

## RESEARCH ARTICLE

# The Effects of Peatland Restoration on Water-Table Depth, Elemental Concentrations, and Vegetation: 10 Years of Changes

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## Abstract

We studied the effects of restoration on water-table depth (WTD), element concentrations of peat and vegetation composition of peatlands drained for forestry in southern Finland. The restoration aimed to return the trajectory of vegetation succession toward that of undisturbed systems through the blockage of ditches and the removal of trees. Permanent plots established on a bog and a fen were sampled 1 year before, and 1, 2, 3, and 10 years after the restoration. The restoration resulted in a long-term rise of the water-table in both peatlands. Ten years after restoration, the mineral element concentrations (Ca, K, Mg, Mn, and P) of peat corresponded to those reported from comparable pristine peatlands. In particular, the increase of K and Mn concentrations at both sites suggests the recovery of ecosystem functionality in terms of nutrient cycling between peat and plants. The restoration resulted in the

succession of plant communities toward the targeted peatland vegetation of wetter condition at both sites. This was evident from the decreased abundance of species benefiting from drainage and the corresponding increase of peatland species. However, many species typical of pristine peatlands were missing 10 years after restoration. We conclude that the restoration led to a reversal of the effects of drainage in vegetation and studied habitat conditions. However, due to the slow recovery of peatland ecosystems and the possibility that certain failures in the restoration measures may become apparent only after extended time periods, long-term monitoring is needed to determine whether the goals of restoration will be met.

**Key words:** anthropogenic disturbance, ecosystem function, ecosystem structure, DCA, long-term monitoring, wetland.

## Introduction

In Europe, peatlands have historically covered nearly 100 million hectares, that is, 20% of the land area (Lappalainen 1996). Human influence on peatlands has been strong, impacting approximately 60% of their original area through agricultural, forestry, and peat extraction activities (Joosten 1997). The degradation of pristine peatland ecosystems causes a significant threat to the international goal of achieving a significant reduction in the current rate of biodiversity loss at the global, regional, and national levels (SCBD 2006). In Finland

alone, there are 217 species confined to peatland habitats that are classified as threatened, and 4.5% of all endangered species live only in undegraded peatlands (Rassi et al. 2001). In addition to the threat to these species, human activities have caused changes at the ecosystem and habitat levels. For example, in a recent assessment of the threatened habitat types in Finland, 51 out of 70 (73%) of peatland habitat types were classified as threatened or nearly threatened (Kontula & Raunio 2009).

Finland as a country has the highest proportion of peatlands worldwide (Vasander et al. 2003), and drainage for forestry has been the main factor behind their degradation. Altogether, 4.8 million hectares (54% of peatland area) have been drained for forestry (Finnish Forest Research Institute 2008). Drainage lowers the water-table level on average by 20–60 cm (e.g. Laine & Vanha-Majamaa 1992), which increases aeration and thus promotes the decomposition and nutrient mineralization of the peat matrix. Typically, this results in elevated total concentrations of N and P and decreased concentrations of major exchangeable cations such as K, Ca, and Mg in the surface peat (Laiho et al. 1999; Sundström et al. 2000). Increase in the concentrations of other mineral elements such as Al, Fe, and Mn in the pore water has also been observed

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(Berry & Jeglum 1991). Following drainage, mineralized nutrients are either leached or relocated in the ecosystem, where stands of trees become a considerable nutrient sink (Finér 1989; Sallantausta 1992).

Changes in these abiotic factors are reflected in the species composition of the drained peatland ecosystems. Most visible changes take place in vegetation, where typical peatland species are replaced by common forest species (e.g. Laine et al. 1995). Of the original peatland vegetation, hummock species may benefit from the drainage, whereas species demanding prolonged inundated conditions soon disappear after drainage (Vasander 1982; Heikkilä & Lindholm 1995a).

Ecological restoration aims at reversing the trend of degradation by partial rehabilitation or complete restoration of the original structure (species composition) and function (e.g. hydrology, nutrient cycling) of the ecosystem (Bradshaw 1990; Dobson et al. 1997; Vanha-Majamaa et al. 2007). The restoration of peatlands drained for forestry is implemented by damming or filling in the ditches with peat and the removal of trees grown after drainage. The aim of these actions is to initiate a process that will restore functional peatland ecosystems that can maintain viable populations of species characteristic to these habitats (Aapala et al. 2008). Approximately 16,200 ha of peatlands drained for forestry have been restored in Finland since the early 1990s (Aapala & Hyvärinen 2009). Globally, there is an increasing trend in peatland restoration projects as seen in examples from Northern England and Belarus (Ramchunder et al. 2009; Thiele et al. 2009).

The short-term effects of restoration include a rapid rise of the water-table and changes in peat chemistry (Jauhiainen et al. 2002; Worrall et al. 2007). For example, concentrations of Al and Ca increased in the surface peat of a fen, whereas concentrations of K and P decreased after restoration (Jauhiainen et al. 2002). Restoration can also cause relatively rapid changes in vegetation composition such as a decline in the number of forest species and an increase in the abundance of peat-forming *Sphagnum* mosses (Sphagnaceae) and some other typical peatland species (Jauhiainen et al. 2002; Aapala & Tukka 2008).

There are relatively few reports identifying the longer term responses of peatland ecosystems to restoration after forestry drainage. Despite the encouraging short-term effects, it has been suggested that rewetting was successful only for every second peatland system in Finland, and that many species typical of pristine peatland communities were missing 7–10 years after restoration (Tahvanainen 2006). Moreover, unwanted redrying can impact on succession despite promising early rewetting, probably due to failures of the dams in maintaining prolonged high water levels (Haapalehto et al. 2006). Long-term studies using permanent plots are needed to assess both the technical and ecological success of these restoration initiatives.

Here we studied the effects of restoration on vegetation and habitat conditions of two drained *Sphagnum* peatlands in southern Finland by comparing the restored area of each peatland to an unrestored area of the same peatland. The data were gathered over a 10-year period after the restoration. The

goals were to determine whether restoration had resulted in a reversal of the effects of drainage on (1) the depth of the water-table (WTD), (2) the concentrations of the elements in the surface peat, (3) the structure of vegetation communities and (4) the abundance of individual plant species. In addition to the above comparisons, reference values from literature and the occurrence of indicator species were used to evaluate whether the restored peatlands had evolved toward the desired natural target state.

## Methods

### Study Sites

Two *Sphagnum* peatlands, an ombrotrophic bog (lat 61°51'N, long 24°14'E, 160 m a.s.l.) and a minerotrophic fen (lat 61°48'N, long 24°17'E, 155 m a.s.l.), located 5 km apart from each other in southern Finland, were selected as study sites. The mean annual precipitation in the area is 675 mm and the long-term annual mean temperature is +3.4°C (Finnish Meteorological Institute 2008).

