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RESTORATION OF A FEN PLANT COMMUNITY AFTER PEAT MINING

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RÉSUMÉ

Cette étude porte sur la restauration des fens dominées par les Cyperaceae après l'exploitation de dépôt de la tourbe. L'hypothèse de cette recherche est que la restauration d'une communauté de plantes typiques des fens est possible par l'application des techniques développées en Amérique du Nord pour les bogs. Nous avons choisi une approche expérimentale qui consiste à manipuler les conditions environnementales et les facteurs biologiques. L'application de paille et la réintroduction de fragments de plantes de tourbières ont un effet positif sur le rétablissement d'un couvert de végétation et d'une diversité d'espèces représentative des fens. Ces résultats suggèrent que la restauration d'une communauté typique de fens est possible avec l'application des techniques élaborées pour la restauration des bogs. Néanmoins, une attention particulière devrait être portée aux conditions chimiques de la tourbe, ainsi qu'aux conditions hydrologiques du site à restaurer, afin de s'assurer que ces conditions sont similaires à celles des fens naturels.

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ABSTRACT

This study was concerned with the restoration of a fen plant community, dominated by *Carices*, on sedge peat surfaces after peat mining. We hypothesized that the restoration of a fen plant community was possible by applying techniques developed in North America for the restoration of bogs. We chose an experimental approach to manipulate the environmental conditions and biological factors. The application of straw and the introduction of donor seed banks from natural fens had a positive effect on the re-establishment of fen plants, in terms of percentage cover and species richness. These results suggest that the restoration of fens is possible with the application of techniques created for restoring bogs. However, chemical and hydrological conditions of the peat at the restoration site require further restoration measures to match conditions observed at natural fens in the study region.

PREFACE

The first chapter is an introduction and literature review to my thesis. The third and fourth chapters are written in the form of an article. I am entirely responsible for the research and writing of these chapters.

The second chapter is in the form of an article and has been prepared for submission to the journal *Applied Vegetation Science*. I am the first author, Dr. Line Rochefort is the second author, and Dr. Jonathan Price is the third author. I am responsible for the conception of the research hypotheses, Dr. Line Rochefort and Dr. Jonathan Price assisted with the projects design. I was responsible for data collection, data analysis, and manuscript preparation, including writing the initial draft, making corrections, and organizing correspondence. Dr. Line Rochefort and Dr. Jonathan Price assisted with data interpretation and provided editorial comments.

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1 Introduction

Peat moss deposits are extracted from *Sphagnum* dominated peatlands across Canada by commercial peat moss producers that sell the peat to the horticultural industry. Peat mined sites are typically abandoned once the high quality peat of the uppermost layers has been extracted. The abandoned sites are characteristic of a bog peatland type with *Sphagnum*-based peat at the surface and ombrotrophic conditions. Restoration of these sites has focused on establishing a *Sphagnum* moss carpet in order to bring a system to the state development similar to pre-disturbance (Rochefort *et al.* 2003). Recently, several peat-mined fields have been extracted to deeper depths. These sites are characteristic of a fen peatland type with sedge-based peat at the surface and minerotrophic conditions (Wind-Mulder *et al.* 1996; Wind-Mulder & Vitt 2000). The restoration of these sites towards a fen peatland type (i.e. an earlier successional stage) has been recommended (Wheeler & Shaw 1995; Wind-Mulder *et al.* 1996) and is largely unstudied.

This literature review is concerned with the restoration of a fen ecosystem on sites with exposed sedge peat and minerotrophic conditions after peat extraction. I first review information pertaining to natural peatlands in order to better understand the development of these systems, as well as to identify the environmental and biological conditions that define the two main types of peatlands – bogs and fens. I then describe the current state of the peat mining industry in Canada, and the environmental and biological conditions that characterize sites disturbed by peat extraction. Thirdly, I attempt to summarize current approaches and techniques for restoring peatlands disturbed by peat mining. I finally establish specific research hypotheses and objectives in light of this knowledge.

Nomenclature

Vascular plants (Gleason & Cronquist 1991); Sphagnopsida (Anderson 1990); other mosses (Anderson *et al.* 1990).

1.1 Natural peatlands

Peat is the partially decomposed remains of plants that form when the rate of production exceeds the rate of decomposition (Clymo 1983). Even though the definition of peatlands varies for different countries (Bridgham *et al.* 1996), the *Canadian Wetland Classification System* defines them as organic wetlands where greater than 40 cm of peat has accumulated (NWWG1997). Peatlands are classified according to properties that reflect their ontogeny and topography, including hydrology, water chemistry, and plant-community composition (Table 1.1). The Canadian Wetland Classification System characterizes the two main classes of peatlands as fens and bogs (NWWG1997) as following.

Table1.1 Key distinguishing features of fens and bogs.

	Fen	Bog
Water source	Precipitation & telluric supplies (geogeneous or minerogenous)	Precipitation (ombrogeneous)
Mineral status	Minerotrophic (rich in minerals)	Ombrotrophic (poor in minerals)
Major nutrient status (N,P,K)	Eutrophic - Oligotrophic (nutrient rich - poor) May be N-limited, P-limited or N- & P-limited, rarely K-limited	Oligotrophic (nutrient-poor) Typically P-limited
pH	Moderately acidic (4.5 – 5.5) to circumneutral (5.5 – 8.0)	Very acidic (3.5 – 4.5)
Peat types	Mixtures of sedges, grasses, herbs, woody species, mosses	Typically *Sphagnum* moss dominated with some sedges, herbs and woody species
Vegetation diversity	Low to high (largely dependent on nutrient status)	Low
Characteristic species	Graminoids, herbs, woody species, and brown mosses,	*Sphagnum* mosses, ericaceous shrubs, and/ or tree species

Peatland development occurs by two main processes – terrestrialisation, where a body of water is gradually in filled with organic remains to become a peatland, and paludification which is the formation of a peatland over dry land by “swamping” or water logging the land (Gore 1983). Peat gradually accumulates, layer by layer, over long periods, preserving the partial remains of flora and fauna (Tallis 1983). Thus, peatlands contain a direct record of the antecedent communities for present communities. Analysis of stratigraphic sequences can be used to determine the successional pathway of a peatlands development (Tallis 1983) (e.g. Figure 1.1).

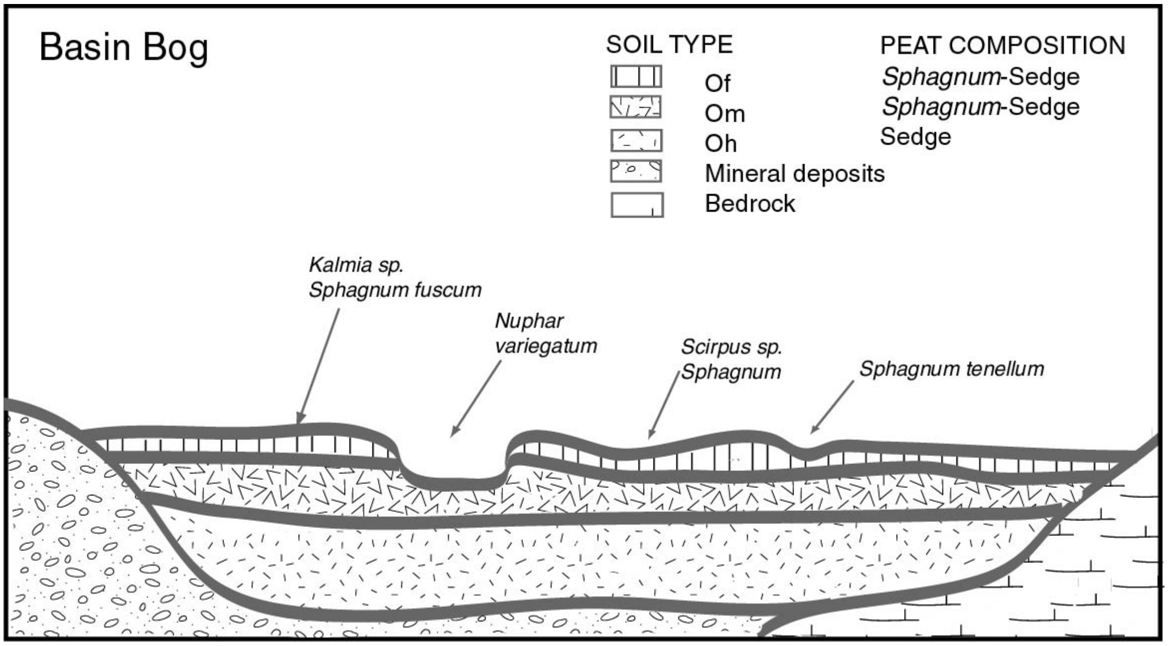

Figure 1.1 A cross section of a peatland showing layers of peat that has accumulated during its formation. Note the sedge peat at the base of the peat deposit and *Sphagnum* - sedge peat near the surface of the present day bog (NWWG1997).

Peatland development does not follow a single pathway. However, most North American peatlands involve an early seral stage with vegetation composed principally of *Carex spp*. forming sedge peat (Tallis 1983) (Figure 1.2). Sedge peat forms where grasses and sedges dominate the flora, tending to occur under minerotrophic fen conditions (Clymo 1983). Under suitable climatic conditions and geomorphic setting, peat may develop with sufficient quantities to impede the drainage of precipitation. The gradual accumulation of peat beyond the influence of minerogenous water, results in ombrotrophic conditions characteristic of a bog, with *Sphagnum* peat. Thus, the hydroseral changes of wetland communities tend to include minerogenous fens as an early stage that proceeds in the forward direction towards an ombrogenous bog (Figure 1.2).

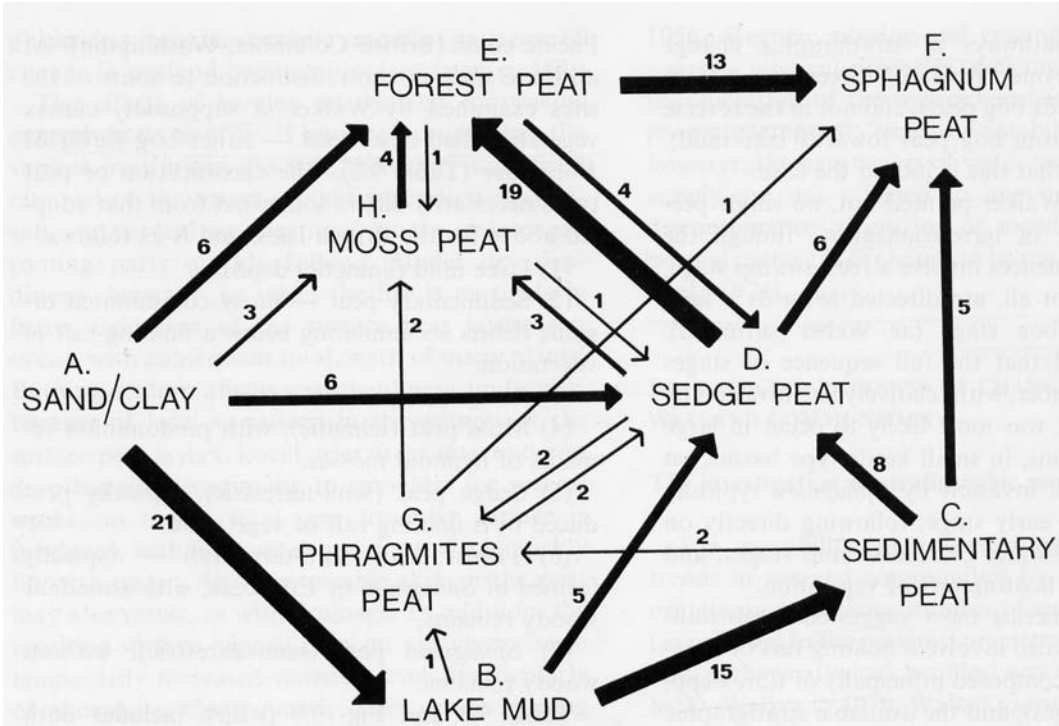

Figure 1.2 Stratigraphic sequences (36) in peat profiles from North America. The arrows connect pairs of superposed strata in published profile descriptions; the number against each arrow gives the number of recorded instances of that particular transition (Tallis 1983).

1.1.1 <u>Fens</u>

Fens are minerogenous peatlands, receiving surface water and groundwater from the surrounding mineral soils in addition to precipitation (Bridgham *et al.* 1996). The vegetation community of fens is highly variable, depending largely on the depth of the water table and the water chemistry (Bridgham *et al.* 1996). Fens are often classified as “poor fens” and “rich fens” based on plant-community composition and water chemistry. Poor fens (also termed “mesotrophic bogs” or “transition fens”) have a pH between 4.5 and 5.5, low mineral status, and vegetation dominated by *Sphagnum* mosses, herbs, and shrubs (Wheeler & Proctor 2000). Indicator moss species of poor fens in Québec include *Sphagnum riparium*, *Drepanocladus* spp. and *Tomentypnum* spp.. Several herbaceous species such as *Calamagrostis canadensis*, *Utricularia* spp., *Carex canescens, C. stricta, C. aquatilis*, *Juncus filiformis*, *Viola macloskeyi*, and *Epilobium leptophyllum* may also be found. Shrubs characteristic of poor fens include *Myrica gale*, *Nemopanthus mucronata*, and *Lonicera villosa*; trees include *Picea mariana*, and *Larix laricina* (Garneau 2001).

In contrast, "rich fens" have a pH greater than 5.5, rich mineral status, and are dominated by graminoids and brown mosses (Charman 2002). Indicator species of intermediate - rich fens in Quebec include several mosses from the Amblystegiaceae family such as *Scorpidium scorpioides, Drepanocladus* spp.*, Campylium* spp. and *Calliergon* spp. Herbaceous plants are typically abundant and diverse including *Carex aquatilis, C. utriculata, C. leptalea, C. lasiocarpa, Eleocharis smallii,* and *Solidago spp*. Shrubs such as *Salix spp.*, *Betula spp*., and *Potentilla fruticosa* may occur, in addition to trees such as *Larix laricina*, and *Thuja occidentalis* (Garneau 2001).

The productivity of vegetation in fens may be limited by nitrogen or phosphorus. The main source of nitrogen available to plants is from microbial fixation of atmospheric nitrogen. This tends to be low in early successional stages due to the absence of previously stored organic nitrogen. Thus, early successional rich fens tend to be nitrogen limited (Verhoeven *et al.* 1996). Fens tend to have high inputs of phosphorus due to water inputs with phosphorus derived from rock weathering. However, in alkaline conditions phosphorus is often precipitated with calcium minerals and may become limiting (Schlesinger 1997). The biodiversity of fen plant communities may be extremely high or low. Biodiversity has been positively correlated with increasing nutrient availability, until a threshold value, beyond which it declines (Bedford *et al.* 1999).

1.1.2 Bogs

The peat surface of a bog is generally raised or level with the surrounding landscape. Consequently, bogs receive water almost solely from precipitation, and are virtually unaffected by mineral water sources (Ingram 1983). Peatlands that are poor in minerals are referred to as “oligotrophic”. Precipitation does not contain dissolved minerals and is mildly acidic, and as a result, bogs have low mineral status and high acidity (Bridgham *et al.* 1996). The surface water pH of bogs is typically between 3.5 and 4.5 (Gorham & Janssens 1992; Zoltai & Vitt 1995). Plant productivity is generally limited by the availability of phosphorus (Verhoeven *et al.* 1996). The short supply of phosphorus for plant growth is not surprising, since bogs receive little or no water runoff from the surrounding land and phosphorus originates from weathering of rocks (Schlesinger 1997).

Sphagnum mosses, the dominant species in most bogs, are considered “ecosystem engineers” (van Breemen 1995). They promote bog development through autogenic processes. Functionally, *Sphagnum* species increase the acidity of peatlands due to the high cation exchange capacity of their live tissues, and their release of organic acids during decomposition. *Sphagnum* peat conducts heat poorly that effectively reduces the growing season for vascular plants. In addition, Sphagna preferentially sequester nutrients and transport them to the apical parts of the plant. The harsh environmental conditions created by *Sphagnum* species are unsuitable for many other species. Reduced competition with other species in turn stimulates positive feedback to the growth of *Sphagnum* (van Breemen 1995). Plants growing in bogs have unique adaptations to withstand the low nutrient environments, including mechanisms such as evergreenness, schlerophylly and defensive compounds to reduce grazing losses, nutrient translocation before leaf abscission, high nutrient-use efficiency and high shoot: root ratios (Bridgham *et al.* 1996).

Bogs are easily identified by their plant community composition. Sphagnum mosses, ericaceous shrubs and / or conifers dominate bogs (Bridgham et al. 1996). Several *Sphagnum* species are common to bogs in Québec including *Sphagnum angustifolium, S. capillifolium, S. fuscum, S. magellanicum*, and *S. rubellum*. Other mosses that occupy bogs include *Dicranum spp., Pohlia nutans*, and *Polytrichum strictum*. Ericaceous shrubs indicative of bogs are *Vaccinium spp., Ledum groenlandicum, Kalmia angustifolium, K. polifolia, Andromeda glaucophylla*, and *Chamaedaphne calyculata*. Trees such as *Picea mariana* and *Larix laricina* may be present. Herbaceous plants are not typically prominent, but may be present including *Eriophorum spp*. and *Carices*, and insectivorous plants such as *Drosera* spp., and *Sarracenia purpurea* (Garneau 2001).

1.2 Impact of industrial peat extraction

Peat mining affects approximately 17 000 hectares of Canada's 113 million hectares of peatlands (Daigle & Gautreau-Daigle 2001). Over 1 million tonnes of peat are extracted annually, worth approximately 170 million dollars. Canada sells peat for horticultural purposes (i.e. peat moss for use in gardens and greenhouses) and ranks second internationally in the global extraction of horticultural peat, after Germany. The horticultural market demands weakly decomposed peat comprised mainly of *Sphagnum* mosses, which can be found underlying bog ecosystems. Peat mining operations in Canada rely almost exclusively on modern milling techniques. This process involves vacuuming off dry peat from the surface of bare peat fields. Because only thin layers of peat are removed at one time, large tracts of land must be worked to ensure that operations are profitable. In addition, sites must contain deep *Sphagnum*-based peat deposits, and occur in regions with appropriate climate and transport facilities. Peat milling techniques requires long periods of consecutive days without rainfall to dry sufficiently the peat surface for collection. Within Canada, peat-mining operations are concentrated in the St. Lawrence lowlands of Québec and coastal regions of New Brunswick. They also occur sporadically in the provinces of Nova Scotia, Manitoba and Alberta.

