

# Fen restoration: defining a reference ecosystem using paleoecological stratigraphy and present-day inventories

Vicky Bérubé, Line Rochefort, and Claude Lavoie

**Abstract:** Choosing past and present-day indicators could strengthen the reference ecosystem used for ecological restoration projects. Based on the paleoecological analysis of four peat cores and the characterization of 13 contemporary natural sites, the reference ecosystem for minerotrophic peatlands in southeastern Canada is composed of two broad categories of plant assemblages described as tall-sedge and *Sphagnum–Thuja*/brown moss. In paleoecological peat profiles, tall-sedge communities were found at the transition between aquatic and terrestrial, and were associated with high graminoid production, riverine peatlands, and elevated water table in the present-day analyses. *Sphagnum–Thuja* communities resemble the present-day vegetation found in natural basin type peatlands. Except for *Sphagnum warnstorffii* Russ., these communities, with high taxonomical diversity, contain more generalist species from boreal peatland vegetation, such as *Rhododendron groenlandicum* (Oeder) Kron & Judd, *Thuja occidentalis* L., *Linnaea borealis* L., and *Maianthemum trifolium* L. They grow in dryer or shady habitats such as hummocks or forest understory. The importance of brown mosses was revealed by paleoecological analysis. Overall, findings from both approaches are complementary: paleoecological stratigraphy informs us about past ecosystem dynamics, while present-day inventories allow us to define current plant communities and their major environmental characteristics. The range of variability of vegetation and environmental variables found in these studies are essential tools for fen restoration projects.

**Key words:** rehabilitation, minerotrophic peatland, plant assemblages, plant dynamics, indicator value index (IndVal).

**Résumé :** Lors d'un projet de restauration écologique, la description de l'écosystème de référence est renforcée par l'utilisation d'indicateurs combinant les études temporelles et spatiales. Se basant sur une étude paléoécologique et la caractérisation de treize sites naturels, la référence pour une tourbière minérotrophe modérément riche située dans le sud-est du Canada se compose de deux types d'assemblages d'espèces végétales : l'assemblage des Grandes-cypéracées et l'assemblage général de *Sphagnum–Thuja*/mousse brune. Dans l'étude paléoécologique, la communauté des Grande-cypéracées est observée à la transition entre les écosystèmes de marais et de tourbières. Dans l'étude des sites naturels, cette communauté est associée à une production de biomasse élevée, un niveau d'eau élevé et aux fens de type riparien. L'assemblage *Sphagnum–Thuja* est semblable à ceux inventoriés dans les fens de type bassin. En faisant l'exception du *Sphagnum warnstorffii* Russ., cet assemblage, très diversifié, inclut surtout des espèces généralistes provenant des tourbières boréales telles le *Rhododendron groenlandicum* (Oeder) Kron & Judd, le *Thuja occidentalis* L., le *Linnaea borealis* L. et le *Maianthemum trifolium* L. L'importance des mousses brunes est révélée dans l'analyse paléoécologique. En somme, les deux approches sont complémentaires : la paléoécologie nous informe sur la dynamique de l'écosystème tandis que les inventaires des sites naturels nous a permis de définir les assemblages et leurs associations aux conditions environnementales. Les étendues de variabilité définies dans ce travail sont un outil à la décision essentiel dans un projet de restauration écologique de fen.

**Mots-clés :** réhabilitation, tourbière minérotrophe, assemblage de végétaux, dynamique des communautés, index valeur indicatrice (IndVal).

## Introduction

Reference ecosystems act as a standard of comparison and evaluation for any restoration project (SERI 2004; Clewell and Aronson 2013; Shackelford et al. 2013). Se-

lecting references among other similar ecosystems located in the same region is a usual starting point for establishing standards (White and Walker 1997; SERI 2004; Hallett et al. 2013). However, the absence or limited

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V. Bérubé and L. Rochefort. Département de phytologie and Centre d'études nordiques, Université Laval, 2425, rue de l'Agriculture, Québec, QC G1V 0A6, Canada.

C. Lavoie. École supérieure d'aménagement du territoire et de développement régional and Centre d'études nordiques, Université Laval, 2325, rue des Bibliothèques, Québec, QC G1V 0A6, Canada.

**Corresponding author:** Vicky Bérubé (email: [vicky.berube.1@ulaval.ca](mailto:vicky.berube.1@ulaval.ca)).

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knowledge of pre-disturbance states, ecosystem dynamics, and cultural services may be responsible for an inappropriate reference selection (Aronson et al. 1995; Birks 1996; Higgs 1997; Jackson and Hobbs 2009; Balaguer et al. 2014). An in-depth knowledge of ecosystem variability with temporal and spatial perspectives is thus recommended for making better choices. Moreover, multiple sources of information improve the quality and accuracy of the data that can be used to build a reliable picture of a reference ecosystem (White and Walker 1997; SERI 2004), and may help to better define realistic restoration objectives (Ehrenfeld 2000; SERI 2004; Hallett et al. 2013; Shackelford et al. 2013).

Data useful for defining a reference ecosystem can be collected using historical records, by studying similar undisturbed ecosystems, or both (Birks 1996; White and Walker 1997; Balaguer et al. 2014). Historical tools include paleoecological studies, past plant or faunal surveys, and aerial and in-situ photographs. One of the advantages of using historical data are that they provide insights into ecosystem dynamics over time (Jackson and Hobbs 2009; Balaguer et al. 2014). They also help to choose the developmental stage at which the ecosystem should be restored. Paleoecological studies are widely used in peatlands to assess past vegetation patterns and succession dynamics (e.g., Glaser and Janssens 1986; Kuhry et al. 1993; Bauer et al. 2003) or climate change (e.g., Halsey et al. 1998; Yu 2006). They proved to be useful to provide a better understanding of the ecosystem functioning over the long term, which can help to evaluate the time frame over which restoration objectives could reasonably be achieved (Lavoie et al. 2001; Blundell and Holden 2015). The response of ecosystems to past disturbance events can also inspire restoration engineers by providing clues on natural recovery processes (Benscoter and Vitt 2008). On the other hand, present-day studies on the existing composition and distribution of species are essential to complete the reference picture. They represent a timescale snapshot but can nevertheless give a precise overview of the spatial distribution of species along a wide range of biotic and abiotic conditions.

In North America, most peatlands that have actively been restored are cutover bogs (ombrotrophic peatlands; Poulin et al. 2013; González et al. 2014). The peat was extracted for horticultural purposes. The most widely used extraction method (vacuum) consists of draining the peatland, removing the living vegetation, and harrowing and vacuuming a thin layer of dried peat. Peat extraction activities cease when peat quality does not correspond to industrial standards (too decomposed, too woody, etc.) and, until recently, at least 50 cm of *Sphagnum* peat was left on-site to ensure that enough material is left for restoration. However, in some cases, the peat industry produces specific growing substrates with the deeper, older peat layers of peatlands. In such cases, ancient minerotrophic peat layers composed of the re-

mains of sedges and brown moss become exposed (Graf et al. 2008). Because the chemical properties of the water and peat then become similar to fen habitats (Wind-Mulder and Vitt 2000), the rehabilitation of the peatland to a minerotrophic developmental stage could be considered if ecological restoration is planned. The challenge under those conditions is to define a reference ecosystem from a series of developmental stages that the ecosystem has experienced in the past, and could experience in the future.

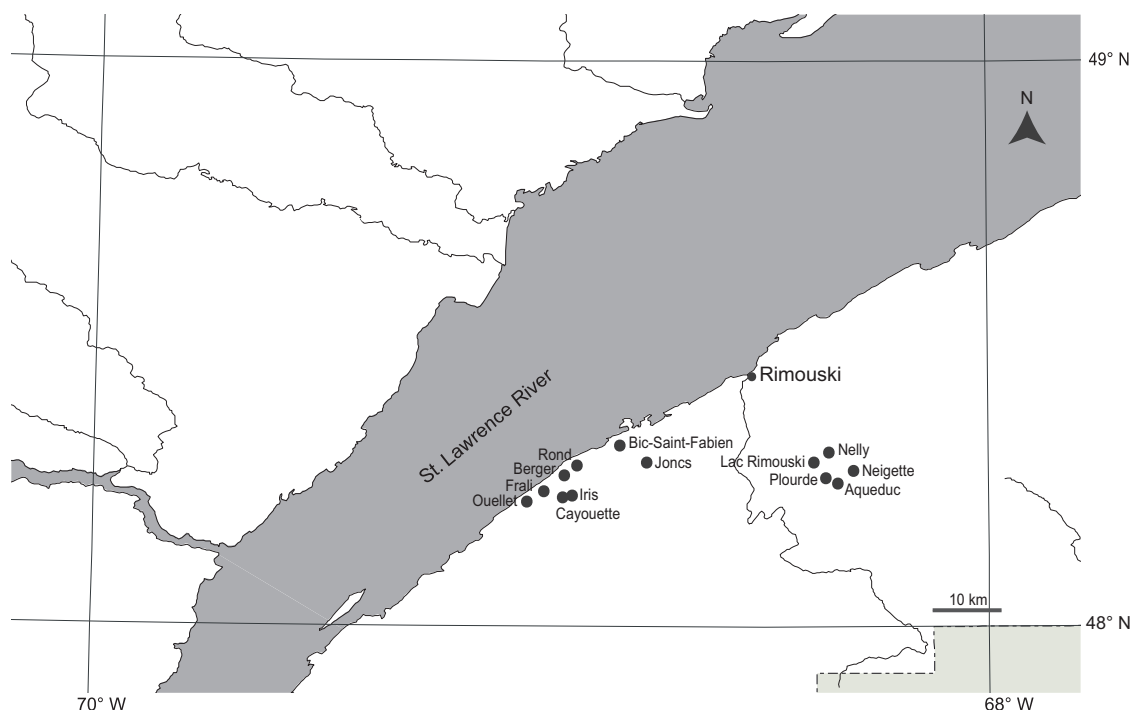
The goal of this study was to build a reference ecosystem in the context of fen restoration in southeastern Canada, using paleoecological studies and current regional plant surveys. Specifically, we wanted (i) to describe the historical trajectory of a minerotrophic peatland via its developmental states and its associated plant assemblages; (ii) to assess whether modern plant assemblages are representative of the past; (iii) to evaluate the importance of bryophytes for fen development; (iv) to study natural disturbance regimes and the resilience process, and finally (v) to identify the state to which the disturbed peatland has regressed after the cessation of peat harvesting activities. If the present plant assemblage is representative of a past assemblage, the modern plant surveys should complete the reference by determining the variability in species composition and environmental condition of moderately rich fens. Our overall intent with this study was to demonstrate the usefulness of both approaches to establish precise restoration objectives.

