Examination of Wet Meadow Creation as a Restoration Option for Extracted Peatland Sites in Alberta

by

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Abstract

Question: Can a wet meadow plant community be established on abandoned peatlands through broadcast plant diaspore spreading in western Canada? Does fertilization impact the development and establishment of the wet meadow?

Location: Evansburg, Alberta, Canada

Methods: Wet meadow vascular and non-vascular species were spread under 2 treatments on an abandoned peatland. The effect of spreading and fertilization treatments were tested using a factorial randomized unbalanced design repeated six times. Treatments were used to statistically test main effects and interactions. A barley (*Hordeum vulgare*) straw mulch cover was applied on all experimental units.

Results: Plant spreading was an effective restoration method for establishing vascular wet meadow vegetation on sites with sufficient moisture. The treatments that included plant spreading had 80% coverage by wetland dependent vegetation and a distinct decline in agronomic and upland species. Fertilization had no significant effect on plant cover.

Conclusion: The use of a modified Sphagnum moss layer transfer method to establish a wet meadow plant community on a post-abandoned peatland was successful and contributes to the development of a new approach towards managing abandoned peatlands in mid-continental boreal Alberta, Canada.

Keywords: Peatland, diaspore; wet meadow; spreading; restoration; re-vegetation; transfer method; fertilization.

Acknowledgements

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Chapter I: General Introduction

Peatlands are an important resource on a global scale supporting carbon sinks and fresh water filters. Peatlands encompass approximately 1.24 X 10^8 ha or 8% of Canada's land area (Joosten and Clarke 2002) of which 1.03 X 10^7 ha are contained within the province of Alberta representing 16% of the province's land area (Daigle and Gautreau-Daigle 2001). In North America, peat is extracted for horticultural purposes and peatlands are often disturbed by forest and energy exploration and extraction activities (Turetsky and St. Louis 2006). Disturbances typically lead to extended periods of drainage, compaction, removal of the acrotelm, and considerable loss of the catotelm (Ferland and Rochefort 1997; Wind-Mulder *et al*. 1996).

As a result of the *in situ* anthropogenic disturbance clear management goals are required for the highly altered peatland sites. Managing for reclamation goals would entail the stabilization of terrain, assurance of public safety, aesthetic improvement and a return to equivalent land capability for a specified end land use (SER 2002). Following the reclamation approach, there is no specific requirement to develop a wetland ecosystem and these areas may be converted to agricultural or commercial use. An alternate and appropriate management goal entailing site restoration would include a process of restoring one or more valued attributes of the original landscape and assisting the recovery of the ecosystem components that were damaged, degraded or destroyed (SER 2002; Davis and Slobodkin 2004). Using a restoration management paradigm emphasizes the linkage to the original landscape and favours development of techniques that reinforce wetland ecosystem processes. Although the restored wetland may not be identical to the original site, it is an approximation that supports the essence of wetland function and may provide opportunity to develop towards a system with minimal anthropogenic input.

Canadian peatland restoration literature focuses predominantly on cut-over ombrotrophic sites in eastern Canada. However, many western Canadian continental sites are abandoned with limited restoration effort. In Alberta, horticultural extraction processes remove up to several meters of peat substrate over timelines extending into decades which change the hydrology and chemistry of each site in a unique fashion (Kuhry *et al*. 1993). As such, there is a change in the successional stage from an ombrotrophic peatland prior to extraction to a cut-away minerotrophic peatland after extraction (Kuhry *et al*. 1993; Wind-Mulder *et al*. 1996; Graf *et al*. 2008). Cut-away minerotrophic peatlands are distinct in hydrology, substrate chemistry and seed bank content when compared to cut-over peatlands that retain similar conditions to the undisturbed site; as such, unique restoration techniques are required to alter the successional development potential of the site towards wetland restoration targets (Wind-Mulder *et al*. 1996; Graf and Rochefort 2008). Unmanaged, abandoned peatlands may be classified as marsh sites where vegetation may be more closely linked to the underlying mineral layers than to the thin remaining peat layers; this is particularly true in cut-away peatlands. The resulting perception of this site change is an historic belief that peatland restoration techniques used in eastern Canada will not apply to the Alberta sub-humid climate where evaporation exceeds precipitation. As a result of the sub-humid conditions, Alberta peatlands are unique in type and distribution when contrasted to the provinces of Quebec and New Brunswick. The dominant Alberta peatland is minerotrophic rather than ombrotrophic (Nicholson *et al*. 1992). Although there are limited studies on Alberta cut-away peatlands, several researchers have addressed water and substrate chemistry, classification, and small scale restoration in minerotrophic sites (Kuhry *et al*. 1993; Vitt and Chee 1990; Windmulder *et al*. 1996; Cobbaert; 2003, Cobbaert *et al*. 2004; Graf and Rochefort 2008). Currently there have been no complete and comprehensive studies undertaken in western Canadian peatlands that address large scale restoration on sub-humid continental minerotrophic sites.

