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Vegetation dynamics on an abandoned vacuum-mined peatland: 5 years of monitoring

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Abstract

We studied from 1998 to 2002 the fine-scale vegetation dynamics of a poorly regenerated vacuum-mined bog located in southern Quebec. We selected mined sites that have been abandoned for 14 years and monitored the vascular and non-vascular plants, and some hydrological characteristics. We focussed our study on the monitoring of cotton-grass (*Eriophorum vaginatum* L.) tussocks. Major changes in the plant cover were observed during the five-year period, such as a decrease (26–31%) in the number of cotton-grass tussocks and an increase in the ericaceous shrub cover. The water table level (lower than 40 cm below the soil surface) and frost heaving appear to be the main factors explaining the failure of cotton-grass and of other typical bog plant species to colonize abandoned mined surfaces. The ericaceous shrub cover, although increasing, was still sparse even after two decades of abandonment, and it may take several additional decades before a complete shrub cover establishes itself. While the natural revegetation process of this vacuum-mined bog is still dynamic after two decades of abandonment, there is no evidence to suggest that vegetation assemblages will eventually resemble those of undisturbed peatlands.

Introduction

Peatland mining is a widespread industrial activity, especially in Canada, Germany, and more recently in the Baltic States where peat is used for horticulture or agriculture, and in Finland, Ireland and Russia where peat is used for domestic heating or energy generation. Extraction activities cover approximately 50,000 km^2 and are responsible for 10% of peatland area losses in the non-tropical world (Joosten and Clarke 2002). Peatland mining is a major anthropogenic disturbance – bogs are drained, vegetation is removed, and a thick layer of soil is extracted – and usually occurs over a period of several decades. A post-mined peatland is a harsh environment for plants, and modern mining techniques (vacuum) seriously hamper the natural capacity of bog ecosystems to regenerate after a disturbance. The revegetation of vacuum-mined bogs is particularly poor compared to that of block-cut peatlands, i.e., sites that have been mined by hand using shovels (Lavoie et al. 2003). In those sites, mined trenches have rapidly (less than 5 years) been revegetated (>90% plant cover) by typical peatland species, although the *Sphagnum* cover remained low (10%) even after 20 years of abandonment (Girard et al. 2002).

Several causes have been identified to explain the slowness of natural processes to regenerate mined bog ecosystems: dryness of the upper peat layers, soil oxidation and compression, frost heaving, wind erosion and low dispersal abilities of bog plants (Salonen 1987; Campbell et al. 2002, 2003; Groeneveld and Rochefort 2002; Waddington and McNeil 2002; Price et al. 2003). Recent studies indicate that highly disturbed peatlands, particularly those that have been mined using tractor-drawn vacuum machines, are rarely revegetated by typical bog plant species even after decades of abandonment (Lavoie et al. 2003). Consequently, peatland managers strongly advocate active restoration of abandoned bogs (Rochefort et al. 2003). However, an effective natural revegetation process might be in action, albeit at a very slow pace, and in the long term, the vegetation of vacuum-mined bogs may fully redevelop through natural processes alone (Bérubé and Lavoie 2000). It is thus essential to determine whether such natural processes are occurring, and how much time is required to obtain a vegetation cover similar to undisturbed sites.

We studied the fine-scale vegetation dynamics of a poorly regenerated vacuum-mined bog. We selected mined sites that have been abandoned for 14 years and monitored vascular and non-vascular plants, and some hydrological characteristics during a five-year period. We focussed our study on cotton-grass (*Eriophorum vaginatum* L.) tussocks, one of the few vascular plants that exhibit some success recolonizing dry peat deposits (Campbell et al. 2003; Lavoie et al. 2003, 2005). Cotton-grass tussocks are presumed to create microclimatic conditions facilitating the establishment and growth of other plants, particularly *Sphagnum* species (Grosvernier et al. 1995; Matthey 1996; Marcoux 2000; Lavoie et al. 2003). We did not select sites that were recently (< 5 years) abandoned (e.g., Salonen et al. 1992) because we theorized that the early phases of the revegetation process do not necessarily give a good indication of future regeneration success, since some plant species rapidly decline after running out of available nutrient resources (Bérubé and Lavoie 2000). We hypothesized that although the establishment of typical bog plant species in vacuum-mined bogs is a very slow process, vegetation assemblages will eventually resemble those of undisturbed peatlands after a few decades.

Study area and methods

The largest concentration of abandoned mined bogs in North America is located in the Bas-Saint-Laurent region, eastern Quebec, Canada (Desrochers et al. 1998; Lavoie and Saint-Louis 1999). The Rivière-du-Loup (RDLP) peatland, the largest bog in this region, was selected for study. It is located 206 km northeast of Quebec City (47°48′ N, 69°30′ W) and covers 3375 ha. This large ombrotrophic peatland is dominated by black spruce (*Picea mariana* [Mill.] B.S.P.), ericaceous shrubs and *Sphagnum* species (Gauthier and Grandtner 1975). In the RDLP region, the climate is wet and continental. Data from the meteorological station at Saint-Arsène, located near (12 km) the study site, indicate that the mean annual temperature is 3 °C. January is the coldest month (mean temperature: −12 °C) and July, the warmest month (mean temperature: 18 °C). The mean annual precipitation is 924 mm, 27% of which falls as snow (Environnement Canada 1993).

Peat mining began on the site in the 1940s. By 1995, approximately 62% of the RDLP bog area had been mined for the production of horticultural peat (Desaulniers 2000). All abandoned vacuum-mined sites are at present poorly revegetated. For this study, we selected two sites (RDLP1 and RDLP2) mined between 1963 and 1984 (Figure 1). These sites were located 200 m from each other and were separated by a mineral ridge and a narrow strip of forest. Approximately 90% of the surrounding area was either currently being mined or have been abandoned after two decades of mining. No drain-blocking or restoration activities were conducted in this sector to regenerate these abandoned sites. On the basis of their vegetation cover, RDLP1 and RDLP2 sites were deemed representative of most abandoned vacuum-mined sites of the Bas-Saint-Laurent region.

