11.3	43.1	21.3	3429

NORTH FINLAND

Inventory	Period	Undrained	% of peatland area Newly ditched peatland	Trans-forming peatland	Transformed peatland	Total peatland area (1000 ha)
VMI3	(1951-53)	96.1	1.6	2.2	0.1	5896
VMI5	(1969-70)	78.8	16.0	4.1	1.1	5821
VMI6	(1975-76)	69.3	16.0	12.1	2.6	5707
VMI7	(1982-84)	63.5	12.3	21.6	2.5	5571
VMI8	(1992-94)	61.7	12.0	22.5	3.8	5498

increments of five years preceding the inventory, where measurement is before August 1st). The volume increment estimate of a single sample tree is the difference of the present volumes and the volume five years ago. Diameter increment (based on boring), height increment, bark thickness and a model of its development, as well as taper curve models, ar used in the estimation. The increment of the drai (cutting removals + cutting waste + natural mortality during the period of increment estimation (5 years is taken into account. The details are given in Kujal (1980) and Tomppo *et al.*, (1997).

Some changes have been made in the field measurements and estimation methods since the 3rd inventory. The effects of these changes on the estimates have been taken into account and all figures presented in this paper are comparable, except that the increment of trees with breast height diameter less than 2.5 cm has been measured since the 7th inventory (1977-84). Those trees accounted for 2% of the total increment of the growing stock in the 8th inventory (Tomppo & Henttonen 1996). The standard errors of the estimates are estimated using local quadratic forms, as suggested by Matérn (1960).

RESULTS

Areas

The results in this paper are given for the whole country and, in some cases, separately for South and North Finland, the border between these regions lying around the latitude of 64 degrees. (Finland is located between the latitudes of 60 and 70 degrees N.) The forestry land areas of South and North parts in 1989-94 were 12.54 million hectares and 13.73 million hectares, respectively. The forest land areas were 11.50 million hectares and 8.53 million hectares (for definitions, see Introduction).

The peatland area of the whole country in the beginning of 1950s was 9.74 million hectares. The clearance of forestry land for arable land and the construction of artificial lakes have decreased the peatland area by about 150,000 ha since the beginning of 1950s (Paavilainen & Tiihonen, 1988). About 660,000 ha of earlier mires with shallow peat layer, or without peat layer, are currently classified as mineral soils. Note the definition of peatland in NFI (see Introduction). The present peatland area is 8.93 million hectares (Table 1). The area of spruce mires is 2.28 million hectares, pine mires 4.93 million hectares and treeless bogs and fens 1.71 million hectares.

The peatland drainage operations were carried out in Finland mainly between the mid 1950s and the early 1980s. The proportion of drained peatlands was 9% (917,000 ha) in 1951-53 and 53% (4.70 million hectares) in 1989-94. (Note that the total peatland area had decreased during that time.) The drainage operations were a part of extensive forestry programs which aimed to lower the water level of mires and so increase the increment and the volume of the growing stock and, together with intensified sylvicultural measures, improve present and future cutting possibilities. Drained peatland will develop towards a stage called transformed peatland where tree growth has recovered and the ground vegetation resembles that of mineral soil. The development towards the transformed peatland stage has been rapid in South Finland during the 1960s and 1970s and in North Finland (where changes are slower) during the 1970s and 1980s (Table 1).

In 1989-94, the area of newly drained peatlands (which do not yet exhibit changes in the growth condition or ground vegetation) was 1.05 million hectares, transforming peatland 2.71 million hectares and transformed peatland 0.94 million hectares (Table 1). The area of drained peatlands which on the basis of site fertility remain too poor for wood production was 450,000 ha, i.e. 10% of the drained area.

The majority of drained peatlands were originally classified as forest land with a minor part also classified as other wooded land or even as waste land (treeless bogs). The drainage was considered acceptable only if it was likely that the site would develop into forestland. Of all the peatlands, 4.81 million hectares are currently classified as forestland, 2.01 million hectares as other wooded land and 2.11 million hectares as waste land. Of all the drained peatlands, 84% are classified as forestland and 14% as other wooded land.

In the 7th inventory, the area of forest and other wooded land on peatlands was 6.90 million hectares (South Finland 3.14 million hectares and North Finland 3.76 million hectares). The corresponding figure in the inventory 8+ (1989-94) was 6.82 million hectares (South Finland 3.14 and North Finland 3.68 million hectares).

The peatland sites are divided into site fertility classes on the basis of indicator species. Of all the undrained mires (4.2 million hectares), 17% is the herb-rich type or more fertile type, 32% the *Vaccinium myrtillus* or tall-sedge type, 24% the *Vaccinium vitis-idaea* or small-sedge type, 22% the cotton grass and dwarf-shrub type and 5% the *Sphagnum fuscum* type. The corresponding figures for drained peatlands are 16%, 25%, 34%, 23% and 2%.

Volumes

The volume of the total growing stock of the Finnish forests in the latest inventory (1989-94) was 1 937 10^6 m^3. This is 26% more than the 1 538 10^6 m^3 in the beginning of 1950s (Table 4). The relative standard errors were 0.57% and 0.36%, respectively. The volume of growing stock of peatland forests is 394

10^6 m^3. The tree species proportions are pine (*Pinus sylvestris* L.) 46%, spruce (*Picea abies* (L.) Karsten) 27%, birch (*Betula pendula* Roth and *Betula pubescens* Ehrh.) 25% and other tree species (mainly *Populus tremula* L. and *Alnus* spp.) 2% (Table 2). The volume of growing stock on drained peatlands is 307 10^6 m^3.

Because only temporary field plots were utilised in the Finnish national forest inventories until the year 1992, the changes on the earlier peatland areas can only be estimated indirectly. The increase of the volume of growing stock on the area classified as peatland in the beginning of 1950s can be estimated utilising current information concerning drained mineral soils because earlier peatlands which are now classified as mineral soils are very likely drained mineral soils. It is assumed that the growing stock of the forests that has changed from peatland to mineral soil is, on average, similar to that of all drained mineral soils. The total area of drained mineral soil is 1.07 million hectares, of which 0.7 million hectares was earlier classified as peatland. The estimated growing stock volume on this area is 55 10^6 m^3 and 449 10^6 m^3 on the sites previously classified as peatland. The volume in 1951-53 was 252 10^6 m^3. The volume has therefore nearly doubled (Table 4). The increase of the growing stock was most rapid in the late 1970s and in the 1980s particularly in South Finland.

Increment

The annual increment of the whole growing stock of the Finnish forests in 1989-94 was 75.4 10^6 m^3 (Table 4). The relative standard error was 0.80%. The annual increment of the current peatland forests is 17.7 10^6 m^3. The tree species distribution is: pine 45%, spruce 23%, birch 29% and other tree species 3% (Table 3). The total annual increment of all forests at the beginning of the 1950s was 55.2 10^6 m^3 and the increment of peatland forests 9.9 10^6 m^3. The estimated volume increment of the growing stock in the area which was classified as peatland in 1951-53 and as mineral soil in 1989-94 (0.7 million hectares) is 2.6 10^6 m^3. Thus the annual increment of the growing stock on the land classified in the 3rd inventory as peatland is 20.3 10^6 m^3 (Table 4). Drainage and other peatland forestry measures have increased the annual total volume increment of the growing stock by 10.4 10^6 m^3. This is slightly more than the 9.8 10^6 m^3 increase of increment of the Finnish mineral soil forests.

Table 2: The mean and total volumes of the growing stock of forest and other wooded land on peatlands according to the 7^{th} and $8+^{th}$ inventories, by tree species groups.

	The 7^{th} inventory (1977-84)						The $8+^{th}$ inventory (1989-94)					
	South Finland 1977-82		North Finland 1982-84		Total 1977-84		South Finland 1989-94		North Finland 1992-94		Total 1989-94	
	m^3 ha^{-1}	1000 m^3	m^3 ha^{-1}	1000 m^3	m^3 ha^{-1}	1000 m^3	m^3 ha^{-1}	1000 m^3	m^3 ha^{-1}	1000 m^3	m^3 ha^{-1}	1000 m^3
Pine	25.3	79500	12.5	47900	18.4	127400	36.6	114600	18.6	68400	26.8	183000
Spruce	19.6	61700	5.2	19600	11.8	81300	26.1	81700	6.7	24800	15.6	106500
Non-coniferous	16.3	51200	7.8	29500	11.7	80700	22.3	69700	9.5	35000	15.4	104700
Total	61.2	192400	25.5	97000	41.9	289400	85.0	266000	34.8	128200	57.8	394200

Table 5. The mean and total annual increments of the volume of growing stock of forest and other wooded land on peatlands according to the 7^{th} and $8+^{th}$ inventories, by tree species groups.

	The 7^{th} inventory (1977-84)						The $8+^{th}$ inventory (1989-94)					
	South Finland 1977-82		North Finland 1982-84		Total 1977-84		South Finland 1989-94		North Finland 1992-94		Total 1989-94	
	$m^3 ha^{-1}$	1000 m^3	$m^3 ha^{-1}$	1000 m^3	$m^3 ha^{-1}$	1000 m^3	$m^3 ha^{-1}$	1000 m^3	$m^3 ha^{-1}$	1000 m^3	$m^3 ha^{-1}$	1000 m^3
Pine	1.3	4130	0.5	2020	0.9	6150	1.7	5360	0.7	2690	1.2	8050
Spruce	1.0	3010	0.2	590	0.5	3600	1.1	3410	0.2	710	0.6	4120
Non-coniferous	1.1	3510	0.4	1600	0.7	5110	1.2	3690	0.5	1870	0.9	5560
Total	3.4	10650	1.1	4210	2.1	14860	4.0	12460	1.4	5270	2.6	17730

Carbon Storage of Peats

The depth of the peat layer was measured to a depth of 4 metres in the beginning of the 1950s and again in the ongoing 9[th] inventory (1996-). Peatlands play an important role in the carbon balance in the boreal region. The estimated carbon storage of Finnish peatlands is 4.8 million Gg. This is seven times greater than the 0.68 million Gg in the growing stock of all the Finnish forests (Kauppi *et al.*, 1997).

Accomplished and Required Measures

From the wood production point of view, extensive thinning cuttings need to be applied to peatland forests, totally to an area of 2.3 million hectares, during the next 10-year-period. Non-commercial thinning cuttings are needed in area of 0.63 million hectares and first thinning cuttings in an area of 0.77 million hectares, mainly because of the large area of young forests.

The drainage of pristine peatlands has practically been stopped at present. However, 780,000 ha of undrained peatlands (4.2 million hectares) would be suitable for wood production, at least after a drainage operation. The criterion is that the amount of available nutrient in the peat is usually high enough to support an increase in the tree growth with an economically justified amount (Päivänen & Paavilainen 1996). Out of the total drained area of 4.7 million hectares, complementary ditching or cleaning of ditches is needed in an area of 1.46 million hectares.

Conclusions

The inventory results show that peatland forestry has resulted in a considerable increase in the volume of the growing stock in Finland and especially in the annual volume increment of the growing stock. At the same time, the nature of peatland forests has changed over wide areas. Some of the sites classified as peatlands in the beginning of 1950s are now classified as mineral soils. The contribution of peatland forestry to the future cutting potential largely depends on how the necessary thinning cuttings and maintenance of ditches will be implemented.

Acknowledgements

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Table 4: The volume of growing stock and annual volume increment on mineral soils and on peatlands in 1951-53 and in 1989-94. The figures are also given for these areas which were classified as peatland in 1951-53 but are classified as mineral soil in 1989-94.

	Volume (mill. m^3)				Increment (mill. m^3a^{-1})			
	Mineral soils	Peatland	Peatland 1951-53	Total	Mineral soils	Peatland	Peatland 1951-53	Total
1951-53	1286	252	252	1538	45.3	9.9	9.9	55.2
1989-94	1543	394	449	1937	57.7	17.7	20.3	75.4
Increase (%)	19.3	56.4	78.2	25.9	27.4	78.8	105.1	36.6

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SPATIOTEMPORAL SIMULATION OF WATER AND NITROGEN DYNAMICS AS A TOOL IN FEN RESTORATION

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SUMMARY

Nitrogen transformation processes are calculated with the model system WASMOD for different landuse types on minerotrophic peat soils from Northwest Germany with a high spatiotemporal resolution. Nitrogen budgets are simulated for three sites. Abandoned Magnocaricion has a stable nitrogen budget. The nitrogen balance is negative in the two agricultural sites with harvest and denitrification as the main output sources. Leaching rates from minerotrophic peat soils increase with drainage depth and landuse intensification. The model is also applied on a subcatchment scale in the drained Pohnsdorfer Stauung peatland, a terrestrialisation fen in Northwest Germany. Mineralization rates within the peatland show high spatial variation related to water table depths. The results are used to predict the effect of future management activities on the water and nutrient cycle and are therefore valuable in environmental landscape planning and fen restoration.

Keywords: minerotrophic peatland, modelling, ecosystem functioning, restoration.

INTRODUCTION

Peat forming fens and bogs are amongst the most threatened ecosystems in Germany. Peatlands in which peat is no longer accumulating, however, cover 10,810 km² (about 9.2%) of the land surface in Northern Germany, mostly in Schleswig-Holstein, Niedersachsen, Mecklenburg-Vorpommern and Brandenburg (Grosse-Brauckmann, 1997). Recent inventories determined the quality of these peatlands. The majority are under agricultural use, mainly intensive dairy farming, with high fertiliser input. In Schleswig-Holstein only 17.3% of the minerotrophic peat soils are regarded as 'ecologically valuable' and in only half of these are water levels near to the surface (Trepel & Schrautzer, 1998). In Mecklenburg-Vorpommern only 2.8% of the original mire area has remained undrained (Lenschow, 1997).

In the past, peatland conservation focused mainly on the maintenance of species and habitats as the last remnants of a formerly widely distributed landscape type. Management activities focused on restoring the hydrology of drained bogs or establishing low disturbance grazing and mowing regimes in order to maintain low-productivity, mesotrophic species (Pfadenhauer & Klötzli, 1996). In contrast, intensive agricultural use and drainage of minerotrophic peatlands causes severe environmental problems including subsidence of the surface owing to shrinkage and oxidation of peat, nutrient leaching and a loss of species adapted to wet and nutrient poor habitats (Succow, 1988). At present, the benefits from hydrologically unaffected wetlands such as water flow regulation, improvement of water quality and habitat structure have become more valuable for society. Quantification of the hydrological, biogeochemical, ecological and social functioning of ecosystems is a prerequisite to the development of integrated peatland management with the aim of stabilising ecosystem services.

Quantifying ecosystem functioning requires a good understanding of the complex processes and interactions of all landscape units and is clearly scale dependent. Models can help to describe and quantify the structure and functioning of ecosystems on various scales.

In this study the effect of landuse on the nitrogen cycle in minerotrophic peat soils is calculated using the simulation model WASMOD (Reiche, 1994; 1996). The application of this model allows quantification of biogeochemical processes in the water and nutrient cycle. These results can be used by decision makers to establish a future management regime with low environmental impact.

MATERIAL AND METHODS

Modelling

Models are a widely used tool in ecology to describe the complex interactions of biological systems (Jørgensen, 1994). Modelling the water and nutrient dynamics of peatland ecosystems is a scale dependent problem. On a microscale, nitrogen transformations such as mineralization or denitrification are a function of soil water content and soil properties; they interact quickly with weather conditions. On a mesoscale, the system reacts slower and interaction between microsites is controlled by other properties including slope and hydrological conductivity. The simulation system WASMOD describes water fluxes and nutrient transformations of ecosystems and the interaction between them in a hierarchically structured way with a high spatiotemporal resolution (Reiche, 1994, 1996; Schimming *et al.*, 1995; Göbel, 1997). WASMOD consists of several submodels, which simulate soil water and groundwater dynamics, surface runoff, soil heat budget and organic carbon and nitrogen transformation processes for single sites as well as for entire catchments (Fig. 1) (Reiche, 1994). Water, nitrogen and carbon budgets are given as a result. The primary spatial units are determined with the help of a GIS system based on the heterogeneity of soil properties, elevation, vegetation and landuse.

Corresponding to the objectives mentioned above the model includes processes that belong to different spatial and temporal scales. Watersheds and subwatersheds represent in this context the superior system units. The atmosphere represents the upper system boundary, characterised by daily measured meteorological data of air temperature, air humidity and precipitation. The vegetation is considered by calculating daily rates of biomass increase, interception by the plant canopy, transpiration and nitrogen uptake. Organic matter turnover of plant residues is also taken into account. The soil mineral nitrogen processes in the model include nitrification, denitrification, uptake by plant roots, and vertical movement in the soil profile. Nitrification is simulated by applying first order kinetics assuming the rate coefficient to be influenced by soil temperature and soil water content. Denitrification is simulated by defining a potential denitrification rate assumed to be related to the carbon dioxide evolution rate in the soil and soil temperature (Hansen *et al.*, 1991). Plant canopy and rooting depth are simulated according to their annual development. Soil texture, mineral content, bulk density and organic matter con-

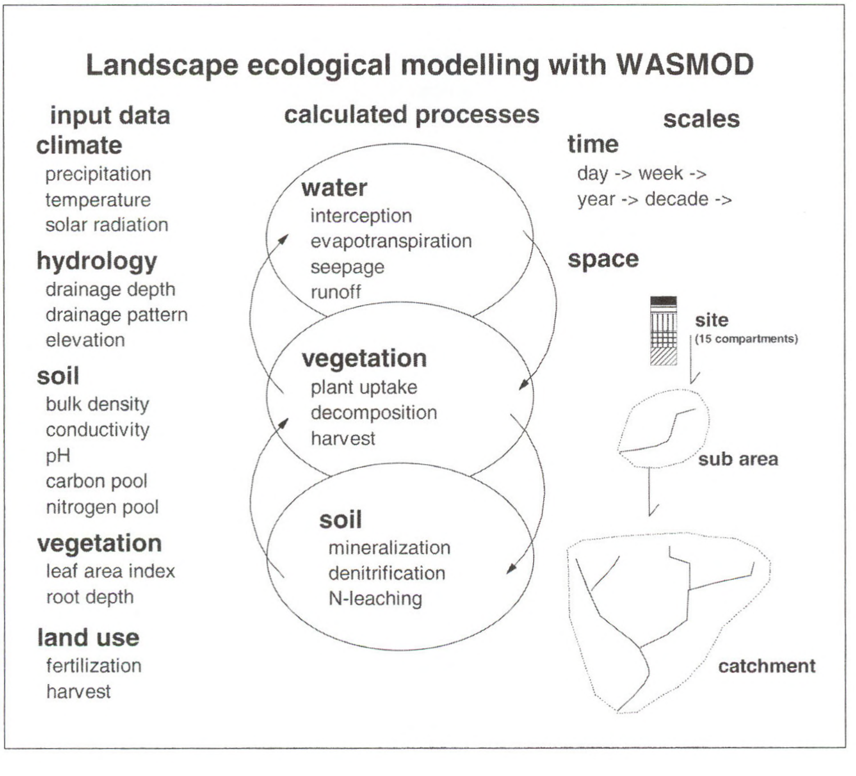

Fig. 1: Important input parameters and simulated processes in the WASMO1 model.

tent are highly variable in the upper soil layer. Consequently soil water content, soil temperature and the concentration of nitrogen show the highest temporal gradients at the surface.

Data Acquisition and Study Site

The model was applied in a small terrestrialisation fen, the 'Pohnsdorfer Stauung' (54°15' North - 10°12' East), 10 km southeast of Kiel, Germany. The fen is situated in the catchment of the Neuwührener Au-brook near its outlet to Lake Postsee in the Weichselian moraine landscape. The fen developed between two ice advances in a dead ice depression. After melting of the ice in the Late Weichselian, a lake formed in the area. This lake was filled by meltwater sands and varved clay in the Late Weichselian, by gyttja in the Early Holocene (locally more than 7 m) and, from the Atlantic onwards, with a layer of peat 1 to 4 m thick. The peat consists of strongly humified *Alnus* peat, overlain by a thin layer of *Phragmites peat* (Weerts, 1997). At the surface, a layer of moderately humified sedge peat (about one metre) and, locally, a slightly humified *Sphagnum* peat layer indicate a successional shift to mesotrophic conditions. A small part of the fen was drained by ditches and used as a wet meadow before 1953. In 1953, a pumping station was built and the entire area cultivated for agricultural use. In 1992, a private nature conservation foundation bought the area and started a programme of restoration.

The Pohnsdorfer Stauung peatland (~100 ha) contains a great variety of vegetation types, including intensively used pastures, species rich wet meadows, *Phragmites* and *Carex* reedbeds and *Alnus* carr. The vegetation types and landuse forms in the Pohnsdorfer Stauung are typical for peatland ecosystems in the Weichselian moraine landscape (Table 1).

The actual hydrological condition in the peatland is controlled by the pumping station. Immediately after the installation of the pumping station in 1953, subsidence shrinkage of about 1 to 1.8 m was measured. Until 1992 subsidence amounted to 2.4 m. Subsidence in drained peat areas is a continuous process caused by lowering the water level to create optimal conditions for agriculture. At present, the annual mean water level in about 30% of the minerotrophic peat soil is less than 0.3 m below surface.

The geology, soil, vegetation, hydrology and landuse of the 'Pohnsdorfer Stauung' were studied in the field and data stored in a database of a GIS system. To determine the elevation, a digital elevation model with a 12.5 m resolution was used. Water levels were monitored fortnightly in groundwater wells. For calibration of the model, soil properties were determined in the field and compared with literature values from Northern Germany. Climate data (daily measurements for precipitation, maximum-, minimum-temperature and calculated solar radiation) were measured at the nearest weather station in Ruhwinkel 15 km southeast of the Pohnsdorfer Stauung.

The water and nutrient dynamics were first simulated for three sites with minerotrophic peat soils differing in soil properties and landuse (Table 1) and secondly for the total subcatchment area of the peatland 'Pohnsdorfer Stauung'. The catchment area was classified into hierarchically ordered hydrological subunits based on the drainage pattern, hydrological conductivity and elevation. For the spatial analysis, the catchment area (413 ha) was divided into 632 polygons with a mean area of 0.65 ha (0.87 std), that are homogenous with respect to soil properties, vegetation, elevation, hydrology and landuse. Actual landuse in the catchment consists of 40% forest (mainly *Fagus sylvatica*), 15% reedbed, 20% pasture or meadow and 15% farmland.

Site A is an abandoned wet meadow with high water levels. The vegetation consists of tall *Carex* species and the grass *Phalaris arundinacea*. During re-

Table 1: Characterisation and selected input parameters for the simulation.

Site	A	B	C
Vegetation type	Magnocaricion	Calthion	Lolio-Potentillion
Landuse	abandoned	mowing in late July	mowing 3 times a year
Fertilisation [kg N ha^{-1} yr^{-1}]	0	0	160
Drainage depth [m below surface]	0.1	0.2	0.5
Bulk density 0 - 10 cm [g cm^{-3}]	0.2	0.3	0.5
pH [KCL]	4.6	5.5	5.5

cent years the area of abandoned meadow in Germany has increased for economic reasons. Before abandonment, agricultural use was of low intensity because of high water levels and low productivity. Abandonment initiates a replacement of low-productive, short living species by high-productive, long living species (Jensen, 1997).

The Calthion stand B is cut only once a year in late July. The vegetation structure and species composition represent the influence of high water levels (drainage depth at 0.2 metres) and a low disturbance regime. In the vegetation many Calthion and Molinietalia species, for example, *Caltha palustris*, *Lychnis flos-cuculi*, *Cirsium palustre* and *Geum rivale* occur. Calthion stands have become rare as a result of agricultural intensification in Northwest Germany (Schrautzer & Wiebe, 1993). In some peatlands special management efforts are initiated to maintain vegetation structure and biodiversity (Pfadenhauer & Klötzli, 1996).

Site C is cut three times a year in the middle of May, July and in late August. It is fertilised with 160 kg N annually. This intensive agricultural use is one of the most frequent types of land use on minerotrophic peat soils. The vegetation is dominated by the high productive grasses *Alopecurus pratensis*, *Lolium perenne* and *Holcus lanatus*. Molinietalia species are absent because they cannot regenerate under a high disturbance regime (Schrautzer & Trepel, 1997). This landuse type is of minor value for species diversity and conservation.

The simulation was run twice with a 30 year long climate data set from the regional weather station Hamburg-Fuhlsbüttel. During this time, nutrient and moisture content establish an equilibrium depending on interactions between carbon and nitrogen pools and the vegetation (Hansen *et al.*, 1991; Jensen *et al.*, 1996). Subsequently, the water and nutrient dynamics were simulated using meteorological data for the period October 1988 to September 1997. Only these simulation results were evaluated.

RESULTS

The simulated nitrogen balances for the three sites show the principal pattern of nitrogen dynamics in minerotrophic peat soils (Fig. 2). The intensity of the nitrogen transformation is strongly influenced by moisture content and pH. All sites receive a nitrogen input of 20 kg N through atmospheric deposition. Site C is fertilised additionally with 160 kg N $ha^{-1}yr^{-1}$. Nitrogen mineralization is the most important internal source at the unfertilised sites. At site A with a drainage depth of 0.1 m, mean nitrogen mineralization amounts to 59 kg N ha^{-1} yr^{-1}. Nitrogen mineralization at site C is nearly four times higher owing to a slightly higher pH and deeper drainage level. The higher nitrogen input in site C results in a more productive vegetation. The meadow is cut three times a year with a mean yield of 210 kg N. In the budget denitrification, harvest and leaching are considered as outputs. The simulation results indicate that denitrification of 19 kg N ha^{-1} yr^{-1} is the most important output process at the unused site A. More than 90% of the atmospheric nitrogen input leaves the system through denitrification. In the simulation denitrification rates increase with drainage depth, caused by higher mineralization and nitrification rates in better aerated soils. The calculated mean nitrogen loss by denitrification at site C amounts to 84 kg N ha^{-1} yr^{-1}.

Harvest is the most important output at the agricultural sites B and C. Only the abandoned wet meadow shows a stable nitrogen budget. In wet years a nitrogen accumulation is possible. The nitrogen budgets at the two agricultural sites B and C show negative values of 75 and 135 kg N ha^{-1} yr^{1}, respectively. These sites act as nitrogen sources in the landscape with harvest as the main output source. Nitrogen leaching of 7 kg in the Calthion stand B is low compared to the 21 kg N ha^{-1} yr^{-1} of the fertilised site C.

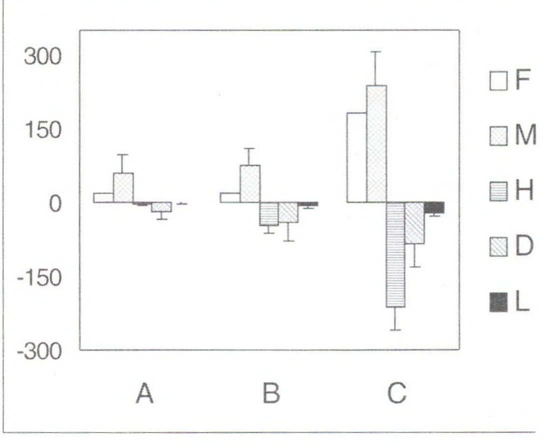

Fig. 2: Simulated Nitrogen balance for three sites on minerotrophic peat soils for the period 10. 1988 - 9.1997. Site description see Table 1; F = fertilisation, M = mineralization; H = harvest; D = denitrification, L = leaching; all values in [kg N $ha^{-1}yr^{-1}$].

The annual simulation results for the different nitrogen transformation processes show a high variation. This is caused by the large climatic variation in the meteorological data used for the simulations. The data embrace very warm and dry phases as well as extremely wet periods. Moisture and temperature are the forcing functions in the nitrogen cycle. As an example of the climatic influence on water balance and nitrogen dynamic, the weekly simulated downward seepage and leaching rates from site C are shown for a period of 5 years (Fig. 3).

Normally downward seepage is highest in winter and low in summer owing to higher evapotranspiration during the latter. This annual pattern cannot be seen clearly in the simulation results, because the real climate conditions used in the simulation differ from the long term conditions. In autumn 1992 seepage rates increased slowly owing to dry summer conditions. During the wet summers of 1993 and 1994 seepage rates decreased in the beginning of June. From October 1995 to January 1997 precipitation was extremely low, causing seepage rates to decrease to zero and they increased only slightly from January 1997 onwards. Nitrogen leaching is strongly connected with seepage rates. In the simulation, nitrogen leaching rates differ only slightly within the year but with peaks that are correlated strongly with fertiliser treatment. Fertiliser application leads to an immediate increase in nitrogen leaching depending on the seepage rates. In the dry summer of 1996 nitrogen leaching was not calculated because of the water deficit.

Annual average nitrogen mineralization in the total peatland is 115.7 ± 71.3 kg N ha^{-1} yr^{-1} with a high spatial variation (Fig. 4). The spatial pattern of the computed nitrogen mineralization depends mainly on the hydrological conditions within the peatland. In the lower parts with constantly high water levels, mean nitrogen mineralization is below 100 kg ha^{-1} yr^{-1}. At the peatland margins, however, mineralization rates are up to 280 kg N ha^{-1} yr^{-1} higher than the average.

The simulation results indicate clearly that recent hydrological conditions are not suitable for development of a peat accumulating fen. Decomposition of organic matter is high owing to continuous drainage. Nitrogen accumulation occurs only in small spots in the peatland, where the water level is constantly near the surface.

Discussion

In this study the ecological functioning of peatland ecosystems has been quantified with a high spatiotemporal resolution. The simulation results agree well with measurements both in their temporal dynamic as well as in their range. Nitrogen mineralization in minerotrophic peat soils increases with drainage depth (Koerselman & Verhoeven, 1992). In reed stands with a mean groundwater level between 0 - 10 cm below

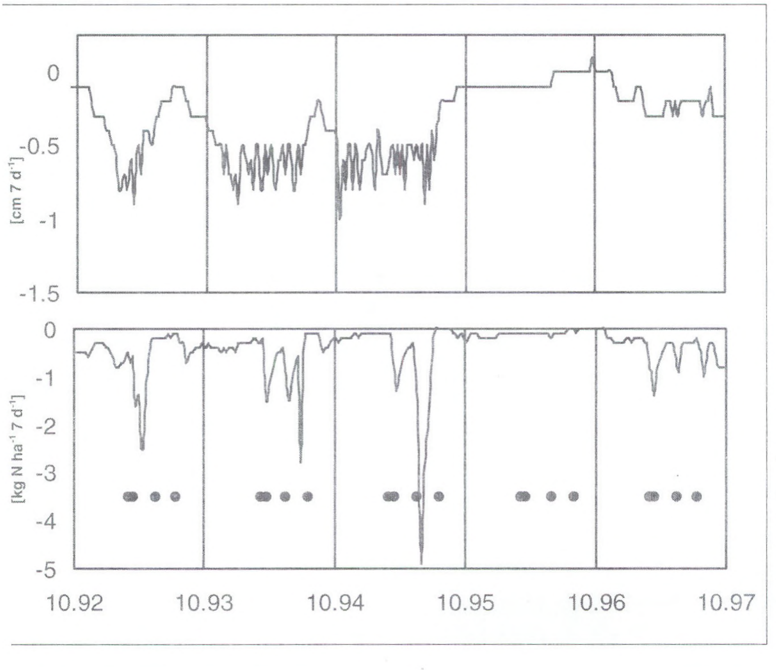

Fig. 3: Simulated downward seepage [cm 7 d^{-1}] and nitrogen leaching [kg N ha^{-1} 7d^{-1}] from an intensively used meadow on minerotrophic peat soil over a period of 5 years. ● indicates fertiliser application of 40 kg ha^{-1}.

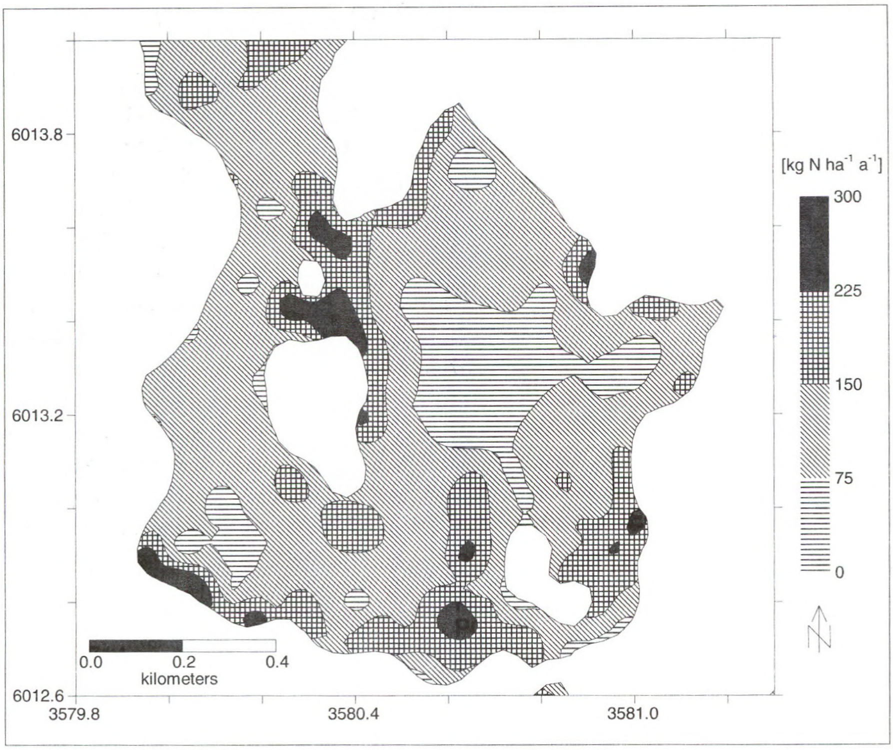

Fig. 4: Simulated mean nitrogen mineralization [kg N ha^{-1}yr^{-1}] in the Pohnsdorfer Stauung peatland over a period of 9 years. White area = mineral soil.

surface, nitrogen mineralization amounts to 100 kg N ha^{-1} yr^{-1}. In wet Calthion stands, nitrogen mineralization ranges between 50 - 150 kg N ha^{-1} yr^{-1}, increasing up to 300 kg N ha^{-1} yr^{-1} in deeply drained and intensively used meadows and pastures (Trepel & Schrautzer, 1998; Rosenthal *et al.*, 1998).

Ross *et al.* (1995) also report an increase in nitrogen leaching with fertiliser application and drainage depth in dry lysimeter experiments. They measured under undrained and unfertilised conditions an inorganic nitrogen loss similar to the simulation results below 1 kg. With fertiliser application nitrogen losses from the system increased slightly up to 14 kg N ha^{-1} yr^{-1}.

Denitrification increases also with drainage depth in peat soils owing to higher nitrification rates in better aerated soils (Augustin *et al.*, 1996; Lång *et al.*, 1995). The simulated denitrification rates for the drained site C are high compared to other studies, but Davidsson & Leonardson (1997) report for a drained meadow an annual N_2O+N_2 production of higher than 108 kg N ha^{-1}. Measuring total denitrification is connected with many methodological problems. The climatic influence can only be studied in long-term monitoring programmes, which do not exist at the moment. Knowledge about denitrification processes in long-term studies is limited. In this context, models can be applied as a tool to review recent knowledge with the aim of finding gaps ir present understanding. In this way, simulations rur by validated models can be used to define futur research activities clearly.

The simulation results can be used in landscap planning and fen restoration to predict the effect o a future landuse on the nutrient cycle and to devel op guidelines for future management. Undraine fens act as nitrogen sinks in the landscape. They re duce the nitrogen inflow via ground-, surface- o rainwater and atmospheric deposition by accumula tion and denitrification. Annual plant production i greater in undrained mires than decay. The remair ing plant material is accumulated as peat. The mea annual long-term nitrogen accumulation in fen pe is probably lower than 1.5 kg per hectare based o a carbon accumulation rate of 0.25 t ha^{-1} yr^{-1} (Mal by & Immirzi, 1993) and a C/N ratio of 15 (ow measurements). In some peatlands, denitrificatio rates may be much higher, especially if there is a sig nificant nitrogen import via surface- or groundw ter inflow (Blicher-Mathiesen *et al.*, 1998; Haycoc *et al.*, 1993).

Owing to the fact that undrained fens are nea ly extinct in North Germany, they cannot fulfill the function as nutrient sinks in the landscape cycle. I general, the farmer prefers well drained soils to pr duce a high yield. Normally, the ecological services peat soils are not taken into account by the farmer nor honoured by society. The simulation results can help to develop guidelines for a wise use on peatsoils with low environmental impact. Nitrogen leaching, for example, can be reduced if nitrogen fertiliser application is stopped. Nitrogen fertilisation is not necessary on minerotrophic peat soils with a high natural nitrogen content. The added fertiliser stimulates nitrogen mineralization through the so called 'priming-effect' described by Jenkinson *et al.* (in Ross *et al.*, 1995). The simulation results indicate that any agricultural use of drained peat soils is combined with a more or less significant peat loss owing to oxidation depending on water level depths. Any drained peatland therefore acts as a nitrogen source in the landscape with harvest and gaseous losses as main output pathways. With the help of models, the effect of higher water levels can be simulated over long time periods with different climate. The results are useful to define different management zones within the peatland. The parts with groundwater levels above or near the surface are suitable for restoration of a fen with a new peat forming vegetation. These areas can be surrounded by meadows or pastures to be used only if hydrological conditions are suitable. In some areas it is not possible to raise the water level above the peat surface; these areas should be used for agriculture in a peat conserving way.

Conclusions

The quantification of processes of peatland ecosystems is a prerequisite for the development of integrated peatland management options. In this study, the application of the model WASMOD allows a quantification of the biochemical processes and nutrient losses from minerotrophic peat soils under different types of land use. The results can be used to define peat conserving management practices with low environmental impact. The application of the model in combination with a GIS visualises the intensity of biochemical processes (eg. mineralization) with a high spatial resolution and enables the definition of different management zones.

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CALIBRATION OF THEMATIC MAPPER IMAGERY FOR DENSITY MAPPING OF PRODUCTION BOGS IN IRELAND

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SUMMARY

Peat is a valuable natural resource. It is exploited commercially in many countries as a horticultural product. It is also exploited, in a smaller number of countries, as a fuel for energy production where it must compete with alternative energy sources. Ongoing research is required in all aspects of the peat production industry for peat to remain a competitive product. An element of the research currently being undertaken to improve the efficiency of large scale peat production involves the creation of reliable peat density variation maps for production bogs. For this purpose a combination of modern technologies including satellite positioning, satellite imaging and digital image processing was investigated. A Landsat Thematic Mapper (TM) satellite image was successfully calibrated for variations in peat density on production bogs using two approaches that reflected different perspectives on the development of the bog. The first approach involved discriminant analysis using the maximum likelihood estimator algorithm while the second approach used multiple regression analysis. Three distinct density classes were identifiable. Overall classification accuracies in the region of 80% to 90% were achieved, depending on the measure of accuracy adopted, with no significant difference in accuracy between the two approaches. A "leave-one-out" cross-validation approach was employed to obtain a reliable accuracy assessment from the ground truth data available. The efficacy and efficiency of the Global Positioning System (GPS) for the purposes of geo-location on production bogs were demonstrated.

