

# Ecohydrological and vegetational changes in a restored bog and fen

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The vegetation of two boreal mires drained for forestry was studied prior to and after restoration (removal of tree stand and filling in of ditches). The restoration induced a rapid rise in the water table level and caused relatively rapid changes in plant species composition and cover. On the minerotrophic fen site, the number of forest species declined and the cover of *Eriophorum vaginatum* increased five-fold, reaching over 50% cover in three years. On the ombrotrophic bog site, the terrestrial lichens disappeared, while the cover of *Empetrum nigrum*, *Calluna vulgaris*, *Eriophorum vaginatum*, and *Sphagnum balticum* increased. Changes in water table level and vegetation indicate a change towards a functional mire ecosystem.

Keywords: drainage, hydrological change, mire vegetation, peatland restoration

## Introduction

Europe, with ca. 20%, or 96 million ha, of the land area covered by peat, is the most peat-covered part of the world (Lappalainen 1996). Large-scale drainage of mires for agriculture (30% of the original area), forestry (15%) and peat harvesting (6%) has reduced the area of natural mires to less than half (Joosten 1997). In Finland, mires have been drained primarily for forestry purposes: almost 60% of the original area (e.g. Aapala *et al.* 1996). In several mire types, vegetation has started to develop towards forest vegetation (Laine *et al.* 1995) and tree growth has increased. Some drained mires, however, have been unsuited for forestry. Some

of these mires as well as those belonging to key biotopes, threatened site types, or regionally important mires are favorable sites for active restoration to increase regional diversity of mire habitats (Aapala *et al.* 1996). Marginal parts of mires drained for forestry may be restored as buffer zones to diminish the effects of forestry on waterways (Sallantausta *et al.* 1998).

The aim of active peatland restoration is to re-establish mire plant communities and the functioning mire ecology that was present before drainage (Fojt 1995). Active restoration means that ecological and vegetational changes are induced or accelerated by blocking ditches with dams or filling them to elevate the water table to the pre-drainage level. In forested mires, trees are

also often cut to diminish biological evaporation (Heikkilä & Lindholm 1994). Restoration of forested mires is a relatively recent nature conservation operation; thus, few studies exist concerning restoration-driven vegetation succession (Heikkilä & Lindholm 1997, Komulainen et al. 1998).

In this study, two different mire sites were actively restored: ditches were filled, and trees were cut. Our objectives were to investigate how these measures, aimed at reverting the secondary succession induced by forest drainage, would influence (1) ecohydrology, and (2) vegetation composition and cover during the first three years after restoration. Our hypothesis was that if we were able to restore the water table levels close to the soil surface, mire vegetation would start to spread spontaneously by wind-, animal- and waterflow- mediated dispersal, and also from the existing small seedbank (Jauhiainen 1998).

## Materials and methods

### Study sites

The study sites are located in an eccentric raised bog region (Ruuhijärvi 1983) about 60 km north-east of Tampere in southern Finland. Mean annual precipitation in the area is 709 mm, and mean daily temperature in July is 15.3 °C and in January –8.9 °C. The effective temperature sum between 1961 and 1990 using a +5 °C threshold is 1060 degree days (according to the Finnish Meteorological Institute, Juupajoki station). Precipitation sums for the period May–September in 1994 (before restoration) and 1997 (after restoration) were 286 mm and 359 mm, respectively.

The minerotrophic Konilampi mire site is located on the northern part of Hanhisuo mire, (61°48'N, 24°17'E, 155 m a.s.l.), between an esker and a lake (Hanhijärvi). This tall-sedge pine fen (*sensu* Laine & Vasander 1996) was drained for forestry in 1955 by using a ditch spacing of 50 m and ditch depth of ca. 90 cm. Ditch maintenance was carried out in 1965 and 1988. At the time of restoration, the mire supported a pine-dominated (*Pinus sylvestris*) stand with some spruce (*Picea abies*) and birch (*Betula pubescens* and *B. pendula*). The total

stand volume was about 100 m<sup>3</sup> ha<sup>-1</sup>, and the average tree age was 85 years. Ground vegetation was composed of common forest shrubs, such as *Vaccinium myrtillus*, *V. uliginosum*, and *V. vitis-idaea*, with some mire dwarf shrubs such as *Ledum palustre* and *Betula nana*. Moss layer was composed of *Sphagnum angustifolium*, *S. magellanicum*, and *S. russowii* with the forest moss *Pleurozium schreberi*.

The ombrotrophic Viheriäisenneva mire site (61°51'N, 24°14'E, 160 m a.s.l.) was originally a low-sedge bog. The site was drained for experimental purposes in 1955 with ditch spacing of 30 m and ditch depth of 90 cm. Ditch maintenance was done in 1988. In 1966, the nearly treeless site was planted with pine seedlings (*Pinus sylvestris* and *P. contorta*), but in 1994 the tree layer was still stunted, most seedlings being shorter than 1 m. The stem number was about 4400 ha<sup>-1</sup>. The site had a mosaic-like vegetation structure consisting of hummocks, lawns, and hollows. Before the restoration, the field layer was characterized by dwarf shrubs *Calluna vulgaris*, *Empetrum nigrum*, and *Vaccinium uliginosum*, and the ground layer was dominated by *Cladonia* species and ombrotrophic *Sphagna* such as *S. fuscum*, *S. rubellum*, and *S. balticum*.

Both study sites have been intensively monitored for forest growth and yield research since the 1950s (Sarkkola & Päivänen 2001). Nomenclature follows Hämet-Ahti et al. (1998) for vascular plants, Koponen et al. (1977) for bryophytes, and Ahti (1993) for lichens.

### Restoration arrangements

Restoration was initiated in summer 1994 by measuring trees, constructing ground water wells, and establishing vegetation sample plots. On the fen site, trees (> 7 m tall) were measured for height, diameter at breast height (1.3 m DBH), and diameter at 6.0 m. Smaller trees were measured only for height. An area of 1.1 ha was rewetted, of which 0.6 ha was further clear-cut, and trees and logging slash were removed from the site. This 0.6 ha, located between two filled-in ditches with a 50-m spacing, will be referred to as the restoration site. On the bog site, the area

rewetted comprised altogether 10.5 ha. Only the height of stunted trees was recorded, and they were removed from an area of 0.5 ha (restoration area, between two filled-in ditches with a 30-m spacing).

