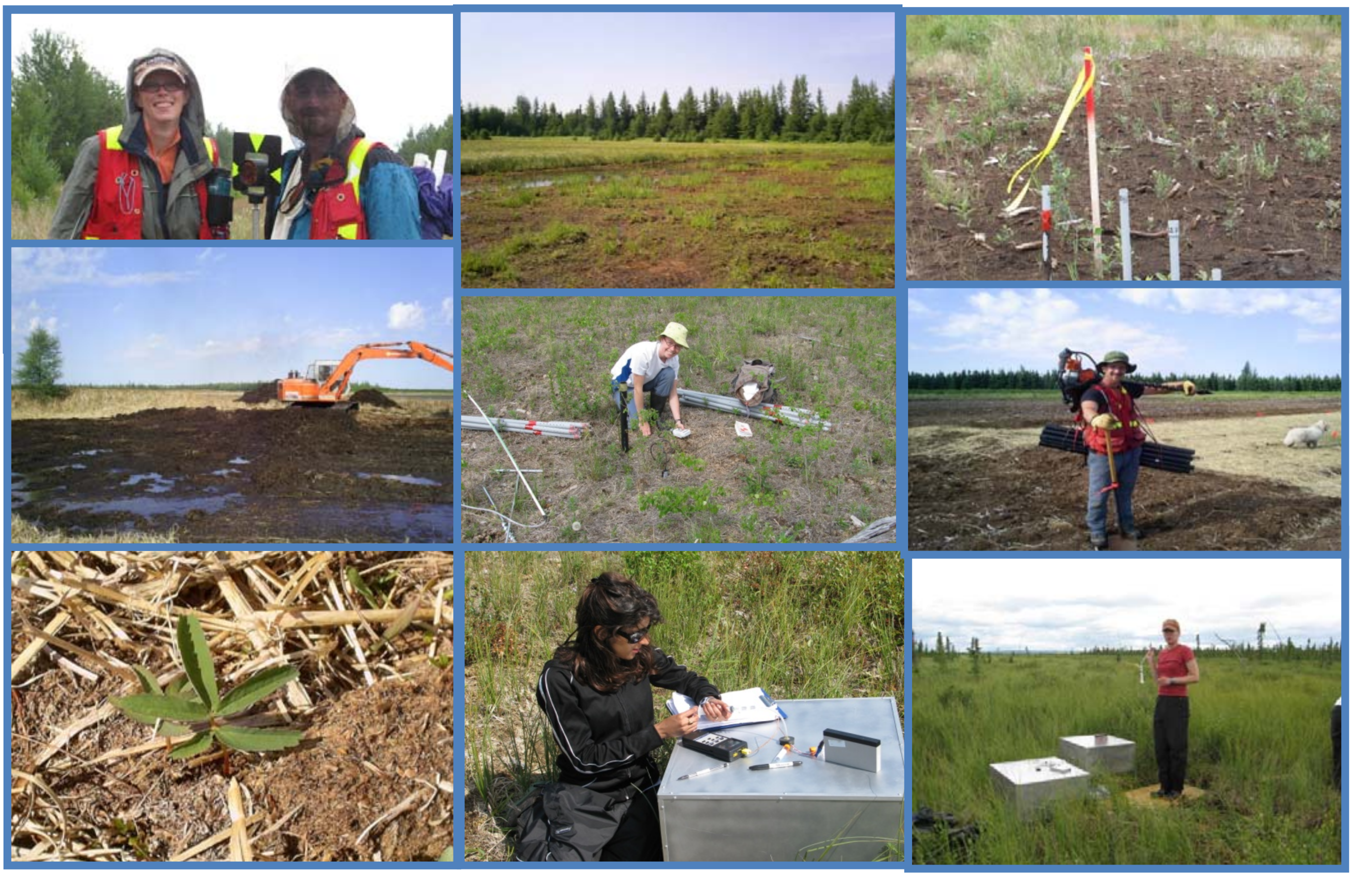
Restoring peatlands in Alberta: a case study of Evansburg North



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Prepared for Sun Gro Horticulture

December 2009

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Groupe de recherche en écologie des tourbières

Peatland Ecology Research Group

NSERC'S INDUSTRIAL RESEARCH CHAIR IN PEATLAND MANAGEMENT

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Executive summary

In western Canada peatlands are being disturbed by the energy, forestry and peat industrial sectors. However, no comprehensive studies have been undertaken to develop restoration techniques for western peatlands. Although extensive work has been carried out in eastern Canada, it is unknown if these techniques will be applicable in the sub-humid climate of western Canada. This project will be one of the first monitored peatland restorations in western Canada. The aim of this report is to establish baseline information on the Evansburg North peatland to provide a reference before a large-scale restoration is carried out. The vegetation, hydrology as well as carbon and methane dynamics were studied. Additionally, a field-scale experiment was carried out to test various reintroduction techniques.

The vegetation has developed greatly in the last 5 years, increasing from 5 to 25% cover. The majority of these plants are facultative or obligate wetland plants. Areas where vegetation was reintroduced as part of the field-level experiment had a higher vegetation cover (50%). The most successful recolonizer of the site is *Salix discolor*, a species common to shrubby fens. However, in comparison to undisturbed fens, bryophytes are not present on the cutaway site. Reintroduction techniques should concentrate on reintroducing bryophyte species due to their acknowledged role in C accumulation.

The field-scale experiment tested four reintroduction techniques: 1) diaspores introduction, 2) diaspores introduction with fertilization, 3) fertilization only, and, 4) no planting or fertilization. The treatments where propagule was introduced (both with and without fertilizer) showed a significantly higher vegetation cover as well as species richness. Although bryophytes were not present in the donor propagule material, early succession moss species did establish within 2 years of restoration.

A study of the hydrology of the Evansburg North site revealed that the area in the northwest corner of the site, close to the blocked drainage canal, showed a wetness that was closest to the hydrology of a nearby undisturbed fen. A more thorough blocking of the drainage ditches at 75 meter intervals would improve the hydrology of the site.

A study of carbon fluxes showed that a nearby undisturbed fen was a larger sink of CO_2 than the cutaway peatland. Emission of CO_2 via ecosystem respiration as maximized when volumetric water content was 55%. Restoration measures should try to increase peat moisture to above this threshold. Methane fluxes were much higher on the undisturbed site than on the cutaway peatland. Raising the water table to -10 to -30 cm should maximize CO_2 sequestration, while keeping the methane emissions one order of magnitude lower than the undisturbed fen.

The authors suggest that a large-scale restoration of the site should include a more comprehensive blocking of drainage ditches. A donor site which has a dominant bryophyte vegetation layer is suggested. Additionally, the authors suggest testing three reintroduction strategies: 1) removal of spontaneous vegetation and then introducing diaspores, 2) introducing diaspores without removing existing vegetation, and 3) no reintroduction of diaspores.

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1. Introduction

Peatland restoration in Canada has focused on restoring peatlands of the Eastern provinces of Québec and New Brunswick (Quinty and Rochefort 2003; Rochefort *et al.* 2003). However, peatlands are currently being disturbed through peat, forestry and energy industries in Alberta (Turchenek 1990; Turetsky and St. Louis 2006) and restoration techniques adapted to the western climate need to be developed. Disturbed peatlands are often not restored because it is believed that peatland restoration is not feasible in Alberta's sub-humid climate (Oil Sands Wetland Working Group 2000; Reclamation Criteria Advisory Group 2008). Can the restoration techniques developed for the humid climate of the Eastern provinces be applied to sub-humid Albertan peatlands?

Alberta peatlands also differ from their Eastern counterparts because they are younger and peat deposits are relatively shallow (Kuhry *et al.* 1993). Fens, minerotrophic peatlands, are common (Vitt *et al.* 1996). Therefore, fens are more likely to be affected by disturbances and mined peatlands are more likely to have shallow residual peat. Being able to restore fen vegetation communities is important to the Albertan context. Although some fen restoration projects have been carried out in North America (Cooper and MacDonald 2000; Cobbaert *et al.* 2004; Graf and Rochefort 2008), fen restoration has not been attempted in the boreal plains.

The overarching goal of this study is to conduct a baseline study of vegetation, hydrology and carbon dynamics of a cutaway peatland before large-scale restoration of a fen vegetation community. This data will be used to understand restoration success and failures and to document changes brought about by restoration. A field-level experiment was also conducted to test the application of the *Sphagnum* moss layer transfer technique, used to restore peatlands of Eastern Canada. Additionally, this data will be used to create restoration strategies for western peatlands as well as suggestions for the large-scale restoration of this site.

1.1 Format of document

This report is a collaborative approach from several researchers. Each topic will be described in separate sections. Martha Graf conducted the research on the vegetation and hydrology, Dave Critchley carried out an experiment on reintroduction techniques, and Maria Strack conducted the carbon dioxide and methane exchange research. This report is intended for internal use only as it contains unpublished data and all authors retain their intellectual ownership over results shown in this report. The authors brainstormed on suggestions for a large-scale restoration of the site. Because all studies deal with the same site, a site description will be included in the introduction of this document.

1.2 Site description

The Evansburg North (53°38'N 115°06'W) site is located in the province of Alberta, *circa* 100 km west of Edmonton. It is located in the ecoregion boreal mixed wood and in the subregion moist mixed wood (Strong and Leggat 1981). The moist subregion has mean summer temperatures that range from 10.5 to 13.5 °C. This zone is sub-humid, meaning evaporation exceeds precipitation (Devito and Mendoza 2007). Climatic moisture deficits in the boreal mixedwood zone are high with an annual water deficit between 0 and 200 mm (Strong and Leggat 1981). The climate is characterized by dry autumns and winters and wet summers. 70% of the year's precipitation falls within the summer, where July is usually the wettest month (Strong and Leggat 1981).

The environmental parameters (shown in Table 1.1) indicate that the residual peat corresponds to a moderate-rich fen (Gorham and Janssens 1992; Vitt 2006). A macroanalysis of the residual peat showed that 37% was ligneous residue, 32% *Drepanocladus* (brown moss species), 18% *Sphagnum*, 10% roots, and 3% from the sedge family (Cyperaceae) (Graf *et al.* 2008), which also corresponds to a moderate-rich fen. The site is approximately 70 ha in area and mining activities ceased in 1999. The points sampled in sections 1, 3 and 4 are shown in figure 1.2 and the GPS coordinates for these points are listed in appendix 3.

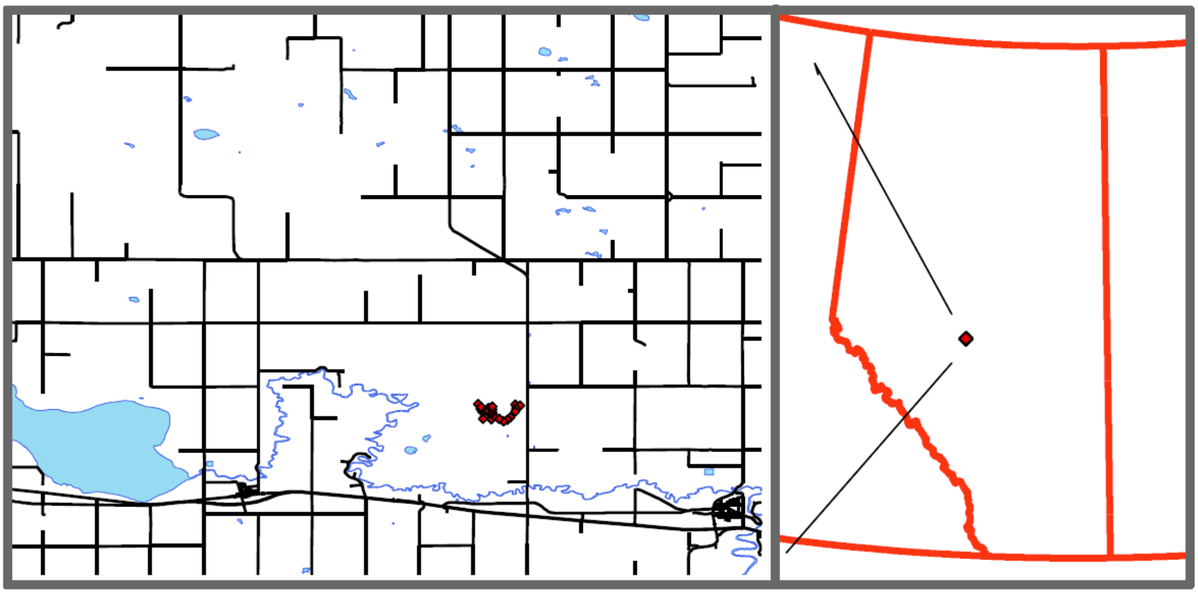

Figure 1.1: Location of Evansburg North in the province of Alberta (right) and on a regional map (left).