Preparing a site for peat mining involves the installation of deep drainage ditches around the perimeter, and denudation of surface vegetation (Daigle & Gautreau-Daigle 2001). Shallower drainage ditches are then created to drain water from surface peat into the deeper perimeter ditches. Typically, these ditches are formed parallel to one another and spaced 30 m apart. The ditches function to reduce the water content of the peat, enabling it to bear the weight of heavy machinery. The surface peat is harrowed or "milled" with large milling machines. This acts to break the capillary flow of water and enhances the drying process. Once the surface peat layer (15-50 mm) is sufficiently dried (requiring one to three days) it is collected with large vacuum machines. The peat is then transported to a processing plant where it is screened and packaged into compressed bales.

Peat deposits are typically abandoned after several decades when the weakly decomposed layers of *Sphagnum* peat have been exhausted. Recently, several mined peat fields have been extracted to deeper depths, until sedge peat is exposed. The water and peat chemistry of such sites has been observed to be similar to poor or moderate-rich fens rather than bogs (Wind-Mulder *et al.* 1996; Wind-Mulder & Vitt 2000). Approximately

2300 hectares of mined peat fields have been abandoned and are available for restoration (Dr. Line Rochefort, personal communication).

1.2.1 Environmental conditions

The environmental conditions of an abandoned mined peat field are extremely harsh for plant re-establishment (Salonen 1987, 1992). The physical and chemical properties of peat deteriorate due to the effects of long-term drainage and compression from heavy machinery (Okruszko 1995; Price *et al.* 2003). Peat extraction removes the surface layer of peat, which is biologically active and more water-permeable, referred to as the "acrotelm". The subsurface fossilized layers, referred to as the "catotelm", become exposed (Ingram 1978). The catotelm has a higher bulk density and a lower water storage capacity compared to the acrotelm (Price *et al.* 2003). Removal of the acrotelm results in a deeper and more variable water table throughout the growing season, and decreased soil moisture and increased soil-water tension (Price 1997; Price & Schlotzhauer 1999; Price & Whitehead 2001).

Peat mining also changes the chemical properties of peat. Drying induces biochemical oxidation, mineralization, and release of hydrogen ions and nutrients (Wheeler & Shaw 1995). Mineralization, is the transformation of nutrients from organic (plant-unavailable forms) to inorganic (plant available forms) by soil microbes (Grootjans & Van Diggelen 1995). Mineralization processes are accelerated by drying of the peat and nutrients, particularly nitrate, become available in large quantities, even excessive amounts. The concentration of solutes is higher and more variable in peat mined surfaces compared to undamaged bogs (de Mars *et al.* 1996; Wind-Mulder & Vitt 2000). The increased fluctuations of solute concentrations are largely due to increased fluctuations of the hydrological regime, which have significant control over the water and soil chemistry. The permanence of these effects is largely unknown (Wheeler & Shaw 1995).

1.2.2 Biological conditions

Even after several decades post-abandonment, little spontaneous vegetation may occur on mined peat fields (Lavoie *et al.* 2003). The restoration of a fen plant community on peat mined sites is likely to be constrained by the availability of suitable diaspores. The residual peat is devoid of plants and viable seed banks (Salonen 1987) and natural areas surrounding peat mined sites are typically bogs with few to no herbaceous species present (Poulin *et al.* 1999; Campbell *et al.* 2003). Peat mined sites with shallow peat deposits tend to be spontaneously recolonised with non-peatland species, particularly annual weeds (Salonen 1990; Rowlands 2001). Introducing suitable species to mined peat fields may be necessary to promote the development of a fen plant community (Wheeler & Shaw 1995).

The development of a plant community is determined by the availability of viable seeds or other diaspores at a site, as well as appropriate environmental conditions for germination and subsequent growth (Bakker & Berendse 1999; Mitsch & Gosselink 1993). In an elegant study by Salonen (1987), the relationship between seed rain and plant establishment on peat-mined sites was examined. He found no relationship between the numbers of seeds dispersed and individual plants of the same species in pioneer populations. This indicates that unfavourable site conditions may be a crucial factor limiting plant establishment on mined peat sites.

1.3 Restoring peatlands

In North America, restoration of peatlands has tended to focus on the restoration of a bog ecosystem because the majority of mined peat sites in need of restoration have acidic, nutrient-poor conditions suitable for the establishment of *Sphagnum* mosses (Rochefort *et al.* 2003). Restoration efforts have focused on establishing a *Sphagnum* carpet, as *Sphagnum* species are seen as the great engineers of peatland formation and are suitable for acidic residual conditions (Rochefort *et al.* 2003).

Recently, a few peat-mined sites in eastern Canada have been extracted to deeper depths, exposing more basic, nutrient-rich peat. Peat mining extracts layers of peat that have accumulated over time, so that the surface of the original ecosystem is cut back to an earlier stage in development. The presence of sedge peat and minerotrophic conditions at the exposed surface indicates that a fen ecosystem historically occurred at the site. This project aims to restore a fen plant community (i.e. a historical plant community) on abandoned sedge peat with minerotrophic conditions (i.e. a site degraded to its historical conditions), as has been proposed by others (Wheeler & Shaw 1995; Wind-Mulder *et al.* 1996). However, fen restoration research is still in its infancy and few projects have attempted to do so (Chalmer 2002).

There are two major requirements of peatland restoration: (i) the effective rewetting of a peat surface, and (ii) the establishment of suitable recolonist species (Wheeler & Shaw 1995). Below I will summarize the techniques and impacts of rewetting and vegetation reintroduction employed in restoring peatlands.

1.3.1 Rewetting

Effective rewetting has been identified as one of the most important prerequisites for short-term regeneration of peat mined sites (Sliva & Pfadenhauer 1999; Rochefort 2000). Rewetting refers to all measures that result in wet conditions of the surface peat (Wheeler & Shaw 1995). The main requirements are to create a high and stable water table with the surface saturated throughout the year, appropriate microclimate conditions, and for the water to be of suitable quality. The measures needed to achieve these conditions are extremely variable from one site to another. Possible measures include recontouring and reshaping the site, ditch blocking and/ or filling, sealing the edges of the site, and pumping additional water from a reservoir (Charman 2002). Blocking or filling in the drainage ditches is needed to raise the ground water level, and to help reduce runoff during dry and moderately wet conditions (Price *et al.* 2003). In addition, bunds or small embankments may be built. This helps store surface water, typically precipitation, more evenly on the site for longer periods (LaRose *et al.* 1997; Money & Wheeler 1999; Price *et al.* 2003).

Approaches to peatland restoration vary and rewetting strategies are adjusted accordingly. In northeastern Germany restoration attempts to simulate terrestrialization processes towards the natural development of a peatland. Large bunds are created to impound large quantities of water and create flooded conditions (Joachim Blankenburg, personal communication). Such conditions favour the natural recolonization of aquatic *Sphagnum* species, such as *Sphagnum cuspidatum* (Chirino and Rochefort, unpublished data) and *Sphagnum fallax*. In contrast, restoration in North America attempts to simulate paludification processes towards the development of a peatland. This approach demands that rewetting techniques create soaked surface conditions. Such conditions are favourable for terrestrial *Sphagnum* species, such as *S. fuscum* and *S. capillifolium* (Chirino and Rochefort, unpublished data; Campeau, Rochefort and Price, unpublished data). In North America bunds that withhold 'excess' water at a site are often associated with excessive water fluctuation and are considered problematic (Price *et al.* 2003).

These strategies have been developed primarily to rewet *Sphagnum* peat. Rewetting strategies for fen restoration need to consider the input of minerogenous water (Wheeler & Shaw 1995). The minerogenous supply that characterizes fens varies in its chemical quality and seasonality, and it is of fundamental importance to their ecology (Ingram 1983). Although the presence of fen peat indicates that minerotrophic water was historically supplied to the site, the sources of minerogenous water may no longer be available or may no longer exist. Potential sources of minerotrophic water vary (e.g.

springs, river inundation, lakes) and may be difficult to identify today due to changes to the landscape (Wheeler & Shaw 1995). Still, rewetting the peat with techniques used for bog restoration may be sufficient for fen restoration (e.g. Cooper *et al.* 1998). However, in some cases additional measures may be necessary to ensure that minerogenous water flows through the peat surface (Grootjans & Van Diggelen 1995; Charman 2002).

The microclimatic conditions at mined peat sites are harsh due to an absence of vegetative 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; Price *et al.* 2003). Mulches have increased the establishment of herbaceous plants (Roth *et al.* 1999; Sliva &

Pfadenhauer 1999), and mosses, particularly *Sphagnum* species (Quinty & Rochefort 1997; Rochefort *et al.* 1997) on abandoned mined peat surfaces.

1.3.2 Re-establishing vegetation

Reintroducting vegetation is considered necessary where a landscape is fragmented to the extent that seed dispersal from a source sites can no longer be transferred to the restoration site (Middleton 1999a). Campbell *et al.* (2003) found the immigration potential of herbaceous vegetation to be low at peat mined sites, and their reintroduction was recommended. In contrast, the immigration potential of several trees, shrubs and mosses was estimated to be moderate to high. Restoration of these species should focus on creating microenvironmental conditions suitable for their establishment. Site preparation may be important to ensure that environmental conditions meet the biological requirements of the target species at all stages of maturation for successful establishment (Whisenant 1999). Similarly, it is important to select target plants that match the environmental conditions at the restoration site (Whisenant 1999).

The establishment of sedges, a dominant plant in many fens, from seed is considered very difficult (Budelsky & Galatowitsch 1999; Sliva & Pfadenhauer 1999; van der Valk *et al.* 1999). Reintroduction attempts with *Carices* from seed may fail because of a poor seed set in source populations, and low seed viability (Galatowitsch & van der Valk 1994; van der Valk *et al.* 1999). Low germination rates have been observed in growth chamber germination experiments, whereas field germination experiments with the same seed population have succeeded (Patzelt *et al.* 2001). This indicates that *Carices* may have complex dormancy cycles that are not easily broken with standard seed treatment techniques. Occasionally high germination rates have been observed with *Carex* species introduced by seed sowing. Successful germination was attributed to the creation of appropriate hydrological conditions (Roth *et al.* 1999). In another study (Bohnen *et al.* 2002), twenty *Carex* species were introduced to a wet meadow as seeds and as mature plants. Establishment levels were low with both techniques. However, several of the seeds dispersed via water to higher zones, where they successfully established. This led the authors to recommend seeding as a superior method.

Species that do not establish well from seed are almost always transplanted (Middleton 1999a). Plants tend to be more tolerant of extreme environmental conditions as mature individuals (Middleton 1999b). Transplantation of ramets, rhizomes, juvenile or mature plants has been an effective technique for establishing several fen species, including *Carices* (e.g. Sliva & Pfadenhauer 1999; Yetka & Galatowitsch 1999; Budelsky & Galatowitsch 2000; Wild *et al.* 2001; Isselstein *et al.* 2002). A field experiment conducted by Roth *et al.* (1999) observed higher establishment rates for fen species introduced as transplanted juveniles and mature plants than as seeds. However, failures have also occurred with transplanted fen plants, which were attributed to acidified site conditions that did not match the biological requirements of the species (van Duren *et al.* 1998).

Another method of reintroducing plants is by importing substrate and its seed bank from a nearby donor wetland community (Middleton 1999a). Donor seed bank is the surface layer and rooting zone of a plant community, and contains a variety of species and types of diaspores including seeds, ramets, rhizomes, stolons, and diaspores. This variety of diaspores 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. The inclusion of the substrate with the seed bank 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 (Cooke & Lefor 1998; Turner & Friese 1998). Another advantage of this method is that donor seed bank collected in the spring has undergone the natural dormancy cycle. This is particularly important for *Carex* species, which have complex dormancy cycles and species-specific germination traits (Baskin *et al.* 1996; Schultz 1998; Patzelt *et al.* 2001). Donor seed bank has proven to be a successful restoration technique for bogs (Rochefort *et al.* 2003), and marshes (Brown & Bedford 1997; Stauffer & Brooks 1997).

1.4 Restoration monitoring and evaluation

The clear definition of goals for restoration projects is necessary. Furthermore, the periodic assessment of restoration goals is necessary to improve the predictability of restoration procedures, and ultimately to progress the science of restoration ecology (Zedler 2000). A reference ecosystem is recommended to define restoration goals, determine the restoration potential of sites, and evaluate the success of restoration efforts (White & Walker 1997). Ideally, a reference ecosystem is built from multiple types of data, collected from a variety of reference sites, to account for ecosystem variability (White & Walker 1997). This information can be used to define precise objectives of the restoration project, and later as a reference point to evaluate its success (SER 2002).

There are three strategies for conducting an evaluation of a restoration project: direct comparison, attribute analysis, and trajectory analysis (SER 2002). The direct comparison analysis uses a carefully selected suite of ecosystem traits, including abiotic and biotic parameters, to measure and compare the reference and restoration sites. Attribute analysis assesses whether a restoration site is recovered by examining attributes of restored ecosystems defined by the Society of Ecological Restoration (2002). Trajectory analysis uses data collected periodically from the restoration site to plot the trajectory of the site compared to its intended trajectory towards the reference ecosystem.

1.5 Research Aims and Objectives

I sought to develop a comprehensive study on the restoration of a fen plant community on sedge peat exposed by peat mining in eastern Canada. Firstly, I tested the hypothesis that techniques used in North America for restoring bog vegetation on *Sphagnum* peat surfaces could be applied to restoring fen vegetation on sedge peat surfaces. An experimental and descriptive approach was used to determine the effect of different vegetation treatments, mulch treatments, and environmental conditions, on establishing a fen plant community. Secondly, I sought to define conditions that could maximize the success of introducing fen species with donor seed bank techniques. Towards this end, I set up experiments in the green house and tested how manipulating the hydrological regime and the source of donor seed banks affected seedling emergence. Finally, I aimed to define a reference ecosystem as a goal for fen restoration in the study region, and as a point of reference to evaluate fen restoration success. A composite reference ecosystem was created from inventories of several natural fens in the study region. This information was used as a point of comparison to evaluate the progress of the restored plant community. Different strategies for evaluating restoration were used, including direct comparisons and trajectory analysis.

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2 Experimental restoration of a fen plant community after peat mining

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Abstract

The aim of this project was to restore a minerotrophic peat surface abandoned after peat mining with a fen plant community. A descriptive and experimental research approach was used to determine environmental and biological factors favouring fen restoration. The effectiveness of introducing fen plants with the application of donor seed bank was tested. The donor seed bank, containing seeds, rhizomes, moss fragments, and other plant diaspores, was collected from two different types of natural fens. A straw mulch treatment was applied to test its effects on fen plant establishment and richness. 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 revegetation. All experimental treatments were tested within a factorial split-plot design. Applying donor seed bank from natural fens was found to increase significantly the fen plant cover and richness after two growing seasons. Straw mulch proved to significantly increase fen plant richness. The intermediate terrace level (40 cm) had the highest fen plant establishment. The low terrace level (15 cm) was richer in base cations compared to the reference sites, while the high terrace level (56 cm) was too dry and nitrate rich, perhaps explaining the lower success of plant establishment.

Nomenclature

Vascular plants (Gleason & Cronquist 1991); Sphagnopsida (Anderson 1990); other mosses (Anderson *et al.* 1990).

2.1 Introduction

Restoration ecology research on mined peatlands in North America has focused on the rehabilitation of *Sphagnum*-dominated peatlands because most of the abandoned sites have *Sphagnum*-based peat surfaces (Rochefort *et al.* 2003). Specifically, these procedures include the introduction of donor seed bank material from natural bogs, rewetting the site by blocking drainage canals and creating bunds, applying phosphorus fertilizer, and applying straw mulch to improve the microclimate of the peat surface (Quinty & Rochefort 2003; Rochefort 2001; Rochefort *et al.* 2003). Occasionally, peat-mining operations in Canada cease with a peat surface that is comprised mainly of sedge fragments, as is characteristic of a fen wetland type. The water and peat chemistry of these sites is similar to poor or moderate-rich fens rather than bogs. In such cases, the restoration of abandoned minerotrophic peat towards a fen ecosystem has been recommended (Wind-Mulder *et al.* 1996; Wind-Mulder & Vitt 2000), yet is largely unstudied in North America. Fen species have developed on bogs after deep peat mining in Europe, often in minerotrophic seepage areas (Grootjans & van Diggelen 1995). However, these studies have been descriptions of spontaneous fen development rather than restoration attempts. Charman (2002) commented how it is surprising that there has not been more attention given to fen restoration in the past, particularly in Western Europe where the destruction of fen habitats is at least as severe as ombrotrophic bogs. We sought to test the effectiveness of applying bog restoration procedures to restoring a fen plant community on abandoned minerotrophic peat.

The availability of viable seeds or other diaspores at a site determines the initial development of a plant community (Bakker & Berendse 1999; Campbell *et al.* 2003;

Mitsch & Gosselink 1993). The spontaneous colonization of fen plants on mined peat sites is constrained by a lack of suitable diaspores. The residual peat is devoid of plants and a viable seed bank (Salonen 1987), and natural areas surrounding mined peat sites in North America are typically bogs with few or no fen species present (Poulin *et al.* 1999). Previous fen restoration studies have reintroduced fen species by sowing seeds, or transplanting seedlings, rhizomes or plant cuttings (van Duren *et al.* 1998; Roth *et al.* 1999; Cooper & MacDonald 2002). Another method of introducing plants is by importing substrate and its seed bank from a nearby donor wetland community (Mitsch & Gosselink 1993). The application of donor seed bank has proven to be a successful plant introduction technique for bog restoration (Rochefort *et al.* 2003), and marsh restoration (Brown & Bedford 1997; Stauffer & Brooks 1997).