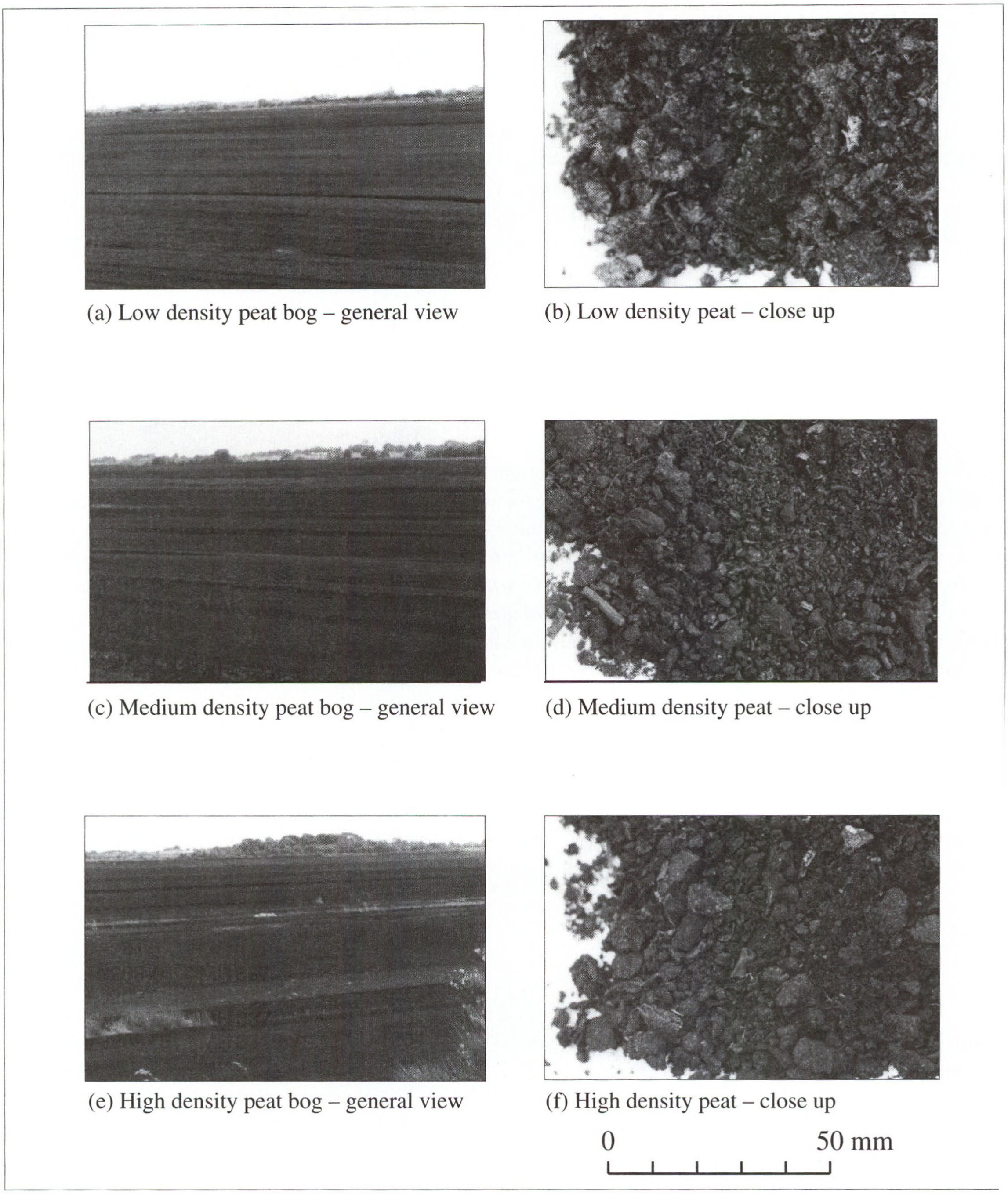
Keywords: satellite imagery; peat density classification.

INTRODUCTION

Peat is defined as a soil type containing >75% organic matter (Andrejko *et al.*, 1983) and it forms in areas where suitable internal and external conditions exist. Peatlands (bogs) constitute 17.2% of the land cover of the Republic of Ireland. Of these peatlands, 13.8% are worked by Bord na Móna (The Irish Peat Board) for the production of horticultural and energy products (Leinonen *et al.*, 1997). Currently 15% of the electricity produced in Ireland is derived from peat. It has been estimated that the reserves of Irish peatlands designated for industrial production are worth 3,640 billion dollars as an energy source (assuming an oil price of $18 per barrel) and that they will last another 30 years (O'Connor, 1994). As the supplier of the only major indigenous fuel used for electricity production, the Irish peat industry is of strategic national importance. The industry is also of importance because it provides employment in areas that have traditionally suffered economic disadvantage. Nevertheless, improved efficiencies in all operational aspects of the industry are required for the peat industry to survive and develop in an era of increasing competitive pressures.

The peat industry in Ireland generally classifies peat into three categories based on density, *i.e.* low, medium and high density. Examples of the categories are shown in Fig. 1.

Knowledge of peat density variation is required for management of the resource in areas such as the monitoring of reserves and the development of harvesting strategies. Peat density is also required as input to scientific studies, *e.g.,* carbon sequestration model development, and to developments in production methods such as the Precision Peat Produc-



(a) Low density peat bog – general view

(b) Low density peat – close up

(c) Medium density peat bog – general view

(d) Medium density peat – close up

(e) High density peat bog – general view

(f) High density peat – close up

Fig. 1: Examples of the three peat density categories used by the Irish peat industry. Note variations in colour and texture between categories.

tion (PPP) system proposed by Ward and Holden (1998). Peat destined for energy production in Ireland is milled into a friable crumb which is then left to dry on the bog surface for a set period of time (usually four days) before being collected into linear stockpiles (Fig. 2). This is known as the PECO production system.

The milled peat is stored in the stockpiles unti sold and it is then removed to the power station. Th price paid for the product is reduced if specified fig ures for mean final water content and final wate content variability are not achieved. Ward and Hold en (1998) have proposed a method of improvin control over the final water content of milled pea

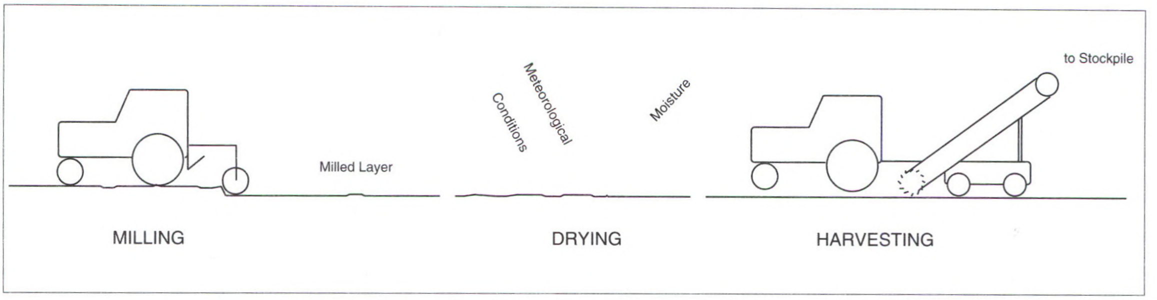


Fig. 2: The stages in milled peat production.

This involves automatic real-time control of the miller with peat density among the required inputs. Significant improvements in financial return to the producer have been predicted if such a system could be successfully implemented.

Traditional methods of determining peat density variations are based on ground sampling. These are considered both time consuming and labour intensive and so an alternative approach using satellite imagery and satellite positioning was considered. Satellite imagery has been used to study natural peatlands (e.g., Pala, 1984; Cruickshank and Tomlinson, 1990; Allison *et al.*, 1995). However, with the exception of some preliminary studies (Clark, 1984; Collins and Mac Siúrtáin, 1987; ERA, 1987), little work has been carried-out on the application of satellite imagery to production bogs. Accurate geo-location on any large areas which lack specific identifiable features has been a problem in the past but modern satellite positioning systems offer a solution. The objective of this research was to assess the use of satellite imagery, used in conjunction with digital image processing and satellite positioning, for producing milled peat density variation maps of production bogs.

METHODS AND MATERIALS

Imagery Selection and Spatial Referencing

The TM is a satellite-based sensor that records the relative strength of energy reflected from the surface of the earth on six electromagnetic (EM) bands with a spatial resolution of 30 m 30 m (this represents one pixel). Energy readings are in the range 0-255 and are referred to as digital number (DN) values. The reflective bands represent the visible part of the EM spectrum (bands 1, 2 and 3) and the infrared part of the EM spectrum (bands 4,5 and 7). Band 6 records thermal infrared energy transmitted from the surface of the earth with the same DN range but with a reduced spatial resolution of 120 m 120 m.

Landsat Thematic Mapper imagery was selected because it has good spectral and radiometric resolution and previous studies had demonstrated its potential in this area of research (ERA, 1987). The spatial resolution was considered adequate because speed of miller depth adjustment combined with forward motion and the small number of density classes envisaged meant that a fine spatial resolution was not required. A 50 km 50 km subscene of TM scene [Path207, Row23] taken on 10th April 1997, and centred on Longitude 8 00' W, Latitude 53 15' N, was acquired. This scene was 95% cloud free and included the entire complex of bogs comprising the Boora Works in Co. Offaly, Ireland (Fig. 3).

A date prior to the start of the milled peat production season was chosen to avoid degradation of the image by the production process. Met Éireann (The Irish Weather Service) confirmed that the weather was dry and warm for the days prior to, and including, that of the satellite overpass. The imagery was rectified to the Irish National Grid (ING) by reference to the Ordnance Survey Ireland (OSI) digital vector map of the area. This digital map series is derived from the OSI 1/10,560 scale paper map series and is thus of suitable planimetric accuracy for this application. Twenty well-defined points, common to both the satellite image and the OSI map, were identified and their common coordinates were used to compute first-order transformation parameters between the satellite image file reference system and ING. The resulting transformation had a Root Mean Squared (RMS) error of 21 m, or 0.7 of a pixel, and was used to rectify the TM image to the ING coordinate system. The Nearest Neighbour resampling algorithm was used to retain the integrity of the spectral responses.

Fig. 3: General location of Boora Bogs complex, Co. Offaly, Ireland, with ground truth sample locations inset.

Ground Truth Acquisition and Image Enhancement

The density of milled peat is the variable for calibration, and therefore milled peat rather than raw bog was used to relate the spectral responses at the satellite to density. Milled peat is stored on the bog surface in linear stockpiles spaced up to 150 m apart and up to 2 km in length. A sample of milled peat taken from a stockpile can be considered a mix of the milled peat from the surrounding area, up to a radius of *ca.* 75 m around the sample location. On this basis, the dry weight density of milled peat was determined at 71 stockpile locations at the Boora Works complex of bogs to provide ground truth data for image classification (Fig. 3). These locations were evenly distributed, based on local knowledge, through the three density classes used by the industry.

At each location, the stockpile was sampled 12 times on a 2 m 4 m grid. For each sample the dry weight density of the milled peat was determined. The average of the 12 samples gave the dry weight density at that location. The field work was undertaken in November 1997 and April 1998 and, accordingly, there was a full production season (one summer) between the date of the imagery and the acquisition of the ground truth data. Given the slow rate at which production bogs are cut (max. 0.15 m per season), the milled peat sampled from the stockpiles was considered to be drawn from the top layer of the bog at the time of image acquisition and a direct relationship between the two was hypothesised.

The ING coordinates of the sample locations were required so that they could be overlaid on the rectified satellite image. A planimetric accuracy of 5 m was considered appropriate given the resolution of TM imagery. A pair of Trimble 4000SSi dual-frequency carrier-phase GPS receivers was used to coordinate the sample locations. One of the receivers was located at a secure base station whose position had been previously connected to the ING. The second receiver was used to observe GPS baselines between the individual sample locations and the base station. The GPS baselines were computed by post-processing and the ING coordinates of the sample locations were determined. Centimetre-level relative accuracies though not necessary for the purposes of this study were achieved.

The relationship between peat density and the satellite image was to be established by relating the peat density at each sampling location to the image pixel values at the corresponding point on the satellite image. However, before this procedure was carried out smoothing of the satellite image was considered necessary, for the following reasons: (1) Dikshit and Roy (1996) have estimated that the Nearest Neighbour resampling algorithm can introduce positional errors in the resampled image of close to one pixel. When these are combined with errors due to the rectification process, total positional errors of up to 1.75 pixels can be introduced in the satellite image. (2) A satellite image of a production bog can contain noise from a number of sources including system hardware limitations, atmospheric propagation uncertainties and non-homogeneity of the bog surface due to milled peat debris, stockpiles, drains and various access tracks. (3) Samples taken at a particular location are presumed to be a mix of milled peat from an area of radius 75 m around the location and so the DN values for the location should be representative of this area. A 5 5 mean value averaging filter was found to remove most of the high frequency variations in spectral response on the image and was accordingly applied.

Image Classification and Accuracy Assessment

Two approaches to image classification and testing were used. The first approach presumed that the milled peat could be separated into the three distinct categories used by the industry and related to the genetic origin of the bog. For this approach a classifier that assigned pixels to predefined discrete classes, a "crisp" classifier, was appropriate. Of the many such classification algorithms employed in remote sensing, the maximum likelihood classifier is considered the most advanced (Estes *et al.,* 1983; Luman and Minhe Ji, 1995). The effectiveness of the maximum likelihood estimator depends on reasonably accurate determination of the mean vector and the covariance matrix for each class drawn from normally distributed unimodal data. This, in turn, depends on having a sufficient number of training pixels in each class. Campbell (1996) recommends at least 100 training pixels per class while Joyce (1978), writing in the context of Landsat Multi-Spectral Scanner (MSS) imagery, recommends at least 90 pixels per class. The effort involved in gathering ground truth information for this application of satellite remote sensing was considerably greater than that required for the more usual land-cover-assessment type applications where the class of pixels comprising whole fields on the ground can be determined by visual inspection. The number of sample locations in this case was 71, with the reported value for each location being the mean of 12 individual determinations of density. Training statistics were extracted from the image by centring a 60 m 60 m box on the satellite image over the GPS-located sample location and recording the DN values in all seven TM bands of all pixels inside the box. Given that the image was already smoothed, this procedure was designed to satisfy the requirement of determining DN values representative of the area surrounding the stockpile from which the milled peat was sourced and to provide a sufficient number of pixels from which to generate reliable statistics for the classification. A further class, referred to as "Not in Production", was added to the classification. This represented areas within the original bog but not in production such as uncut and cutaway bogs, access roads and areas where change of use has occurred. These areas had spectral signatures significantly different to any of the production bog classes so their inclusion did not influence the classification of the production bogs. Non-bog areas were masked off and a supervised classification was carried out using the maximum likelihood estimation algorithm. All image processing was carried-out using Imagine v. 8.2 software (ERDAS Inc., 1996).

The second approach to image classification was based on the assumption that variations in peat density were continuous in nature. Accordingly, multiple regression analysis was used to develop a regression equation which would predict the density at a point on the bog based on the spectral responses of the TM bands of the pixel corresponding to that point. Appropriate density classes could then be determined by thresholding the predicted data.

An assessment of the accuracy of the results from the above methods was required to enable valid comparisons to be made and to provide end-users with a measure of confidence in the results. The standard approach to accuracy assessment in remote sensing is through the empirical comparison of the derived thematic (classification) map with independent ground truth information. Using the same ground truth information for accuracy assessment as was used for developing the classification is not recommended as this introduces a bias in the procedure resulting in an over-optimistic determination of accuracy (Hair *et al.*, 1992). Considering the effort required in gathering ground truth information for this application of remote sensing, a significant increase in the ground truth requirement could affect its viability as a cost effective approach to density variation mapping. Splitting the available ground truth information in half to provide separate samples for training and testing would, given the number of samples involved, significantly reduce the effectiveness of both procedures. Accordingly, for both classification approaches, the full dataset was used to develop the classification and accuracy was assessed using an approach referred to as "leave-one-out" cross-validation. This form of cross-validation involves removing each sample in turn from the dataset and developing the classifier or predictor using the remaining samples. The sample left out is then used to test against the classification thus developed, which can be considered a very close approximation to the classification derived from the full dataset. In this way, all the original data is used to develop the classification and subsequently to provide an independent assessment of the accuracy of the classification. "Leave-one-out" cross-validation has the effect of "squeezing the data almost dry" (Mosteller and Tukey, 1977) and has become a viable approach with the advent of modern computers.

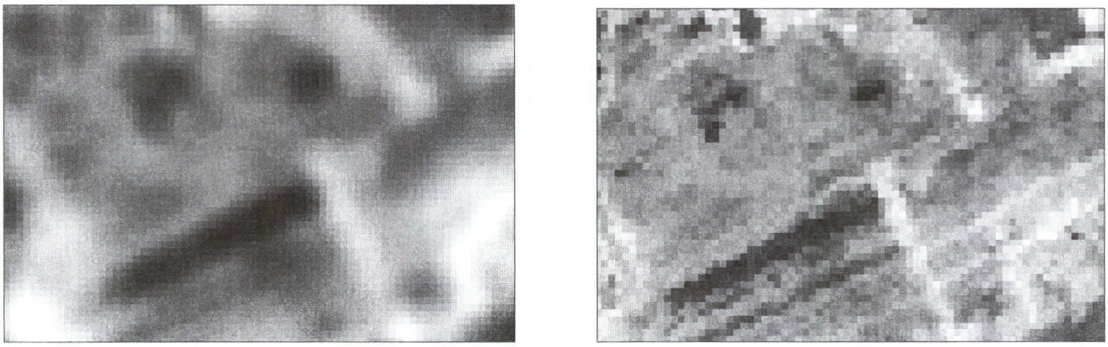
Results and discussion

Fig. 4 shows the effect of pre-classification smoothing on a section of the image. From the great variability in neighbouring pixels apparent in Fig. 4a, it is clear that any misregistration between the image and the ground truth locations could cause difficulties in the classification. In Fig. 4b it can be seen that the effects of any such misregistration are minimised demonstrating the importance of smoothing for this application.

Of the two approaches to image classification, the first approach presumed that the peat could be divided into the three discrete density classes used by the industry. Classification using class signature generation based on ground truth data, and applied using the maximum likelihood estimator algorithm, was used. To perform a supervised classification, the classes to be produced had to be identified. From inspection it was apparent that bands 4 and 5 showed the greatest range of DN values for points on the bogs. Accordingly, using each point in the dataset, a feature space plot of band 4 v. band 5 DN values was plotted (Fig. 5).

With the exception of one apparent outlier, three distinct clusters could be discerned, as hypothesised, and these were partitioned on the plot. These clusters corresponded to the following density ranges: 0.08-0.12 Mg m^{-3}, 0.13-0.18 Mg m^{-3}, 0.19-0.30 Mg m^{-3} and were referred to as low, medium and high density respectively. Although the three clusters were clearly distinct, significant feature space variability was noted within the clusters. The spectral signature statistics (mean, variance and covariance) for each of these classes were developed from the DN values of the points contained within their respective clusters. Fig. 6 shows the mean DN values for each of the three classes with the 95% confidence intervals indicated.

The following conclusions were drawn from Fig. 6:
(1) Separation between the signatures in bands 1 and 2 was evident but marginal. The higher average, and



(a) (b)

Fig. 4: Example of smoothing a 2.2 km x 1.4 km extract of Landsat TM image, 10th April 1997, of Boora Bogs complex, Co. Offaly, using a 5 x 5 mean value averaging filter. (a) Original image. (b) Smoothed image.

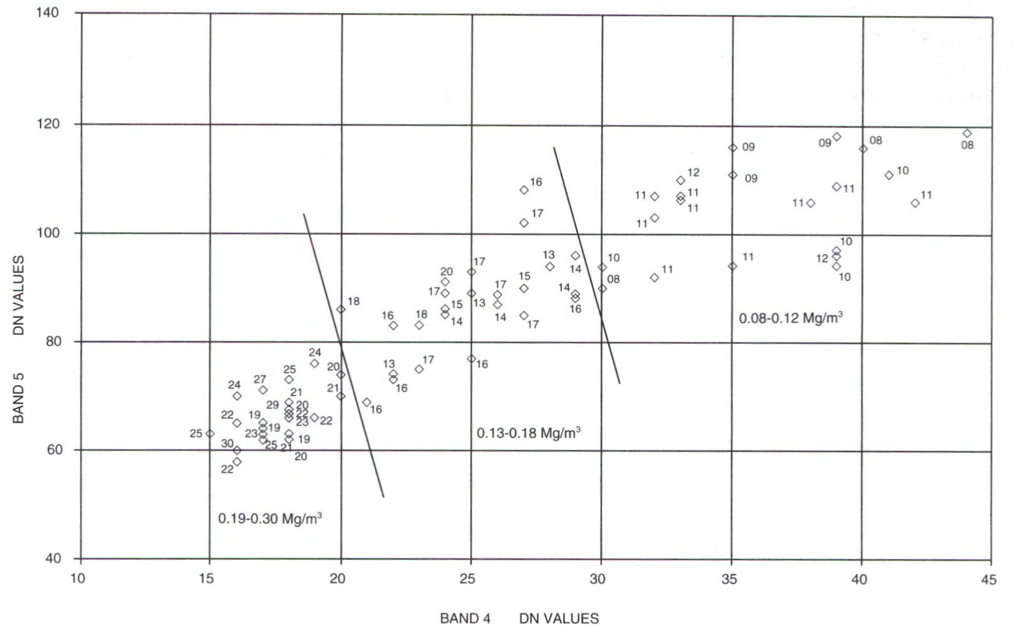


Fig. 5: Feature space plot of Band 4 v. Band 5 DN response values for ground sample points showing partitioning into three classes. à16 = sample position in feature space and density Mg m^{-3} (x 100).

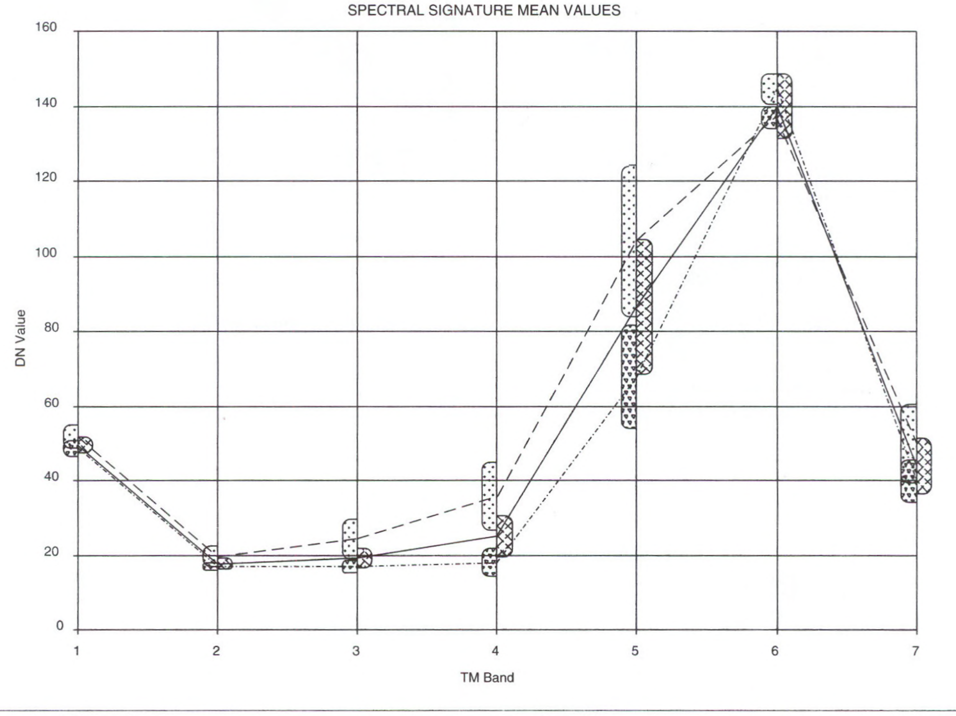


Fig. 6. Spectral response of signature means for three density classes with 95 % confidence intervals shown. – – – – – – = Low Density, ———— = Medium Density, –· – · – · – · – = High Density

greater spread, of DN values in band 1 was attributed to the increased effects of atmospheric backscatter at lower wavelengths (band 1 corresponds to the blue portion of the visible electromagnetic spectrum).
(2) Bands 3, 4 and 5 showed clear separation between the three class means. The bands did, however, show some overlap at the 95% level. Band 5 had the greatest range of values across signatures but band 4 showed the best separability with only a small amount of overlap between class signatures.
(3) Band 6, the thermal band, showed a reversal in the order of the signatures compared to the other bands. This was in accordance with the so-called "colour effect" on soil temperatures (Bowers and Hanks, 1965) whereby darker soils absorb more incident energy and thus have elevated daytime temperatures.
(4) Band 7 exhibited a greater degree of separation between signature means than band 3 but, because of the larger standard deviations, offered reduced powers of discrimination.
(5) Generally, while bands 1, 2, 6 and 7 appeared capable of reasonably discriminating only two classes, their inclusion in the classification was warranted because the maximum likelihood estimator, by virtue of its consideration of the covariances of the signatures, effectively weights the contribution of the different bands when carrying out the classification.

A quantitative assessment of signature separability using the Jeffries-Matusita distance algorithm (Swain and Davis, 1978) confirmed that the inclusion

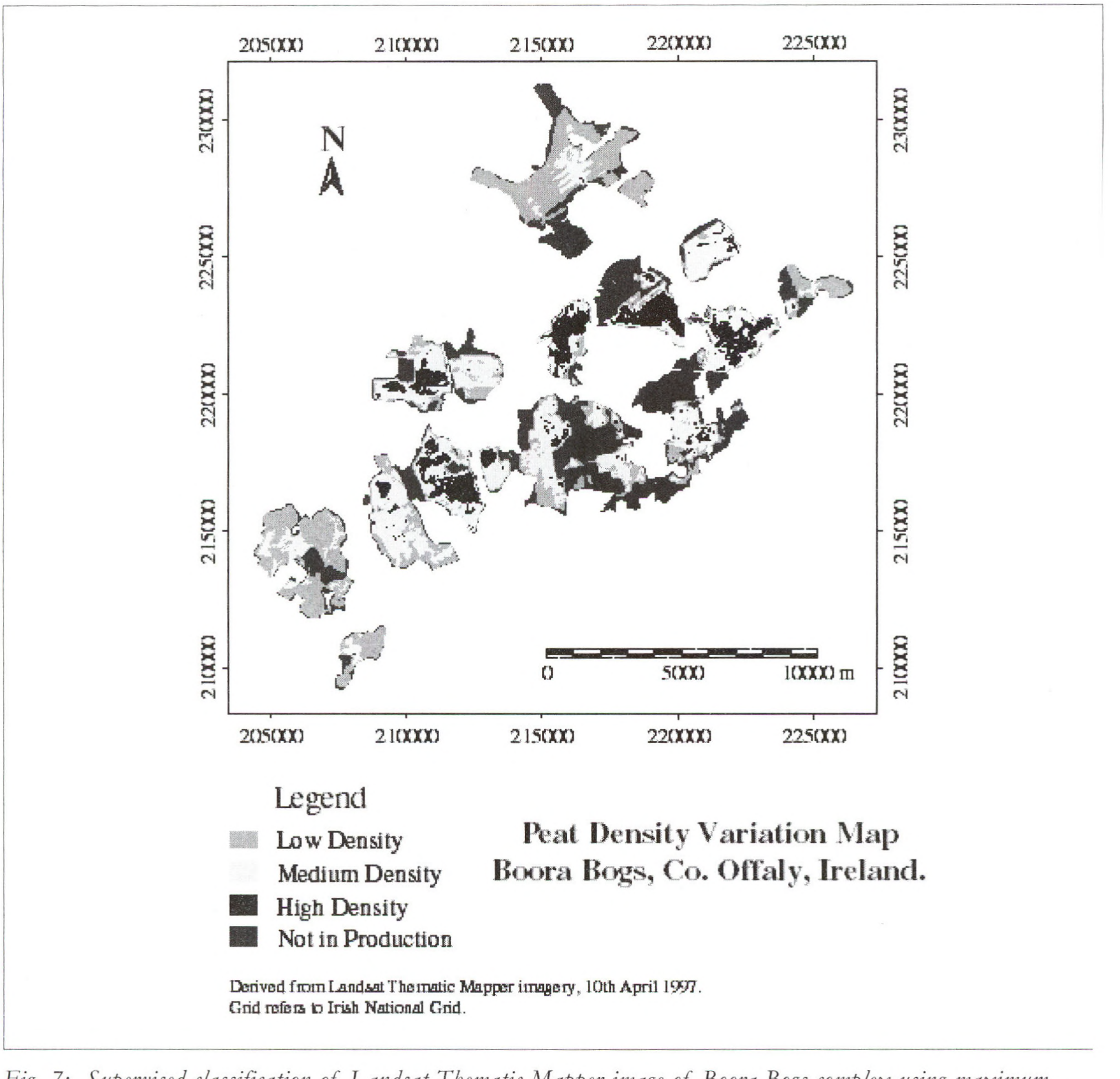


Fig. 7: Supervised classification of Landsat Thematic Mapper image of Boora Bogs complex using maximum likelihood estimator algorithm.

of all seven bands yielded the best separability and that band 4 was the single most discriminating of all the bands. Chiao *et al.,* (1986), in a study of forest cover types classification, also found that the use of all available bands optimised results. Due to the limited amount of training data and the importance of normally distributed training statistics for the maximum likelihood estimator, the training data were tested for normality. The R test (Ryan and Joiner, 1976), equivalent to the Shapiro-Wilk W test for small samples, was employed and confirmed that the training data were normally distributed. The satellite image was classified using the maximum likelihood estimator algorithm and the resulting milled peat density variation map is shown in Fig. 7.

"Leave-one-out" cross-validation, as previously described, was employed to assess the accuracy of classifications. The error matrix thus derived for the classification using the maximum likelihood estimator algorithm is shown in Table 1.

The overall accuracy of the derived thematic map, expressed as the percentage of correctly classified points to the total number of points used for accuracy assessment, was 87%. The lower limit of the overall accuracy, obtained by using the 95-percent one-tailed lower confidence limit (Snedecor and Cochran, 1967), was 80%. Hord and Brooner (1976) recommend that this lower figure should be quoted as the single figure estimate of overall map accuracy to ensure user confidence in the map. Another measure of map accuracy, recommended by Congalton *et al.,* (1983) and Rosenfield and Fitzpatrick-Lins (1986), is the Kappa coefficient. This is defined in terms of all the elements of the error matrix, and effectively adjusts the overall percentage correct measure by subtracting the estimated contribution of chance agreement in that measure. The adjusted overall accuracy as indicated by the Kappa coefficient was 81% and the variance of the coefficient was 0.026 using the formula given by Bishop *et al.,* (1975). The Kappa coefficient has gained wide acceptance although both Foody (1992) and Ma and Redmond (1995) have suggested that Kappa overdetermines the contribution of chance and thus underdetermines the adjusted overall accuracy. Regardless of what measure of accuracy is adopted, it was concluded that an overall classification accuracy of *at worst* 80% was achieved.

In the context of remote sensing, a production bog can be considered a relatively controlled environment. The surface is flat and therefore not subject to the topographical effects of uneven solar illumination and shadowing. Also, there is no plant growth on the surface so phenological variations are not an issue. Accordingly, a robust classification of production bogs within general land cover types can be expected. However, determining variations within such a relatively homogeneous surface is potentially more difficult and so the accuracy achieved can be considered better than might be expected. This degree of success is attributed to a robust field sampling methodology and precise GPS-derived location of ground truth data points on the satellite image.

Congalton (1991) used the terms "producer's accuracy" (P.A.) and "user's accuracy" (U.A.) to describe within-class measures of classification accuracy and thus provide a breakdown of the figure for

Table 1: Error Matrix and statistical results for maximum likelihood estimator algorithm classification and "leave-one-out" cross-validation testing.

Classified Data

Ground truth data		Low Density	Med. Density	High Density	Σ	P.A. %
	Low Density	20	2	0	22	91
	Med. Density	3	21	1	25	84
	High Density	0	3	21	24	88
	Σ	23	26	22	71	
	U.A. %	87	81	9		

Overall accuracy = 87 %, Lower 95 % confidence limit = 80 %.
Kappa coefficient ($\hat{k}$) *= 0.81, Variance of* k *= 0.026*
Adjusted overall accuracy = 81 %, P.A. % = Producer's accuracy, U.A. % = User's accuracy

overall accuracy in the error matrix. P.A. gives the analyst an estimate of how successful the classification procedure is in the different classes. U.A. gives the user an estimate of how reliable the thematic map is as a predictive device in the different classes. In this case it can be seen that the class of medium density performed worst in both categories, P.A. = 84% and U.A. = 81%, and was thus considered the least reliable of the classes. This was to be expected because the class of medium density inevitably overlapped with adjoining classes at both ends of its density range whereas the other two classes only overlapped at one end of their respective density ranges.

The second approach to image classification considered the variation in peat density as a continuum from low density to high density. Multiple regression analysis was used to regress dry weight density of milled peat on TM sensor DN values using all the available ground truth data. A "best subsets" regression analysis was first carried out to determine the relative contribution of the seven TM bands to the regression model. This determines the R-squared values for all possible combinations of predictors and extracts, for each order of combination, the best subset based on the largest R-squared criterion. The results are shown in Table 2.

Table 2. Best subsets, based on the maximum R-squared criterion, for increasing number of predictors.

TM band subset	R-squared (%)
4	75.2
5,7	80.7
3,5,7	83.3
2,3,5,7	84.0
2,3,5,6,7	84.2
2,3,4,5,6,7	84.2
1,2,3,4,5,6,7	84.3

From Table 2 it can be seen that band 4 had the highest discriminating power of all the bands and accounted for 75% of the total variability in density. The inclusion of all seven bands provided the strongest regression, although after the best three band combinations had been entered the contributions of subsequent bands were relatively small. These results were in line with those reached previously using discriminant analysis. With today's computing power there is generally no need to minimise the number of bands included in the regression for the sake of computational efficiency, so all seven bands were included. The derived regression equation, explaining 84% of the variability, was:

$$\text{Dry weight milled density} = 0.365 - 0.00168*TM1 + 0.0114*TM2 - 0.00679*TM3 - 0.00162*TM4 - 0.00504*TM5 - 0.00114*TM6 + 0.0101*TM7 \quad (1)$$

R-squared = 0.84

where TM# = DN value of the numbered Thematic Mapper band. The confidence interval for a value predicted by a multiple regression equation is a function of the particular set of input variables and so it is not possible to quote a global confidence interval for density values predicted using Equation 1. However, "leave-one-out" cross validation was employed to determine a reasonable estimate of the global confidence interval that can be expected. Each point, in turn, was removed from the dataset and the remaining points were used to develop a multiple regression equation of density on satellite DN response. The DN values of the point that was left out were then used in the regression equation thus developed to predict the density, and associated 95% confidence interval, at that point. The average of the 71 confidence intervals was 0.016 Mg m^{-3} 0.002 Mg m^{-3}. Therefore, 95% of the confidence intervals were between 0.012 Mg m^{-3} and 0.020 Mg m^{-3} and 0.02 Mg m^{-3}, the upper interval, was accordingly adopted as a reasonable 95% confidence interval for any peat density value predicted by Equation 1.

To enable a direct comparison to be made between the two approaches, the predicted densities from the regression analysis were thresholded into the same density classes as used in the discriminant analysis and, using "leave-one-out" cross-validation in the manner described above, an error matrix was produced (Table 3).

From the error matrix an overall accuracy of 90% was determined with a lower 95% confidence limit of 83%. The Kappa coefficient was 0.85, with a variance of 0.028, which indicated an adjusted overall accuracy of 85%. As before, a breakdown of the overall accuracy figures into producer's accuracy and user's accuracy showed that the medium density class was the least reliable of the three classes. The above figures are of the same order as those achieved using the discriminant approach (Table 1). However all the indicators of overall accuracy sug-

Table 3: Error Matrix and statistical results for multiple regression analysis classification with "leave-one-out" cross-validation testing.

Classified Data

Ground truth data	Low Density	Med. Density	High Density	Σ	P.A. %
Low Density	20	2	0	22	91
Med. Density	1	21	3	25	84
High Density	0	1	23	24	96
Σ	21	24	26	71	
U.A. %	95	87	88		

Overall accuracy = 90%, Lower 95% confidence limit = 83%.
Kappa coefficient (k) = 0.85, Variance of k = 0.028
Adjusted overall accuracy = 85%, P.A.% = Producer's accuracy, U.A.% = User's accuracy.

gested that the multiple regression approach yielded a more accurate classification than the discriminant analysis approach. Cohen (1960) described a pair-wise test of significance between two Kappa coefficients wherein if Z, the test statistic, exceeded 1.96, then the difference between the two coefficients was significant at the 95% probability level. Using the formula given by Cohen (1960), a value for the test statistic of Z = 0.17 was determined for the Kappa coefficients of the two classification approaches. This indicated that there was no significant difference between the two classification approaches in terms of accuracy.

There are, nevertheless, inherent differences between the two approaches to classification resulting from the different algorithms employed. In the case of the discriminant analysis approach using the maximum likelihood estimator, the area of interest on the satellite image is classified into the classes identified earlier at the training stage. The classes can be related, e.g. different peat densities, or unrelated, *e.g.*, peatlands and forest. In the regression analysis approach, however, the image is not classified but rather the density of the peat on the production bog is predicted for each pixel using the derived regression equation. If required, thresholds can be set within which the predicted density values can be placed to create density variation classes. Clearly then the regression equation can only be applied to pixels on the image that correspond to exposed peat on the ground.

In the context of the Precision Peat Production System (PPP) for milled peat, as previously discussed, of the two approaches considered, the regression analysis approach would appear to be more advantageous because peat density at a point on the bog is the requirement as input from this element of the PPP. Setting thresholds into which the individual density values are grouped effectively dilutes the information available in the original data. However, because of the nature of the production bog environment, the mechanical element of this part of the PPP, which adjusts the milling depth according to a number of system inputs (including density), cannot reliably make very small adjustments to the depth of the milled layer. Accordingly three depth settings probably reflects the sensitivity of the system. As a consequence, the discriminant analysis approach would appear to be generally the better option because it achieves the required sensitivity of classification within the bog area and can, at the same time, classify for other non-production bog areas which may be useful for a variety of applications related to the management of the overall bog resource. There are, however, specific applications, *e.g.* as input to milled peat rewetting model or peatland carbon sequestration models, where the greater resolution of the regression approach, in terms of derived peat density, may be of benefit. Both the class signatures for the discriminant function and the regression equation derived in this research should be applicable to TM images of all the production Midland raised bogs operated by Bord na Móna, whether separated in space or time, without further field work or processing but subject to successful inter-image spectral normalisation. This would make for an extremely cost effective method of determining density variability across those bogs. Further research is, however, required to validate this hypothesis.