Ditches around the restoration areas were either partly blocked with dams or filled during winter 1995. On the fen site, an additional 'feeder ditch' was excavated to direct water flow from the catchment to the restoration area.

### **Vegetation and water table**

For vegetation mapping, 12 systematically laid permanent sample plots (50 × 100 cm) on restoration sites and 9 sample plots on control sites were marked. On the restoration site the sample plots were located on four transects between the filled ditches. On the control site there were three transects between the (unfilled) ditches. In all transects the sample plots were situated 5 m apart from each other. Ground water wells were set up close to each vegetation plot.

Vegetation was determined visually as percentage cover of each species, with an accuracy of 1%. This was done at the end of July every year during 1994–1997, always by the same person (S.J.). Species that existed with a cover less than 0.5% were recorded as 0.1%. In 1997, in addition to mapping of permanent plots, vegetation was mapped from five parallel transects across the whole restoration areas of both mires to obtain a more complete view of the succession stage after three restoration years. Vegetation cover was recorded in 1-m<sup>2</sup> areas along both sides of the transects at 15-m intervals. The water table level was measured weekly during the growing seasons.

### **Chemical analysis**

Peat cores for chemical analysis were taken near each permanent vegetation plot in August 1994 before the restoration. Coring was repeated in August 1997. Cores were obtained using a volumetric sampler (area 8.3 × 8.4 cm) to a depth of 20 cm from the mire surface. They were divided into 10-cm pieces in the field and packed into

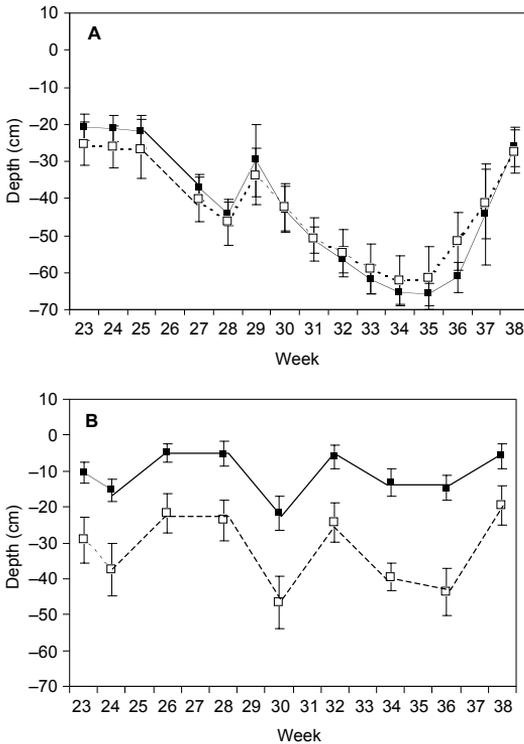
plastic bags. The peat samples were dried at 105 °C to a constant mass and ground through a 2-mm sieve. Subsamples were digested with HNO<sub>3</sub>–H<sub>2</sub>SO<sub>4</sub>–HClO<sub>4</sub> at 200 °C, and total concentrations of Al, Ca, Fe, K, Mg, Mn, and P of peat were analyzed at The Finnish Forest Research Institute, Vantaa Research Center laboratory with an ARL 3580 vacuum ICP plasma emission spectrometer (in 1994) and with an IRIS Advantage ICP spectrometer (in 1997). The pH was measured from a soil–CaCl<sub>2</sub> (0.01 M) suspension 1:2.5 v/v, and from soil water from both mires.

### **Data analysis**

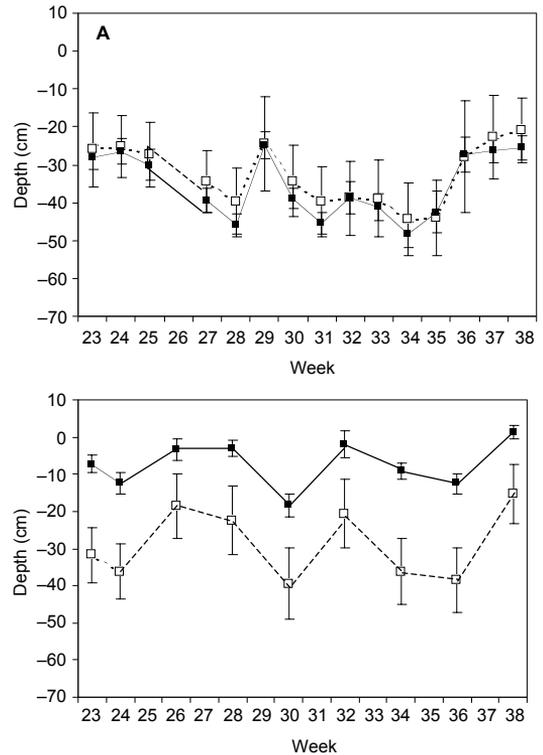
Pre- and posttreatment element concentrations in the soil (0–10 cm layer) were compared using the sign test. The same test was also applied to examine whether pretreatment concentrations in the control vs. restored sites differed from each other. The soil element concentrations on the fen vs. the bog site were compared using the two-sample Kolmogorov-Smirnov (K-S) test. Systat 10 for Windows was used for these tests. No corresponding *t*-tests were applied because element concentrations did not follow normal distributions.

Changes in species cover were analyzed using CCA (canonical correspondence analysis) of the CANOCO program, version 4.0 (ter Braak 1998). To capture the restoration-induced change, we combined data sets of 1994 (pre-restoration) and 1997 (three years after restoration). Percentage cover of species in 1994 and 1997 formed species data, and water table level (WT) and concentrations of Al, Ca, Fe, K, Mg, Mn, and P in peat were used as environmental variables. The influence of element concentrations in 10- and 20-cm peat layers on vegetation was studied. Using concentrations of the 10-cm layer produced more consistent results and are thus presented. This was likely because the concentrations of some elements, especially K and Mn, were extremely low in the 20 cm layer.

Annual changes (1994–1997) in vegetation composition at sample plot level were studied by using DCA (Detrended correspondence analysis) of the CANOCO program.