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 reestablishment (Salonen 1987, 1992; Campbell *et al.* 2002). The physical and chemical properties of peat deteriorate due to the effects of long-term drainage and compression from peat mining operations (Okruszko 1995; Price *et al.* 2003). Effective rewetting has been identified as one of the most important prerequisite for 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). In that respect, 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 minerotrophic peat surface abandoned after peat mining with a plant community dominated by fen species. 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 seed bank from natural fens would increase the cover and richness of fen species compared to control plots. Secondly, the usefulness of straw mulch in promoting the establishment of fen plants was tested. We hypothesised that the application of straw mulch would increase fen species cover and biodiversity compared to control plots without straw mulch. Thirdly, the creation of terraces of different peat depths was used to vary the chemical and hydrological conditions. 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.

2.2 Materials and methods

2.2.1 Site Description

2.2.1.1 Restoration site

The restoration site is part of the Rivière-du-Loup peatland, located approximately 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, alt. 100 m). It is classified as a low boreal peatland (NWWG1988), 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 1993).

The restoration site included two adjacent fields (30 m x 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. The peat was composed of matted sedges interspersed with coniferous wood. Preliminary chemical analyses indicated that the peat was characteristic of a minerotrophic fen with an average pH value = 5.9. The underlying mineral soil was primarily clay with deposits of sand, gravel, and occasional boulders.

2.2.1.2 Donor sites

Field reconnaissance to locate donor sites revealed that there were few natural fens nearby the restoration site. The lack of natural fens in the region partly reflects the gentle topography of the Lower St. Lawrence floodplain and the long period since deglaciation. Paleoecological 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, approximately 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 vegetation communities and environmental site conditions.

The first donor site is a basin fen (NWWG1997) dominated by *Sphagnum* species (hereafter referred to as *Sphagnum* fen). It is a small fen 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 x 25 m) was positioned in the centre of the peatland where the peat depth averaged 86 cm. The chemistry of the water indicate that it is a poor fen with an average pH = 5.5 (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 (NWWG1997), dominated by *Calamagrostis canadensis* (hereafter referred to as *Calamagrostis* fen). It is a small fen 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 peat depth averages 85 cm in the reference landscape unit (25 m x 25 m) in the centre of the fen. The fen is a transitional or moderate fen with a pH = 5.8 (Zoltai & Vitt 1995). Other plants that dominate the site are *Warnstorfia exannulata, Carex utriculata, Scirpus cyperinus, Utricularia minor,* and *Calla palustris*.

2.2.2 Experimental design

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

Terrace levels were treated as main plots and were divided into three blocks to determine effects within the site. The vegetation and straw treatments were treated as subplots and were randomly assigned within the blocks (Appendix A).

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. Each terrace was 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 approximately 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 microtopgraphy 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 by 5 m) were established on the terraces and were separated by a 1 m buffer. The vegetation treatments were (1) donor seed bank from the *Sphagnum* fen, (2) donor seed bank from the *Calamagrostis* fen, and (3) a control, without donor seed bank applied. The donor seed bank was collected from eighteen (1.25 m x 1.25 m) random quadrats located within the donor area (25 m x 25 m). The ratio of donor seed bank 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. Care was taken to spread the donor material evenly between all plots. The mulch treatments were (1) straw, and (2) a control without straw. The straw was applied with a density of 1500 kg/ha 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 May 7-11, 2001.

2.2.3 Site monitoring

2.2.3.1 Vegetation surveys

Percent cover of the vegetation at the restoration site was sampled from October 10-14, 2001 and from August 8-13, 2002. Ten quadrats (30 cm x 30 cm) in each experimental plot were systematically 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 sampled on August 13, 2002. Three transects were randomly placed within the donor area, along which ten quadrats (30 cm x 30 cm) were sampled systematically. The quadrats were sampled for the percent cover (visually estimated to the nearest percent) of each plant species present, and a species list was compiled including any additional species that were noted within the donor area.

A mined peat field that had been abandoned five years previously was located nearby the restoration site (approximately 20 m away on the nearest edge). This field was surveyed for vegetation to determine which plants had the potential to spontaneously colonize the restoration site. 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, 1m 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 sampling rod of infinite height.

2.2.3.2 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 1993).

The water table depth and soil water pressure (-5 cm depth) were measured following the methodology of Price *et al.* (2002). Three wells and 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 samples for chemical analyses were taken 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 the donor fen sites; however, the dry conditions of the surface peat prevented their collection at the restoration site where peat samples were collected instead. Water chemistry was then obtained by adding distilled water to the peat sampled, and extracting the solution with a filter and a vacuum apparatus. All samples were taken to laboratory immediately for analysis or stored in a refrigerator at 4 C until they could be analysed. 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, PO4-P, Fe, Cu, Mn, and Zn (Golterman *et al.* 1978). Peat samples were similarly analysed for the total concentrations these elements after standard dry ashing procedures at 500° C (ex. (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).

2.2.4 Data analyses

The plant species at the restoration site after the first and second growing season were listed by percent cover. Potential sources for the plant species at the restoration site were identified as introduced via donor seed bank, spontaneously recolonized from neighbouring sites, or introduced via the straw mulch. These assessments were based on the plant surveys at the donor sites, the fields neighbouring the restoration site, and knowledge of common agricultural species in the area.

Fen plant cover and fen plant richness (total number of fen species) were averaged for each plot at the restoration site. Species were regarded as a fen species if they were found in the donor site surveys. Due to the vegetative (i.e. non-random and competitive) growth of two non-target species, *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 analysis of simple main effects (Winer *et al.* 1991). All statistical operations were performed with SAS software systems, version 4.0 (SAS Institute 1988).

2.3 Results

2.3.1 Restored vegetation

The total species cover at the restoration site doubled during the course of the second year from 12 to 35 %. The abundance of fen plants within the community increased from 5 % in the first year to 20 % in the second year, and thus represented the majority of plants in terms of both percent cover and richness (Table 2.1). The plant community was composed primarily of forbs and graminoids. There was a small component of woody plants, while bryophytes were largely absent. There was a general decline in the richness of the plant community, including fen species, from the first to second year (Table 2.1). Trace amounts of bryophyte species observed in several plots in the first year were no longer present in the second year, contributing to the decline of plant richness.

Table 2.1 Mean abundance (percent cover) of all plant species at the restoration site after the first (2001) and second (2002) growing seasons. The species are categorized (x) as fen, spontaneous, and straw, based on surveys of the natural fen donor sites, a field neighbouring the restoration site, as well as knowledge of common agricultural species, respectively.

	% cover		Source		
Species	2001	2002	Fen	Spontaneous	Straw
Agrostis hyemalis	0.5	5.3	x	x	-
Equisetum arvense	0.8	5.0	-	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 effuses	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 sp.	0.2	0.6	-	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	12.1	34.8			
Fen cover	4.7	20.3			
Total richness	20.5	18.1			
Fen richness	12.3	10.6			

Several fen species established at the restoration site from the donor fen seed bank, 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 introduced via the donor seed bank and were also present at the field neighbouring the restoration site, including *Agrostis hyemalis, Scirpus cyperinus, Juncus effusus,* and *Lycopus uniflorus* (Table 2.1). *Equisetum arvense, Tussilago farfara*, and *Euthamia graminifolia* were abundant at the restoration site and probably naturally dispersed to the site from the local seed rain (i.e. they were present in the neighbouring fields). 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 abundant.

2.3.1.1 Donor seed bank treatments

Donor seed bank treatments (from *Sphagnum* fen and *Calamagrostis* fen) increased the abundance of fen species after the first and second growing seasons compared to plots without donor seed bank (Figure 2.1). During the first year, there was an interaction between the donor seed bank and straw mulch treatments (Table 2.2). The combination of *Sphagnum* donor seed bank and straw mulch treatments significantly increased the abundance of fen species cover and produced the highest total fen species cover of all experimental treatments (9 ± 1 %) (Figure 2.1b). Several herbaceous species proliferated with the combined treatments of *Sphagnum* fen seed bank and straw mulch including *Viola macloskeyi, Lycopus uniflorus,* and *Galium trifidum*. After the second growing season plots treated with *Calamagrostis* fen seed bank (31 ± 5 %) tended to have higher fen cover than *Sphagnum* fen seed bank (22 ± 3 %), although there were no significant differences between donor seed bank type.

Regarding fen plant richness, after the first growing season it was significantly highest where *Sphagnum* fen seed bank (18 ± 1 taxa) had been applied, intermediate with the application of *Calamagrostis* fen seed bank (13 ± 1 taxa) and lowest without the application of donor seed bank (6 ± 1 taxa) (Figure 2.2a). 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 seed bank applied (*Sphagnum* fen = 13 ± 0.9; *Calamagrostis* fen = 12 ± 1 taxa). Nevertheless, the application of donor seed bank increased the fen plant richness compared to the control (7 ± 1 taxa) (Figure 2.2b).

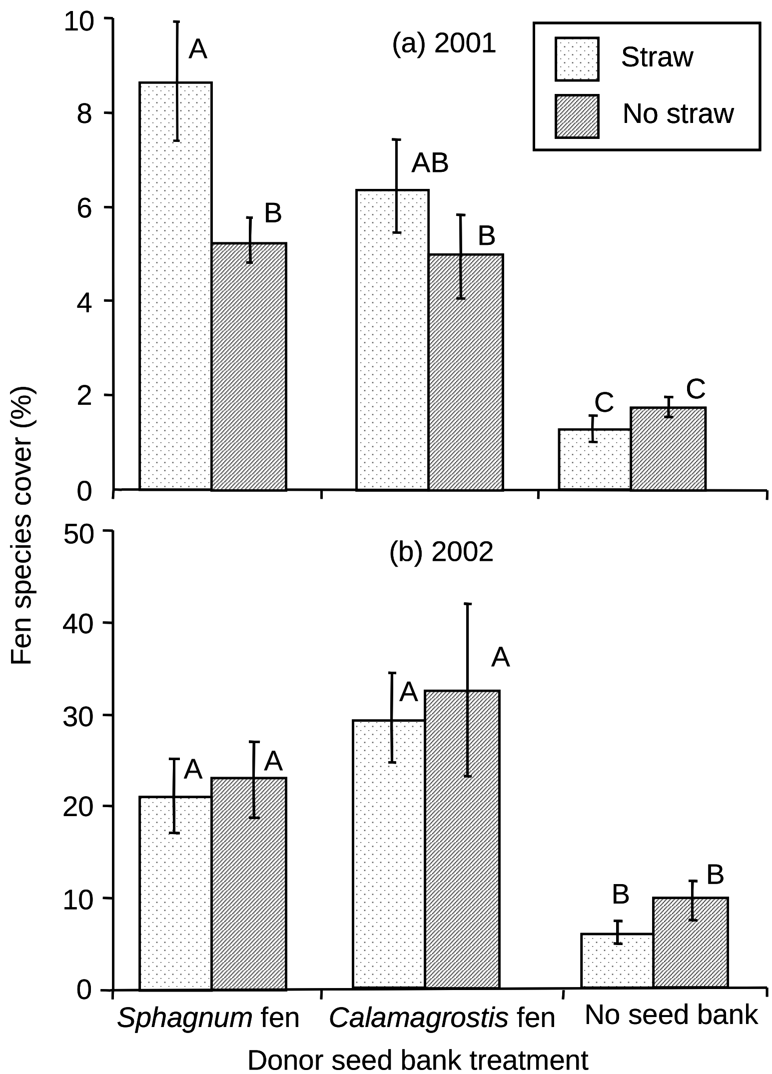

Figure 2.1 Effect of donor seed bank and straw mulch treatments on fen species cover (%) after one growing season (a), and the second growing season (b). Error bars show SE. Different letters within graphs represent significant differences between treatments identified by the splice function for significant two-way interactions (a), and Tukey tests for within treatments without significant interactions (b).

Table 2.2 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 after the second growing season. F-ratios are followed by P-values in parentheses. Significant P-values ($P < 0.05$) are indicated in bold type.

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

2.3.1.2 Straw mulch

The application of straw mulch did not improve the establishment of fen plant cover after two growing seasons. Only during the first year did straw mulch statistically improve the cover of fen plants in combination with *Sphagnum* fen seed bank (9 ± 1 %), compared to *Sphagnum* donor seed bank plots without straw mulch (5 ± 1 %) (Figure 2.1a). 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 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) (Figure 2.2b).

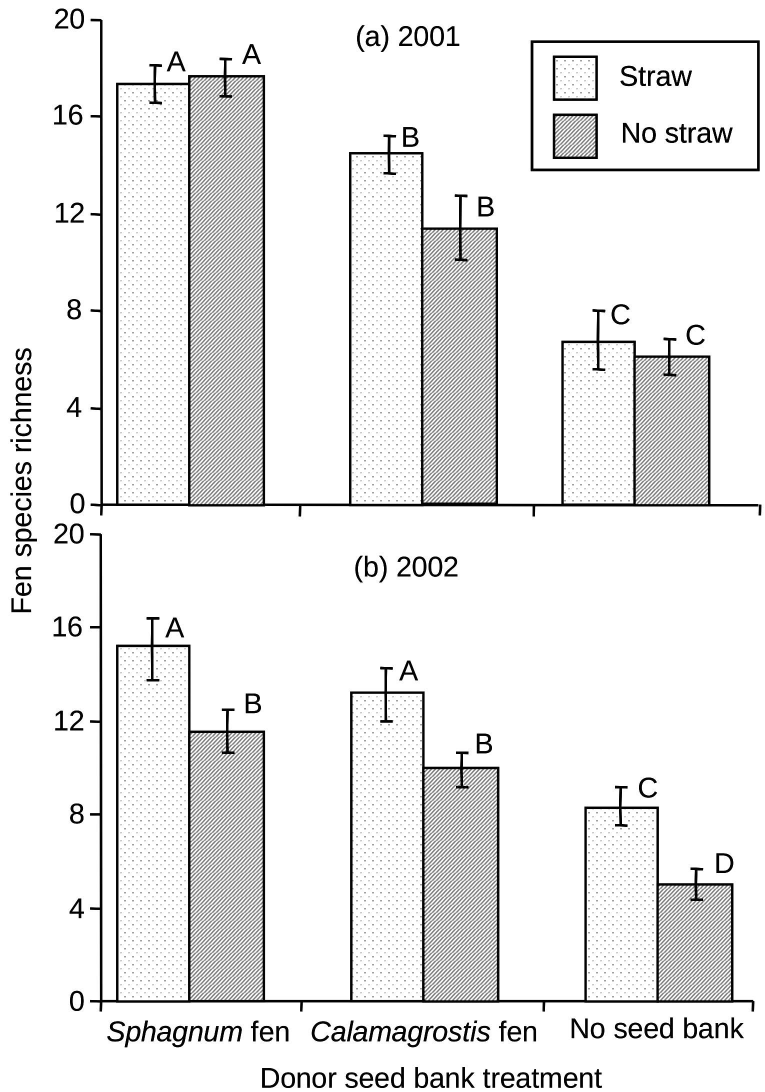

Figure 2.2 Effect of donor seed banks and mulch treatments on fen species richness after one growing season (a), and two growing seasons (b). Error bars show SE. Different letters represent significant differences within treatments identified by a Tukey test within treatments. There were no significant interactions between treatments.

2.3.1.3 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 (Figure 2.3a).

Equisetum arvense and *Tussilago farfara* were the second and third most dominant species after two growing seasons (Table 2.1). These species were dominant on the low terrace level (26 % ± 8%), whereas they formed only a minor component of the plant communities on the middle (1 % ± 0.5 %), and high terrace (2 % ± 1 %) levels (Figure 2.3b). No other experimental treatments had an effect on the establishment of these non-typical fen species (Table 2).

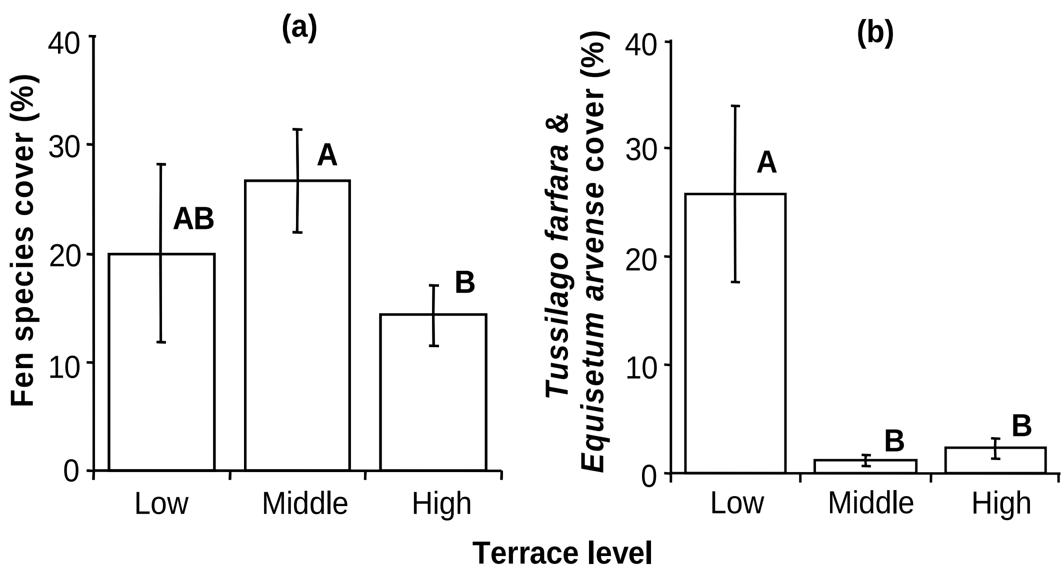

Figure 2.3 Effect of terrace levels on fen species cover (a), and *Tussilago farfara* and *Equisetum arvense* cover (b) after two growing seasons (2002). Error bars show SE. Different letters represent significant differences within treatments identified by Tukey tests.