Conclusions

A Landsat Thematic Mapper satellite image of the Boora Works complex of production bogs, situated in the Midlands of Ireland, was successfully calibrated for variations in the density of the peat using satellite positioning, satellite imagery and digital image processing. It was determined that three density classes could be reliably distinguished from the satellite image corresponding to the following density ranges: 0.08-0.12 Mg m^{-3}, 0.13-0.18 Mg m^{-3}, 0.19-0.30 Mg m^{-3}. TM bands 3, 4 and 5 (red, near infrared and mid infrared) showed clear discrimination between the three classes. Band 4 was the single most discriminating of all the bands, accounting for 75% of the total variability in density. Two approaches to classification were employed and compared. Firstly, discriminant analysis using the maximum likelihood estimator algorithm was used. Because of the limited number of training samples, a "leave-one-out" cross validation approach was employed to obtain a reliable accuracy assessment. An overall classification accuracy of 87% was achieved with a lower 95% confidence limit of 80% and a Kappa-adjusted overall accuracy of 81%. These were considered good results and the success was attributed to a robust field sampling methodology and precise GPS-derived location of ground truth data points on the satellite image. Secondly, multiple regression analysis was applied to the data and a regression equation that predicted the peat density at a point on the bog from the satellite image responses was developed. This regression equation had a coefficient of determination (R^2) of 0.84 and a 95% confidence interval of 0.02 Mg m^{-3} for densities predicted by this equation. Thresholding of the predicted densities into the same classes as used for discriminant analysis allowed for a direct comparison to be made between the two approaches. This showed that there was no significant difference between the two approaches in terms of accuracy.

It was concluded that the discriminant analysis approach would be more beneficial for applications concerned with general management of the bog resource because of its ability to classify, subject to adequate training, not only production areas but also non-production areas within the bog and areas outside the bog. The regression analysis approach was considered the more suitable for specific applications within the production bog area, *e.g.,* as input to the Precision Peat Production system, and as input to scientific studies involving peat, *e.g.,* carbon sequestration and peat rewetting models.

Acknowledgements

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TREE STAND STRUCTURE ON PRISTINE PEATLANDS AND CHANGES AFTER FOREST DRAINAGE

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SUMMARY

In Finland the trophic status of pristine mires is reflected in the number of stems per hectare. On sites rich in nutrients, the stem numbers are considerably higher than on poor sites. The tree stands preserve the same external appearance over decades, and increment is eliminated by natural removal, mainly through the death of older trees. Thus, tree stands on pristine mires, both Norway spruce (*Picea abies* (L.) Karst) on richer sites and Scots pine (*Pinus sylvestris* L.) on poorer sites, can be considered climax forests having a highly uneven-aged structure.

A typical feature for peatland forests is their great variation in tree age and size. Different approaches to study the development of the stem size distribution after forest drainage are discussed in the paper. Another feature is that pubescent birch (*Betula pubescens* Ehrh.), a typical pioneer tree, is able to occupy nutrient-rich sites. Later on spruce, being shade-tolerant, can grow as the lower layer in such two-storey stands, with birch acting as a nurse stand for the regenerating spruce. The best quality birches may even be grown up to timber size.

Keywords: mires, uneven-aged stands, even-aged stands, Picea abies, Pinus sylvestris, Betula pubescens.

INTRODUCTION

Some peatlands support tree growth naturally in Finland, but forest drainage enables economic wood production from a much more extensive peatland area. Forest improvement, particularly the draining of peatlands and waterlogged mineral soil sites, has been shown to significantly increase the growth of tree stands (volume basis) and thereby contribute to the sustainable forestry practised on these areas - provided the measures have been directed towards appropriate sites (Paavilainen & Päivänen, 1995; Tomppo, 1999). Very little is known, however, about the tree stand structure before and after forest drainage, and the quality of wood raw material and its suitability for different purposes. The question is especially important in Finland where about 5.4 million ha of peatlands, more than half of the total peatland area, has been drained for forestry.

The aim of this paper is to describe the general features of tree stands on peatlands and discuss the approaches to studying their development after forest drainage. This is done partly by reviewing relevant literature and partly by preliminary graphical analysis of some successive measurements in permanent sample plots. The data analysed originate from permanent sample plots established by the Department of Forest Ecology (see Ekola & Päivänen, 1991), University of Helsinki, on forest drainage areas in the surroundings of Hyytiälä Forestry Station in southern Finland (61° 50′ N; 24° 17′ E).

CONCEPT OF PEATLAND AND MIRE

The main environmental factor influencing the initiation of mire vegetation is water, both its quantity and quality. In the boreal zone, in particular, wetland ecosystems are often characterized by the accumulation of organic matter, which is produced and deposited at a greater rate than it is decomposed, leading to the formation of peat (Gore, 1983). A prerequisite for peat formation is excess water on the surface of the soil. Usually this means that peatlands are located in depressions or in flat areas. Peatlands can, however, also develop on slopes where precipitation is much greater than evapotranspiration (Verry, 1997).

These sites, called *peatlands* or *mires*, are usually supported by humid climate and high water table levels, leading to a low level of microbial activity in the soil. The term mire can be considered a slightly wide concept than peatland, because it encompasses all peat-forming habitats, irrespective of peat thickness. In an ecological context the term mire has been preferred, whereas in connection with forest amelioration and management, the more traditional term peatland has been used (Paavilainen & Päivänen, 1995).

Peatlands can be broadly classified into two groups according to the origin and quality of the water supporting the vegetation. *Ombrotrophic* peatlands (*bogs*) receive water and chemical elements from atmospheric deposition only. *Minerotrophic* peatlands (*fens, swamps*) receive additional nutrients from water that has passed through the upland (mineral soil) part of the catchment.

TREE STANDS ON PRISTINE PEATLANDS

A mire site may be forested, sparsely forested, or treeless. Sparsely forested sites have also been referred to as *treed fens* or *treed bogs* (Jeglum, 1991).

In Finland some forested mires support commercial size tree stands. Minerotrophic sites are dominated usually by Norway spruce (*Picea abies*) although the proportion of pubescent birch (*Betula pubescens*) on these by volume may be considerable. Ombrotrophic sites are dominated by Scots pine (*Pinus sylvestris*).

Typically for pristine mire tree stands, the number of stems is very high in small diameter classes and decreases abruptly with increasing diameter (Gustavsen & Päivänen, 1986; Norokorpi *et al.*, 1997). Only trees growing on the best micro-sites are able to continue their growth. Owing to the high water table and overgrowth of *Sphagnum* mosses, most of the trees die sooner or later. The result is that both the age and the size structures of tree stands on pristine mires are uneven. They represent a dynamic stability with new individuals continually emerging while others are dying. Small-scale disturbances, caused mainly by single tree falls, make it possible for some of the younger and suppressed individuals to develop (Hörnberg *et al.,* 1995). A tree stand can maintain the same external appearance over decades; increment is eliminated by natural mortality. Thus, tree stands on pristine mires, both Norway spruce on richer sites and Scots pine on poorer sites, can be considered climax forests (Heikurainen, 1971).

In general, the stem diameter distributions on pristine forested mires and in mineral soil sites are totally different (Fig. 1). The mire site (bar graph) represents a mean for several Scots pine stands on dwarf shrub pine mires (Gustavsen & Päivänen, 1986) while the mineral soil site data (curve) are from a 60 year old unmanaged Scots pine stand on a *Vaccinium vitis-idaea* site type (Ilvessalo, 1965).

Very little tree harvesting is carried out on pristine, forested mires in Finland. The cuttings, if any, should be "light", otherwise there is a risk that the growing conditions for the trees to be left will become worse owing to the groundwater rise (Päivänen, 1982). Only trees that would die in the near future should be harvested. Indeed, pristine, forested mires may be one of the few cases where silviculture to promote an uneven-aged stand structure could be ecologically recommended, although Lähde (1992) is applying this method to fertile sites on mineral soils. The idea in this kind of single tree selection (continuous cover) is to retain an all-sized (uneven-aged) mixed stand structure (Lähde *et al.*, 1992). Harvesting of single trees on pristine mires is, however, usually not economical (Paavilainen & Päivänen, 1995) because of the low yield and high harvesting costs.

EMPHASIS ON FOREST DRAINAGE

In the case of pristine mires, excess water in the substrate checks root growth and microbial activity, and may lead to unfavourable biochemical phenomena. One of the most important goals of draining is

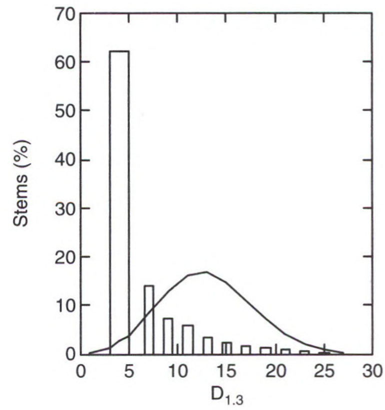

*Fig. 1: Diameter ($D_{1.3}$) distribution for Scots pine (*Pinus sylvestris*) stands on pristine dwarf shrub pine bog (bar graph) and on mineral soil representing* Vaccinium vitis-idaea *site type (curve). (Redrawn from Gustavsen & Päivänen, 1986, and Ilvessalo, 1965).*

therefore to adjust the water content of the peat soil to a level that ensures sufficient aeration (Päivänen, 1973).

The actual forest drainage is carried out using the cross-drainage principle; that is, the bulk of the ditches in the system drain water by collecting it as it moves laterally through the surface and sub-surface soil layers. The most commonly used machine for forest drainage has been the tractor digger. Operational recommendations for ditch spacing (using about 90 cm deep ditches) in boreal conditions have been 50 m for shallow peat soils, 40 m for forested thick peat soils and 30 m for sparsely treed mires (Rosen, 1989; Paavilainen & Päivänen, 1995).

Tree Stand Development After Drainage

Response Period

Much of the forest drainage in Fennoscandia has been on naturally tree-covered areas. Treeless peatlands have usually been drained only if it has been supposed that the area will be afforested naturally. This means that in most cases the ameliorated tree growth starts from a prevailing but moribund tree stand.

The term response refers to the difference between post-drainage and pre-drainage tree growth. Usually both the height growth and radial growth response of short trees and trees with a small diameter have been found to be better than those of tall trees (Heikurainen & Kuusela 1962). This means that the uneveness in size structure between the trees prevailing on the site at the time of drainage may diminish after drainage. Tree size cannot be used, however, as the only indicator of response capability. The existence of a clearly discernible top shoot, sufficient crown size as well as a green needle colour may be better indicators of a tree's physiological condition and its response capability. Minerotrophic mire site types will develop mixed stands of pubescent birch (*Betula pubescens*), Norway spruce and Scots pine after drainage. The more nutrient-poor sites support mainly Scots pine (Keltikangas *et al.*, 1986).

Two Storied Stands

In drained peatlands, pubescent birch is a typical pioneer tree species, which can quickly occupy sites that for some reason have free space as a result of ditching or clear-cutting. Since birch is a rather short-lived light-demanding species, it gives way to spruce in time. Spruce, being shade-tolerant, can grow as the lower layer in these two-storey stands, with birch acting as a nurse stand for the naturally regenerating spruce (Fig. 2). If the aim is to grow a predominantly coniferous stand as soon as possible, Norway spruce should be released at the height of 4 m (Heikurainen, 1985). It is also possible to grow two-storey stands where a part of the birch will reach timber size later after thinning. The standards (nurse trees) have to be harvested in winter during thaw so that spruce forming the next generation on the site will not be damaged. These two-storey stands do not, however, fulfil the concept of an uneven-aged stand.

Stand Structure Development

Surveys Based on Temporary Sample Plots

Owing to several factors, the typical features of peatland forests at the time of drainage (i.e., their great variation in tree age and size) may remain, or are emphasized, during the first decades after draining but before the first thinning (see Hökkä & Laine, 1988, p. 162, Fig. 6.6; Uuttera *et al.*, 1996). Apart from genuine, forested hardwood-spruce mires, the tree cover on virgin mires is sparse and unevenly scattered over the area. The latter situation is particularly true for sparsely forested composite pine mires and hardwood-spruce mires. As growth conditions improve after ditching, openings fill up with

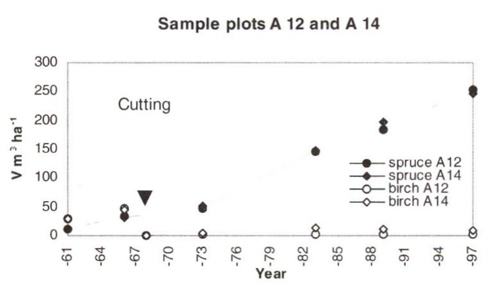

*Fig. 2: The development of the stand volume on sample plots A12 and A14 on minerotrophic peat soil (according to Laine (1989): herb-rich type) drained for forestry. Norway spruce (*Picea abies*), naturally regenerated under the pioneer birch (*Betula pubescens*), is forming the second rotation. For detailed stand descriptions, see Ekola & Päivänen (1991).*

seedlings and stand density increases. In nutrient-rich composite types in particular, an immediate reaction to drainage is observed in which there is a large increase in the quantity of small diameter trees, especially pubescent birch. This increase slows when the stands have reached full density in respect to the site and the development class of the stand. If correct, these observations may also indicate that treatment of young stands and first commercial thinning have been delayed or totally omitted in forest drainage areas.

Another typical feature of tree stands in drained areas is that the trees grow in groups. This is particularly true for the composite types of hardwood-spruce and pine mires. The gaps are gradually filled in after drainage. Drainage may, however, also lead to unevenness in tree growth within the stand. Tree growth is usually better closer to the ditches (Seppälä, 1972; Miina, 1994). Thus, trees growing in the middle of the strips between ditches will be outgrown by those growing near the ditches (Miina *et al.*, 1991). This difference in growth is clearer the greater the ditch spacing or the more defective the drainage system.

The number of stems reaching saw-timber dimensions in the first stand after draining depends on both the geographical location and the fertility of the site. Nutrient-rich composite mire sites have shown particularly high potential to produce saw-timber dimensions during the first 50 years after forest drainage (Hökkä & Laine, 1988).

Approach Based on Permanent Sample Plots

The uneven structures of age and size described for tree stands on both pristine mires and drainage areas should be further discussed. The stem distribution curves of single stands based on temporate sample plots (Gustavsen & Päivänen, 1986; Hökkä & Laine, 1988; Norokorpi *et al.*, 1997) may, however, give a biased view if extended to describe the situation for large forest areas (e.g. Uuttera *et al.*, 1997).

This is demonstrated by examining the diameter distributions of three Scots pine stands growing on the same drained peatland forest site type (according to Laine (1989) *Vaccinium vitis-idaea* type II) but representing different development stages. When the three different stages are combined, the relative stem size distribution forms a reversed J-shape, although the individual stands, especially in the more advanced stages, show fairly normal size distribution (Fig. 3). This is because the stem number in young stands is always greater than in the thinning stage

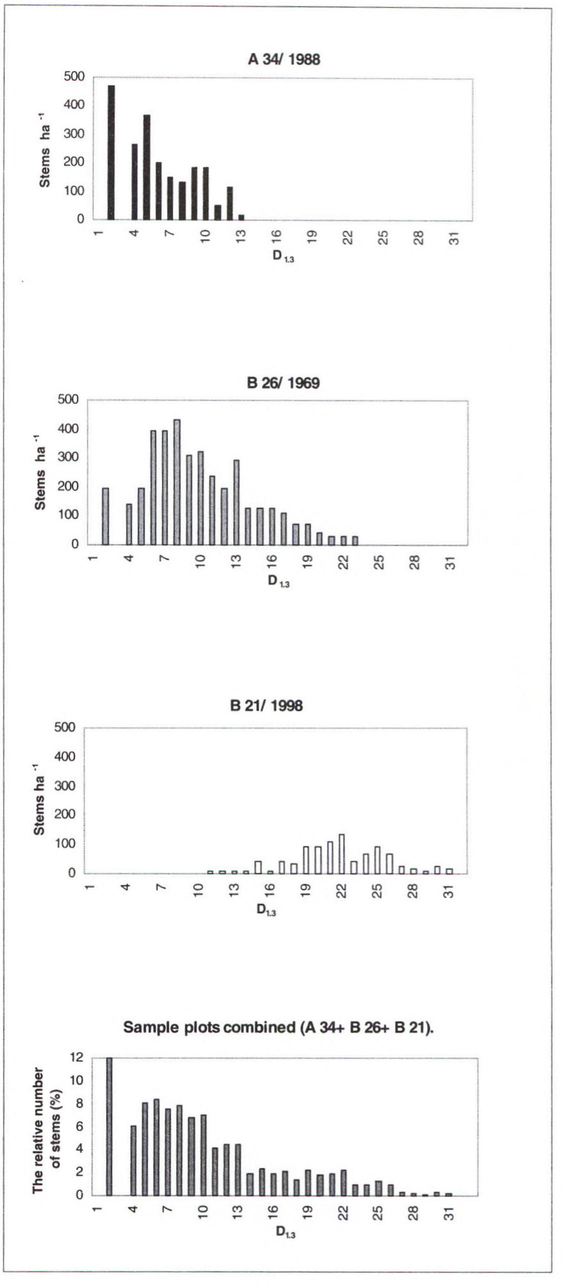

*Fig. 3. Diameter ($D_{1.3}$) distributions of three Scots pine (*Pinus sylvestris*) stands on drained peatland representing different development stages: A34/1988 = Naturally regenerated seedling stand 20 years after the harvest of seed trees; B26/1969 = Naturally regenerated stand without any man-made thinnings; B21/1998 = Mature stand where most of the stems reached sawwood dimensions. The site slightly minerotrophic (according to Laine (1989):* Vaccinium vitis-idaea *type II). For detailed stand descriptions, see Ekola & Päivänen (1991). In the fourth sub-figure the sample plots are combined and the relative diameter distribution form a reversed J-shape.*

stands, which again have more stems than the mature stands.

The more correct approach would be to study the development of the stem diameter distribution based on successive measurements on permanent sample plots (see e.g. Hökkä *et al.*, 1991). Growth and yield studies based on successively measured sample plots do not usually, however, give the information about the stand structure (e.g. Hökkä *et al.*, 1997; Gustavsen *et al.*, 1998).

In this connection the development of diameter distribution in a Scots pine dominated permanent sample plot on an ombrotrophic drained mire is visualized based on successive measurements (Fig. 4). The mire had originally been a dwarf shrub pine bog (see Laine & Vasander, 1997) and it had been drained as early as 1915. In the early phase only light thinning from below has been performed, and before measurement in 1988 about 41 m^3 ha^{-1} had been removed from the sample plot. At the time of the last measurement in 1998 there were 622 trees per hectare and the volume of the stand was 184 m^3 ha^{-1}.

The diameter distribution becomes more or less normal when several decades have passed since drainage. Over the decades the mean diameter moves logically towards the bigger diameter classes.

Discussion

Pristine mires have only a marginal importance for wood production in Fennoscandia. Thus their uneven-aged (uneven-sized) structure and its maintenance, which from the ecological point of view seems to be sensible, is only of minor interest.

The approach to mire utilization for forestry has for decades been not only sustainable but progressive management, that is drainage, thinning, ditch cleaning, final harvesting and regeneration (Päivänen & Paavilainen, 1997). On the operational scale two storied stands (Norway spruce under pubescent birch) have been grown successfully. As mentioned the early study results showing a high frequency of uneven-sized stand structures also on drained sites may, however, be partly biased.

The preliminary results from old forest drainage areas (Fig. 3 and 4) indicate that the stand structure after the revival period approaches a "normal" distribution, at least on unfertile sites with a predominance of Scots pine. This is logical because natural regeneration has been found to be hampered by feather mosses on old drainage areas (see discussion in Paavilainen & Päivänen, 1995).

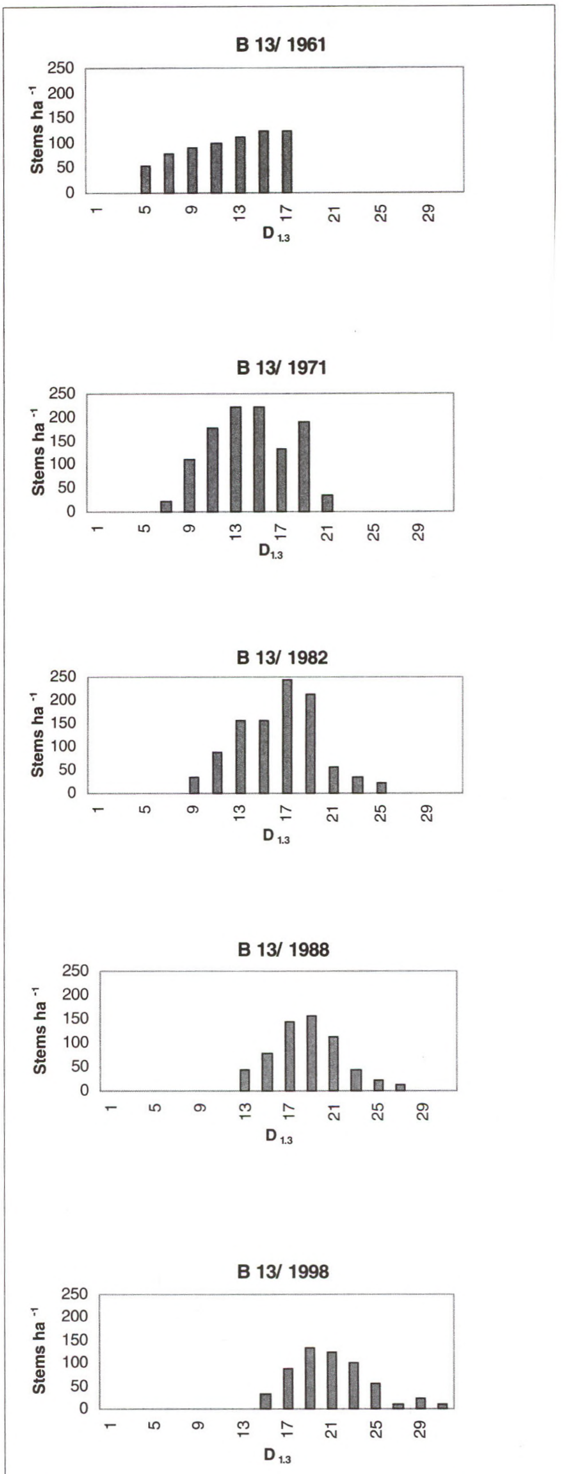

*Fig. 4 Diameter ($D_{1.3}$) distributions for a Scots pine (*Pinus sylvestris*) stand on a permanent sample plot (B13) situated on an old drainage area. The original site type at the time of drainage (1915) was ombrotrophic dwarf shrub pine bog. For detailed stand descriptions, see Ekola & Päivänen (1991).*

The structure and development of tree stands on drained sites as well as the usability of the wood material produced for different purposes (e.g. saw timber, pulp or energy) is not yet fully understood but a fairly thorough study program has been initiated quite recently. This includes a survey of the development of the stem distribution based on a great number (> 400) of successively measured permanent sample plots. The measured diameter distribution will be fitted with the Weibull-function. After that the correlation between the distribution parameters and tree stand characteristics can be tested (Hökkä *et al.*, 1991). In the other part of the study consort the quality of the raw wood material will also be examined.

The ditch network and its maintenance in forest drainage areas must also be taken into consideration. This fact may support the suggestion for more intensive but less frequent thinning, and natural or artificial regeneration aiming at more or less even-aged management.

ACKNOWLEDGEMENTS

This paper would not have been possible without the far-sighted teachers and scientists in the former Department of Peatland Forestry (at present the Department of Forest Ecology), University of Helsinki, who established and managed for decades the permanent sample plots on drained peatlands in the vicinity of the Hyytiälä Forestry Station. The last measurements and calculations were done by Mr. Sakari Sarkkola. The project, to which the paper serves as a kind of introduction, is financed by the Ministry of Agriculture and Forestry, Finland.

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ECOSYSTEM RECOVERY AND NATURAL DEGRADATION OF SPILLED CRUDE OIL IN PEAT BOG ECOSYSTEMS OF WEST-SIBERIA

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SUMMARY

The peat bog ecosystems in the southern part of the West-Siberian Plain situated over oil fields are subject to various types of pollution related to oil exploration and exploitation activities. In a co-operative research project between the Universities of Tomsk and Utrecht, 50 well dated polluted locations and 10 undisturbed reference sites were selected for sampling peat and peat-water and for making vegetation surveys. The data range covers a period of 30 years. Through biochemical breakdown and by physical chemical processes the concentration of oil components in aerated water was reduced to the regional background level within 20-25 years. In the anoxic environment of the drill mud, no decrease of any oil components was detected within 30 years. Recovery of bog vegetation after pollution may last for decades up to centuries depending on the intensity of the pollution and the site characteristics.

Keywords: peat, oil, biodegradation, vegetation, habitat recovery.

INTRODUCTION

Within the extensive peat bog ecosystems of the West-Siberian Plain, an increasing number of locations are subject to pollution resulting from oil exploration, oil exploitation, and related activities. This pollution is a matter of concern for the Tomsk provincial authorities. Where spills occur close to running surface water, action is taken in order to avoid further spread and impacts upon water quality. Locations in remote peat areas, however, are left untreated as the prevailing opinion is that oil, in common with peat, is a product of nature and therefore the pollution will disappear by natural breakdown processes.

In Europe and America these pollution sites normally have to be cleaned up because they conflict with other land uses. The best solution in many cases is the removal of the contamination by digging away the polluted soil and replacement with clean soil. Another solution is *in situ* stimulation of biodegradation processes. The latter solution brings uncertainties about the oil concentration after the costly process is ended, because bacterial breakdown (artificial biodegradation) cannot clean the soil and groundwater completely. The question then becomes are the concentrations below the acceptable contamination standards established by law? As a result of strict legislation in the West, decades-long experiments to determine the natural biodegradation processes for prediction of the time elapsed until standard concentrations will be reached, are out of the question. Another experimental problem is that, in all temperate areas in Europe and North America locations cannot be found where oil pollution can be studied solely, because of interference with other, point and diffuse sources of pollution.

The West Siberian Plain, which consists of vast almost undisturbed areas apart from oil activities, does not have these limitations for experimental research. In effect, the exploration and exploitation of oil fields within this uninhabited area produces ideal sites for studying natural remediation processes, including biodegradation and vegetation recovery, following oil pollution.

Study Area and Its Vegetation

The study area is situated in the southern part of the West Siberian Plain in the Province of Tomsk in the western part of the river Ob catchment (Lat. N 57-61°, Long. E 76-78°,) and includes five oil deposits near Igol, Olenje, Pervomajskoje, and Strezjevoj (Fig. 1). The climate is continental with only a 3-month growing season (mean summer month temperatures up to 17.4°C) and severe winter (coldest mean month temperatures -21°C), with long lasting snow cover. Precipitation, snow included, is 450-700 mm yr^{-1}. Surface elevation varies from 40-140 metres above mean sea level.

The Pleistocene loamy sand soils in 60% of the area are covered with peat. At the watersheds, extensive ombrotrophic open bogs ('Aapa-mires') have developed with a maximum thickness of 6 metres. On the characteristic ridges ('ryam'), communities of *Sphagnum fuscum* with *Ledum palustre*, *Chamaedaphne calyculata*, *Andromeda polifolia*, *Rubus chamaemorus*, *Oxycoccus microcarpus and Drosera rotundifolia* grow, with scattered *Pinus sylvestris* trees of 0.5-2 m in height. In the hollows, various Rhynchosporion albae marsh communities dominate, consisting commonly of *Carex limosa*, *Scheuchzeria palustris*, *Rhynchospora alba*, *Sphagnum balticum*, *Sphagnum jensenii* and, more rarely, *Sphagnum lindbergii and Sphagnum compactum*. Toward the edges of the watershed bog-complexes, the pine trees in the pinus-dwarf shrub-sphagnum ridges grow higher and grasses appear. There, waterlogged mesotrophic and meso-oligotrophic, sedge-sphagnum communities consisting of *Carex rostrata*, *C. lasiocarpa*, *C. limosa*, *Equisetum fluviatile*, *Sphagnum fallax* and *S. obtusum* have developed. Further down, partly paludified woodlands of *Pinus sylvestris* and mixtures of *Pinus sylvestris, Pinus sibirica*, *Betula pendula* and *B. pubescens occur.* In the understorey *Vaccinium*

Fig. 1: The study area.

vitis-idaea, *V. myrtillus* and taiga low growing herb species (*Maianthemum bifolium, Trientalis europaea, Oxalis acetosella,* and *Goodyera repens*) prevail. Also present are *Lycopodium annotinum, L. clavatum, Diphaziastrum complanatum, Equisetum sylvaticum, E. pteridophyte and Melampyrum pratense.* In the wetter boggy forest, *Ledum palustre* forms a dwarf shrub layer (Bootsma *et al.,* 1995).

The dry mixed taiga woodland on the river terraces and the eutrophic floodplain ecosystems were excluded from this research.

Oil Production Methods in Tomsk Province

The current procedures for drilling and oil production applied in the Tomsk Province lead to heavy pollution with salty production water, drill mud and crude oil from exploitation tests, leakage, and pipe bursts.

Additionally, for prospect drilling, a surface area of 2-4 ha is cleared of trees and a basin is dug out, creating a reservoir (in Russian: 'ambar') of about 400 m^3 for the storage of drill mud, composed of drill clay and salty groundwater (brine) in variable compositions. Crude oil from productivity tests (up to several tens of m^3) is also dumped into the ambar. Until 1994 the wastes from prospective drillings on remote oligotrophic peat bogs were spread over the surrounding areas, without the use of any ambar. By this last practice, surface areas up to some tens of hectares were polluted.

For oil production a limited number of sites are selected, from which the oil bearing layers are reached by several, sometimes inclined, drillings. For the construction of a typical oil production platform (Russian: 'kyst') the peat soil is partly or completely dug away and the surface elevation is increased with a 1-2 m sand layer, and a big ambar, surrounded by sand dykes, is dug out. Ecosystems downhill from both types of ambars are occasionally contaminated by the waste materials stored in the ambar, as their sand dykes can break in peat areas simply by subsidence in the peat.

Kysts are connected by temporary roads and high-pressure pipelines for transportation of the oil and for reuse of the salt water derived from the oil-bearing formation. Spills of crude oil and salt 'formation water' (Table 1) occur frequently owing to pipe bursts.

Recently, forced by regional environmental legislation, ambars situated close to inhabited areas have to be 'recultivated' after the termination of drilling activities. In practice this means that these ambars are filled up with sand. Remote sites, in particular those used for exploration drilling, are abandoned, giving nature the possibility to restore itself. As most exploration drillings are well documented and situated in the middle of pristine bogs and taiga forests, these sites are excellent places to study the natural recovery processes by comparison of spills of different age.

Objectives

The objective of this research was to investigate drilling sites within the southern taiga zone of West Siberia with the purpose of determining the recovery of peat ecosystems and the natural breakdown processes in soil and water and soil-waters polluted with oil and brine resulting from oil exploration and

Table 1: The composition of formation water (production water) and several water types in the study area. Formation water is derived from the oil-containing Cenomanien layers.

		Formation water	Soil water	River	Rain	Snow
Na^+	(mg l^{-1})	2537	8.6	29.0	6.1	1.7
Ca^{2+}	(mg l^{-1})	280	77.8	50.1	1.0	1.5
Mg^{2+}	(mg l^{-1})	24	13.8	9.9		
NH_4^+	(mg l^{-1})	-	1.8	1.3	1.5	0.3
Cl^-	(mg l^{-1})	6027	1.8	12.9	1.7	1.1
HCO_3^-	(mg l^{-1})	134	317	259	21.4	10.7
oil product	(mg l^{-1})	-	0.016	0.030	-	0.021
pH		8.2	7.2	7.0	5.0	5.7

exploitation activities. A second aim was the development of guidelines for oil production that would result in less damage to and pollution of ecosystems.

METHODS

Based on the Oil Company records of exploration and exploitation drillings and pollution events, 85 sites suitable for research were selected. Essential in this choice was the availability of the date (year) of drilling or pollution event. Only sites on peat soils (bog, fen or taiga woodland) were selected.

During summer expeditions in 1995 and 1996 a total of 40 sites (Strzjevoj: 5; Olenje: 11; Pervomajskoje: 17; and Igol: 7) were visited for analysis and sampling. These 40 sites cover a pollution history of 25 years. At each site a landscape ecological description was made, including the hydrological position of the pollution and the soil strata. Also at each site, vegetation surveys were made and soil, soil-water, and water samples were taken in 3 types of locations: (a) in the ambar, expected to be polluted; (b) downhill from the ambar, possibly polluted; and (c): unpolluted reference sites.

Water samples (2 litres) were taken from just below the water surface in the ambar and from small lakes. Soil water and peat water samples were obtained from 20-40 cm deep pits. Acidity (pH), electrical conductivity (EC), temperature and alkalinity were measured in the field. Macro-ions were analyzed mostly in Russia by titration and a limited number of samples was analyzed in Utrecht (Laboratory of Physical Geography) by atomic absorption spectrophotometry colorimetry using a (SCALAR) autoanalyzer and for metals by Inductively Coupled Plasma Atomic Emission Spectrometry (ICP-AES). Oil compound composition was determined by combined Gas Chromatography - Mass Spectrography (GC-MS) analysis (Tomsk Oil Laboratory) following pretreatment of samples by extraction in a 30 ml solution of chloroform within 6 hours after collection (United States Environmental Protection Agency: EPA method 625). Between pretreatment and analyses the samples were stored at 4°C in a refrigerator.

Soil samples were taken from the ambar bottom by a pulse or hand drill and excised peat soil samples were sealed in plastic bags. For GC-MS analyses 100 g soil material was air-dried for several days. From this dried soil, 20 g was treated for 12 hours using a Soxhlet-device with chloroform to extract the oil compounds.

RESULTS

Water

In ambar water the total concentration of hydrocarbons from oil ('oil product') is related to the date of the original pollution (Fig. 2). The concentration of 'oil product' decreases to the reference level within a period of 20 years in the aerobic conditions of open water in the 'ambar'. This decrease can result from: (1) microbiological breakdown of oil components (Leahy & Colwell, 1990; Morgan & Watkinson, 1994); (2) dilution through the yearly input of precipitation (rain, snow); (3) percolation to the groundwater; (4) evaporation of volatile components; and/or, (5) by settling of heavy oil components. Both heavy and light components will be formed by biochemical breakdown of oil. The effects of precipitation seem

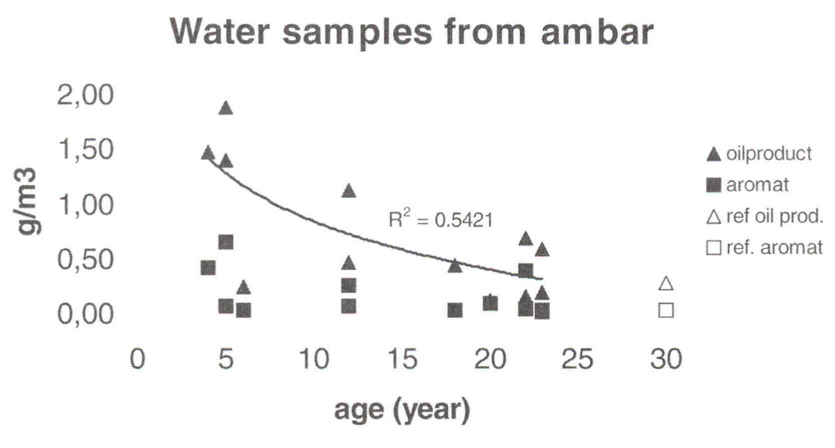


Fig. 2. Results of biochemical analyses of water samples from sites polluted in different years and analyses of reference samples. Aromat = sum of aromatic hydro-carbons; Line and R²: represent log.regression through 'oil product' data.

to be small, because these should affect all dissolved oil component concentrations while the aromatic hydrocarbon concentrations do not decrease significantly (Fig. 2). The constancy over time of aromatic hydrocarbons confirms that they are more difficult to break down bacteriologically (Hunt, 1996).

Ambar Soil

The patterns of concentrations of 'oil product' and 'aromats' for soil samples do not show any relation to date of pollution event (Fig. 3). Within the examined time period, no significant breakdown of oil components occurred. Because the initial quantities of emitted crude oil are not known, this could mask decreases in concentration. The concentrations of the biomarkers pristane and phytane were also examined (Britton, 1984; Peters & Moldowan, 1993). The ratios of neither Pr/n-C17 nor Ph/n-C18 revealed any time related changes. This lack of degradation might be attributed to the anoxic environment in the bottom soils, which mostly have high clay content (from drilling solution). Bacteria living in these environments usually have a lower capacity for the breakdown of oil components (Leahy & Colwell, 1990). The clay content of the ambar soil also decreases the availability of oil components for micro-organisms (Sugiura *et al.,* 1997) because of the compacted nature of the bentonite-oil mixture. In this way the drilling solution shields even simple aromatic hydrocarbons from attack by micro-organisms.

The conclusion is therefore that the pollution in ambar bottoms will remain for a considerable time period, as long as measures are not taken to drastically change their environment.

Peat Water Samples

The peat ecosystems downhill from many ambars have been polluted by dyke bursts. In the case of exploration sites, pollution plumes consisting of the lighter oil components floating on the ambar water covered the surrounding peat to variable extents. Occasionally the lighter oil mass was mixed with eroded ambar bottom material, including clay and pure oil products. In exploration sites without ambars, all drill solutions together with crude oil from production tests have been spread over the surrounding peat. After ending the drilling activities, a decrease of the oil component concentrations and of the salty drill solution by dilution and degradation processes was expected.

In Fig. 4 some characteristics of the sampled peat water and the concentrations of oil product and aromatic hydrocarbons are presented for two time periods since the pollution event. Because the total number of analyzed peat-water samples is small (n=12) the median concentrations are presented. For comparison also the maximum concentrations (from the 3-6 year group) are given. Reference concentrations are from water samples taken from not directly polluted peat water and small lakes. The references do not represent real base line values, but give an impression of the local background, which is influenced by the spreading and deposition of atmospheric pollution originating from the oil processing activities and gas torches. Background concentrations of undisturbed peat bog waters are given in Table 2.

A clear decrease in the salt concentration (EC, Na+K) results from dilution with precipitation water after pollution. However, this trend is not seen in the

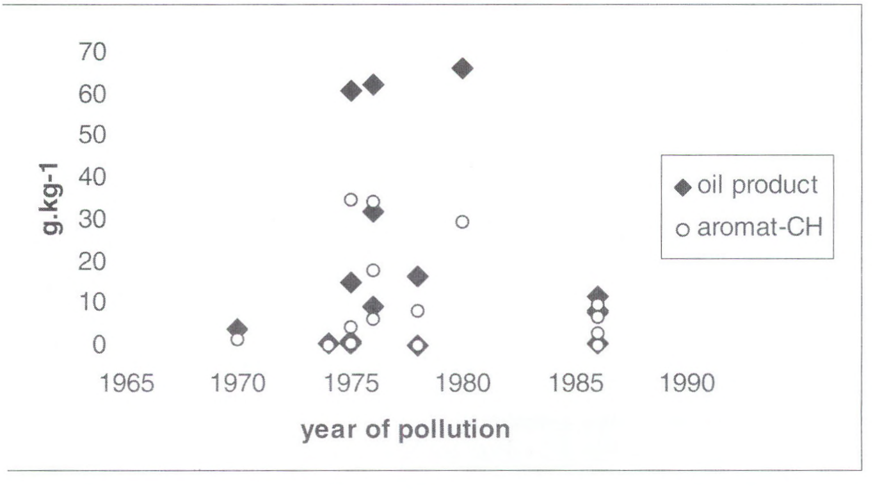
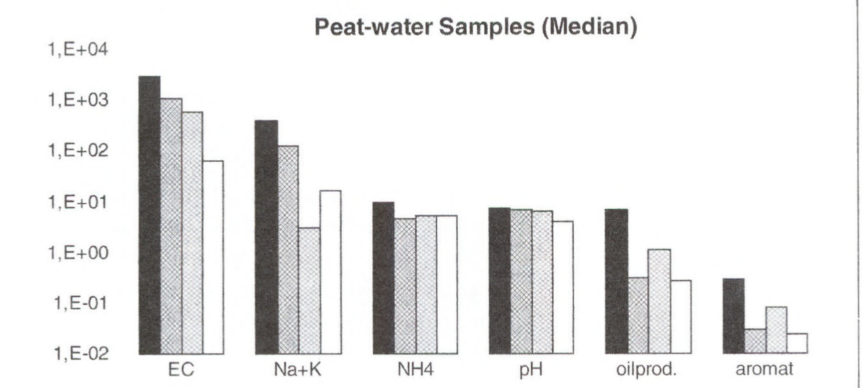


Fig. 3: Results of biochemical analysis of samples from 'ambar' bottom (5-40 cm below the top of the soil) and reference soil samples (1970).