**Fig. 1.** Mean level of water table on the fen site during the growing season with error bars (SD). — **A:** 1994 (pre-restoration). — **B:** 1997 (post-restoration). Dashed line with open squares = the control site ( $n = 9$ ), solid line with filled squares = restored site ( $n = 12$ ).



**Fig. 2.** Mean level of water table on the bog site during the growing season with error bars (SD). — **A:** 1994 (pre-restoration). — **B:** 1997 (post-restoration). Dashed line with open squares = the control site ( $n = 9$ ), solid line with filled squares = restored site ( $n = 12$ ).

## Results

### Water table level and peat chemistry

In drained, pre-restoration condition (1994), the water table level (WT) on the fen site ranged between 20 and 65 cm below mire surface during the growing season, being at its lowest level in weeks 32–36 (Fig. 1A). One year after the restoration, the average WT ranged between 10 cm (early summer) and 40 cm (late summer). Near the feeder ditch, the rewetting was faster and the WT did not go lower than 20 cm, even during the driest period, but at the other end of the restored site it still descended to 40 cm. Three years after the restoration, in 1997, the WT of the restored site ranged between 5 and 20 cm (Fig. 1B).

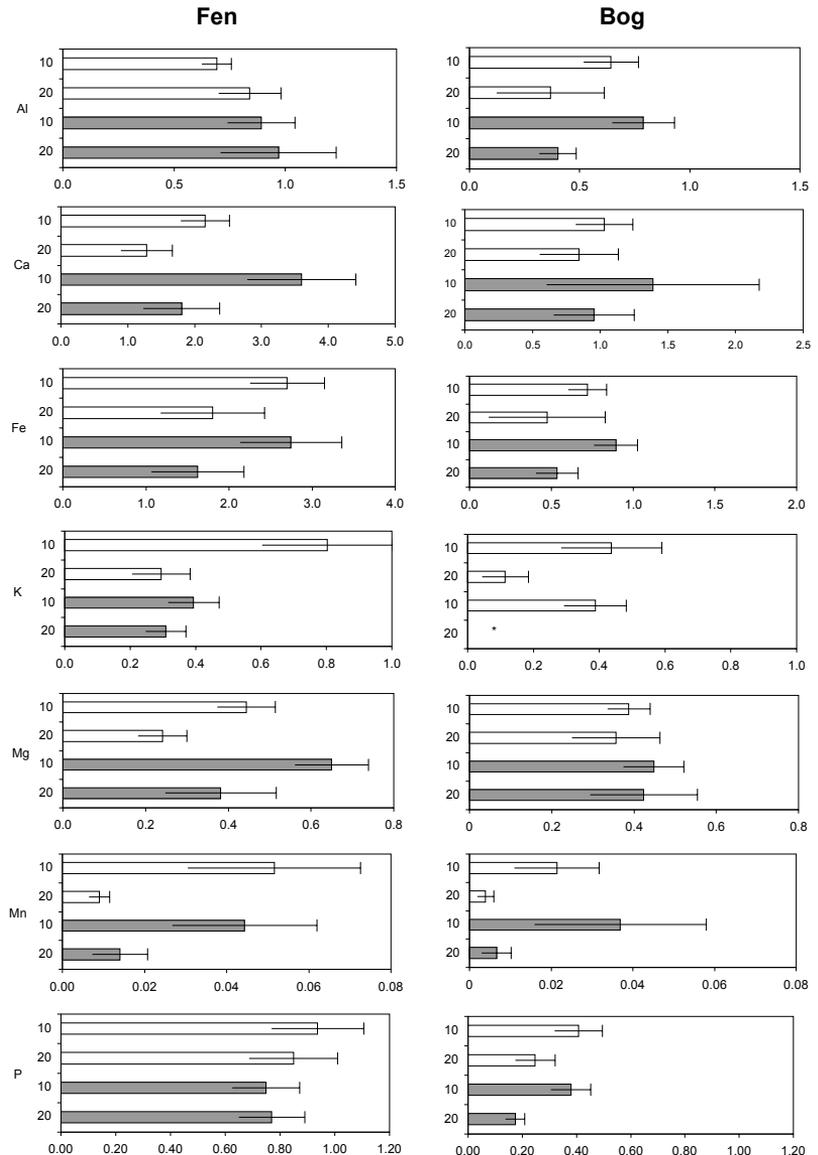
On the bog site, the prerestoration WT ranged between 20 and 45 cm (Fig. 2A). Three years after the restoration, the WT ranged from 18 cm

to a level sufficient to submerge the vegetation (Fig. 2B). On the control site, the WT remained at prerestoration levels.

Mineral concentrations in the fen peat were significantly higher than those in the bog peat (K-S  $p < 0.001$ ), except for Al and Mg (Fig. 3). Prerestoration concentrations of all elements were similar on the restored and control sites. Concentrations of all elements, excluding fen Al, decreased with increasing depth.

During the three years after the restoration, Al and Ca increased significantly in the surface peat of the fen, whereas K and P decreased (sign test  $p \leq 0.012$ ). On the bog site, Al and Fe concentrations increased ( $p = 0.007$ ). A noticeable change occurred in K, the concentration of which dropped below the detection limit in the 10–20 cm layer.

The pH of (dry) peat was very low on both sites, being 2.77 and 2.83 in the 0–10 cm layer



**Fig. 3.** Element concentrations ( $\text{mg g}^{-1}$ ) in 0 to 10-cm and 10 to 20-cm peat layers in restored sites of both mires: 1994 (open bars) and 1997 (filled bars).

on the fen site, and 2.69 and 2.75 in the bog in 1994 and 1997, respectively. The pH measured from soil water changed from 4.12 to 3.87 on the fen and from 4.77 to 3.97 on the bog site.

