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Experimental restoration of a fen plant community after peat mining

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Abstract

Question: Which restoration measures (introduction of donor diaspore material, application of straw mulch, alteration of residual peat depths) contribute to the establishment of a fen plant community on minerotrophic surfaces after peat mining?

Location: Rivière-du-Loup peatland, southern Québec, Canada at 100 m a.s.l.

Methods: The effectiveness of introducing fen plants with the application of donor diaspore material was tested. The donor diaspore material, containing seeds, rhizomes, moss fragments, and other plant propagules, was collected from two different types of natural fens. We tested whether the application of straw mulch would increase fen species cover and biodiversity compared to control plots without straw mulch. Terrace levels of different peat depths (15 cm, 40 cm, and 56 cm) were created to test the effects of different environmental site conditions on the success of re-vegetation.

Results: Applying donor seed bank from natural fens was found to significantly increase fen plant cover and richness after the two growing seasons. Straw mulch proved to significantly increase fen plant richness. The intermediate terrace level (40 cm) had the highest fen plant establishment. Compared to reference sites, the low terrace level (15 cm) was richer in base cations, whereas the high terrace level (56 cm) was much drier.

Conclusions: The application of donor diaspore material was demonstrated as an effective technique for establishing vascular fen plants. Further rewetting measures are considered necessary at the restoration site to create a fen ecosystem rather than simply restoring some fen species.

Keywords: Diaspore; Fen restoration; Peat depth, Re-vegetation; Straw mulch, Water regime.

Nomenclature: Gleason & Cronquist (1991) for vascular plants; Anderson (1990) for *Sphagnopsida*; Anderson et al. (1990) for other mosses.

Introduction

Even after several decades post-abandonment, little spontaneous vegetation may occur on mined peat fields in North America (Lavoie et al. 2003). Restoration ecology research has focused on the rehabilitation of a *Sphagnum*-dominated peatland because these sites typically have *Sphagnum*-based peat surfaces (Rochefort et al. 2003). Specifically, these procedures include: the introduction of donor diaspore material from natural bogs, rewetting the site by blocking drainage canals and creating bunds, and applying straw mulch to improve the microclimate of the peat surface (Quinty & Rochefort 2003). Occasionally, mined peat sites in Canada have been abandoned with a peat surface that is comprised of sedge fragments and water and peat chemistry similar to moderate-rich fens rather than bogs (*sensu* Du Rietz 1949; Zoltai & Vitt 1995). In such cases, the restoration of abandoned minerotrophic peat towards a moderate-rich fen has been recommended (Wind-Mulder et al. 1996; Wind-Mulder & Vitt 2000), yet few such studies exist in North America (Cooper et al. 1998; Cooper & MacDonald 2002).

The availability of viable seeds or other propagules at a site determines the initial development of a plant community (Bakker & Berendse 1999; Mitsch & Gosselink 2000). The spontaneous colonization of fen plants on mined peat sites is constrained by a lack of suitable propagules. The residual peat is devoid of plants and a viable seed bank (Salonen 1987), and natural areas surrounding mined peat fields are typically bogs with few or no fen species present (Poulin et al. 1999; Campbell et al. 2003). Introducing suitable species to cut-over sites may be necessary to promote the development of a fen plant community (Wheeler & Shaw 1995). Previous fen restoration studies have revegetated sites by sowing seeds, or transplanting seedlings, rhizomes, plant cuttings, or hay (Patzelt 1998; van Duren et al. 1998; Pfadenhauer & Grootjans 1999; Roth et al. 1999; Cooper & MacDonald 2002).

Another method of introducing plants is by importing substrate and its diaspores from a nearby donor wetland community (Mitsch & Gosselink 2000). The application of donor diaspores has proven to be a successful plant introduction technique for the restoration of bogs (Rochefort et al. 2003), marshes (Brown & Bedford 1997; Stauffer & Brooks 1997) and fens (Kratz & Pfadenhauer 2001).

The environmental conditions of a restoration site must match the biological requirements of the target species (Pfadenhauer & Grootjans 1999). Following peat mining, the environmental conditions of an abandoned field are extremely harsh for plant re-establishment (Salonen 1987, 1992; Campbell et al. 2003). The physical properties of peat deteriorate due to the effects of long-term drainage and compression from peat mining operations (Okruszko 1995; Price et al. 2003). Rewetting has been identified as the most important prerequisite for rapid regeneration of vegetation on mined peat surfaces (Sliva & Pfadenhauer 1999; Rochefort 2000). The microclimatic conditions are harsh due to an absence of vegetation cover, and the surface peat may form impenetrable crusts prone to frost heaving (Salonen 1987; Groeneveld & Rochefort 2002). The application of mulches improves the microclimatic conditions by moderating the surface temperatures and increasing the relative humidity and soil moisture (Price et al. 1998).

The aim of this project was to restore a moderate-rich fen plant community on a minerotrophic peat surface. An experimental approach was taken to determine the effect of different vegetation treatments, mulch treatments, and different residual peat depths, on the establishment of fen vegetation. The first hypothesis was that the application of donor diaspores from natural fens would increase the cover and richness of fen species compared to control plots. Secondly, we hypothesised that the application of straw mulch would increase fen species cover and biodiversity compared to control plots without straw mulch. Thirdly, using different terrace levels, we hypothesised that fen species cover and biodiversity would be highest on the terrace level that most closely matched the environmental conditions of the donor sites. In conjunction, a descriptive comparison of the abiotic factors at the restoration site and two natural fens was used to determine the environmental conditions that were potentially limiting fen restoration.

Material and Methods

Site description

Restoration site

The restoration site is part of the Rivière-du-Loup peatland, located ca. 155 km east of Québec City, Canada, between the south shore of the St. Lawrence River and the Appalachian foothills (47°50' N, 69°25' W, ca. 100 m a.s.l.). It is classified as a low boreal peatland (Anon. 1988a), and is a large complex of ombrotrophic bogs dissected with *Alnus* swamps (Gauthier & Grandtner 1975). The peat lies on marine clays of the Goldthwait Sea and the thickness of the deposit may reach 10 m in undisturbed sites (Dionne 1977). The regional climate is characterized by cold winters and warm summers with January and July mean temperatures of –12 and 18 °C, respectively. The mean annual precipitation is 924 mm, of which 73% falls as rain (Environment Canada; Anon. 1993).

The restoration site included two adjacent fields (30 m × 60 m) separated by a central drainage ditch. Residual peat at the centre of the fields averaged 65 cm, and decreased towards the drainage ditches, where the residual peat averaged 20 cm. No vegetation was present on the fields, which were abandoned in September 2000. The peat was composed of matted sedges interspersed with coniferous wood. Preliminary chemical analyses indicated that the peat was characteristic of a fen. The underlying mineral soil was primarily clay with deposits of sand, gravel, and occasional boulders.

Donor sites

Field reconnaissance to locate donor sites revealed that there were few natural fens nearby the restoration site. This lack partly reflects the gentle topography of the Lower St. Lawrence floodplain and the long period of time since deglaciation. Palaeo-ecological studies indicate that fens were once common in landscape depressions of the Lower St. Lawrence River, forming an early seral stage of today's ombrotrophic bogs, including the Rivière-du-Loup bog (Lortie 1983; Garneau 1998; Lavoie et al. 2001). Two natural fens were found in the foothills of the Appalachian Mountains, ca. 25 km southwest of the restoration site. These fens were chosen as donor sites based on their proximity to the restoration site, accessibility, and contrasting plant communities and environmental site conditions.