	EC	Na+K	NH4	pH	oilprod.	aromat
■ max	2980,00	381,80	10,00	7,46	6,69	0,30
▨ 3-6 yr	1073,00	128,00	4,50	6,78	0,31	0,03
▨ 12-24 yr	584,00	3,00	5,30	6,50	1,09	0,08
□ reference	62,00	16,18	5,20	3,91	0,27	0,02

Fig. 4: Selected physical-chemical characteristics and concentrations of total oil products (oil prod.) and total aromatic hydrocarbons (aromat) of peat-water samples (max: max value of all samples; yr: number of years between pollution event and sample year; reference: values measured in samples from unpolluted surface water in the same area). (EC: ΦS cm^{-1}, ions, oil product and aromat: g m^{-3})

Table 2. Base line characteristics of peat bog water. Samples (N=28) are from Vasyugan bog, 200 km SE from study area.

		Mean	SD
Na + K	g m^{-3}	5.5	6.2
NH_4	g m^{-3}	1.1	3.8
pH		4.4	1.0
EC	ΦS cm^{-1}	77	60

oil hydrocarbons. Apparently the breakdown of oil components occurs soon after the pollution event, as the concentrations in the 3-6 year group do not differ much from the reference value.

Peat Soil Samples

The pollution in samples from peat soil is of the same origin as the pollution of the peat water. By cation adsorption the salt concentration (Na+K) decreases more slowly over time than in the peat water and does not reach background values within 25 years (Fig. 5).

The concentrations of oil components in the peat soil samples are generally higher than in the peat water, which can be attributed to absorption by the organic material (peat). The breakdown of 'oil product' (mostly alkanes) is much slower than of the aromatic compounds. The reason for this might be that oxygen is only available in the thin top-layer of the peat. Deeper within the peat and inside of the peat structural elements, the environment is anoxic. There, the breakdown of aromatics (especially simple ones) is possible (Cozzarelli *et al.*, 1990). Aromatics were present in only 15% of the peat samples and alkylbenzenes, for example, were detected only in peat samples polluted less then 4 year before sampling.

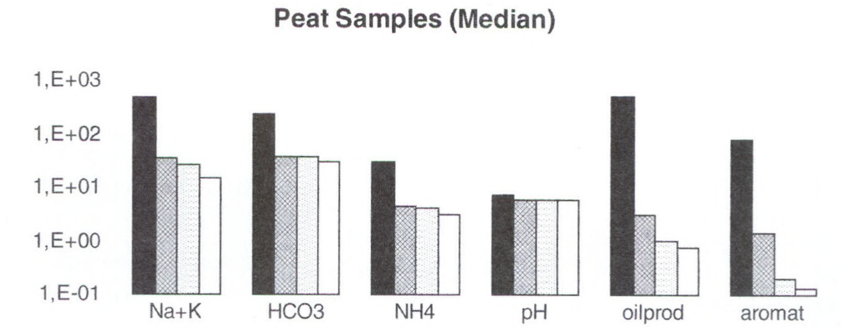
Vegetation

Based on the vegetation surveys made on the polluted sites and unpolluted surrounding areas indications of vegetation recovery could be established. Table 3 gives a compilation of the surveys of oligotrophic bogs related to ecosystem type and pollution age. Species diversity appears to increase afte pollution at ridge and hollow parts of bog complexes, as a result of the change in nutrient availability In all cases the number and abundance of typica bog plant species decreases after pollution with a replacement by some eutrophic indicator species. Indications of recovery occur after long periods (survey cluster No. 7).

Vegetation recovery after flooding with drilling solution, mixed with oil, is presented schematicall in Fig. 6. Depending on the level of vegetatio damage, time-dependent vegetation stages have bee determined.

As a result of flooding with mixtures of oil an drill mud, the peat vegetation close to the origin (am bar) of the pollution plume is completely covere

Fig. 5: Selected physical-chemical characteristics and concentrations of total oil products (oil prod.) and total aromatic hydrocarbons (aromat) of peat samples (max: max value of all samples; yr = number of years between pollution event and sample year; reference: values measured in samples from unpolluted peat bog in the same area). (ions, oil product and aromat: g.m^{-3})



	Na+K	HCO3	NH4	pH	oilprod	aromat
max	504,8	244,0	30,0	7,3	522,0	81,6
3-6 yr	35,5	38,1	4,5	6,0	3,3	1,5
12-24 yr	27,8	38,1	4,3	5,9	1,0	0,2
reference	14,8	30,5	3,3	5,9	0,8	0,1

and compressed (lower right corner in Fig. 6). Here all plants are dead. Depending on the degree of flooding, the surface covered with oil and mud occupies 0.1-0.5 hectares. Vegetation recovery in such places is hampered by the lack of oxygen below the oil layer. Vegetation recovery starts with colonization by the few species able to root in this kind of environment. The raised nutrient availability (Fig. 5) compared to the nutrient poor peat bog environment (Table 2) supports the growth of these species.

Within 5 years after pollution the first patches of *Juncus* species appear. After 10-15 years the soil is covered with *Equisetum* mixed with *Phragmites, Bulboschoenus* and *Typha*. Later this vegetation changes into sedge communities, dominated by *Carex rostra-*

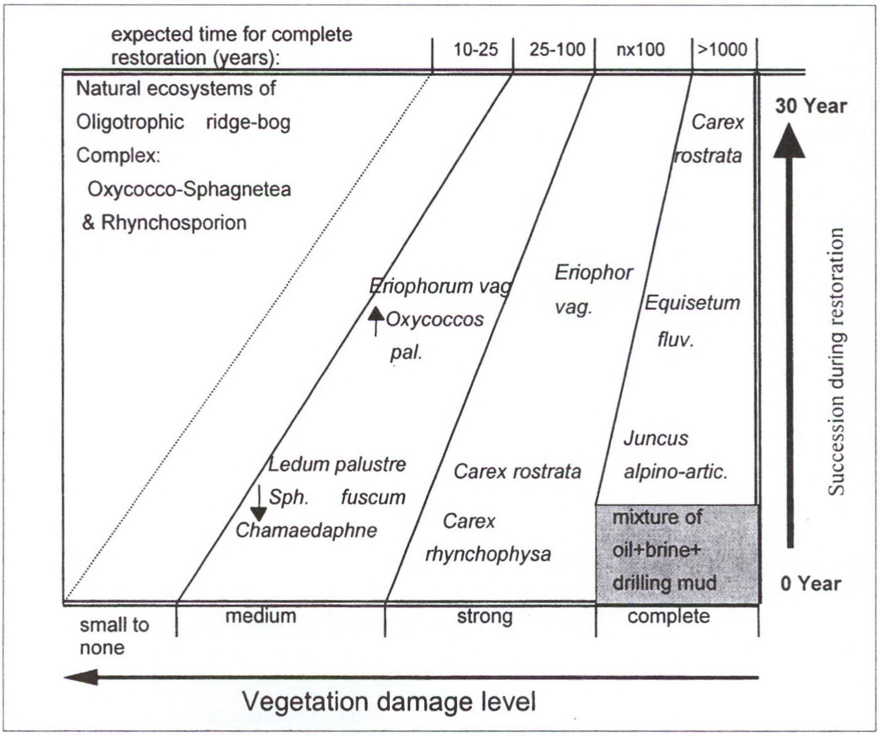


Fig. 6: Vegetation recovery after heavy pollution with a mixture of crude oil, salt process water and drill mud.

Table 3: Vegetation surveys of 4 types of ecosystems from oligotrophic complex bogs at different stages of restoration. Only phyto-coenological important species are presented. No. 1: ridge site, undisturbed; No. 2: ridge site after oil-mineral pollution; No. 3: hollow site, undisturbed; No. 4: hollow site, after oil-mineral pollution; No. 5: flooded with mixtures of oil and drill solution, early stage of recovery (5-10 years after pollution); No. 6: flooded with mixtures of oil and drill solution, middle stage of recovery (20-25 years after pollution); No. 7: flooded with mixtures of oil and drill solution, later stage of recovery (5-10 years after pollution); No. 8: Overgrowing of 'ambars' on peat bogs (age of ambar: 1-5 years); No. 9: Overgrowing of 'ambars' in paludified forest (age of ambar 11-22 years); +: present in less than 10% of the surveys; I: in 11-29%; II: in 21-40%; III: in 41-60%; IV: in 61-80%; V: in 81-100%.

No.	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**
Number of surveys	**6**	**10**	**8**	**6**	**8**	**6**	**5**	**24**	**12**
Number of species	**23**	**54**	**13**	**24**	**20**	**41**	**54**	**13**	**90**
Pinus sylvestris	V	V	.	.	.	.	.	.	.
Oxycoccus microcarpus	V	V	.	.	.	.	.	.	.
Sphagnum fuscum	V	IV	.	.	.	.	.	.	.
Rubus chamaemorus	V	IV	.	.	.	.	.	.	.
Chamaedaphne calyculata	V	IV	.	.	.	.	III	.	.
Cladina sp.	V	IV	.	.	.	.	.	.	.
Cladonia sp.	V	III	.	.	.	.	.	.	.
Ledum palustre	V	II	.	.	.	.	.	.	.
Drosera rotundifolia	V	II	I	I	.	.	III	.	.
Mylia anomala	V	II	.	.	.	.	.	.	.
Pleurozium schreberi	V	II	.	.	.	.	III	.	I
Sphagnum angustifolium	V	II	.	.	.	.	.	.	.
Vaccinium uliginosum	V	I	.	.	.	.	III	.	.
Dicranum bergeri	V	I	.	.	.	.	.	.	.
Sphagnum magellanicum	IV	I	.	.	.	.	.	.	.
Polytrichum strictum	III	III	.	.	.	.	.	.	.
Dicranum polysetum	III	II	.	.	.	.	.	.	.
Sphagnum balticum	II	I	V	I	.	.	.	.	.
Drosera anglica	.	.	V	I	.	.	.	.	.
Scheuchzeria palustris	.	.	V	II	.	.	.	.	.
Carex limosa	.	.	V	II	.	.	III	.	.
Sphagnum jensenii	.	.	V	III	.	.	.	.	.
Sphagnum lindbergii	.	.	IV	I	.	.	.	.	.
Rhynchospora alba	.	.	I	II	.	.	.	.	.
Eriophorum russeolum	.	.	I	IV	.	.	.	.	.
Sphagnum papillosum	.	.	III	IV	.	.	.	.	.
Eriophorum vaginatum	IV	V	II	IV	.	.	V	.	.
Andromeda polifolia	IV	V	III	III	.	.	.	.	.
Oxycoccos palustris	.	V	III	II	.	.	III	.	.
Typha latifolia	.	V	.	IV	V	II	III	IV	V
Epilobium palustre	.	V	.	IV	III	II	V	I	V
Bryum creberrimum	.	III	.	V	IV	I	III	.	V
Agrostis gigantea	.	III	.	.	IV	II	V	II	III
Marchantia polymorpha	.	+	.	+	.	II	V	.	II
Ceratodon purpureus	.	V	.	V	.	+	.	.	IV
Agrostis clavata	.	III	.	I	+	.	III	.	I
Juncus alpino-articulatus	.	III	.	V	V	.	.	II	I
Bidens radiata	.	II	.	III	IV	.	.	I	III
Typha laxmannii	.	I	.	.	II	I	.	III	I
Phragmites australis	.	I	.	.	.	.	III	III	II
Funaria hygrometrica	.	I	.	.	IV	.	.	.	II
Epilobium adenocaulon	.	IV	.	.	II	.	.	+	II
Juncus bufonius	.	III	.	IV	V	.	.	III	.
Blasia pusilla	.	.	.	I	IV	.	.	.	.
Equisetum fluviatile	.	.	.	.	.	III	V	.	V
Drepanocladus aduncus	.	.	.	.	.	II	V	.	V
Warnstorfia pseudostraminea	.	.	.	.	.	II	V	.	IV
Salix cinerea	.	.	.	.	.	I	V	.	II
Salix pentandra	.	I	.	.	.	II	V	.	II
Carex rostrata	.	.	.	.	.	II	V	.	II
Campylium stellatum	.	.	.	.	.	II	V	.	I
Populus tremula	.	I	.	.	.	I	V	.	I
Poa palustris	.	.	.	.	.	+	V	.	II
Leptobryum pyriforme	.	IV	.	.	.	+	V	.	IV
Equisetum palustre	.	.	.	.	.	+	V	.	III
Aneura pinguis	.	.	.	.	.	.	V	.	III
Helodium blandowii	.	.	.	.	.	.	V	.	II
Aulacomnium palustre	.	II	.	III	.	.	V	.	I
Betula pubescens	.	III	.	.	.	.	V	.	I
Dactylorhiza hebridensis	.	.	.	.	.	.	V	.	.
Sphagnum squarrosum	.	.	.	.	.	.	III	.	.
Hypnum lindbergii	.	.	.	.	.	.	III	.	.
Hypnum pratense	.	.	.	.	.	.	III	.	.
Menyanthes trifoliata	.	.	.	.	.	.	III	.	.
Juncus filiformis	.	.	.	.	.	.	III	.	.
Petasites frigidus	.	.	.	.	.	.	III	.	.
Salix pyrolifolia	.	.	.	.	.	+	III	.	I
Brachythecium mildeanum	.	.	.	.	.	.	III	.	I
Cirsium setosum	.	.	.	.	.	.	III	.	I
Carex vesicaria	.	.	.	.	.	+	III	.	II
Calliergon cordifolium	.	.	.	.	.	.	III	.	II
Campylium polygamum	.	.	.	.	.	+	III	.	IV
Brachythecium salebrosum	.	.	.	.	.	I	III	.	IV
Amblystegium serpens/juratzka	.	.	.	.	.	I	III	.	III
Carex canescens	.	.	.	.	.	II	III	.	III
Bryum pseudotriquetrum	.	.	.	.	.	I	III	.	III
Campylium sommerfeltii	.	.	.	.	.	.	III	.	III
Lemna minor	.	.	.	.	.	.	.	.	V
Carex rhynchophysa	.	.	.	.	.	.	.	.	III
Calamagrostis langsdorffii	.	.	.	.	.	.	.	.	III
Rorippa palustris	.	I	.	.	.	.	.	.	III
Carex loliacea	.	.	.	.	.	.	.	.	III
Calamagrostis phragmitoides	.	III	.	.	.	.	.	.	I
Pinus sibirica	II	I	.	.	.	.	III	.	.
Pohlia nutans	.	III	.	.	.	+	.	.	II
Salix caprea	.	I	.	.	.	+	.	.	I

ta. Peat development starts again after 25-30 years as is shown by the presence of peat accumulating species (*Chamaedaphne calyculata, Vaccinium uliginosum, Oxycoccus palustris, Drosera rotundifolia*). Based on palaeo-botanical studies in the area, it was expected that complete recovery of the original vegetation would require a period of more then 500 years (Lapshina & Bleuten, 1998).

Around or downstream from the completely destroyed ecosystems, a larger area (0.2-1.0 ha.) is partially damaged by the flow of oil and mud water through the plant hummocks ('strong damage' in Fig. 6). Starting with strong damage the vegetation re-develops through various successional stages. Within a few years tall-sedge communities (*Carex rhynchophysa, Carex rostrata*) appear. This is followed in 10-20 years by *Eriophorum vaginatum*.

It was noticed that regardless of the pollution type (crude oil, oil-mineral, or mineral), low to moderate levels of pollution positively influenced the development of certain species. *Andromeda polifolia, Oxycoccus palustris* and *Eriophorum vaginatum* increase in abundance and quite often can dominate. Oligotrophic dwarf shrub species (*Ledum palustre, Chamaedaphne calyculata*) and bog mosses (*Sphagnum fuscum, S. magellanicum*) respond to the stressed conditions of oil pollution by a decrease in their abundance. *Betula nana, Rubus chamaemorus* and *Oxycoccus microcarpus* react rather indifferently.

In places heavily flooded with salt production water (brine) the vegetation, except for some lichen species, disappears completely. After 2-3 years vegetation recovery starts in the lower sites (hollows) with the growth of algae. The high salt concentration must be diluted before other plants will re-establish. This process takes about 10 years. The salt concentrations are by then still raised compared with the background level but are obviously below the critical levels for most plants.

CONCLUSIONS

Data on the quantities of crude oil dumped in the ambars were not available, therefore the time-dependent decrease in concentration by breakdown processes could not be established completely. In addition, as a result of logistical problems, the number of samples was too limited for profound statistical analyses. Nevertheless, some conclusions can be made.

The total concentration of 'oil product' in water samples taken from dump reservoirs (ambars) shows a clear relation to the date of pollution. The 'oil product' concentration decreases to the reference level within a period of 20 years in this open water environment rich in oxygen. Aromatic hydrocarbon concentrations do not decrease significantly during this time. The constancy of aromatic concentrations confirms that they are more difficult for bacteria to break down.

The concentrations of oil components (alkanes and aromatics) in ambar sub-aquatic reservoir soil did not decrease significantly within the whole 30 year period. It is believed that drill mud (Bentonite clay) in the ambar encloses the oil and creates anoxic conditions.

In peat soils polluted with oil, the aromatic hydrocarbons disappear within a few years and 'oil product' concentrations reach background levels in about 10 years.

In general, vegetation recovery is much slower than the degradation of oil components. Within the 30-year period after complete burial with oil and drill mud layers, recovery of the original bog vegetation had just started. Complete recovery by succession is expected to last many hundreds of years.

Ridge sites of ryam vegetation appear to be more sensitive compared to hollow and mire sites. Although the lower hollow parts degrade to a greater extent, they recover faster than the higher ridges.

The damage caused by the overflow of salt production water (brine) is, except for some lichens, complete. Although the salt content of the peat-water decreased almost to background levels within 10 years, the restoration of the vegetation in these areas takes several tens of years.

The use of reservoirs (ambar) for storage of drill mud and oil may be acceptable if reliable dykes are constructed around them. The pollution will remain in the ambar-soil for an undefined time, but when enclosed by drill clay layers is immobile and will not affect the surrounding areas. The alternative, that is, allowing the contaminants to spread over the peat bog, results in extreme degradation of the peat ecosystems over large areas and requires very long recovery times.

By monitoring over several decades a selection of abandoned exploration sites, which are not located near pipelines for the transportation of oil or brines, knowledge of the biochemical degradation processes and ecosystem recovery can be enhanced substantially. The remoteness of these sites assures the reliability for such research.

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EFFECTS OF FERTILIZATION AND REMOVAL OF OVERSTOREY ON FOLIAR NUTRIENT STATUS AND CHLOROPHYLL FLUORESCENCE OF NORWAY SPRUCE [*PICEA ABIES* (L.) KARST.] UNDERWOOD ON DRAINED PEATLANDS

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SUMMARY

The effect of an overstorey cutting with or without simultaneous PK-fertilization on Norway spruce underwood on a drainage area was investigated. The nutrient status of the spruce was studied by analysing the nutrient and free polyamine concentrations of the needles. A possible effect of light and nutrient conditions on photosynthetic efficiency was studied by measuring the ratio of variable to maximum chlorophyll *a* fluorescence (Fv/Fm) in the needles during the growing season. The fertilization increased only potassium concentrations, whereas the release cutting increased nitrogen concentrations in the needles. Although the PK-fertilization, coincident with the removal of overstorey, improved the nutrient status of the trees after the release, it did not improve the adaptation of the trees to increased light intensity. According to this, and the better recovery of fertilized, suppressed spruce, the best time for PK-fertilization might be at least one year before cutting.

Keywords: nutrition, polyamines, chlorophyll fluorescense, spruce underwood, drained peatlands.

INTRODUCTION

In several studies of shade-grown trees, Norway spruce (*Picea abies*) has been found to be quite adaptable to new conditions after the removal of overstorey (Cajander, 1934; Katrusenko, 1965; Skoklefald, 1967; Bergan, 1971; Andersson, 1988; Koistinen & Valkonen, 1993). There are also opposing observations, according to which spruce has suffered severe damage (Ståfelt, 1935; Andersson, 1984). Changes in soil moisture (Heikurainen & Päivänen, 1970; Päivänen, 1974; 1982), air humidity and temperature conditions (Multamäki, 1942; Perttu, 1974; Leikola, 1975; Leikola & Rikala, 1983) after cutting may affect the adaptation of spruce. Many of the chlorotic symptoms after the release may be explained by environmental factors other than the increased illumination (Katrusenko, 1965; Gnojek, 1992; Seryakov, 1994). Differences in the viability, height and earlier growth of spruce as well as weather conditions before and after the release have a strong influence on the survival and growth reaction of spruce (Cajander, 1934; Koistinen & Valkonen, 1993).

The photosynthetic efficiency of the needles may be reduced because of photoinhibition after the highly increased illumination. Water stress and lack of some mineral nutrients such as potassium, magnesium and zinc can make plants more susceptible to photoinhibition and chlorophyll bleaching (Laatsch & Zech, 1967; Björkman & Powles, 1984; Marschner & Cakmak, 1988). Chlorophyll *a* fluorescence has been widely used as a probe to assess the physiological state and intactness of the photosynthetic system (Lichtenthaler & Rinderle, 1988; Bolhar-Nordenkampf *et al.,* 1989; van Kooten & Snel, 1990; Krause & Weis, 1991). The ratio of variable to maximum fluorescence (Fv/Fm) is widely used as an indicator of photoinhibition (Öquist & Wass, 1988). Potassium deficiency has been shown to decrease the photosynthetic rates of Norway spruce (Baillon *et al.,* 1988; Lütz *et al.*, 1992) and some effects of magnesium and potassium deficiency on chlorophyll *a* fluorescence in spruce have been reported (Baillon *et al.,* 1988).

External stress has been shown to affect endogenous polyamine levels in plants (Flores, 1991), which has been considered an adaptive and protective response (Galston & Kaur-Sawhney, 1990). The most common polyamines in plants are putrescine, spermidine and spermine. Potassium deficiency has been reported to cause accumulation of putrescine in Scots pine (*Pinus sylvestris*) and Norway spruce needles (Sarjala & Kaunisto, 1993; 1996; Kaunisto & Sarjala, 1997). The homeostatic effect of poly-amines, i.e. the maintenance of cellular pH in plants, is one of the proposed mechanisms in the action of polyamines under mineral deficiencies (Altman & Levin, 1993).

The increased illumination as well as other environmental changes after the removal of overstorey may affect foliar nutrient concentrations. In earlier studies of Norway spruce, nitrogen and phosphorus concentrations have been shown to increase and potassium concentrations decrease (Katrusenko, 1967; Koshelkov, 1982; Saarinen, 1996) after release cutting.

A shortage of mineral nutrients has been reported in trees in some old drainage areas (Kaunisto, 1989; Kaunisto & Paavilainen, 1988). On those peatlands the main nutrient imbalance of the trees may become more critical after overstorey cutting. In the earlier study of Saarinen (1996) this problem was investigated in an experiment established in an old drainage area of previously sparsely stocked wet pine mire. In spite of the site type having a potentially unbalanced main nutrient regime, the concentration of potassium was found to be fairly high in the spruce underwood. After removal of overstorey, however, needle potassium concentrations decreased strongly down to the deficiency limit possibly owing to a dilution effect or translocation of potassium to the roots (Saarinen, 1996). For the present study a new experiment was established in the same type of drainage area where, however, the underwood suffered from a strong and visible shortage of potassium. In this case a PK-fertilization treatment was carried out in order to study the effect of increased P and K availability on the recovery of the underwood. The effect of the removal of overstorey on the nutrient status of the trees was studied by analysing the foliar chemical composition. Polyamines were analysed also to reveal the potassium status of the trees during the growing season and dormancy. The ratio of variable to maximum chlorophyll *a* fluorescence (Fv/Fm) in the needles was recorded during the growing season in order to indicate possible effects of the light and nutrient conditions on photosynthetic efficiency after the release cutting and fertilization.

MATERIAL AND METHODS

The experimental site is located in Honkajoki, southern Finland (61° 56' N, 22° 02' E), 60 km northwest of the previous experiment (Saarinen, 1996) in Kiikoinen. According to the Finnish mire classification by Heikurainen and Pakarinen (1982), the site was originally a wet and sparsely stocked herb-rich sedge birch-pine swamp. Drainage was carried out in the 1950s. The stock before cutting consisted mainly of pine (*Pinus sylvestris*) and downy birch (*Betula pubescens*) with mean height of 17 m and stand volume of 83 m^3 ha^{-1} . This mixed stand suppressed a relatively even-aged Norway spruce (*Picea abies*) underwood, the height of which varied from 4 to 6 metres.

The experimental area was composed of three adjacent blocks one of which was a mesotrophic site and the others meso-oligotrophic sites. The whole area was ditched using 40-metre spacings. Each of the three blocks consisted of two adjacent strips. In winter 1995 the mixed pine-birch overstorey was removed from one of the strips, whereas the other one remained as a control. Both of these strips were divided into two 30-40-meter sub-plots for a fertilization treatment. This two level treatment (control and fertilization with 45 kg ha^{-1} of phosphorus and 80 kg ha^{-1} of potassium) took place in May 1995. At the same time all ditches were cleaned with an excavator.

Needle samples from ten randomly selected spruces from each plot were taken for analysis. Needles were collected by cutting one branch from the third branch whorl. Last three needle year classes were separated later in the laboratory. Shoots of each sample spruce on each plot were combined to represent three different needle year classes. Needles were collected in December 1994 before the overstorey removal (March 1995) and in three years after it (1995–1997).The same trees were sampled every year. Analysis of foliar potassium was performed by flame atomic spectrophotometry (Varian AA-30) after HCl extraction of dried and ground material. Phosphorus was determined spectrophotometrically in dry ashed material. Nitrogen was determined with the Kjeldahl method (Halonen *et al.,* 1983). Also the dry mass of one hundred needles and the height of the sample spruces were measured annually. The height was measured also in 1998 after the end of the fourth growth period after the release.

Five of the ten sample trees were used for collecting needles for polyamine analyses and fluorescence measurements every July and September in the three year period (1996-1998) after the removal. The needles were kept in a deep freeze (-80°C) until analysed. The spruce twigs for chlorophyll fluorescence measurements were kept in a cold room (+6°C) overnight before measurement. Free polyamines (putrescine, spermidine and spermine) were analysed with HPLC as described by Sarjala & Kaunisto (1993).

The chlorophyll fluorescence measurements from spruce needles were performed with a Plant Efficiency Analyser (Hansatech Instruments Ltd., Norfolk, England). The needles were dark adapted in a dim room at a temperature of +20°C in leafclips supplied by the analyser manufacturer for at least 30 min before measurement. The fluorescence parameters Fo (initial fluorescence level) and Fm (maximum fluorescence level) were measured and the ratio Fv/Fm (where Fv = Fm - Fo) was determined with settings of 15 seconds at 100% intensity level of photon flux density (4000 mol $m^{-2} s^{-1}$) during the measurement.

Multivariate analyses of variance with repeated measures (BMDP 4V) were used for testing the parameters of the following two models:

$$Y_1 = R + F + N + T + T^*R + (TR)^*F + (TR)^*N + \varepsilon$$

$$Y_2 = R + F + T + T^*R + (TR)^*F + \varepsilon$$

Y_1 = foliar concentrations of N, P and K and foliar dry mass.
Y_2 = chlorophyll fluorescence, foliar concentrations of polyamines and annual height increment.
R = release cutting, F= fertilization, N = needle year class, T = growth period, ε = error.

The design had two grouping factors (R and F) and two repeated measure factors (N and T). Contrasts across time were used for testing the effect of the release on the nutrient concentrations and the dry mass of the current needles and annual height increment during each separate growth period after the cuttings. These contrasts were used also for testing the same effect on the total amounts of nutrients per one hundred needles (mg per dry mass).

The release and fertilization effects on the foliar nutrient concentrations and dry mass were tested in the form of the following four hypotheses and the same effects on the chlorophyll fluorescense, polyamine concentrations and height increment in the form of the first three:

1. Changes in the follow-up period (time effect; T) on the released plots differ from those on the control ones (release effect; R). The release-by-time interaction [T*R] is significant.
2. Changes in the follow-up period on the fertilized plots (fertilization effect; F) differ from those on the non-fertilized ones. The fertilization-by-time interaction [T*F] is significant.
3. The release-by-time interaction [T*R] depends on the fertilization (F) treatment. Interaction between the fertilization, time and release effects [(TR)*F] is significant.
4. Changes in the time after the release depend on the needle year classes. Interaction between the needle (N), time and release effects [(TR)*N] is significant.

RESULTS

Height Increment and Main Nutrient Concentrations

The PK-fertilization had a slight but statistically significant effect on the height increment of underwood during the same follow-up period (1994-1997) when the nutrient status was measured (Tables 1 and 3). Fertilized and suppressed spruces increased their height increment 32% during this period, whereas there were no increases of height increment on the fertilized and released plots (Fig. 1). At the same time, however, those suppressed spruces without fertilization decreased their height increment. There was also a slight decrease in height increment in the released and non-fertilized spruces until they started to recover during the third year after the release. The removal of overstorey had no statistically significant effect on the height increment during this three year period after cutting. Including also the year 1998 (fourth after the release) into the follow-up period, the release effect (release-by-time interaction) was statistically significant. This was caused by the onset of recovery of height increment in the released spruces (Fig. 1).

Nitrogen concentrations increased strongly during the growth periods after the release (Fig. 2 , Table 2). They were between 14.3-14.9 mg g^{-1} before the release cuttings and increased up to 17.5 mg g^{-1} after the release. There was only a slight decrease during the following years. Phosphorus increased slightly but not significantly during the first growth period. Potassium concentrations were only 3–4 mg g^{-1} before the release. The following years showed a slight decrease in the suppressed control (non-fertilized) plots, whereas on the released plots a slight increase took place. There was no similar difference on the fertilized plots. PK-

Table 1: Results of Greenhouse-Geisser tests of three hypotheses explained in text. F-, p- and adjusted df-values of height increment, concentrations of foliar nitrogen (N), phosphorus (P), potassium (K), and polyamines, 100 needle dry mass (DM), and chlorophyll fluorescence.

	Hypothesis (interaction)											
	1 Time*Release			2 Time*Fertilization			3 (TR)*Fertilization			4 (TR)*Needle		
	F	p	df	F	p	df	F	p	df	F	p	df
Height increment (1994 - 1997)	1.0	0.4170	2.39	3.8	0.0351	2.39	1.9	0.1768	2.39	—	—	—
Main nutrient concentrations in three needle year classes:												
N mg g^{-1}	21.2	0.0001	1.68	1.6	0.2398	1.68	0.1	0.9288	1.68	1.7	0.1853	3.16
P mg g^{-1}	0.9	0.4341	1.94	1.3	0.3031	1.94	2.8	0.0911	1.94	2.1	0.1497	2.17
K mg g^{-1}	5.6	0.0077	2.58	30.0	0.0000	2.58	0.5	0.6407	2.58	1.6	0.2266	2.37
Needle dry mass												
DM	3.2	0.0688	2.05	0.8	0.4623	2.05	1.3	0.2958	2.05	1.1	0.3579	2.30
Chlorophyll fluorescence in current needles (Fv/Fm):												
July	8.5	0.0080	1.48	1.0	0.3693	1.48	0.1	0.8304	1.48	—	—	—
September	1.9	0.1973	1.82	0.6	0.5344	1.82	0.7	0.5114	1.82	—	—	—
Polyamine concentrations in current needles (nmol g^{-1} FW):												
Putrescine	0.8	0.4727	1.82	3.8	0.0499	1.82	1.8	0.2004	1.82	—	—	—
Spermidine	2.2	0.0928	3.85	4.3	0.0079	3.82	1.5	0.2241	3.85	—	—	—
Spermine	1.5	0.2603	2.26	1.7	0.2186	2.26	0.7	0.5380	2.26	—	—	—

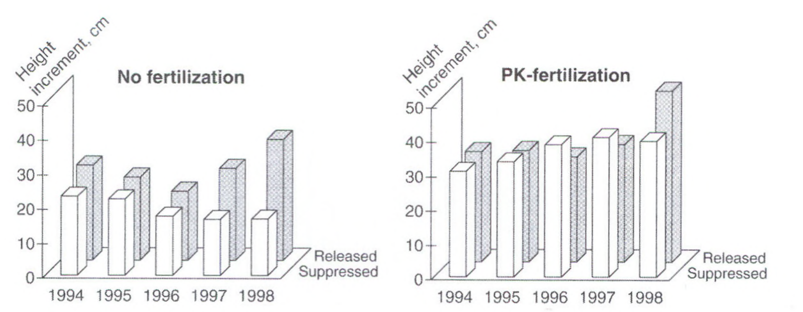

Fig. 1: Annual height increment of sample spruces.

fertilization had an effect only on the potassium concentration (Table 1). The concentration increased to 5.6-6.7 mg g^{-1}.

All these changes as a result of the release (hypothesis 1: interaction between the time and release treatments) were significant only in the case of nitrogen and potassium (Table 1). The fertilization effect was significant only for potassium (Table 1 and 3) . Interaction between the fertilization, release and time [(TR)*F] was not significant (Table 1), which means that the fertilization effect was not dependent on the effect of release across time. It is important to notice that the effect of the release and fertilization on the dry mass of needles was not significant either (Table 1). The release effects across time on the total amounts of nutrients per one hundred needles (mg per dry mass) were nearly the same as described above.

Free Polyamine Concentrations

The needle putrescine concentrations in individual trees varied from 50 nmol g^{-1} FW to over 5000 nmol g^{-1} FW indicating that the potassium nutrition of the trees varied from a satisfactory one to a severe deficiency. The spermidine concentrations varied from 30 to 260 nmol g^{-1} FW and spermine from 10 to 122 nmol g^{-1} FW.

Putrescine levels in the needles were highest in December and lowest in July (Fig. 3). Before the fertilization treatments in May 1995 the average putrescine levels indicated an unsatisfactory potassium nutrition in all the plots. The PK-fertilization caused a statistically significant effect on the pattern of fluctuation in putrescine by decreasing the concentrations during the dormancy (fertilization-by-time interaction was significant; Table 1). The release cut-

Table 2: Time contrasts of the release effect (F- and p- values of the statistical tests) concerning height increment, foliar nutrient concentrations and needle dry mass in sample spruces.

	1994 vs the years after release cuttings								
	94 vs 95			94 vs 96			94 vs 97		
	F	p	df	F	p	df	F	p	df
Height increment	1.1	0.317	1	2.1	0.184	1	0.1	0.723	1
Concentrations in current needles:									
N mg g^{-1}	11.3	0.010	1	55.0	0.000	1	59.6	0.000	1
P mg g^{-1}	1.1	0.327	1	0.0	0.956	1	0.2	0.712	1
K mg g^{-1}	4.4	0.068	1	9.5	0.015	1	14.9	0.005	1
Needle dry mass:									
DM	1.4	0.276	1	7.4	0.026	1	8.4	0.020	1

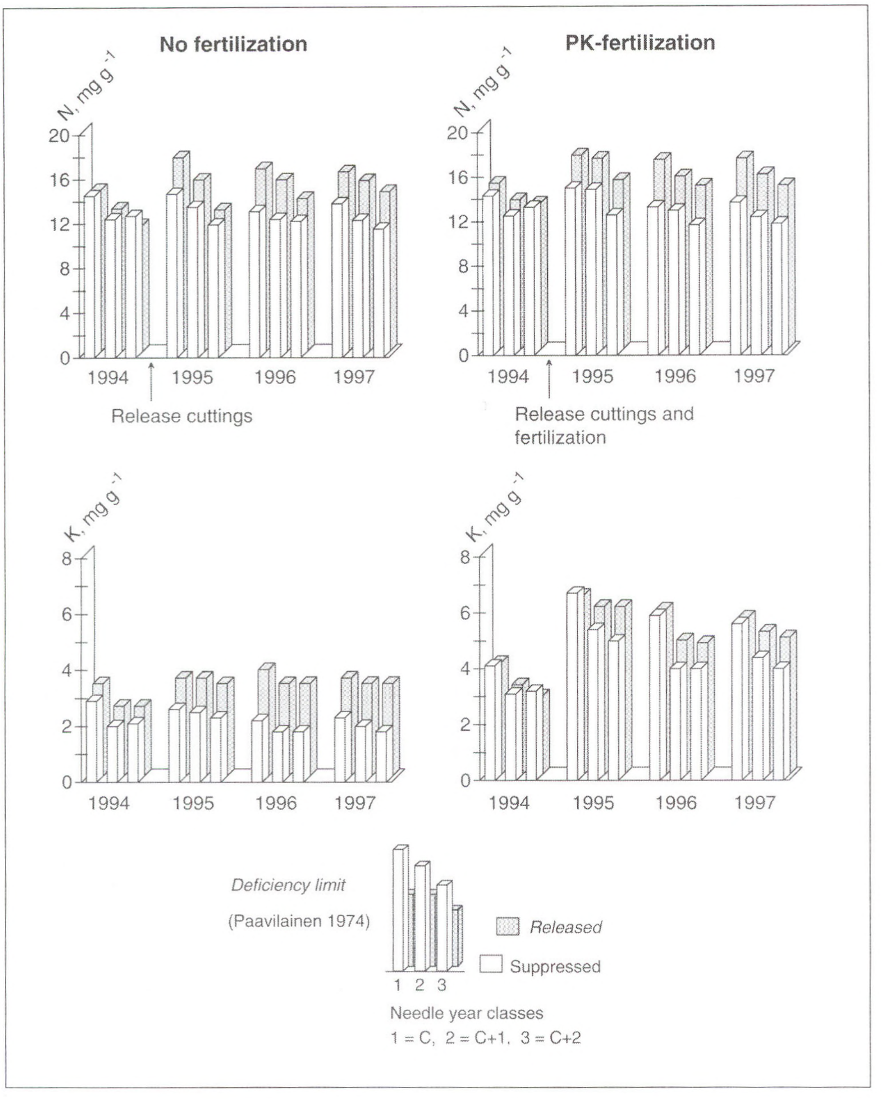

Fig. 2: Foliar nitrogen (N), and potassium (K) concentrations in three needle year classes of sample spruces.

ting had a significant decreasing effect on putrescine concentrations only in the non-fertilized plots during the dormancy.