## Materials and methods

### Study sites

All of the peatlands investigated in this study were located near the south shore of the St. Lawrence River, in the Bas-Saint-Laurent region, Quebec, Canada (Fig. 1). The area was covered by a glacier until 14 000 years BP, and then by the Goldthwait Sea until 10 000 years BP (Locat 1977), leaving clay, sand, and gravel as surficial marine deposits over the St. Lawrence River Lowlands (Fulton 1995). The paleoecological study was conducted in the Bic-Saint-Fabien (BSF) peatland (48°19'22"N, 68°49'52"W). It is part of a peatland complex that developed in a narrow valley (Fig. 2). BSF, which is now included in the Bic National Park, is bordered by steep calcareous hills (340 m a.s.l.) to the north and by agricultural fields to the south. A large pond (about 40 ha) was located in the central part of the peatland but was drained in 1954. The section of the peatland east of the pond was formerly a *Sphagnum* bog (Scoggan 1942) with a well-defined lagg on its margins. This eastern section had largely been used for peat extraction. The section west of the pond is now a mosaic of peat extraction and few natural (no major disturbance) sites. Peat extraction began in 1946 and ended in the 1990s. The 15 ha area that was targeted for a restoration project has a peat substrate with the following minerotrophic characteristics, as assessed in summer 2011 (L. Rochefort, unpublished data;  $n = 111$ ): the water pH was

**Fig. 1.** Locations of the 13 sites used for present-day surveys of fen vegetation in the Bas-Saint-Laurent region, Quebec (Canada). Map was re-used with the permission of ©Gouvernement du Québec (2006).



$6.0 \pm 0.4$  (mean  $\pm$  SD) and the electroconductivity was  $229 \pm 141 \mu\text{S}\cdot\text{cm}^{-1}$ . The central section was spontaneously revegetated, mostly with *Typha latifolia*, *Scirpus cyperinus*, *S. atrocinctus*, and *Salix* species, essentially along former drainage ditches. Bare peat covered the last four peat fields of this eastern section.

The northern and eastern sections of BSF bordering the extraction zone still support a moderately rich fen plant community (sensu Chee and Vitt 1989 and Locky and Bayley 2005) dominated by *Thuja occidentalis* (Gauthier and Grandtner 1975). In these sections, the water pH was  $6.5 \pm 0.8$  (mean  $\pm$  SD) and the electroconductivity  $237 \pm 136 \mu\text{S}\cdot\text{cm}^{-1}$  (L. Rochefort, unpublished data;  $n = 41$ ).

#### Paleoecological analyses

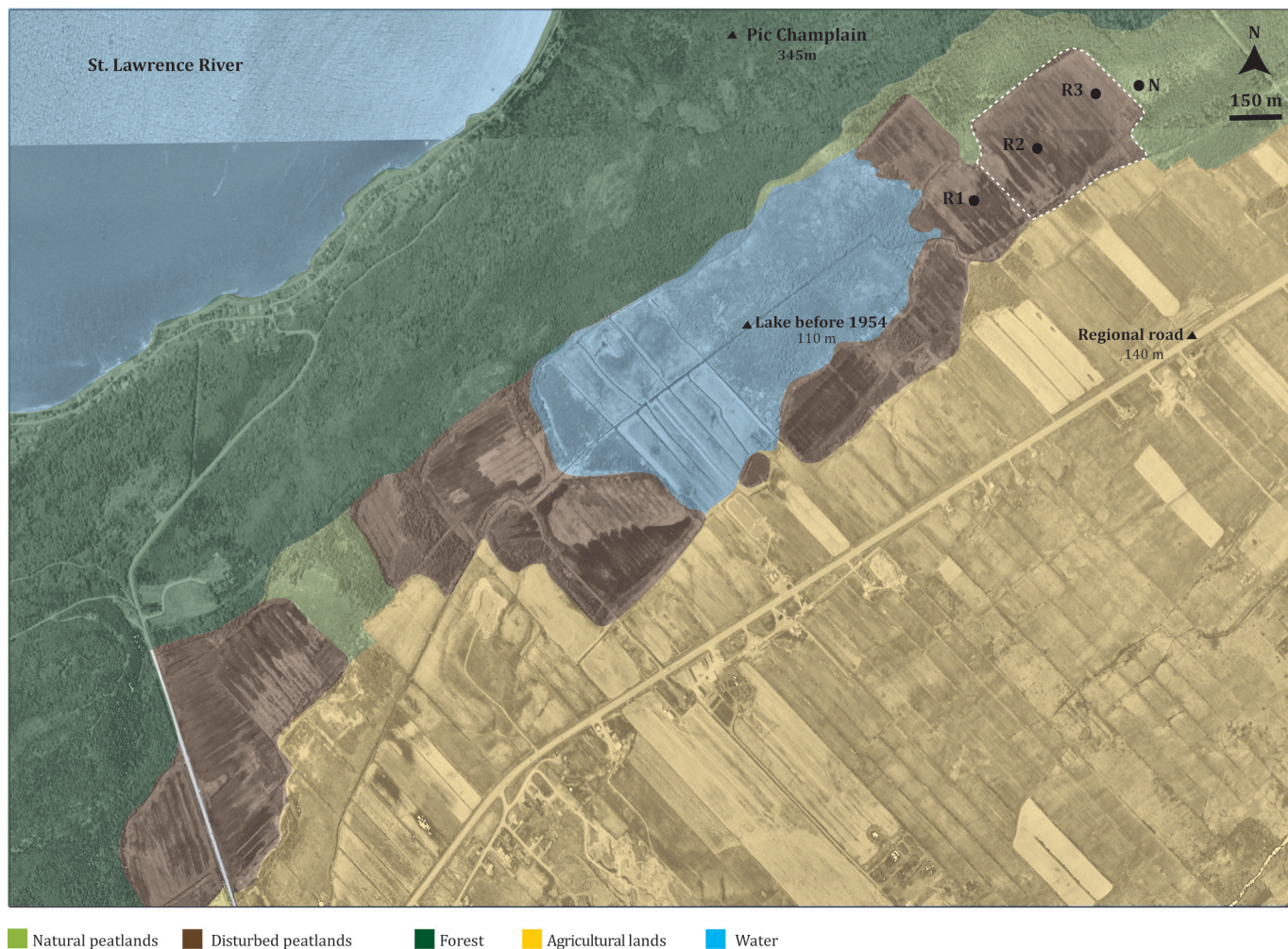
For this study, only the eastern section of the BSF peatland complex has been considered, excluding the former lakebed. Efforts for an ecological restoration project were concentrated in this section (Fig. 2). The condition of the disturbed sector was heterogeneous if we consider the time of abandonment, elevation, and spontaneous vegetation cover. To get a good representation of the site, we divided the area into three more homogeneous subsectors for sampling. One peat core was extracted from each of the three subsectors (R for restored section: R1, R2, and R3). The peat cores were respectively about 700, 950, and 1200 m from the location of the drained pond (Fig. 2). A fourth core (N for natural section) was sampled at a distance of 1300 m, but in the part of the peatland with no peat extraction.

A side-wall peat corer (Jowsey 1966) was used to extract the peat cores from the soil surface to the top of the surficial marine deposit. Each core was cut every 5 cm into subsamples, and each subsample was washed through a series of sieves (2, 1, and 0.5 mm meshes) and then dried. Fossil pieces were sorted and identified to the finest taxonomical level (family, genus, or species when possible). Nomenclature followed the Integrated Taxonomic Information System (2015) for vascular plants and animals, Faubert (2007) for bryophytes, and LoBuglio (1999) for fungi. Pieces were counted under a low power (50 $\times$ ) microscope. When the fossil pieces were too numerous, a subsample of 0.5 g of peat was extracted, the pieces counted, and the total number of pieces was estimated using the total dry weight of the sample. All the results were adjusted to a volume 100 cm<sup>3</sup> of peat. Macrofossil zones were defined visually according to changes in the relative abundance of plant remains. Samples at the limits of the zones and at the surface of the peat (for R1, R2, and R3) were dated using accelerator mass spectrometry (AMS) radiocarbon dating. Samples were prepared by the Radiochronology lab at Université Laval, Quebec, Canada, and analyzed by Keck Carbon Cycle AMS Facility (University of California – Irvine, California, USA) Radiocarbon dates were calibrated with CALIB REV 7.0.1 software (Stuiver et al. 2005).

#### Natural site surveys

A total of 13 fens were found in the vicinity of the BSF peatland (from a 5–30 km distance), with the help of a local bryologist and by looking at satellite maps (Fig. 1).

**Fig. 2.** Aerial photograph of the Bic-Saint-Fabien (BSF) peatland complex and of the surrounding landscape. The peatland developed in a valley surrounded by a mountain in the north (dark green), part of Bic National Park, and agricultural lands in the south (pale yellow). An ancient lake located in the central part of the peatland, as seen on an aerial photograph taken in 1925, is overlaid in blue. The lake was drained in 1954 and is now covered, in the sections that have not been used for peat extraction, by dense shrubs and wet meadows. The dotted white line delimits the 15 ha area claimed for a restoration project. The peat core locations (R1, R2, R3, N) are also indicated. Source: Orthophoto was used with the permission of the ©Gouvernement du Québec (2001).



The final selection was based on the presence of indicator plant species of moderately rich fens and water pH above 5.5 (Chee and Vitt 1989). A survey of modern vegetation communities and associated environmental conditions was made during summer 2012. Six of the 13 peatlands were subdivided into two homogeneous units on the basis of their tree cover (with or without) or because they represent different sectors within a peatland complex. The number of quadrats depended on the area of the unit. Between 10 and 12 circular quadrats of 70 cm diameter were positioned equidistant along a linear transect passing throughout the longest axis of the fen (unit), for a total of 200 quadrats. The cover occupied by each species present within the quadrats was visually estimated with the accuracy of  $\pm 5\%$  (for more details, see Rochefort et al. 2013). Environmental variables were measured for each peatland unit. The sampling was located in the central part of the peatland unit. Species

richness is defined as the number of species in each quadrat, and diversity was also calculated for each quadrat using the Shannon diversity index (Whittaker 1972). All of the variables are described in Table 1, along with analytical methods used.

#### Data processing and statistical analyses

The cluster method, specifically *K*-means partitioning, and the indicator value index (IndVal) method (Borcard et al. 2011) were used to define modern plant communities and to identify their associated indicator species. *K*-means are based on species similarity found in each quadrat. To achieve this goal, a truncated species dataset was used for all statistical analyses to avoid biases potentially caused by rare species (Legendre and Gallagher 2001; Borcard et al. 2011); only species with a frequency (occurrence among all the quadrats from all peatlands) and (or) a mean cover (among the quadrats with the species)

**Table 1.** Description of 23 environmental variables, code names, and units measured in each of the fens (units) surveyed in the Bas-Saint-Laurent region in 2012.