Although there are numerous European fen restoration studies; intensive agricultural use and unique project goals often focused on rare species establishment preclude extensive knowledge transfer to Alberta (van der Hoek *et al*. 1998; VanDuren *et al*. 1998; Lamers *et al*. 2002; Klimkowska *et al*. 2010). The most applicable restoration approach to Alberta's minerotrophic site is the North American Approach to peatland restoration developed by Rochefort *et al*. (2003).

A cut-away peatland will have richer mineral and nutrient content and an elevated pH which creates poor conditions for ombrotrophic peatland restoration (Wind-Mulder et al. 1996; Graf and Rochefort 2008). Although the current North American restoration approach is specific to

Sphagnum dominated ombrotrophic peatlands; modifications that align to cut-away conditions are feasible. As restoration is focused on the establishment of one or more valued attributes of the original landscape the target endpoint of the North American Approach can be altered to focus on species and conditions equated to fen and meadow communities. Rather than concentrating on the final product of an ombrotrophic peatland, intermediary steps focused on minerotrophic species establishment may be preferred for sub-humid continental peatlands found on the southern fringe of the boreal ecozone. This will encompass wet meadow communities as they include graminoid, graminoid-like, and wetland forb species growing in seasonally waterlogged soils (Raab and Bayley 2013) with minimal bryophyte diversity. Once established, the deep rooting potential of many graminoid like species and forbs will assist in low water periods as they can readily access deeper water inaccessible to bryophytes. Ultimately, the restoration of minerotrophic sites requires the cessation of continued degradation including improvement of hydrology, reduction of surface isolation and allowance for suitable vegetation establishment. This anthropogenic alteration will change the successional trajectory of the site by reducing the occurrence of bare peat and early successional species dominance. Spreading vegetation materials, from a donor site, and fertilizing will promote the rapid development of mid to late seral species indicative of minerotrophic vascular communities. Given reasonable timelines, wet meadow vascular communities will establish and allow for traditional development along the wet meadow to fen meadow or ombrotrophic peatland trajectory (Figure 1).