The first donor site is a basin fen (National Wetlands Working Group, Anon. 1997) dominated by *Sphagnum* species (hereafter referred to as *Sphagnum* fen). It is a small fen (0.5 ha, 47°77' N, 52°83' W, ca. 274 m a.s.l.) receiving minerotrophic water from a small stream to the north and surface runoff from a slope on its western side. The donor area (25 m × 25 m) was positioned in the centre of the peatland where the peat depth averaged 86 cm. The site is characteristic of a poor fen based on the vegetation composition and water and peat chemistry (Zoltai & Vitt 1995). The main species (in order of dominance at the site, cover > 2%) are *Sphagnum centrale, Sphagnum flexuosum, Utricularia minor, Polytrichum strictum, Calamagrostis canadensis, Salix pyrifolia, Picea mariana, Glyceria canadensis, Sphagnum capillifolium, Carex canescens,* and *Sphagnum magellanicum*.

The second donor site is a riparian stream fen (Anon. 1997), dominated by *Calamagrostis canadensis* (hereafter referred to as *Calamagrostis* fen). It is a small fen (1.5 ha, 48°19' N, 52°81' W, ca. 320 m a.s.l.) receiving minerotrophic water from a stream entering the peatland on the north side, coursing through the main body of the fen and emptying into a small pond on the southern end. A beaver dam was found upstream of the fen in June 2001, which caused the water table to rise from below the surface to create flooded conditions for the remainder of the study period. The donor area (25 m × 25 m) was positioned in the centre of the fen where the peat depth averaged 85 cm. The fen is a moderate to rich fen (Zoltai & Vitt 1995). Other species that dominate (cover > 2%) the site are *Warnstorfia exannulata, Carex utriculata, Scirpus cyperinus, Utricularia minor* and *Caltha palustris*.

Experimental design

The experiment was a split-plot factorial design. In total, 54 plots (3 terrace levels × 3 blocks (replicates) × 3 vegetation treatments × 2 straw mulch treatments) were established.

Terrace levels were treated as main plots and were divided into three blocks to minimize the effect of variance within the site. The vegetation and straw treatments were treated as subplots and were randomly assigned within the blocks (App. 1).

The installation of the experiment commenced in April 2001, just after snow melt. The convex shape of the abandoned fields was modified to create three terraces of decreasing elevation, with different peat depths, on either side of and parallel to the main drainage ditch. The terraces were levelled with a machine grader that scraped excess peat off the site. The terrace levels are referred to as high, middle, and low, with an average peat depth of 56, 40 and 15 cm, respectively. The terrace levels could not be randomly positioned due to topographic constraints of the site. The central drainage canal was blocked, while a secondary ditch upslope of the site was unblocked. Peat mining operations continued on fields' upslope of the restoration site throughout the study period, and blocking of these drainage ditches was not permitted. Berms were created on the down slope side of the terrace levels to hold water on the site, and prevent erosion. Each berm was ca. 0.5 m wide and 0.3 m in height.

Prior to the application of the vegetation and straw treatments, plots were raked to break up the surface crust, minimize inconsistencies of compaction, and reduce microtopography resulting from the machinery. Phosphorus fertilizer (2 g.m^{-2}) was subsequently applied, as recommended for bog restoration to favour vascular plant establishment (Rochefort et al. 2003). Experimental plots (5 m × 5 m) were established on the terraces and were separated by a 1-m buffer. The vegetation treatments were (1) donor diaspore material from the *Sphagnum* fen, (2) donor diaspore material from the *Calamagrostis* fen, and (3) a control, without donor diaspore material applied. The donor diaspore material was collected from 18 (1.25 m × 1.25 m) random quadrats located within the donor area (25 m × 2 m). The ratio of donor diaspore collection area to restored area (1:16) was similar to that suggested for bog restoration (Campeau & Rochefort 1996). The top 10 cm of substrate and vegetation from each donor quadrat was collected by hand and transported to the restoration site, where it was broken into small pieces and spread by hand. The mulch treatments were (1) straw, and (2) a control without straw. The straw was applied with a density of 1500 kg.ha^{-1} and was spread to exceed the plot boundary to minimize edge effects. Vegetation and mulch treatments were applied to the restoration site during the week of 7-11 May 2001.

Site monitoring

Vegetation surveys

Percent cover of the vegetation at the restoration site was surveyed from 10-14 October 2001 and from 8-13 August 2002. Ten quadrats (30 cm × 30 cm) in each experimental plot were sampled. The percent cover (visually estimated) for each plant species within each quadrat was recorded. Sampling omitted the border area (0.5 m on each side) of the plots to minimize the observation of edge effects.

The donor sites were surveyed on 13 August 2002. Three transects were randomly placed within the donor area, along which ten quadrats (30 cm × 30 cm) were sampled every 30 cm for a total of 30 quadrats. The percent cover (visually estimated) of each plant species was recorded, and a species list was compiled including any additional species that were noted within the donor area.

A mined peat field (30 m × 120 m, surrounded by drainage ditches) adjacent to the restoration site (approximately 20 m away on the nearest edge) was surveyed to determine which plants had the potential to spontaneously colonize the restoration site. The field had been abandoned five years previously. The field was systematically sampled with the line transect method (Bonham 1989). A transect was set every 8 m along the field, and points were measured at the centre of the ditch, 1 m from the ditch, every 5 m along the field, and in the centre of the ditch on the far side, for a total of 216 sampling points. A species was recorded as present if it contacted the front side of a 2-mm diameter sampling rod of infinite height.

Environmental site conditions

The regional precipitation during the 2001 and 2002 growing seasons were assessed by comparing rainfall data collected from a meteorological station at the Bois-des-Bel peatland (15 km northeast of the current study site) to 30 year averages collected nearby at the St. Arsene meteorological station (Environment Canada, Anon. 1993).

The water table depth and soil water pressure (–5 cm depth) were measured following the methodology of Price et al. (2002). Six wells and six tensiometers were placed equidistantly along the centre-line of each terrace, for a total of 18 hydrological stations at the restoration site. Both the water table depth and soil water pressure were measured twice a week during the 2001 growing season. The water table depth at the donor sites was measured periodically throughout the first growing season from three wells that were placed equidistantly across the donor sites.