The variation in the spermidine concentrations between the dormant and growing seasons was not as big as for putrescine (Fig. 3) . The fertilization, but not the release cutting, had a statistically significant effect on the pattern of fluctuation in spermidine. In the suppressed plots higher spermidine concentrations in the fertilized trees than in the non-fertilized ones were found in December and September, but not during the growing season in July. In fact, the fertilized suppressed trees had lower spermidine levels than the other trees during the growing season. They did not show any particular fluctuation between the growing seasons and dormancy although an increasing trend in spermidine levels could be seen during the three subsequent growing seasons.

The spermine concentrations were highest in December and September and lowest in July following mainly the pattern of the putrescine concentrations (Fig. 3). Statistically significant differences were not found between the treatments.

Chlorophyll Fluorescence

The Fv/Fm values that indicated photochemical efficiency were slightly lower on released plots (Fig. 4) There were no measurements, however, of chlorophyll fluorescence before release cuttings in summer 1994. Thus it is impossible to say for sure whether the difference can be ascribed to cutting

Table 3: Time contrasts of the fertilization effect (F- and p- values of the statistical tests) concerning height increment, foliar nutrient concentrations and needle dry mass in sample spruces.

	1994 vs the years after release cuttings								
	94 vs 95			94 vs 96			94 vs 97		
	F	p	df	F	p	df	F	p	df
Height increment	1.7	0.228	1	6.2	0.038	1	36.4	0.000	1
Concentrations in current needles:									
N mg g^{-1}	0.9	0.383	1	0.4	0.533	1	0.2	0.638	1
P mg g^{-1}	0.0	0.967	1	0.6	0.479	1	2.1	0.185	1
K mg g^{-1}	72.0	0.000	1	23.2	0.001	1	50.9	0.000	1
Needle dry mass:									
DM	0.1	0.784	1	2.5	0.154	1	1.7	0.231	1

Nevertheless the release-by-time interaction after release cuttings was significant (Table 1). Lower values in released ones could be seen both in the non-fertilized and fertilized trees and no differences were found as a result of fertilization. The average levels of Fv/Fm in the suppressed spruce trees were above 0.80 and between 0.75-0.80 in the released ones. In September the Fv/Fm levels were higher compared with the values in July and the difference between the released and suppressed spruces was significant only in July.

DISCUSSION

The foliar nutrient composition before the release cutting indicated satisfactory nitrogen and phosphorus nutrition, whereas the potassium concentrations were below the deficiency limits (Paavilainen, 1974, Reinikainen *et al.,* 1998).

Improvement of the nitrogen status of released spruces was similar to that in the Kiikoinen experiment (Saarinen, 1996). It may be related to a decreased competition for nutrients and increased mineralization in warmer soils. Also according to some Russian studies the increase in the foliar nitrogen concentrations is related to released soluble nitrogen in the soil and to an improved capability of nutrient uptake due to the strongly developed root system (Koshelkov, 1982). These studies indicated that the temporary increase of both nitrogen and phosphorus was supposedly related to the internal nutrient translocation of trees. A corresponding translocation from one and two year old shoots (C+1 and C+2) to the current (C) ones could not be discerned in this experiment. Concurrent with the increase of the concentrations in the youngest needles (current), an improvement in the nutrient status in two older needle classes was also found. It is, however, possible that nitrogen and phosphorus have been translocated from shoots over two years old (C+3 or older).

When comparing the release effect on the potassium concentrations in the Kiikoinen experiment (Saarinen, 1996) an opposite reaction was found in the present study. The concentrations increased slighty in contrast to the strong decrease in Kiikoinen. Also the reaction in height increment of the underwood differed from the previous experiment. In this study the spruces did not show a similar strong recovery after the release and it started later than in Kiikoinen. The difference between these two experiments before the release cutting was the potassium concentration of the underwood. In the Kiikoinen study this was much higher and above the deficiency limit. In the present study the spruce trees suffered from a severe and visible shortage of potassium. Thus it is obvious, that owing to poor recovery after the release, a dilution effect or potassium translocation to the roots, similar to that in the Kiikoinen experiment, has not taken place yet. The needle nutrient and polyamine concentrations showed that the fertilization effect on the potassium status was very strong both

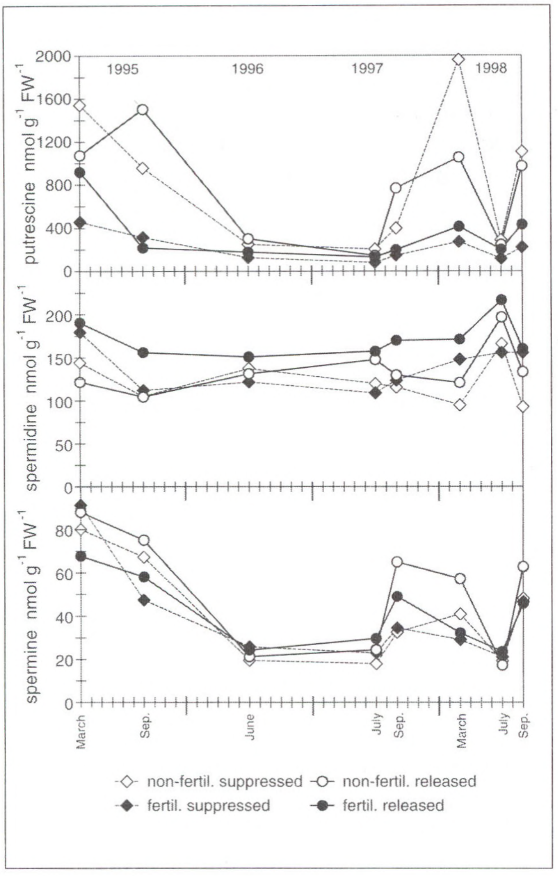

Fig. 3: Free polyamine concentrations in current needles of sample spruces.

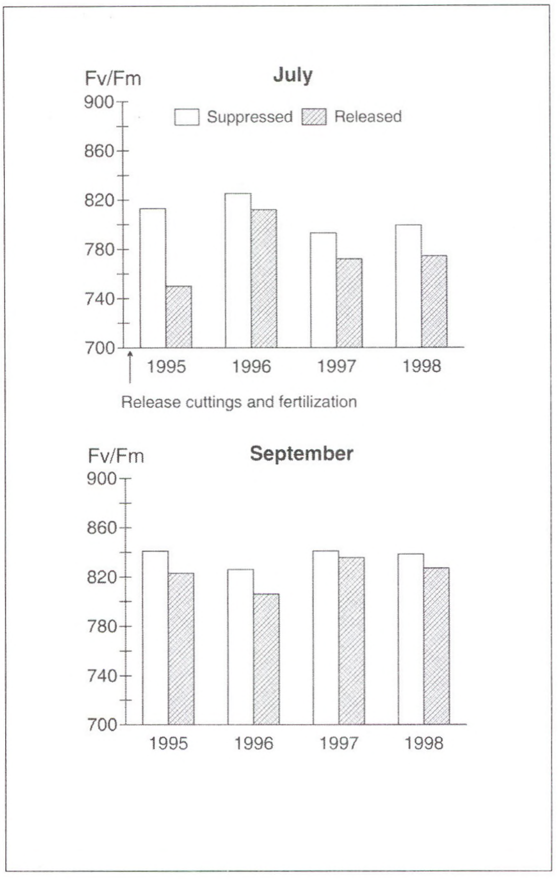

Fig. 4: Annual chlorophyll a *fluorescence (Fv/Fm) in current needles of sample spruces in July and September.*

in the released and suppressed trees immediately after the release cutting during the following growth period. The putrescine concentrations showed that the removal of the overstorey did not much affect the potassium nutrition except that significantly lower putrescine levels were found on non-fertilized, released plots than on non-fertilized, suppressed ones. The accumulation of putrescine in potassium deficient needles was greater during dormancy than in the growing season which is in line with earlier observations with Scots pine (Sarjala & Kaunisto, 1996).

A more distinct fluctuation in spermidine was observed in the non-fertilized trees compared with the fertilized ones. Spermidine has been shown to affect cell division and growth (Smith, 1985; Evans & Malmberg, 1989) and to correlate positively with the growth rate of Scots pine seedlings (Sarjala, 1996). Interestingly, the fertilization of the suppressed trees in this study increased spermidine only in December and September, but not in July. This may be owing to the more pronounced negative effect of the overstorey on light conditions in July than in December or September. Thus the ability of the suppressed trees to utilize the added nutrients for growth as efficiently as the released trees is decreased. This means that the limiting light conditions did not allow the suppressed trees to utilize the nutrient addition to an optimal degree.

The Fv/Fm values, which indicated photochemical efficiency, decreased slightly after removal of the overstorey both in the non-fertilized and fertilized trees. Decreased Fv/Fm values for released spruces have also been reported by Gnojek (1992). The effect of fertilization on the Fv/Fm values was not significant (Table 1). This suggests that in this study the light conditions rather than the nutrient status of the trees affected the photochemical efficiency and that it could not be improved by fertilization. Improved potassium nutrition of Scots pine seedlings increased the Fv/Fm values under moderate light intensity (Savonen & Sarjala, 1998). The average levels of

Fv/Fm in the suppressed spruce trees were above 0.80 corresponding to the levels measured from Scots pine seedlings under a good potassium nutrition and moderate light intensity (Savonen & Sarjala, 1998). Fv/Fm between 0.75-0.80 in the released spruce trees corresponds to the values measured from potassium deficient Scots pine seedlings under moderate light intensity (Savonen & Sarjala, 1998). The difference in Fv/Fm levels between the suppressed and released trees was significant in July, whereas in September the difference decreased to a non-significant level (Table 1). This is in line with the observations of Savonen & Sarjala (1998) concerning Scots pine seedlings under decreasing light intensity and photoperiod. According to Gnojek (1992) a partial recovery of Fv/Fm in released spruces in September is expected because of the lower photosynthetic photon flux density at the end of the summer.

In conclusion, the present study, together with the earlier results of Saarinen (1996), shows that the response of Norway spruce underwood to release cutting depends on the nutrient status of the trees before the cutting. Application of PK-fertilizer at the time of removal of overstorey improved the nutrient status of the trees after release, although it did not have an effect upon adaptation of the remaining trees to the increased light intensity. According to this, and the better recovery of fertilized and suppressed spruces, the best time for PK-fertilization might be at least one year before release cutting.

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DIVERSITY OF VEGETATION IN PRISTINE AND DRAINED FORESTED MIRE MARGIN COMMUNITIES IN FINLAND

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SUMMARY

Within the Scandinavian boreal landscape, forested mire margin communities (e.g. swamp forests) are natural centres of biodiversity. The change in diversity of vegetation, caused by drainage, of forested mire margin communities (paludified forests, mire margin forests and forested pine and spruce mires) in south and central Finland was examined. Data for the sample plots of the 8th National Forest Inventory (1985-86) were used in this study. The diversity of the vegetation was analysed at community level (b-diversity) with multivariate methods of ordination (NMDS) and classification (TWINSPAN). The diversity at species level (a-diversity) was also analysed by species richness and diversity indices. The main compositional gradient in the NMDS-ordination of the pristine and drained sites was site fertility or trophic gradient, from ombro-oligotrophy to meso-eutrophy. The drainage state of the sites determined the secondary gradient. The third interpretable gradient was related to tree stand succession and peat thickness. In general, the results showed that species diversity was relatively high in mire margin forests. When estimated only by numbers of species, the species diversity remains high after drainage, but the species composition of these communities changes towards forest vegetation dominated by mesic and herb-rich forest species with mire species present only as relicts.

Keywords: forest vegetation, mire vegetation, forested peatland, vegetation diversity, forest drainage.

INTRODUCTION

Forested mire margin communities, in the boreal region, are mosaics that include both upland forest and peatland vegetation (spruce swamp, bog, marsh and spring) (Eurola, 1962; Ruuhijärvi, 1960) that show high species richness and diversity (Korpela & Reinikainen, 1996a,b). Within the boreal landscape in Scandinavia, swamp forests are natural centres of biodiversity (e.g. Ohlsson *et al.*, 1997). In Finland these biotopes have, to a large extent, been drained owing to their potential for wood production (Gustavsen & Päivänen, 1986) and in Sweden they have also been exploited extensively (Ohlsson *et al.*, 1997). According to the data of permanent sample plots of the 8th National Forest Inventory (8th NFI), the area of undrained mire margin sites in 1985-86 was half of what it had been in the early 1950s compared to data from the 3rd NFI (Ilvessalo, 1956, 1957).

In their undrained condition these wetland habitats have remained relatively pristine in their stand structure and ground vegetation, although they have been often subjected to some silvicultural management. Thus on the community and species level they are assumed to be important for local and regional diversity.

Mire margin forests and forested pine and spruce mires are included in so-called genuine forested mire sites, which are characterized by hummock and intermediate level vegetation resembling forest vegetation, especially at the more nutrient-rich end of the vegetation continuum. These sites have often developed from the paludification of forest land. The more nutrient-rich types are called spruce mires, with vegetation characterized by shade-tolerant herbs and dwarf shrubs. The sites with lower nutrient levels are called pine mires with characteristic hummock dwarf shrub vegetation (Vasander, 1996). The vegetation gradient between mire margin and mire expanse communities is one of the main ecological gradients in addition to the gradients of nutrient availability (trophic status) and hydrology, in the ecology and classification of boreal mires in Finland and Sweden (Heikurainen & Pakarinen, 1982, Eurola *et al.*, 1984 Malmer, 1985). A distinction between mire expanse and mire margin vegetation was made in early mire vegetation studies (Sjörs, 1948). According to Finnish environmental characterisation mire margin vegetation occurs often on sites with a thin peat layer that receive a supplementary input of mineral nutrients from surrounding mineral soil and are thus minerotrophic (Eurola *et al.,* 1984). This differs from the Scandinavian interpretation (Sjörs, 1983, Økland, 1989, 1990), according to which marginal pine forests of bogs are included in mire margin sites despite being ombrotrophic. In this study the Finnish definition (Eurola *et al.,* 1984) of the mire margin concept is used. Mire margin vegetation occurs typically in a zone between proper upland vegetation and mire expanse vegetation. The sites include (1) paludified forests (= paludified mineral soil forests), (2) mire margin forests and (3) forested pine and spruce mires (Lumiala, 1937; Tuomikoski, 1942).

There has been a demand for a practical site classification of all drained peatlands in Finland (about 6 million hectares during the years 1950-1990 (Sevola, 1997)), which undergo different stages of post-drainage succession. The system for this classification is based on the gradual decrease of mire plants along the succession following drainage. The criteria for distinguishing the four different drainage phases are: (1) *pristine mire* vegetation, (2) *recently drained phases* with unchanged vegetation, (3) *transforming phase* with a 25-75% proportion of mire species of the original community, and (4) *transformed phase* with less than 25% of mire species left (Sarasto, 1961; Heikurainen & Pakarinen, 1982; Paavilainen & Päivänen, 1995).

According to Cajander (1913), after drainage a mire community will change until it finally resembles a mineral-soil forest community of the corresponding fertility level. This assumption has generally proved to be correct, but present experience shows that special characteristics of the original mire and peat substrate remain (Laine, 1989; Reinikainen, 1990). The time required for this development varies between 15 and 50 years, depending mainly on the fertility, moisture and tree stand of the original mire type (Laine *et al.*, 1995).

Previous studies of the effect of forest drainage on vegetation have concentrated either on sparsely forested or treed pine mires (treed pine bogs) (Laine & Vanha-Majamaa, 1992; Laine *et al.*, 1995) or on a broader scale of all peatland sites, also including treeless mires (Sarasto, 1961; Reinikainen, 1988; Hotanen & Vasander, 1992). After drainage, with the subsequent changes in the growth substrate and tree layer, the composition of the lower layers of vegetation also change drastically. Plant species adapted to wet habitats are the first to disappear, while hummock-dwelling species (e.g. dwarf shrubs) may benefit from drainage (Sarasto, 1961; Eurola *et al.*, 1984; Laine *et al.*, 1995). Only a few studies have concentrated solely on the influence of mire margin on drained sites (Eurola *et al.*, 1995).

The objectives of this study were to describe and analyse changes in the diversity of the vegetation of forested mire margin communities. The vegetation diversity was analysed at both species and community levels on pristine and drained sites.

Materials and Methods

Sample Plots and Study Area

The material in this study consists of data collected from the permanent sample plots of the 8th National Forest Inventory (1985-86) in southern and central Finland (S of 66° N, 2618 plots). The following criteria had to be fulfilled: (1) location S of the 66th latitude, (2) site type either undrained paludified forest, mire margin forest, forested pine mire or spruce swamp, or (3) drained mire margin forest, drained forested pine mire or spruce swamp, (4) the whole plot classified as a single site type, (5) vegetation sampled from four (2 m^2) vegetation sample quadrats on the plot, (6) tree stand untreated for at least ten years before the inventory and (7) forest development class from 4 (young thinning stands) to 6 (mature stands). A total of 156 plots met all these criteria: 82 undrained and 74 drained for forestry (Fig. 1, Table 1). On each of the circular plots (300 m^2) on four 2 m^2 sample quadrats percentage cover of the understorey vegetation (including tree and shrub species lower than 50 cm) was estimated visually. The canopy cover of tree (>1.5 m high) and shrub (0.5-1.5 m high) species was estimated for the whole plot. The study area, field work and vegetation surveys are described in more detail in previous articles (Korpela & Reinikainen, 1996a, b).

The traditional "Finnish-forest-type approach of Cajander" (Cajander, 1926; Frey, 1973) was used to separate the sample plots into site classes. The paludified forest site types were classified according to Kalela (1973). The mire site types were classified using the site types of Heikurainen and Pakarinen (1982). The wood production potential of the site types was estimated using the six-scale system of Huikari (1974) in which site fertility class I is the

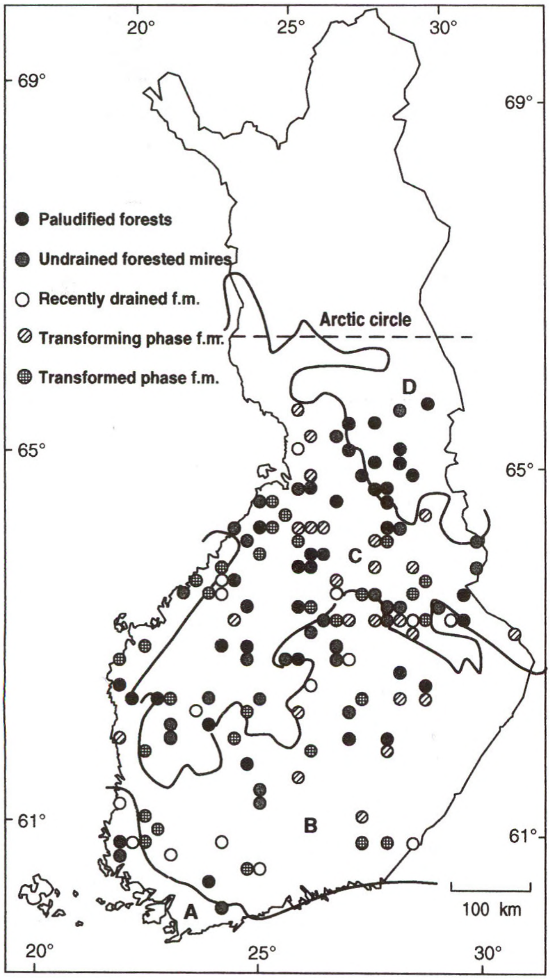

Fig. 1: Distribution of sample plots within A = hemiboreal, B = southern boreal, C = middle boreal and D = northern boreal subzones (Ahti et al., *1968). The drainage phase of the sample plots is also presented according to Heikurainen & Pakarinen (1982).*

richest and VI the poorest (Table 1). The drainage phase of the sites was determined according to the system described by Sarasto (1961) and by Heikurainen & Pakarinen (1982). The species were classified into forest or mire plants and placed into ecological groups according to Kalela (1973) and Eurola *et al.* (1994).

The nomenclature for vascular plants follows that of Hämet-Ahti *et al.* (1998), for bryophytes Koponen *et al.* (1977) and for lichens Vitikainen *et al.* (1997).

Data Analyses

Mean percentage cover for each understorey species was calculated from the four surveyed quadrats on each plot. If a species was present in only one or two plots, it was excluded from the numerical analyses, except for the species richness values and diversity indices. This left a total of 124 plant species (from a total of 226 species, see Appendix 1) on the 156 sample plots. The data were analysed using multivariate ordination and classification methods. The classification analysis was used to reveal the structure of the vegetation communities. The TWINSPAN (two-way indicator species analysis) method of classification (Hill 1979) was applied using default options for minimum group size (5); maximum number of indicator species (7) and pseudo-species cut levels (0, 2, 5, 10, 20 cover %). Four levels of division were used.

The data were analysed by multivariate ordination in order to find the main gradients in the variation of species composition and to interpret the gradients according to environmental variables. Multivariate ordination was also used to reveal community level diversity. In simulation tests, non-metric multidimensional scaling has been found to be the most efficient method of ordination (Prentice, 1980, Minchin, 1987) for showing the floristic relationships between sample plots. The Bray-Curtis (Czekanowski or Sørensen) dissimilarity index, which is robust to random variation or noise in the data (Faith *et al.* 1987) was used as a measure of dissimilarity between the sample plots. A global non-metric multidimensional (GNMDS) method of ordination, which is included in the DECODA program package version 2.04 (Minchin, 1991), was used. This matrix of dissimilarities between sample plots was calculated from abundance values of species. Standardization was no applied except for logarithmic transformation of th values for species percentage cover. One- to four dimensional GNMDS solutions were carried out usin ten randomly generated starting configurations. Th minimum stress configurations were compared by th Procrustean analysis (Schöneman & Carrol 1970). A Monte-Carlo approach (in DECODA) was used t test the significance of the maximum correlation fc environmental variables through the correlation. T reveal ecological gradients in the data, variation b species composition and environmental variables wa also examined when the number of dimensions wa determined. According to this procedure, a three dimensional GNMDS-ordination solution wa selected for further analysis by the lowest minimu

Table 1: Site types studied and their Finnish abbreviations within 1) paludified upland forests according to Kalela (1973), 2) mire margin forests and 3) forested mires according to Heikurainen & Pakarinen (1982). Fertility class of the site types are according to Huikari (1974) and drainage phase groups (1 = undrained, 2 = recently drained, 3 = transforming phase, 4 = transformed phase) according to Paavilainen & Päivänen (1995).

Site types	Abbrev.	Fert. class	Drainage phase groups (n) 1	2	3	4
1. Paludified forests						
Empetrum-Vaccinium type	sEVT	IV	9			
Vaccinium type	sVT	IV	4			
Vaccinium-Myrtillus type	sVMT	III	10			
Myrtillus type	sMT	III	14			
Deschampsia-Myrtillus type	sDeMT	III	2			
Oxalis-Myrtillus type	sOMT	II	4			
2. Mire margin forests						
Paludified pine forests	KgR	IV	5	2	3	1
Oligo-mesotrophic paludified spruce forest	KgK	II,III	10	4	-	8
Eutrophic paludified hardwood-spruce forest	LhK	I	2	-	-	1
3. Forested pine and spruce mires						
Spruce-pine swamp	KR	IV	5	2	9	3
Spruce swamp	VK	III,IV	14	5	14	9
Herb rich hardwood-spruce swamp	RhK	II	3	2	2	9
Sample plots total 156			82	15	28	31

stress value (0.1592), which was achieved with all ten starting configurations. The weighted averages of all the species included were derived from the sample plot ordination by calculating the means of the scores for the sample plots in which a species occurred and then weighting them according to species abundance.

The relationship between the ordination pattern and the selected explanatory variables was obtained using a vector-fitting procedure. This option calculates a vector for each variable through the ordination configuration, along which the scores of the sample plots have maximum linear (Pearson) correlation with the variable in question (Kantvilas & Minchin, 1989; Minchin, 1991).

The environmental and other explanatory variables used to explain the results of the ordination and classification were site variables (temperature sum, fertility class, drainage phase and peat thickness), tree stand variables (basal area, mean diameter, dominant height, stand age and stand development class) and species diversity variables (species richness, diversity indices; Shannon diversity index, H', Simpson heterogeneity index D, and Pielou's evenness index J, according to Magurran (1988)). Spearman correlation coefficients between the three dimensions of GNMDS ordination and explanatory variables were calculated. One-way analysis of variance (ANOVA) was used to test the TWINSPAN clusters for environmental differences.

Results

TWINSPAN Classification and Its Ecological Interpretation

The analysis by TWINSPAN classification resulted in eight vegetation clusters that could be interpreted as ecologically distinct. On the first division level, a cluster of ten (cluster H) sample plots was separated from the rest of the 156 sample plots. These plots were drained, were in the transformed phase and had herb-rich or mesic forest species vegetation (Fig. 2, Table 2, Appendix 1). The indicator species were *Dryopteris carthusiana* (the most dominant species in this cluster), *Trientalis europaea* and the genus *Brachythecium* (the most abundant moss taxon in this cluster). Marsh and spring species (e.g. *Filipendula ulmaria, Viola palustris, Cirsium palustre, Calamagrostis purpurea, Deschampsia cespitosa* and *Sphagnum riparium*) were most abundant in this cluster, while xeric forest species such as dwarf shrubs and lichens were rare. Only 14% of the species occurred in the rest of the sample plots (Fig. 2, Appendix 1). The drainage phase and the site fertility class of these plots were the variables that best explained the early separation from the rest of the plots. The age of the tree stands on these sample plots was also clearly lower than those of other plots (Table 3).

In the second division the remaining 146 plots were divided into two main cluster groups containing 85 and 61 plots. This main division was made according to trophic status, between meso-eutrophy and ombro-oligotrophy. The group of 85 sample plots (E-G, Fig. 2, Table 2) clearly represented meso-eutrophic spruce mire types (KgK, VK, RhK, LhK) and paludified southern mesic forest types (sMT) and herb rich forest types (sOMT). The most important indicator species of these meso-eutrophic plots was *Sphagnum girgensohnii,* which is typical of spruce mires.

The group of 61 sample plots (clusters A-D) were separated into xeric paludified forest types (sEVT, sVT), northern paludified mesic forest types (sVMT) and ombro-oligotrophic pine mire types (KgR, KR), including some of the oligotrophic spruce mire types. These ombro-oligotrophic plots had bog indicator species (e.g. *Vaccinium uliginosum, Ledum palustre, Empetrum nigrum, Sphagnum angustifolium* and *S. russowii*) (Fig. 2, Table 2).

On the meso-eutrophic side of the second division level (n = 85), the eutrophic cluster G (n = 22) was separated on the third division level by the indicator species *Trientalis europaea* (mesic forest species) and *Viola palustris* (marsh and spring species). The most abundant species in this cluster, however, were the spruce mire species *Sphagnum girgensohnii*

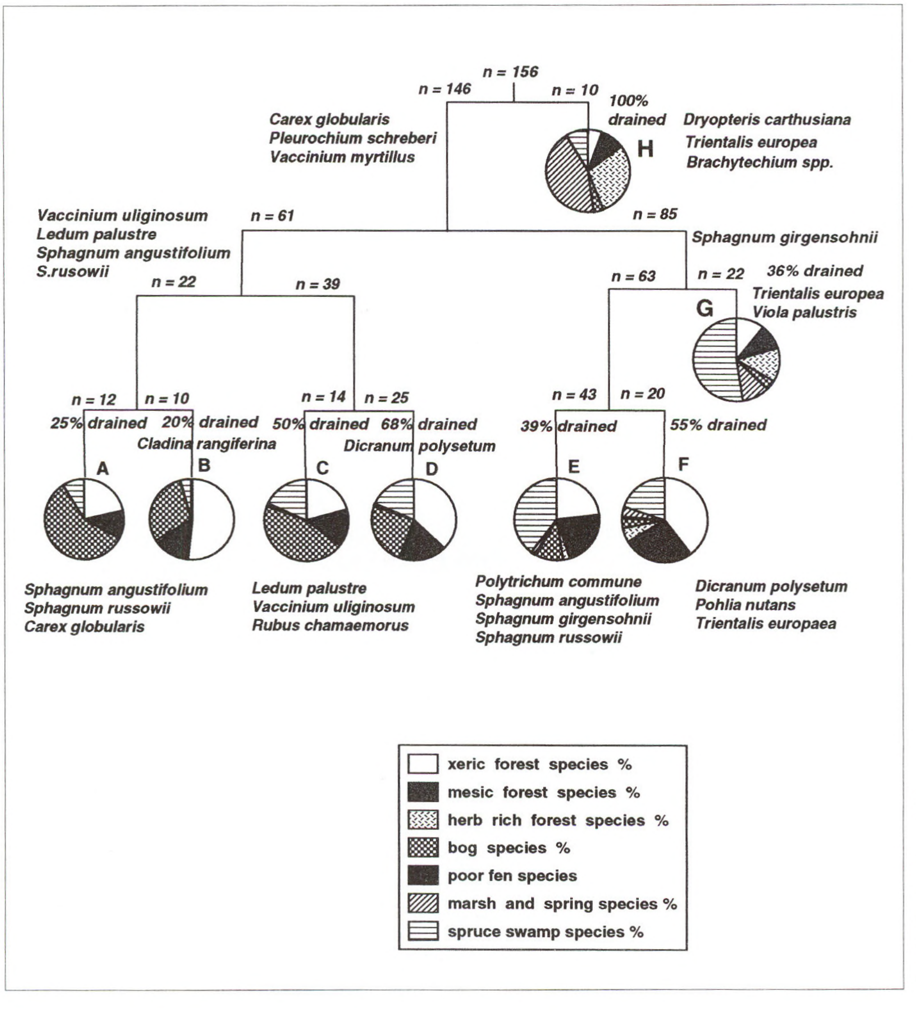

Fig. 2: Division of th TWINSPAN vegetation clusters and distribution of the ecological groups in each cluster with their indicator species.

Table 2. Distribution of site types of paludified forests (1.) and site types and phases of drainage of mire margin forests (2.) and forested mires (3.) in different TWINSPAN-clusters. The numbers of undrained sample plots are marked on the same row as the abbreviations of site types (see Table 1) and corresponding trophic status.

Site types/ Drainage phase	Trophic status	TWINSPAN-clusters A	B	C	D	E	F	G	H	n
1. sEVT	oligo	4	4				1			9
1. sVT	oligo		2		1	1				4
1. sVMT	meso-oligo	1	1	2	3	3				10
1. sMT	meso-oligo			1	1	6	4	2		14
1. sDeMT	meso-oligo					2				2
1. sOMT	meso						3	1		4
2. KgR	oligo	4	1							5
recently dr.		2								2
transforming					2	1				3
transformed			1							1
2. KgK	meso-oligo				2	6		1	1	10
recently dr.						2	2			4
transforming										-
transformed					1	2	1	2	2	8
2. LhK	eutrophic							2		2
recently dr.										-
transforming										-
transformed									1	1
3. KR	oligo			3			1	1		5
recently dr.					1					1
transforming		1		3	4	1				9
transformed					2			1		3
3. VK	meso-oligo			1	1	8		4		14
recently dr.				1	1	3	1			6
transforming			1	2	5	3	2	1		14
transformed				1	1	4	2		1	9
3. RhK	meso							3		3
recently dr.								2		2
transforming						1	1			2
transformed							2	2	5	9
Number of plots		12	10	14	25	43	20	22	10	156

and *Equisetum sylvaticum*. This cluster included the undrained sites of eutrophic paludified hardwood-spruce forest and herb rich and mesotrophic spruce mires. Most of the drained sample plots (36%) in this cluster were herb-rich spruce mires in the transformed phase (Fig. 2, Table 2). A total of 52 herb species, 21 graminoid species and 11 *Sphagnum* species occurred and altogether 136 species were present in this cluster (Appendix 1). The richness of tree and shrub species differed significantly from clusters A-B and clusters E-F and species richness differed from clusters A-B (Table 3).

Divisions at the fourth level occurred most obviously according to moisture gradient. The mesotrophic plots (n = 63) were divided on the fourth division level into clusters E (n = 43) and F (n=20).

Table 3: The mean values (± S.E.) of 1) site quality, 2)tree stand and 3)species diversity variables in the eight final TWINSPAN-vegetation groups (A-H). For each variable TWINSPAN groups with different letters are significantly different (P < 0.05).

	A	B	C	D	E	F	G	H	F-values
Site quality									
1. Temperature sum (dd)	1033.3 (9.8)	1034.0 (8.8)	1015.0 (8.8)	1081.6 (7.03)	1089.3 (11.2)	1155.5**c** (9.81)	1153.2 (11.63)	1117.0 (9.63)	4.74***
1. Site fertility class	3.8 (0.4)	3.9 (0.3)	3.7 (0.6)	3.6 (0.5)	3.1**a-d** (0.3)	3.1**ab** (0.6)	2.5**a-d** (0.8)	2.0**a-df** (0.5)	22.19***
1. Drainage phase class	1.9 (1.0)	1.8 (1.5)	2.8 (1.4)	3.2**a** (1.4)	2.5 (1.4)	2.8 (1.7)	2.7 (1.5)	4.7**a-g** (0.9)	4.56***
1. Peat thickness (dm)	23.6 (26.2)	18.6 (29.9)	43.6 (31.6)	47.6 (36.0)	29.8 (27.0)	31.0 (32.3)	32.5 (31.9)	43.5 (35.3)	1.71
Tree stand									
2. Basal area (m^2/ha)	15.3 (6.3)	16.4 (6.3)	16.6 (6.7)	20.9 (6.0)	24.9**abc** (7.7)	24.7**a** (9.5)	25.1**abc** (7.1)	23.1 (6.6)	5.43***
2. Mean DBH (cm)	12.8 (4.6)	12.5 (4.6)	9.7 (4.6)	10.8 (3.1)	14.3 (6.5)	14.7 (6.1)	15.3**d** (5.6)	14.4 (6.2)	2.53*
2. Dominant height (m)	11.5 (3.7)	12.6 (2.7)	12.6 (3.1)	12.8 (4.0)	16.2 (5.1)	20.2**a-d** (3.2)	20.9**a-e** (5.2)	24.2 (5.50)	12.94***
2. Development class	5.0 (1.0)	5.4 (0.79	4.7 (0.9)	4.9 (0.7)	5.1 (0.8)	5.0 (0.7)	5.1 (0.8)	4.9 (0.7)	0.93
2. Stand age (years)	108.3 (47.4)	113.0 (32.3)	95.1 (48.9)	94.6 (27.6)	101.6 (48.1)	77.5 (31.1)	75.1 (32.3)	58.0**bde** (20.4)	3.19**
Species diversity									
3. Tree and shrub species richness	4.2 (1.0)	4.1 (0.9)	4.2 (1.3)	5.3 (1.8)	4.6 (1.6)	5.3 (1.6)	6.1**a-ce** (1.8)	7.0 (3.3)	5.06***
3. Species richness	15.3 (1.9)	16.5 (4.2)	18.4 (4.1)	18.6**a** (4.4)	16.0 (4.3)	19.9 (6.6)	22.4**ae** (8.6)	22.9**ae** (5.6)	4.93***
3. D - index	0.82 (0.069	0.75 (0.11)	0.81 (0.08)	0.81 (0.07)	0.76 (0.11)	0.80 (0.09)	0.72 (0.16)	0.84**e** (0.06)	2.90**
3. H'- index	2.02 (0.23)	1.83 (0.33)	2.03 (0.32)	2.02 (0.34)	1.82 (0.36)	2.03 (0.41)	1.92 (0.57)	2.32**be** (0.30)	2.53*
3. J - index	0.75 (0.09)	0.66 (0.10)	0.70 (0.10)	0.70 (0.10)	0.67 (0.12)	0.69 (0.09)	0.63 (0.17)	0.75 (0.07)	2.15*
Number of sample plots	12	10	14	25	43	20	22	10	

The sample plots of cluster E included most of the paludified forests and spruce mires. The paludified forests accounted for 30% and the spruce mires for 70% of the sample plots of cluster E; of these spruce mire plots 60% were undrained and 40% drained. The sample plots of cluster F (n = 20) were drained spruce mires and paludified mesic and herb-rich forests (Fig. 2, Table 2).

In cluster E the indicator species were the oligo-mesotrophic *Sphagnum* mosses *S. angustifolium* and *S. russowii*, although *Polytrichum commune* was most abundant in this cluster. The typical spruce mire species *S. girgensohnii* was almost as abundant here as in the meso-eutrophic cluster G. Furthermore, the typical spruce mire herb species *Equisetum sylvaticum* was almost as abundant as in cluster G.

The species composition in cluster F was more forest-like. The indicator and also the most abundant species in this cluster was the forest carpet moss *Dicranum polysetum,* which is typical of xeric forests. Other xeric forest species (e.g. *Calluna vulgaris, Vaccinium vitis-idaea, Deschampsia flexuosa, Pleurozium schreberi, Polytrichum juniperinum* and *Cladonia* spp.) were also more abundant than in cluster E. In addition, the number of herb species was higher than in cluster E. The mesic forest species *Trientalis europaea* was an indicator and other typical mesic forest herbs (e.g. *Lycopodium annotinum, Orthilia secunda, Rubus saxatilis*) and herb rich forest species (e.g. *Oxalis acetosella, Maianthemum bifolium*) and marsh and spring species (e.g. *Caltha palustris, Potentilla erecta*) also occurred in this cluster. Other graminoids, in addition to *Deschampsia flexuosa,* were slightly more abundant here as well. The cover of *Sphagnum* species was clearly less than in cluster E.

In the fourth level division sample plots of the ombro-oligotrophic group (n= 61) were divided into clusters A (n=12), B (n = 10), C (n = 14) and D (n=25). Clusters A and B consisted mainly of undrained paludified xeric forest types and forested pine mire types. Clusters C and D included most of the drained ombro-oligotrophic pine mire types, which mainly were in the transforming phase (Fig. 2, Table 2). There were clear differences in the phase of drainage and in the peat thickness between cluster groups A-B and C-D (Table 3).

The composition of species in cluster A was dominated by bog species; the most abundant species were the indicator species *Sphagnum russowii* and *S. angustifolium.* In cluster B xeric forest species dominated, with indicator species *Cladina rangiferina* being the most abundant of the reindeer lichens. In cluster C both undrained and drained spruce-pine mire types were found. The indicator species here were dwarf shrubs typical of bogs (*Ledum palustre, Vaccinium uliginosum)* and the herb *Rubus chamaemorus,* which was most abundant in this cluster. Other bog species (e.g. *Eriophorum vaginatum, Sphagnum angustifolium*) were also abundant in this cluster. Cluster D contained most of the drained spruce-pine mires and some of the drained spruce mires; the few undrained sites in this cluster were paludified xeric and mesic forests. The indicator species was the xeric forest moss *Dicranum polysetum* which, as well as other forest carpet mosses, was more abundant here than in cluster C. The xeric forest dwarf shrub *Vaccinium vitis-idaea* was also more abundant in cluster D.

On the main division level, the cluster group A-D differed significantly from cluster group E-H in terms of their site fertility classes (F-value 39.6, p = 0.0000) and those tree stand variables that indicate site productivity, such as dominant height (F-value 19.6, p = 0.0000) and basal area (F-value 8.7, p = 0.0000). There was also a significant difference in temperature sums (F-value 5.5, p = 0.0003) between cluster groups A-D and E-H. This main division happened also according to pine and spruce dominance in the canopy layer (see Appendix 1).