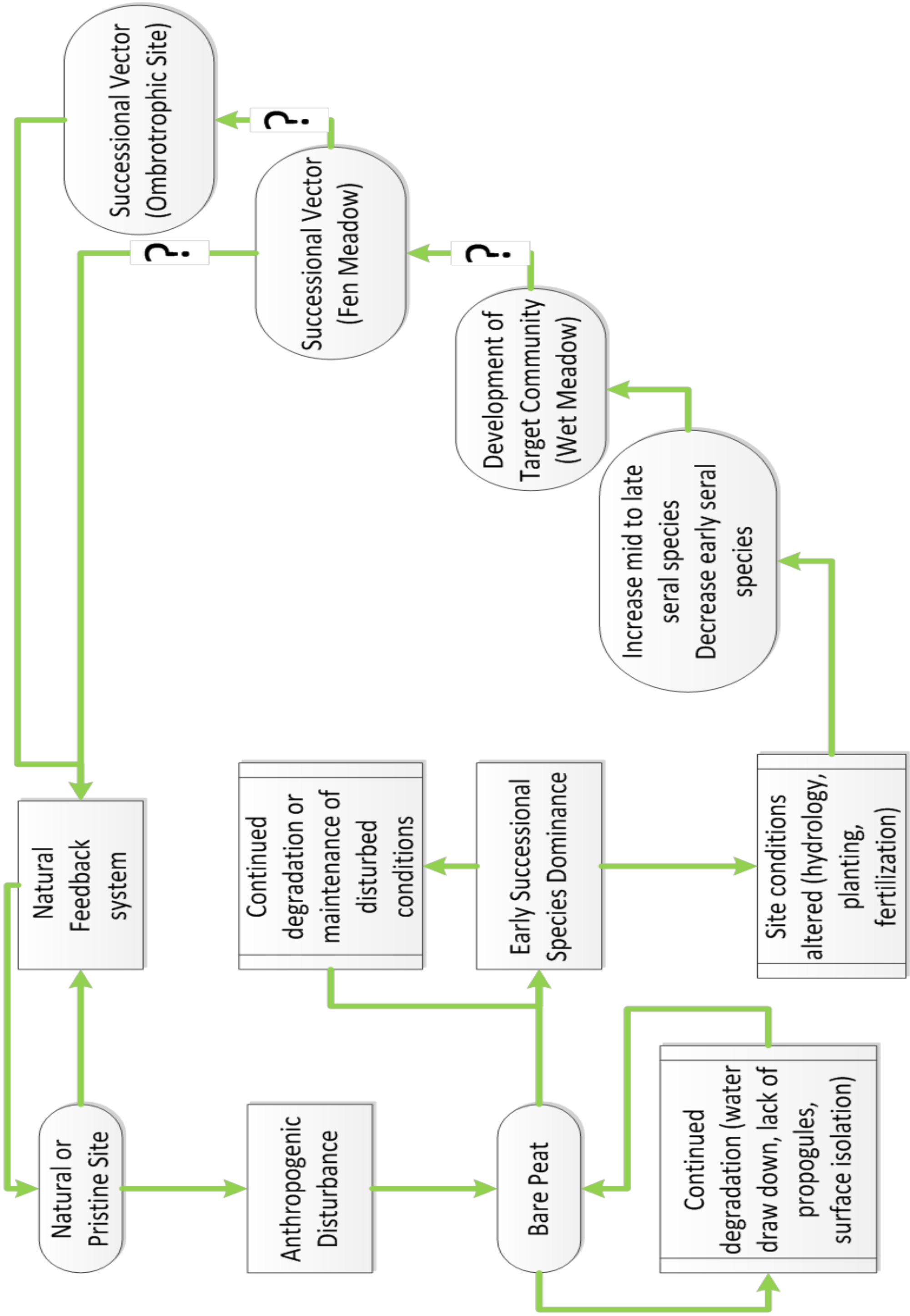

Figure 1 Hypothesized development pathways for Alberta cut-away peatland restoration sites.

To initiate the restoration process on western Canadian cut-away peatlands establishment of a new vegetation community is required. To achieve successful restoration two major questions need to be determined: 1.) Can wet meadow plant communities be established on abandoned peatlands and 2.) Does fertilization positively impact the establishment of these wet meadow communities? The response to both of these questions, combined with the knowledge from various management options provided by the Peatland Ecology Research Group and various other international organizations will establish additional restoration pathways for Alberta peatlands.

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Chapter II:

Wet meadow creation: a restoration option for Alberta's horticultural peatlands?

Critchley, D; Foote, A.L. and Rochefort, L

INTRODUCTION

Peatlands are an important resource and ecologically active land cover recognized on a global scale as important storage sites of organic carbon and filters of fresh water. They encompass approximately 1.24 X 10^8 ha or 8% of Canada's land area (Global peatland database of the International Mire Conservation Group) of which 1.03 X 10^7 ha of peatland is contained within the province of Alberta making up 16% of the land area (Daigle and Gautreau-Daigle 2001). In North America, peat is extracted for horticultural purposes or disturbed by forest and energy sector activities (Turetsky and St. Louis 2006). Extraction requires extended periods of drainage, compaction, removal of the acrotelm, and removal or disturbance of the catotelm (Ferland and Rochefort, 1997; Wind-Mulder *et al*. 1996). Extraction removes up to several metres of peat substrate changing the successional position from an ombrotrophic peatland prior to extraction to a minerotrophic peatland (Wind-Mulder *et al.* 1996; Graf *et al*. 2008) or possibly a mineral-based wet meadow. Horticultural peat extraction timelines extend into decades which, combined with peat removal, changes the hydrology and chemistry of each site in a unique fashion (Kuhry *et al*. 1993). Cut-away peatlands have depauperate seed banks and are often colonized by ex-situ pioneer species (Wind-Mulder *et al*. 1996), resulting in inefficient or unsuccessful regeneration of peat extraction sites.

Prior to human intervention on these sites, moisture content, surface oxidation and local environmental chemistry spatially restricted most species establishment and regeneration to surface cracks and drainage ditches (Salonen 1987; Campbell *et al*. 2002; Groeneveld and Rochefort 2002; Waddington and McNeil 2002; and Price *et al*. 2003). Little non-vascular establishment occurred on the restoration sites prior to experimental manipulation. The presence of an isolated and desiccated surface was functionally comparable to the closed vegetation canopy associated with abandoned peat meadows, post agricultural use, in European countries which reduce survivorship of native species (Isselstein *et al*. 2002; Kleijn 2003; van Dijk *et al*. 2007).

Peatland restoration is further affected by microsite-specific concerns of drying as well as impacts resultant from the origin of the peatlands, depth of extraction, the availability of water to the rooting zone, and the type and age of residual peat (Ferland and Rochefort 1997). The post extraction conditions provide opportunity to assess various restoration pathways.