The total area (200 ha) of the studied bog was drained for forestry in 1955 using 30 m ditch spacing. In the central part of the bog, an area of 10.5 ha was re-wetted in 1995. The rewetting was done mainly by filling in the ditches with peat excavated from the ditch banks and the bog surface, but in places where volumes of peat in the ditch banks were insufficient for filling the ditches completely, more localized dams were constructed from peat. Inside the re-wetted part of the bog, an area of 0.5 ha was chosen for monitoring. To regain the openness typical of natural bogs, the monitoring area (Restored Bog) was clear-cut and the logging slash was removed. Approximately 100 m away from the Restored Bog site, a similar but unwetted and uncut drained area was selected as an experimental control location (Drained Bog). The vegetation type of the bog prior to the drainage had been ombrotrophic short-sedge bog (sensu Laine & Vasander 1996), dominated by *Eriophorum vaginatum* (Tussock cottongrass), *Calluna vulgaris* (Heather), *Empetrum nigrum* (Crowberry) and *Sphagnum* species of wet hollows, such as *Sphagnum balticum* (Baltic bog moss). At the time of restoration, the field layer was characterized by *C. vulgaris*, *E. nigrum*, and *Vaccinium uliginosum* (Bog bilberry). Lichens (especially *Cladonia* spp. [Cladoniaceae]), *S. fuscum* (Sphagnaceae), and *S. rubellum* (Sphagnaceae) dominated the ground layer with some remnant populations of *S. balticum* present.

The minerotrophic fen (150 ha) was drained for forestry in 1955 with a ditch spacing of 50 m. In 1995, an area of 1.1 ha was re-wetted by filling in one ditch with peat and damming an adjacent ditch with water-tight peat dams. The peat was excavated from the fen surface close to the ditches. A short feeder ditch was excavated upstream to enhance water flow from a nearby pond and mineral soils. Inside the re-wetted area, an area of 0.6 ha was selected for monitoring (Restored Fen). This monitoring area was clear-cut and the logging slash and trees were removed. Approximately 50 m away from the Restored Fen site, a similar but unwetted and uncut area was selected

as an experimental control location (Drained Fen). Prior to drainage, the fen was of the tall-sedge pine fen type (sensu Laine & Vasander 1996). At the time of restoration, the sites supported a tree stand dominated by *Pinus sylvestris* (Scots pine). During the drainage phase, the field layer had changed from tall-sedge (e.g. *Carex rostrata* [Bottle sedge]) dominated, clearly minerotrophic peatland vegetation into communities of common forest plants, such as dwarf shrubs *V. myrtillus* (Blueberry), *V. uliginosum*, *V. vitis-idaea* (Lingonberry), *Ledum palustre* (Wild rosemary), and *Betula nana* (Dwarf birch). The ground layer was dominated by *Pleurozium schreberi* (Big red stem moss) along with *S. angustifolium* (Sphagnaceae), *S. magellanicum* (Sphagnaceae), and *S. russowii* (Sphagnaceae).

### Experimental Set-up and Data Collection

In 1994, systematically laid vegetation sample plots (100 × 50 cm) were permanently marked on the restored and drained sites of the bog and the fen (Table 1). The plots, which had a minimum distance of 5 m from each other, were located in transects between the ditches and running coaxial to the ditches. A visual estimation of the cover of each plant species and lichens in the sample plots was used as a measure of their abundance. The sampling was done by the same person (S. Jauhiainen) at the end of July in 1994, 1995, 1996, 1997, and 2005. Pipe wells (polyvinyl chloride, PVC) were established to measure the WTD. Measurements were taken covering the snow free periods (Table 1), and on each occasion all measurements (cm) were done during the same day at all sites.

To study the elemental concentrations in the peat, 20 cm deep cores of surface peat (8.3 × 8.4 cm in width) were taken near each sample plot in August 1994, 1997, and 2006. The samples were divided into two segments (0–10 and 10–20 cm) and stored in plastic bags. Peat samples were dried at 105°C to a constant mass and ground through a 2 mm sieve. Samples were digested with HNO<sub>3</sub>–H<sub>2</sub>SO<sub>4</sub>–HClO<sub>4</sub> at 200°C, and the total concentrations of Al, Ca, Fe, K, Mg, Mn, and P were analyzed at the Vantaa Research Unit laboratory of the Finnish Forest Research Institute, using an ARL 3580 vacuum ICP plasma emission spectrometer (1994) and a TJA IRIS Advantage ICP spectrometer (1997, 2006). Meteorological data (precipitation, temperature) were acquired from the Finnish Meteorological Institute (Finnish Meteorological Institute 2008). The nomenclature of scientific names

follows Euroala et al. (1992) for vascular plants, bryophytes, and lichens.

### Data Analysis

Repeated measures analysis of variance (ANOVA) was used to analyze the effect of time and the restoration treatment on the WTD, elemental concentrations in the peat, and the abundance of individual plant species. The treatment and the depth of the peat sample, in the case of testing for elemental concentrations, were used as fixed between-subject factors and sampling years formed the levels of the within-subject factor (time). In cases where the assumption of sphericity was violated (tested by Mauchley's test), the Greenhouse–Geisser correction was used to adjust the degrees of freedom. To determine the change in the depth of the water-table over time, yearly averages were calculated for the drained and the restored sites. The average daily rainfall (April to September) and the average monthly temperature (measured daily at 8 pm, April to September) were calculated and Pearson correlation was used to study the relationship between rainfall, temperature and WTD.

Changes in the vegetation community structure were analyzed employing detrended correspondence analysis (DCA) in PC-ORD 5.0 (McCune & Mefford 1999). The aim of the DCA was to reveal the major community gradients within the data sets and to test the temporal movement of communities within monitoring plots along gradients. The DCAs were completed without downweighting rare species, because the species in question were particularly valuable for the identification of the community gradients.

DCA was selected among the unconstrained ordination techniques for specific reasons. After preliminary data exploration, it was evident that the main gradients involved a significant turn-over of community composition and unimodal sequences of species responses (high gradient length in DCA). Another preliminary observation was that the differences in vegetation changes between the restored and the drained sites were not easily identified. Thus, a repeatable ordination method with a unimodal response model was required. Furthermore, an objective comparison of the distances between sample pairs (before–after) and between samples and fixed goals within interpretable ordinations was intended for subsequent analyses. In DCA, the separate axes are, in principle, hierarchical and comparable in units of gradient length (see Økland 1986).