Table 1.1: Mean physicochemical data (±SE) of for Evansburg peatland sampled in August 2005 (n=25). See Graf *et al.* (2008) for details about methodology.

Parameter	Units	Average	SD
water table	(cm)	-80.1	53.7
peat depth	(cm)	134.3	36.7
vanPost		7	0
pH		5.1	0.3
electrical conductivity	(μS cm^{-1})	368.8	351.0
bulk density	(g cm^{-3})	0.107	0.030
P_{sol}	(mg kg^{-1})	9.6	2.5
Ca	(mg g^{-1})	9.7	1.2
Mg	(mg g^{-1})	2.4	0.2
Na	(mg g^{-1})	0.7	0.14

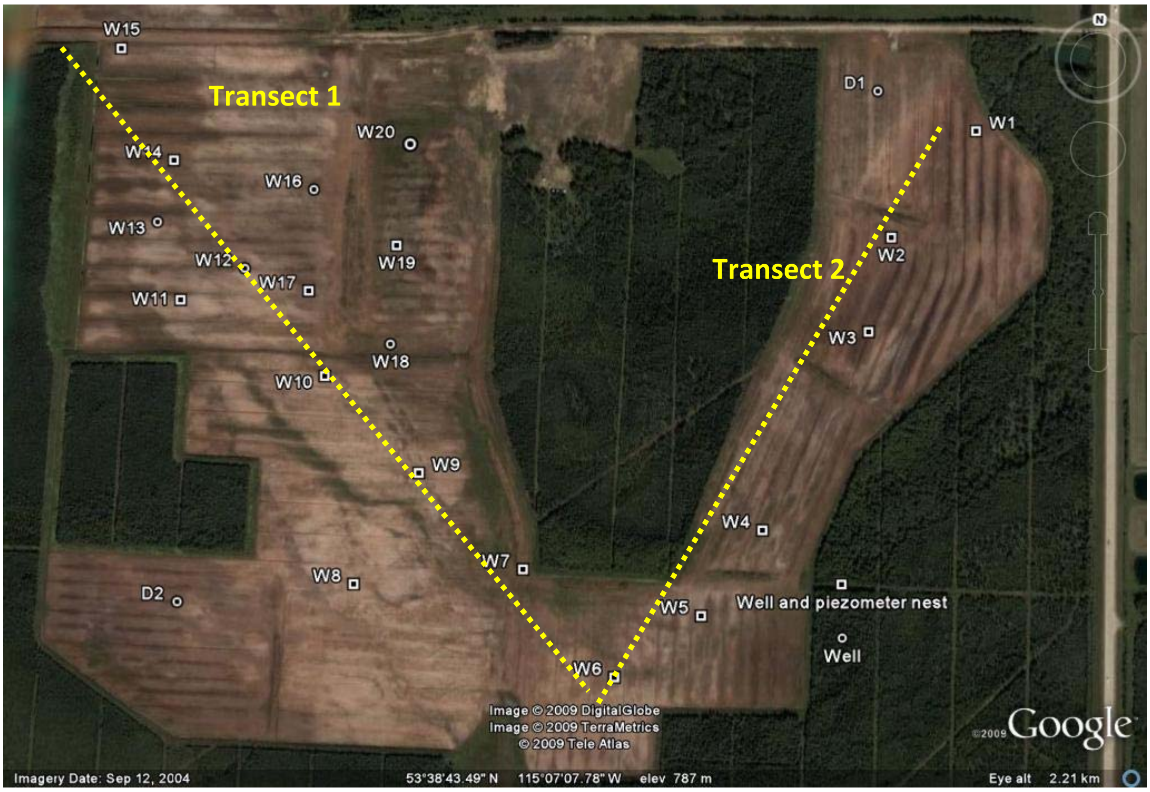

Figure 1.2: Location of points sampled for hydrology, vegetation as well as carbon and methane dynamics. These points were located along two transects.

2. Vegetation

2.1 Introduction

Vegetation plays an essential role in the restoration process of peatlands because the ecological functions of the top peat layers depend on the species composition. Therefore, the establishment of the appropriate species is imperative for the return of the ecosystem functions.

Cutover peatlands, which have ombrotrophic or bog residual peat, can remain void of vegetation for decades (Lavoie and Rochefort 1996). However, cutaway peatlands, where the residual peat is minerotrophic or sedge peat, are rapidly recolonized by vegetation. Famous *et al.* (1991) and Graf *et al. (*2008) found that cutaway peatlands revegetated significantly faster than cutover peatlands. Famous *et al.* (1991) and Graf *et al. (*2008) showed that richer, more humid sites are colonized more quickly than drier, poorer sites. Graf *et al.* (2008) found that, although wetland vascular plants do readily colonize cutaway peatlands, *Carex* species and fen bryophytes, the dominant vegetation groups in natural fens, do not.

The objectives of this study were to:

- Follow the successional development of a cutaway peatland site, and
- Identify which fen/wetland species have spontaneously recolonized the site.

2.2 Methods

The vegetation of the Evansburg North site was surveyed in 2005, 5 years after abandonment and again in 2009, nine years after abandonment. In 2005, 25 1-m^2 quadrats were equidistantly sampled across each site along transects arranged in a "W", which ensured that borders as well as the center of the fen was sampled. In 2009, twenty-two 1-m^2 quadrats were randomly sampled within a 5 m radius from each hydrological station (see Figure 1.2). Of the 22 sampled, four were in areas where vegetation had been reintroduced. Each species and its percentage cover (to the nearest 2% for covers less than 10% and to the nearest 5% for covers greater than 10%) were noted within each 1-m^2 quadrat. The nomenclature and wetland indicator status used for the vegetation follows USDA plants database (Table 2.2).

2.3 Results

Five years after peat extraction ceased, very little vegetation had spontaneously recolonized the site (Figure 2.1). The percentage of vegetation cover increased from near 5 % five years after abandonment to circa 25% nine years after abandonment without the re-introduction of vegetation. The four plots where vegetation had been introduced had the highest vegetation cover, which was close to 50%. The plants that spontaneously recolonized the site were mainly obligate or facultative wetland species (Figure 2.2).

The Evansburg North site has been mainly recolonized by shrub and tree species (Table 2.1). The most dominant species on the site is *Salix discolor*, a common shrub found in fens. *Carex aquatilis*, a species found frequently in natural fens, was the most frequently found herb. *C. aquatilis*, not found on the site 5 years after abandonment, was able to quickly colonize and spread across the site. Appendix 1 includes a complete list of the species found in the quadrats and appendix 2 lists additional species observed on the site which were not found in the systematic surveyed quadrats. The species lists shown in Table 2.1, Appendix 1, and 2 indicate a high diversity of plants. In total 58 species were observed 9 years after abandonment.

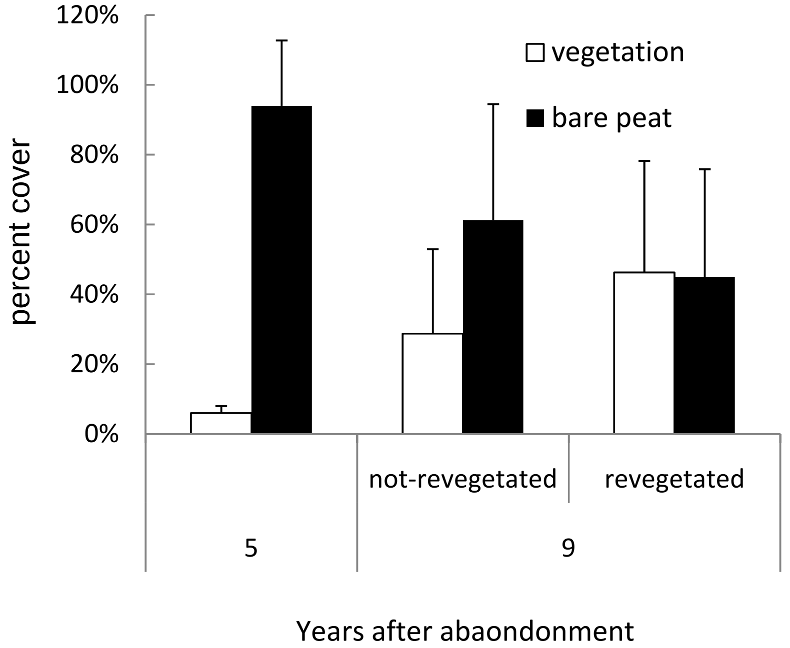

Figure 2.1 Average percentage cover of vegetation for quadrats sampled 5 years after abandonment (n=25) and 9 years after abandonment. For the quadrats sampled 9 years after abandonment, some of the quadrats were located in experimental plots were vegetation had been reintroduced (revegetated, n=4). Most were in areas where no vegetation was reintroduced (not-revegetated, n=18).

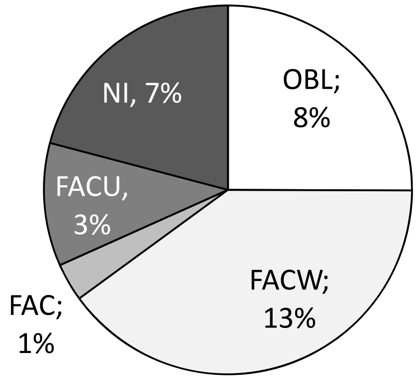

Figure 2.2: Pie graph showing the percent cover of different wetland classes of vegetation found on the site in 9 years after abandonment. See Table 2.2 for a description of the classes. The total percentage cover was approximately 30%.

Table 2.1: List of species observed in quadrats 5 and 9 years after abandonment. Species are arranged in decreasing order of frequency. Codes used to describe the wetland indicator status are shown in Table 2.3. A complete list with botanical authorities is shown in the appendix 1.

				Years after abandonment			
				9		5	
Latin name	**Gen.spe**	**Common name**	**Wetland Indicator Status**	**Frequ-ency**	**% cover**	**Frequ-ency**	**% cover**
Salix discolor	Sal.dis	Pussy willow	FACW	15	6.05	7	2.85
Betula papyrifera	Bet.pap	Paper birch	FACU	9	2.41	6	0.25
Populus tremuloides	Pop.tre	Quaking aspen	NI	8	2.09	7	0.71
Carex aquatilis	Car.aqu	Water sedge	OBL	7	3.36		
Chamerion angustifolium ssp. angustifolium	epi.ang	Fireweed	FAC	7	0.78	13	0.92
Deschampsia cespitosa	Das.cae	Tufted hairgrass	FACW	6	3.19	7	0.23
Picea mariana	Pic.mar	Black spruce	FACW	6	0.20	3	0.08
Salix petiolaris	Sal.pet	Meadow willow	OBL	4	2.18	1	0.08
Fragaria virginiana	fra.vir	Viginia strawberry	FACU	3	0.10	1	0.02
Geum macrophyllum	Geu.mac	Largeleaf avens	FACW	3	0.15		
Hieracium umbellatum	Hie.umb	Narrowleaf hawkweed	NI	3	0.10		
Lycopus uniflorus	Lyc.uni	northern bugleweed	OBL	3	0.32		
Sonchus arvensis	Son.arv	field sowthistle	FAC	3	0.24		
Calamagrostis canadensis	Cal.can	bluejoint	FACW	2	0.36		
Calamagrostis stricta ssp. inexpansa	cal.str	northern reedgrass	FACW	2	1.36	5	0.52

Table 2.2: Description of the wetland indicator codes used in Fig. 2.2 and Table s 2.1, appendix 1 and 2 (USDA 2009).