## Vegetation changes

### The fen site

After the restoration (tree removal, raising WT), the cover of the shrub-size *Betula pubescens*,

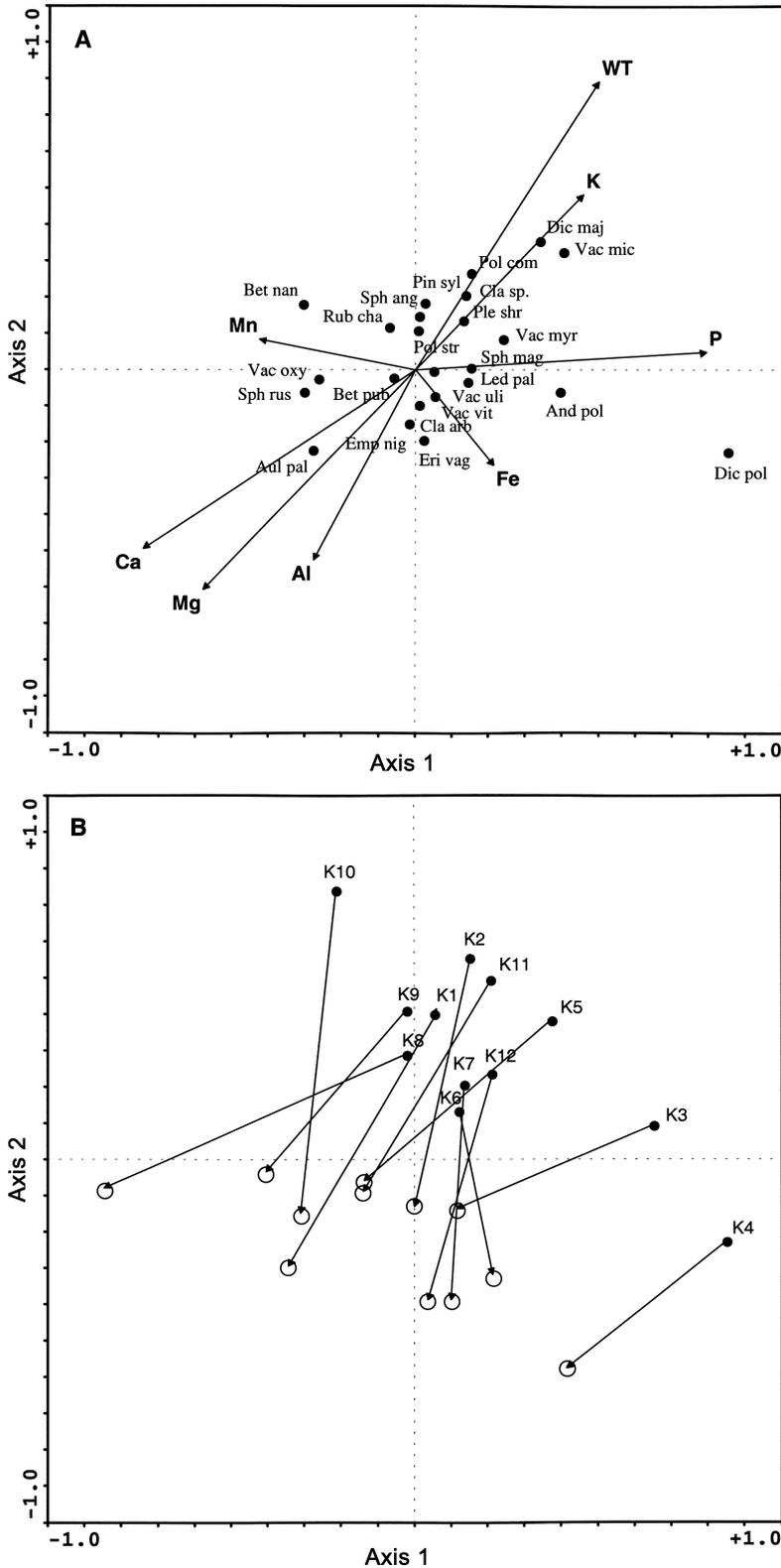
and dwarf shrubs *B. nana*, *Empetrum nigrum*, *Ledum palustre*, *Vaccinium uliginosum*, and *V. vitis-idaea* increased, whereas *V. myrtillus* declined (Table 1). The cover of *Eriophorum vaginatum* increased fivefold during the restoration, with over 50% cover in 1997. In the moss layer, the cover of *Sphagnum angustifolium* and *S. russowii* increased, whereas *S. magellanicum* decreased. Outside the vegetation plots, mire species, such as *Carex rostrata*, *Calla palustris*, and *Potentilla palustris*, among others, were found.

**Table 1.** Mean percentage cover of vegetation for restored ( $n = 12$ ) and control ( $n = 9$ ) sites in 1994 and 1997 in both mires. Species listed below the table, identified on transect lines in 1997 (see Materials and methods), did not grow in the vegetation sample plots.