Ordination of Plant Communities and the Main Ecological Gradients

In the ordination space of the three dimensions, the sample plots separated well according to their site trophic level. This main gradient was interpreted to describe a fertility gradient. The oligotrophic mire sites and paludified xeric forest sites were situated at one end and the meso-eutrophic and herb rich sites at the opposite end of this gradient in the ordination space (Fig. 3a). Site-fertility class and tree-stand variables such as dominant height had the strongest correlations with this direction in the ordination space (Fig. 3a, Table 4). Inter-correlation between tree stand variables: dominant height, basal area and mean diameter, was also strong (0.494 ***, 0.435 ***) and the correlation between them and site fertility class was significant (-0.494 ***, -0.410 ***, -0.277 ***).

The second gradient was interpreted to describe the post-drainage succession. Most of the undrained forested mire sites were situated at one end and most of the drained forested mire sites, in the transformed phase, were situated at the opposite end of this gradient in the ordination space (Fig. 3a). The

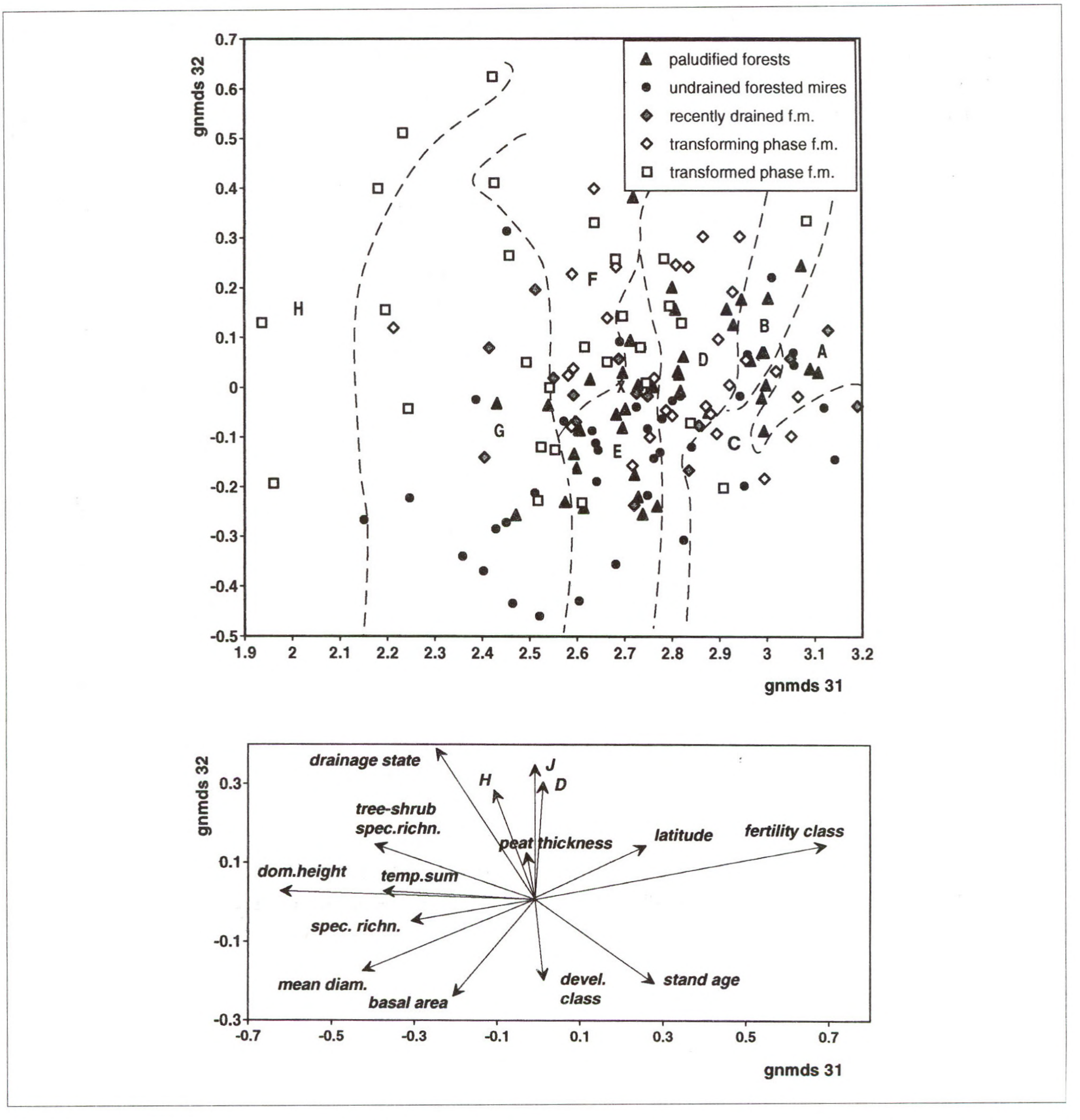

Fig. 3a: GNMDS ordination with the first and second dimensions of the three dimensions. The sample plot symbols relate to drainage phases. Sample plots are divided by dotted lines into TWINSPAN vegetation groups indicated by corresponding letters, which are placed in the centers of the groups. The directions of the maximum correlations of the environmental variables are indicated with arrows. The length of the arrow describes the strength of the correlation (the midpoint of the arrows = x in the upper figure).

paludified forests were situated between these two groups. State of drainage was best correlated with the first and the second directions in the ordination space (Fig. 3a, Table 4). Species richness as well as richness of tree and shrub species correlated significantly with the direction of the fertility gradient and with each other. Species diversity indices correlated best with the direction of drainage-state gradient (Fig. 3a, Table 4). Inter-correlation between state of drainage and peat thickness was strong (0.640 ***). State of drainage correlated significantly also with Shannon (H') diversity-index (0.289 ***) and with Simpson (D) heterogeneity-index and Pielou's (J) evenness-index (0.214 **, 0.240 **).

A gradient appeared to be related to peat thickness, although the maximum correlation of peat thickness was not significant in the three-dimensional ordination (Fig. 3b, Table 4). On this gradient peat

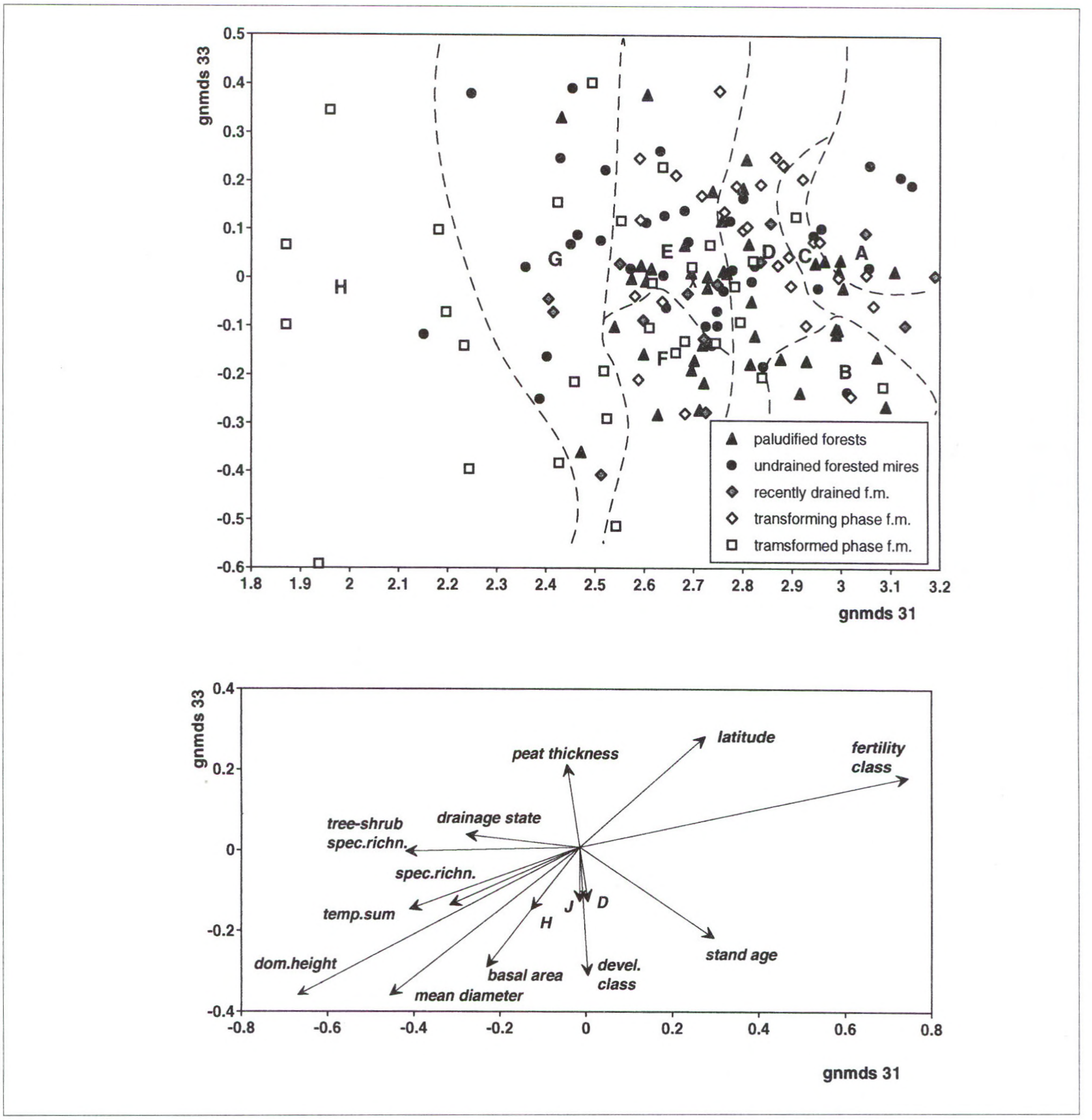

Fig. 3b: GNMDS-ordination with the first and third dimensions of the three dimensions. The sample plot symbols relate to drainage phases. Sample plots are divided by dotted lines into TWINSPAN vegetation groups indicated by corresponding letters, which are placed in the centers of the groups. The directions of the maximum correlations of the environmental variables are indicated with arrows. The length of the arrow describes the strength of the correlation (the midpoint of the arrows = x in the upper figure).

thickness increased towards most of the undrained and some of the drained forested mires in the transforming and transformed phase. The tree-stand variables (development class, basal area, mean diameter) also had significant negative correlations with this gradient (Fig. 3b, Table 4). Peat thickness had significant negative correlation with stand development class, mean diameter and dominant height (-0.272 ***, -0.273 ***, -0.227 **).

Species Indicating the Main Gradients

The weighted averages for species in the GNMDS ordination of the sample plots also showed that the main gradient was related to site fertility. Species common for xeric forests and ombro-oligotrophic bogs (e.g. *Calluna vulgaris, Vaccinium vitis-idaea, Empetrum nigrum, Ledum palustre, Vaccinium uliginosum, V. microcarpum, Eriophorum vaginatum, Sphagnum nemoreum,*

*Table 4: Spearman rank correlations between the explanatory variables and the dimensions of the three-dimensional GNMDS ordination (p = * < 0.05, ** < 0.01, *** < 0.001).*

Variables	Dimension 1		Dimension 2		Dimension 3		
	r	p	r	p	r	p	N
Latitude	0.35	***	0.04		0.14		156
Temperature sum	-0.44	***	-0.05		-0.05		156
Peat thickness	0.06		-0.04		0.12	-	156
Site fertility class	0.73	***	0.29	***	0.10		156
Drainage state	-0.14	*	0.17	*	0.01		156
Basal area	-0.48	***	-0.27	**	-0.31	***	156
Mean diameter	-0.37	***	-0.05		-0.32	***	156
Dominant height	-0.65	***	-0.12	-	-0.30	***	132
Development class	-0.12	-	-0.12	-	-0.29	***	156
Stand age	0.24	**	-0.05		-0.19	*	156
Tree-shrub species richness	-0.29	***	0.074		0.14	*	156
Species richness	-0.15	*	-0.11		0.03		156
H'-index	-0.01		0.21	**	0.01		156
D-index	0.11	-	0.28	***	-0.01		156
J-index	0.10		0.31	***	-0.02		156

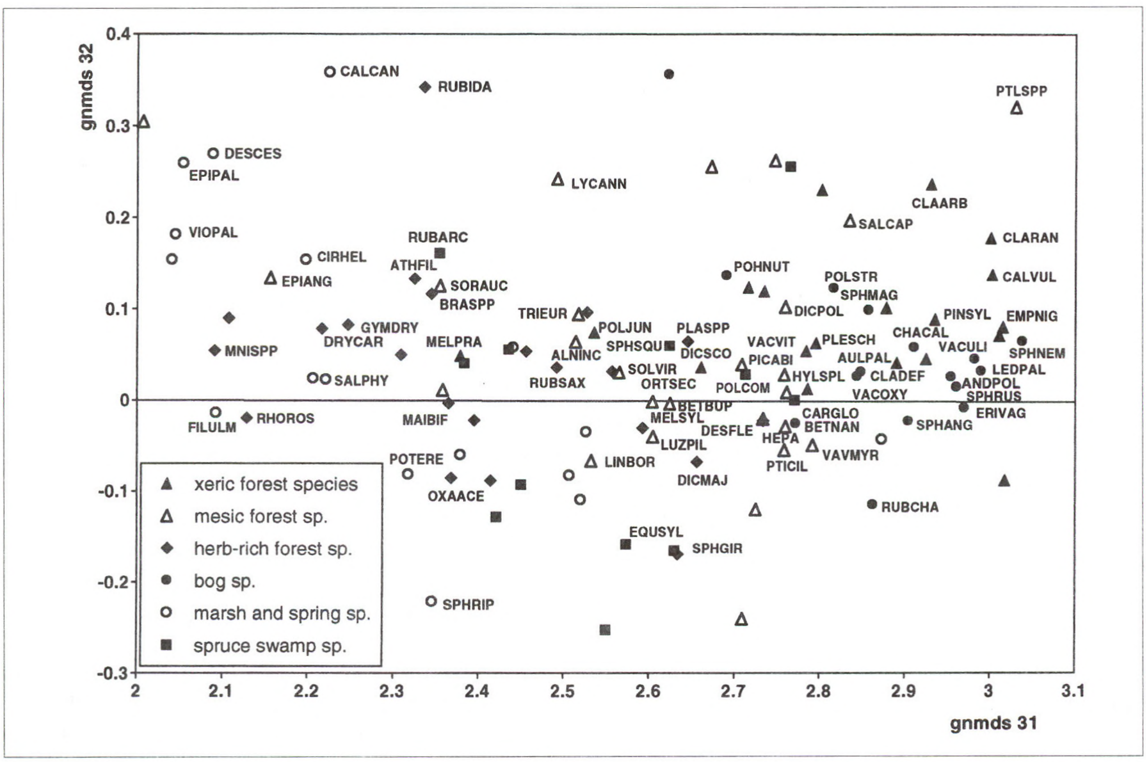

Fig. 4a: Optima of the species investigated in the ordination space of the first and the second dimensions of three dimensional GNMDS-ordination. The species are in ecological groups according to Eurola et al. (1994). Species names are the first three letters of the genus and the first three letters of the species (only the species which occur at least on ten sample plots are marked by letters, for full species names see Appendix 1).

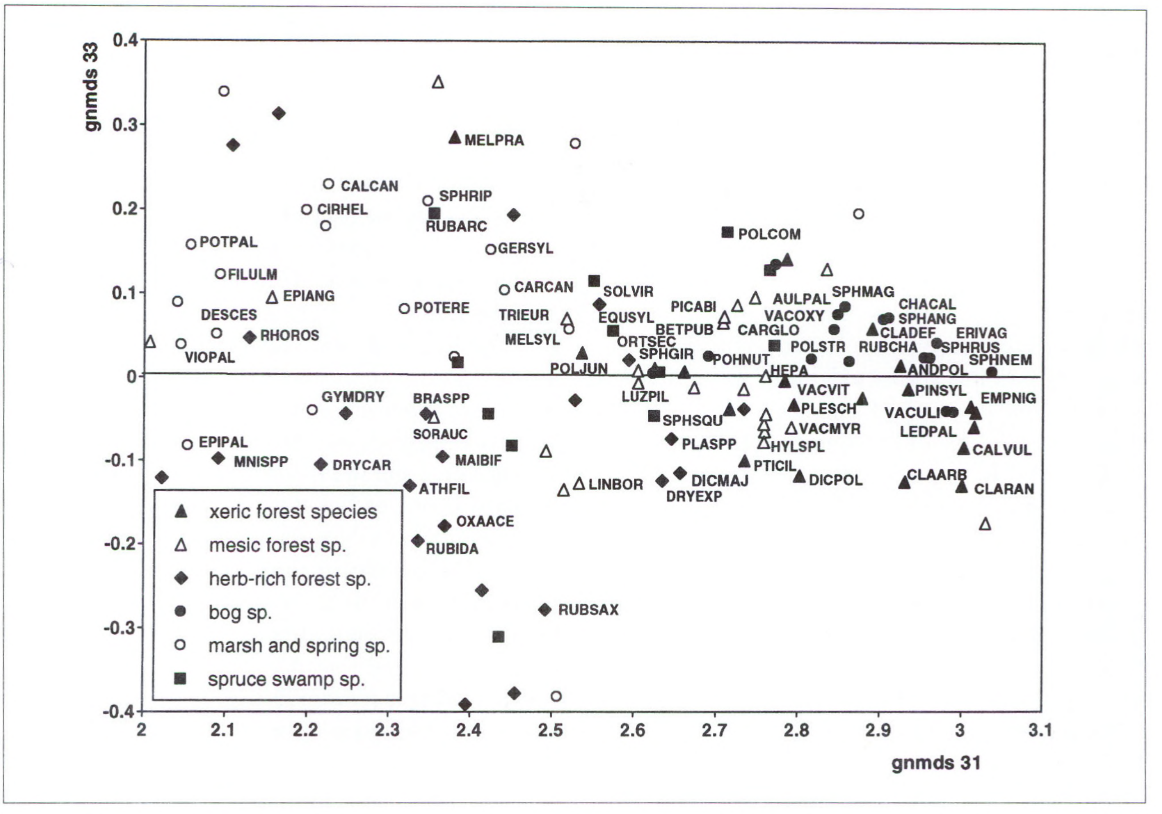

Figure 4 b: Optima of the species investigated in the ordination space of the first and the third dimensions of three dimensional GNMDS-ordination. Species are in ecological groups according to Eurola et al. *(1994). Species names are the first three letters of the genus and the first three letters of the species (only the species which occur at least on ten sample plots are marked by letters, for full species names see Appendix 1).*

S. russowii, S. angustifolium, Cladonia spp.) were situated towards one end of the fertility gradient and herb-rich forest species (e.g. *Oxalis acetosella, Maianthemum bifolium, Athyrium filix-femina, Dryopteris carthusiana, Rhodobryum roseum, Mniaceae* spp.) and mesotrophic and eutrophic marsh and spring species (e.g. *Viola palustris, Potentilla palustris, Epilobium palustris, Deschampsia cespitosa, Cirsium helenioides, Calamagrostis canescens, Sphagnum riparium*) at the opposite end of this gradient. Most of the constant species, for example, mesic forest plants (e.g. *Vaccinium myrtillus, Hylocomium splendens, Linnaea borealis, Trientalis europaea*) and mesotrophic spruce swamp species (e.g. *Sphagnum girgensohnii, Equisetum sylvaticum, Polytrichum commune*), had their optima in the centre of the gradient (Fig. 4a and 4b). The third gradient, indicating moisture or peat-thickness separated forest and mire species better than the second gradient. Most of the xeric, mesic and herb-rich forest species were situated at one end and most of the bog, spruce swamp as well as marsh and spring species at the opposite end of this third gradient. (Fig. 4b).

DISCUSSION

Changes in Diversity on the Community Level (β-diversity)

In the analysis of the main compositional gradients, the multivariate ordination (GNMDS) method revealed that the main gradient was a fertility gradient. This is a gradient that strongly dominates the patterns of boreal forest and mire vegetation (Eurola *et al.*, 1984; Kuusipalo, 1985; Lahti & Väisänen, 1987; Tonteri *et al.*, 1990; Korpela & Reinikainen, 1996a & b). The sample plots at the opposite ends of this gradient still had about 10% of the species in common (see Appendix 1). It is important to notice that when forested site types with mire margin (supplementary nutrients) influence are concerned, all the trophic levels from oligotrophy to eutrophy are minerotrophic and that these usually are richer in nutrients than sites with mire expanse (mire inherent) influence (Eurola *et al.*, 1995).

Compared to the ordination of undrained sites in an earlier study (Korpela & Reinikainen, 1996a), the variation in species composition along the site fertility gradient was not affected by drainage. In addition, the gradient between mire margin and mire expanse, which is one of the main gradients in Scandinavian mire vegetation (Sjörs, 1983; Eurola *et al.*, 1984; Malmer, 1985), was parallel to the site fertility gradient. Together these two gradients illustrate the mixture caused partly by indicators of mire margin influence, which are mesotrophic forest and wetland species, and partly by mire expanse indicators, some of which are xeric forest species. Generally it is difficult to separate these gradients (e.g. Eurola *et al.*, 1994; Eurola *et al.*, 1995).

After drainage, correlation between the variables that describe tree-stand productivity and the fertility gradient, remained equally high or increased compared to the same variables on the undrained sites (Korpela & Reinikainen, 1996a). When compared to studies of mires with sparse tree stands before drainage (e.g. Laine *et al.*, 1995), the importance of the tree stand becomes more dominating after drainage. In this study of forested mire margin sites it is equally dominating on undrained and drained sites. Age of the tree stand was, in general, lower on drained sites both at the ombro-oligotrophic and the meso-eutrophic ends of the fertility gradient.

Temperature sum and latitude also correlated with the fertility gradient, reflecting the differences in distribution of the site types along the S-N gradient. This is because most of the paludified xeric forest sites, at the lower nutrient level end of the fertility gradient, belong to northern forest site types where bog dwarf shrubs are typical (Kalela, 1973) while most of the herb-rich, drained sites of the transformed phase, at the higher nutrient level end, are located in the south.

The undrained and drained sites were in a continuum along these main gradients, except for the ten herb-rich sites in cluster H, which clearly differed from the rest of the sites in both the ordination and classification analyses. In general, the trophic status (nutrient level) of the forested mire sites in the transformed phase was higher (see Table 1). One of the reasons for this is that usually the most productive sites have been drained first. On the other hand, the change in species composition has been fastest on the more nutrient-rich site types (Sarasto, 1961; Pienimäki, 1982; Hotanen *et al.* 1999). In addition, these sites reach the final transformed phase in a shorter time (Hotanen & Vasander, 1992; Laine *et al.*, 1995). In the early works of Fenno-Scandinavian forest scientists (Tanttu, 1915; Melin, 1917; Cajander, 1926; Lukkala, 1929) it was evident even then that after drainage the effect of moisture diminishes while that of nutrient status increases. On pristine forested mire margin sites the moisture gradient is shorter than on mires in general. The effect of drainage further shortens this gradient and accentuates the importance of the nutrient status according this study.

The length of the second gradient, which describes post-drainage succession, is shorter than the first one; about 40% of the species occur at both ends of this gradient. Most of the undrained oligo-mesotrophic and meso-eutrophic spruce mires were situated at one end of this gradient and the drained spruce mires, mostly in the transformed phase, at the other. The paludified mesic and herb-rich forests were situated in the middle of the gradient, together with mesotrophic drained spruce mires in the transforming and transformed phase. The situation of these sample plots in different drainage phases in the ordination space showed the degree of change from mire vegetation to forest vegetation along the post-drainage succession.

The third gradient was as long as the second and related more to peat thickness or paludification and to development of the tree stand. In this study of mire margin forested sites, however, peat thickness was not a statistically significant variable. The clearest difference in peat thickness was at the ombro-oligotrophic end of the fertility gradient between paludified xeric forests and undrained forested pine mires.

Changes in the Diversity of Species (α-diversity)

Diversity of species is expressed as numbers of species (species richness) and is related to the fertility gradient. At the ombro-oligotrophic end of the fertility gradient the mean richness of species was highest in cluster (D) that had the most sample plots in the transforming phase. At the meso-eutrophic end of the gradient the lowest mean richness of species was in cluster (E) with undrained mesotrophic spruce mire sites. According to Vasander (1987) and Vasander *et al.* (1997) the number of plant species is highest some years after drainage, at which time three groups of plants are found on the sites: original mire species colonists and forest species. These groups of species were most evenly distributed in cluster D. The opposite situation prevails in cluster (E) with the least species richness and the lowest indices of diversity, e.g. the group of undrained mesotrophic spruce mires, where typical, constant spruce mire species dominated. For example, clusters E and F had the same number of ecological groups of species (Fig. 2). In cluster E the spruce swamp species were dominant, followed by xeric and mesic forest and bog species whilst herb-rich forest and marsh and spring species had much lower abundance. The drained mesotrophic sites in cluster F had a more evenly distributed abundance of xeric and mesic forest species, spruce swamp species and, on a smaller scale, an even distribution of herb-rich forest, bog, and spring and marsh species. Cluster E had the highest number of mire species.

The highest mean species richness was in the eutrophic end of the fertility gradient where the herb-rich forest species and spring and marsh species were most abundant. In the drained, herb-rich sites in the transformed phase (cluster H) the marsh and spring species were most abundant and dominated together with herb-rich forest species. In the undrained eutrophic and herb-rich sites (cluster G) spruce mire species were most dominant. The indices of diversity of species were the highest in the drained sites but there was no significant difference between drained and undrained communities. Only the Shannon-index (H'), which weights rare species (Hill, 1973), differed significantly between the most herb-rich, transformed phase sites of cluster H, the most paludified xeric forest sites in cluster B and the undrained mesotrophic spruce mire sites in cluster E (Table 3). The diversity indices have not proven to be distinct enough to show differences between undrained and drained peatland sites (Vasander *et al.*, 1997). Many scientists have pointed out the weaknesses of the theory behind and the uncertain biological purpose of the indices of diversity (Whittaker, 1972; Peet, 1974; Alatalo, 1981; van der Maarel, 1988). In this study the ordination methods used in analysing the beta-diversity proved to be more relevant in revealing the differences between pristine and drained mire margin sites.

Changes in the Composition of Species

A list of species coverage by clusters is presented in Appendix 1.

Along the sample plots groups of oligotrophic mire sites (from undrained to different successional phases of drainage) there was an apparent decrease in mire dwarf shrubs such as *Vaccinium uliginosum* and a corresponding increase in forest dwarf shrubs, for example, *Vaccinium vitis-idaea.* The abundance of *Vaccinium myrtillus*, a dwarf shrub of mesic forests, were quite even throughout the main gradients; only on the most eutrophic sites was abundance very low compared to an earlier study of undrained forested mire margin sites (Korpela & Reinikainen, 1996a).

Carex species usually disappear during the post-drainage succession because of increasing shading from trees and dwarf shrubs and the lack of nutrients (Aapala & Kokko, 1988; Laine *et al.*, 1995). The most abundant *Carex* species on forested mire margin sites with a thin peat layer is *Carex globularis* (Eurola, 1962; Ruuhijärvi, 1960). This species seemed to be indifferent to the effect of drainage. It was most abundant on drained, oligotrophic forested mires in the transforming phase but occurred sparsely even on the most nutrient-rich forested mires in the transformed phase. Most of the *Carex* species were found on undrained meso-eutrophic spruce mire sites, but the intermediate and flark level tall sedges (e.g. *Carex lasiocarpa, C. chordorriza*) were found as relicts on drained forested mires in the transforming phase. The hummock level species *Eriophorum vaginatum* was most abundant on undrained oligotrophic forested mire sites and was found in small numbers on drained sites in the transforming phase, although it is quite resistant to or even benefits from disturbance (Chapin *et al.*, 1979).

The most common xeric forest graminoid species was *Deschampsia flexuosa* and it was most abundant on xeric and mesic paludified forest sites and drained spruce mire sites in different drainage phases. The forest graminoid species that indicate marsh and spring influence, such as *Calamagrostis purpurea, C. canescens* and *Deschampsia cespitosa,* were more abundant on herb-rich sites in the transformed phase. The increase of grasses is an indication that sufficient light, growth space and mineral nitrogen is available, as well as influnce of surface water for some species (Holmen 1964; Platonov 1976).

The herbs were clearly related to the meso-eutrophic side of the fertility gradient. The only herbs present throughout the main gradients were *Rubus chamaemorus* and, very sparsely, *Melampyrum pratense. Rubus chamaemorus* is a hummock species of nutrient-poor mire sites (Eurola *et al.*, 1994) and it benefits from the disappearance of competitors (Aapala & Kokko, 1988). In this study, this species was clearly most abundant on undrained oligotrophic forested mire sites, and its abundance decreased towards drained and more nutrient rich forested mire sites, but was still found on herb-rich sites in the transformed phase. *Epilobium angustifolium* had clearly benefitted from drainage and was most abundant on herb-rich forested mires in transformed phase. *Equisetum sylvaticum* was most abundant on undrained meso-eutrophic spruce mires. Of the forest herbs *Maianthemum bifolium* was most abundant on undrained, eutrophic paludified hardwood-spruce forests and *Oxalis acetosella* on herb-rich forested mires in the transformed phase. According to Laine *et al.* (1995), the cover of mesic forest herbs (e.g. *Dryopteris carthusiana* and *Trientalis europaea*) clearly increased with increasing drainage-age; this was also apparent in the present study. On the other hand, wetland herbs such as *Cirsium palustre* and *Viola palustre* may even become more abundant after drainage and also grow well on forested mire sites in the transformed phase (Aapala & Kokko, 1988).

In this study the decrease of *Sphagnum* species along post-drainage succession was clear. Many forest mosses including *Pleurozium schreberi* and *Hylocomium splendens* already occur on hummocks in undrained forested mires and, after drainage, they increase in abundance (Laine *et al.*, 1995). The forest carpet moss *Dicranum polysetum* also becomes more abundant. In this study it was found in all clusters, but not so abundantly as *Pleurozium schreberi*. An increase in the abundance of *Brachythecium* species on the meso-eutrophic side of the fertility gradient was also apparent. *Sphagnum riparium* and *S. squarrosum,* which indicate spring and marsh influence, were most abundant on sites in the transformed phase. Usually they are found in or near ditches, where optimal growth conditions still prevail. *Sphagnum angustifolium* was still found in all clusters and *S. girgensohnii* also occurred sparsely on the drained eutrophic sites of the transformed phase, although greatly decreased compared to its abundance on undrained herb-rich and eutrophic sites (Korpela & Reinikainen, 1996a).

CONCLUSIONS

The main gradients on undrained mires are nutrient level and hydrological condition. After drainage hydrological conditions become more uniform and, consequently, the influence of the tree stand, which depends on the nutrient level, gradually becomes more dominating. Along the post-drainage succession, mire species (e.g. *Sphagnum* spp.) are gradually replaced by forest species, which already dominate in the surrounding forests (Laine *et al.*, 1995). In this study, site fertility determined the main gradient on both undrained and drained sites. Nutrient availability was not restricted even on the undrained mire margin sites. This main gradient correlated with site productivity and with stand variables such as dominant height.

The second gradient representing the drainage state described the hydrological conditions better than peat thickness, because already when undrained, forested mire margin sites have a thin peat layer. The drainage phase is determined according to the proportions of mire and forest species. The effect of drainage was stronger on the meso-eutrophic forested mire margin sites than on more oligotrophic sites. Similar results have been obtained in other studies where pristine and drained mire sites have been investigated (Hotanen & Vasander 1992; Laine *et al.* 1995; Hotanen *et al.* 1999). On undrained forested mire margin sites the influence of tree stands is dominating. It was not even possible to determine whether drainage increased the effect of tree stand on the vegetation.

The mixing of mire margin features, such as spring and marsh and spruce swamp, results in high floristic diversity by associations with the forest and bog species. After drainage, the diversity of species still remains high when only species numbers are examined. Structurally, the communities change towards forest vegetation where mesic and herb-rich forest species dominate, with mire species present as relicts. These changes indicate that the composition of the vegetation will resemble corresponding forest sites at the same or higher level of fertility. It remains to be seen whether the tree stand structure of the drained sites also will become more forest like.

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Appendix 1.

Tree and shrub layer and field and ground layer vegetation in the eight TWINSPAN-clusters (A-H). Mean percentage cover is presented for each species in each cluster. Total number of field and ground layer species included was 226. Total number of species present in each cluster by growth-form groups are also presented. The number of sample plots in each cluster = (n).

	TWINSPAN-clusters							
	A (n = 12)	**B** (n = 10)	**C** (n = 14)	**D** (n =25)	**E** (n = 43)	**F** (n = 20)	**G** (n = 22)	**H** (n = 10)
Tree and shrub layer								
Alnus glutinosa	-	-	-	-	-	-	1.00	0.8
Alnus incana	-	-	0.04	0.5	0.4	1.95	1.88	3.38
Betula pendula	-	-	0.47	0.05	1.49	2.90	1.36	0.60
Betula pubescens	7.92	6.00	23.32	20.26	14.88	15.84	25.05	34.19
Daphne mezereum	-	-	-	-	-	-	-	0.02
Frangula alnus	-	-	-	0.02	0.02	-	0.31	0.05
Juniperus communis	-	-	0.29	1.02	0.58	0.66	0.20	0.70
Picea abies	13.55	6.88	13.93	18.51	38.18	38.89	26.20	16.38
Pinus sylvestris	21.31	22.12	14.46	18.18	7.43	6.99	9.26	6.75
Populus tremula	0.01	0.15	2.86	0.160.	0.20	0.05	1.26	0.66
Prunus padus	-	-	-	-	-	-	0.01	0.31
Salix caprea	0.01	0.07	0.04	0.29	0.21	0.15	0.44	2.22
Rubus idaeus	-	-	-	-	0.02	0.23	-	2.00
Sorbus aucuparia	-	0.16	0.37	0.34	0.56	1.10	0.84	1.16
Salix spp.	2.98	1.13	1.17	1.38	0.89	0.41	1.61	11.59
Total number of species	6	7	10	11	12	11	13	15
Field layer								
Trees and shrubs:								
Alnus incana	-	-	-	-	0.02	0.11	0.08	-
Betula pubescens	-	0.01	0.50	0.09	0.16	0.10	0.43	0.22
Frangula alnus	-	-	-	-	-	-	0.13	0.02
Juniperus communis	-	-	-	0.17	0.10	-	0.03	0.10
Picea abies	-	0.08	0.11	1.01	0.41	0.20	0.21	0.24
Pinus sylvestris	0.01	0.11	0.06	0.15	0.02	0.01	-	-
Populus tremula	-	-	-	-	-	-	0.01	-
Prunus padus	-	-	-	-	-	-	-	0.05
Rubus idaeus	-	-	-	-	-	0.11	0.01	0.28
Salix aurita	0.01	-	0.07	0.82	-	0.01	0.05	-
Salix caprea	-	-	-	0.35	0.01	0.03	0.02	-
Salix phylicifolia	-	-	-	0.03	0.04	-	0.10	0.60
Salix repens	-	-	-	0.02	-	0.01	-	-
Salix rosmarinifolia	0.33	-	-	-	-	-	-	-
Salix spp.	-	-	-	-	0.07	-	-	1.55
Sorbus aucuparia	-	0.03	0.09	0.05	0.10	0.14	0.09	0.89
Total number of species	3	4	5	10	9	9	11	9
Dwarf-shrubs:								
Andromeda polifolia	0.12	0.05	0.28	0.02	0.01	-	0	-
Arctostaphylos uva-ursi	-	-	-	-	-	0.01	-	-

	TWINSPAN-clusters							
	A (n = 12)	B (n = 10)	C (n = 14)	D (n =25)	E (n = 43)	F (n = 20)	G (n = 22)	H (n = 10)
Betula nana	0.06	-	0.01	0.26	0.44	-	-	-
Calluna vulgaris	2.14	3.79	0.01	0.01	-	0.63	-	-
Chamadaphne calyculata	-	-	0.33	0.55	0.02	-	-	-
Empetrum hermaphroditum	-	-	1.24	2.07	-	-	-	-
Empetrum nigrum	6.67	7.27	1.74	0.65	0.01	0.08	0.00	-
Ledum palustre	8.66	11.52	10.43	0.91	0.05	0.34	-	-
Vaccinium microcarpum	0.02	-	-	-	-	-	-	-
V. myrtillus	11.26	14.11	14.59	13.55	16.62	13.42	3.56	0.94
V. oxycoccus	0.04	0.00	0.09	0.07	0.02	0.02	-	-
V. uliginosum	10.28	7.79	8.47	2.23	0.09	0.46	0.05	0.20
V. vitis-idaea	6.57	10.82	6.42	11.83	6.30	7.27	3.42	0.59
Total number of species	10	8	11	11	9	8	6	3
Graminoids:								
Agrostis canina	-	-	-	-	-	-	-	1.5
A. capillaris	-	-	-	-	0.01	0.02	0.14	1.95
A. tenuis	-	-	-	-	0.010	0.04	-	-
Calamagrostis arundinacea	-	-	0.14	-	0.01	0.20	0.01	0.01
C. canescens	-	-	-	-	0.01	0.21	0.50	2.10
C. purpurea	-	-	0.01	0.01	0.08	0.29	0.98	5.68
Carex brunnescens	-	-	-	0.01	0.01	0.03	-	0.08
C. canescens	-	-	0.22	-	0.20	-	0.63	0.02
C. chordorriza	-	-	-	0.01	-	-	-	-
C. digitalis	-	-	-	-	0.01	-	0.03	-
C. echinata	-	-	-	0.01	-	-	-	-
C. flava	-	-	-	-	-	-	0.02	-
C. globularis	3.10	0.42	2.68	6.45	3.36	3.62	1.36	0.01
C. lasiocarpa	-	-	-	0.12	-	-	-	-
C. loliacea	-	-	-	-	-	-	0.01	-
C. nigra	-	-	-	0.24	-	0.08	0.70	0.02
C. ovalis	-	-	-	-	-	0.01	-	-
C. pallescens	-	-	-	-	-	-	0.57	-
C. spp.	-	-	-	-	0.01	0.10	0.13	0.80
C. vaginata	-	-	-	-	-	0.03	-	-
Deschampsia cespitosa	-	-	-	-	0.12	0.53	0.18	8.60
Deschampsi flexuosa	0.01	0.10	1.91	0.34	0.28	2.25	0.21	0.45
Eriophorum vaginatum	0.01	-	2.59	0.25	0.02	-	0.05	-
Festuca spp.	-	-	-	-	-	-	0.05	-
Juncus filiformis	-	-	0.01	0.12	0.04	-	-	0.72
Luzula pilosa	-	-	0.18	-	0.07	0.12	0.09	0.07
Melica nutans	-	-	-	-	-	-	0.02	-
Molinia caerulea	-	-	-	-	-	-	0.03	-
Phragmites australis	-	-	-	-	0.07	-	-	-
Poa nemoralis	-	-	-	-	-	-	0.19	-
P. pratensis	-	-	-	-	-	-	-	0.10
P. spp.	-	-	-	-	-	0.06	0.05	-
Total number of species	3	2	8	10	16	16	21	15