Group	Variables	Code	Unit	Range	Notes
Descriptive	Peatland classification*	type	factor	Riverine, basin	Estimated in the field (NWWG 1997)
	Distance to the St. Lawrence River*	dist_river	km	1 to 21	Estimation based on satellite photographs (Google earth V7.1.5.1557)
Physical	Disturbance*	disturb	factor	None, limited	Estimated in the field
	Water level	WT	cm	-35 to 17	Used a ruler within a dug hole
	Peat depth*	depth	cm	42 to 612	Metal rod inserted in the central part of the peatland (unit)
	Bulk density*	BD	g·mL <sup>-1</sup>	0.13 to 0.30	Peat samples dried and weighed in the laboratory
Water chemistry	Organic mater	OM	%	78.6 to 91.9	Dry-ash method (JAOAC 1967)
	pH	pH	—	5.8 to 8.1	Measured in the laboratory (Fisher Scientific Accumet AB150)
	Electroconductivity*	EC	μS·cm <sup>-1</sup>	94 to 275	Measured in the laboratory (Orion Model 122) and corrected according to Sjörs (1950)
	P	P	mg·L <sup>-1</sup>	0 to 1.6	ICP <sup>†</sup>
	Ca*	Ca	mg·L <sup>-1</sup>	7.16 to 45.75	ICP
	Mg*	Mg	mg·L <sup>-1</sup>	0.75 to 7.01	ICP
	Fe	Fe	mg·L <sup>-1</sup>	0 to 0.26	ICP
	K*	K	mg·L <sup>-1</sup>	0.21 to 7.90	ICP
	Na*	Na	mg·L <sup>-1</sup>	1.82 to 12.22	ICP
	NH <sub>4</sub>	NH4	mg·L <sup>-1</sup>	0.05 to 5.58	FIA <sup>‡</sup>
Vegetation	NO <sub>3</sub> *	NO3	mg·L <sup>-1</sup>	0.01 to 0.14	FIA
	SO <sub>4</sub> *	SO4	mg·L <sup>-1</sup>	1.13 to 21.64	FIA
	Shannon diversity index*	shannon	—	0 to 2.53	$H' = \sum[(p_i) \times \log_2(p_i)]$ calculated with all species per quadrat
	Richness*	rich	—	4 to 22	Number of species per quadrat
	Biomass (graminoid)*	b_gram	g·m <sup>-2</sup>	2.92 to 296.50	All living vegetation cut from two
	Biomass (forbs)*	b_forb	g·m <sup>-2</sup>	2.92 to 63.71	50 cm × 50 cm quadrats randomly placed
	Biomass (shrubs)*	b_shrub	g·m <sup>-2</sup>	0 to 84.50	in a representative homogenous section, dried, and weighed
General tree cover*	cover	factor	Variable, open	Estimated in the field for each quadrat	

**Note:** All of the measurements were conducted at the end of July 2012, except for water sampling in Plourde and Aqueduc peatlands, which were conducted in mid-July 2009. Water chemistry samples were analyzed at Laval University. An asterisk (\*) indicates the variables kept for the RDA model after forward selection.

<sup>†</sup>ICP, inductively coupled plasma spectroscopy (Optima 4300 DV; Perkin Elmer).

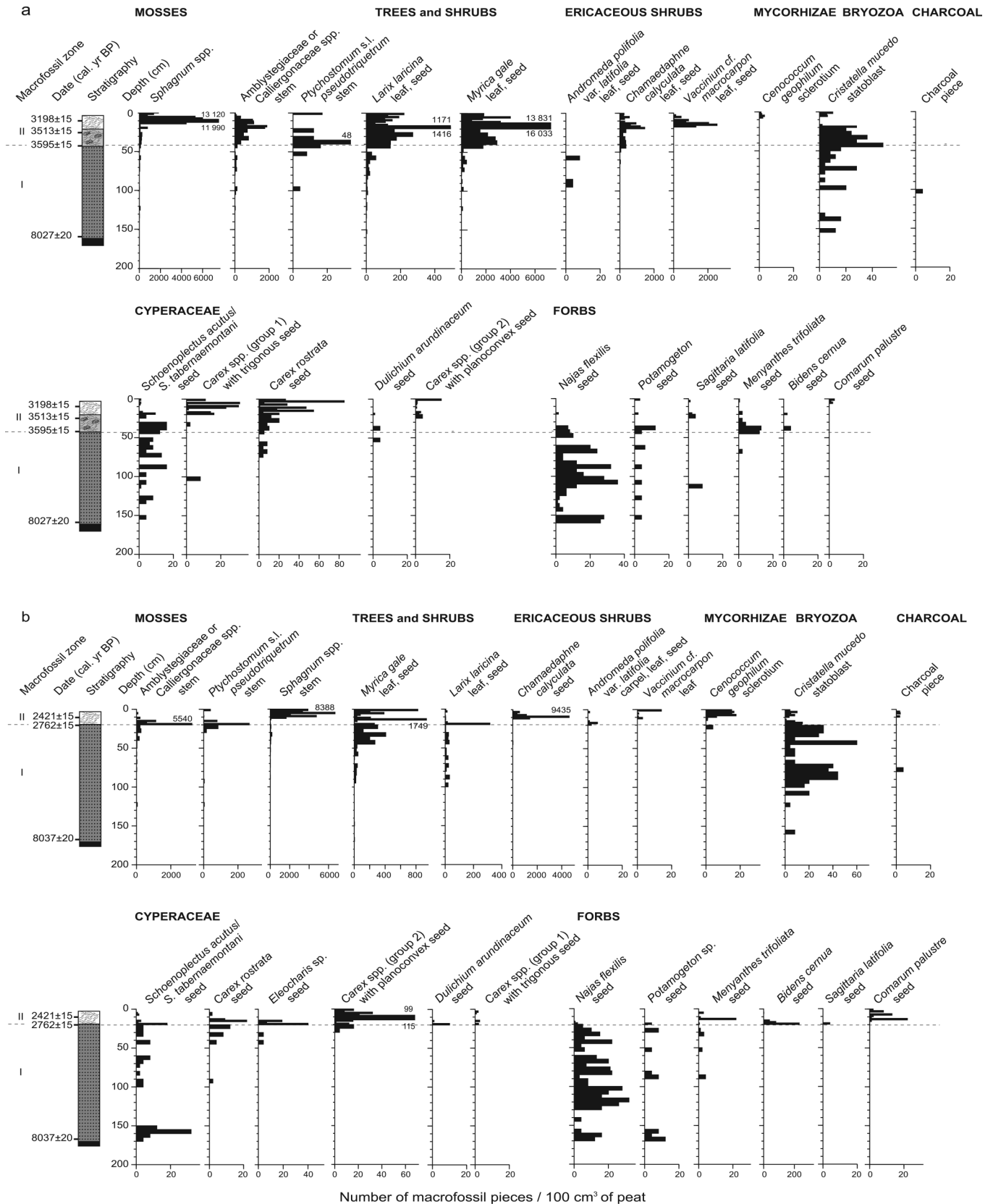
<sup>‡</sup>FIA, flow injection analysis (Quikchem 4000, Lâchât Instrument Division, Milwaukee, Wisconsin, USA).

>5% were kept. Hellinger's transformation was conducted to take into account the importance of double zeros and common species (Legendre and Gallagher 2001).

The Calinski-Harabasz criterion (Milligan 1996) was used to determine the adequate number of clusters (plant assemblages). To facilitate the categorization of assemblages, indicator species in each group were identified with the indicator value index (IndVal) developed by Dufrene and Legendre (1997). This index computes the product of specificity (abundance of a species in all the quadrats of a cluster) and fidelity (presence of a species in a majority of quadrats in a cluster) for every species in each cluster previously delineated with K-means partitioning. IndVal values range from 0 to 1; a value close to 1 for a particular species indicates that the species is present exclusively in this cluster and has a high occurrence. Only species with IndVal values > 0.25, and an associated  $P < 0.001$ , were used to identify plant communities.

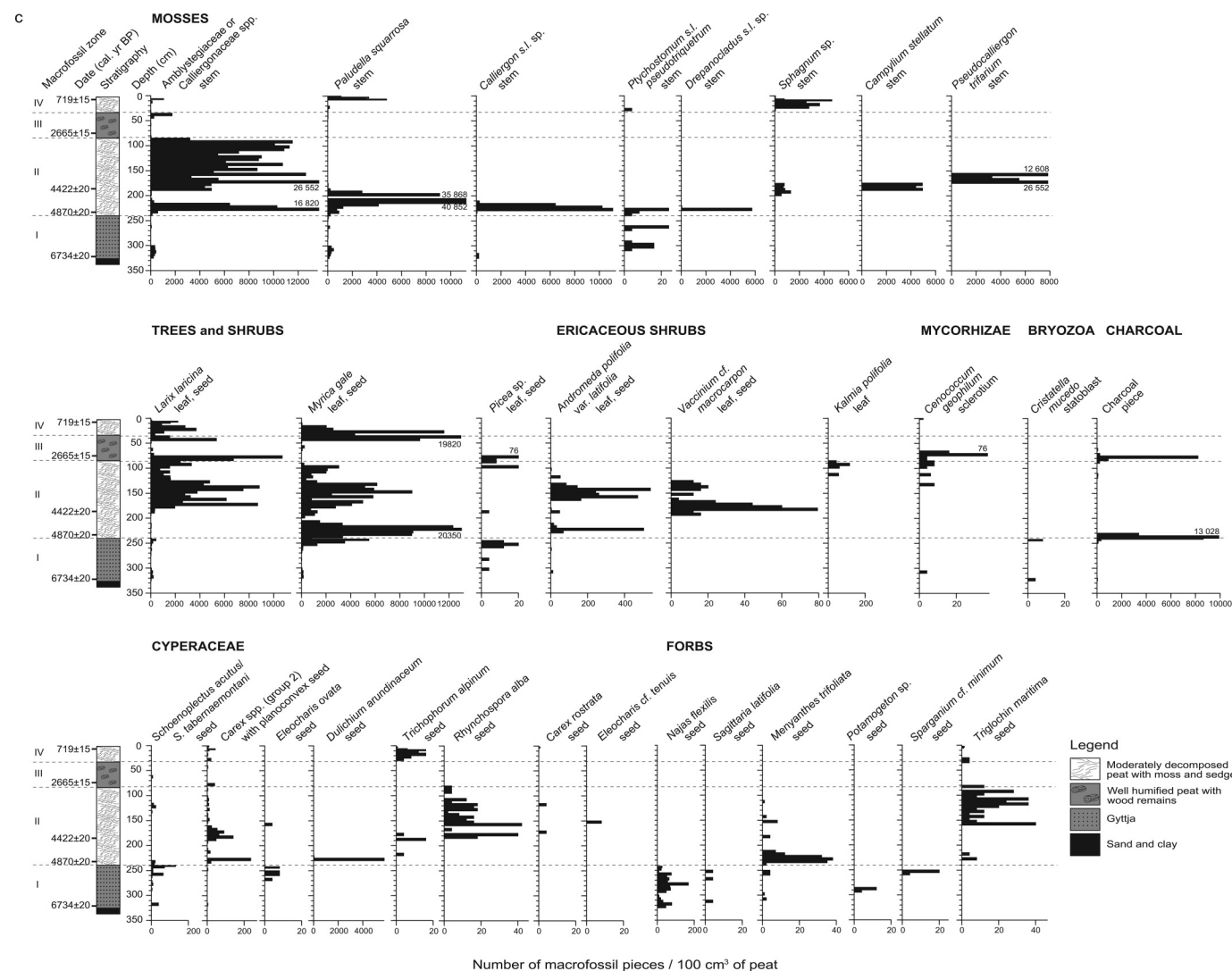
A redundancy analysis (RDA) was performed to understand the influence of environmental variables on plant communities. A forward selection was applied to reduce type I error and select the significant environmental variables. Cumulative  $r^2$  adjusted from this selection must be lower than the  $r^2$  adjusted from the RDA model previously run with all the variables to respect the two stopping criteria (Blanchet et al. 2008). The significant environmental variables issued from the forward selection were used to perform a second RDA and improve model variation. To view the position of the K-means clusters, equiprobability ellipses were created, including 95% of the sites from the same plant community. Finally, the RDA model was submitted to a permutation test with 9999 randomized runs to assess its significance (Legendre and Legendre 2012). All statistical analyses were conducted using R software version 3.0 (R Development Core Team 2016), with the function RDA, Ordistep, and CascadeKM from the vegan package (Oksanen et al. 2011),

**Fig. 3.** Macrofossil diagrams (selected taxa) of the Bic-Saint-Fabien BSF peatland. (a) R1, (b) R2, (c) R3, and (d) N. The broken lines mark the different vegetation zones.