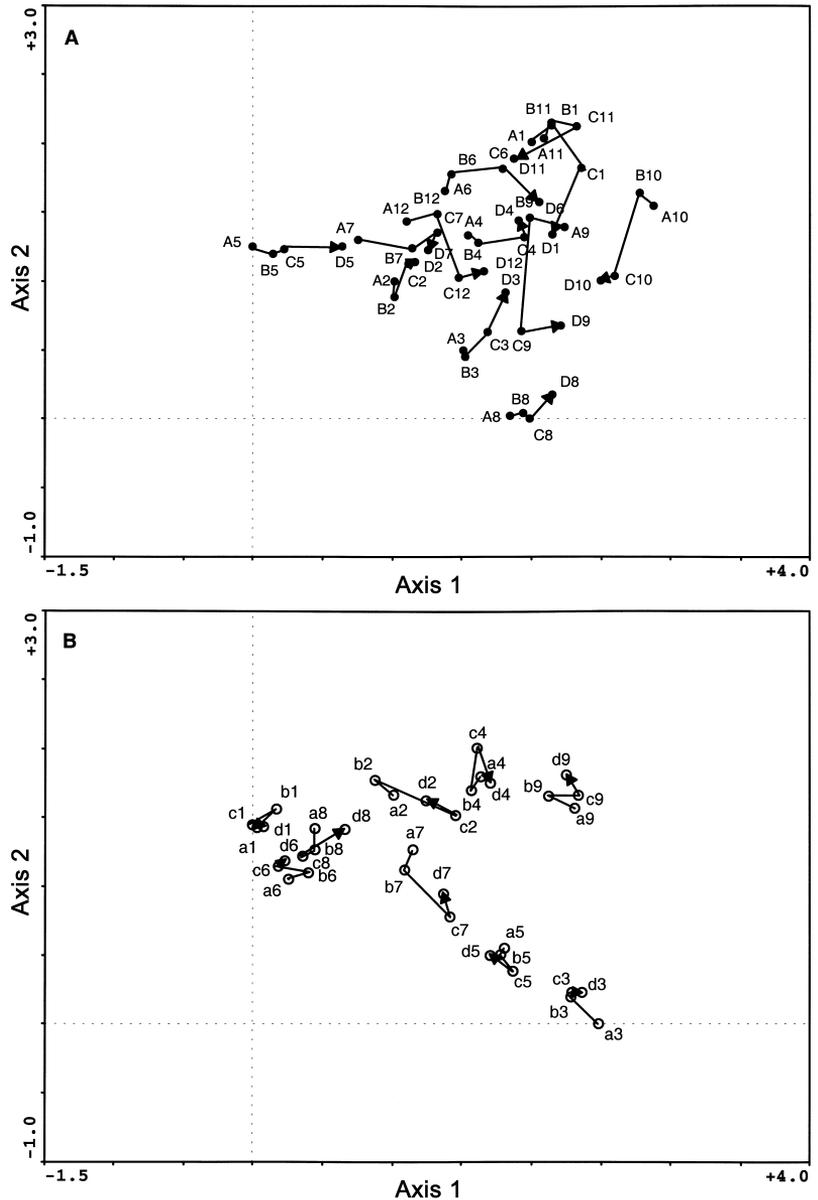
	Fen(r) 94		Fen(c) 94		Fen(r) 97		Fen(c) 97		Bog(r) 94		Bog(c) 94		Bog(r) 97		Bog(c) 97	
	Cover%	Freq														
<i>Andromeda polifolia</i>	0.4	6														
<i>Aulacomnium palustre</i>	0.1	1	1.1	1	1.1	2	2.0	1	2.7	10	0.9	8	8.9	10	0.9	8
<i>Betula nana</i>	18.8	5	1.4	3	5.0	2	1.9	2								
<i>Betula pubescens</i>	6.5	3	0.1	1	11.7	6			7.2	8	2.1	5	8.7	7	2.1	5
<i>Calluna vulgaris</i>									22.6	8	25.9	8	22.8	7	25.9	8
<i>Cladina arbuscula</i>	0.1	1			1.0	1			0.3	5	0.6	4	1.0	3	0.6	4
<i>Cladina rangiferina</i>							1.2	1	19.3	2	3.9	6	16.8	2	3.9	6
<i>Cladina stellaris</i>									0.4	4	0.8	3	0.1	2	0.8	3
<i>Cladonia cenotea</i>									0.7	3	5.6	2			5.6	2
<i>Cladonia chlorophaea</i>									1.0	1						
<i>Cladonia cornuta</i>									1.0	3	0.5	1	1.1	3	0.5	1
<i>Cladonia crayi</i>																
<i>Cladonia deformis</i>									12.1	6	20.7	3	8.2	5	20.7	3
<i>Cladonia fimbriata</i>																
<i>Cladonia sp.</i>	0.3	1														
<i>Cladonia squamosa</i>									0.5	1			0.2	1		
<i>Cladonia stygia</i>									9.5	2						
<i>Cladonia sulphurina</i>									7.8	7			3.6	6	2.0	2
<i>Dicranum majus</i>	3.0	3														
<i>Dicranum polysetum</i>	0.8	1														
<i>Dicranum scoparium</i>			6.8	7	11.2	8	2.9	2	1.7	4	0.8	6	3.8	2	0.8	6
			2.0	2	0.7	1			0.7	1						

<i>Empetrum nigrum</i>	4.1	8	22.1	8	3.9	4	5.5	4	23.9	10	46.0	6	11.5	5	17.5	4
<i>Eriophorum vaginatum</i>	11.0	12	50.5	12	4.2	7	10.7	8	4.7	11	14.2	11	1.1	9	4.9	9
<i>Ledum palustre</i>	14.3	11	20.1	11	7.8	7	8.2	7								
<i>Picea abies</i>					3.5	2	0.1	2								
<i>Pinus contorta</i>													10	1	12.0	1
<i>Pinus sylvestris</i>	1.5	1	0.7	3	0.1	1	0.4	2	0.0	1	0.0	2				
<i>Pleurozium schreberi</i>	19.9	9	22.4	5	22.4	8	28.7	9	31.4	4	30.8	4	8.9	4	11.5	3
<i>Pohlia nutans</i>													0.1	3		
<i>Polytrichum commune</i>	4.4	3	0.2	1												
<i>Polytrichum strictum</i>	3.1	4	4.0	3	5.0	3	8.3	2	5.5	5	7.0	3	2.1	2	1.0	2
<i>Rubus chamaemorus</i>	13.4	12	15.1	12	4.7	8	7.1	7	2.4	10	5.6	8	1.7	8	2.8	8
<i>Sphagnum angustifolium</i>	36.3	9	37.0	7	32.0	4	38.0	5	52.6	4	49.1	5	6.2	7	6.5	5
<i>Sphagnum balticum</i>									1.5	2	2.5	1	12.5	1	12.0	1
<i>Sphagnum capillifolium</i>											3.0	1				
<i>Sphagnum fuscum</i>									5.6	4	5.3	2	30.0	2	48.0	1
<i>Sphagnum magellanicum</i>	18.4	8	12.7	10	13.4	5	16.2	5	1.1	1	1.5	1	0.5	1		
<i>Sphagnum rubellum</i>											2.0	1	4.0	1	1.8	2
<i>Sphagnum russowii</i>	18.0	4	38.9	7	6.0	2	6.3	2	12.4	7	4.8	8	1.7	3	6.0	2
<i>Vaccinium microcarpum</i>	1.5	2			0.0	1										
<i>Vaccinium myrtillus</i>	13.1	10	3.8	10	11.9	8	11.1	9								
<i>Vaccinium oxycoccos</i>	1.0	3	1.7	6	0.8	2	1.3	2	1.3	3	0.0	1	0.2	2		
<i>Vaccinium uliginosum</i>	5.6	9	10.1	7	3.5	6	9.9	4	7.0	6	11.2	5	4.5	2	15.5	2
<i>Vaccinium vitis-idaea</i>	6.4	11	10.9	2	6.3	7	6.5	9								