	TWINSPAN-clusters							
	A (n = 12)	B (n = 10)	C (n = 14)	D (n =25)	E (n = 43)	F (n = 20)	G (n = 22)	H (n = 10)
Herbs:								
Aegopodium podagraria	-	-	-	-	-	-	-	0.08
Anemone nemorosa	-	-	-	-	-	-	0.05	-
Angelica sylvestris	-	-	-	-	-	-	-	0.31
Athyrium filix-femina	-	-	-	-	-	-	0.27	0.50
Calla palustris	-	-	-	-	-	-	0.01	-
Caltha palustris	-	-	-	-	-	0.05	0.02	0.80
Campanula rotundifolia	-	-	-	-	-	0.01	-	-
Cardamine pratensis	-	-	-	-	-	-	-	0.01
Centaurea jacea	-	-	-	-	-	-	-	0.05
Cirsium helenoides	-	-	-	-	-	-	0.03	0.55
C. palustre	-	-	-	-	-	-	0.01	0.27
Cornus suecica	-	-	0.11	-	0.22	-	-	-
Crepis paludosa	-	-	-	-	-	-	0.11	0.01
Dactylis maculata	-	-	-	-	0.01	0.01	-	-
Dryopteris expansa	-	-	-	-	0.48	0.55	0.06	-
D. carthusiana	-	-	-	0.41	0.38	0.25	2.58	10.6
Epilobium angustifolium	-	-	-	0.11	0.03	0.29	0.14	2.85
E. palustre	-	-	-	-	-	0	0.01	0.07
E. spp.	-	-	-	-	-	-	0.03	-
Equisetum arvense	-	-	-	-	-	0.09	0.27	-
Equisetum palustre	-	-	0.04	0.05	-	0.05	0.23	0.11
E. pratense	-	-	-	-	-	-	0.01	-
E. sylvaticum	0.02	-	0.31	0.96	2.53	0.66	3.63	1.44
Filipendula ulmaria	-	-	-	-	-	-	0.61	3.18
Fragaria vesca	-	-	-	-	-	-	0.01	0.06
Galium boreale	-	-	-	-	-	-	0.08	-
Galium palustre	-	-	-	-	-	-	0.02	0.08
G. spp.	-	-	-	-	-	-	0.01	-
Geranium sylvaticum	-	-	-	-	-	-	0.03	1.70
Geum rivale	-	-	-	-	-	-	0.07	0.06
Goodyera repens	-	0.01	-	-	-	0.05	-	-
Gymnocarpium dryopteris	-	-	-	-	0.29	0.75	0.83	4.36
Huperzia selago	-	-	-	-	-	-	-	0.01
Linnaea borealis	-	0.02	0.10	0.02	0.15	0.72	0.46	0.14
Listera cordata	-	-	-	-	0.01	-	-	-
Lychnis viscaria	-	-	-	-	-	-	-	0.29
Lycopodium annotinum	-	-	0.11	0.01	0.03	0.56	0.03	0.58
Lysimachia vulgaris	-	-	-	-	-	-	0.09	-
Maianthemum bifolium	-	-	0.06	0.05	0.17	0.52	2.45	0.25
Melampyrum pratense	0.01	0.02	0.08	0.09	0.07	0.10	0.49	0.30
M. sylvaticum	-	-	-	0.01	0.03	0.03	0.02	0.02
Menyanthes trifoliata	-	-	-	0.01	-	0.05	-	-
Orthilia secunda	-	-	0.02	0.27	0.20	0.57	0.21	0.07
Oxalis acetosella	-	-	-	-	0.13	0.22	0.74	0.96
Polygonum viviparum	-	-	-	-	-	-	0.01	-
Potentilla erecta	-	-	-	-	-	0.03	0.10	0.07
P. palustris	-	-	-	-	0.06	-	0.36	1.17

	TWINSPAN-clusters							
	A	B	C	D	E	F	G	H
	(n = 12)	(n = 10)	(n = 14)	(n =25)	(n = 43)	(n = 20)	(n = 22)	(n = 10)
Prunella vulgaris	-	-	-	-	-	-	0.01	-
Pyrola chlorantha	-	-	-	-	-	0.01	-	-
P. minor	-	-	-	-	-	-	0.03	-
P. rotundifolia	-	-	-	0.02	-	-	0.09	0.55
Ranunculus acris	-	-	-	-	-	-	0.01	1.26
R. auricomus	-	-	-	-	-	-	0.01	-
R. repens	-	-	-	-	-	0.75	-	0.05
R. spp.	-	-	-	-	-	-	-	0.05
Rubus arcticus	-	-	-	0.02	0.09	0.11	0.65	1.58
R. chamaemorus	1.39	0.48	8.86	1.67	1.75	0.36	0.98	0.05
R. saxatilis	-	-	-	-	-	0.35	0.3	0.13
Rumex acetosa	-	-	-	-	-	-	-	0.01
Solidago virgaurea	-	-	0.14	0.01	0.15	0.01	0.17	0.55
Stellaria graminea	-	-	-	-	-	-	-	0.01
S. longifolia	-	-	-	-	-	-	-	0.07
S. uliginosa	-	-	-	-	-	0.01	-	-
S. palustris	-	-	-	-	-	-	0.01	-
Thelypteris paludosa	-	-	-	-	-	0.15	0.51	-
Veronica serpyllifolia	-	-	-	-	-	-	0.02	-
Viola canina	-	-	-	-	-	-	0.01	-
V. epipsila	-	-	-	-	-	-	-	1.90
V. palustris	-	-	-	-	-	-	0.42	4.06
V. riviniana	-	-	-	-	-	-	0.02	-
Thelypteris phegopteris	-	-	-	-	0.06	-	0.05	0.20
Trientalis europaea	-	-	0.96	0.06	0.17	0.83	1.60	1.86
Urtica dioica	-	-	-	-	-	-	-	0.40
Total number of species	3	4	11	16	21	30	52	47
Ground layer								
Sphagnum mosses:								
Sphagnum angustifolium	31.35	1.85	20.77	11.89	6.56	0.87	1.62	2.23
S. centrale	-	-	-	-	0.27	-	1.28	0.10
S. compactum	-	-	-	0.04	-	-	-	-
S. fallax	-	-	-	-	0.35	-	-	-
S. fuscum	-	1.00	-		-	-	-	
S. girgensohnii	-	4.50	12.49	2.78	24.11	14.35	28.56	2.38
S. magellanicum	0.04	-	1.50	1.88	0.29	0.08	0.01	-
S. nemoreum	11.31	4.95	0.20	1.20	0.04	-	0.17	-
S. papillosum	-	-	0.01	-	-	0.75	0.09	-
S. riparium	-	-	1.07	-	0.21	-	2.73	5.01
S. rubellum	-	0.20	-		-	-	-	-
S. russowii	20.23	9.13	9.52	8.54	2.98	-	0.15	-
S. spp.	-	-	-	-	-	0.01	-	0.02
S. squarrosum	-	-	-	-	1.16	0.25	0.24	0.45
S. subsecundum	-	-	-	-	-	-	-	0.02
S. teres	-	-	-	-	-	0.25	0.07	-
S. wulfianum	-	0.01	-	-	0.05	-	1.23	-
Total number of species	4	7	7	6	10	7	11	7

	TWINSPAN-clusters							
	A	B	C	D	E	F	G	H
	(n = 12)	(n = 10)	(n = 14)	(n =25)	(n = 43)	(n = 20)	(n = 22)	(n = 10)
Carpet mosses:								
Dicranum fuscescens	0.01	-	-	0.01	0.01	0.10	0.02	-
D. majus	-	0.06	0.07	0.15	0.97	0.38	0.61	0.02
D. polysetum	0.23	2.01	0.19	3.54	1.43	5.54	0.38	0.70
D. scoparium	0.02	0.46	0.56	0.93	1.87	1.42	0.9	0.31
D. spp.	-	-	-	-	-	-	0.05	-
Hylocomium splendens	5.29	2.88	1.57	6.06	7.08	7.20	1.34	0.36
Pleurozium schreberi	15.99	39.16	18.11	26.63	16.33	22.83	4.08	2.83
Ptilium crista-castrensis	0.02	0.05	-	0.01	0.32	0.14	-	-
P. spp.	-	-	-	0.01	-	-	-	-
Rhytidiadelphus triquetrus	-	-	-	-	-	0.06	0.16	0.05
Total number of species	6	6	5	8	7	8	8	6
Other Bryidae:								
Atrichum undulatum	-	-	-	0.06	-	-	0.01	0.05
Aulacomium palustre	0.87	1.42	0.22	0.58	0.13	0.22	0.24	0.27
Brachythecium albicans	-	-	-	-	-	0.10	-	-
B. curtum	-	-	-	-	0.01	-	-	-
B. oedipodium	-	-	-	0.02	-	0.05	-	-
B. spp.	-	0.03	0.03	0.08	0.31	1.36	0.62	6.08
Calliergon cordifolium	-	-	-	-	-	-	-	0.90
C. cuspidatum	-	-	-	-	-	-	-	0.01
C. spp.	-	-	-	-	-	-	0.01	-
C. stramineum	-	-	-	-	-	-	-	0.10
Campylium spp.	-	-	-	-	-	-	0.01	-
Ceratodon purpureus	-	-	-	-	-	0.01	-	-
Climacium dendroides	-	-	-	-	-	-	0.51	0.01
Gymnodontium strumiferum	-	-	-	-	-	0.01	-	-
Mnium spp.	-	-	-	-	0.01	0.15	0.10	0.54
Musci spp.	-	-	-	-	-	-	0.01	-
Paraleucobryum longifolium		-	-	-	-	0.01	-	-
Plagiochila asplenioides	-	-	-	-	-	0.10	0.03	0.20
Plagiomnium affine	-	-	-	-	-	0.01	-	-
P. ellipticum	-	-	-	-	-	0.01	-	-
Plagiomnium spp.	-	-	-	-	0.02	0.09	0.01	0.06
Plagiothecium spp.	-	-	-	0.14	0.07	0.09	0.07	0.10
Pogonatum spp.	-	-	-	-	-	0.01	-	-
Pohlia cruda	-	-	-	0.01	-	0.01	-	-
P. nutans	-	0.01	0.13	0.28	0.02	0.64	0.10	0.24
Polytrichastrum formosum	-	-	-	-	-	-	0.01	0.01
P. spp.	-	-	-	-	-	-	0.14	-
Polytrichum commune	10.05	0.58	10.09	13.11	16.23	1.27	9.52	0.45
P. juniperum	0.01	0.13	0.09	0.15	0.99	1.50	-	0.35
P. longisetum	-	-	-	-	0.02	-	-	-
P. strictum	0.91	0.22	0.32	0.66	0.02	0.68	0.01	0.50
Pseudobryum cinclidioides	-	-	-	-	-	0.75	-	-
Rhodobrym roseum	-	-	-	-	-	-	0.01	0.04
Rhizomnium spp.	-	-	-	0.01	-	-	0.01	0.05

	TWINSPAN-clusters							
	A (n = 12)	B (n = 10)	C (n = 14)	D (n =25)	E (n = 43)	F (n = 20)	G (n = 22)	H (n = 10)
Tetraphis pellucida	-	-	-	0.02	-	-	0.02	-
Warnstofia fluitans	-	-	-	-	0.02	-	-	-
Total number of species	4	6	6	12	13	19	19	18
Hepatics:								
Barbilophozia barbata	-	-	-	-	0.01	-	-	-
B. lycopodioides	-	-	0.09	-	0.01	-	0.03	-
B. spp.	-	-	-	0.04	-	-	0.11	-
Blepharostoma trichopyllum-		-	-	-	-	0.01	-	-
Hepaticae	0.07	0.01	0.10	0.24	0.10	0.01	0.17	0.02
Mylia anomala	-	-	0.01	-	-	-	-	-
Pellia spp.	-	-	-	-	-	0.25	-	-
Ptilidium ciliare	-	-	0.07	0.09	0.03	-	0.04	-
P. spp.	-	1.50	-	0.04	0.02	0.03	0.01	-
Total number of species	1	2	4	4	6	3	5	1
Lichens:								
Cladina arbuscula	-	0.72	0.01	0.05	0.01	0.15	-	-
C. rangiferina	0.01	0.13	0.52	0.03	0.01	-	-	-
C. stellaris	-	0.38	-	-	-	-	-	-
Cladonia cariosa	-	-	0.01	0.04	-	-	-	-
C. chlorophaea	-	-	0.01	-	-	0.01	-	-
C. cornuta	-	0.16	-	0.01	0.01	0.06	-	-
C. deformis	0.01	0.01	0.08	0.04	0.01	0.01	-	-
C. fimbriata	-	0.01	-	0.01	-	-	0.01	-
C. furcata	-	-	-	-	-	-	0.01	-
C. gracilis	-	0.01	0.11	-	-	0.01	-	-
C. spp.	-	-	-	0.05	0.01	0.12	0.01	0.01
C. sulphurina	-	-	-	0.1	-	0.01	-	-
Peltigera aphtosa	-	0.20	0.16	-	-	-	-	-
Total number of species	2	8	7	8	5	8	3	1
Total number of species present	36	45	64	85	96	108	136	107

EFFECTS ON HYDROLOGY AND SURFACE WATER CHEMISTRY OF REGENERATION CUTTINGS IN PEATLAND FORESTS

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SUMMARY

In the Nordic-Baltic region, forestry is a land-use activity, that influences extensive areas. Sylviculture includes practices such as cutting, scarification, drainage and reforestation all of which affect hydrological and hydrochemical conditions. Cutting results in elevated groundwater levels, altered water pathways and increased runoff. Scarification, drainage and forest growth lower the water level and intensify organic matter decomposition. Cutting and drainage result in leaching of chemical substances while forest growth reduces discharge and alters water quality.

Especially important in catchment water and chemical turnover are downslope sites, close to watercourses. These sites can be wet mineral soils or peatlands where more careful forestry measures should be performed. Instead of clear-cutting, shelterwood cutting is one alternative practice, which is supposed to produce no or low drainage activity. In order to study the effects of this alternative forestry practice, catchment investigations were performed including clear-cut drained and undrained areas; shelterwood cut areas and natural peatland forests acted as controls.

Results from the investigations showed elevated groundwater levels in both clear-cut and shelterwood cut areas. In the shelterwoods, even small elevations in water level had a decisive influence on tree roots. Forest cutting was followed by both increased runoff and leaching of chemical substances. The contribution to element outflow was mainly in the form of increased chemical concentrations. Observations during this first short period after forest harvesting measures indicated little or no benefits from shelterwood cutting.

Keywords: groundwater, leaching, shelterwood, water quality, wetland forestry.

INTRODUCTION

Forests cover extensive areas in the Nordic-Baltic region. For example, in Estonia, Finland and Sweden there are 2, 20 and 24 million hectares (Mha), respectively (40-65% of the national area). Substantial areas consist of forested peatlands. In both Finland and Sweden, forested peatlands extend over at least 5 Mha, i.e. 20-25% of the total forest area (Hånell, 1990; Aarne, 1994). In the Baltic States, drained wetland forests also constitute large parts of the forested land. In Estonia, for example, the fraction is 25%, which is approximately half of the total peatland area. In all countries in the region, substantial parts of the forests comprise highly productive stands with considerable stand volumes of 300-400 $m^3 ha^{-1}$. Other wetland forests also exist, however, and some of these host a high degree of biodiversity. In the early part of this century, these forests might have been the origin of today's productive wetland forests (Hånell, 1993). Drainage, a necessary prerequisite, resulted in increased forest growth. At present, many of these forests have reached an age when harvesting is required and future problems of regeneration need to be identified.

Traditionally, harvesting has been performed as clear-cutting which gives rise to problems of high groundwater level, frosts and competing field and bush vegetation. Apart from the elevated groundwater levels following cutting (Lundin, 1979; Päivänen, 1982), there are also altered water pathways, higher runoff and increased outflow of most chemical substances (Grip, 1982; Ahtiainen, 1992). Measures to facilitate regeneration include drainage and other soil treatments such as mounding. These forestry practices, however, also influence the hydrology and water chemistry of the site. Drainage lowers water levels and intensifies organic matter decomposition, resulting in release of chemical compounds (Sallantaus, 1989). Altogether, these forestry activities have been shown to result in increased chemical leaching to surface waters (Lundin, 1988).

Areas of interest for drainage, such as wet mineral soils or peatlands, are often located in low-lying areas, such as depressions or downslope situations, close to surface water formation. Therefore these areas often have considerable influence on stream hydrology and water chemistry (Bishop, 1991). Such wetlands have been described as "the kidneys of the landscape" (Mitsch & Gosselink, 1993). This fact, in conjunction with the recognition of wetland values (Kuntze, 1994; Wheeler & Shaw, 1995) and with identified forest regeneration problems, has resulted in attempts to develop alternative forest practices involving more careful operations. Shelterwood cutting has, to a large extent, replaced conventional clear-cutting. Shelterwood cutting involves leaving a number of trees of the existing stand (300-400 stems per hectare) in order to minimize water level elevation, prevent frosts and shelter new plants from over exposure to direct sunlight. Leaching of nutrients is also reduced as a result of shelterwood tree uptake. It is not yet clear, however, what benefits shelterwood cutting has for hydrology and hydrochemistry.

The objectives of this study were to determine environmental effects and forest regeneration success of shelterwood cutting without drainage, compared to traditional clear-cutting with and without drainage. Research focused on hydrology, water chemistry and leaching aspects.

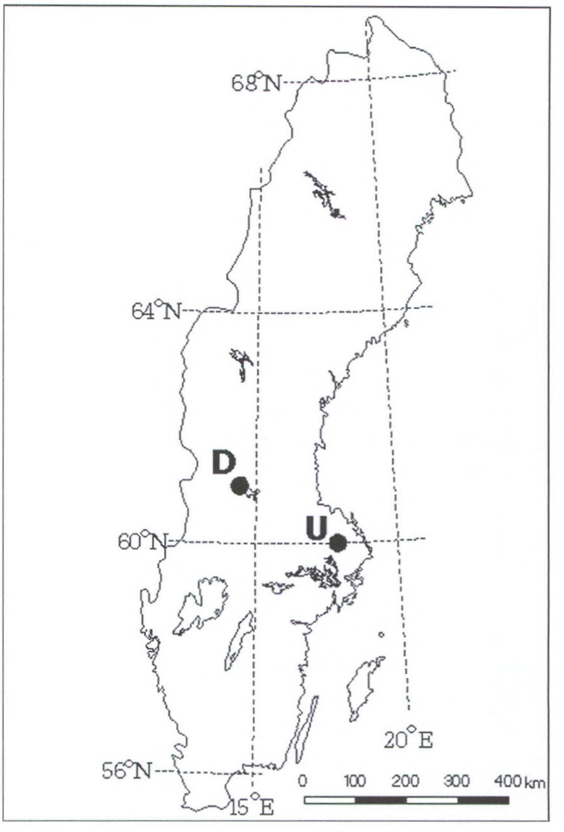


Fig. 1: Geographical location of the investigation areas located in Dalarna county (D), central Sweden and in Uppland county (U), east-central Sweden.

METHODS

The investigations were performed employing a catchment approach, with peatland forest stands at low-lying locations surrounded by mineral soil uplands. A consequence of this is that discharge and streamwater chemistry relate both to the peatland stand and the stands on the uplands. Forestry activities were not performed in the surrounding uplands.

The investigation sites were located in two regions in Sweden (Fig. 1). In each region, five similar catchments were studied. The peatland forest stands formed different treatments: one untreated control; two clear-cuts scarified by mounding, one of which was also drained; and two shelterwoods.

Investigation periods were 1-2 years before measures and 3-4 years after treatments. In the Uppland areas 1992-93 represented the calibration period and 1994-97 the impact period. In the Dalarna areas, these periods were 1993-94 and 1995-97, respectively.

Effects of forestry measures (cutting, drainage and mounding) in the impact areas were determined by the comparison method with a calibration period. The calibration period provided paired regression correlations between control and impact areas. Relationships obtained were used for the period after treatments. Using these relationships and values from the control area, the presumptive untreated conditions in the impact area were estimated. These conditions were compared to measured values for the impact area and deviations were evaluated as effects of measures.

Measurements comprised precipitation, soil physical and chemical characteristics, groundwater levels, runoff and streamwater quality. Precipitation was determined for two-week periods at one location in each region, close to the stands investigated. These values were correlated to results from the Swedish precipitation survey, using daily observations from a meteorological station close to the investigation sites. From these comparisons, daily values at the catchments were obtained. Groundwater levels were measured manually at fortnightly intervals in ten piezometers installed to 1-2 m depth. The upper piezometers showed the actual groundwater levels. Runoff was calculated for each catchment at one discharge station, equipped with a V-notch weir and a water level recorder. In the Dalarna region, problems with discharge determinations occurred, resulting in periods of uncertainty and with difficulty in estimating leaching.

Streamwater chemical composition was determined from analyses of monthly water samples and comprised pH, HCO_3, DOC, K, Ca, Fe, NO_3, NH_4, tot-N, tot-P and water colour. Analyses were performed according to Swedish Standards (SIS, 1986); bicarbonate alkalinity by titration with HCl, DOC on a Shimadzu element analyser, K, Ca and Fe by ICP, N and P by flow injection analysis (FIA) and water colour as absorption (420 nm) by spectrophotometry.

Leaching of substances from the catchments was determined from daily values for discharge and interpolated water chemistry concentrations from the monthly samples. The flows of substances were added to provide monthly totals. Results for the Dalarna area are not reported in this paper.

SITE DESCRIPTION

Investigations were performed in two regions in Sweden, one in Uppland County, 60°08'N; 17°43'E, representing fertile lowlands in a favourable climate and the other in Dalarna County, 60°53'N; 14°23'E, 250 km NW of the Uppland area. In the Dalarna region, conditions were less rich than at Uppland and climatic similarities with northern Sweden prevailed. Norway spruce (*Picea abies*) dominated tree stands with stand volumes at 300-400 $m^3\ ha^{-1}$.

As the peatland forest stands existed on wetlands of minerotrophic fen type, it is natural that mineral soil uplands influenced both hydrology and streamwater quality. The case study peatlands, mainly of the fen type, acted as discharge areas for outflowing groundwater from the mineral soil uplands.

In general, the catchments in Uppland were larger than the catchments in Dalarna, with areas between 14 ha and 140 ha including peatlands of mainly low herb type, comprising 3-26 ha, 15-27% of the catchments. In Dalarna, catchments were 1-13 ha, including peatlands of low herb or bilberry-horsetail type of 0.5-3.5 ha, 25-50%.

Thickness of the peat layer was 0.2-2.0 m on top of calcareous clay in Uppland. The peat overlies Granitic till in Dalarna where peat thickness is thinner and the main areas have 0.2-0.5 m peat. Catchment location altitudes were approximately 30 m in Uppland and 250 m above mean sea level in Dalarna. At the latter, the climate was colder with slightly higher precipitation and 50% higher runoff due to evapotranspiration being 60 mm lower than in Uppland (Table 1).

RESULTS

Precipitation

Precipitation during the years of investigations, 1992-1997, varied between 457 mm and 700 mm in Uppland and between 627 mm and 743 mm in Dalarna. Precipitation during the years of calibration (641 and 576 mm) did not differ greatly from that in the years after forestry measures (457-700 mm). However, 1994 and 1996 showed fairly low precipitation, with 1996 having 80% of the six-year mean.

Groundwater

The groundwater level varied in the forested condition between 0.2 and 0.9 m below the ground surface, thus occurring mainly within the peat layer. Annual average levels in Uppland were 30-50 cm

Table 1: Climate, hydrological characteristics and potential post-drainage site quality class in the Uppland and Dalarna areas of Sweden. (1. Alexandersson et al., 1991; 2. Odin et al., 1983; 3. Eriksson, 1983; 4. SMHI, 1979; 5. Hånell, pers. comm.)

	Uppland	Dalarna
Annual mean temperature °C (1)	5.0	3.1
Growth period, days with T > 5 °C (2)	190	175
Snow covered period, days	110	150
Precipitation, mm (3)	690	755
Runoff, mm (4)	250	375
Evapotranspiration, mm	440	380
Post-drainage quality class, $m^3\ ha^{-1}yr^{-1}$ (5)	10	7

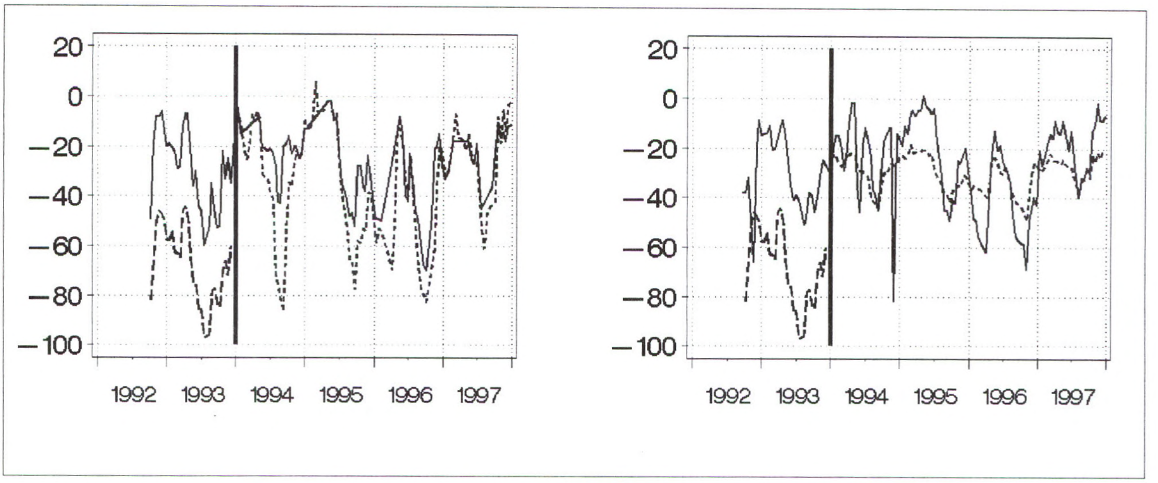


Fig. 2: Groundwater levels at one clear-cut area (left) and one shelterwood cut area (right) in the Uppland region during the period before cutting (1992-93) and after cutting (1994-97). Time of treatments = vertical bar; measured levels = solid line; control area (before cutting) = long dashed line; calculated levels as forested = short dashed line (after cutting).

below the ground surface, with a range of 30-80 cm in the period of May-September. In Dalarna, annual average levels were 40-50 cm below the peat surface with levels in May-September ranging between 40 cm and 80 cm, thus occasionally reaching into the layers of underlying mineral soils. In this region, larger differences in water levels occurred between the peatlands studied, with a range in depth of 23-79 cm. In Uppland, the groundwater levels were similar across the sites.

After cutting in Uppland, annual mean water levels were elevated 0-17 cm with four-year mean rises of 5-11 cm, resulting in a groundwater level 10-30 cm below the ground surface. Under forested conditions, the corresponding levels were expected to be at 30-50 cm (Fig. 2).

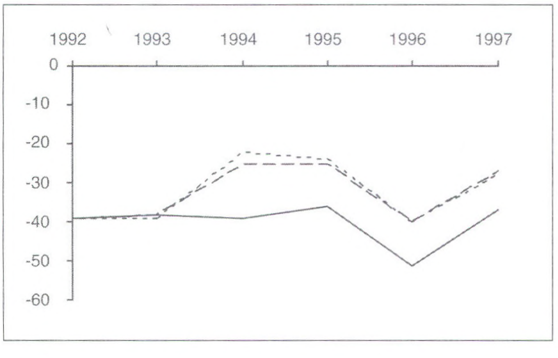


Fig. 3: Annual mean groundwater level at the control (solid line), at clear-cut (short dashed lines) and shelterwood-cut (long dashed line). Range of variation: control 20-100 cm, clear-cut 0-85 cm, shelterwood-cut 0-80 cm.

Drainage lowered the water level with an insignificant change compared to forested conditions. Clear-cutting and shelterwood-cutting changed water levels with similar amounts, with differences of only a few cm (Fig. 3).

In the Dalarna areas, the levels after cutting were elevated by 0-13 cm in the first two years while no differences were observed in the third year. Similar results were also obtained in comparisons between clear-cut and shelterwood cut areas. Drainage at one area lowered the water table to 10 cm below the level expected for forested conditions (Table 2). However, there were differences between stands and the shelterwood adjacent to the control showed 10 cm higher groundwater levels, compared to expected forested conditions, during the main vegetation period (May to August) in all three years after cutting.

Runoff

Runoff from catchments in both regions showed a typical pattern with a high spring snowmelt peak, a lower autumn rain peak and frequent low discharges in periods in-between. Annual runoff in Uppland estimated from a national survey (SMHI, 1979) was approx. 250 mm but runoff measured from the catchments varied between 53 mm and 262 mm. Mean annual runoff for the six years at the control site was 148 mm.

Annual runoff in Dalarna estimated from the national survey (SMHI, 1979) was 375 mm but was measured as 244 mm for 1993-97, with a variation between years from 177 mm to 284 mm from a con-

Table 2: Annual mean groundwater levels (below ground surface) three years after cutting and changes (D) cm at one undrained clear-cut area (CC) at one clear-cut and 1996 drained area (CCD) and two shelterwood forests (SWF1 and 2). + indicates elevated levels, - indicates lowered levels.

	CC, Δ	CCD, Δ	SWF1, Δ	SWF2, Δ
1995	10, +6	11, +22	27, +9	37, 0
1996	20, +1	40, +4	39, +9	57, -8
1997	28, -2	63, -10	50, +9	69, -9

trol catchment. Differences between the impact catchments were even larger, with annual discharges between 221 mm and 535 mm. Uncertainties in measurements were considerable but runoff reported here was based on more reliable stations and periods.

Effects of cutting and drainage were mainly increased runoff, especially at low discharge rates. In Uppland, increases at the clear-cut undrained site were 4-47 mm, 4-55%. Cutting and drainage caused increases between 18 mm and 85 mm, 3-100%. Shelterwood cutting also resulted in higher annual runoff of 7-125 mm, 3-113%. There were comprehensive differences between the years both concerning total runoff and the changes caused by the forestry measures (Fig. 4).

In Dalarna, increases in runoff after clear-cutting varied between 16% and 36%. There was one exception, a decrease of 13% at one catchment in 1997. From one of the shelterwoods, an increase of 1.8 l s^{-1} km^{-2} or 26% was observed. A comparison between the shelterwood and the clear-cut areas showed 8% lower runoff from the shelterwood. In Uppland, discharge measurements provided the potential to investigate effects on low and high flows. Influences on peak flows at moderately high discharges showed both increases and decreases. Lower peaks occurred from the undrained clear-cut, and from all areas during the year 1997 but higher peaks were observed from drained and also from shelterwood areas during the first three years.

Streamwater Quality

Streamwater quality from the catchments was characterised by a considerable degree of water colour but also a considerable content of base cations, especially in Uppland (Tables 3 and 4). Humic substances, partly

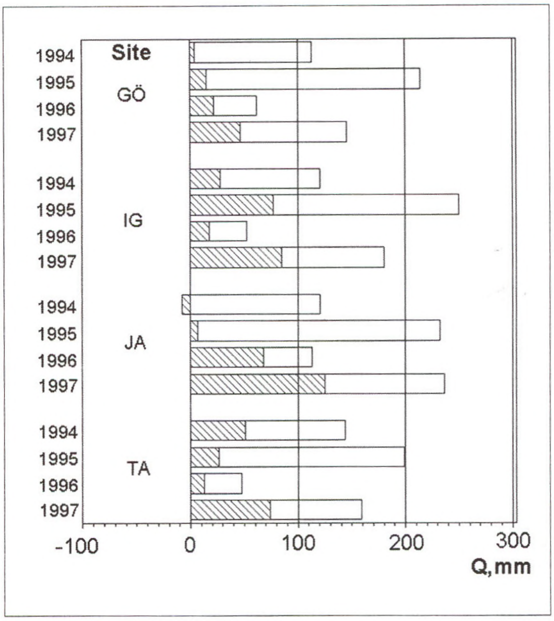


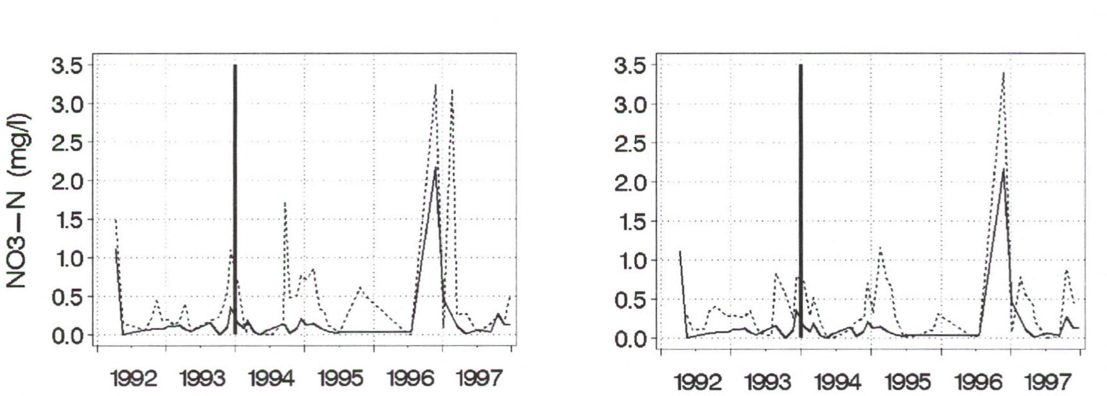
Fig. 4: Annual runoff after measures from the four Uppland impact areas. Total bar = measured runoff and hatched part of bar = runoff change. GÖ = clear-cut, IG = cut and drained, JA & TA = shelter-wood cut.

indicated by high DOC contents but also the presence of metals, especially iron, influence water colour from peatlands. In Uppland, the water colour showed values of 100-300 mg Pt l^{-1}, while in Dalarna values of 50-120 mg Pt l^{-1} were observed. DOC contents were 30-40 mg l^{-1} and 10-30 mg l^{-1}, respectively. Fe concentrations in Uppland were 0.1-0.6 mg l^{-1} and in Dalarna 0.4-1.2 mg l^{-1}. The higher content in Dalarna reflects the acidic conditions with pH being 4.3-5.3. In Uppland acidity was low, with a pH of 5.0-7.0. Discharging mineral soil groundwater influenced pH and base cation content. In Uppland, water partly emanating from underlying clay furnished base cation (BC) contents of 0.5 me l^{-1} and up to as high as 2 me l^{-1}. Poorer conditions prevailed in Dalarna, with a BC content range of 0.1-0.2 me l^{-1}, indicating influences from the nutrient-poor bedrock, granites and porphyry at this site (Tables 3 and 4).

Important nutrients for vegetation growth are potassium, phosphate, nitrate and ammonia and in peatlands K and P are often limited. Potassium concentrations were especially low in three of the Dalarna catchments with only 0.1-0.6 mg l^{-1}, compared with 0.7-1.3 mg l^{-1} in Uppland. Phosphorus content was low in both areas but values obtained were common for Swedish forestland waters with 10-20 g l^{-1}. Nitrogen content in the streamwater from the Upp-

*Table 3: Mean values of streamwater chemistry at three Uppland catchments during two periods after forest operations. D= change compared to forested conditions. I = first two years, II = second two years. CC (GO) = only clear-cut, CCD (IG) = clear-cut and drained, SWF (JA) = shelterwood cut. (mg l^{1}; colour mgPt l^{1}) Significance (ANOVA 95% level) in control and treatment area means showed by * = equal in calibration period and different after treatment; ** = different both before and after treatment.*

		CC	Δ	CCD	Δ	SWF	Δ
PH	I	6.4	-0.1	6.8	+0.3	6.8	+0.1
	II	6.8	+0.1	7.1	+0.3	6.9	-0.1
HCO_3	I	1.00	+0.4	1.61	+0.6	1.28	+0.6
	II	1.27	+0.5	2.23	+0.9	1.53	+0.7
Colour	I	276	+141	163	+35	176	+69
	II	195	+101	117	+25	125	+47
DOC	I	50	+21	40	+5	33	+7
	II	43	+17	33	+1	29	+7
Fe	I	0.57	+0.4	0.15	+0.1	0.26	+0.1
	II	0.61	+0.3	0.13	-0.1	0.37	+0.1
K	I	1.3	+0.8	1.2	+0.3	0.7	+0.1
	II	0.7	+0.1	1.3	+0.4	0.8	+0.2
NO_3-N	I	0.35	+0.15	0.43	-0.01	0.33	+0.13
	II	0.80	+0.15	1.43	-0.01	0.77	+0.03
Tot-N	I	2.6	+0.9	2.4	+0.4	1.8	+0.2
	II	2.6	+0.1	2.9	-0.7	2.1	-0.2
Tot-P	I	0.031	+0.02	0.021	+0.01	0.026	+0.01
	II	0.019	+0.01	0.014	-0.01	0.020	0.00



ig. 5: Monthly nitrate concentrations in streamwater from clear-cut (left) and shelterwood-cut (right) areas (dashed nes) in the Uppland region compared to control (solid line). Vertical bar indicates cutting.

Table 4. Average streamwater chemistry at four Dalarna catchments during three years after measures.D = change compared to forested conditions. CC = only clear-cut, CCD = clear-cut and drained, SWF1 and 2 = shelterwood cut. (mg l^{-1}; colour mgPt l^{-1})

	CC,	Δ	CCD,	Δ	SWF1,	Δ	SWF2,	Δ
pH	4.8,	+0.1	5.3,	+0.2	4.2,	0	4.3,	+0.1
Colour	93,	+25	60,	+40	109,	+3	127,	+55
DOC	17,	+3	14,	+6	25,	+4	26,	+7
Fe	0.8,	+0.2	0.8,	+0.6	0.5,	+0.1	0.5,	+0.2
K	1.0,	+0.7	1.4,	+1.2	0.5,	+0.3	0.3,	+0.2
Ca	1.2,	-0.4	2.5,	0	3.0,	-0.6	2.0,	-0.7
NO_3-N	0.014,	+0.003	0.63,	+0.60	0.015,	+0.005	-	
NH_4-N	0.05,	+0.01	0.47,	+0.4	0.04,	+0.01	-	
Tot-N	0.7,	+0.2	1.8,	+1.1	1.3,	+0.06	1.4,	+0.3
Tot-P	0.015,	+0.005	0.012,	+0.006	0.011,	-0.001	0.015,	+0.008

land catchments (1-2 mg l^{-1}) was about double the concentrations in Dalarna (0.5-1.0 mg l^{-1}), both ranges being common for Swedish forest streams. Organic nitrogen comprised considerable parts of total nitrogen, making up 80% in Uppland and 95% in Dalarna. Obvious differences in inorganic nitrogen compositions in the two regions were a totally dominating nitrate content in Uppland, while ammonia dominated in Dalarna and nitrate showed low concentrations (Tables 3 and 4).

Effects of forestry measures were similar in the two regions, resulting in increased concentrations during the impact period of most substances except for hydrogen and sulphate. In Uppland, small increases in pH were found but larger changes occurred in alkalinity and base cations. Water colour, dependent on organic matter and iron content, also increased. Higher concentrations were found for phosphorus and nitrogen during the first two years but these changes were less pronounced during the second two-year period after forestry measures (Table 3).