Indicator Code	Wetland Type	Comment
OBL	Obligate Wetland	Occurs almost always (estimated probability 99%) under natural conditions in wetlands.
FACW	Facultative Wetland	Usually occurs in wetlands (estimated probability 67%-99%), but occasionally found in non-wetlands.
FAC	Facultative	Equally likely to occur in wetlands or non-wetlands (estimated probability 34%-66%).
FACU	Facultative Upland	Usually occurs in non-wetlands (estimated probability 67%-99%), but occasionally found on wetlands (estimated probability 1%-33%).
NI	No indicator	Insufficient information was available to determine an indicator status.

2.4 Implications for restoration

The percentage cover of vegetation has risen significantly over the last four years. This may be due to the blocking of the main drainage canal and the subsequent partial restoration of the site's hydrology (see section 3 for a detailed description). Graf *et al.* (2008) and Famous *et al.* (1991) found that restoring hydrology was the most important factor in aiding the natural succession of a cutaway peatland. Keddy (1999) found that water level was the most important factor controlling wetland composition. The natural revegetation of the Evansburg North site has been developing towards a community of wetland plants. The diversity of the site is high, as is often observed in early succcessional sites, because pioneer species as well as wetland species have colonized the site.

When this community is compared to undisturbed fens, the main vegetation group missing is fen bryophytes. This same trend was observed on 28 cutaway peatlands in North America (Graf *et al.* 2008). Restoration measures should aim to reintroduce fen bryophytes to the site, as they are believed to be crucial to the return of important ecosystem processes (Rochefort 2000).

3. Field experiments for restoration techniques

3.1 Introduction

In North America, peat is extracted for horticultural purposes or disturbed by forestry and energy resource extraction (Turetsky and St. Louis 2006). These disturbances result in drainage, compaction, removal of the acrotelm, and loss or disturbance of the catotelm (Ferland and Rochefort 1997; Wind-Mulder et al. 1996). The extraction process removes several meters of peat substrate which changes the successional position from an ombrotrophic bog prior to extraction to a minerotrophic fen (Wind-Mulder *et al.* 1996; Graf *et al.* 2008). Extraction of peat changes the hydrology and chemistry of each site in a unique fashion. A cutaway peatland is devoid of a seed bank and is therefore colonized by ex-situ pioneer species (Wind-Mulder *et al.* 1996), which may result in inefficient or unsuccessful regeneration of peat extracted sites.

Restoration methods, including blocking drainage and introducing plant diaspores, restarts the wetland successional pathway and facilitates vegetation establishment. The Evansburg North site displayed vegetation characteristics similar to cutover peatlands immediately post abandonment. Prior to anthropogenic intervention, moisture content, surface oxidation and local environmental chemistry spatially restricted most species establishment and regeneration to oxidization cracks and drainage ditches as found in cut over peatlands (Figure 3.1) (Salonen 1987; Campbell *et al.* 2002, Groenveld and Rochefort 2002; Waddington and McNeil 2002; and Price *et al.* 2003). Considering these challenges; there is need for active management of abandoned peatlands (Rochefort *et al.* 2003) and restoration techniques need to be evaluated to ensure end goals are met within reasonable timelines (Lavoie and Rochefort 1996; Bugnon *et al.* 1997) The introduction of plant diaspores is intended to expedite the establishment process and enhance the plant diversity of the site.

Figure 3.1: Pre-restoration vegetation along lateral drainages and within surface oxidation cracks, Evansburg North.

This project targeted wet meadow establishment to provide an option for areas not suitable to bog restoration. A wet meadow is grassland with waterlogged soil near the substrate surface that lacks standing water for the majority of the year (Mitsch and Gosselink 2000) and contains minimal bryophyte diversity. The research site is suitable for development as a wet meadow to assist in the preliminary recovery of the degraded ecosystem. The establishment of a wet meadow is hoped to assist in the establishment of vascular fen species and provide suitable cover to nurse bryophyte fen species given time.

To meet the ecological requirements for a wet meadow, a drainage canals was blocked in conjunction with a modified *Sphagnum* moss layer transfer method (Quinty and Rochefort 2003). We conducted a study to determine the establishment success of wet meadow species and the effect of farm-grade fertilization on a cutaway peatland.

3.2 Methods

Historically, research focused on the cutover test areas of Eastern Canada; leaving limited quantitative information available regarding specific restoration success and potential for cutaway Canadian peatlands (but see Wind-Mulder 1998). A modified version of the *Sphagnum* moss layer transfer method for wet meadow establishment was employed in this project. The primary modification was a shift in target species from bryophytes to vascular species to meet the goals of wet meadow establishment. The secondary modification was a shift in fertilizer from a rock phosphate that assists in the establishment and perpetuation of *Polytrichum strictum* which is a *Sphagnum* nurse species (Sottocornola et al. 2007) to a 20-10-10-10 (NPKS) mixture intended to favour herbaceous establishment, while not overloading the system with nitrogen.

The donor vegetation site was selected at 53° 38' 25.774" N 115° 7' 12.699" W which is centrally located and within the Evansburg North lease (Figure 3.2). The donor site was a graminoid wet depression not directly impacted by peat extraction activities. The dominant vegetation included *Carex* sp. and various facultative wetland species with a lag of *Salix discolor* and *Picea mariana*.

Figure 3.2: Wet depression donor site and donor materials removed and piled for transport to experimental trials 2006.

The effect of plant reintroduction and fertilization treatments were tested using a factorial randomized unbalanced design repeated six times. The physical treatments included (1) plant reintroduction with fertilization (n=10), (2) plant introduction only (n=8), (3) fertilizer introduction only (n=9), and (4) a control where no plant or fertilizer was introduced (n=8). Treatments were used to test the effect of planting versus no-planting, fertilization versus no-fertilization and to determine if there is interaction of any significant level between the two factors. The plants were introduced at a ratio of approximately 1:15, and spread to a mean depth of three centimetres; fertilizer treatments were applied at a dose of 17g m^{-2}. Additionally, the interaction between fertilization and planting was investigated within the treatment structure. Mulch cover was standard on all treatment areas. The straw mulch cover involved mechanical and manual spreading of Barley straw (*Hordeum vulgare*) at a rate consistent with Quinty and Rochefort's (2003) recommendation.

To evaluate the success of establishment, vegetation cover was visually estimated within 1 m^2 quadrats. Cover percentage was estimated for each individual species thus layers superimposed over the sampling quadrat; consequently, percent cover can reach values greater than 100% in the analysis. One and two-years post-restoration, 16 quadrats per experimental unit were evaluated, in the third year it was reduced to 8 quadrats. Sampling followed the Braun-Blanquet (1965) method of cover sampling to ensure consistency between observers and days. Vegetation sampling points were located within the perimeter of the experimental unit by a minimum of 0.5 m to reduce bias based on treatment application and edge effects.

Final analysis pooled vegetation species into functional groups to improve comprehension and inference ability (USDA 2009). The functional groups presented include obligate wetland species, facultative species (see table 2.2 for descriptions), bryophytes, agronomic and upland species and bare peat. Bryophytes include specimens of the moss and liverwort groups; the agronomic and upland species group forest, invasive, weedy and agricultural species that are typically non-target wetland species; and the bare peat is un-vegetated areas within each experimental unit.

Site selection and preparation began in May 2006; planting, fertilization, and mulch cover occurred in May and June 2006; vegetation surveys were carried out in August and September 2006, 2007, and 2008.

Statistical Analysis

A two factor analysis of variance was conducted using a proc Mixed procedure of SAS 9.1. The two factors of interest include planting and fertilization as well as their interaction. Graphical evaluation, the Levene's test for homogeneity of variance and the Shapiro Wilk's test of normality were used to determine if the assumptions for the proc mixed procedure were met. A Log (x+1) transformations was used to improve normality and tighten heterogeneity of variances to meet assumptions of the mixed model.

3.3 Results

The total vegetation cover (including weedy agronomics and targeted facultative species) increased from 92% in the first growing season (2006) to 136% in the second year and stabilized at 132% in 2008 (Table 3.1). Straw mulching resulted in dense barley (*Hordeum vulgare*) establishment and it is believed that the mulch was instrumental in enhancing revegetation potential and moisture retention at the growing surface. *H. vulgare* established within the experimental units from the mulch application during the first growing season (mean cover of 22%). There was no measurable change in subsequent years within the sampling area. Table 3.1 also presents species and cover types with a mean cover value greater than 1% over all experimental units during each sampling year. The moss grouping increased from below 1% overall coverage in 2006 to 19% in 2007 and 27% in 2008. Although not statistically different between treatments, the bryophytes were notable contributors to the vegetation during the second and third growing season. It is important to recognize that the bryophytes consistently developed in all treatments with or without planting and fertilization (Figure 3.3 and Figure 3.4)

Absolute species richness within the combined treatments is highest followed closely by the reintroduction treatment (Table 3.1 and Table 3.2). There is no significant impact of fertilization on planting related to species richness during the first and last growing season. However, there was a significant interaction between fertilizer and planting treatments during the second growing season for both facultative and total species richness (Table 3.2). Planting provided a consistent and significant gain in facultative and total species richness over all growing seasons (Table 3.2). The non planted species richness consistently displayed the lowest absolute species richness. These patterns are noted spatially between treatments and temporally over the three years of the experiment.

Table 3.1: Estimated mean percent cover all treatments confounded for species and cover groups with an estimated mean response greater than 1%. Probable species source is indicated with an “x” based on vegetation surveys and presence within the experimental areas. Sources include donor area for reclamation trial, spontaneous vegetation either found on site through adjacent seed dispersal or from mulch. Total species richness is categorized at the base of the table.

Cover type	**% Cover during growing seasons**			**Presence**		
	1	**2**	**3**	**Donor**	**Sponta-neous**	**Mulch**
Mulch	53.57	53.75	54.09	x	x	x
Moss	<1.0	19.1	26.83	x	-	-
Potentilla norvegica	1.76	10.37	2.54	x	x	-
Salix sp.	2.8	9.75	1.73	x	x	-
Epilobium angustifolium	<1.0	7.55	5.47	x	-	-
Bidens cernua	6.59	6.29	<1.0	x	-	-
Rorippa islandica	2.39	5.46	<1.0	x	-	-
Poa sp.	<1.0	5.04	1.43	x	-	x
Festuca saximontana	<1.0	4.41	13.23	x	x	-
Calamagrostis canadensis	2.37	3.68	<1.0	x	-	-
Phleum pratense	<1.0	2.91	2.23	-	-	x
Cirsium arvense	<1.0	1.79	1.59	-	-	x
Populus balsamifera	<1.0	1.3	1.82	-	x	-
Carex utriculata	<1.0	1.19	2.39	x	-	-
Polygonum convovulvum	1.4	1.11	<1.0	-	-	x
Rumex occidentalis	<1.0	1.07	<1.0	x	-	-
Trifolium repens	<1.0	1.05	1.62	-	-	x
Aster puniceus	<1.0	<1.0	1.19	x	-	-
Carex Aenea	<1.0	<1.0	2.88	-	x	-
Galium trifidum	<1.0	<1.0	1.86	x	-	-
Geum rivale	<1.0	<1.0	3.69	x	x	-
Hordeum vulgare	21.56	<1.0	<1.0	-	-	x
Lycopus asper	<1.0	<1.0	1.2	x	-	-
Populus tremuloides	<1.0	<1.0	1.63	-	x	-
Sonchus arvense	<1.0	<1.0	1.21	x	-	x
Stellaria media	<1.0	<1.0	1.57	-	-	x
Taraxacum officinale	<1.0	<1.0	1.07	-	-	x
Typha latifolia	<1.0	<1.0	1.03	x	-	-

Total Cover	92	136	132
Species Richness			
Total of Combined Treatments	44	58	45
Reintroduction x Fertilizer	36	50	43
Reintroduction Richness	32	47	38
Fertilization Richness	28	41	37
Control Richness	27	38	34

For the plant reintroduction treatments, facultative wetland species established a significant cover consistently over the three year sampling period (Table 3.2). In the first growing season the estimated mean percent cover of facultative wetland species was 23% (Figure 3.3 and Figure 3.4) and increased during the second growing season 3 fold to 72%. During the second growing season, there was no significant difference in total vegetation cover between the combined reintroduction and fertilization treatment and the reintroduction only treatment (Table 3.2); there is no treatment interaction. Sites where no vegetation was reintroduced and there was no fertilization had facultative and obligate wetland species cover between 15 and 22% over the duration of the field experiment. These species were spontaneous colonizing plants (Figure 3.3 and Figure 3.4).