At each site, we delineated a 10 × 20 m quadrat that was subdivided into fifty 2 × 2 m subquadrats. In each subquadrat, the percent cover of all vascular and non-vascular plants was estimated each year (in June) from 1998 to 2002 using a semi-quantitative scale: 0%, < 1%, 1–10%, 11–25%, 26–50%, 51–75% and 76–100%. The species cover was compared year to year by calculating the median value of the cover class occupied by the species in each subquadrat, and totalling all

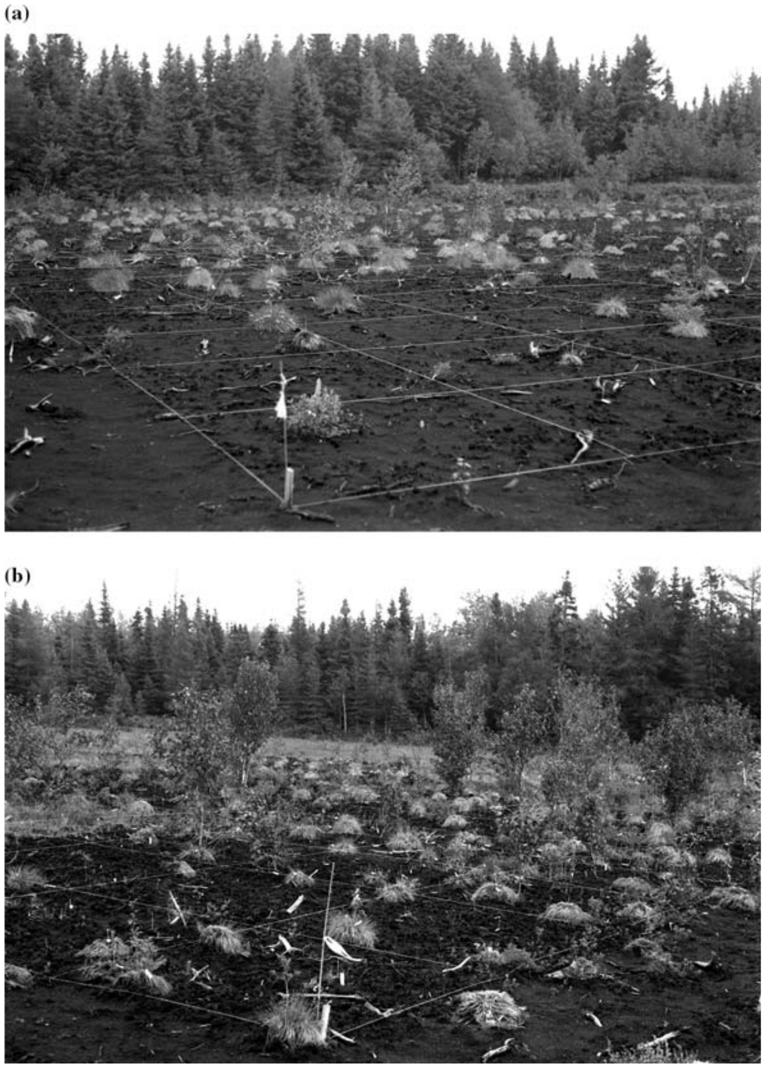

Figure 1. Vacuum-mined sites in the Rivière-du-Loup ombrotrophic peatland (southern Quebec, Canada) that were abandoned in 1984 (photographs taken in 1998): (a) RDLP1 site, sparsely revegetated by cotton-grass (*Eriophorum vaginatum*) tussocks; (b) RDLP2 site, with cotton-grass tussocks and some birch (*Betula papyrifera*) individuals. Disturbance of the soil surface caused by frost heaving is visible in both sites, especially at RDLP2.

median values from all subquadrats. The position of each cotton-grass tussock (Figure 1) was mapped using an $X - Y$ coordinate system, and its status (living or dead, i.e., without green shoots) monitored each year. New tussocks established after 1998 were also monitored, but it is important to note that this growth form can only be clearly distinguished from seedlings after one or two growing seasons. The number of infructescences produced each year by the tussocks was counted. The number of emerging seedlings was also estimated in five randomly selected subquadrats. Using SPSS software (SPSS Inc. 2001), Pearson's product moment correlation coefficients (Gilbert and Savard 1992) were calculated between the number of infructescences and the number of seedlings emerging one year later. Individual trees present in 1998 were mapped, and their status (living or dead) was monitored each subsequent year. The total height of individual trees was also measured. Trees established after 1998 were not monitored on an individual basis because they were too numerous. Great care was taken to do not trample the vegetation, which was relatively easy because of the low cover occupied by plants during all the monitoring period. Nomenclature follows Hinds (2000) for vascular plants and Anderson et al. (1990) for mosses.

One small peat sample from the surface (12 cm) peat layer was taken at the centre of each quadrat for chemical analyses, conducted in the Premier Horticulture Laboratory (Rivière-du-Loup, Québec) using the standard methods suggested by Day et al. (1989). Total peat thickness was estimated using an iron rod driven into the soil at eight different locations in each quadrat (near all corners and mid-points between corners). Precipitation data during the monitoring period were taken from the Saint-Arsène meteorological station until August 1999 (Environnement Canada, unpublished data), and from an automated station of the Peatland Ecology Research Group (J.S. Price, unpublished data) located in the Bois-des-Bel peatland (14 km northeast of the RDLP bog) for the summers of 2000, 2001 and 2002. No precipitation data were available from the Saint-Arsène station after August 1999 nor for the fall–spring seasons from the Bois-des-Bel station. Available precipitation data were compared to a reference period (Saint-Arsène station, 1963–1990; Environnement Canada 1993).