Peat and water chemistry was determined on several occasions throughout the first growing season. Three random samples were collected along each terrace at the restoration site, and from each donor site. On two occasions during the first growing season, and once during the second growing season, samples were collected from each experimental plot at the restoration site and from three random locations at each donor site. Surface peat samples (0-5 cm depth) were collected from the restoration site and the donor fen sites. Water samples were collected from surface water at the donor fen sites; however the dry conditions of the surface peat prevented their collection at the restoration site. All samples were taken to laboratory immediately for analysis or stored in a refrigerator at 4 °C until they could be analysed, typically within one week. Water chemistry samples were obtained from the peat samples collected at the restoration site with the addition of distilled water, and extraction of the solution with a filter and a vacuum apparatus. Water samples collected from the donor sites were analysed directly. The pH was measured using a pH meter (Accumet pH meter Model 950). The electrical conductivity was measured with a conductivity cell (YSI Model 32), adjusted to 20 °C, and corrected for hydrogen ions (Sjörs 1952). An inductively coupled argon plasma spectrophotometer (ICP-OES Optima 4300DV of Perkin Elmer) was used to determine the concentrations of Na, K, Ca, Mg, PO_4-P, Fe, Cu, Mn and Zn (Golterman et al. 1978). Peat samples were similarly analysed for the total concentrations of these elements after standard dry ashing procedures at 500 °C (Van Loon 1985). The concentrations of nitrogen (total nitrogen, nitrate-nitrogen, and ammonia-nitrogen) were determined with colorimetric methods, using NaOH, Rochelle's salt and Nessler's reagent (Golterman et al. 1978).

Data analyses

The plant species at the restoration site after the first and second growing season were listed by percent cover and assigned to have been introduced via donor diaspore material, spontaneously recolonized from neighbouring sites, or introduced via the straw mulch. These assessments were based on the plant surveys at the donor sites, the field neighbouring the restoration site, and known agricultural species (Frankton & Mulligan 1993).

Fen plant cover and fen plant richness (total cover and number of species introduced via donor diaspore material) were averaged for each plot at the restoration site. Due to the vegetative (i.e. non random and competitive) growth of *Tussilago farfara* and *Equisetum arvense* and their predominance on the low terrace level, their cover was averaged and analysed separately. Percent cover data was $\log_{10}$ transformed to improve the normality of the residuals. A split-plot ANOVA with a randomized block design was applied to test the effect of the experimental treatments on the dependent variables. A Tukey test was used to isolate differences between the treatments effects when no interactions were significant (Zar 1984). Significant interactions were tested for treatment effects with the splice technique. All statistical operations were performed with SAS version 4.0 (Anon. 1988b).

Results

Vegetation

Two growing seasons after the re-introductions the majority of established plants were fen species, in terms of percent cover and richness. The total species cover at the restoration site doubled during the second year from 12 to 35%. The abundance of fen plants within the experimental plots increased from 5% in the first year to 20 % in the second year (Table 1). The plant community was composed primarily of forbs and graminoids. There

Table 1. Mean abundance (percent cover) of plant species at the restoration site after the first (2001) and second (2002) growing seasons. The probably source for each species is categorized (x) as donor fen, spontaneous, and straw, based on plant surveys of the natural fen donor sites, a field neighbouring the restoration site, and species common to agricultural areas, respectively.

Species	% cover (2001)	2002	Donor fen	Spontaneous	Straw
Agrostis hyemalis	(0.5)	5.3	x	x	-
Equisetum arvense	(0.8)	5.0	-	x	x
Tussilago farfara	(2.7)	4.9	-	x	-
Carex crawfordii	(0.2)	3.7	x	-	-
Scirpus cyperinus	(0.4)	2.9	x	x	-
Glyceria canadensis	(0.3)	1.9	x	-	-
Rorippa palustris	(0.5)	1.4	-	-	x
Juncus effusus	(0.2)	1.4	x	x	-
Carex canescens	(0.2)	0.9	x	-	-
Lycopus uniflorus	(0.2)	0.9	x	x	-
Euthamia graminifolia	(0.2)	0.8	-	x	-
Polygonum hydropiper	(1.1)	0.7	-	x	-
Hieracium spec.	(0.2)	0.6	-	x	x
Salix spp.	(0.2)	0.5	x	x	-
Galium trifidum	(0.9)	0.5	x	-	-
Calamagrostis canadensis	(0.3)	0.4	x	-	-
Viola macloskeyi	(0.2)	0.3	x	-	-
Ranunculus pensylvanicus	(0.2)	0.3	x	-	-
Bidens cernua	(0.4)	0.3	-	x	-
Fragaria virginiana	(> 0.2)	0.3	x	-	-
Epilobium ciliatum	(> 0.2)	0.2	x	-	-
Juncus brevicaudatus	(> 0.2)	0.2	x	-	-
Secale cereale	(0.6)	> 0.2	-	-	x
Dicranella cerviculata	(0.3)	> 0.2	-	x	-
Avena sativa	(0.3)	> 0.2	-	-	x
Total cover	**(12)**	**35**			
Fen cover	**(5)**	**20**			
Total richness	**(21)**	**18**			
Fen richness	**(12)**	**11**			

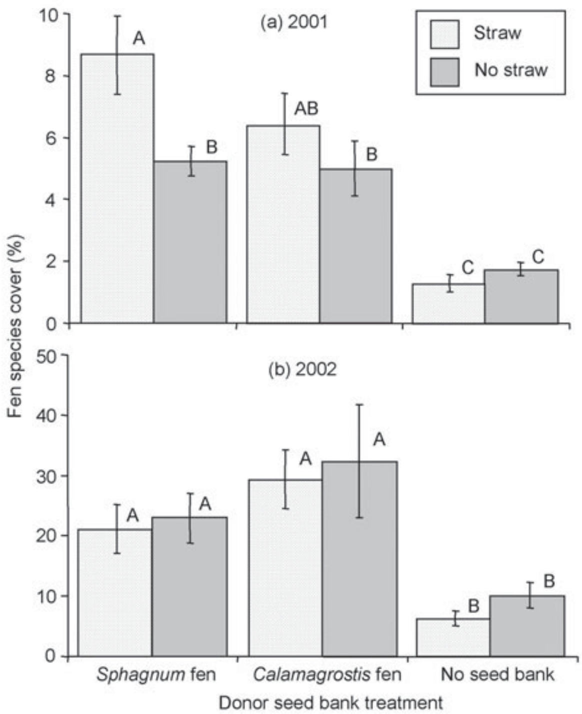

Fig. 1. Effect of donor seedbank and straw mulch treatments on fen species cover (%) after one growing season (**a**), and the second growing season (**b**). Error bars show SE. Different letters within graphs represent significant differences between treatments identified by the splice function for significant two-way interactions (**a**), and Tukey tests for within treatments without significant interactions (**b**).

was a small component of woody plants, while bryophytes were largely absent. Trace amounts of bryophyte species were observed in several plots in the first year that were no longer present in the second year. Control plots without straw or diaspore material were primarily devoid of vegetation, while many treatment plots had abundant growth.

Several species established at the restoration site from the donor fen diaspore material, including *Glyceria canadensis, Carex canescens, Galium trifidum, Calamagrostis canadensis, Viola macloskeyi, Ranunculus pensylvanicus, Fragaria virginiana, Epilobium ciliatum,* and *Juncus brevicaudatus*. Several other fen species were either introduced via the donor diaspore material or present in the local seed rain, such as *Agrostis hyemalis, Scirpus cyperinus, Juncus effusus,* and *Lycopus uniflorus* (Table 1). *Equisetum arvense* and *Tussilago farfara* established an extensive cover on the low terraces. These species are commonly observed growing along the ditches of the cut-over peat fields; their rhizomes were likely left in the ground during preparation of the terraces. Straw mulch introduced a few agricultural species including *Secale cereale, Rorippa palustris,* and *Avena sativa*. By the end of the second year only *Rorippa palustris* was still present.