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Fig. 3 (continued).



and IndVal from the labdsv (Roberts 2013) and ellipse package (Murdoch and Chow 2007).

## Results

### Paleoecological analyses

Four major vegetation zones were identified in N and R3, while only two were observed in R1 and R2. Limnic deposit known as gytija or coprogenous earth (Groupe de travail sur la classification des sols 2002; Meier-Uhlherr et al. 2015) started to accumulate over a sand and clay marine deposit between 8037 years BP at R1 and 7090 years BP at N (Fig. 3; Table 2; Supplementary data, Fig. S1<sup>1</sup>). The closer a peat core was to the pond of BSF peatland, the later peat accumulation was initiated. Rates of peat accumulation for R3 and N varied between 0.31 to 1.12 mm per year, depending on the core sampling location and vegetation assemblage. Peat composed of brown mosses (zones II

and IV) accumulated faster than peat composed only of wood remains (zone III).

At least three major fire events occurred. The peatland burned about 4870 years BP and 2700 years BP at the R3 location. The first fire coincided with the initiation of peat accumulation at this location, and the second with the end of zone II and the beginning of zone III. Around 1930, a fire burned part of the peatland (Scoggan 1942) and left charcoal pieces near the surface of core N. High abundance of *Campylium stellatum* and *Scorpidium cossonii* remains was noticed after this fire. Macrofossils of *Thuja occidentalis* also appeared after the 1930 fire.

The four peat cores followed a similar trajectory concerning the evolution of vegetation assemblages (Fig. 3, and Supplementary data, Fig. S1). Consequently, descriptions of each homogeneous zone identified through time are grouped together to avoid repetition.

<sup>1</sup>Supplementary data are available with the article through the journal Web site at <http://nrcresearchpress.com/doi/suppl/10.1139/cjb-2016-0281>.

Fig. 3 (concluded).

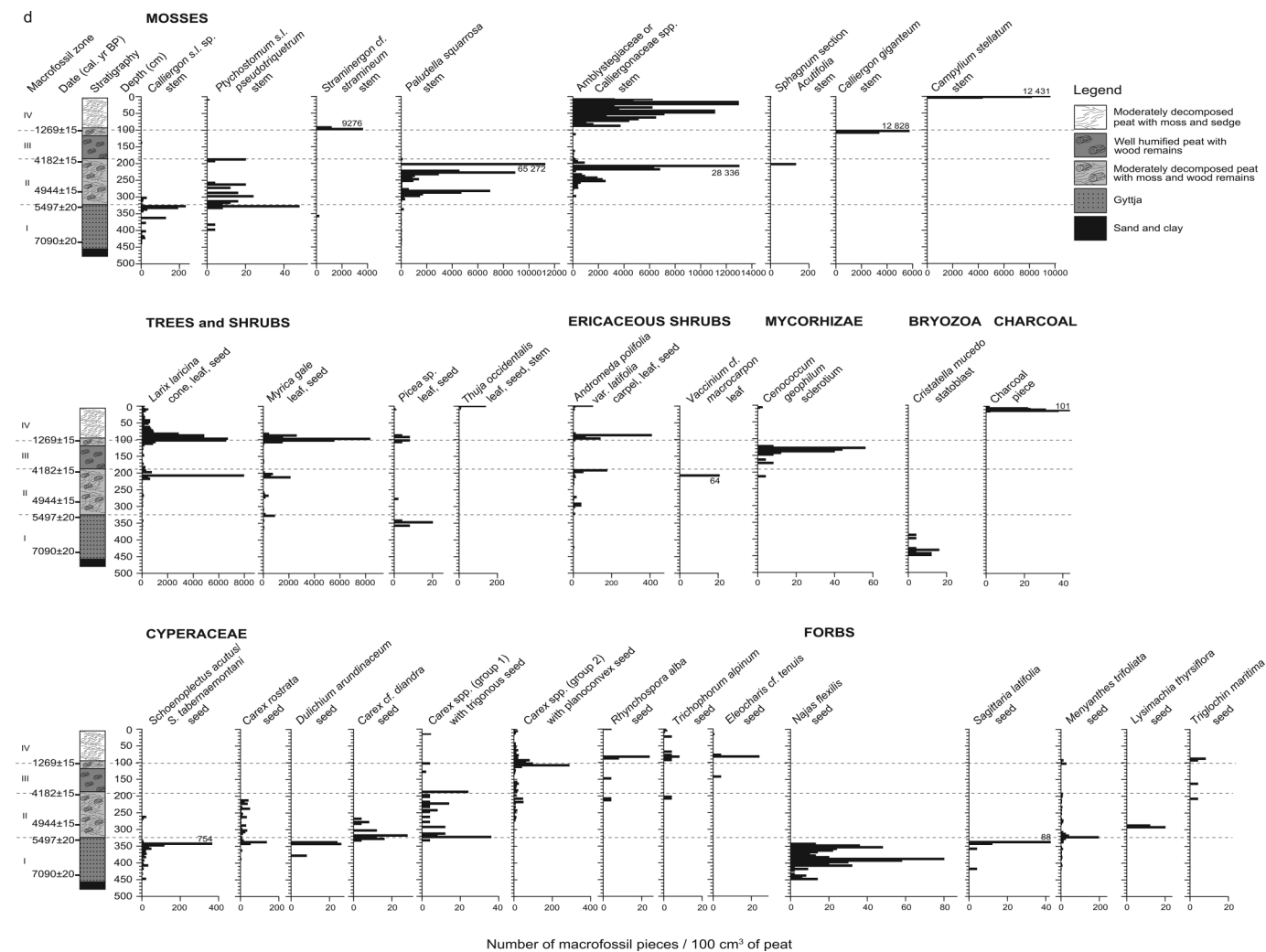


Table 2. Radiocarbon dates from the four peat cores extracted at the Bic-Saint-Fabien (BSF) peatland.

Core site and depth (cm)	Laboratory No.	Age ( <sup>14</sup> C year BP)	1-sigma range (cal. year BP)	Calibrated age (cal. year BP)	1-sigma range (cal. year)	Calibrated age	Material dated
R1, 2–4	ULA-1627/UCIAMS-76677	3015±15	3175–3221	3198	1272–1226 BC	1249 BC	Moss fragments
R1, 18–20	ULA-1617/UCIAMS-76675	3300±15	3492–3533	3513	1584–1543 BC	1564 BC	Moss fragments
R1, 35–40	ULA-1616/UCIAMS-76674	3360±15	3575–3614	3595	1665–1627 BC	1646 BC	Moss fragments
R1, 155–160	ULA-1626/UCIAMS-76676	7235±20	8001–8052	8027	6103–6052 BC	6078 BC	Wood fragments
R2, 2–4	ULA-1615/UCIAMS-76673	2425±15	2376–2465	2421	516–427 BC	472 BC	Moss fragments
R2, 16–18	ULA-1614/UCIAMS-76672	2660±15	2754–2769	2762	820–805 BC	813 BC	Moss fragments
R2, 165–170	ULA-1582/UCIAMS-76658	7260±20	8019–8055	8037	6106–6070 BC	6088 BC	Wood fragments
R3, 2–4	ULA-1613/UCIAMS-76671	820±15	702–735	719	1215–1248 AD	1232 AD	Moss fragments
R3, 75–80	ULA-1581/UCIAMS-76657	2460±15	2635–2694	2665	754–686 BC	720 BC	Wood fragments
R3, 185–190	ULA-1584/UCIAMS-76660	3955±20	4412–4431	4422	2489–2461 BC	2475 BC	Moss fragments
R3, 235–240	ULA-1564/UCIAMS-75742	4340±20	4854–4886	4870	2937–2905 BC	2921 BC	Moss fragments
R3, 320–325	ULA-1583/UCIAMS-76659	5915±20	6716–6752	6734	4803–4767 BC	4785 BC	Gyttja sediments
N, 105–110	ULA-1567/UCIAMS-75745	1300±15	1256–1281	1269	669–674 AD	672 AD	Moss fragments
N, 195–200	ULA-1566/UCIAMS-75744	3815±15	4158–4205	4182	2256–2209 BC	2233 BC	Moss fragments
N, 275–280	ULA-1565/UCIAMS-75743	4370±15	4927–4960	4944	3011–2978 BC	2995 BC	Moss fragments
N, 335–340	ULA-1563/UCIAMS-75741	4815±20	5488–5505	5497	3556–3539 BC	3548 BC	Wood fragments
N, 445–450	ULA-1562/UCIAMS-75740	6205±20	7067–7112	7090	5163–5118 BC	5141 BC	Wood fragments



### Zone I (R1, R2, R3, N)

Gyttja sediments attested to the terrestrialization (in-filling) origin of the peatland (Łachacz et al. 2009). The jelly-like sediments included plant species usually found in a shallow water habitat, such as *Najas flexilis*, *Nymphaea* spp., *Potamogeton* spp., *Sparganium* spp., and the bryozoan *Cristatella mucedo*. An assemblage composed of emergent species characterized the end of this zone. Marsh species (*Bidens cernua*, *Dulichium arundinaceum*, and *Sagittaria latifolia*) were sporadically found. The abundance of *Carex* cf. *diandra*, *Carex rostrata*, and *Schoenoplectus* s.l. *acutus* suggested a transition from a shallow lake ecosystem to a fen.

### Zone II (R1, R2, R3, N)

Bryophytes were the main components of zone II. High abundance of *Paludella squarrosa*, *Calliergon* spp., and other brown moss fragments from Amblystegiaceae and Calliergonaceae families were recovered. The peat cores closer to the ancient pond (R1 and R2) also included *Sphagnum* stems from Acutifolia section species. At the beginning of zone II, the abundance of remains of mat-forming species (*Menyanthes trifoliata*, *Comarum palustre*, *Bidens cernua*) decreased. Cyperaceae species such as *Carex diandra*, *C. rostrata*, and *Rhynchospora alba*, along with other unidentified *Carex* species with planoconvex and trigonous shaped seeds, were well-represented for the entire zone.

An abrupt change in plant assemblages was detected in this moss-rich zone of R3, and was characterized by (i) a decrease in the number of *Calliergon* sp., *Ptychostomum* s.l. *pseudotriquetrum*, and *Drepanocladus* sp. s.l. remains, followed by (ii) the disappearance of *Paludella squarrosa*, and then by (iii) the establishment of other brown moss species (Amblystegiaceae spp.) dominated by *Campylium stellatum* and *Pseudocalliergon trifarium*.

### Zone III (R3, N)

A major change in the vegetation composition of the peatland occurred about 2665 years BP at R3 and 4182 years BP at N, when bryophytes suddenly disappeared. The highly decomposed peat was mainly composed of small woody debris and no seeds or discernable plant parts were noticed, except for sclerotia of *Cenococcum geophilum*.

### Zone IV (R3, N)

A rapid increase in the abundance of bryophytes characterizes the beginning of zone IV. *Sphagnum* spp. were especially abundant at R3, which was also the case for *Paludella squarrosa* at the upper end of the profile. N was dominated by brown mosses (*Campylium stellatum*, *Calliergon giganteum*, and *Straminergon* cf. *stramineum*). The vascular plant assemblages included *Rhynchospora alba*, *Trichophorum alpinum*, *Eleocharis* cf. *tenuis*, *Triglochin maritima* and *Andromeda polifolia* var. *latifolia*.