The Alberta Wetland Inventory Classification system version 2.0 (Halsey *et al*. 2004) describes peatlands as having potential to grade into wet meadows in open non-patterned fen wetlands. This concept provides alternate restoration pathways and is further supported by the adoption of the Stewart and Kantrud (1971) wetland classification system that is frequently applied to the white zone wetlands of Alberta. As such, wet meadows may provide a pivotal role in the initial establishment and development of peatland restoration projects.

A wet meadow is a compositionally diverse grassland with graminoid, graminoid-like and wetland forb species that experiences seasonally waterlogged soil near the surface, lacks standing water for the majority of the year (Stewart and Kantrud 1971; Raab and Bayley 2013), and contains minimal bryophyte diversity and limited peat accumulation. The Evansburg 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 anticipated to assist in the establishment of vascular fen species and provide suitable cover to nurse bryophyte fen species given time.

The North American *Sphagnum* moss layer transfer method of peatland restoration was pioneered by Rochefort et al. (2003) and includes the following stages: (1) site preparation; (2) diaspore collection; (3) diaspore transfer; (4) mulch covering; and (5) potash fertilization. Our research is intended to adapt this technique to sub-humid minerotrophic continental peat conditions on post-industrial extraction sites in western Canada. To date there have been no operational scale restoration projects in this geographic region focused on cut-away peatlands.

Objectives

Although long-term goals of peatland restoration typically focus on ombrotrophic systems with peat accumulation from a suite of peatland species; prevention of further degradation of abandoned sites should include intermediate goals that may facilitate peat accumulation in the future. The early stages of restoration may neither be peat-accumulating nor bryophyte dominated. Wet meadow vegetation was selected as a target community because the post-extracted sedge peat conditions of many Alberta peatlands are not immediately compatible with ombrotrophic peatland restoration species (Harkonen 1985; Zoltai and Johnson 1985; Vitt and Chee 1990; and Wind-Mulder *et al*. 1996). The research was driven by the following questions: 1) will wet meadow vegetation communities establish on a cut-away peatland, and 2) will fertilization enhance wet meadow vegetation community establishment on a cut-away peatland. Success in the project was designated as having a large proportion of native mid to late seral wetland vegetation species established on the research area.

METHODS

Site Description

The field experiment was conducted over three growing seasons on an abandoned, cut-away peatland approximately 70 hectares (ha) in size and located ca. 115 Kilometres (km) west of Edmonton, Alberta, Canada in the continental mid-boreal wetland region of Canada (NWWG, 1988; NWWG, 1986) (11U 624148.65E 5945781.37N, ca. 787m a.s.l.) (Figure 2). The peatland complex is within a non-patterned open minerotrophic peatland system dominated by *Carex* sp., *Salix* sp., *Betula* sp., and *Picea* spp. (Nicholson *et al*. 1992). The greater research area has periodic breaks of peatland vegetation that exhibit minerotrophic marsh characteristics. The restoration site, which was abandoned seven years prior to the experiment, has remnant sedge peat overlaying a clay base from an undulating high land form (Wind-Mulder *et al*. 1996; MacMillan and Pettapiece 2000). The mean annual growing season temperature (May-September) was 14 degrees °C and mean cumulative rainfall was 282 millimeter (mm) over the study period. Strong and Leggat (1981; 1992) describe the area as receiving 70 % of the year's precipitation during the summer months with July being the wettest month.