Donor diaspore treatments

Donor diaspore treatments (from *Sphagnum* fen and *Calamagrostis* fen) increased the abundance of fen species after the first and second growing seasons (Fig. 1). During the first year the combination of *Sphagnum* donor diaspore and straw mulch treatments favoured the abundance of fen species cover and produced the highest total fen species cover of all experimental treatments (9 ± 1%) (Table 2; Fig. 1a). Several herbaceous species proliferated with the combined treatments of *Sphagnum* fen diaspore material and straw mulch including: *Viola macloskeyi, Lycopus uniflorus,* and *Galium trifidum*. After the second growing season plots treated with *Calamagrostis* fen diaspore material (31 ± 5%) tended to have higher fen cover than *Sphagnum* fen diaspore material (22 ± 3%), although these differences were not significant. Both diaspore treatments were significantly higher than control plots (8 ± 1%) (Fig. 1b).

Table 2. Split plot ANOVA results for the effect of experimental treatments on fen plant cover and richness after the first and second growing season, and *Tussilago farfara* and *Equisetum arvense* cover (*T+E*) after the second growing season. F-values are followed by *p*-values in parentheses. Significant *p*-values ($p < 0.05$) are indicated in bold type.

Year		2001	2001	2002	2002	2002
Source of variation	d.f.	Fen cover	Fen richness	Fen cover	Fen richness	*T+E* cover
Terrace	2	1.68 (0.16)	1.89 (0.82)	10.76 (0.38)	4.05 (0.99)	18.19 **(0.01)**
Block	2	3.02 (0.29)	0.21 (0.26)	1.27 **(0.025)**	0.01 (0.11)	0.75 (0.53)
Terrace*Block	4					
Diaspore material	2	61.54 **(0.0001)**	99.08 **(0.0001)**	15.90 **(0.0001)**	28.89 **(0.0001)**	0.09 (0.91)
Straw	1	2.62 (0.12)	3.39 (0.07)	1.13 (0.30)	20.73 **(0.0001)**	1.27 (0.27)
Diaspore material*straw	2	4.82 **(0.015)**	2.70 (0.08)	0.52 (0.60)	0.03 (0.97)	1.55 (0.23)
Diaspore material*terrace	4	1.97 (0.15)	1.14 (0.36)	0.27 (0.90)	1.80 (0.15)	0.88 (0.49)
Straw*terrace	2	0.04 (0.96)	2.11 (0.13)	0.08 (0.92)	0.41 (0.66)	0.17 (0.84)
Diaspore material*straw*terrace	4	0.56 (0.69)	0.69 (0.60)	1.90 (0.14)	1.45 (0.24)	0.02 (0.99)
Error	30					

After the first growing season, the fen plant richness was significantly highest where *Sphagnum* fen diaspore material (18 ± 1 taxa) had been applied, intermediate with the application of *Calamagrostis* fen diaspore material (13 ± 1 taxa) and lowest without donor diaspore material (6 ± 1 taxa) (Fig. 2a). The richness of fen plants decreased from the first to second year, and there was no longer a significant difference between the types of donor diaspore material applied (*Sphagnum* fen = 13 ± 1; *Calamagrostis* fen = 12 ± 1 taxa). Nevertheless, the application of donor diaspore material increased the fen plant richness compared to the control (7 ± 1 taxa) (Fig. 2b).

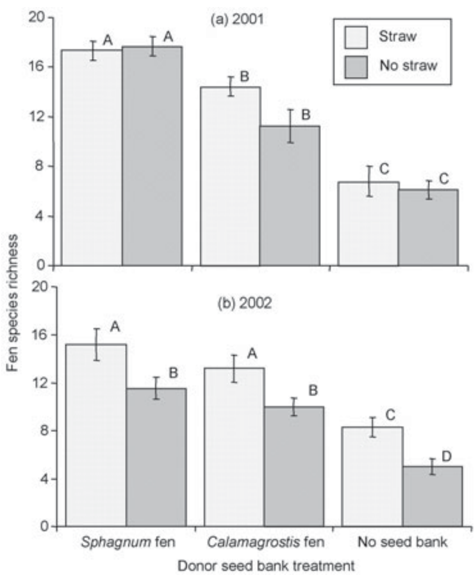

Fig. 2. Effect of mulch treatments on fen species richness after one growing season (**a**), and two growing seasons (**b**). Error bars show SE. Different letters represent significant differences within treatments identified by a Tukey test. There were no significant interactions between treatments.

Straw mulch

The application of straw mulch did not improve the abundance of fen plants after two growing seasons. Only during the first year did straw mulch improve the cover of fen plants from 5 to 9 % in combination with *Sphagnum* fen diaspore material (Fig. 1a). These initial increases in fen plant cover did not extend to the second year. More notably, straw mulch clearly increased the richness of fen species from both diaspore materials after two years. Fen plant richness was higher for plots treated with straw mulch (12 ± 1 taxa) compared to plots without straw mulch (9 ± 1 taxa) (Fig. 2b). *Rorippa palustris* was the only species introduced with the straw mulch that persisted over two growing seasons at the restoration site (Table 1).

Terrace level

After two growing seasons there was significantly more fen species cover on the middle terrace level (27 ± 5%) than the high terrace level (14 ± 2%). The mean fen species cover on the low terrace level (20 ± 4%) was between the values observed at the middle and high terrace levels, and was not significantly different from

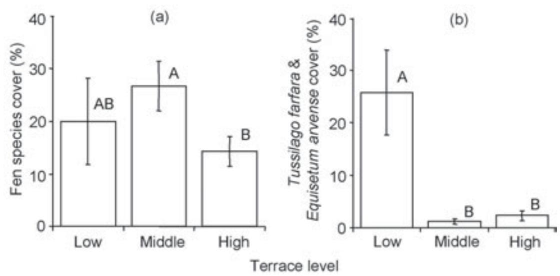

Fig. 3. Effect of terrace levels on fen species cover (**a**), and *Tussilago farfara* and *Equisetum arvense* cover (**b**) after two growing seasons (2002). Error bars show SE. Different letters represent significant differences within treatments identified by Tukey tests.

Table 3. Means and standard deviations of the environmental conditions at the terraces of the restoration site and the natural fen donor sites.