Two hydrological characteristics were monitored on a weekly basis during the summers (May–August) of 2000 and 2001. The water table level was measured using well pipes slotted along their entire length and covered with a geotextile screen. Pipes were inserted into holes that were hand-drilled at three different locations in each quadrat, i.e., at the centre and at each extremity of the quadrat. All pipe bases were set above the peat-mineral interface. The distance from the top of the well to the soil surface was measured throughout the summer to record any possible shifting of the pipe due to peat shrinkage. The volumetric soil moisture content of the surface (12 cm) peat layer was also monitored near each well using a HydroSense^{TM} soil water content measurement system (Campbell Scientific, Logan, Utah). Volumetric soil moisture content data were calibrated for peaty soils in the laboratory using peat samples from southern Quebec peatlands (Marcoux 2000). Hydrological data from RDLP1 were compared to those of RDLP2 using a two-factor analysis of variance with repeated measures (Zar 1984) calculated using SPSS software (SPSS Inc. 2001).

Results

Abiotic characteristics

Both RDLP1 and RDLP2 sites had a thick (167–252 cm) acidic (pH: 3.3–3.6) peat deposit with high *Sphagnum* content. The peat chemistry of both sites was quite similar. In general, chemical properties of the peat were similar to those recorded from abandoned mined bogs (Wind-Mulder et al. 1996): Ca^{2+}, Cl^{-}, Mg^{2+}, Na^{+}, NH_4^{+}–N and NO_3^{-}–N values ranged from 1 to 17 mg l^{-1}.

During the summers of 2000 and 2001, the water table levels at RDLP1 and RDLP2 (Figure 2) were similar, although significantly lower ($p = 0.043$ in 2000 and $p = 0.026$ in 2001) at RDLP1 than at RDLP2. The water table level range for a specific day was also larger at RDLP1. Except for some very brief periods, the water table level at RDLP1 remained at least 40 cm below the soil surface or lower, a threshold value that is considered critical for the rapid re-establishment of *Sphagnum* colonies in mined peatlands; below this value, almost no *Sphagnum* colonies establish or survive on peaty soils (Schouwenaars 1988; Girard et al. 2002; Price et al. 2003). At RDLP2, the water table level was above this 40-cm value during the first half of both summers. Volumetric peat water content values at RDLP1 and RDLP2 were also similar (Figure 2). The volumetric peat water content was significantly lower ($p = 0.034$) at RDLP1 than at RDLP2 only in 2000. No significant difference between sites was found in 2001 ($p = 0.326$). Volumetric peat water content values were always above 50%, which is considered to be another critical threshold for the rapid re-establishment of *Sphagnum* colonies; below this value, few *Sphagnum* colonies establish on mined bog surfaces (Price and Whitehead 2001). However, it is important to note that summers 2000 and 2001 were particularly dry compared to the 1963–1990 reference period (Figure 3).

Cotton-grass tussocks and tree individuals

The spatial distribution of cotton-grass tussocks was fairly homogeneous at RDLP1 and RDLP2 (Figure 4). At the beginning of the monitoring period (1998), the density of living tussocks was 4300 and 12,900 tussocks per hectare at RDLP1

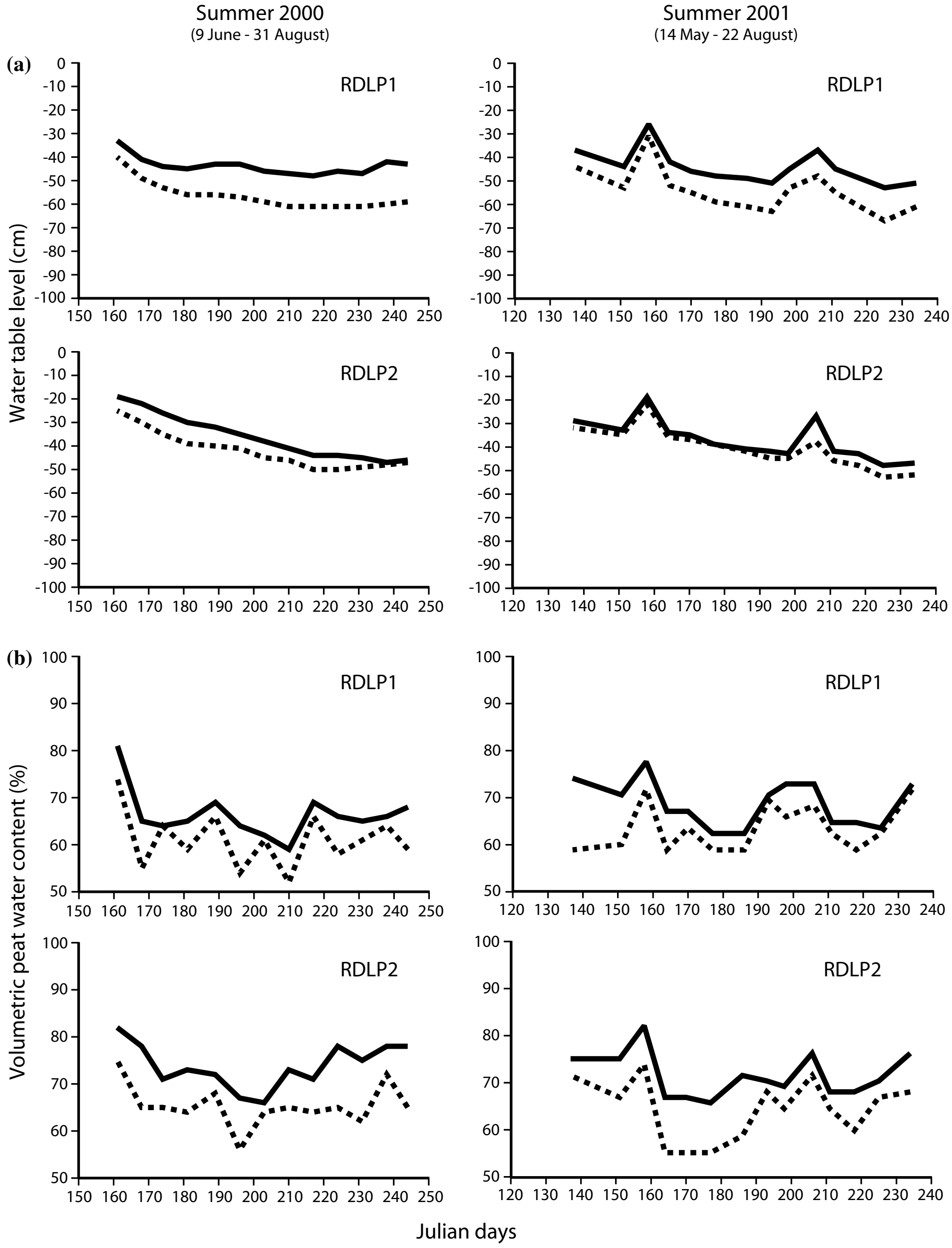

Figure 2. Monitoring of the (a) water table level and (b) volumetric peat water content at RDLP1 and RDLP2 abandoned vacuum-mined sites during summers 2000 and 2001. Highest (continuous lines) and lowest (dotted lines) values recorded are indicated.