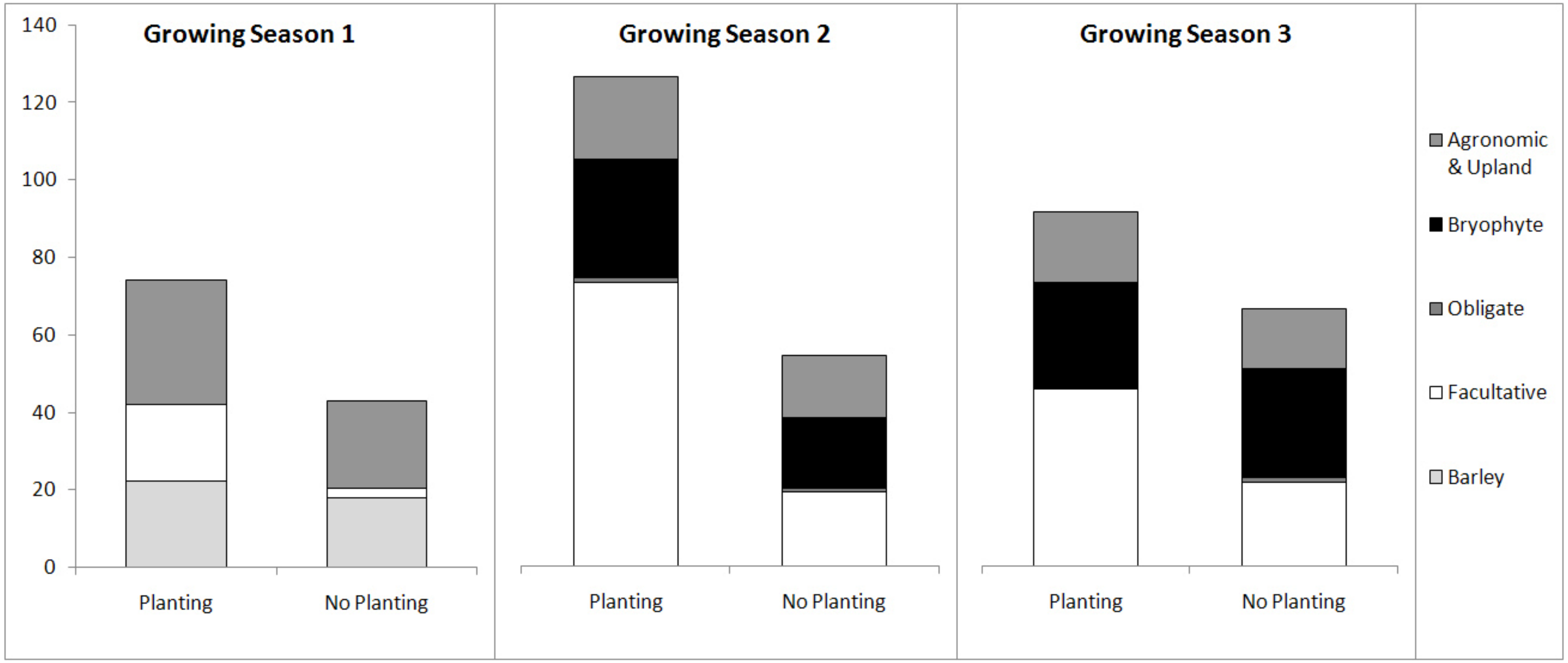

Figure 3.3: Estimated mean vegetation cover (%) sectioned into associated functional groups relative to growing season and target response type. Treatments include Planting and fertilization (n=10), No planting and no fertilization (n= 8), Fertilizer only application (n= 9), and plant introduction only, (n=8). Treatments were regrouped to show the effects of planting because there were no significant interactions noted (see table 3.2).

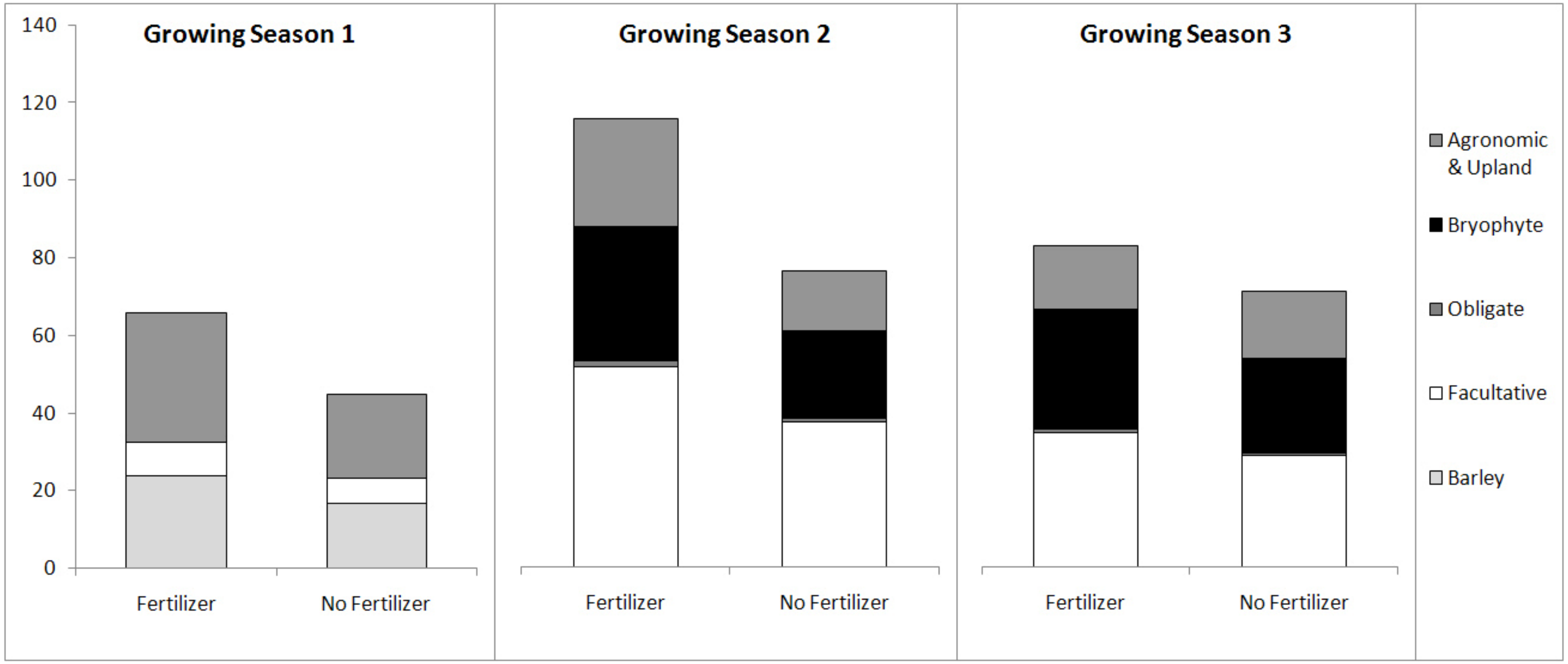

Figure 3.4: Estimated mean vegetation cover (%) sectioned into associated functional groups relative to growing season and target response type. Treatments include Planting and fertilization (n=10), No planting and no fertilization (n= 8), Fertilizer only application (n= 9), and plant introduction only, (n=8). Treatments were regrouped to show the effects of fertilizer because there were no significant interactions noted (see table 3.2).

Table 3.2: Proc mixed ANOVA output results for the effects of planting and fertilization on wet meadow species cover and richness functional groupings over three years post reclamation (2006, 2007, and 2008). Treatments include Planting and fertilization (n=10), No planting and no fertilization (n= 8), Fertilizer only application (n= 9), and plant introduction only (n=8).

			Facultative sp. Cover (log(x+1))		Total Veg sp. Cover (log(x+1))		Bryophyte sp. Cover		Facultative sp. Richness (log(x+1))		Total Sp. Richness		Agronomic Sp. Richness	
	Source	d.f.	F	P	F	P	F	P	F	P	F	P	F	P
2006	Planting Trtm	1	43.8600	**<0.0001**	19.6500	**0.0001**	Data does not meet analysis criteria		20.7600	**<0.0001**	36.3600	**<0.0001**	1.8800	0.1797
	Fertilizer Trtm	1	1.0300	0.3177	2.5600	0.1197			0.3600	0.5534	1.1600	0.2897	1.4000	0.2465
	Planting * Fertilizer	1	0.7500	0.3941	1.6700	0.2064			0.1900	0.6654	0.0000	0.9687	0.1000	0.7597
2007	Planting Trtm	1	52.7400	**<0.0001**	27.8200	**<0.0001**	3.8000	0.0602	24.7300	**<0.0001**	16.0900	**0.0004**	0.0100	0.9256
	Fertilizer Trtm	1	3.0300	0.0919	5.0700	**0.0315**	2.5700	0.1190	0.2600	0.6110	0.0200	0.8915	0.0000	0.9904
	Planting * Fertilizer	1	1.8700	0.1814	0.7400	0.3955	0.0000	0.9654	6.7500	**0.0142**	5.6700	**0.0236**	0.5000	0.4827
2008	Planting Trtm	1	21.3300	**<0.0001**	4.9400	**0.0337**	0.0400	0.8524	18.6500	**0.0001**	7.3800	**0.0107**	0.0400	0.8484
	Fertilizer Trtm	1	1.2500	0.2720	1.8700	0.1809	0.9200	0.3442	0.7200	0.4030	1.6100	0.2143	0.0500	0.8204
	Planting * Fertilizer	1	0.4700	0.4986	0.0900	0.7717	1.1500	0.2910	1.6900	0.2028	2.3800	0.1327	0.4500	0.5073
	Error	31												

3.4 Implications for restoration

Rochefort (2000) has argued that the ecological restoration of degraded peatlands should aim to restart the successional pathway that will in the long term return a damaged peatland to a peat accumulating ecosystem. This current project deals with post-extraction conditions that are no longer conducive to restoring Sphagnum dominated peatlands. The short term objectives of this project were to ensure establishment of wet meadow species with the anticipation of long term perpetuation and the potential for a successional switch towards convergence with a peat accumulating system dominated by bryophytes.

The hydrogeochemical properties of residual peat of central Alberta is more suited to wet meadow and fen restoration than bog restoration. This project was designed to test an adaptation of the moss layer transfer method (Rochefort *et al.* 2003) for a situation where bog restoration was not feasible because of the presence of rich nutrient soil conditions (Table 1.1). Restoring wetlands is a better alternative to cropland or nursery development in terms of biodiversity and restoring ecosystem functions and services.

Considering that the target vegetation for this project was vascular species; the emergence of early successional bryophytes is encouraging. The emergence of bryophytes was observed to be linked to mulch cover as there was no significant impact from the fertilizer or planting treatments. Further research trials targeting a representative combination of non-vascular and vascular species will assist in both the ecological and legislative goal setting process, specific to Alberta peatland management. As noted by Graf *et al.* (2008), active management of drainage is key to successful restoration measures. Furthermore, early establishment of a facultative wetland vegetation community immediately after abandonment will likely involve mulch cover and a modified *Sphagnum* moss layer transfer method. It is clear that a vegetation transfer method is critical to enhancing the species richness and the vegetation cover on the Evansburg North site.

Fertilization did not favour greater establishment of the vegetation over most growing seasons; however, observations of vertical structure and speed of establishment made by the field team suggest a benefit that needs further study. Future study of fertilization use requires focus to determine appropriate levels and timing.

4. Hydrology

4.1 Introduction

Restoring hydrology is the most important component in restoring wetlands (Wheeler and Shaw 1995; Keddy 1999). In the sub-humid climate of the boreal plains, hydrology plays an even more important role (Devito and Mendoza 2007). Most years there is a water deficit in the water budget of approximately between 40 to 60 mm (Johnson and Miyanishi 2008). The factors that make wetlands possible in such a climate are 1) inputs from groundwater flow, 2) deep soils, and 3) lower actual evaporation compared with potential evaporation (Devito and Mendoza 2007). It is unknown whether peatland restoration will be feasible in a climate where the water balance is so fragile.