	Restoration site				Donor sites		
	n	Low terrace	Middle terrace	High terrace	*n*	*Sphagnum* fen	*Calamagrostis* fen
Hydrology							
Water table (cm)	222	– 28.6 ± 15.1	– 33.5 ± 15.6	– 45.2 ± 17.2	3	– 3.5 ± 3.2	7.6 ± 8.1
Soil-water tension (mb)	223	– 41.3 ± 5.2	– 62.5 ± 86.3	– 66.3 ± 77.0	0	Not measured	Not measured
Water chemistry							
pH	78	5.9 ± 0.5	6.0 ± 0.3	5.9 ± 0.2	14	5.5 ± 0.4	5.8 ± 0.3
Electrical conductivity (µS.cm^{-1})	78	338 ± 495	132 ± 117	134 ± 106	14	27 ± 9	40 ± 30
P total (mg.l^{-1})	54	0.6 ± 0.5	0.5 ± 0.4	0.5 ± 0.4	5	0.3 ± 0.5	0.2 ± 0.2
NH_4^+ (mg.l^{-1})	54	1.3 ± 2.1	3.0 ± 3.3	3.6 ± 3.8	5	1.2 ± 2.3	1.0 ± 2.0
NO_3^- (mg.l^{-1})	54	1.5 ± 1.5	2.1 ± 1.8	2.2 ± 2.1	5	0.9 ± 0.7	1.6 ± 1.1
K^+ (mg.l^{-1})	54	6.5 ± 4.9	3.3 ± 2.2	3.0 ± 1.8	5	1.4 ± 0.3	0.4 ± 0.4
Ca^{2+} (mg.l^{-1})	54	46.8 ± 73.9	9.5 ± 18.2	5.8 ± 4.3	5	1.8 ± 2.2	3.4 ± 1.6
Mg^{2+} (mg.l^{-1})	54	28.1 ± 58.2	4.6 ± 10.2	2.7 ± 2.9	5	0.4 ± 0.5	0.3 ± 0.1
Na^+ (mg.l^{-1})	54	50.1 ± 34.8	26.1 ± 13.5	25.9 ± 15.6	5	2.0 ± 1.1	2.7 ± 1.7
Fe^{3+} (mg.l^{-1})	54	0.5 ± 2.5	0.4 ± 0.3	0.6 ± 0.3	5	0.1 ± 0.1	0.7 ± 0.4
Cu^- (mg.l^{-1})	54	0.4 ± 1.36	0.2 ± 0.1	0.2 ± 0.1	5	0.1 ± 0.0	0.2 ± 0.1
Peat chemistry							
P (mg.g^{-1})	42	0.42 ± 0.26	0.27 ± 0.21	0.33 ± 0.36	7	0.58 ± 0.15	1.18 ± 0.61
N (mg.g^{-1})	42	19.94 ± 23.16	18.98 ± 30.26	14.96 ± 36.79	7	10.11 ± 16.99	19.92 ± 35.55
K (mg.g^{-1})	42	0.61 ± 0.34	0.22 ± 0.38	0.38 ± 0.16	7	0.79 ± 0.36	0.57 ± 0.40
Ca (mg.g^{-1})	42	9.92 ± 1.79	10.57 ± 12.67	10.13 ± 15.10	7	7.58 ± 1.46	5.15 ± 3.03
Mg (mg.g^{-1})	42	2.58 ± 0.42	3.80 ± 5.08	3.55 ± 4.15	7	1.01 ± 0.27	0.78 ± 1.07
Na (mg.g^{-1})	42	0.67 ± 0.19	0.43 ± 0.18	0.35 ± 0.10	7	0.24 ± 0.45	0.29 ± 0.94
Fe (mg.g^{-1})	42	4.17 ± 0.98	4.57 ± 0.74	4.34 ± 0.73	7	1.73 ± 0.51	2.17 ± 0.84

either (Fig. 3a). *Equisetum arvense* and *Tussilago farfara,* the second and third most dominant species after two growing seasons (Table 1), were dominant on the low terrace level (26 ± 8 %), whereas they formed only a minor component of the plant cover on the middle (1 ± 1%), and high terrace (2 ± 1%) levels (Fig. 3b). No other experimental treatments had an effect on their establishment (Table 2) and their dominance on the low terrace is presumed to be due to their establishment along the ditches prior to the experiment, rather than a treatment effect.

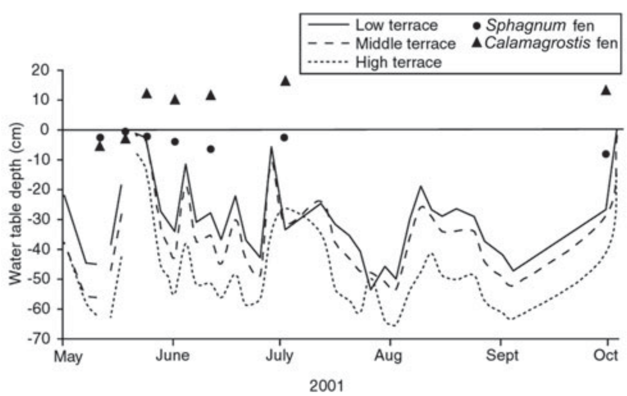

Fig. 4 . Water table depth (cm) for the terrace levels at the restoration site and the natural fen donor sites throughout the first growing season.

Environmental conditions

Hydrology

From May to August 2001 and 2002 the total rainfall was 286 and 253 mm, respectively, compared to the mean 30-yr seasonal total of 353 mm (Environment Canada 1993). The lower terraces were more proximal to the water table and to the underlying clay substrate. The mean depth to water table was –29, –34 and –45 cm for low to high terraces, respectively (Table 3). This resulted in a water table that sloped toward the central drainage ditch, with a gradient of approximately 0.032 and 0.048 during the wettest (June 4, 2001) and driest (August 16, 2001) period, respectively. Except for brief periods immediately following rain events, the water

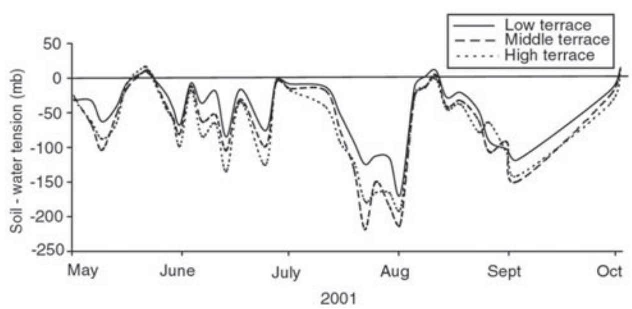

Fig. 5. Soil water tension (mb) for the terrace levels at the restoration site throughout the first growing season.

table in the lowest terrace was always within the clay substrate. In the middle and upper terrace the water table was generally within the peat layer except for during the driest periods. The water table depth at the restoration site was far lower than the donor sites throughout the 2001 growing season (Fig. 4). The water table at the donor sites was around – 4 cm at the *Sphagnum* fen and + 8 cm at the *Calamagrostis* fen (Table 3).

Mean soil-water pressure decreased below –100 mb from mid-July to mid-August (Fig. 5). Soil-water pressure was less than – 100 mb at the low, middle, and high terraces 16%, 24%, and 24% of the time, respectively. Soil-water pressure in the upper and middle terrace was similar (averaging – 66 and – 62 mb, respectively), in spite of a notable difference in water table.

Water chemistry

In general, the concentrations of minerals and nutrients at the restoration site were higher than at the natural fen donor sites (Table 3). The mean pH levels of the terraces at the restoration site did not vary greatly (ca. 5.9), and were similar to the *Calamagrostis* fen donor site (5.8), while the *Sphagnum* fen donor site had a lower mean pH (5.5). The electrical conductivity of the restoration site was higher than that of the natural fens and the low terrace values were higher than the other two terraces (Table 3). The major cations followed a pattern similar to the electrical conductivity.