and RDLP2, respectively. At both sites, the density decreased to reach 3200 (RDLP1) and 8950 (RDLP2) tussocks per hectare in 2002, a 26–31% decrease (Figure 5). The annual mortality rate of tussocks ranged from 11 to 16%, except at RDLP1 in 2000 (5%). It should be noted that all dead tussocks, except very small ones, remained on site during the monitoring period. Tussocks from RDLP1 produced more infructescences than those from RDLP2 between 1998 and 2001, but RDLP2 tussocks were more productive in 2002 (Figure 6). There were large annual fluctuations in the number of cotton-grass seedlings, with maxima recorded in 2000 and minima in 2001 and 2002. There was no significant correlation coefficient between the number of infructescences and the

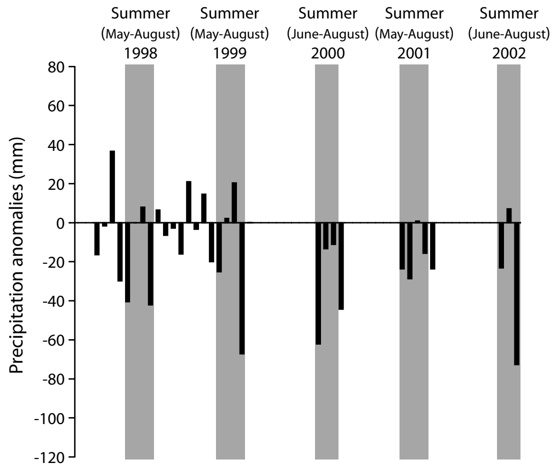

Figure 3. Precipitation anomalies in the Rivière-du-Loup area from January 1998 to August 2002. Vertical bars: difference between precipitation recorded for a particular month and mean precipitation for the same month during the 1963–1990 period. Summer months (May–August) are indicated by grey zones. No data were available for fall–spring seasons in 2000, 2001 and 2002.

number of seedlings emerging one year after in either RDLP1 (p = 0.057) or RDLP2 (p = 0.615), but a longer time series would probably be necessary to detect a significant correlation.

As with the tussocks, individual trees were much more numerous at RDLP2 than at RDLP1 (Figure 4). Only 36 and 46% of trees with a total height $\leq$ 10 cm in 1998 were still living in 2002 at RDLP1 and RDLP2, respectively. On the other hand, more than 83 (RDLP1) and 89% (RDLP2) of trees with a total height $>$10 cm in 1998 were still alive in 2002. However, 36% of birch (*Betula papyrifera* Marsh.) individuals displayed evidence of decline (foliage loss) at RDLP2 as early as 1999.

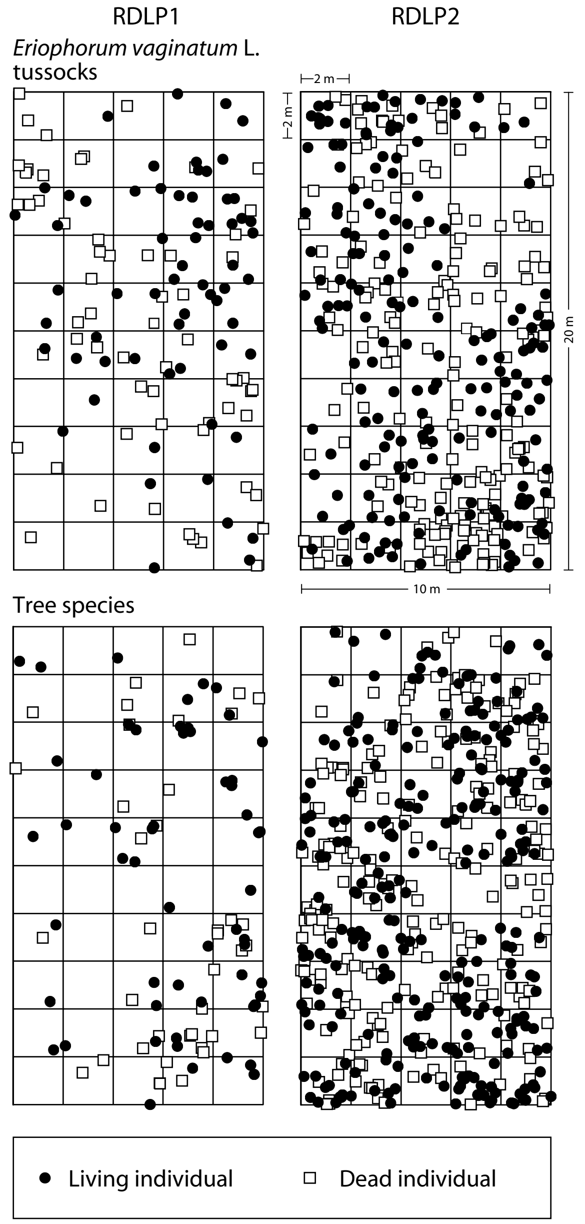

Figure 4. Spatial distribution of cotton-grass (*Eriophorum vaginatum*) tussocks and tree individuals at RDLP1 and RDLP2 abandoned vacuum-mined sites in 2002. Only tree individuals present in 1998 are shown, with their status (living or dead) in 2002.