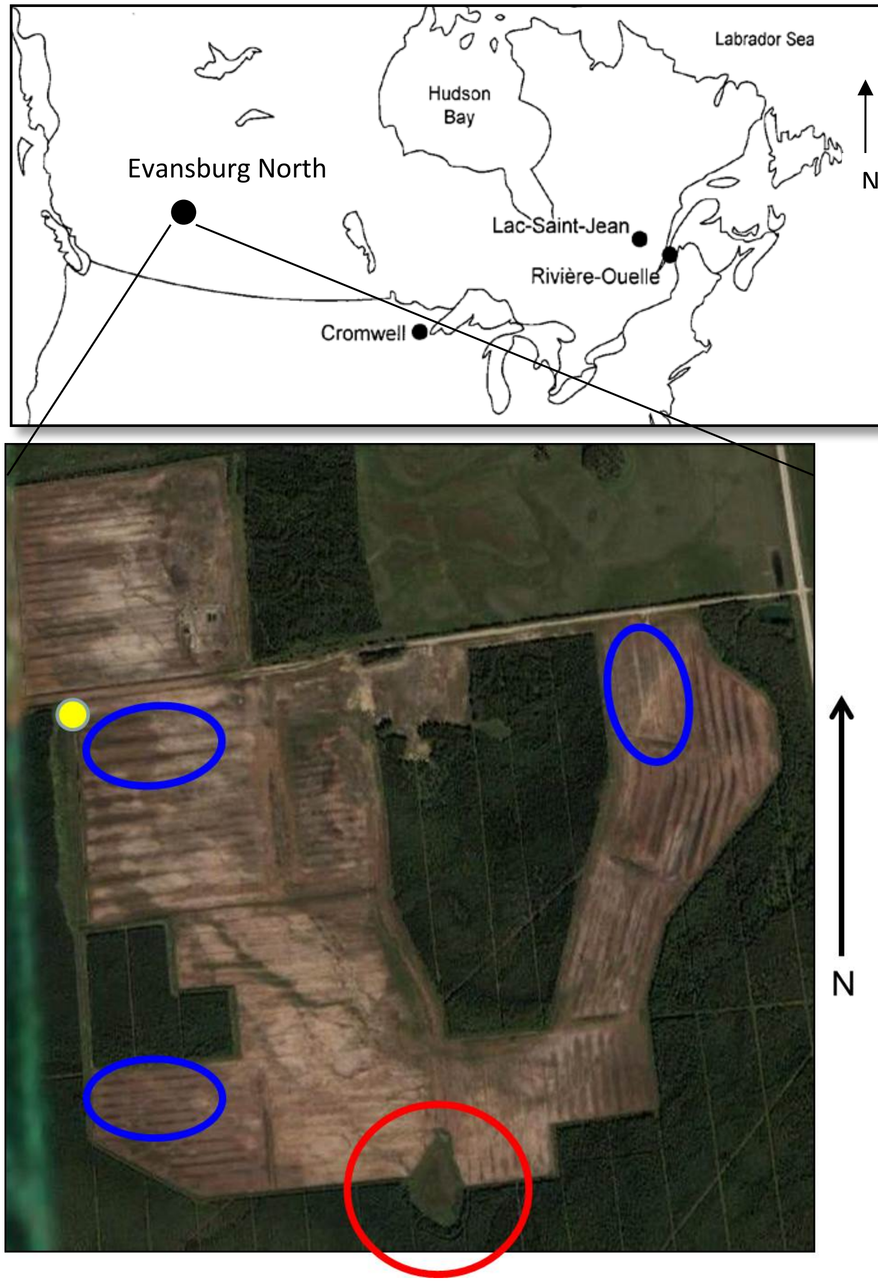

Figure 2. Study site location relative to prominent North American peatland research sites. (yellow circle is the location of the peat damn, blue circles are field test sites and the red circle denotes the donor area)

To meet the ecological requirements for a wet meadow on this cut-away peatland, the main drainage canal was blocked at the northwest corner of the lease with a wet peat dam (Figure 2).

Donor Site

The 1-ha donor site was located on the southern portion of the Evansburg lease (11U 624272.84E 5945167.42N, ca. 786 m a.s.l.) no more than 1.5 km from the treatment areas (Figure 2). The donor site was selected because of similar environmental parameters and proximity to the restoration plots. The pH of the donor site was 4.8 and contained *Carex* sp. and various facultative wetland species with a fringe of *Salix discolour* and *Picea mariana*.

Site Monitoring

Environmental Conditions

I measured local precipitation over the growing period using standard rain gauges and compared these records with an Environment Canada weather station in Entwistle, Alberta, Canada (Environment Canada 2009). The location of the weather station is ca. 11 km southeast of the research site (11U 633658.52E 5940890.11N, ca. 779m a.s.l)

The water level was measured using three water wells placed in each experimental unit to a depth of 1 meter. Depth-to-water was calculated using a conductivity probe and recorded monthly.

After treatments were applied, peat substrate chemical analysis was conducted by Sun Gro Analytical Services (Micro Macro International; Athens, Georgia, United States of America) to quantify the change associated with fertilization treatments and plant spreading.

Vegetation Cover and Composition

Visual estimates of percent cover and species composition were made on the research site during August and September for three years: 2006, 2007, and 2008. Percent cover for facultative wetland species, obligate wetland species, agronomic and upland species, bryophytes and barley were calculated for each experimental unit (Table 1). Vegetation cover and composition were estimated using 16 (100 cm X 100 cm) quadrats per experimental unit. The sampling intensity was lowered in 2008 to eight - (100 cm X 100 cm) quadrats per experimental unit due to preliminary analyses suggesting low variance between quadrats. Vegetation sampling points were located within the perimeter of the experimental units by a minimum of 100 cm to reduce bias based on treatment application and edge effects.