2.3.2 Environmental conditions

2.3.2.1 Hydrology

From May to August 2001 and 2002, the total rainfall was 286 and 253 mm, respectively, compared to the mean 30-year seasonal total of 353 mm (Environment-Canada 1993). Site preparation was during a rather dry period in early May 2001, following the snowmelt. Removal of the surface layers of peat to successively greater depths to create the lower terraces resulted in the local surfaces being 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 2.3). This resulted in a water table that sloped toward the central drainage ditch, with a gradient of approximately 0.032 when the conditions were wettest (June 4, 2001) and 0.048 during the driest period in mid-August (August 16, 2001). Except for brief periods immediately following rain events, the water 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 except for during the driest periods. The water table depth at the restoration site was far lower than at the donor sites throughout the 2001 growing season (Figure 2.4). The water table at the donor sites was consistently just below the surface or above the surface throughout the growing season (*Sphagnum* fen: -4 cm ± 3 cm; *Calamagrostis* fen: 8 ± 8 cm) (Table 2.3).

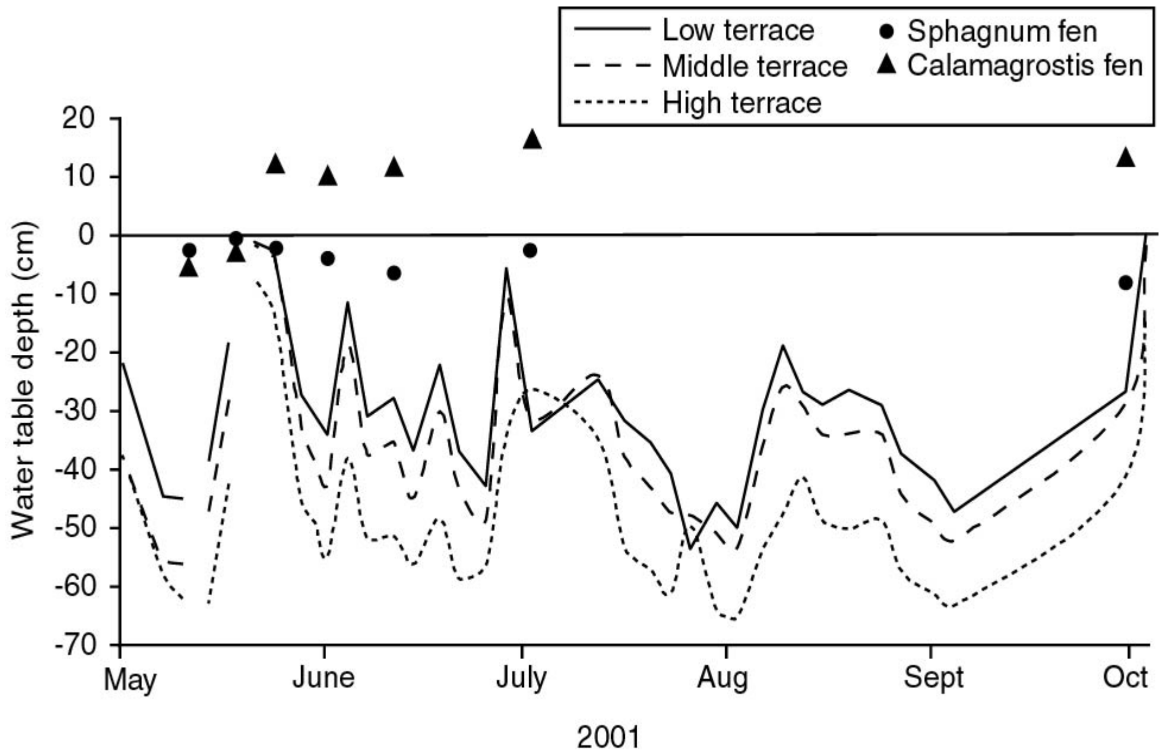

Figure 2.4 Water table depths (cm) for the terrace levels at the restoration site and the natural fen donor sites throughout the first growing season.

Mean soil-water pressure was above -100 mb until the middle of July for all terraces and decreased below -100 mb from the middle of July to the middle of August (Figure 2.5). The percentage of the time for which measures were less than -100 mb at the low, middle, and high terraces was 16%, 24%, and 24%, respectively. Soil water pressure is controlled partly by the strength of the capillary connection to the water table, and partly by the redistribution of water stored and released by rainfall infiltration and evaporative loss. Soil-water pressure in the upper and middle terrace was similar (averaging -66.3 and -62.5 mb, respectively), in spite of a notable difference in water table. This suggests the water storage and release processes are dominantly occurring in the upper layer of soil, and that capillary water flow in the middle terrace is insufficient to elevate the soil water pressure. In the lowest terrace soil water pressure was higher (averaging -41.3 mb), but below the equilibrium pressure defined by the water table (Table 2.3). With the water table there most frequently in the clay, little capillary water flow from that source can occur. Again, water storage and release processes in the upper layer predominate. In this lower terrace location, however, where the clay limits deeper water percolation, more complete resaturation of the peat occurred after significant rainfalls. 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 there.

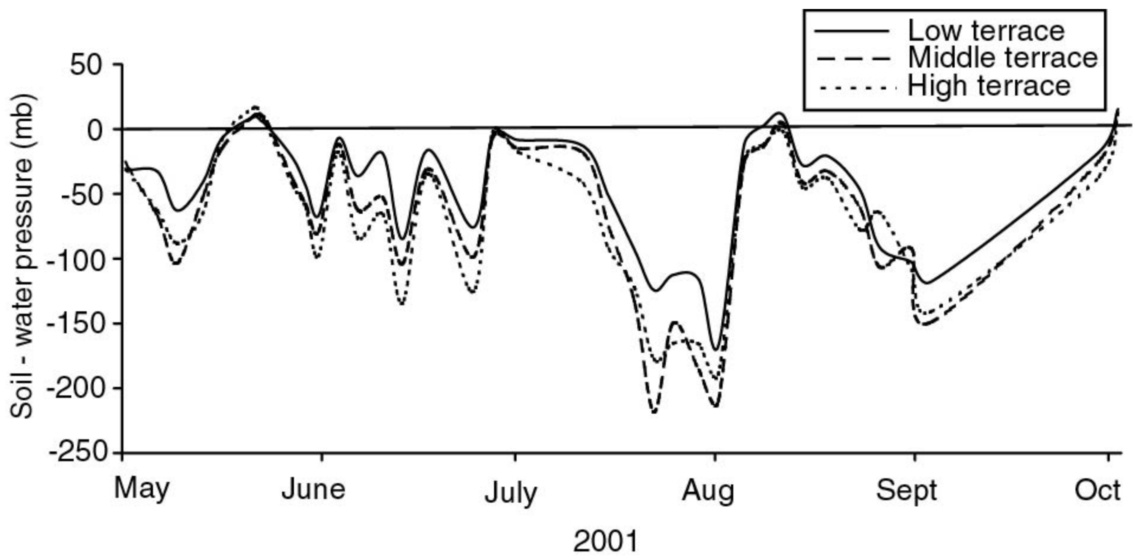

Figure 2.5 Soil-water pressures (mb) for the terrace levels at the restoration site throughout the first growing season.

Table 2.3 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 [1]							
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	78	338 ± 495	132 ± 117	134 ± 106	14	27 ± 9	40 ± 30
P total	54	0.6 ± 0.5	0.5 ± 0.4	0.5 ± 0.4	5	0.3 ± 0.5	0.2 ± 0.2
NH_4^+	54	1.3 ± 2.1	3.0 ± 3.3	3.6 ± 3.8	5	1.2 ± 2.3	1.0 ± 2.0
NO_3^-	54	1.5 ± 1.5	2.1 ± 1.8	2.2 ± 2.1	5	0.9 ± 0.7	1.6 ± 1.1
K^+	54	6.5 ± 4.9	3.3 ± 2.2	3.0 ± 1.8	5	1.4 ± 0.3	0.4 ± 0.4
Ca^{2+}	54	46.8 ± 73.9	9.5 ± 18.2	5.8 ± 4.3	5	1.8 ± 2.2	3.4 ± 1.6
Mg^{2+}	54	28.1 ± 58.2	4.6 ± 10.2	2.7 ± 2.9	5	0.4 ± 0.5	0.3 ± 0.1
Na^+	54	50.1 ± 34.8	26.1 ± 13.5	25.9 ± 15.6	5	2.0 ± 1.1	2.7 ± 1.7
Fe^{3+}	54	0.5 ± 2.5	0.4 ± 0.3	0.6 ± 0.3	5	0.1 ± 0.1	0.7 ± 0.4
Cu^-	54	0.4 ± 1.36	0.2 ± 0.1	0.2 ± 0.1	5	0.1 ± 0.0	0.2 ± 0.1
Peat chemistry [2]							
P	42	0.42 ± 0.26	0.27 ± 0.21	0.33 ± 0.36	7	0.58 ± 0.15	1.18 ± 0.61
N	42	19.94 ± 23.16	18.98 ± 30.26	14.96 ± 36.79	7	10.11 ± 16.99	19.92 ± 35.55
K	42	0.61 ± 0.34	0.22 ± 0.38	0.38 ± 0.16	7	0.79 ± 0.36	0.57 ± 0.40
Ca	42	9.92 ± 1.79	10.57 ± 12.67	10.13 ± 15.10	7	7.58 ± 1.46	5.15 ± 3.03
Mg	42	2.58 ± 0.42	3.80 ± 5.08	3.55 ± 4.15	7	1.01 ± 0.27	0.78 ± 1.07
Na	42	0.67 ± 0.19	0.43 ± 0.18	0.35 ± 0.10	7	0.24 ± 0.45	0.29 ± 0.94
Fe	42	4.17 ± 0.98	4.57 ± 0.74	4.34 ± 0.73	7	1.73 ± 0.51	2.17 ± 0.84

[1] Units for electrical conductivity are in µS/cm and chemical elements are in mg/l.
[2] Units are in mg/g.

2.3.3 Water chemistry

In general, there were higher concentrations of nutrients and minerals at the restoration site than at the natural fen donor sites (Table 2.3). The mean pH levels of the terraces at the restoration site did not vary greatly (circa 5.9). The pH of the *Calamagrostis* fen donor site (5.8) was similar to the restoration site, while the *Sphagnum* fen donor site had a lower mean pH (5.5). The electric conductivity of the restoration site was much higher than the natural fens, and there was a strong gradient of increasing electric conductivity with decreasing peat thickness. The electric conductivity for the *Sphagnum* fen and the *Calamagrostis* fen averaged 27 and 40 µS/cm, respectively. In contrast, the restoration site had an average electric conductivity of 201 µS/cm. The mean electric conductivity of the low terrace level was far greater with more variance (338 ± 495 µS/cm), than the middle (132 ± 117 µS/cm), and the high terraces (134 ± 106 µS/cm). The major cations followed a similar pattern as the electric conductivity, with higher concentrations at the restoration site compared to the natural fens, and a negative concentration gradient with increasing peat depth. Available iron at the restoration site did not exhibit a pattern along the terrace levels, and the mean concentrations (0.5 mg/L) were within the range of the natural fens (0.1 mg/L and 0.7 mg/L, for the *Sphagnum* fen and *Calamagrostis* fen, respectively). The concentration of copper in solution at the middle and high terrace levels (0.2 ± 0.1 mg/L) was similar to the natural fen concentrations (0.1 - 0.2 mg/L). In contrast, the low terraces had double the mean concentration of copper ions (0.4 ± 1.36 mg/L) with extremely high variability (Table 2.3).

For nutrients, the concentration of available phosphorus was higher at the restoration site (0.5 ± 0.3 mg/L) than at the *Sphagnum* fen (0.3 ± 0.5 mg/L) and at the *Calamagrostis* fen (0.2 ± 0.2 mg/L). Ammonium and nitrate concentrations were higher at the restoration site compared to the natural fens. Their concentration increased with increasing peat thickness, so that the highest terraces had the highest concentrations of ammonium and nitrate. The concentration of ammonium on the lowest level (1.3 ± 2.1 mg/L) was similar to the concentration at the *Sphagnum* fen (1.2 ± 2.3 mg/L) and the *Calamagrostis* fen (1.0 ± 2.0 mg/L). Whereas the other two terrace levels had higher concentrations (3.0 to 3.6 mg/l) than the natural fens. Similarly, the nitrate concentrations at the lowest terrace level (1.5 ± 1.5 mg/L) were between the natural levels found at the *Sphagnum* fen (0.9 ± 0.7 mg/L), and the *Calamagrostis* fen (1.6 ± 1.1 mg/L). The middle (2.1 ± 1.8 mg/L) and high terrace levels (2.2 ± 2.1 mg/L) had higher nitrate concentrations than the natural fens (Table 2.3).

2.3.4 Peat chemistry

The concentrations of major nutrients in peat tended to be lower or equal at the restoration site compared to the reference sites (Table 2.3). The mean total nitrogen concentration at the restoration site (17.96 mg/g) was between the mean concentration of the *Sphagnum* fen (10.11 mg/g) and *Calamagrostis* fen (19.92 mg/g). The concentration of total nitrogen decreased at the restoration site with decreasing peat thickness, with 19.94 mg/g, 18.98 mg/g, and 14.96 mg/g, for low, middle, and high terrace levels, respectively. Total phosphorus concentrations of the peat were lower at the restoration site (0.34 mg/g) than at the *Sphagnum* fen (0.58 mg/g), and the *Calamagrostis* fen (1.18 mg/g). There was no distinct pattern for the total phosphorus concentrations along the terrace levels. Potassium concentrations were generally lower at the restoration site (0.40 mg/g) than the natural fens (0.79 mg/g and 0.57 mg/g at the *Sphagnum* and *Calamagrostis* fens, respectively). The low terrace level had higher potassium concentrations (0.61 mg/g) that were similar to the natural fen concentrations. The middle (0.22 mg/g), and high (0.38 mg/g) terrace levels exhibited lower potassium concentrations than the natural fen concentrations.

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. These elements also tended to increase in concentration with decreasing peat thickness, indicating their mineral subsoil origin. Only iron did not exhibit a strong concentration gradient with peat thickness.

2.4 Discussion

2.4.1 Establishment of fen vegetation

The application of donor seed bank from natural fens clearly increased the cover and richness of fen species compared to control plots, supporting our first hypothesis. To the authors' knowledge, this is the first fen restoration project to test experimentally the effectiveness of applying donor seed bank as a plant reintroduction technique. The advantages of this plant introduction technique are numerous. Firstly, the variety of diaspore species and types contained within the donor seed bank 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 seed bank 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 (Cooke & Lefor 1998; Turner & Friese 1998). 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 seed bank is collected in the spring, it allows diaspores to fulfill 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). One disadvantage of this method is the disturbance caused to the donor wetland community during the seed bank removal. However, informal observations of the disturbed quadrats (< 4% of the reference unit) of the current study revealed that there was 25-40 % recovery by the end of the second growing season.

Several non-target species established at the restoration site, most notably *Equisetum arvense* and *Tussilago farfara*. These species dominated the lowest terraces of the restoration site after two years, but were largely absent on the higher levels. Their rapid and competitive growth appeared to limit the establishment of fen species on the lowest terrace level. These perennial species are able to quickly colonize due to their ability for expansive vegetative reproduction and their ability to produce a high number of spores or seeds rapidly. Future studies are required to determine whether these species are responding to the hydrological or chemical differences between the terrace levels. *Equisetum arvense* and *Tussilago farfara* naturally colonized abandoned minerotrophic peat surfaces in Finland. In most cases, their abundance was lower on older peat fields, suggesting a decrease in dominance over time (Salonen 1990).

The mulch treatment increased the diversity of fen plants after two years. However, it contributed little to increasing the abundance of fen species. Only during the first year was there a synergistic effect with straw mulch and *Sphagnum* fen seed bank together. In previous studies, mulches have been shown to improve the germination of several graminoid species, although others (ex. *Eriophorum vaginatum*) failed to respond to the same treatments (Sliva & Pfadenhauer 1999). Straw mulch has been demonstrated to improve moss diaspore establishment, particularly *Sphagnum* mosses. In contrast, vascular plants failed to show improvements (Rochefort *et al.* 2003). Mulches improve microclimatic conditions by increasing soil moisture and moderating surface temperatures (Price *et al.* 1998). The surface peat was dry at the restoration site for a large portion of the growing season. Under wetter conditions, the straw mulch may have functioned to retain moist conditions and increase the number of safe sites available, thereby promoting the abundance of fen plants.

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 highly minerotrophic (saline) conditions of the lowest terrace level. Extremely dry conditions and relatively saline conditions were not observed at the donor sites.

2.4.2 Environmental conditions limiting fen restoration

While the water table sloped across the terraces, there was not a uniform flux of water toward the ditch. 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 Rivière-du-Loup region (Van Seters & Price 2002). Thus lateral water flow across the lowest terrace was insignificant except for a short period following major rainstorms. In contrast, water flow through the higher hydraulic conductivity peat may have helped by maintaining a higher water table in the middle terrace than would otherwise occur. The corollary of this is an even lower water table in the upper terrace.

The low water table position and geometry of the terraces indicate that vertical and lateral drainage of the upper and middle terraces occurs, which enhances flushing of solutes present in the peat, reducing their concentration. The thinner peat layer (only 15 cm of peat), and limited lateral drainage at the lower terrace reduced deep percolation at this site, thus solutes were not leached away as in the upper terraces. This partly explains the higher concentration of solutes at the lower terrace, which will be a factor in the restoration of any cutover peatland with a small residual peat depth. In this experiment, the higher concentration of solutes in deeper peat is a natural occurrence resulting from the diffusion of salts from the marine clay (Van Seters & Price, unpublished data for Cacouna peatland, 1999). Deeper excavation to the level of the lower terraces at the commencement of the experiment exposed peat with higher solute concentration. Over time the higher concentration of solutes at all terraces are expected to decline. Nevertheless, the ecological response measured in this experiment reflected these more saline conditions, and showed the importance of considering and perhaps managing the ambient concentration of solutes in the peat substrate. Obtaining the correct water quality is critical to achieve the desired fen plant community (Charman 2002; Lamers *et al.* 2002). The concentrations of base cations at the low terrace level are currently typical of saline marshes (Zoltai & Vitt 1995). The high concentrations of major cations may restrict the growth of some fen species, as fen plant communities have been shown to vary along base cation concentration gradients (ex. (Bridgham *et al.* 1996). Pore water from isolated fens is more enriched than that of fens connected to surface water bodies (i.e. lakes, rivers, and perennial streams) (Godwin *et al.* 2002). The base cation concentrations of the restoration site could be reduced by increasing potential hydrological inputs. This would hasten the flushing of cations from the peat substrate.