Plant cover

Major changes in the plant cover occurred during the five-year monitoring period. The cover of birch, black spruce and blueberry (*Vaccinium angustifolium* Ait.) increased at RDLP1, while that of cotton-grass decreased, especially in 2001 and 2002 (Table 1; Figure 7). More plant species were recorded at RDLP2 (Table 1), including some typical bog species never found at RDLP1 (*Empetrum nigrum* L., *Eriophorum angustifolium* Honckeny, *Drosera rotundifolia* L., *Gaultheria hispidula* [L.] Muhl., *Kalmia polifolia* Wangenh., *Sphagnum* cf. *rubellum* Wils.). The presence of five small *Sphagnum* colonies, two of them present in 1998, is noteworthy, but during the monitoring period, none of these colonies expanded; one colony died in 2002 and the others showed strong evidence of

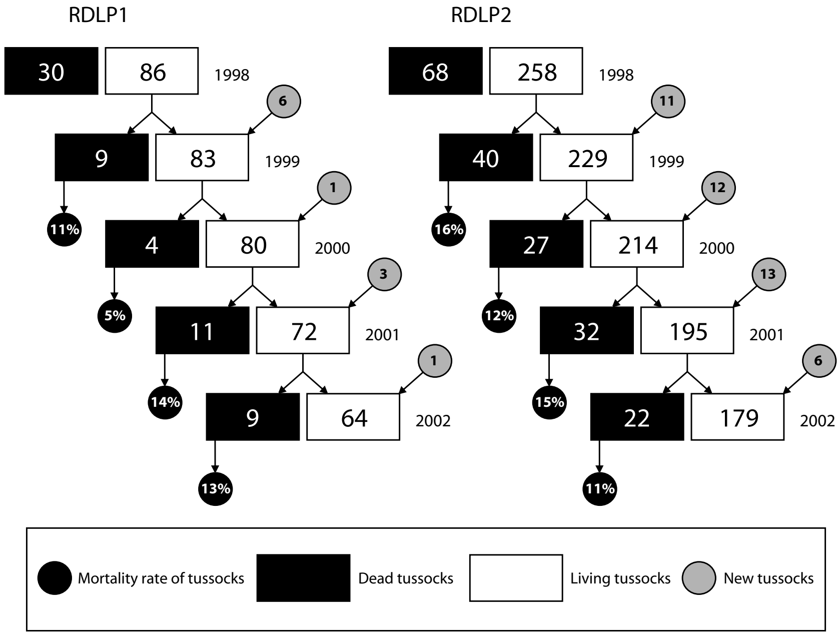

Figure 5. Evolution of the number of cotton-grass (*Eriophorum vaginatum*) tussocks from 1998 to 2002 at RDLP1 and RDLP2 abandoned vacuum-mined sites (10 × 20 m quadrat). For example, of the 86 living tussocks present in 1998 at RDLP1, 9 were dead in 1999. However, since 6 additional tussocks emerged in 1999, the total number of living tussocks was 83 that year. It should be noted that all dead tussocks, except very small ones, remained on site during the monitoring period.

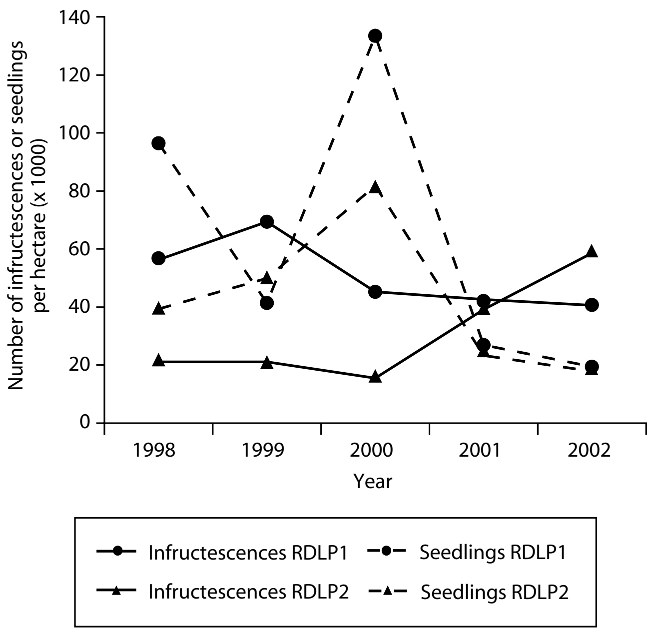

Figure 6. Number of cotton-grass (*Eriophorum vaginatum*) infructescences and seedlings per hectare estimated each year from 1998 to 2002 at RDLP1 and RDLP2 abandoned vacuum-mined sites.

Table 1. Evolution of the cover of main plant species found in two abandoned vacuum-mined sites of the Rivière-du-Loup (RDLP) peatland, southern Quebec, Canada, from 1998 to 2002.

Site	RDLP1	RDLP2
Total number of plant species (vascular plants, mosses, liverworts) in 1998	13	20
Total number of plant species (vascular plants, mosses, liverworts) in 2002	12	20
Increase or decrease (%) of plant cover from 1998 to 2002		
Betula papyrifera Marsh	+104	+5
Dicranella cerviculata (Hedw.) Schimp.	N	−66
Eriophorum angustifolium L.	–	−79
Eriophorum vaginatum L.	−16	−46
Kalmia angustifolia L.	N	+168
Kalmia polifolia Wangenh.	–	+355
Larix laricina (Du Roi) Koch	N	+170
Picea mariana (Mill.) B.S.P.	+142	+31
Polytrichum strictum Brid.	N	+254
Rhododendron canadense (L.) Torr.	N	+112
Rhododendron groenlandicum (Oeder) Kron & Judd	N	+106
Vaccinium angustifolium Ait.	+291	+129

N – plant species with a negligible cover during all the five-year period.

desiccation. At RDLP2, the cover of some plant species (*Dicranella cerviculata* [Hedw.] Schimp., *E. angustifolium*) decreased between 1998 and 2002, while the cover of all ericaceous shrub species and *Polytrichum strictum* Brid. increased (+106% to +355%) during the same period (Table 1, Figure 8). However, it is important to note that few 2 × 2 m subquadrats had ericaceous shrub or moss species with a cover >10%, and none with a cover >25%. The cotton-grass cover decreased from 1998 to 2000 because old tussocks in the lower left corner of the quadrat died out, but then increased in 2001 and especially in 2002 because new tussocks became established in the upper part of the quadrat (Figure 8).