Table 1. Description of the vegetation functional grouping used in experiment (USDA 2009).

Code	Functional Group	Comments
OBL	Obligate wetland	Species that almost always occur under natural conditions in wetlands (estimated probability 99%)
FAC	Facultative wetland	Species that usually occur in wetlands (estimated probability 67-99%) as well as those that are likely to occur in wetlands or non-wetlands (estimated probability 34-67%)
BRYO	Bryophytes	Species of mosses and liverworts
AGRO	Agronomic and upland	Forest, invasive, weed and agricultural species that are typically non-wetland species
BARL	Barley	*Hordeum vulgare* introduced as mulch
PEAT	Bare peat	Non-vegetated areas

Statistical Analysis

A two-factor unbalanced analysis of variance (ANOVA) was conducted using the proc mixed procedure of SAS 9.1 (SAS 2004). The treatments tested included 1.) plant spreading (n=18) or no plant spreading (n=17). The plants were introduced at a ratio of approximately 1:15, and spread to a mean depth of three centimetres; and 2) fertilization (n=19) or no fertilization (n=16), applied at a dose of 17 g m^{-2}. 560-1m^2subsamples (16 per experimental unit) were evaluated annually to capture vegetation cover response. Additionally, the interaction between fertilization and planting was investigated within the treatment structure. A Log (x+1) transformations was used to improve normality of data and reduce heterogeneity of variances to meet assumptions of the model.

I conducted statistical analysis of chemical parameters individually with a generalized linear model (GLM) over replicates and between treatments.

Each 30 m x 30 m experimental unit was leveled, scraped and bermed around the perimeter to remove spontaneous vegetation and enhance water capture. Additionally, mulch cover was standard on all treatment areas and applied at a rate of 3000 Kg ha^{-1}.

Null Hypotheses

1. There will be no significant mean difference in vegetation cover between spreading treatments.
2. There will be no significant mean difference in vegetation cover between fertilizer treatments.
3. There will be no significant mean difference in vegetation richness between spreading treatments.
4. There will be no significant mean difference in vegetation richness between fertilizer treatments.

RESULTS

Vegetation cover

Graminoids, graminoid-likes and forbs were the primary vegetation types established during this experimental restoration with some woody species establishing during the second and third growing seasons. Vegetation cover increased and composition changed from the year of establishment to the third growing season. The main contributors to the wet meadow species composition in the first growing season included *Potentilla norvegica, Bidens cernua, Calamagrostis canadensis, Rorippa islandica* and *Salix sp.* These are typical early successional species found in recently disturbed wet zones and represent constituents in the transferred donor materials. The total vegetation cover increased from 38% in the first growing season to 82% in the second year and stabilized at 78% in the final sampling season in 2008 (Table 2). The second growing season contained the same species as well as additional colonizers: *Festuca saximontana, Carex utriculata, Epilobium angustifolium, Phleum pratense, Poa sp*. and various non-vascular species.

Table 2. Estimated mean percent cover for each growing season including species and cover groups with an estimated mean response >1%. Probable species source is indicated with an "x" based on vegetation surveys and presence within the experimental areas.

Cover Type	% Cover			Presence		
	2006	**2007**	**2008**	**Donor**	**Spontaneous**	**Mulch**
Moss	<1.0	19.1	26.83	x	-	-
Festuca saximontana	<1.0	4.41	13.23	x	x	-
Epilobium angustifolium	<1.0	7.55	5.47	x	-	-
Geum rivale	<1.0	<1.0	3.69	x	x	-
Potentilla norvegica	1.76	10.37	2.54	x	x	-
Carex utriculata	<1.0	1.19	2.39	x	-	-
Galium trifidum	<1.0	<1.0	1.86	x	-	-
Salix sp.	2.8	9.75	1.73	x	x	-
Poa sp.	<1.0	5.04	1.43	x	-	-
Sonchus arvense	<1.0	<1.0	1.21	x	-	-
Lycopus asper	<1.0	<1.0	1.2	x	-	-
Aster puniceus	<1.0	<1.0	1.19	x	-	-
Typha latifolia	<1.0	<1.0	1.03	x	-	-
Calamagrostis canadensis	2.37	3.68	<1.0	x	-	-
Rorippa islandica	2.39	5.46	<1.0	x	-	-
Bidens cernua	6.59	6.29	<1.0	x	-	-
Rumex occidentalis	<1.0	1.07	<1.0	x	-	-
Carex Aenea	<1.0	<1.0	2.88	-	x	-
Populus balsamifera	<1.0	1.3	1.82	-	x	-
Populus tremuloides	<1.0	<1.0	1.63	-	x	-
Mulch	53.57	53.75	54.09	-	-	x
Hordeum vulgare	21.56	<1.0	<1.0	-	-	x
Polygonum convovulvum	1.4	1.11	<1.0	-	-	x
Stellaria media	<1.0	<1.0	1.57	-	-	x
Taraxacum officinale	<1.0	<1.0	1.07	-	-	x
Trifolium repens	<1.0	1.05	1.62	-	-	x
Cirsium arvense	<1.0	1.79	1.59	-	-	x
Phleum pratense	<1.0	2.91	2.23	-	-	x
Total Vegetation Cover	38.86	82.08	78.2			