Several recent studies have indicated that the hydrological conditions, particularly water table depth, are an important factor affecting fen species establishment (Roth *et al.* 1999; Budelsky & Galatowitsch 2000). Reintroductions by plantings have been the most effective with water table levels slightly below the surface or with shallow standing water (Cooper & MacDonald 2002). *Sphagnum* mosses and other bryophytes were a dominant component of the donor fen sites, but failed to establish at the restoration site. This is likely due to harsh hydrological conditions. Price & Whitehead (2001) identified hydrological thresholds for the establishment of *Sphagnum* mosses. Specifically, abandoned mined peat sites with *Sphagnum* mosses present have been found to have soil-water pressure values greater than –100 mb, and water table depths greater than -40 cm (Price & Whitehead 2001). *Sphagnum* is unable to extract moisture from the soil when the soil-water pressure is below -100 mb because it can not generate enough capillary force (Price 1997). The restoration site exceeded these thresholds at all terrace levels for a portion of the first growing season. While the climatic conditions were dryer than normal, they were not atypical, and restoration measures must be designed to accommodate a realistic range of conditions. Further rewetting measures are necessary, therefore, to create fen-like hydrological setting at the restoration site. At this site, the freedom to manipulate the hydrology was constrained by drainage requirements from ongoing adjacent extraction activities.

Another water quality factor affecting the restoration potential of the fen plant community is the high concentration of nitrate at the restoration site. The high nitrate concentrations are likely due to the drier hydrological conditions of the restoration site. Decreased soil moisture promotes microbial activity, which increases mineralization processes. Nitrate, in particular, may become available in large quantities, even excessive amounts on drained peat surfaces (de Mars *et al.* 1996; Wind-Mulder & Vitt 2000). High nitrate concentrations have been correlated with low plant diversity in natural fens (Drexler & Bedford 2002), and on abandoned minerotrophic sites after peat mining (Rowlands 2001). Moreover, fertilization studies on sedge meadow communities have shown that community diversity and evenness declined with increasing nitrate levels (Green & Galatowitsch 2002). It follows that restoration measures aimed at raising the water table are likely to lower the concentration of nitrate and facilitate the establishment of a more diverse plant community.

2.5 Conclusions

The application of donor seed bank was clearly demonstrated as an effective introduction technique for restoring fen plants. The dominance of fen plants at the restoration site increased from the first to the second year, indicating that site is advancing towards a fen plant community. Despite the establishment success of several fen species, bryophytes were absent after two years due to insufficient rewetting. Further management of the site is required to create hydrologic conditions that can support bryophytes establishment. Under wetter conditions, straw mulch may have improved the establishment of mosses, as has been found in bog restoration.

The application of straw mulch improved the richness of fen plant species, and aided in the proliferation of some fen plants in the early stages of community development. Straw mulch may function to increase fen plant establishment during the early stages of plant community development.

The terrace levels affected the establishment of fen plants, which were more abundant on the intermediate level. This treatment may have promoted the establishment of fen plants because of its intermediate moisture regime (i.e. not too dry), and its intermediate mineral status (i.e. not too saline). Further rewetting of the restoration site may reduce base cation and nutrient concentrations, creating similar conditions to the donor fens. The low terrace level has base cation concentrations that are similar to a saline marsh, and it is considered unsuitable for a fen plant community.

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3 Effects of water regime and donor seed bank source on the seedling emergence of fen species

Abstract

The effects of different donor seed bank treatments and water regimes on seedling emergence were examined in a green house experiment. Two donor seed banks were collected from natural fens with contrasting vegetation communities – a poor fen, and a moderate-rich fen. Two water regime treatments were tested - saturated and flooded conditions. The total number of seedlings that emerged from the seed bank was significantly higher from the moderate-rich fen, compared to the poor fen. The saturated water regime yielded more seedlings than the flooded water regime, although the results were not statistically significant. The results suggest that the diversity of species that emerge from donor seed bank could be maximized by creating a diversity of hydrological conditions in the field and applying donor seed bank from more than one source.

Nomenclature

Vascular plants (Gleason & Cronquist 1991); Sphagnopsida (Anderson 1990); other mosses (Anderson *et al.* 1990).

3.1 Introduction

Understanding the factors that control seed germination is important for restoration projects that attempt to introduce plants with donor seed bank. A seed bank contains a variety of species with different germination requirements that allow different species to occupy different niches (Leck *et al.* 1989). The manipulation of different factors in laboratory experiments can provide invaluable information for predicting vegetation composition in the field and for applying appropriate management regimes to favour target species (Keddy *et al.* 1989).

Hydrology is considered the most important environmental factor controlling the community structure and composition of wetlands (Keddy 1999). Water table levels have been shown to affect seedling emergence from wetland seed banks (Leck 1989; Willis & Mitsch 1995). The establishment of sedges, a dominant species in many fens, from seed is considered very difficult (Budelsky & Galatowitsch 1999; Sliva & Pfadenhauer 1999; van der Valk *et al.* 1999). Occasionally high germination rates have been observed in the field by sowing seeds of *Carex* species. Successful germination was attributed to the creation of appropriate hydrological conditions (Roth *et al.* 1999). Determining the hydrological conditions that favour fen plant species emergence is of crucial importance for the management and restoration of fen plant communities.

In this paper, I manipulated different factors in a greenhouse experiment to better understand the relationships between donor seed bank dynamics and plant community composition. The first objective was to determine how donor seed bank from natural fens with contrasting plant communities affected the diversity and abundance of species that established. The second objective was to determine what effect saturated versus flooded hydrological regimes had on the species that emerged from the donor seed banks. I then considered how these results could be applied to better manage restoration projects in the field.

3.2 Materials and Methods

The seedling emergence technique was used to measure differences in the relative response of fen species under different experimental treatment. This technique provides an estimate of the number of viable seeds in a soil seed bank based on the emergence of seedlings under conditions favourable to their germination (Simpson *et al.* 1989). This technique may greatly underestimate viable seed abundances in a soil seed bank because ideal germination conditions are rarely met due to the sensitivity of germination patterns to light, fluctuating temperatures, oxygen availability, and substrate texture (Simpson *et al.* 1989). Despite these limitations, it is the technique considered the most appropriate for measuring the relationships between seed bank composition and field recruitment of wetland plants (Brown 1998).

The experiment was a two factors complete factorial design. The effect of donor seed bank collected from different plant communities and the effect of different hydrological conditions were tested. Donor seed bank was collected from two natural fens used in a corollary field restoration experiment (Refer to chapter 2). One collection site was a poor fen with low herbaceous cover (*Sphagnum* fen) and the second site was a moderate-rich fen with high herbaceous cover (*Calamagrostis* fen). A vegetation control treatment (no donor seed bank material) was established to evaluate seed contamination within the greenhouse. The hydrology regime treatments were flooded and saturated conditions (see below). There were three replicates for each combination of vegetation treatments and water regimes, totalling 18 plots (3 replicates x 3 seed bank treatments x 2 water regimes).

Eighteen trays (25 cm x 25 cm) were randomly positioned on the same table within a greenhouse. Each tray received 750 ml of sterilized sand (approximately 1cm depth). The flooded water table treatment was created by adding 750 ml of composited donor seed bank material on top of the sterilized sand, while the saturated water table treatment had 1500 ml of donor seed bank material added. Approximately equal amounts of water were added to the trays each day. Due to the differences in the original quantity of donor seed bank material applied, the water table of the flooded treatments was approximately 1 cm above the surface, while the water table of the saturated treatments was approximately 0.5 cm below the surface. Donor seed bank was collected from the natural fen sites two weeks after the field restoration experiment commenced, in the last week of May 2001 (refer to Chapter 2). Therefore, the state of the donor seed bank in the seedling emergence experiment approximates its state at the time of restoration. Thirty-six seed bank samples were randomly collected from each donor site. Samples were collected with a soil corer (3.5 cm radius, 5 cm depth) that was wiped clean between sites. The subsamples from each site were combined and stored at 4ºC for one week. Any seedlings, live and dead roots, rhizomes, sticks, leaves, and other macroscopic plant diaspores were removed from the peat samples to isolate the seed bank. Seedling emergence was recorded at the end of an 11-week period (June 4- August 19, 2001). Each individual stem was counted, and mature specimens were identified to the species level. Immature plants were identified to the closest identifiable taxon.

3.2.1 Analysis

The experiment was analyzed with a two-way analysis of variance (ANOVA) using Microsoft® EXCEL 2002 (Microsoft Corporation 2002). The hydrological regime and seed bank source were treated as main effects. The control treatment of donor seed bank was not included in the analysis because the experiment was designed to test for the effect of different donor seed bank sources. The saturated controls produced four individuals of *Tussilago farfara* and one unidentified herb, indicating contamination from local seed rain. These species were omitted from seed bank analysis. The total number of seedlings was used as the response variable. The mean and standard deviation of seedling emergence at the individual species or taxon level were listed by treatments to determine their effects.

3.3 Results

Twelve species emerged during the experiment (Table 3.1). A few species were abundant (e.g. *Juncus spp., Glyceria canadensis, Agrostis hyemalis*), others were moderately abundant (e.g. *Galium trifidum, Scirpus cyperinus*, Cyperaceae (immature)), and several were uncommon. *Sparganium chlorocarpum, Potamogeton* cf. *pusillus, Drosera rotundifolia* plants emerged in the greenhouse experiment (Table 3.1). However, these species were not observed in field restoration experiments using the same seed bank treatments (Chapter 2).

Table 3.1 Seedling emergence (means and standard deviations) by species according to donor seed bank and water regime treatments.

Species	*Sphagnum* fen		*Calamagrostis* fen	
	Saturated	Flooded	Saturated	Flooded
Agrostis hyemalis			37 ± 17	72 ± 28
Cyperaceae (immature)	1 ± 2	6 ± 9	9 ± 14	25 ± 15
Drosera rotundifolia		0 ± 1		
Galium trifidum	0 ± 1	2 ± 1.5	28 ± 25	7 ± 5
Glyceria Canadensis	86 ± 26	56 ± 10	8 ± 12	5 ± 8
Gramineae (immature)	22 ± 27	15 ± 9	1 ± 1	7 ± 8
Juncus spp.			135 ± 62	71 ± 20
Potamogeton cf. pusillus				1 ± 1
Salix spp.	1 ± 2	1 ± 0.6	1 ± 1.7	1 ± 2
Scirpus cyperinus			21 ± 3	14 ± 6
Sparganium chlorocarpum			4 ± 3	9 ± 3
Viola macloskeyi	6 ± 3	7 ± 4		1 ± 1
Total seedlings emerged	118 ± 60	89 ± 21	299 ± 71	209 ± 48

3.3.1 Effect of seed bank source on seedling emergence

Significantly more seedlings emerged from the *Calamagrostis* fen seed bank (254 ± 73) compared to the *Sphagnum* fen seed bank (103 ± 39) (Table 3.2). The species that emerged from the donor seed banks differed. The most abundant species that emerged from the *Calamagrostis* fen seed bank were *Juncus spp., Agrostis hyemalis, Galium trifidum*, Cyperaceae (immature), and *Scirpus cyperinus* (Table 3.1). Species that emerged solely from the *Calamagrostis* fen seed bank included *Agrostis hyemalis, Potamogeton cf. pusillus, Scirpus cyperinus*, and *Sparganium chlorocarpum*. The *Sphagnum* fen seed bank yielded high numbers of *Glyceria canadensis,* Gramineae (immature), and *Viola macloskeyi*, in addition to *Drosera rotundifolia*, which did not emerge from the *Calamagrostis* fen seed bank. The total number of species that emerged from the *Calamagrostis* fen seed bank was 11 species, compared to seven species from the *Sphagnum* fen seed bank (Table 3.1).

Table 3.2 Two-way ANOVA results for the effect of seed bank and water regime treatments on total seedling emergence. F-ratios are followed by P-values in parentheses. Significant P-values ($P < 0.05$) are indicated in bold type.

Source of variation	Sum of squares	D.F.	Mean square	F-ratio (P)
Seed bank	67950.75	1	67950.75	25.90 **(0.001)**
Water regime	10620.75	1	10620.75	4.05 (0.07)
Seed bank * water regime	2730.08	1	2730.08	1.04 (0.33)
Error a	20987.33	8	2623.41	
Total	102288.9	11		

3.3.2 Effect of water regime on seedling emergence

There was no statistical difference between the numbers of seedlings that emerged from the water regime treatments (Table 3.2). However, the total number of seedlings that emerged was greater under the saturated conditions (208 ± 114) compared to the flooded conditions (149 ± 74). All species emerged in greater densities from the saturated water regime, except Cyperaceae (immature), *Drosera rotundifolia*, *Sparganium chlorocarpum*, and *Viola macloskeyi* (Table 3.1).

3.4 Discussion

3.4.1 Effect of seed bank on seedling emergence

The *Calamagrostis* fen seed bank produced a higher seedling density and species richness compared to the *Sphagnum* donor seed bank. The diversity of established vegetation at the donor sites was similar, with 37 species at the *Sphagnum* fen and 34 species at the *Calamagrostis* fen, suggesting that the diversity of species in the seed bank was not the factor most strongly affecting the diversity of species that emerged. Species in the *Calamagrostis* fen seed bank may have a higher reproductive capacity, higher persistence, and broader tolerance limits to environmental conditions, than species in the *Sphagnum* fen seed bank. Wetland species vary in their reproductive capacity (i.e. number of seeds produced) and germination strategies (persistent or transient) and rates, and response to environmental factors (Leck 1989). In addition, the timing of the soil collection or the conditions for emergence of the *Sphagnum* fen seed bank may not have been suitable (Parker *et al.* 1989).

Since different species emerged from the different sources of donor seed bank, using multiple sources of donor seed bank material may increase the odds of obtaining a more diverse species assemblage at a restoration site.

3.4.2 Effect of water regime on seedling emergence

A greater number of seedlings emerged under saturated soil conditions than flooded soil conditions, although differences were not statistically significant. It is worth noting that a flaw in the experimental design likely overestimated the number of seedlings that emerged from the flooded treatments. The flooded treatment was created by adding double the quantity of donor seed bank material, which effectively doubled the number of seeds that had the potential to germinate. Both treatments should have received an equal quantity of seed bank material to isolate the effect of the water regime. Despite the inherent bias towards the flooded treatment, the saturated treatment produced more seedlings, allowing us to be confident that the saturated treatment was more effective. Restoration projects should attempt to create saturated soil conditions to maximize the emergence of seedlings.

Flooded conditions act as a strong environmental filter hindering the germination of many wetland species (Willis & Mitsch 1995) and affecting the plant community composition of wetlands (Keddy 2000). The donor seed bank treatments yielded several species that germinated in greater quantities under the flooded conditions, including *Sparganium chlorocarpum, Potamogeton* cf. *pusillus,* and *Drosera rotundifolia*. It is worth noting that none of these species were recorded in the field restoration experiment using the same donor seed bank (Chapter 2). This study suggests that these species failed to emerge in the field experiment due to a lack of flooded conditions. This is not surprising for *Drosera rotundifolia* because germination and growth generally start while the peatland is covered by meltwater in the spring. The water table level is critical for *Drosera rotundifolia,* which normally ranges from 2 cm above the ground surface to 40 cm below, and several weeks of flooding can be endured, whereas long periods of drought cannot be tolerated (Crowder *et al.* 1990).

These results support the work of other studies showing that differences in micro environmental conditions could result in different vegetation composition (Galinato & van der Valk 1986; Leck 1996). Restoration protocols that create a variety of hydrological conditions are likely to create a more diverse community assemblage.

3.5 Conclusion

The *Calamagrostis* fen seed bank produced a higher seedling density and species richness compared to the *Sphagnum* donor seed bank. A greater number of seedlings emerged under saturated soil conditions than flooded soil conditions. Since different species emerged from the donor seed banks and hydrology treatments, restoration protocols should introduce a variety of seed bank sources and create a variety of hydrological conditions to create a more diverse community assemblage.

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3.6 References

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4 Monitoring and evaluating fen restoration success

Abstract

Restoration efforts to restore a fen plant community after peat mining were monitored and evaluated. Surveys of natural fens in the study region were used to build a composite model of a reference ecosystem. Direct comparisons with natural fens in the study region revealed that several environmental and biological factors have not yet been restored. Total fen plant cover and richness at the restoration site were below natural fen levels. In addition, water table levels and solute concentrations were outside the ranges observed at reference sites. These environmental conditions require further restoration measures to create conditions favourable for a fen plant community. Trajectory analysis was used to monitor and evaluate the plant community composition at the restoration site relative to plant communities at unrestored sites and natural fens. The analysis clearly separated disturbed sites from natural fen sites based on plant community composition. Several fen plant species were not found at the disturbed sites, including several *Carex* species. Further monitoring of the restoration site is recommended to determine whether the plant community proceeds towards the target reference ecosystem.

Nomenclature

Vascular plants (Gleason & Cronquist 1991); Sphagnopsida (Anderson 1990); other mosses (Anderson *et al.* 1990).

4.1 Introduction

The goal of restoration is to return a damaged site to its historical trajectory (i.e. a more natural condition) (SER 2002). Monitoring and evaluating the success of restoration projects is important to help guide future restoration efforts. The term "trajectory" has been adopted in restoration ecology to describe the path of restoration site development through time towards (or away from) its target ecosystem (Zedler & Callaway 1999). Ideally, a target or reference ecosystem is based on several wetlands within a specific geographic region to encompass the known variation of the group or class of wetlands of interest. Studies of single sites of pairs or small number of sites do not provide results that can be extrapolated to a broader range of circumstances and conditions. The reference ecosystem is used to define restoration goals and later to assess their success (Brinson & Rheinhardt 1996; Kentula 2000). Natural ecosystems provide direct evidence of later successional stage ecosystems under undisturbed conditions and can serve as the goal for restoration (White & Walker 1997). Disturbed sites can act as an experimental control, serving as a reference point to which the relative effectiveness of the restoration measures can be assessed. This information can also be used to identify potential barriers limiting restoration, such as dispersal barriers or abiotic conditions.