Species existing along the transect lines — Fen: *Betula pendula*, *Carex canescens*, *C. echinata*, *C. nigra*, *C. rostrata*, *Calamagrostis* sp., *Calla palustris*, *Calliergon stramineum*, *Dicranum bergeri*, *Dactylorhiza maculata*, *Deschampsia caespitosa*, *Dicranella cerviculata*, *Dryopteris carthusiana*, *Epilobium angustifolium*, *Polytrichum juniperinum*, *Plagiothecium laetum*, *Polytrichastrum longisetum*, *Populus tremula*, *Potentilla palustris*, *Sphagnum fallax*, *S. flexuosum*, *S. riparium*, *S. squarrosum*, *Warnstorfia fluitans*; — Bog: *Dicranum bergeri*, *Myliia anomala*, *Sphagnum cuspidatum*, *S. fallax*, *S. majus*, *S. tenellum*



**Fig. 4.** Canonical Correspondence Analysis (CCA) for the fen site. — **A:** Species of the restored area with environmental variables. — **B:** Shifts of the vegetation sample plots of the fen in the ordination space caused by restoration. Sample plots (K1–K12) in 1994 (open circles) and 1997 (filled circles).

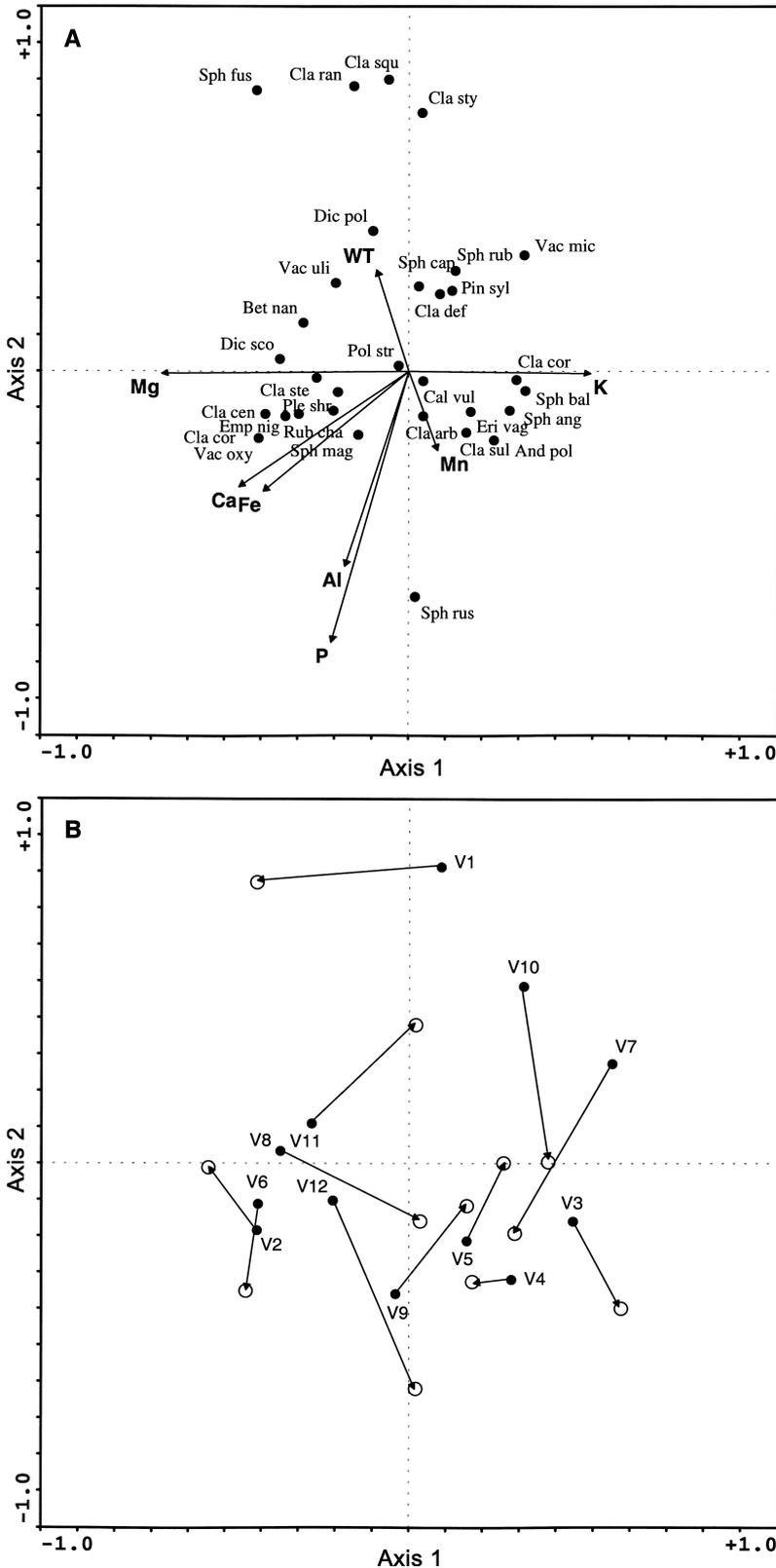


**Fig. 5.** Detrended correspondence analysis (DCA) for the fen site. Annual change in the vegetation composition of the fen sample plots indicated as shifts in the ordination space. — **A:** Restored site (letters indicate the year: A = 1994; B = 1995; C = 1996; D = 1997). — **B:** Control site (letters indicate the year: a = 1994; b = 1995; c = 1996; d = 1997). Numbers indicate the specific vegetation sample plots.

In the ordination analysis, WT together with K and the concentrations of Ca, Mg, and Al formed the main gradient on the fen site (Fig. 4A). Species, such as *Dicranum majus*, *Vaccinium microcarpum*, and *Polytrichum commune*, correlated with low WT. The eigenvalues for the first and second axes were 0.186 and 0.154, respectively, and 20.2% of the species occurrence and 51% of the species-environment relation were explained by these two axes.

The vegetation changed in accordance with the hydrological gradient between the years 1994 and 1997 (Fig. 4B).