**Table 1.** The number of vegetation plots and pipewells established and sampled.

	Restored Bog	Drained Bog	Restored Fen	Drained Fen
Vegetation plots established and sampled	12	9	12	9
Pipe wells established 1994	19	11	20	14
Pipe wells sampled 1994–1997	12 <sup>a–e</sup>	9 <sup>a–e</sup>	12 <sup>a–e</sup>	9 <sup>a–e</sup>
Pipe wells sampled 2005	7 <sup>a</sup> , 18 <sup>b</sup> , 12 <sup>c</sup> , 12 <sup>d</sup> , 19 <sup>e</sup>	4 <sup>a</sup> , 13 <sup>b</sup> , 9 <sup>c</sup> , 9 <sup>d</sup> , 13 <sup>e</sup>	8 <sup>a–e</sup>	5 <sup>a</sup> , 8 <sup>b</sup> , 8 <sup>c</sup> , 9 <sup>d</sup> , 14 <sup>e</sup>

<sup>a</sup> June.

<sup>b</sup> July.

<sup>c</sup> Beginning of August.

<sup>d</sup> End of August.

<sup>e</sup> Beginning of September and end of November.

To objectively interpret the DCA pattern, we analyzed the Euclidean distances (the square root of the sum of the squared differences between values for the items) between years for vegetation plots based on the site scores and the species scores along the axes 1–3 in the ordination. The site scores were first used to analyze the overall effect of restoration in the ordination pattern. The site scores and the species scores of selected indicator species were then used to test if vegetation at the restored sites had changed toward the desired natural target state (wetter communities) or whether the change toward vegetation communities tolerant of drier conditions had continued. Based on the studies on peatland plant ecology and species responses to drainage (e.g. Heikkilä & Lindholm 1995b; Tahvanainen et al. 2002), *S. balticum* and *S. angustifolium* were chosen as the indicator species of wet conditions for the bog and the fen, respectively. *P. schreberi* was selected as the species indicative of dry conditions at both sites.

First, we calculated the Euclidean distance of site scores for each vegetation plot between years 1994 and 2005 to determine if their changes were dependent on the restoration treatment. Second, to determine whether the vegetation had changed toward wetter or dryer communities in the ordination, we calculated the Euclidean distances between the site scores of each plot and the species scores of the selected indicator species (wet or dry conditions) for both 1994 and 2005. By subtracting the distance value of 2005 from 1994, we can infer the direction and length (in the gradient length units of DCA axes) of community change in relation to the indicator species. If, for example, the difference between treatments was significant and positive, the sites had moved toward the species indicative of wet conditions. As separate DCA axes have the same unit of length and are hierarchical, scores for the first axes in both ordinations bore the greatest contribution to the calculated Euclidean distances. Therefore, the delimitation of the number of axes was not as critical as it would have been if the non-metric multidimensional scaling-method had been used. Differences between the treatments were analyzed with nonparametric Mann–Whitney test. Statistical analyses were performed using SPSS 15.0.

## Results

### Depth of the Water-Table

The water-table rose soon after restoration and remained higher than at the drained sites 10 years after restoration (Fig. 1). However, the significant quadratic year by treatment within-subjects interaction contrasts indicate that the difference in the WTD between restored and drained sites diminished with time (bog :  $f_{[1,81]} = 73.23$ ;  $p < 0.001$ ; fen :  $f_{[1,77]} = 93.59$ ;  $p < 0.001$ ). The significant year by treatment interactions indicate that the changes in the WTD depended on the restoration treatment (bog :  $f_{[2,10,169,71]} = 35.28$ ;  $p < 0.001$ ; fen :  $f_{[2,26,173,61]} = 28.43$ ;  $p < 0.001$ ). There were no significant relationships between the WTD and precipitation or temperature (data not shown), suggesting that the rise of

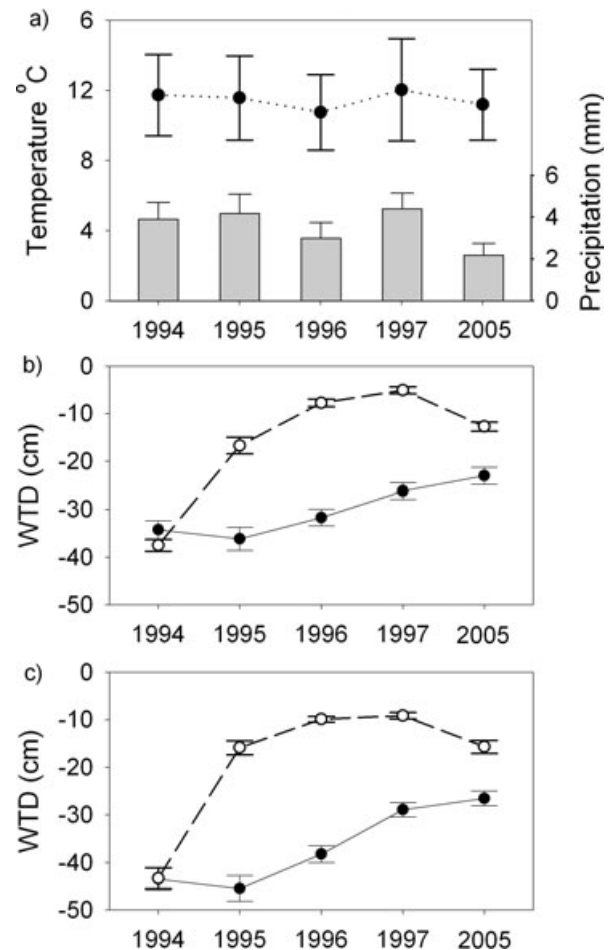


Figure 1. Temperature and precipitation (a), and the depth of the water-table (WTD) at the bog (b) and at the fen (c). For temperature and precipitation, the monthly average of daily values measured at 8 pm (°C,  $\pm$  standard error) and the average of daily mean values in April to September (mm,  $\pm$  standard error) were calculated, respectively. For the WTD, average values of June to October (cm,  $\pm$  standard error) were calculated. Solid lines with filled circles and dotted lines with open circles indicate drained and restored sites, respectively.

the water-table at the drained sites was not caused by these climatic factors.

### Elemental Concentrations in the Peat

In general, the effect of treatment on the elemental concentrations was more pronounced in the 0–10 cm layer (Table 2). The significant three-way within-subjects interactions among year, treatment, and depth indicate that the effect of restoration treatment on changes in elemental concentrations was different at the two depths for Al and K in the fen (Table 3).