### Present-day fen surveys

The 13 sampled fens (19 peatland units) had a peat thickness of  $345 \pm 169$  cm (mean  $\pm$  SD), water table depth of  $10 \pm 13$  cm below the soil surface, pH of  $6.7 \pm 0.6$ , corrected electroconductivity of  $159 \pm 53$   $\mu\text{S}\cdot\text{cm}^{-1}$ , and Ca concentration of  $18.1 \pm 11.3$   $\text{mg}\cdot\text{L}^{-1}$ . Based on The Canadian Wetland System Classification (NWWG 1997), hydrological regime (terrigenous or littogenous), and topography, there were two main types of fens: basin ( $n = 8$  peatlands or units) and riverine ( $n = 11$ ). The basin fens were richer in species (17 versus 11), had a deeper water table depth (18 versus 5 cm), a lower pH (6.4 versus 7.0), and a lower electroconductivity (143 versus 171  $\mu\text{S}\cdot\text{m}^{-1}$ ) than the riverine fens. Levels of elements in the water were also 35% lower (on average) in the basin than in the riverine fens, except for the levels of P and Na, which were higher in the basin fens. Basin fens also occupied a smaller area ( $3 \text{ ha} \pm 3$  [SD] versus  $5 \pm 4$  [SD]) than the riverine fens.

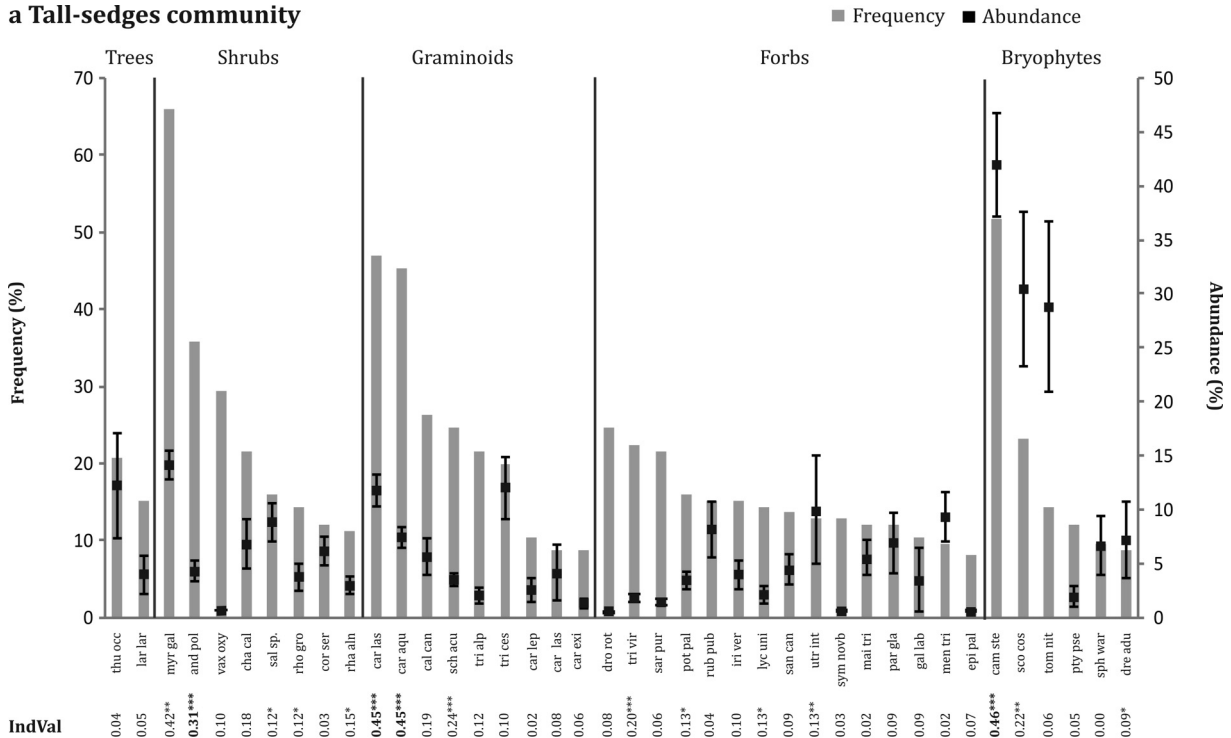
Plant species found in the quadrats ( $n = 200$ ) surveyed in 2012 were grouped into two vegetation communities (Fig. 4; the complete list of recorded species is available in the Appendix, Table A1), according to the K-means analysis with the highest Calinsky–Harabasz criterion. RDA also divided plant quadrats into two distinct groups (Fig. 5). The first group, named tall-sedge community, was the most widely represented, with 126 quadrats out of 200. *Campylium stellatum*, *Carex lasiocarpa*, and *C. aquatilis* were the indicator species of this group, as revealed by the IndVal analysis (Fig. 4). Other associated species were *Myrica gale*, *Andromeda polifolia* var. *latifolia*, *Schoenoplectus acutus* var. *acutus*, and *Scorpidium cossonii*. These plants were mainly found in sites with a water table close to the surface and open habitats. The high production of graminoid biomass ( $>120$   $\text{g}\cdot\text{m}^{-2}$ ), which characterized this group, was positively correlated with electroconductivity, Mg, Ca, and distance to the St. Lawrence River, but negatively with  $\text{SO}_4$  concentration, bulk density, and the Shannon diversity index.

The second group, named *Sphagnum–Thuja* community, pooled the other 74 quadrats. *Sphagnum warnstorffii* had the highest species indicator value. Other indicator species were mainly woody species (*Linnaea borealis*, *Rhododendron groenlandicum*, *Thuja occidentalis*, and *Vaccinium oxycoccus*) or forbs (*Maianthemum trifolium* and *Rubus pubescens*). These species usually grew in drier microhabitats like hummocks. *Carex leptalea* and *C. disperma*, also indicator species, were usually associated with mossy and shady forests.

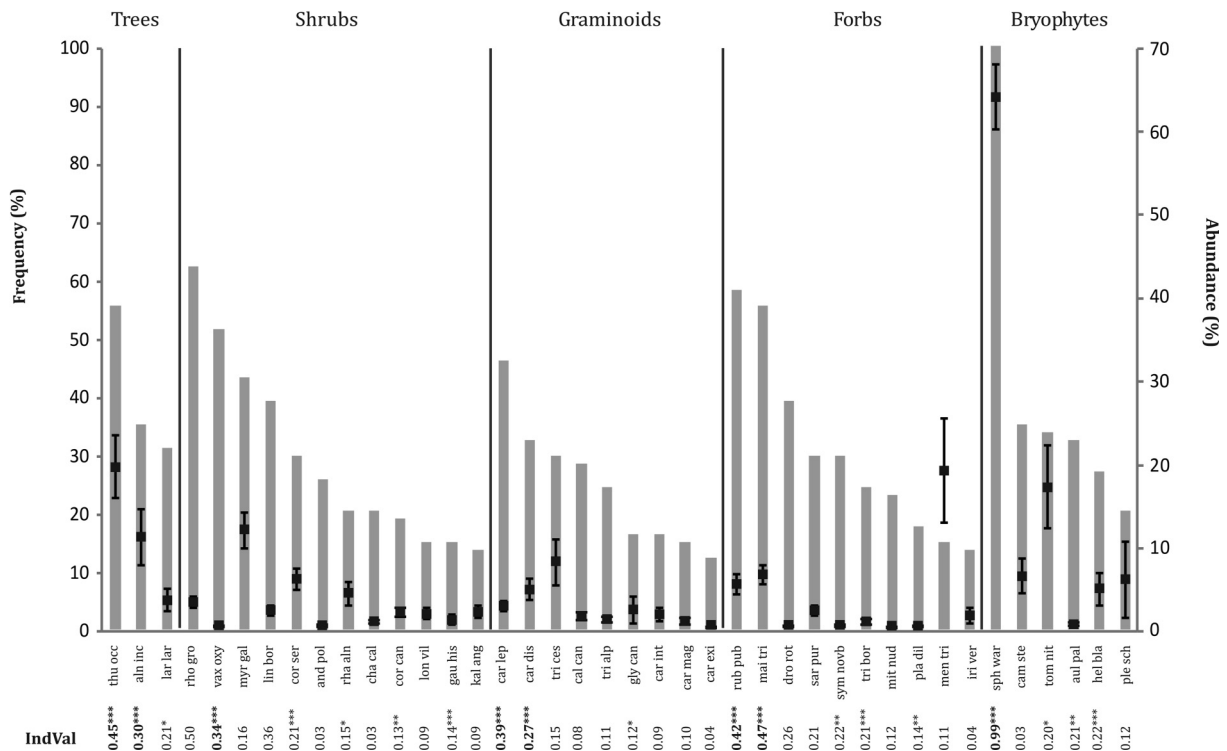
### Discussion

We used paleoecological tools and modern (present-day) plant surveys to gain knowledge on plant assemblages occurring in moderately rich fens of St. Lawrence Lowlands. The presence in the gyttja sediments of species thriving in ponds and completing most of their life cycle submerged, such as *Cristatella mucedo* and *Najas flexilis* (Ricciardi and Reisinger 1994; Wingfield et al. 2005)