The objective of this research was to study the hydrology of the site in order to 1) improve our ability to plan a large-scale restoration and 2) collect baseline information which will act as a pre-restoration reference. In order to achieve these objectives the following measurements were carried out:

- Hydraulic conductivity was measured to calculate the speed by which water can flow through the site;
- Water table was measured weekly throughout the season to understand the seasonal and geographic variation of the water level;
- Actual evaporation was measured weekly;
- Peat movement (swelling and shrinking) was also measured weekly to understand the 'mooratmung' of the site;
- Water head was measured weekly to determine groundwater flux.

4.2 Methods

Wells were installed in May of 2009 and piezometer nests were installed in June, once the peat had thawed sufficiently for installation (see Figure 1.2 for locations). The piezometer nests were mainly located along two main transects that created a 'v' form on the site. A piezometer nest was not included for W13 along transect 1, because the clay layer was too close to the surface (*circa* -50 cm) to measure conductivity or hydraulic head in peat. Instead a piezometer nest was installed nearby at W11.

The water levels were measured weekly between May 17th and September 15th. Wells were constructed using PVC pipes with perforations every 2.5 cm and were covered with nylon stockings. Volumetric water content was also measured weekly by taking 5 measurements which were later averaged within a 2 m radius of each well using a WET sensor (Model 1.2

Delta-T Devices Ltd., Cambridge, U.K.) connected to a moisture meter type HH2 (Model 3.0, Delta-T Devices Ltd.).

Piezometer in nests were installed at -50 cm, -75 cm, -100 cm, and -150 cm. The depth of -150 cm was only installed for piezometer nests where the residual peat was deep and where the water level was often below -75 cm (W7, W8, W9, and W10). Hydraulic head was measured weekly. Hydraulic conductivity was measured at each of the piezometer nests between 10th and 13th of August 2009. Bail tests (Hvorslev, 1951) were used to determine hydraulic conductivity (K) for each piezometer. The K values were calculated as outlined in Freeze and Cherry (1979) based on Hvorslev (1951):

$$K = r^2 \ln (L/R) / 2LT_0 \qquad \text{Equation 4.1}$$

where *r,* and, *R,* are the internal and external radii of the piezometer, *L,* is the length of the slotted intake, and T_0, is the basic lag time parameter, which is calculated from the head recovery curve of the bail or slug test.

Four lysimeters (located next to W2, W7, W10 and W18) were used to estimate evaporative losses. This was calculated by measuring mass changes due to precipitation and evapotranspiration (Kelemen and Ingram 1999). The lysimeters were constructed from plastic containers, which were circular 7.57 L buckets. One bucket was perforated at the bottom, and nested into another, identical non-perforated bucket. Peat monoliths with vegetation were put into the perforated container. Lysimeters were weighed once a week. Each week, the volumetric water content was measured using a WET sensor (same as above) inside and outside the bucket. If the surrounding area was much drier (more than 10% drier), water that had accumulated in the non-perforated bucket would be removed. If the peat monolith in the bucket was drier, then water was added. The volume of water added was always noted.

Lines of elevation sensor rods (Price 2003) or squishometers (Whittington 2005) were installed in 3 locations on the site to measure the swelling and shrinking (mooratmung) on the site. Rebar poles were pounded into the clay substrate below. The squishometers were located at W2, W7 and W14 (see figure 1.2). A hole was drilled into a plastic disc which was slid down the rebar and rested on the peat surface. The disc provided a flat surface to measure the distance between the peat surface and the top of the rebar. Any vegetation was cleared away from beneath the disc. The height of the rebar was measured weekly.

4.3 Results

Hydraulic conductivity

The hydraulic conductivity in peat on the Evansburg North site ranged from 1.12×10^{-4} to 4.45×10^{-2}. The hydraulic conductivity of the wettest area (nest number 15) was 1-2 orders of magnitude greater than drier areas at similar depths. The highest hydraulic conductivity was observed on a nearby undisturbed fen, where the hydraulic conductivity was one order of magnitude greater the wettest areas of the cutaway site (Table 4.1).

Table 4.1: Hydraulic conductivity (cm s^{-1}) for piezometer nests (see Figure 1.2 for locations) at four different depths between August 10th and 13th 2009.

	Depth of measurement (cm)			
Nest Number	**50**	**75**	**100**	**150**
W1	dry	1.12×10^{-4}	2.39×10^{-4}	NM
W2	dry	dry	3.45×10^{-3}	NM
W3	7.38×10^{-4}	2.33×10^{-3}	5.47×10^{-3}	NM
W4	dry	3.25×10^{-4}	2.28×10^{-3}	NM
W5	dry	dry	1.59×10^{-4}	NM
W6	dry	dry	1.49×10^{-2}	NM
W7	dry	dry	dry	6.45×10^{-3}
W8	dry	dry	dry	1.39×10^{-2}
W9	dry	dry	dry	6.87×10^{-3}
W10	dry	dry	dry	2.13×10^{-3}
W11	dry	2.91×10^{-2}	4.13×10^{-6} *	NM
W14	3.79×10^{-2}	2.65×10^{-3}	8.46×10^{-4}	NM
W15	1.63×10^{-2}	4.45×10^{-2}	1.14×10^{-3}	NM
W16	dry	4.15×10^{-4}	-4.10×10^{-7}*	NM
W19	1.67×10^{-4}	4.63×10^{-4}	8.27×10^{-7}*	NM
Undisturbed fen	NM	5.05×10^{-1}	6.23×10^{-2}	NM

*Piezometers were located in clay; NM= depths that were not measured. See methods section for explanation.

Water level and climate data

The growing season in 2009 was drier than average. The total precipitation measured was measured on site from mid-May to mid-September was 231 mm. The 30-year average precipitation for the same time period is 337 mm for Stony Plain weather station, located *circa* 70 km directly to the east of Evansburg (Environment Canada 2002). The precipitation events were not evenly distributed (Figure 4.1a). June was especially dry; the site received 57 mm, compared with the 30-year average of 98 mm (Environment Canada 2002). During the wet periods, the water table rose to a maximum of 2 cm above the peat surface for the wettest areas of the site (W14; Figure 4.1b). As is typical for the boreal plains, the driest period was mid-September where the water level ranged from 60 cm under the surface for the wettest area to 150 cm for the driest (Figure 4.1b).

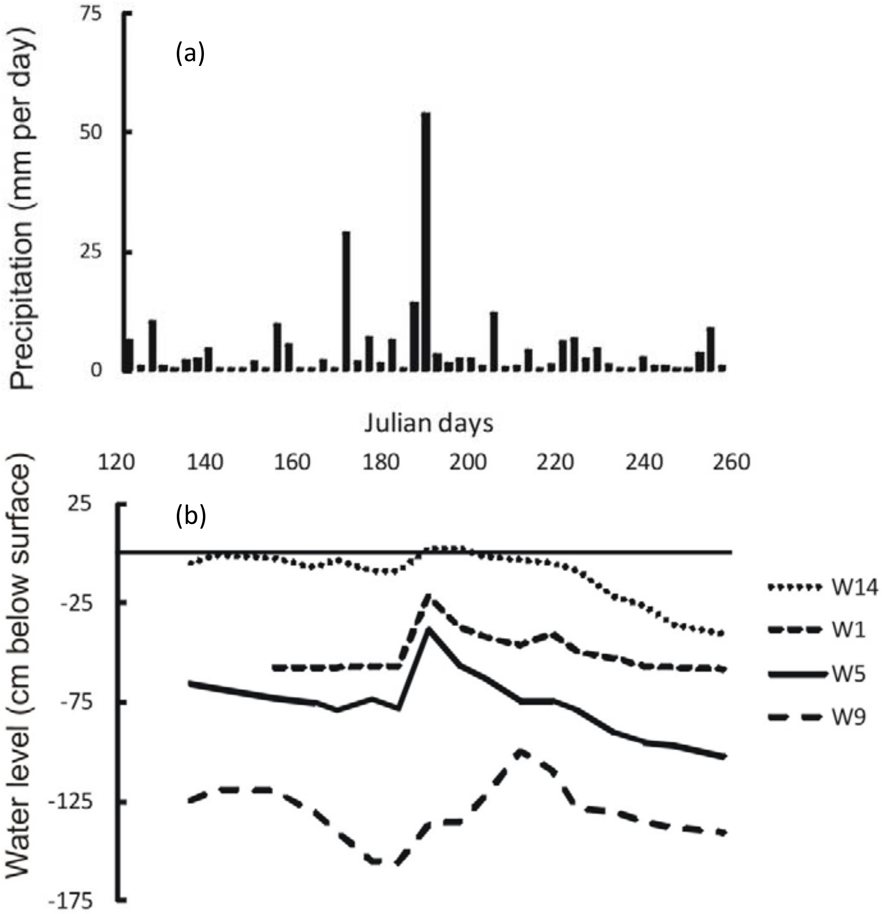

Figure 4.1: Precipitation (a) and water table relative to the peat surface (b) for 4 wells which represent the geographic distribution and hydrological gradient (see figure 4.2) of the Evansburg North site. Measurements were taken on the site from early May to mid-September 2009.

There was a clear difference in the water level in relation to the peat surface of the site. The area where the water level was closest to the surface (the northwest corner near W14) was next to a large drainage canal that had been blocked in 2006 (See section 3 for more details). The driest area was close to W9, where the elevation was the highest and the residual peat appeared to be the deepest. A nearby undisturbed fen had a water level of -4 (±2) cm.

Evaporation

The average actual evaporation measured on site for 2009 was -0.017 (±SD 0.65) mm day^{-1}. The total evaporation rate for the measured season was -1.70 mm. This value is rather low for this area where the summer climatic moisture index varies between 0 and -200 mm deficit (Strong and Leggat 1981).

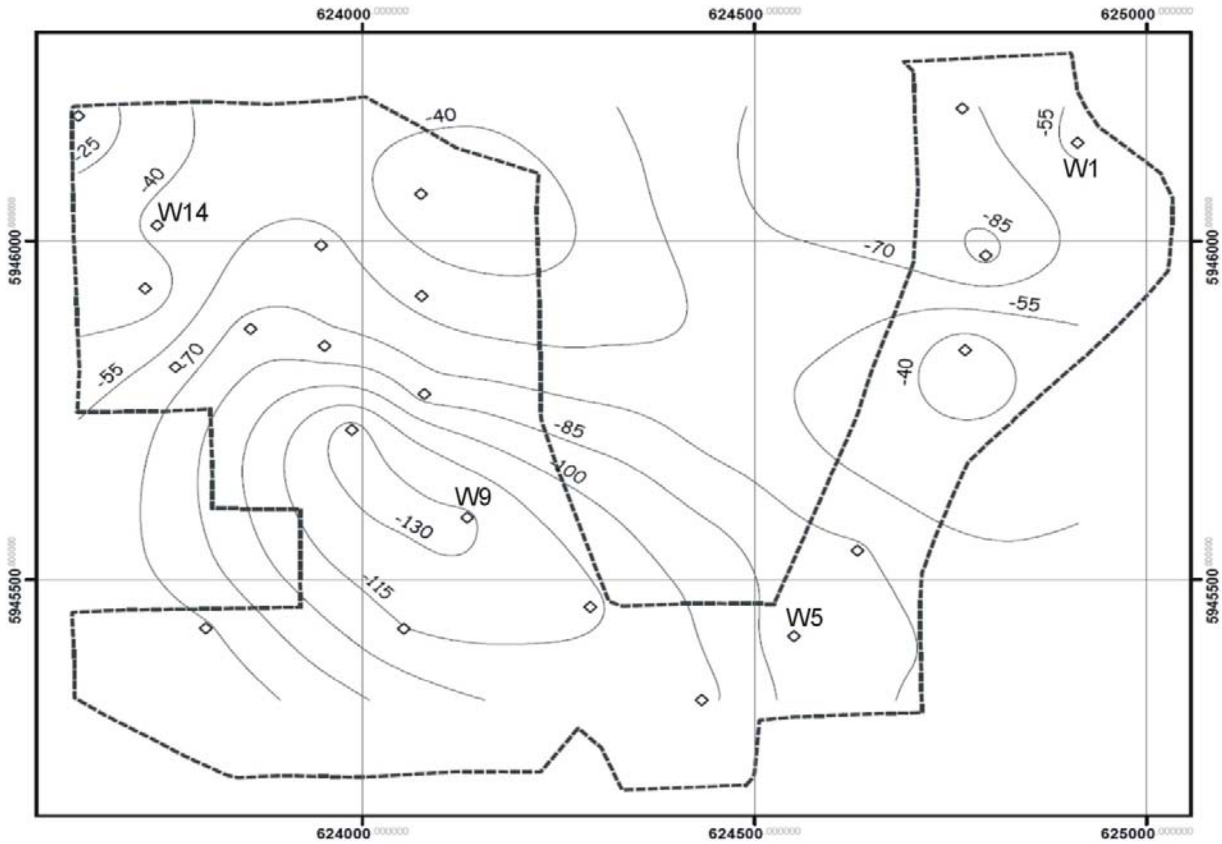

Figure 4.2: Contour map of the average water level values (cm below surface) for wells measured from mid-May to mid-September. Seasonal variation for wells 14, 9, 5 and 1 can be seen in Figure 4.1.