In general, the elemental concentrations at the restored peatlands increased with time, as compared to the unrestored sites (Table 2). However, the concentration of K in the surface peat in the fen initially decreased and then increased 10 years after restoration. A similar but less distinct pattern was recognized in P concentrations in the bog. The significant

**Table 2.** Summary of element concentrations (mg/g) in the peat of the Bog and the Fen.

Depth (cm)	1994				1997				2005				
	Drained		Restored		Drained		Restored		Drained		Restored		
	Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD	
<b>Bog</b>													
Al	0–10	0.76	0.09	0.64	0.12	0.89	0.23	0.79	0.14	0.70	0.09	0.49	0.24
	10–20	0.52	0.16	0.37	0.24	0.62	0.15	0.40	0.09	0.55	0.09	0.51	0.14
Ca	0–10	1.61	0.33	1.03	0.21	1.71	0.45	1.39	0.78	2.21	0.89	2.47	0.74
	10–20	1.34	0.21	0.84	0.29	1.29	0.25	0.95	0.32	1.63	0.38	1.52	0.23
Fe	0–10	1.06	0.18	0.72	0.12	1.09	0.30	0.90	0.13	1.09	0.27	0.74	0.31
	10–20	0.57	0.19	0.47	0.36	0.74	0.39	0.54	0.13	0.81	0.33	0.81	0.42
K	0–10	0.44	0.07	0.44	0.15	0.40	0.10	0.39	0.09	0.67	0.17	1.22	0.49
	10–20	0.12	0.06			0.22	0.04			0.20	0.10		
Mg	0–10	0.52	0.05	0.39	0.05	0.53	0.11	0.45	0.07	0.62	0.10	0.69	0.14
	10–20	0.45	0.08	0.36	0.11	0.46	0.16	0.43	0.14	0.54	0.22	0.53	0.15
Mn	0–10	0.03	0.04	0.02	0.01	0.04	0.03	0.04	0.02	0.07	0.08	0.15	0.09
	10–20	0.003	0.001	0.004	0.002	0.01	0.004	0.01	0.004	0.01	0.004	0.02	0.01
P	0–10	0.45	0.04	0.41	0.09	0.49	0.11	0.38	0.07	0.49	0.07	0.43	0.11
	10–20	0.25	0.08	0.25	0.07	0.29	0.07	0.17	0.03	0.31	0.09	0.27	0.05
<b>Fen</b>													
Al	0–10	0.74	0.10	0.69	0.07	0.79	0.06	0.89	0.15	0.54	0.13	0.41	0.13
	10–20	0.76	0.11	0.84	0.14	0.94	0.14	0.97	0.26	0.52	0.05	0.63	0.12
Ca	0–10	2.27	0.54	2.16	0.37	2.42	0.49	3.60	0.81	2.08	0.62	3.24	0.52
	10–20	1.35	0.41	1.28	0.38	1.38	0.23	1.80	0.57	1.29	0.27	2.62	0.97
Fe	0–10	5.65	2.24	2.70	0.45	4.35	1.86	2.75	0.61	3.86	2.61	3.60	1.87
	10–20	4.46	2.65	1.80	0.63	3.74	2.52	1.62	0.56	4.48	1.72	2.36	0.78
K	0–10	0.69	0.08	0.80	0.20	0.57	0.10	0.39	0.08	0.63	0.27	1.16	0.34
	10–20	0.26	0.08	0.30	0.09	0.26	0.01	0.31	0.06	0.31	0.06	0.31	0.07
Mg	0–10	0.39	0.05	0.44	0.07	0.38	0.07	0.65	0.09	0.38	0.11	0.69	0.07
	10–20	0.16	0.05	0.24	0.06	0.17	0.03	0.38	0.13	0.16	0.05	0.49	0.12
Mn	0–10	0.07	0.05	0.05	0.02	0.08	0.06	0.04	0.02	0.07	0.05	0.11	0.05
	10–20	0.01	0.01	0.01	0.003	0.02	0.004	0.01	0.01	0.01	0.01	0.02	0.01
P	0–10	0.99	0.13	0.94	0.17	0.92	0.15	0.75	0.12	0.85	0.26	0.76	0.09
	10–20	0.80	0.14	0.85	0.16	0.71	0.13	0.77	0.12	0.80	0.15	0.76	0.14

Bog, 0–10 cm: For all elements,  $n(\text{drained}) = 9$ ,  $n(\text{restored}) = 12$ .

Bog, 10–20 cm: For Al, Ca, Fe, Mg, Mn, P  $n(\text{drained}) = 9$ ,  $n(\text{restored}) = 10$ , for K  $n(\text{drained}) = 3$ ,  $n(\text{restored}) = 0$ .

Fen, 0–10 cm: For all elements,  $n(\text{drained}) = 8$ ,  $n(\text{restored}) = 12$ .

Fen, 10–20 cm: Al, Ca, Fe, Mg, Mn, P  $n(\text{drained}) = 7$ ,  $n(\text{restored}) = 12$ , for K  $n(\text{drained}) = 3$ ,  $n(\text{restored}) = 11$ .

within-subjects year by treatment interactions indicate that the change in the concentration of the element was dependent on the restoration for Ca, K, Mg, Mn, and P in the bog and for Ca, K, Mg, and Mn in the fen (Table 3).

### Vegetation Dynamics

**Detrended Correspondence Analysis (DCA).** The Euclidean distances (calculated from the values of the extracted ordination axes) within the three-dimensional DCAs explained 64 and 65% of the variation in the  $n$ -dimensional species matrices for the bog and fen, respectively ( $n$  = number of species). The gradient lengths of the three DCA axes were 4.0, 2.9, and 2.2 in the bog ordination, and 3.1, 2.6, and 2.4 in the fen ordination. In both ordinations, axis 1 represented the water-table gradient, separating the samples according to the abundance of species characteristic of dry and wet conditions. The subsequent axes reflected differences between the vegetation plots in the

abundance of lichens versus bryophytes, and also between typical spruce-forest versus pine-forest species for the fen data.

In the fen, there was a significant overall effect of restoration on the Euclidean distances calculated from the site scores between years ( $n_{\text{restored}} = 12$ ,  $n_{\text{drained}} = 9$ ,  $z = -3.492$ ,  $p < 0.001$ ) (Fig. 2). In other words, the change in the community structure of the vegetation plots was dependent on the restoration treatment. However, in the bog system, there was no overall effect of treatment on the vegetation community change in the ordination ( $n_{\text{restored}} = 12$ ,  $n_{\text{drained}} = 9$ ,  $z = -0.995$ ,  $p = 0.320$ ) (Fig. 3).