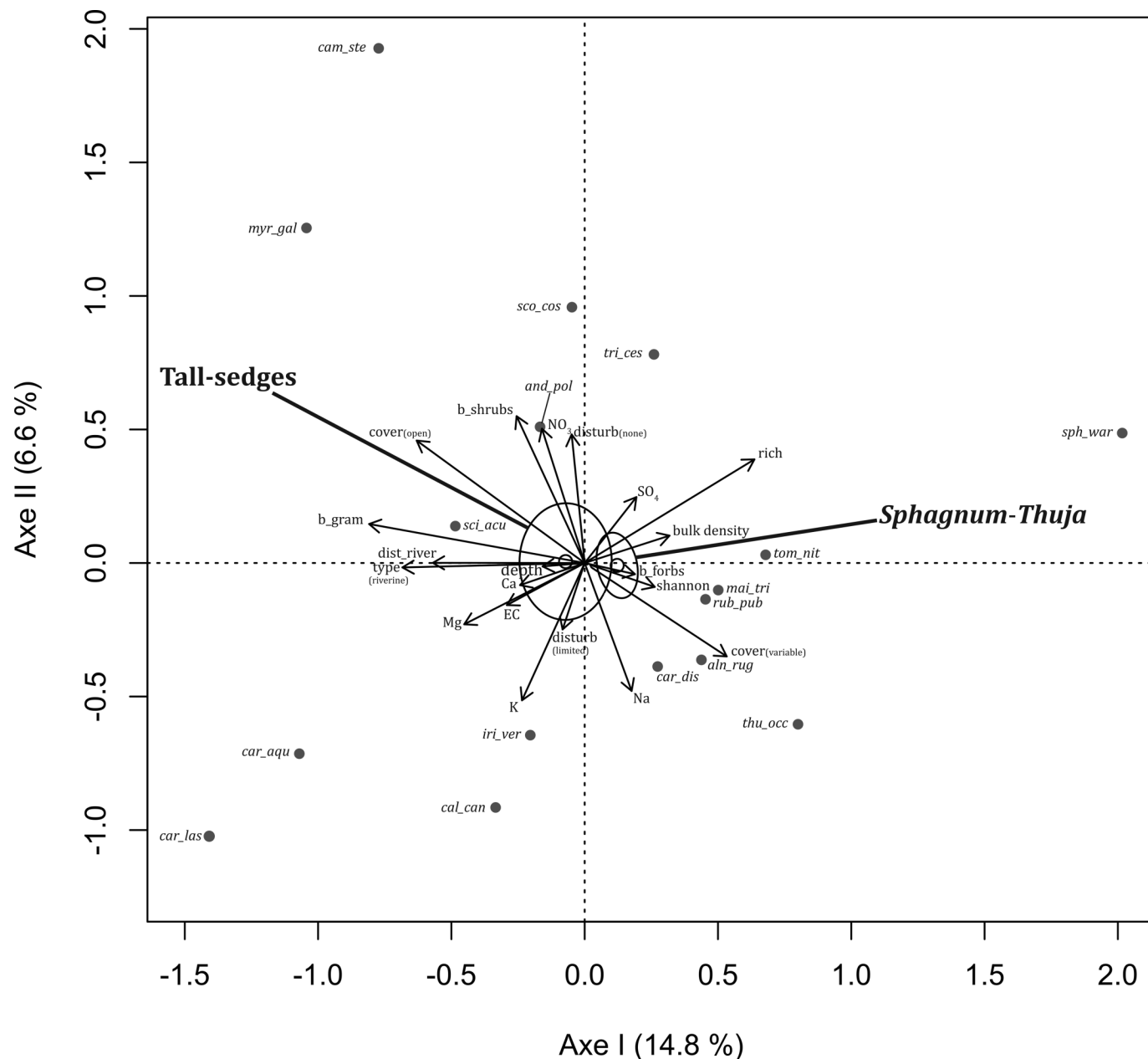
**Fig. 4.** The 40 most frequent species found in fens of the Bas-Saint-Laurent region, classified by life form and grouped into two communities obtained by *K*-means partitioning: (a) tall-sedge and (b) *Sphagnum-Thuja* communities. Frequency (bar chart) represents, in percentage, the occurrence of each species in all quadrats. Abundance (dots), also in percentage, is the average cover ( $\pm$ SD) per quadrat when a species is present. Significant indicator species obtained with the IndVal method are in bold. We retained species with an IndVal > 0.25 and  $P < 0.001$  of each community; (\*\*,  $P < 0.01$ ; \*,  $P < 0.05$ ). The full name of species, coded here by the first three letters for the genus and species, are available in the [Appendix Table A1](#).



**b Sphagnum-Thuja community**



**Fig. 5.** Redundancy analysis (RDA) triplot performed with 17 significant environmental variables. RDA explained 33.1% of the total variation ( $r^2 = 0.36$ ,  $F = 6.348$ ,  $P < 0.001$ ). Open circles represent centroids of the ellipses and include 95% of the quadrats from the two plant communities obtained by *K*-means partitioning. Only the 16 species with the highest scores (most significant) are shown for clarity. Codes name for species are available in the [Appendix Table A1](#), and for the environmental variables in [Table 1](#).



suggests a limnogenous origin for the Bic-Saint-Fabien peatland. Peatland development followed a successional pathway from a shallow water body to a bryophyte-dominated habitat. This hydrosere succession is a common form of autogenic development (Kuhry et al. 1993; Charman 2002). In short, paleoecological analyses revealed the existence of three distinct plant assemblages: (i) an emergent plant assemblage in the first phase of the peatland history (dominated by *Schoenoplectus acutus* or *S. tabernaemontani* and *Menyanthes trifoliata*); (ii) a tall-sedge assemblage (dominated by *Carex rostrata* and *Carex diandra*); and (iii) a brown moss assemblage with small

herbaceous plants, including *Paludella squarrosa*, various species of Amblystegiaceae and Calliergonaceae, *Rhynchospora alba*, and *Trichophorum alpinum*.

Modern plant surveys showed the existence of two distinct plant communities dominated by (i) tall sedges or (ii) *Sphagnum* species with *Thuja occidentalis*. The first community was associated with *Carex aquatilis*, *Carex lasiocarpa*, and *Campyllum stellatum*, all species performing better in a high water table and with a high pH (Gignac 1992; Gignac et al. 2004). The analogs to the tall-sedge community were found in paleoecological assemblages, especially in zone II. The second community was

strongly associated with *Sphagnum warnstorffii* along with generalist wetland boreal species found in dryer peatland microhabitats (Kenkel 1987; Locky and Bayley 2005).

#### Relevance of paleoecological analyses

Paleoecological studies are usually used to assess plant succession through the development of the peatland and to identify processes driving this succession. Here, the results were used in a restoration perspective. First, radiocarbon dates of surface peat may provide indications on the amount of time it was necessary to form the peat deposit that was extracted. The 50 years of peat harvesting activities resulted, at R1 and R2, in a loss of about 2500 to 3200 years of peat accumulation. At R3, 10 years of peat harvesting resulted in a loss of about 800 years of peat. At the time peat harvesting stopped, the BSF site at the location of the R1 and R2 cores regressed, from a peat stratigraphy point of view, from a *Sphagnum* bog to a poor fen. The plant assemblage found at the top part of these cores included species representative of moderately rich to rich fen (e.g., mosses from Amblystegiaceae family, *Ptychostomum pseudotriquetrum*) but also from poor fen (e.g., Ericaceous species, *Comarum palustre*). The richness status could be interpreted as the transitional between the two types of fens (rich and poor). The R3 core most probably regressed to a rich fen. According to successional patterns observed in the paleoecological analyses, target plant assemblage for reference purposes should be inspired from moderately rich fen in the eastern part (R3 location). The same assemblage could be introduced in the central part (R1 and R2) as long as the water chemistry and hydrology fall into the range seen in natural fens. As the paleoecological status was not clear-cut, if the environmental conditions at BSF are closer to poor fen characteristics, we advised targeting this specific assemblage.

High occurrence of bryophytes in zones II and IV generated a higher peat accumulation rate, averaging respectively 1.1 and 0.6 mm per year compared with 0.05 mm for zone III where bryophytes were absent. Even though brown mosses decompose at a faster pace than *Sphagnum* species (Vitt 1990), they decompose at a slower rate than vascular plants (Bartsch and Moore 1985), and thus generate significant peat accumulation. If the most important objective of a peatland restoration plan is to restore the carbon sequestration function, then particular attention should be given to the return of the bryophytes.

Natural disturbance events are of interest for restoration ecologists. By looking at the evolution of an ecosystem after disturbance, one may detect which species help system recovery and the potential trajectories that a restored ecosystem can potentially take (Robert et al. 1999). Fires are a common natural disturbance in peatlands (Turetsky and St. Louis 2006; Lavoie and Pellerin 2007) and past fire events (charcoal layers) are easily detectable in paleoecological analyses. For instance,

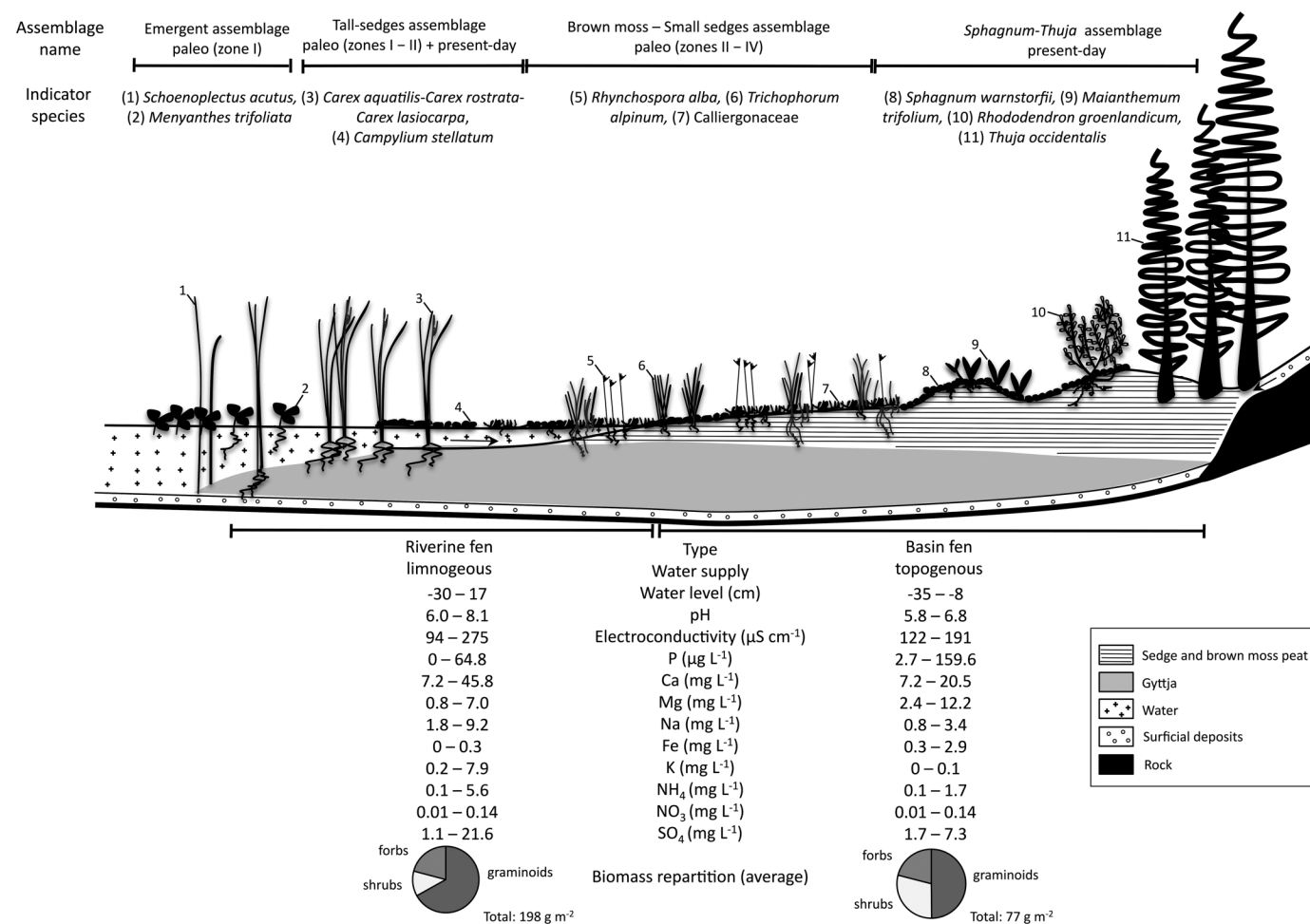
*Polytrichum strictum* grew in abundance after fire in ombrotrophic peatlands (as indicated by paleoecological studies: Kuhry et al. 1993, Lavoie et al. 2001, Bencotter 2006), which suggests that it helps to reestablish the plant cover. The same species was also found to be an excellent nurse plant for *Sphagnum* reestablishment during the restoration of harvested peatlands (Rocheffort et al. 2003; Groeneveld et al. 2007; González et al. 2014). Fire events were not frequent at BSF, probably because the water level in the peatland was probably very close to the surface most of the time (as inferred by the species assemblages), or because the tree cover was too sparse to generate detectable amounts of charcoal. However, at N, *Campylium stellatum* appeared just after the last fire (1930). This species often colonizes bare peat surface soon after a disturbance (Tyler 1984; Arnesen 1999; Mälson et al. 2010) and can grow on mineral ground (Gauthier 2014; Borkenhagen and Cooper 2016). *Campylium stellatum* was observed, during vegetation surveys in 2012, growing abundantly in disturbed locations such as all-terrain vehicles trails where the surface vegetation cover disappeared. Even if there was no evidence of its role in the facilitation of plant reestablishment, *C. stellatum* could be a candidate to test in this respect.