Table 2 displays potential vegetation species sources in the colonizing of the restoration site: donor site, mulch introduction, or spontaneous colonization from adjacent seed rain. Considering the composition of the vegetation community, Table 2 displays a pattern where greater than 60% of the species established from spreading donor materials. There is some overlap in seed source between these three categories. The straw mulch was a seed source for several agricultural and weedy species including: *Hordeum vulgare, Cirsium arvense, Phleum pratense, Polygonum convolvulus, Stellaria media, Taraxicum officinale, Trifolium repens, and Sonchus arvense*. These species occupied <2% of plant cover and were mostly in the non-spreading experimental units where competition was limited and abundant light and space were available. Table 2 also presents species and cover types with a mean cover value greater than 1% over all experimental units during each sampling year.

Within the spreading treatments, facultative wetland species established a significant cover over the three-year sampling period (Table 3). In the first growing season, the estimated mean percent cover of the facultative wetland species was 22.6%+/- 1.37 SE (Figure 3) and increased three-fold during the second growing to 71.9% +/- 1.86 SE (Figure 3). Over all growing seasons, there was a significant difference in total vegetation cover in the spreading treatments (Table 3). Facultative species cover showed a significant overall response to spreading treatments over all three growing seasons (2006 $F=43.86$, $p<0.0001$; 2007 $F=52.74$, $p<0.0001$; 2008 $F=21.33$, $p<0.0001$) (Table 3). There was no statistically significant influence from fertilization or any interaction between fertilization and spreading treatments with respect to facultative groupings.

The overall effect of spreading on total plant cover was significant during the first growing season ($F=19.65$, $p<0.0001$); during the second growing season ($F=27.82$, $p<0.0001$); and during the third growing season ($F=4.94$, $p<0.0337$) (Table 3). Although fertilization only caused a significant difference in total vegetation cover during 2007, ($F=5.07$, $p=0.0315$) (Table 3), total vegetation appeared denser and more vigorous in the spreading and fertilization sites during all growing seasons.

Table 3: Proc mixed ANOVA output results for the effects of spreading and fertilization on wet meadow species cover and richness functional groupings over three years post restoration (2006, 2007, and 2008). Treatments include Spreading and fertilization (n=10), No Spreading and no fertilization (n= 8), Fertilizer only application (n= 9), and Spreading 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												

Mulch application resulted in a 22% cover of barley (*Hordeum vulgare*) (Table 2) established from seeds contained in the straw in the first growing season. *H. vulgare* did not occur in subsequent years (Figure 3). By the third growing season, there was a decline in several early successional species and a slight shift in species composition: *Bidens cernua, Calamagrostis canadensis*, *Polygonum convolvulus,* and *Rorrippa islandica* all declined to an estimated mean levels lower than 1% of plant cover.

The bryophyte cover group increased from less than 1% (primarily as protonema) overall coverage in 2006 to 19% in 2007 and 27% in 2008 (Table 2). Although not statistically different between treatments, the bryophytes were notable contributors to the vegetation during the second and third growing season. The bryophytes consistently developed in all treatments with or without spreading and fertilization (Figure 3).