The DCA analysis for the vegetation sample plots also showed that the vegetation composition of all the plots was changing similarly towards a species composition indicative of moist conditions (Fig. 5A). Relatively large variation was present between the vegetation plots on both the control site (2.5 units) and



**Fig. 6.** Canonical Correspondence Analysis (CCA) for the bog site. — **A:** Species of the restored area with environmental variables. — **B:** Shifts of the vegetation sample plots of the fen in the ordination space caused by restoration. Sample plots (V1–V12) in 1994 (open circles) and 1997 (filled circles).

the restored site (3.0 units), whereas within-plot variation was small on the control site, and no clear direction of change was seen (Fig. 5B).

### The bog site

The changes in the bog vegetation were not as remarkable as on the fen site. At an early stage of the restoration, the lichens *Cladina* and *Cladonia* species almost disappeared and the condition of more resistant species became poor (Table 1). Dwarf shrubs, such as *Betula nana*, *Vaccinium uliginosum*, and especially *Empetrum nigrum*, increased in cover. Moreover, the percentage cover of *Andromeda polifolia*, *Rubus chamaemorus*, and *Eriophorum vaginatum* increased. Relatively small changes occurred in the moss layer (Table 1).

Ordination analysis (Fig. 6A) showed that Mg and K formed the first environmental axis (eigenvalue 0.474), and WT and Mn formed the second axis (eigenvalue 0.335). *Cladina rangiferina*, *Cladonia squamosa*, *C. stygia*, *Sphagnum fuscum* and *Dicranum polysetum* correlated with low WT. On this site, 23.6% of species occurrence and 60.4% of the species-environment relation were explained by these two axes. Vegetation in all the sample plots had changed during the restoration (Fig. 6B) but not as uniformly into certain direction as on the fen site.

DCA analysis revealed large variation between the vegetation sample plots (Fig. 7A), which was approximately the same on both the restored and control sites (0–5 and 0–4.5 units, respectively). Sample plots with vegetation indicating dry conditions appear on the left side of the ordination diagram, while sample plots on former hollows appear on the right (Fig. 7A). After the restoration, no clear change in vegetation was apparent in the former hollows or in the driest sample plot (no. 8 in Fig. 7A). On the intermediate lawn, however, vegetation composition resembled that of the former preres-toration hollows. On the control site, the annual changes were nondirectional and generally small (Fig. 7B).

## Discussion

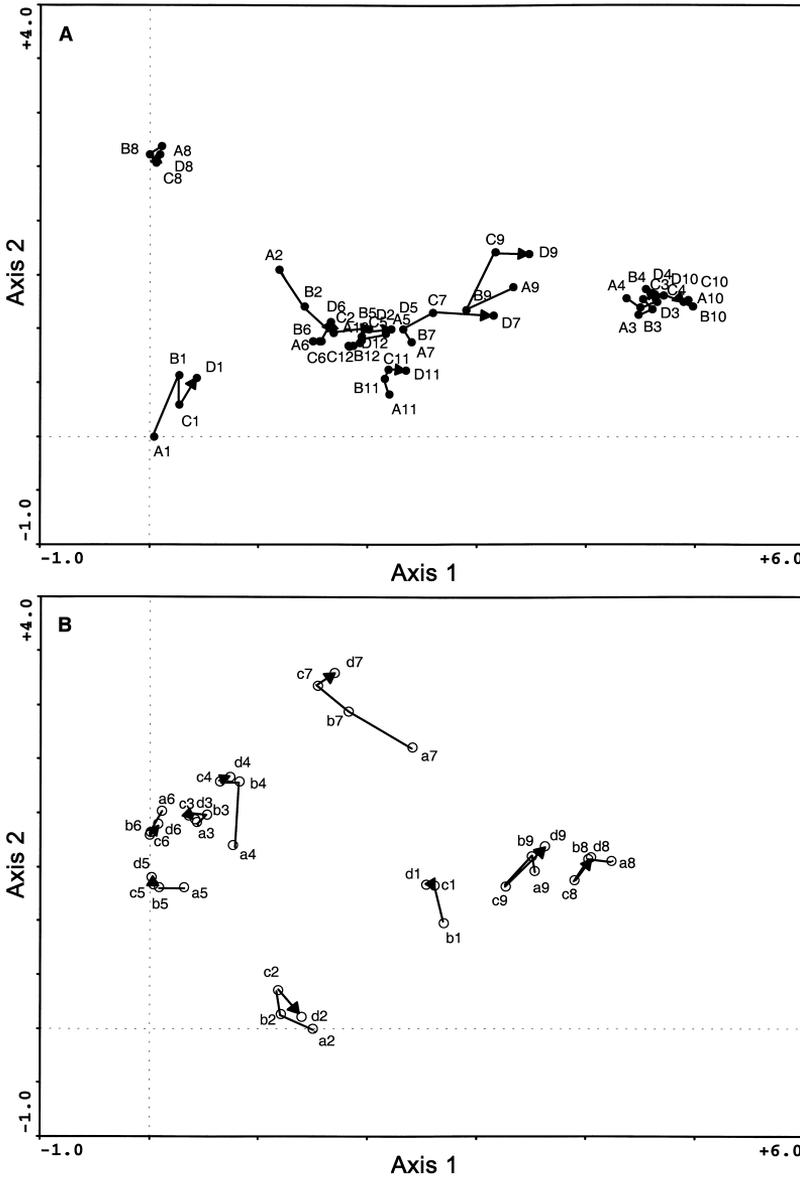
### Ecohydrological changes

Evapotranspiration of the tree stand was likely the main factor behind the large variation in WT on the fen site before the restoration (Paavilainen & Päivänen 1995). The relatively low precipitation during the growing season in 1994 (Lindholm & Markkula 1984, Reinikainen *et al.* 1984) and the density and depth of the ditches (Paavilainen & Päivänen 1995, Lundin 1999) affected the lowest WT. Successful restoration and lack of evaporation after clear cutting elevated the WT of the fen site to a level typical of pristine mires in the region (Lindholm & Markkula 1984, Reinikainen *et al.* 1984) within three years. The feeder ditch made it possible to rapidly achieve the hydrological characteristics of a minerotrophic fen.