Approaches to evaluate restoration projects include direct comparison analysis and trajectory analysis. Direct comparison analysis uses a carefully selected suite of ecosystem traits to measure and compare the reference and restoration sites. Abiotic and biotic parameters are carefully selected to describe collectively the reference ecosystem. Trajectory analysis plots data collected periodically from the restoration site to determine its development through time relative to its target ecosystem (SER 2002).

This paper aims to establish a framework for evaluating the success of restoration efforts to restore a fen plant community (Chapter 2). The first objective is to define a target ecosystem to define a restoration goal and evaluate its success. Surveys of natural fens in the study region were conducted to define the natural variability of plant community composition, and their chemical and hydrological characteristics. The second objective is to determine the relative effects of restoration efforts compared to unrestored sites that were similarly disturbed by peat mining. These comparisons are used to establish a point of reference for future monitoring. Finally, the abiotic and biotic characteristics of the target reference ecosystem are compared to the restored and unrestored sites disturbed by peat mining. The effectiveness of restoration measures after two years are evaluated using direct comparisons and trajectory analysis.

4.2 Methods

4.2.1 Restoration site

The restoration site is part of the Rivière-du-Loup peatland, located approximately 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, alt. 100 m). The site was mined for peat until layers of sedge peat was exposed, at which point it was abandoned. A restoration experiment was established at the site immediately after abandonment in spring 2001. Several restoration procedures were tested including altering the depth of residual peat, introducing donor seed bank from natural fens, and applying straw to improve microclimate conditions (refer to Chapter 2).

The percentage cover of the vegetation at the restoration site was sampled from October 10-14, 2001 and from August 8-13, 2002. Ten quadrats (30 cm x 30 cm) in each experimental plot were systematically sampled. The percent cover (visually estimated to the nearest percent) 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 percent cover of the experimental plots at the restoration site was averaged for each year.

The water table depth was measured twice a week during the 2001 growing season. 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 analyzed for water chemistry (refer to Chapter 2 for a detailed description of techniques used).

4.2.2 Spontaneously revegetated sites

Four sites were identified within the Rivière-du-Loup peatland that had sedge peat exposed from peat mining activities. The sites had been abandoned 7-20 years earlier without restoration efforts. Thus, the vegetation present had spontaneously recolonized the sites. Vegetation was surveyed using the point sampling method (Bonham 1988) on a grid with points intersecting every 8 m along the length of the fields and every 5 m along the width of the fields. Each field was approximately 200 m long and 30 m wide. The presence/ absence of all species in contact with one side of an infinitely long 0.3 cm rod were recorded. The relative abundance of each species was averaged for each site.

4.2.3 Natural fens

Field reconnaissance to locate natural fen sites revealed that there were few natural fens nearby the restoration site. Fifteen relatively undisturbed fens were found in the foothills of the Appalachian Mountains, within 35 km of the restoration site. These fens were chosen based on their proximity to the restoration site, and accessibility. Vegetation communities within the peatlands were subjectively chosen based on contrasting plant communities amongst peatlands. A total of sixty quadrats were surveyed from the fifteen peatlands. The percentage cover of each species in a 10 m by 10 m quadrat was visually estimated, as was the total percent cover. Smaller sized quadrats are generally recommended for bryophytes (e.g. Kent & Coker1992), however time constraints did not allow for more detailed sampling. This sampling protocol was considered adequate for determining the presence and relative abundance of species within the vegetation communities.

Additional species that were not in the quadrat but considered to be part of the same vegetation community were recorded to ensure a more complete species list. The depths of the water table and the peat were measured at three random locations within the quadrat and subsequently averaged. Water was collected from the surface of the water table and analyzed for water chemistry variables (refer to Chapter 2 for techniques).

4.2.4 Analyses

Descriptive summary statistics were compiled for environmental and biological variables at the restoration site in 2002 and at the natural fen sites. For each variable a direct comparison values at the restoration site were compared to the range of variation observed at the natural fen sites. The mean of the restoration site was considered similar to the natural fens if it was within the standard deviation of the natural fens. Similar analysis has been used to determine the restoration success of salt marsh and mud flat habitats (Short *et al.* 2000).

An ordination was performed using Detrended Correspondence Analysis (DCA) of abundance data from the restored site (2002), unrestored sites, and natural fen sites. DCA is an indirect ordination technique and is recommended for exploring community structure and gradients (ter Braak 1995). The fifty most abundant species from all the sites were used in the statistical analysis. Species data was log transformed to reduce the impact of outliers (Zar 1984). DCA was performed with CANOCO for Windows version 4.5 (ter Braak & Smilauer 2002) using the default options. A biplot with species and sites was constructed to allow an assessment of relationships among ordination results and specific taxa and sites (ter Braak 1995).

4.3 Results

4.3.1 Direct comparison

Surveys of the 15 natural fens yielded 224 plant species (Appendix B). In comparison, 96 species were found at the unrestored sites, of which 83 species were fen species. The restoration site had seventy-two species after two years, of which 63 were fen species (Appendix B).

The restoration site differed from the natural fens for eight of fifteen environmental and biological variables (Table 4.1). The mean peat depth and water table level of the restoration site were outside of the range of the standard deviation of the natural fens. Several water chemistry variables were within the natural range including ash (%), pH, electrical conductivity, total phosphorus, nitrate, ammonium, and iron. However, concentrations of dissolved solutes including potassium, calcium, magnesium and sodium were higher the standard deviation observed for the natural fens. The total fen cover and richness at the restoration site was far lower than the values observed at the natural fens (Table 4.1).

Table 4.1 The mean, standard deviation, and range (minimum – maximum) of values from the natural fen surveys and the restoration site for environmental and biological variables. The mean of the restoration site was compared to the standard deviation of the natural fens to determine if the restoration was similar.

Variable	Natural fens	Restoration site	Similar?
Peat depth (cm)	86 ± 35 (10 - <135)	37 ± 18 (7 - 66)	No
Water table depth (cm)	-3.7 ± 15.0 (-38.3–(31.7))	-35.8 ± 17.4 (2.0–(-93.0))	No
Ash (%)	18 ± 35 (1-70)	15 ± 5 (9-37)	Yes
pH	6.2 ± 0.9 (3.6 – 7.1)	5.9 ± 0.4 (3.3 – 6.5)	Yes
Electrical conductivity (µS/cm)	116 ± 136 (30 – 680)	187 ± 214 (35 – 1281)	Yes
P total (mg/L)	0.4 ± 0.4 (0.0 -1.4)	0.5 ± 0.4 (0.0-1.7)	Yes
NH_4^+ (mg/L)	1.5 ± 1.0 (0 - 4.2)	1.9 ± 1.8 (0.0- 9.3)	Yes
NO_3^- (mg/L)	1.4 ± 2.6 (0.0 – 9.8)	2.7 ± 3.3 (0.0 – 23.7)	Yes
K^+ (mg/L)	0.9 ± 0.7 (0 - 4.1)	4.3 ± 3.6 (0.4 – 31.1)	No***
Ca^{2+} (mg/L)	8.3 ± 8.7 (0.5 – 42.5)	20.5 ± 47.4 (0.0 – 381.1)	No
Mg^{2+} (mg/L)	0.4 ± 0.6 (0.0 – 2.6)	11.7 ± 35.8 (0.0 – 385.7)	No***
Na^+ (mg/L)	8.1 ± 17.9 (0.3 – 100.1)	34.0 ± 25.9 (7.7 – 149.5)	No
Fe^{3+} (mg/L)	1.9 ± 3.7 (0.0 – 17.7)	0.5 ± 1.4 (0.0 – 18.3)	Yes
Fen plant cover 2002 (%)	80 ± 19 (15 – 100)	20 ± 17 (1 - 98)	No
Fen plant richness 2002	24 ± 9 (8 - 46)	11 ± 4 (2 – 21)	No

*** Indicates that the mean value at the restoration site is not within the range (minimum – maximum) of values observed at the natural fen sites.

4.3.2

Trajectory analysis

The sites were well separated along an extremely strong primary gradient and a lesser secondary gradient (5.221 and 2.926 SC units, respectively) (Table 4.2) (ter Braak 1995). Overall, eigenvalues were quite large, and the values for the first two axes explained 19.9 % of the cumulative variation in the plant community data (Table 4.2).

Table 4.2 DCA summary statistics of plant communities composition of the natural fens sites, unrestored sites, and the restoration site.

Axis summary statistics	DCA axis 1	DCA axis 2
Eigenvalues	0.569	0.285
Lengths of gradient	5.221	2.926
Cumulative percent variance of species data	13.3	19.9

The primary axis is interpreted as a gradient of disturbance and clearly separated natural fen sites from disturbed sites (Figure 4.1). The natural fens are positioned on the left hand side of the biplot diagram, in distinct contrast to the disturbed sites (both restored and unrestored), which are positioned on the right hand side of the biplot diagram. The second axis differentiates the disturbed sites, positioning the restored sites towards the bottom of the second axis and the unrestored sites towards the top of the second axis (Figure 4.1).

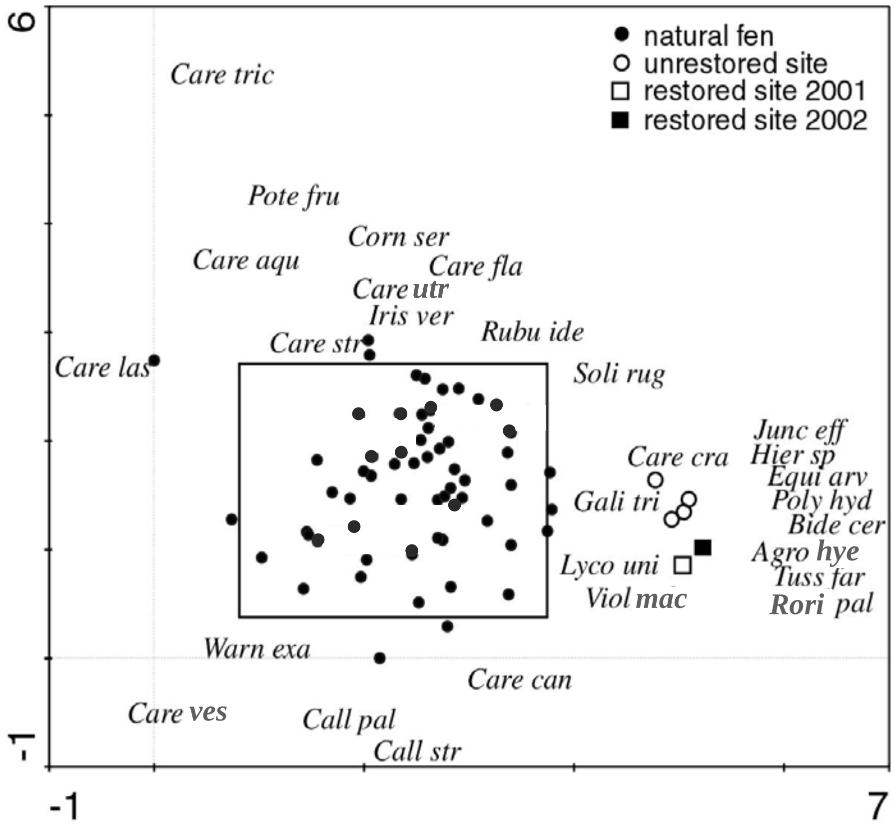

Figure 4.1 Biplot diagram of plot scores and species scores along axis 1 and 2 based on DCA of plant abundance data from natural fen sites, unrestored fen sites, and restored fen sites. The inner box represents the reference ecosystem and is defined on the 95 % percentile of the site scores for the natural fens. Species codes are the first four letters of the genus and the first three letters of the species. Species scores within the box were excluded for clarity.

The species most strongly associated with the disturbed sites (highest species scores from the first DCA axis) were *Rorippa palustris, Tussilago farfara*, *Polygonum hydropiper*, and *Juncus effusus* (Figure 4.1; Table 4.3). All of these species were present at the disturbed sites, whereas none were present at the natural fen sites (Table 4.3). The species most strongly associated with the natural fen sites (lowest species scores from the first DCA axis) were *Carex lasiocarpa, Carex vesicaria, Warnstorfii exanulata,* and *Carex aquatilis*, which were not present at the disturbed fen sites (Table 4.3).

The second axis separated natural fen sites into those with *Carex trichocarpa, Potentilla fruticosa, Cornus sericea*, and *Carex aquatilis* (top of biplot diagram) from those with *Calliergon stramineum, Calla palustris*, and *Carex vesicaria* (bottom of biplot diagram) (Figure 4.1). *Carex canescens*, *Rorippa palustris, Viola macloskeyi*, and *Tussilago farfara* were more strongly associated with the restored sites than the unrestored sites, according to the species scores on the second DCA axis (Figure 4.1).

Table 4.3 Species used in the DCA based on the most abundant species from the natural fen sites, spontaneously revegetated sites, and restored sites (2002). Species are ranked according to species scores from the first DCA axis. The number of plots for which a species was present and the mean cover (%) are listed.

		Natural fens		**Unrestored**		**Restored 2002**	
Species name	**species scores**	**(#/60 sites)**	**cover**	**(#/4 sites)**	**Rel. cover**	**(#/54 plots)**	**cover**
Rorippa palustris	6.17	0	0	1	0.05	46	1.44
Tussilago farfara	6.10	0	0	1	0.22	39	4.86
Polygonum hydropiper	5.95	0	0	3	3.27	38	0.67
Juncus effuses	5.90	0	0	4	5.77	16	1.43
Bidens cernua	5.89	1	0.02	4	2.07	34	0.28
Equisetum arvense	5.79	2	0.03	4	24.40	19	5.02
Hieracium spp.	5.77	1	0.02	3	8.82	36	0.58
Euthamia graminifolia	5.46	5	0.17	4	21.46	48	0.83
Fragaria virginiana	5.44	3	0.07	3	7.08	13	0.22
Agrostis hyemalis	5.44	6	0.11	4	9.26	49	5.34
Carex crawfordii	5.29	3	0.08	1	0.11	38	3.67
Polytrichum strictum	5.26	9	0.15	4	11.55	7	0.01
Scirpus cyperinus	5.08	14	0.69	4	20.04	43	2.93
Pohlia nutans	4.74	16	0.37	4	4.90	12	0.02
Lycopus uniflorus	4.68	14	0.49	4	1.63	20	0.93
Solidago rugosa	4.58	14	0.41	4	4.36	10	0.01
Galium trifidum	4.19	17	0.30	1	0.22	26	0.45
Viola macloskeyi	4.12	9	0.17	1	0.11	21	0.31
Drepanocladus aduncus	3.83	8	0.73	0	0	0	0
Calamagrostis canadensis	3.81	40	13.36	4	11.76	27	0.39
Glyceria canadensis	3.76	24	1.10	0	0	32	1.88
Triadenum virginicum	3.43	15	0.90	2	1.63	0	0
Salix spp.	3.33	46	3.70	3	3.20	38	0.47
Typha latifolia	3.23	17	5.29	2	0.98	0	0
Rubus idaeus	3.20	13	0.86	1	0.33	4	0.01

Carex canescens	3.17	16	2.25	0	0	18	0.94
Spiraea alba var. latifolia	3.08	41	8.13	3	8.28	9	0.01
Carex flava	3.02	5	0.68	0	0	0	0
Acer rubrum	2.94	12	0.80	0	0	0	0
Carex utriculata	2.77	14	1.03	0	0	0	0
Calliergon cordifolium	2.73	9	0.78	0	0	0	0
Carex intumescens	2.68	1	0.01	0	0	0	0
Calliergon stramineum	2.56	12	1.80	0	0	9	0.01
Alnus incana	2.54	44	11.90	4	0.87	0	0
Iris versicolor	2.45	20	1.11	0	0	0	0
Cornus sericea	2.35	22	1.44	1	0.05	0	0
Picea mariana	2.20	14	0.88	2	0.22	3	0.00
Calla palustris	2.12	16	1.45	0	0	0	0
Carex trisperma	2.08	16	0.71	0	0	0	0
Campylium stellatum	2.06	17	0.94	0	0	0	0
Sphagnum spp.	1.89	44	33.27	2	0.30	11	0.04
Carex stricta	1.81	9	3.75	0	0	0	0
Larix laricina	1.54	17	1.93	0	0	0	0
Potentilla fruticosa	1.42	7	0.65	0	0	0	0
Myrica gale	1.39	40	11.87	0	0	0	0
Carex trichocarpa	1.03	1	0.83	0	0	0	0
Chamaedaphne calyculata	0.98	25	7.49	1	0.05	0	0
Carex aquatilis	0.94	8	1.31	0	0	0	0
Warnstorfia exannulata	0.54	13	4.25	0	0	0	0
Carex vesicaria	0.28	1	1.17	0	0	0	0
Carex lasiocarpa	-0.43	3	1.17	0	0	0	0

4.4 Discussion

After two years, the vegetation at the disturbed sites still differed considerably from the natural fens in terms of the total fen plant cover and richness and plant community composition. This is not surprising since the disturbed sites are in the earliest stage of development whereas the natural fens in the region developed over thousands of years (Lortie 1983; Garneau 1998). The length of the monitoring period for restoration varies with the type of the wetland and the goals of the project. Wetland functions may need 15-20 years to establish, although peatlands and other wetland types may require longer (Mitsch & Wilson 1996). Fens may need between 20-100 years to develop functional equivalency due to their species rich systems, and special water quality requirements (Zedler & Callaway 1999). Kentula (2000) reminds us that existing projects are ecologically young and the final verdict on restoration success may be premature. She suggests using trajectories or performance curves to understand and evaluate restoration projects, and using adaptive management for systematically assessing and improving the performance of restored systems. This project is valuable because it establishes a broad range of wetlands for evaluating success, and documents the initial performance of the restoration project. The value of the project will increase with time as it develops into a long-term data set.