The volumetric water content (Figure 4.3) shows the same wet and dry areas as the average water level values. The area in the northwest corner, next to the main drainage canal that was blocked, was the wettest and the area around W9, W10 and W11 are the driest.

Peat swelling and shrinking

During the growing season of 2009 peat swelled slightly (Table 4.2). The greatest amount of swelling was located in the driest area (W7) and the smallest swelling in the wettest area (W14). The maximum swelling occurred on June 5th (Julian day 156), which does not coincide with a big rain event (Figure 4.1). The greatest compaction occurred on the 10th of July (Julian day 191), directly after a big rain event.

Table 4.2: Change in peat depth at 3 locations on the site. Positive values indicate peat swelling; negative values indicate peat shrinking.

	Change in peat depth (cm)			
Location	Average	Maximum	Minimum	±SD
W2	+0.29	+1.5	-0.5	0.60
W7	+0.38	+2.6	-1	0.88
W14	+0.16	+2.5	-1	0.87

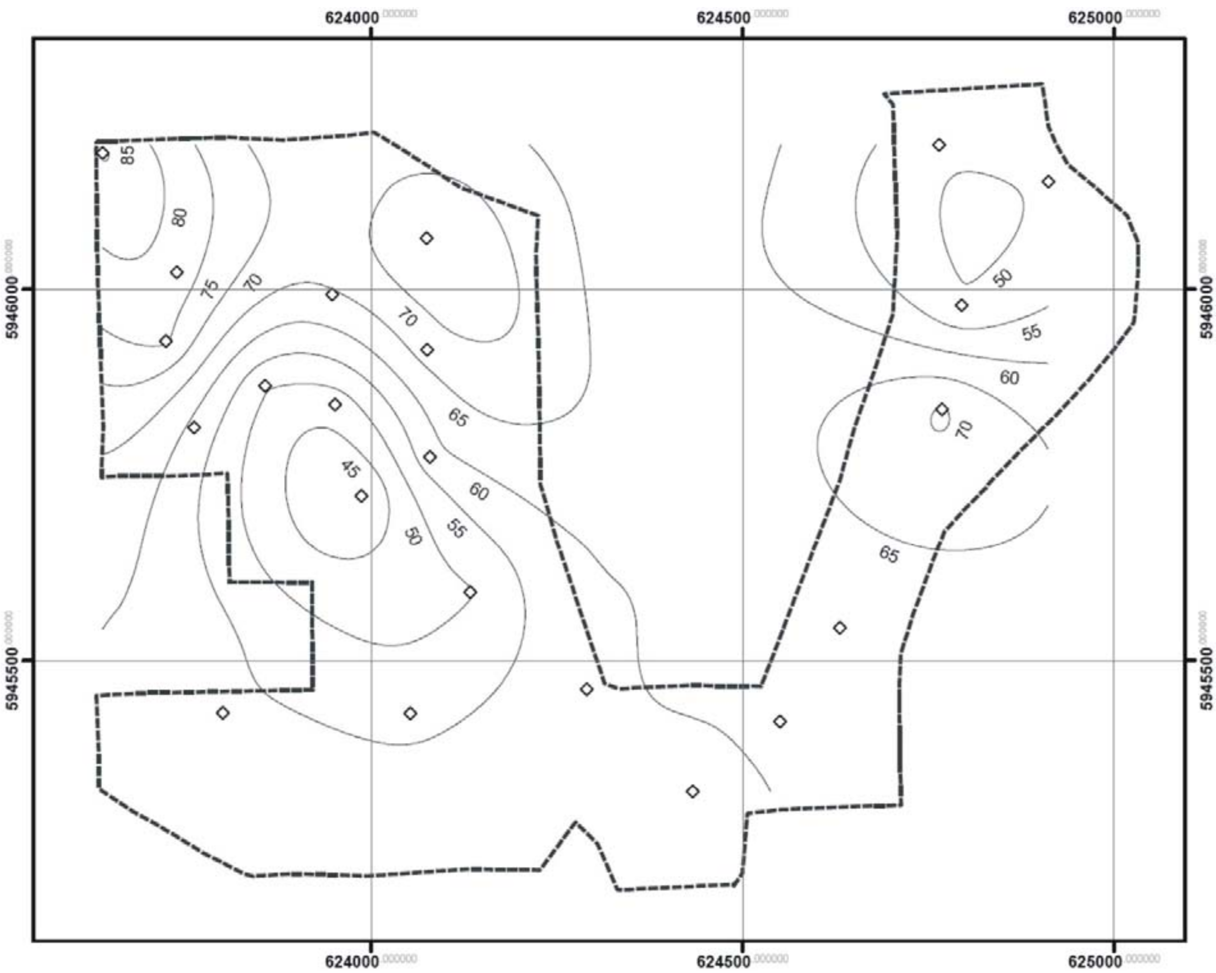

Figure 4.3: Contour map showing the average volumetric water content for the well sites tested measured between mid-May and mid-September.

Groundwater flux

The groundwater flux diagrams indicate that drainage ditches are still affecting the hydrology in transect 1. In general, the water is flowing from the southeast to the northwest in transect 1.

Transect 2 does not appear to be affected by drainage canals, and the water flows from southwest to northeast. There was not a great difference in groundwater flux between wet and dry periods. Flow is predominantly lateral through the remaining peat deposit. This suggests that the site is neither acting a groundwater recharge zone, nor is groundwater discharge providing water input to the site.

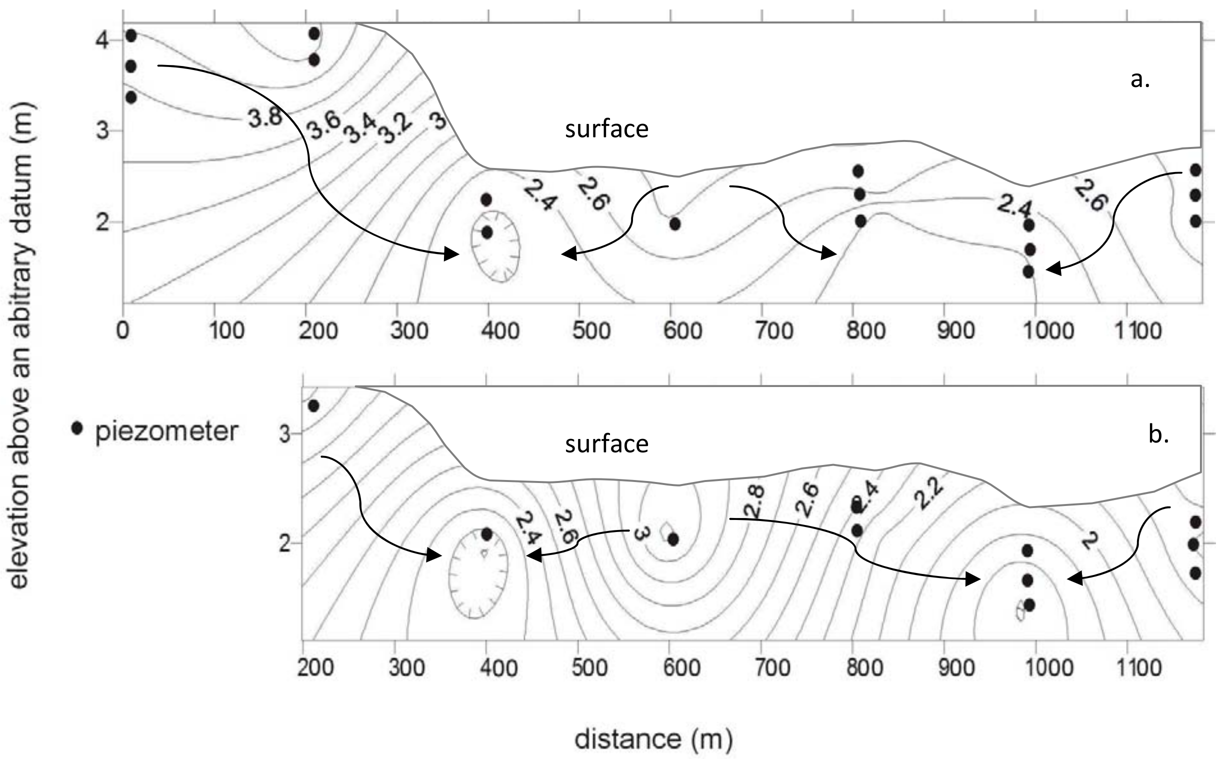

Figure 4.4: Groundwater flow net for transect 1 during (a.) wet conditions (July 10th) and (b.) dry conditions (September 14). The topography shown here is exaggerated because of differences in scale between the x and y axis. Transect is shorter in September because some piezometers were dry.

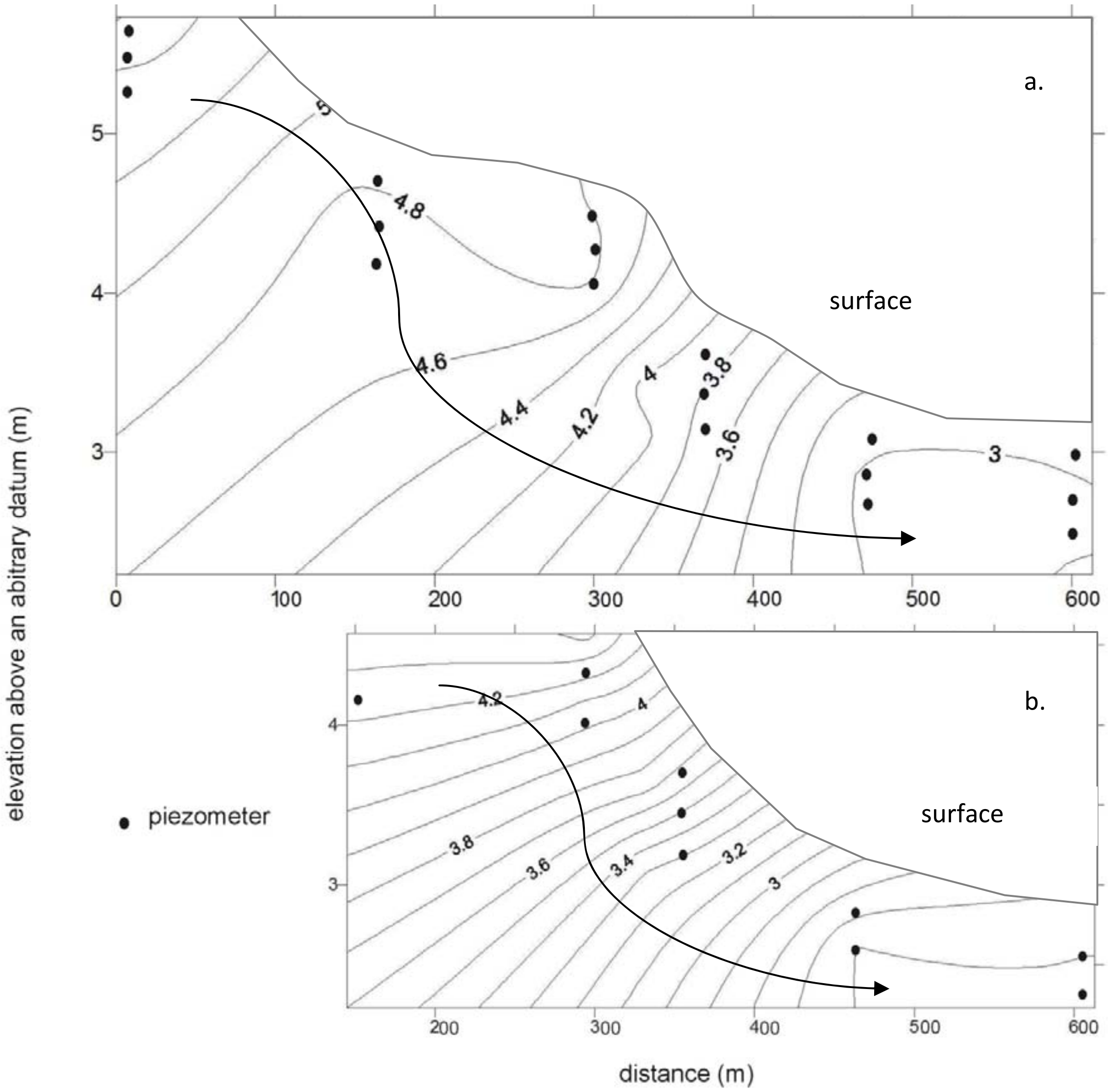

Figure 4.5: Groundwater flow net for transect 2 during (a.) wet conditions (July 10th) and (b.) dry conditions (September 14). The topography shown here is exaggerated because of differences in scale between the x and y axis. Transect is shorter in September because some piezometers were dry.