At both sites, the difference in Euclidean distances between the 1994 and the 2005 site scores, and the scores of the indicator species of wet conditions, depended on the treatment (Table 4). For both the bog and fen system, the difference between the treatments was positive for restored plots, indicating that the vegetation had changed toward the wetter target condition. Also, the difference in the Euclidean distances

**Table 3.** Summary of Repeated measures ANOVA for element concentrations in the Bog and in the Fen.

Source	df	Al		Ca		Fe		K		Mg		Mn		P	
		f	P	f	p	f	p	f	p	f	p	f	p	f	p
<b>Bog</b>															
Between subjects															
T	1	18.19	<0.001	6.60	0.015	13.87	0.001	8.94	0.007	2.16	0.151	2.23	0.144	13.44	0.001
D	1	45.23	<0.001	21.07	<0.001	26.88	<0.001	12.7	0.002	5.38	0.026	43.3	<0.001	107.5	<0.001
T × D	1	0.38	0.846	0.24	0.625	3.43	0.072			0.00	0.983	0.98	0.328	0.29	0.592
Error df		36		36		36		21		36		36		36	
Within subjects															
Y	2 <sup>1</sup>	6.68	0.002	30.47	<0.001	3.48	0.036	13.05	0.001	34.93	<0.001	18.75	<0.001	4.33	0.017
Y × T	2 <sup>1</sup>	0.14	0.870	4.53	0.014	0.05	0.954	10.05	0.003	6.13	0.000	6.01	0.013	4.84	0.011
Y × D	2 <sup>1</sup>	7.77	0.001	3.36	0.04	3.05	0.053	0.93	0.362	2.25	0.112	12.36	0.001	0.53	0.588
Y × T × D	2 <sup>1</sup>	2.42	0.096	0.67	0.515	1.10	0.338			1.58	0.214	3.52	0.059	0.32	0.725
Error df		72		72		72		25.21		72		43.95		72	
<b>Fen</b>															
Between subjects															
T	1	0.46	0.500	22.16	<0.001	25.25	<0.001	4.57	0.041	99.31	0.000	0.65	0.426	1.47	0.233
D	1	9.17	0.005	52.47	<0.001	3.63	0.065	94.28	<0.001	110.3	<0.001	78.76	<0.001	5.96	0.020
T × D	1	2.53	0.121	0.42	0.523	0.79	0.381	2.22	0.147	0.01	0.922	0.15	0.701	3.27	0.079
Error df		35		35		35		30		35		35		35	
Within subjects															
Y	2 <sup>2</sup>	89.83	<0.001	14.81	<0.001	1.85	0.180	11.38	0.001	32.48	<0.001	2.29	0.132	8.74	0.002
Y × T	2 <sup>2</sup>	0.95	0.391	17.57	<0.001	3.56	0.057	6.44	0.010	33.79	<0.001	4.97	0.024	0.61	0.500
Y × D	2 <sup>2</sup>	0.12	0.884	5.22	0.008	0.80	0.401	9.42	0.002	0.53	0.591	2.15	0.146	1.84	0.177
Y × T × D	2 <sup>2</sup>	4.10	0.021	2.44	0.095	1.59	0.217	8.81	0.003	1.03	0.361	3.15	0.074	1.21	0.296
Error df		70		70		43.58		38.27		70		43.82		51.49	

T = Treatment, D = Depth in peat profile, Y = Year.

If the assumption of sphericity was violated, the Greenhouse–Geisser corrected degrees of freedom were used for the <sup>1</sup>Bog (1.20 and 1.22 for K and Mn, respectively) and for the <sup>2</sup>Fen (1.25, 1.28, 1.25, and 1.47 for Fe, K, Mn, and P, respectively).

Note that the between-subjects effect of treatment (T) is not necessarily indicative of the effect of the restoration. This is because the first measurements were conducted before the restoration treatment, and thus the effect may describe differences between the sites that are not associated with restoration. This note applies also to the between-subjects treatment by depth interaction.

between the 1994 and the 2005 site scores, and the indicator species of dry conditions, depended on the treatment (Table 4). The difference was negative for the restored sites, suggesting that the composition of the vegetation had moved in a trajectory away from the drier conditions.

The differences in the Euclidean distances were the opposite in the drained sites, as compared to the restored sites (Table 4). This indicates that even though the water-table at the drained sites had possibly risen after 10 years (Fig. 1), and the difference relative to the restored sites had decreased, the rise of the water-table was not enough to reverse the trajectory of plant community development at the drained sites.

### Plant Species Dynamics

A significant year by treatment within-subjects effect indicates that the restoration had affected the abundance of *Eriophorum vaginatum*, *Sphagnum balticum* and the total *Sphagnum* spp. cover in the Bog; and the abundance of *Vaccinium myrtillus*, *Rubus chamaemorus* (Cloudberry), *E. vaginatum*, *Pleurozium schreberi* and the total forest-moss cover in the fen (Table 5; Fig. 4).

### Discussion

The damming and filling of the ditches with peat resulted in a raised water-table in the restored bog and fen systems. The rise took place immediately after ditch blocking and the elevated water-table remained higher than at the drained sites 10 years after restoration. However, drainage may cause irreversible changes in the physical properties of the peat, which may lead into changes in the hydrological routing of water through the peat and deteriorate the success of restoration (Grootjans et al. 2002a; Holden & Burt 2002). For example, the subsidence of peat due to drainage is particularly effective near the ditches. If the areas remain lower than surrounding areas even after the restoration, they may form localized water-flow paths for surface water along the filled-up ditches (Tahvanainen 2006). Such a pattern may have contributed to the observed decreasing trend in elevated water-table levels at both restored sites 10 years after restoration.

The water-table at both of the drained sites increased between 1995 and 2005, possibly due to the natural overgrowth of the ditches by *Sphagnum* (e.g. Holden et al. 2007).

**Table 4.** Summary of Euclidean distances between 1994 and 2005 site scores and species scores of indicator species of wet (Sph bal, Sph ang) and dry (Ple sch) conditions in the DCAs.

		Bog		Fen	
		ED(1994/Sph bal)– ED(2005/Sph bal)	ED(1994/Ple sch)– ED(2005/Ple sch)	ED(1994/Sph ang)– ED(2005/Sph ang)	ED(1994/Ple sch)– ED(2005/Ple sch)
Restored	Mean	56.25	–34.42	41.00	–56.63
	SD	50.74	78.33	51.54	44.63
	<i>n</i>	12	12	12	12
Drained	Mean	–38.85	51.70	–3.62	6.96
	SD	58.00	69.88	20.68	23.14
	<i>n</i>	9	9	9	9
	<i>z</i>	–3.269	–2.843	–2.416	–3.198
	<i>p</i>	0.001	0.004	0.016	0.001

ED = Euclidean distance, Sph bal = *Sphagnum balticum*, Ple sch = *Pleurozium schreberi*, Sph ang = *Sphagnum angustifolium*. Mann Whitney *U* statistic (*z*) of tests of differences between the restored and drained sites are given below each comparison.