#### Usefulness of modern plant surveys

We surveyed a wide range of minerotrophic peatland types in the BSF region, but only two distinct plant communities were identified, i.e., tall-sedge and *Sphagnum*–*Thuja* communities. These communities are similar in species composition to those found in open, moderately-rich to extremely-rich fens of western boreal Canada (Chee and Vitt 1989; Locky and Bayley 2005), and in the forested cedar swamp of northern Ontario (Kenkel 1987; Jeglum 1991). Although similarities have been found, reference system determination should essentially be based on local surveys, because species habitat requirements may change from region to region (Clewell and Aronson 2013). Moreover, modern plant surveys may also highlight discrepancies between past vegetation assemblages (which are always only a partial record of real communities, because not all plants produce well-preserved macrofossils) and modern vegetation. For instance, *Thuja occidentalis* was very abundant in the peatlands in 2012 but was not found in the macrofossil analyses.

Knowledge of the environmental variables in undisturbed natural sites can be used as a guideline for evaluating the success of a restoration project (Ruiz-Jaen and Aide 2005; Shackelford et al. 2013). In Table 1 and summary Fig. 6, we showed the range of variability for several environmental variables for fens of the BSF area. The characteristics of the restored site should perhaps fall within this range. The nutrients and mineral water concentrations fall within the range seen in the literature for fen (compilation from Andersen et al. 2011), especially moderately rich fen (Chee and Vitt 1989). Among interesting observations that should be considered

**Fig. 6.** Schematic representations of the plant assemblages and their associated habitats within a moderately rich fen ecosystem as described by the paleoecological stratigraphies and present-day studies. The range of variability of the main water chemical elements and biomass are also summarized for two main fen types.



during the restoration, there was the higher nitrate concentrations that favored shrub growth. Although it is recognized that high nitrate concentrations can negatively impact plant diversity in fens (Drexler and Bedford 2002; Bobbink et al. 2010), that was not the case in our study. However, in the long term, shrub expansion might be detrimental to plant diversity, by diminishing moss abundance for example (Pedrotti et al. 2014). As moss cover is a difficult characteristic to re-establish in disturbed fens, nitrogen concentration should be considered carefully before implementing a restoration project. A second observation concerns the possible link between the high sulfate concentrations in water, such as seen in riverine fens, and poor species richness. There are numerous factors that affect species richness, and we do not suggest that sulfate is the explanation. However, there is a paucity of research on the direct impact of sulfate on vegetation communities in peatlands, as most studies on the influence of the sulfur cycle looked at microorganisms (Andersen et al. 2013) and the impact on the C cycle (Vile and Novák 2006). Since sulfate concentrations increased with time after the restoration

(Andersen et al. 2010), the interaction between sulfate and vegetation should be examined closer.

In summary, our present-day study succeeded in establishing relationships between plant assemblages and a set of environmental conditions (Fig. 6). These relationships are the foundation of a reference system for restoration. Therefore, to expect success, the reference system for restoration should match up with the environmental conditions in the disturbed site to be restored. In the BSF case study, for example, if a high water table is predicted, tall-sedge assemblage will be more successful than a *Sphagnum warnstorffii* dominated assemblage. Thus, it is only after evaluation of the disturbed site conditions that we can determine the best plant assemblage to introduce (Clewel and Aronson 2013).

#### Application of the results: BSF case-study

As seen previously in both paleoecological and present-day studies, a total of four fen plant communities were described as possible references for the BSF restoration project. We should then target one or several of these communities for this project. At BSF, the topography of

the disturbed area is one of the main constraints influencing the choice of an appropriate reference. The central section of BSF, where R1 and R2 were cored, is in a lower position compared with the eastern section (R3, N). The monitoring of the water table depth and water chemical analyses indicated that the blockage of the drainage ditches in 2009 consistently maintained a high water table in the central section (Malloy and Price 2014), which was facilitated by the fact that mineral-rich water is continuously flowing toward this section (L. Rochefort, unpublished data). Spontaneous revegetation, mostly with *Typha latifolia*, *Scirpus cyperinus*, *S. atrocinctus*, and *Salix* species, already occurred along former drainage ditches in the central section. Reintroducing small herbaceous plants and bryophytes could be risky because they will likely be out-competed. Paleoecological analyses would suggest the restoration of a poor fen assemblage in the central section (R1 and R2 cores), but modern data would instead point toward the restoration of a tall-sedge community. Species from this community are probably competitive enough to cope with the other species already established.

Peat harvesting did not last long in the eastern section of BSF where R3 was cored. However, negative impacts are important: peat compaction and subsidence on the first metre of the peat profile, low water table depth, frost heaving, no spontaneous vegetation, etc. These impacts are commonly seen in post-harvested cutover peatlands (Price 1996; Poulin et al. 2005). At R3, a *Sphagnum*-brown moss community with more hummock species and with a better survival in driest conditions would consequently be an objective that can realistically be achieved with the moss layer transfer technique currently used in North American bogs (Rochefort et al. 2003).

#### Complementarity of both methods

The selection of species to introduce during the restoration process is the key to reach broader functional and ecosystem stability objectives (Clewel and Aronson 2013; Shackelford et al. 2013). We focused on the gaining information on minerotrophic peatland plant communities, as this is becoming a more prominent goal as well as a tool for peatland restoration planning. Present-day studies produce more applied outcomes but only illustrated a snapshot in the sequence of changing species. Paleoecological tools informed us about the succession of communities and link them to autogenic or allogenic processes. In our study, long persistence in some communities indicated stability. Emergent species were replaced by mat species, then bryophytes and small herbaceous plants. The present-day vegetation at BSF still supports this last community. The disadvantages of paleoecological analyses center on the time required to process cores, and the need to train people. Therefore, paleoecological analyses are less applicable or accessible for smaller restoration projects. But, once it is done in a region, one can assume similar dynamics for similar hydrogeo-

morphological settings of peatlands (e.g., Swinehart and Parker 2000; Arlen-Pouliot 2009). In this sense, field vegetation surveys have the benefits of being faster and have more direct applied outcomes than paleoecological studies, mostly to describe environmental variables associated with vegetation. Paleoecology extrapolates peatland development from one precise point, and the interpretation of cores is based on actual knowledge of the peatland ecology. Still, understanding ecosystem dynamics is an important component of restoration, (SERI 2004; Shackelford et al. 2013), and this understanding is best achieved with paleoecology. For example, in our study, the structure of a plant community was shown to be stable in time where tall-sedge community did initiate peat accumulation and brown moss was an essential component. From our present-day study, we were able to compare distance between reference and disturbed sites and make clearer restoration objectives.

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## Appendix A

Appendix Table A1 appears on the following pages.

**Table A1.** List of species identified during present-day inventories in 19 moderately rich to rich minerotrophic peatlands (and units) located around Rimouski, Quebec, Canada.

Form	Code	Species names	Authority names
Bryophyte	aul_pal	<i>Aulacomnium palustre</i>	(Hedw.) Schwaegr.
Bryophyte	pty_pse	<i>Ptychostomum s.l. pseudotriquetrum</i>	(Hedw.) J.R. Spence & H.P. Ramsay
Bryophyte	bryo_sp	<i>Bryophyte</i> sp.	
Bryophyte	cal_cus	<i>Calliergonella cuspidata</i>	(Hedw.) Loeske
Bryophyte	cal_gig	<i>Calliergon giganteum</i>	(Schimp.) Kindb.
Bryophyte	cal_lin	<i>Calliergonella lindbergii</i>	(Mitt.) Hedenäs
Bryophyte	calli_sp	<i>Calliergon</i> s.l. sp.	(Sull.) Kindb.
Bryophyte	cam_ste	<i>Campylium stellatum</i>	(Hedw.) Lange & C.E.O. Jensen
Bryophyte	cli_den	<i>Climacium dendroides</i>	(Hedw.) Web. & Mohr
Bryophyte	dic_pol	<i>Dicranum polysetum</i>	Sw. ex anon.
Bryophyte	dre_adu	<i>Drepanocladus aduncus</i>	(Hedw.) Warnst.
Bryophyte	dre_unc	<i>Sanionia uncinata</i>	(Hedw.) Loeske
Bryophyte	drep_sp	<i>Depanocladus</i> s.l. sp.	(C. Müll.) G. Roth
Bryophyte	fis_sp	<i>Fissidens</i> sp.	Hedw.
Bryophyte	ham_ver	<i>Hamatocaulis vernicosus</i>	(Mitt.) Hedenäs
Bryophyte	hel_bla	<i>Helodium blandowii</i>	(Web. & Mohr) Warnst.
Bryophyte	hep_sp	<i>Hepatic</i> sp.	
Bryophyte	hyl_spl	<i>Hylocomium splendens</i>	(Hedw.) Schimp.
Bryophyte	hypn_sp	<i>Hypnaceae</i> sp.	
Bryophyte	mni_pun	<i>Rhizomnium punctatum</i>	(Hedw.) T.J. Kop.
Bryophyte	mni_sp	<i>Mnium</i> sp.	Hedw.
Bryophyte	pal_squ	<i>Paludella squarrosa</i>	(Hedw.) Brid.
Bryophyte	ple_sch	<i>Pleurozium schreberi</i>	(Brid.) Mitt.
Bryophyte	pol_str	<i>Polytrichum strictum</i>	Brid.
Bryophyte	pti_cri	<i>Ptilium crista-castrensis</i>	(Hedw.) De Not.
Bryophyte	rhy_tri	<i>Rhytidiadelphus triquetrus</i>	(Hedw.) Warnst.
Bryophyte	sco_cos	<i>Scorpidium cossonii</i>	(Schimp.) Hedenäs
Bryophyte	sco_sco	<i>Scopidium scorpioides</i>	(Hedw.) Limpr.
Bryophyte	sph_ang	<i>Sphagnum angustifolium</i>	(C. Jens. ex Russ.) C. Jens.
Bryophyte	sph_cen	<i>Sphagnum centrale</i>	C. Jens
Bryophyte	sph_fim	<i>Sphagnum fimbriatum</i>	Wils.
Bryophyte	sph_fus	<i>Sphagnum fuscum</i>	(Schimp.) Klinggr.
Bryophyte	sph_mag	<i>Sphagnum magellanicum</i>	Brid.
Bryophyte	sph_pal	<i>Sphagnum palustre</i>	L.
Bryophyte	sph_squ	<i>Sphagnum squarrosum</i>	Crome.
Bryophyte	sph_sub	<i>Sphagnum subsecundum</i>	Nees.
Bryophyte	sph_ter	<i>Sphagnum teres</i>	(Schimp.) Angstr.
Bryophyte	sph_war	<i>Sphagnum warnstorffii</i>	Russ.
Bryophyte	thu_del	<i>Thuidium delicatulum</i>	(Hedw.) Schimp.
Bryophyte	tom_nit	<i>Tomenthypnum nitens</i>	(Hedw.) Loeske.
Bryophyte	tor_tor	<i>Tortella tortuosa</i>	(Hedw.) Limpr.
Graminoid	cal_can	<i>Calamagrostis canadensis</i>	(Michx.) Nutt.
Graminoid	car_aqu	<i>Carex aquatilis</i>	Wahl.
Graminoid	car_aur	<i>Carex aurea</i>	Nutt.
Graminoid	car_dia	<i>Carex c.f. diandra</i>	Schrank
Graminoid	car_dis	<i>Carex disperma</i>	Dewey

Table A1 (continued).