Discussion

The vegetation assemblages of abandoned vacuum-mined bogs are surprisingly dynamic, considering the limited extent of plant cover on those sites even two decades after abandonment. Major plant cover changes were observed at RDLP bog during a five-year period. The decline in the number of cotton-grass tussocks at both sites is especially interesting, considering that cotton-grass usually forms long-lived (>100–200 years) tussocks (Mark et al. 1985). Furthermore, cotton-grass has many characteristics facilitating its establishment and survival in nutrient-poor environments such as mined bogs that have been drained. This plant species can tolerate prolonged drought periods because of its deep root system (Wein 1973). Numerous cotton-grass seeds are produced each year and are easily dispersed by wind (Salonen 1987; Campbell et al. 2003). Seeds germinate at high (23–35 °C) temperatures (Wein and MacLean 1973; Gartner et al. 1986), and during the summer season such temperatures are frequent just above the peat surface of abandoned mined bogs (Matthey 1996; Marcoux 2000). The growth of new leaves is supported almost entirely by nutrient retranslocation from older leaves that are senescing (Jonasson and Chapin 1985). Furthermore, not only this non-mycorrhizal plant can absorb organic nitrogen directly, but it can use it as its sole nitrogen source (Chapin et al. 1993).

It seems that the competitive advantages of cotton-grass are insufficient to counteract the harsh environmental conditions of mined bogs, especially at RDLP1 where the water table level is particularly low. Lavoie et al. (2005) showed that a water table level <30–40 cm below the soil surface and a volumetric peat water content <70% during most of the summer season prevent extensive establishment of cotton-grass on abandoned mined bogs. Furthermore, experimental studies conducted near the study sites showed that frost heaving is particularly active in abandoned mined surfaces of the RDLP bog (Figure 1; Groeneveld 2002). Numerous cotton-grass and tree seedlings, and small cotton-grass tussocks, were found uprooted at RDLP1 and RDLP2. Frost heaving is the most probable cause of uprooting, and this may explain why so few new tussocks established, although numerous seeds were produced and seedlings germinated each year. For instance, the greatest cotton-grass seedling densities in the Alaskan tussock tundra were found where frost heaving was reduced because of the presence of a hepatic mat stabilizing the soil surface (Gartner et al. 1986). Frost heaving and the high peat oxidation rate measured in peatlands of the Bas-Saint-Laurent region also probably explain, in part, the high mortality rate of small trees (Campbell et al. 2002; Groeneveld 2002; Waddington and McNeil