The disturbed sites were characterized with several weedy species such as *Rorippa palustris*, *Tussilago farfara*, *Polygonum hydropiper*, and *Juncus effusus*. Weeds are common on fen peat sites disturbed by peat mining; similar species have been observed on peat-mined sites in Finland and Ireland (Salonen 1990; Rowlands 2001). Evidence of successful establishment of three *Carices*, including *Carex canescens, Carex crawfordii*, and *Carex stipata* was observed at the restoration site, whereas these species were absent from the unrestored sites, except for one encounter with *Carex crawfordii* (Appendix B). The general absence of several fen species, particularly *Carices*, at the disturbed sites suggests that diaspore dispersal is constraining the development of a natural fen plant community. Similar observations have been made for during the restoration of freshwater marshes (Reinartz & Warne 1993) and prairie potholes (Galatowitsch & van der Valk 1996), and the reintroduction of *Carices* was considered necessary (Cronk & Fennessy 2001).

This study directly compared the environmental conditions of the restoration site to a variety of natural fens in the study region. The variety of natural fens sampled ensures that a range of temporal and spatial conditions are represented, providing a more comprehensive basis for comparison. In contrast, the more detailed comparison of the restoration site to donor fen sites (Chapter 2) was important to determine whether the site conditions were similar to those that support the donor species. Comparisons of the restoration with the regional natural fens yielded similar conclusions to those derived from the donor site comparisons. The water table was considerably lower than natural fens levels indicating that further rewetting measures are necessary to create fen-like hydrological setting at the restoration site. Similarly, solute concentrations were extremely high at the restoration site compared to the natural fens. This provides evidence that further measures to reduce solute concentrations are necessary, such as increasing hydrological inputs to hasten the flushing of cations from the peat substrate. Whereas comparisons with the donor fen sites suggested that the concentrations of nitrate and the electrical conductivity were too high at the restoration site, comparisons with the regional fens suggested that these concentrations are suitable for supporting a fen plant community. Therefore, management efforts to reduce nitrate levels at the restoration site should be given lower priority.

4.5 Conclusions

Initial monitoring of the restoration site indicates that the plant community is not yet restored. Total fen plant cover and richness were lower and several plant species at the restoration site were not observed in natural fens. However, several fen species did establish at the restoration site after two years, including several *Carex* species, which were not found on the unrestored sites. Direct comparisons of the environmental conditions at the sites indicates that further restoration measures are necessary to increase the water table level and lower solute concentrations at the restoration site. Further monitoring of the plant community is recommended to determine the effects of restoration treatments in the long term.

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5 Conclusions

Ultimately, restoration ecology strives to predict the outcomes of restoration actions; however the need for restoration guidelines has outpaced the science (Zedler 2000). Ecological principles should be sought using experimental approaches, and predictability should improve where the restoration context and specific restoration actions are held constant. Zedler (2000) recently proposed ten ecological principles that are often ignored in restoration research and suggested where more restoration research is needed. Here I will review how the current restoration project addressed some of these ecological principles and highlight information gaps for further investigation.

- *The specific hydrological regime is crucial to restoring biodiversity and function.*

The hydrological regime is widely regarded as the most important determinant of the establishment and maintenance of specific types of wetlands and wetland processes (Mitsch & Gosselink 1993). Fen restoration is particularly complex due to the variety of water regimes that exist naturally (e.g. limnogenous, soligenous) and the water quality of the water supply is critical to vegetation development (Charman 2002). Obtaining the correct water table levels and nutrient and base cations levels is therefore a careful balancing act that needs careful manipulation and monitoring to achieve particular abiotic conditions that will favour fen vegetation development (Charman 2002). Alterations to the residual depth of peat at the restoration site in the current study affected the water table depth and aspects of water quality. The establishment of fen plants was greatest on the intermediate terrace levels. This level may have promoted the establishment of fen plants because of its intermediate moisture regime (i.e. not too dry) and its intermediate mineral status (i.e. not too saline). The restoration site was clearly drier than natural fens in the study region, and additional restoration measures are required to create fen-like hydrological conditions. Further research is needed to understand the hydrological processes at sites with minerotrophic sedge peat exposed at the surface.

- *Seed banks and dispersal can limit the recovery of plant richness.*

The spontaneous colonization of fen plants on mined peat sites is constrained by a lack of suitable diaspores. The residual peat is devoid of plants and a viable seed bank (Salonen 1987), and natural areas surrounding mined peat sites in North America are typically bogs with few or no fen species present (Poulin *et al.* 1999). The immigration potential of herbaceous plants (a dominant component of most fens) to recolonize peat mined sites in Québec is low (Campbell *et al.* 2003). Reintroducting vegetation is considered necessary where a landscape is fragmented to the extent that seed dispersal from a source sites can no longer be transferred to the restoration site (Middleton 1999). Comparisons of the disturbed sites (restored and spontaneously revegetated sites) to natural fens in the study region showed that several species did not overlap. Several fen species did not occur at disturbed sites, particularly *Carices,* indicating that dispersal constraints limit spontaneous recolonization. Additionally, several weedy non-fen species were found on disturbed sites, such as *Tussilago farfara* and *Equisetum arvense*. Some native plants and many exotics are aggressive colonists. Longer term monitoring of the restoration site is required to determine the development of the plant community over time, especially to compare target fen species versus exotics. In terms of specific restoration actions to restore native plant biodiversity the introduction of fen plants with donor seed bank proved to be an effective method, increasing both total abundance and richness of fen species.

- *Predicting restoration begins with succession theory*

Restoration ecology attempts to return a degraded site to its historical trajectory (SER 2002). This study aimed to return a site degraded to an earlier stage in development (fen) to a historical plant community. The approach to restore minerotrophic peat surfaces to an earlier successional stage has been proposed by others (Wheeler & Shaw 1995; Wind-Mulder *et al.* 1996). However, fen restoration research is still in its infancy and few projects have attempted to do so (Charman 2002). Trajectory analysis and direct comparisons with natural fens indicates that restoration site has not yet been restored in this case study. Long term monitoring of the restoration site is important to determine the outcome of specific restoration actions relative to the target ecosystem.

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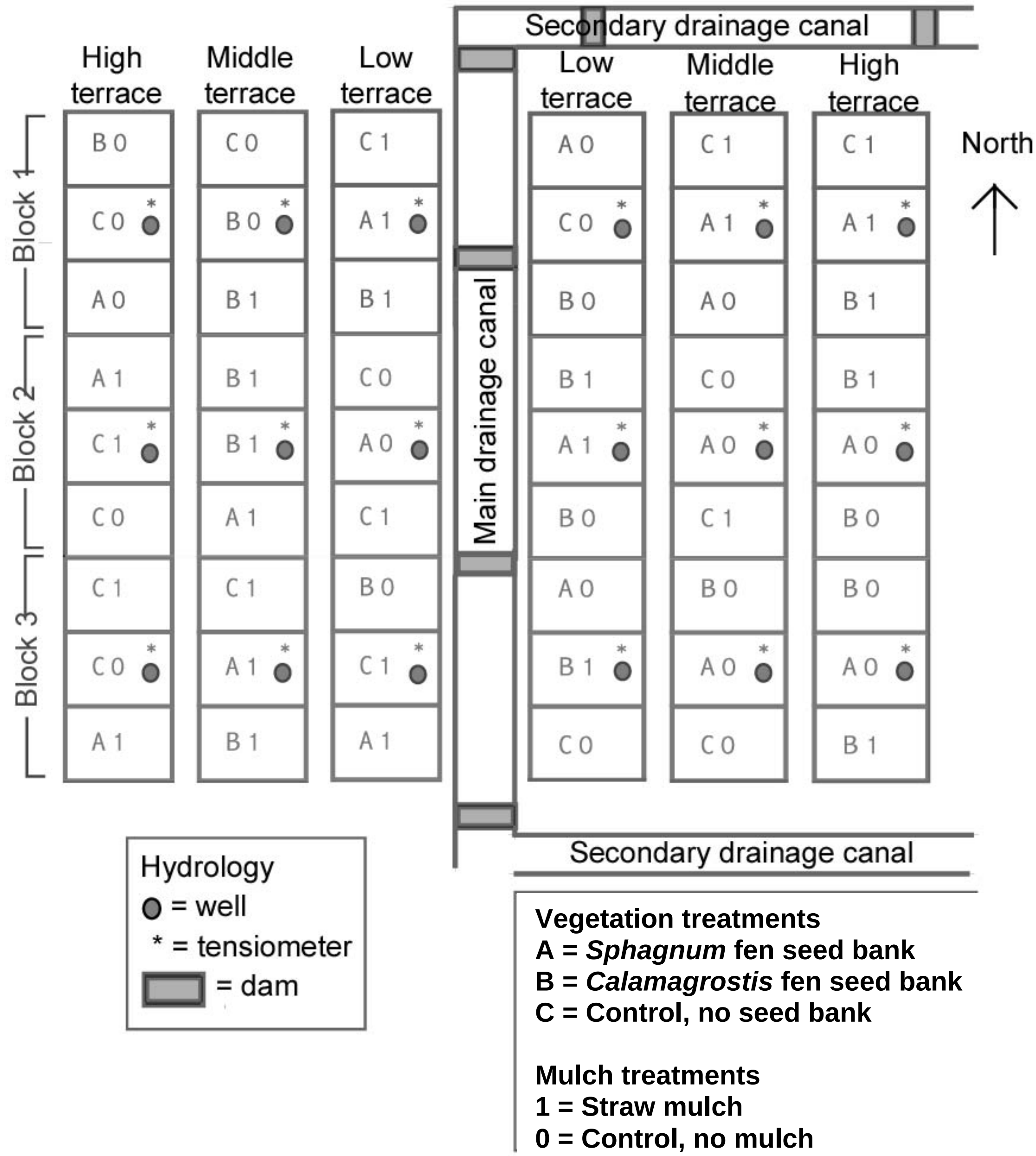
APPENDICES



Appendix A. Layout of terraces, vegetation and mulch treatments, and hydrological stations and features at the restoration site.

Appendix B Species list from surveys of natural fens, unrestored fens, and restored site 2002. The presence of species at each plot is indicated, and the cover (%). Rel = relative.

		Natural fens		Unrestored		Restored 2002	
Species name	**Authority**	**Presence n = 60**	**Cover (%)**	**Presence n = 4**	**Rel.cover (%)**	**Presence n = 54**	**Cover (%)**
Abies balsamea	(L.) Mill.	3	0.32	0	n.p.	0	n.p.
Acer rubrum	L.	12	0.80	0	n.p.	0	n.p.
Agrostis hyemalis	(Walt.) B.S.P.	6	0.11	4	9.26	49	5.34
Alnus incana	(L.) Moench.	44	11.90	4	0.87	0	n.p.
Amblystegium serpens	Schimp. in B.S.G.	1	0.01	0	n.p.	0	n.p.
Amelanchier bartramiana	(Tausch) Roemer	4	0.10	1	0.05	0	n.p.
Anaphalis margaritacea	(L.) Benth. & Hook.	2	0.01	2	0.98	11	0.01
Andromeda glaucophylla	Link	13	0.36	1	0.05	0	n.p.
Aralia hispida	Vent.	0	n.p.	1	0.05	0	n.p.
Aralia nudicaulis	L.	0	n.p.	1	0.05	0	n.p.
Aronia melanocarpa	(Michx.) Elliott	0	n.p.	1	0.05	0	n.p.
Aster nemoralis	Aiton.	1	0.02	0	n.p.	0	n.p.
Aster novae-angliae	L.	1	0.03	1	0.11	0	n.p.
Aster spp.	L.	4	0.06	0	n.p.	0	n.p.
Aster umbellatus	Mill.	7	0.33	2	0.98	4	0.00
Atrichum sp.	P. Beauv.	1	0.01	0	n.p.	0	n.p.
Aulacomnium palustre	(Hedw.) Schwaegr.	16	0.44	0	n.p.	1	0.00
Avena sativa	L.	0	n.p.	0	n.p.	3	0.03
Betula populifolia	Marsh.	0	n.p.	1	0.05	0	n.p.
Betula papyrifera	Marsh.	8	0.21	2	1.85	0	n.p.
Betula spp.	L.	0	n.p.	0	n.p.	13	0.01
Bidens cernua	L.	1	0.02	4	2.07	34	0.28
Brachythecium sp.	Schimp. in B.S.G.	1	0.01	0	n.p.	0	n.p.
Calamagrostis canadensis	(Michx.) Nutt.	40	13.36	4	11.76	27	0.39
Calla palustris	L.	16	1.45	0	n.p.	0	n.p.

Species name	Authority	Natural fens Presence n = 60	Natural fens Cover (%)	Unrestored Presence n = 4	Unrestored Rel.cover (%)	Restored 2002 Presence n = 54	Restored 2002 Cover (%)
Calliergon cordifolium	(Hedw.) Kindb.	9	0.78	0	n.p.	0	n.p.
Calliergon stramineum	(Brid.) Kindb.	12	1.80	0	n.p.	9	0.01
Caltha palustris	L.	2	0.03	0	n.p.	0	n.p.
Campylium hispidulum	(Brid.) Mitt.	1	0.01	0	n.p.	0	n.p.
Campylium polygamum	(Schimp. in B.S.G.) C. Jens.	4	0.04	0	n.p.	0	n.p.
Campylium stellatum	(Hedw.) C. Jens.	17	0.94	0	n.p.	0	n.p.
Carex aquatilis	Wahlendb.	8	1.31	0	n.p.	0	n.p.
Carex brunnescens	(Pers.) Poir.	3	0.28	0	n.p.	0	n.p.
Carex canescens	L.	16	2.25	0	n.p.	18	0.94
Carex crawfordii	Fern.	3	0.08	1	0.11	38	3.67
Carex crinita	Lam.	1	0.02	0	n.p.	0	n.p.
Carex disperma	Dewey	1	0.02	0	n.p.	0	n.p.
Carex echinata	Murray	5	0.15	0	n.p.	0	n.p.
Carex flava	L.	5	0.68	0	n.p.	0	n.p.
Carex interior	L. Bailey	6	0.68	0	n.p.	0	n.p.
Carex intumescens	Rudge	1	0.01	0	n.p.	0	n.p.
Carex lacustris	Willd.	1	0.50	0	n.p.	0	n.p.
Carex lasiocarpa	Ehrh. ex Hoffm.	3	1.17	0	n.p.	0	n.p.
Carex oligosperma	Michx.	4	0.25	0	n.p.	0	n.p.
Carex paupercula	Michx.	5	0.08	0	n.p.	0	n.p.
Carex pseudocyperus	L.	5	0.23	0	n.p.	0	n.p.
Carex utriculata	F. Boott	14	1.03	0	n.p.	0	n.p.
Carex stipata	Muhl.	9	0.21	0	n.p.	14	0.14
Carex stricta	Lam.	9	3.75	0	n.p.	0	n.p.
Carex trichocarpa	Muhl.	1	0.83	0	n.p.	0	n.p.
Carex trisperma	Dewey	16	0.71	0	n.p.	0	n.p.
Carex vaginata	Tausch	1	0.02	0	n.p.	0	n.p.

Species name	Authority	Natural fens Presence n = 60	Natural fens Cover (%)	Unrestored Presence n = 4	Unrestored Rel.cover (%)	Restored 2002 Presence n = 54	Restored 2002 Cover (%)
Carex vesicaria	L.	1	1.17	0	n.p.	0	n.p.
Carex sp.	L.	3	0.42	0	n.p.	10	0.05
Cerastium vulgatum	L.	0	n.p.	0	n.p.	8	0.15
Chamaedaphne calyculata	(L.) Moench	25	7.49	1	0.05	0	n.p.
Chelone glabra	L.	1	0.02	0	n.p.	0	n.p.
Chenopodium album	L.	0	n.p.	0	n.p.	7	0.01
Chrysanthemum leucanthemum	L.	0	n.p.	0	n.p.	2	0.04
Cicuta bulbifera	L.	15	0.21	0	n.p.	0	n.p.
Cirsium arvense	(L.) Scop.	0	n.p.	2	0.76	1	0.01
Cladopodiella fluitans	(Nees) Jörg.	0	n.p.	2	0.33	0	n.p.
Cladina sp.	Nyl.	1	0.01	0	n.p.	0	n.p.
Cladonia sp.	P. Browne	1	0.01	2	0.65	0	n.p.
Climacium dendroides	(Hedw.) Web. & Mohr	4	0.52	0	n.p.	1	0.00
Coptis trifolia var. *groenlandica*	(L.) Salisb. (0eder) Fasset.	1	0.01	0	n.p.	0	n.p.
Cornus canadensis	L.	2	0.03	0	n.p.	0	n.p.
Cornus sericea	L.	22	1.44	1	0.05	0	n.p.
Dicranella cerviculata	(Hedw.) Schimp.	0	n.p.	4	3.49	9	0.04
Dicranum polysetum	Sw.	3	0.05	0	n.p.	0	n.p.
Dicranum spp.	Hedw.	2	0.06	0	n.p.	0	n.p.
Dicranum undulatum	Brid.	4	0.13	0	n.p.	0	n.p.
Drepanocladus aduncus	(Hedw.) Warnst.	8	0.73	0	n.p.	0	n.p.
Drosera rotundifolia	L.	15	0.25	0	n.p.	0	n.p.
Dryopteris cristata	(L.) A. Gray.	6	0.13	0	n.p.	0	n.p.
Dryopteris spp.	Adans.	2	0.08	1	0.22	0	n.p.
Dryopteris carthusiana	(Villars) H.P. Fuchs	9	0.17	0	n.p.	0	n.p.