Form	Code	Species names	Authority names
Graminoid	car_ech	<i>Carex echinata</i>	Murray
Graminoid	car_exi	<i>Carex exilis</i>	Dewey.
Graminoid	car_fla	<i>Carex flava</i>	L.
Graminoid	car_gyn	<i>Carex gynocrates</i>	Wormsk. ex Drejer
Graminoid	car_int	<i>Carex interior</i>	L.H. Bailey
Graminoid	car_lac	<i>Carex lacustris</i>	Willd.
Graminoid	car_las	<i>Carex lasiocarpa</i>	Ehrh.
Graminoid	car_lep	<i>Carex leptalea</i>	Wahl.
Graminoid	car_lim	<i>Carex limosa</i>	L.
Graminoid	car_mag	<i>Carex magellanica</i>	Lam.
Graminoid	car_pra	<i>Carex prairea</i>	Dewey ex Alph. Wood
Graminoid	car_pse	<i>Carex pseudocyperus</i>	L.
Graminoid	car_sp	<i>Carex</i> sp.	L.
Graminoid	car_vag	<i>Carex vaginata</i>	Tausch
Graminoid	eri_viri	<i>Eriophorum viridicarinatum</i>	(Engelm.) Fernald.
Graminoid	gly_can	<i>Glyceria canadensis</i>	(Michx.) Trin.
Graminoid	mul_glo	<i>Muhlenbergia glomerata</i>	(Willd.) Trin.
Graminoid	poa_sp	<i>Poaceae</i> sp.	
Graminoid	rhy_alb	<i>Rhynchospora alba</i>	(L.) Vahl.
Graminoid	rhy_fus	<i>Rhynchospora fusca</i>	(L.) W.T. Aiton
Graminoid	sci_acu	<i>Schoenoplectus acutus</i> var. <i>acutus</i>	(Muhl. ex Bigelow) Á. Löve & D. Löve
Graminoid	tri_alp	<i>Trichophorum alpinum</i>	L.
Graminoid	tri_ces	<i>Trichophorum cespitosum</i>	L.
Forb	gal_rot	<i>Galearis rotundifolia</i>	(Banks ex Pursh) R.M. Bateman
Forb	ara_nud	<i>Aralia nudicaulis</i>	L.
Forb	are_bul	<i>Arethusa bulbosa</i>	L.
Forb	ast_novb	<i>Symphotrichum novi-belgii</i> var. <i>novi-belgii</i>	(L.) G.L. Nesom
Forb	ast_pun	<i>Symphotrichum puniceum</i>	(L.) Á. Löve & D. Löve
Forb	ast_sp	<i>Aster</i> s.l. sp.	
Forb	doe_umb	<i>Doellingeria umbellata</i>	(Mill.) Nees
Forb	ath_fil	<i>Athyrium filix-femina</i>	(L.) Roth
Forb	cal_pal	<i>Calla palustris</i>	L.
Forb	cic_bul	<i>Cicuta bulbifera</i>	L.
Forb	cli_bor	<i>Clintonia borealis</i>	(Aiton) Raf.
Forb	con_chi	<i>Conioselinum chinense</i>	(L.) Britton, Sterns & Poggenb.
Forb	cop_tri	<i>Coptis trifolia</i>	L.
Forb	cyp_par	<i>Cypripedium parviflorum</i>	Salisb.
Forb	cyp_reg	<i>Cypripedium reginae</i>	Walter
Forb	dro_int	<i>Drosera intermedia</i>	Hayne.
Forb	dro_rot	<i>Drosera rotundifolia</i>	L.
Forb	epi_pal	<i>Epilobium palustre</i>	L.
Forb	equ_pal	<i>Equisetum palustre</i>	L.
Forb	eup_mac	<i>Eutrochium maculatum</i>	(L.) E.E. Lamont
Forb	fra_vir	<i>Fragaria virginiana</i>	Duchesne.
Forb	gal_lab	<i>Galium labradoricum</i>	(Wiegand) Wiegand
Forb	gal_sp	<i>Galium</i> sp.	L.
Forb	gal_tri	<i>Galium triflorum</i>	Michx.
Forb	geu_riv	<i>Geum rivale</i>	L.
Forb	tri_vir	<i>Triadenum virginicum</i>	(L.) Raf.
Forb	iri_ver	<i>Iris versicolor</i>	L.
Forb	neo_con	<i>Neottia convallariodes</i>	(Sw.) Rich.
Forb	lob_kal	<i>Lobelia kalmii</i>	L.
Forb	lyc_uni	<i>Lycopus uniflorus</i>	Michx.
Forb	mai_can	<i>Maianthemum canadense</i>	Desf.
Forb	mal_uni	<i>Malaxia unifolia</i>	Michx.
Forb	men_tri	<i>Menyanthes trifoliata</i>	L.
Forb	mit_nud	<i>Mitella nuda</i>	L.
Forb	nup_var	<i>Nuphar variegata</i>	Engelm.

Table A1 (continued).

Form	Code	Species names	Authority names
Forb	orch_sp	<i>Orchidaceae</i>	
Forb	osm_cin	<i>Osmundastrum cinnamomea</i>	(L.) C. Presl
Forb	osm_reg	<i>Osmunda regalis</i>	L.
Forb	par_gla	<i>Parnassia glauca</i>	Raf.
Forb	phr_aut	<i>Phragmites australis</i> subsp. <i>americanus</i>	(Cav.) Trin. ex Steud.
Forb	gym_cla	<i>Gymnadeniopsis clavellata</i>	(Michx.) Rydb.
Forb	pla_dil	<i>Platanthera dilatata</i>	(Pursh) Lindl. ex L.C. Beck
Forb	pog_oph	<i>Pogonia ophioglossoides</i>	(L.) Ker-Gawl.
Forb	per_amp	<i>Persicaria amphibia</i>	(L.) DeTree
Forb	com_pal	<i>Comarum palustre</i>	L.
Forb	pota_sp	<i>Potamogeton</i> sp.	L.
Forb	pyr_asa	<i>Pyrola asarifolia</i>	Michx.
Forb	pyr_ell	<i>Pyrola elliptica</i>	Nutt.
Forb	ran_acr	<i>Ranunculus acris</i>	A. Gray
Forb	rub_pub	<i>Rubus pubescens</i>	Raf.
Forb	sag_lat	<i>Sagittaria latifolia</i>	Willd.
Forb	san_can	<i>Sanguisorba canadensis</i>	L.
Forb	sar_pur	<i>Sarracenia purpurea</i>	L.
Forb	scu_gal	<i>Scutellaria galericulata</i>	L.
Forb	sel_sel	<i>Selaginella selaginoides</i>	(L.) Link.
Forb	mai_tri	<i>Maianthemum trifolium</i>	L.
Forb	sol_uli	<i>Solidago uliginosa</i>	Nutt.
Forb	tha_pub	<i>Thalictrum pubescens</i>	Pursh
Forb	the_pal	<i>Thelypteris palustris</i>	Schott
Forb	thy_lat	<i>Typha latifolia</i>	L.
Forb	tof_glu	<i>Tofieldia glutinosa</i>	(Michx.) Pers.
Forb	tri_bor	<i>Trientalis borealis</i>	Raf.
Forb	tri_mar	<i>Triglochin maritima</i>	L.
Forb	utr_int	<i>Utricularia intermedia</i>	Hayne.
Forb	val_uli	<i>Valeriana sitchensis</i> subsp. <i>uliginosa</i>	(Torr. & A. Gray) F.G. Mey.
Forb	vic_cra	<i>Vicia cracca</i>	L.
Forb	vio_sp	<i>Viola</i> sp.	L.
Shrub	and_pol	<i>Andromeda polifolia</i> var. <i>latifolia</i>	Aiton
Shrub	cha_cal	<i>Chamaedaphne calyculata</i>	(L.) Moench.
Shrub	cor_can	<i>Cornus canadensis</i>	L.
Shrub	cor_ser	<i>Cornus sericea</i>	L.
Shrub	gau_his	<i>Gaultheria hispidula</i>	(L.) Muhl. ex Bigelow
Shrub	gay_bac	<i>Gaylussacia baccata</i>	(Wang.) K. Koch.
Shrub	kal_ang	<i>Kalmia angustifolia</i>	L.
Shrub	kal_pol	<i>Kalmia polifolia</i>	Wang.
Shrub	rho_gro	<i>Rhododendron groenlandicum</i>	(Oeder) Kron & Judd
Shrub	lin_bor	<i>Linnaea borealis</i>	L.
Shrub	lon_vil	<i>Lonicera villosa</i>	(Michx.) R. & S.
Shrub	myr_gal	<i>Myrica gale</i>	L.
Shrub	nep_muc	<i>Nemopanthus mucronatus</i>	(L.) Trel.
Shrub	dai_fru	<i>Dasiphora fruticosa</i>	(L.) Rydb.
Shrub	rha_aln	<i>Rhamnus alnifolia</i>	L'Hér.
Shrub	rib_lac	<i>Ribes lacustre</i>	(Pers.) Poir.
Shrub	ros_bla	<i>Rosa palustris</i>	Marshall
Shrub	sal_beb	<i>Salix bebianna</i>	Sarg.
Shrub	sal_dis	<i>Salix discolor</i>	Mühl.
Shrub	sal_sp	<i>Salix</i> sp.	L.
Shrub	spi_lat	<i>Spiraea latifolia</i>	(Ait.) Borkh.
Shrub	vac_ang	<i>Vaccinium angustifolium</i>	Ait.
Shrub	vac_oxy	<i>Vaccinium oxycoccos</i>	L.
Shrub	vib_cas	<i>Viburnum cassinoides</i>	L.
Tree	abi_bal	<i>Abies balsamea</i>	(L.) Mill.
Tree	ace_pen	<i>Acer pensylvanicum</i>	L.

**Table A1** (concluded).

Form	Code	Species names	Authority names
Tree	ace_rub	<i>Acer rubrum</i>	L.
Tree	aln_rug	<i>Alnus incana</i>	(L.) Moench.
Tree	bet_pap	<i>Betula papyrifera</i>	Marsh.
Tree	fra_nig	<i>Fraxinus nigra</i>	Marsh.
Tree	lar_lar	<i>Larix laricina</i>	(DuRoi) Koch.
Tree	pic_mar	<i>Picea mariana</i>	(Mill.) BSP.
Tree	sor_dec	<i>Sorbus decora</i>	(Sarg.) C.K. Schneid.
Tree	thu_occ	<i>Thuja occidentalis</i>	L.

**Note:** Nomenclature follows the [Integrated Taxonomic Information System \(2015\)](#) for vascular plants, and [Faubert \(2007\)](#) for bryophytes.

## References

Faubert, J. 2007. Catalogue des bryophytes du Québec et du Labrador. *Provancheria* : Mémoire de l'Herbier Louis-Marie, no. 30, Université Laval, Qué.

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