Lacking an evaporative tree stand, there was no similar seasonal variation in WT on the bog site. The WT changes were sensitive to the amount of precipitation (Lindholm & Markkula 1984, Reinikainen *et al.* 1984). The relatively larger within-site WT variation of the control as compared with the restored site was possibly caused by the stronger mosaic structure of hummocks and lawns in the former. Damming the ditches elevated the WT of the bog to the level of a natural bog (Lindholm & Markkula 1984). Since the dry lichen-covered surface was overtaken by moist *Sphagnum* vegetation, capillary forces may keep the WT high (Reinikainen *et al.* 1984, Damman 1986), providing the prerequisite for a functioning, self-sustainable ombrotrophic bog.

Surrounding watershed, including an esker, served as the main source of mineral nutrients for the fen site. Thus, most mineral concentrations in the fen site were significantly higher than those in the bog site, which obtained its nutrients from precipitation and dust deposited from the atmosphere (Mörnsjö 1968, Damman 1990).

Increased WT caused changes in element concentrations in the peat. On the fen site, strongly decreased K may have leached from peat after the disturbance of the biological cycle (Damman 1986, 1990, 1995) or been used by vigorously growing *Eriophorum* (Malmer 1958, Damman 1986). In the drained stage, increased leaching



**Fig. 7.** Detrended correspondence analysis (DCA) for the bog site. Annual change in the vegetation composition of the bog sample plots indicated as shifts in the ordination space. — **A:** Restored site (letters indicate the year: A = 1994; B = 1995; C = 1996; D = 1997). — **B:** Control site (letters indicate the year: a = 1994; b = 1995; c = 1996; d = 1997). Numbers indicate the specific vegetation sample plots.

(Sallantaus 1992) and uptake by the tree stand (Laiho *et al.* 1999) of Ca and Mg resulted in a decrease in their concentrations in the peat. After the restoration, their concentrations increased in the peat, reversing the trend induced by drainage succession. The rapid increase in Ca and Mg is partly facilitated by road salting during the winter. The Na of the salt is partly exchanged with Ca and Mg as it flows through the esker, thus the water entering the fen site is enriched with Ca and Mg (T. Sallantaus, unpubl.). These base cations are then exchanged with hydrogen

ions in the mire, which together with the leaching of humic substances causes increased acidity in the mire soil water. The humic substances are largely derived from decomposition processes during the drainage stage. This ‘acidification’ of the soil water is not reflected in the pH of the peat due to increasing base cation concentrations in the peat. Soil acidity is likely to decrease in the future, after the organic acids formed during the drained stage have been flushed out by the water flow.

Because of the low concentrations of mineral

elements in the ombrotrophic peat (Damman 1986) and in precipitation, the postrestoration changes in the surface peat nutrient regime of the bog also remained small. Thus, the change in ecohydrology was less marked than on the fen site.

## Vegetational changes

Vegetation changed thoroughly in both mires during the restoration period. Changes were greater on the fen site, where tree removal changed light conditions and the ecohydrological shift was larger. The pre-restoration stand volume (about 100 m<sup>3</sup> ha<sup>-1</sup>) was slightly lower than the average for similar drained mire sites of that region but was still within normal variation (cf. Laiho & Laine 1994).

After clear-cutting, increased light and moisture conditions favored germination of vegetative propagules, especially of *Eriophorum vaginatum* (Jauhiainen 1998). Being efficient in using the increased nutrients (Komulainen *et al.* 1999) *Eriophorum* started to grow vigorously. Because *Eriophorum* is an opportunistic pioneer species, it can rapidly take over the habitat created by restoration of different kinds of peatlands (Grosvernier *et al.* 1995, Pfadenhauer & Klötzli 1996, Robert *et al.* 1999, Tuittila *et al.* 2000). The *Eriophorum* stage may, however, be a transitional stage towards *Carex*- and *Sphagnum*-dominated mire vegetation (Grosvernier *et al.* 1995, Tuittila *et al.* 2000). In 1997 true fen species, such as *Carex rostrata*, *Calla palustris*, and *Potentilla palustris*, as well as several species of *Sphagnum* were already present on the fen site (Table 1 footnotes).

Increased light also caused enhanced vegetation growth, especially of *Empetrum nigrum* in the neighboring control site (Tybirk *et al.* 2000). Difficulties in settling vegetation plots between trees on the control site and maintaining 5-m distance between plots caused relatively large variation between vegetation plots. There was, for example, a 'base of tree' influence on some plots.

The changes were smaller on the bog than on the fen site. The mire surface was formed from hummocks, intermediate surfaces, and hollows.

The same structure, originating from the pristine stage, remained even after the restoration. In the drained stage, the lawns and hollows were almost evenly lichen-covered. After the restoration, the hollows filled with water, which caused a degradation of lichens both in hollows and lawns as *Sphagnum* started to overgrow them (Vasander 1981). Although *Sphagnum* cover in lawns and hollows already increased during the first three years, a major change can only be expected after several years (Heikkilä & Lindholm 1994). The first stage of the ground layer vegetation change is the recovery of the small and suffering patches of *Sphagnum* present before restoration and the simultaneous degradation of the lichen cover.

With the raising of the water level, the moisture content in the hummocks also increased. Consequently, *Empetrum* growing on hummocks was released from epiphytic lichens and its growth increased strongly.

## Was restoration successful?

Restoration at both sites was initiated successfully. Water table levels rose and mire species started to increase in cover while forest species declined rapidly. Moreover, typical fen species grew in the fen site and bog species in the bog site, though the species compositions still differed from those on pristine sites. Thus we may accept our initial hypothesis.

In contrast to restoration performed in cut-away peatlands, restoration in peatlands drained for forestry is more rapid, partly because there are vegetative parts and dormant buds of plants, ready to react in changed moisture conditions (Huopalainen *et al.* 1998, Jauhiainen 1998). However, if nothing is done to promote higher water levels, spontaneous restoration may take a very long time due to the evapotranspiration of tree stands and the functioning of drainage ditches. We still need longer ecohydrological and vegetational monitoring for evaluating the man-induced restoration succession in mires drained for forestry, before we know whether we will achieve self supporting, functional mire ecosystems without further measures, and how long it will take before the species compositions

resemble those of pristine sites. Moreover, comparative studies on the functioning of pristine, drained, and restored mire ecosystems, such as decomposer populations (Silvan *et al.* 2000, Laiho *et al.* 2001, Jauhiainen 2002) and the carbon cycle (Komulainen *et al.* 1998, 1999) are needed to assess the ecological functioning and idiosyncrasies of restored peatlands.

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