Species name	Authority	Natural fens Presence n = 60	Natural fens Cover (%)	Unrestored Presence n = 4	Unrestored Rel.cover (%)	Restored 2002 Presence n = 54	Restored 2002 Cover (%)
Dulichium arundinaceum	(L.) Britton	10	0.63	0	n.p.	0	n.p.
Eleocharis acicularis	(L.) Roemer & Schultes.	2	0.01	0	n.p.	0	n.p.
Eleocharis palustris	L.	7	0.23	1	0.11	0	n.p.
Epilobium angustifolium	L.	1	0.03	4	2.18	2	0.00
Epilobium glandulosum	Lehm.	2	0.01	1	0.05	34	0.22
Epilobium leptophyllum	Raf.	7	0.13	1	0.05	4	0.01
Equisetum arvense	L.	2	0.03	4	24.40	19	5.02
Equisetum fluviatile	L.	7	0.60	0	n.p.	0	n.p.
Equisetum sylvaticum	L.	1	0.02	0	n.p.	0	n.p.
Eriophorum polystachion	L.	4	0.12	0	n.p.	0	n.p.
Eriophorum tenellum	Nutt.	4	0.37	0	n.p.	0	n.p.
Eriophorum vaginatum	L.	1	0.02	1	0.05	0	n.p.
Eriophorum viridicarinatum	(Engelm.) Fern.	5	0.32	0	n.p.	0	n.p.
Erysimum cheiranthoides	L.	0	n.p.	0	n.p.	1	0.00
Eupatorium maculatum	L.	6	0.40	1	0.11	0	n.p.
Euthamia graminifolia	(L.) Nutt.	5	0.17	4	21.46	48	0.83
Fragaria virginiana	Duchesne	3	0.07	3	7.08	13	0.22
Fraxinus nigra	Marshall	2	0.02	0	n.p.	0	n.p.
Galeopsis tetrahit	L.	0	n.p.	0	n.p.	3	0.01
Galium aparine	L.	6	0.08	0	n.p.	0	n.p.
Galium labradoricum	(Wieg.) Wieg.	1	0.02	0	n.p.	0	n.p.
Galium tinctorium	L.	7	0.10	0	n.p.	0	n.p.
Galium trifidum	Michx.	17	0.30	1	0.22	26	0.45
Geum aleppicum	Jacq.	3	0.08	1	0.01	0	n.p.
Glyceria canadensis	(Michx.) Trin.	24	1.10	0	n.p.	32	1.88
Habenaria sp.	Willd.	1	0.02	0	n.p.	0	n.p.
Hamatocaulis vernicosus	(Mitt.) Hedenäs	1	0.08	0	n.p.	0	n.p.

Species name	Authority	Natural fens Presence n = 60	Natural fens Cover (%)	Unrestored Presence n = 4	Unrestored Rel.cover (%)	Restored 2002 Presence n = 54	Restored 2002 Cover (%)
Hepatic (undetermined)		6	0.14	3	0.76	1	0.00
Hieracium caespitosum	Dumort.	1	0.01	0	n.p.	0	n.p.
Hieracium spp.	L.	0	n.p.	3	8.82	36	0.58
Hippuris vulgaris	L.	2	0.17	0	n.p.	0	n.p.
Hypericum boreale	(Britton) E. Bickn.	2	0.03	0	n.p.	0	n.p.
Hypericum ellipticum	Hook.	4	0.14	0	n.p.	4	0.01
Hypericum sp.	L.	2	0.01	0	n.p.	0	n.p.
Callicladium haldanianum	(Grev.) Crum	1	0.01	0	n.p.	0	n.p.
Hypnum lindbergii	Mitt.	5	0.12	0	n.p.	0	n.p.
Ilex verticillata	(L.) A. Gray	1	0.00	0	n.p.	0	n.p.
Impatiens spp.	L.	5	0.10	1	0.65	0	n.p.
Iris versicolor	L.	20	1.11	0	n.p.	0	n.p.
Juncus brevicaudatus	(Engelm.) Fern.	4	0.08	4	11.11	20	0.22
Juncus bufonius	L.	0	n.p.	2	0.16	0	n.p.
Juncus compressus	Jacq.	0	n.p.	0	n.p.	3	0.00
Juncus effusus	L.	0	n.p.	4	5.77	16	1.43
Juncus filiformis	L.	1	0.03	2	2.40	0	n.p.
Juncus sp.	L.	4	0.06	4	3.49	7	0.21
Juncus tenuis	Willd.	0	n.p.	1	0.05	0	n.p.
Kalmia angustifolia	L.	10	0.22	1	0.11	0	n.p.
Kalmia polifolia	Wang.	7	0.10	0	n.p.	0	n.p.
Lactuca biennis	(Moench) Fern.	0	n.p.	1	0.05	0	n.p.
Larix laricina	(Du Roi) Koch	17	1.93	0	n.p.	0	n.p.
Ledum groenlandicum	Oeder	13	0.38	2	0.11	0	n.p.
Lemna sp.	L.	0	n.p.	1	0.76	0	n.p.
Leontodon autumnalis	L.	0	n.p.	1	0.05	0	n.p.
Leptodictyum humile	(P. Beauv.) Ochyra	3	0.38	0	n.p.	6	0.01
Lichen (undetermined)		5	0.04	1	0.22	0	n.p.

Species name	Authority	Natural fens Presence n = 60	Natural fens Cover (%)	Unrestored Presence n = 4	Unrestored Rel.cover (%)	Restored 2002 Presence n = 54	Restored 2002 Cover (%)
Lonicera villosa	(Michx.) R. & S.	10	0.28	0	n.p.	0	n.p.
Lycopus americanus	Muhl.	1	0.02	0	n.p.	0	n.p.
Lycopus uniflorus	Michx.	14	0.49	4	1.63	20	0.93
Lycopodium annotinum	L.	1	0.01	0	n.p.	0	n.p.
Lycopodium clavatum	L.	0	n.p.	1	0.05	0	n.p.
Lysimachia terrestris	(L.) B.S.P.	10	0.18	1	0.11	0	n.p.
Mentha arvensis	L.	1	0.03	0	n.p.	0	n.p.
Menyanthes trifoliata	L.	1	0.07	0	n.p.	0	n.p.
Moss (undetermined)		1	0.01	3	1.96	7	0.04
Mylia anomala	(Hook.) S. Gray	6	0.06	0	n.p.	0	n.p.
Myrica gale	L.	40	11.87	0	n.p.	0	n.p.
Nemopanthus mucronata	(L.) Trel.	6	0.34	0	n.p.	0	n.p.
Nuphar sp.	J.E. Smith	5	0.12	0	n.p.	0	n.p.
Oenothera perennis	L.	0	n.p.	0	n.p.	1	0.00
Oenothera biennis	L.	0	n.p.	1	0.05	0	n.p.
Oncophorus wahlenbergii	Brid.	1	0.01	0	n.p.	0	n.p.
Onoclea sensibilis	L.	6	0.40	2	0.44	0	n.p.
Osmunda cinnamomea	L.	3	0.12	0	n.p.	0	n.p.
Osmunda regalis	L.	1	0.03	0	n.p.	0	n.p.
Panicum dichotomiflorum	Michx.	0	n.p.	0	n.p.	6	0.12
Pellia sp.	Raddi	1	0.01	0	n.p.	0	n.p.
Phalaris arundinacea	L.	4	0.11	1	0.11	0	n.p.
Phleum pratense	L.	0	n.p.	0	n.p.	8	0.07
Picea mariana	(Mill.) B.S.P.	14	0.88	2	0.22	3	0.00
Plagiothecium denticulatum	(Hedw.) Schimp. in B.S.G.	5	0.08	0	n.p.	0	n.p.
Plantago major	L.	0	n.p.	0	n.p.	1	0.01
Pleurozium schreberi	(Brid.) Mitt.	10	0.55	1	0.05	0	n.p.

Species name	Authority	Natural fens Presence n = 60	Natural fens Cover (%)	Unrestored Presence n = 4	Unrestored Rel.cover (%)	Restored 2002 Presence n = 54	Restored 2002 Cover (%)
Pohlia nutans	(Hedw.) Lindb.	16	0.37	4	4.90	12	0.02
Polygonum amphibium	L.	0	n.p.	1	0.05	0	n.p.
Polygonum convolvulus	L.	0	n.p.	1	0.05	2	0.02
Polygonum hydropiper	L.	0	n.p.	3	3.27	38	0.67
Polytrichum commune	Hedw.	5	0.14	2	0.65	1	0.00
Polytrichum strictum	Brid.	9	0.15	4	11.55	7	0.01
Populus balsamifera	L.	0	n.p.	1	0.11	0	n.p.
Populaus tremuloides	Michx.	1	0.02	2	2.61	34	0.02
Potamogeton epihydrus	Raf.	1	0.05	0	n.p.	0	n.p.
Potamogeton pusillus	(L.)	1	0.02	0	n.p.	0	n.p.
Potamogeton sp.	L.	1	0.01	1	0.05	0	n.p.
Potentilla fruticosa	L.	7	0.65	0	n.p.	0	n.p.
Potentilla norvegica	L.	2	0.07	2	0.11	20	0.29
Potentilla palustris	(L.) Scop.	8	0.11	0	n.p.	0	n.p.
Prunus pensylvanica	L.f.	0	n.p.	1	0.05	0	n.p.
Prunus virginiana	L.	2	0.05	0	n.p.	0	n.p.
Ptilium crista-castrensis	(Hedw.) De Not.	2	0.03	0	n.p.	0	n.p.
Ranunculus gmelinii	DC.	0	n.p.	0	n.p.	3	0.03
Ranunculus pensylvanicus	L.f.	1	0.01	0	n.p.	0	n.p.
Rhamnus alnifolia	L'Hér.	3	0.01	0	n.p.	0	n.p.
Rhododendron canadense	(L.) Torr.	1	0.03	1	0.05	0	n.p.
Rhynchospora alba	(L.) Vahl.	2	0.12	0	n.p.	0	n.p.
Rhytidiadelphus loreus	(Hedw.) Warnst.	1	0.02	0	n.p.	0	n.p.
Rhytidiadelphus subpinnatus	(Lindb.) T. Kop.	1	0.01	0	n.p.	0	n.p.
Ribes glandulosum	Grauer	4	0.05	0	n.p.	0	n.p.
Ribes hirtellum	Michx.	1	0.05	0	n.p.	0	n.p.
Ribes lacustre	(Pers.) Poiret	6	0.07	0	n.p.	0	n.p.

Species name	Authority	Natural fens Presence n = 60	Natural fens Cover (%)	Unrestored Presence n = 4	Unrestored Rel.cover (%)	Restored 2002 Presence n = 54	Restored 2002 Cover (%)
Ribes sp.	L.	1	0.02	0	n.p.	0	n.p.
Rorippa palustris var. fernaldiana	(L.) Besser. (Butters & Abbe) Stuckey	0	n.p.	1	0.05	46	1.44
Rosa acicularis	Lindley	3	0.28	0	n.p.	0	n.p.
Rubus chamaemorus	L.	1	0.03	0	n.p.	0	n.p.
Rubus idaeus	L.	13	0.86	1	0.33	4	0.01
Rubus pubescens	Raf.	7	0.08	0	n.p.	0	n.p.
Rumex acetosella	L.	0	n.p.	4	2.29	5	0.11
Rumex crispus	L.	1	0.02	1	0.11	0	n.p.
Rumex orbiculatus	A. Gray	0	n.p.	1	0.05	2	0.00
Sagittaria latifolia	Willd.	8	0.23	0	n.p.	0	n.p.
Salix bebbiana	Sarg.	7	0.42	2	0.44	0	n.p.
Salix candida	Flüegge	0	n.p.	1	0.05	0	n.p.
Salix discolor	Muhl.	13	0.94	2	0.22	0	n.p.
Salix lucida	Muhl.	0	n.p.	1	0.05	0	n.p.
Salix petiolaris	J.E. Smith	4	0.49	0	n.p.	0	n.p.
Salix pyrifolia	Andersson	21	1.83	0	n.p.	0	n.p.
Salix eriocephala	Michx.	1	0.13	0	n.p.	0	n.p.
Salix spp.	L.	0	n.p.	3	2.40	38	0.47
Sanguisorba canadensis	L.	2	0.04	0	n.p.	0	n.p.
Sanionia uncinata	(Hedw.) Loeske	6	0.60	0	n.p.	0	n.p.
Sarracenia purpurea	L.	7	0.45	0	n.p.	0	n.p.
Scheuchzeria palustris	L.	2	0.34	0	n.p.	0	n.p.
Scirpus cyperinus	(L.) Kunth	14	0.69	4	20.04	43	2.93
Scirpus microcarpus	C. Presl.	13	0.35	4	1.53	0	n.p.
Scorpidium scorpioides	(Hedw.) Limpr.	2	0.03	0	n.p.	0	n.p.
Scutellaria galericulata	L.	12	0.25	0	n.p.	8	0.02

Species name	Authority	Natural fens Presence n = 60	Natural fens Cover (%)	Unrestored Presence n = 4	Unrestored Rel.cover (%)	Restored 2002 Presence n = 54	Restored 2002 Cover (%)
Scutellaria lateriflora	L.	1	0.01	0	n.p.	0	n.p.
Secale cereale	L.	0	n.p.	0	n.p.	1	0.00
Senecio schweinitzianus	Nutt.	5	0.13	0	n.p.	0	n.p.
Sium suave	Walter	1	0.03	0	n.p.	0	n.p.
Smilacina trifolia	(L.) Desf.	5	0.12	0	n.p.	0	n.p.
Solidago canadensis	L.	1	0.02	2	0.54	3	0.00
Solidago rugosa	Miller	14	0.41	4	4.36	10	0.01
Solidago uliginosa	Nutt.	7	0.33	0	n.p.	0	n.p.
Sonchus arvensis	L.	0	n.p.	1	0.54	0	n.p.
Sorbus aucuparia	L.	0	n.p.	2	0.11	0	n.p.
Sparganium chlorocarpum	Rydb.	5	0.35	4	3.59	0	n.p.
Sparganium minimum	(Hartman) Fries	2	0.07	0	n.p.	0	n.p.
Sparganium sp.	L.	3	0.03	0	n.p.	0	n.p.
Spergula arvensis	L.	0	n.p.	0	n.p.	1	0.00
Sphagnum angustifolium	(C. Jens. ex Russ.) C. Jens. in Tolf	2	1.50	0	n.p.	0	n.p.
Sphagnum capillifolium	(Ehrh.) Hedw.	10	0.96	1	0.11	0	n.p.
Sphagnum centrale	C. Jens. in Arnell & C. Jens.	16	4.75	0	n.p.	2	0.00
Sphagnum cuspidatum	Ehrh. ex Hoffm.	2	1.29	0	n.p.	0	n.p.
Sphagnum fallax	(Klinggr.) Klinggr.	4	0.86	0	n.p.	0	n.p.
Sphagnum fimbriatum	Wils. in Wils. & Hook. f. in Hook. f.	12	1.10	0	n.p.	0	n.p.
Sphagnum flexuosum	Dozy & Molk.	15	4.06	0	n.p.	7	0.03
Sphagnum fuscum	(Schimp.) Klinggr.	4	0.19	0	n.p.	0	n.p.
Sphagnum girgensohnii	Russ.	15	2.63	0	n.p.	0	n.p.
Sphagnum magellanicum	Brid.	17	5.68	0	n.p.	0	n.p.
Sphagnum majus	(Russ.) C. Jens.	1	0.05	0	n.p.	0	n.p.

Species name	Authority	Natural fens Presence n = 60	Natural fens Cover (%)	Unrestored Presence n = 4	Unrestored Rel.cover (%)	Restored 2002 Presence n = 54	Restored 2002 Cover (%)
Sphagnum papillosum	Lindb.	3	0.50	1	0.05	0	n.p.
Sphagnum platyphyllum	(Lindb. ex Braithw.) Sull. ex Warnt.	1	0.03	0	n.p.	0	n.p.
Sphagnum pulchrum	(Lindb. ex Braithw.) Warnst.	10	2.22	0	n.p.	0	n.p.
Sphagnum riparium	Ångstr.	6	1.37	0	n.p.	0	n.p.
Sphagnum rubellum	Wils. in Wils. & Hook. f. in Hook. f.	9	1.75	0	n.p.	0	n.p.
Sphagnum russowii	Warnst.	9	0.36	0	n.p.	0	n.p.
Sphagnum squarrosum	Crome	9	0.17	0	n.p.	2	0.01
Sphagnum subsecundum	Nees in Sturm	6	0.22	0	n.p.	0	n.p.
Sphagnum teres	(Schimp.) Ångstr. in Hartm.	17	3.33	0	n.p.	0	n.p.
Sphagnum warnstorfii	Russ.	8	0.25	0	n.p.	0	n.p.
Spiraea alba var. latifolia	Duroi (Aiton) Dippel	41	8.13	3	8.28	9	0.01
Spiranthes romanzoffiana	Cham.	0	n.p.	1	0.05	0	n.p.
Stellaria graminea	L.	0	n.p.	0	n.p.	9	0.05
Taraxacum officinale	Weber ex Wiggers	0	n.p.	0	n.p.	20	0.05
Tetraphis pellucida	Hedw.	1	0.01	0	n.p.	0	n.p.
Thalictrum dioicum	L.	5	0.13	0	n.p.	0	n.p.
Thelypteris palustris	Schott.	1	0.02	0	n.p.	0	n.p.
Thuja occidentalis	L.	4	0.07	0	n.p.	0	n.p.
Triadenum virginicum	(L.) Raf.	15	0.90	2	1.63	0	n.p.
Trientalis borealis	Raf.	3	0.07	0	n.p.	0	n.p.
Trifolium sp.	L.	0	n.p.	0	n.p.	5	0.00
Tussilago farfara	L.	0	n.p.	1	0.22	39	4.86
Typha latifolia	L.	17	5.29	2	0.98	0	n.p.

Species name	Authority	Natural fens Presence n = 60	Cover (%)	Unrestored Presence n = 4	Rel.cover (%)	Restored 2002 Presence n = 54	Cover (%)
Utricularia intermedia	Hayne.	3	0.04	0	n.p.	0	n.p.
Utricularia minor	L.	2	0.01	0	n.p.	0	n.p.
Utricularia spp.	L.	2	0.02	0	n.p.	0	n.p.
Utricularia vulgaris	L.	4	0.36	0	n.p.	0	n.p.
Vaccinium angustifolium	Aiton	6	0.08	1	0.11	0	n.p.
Vaccinium macrocarpon	Aiton	3	0.28	0	n.p.	0	n.p.
Vaccinium oxycoccus	L.	5	0.12	1	0.05	0	n.p.
Viburnum nudum var. cassinoides	(L.) T. & G.	7	0.51	0	n.p.	0	n.p.
Vicia cracca	L.	1	0.03	0	n.p.	0	n.p.
Viola macloskeyi	Lloyd	9	0.17	1	0.11	21	0.31
Warnstorfia exannulata	(Schimp. in B.S.G.) Loeske	13	4.25	0	n.p.	0	n.p.
Warnstorfia fluitans	(Hedw.) Loeske	3	0.05	0	n.p.	0	n.p.