**Table 5.** Repeated measures ANOVA for plant species abundance on the Bog and Fen in 1994–2005.

	df Bog (Treatment, Error)	df Fen (Treatment, Error)	Year				Year × Treatment			
			Bog		Fen		Bog		Fen	
			<i>f</i>	<i>p</i>	<i>f</i>	<i>p</i>	<i>f</i>	<i>p</i>	<i>f</i>	<i>p</i>
<i>Eriophorum vaginatum</i>	1.63, 30.87	2.42, 45.94	5.13	0.017	12.06	<0.001	6.33	0.008	7.03	0.001
<i>Pleurozium schreberi</i>	1.53, 29.12	2.28, 43.26	4.50	0.028	0.62	0.561	2.44	0.116	6.82	0.002
<i>Rubus chamaemorus</i>	2.22, 42.11	2.68, 50.85	4.00	0.022	3.76	0.020	0.46	0.659	3.02	0.044
<i>Sphagnum balticum</i>	1.25, 23.67	—	2.61	0.113	—	—	4.89	0.030	—	—
<i>Vaccinium myrtillus</i>	—	2.09, 39.75	—	—	6.37	0.004	—	—	10.57	<0.001
Forest mosses <sup>a</sup>	1.59, 30.25	2.35, 44.64	4.58	0.025	1.11	0.346	2.94	0.079	9.55	<0.001
<i>Sphagnum</i> spp. <sup>b</sup>	1.27, 24.20	2.08, 39.56	5.27	0.024	3.91	0.027	5.39	0.022	0.91	0.413

Note that only the species or species groups with statistically significant year × treatment effects at either of the sites are shown in the table.

<sup>a</sup> *Dicranum majus*, *D. polysetum*, *D. scoparium*, *Pleurozium schreberi*, *Pohlia nutans*, *Polytrichum commune*.

<sup>b</sup> *Sphagnum angustifolium*, *S. balticum*, *S. capillifolium*, *S. fuscum*, *S. magellanicum*, *S. rubellum*, *S. russowii*.

However, the rate of peat accumulation is too slow (a few mm per year) to result in complete overgrowth within the time-scale relevant to halting of the current loss of peatland biodiversity. Indeed, the hydrological impacts of drainage last for at least 50 years after the drainage (Holden et al. 2006). In addition to the requirement of a high water-table level, the recovery of peatland vegetation depends on water quality (Grootjans et al. 2002a, 2002b). The recovery of vegetation communities in these peatland ecosystems would be limited if partially overgrown ditches continued to provide artificial flow paths for surface water, thus reducing the supply of nutrients to the vegetation outside the ditches. Consequently, the restoration of peatland vegetation cannot be expected through natural overgrowth processes, and active measures are needed to restore the hydrological functions of degraded peatlands (Van Seters & Price 2001). The results of our DCA support this view by showing that a slight decrease in the WTD, possibly caused by the natural overgrowth of ditches, was not enough to significantly influence the vegetation at either of the drained sites.

Restoration resulted in significant changes to the concentrations of Ca, K, Mg, Mn, and P in the bog and to Ca, K, Mg, and Mn in the fen. The concentrations of Ca, K, Mg, and Mn are known to decrease after drainage (Berry &

Jeglum 1991; Laiho et al. 1999; Åström et al. 2001). Therefore, the increase in their concentrations observed at both of the restored sites after the blockage of the ditches indicates the potential for this restoration technique for reversing the trend of degradation. In comparison with the results of studies of comparable natural peatlands (Damman 1978; Pakarinen 1978; Laiho et al. 1999; Minkkinen et al. 1999), the average concentrations of Ca, K, Mg, and Mn in the restored bog reached natural levels within a 10-year period. In the fen, restoration increased the concentrations of Ca close to the values recorded from pristine fens in the region (Laiho et al. 1999; Minkkinen et al. 1999), indicating the recovery of the minerotrophic hydrology. Furthermore, the development of the concentrations of K and Mn toward the concentrations and vertical patterns reported from pristine peatlands indicates the recovery of peatland functionality in terms of K and Mn cycling at both of the restored peatlands (Damman 1978; Pakarinen 1978).

The initial decrease of subsurface P concentration in the Restored bog was consistent with increased leaching of P observed at other re-wetted sites (e.g. Olde Venterink et al. 2002). The subsequent increase to pre-restoration concentrations approximately equal to the levels of comparable peatlands in the region most likely resulted from P conservation

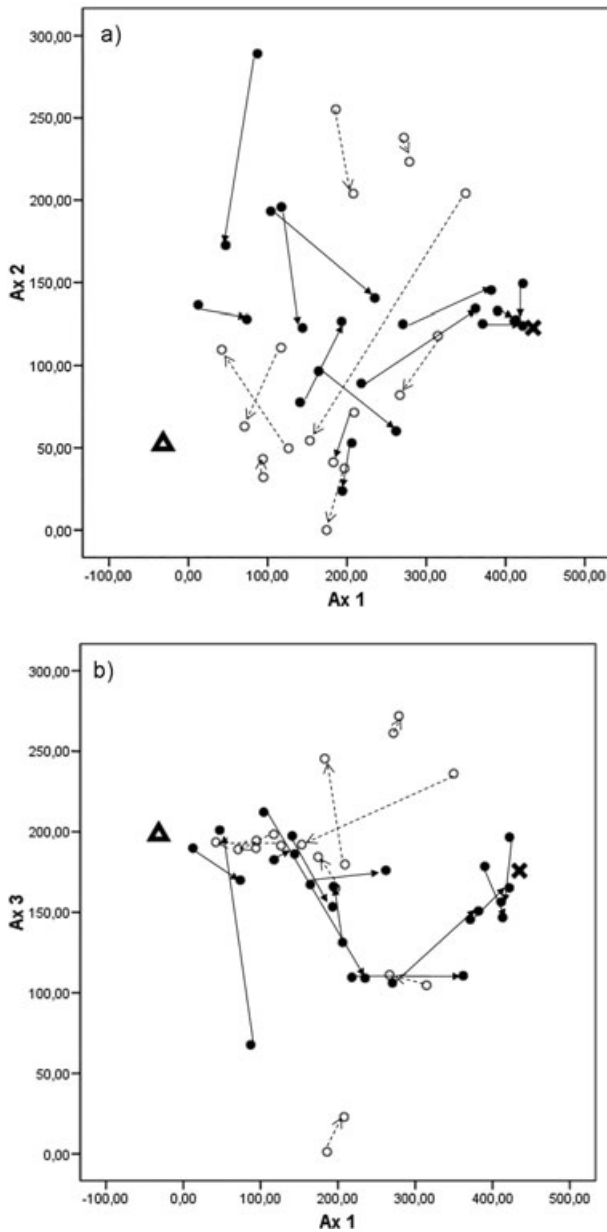


Figure 2. Axes 1 and 2 (a) and axes 1 and 3 (b) of DCA of bog vegetation in 1994–2005. Filled circles indicate plots at the Restored site and open circles plots at the drained site. The triangle and the cross indicate the species scores of the species indicative of dry (*Pleurozium schreberi*) and wet (*Sphagnum balticum*) conditions, respectively. Arrows indicate the movement of each plot from 1994 to 2005 in the ordination. The data of 1995, 1996, and 1997 were also included in the DCA, but to increase the clarity of the figure the scores are not shown. Also very short arrows are not drawn.

by plant biomass and litter (Damman 1978; Pakarinen 1978; Rydin & Clymo 1989; Laiho et al. 1999). In general, the changes in total P concentrations were small, with increased concentrations in the peat of the Restored Bog 10 years after restoration suggesting that the leaching of P decreases with time.