4.4 Implications for restoration

A baseline study of the hydrology of this site indicates that the area close to the blocked drainage canal (northwest corner of the site) has a hydrology that is most similar to that of an undisturbed fen. Its hydraulic conductivity, water level and volumetric water content were the most similar to that of the undisturbed fen. However, the rest of the site remains relatively dry during most of the season. A more thorough blocking of the drainage ditches, i.e. every 75 m as described in Quinty and Rochefort (2003), is recommended.

5. Carbon dioxide and methane exchange

5.1 Introduction

Natural peatlands in the boreal zone are net sinks for atmospheric carbon as uptake of carbon dioxide (CO_2) via plant photosynthesis is greater than emission of carbon as CO_2 via plant and soil respiration and CH_4 via anaerobic decomposition of organic matter below the water table. Gorham (1991) estimated mean annual CO_2 uptake in northern peatlands of 23 g C m^{-2}, and release of CH_4 of 14 g C m^{-2}. Continental peatlands of western North America have been reported to have mean annual carbon accumulation of 19.4 g C m^{-2} (Vitt *et al.* 2000).

Soil wetness is an important controller of the net carbon balance of a site because decomposition is slower under wet, anoxic (low-oxygen) conditions. Following horticultural peat extraction, water tables often remain deep if drainage systems remain active. This encourages continued peat decomposition resulting in large CO_2 emissions from the site. Also, since vegetation re-establishment is often slow on these abandoned peat sites, there is little uptake of CO_2 due to low rates of photosynthesis. Thus, unlike natural peatlands that store CO_2, these disturbed sites are often large source of atmospheric CO_2. Annual releases of over 300 g C m^{-2} have been reported from cutover peatlands in Eastern Canada (Waddington *et al.* 2002). In contrast, the dry conditions result is a substantial reduction in CH_4 emissions and often, abandoned peatlands may actually act as small CH_4 sinks (Waddington and Price 2000).

While general patterns of carbon exchange following extraction and abandonment have been described (e.g. Waddington and Price 2000), most research in North America has focused on moist temperate climates in Quebec, with little known about the magnitude of carbon fluxes from horticultural disturbance of continental peatlands in Alberta. In addition, when planning restoration projects and evaluating their success it is important to understand specific controls on ecosystem function and develop baseline data against which restored ecosystem functions can be assessed.

The objectives of this research were to:

1. Determine rates of CO_2 and CH_4 exchange at the abandoned portion of the Evansburg North peatland in central Alberta to be used as baseline data prior to large scale restoration,
2. Investigate controls on the magnitude carbon fluxes in order to effectively plan restoration activities that will enhance carbon storage at the site,
3. Determine rates of CO_2 and CH_4 exchange at a nearby undisturbed fen to assess effects of disturbance and provide values for target carbon exchange for assessing post-restoration function.

5.2 Methods

At the abandoned site carbon measurements were distributed across the site to capture the variability in ecohydrological conditions. Carbon fluxes were measured near hydrological measurement locations with 16 well areas investigated (Figure 1.2; stations W1-W7, W9, W10, W12, W14-17, D1, and D2 were measured). At each well, at least three square metal collars (60 cm x 60 cm) were installed systematically and encompassed the variability in vegetation cover in the vicinity. Metal collars had a groove at the top that was fitted with a chamber in order to measure carbon flux. An additional six sampling plots were investigated at a nearby (~2 km west) undisturbed poor fen (see Figure 6.1). These sites were chosen to represent the ecohydrologic gradient at the natural site including low-lying moss-dominated hollows and higher, drier shrub-dominated hummocks.

A clear acrylic chamber was used to measure net ecosystem CO_2 exchange (NEE - the balance between photosynthesis and respiration). The chamber was placed on the collar and changes in CO_2 concentration measured over a 2 minutes sampling period using a portable infrared gas analyzer. An opaque shroud was then used to cover the chamber to measure respiration (RE). Gross ecosystem photosynthesis (GEP) was determined as the difference between the measurements.

Methane (CH_4) emissions were measured using dark acrylic chambers. Gas samples were collected 7, 15, 25 and 35 minutes after closing the chamber and CH_4 concentration determined in the laboratory using a gas chromatograph.

At each sampling plot volumetric moisture content (VWC - soil wetness) was determined using a WET sensor (see section 4.2) and vascular plant cover and moss cover were estimated visually. Water table position was measured at the central well at each site. Air temperature was recorded throughout the measurement period at the meteorological station. All measurements were made in one intensive sampling campaign August 11-14, 2009.

5.3 Results

Carbon dioxide exchange

On average, GEP was slightly greater than RE at the abandoned site suggesting that the site is on average ± standard deviation a small sink of 2.8 ± 9.8 g CO_2 m^{-2} d^{-1} under full light conditions (Table 5.1). However, during low-light periods, the site would be a source of CO_2 to the atmosphere suggesting that seasonally CO_2 is being release. However, seasonal measurements under a variety of climatic and light conditions would be required to determine the magnitude of this flux.

Both GEP and RE were highly variable across the site. As has been observed in other studies (e.g. Riutta *et al.* 2007), RE was positively related to air temperature. It was also related to volumetric moisture content (VWC) with maximum RE occurring at when VWC was 55% and declining under both wetter and drier soil conditions. Vascular cover was also related to RE suggesting that at vegetated sites, plant respiration was an important component of total CO_2 emissions from the plots. GEP was primarily related to vascular plant cover (Figure 5.1). Some of the variability was also related to soil moisture with optimum GEP at VWC of 57%. Sites without vegetation cover will act as sources of CO_2 to the atmosphere because GEP is zero; however, even sites with less than 10% vascular plant cover were consistently sources of CO_2 even under full light conditions (Figure 5.1).

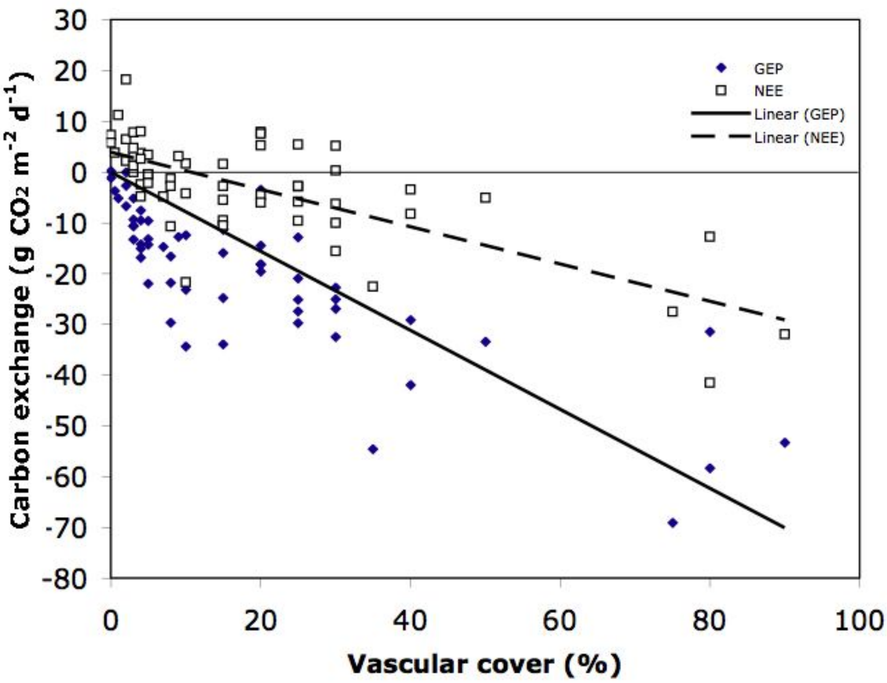

Figure 5.1: Gross ecosystem photosynthesis (GEP) and net ecosystem CO_2 exchange (NEE) under full light conditions as related to vascular plant cover at the cutover site. Negative values of carbon exchange represent an uptake of carbon by the ecosystem.

The undisturbed site was a larger sink of CO_2 than the abandoned site taking up on average 13.8 ± 13.2 g CO_2 m^{-2} d^{-1}. This resulted from the fact that the undisturbed site had higher GEP and lower RE than the abandoned site. Dense vascular cover and nearly continuous moss cover likely resulted in higher productivity while shallow water table position limited soil respiration.

Table 5.1: Mean carbon fluxes from the Evansburg North peatland

Flux *(mean ± standard deviation)*	**Abandoned** n = 57	**Undisturbed** n = 6
Gross ecosystem photosynthesis (GEP) *g CO_2 m^{-2} d^{-1}*	19.3 ± 13.8	27.1 ± 16.3
Ecosystem respiration (ER) *g CO_2 m^{-2} d^{-1}*	16.5 ± 7.9	13.3 ± 4.2
Net ecosystem exchange (NEE) *g CO_2 m^{-2} d^{-1}*	2.8 ± 9.8	13.8 ± 13.2
CH_4 flux *mg CH_4 m^{-2} d^{-1}*	0.2 ± 5.2	22.1 ± 9.0

Methane flux

Methane emissions were extremely low at the abandoned peatland. Average CH_4 flux was only 0.2 ± 5.2 mg CH_4 m^{-2} d^{-1}. Emissions from the abandoned site were related to water table position (Figure 5.2). Sites with water table 10 to 30 cm below the surface emitted on average 2.4 mg CH_4 m^{-2} d^{-1} while sites with deeper water tables had a small uptake of 0.7 mg CH_4 m^{-2} d^{-1}. In contrast, the undisturbed site was a source of 22.1 ± 9.0 mg CH_4 m^{-2} d^{-1}.