The nitrogen and phosphorus levels at the restoration site were moderate and are considered mesotrophic (Bridgham et al. 1996). The mean concentration of available P at the restoration site was higher than the natural fens, but within the observed range of variability. The concentrations of ammonium and nitrate on the lowest terrace level of the restoration site were similar to the natural fens, whereas the middle and high terrace levels tended to have higher concentrations (Table 3).

Peat chemistry

Total nutrients (N, P, and K) of the restoration site tended to be lower or equal to the donor fens (Table 3). The mean total nitrogen concentration at the restoration site (17.96 mg.g^{-1}) was between the mean concentration of the *Sphagnum* fen (10.11 mg.g^{-1}) and *Calamagrostis* fen (19.92 mg.g^{-1}). Total phosphorus concentrations of the peat were lower at the restoration site (0.34 mg.g^{-1}) than the *Sphagnum* (0.58 mg.g^{-1}) and *Calamagrostis* (1.18 mg.g^{-1}) fens. Similarly, potassium concentrations were generally lower at the restoration site than the natural fens.

In contrast, the concentrations of the other mineral elements present in the peat including calcium, magnesium, sodium, and iron, were higher at the restoration site than at the reference sites.

Discussion

Effect of plant diaspore re-introductions

The application of donor diaspore material from natural fens clearly increased the cover and richness of fen species compared to control plots, supporting our first hypothesis.

Diaspore material led to the establishment of several species at the restoration site that were absent from neighbouring abandoned fields and the nearby natural bog. This provides evidence that the establishment of fen vegetation is hampered by dispersal, rather than conditions at the restoration site. After two years, several graminoid and herbaceous species from the *Calamagrostis* fen formed a dominant vegetation cover at the restoration site, despite unfavourable water table conditions. However, several non-target species also became established, most notably, *Equisetum arvense* and *Tussilago farfara,* but these species appear to have spread from the adjacent ditch to the lower terrace. The establishment of invasive, weedy species has been observed elsewhere in Europe (Rowlands 2001) and North America (Cooper & MacDonald 2002) on cut-over sites with minerotrophic peat. In Finland, the abundance of *E. arvense* and *T. farfara* was lower on older minerotrophic peat fields, suggesting a decrease in dominance over time (Salonen 1990), so these species may not be a concern in the long-term. Further monitoring of the vegetation community is required to determine whether there is convergence with the donor fen plant communities over time, or increased dominance of non-target species.

The advantages of applying diaspore material as a plant introduction technique are numerous. Firstly, the variety of propagule species and types contained within the donor diaspore material increases the chances that some of the species biological requirements will match the environmental conditions of a site and the particular climatic conditions of a given year. Secondly, the inclusion of the substrate with the diaspore material means that soil mycorrhizal fungi associated with the plant community are also brought to the site. Mycorrhizal fungi may be of great importance in wetland plant communities (Turner & Friese 1998; Thormann et al. 1999). Thirdly, insect larvae and other disseminules may also be brought to the site within the substrate, further aiding plant community establishment by acting as dispersal agents (Middleton 1999). Finally, if the donor diaspore material is collected in the spring, it allows propagules to fulfil their natural dormancy cycle under their native conditions. This may be of great importance for establishing *Carex* species, an important component of fen plant communities, which have been shown to have complex dormancy cycles and species-specific germination traits (Baskin et al. 1996; Schultz 1998; Patzelt et al. 2001).

A disadvantage of applying donor diaspore material as a reintroduction method is the disturbance caused to the donor wetland community during diaspore collection. However, informal observations of the disturbed quadrats (> 4% of the reference unit) of the current study revealed that there was 25 to 40 % recovery by the end of the second growing season. Monitoring the donor sites is warranted if this method is adopted for large scale restoration projects in the future.

Effect of straw mulch

After two years, the straw significantly increased the diversity of fen plants. Only during the first year was there a synergistic effect on fen species abundance with straw mulch and *Sphagnum* fen diaspore material treatments. The straw mulch may facilitate the germination and early stages of establishment of some species due to improved water conditions (Price et al. 1998), whereas mulch may inhibit other species and later stages of plant development by shading, obstructing plants, attracting predators or promoting pathogens (Xiong & Nilsson 1997). In previous studies, cover treatments improved the germination of vascular plants, including *Eriophorum angustifolium, Molinia* and *Calluna*; however, other species such as *E. vaginatum* failed to respond to the same treatments (Sliva & Pfadenhauer 1999). Straw mulch did not improve the establishment of vascular plants in bog restoration attempts, whereas the establishment of mosses, particularly *Sphagnum* species, was substantially improved (Rochefort et al. 2003). For fen restoration sites where better rewetting can be achieved, mulching could prove to have a greater effect, particularly for helping mosses to establish.

Effect of environmental conditions created by terracing

We had hypothesised that the terrace level with environmental conditions closest to the natural fen donor sites would support the highest fen plant establishment. The intermediate terrace level had the highest fen species cover after two years. The environmental conditions of the middle terrace level may represent a compromise between the extremely dry conditions of the high terrace level, and the more minerotrophic conditions of the lowest terrace level.

Water availability is an important factor affecting fen restoration success (Roth et al. 1999), fen species distribution (e.g. Bridgham & Richardson 1993), and number of plant niches (Silvertown et al. 1999). Reintroduction of fen species by plantings was the most effective with water table levels slightly below the surface or with shallow standing water (Cooper & MacDonald 2002). The extremely dry conditions of the restoration site are likely limiting the establishment of a fen plant community. The water tables at the donor sites were consistently just below or above the surface, whereas the surface peat of the restoration site was dry throughout the growing seasons, except immediately following rain events. Further rewetting measures are considered necessary to create fen-like hydrological setting at the restoration site, and ultimately to create a fen system rather than simply restoring fen species (Grootjans & van Diggelen 1995). However, the freedom to manipulate the sites hydrology was constrained by drainage requirements of adjacent peat extraction activities.

Rainfall at the site was less than normal, but restoration measures must be designed to accommodate such a range of conditions. Vegetation is responsive to the hydrological conditions in the rooting zone, which is controlled by local water flux and storage conditions. The water table at the lowest terrace resided predominantly within the clay substrate, where the hydraulic conductivity is at least several orders of magnitude lower than in the peat in this region (Van Seters & Price 2002). Thus lateral water flow across the lowest terrace was insignificant except for a short period following major rainstorms when the water table rose into the peat layer. Some water redistribution between the upper and middle terrace occurred, where the water table generally resided in the more permeable peat layers. However, the largest water flux in cutover peatlands during the summer is vertical, where upward loss by evaporation generally exceeds percolation (Price 1996). Sustaining soil-water pressure in the rooting zone that is suitable for vegetation relies upon the upward migration of water from the water table, or from soil moisture stored above it, by capillary transport processes.

Soil-water pressure in the upper and middle terrace was similar (averaging –66 and –62 mb, respectively), in spite of a notable difference in water table. In the lowest terrace, soil water pressure was higher (averaging –41 mb), but also below the equilibrium pressure defined by the water table (Table 3). Under equilibrium conditions the soil-water pressure would be equal to the distance above the water table (Dingman 2002). The consistently low soil-water pressure suggests that capillary rise was unable to meet evaporative water losses. The close proximity of the soil surface to the water table helped sustain the water flux by capillary rise and thus elevated soil-water pressure in the rooting zone of the low terraces, compared to the upper and middle terraces. The low terraces had more complete re-saturation of the peat after significant rainfalls because the clay limited deeper water percolation. In comparison, at the upper and middle terraces, water percolation to deeper peat layers deprived the upper layer of water, resulting in lower soil water pressures.