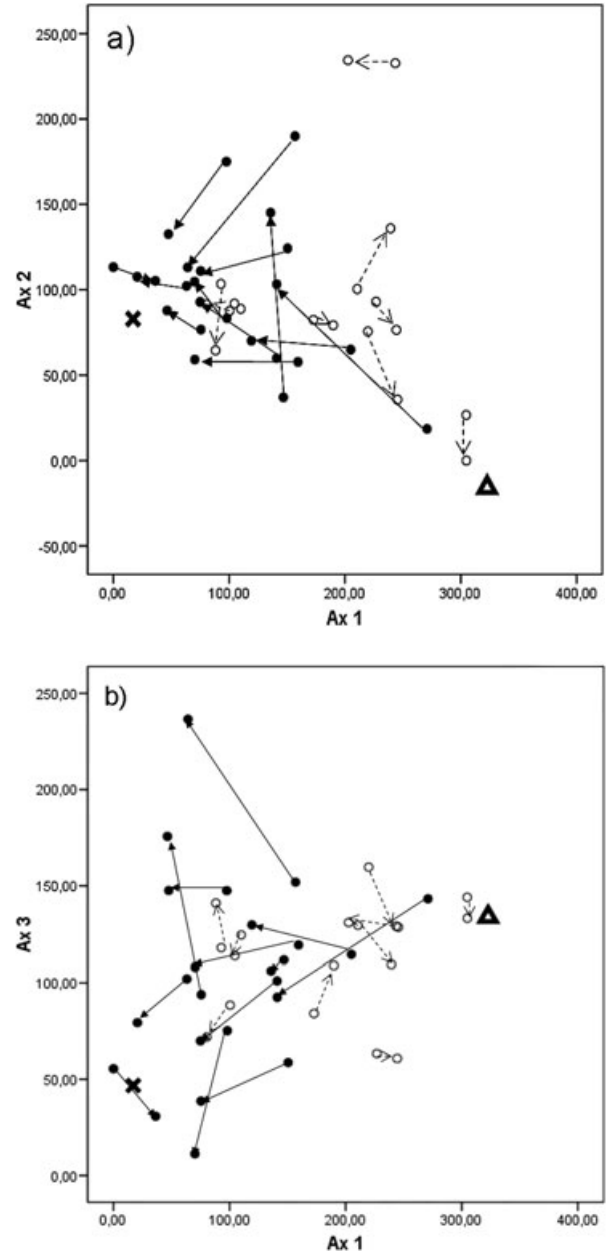


Figure 3. Axes 1 and 2 (a) and axes 1 and 3 (b) of DCA of fen vegetation in 1994–2005. Filled circles indicate plots at the Restored site and open circles plots at the drained site. The triangle and the cross indicate the species scores the species indicative of dry (*Pleurozium schreberi*) and wet (*Sphagnum angustifolium*) conditions, respectively. For further explanations, see Figure 2.

At the drained sites, the vegetation continued to develop toward drier communities with forest species dominant, whereas in the restored sites a successful reversal of this development and a succession toward targeted communities were gained by the restoration treatment. However, the restored and drained sites were still partly mixed in the composition of their plant species. Relatively small changes in species composition were expected for the Restored Bog as bogs are