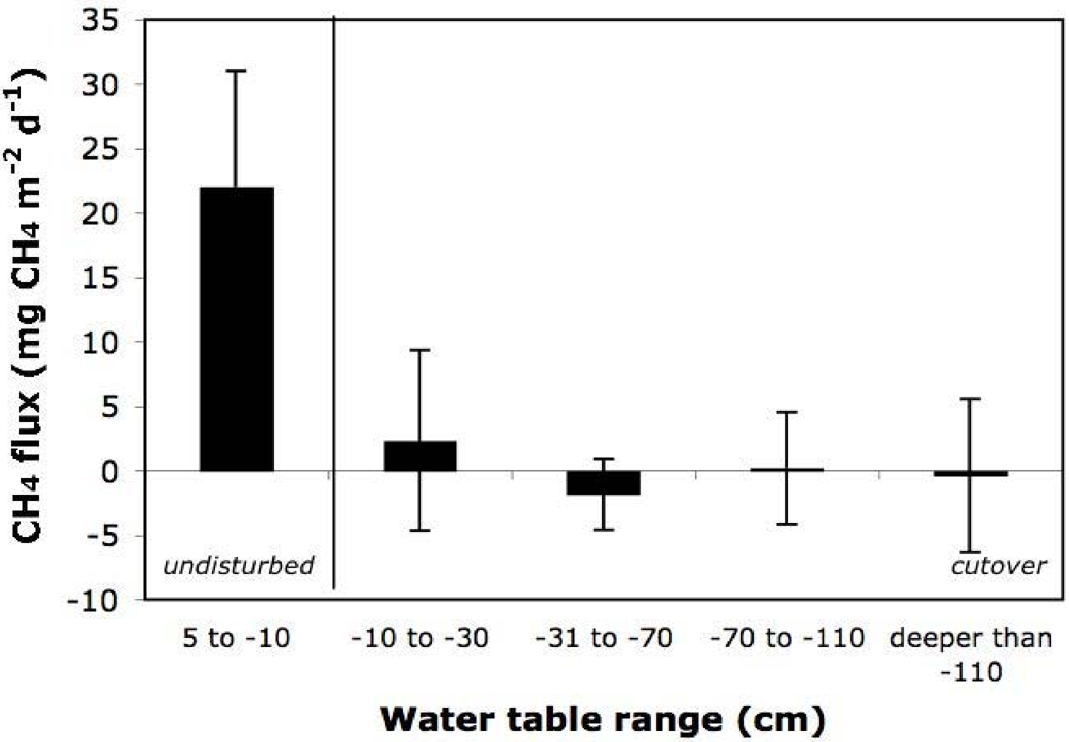

Figure 5.2: Methane flux related to water table position at the undisturbed and cutaway sites. Positive values indicate a release of methane from the peatland to the atmosphere.

5.4 Implications for restoration

Emission of CO_2 via RE was maximized when VWC was 55%. Although this was also close to the optimal soil moisture for GEP, restoration should strive to increase peat wetness above this threshold to minimize soil respiration and thus reduce carbon emission. More important than soil moisture for controlling CO_2 uptake via GEP was plant cover as bare and sparsely vegetated sites (<10% cover) were net sources of CO_2 to the atmosphere. Thus, if one of the goals of restoration is to return the site to a net carbon accumulating system, activities also need to encourage colonization of the site by vegetation. Although raising the water table will likely enhance CH_4 emissions from the site, evidence from this study suggests that even water tables within 10 to 30 of the surface should result in CH_4 emissions that are an order of magnitude lower than the undisturbed site and substantially lower than average CH_4 emissions from northern peatlands reported in literature (Gorham 1991; Saarnio *et al.* 2007).

6. Suggestions for a large-scale restoration of Evansburg North

An essential step in a large-scale restoration of the Evansburg North is a more thorough blocking of drainage ditches to improve the hydrology of the entire site (although not the first action to implement, plant reintroduction has to be done first). All main drainage ditches should be blocked every 75 m. Even then, some areas (around W8, 9 and 10) may still be too dry to support wetland vegetation. One possibility for such areas would be to remove some of the residual peat to bring the water level closer to the surface. Another possibility would be to restore this area for forest.

After hydrology, the second most important component in peatland restoration is the selection of an appropriate donor site. Due to its proximity, abundance of donor vegetation, similar chemistry, a high species richness and a dominance of bryophytes, we recommend an undisturbed fen (53º38'33.32"N 115º9'5.98"W) located on the Sungro lease directly west of the cutaway peatland.

Figure 6.1: Location of a suitable donor site located directly west of the cutaway peatland.

One major difference between fen and bog restoration is the presence of spontaneous vegetation on abandoned fens. Many wetland plants have established on this site without active reintroduction. However, field trials show that reintroducing vegetation increases vegetation cover and species richness. Indeed, plots where vegetation was reintroduced showed a vegetation cover twice as high. Additionally, bryophytes, an important component of fen vegetation, have not spontaneously recolonized the site. Field experiments in Québec have shown that the presence of spontaneous vegetation (vascular plants) increases the percent cover of reintroduced bryophytes.

We suggest that a large-scale restoration could test the influence of spontaneous vegetation by incorporating three treatments into the restoration of this site. The first treatment would be to physically remove spontaneous vegetation with a screw with end or backhoe and to level the peat benches before reintroducing the vegetation. This would simulate restoring a cutaway peatland directly after abandonment where no spontaneous vegetation has established. The second treatment would be to reintroduce vegetation without removing the spontaneous vegetation. The third treatment would be a control where vegetation would neither be introduced nor removed. We recommend that the treatments be implemented on alternating benches (i.e. bench 1 would receive treatment 1, bench 2, treatment 2, etc.) to improve the geographical distribution of the treatments.

We recommend that the experimental plots from Dave Critchley be left untouched to provide an example of the vegetation development of areas restored at an earlier date.

Lastly, we would like to set aside three areas on the site to carry out an experiment to test the relationship between the vegetation structure (mosses, bryophyte or a combination) and the peat-accumulation capacity. The exact areas to be used for this experiment should be discussed with Maria Strack based at University of Calgary.

7. References

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Appendices

Appendix 1: List of species observed in quadrats in 2009 and 2005. Species are arranged in decreasing order of frequency. Codes used to describe the wetland indicator status are shown in Table 2.3.

				2009		2005	
Latin name	**Gen.spe**	**Common name**	**Wetland Indicator Status**	**Frequ-ency**	**% cover**	**Frequ-ency**	**% cover**
Salix discolor Muhl.	Sal.dis	Pussy willow	FACW	15	6.05	7	2.85
Betula papyrifera Marsh.	Bet.pap	Paper birch	FACU	9	2.41	6	0.25
Populus tremuloides Michx.	Pop.tre	Quaking aspen	NI	8	2.09	7	0.71
Carex aquatilis Wahlenb.	Car.aqu	Water sedge	OBL	7	3.36		
Chamerion angustifolium (L.) Holub ssp. Angustifolium	epi.ang	Fireweed	FAC	7	0.78	13	0.92
Deschampsia cespitosa (L.) P. Beauv.	Das.cae	Tufted hairgrass	FACW	6	3.19	7	0.23
Picea mariana (Mill.) Britton, Sterns and Poggenb.	Pic.mar	Black spruce	FACW	6	0.20	3	0.08
Salix petiolaris Sm.	Sal.pet	Meadow willow	OBL	4	2.18	1	0.08
Fragaria virginiana Duchesne	fra.vir	Viginia strawberry	FACU	3	0.10	1	0.02
Geum macrophyllum Willd.	Geu.mac	Largeleaf avens	FACW	3	0.15		
Hieracium umbellatum L.	Hie.umb	Narrowleaf Hawkweed	NI	3	0.10		
Lycopus uniflorus Michx.	Lyc.uni	Northern bugleweed	OBL	3	0.32		
Sonchus arvensis L.	Son.arv	Field sowthistle	FAC	3	0.24		
Calamagrostis canadensis (Michx.) P. Beauv.	Cal.can	Bluejoint	FACW	2	0.36		
Calamagrostis stricta (Timm) Koeler ssp. Inexpansa (A. Gray) C.W. Greene	cal.str	Northern reedgrass	FACW	2	1.36	5	0.52
Carex canescens L.	Car.can	Silvery sedge	OBL	2	0.32		
Carex sp. L.	Car.sp	Carex species	NI	2	0.59		

Ranunculus pensylvanicus L. F.	Ran.pen	Pennsylvania buttercup	FACW	2	0.01		
Salix exigua Nutt.	Sal.exi	Narrowleaf willow	FACW	2	0.32		
Euthamia graminifolia (L.) Nutt. Var. Graminifolia	sol.gra	Flat-top goldentop	FACW	2	0.23		
Triglochin palustris L.	Tri.pal	Marsh arrowgrass	OBL	2	1.41		
Trifolium pratense L.	Tri.pra	Red clover	FACU	2	0.59		
Bidens cernua L.	Bid.cer	Nodding beggartick	OBL	1	0.05		
Carex brunnescens (Pers.) Poir.	Car.bru	Brownish sedge	FAC	1	0.05		
Cinna latifolia (Trevis. Ex Goepp.) Griseb.	Cin.lat	Drooping woodreed	OBL	1	0.00		
Cirsium arvense (L.) Scop.	Cri.arv	Canada thistle	FACU	1	0.23		
Dicranella cerviculata (Hedw.) Schimp.	Dic.cer	Dicranella moss	NI	1	0.45		
Geum aleppicum Jacq.	Gea.ale	Yellow avens	FACU	1	0.00		
Larix laricina (Du Roi) K. Koch	lar.lar	Tamarack	FACW	1	0.14	1	0.02
Polytrichum juniperinum Hedw.	Pol.jun	Juniper polytrichum moss	NI	1	0.23		
Polytrichum strictum Brid.	Pol.str	Polytrichum moss	NI	1	3.18		
Rorippa islandica (Oeder) Borbás	ror.isl	northern marsh Yellowcress	NI	1	0.00		
Salix glauca L.	Sal.gla	grayleaf willow	FACW	1	0.68		
Scirpus cyperinus (L.) Kunth	sci.cyp	woolgrass	OBL	1	0.23		
Taraxacum officinale F.H. Wigg.	Tar.off	Common dandelion	FACU	1	0.09	1	0.08
Typha latifolia L.	Typ.lat	Broadleaf cattail	OBL	1	0.09		
Hordeum jubatum L.	Hor.jub	Foxtail barley	FACW			3	0.48
Lepidium ramosissimum Nelson	lep.ram	Manybranched pepperweed	NI			3	0.15
Potentilla gracilis Douglas ex Hook.	Pot.gra	Slender cinquefoil	FAC			5	0.25
Salix sp.	Sal.sp	Willow species	NI			3	0.58

Appendix 2: Additional species observed on the site in 2009 that were not present in the quadrats sampled.

Latin name	Common name	Wetland Indicator Status
Achillea millefolium L.	Common yarrow	FACU
Alopecurus aequalis Sobol.	Shortawn foxtail	OBL
Bromus inermis Leyss.	Smooth brome	NI
Carex crawfordii Fernald	Crawford's sedge	FAC
Carex siccata Dewey	dryspike sedge	FACW
Carex utriculata Boott	Northwest Territory sedge	OBL
Eleocharis acicularis (L.) Roem. and Schult.	Needle spikerush	OBL
Elymus trachycaulus (Link) Gould ex Shinners	slender wheatgrass	FACU
Equisetum arvense L.	Field horsetail	FAC
Eriophorum vaginatum L.	Tussock cottongrass	OBL
Festuca saximontana Rydb.	Rocky Mountain fescue	NI
Hierochloe odorata (L.) P. Beauv.	Sweetgrass	FACW
Lemna minor L.	Common duckweed	OBL
Parnassia palustris L.	Marsh grass of Parnassus	OBL
Phalaris arundinacea L.	Reed canarygrass	FACW
Pinus contorta Douglas ex Louden	Lodgepole pine	FACU
Rumex aquaticus L. Var. *Fenestratus* (Greene) Dorn	Western dock	OBL
Salix candida Flueggé ex Willd.	Sageleaf willow	OBL
Schoenoplectus tabernaemontani (C.C. Gmel.) Palla	Softstem bulrush	OBL
Spiranthes romanzoffiana Cham.	Hooded lady's tresses	OBL
Symphyotrichum puniceum (L.) A. Löve and D. Löve var. *Puniceum*	Purplestem aster	OBL
Utricularia macrorhiza Leconte	Common bladderwort	OBL

Appendix 3: A list of GPS coordinates for wells and piezometers nests installed on Evansburg North site. See Figure 1.2 for a map of these points.

Well		N			W	
D1	53	38	977	115	6	741
W1	53	38	948	115	6	609
W2	53	38	860	115	6	719
W3	53	38	785	115	6	746
W4	53	38	628	115	6	878
W5	53	38	561	115	6	954
W6	53	38	512	115	7	63
W7	53	38	588	115	7	189
W8	53	38	574	115	7	405
D2	53	38	578	115	7	634
W9	53	38	661	115	7	328
W10	53	38	733	115	7	458
W11	53	38	786	115	7	660
W12	53	38	815	115	7	572
W13	53	38	849	115	7	692
W14	53	38	987	115	7	763
W15	53	38	899	115	7	676
W16	53	38	880	115	7	487
W17	53	38	800	115	7	487
W18	53	38	760	115	7	373
W19	53	38	838	115	7	373
W20	53	38	919	115	7	370