Water quality is considered as important as water availability for fen restoration (Charman 2002; Lamers et al. 2002). In this experiment, the higher concentration of solutes in deeper peat results from the diffusion of salts from the marine clay (Van Seters & Price, unpubl. data for Cacouna peatland, Québec, 1999). Some of these solutes may migrate more quickly above the water table by capillary rise, and become more concentrated by evaporation. During the summer the net direction of solute transport is upwards, similar to the net water flow direction (i.e. driven by evaporative water losses). The concentrations of solutes within the peat substrate were considerably higher than the donor sites and may be one factor that lowered the success of fen plant recolonization. An experiment that separates the effect of water availability and solute concentration on fen plant establishment is recommended to discern their relative importance.

Nitrate has been noted to become available in large quantities, even in excessive amounts, on drained peat surfaces (de Mars et al. 1996; Wind-Mulder & Vitt 2000). The nitrate concentrations at the restoration site were higher than the donor fens; however they are not considered extreme. The high nitrate levels are likely due to the dry conditions at the restoration site, which promote mineralization. Further rewetting of the site may lower nitrate levels to those of the donor fens; however, mineralization has been observed in flooded conditions (Koerselman & Verhoeven 1995). The high nitrate concentrations at the restoration site may decrease over time without intervention, owing to increased uptake by the developing vegetation community.

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References

Anon. 1988a. *Wetlands of Canada*. National Wetlands Working Group, Polyscience Publications, Montréal, CA.

Anon. 1988b. SAS/STAT user's guide: release 6.03 edition. SAS Institute Inc., Cary, NC, US.

Anon. 1993. *Canadian climatic normals, 1961-1990, Québec*. Atmospheric Environment Service, Environment Canada, Ottawa, CA.

Anon. 1997. The Canadian wetland classification system. In: Warner, B.G. & Rubec, C.D.A. (eds.) *The Canadian wetland classification system*. National Wetlands Working Group, Waterloo Research Centre, Waterloo, CA.

Anderson, L.E. 1990. A checklist of *Sphagnum* in North America north of Mexico. *Bryologist* 93: 500-501.

Anderson, L.E., Crum, H.A. & Buck, W.R. 1990. List of the mosses of North America north of Mexico. *Bryologist* 93: 448-499.

Bakker, J.P. & Berendse, F. 1999. Constraints in the restoration of ecological diversity in grassland and heathland communities. *Trends Ecol. Evol*. 14: 63-68.

Baskin, C.C., Chester, E.W. & Baskin, J.M. 1996. Effect of flooding on annual dormancy cycles in buried seeds of two wetland *Carex* species. *Wetlands* 16: 84-88.

Bonham, C.D. 1989. *Measurements for terrestrial vegetation*. Wiley, New York, NY, US.

Bridgham, S.D. & Richardson, C.J. 1993. Hydrology and nutrient gradients in North Carolina peatlands. *Wetlands* 13: 207-218.

Bridgham, S.D., Pastor, J., Janssens, J.A. & Chapin, C. 1996. Multiple limiting gradients in peatlands: a call for a new paradigm. *Wetlands* 16: 45-65.

Brown, S.C. & Bedford, B.L. 1997. Restoration of wetland vegetation with transplanted wetland soil: an experimental study. *Wetlands* 17: 424-437.

Campbell, D.R., Rochefort, L. & Lavoie, C. 2003. Determining the immigration potential of plants colonizing disturbed environments: the case of milled peatlands in Quebec. *J. Appl. Ecol*. 40: 78-91.

Campeau, S. & Rochefort, L. 1996. *Sphagnum* regeneration on bare peat surfaces: field and greenhouse experiments. *J. Appl. Ecol*. 33: 599-608.

Charman, D. 2002. *Peatlands and environmental change*. Wiley, New York, NY, US.

Cooper, D.J. & MacDonald, L.H. 2002. Restoring vegetation of mined peatlands in the southern Rocky Mountains of Colorado, U.S.A. *Restor. Ecol*. 8: 103-111.

Cooper, D.J., MacDonald, L.H., Wenger, S.K. & Woods, S.W. 1998. Hydrologic restoration of a fen in Rocky Mountain National Park, Colorado, USA. *Wetlands* 18: 335-345.

de Mars, H., Wassen, M.J. & Peeters, W.H.M. 1996. The effect of drainage and management on peat chemistry and nutrient deficiency in the former Jegrznia-floodplain (NE-Poland). *Vegetatio* 126: 59-72.

Dingman, S.L. 2002. *Physical hydrology,* 2nd. ed. Prentice Hall, Upper Saddle River, NJ, US.

Dionne, J.-C. 1977. La mer de Goldthwait au Québec. *Géogr. Phys. Quat*. 31: 61-80.