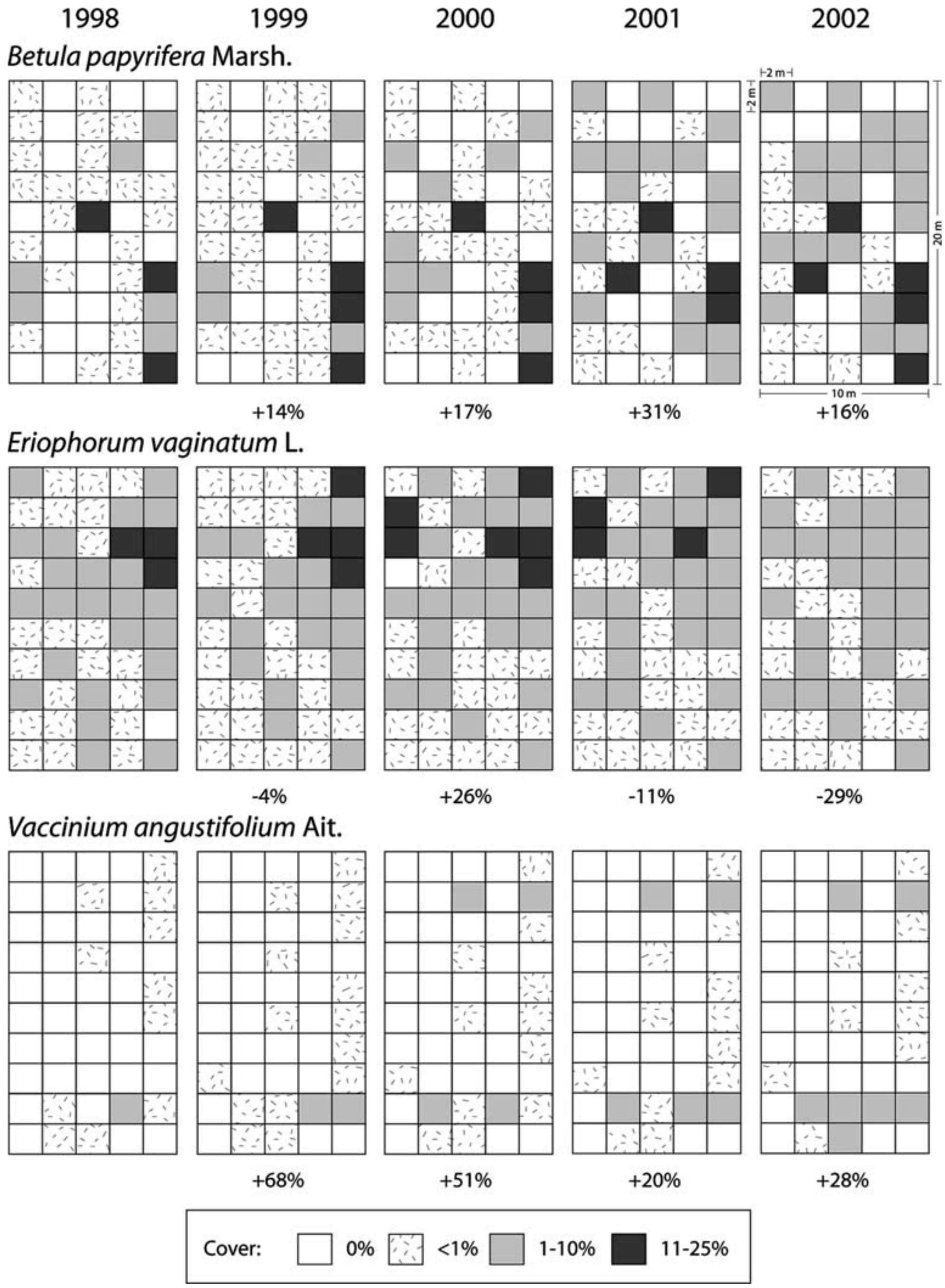

Figure 7. Monitoring of the cover of *Betula papyrifera*, *Eriophorum vaginatum* and *Vaccinium angustifolium* in a 10 × 20 m quadrat installed in the RDLP1 abandoned vacuum-mined site. The quadrat was subdivided into fifty 2 × 2 m subquadrats in which the cover of living individuals was evaluated in June, from 1998 to 2002. Cover classes are indicated in the legend. The species cover was compared year to year by calculating the median value of the cover class occupied by the species in each subquadrat, and totalling all median values from all subquadrats. Percentage change of the species cover, compared to the previous year, is shown for each year from 1999 to 2002.

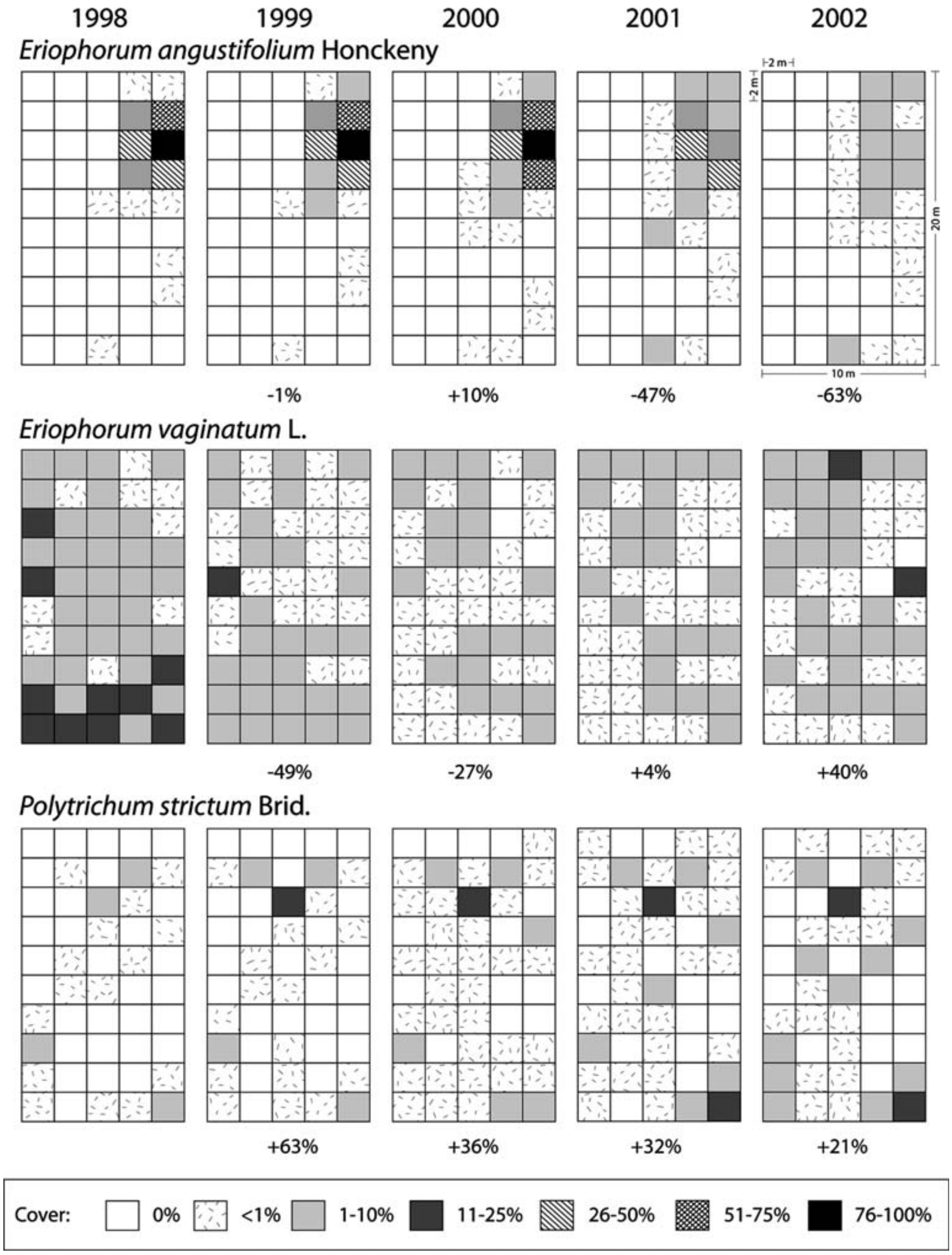

Figure 8. Monitoring of the cover of *Eriophorum angustifolium, Eriophorum vaginatum, Polytrichum strictum, Kalmia angustifolia, Rhododendron groenlandicum* and *Vaccinium angustifolium* in a 10 × 20 m quadrat installed in the RDLP2 abandoned vacuum-mined site. The quadrat was subdivided into fifty 2 × 2 m subquadrats in which the cover of living individuals was evaluated in June, from 1998 to 2002. Cover classes are indicated in the legend. The species cover was compared year to year by calculating the median value of the cover class occupied by the species in each subquadrat, and totalling all median values from all subquadrats. Percentage change of the species cover, compared to the previous year, is shown for each year from 1999 to 2002.