Nitrate, being the major inorganic nitrogen compound in Uppland, was strongly influenced by cutting, both clear- and shelterwood-cutting, and increased in the runoff water during the first two years to 0.35 mg l^{-1} (+75%) and 0.33 mg l^{-1} (+65%), respectively. In the fourth year (1997), NO_3-N concentration in water from the clear-cut was still 40% higher than estimated for forested conditions while at the shelterwood-cut the increase was 10% (Fig. 5).

In Dalarna, negligible changes, or possibly a small increase, were observed for pH. There were increases in DOC (15-40%) and in Fe (20-75%) and, consequently, in water colour (5-60%). Both K and P increased but with decreased concentrations of Ca. From the peatlands with shallow peat layers, the increase in K was more evident. Nitrogen concentrations also increased, with nitrate being low during the first years but reaching higher values with time while ammonia showed the opposite pattern, decreasing with time, especially from the undrained, clear-cut area (Fig. 6).

Leaching

Leaching of chemical substances could mainly be calculated for the Uppland catchments and was strongly influenced by streamwater discharge. With a higher runoff, chemical flow increased for most elements. Alkalinity, base cations, organic matter and nitrogen showed approximately 50% higher flows after treatments and nitrate increased in particular (Table 5). The usual higher fraction of inorganic nitrogen was also observed in these investigations. Major differences in effects between clear-cut and shelterwood cut areas were not detected.

Discussion and Conclusions

These preliminary investigations showed similar effects for both clear-cut and shelterwood cut areas over the first three to four year period after forestry measures. Groundwater levels were elevated in both cases and runoff increased at similar rates. There were only small changes in groundwater levels but since

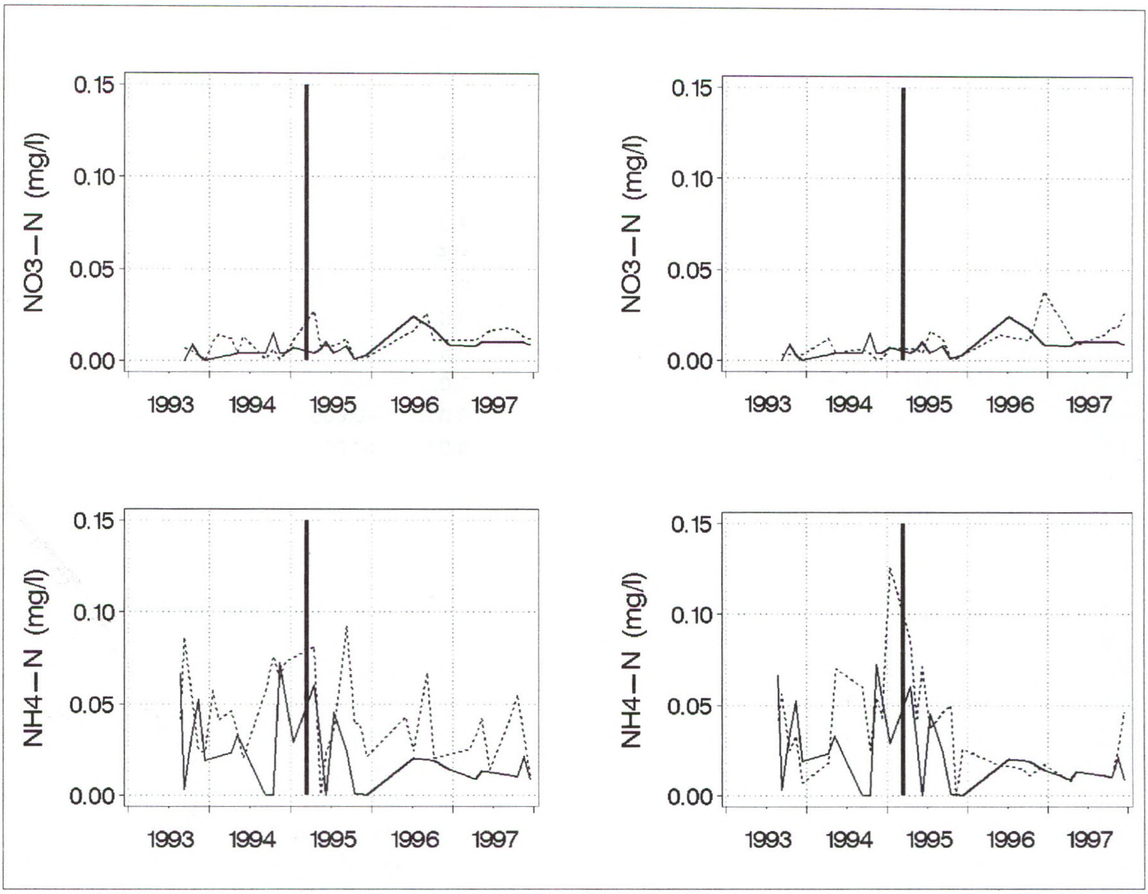


Fig. 6. Monthly inorganic nitrogen concentrations in streamwater in the Dalarna areas, clear-cut (left) and shelterwood cut (right), nitrate above and ammonia below. Concentrations at the control = solid line, concentrations at impact catchment = broken line. Vertical bar indicates time of treatment.

Table 5. Leaching of HCO_3, K, DOC, NH_4-N, NO_3-N and Tot-N (kg ha^{-1} yr^{-1}) and changes (D) compared to the control = REF, clear-cut = CC, clear-cut and drained = CCD, shelterwood cut = SWF during the first four years (1994-97) after measures.

	REF	CC	D	CCD	D	SWF	D
HCO_3	21	19	+ 7	47	+ 12	19	+ 9
K	0.5	1.1	+ 0.7	1.3	+ 0.7	0.9	+ 0.4
DOC	47	61	+ 23	53	+ 18	58	+ 32
NH_4-N	0.03	0.09	+ 0.05	0.09	+ 0.04	0.05	+ 0.01
NO_3-N	0.21	0.64	+ 0.30	0.76	+ 0.15	0.95	+ 0.65
Tot-N	2.3	3.1	+ 0.7	3.5	+ 0.4	3.5	+ 1.4

these peatland forest stands exist on sites with high natural groundwater levels, even small elevations of approximately 10 cm can have considerable effect on rooting depths. Severe forest damage, wind-thrown trees and insect attacks were observed.

Runoff from the shelterwoods was predicted to increase less than from clear-cut areas but the observed differences were small. This could result from the short period studied and the possible negative effects on tree vitality from elevated groundwater levels. The remaining trees in the shelterwood did not have the time to react to the new situation by increasing transpiration. Increasing growth could also have been hampered by the age of the stands, which were around 100 years old. With a younger and more vital forest stand, a stronger reaction could be expected since the trees have to have the ability to increase their canopy size. These young stands would also influence leaching and streamwater quality by increased water and nutrient uptake. Indications of this could be observed during the second two-year period while initially increased concentrations partly turned to smaller changes in the second two-year period in the Uppland shelterwoods. Leaching was always higher in shelterwoods than under forested conditions, however, mainly owing to the increase in discharge in the former. Larger water flows mitigated increases in chemical concentrations, although amounts of chemical substances were higher. After this first four-year period, the benefits of shelterwoods on water conditions could not be shown. Indications of beneficial developments, with further implications for the effects on water and chemical conditions, however, were observed at vital parts of the stands, which were not dominated by spruce.

ACKNOWLEDGEMENTS

The National Board of Forestry, the Swedish Council for Forestry and Agricultural Research and the Swedish Forest Research Foundation supported research.

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PAST AND FUTURE ATMOSPHERIC CARBON GAS (CO_2, CH_4) EXCHANGE IN BOREAL PEATLANDS

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SUMMARY

This paper integrates the results from several recent studies on carbon cycling in boreal peatland ecosystems in Finland. The emphasis is on the exchange of greenhouse gases CO_2 and CH_4. Time series of global warming potential is calculated for a 600 hectare paludifying landscape over the Holocene in order to estimate the atmospheric impact of paludification. The delicate relationship of peatland carbon balance on climate variations is demonstrated by two case studies: a fen ecosystem in a wet year and a bog in a year with an extraordinarily dry summer. The future carbon balance scheme of boreal peatlands is briefly discussed.

Keywords: climate change, CO_2 exchange, CH_4 emission, bog, fen, Holocene, peat accumulation.

INTRODUCTION

Climate change is predicted to affect the atmospheric impact of boreal peatlands in many ways. Primary productivity would increase owing to increased temperature and atmospheric carbon dioxide (CO_2) content (e.g. Melillo *et al.*, 1990), which would enhance carbon (C) sequestration from the atmosphere. More C would thus accumulate in raised bogs, if precipitation stayed at the present level (Tolonen & Turunen, 1996). On the other hand, anaerobic decomposition could increase with more substrates supplied, and CH_4 emissions might rise (Guthrie, 1986; Dacey *et al.*, 1994; Hutchin *et al.*, 1995). An increase in temperature has been shown to promote plant and microbial respiration, which could turn northern peatlands from sinks to sources for atmospheric CO_2 (Billings *et al.*, 1982; Lashof, 1989; Melillo *et al.*, 1990). A possibly decreased water table would result in higher decomposition (Billings *et al.*, 1982; Silvola *et al.*, 1996), with more CO_2, but less CH_4 released from peat (Nykänen *et al.*, 1998). If water levels remain permanently low, however, increased tree and scrub growth could compensate for these respiratory losses, mainly as a result of increased above- and below ground litter production (Laine *et al.*, 1996; Laine and Minkkinen, 1996). In all, these forcing agents have antagonistic effects on gas fluxes, carbon balance and thus the atmospheric interaction of peatlands.

Boreal peatlands manifest great spatial and temporal variability in the atmospheric exchange rates of the carbon containing gases, e.g. CO_2 and CH_4. Much of the spatial variability in gas exchange is attributable to topographic microvariation in peatlands: a mosaic formed by hollows, lawns and hummocks (Bubier *et al.*, 1993; Saarnio *et al.*, 1997; Alm *et al.*, 1997). Although the spatial net ecosystem production seems to be linearly related to relative hummock height above the water table, lawns and hollows with varying vegetation obey their own rules both in CO_2 balance (Alm *et al.*, 1997) and CH_4 efflux (Saarnio *et al.*, 1997). In addition to biotic factors, these flux rates, reflecting the basic functions of a mire ecosystem, are sensitive to annual or diurnal changes in solar irradiation, temperature and moisture. Irradiation controls the photosynthetic production, while changes in temperature and moisture affect the rate of peat decomposition.

At short time scales, hours, days, or years, the gas exchange rates show most prominent dynamics, and there the "risks" in future gas balances with changing climate can be found. Variations in climate, wet summers or summer droughts are immediately reflected in the rates of CO_2 uptake, heterotrophic respiration or CH_4 release. In addition to the short-term responses, a perspective over longer time scales of gas and matter balance, especially the Holocene history of C accumulation and CH_4 release, is needed as a background to understand the present climatic interactions of boreal peatlands.

In this paper gas exchange data from several previous studies are used to illustrate the climatic sensitivity in peatland carbon balance and methane release. Secondly, knowledge of long-term C accumulation and CH_4 release from certain mire site types is applied in a simulation for bog development, and the role of landscape paludification history in mediating the Holocene atmosphere. The possible future atmospheric interactions of boreal peatlands are discussed on the basis of these findings.

MATERIALS AND METHODS

Study Sites

A raised concentric bog complex near the southwest coast of Finland, Reksuo (60°30'N, 23°16'E), was employed as a model site for three-dimensional reconstruction of mire growth and landscape CH_4 release. The region lies in the southern boreal zone of the European phytogeographical system (Ahti *et al.*, 1968). The mire is concentrically domed, largely natural and supports ombrotrophic vegetation (Korhola, 1992). In the years 1989–1993, morphometric measurements and peat analyses were made at 120 points along a network of transects. The samples for radiocarbon dating and bulk density analyses were taken from 16 peat columns using a box sampler (80 x 80 x 1000 mm) for the upper (~1 m) and a Russian peat sampler (85 x 500 mm) for the lower peat section. A total of 44 ^{14}C dates was obtained for the reconstruction at the Helsinki University Dating Laboratory. The dates were converted to calendar years using the program CALIB 3.0.3 (Stuiver & Reimer, 1993).

Methane release and carbon dioxide exchange were studied intensively at two sites, a minerogenic fen (Salmisuo, 62°47'N, 30°56'E) and an ombrogenous bog (Ahvensalo, 62°50'N, 30°53'E), in eastern Finland. The vegetation for the fen site is described in more detail by Saarnio *et al.* (1997) and by Alm *et al.* (1999b) for the bog site.

Modelling of Bog Growth and Atmospheric Interaction

The model of bog growth incorporates the history of both lateral expansion and vertical peat accumulation. The mire was divided into three functional entities: *plateau*, *slope* and *lagg*. Each entity was considered separately in this approach because of the inherent differences in nutrient supply, apparent rate of peat accumulation and organic matter mineralization rate. The lateral growth rate of the peripheral fen lagg, and thus also mire expansion, was calculated as a function of up-slope gradient of the bog basin. The rates were derived from the mire bottom radiocarbon dates and topography of the mire basin, as elevation per distance according to data presented in Korhola (1992, 1994). Grid layer techniques of ARC/INFO (Geographic Information System, GIS) were applied in the modelling of mire growth, see Korhola *et al.* (1996). The annual CH_4 emissions for each simulated mire entity as given for different mire site types by Martikainen *et al.* (1995) were adopted. The CH_4 release rate for fen was 30 g CH_4-C $m^{-2}yr^{-1}$, and 3.8 g for bog, respectively.

GWP (Global Warming Potential in CO_2 equivalents) is a problematic indicator of the atmospheric interaction of natural ecosystems and calls for several simplifying assumptions. GWP was derived from the simulated time series of Holocene development of paludification in the present Reksuo basin. Peat accumulation was assumed to dispose of atmospheric CO_2 and generate a negative (cooling) effect, while emission of CH_4 results in positive (warming) GWP. The value of GWP was calculated for each time step in the simulation as a difference of estimated actual rate of C accumulation (ARCA) and CH_4 release in the landscape (Alm, 1997).

Measurements of CH_4 Release

Closed aluminium chambers (volume 54 dm^3) were used for the CH_4 measurements at Salmisuo fen. Four 40 ml samples were taken using polypropylene syringes during a 20—30 minute sampling period. Gas concentrations were determined within 6 hours at Mekrijärvi Research Station using a Shimadzu GC-14-A (1.8 m Haye Sep Q 80/100 mesh packed metal column) equipped with a flame ionization detector. The flux was calculated from the linear change in the headspace CH_4 concentration, and extrapolated over the hour.

Measurements of CO_2 Exchange

A static chamber technique was applied in measuring the CO_2 surface flux at Salmisuo. Net CO_2 exchange (net ecosystem production, NEP) was measured under prevailing irradiation, and community dark respiration (R_{TOT}) in full darkness. The chamber (59 cm x 59 cm x 18 cm, total airspace 66.8 dm^3) was made of polycarbonate and equipped with a battery-operated fan and a thermostat connected to a cooling apparatus that maintained the headspace air temperature to within approximately 1°C of the ambient temperature. Each dark measurement was made next to the light measurement. The chamber was removed between measurements in order to permit equilibration of the gas concentration.

During the measurements, the CO_2 mixing ratio in the headspace of the chamber was monitored with a portable infrared gas analyzer (LCA-2, Analytical Development Company, England). The gas-mixing ratio was converted to a flux estimate using the linear or near-linear change in headspace CO_2 mixing ratio during the 150—210 second measuring period, and extrapolated over the hour.

Experiments on CO_2 Enrichment

Effects of raised CO_2 on CH_4 release were studied both in glasshouse experiments (Saarnio *et al.,* 1998, Saarnio & Silvola, 1999) and in the field (Saarnio *et al., in press*). For both glasshouse experiments, 36 intact peat monoliths with living vegetation were cored from Salmisuo fen. For the first experiment, the monoliths were cored from lawn dominated by *Carex rostrata* and for the second experiment from lawn with *Eriophorum vaginatum*. The monoliths were distributed into four glass houses and grown at simulated boreal peat temperature conditions generated by refrigerators. Two of the glasshouses had ambient concentration of CO_2 (360 ppm) and in the other two it was kept at doubled concentration (720 ppm).

The effect of raised CO_2 on CH_4 release and CO_2 exchange were studied in the field over a two-year experiment in an *Eriophorum* lawn in Salmisuo. The target concentration of CO_2 in the field was 560 ppm. During the snow-free periods, CH_4 release and CO_2 exchange were followed using chamber techniques described above. Snow time gas releases were determined using a snow pack diffusion method (Sommerfeld *et al.,* 1993, Alm *et al.,* 1999a).

RESULTS

Interaction of Raised Bog Reksuo and the Atmosphere

The paludification history of the landscape presently occupied by Reksuo mire started from a single central locus at the lowest elevation value around 8,600 years ago (Fig. 1). According to the radiocarbon data from basal peat, lateral extension of the mire margin was fastest between 4500 and 3000 BP, after which there was a decline in the rate of expansion (Korhola, 1992). The rate of volumetric extension of Reksuo was very slow in the early Holocene until 3000 BP, although by this time it had reached over 80% of its present mire area (Korhola *et al.,* 1996). As late as 4500 BP, the mire was a thin minerotrophic fen, and only after that about 5.5 metres of peat accumulated in the deepest point of the central plateau. Results clearly suggest that about 50% of the current landscape C stock at Reksuo was accumulated during the last 3000 BP during ombrotrophication (Fig. 2). During the early phase between 8600 BP and 3000 BP, when *Carex* fens dominated, carbon accumulation in the Reksuo was primarily controlled by topography of the basin and consequent variations in the horizontal growth rate.

Under the simulated mire development, 239x10^6 kg organic matter decayed, corresponding to 120x10^6 kg C. Area-based estimate of CH_4 emission from the mire complex over the same period was 334x10^6 kg CH_4-C, i.e., about three times more than the peat loss. This apparent excess of methane can originate in the anaerobic part of the vascular plant rooting zone owing to the production of easily soluble root exudates and fine root litter.

Present Annual C Balances

Carbon balance was calculated for hummock, lawn and hollow communities at an oligotrophic fen in 1993 (Alm *et al.,* 1997) and a poor bog in 1994 (Alm *et al.,* 1999b). The study years represented two extremes: in 1993 water tables in the fen stayed high throughout the growing season, whereas in 1994 the bog experienced a drought of two months in May-June. Carbon balance in the fen during the wet year was strongly positive (Fig. 3), net gain varying from 32 g m^{-2} to 73 g m^{-2}, depending on the microsite. The drought and lowered water tables caused increased oxidation and reduced primary production, resulting

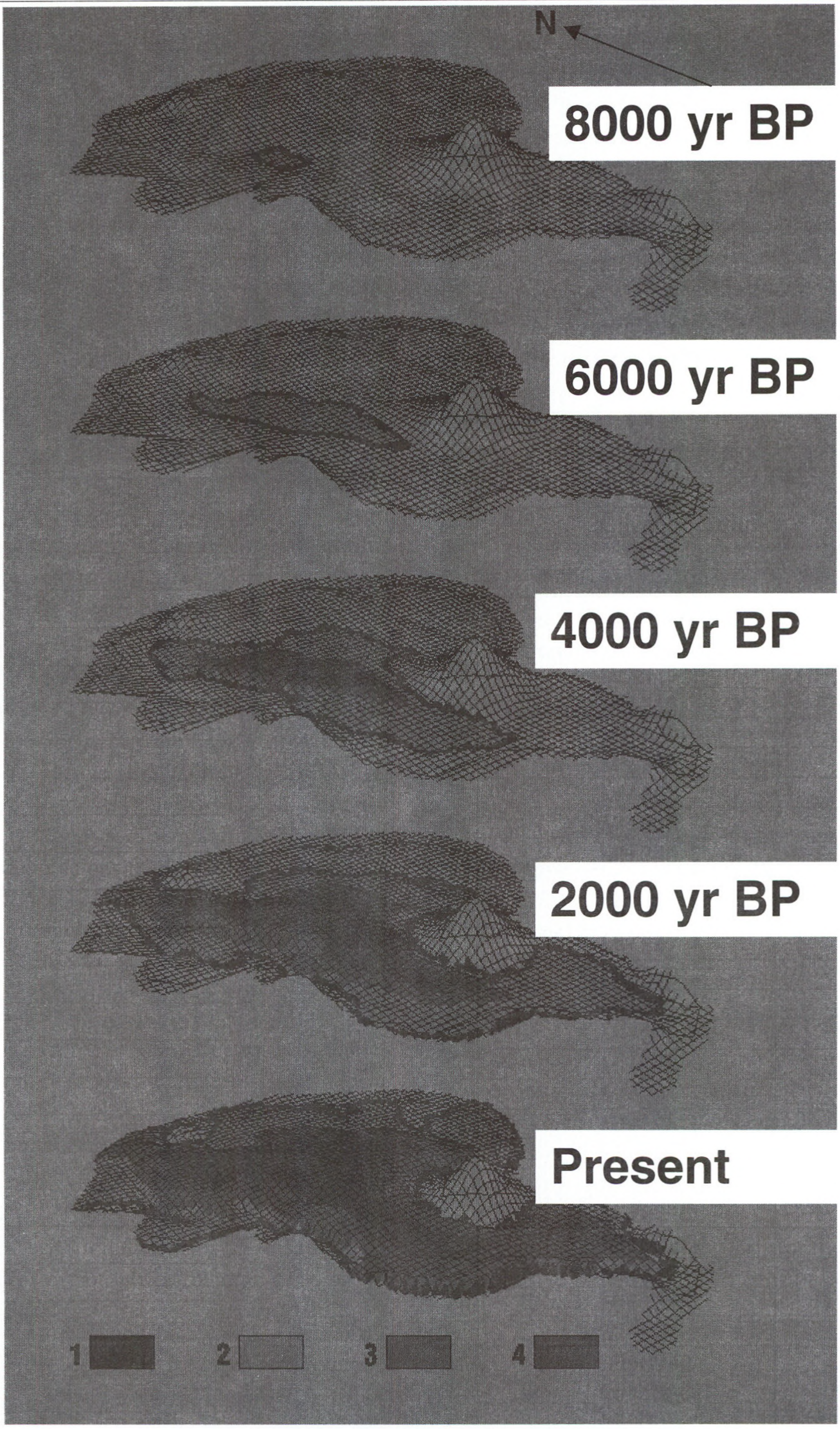

Fig. 1: Simulated paludification history of the landscape presently occupied by Reksuo *bog. The areas of lagg (1), fen (2), marginal slope (3) and* Sphagnum *plateau (4) are plotted at various instances during the Holocene from 6000 BP to present. The elevated form near the centrum of the bog represents upland soil. The thin line around the 50 m-elevation grid marks the present domains of* Reksuo*. The North arrow shows the orientation of the map. Vertical topography is exaggerated by a factor of 50. Originally published in the Journal of Quaternary Science (Korhola* et al.*, 1996).*

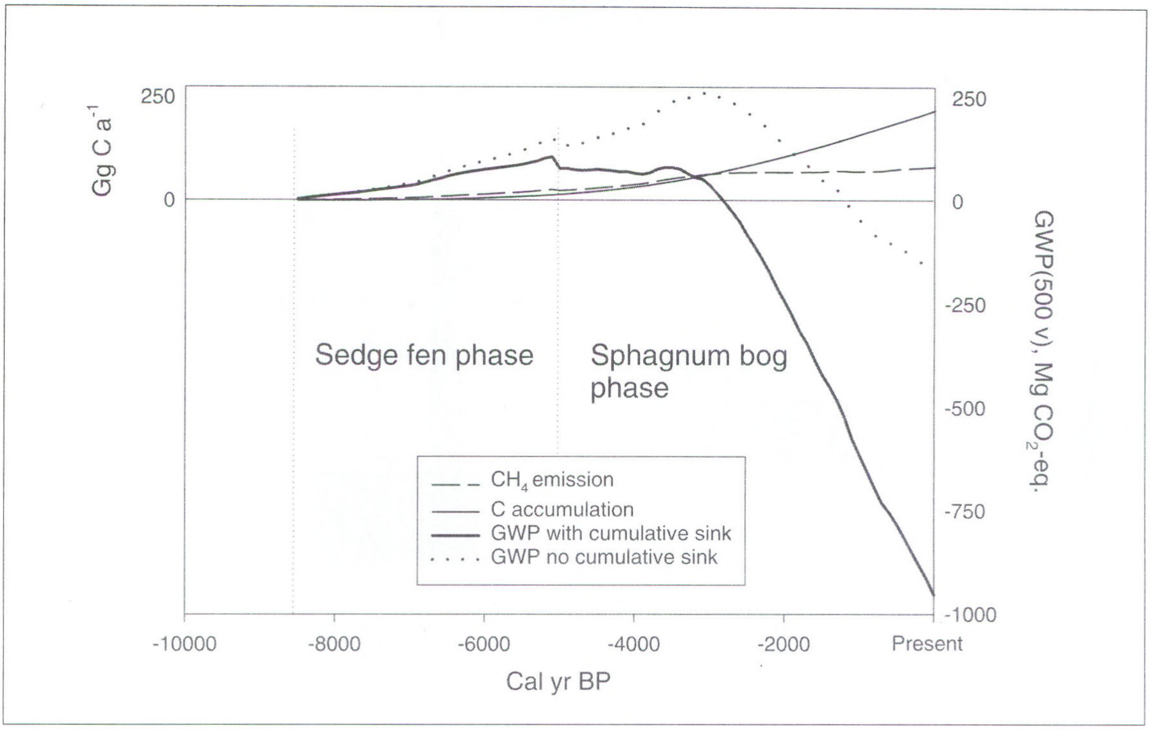

Fig. 2: Carbon accumulation and CH_4 release in a raised bog (Korhola et al., *1996). Global warming potential (GWP) as CO_2-equivalents calculated using IPCC 500-yr coefficient for methane under assumption of no cumulative atmospheric CO_2 sink (oceans replace the CO_2 drawn from the atmosphere, upper dotted line) and with a cumulative atmospheric sink in peat (lower dotted line).*

in a C loss that varies from 4 g m^{-2} in wet lawns to 157 g m^{-2} in hummocks. Winter emissions (Alm *et al.,* 1999a) of 31 g C m^{-2} in the bog and 57-84 g m^{-2} in the fen, and estimated leaching of 6-7 g m^{-2} were included in the balances.

Raised CO_2, Primary Production and Decay

Elevated CO_2 concentrations increased CH_4 efflux rate by a maximum 10–20% in monoliths growing either *C. rostrata* and *Sphagna* (Saarnio *et al.,* 1998) or *E. vaginatum* and *Sphagna* (Saarnio & Silvola, 1999). In the field, the increase in CH_4 release was about 15% for the whole study period, May 1996–July 1998 (Saarnio *et al., in press*), Fig. 4. One indicator of enhanced decomposition was probably the increased total ecosystem respiration under raised CO_2 in Salmisuo. In contrast, net photosynthesis seemed to remain unchanged (unpublished results).

DISCUSSION

Holocene History of Atmospheric Interaction

The Holocene evolution of C accumulation and CH_4 emission in the landscape that is presently called the Reksuo bog complex, Figures 1 and 2, is used to illustrate the atmospheric effect of paludification after the last glaciation. The effect of C accumulation can be assumed to be cumulative in the calculations; CO_2 drawn from the atmosphere is set aside as carbon in peat, the sequestration inducing an immediate cooling effect in the atmosphere. This assumption may actually not hold, because degassing of oceanic CO_2 might compensate for the atmospheric CO_2 loss in peat even within decades (Walker, 1993). Both marginal conditions are indicated in Fig. 2. Anaerobic decay of the deposited C compounds at the same time produces CH_4 that, when emitted to the atmosphere, has a warming effect owing to the molecule's ability to absorb energy from the infrared radiation reflected

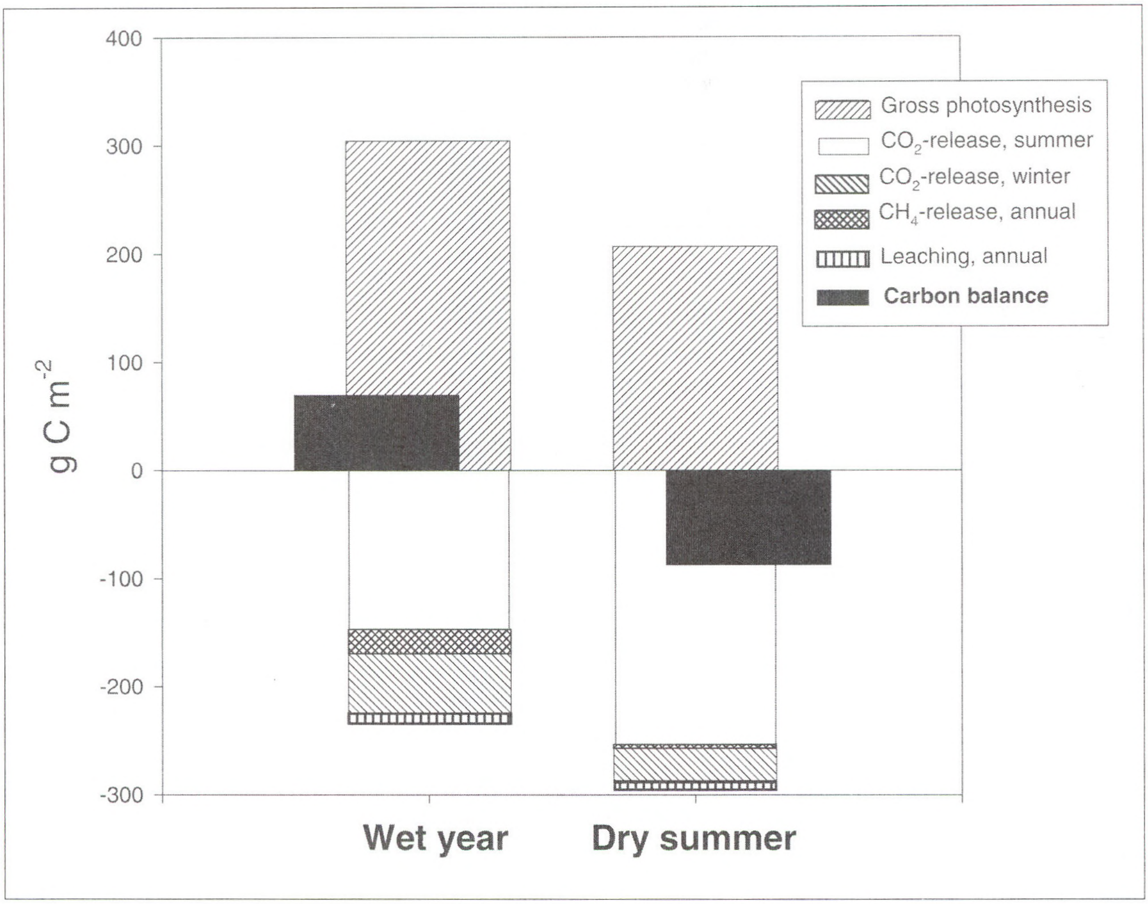

Fig. 3: Comparison of C balance components in a fen during a wet year (Alm et al., *1999a, 1999b, Saarnio* et al., *1997) and in a bog in a year with a 2 months summer drought (Alm* et al., *1999b).*

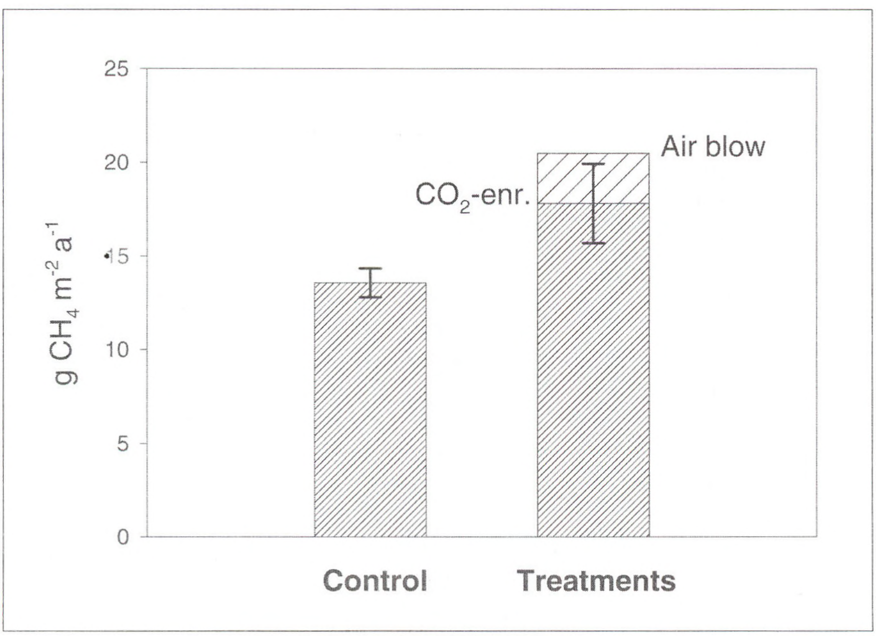

Fig 4: Estimated methane emission with CO_2-enrichment (target 560 ppm) ± S.E., compared with ambient control for 1997 in Salmisuo fen (Saarnio et al., in press*). The measured value in the CO_2-treatment apparently consisted of the effects of both CO_2 and air blow by the Free-Air Carbon dioxide Exposure (miniFACE).*

from the earth's surface (Rodhe, 1990). The net warming effect of a CH_4 molecule is estimated to be 4.6—8.8 times larger than that of CO_2 (IPCC, 1992), the figures holding for a 500 year time window. If the time series of GWP is calculated for the landscape over the Holocene, the CH_4 emission rate (in CO_2 equivalents) can be compared with the rate of CO_2 sequestration at each time step. The GWP history of Reksuo (Fig. 2) illustrates the changing atmospheric role of a domed bog. During the phase of fen domination that lasted about 6000 years, the mire complex exerted a net warming potential of 53 kg carbon dioxide equivalents (CO_2-eq) per square meter. Ombrotrophication, which started 5000 years ago in Reksuo, soon levelled the increase in positive GWP of the landscape, and about 2500 years ago the GWP turned negative, when the sink of CO_2 exceeded the warming effect of the CH_4 emissions.

The expansion of boreal mires has evidently had an antagonistic atmospheric effect during the Holocene: warming by increasing landscape methane release and cooling by C accumulation. Mire fires have contributed to the atmospheric impact by re-releasing C in carbon gases, mainly CO_2 and CH_4. In some areas the C losses could have been about 50% of the long-term apparent rate of carbon accumulation (LORCA), (Pitkänen *et al.,* 1999). Most of the fires occurred prior to the onset of the moist subatlantic climate period 2500 BP. Signs of fires were not detected in peat at Reksuo.

Until present, the net cooling effect per square metre at Reksuo bog has been 224 kg CO_2-eq, while the earlier net warming effect was only 24% of this amount (Alm, 1997). Thus, if it is assumed that the atmospheric CO_2 concentration decreased owing to a terrestrial sink during the Holocene, warming by the CH_4 emissions could not compensate the cooling by C accumulation either during the last 2500 years, or when summed over the whole bog history. It can thus be deduced that most present bogs have a net cooling greenhouse effect, while fens still contribute to the natural atmospheric warming in spite of their role as a carbon sink.

Future C Balance and Climatic Forcing

Lowered water tables cause increased CO_2 release rates from peat (Moore & Dalva, 1993; Silvola *et al.,* 1996), and are reported to cause a net C loss in natural peatlands (Shurpali *et al.,* 1995; Waddington & Roulet, 1996; Carroll & Crill, 1997). On the other hand, there are indications that a long-term drop in water levels would cause increased C accumulation owing to increased litter-fall and fine root production by the trees in drained peatlands (Laine and Minkkinen, 1996). Nutrient-poor bogs, however, can support only limited tree growth, and if natural droughts appear discontinuously, i.e. growing seasons with low water tables alternate with seasons of higher precipitation and water tables, tree growth would probably be suppressed during the wet years, as has happened in the present naturally treeless peatlands. Heavy C losses from the peat would still occur during the dry periods (Alm *et al.,* 1999b).

When the 1994 June–August precipitation sum in eastern Finland was compared to the long-term averages calculated for the same period, it appeared that similar or more severe droughts occurred only in 2 cases (6%) during the 1961—1993 reference period (data from the Finnish Meteorological Institute). Dry summers have been rare during the past three decades, but the results from Ahvensalo bog clearly demonstrate that severe C losses can occur even in years with an average temperature close to and precipitation well above the long-term means. These losses can, however, be compensated in more humid years, when the annual accumulation greatly exceeds the long-term averages (Alm *et al.,* 1997). The ratio between "moist" and "dry" summers should be at least 4/1 to retain a C balance with accumulation that merely compensates for the losses of the order of 100 g C m^{-2} yr^{-1}, and 5/1 to maintain net C accumulation at an average long-term rate of 25 g C m^{-2} yr^{-1} (Alm *et al.*, 1999b).

Annual distribution of drought and precipitation has a key role in the carbon balance of boreal peatlands. Wintertime precipitation is predicted to increase in the Fennoscandian region, while temperatures are assumed to rise, which could mean a longer growing season in the north. Water tables should be high after spring thaw, indicating a positive NEP by *Sphagnum* mosses during early summer. This might thus be the only potentially high C accumulation period before a summer water table draw-down. A lower light regime would restrain the photosynthesis later in summer and autumn.

An increase in atmospheric CO_2 in present summer temperatures seems to increase ecosystem respiration (unpublished results). Experiments under raised CO_2 suggest that there will only be a modest increase in the CH_4 release from boreal peatlands. This result can probably be explained by differences in temperature. In temperate conditions, Dacey *et al.* (1994) reported an increase of 80% in a wetland experiment, and Hutchin *et al.* (1995) an increase of 150% for peat cores. A low temperature regime in the boreal zone thus seems to retard the anaerobic conversion of atmospheric CO_2 into CH_4 (Saarnio *et al., in press*).

Exceptionally early and thick snowfall would insulate the peat surface from freezing and promote early winter CO_2 release from peat, while both soil temperature and CO_2 losses could be decreased by a late or a thin snow layer owing to a warmer winter climate. Relative to annual C balances, however, winter CO_2 emissions probably do not change as much as do the fluxes during the growing season, driven by interseason climatic variability (Alm *et al.,* 1999a).

Conclusions

Boreal peatlands are sensitive ecosystems, reacting instantly to variations in climate. Until now the climate during the Holocene, on average, has favoured peat accumulation and maintained a boreal C sink. The atmospheric cooling effect associated with this sink in the past and also in the future depends on the release of CH_4. Atmospheric changes most probably bring new concerns. Past peat accumulation rates cannot be taken for granted even in the near future. Increased CO_2 concentration could accelerate the C turnover rate in peatlands both in the aerobic and anaerobic regime. Changes in precipitation, irradiation, warming and evapotranspiration certainly affect hydrology in the boreal zone. Changing climate could negate the peat C sink and promote the release of radiatively active gases in natural mires. Thus, the potential future gas balance for natural mires, instead of the past carbon sink scheme, must be considered as a reference for the atmospheric effects exerted by peatland exploitation, such as peat mining, forestry, cultivation or restoration. There is a need for improved knowledge and simulation models dealing with the responses of biogeochemical processes to the predicted atmospheric changes in the boreal zone.

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