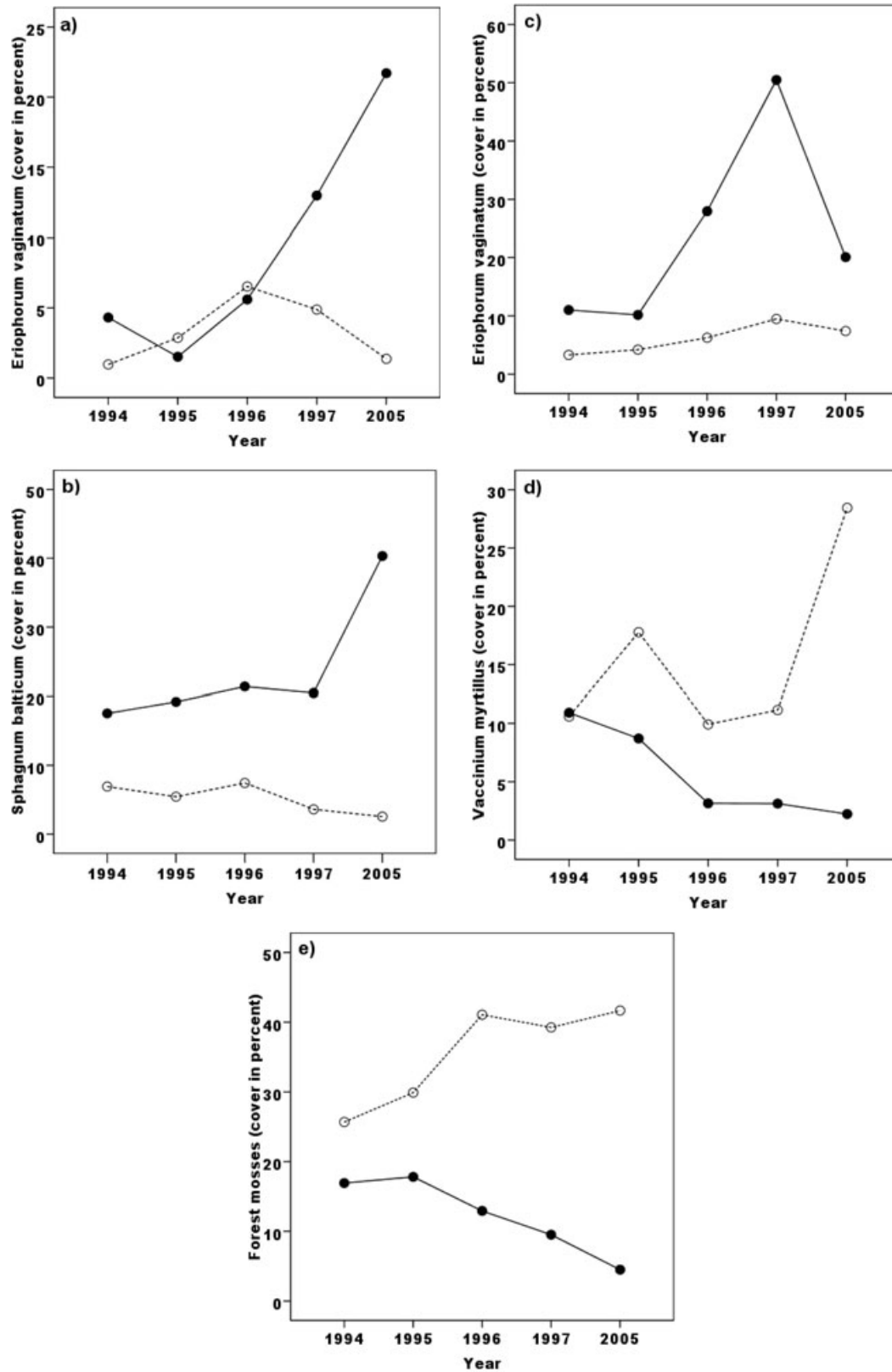


Figure 4. The abundance (cover in percent) of (a) *Eriophorum vaginatum* and (b) *Sphagnum balticum*, in the bog and (c) *Eriophorum vaginatum*, (d) *Vaccinium myrtillus*, and (e) all forest mosses in the fen in 1994–2005. Solid lines with filled circles indicate restored and dotted lines with open circles indicate drained sites. Values are estimated marginal means.

generally species-poor ecosystems and the species turn-over after drainage is rather low (e.g. Vasander 1982). In fens, where species richness is generally higher and the effects of drainage on vegetation are stronger (Minkkinen et al. 1999), relatively stronger impacts could be expected also following restoration. Consistent with this view, the changes in the vegetation composition at the Restored Fen appeared more significant than in the Restored Bog.

The decline of the total forest-moss cover and the cover of *Vaccinium myrtillus*, a species known to be strongly favored by drainage in boreal peatlands (e.g. Laine et al. 1995), further support the conclusion that restoration was successful in reversing the degradation of peatland vegetation caused by drainage at the fen site. The increased abundance of *Sphagnum balticum* similarly verifies the results of ordination analysis for the bog as the species typically grows in wet hollows of pristine bogs and requires a constantly high water-table (Laine et al. 2009).

*Eriophorum vaginatum* was the species with the strongest response to restoration. In the fen, its abundance greatly increased during the first three years after the restoration. As suggested by Jauhiainen et al. (2002), however, the *E. vaginatum*-dominated stage was only a transitional stage toward *Sphagnum*-dominated vegetation. Ten years after the restoration, the abundance of *E. vaginatum* had declined close to the level before restoration in the Restored Fen. In the Restored Bog, the abundance of *E. vaginatum* still continued to increase through the 10-year monitoring period, possibly due to the lower initial abundance or the lower nutrient status in the Bog, as compared to the fen (Komulainen et al. 1999). Our results suggest that the early successional traits of vegetation after restoration may differ between peatland types. As *E. vaginatum* has a strong impact on nutrient cycling (e.g. immobilization of N and P) and the vegetation succession of peatlands (Silvan et al. 2004), initial differences in succession will possibly lead to differences in the longer term as well. Therefore, an assessment of the efficiency of restoration based on the vegetation changes shortly after restoration is unreliable, and long-term monitoring of the effects of restoration is necessary. Care must be taken when the monitoring results of one restored peatland type are generalized to another type of peatland, especially when based on changes in only a few species.

Although most of the results above indicate a successful reversal of the drainage-induced degradation by restoration, there are still clear differences in the composition of plant species between the restored sites and comparable pristine peatlands. In the bog, the total cover of *Sphagnum* mosses increased only from 20 to 50% in a 10-year period after restoration, which is clearly less than expected in the natural condition. Furthermore, certain common vascular plant species typical of wet hollows of bogs in the region (*Scheuchzeria palustris* [Rannoch-rush], *Carex limosa* [Mud sedge]) were absent. In the fen, in addition to certain typical fen vascular plants (e.g. *C. rostrata*), all typical *Sphagnum* species indicating minerogenic influence (e.g. *S. fallax* [Sphagnaceae], *S. flexuosum* [Sphagnaceae]) were still absent 10 years after

restoration. Instead, the ground layer consisted of generalist *Sphagnum* species with wide ecological niches and therefore tolerant of ombrogenic environments also.

The differences in the composition of vascular plants between pristine and restored peatlands may be due to the slow recovery rate of plant communities after restoration, as caused by the poor dispersal potential of the absent peatland specialists, the competitive advantage of the indifferent peatland species survived through the drainage phase, or by the unsuitability of the habitats despite the restoration (Campbell & Rochefort 2003; Seabloom & van der Valk 2003; Mälson et al. 2008). The slow rate of re-establishment of some *Sphagnum* species characteristic of natural peatlands may be explained by the effective vegetative reproduction by the generalist *Sphagna* that has survived the drainage phase at the restored sites, and potentially hinder the immigration of new species by wind-borne diaspores (Rydin 1993). In southern Finland, populations of most *Sphagnum* species exist within comparably short ranges from restoration areas and dispersal by wind-borne spores are probable.

## Conclusions

Restoration has resulted in a long-lasting rise of the water-table in the studied peatlands. Changes in peat elemental concentrations caused by restoration indicated that the trend of degradation by drainage was reversed, and for most elements the concentrations reported from comparable pristine peatlands were reached during the 10-year monitoring period after restoration. The increase in concentrations of K and Mn at both studied sites indicates a recovery of ecosystem functionality in terms of nutrient cycling between the peat and vegetation. Although our results indicate a reversal of the degradation due to drainage, the vegetation of the restored sites clearly differed from the comparable target communities of pristine peatlands 10 years after the restoration. Moreover, all variables studied in this project still showed significant changes during the period of 3–10 years after the restoration, thus demonstrating the need for long-term monitoring in restoration projects.

## Implications for Practice

- Active restoration measures, rather than relying on the natural overgrowth of ditches, are needed if the reversal of the effects of drainage in *Sphagnum* peatlands is desired in a time-scale relevant to the halting of the current loss of peatland biodiversity.
- Our results show that actively filling in and blocking ditches with peat is a useful measure to induce changes in water-table, peat elemental concentrations, and vegetation toward pristine target ecosystems.
- Changes in the ecosystem due to degradation (e.g. subsidence of peat), may cause problems in recovery that can be seen only years after the restoration. Thus, careful planning and execution of restoration measures as well as long-term monitoring of the impacts are needed in the restoration projects.

- Here we present a method for evaluating the recovery of the vegetation composition of degraded ecosystems without natural control sites. The method uses the distances between site scores and selected indicator species scores within DCA ordinations.

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