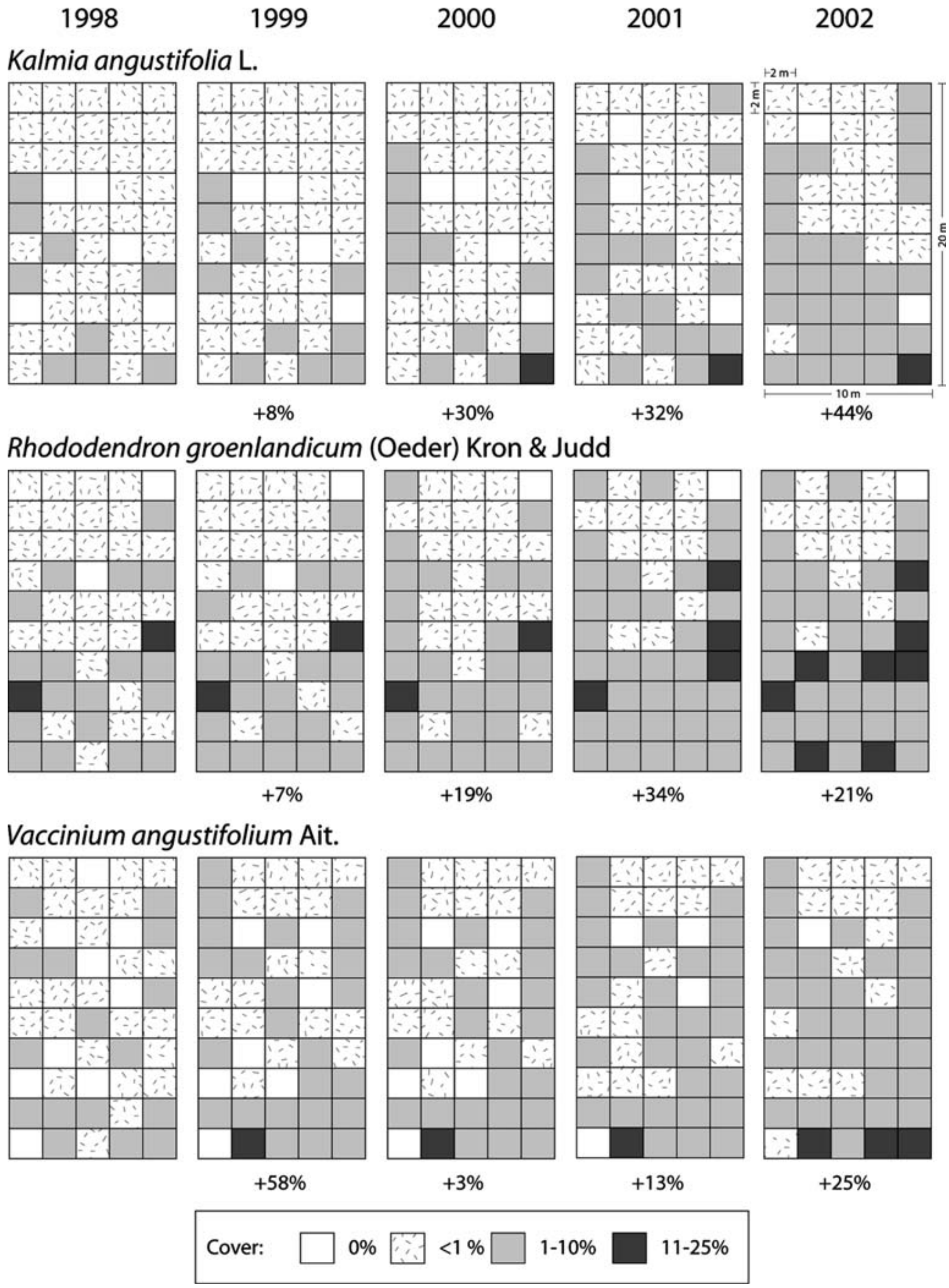

Figure 8. Continued.

2002). Some moss species, such as *Polytrichum strictum*, may contribute by stabilizing the soil surface to facilitate the survival of both vascular plant seedlings and small *Sphagnum* colonies (Grosvernier et al. 1995; Groeneveld and Rochefort 2002; Rochefort et al. 2003). The *P. strictum* cover is increasing at RDLP2, but remains very low even after two decades of abandonment, and probably does little to accelerate the revegetation process.

The other major change noted at both sites was the increase in the ericaceous shrub cover, especially at RDLP2. Similar increases have been noted in other mined sites in southern Québec (Bérubé and Lavoie 2000; Lavoie et al. 2005). This suggests a recovery of the typical shrub layer characterizing most ombrotrophic peatlands of the Bas-Saint-Laurent region (Gauthier and Grandtner 1975). However, it should be noted that the ericaceous shrub cover, although increasing, was still very low even after two decades of abandonment, and it may take several additional decades before a complete shrub cover establishes at RDLP2, and competes with other plant species like cotton-grass.

Conclusions

After two decades of abandonment, the natural revegetation process of the vacuum-mined bogs in the Rivière-du-Loup area is still dynamic, but there is no evidence to suggest that vegetation assemblages will eventually resemble those of undisturbed peatlands. Even the most competitive species (cotton-grass) displayed evidence of decline. It is very unlikely that a moss cover dominated by *Sphagnum* species will re-establish on mined surfaces, although this is an essential condition for the restoration of the main ecological functions of bog ecosystems (Rochefort 2000). This does not mean that natural revegetation processes are useless in vacuum-mined sites: minimal restoration activities, such as blocking drainage ditches, may favour the rapid re-establishment of a dense cotton-grass cover and of many other typical bog plant species (Tuitilla et al. 2000; Chapman et al. 2003; Lavoie et al. 2003, 2005). In Europe, flooding of mined bogs has been used to accelerate the spontaneous re-establishment of vascular plants and of some *Sphagnum* species (Smart et al. 1989; Meade 1992; Farrell and Doyle 2003). Even a slight increase in the water table level seems to improve growth conditions for many bog species, as observed at the RDLP2 site. However, at most mined sites located in North America, flooding cannot be considered as a restoration option, because mining operations are still active near abandoned sites. Alternative restoration procedures, such as the spreading of *Sphagnum* diaspores combined with the use of straw mulch to protect diaspores from desiccation, and of fertilizers to accelerate the growth of companion plants (Price et al. 2003; Rochefort et al. 2003), certainly represent the best management option to rapidly re-establish a plant cover typical of ombrotrophic peatlands.

Acknowledgements

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