Du Rietz, G.E. 1949. Huvudenheter och huvudgränser i svensk myrvegetation. *Sven. Bot. Tidskr.* 43: 274-309.
Frankton, C., & Mulligan, G.A. 1993. *Weeds of Canada.* NC Press, Toronto, CA.
Garneau, M. 1998. *Paléoécologie d'une tourbière littorale de l'estuaire maritime du St.-Laurent, L'Isle-Verte, Québec.* Ressources Naturelles Canada, Ottawa, CA.
Gauthier, R. & Grandtner, M.M. 1975. Étude phytosociologique des tourbières du bas Saint-Laurent, Québec. *Nat. Can.* 102: 109-153.
Gleason, H.A. & Cronquist, A. 1991. *Manual of vascular plants of northeastern United States and adjacent Canada.* 2nd. ed. The New York Botanical Garden, New York, NY, US.
Golterman, H.L., Clymo, R.S. & Ohnstad, M.A.M. 1978. *Methods for physical and chemical analysis of fresh waters.* 2nd ed. Blackwell Scientific, London, UK.
Groeneveld, E.V.G. & Rochefort, L. 2002. Nursing plants in peatland restoration: on their potential use to alleviate frost heaving problems. *Suo* 53 (3-4): 73-85.
Grootjans, A. & van Diggelen, R. 1995. Assessing the restoration prospects of degraded fens. In: Wheeler, B.D., Shaw, S.C., Fojt, W.J. & Robertson, R.A. (eds.) *Restoration of temperate wetlands*, pp. 73-90. Wiley, Chichester, UK.
Koerselman, W. & Verhoeven, J.T.A. 1995. Eutrophication of fen ecosystems: external and internal nutrient sources and restoration strategies. In: Wheeler, B.D., Shaw, S.C., Fojt, W.J. & Robertson, R.A. (eds.) *Restoration of temperate wetlands*, pp. 91-112. Wiley, Chichester, UK.
Lamers, L.P.M., Smolders, A.J.P. & Roelofs, J.G.M. 2002. The restoration of fens in the Netherlands. *Hydrobiologia* 478: 107-130.
Lavoie, C., Zimmerman, C. & Pellerin, S. 2001. Peatland restoration in southern Québec (Canada): A paleoecological perspective. *Ecoscience* 8: 247-258.
Lavoie, C., Grosvernier, P., Girard, M. & Marcoux, K. 2003. Spontaneous revegetation of mined peatlands: an useful restoration tool? *Wetland Ecol. Manage.* 11: 97-107.
Lortie, G. 1983. Les diatomées fossiles de deux tourbières ombrotrophes du Bas-Saint-Laurent, Québec. *Géogr. Phys. Quat.* 37: 159-177.
Kratz, R. & Pfadenhauer, J. 2001. *Ökosystemmanagement für Niedermoore – Strategien und Verfahren zur Renaturierung.* Ulmer, Stuttgart, DE.
Middleton, B. 1999. Revegetation alternatives. In: Middleton, B. (ed.) *Wetland restoration, flood pulsing, and disturbance dynamics*, pp. 191-211. Wiley, New York, NY, US.
Mitsch, W.J. & Gosselink, J.G. 2000. *Wetlands.* 3rd. ed. Wiley, New York, NY, US.
Okruszko, H. 1995. Influence of hydrological differentiation of fens on their transformation after dehydration and on possibilities for restoration. In: Wheeler, B.D., Shaw, S.C., Fojt, W.J. & Robertson, R.A. (eds.) *Restoration of temperate wetlands*, pp. 113-119. Wiley, Chichester, UK.
Patzelt, A. 1998. Vegetationökologische und populationsbiologische Grundlagen für die Etablierung von Magerwiesen in Niedermooren. *Diss. Bot.* 297: 1-215.
Patzelt, A., Wild, U. & Pfadenhauer, J. 2001. Restoration of wet fen meadows by topsoil removal: vegetation development and germination biology of fen species. *Restor. Ecol.* 9: 127-136.
Pfadenhauer, J. & Grootjans, A. 1999. Wetland restoration in central Europe: aims and methods. *Appl. Veg. Sci.* 2: 95-106.
Poulin, M., Rochefort, L. & Desrochers, A. 1999. Conservation of bog plant species assemblages: assessing the role of natural remnants in mined sites. *Appl. Veg. Sci.* 2: 169-180.
Price, J., Rochefort, L. & Quinty, F. 1998. Energy and moisture considerations on cutover peatlands: surface microtopography, mulch cover and *Sphagnum* regeneration. *Ecol. Engin.* 10: 293-312.
Price, J.S., Rochefort, L. & Campeau, S. 2002. Use of shallow basins to restore cutover peatlands: Hydrology. *Restor. Ecol.* 10: 259-266.
Price, J.S., Heathwaite, A.L. & Baird, A.J. 2003. Hydrological processes in abandoned and restored peatlands: An overview of management approaches. *Wetland Ecol. Manage.* 11: 65-83.
Quinty, F. & Rochefort, L. 2003. *Peatland restoration guide.* 2nd ed. Université Laval, Québec, CA.
Rochefort, L. 2000. *Sphagnum* – A keystone in habitat restoration. *Bryologist* 103: 503-508.
Rochefort, L., Quinty, F., Campeau, S., Johnson, K. & Malterer, T. 2003. North American approach to the restoration of *Sphagnum* dominated peatlands. *Wetland Ecol. Manage.* 11: 3-20.
Roth, S., Seeger, T., Poschlod, P., Pfadenhauer, J. & Succow, M. 1999. Establishment of helophytes in the course of fen restoration. *Appl. Veg. Sci.* 2: 131-136.
Rowlands, R.G. 2001. *The ecological restoration through natural revegetation of industrial cutaway peatlands in Ireland.* Ph.D. Thesis, University College Dublin, Dublin, IE.
Salonen, V. 1987. Relationships between the seed rain and the establishment of vegetation in two areas abandoned after peat harvesting. *Holarct. Ecol.* 10: 171-174.
Salonen, V. 1990. Early plant succession in two abandoned cut-over peatland areas. *Holarct. Ecol.* 13: 217-223.
Salonen, V. 1992. Effects of artificial plant cover on plant colonization of a bare peat surface. *J. Veg. Sci.* 3: 109-112.
Schultz, W. 1998. Seed dormancy cycles and germination phenologies in sedges (*Carex*) from various habitats. *Wetlands* 18: 288-297.
Silvertown, J., Dodd, M.E., Gowing, D.J.G. & Mountford, J.O. 1999. Hydrologically defined niches reveal a basis for species richness in plant communities. *Nature* 400: 61-63.
Sjörs, H. 1952. On the relation between vegetation and electrolytes in north Swedish mire waters. *Oikos* 2: 241-258.
Sliva, J. & Pfadenhauer, J. 1999. Restoration of cut-over raised bogs in southern Germany – a comparison of methods. *Appl. Veg. Sci.* 2: 137-148.
Stauffer, A.L. & Brooks, R.P. 1997. Plant and soil responses to salvaged marsh surface and organic matter amendments at a created wetland in central Pennsylvania. *Wetlands* 17: 90-105.
Thormann, M.N., Currah, R.S. & Bayley, S.E. 1999. The mycorrhizal status of the dominant vegetation along a

peatland gradient in southern boreal Alberta, Canada. *Wetlands* 19: 438-450.

Turner, S.D. & Friese, C.F. 1998. Plant - mycorrhizal community dynamics associated with a moisture gradient within a rehabilitated prairie fen. *Restor. Ecol.* 6: 44-51.

van Duren, I.C., Strykstra, R.J., Grootjans, A.P., ter Heerdt, G.N.J. & Pegtel, D.M. 1998. A multidisciplinary evaluation of restoration measures in a degraded *Cirsio-Molinietum* fen meadow. *Appl. Veg. Sci.* 1: 115-130.

Van Loon, J.C. 1985. *Selected methods of trace metal analysis: biological and environmental samples*. Wiley, New York, NY, US.

Van Seters, T.E. & Price, J.S. 2002. Towards a conceptual model of hydrological change on an abandoned cutover bog, Quebec. *Hydrol. Process.* 16: 1965-1981.

Wheeler, B.D. & Shaw, S.C. 1995. A focus on fens – Controls on the composition of fen vegetation in relation to restoration. In: Wheeler, B.D., Shaw, S.C., Fojt, W.J. & Robertson, R.A. (eds.) *Restoration of temperate wetlands*, pp. 49-72. Wiley, Chichester, UK.

Wind-Mulder, H.L. & Vitt, D.H. 2000. Comparisons of water and peat chemistries of a post-harvested and undisturbed peatland with relevance to restoration. *Wetlands* 20: 616-628.

Wind-Mulder, H.L., Rochefort, L. & Vitt, D.H. 1996. Water and peat chemistry comparisons of natural and post-harvested peatlands across Canada and their relevance to peatland restoration. *Ecol. Engin.* 7: 161-181.

Xiong, S.J. & Nilsson, C. 1997. Dynamics of leaf litter accumulation and its effects on riparian vegetation: A review. *Bot. Rev.* 63: 240-264.

Zar, J.H. 1984. *Biostatistical analysis*. 2nd. ed. Prentice Hall, Upper Saddle River, NJ, US.

Zoltai, S.C. & Vitt, D.H. 1995. Canadian wetlands: Environmental gradients and classification. *Vegetatio* 118: 131-137.

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