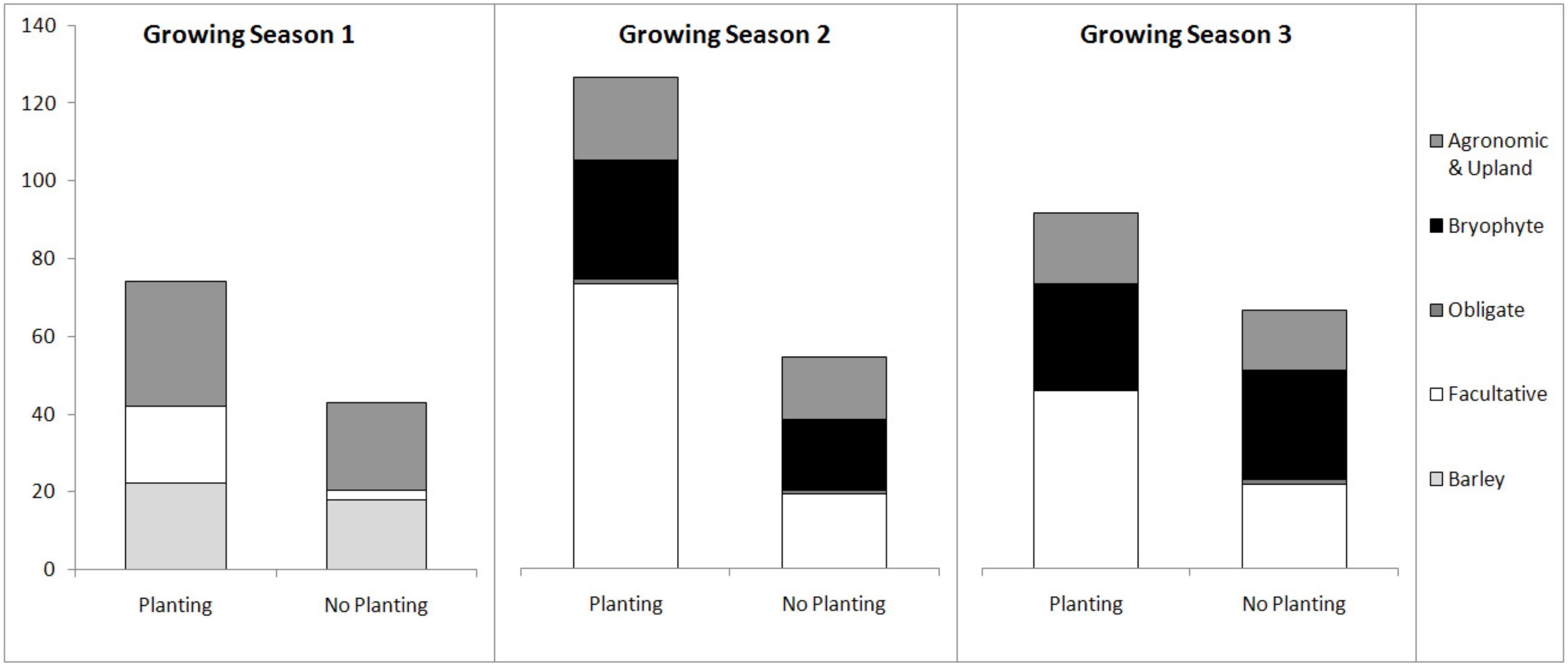

Figure 3: Estimated mean vegetation cover (%) sectioned into associated functional groups relative to growing season and target response type. Treatments include spreading and fertilization (n=10), no spreading and no fertilization (n= 8), fertilizer only application (n= 9), and spreading only, (n=8). Treatments were regrouped to show the effects of fertilizer because there were no significant interactions noted.

Vegetation Species Richness

Absolute species richness within the combined treatments was highest followed closely by the spreading treatment (Table 4). The total species richness for the spreading with fertilization was 36 species in the first growing season, 50 in the second and 43 in the third (Table 4). In contrast, the control species richness was 27 species in year one, 38 in year two, and 34 in year three. There was a significant interaction between fertilization and spreading treatments during the second growing season for both facultative and total species richness (Table 3).

Table 4. Total vegetation species richness (number of species) delineated by treatment groupings and growing season.

Vegetation Species Richness			
	2006	**2007**	**2008**
Spreading x Fertilizer	36	50	43
Spreading	32	47	38
Fertilizer	28	41	37
Control	27	38	34
Total	44	58	45

Spreading provided a consistent and significant gain in facultative and total species richness over all growing seasons (Table 3). The non-spreading species richness consistently displayed the lowest absolute species richness. Facultative species richness showed a significant treatment effect over every year of the field experiment (2006 $F=20.76$, $p<0.0001$; 2007 $F=24.73$, $p<0.0001$; 2008 $F=18.65$, $p=0.0001$) (Table 3). All spreading treatments displayed a positive significant difference (2 and 3-fold) as compared with the non-spreading treatments.

Agronomic species richness did not respond to any treatments during the field experiment; however, it maintained status quo in numbers thereby influencing the group composition. The greatest agronomic influence was observed during the first growing season immediately post-mulch application when barley covered all mulched plots (Figure 3 and Figure 4).

Figure 4 shows the low variability among the facultative species richness between treatments over three years of data collection and richness of facultative species tended to increase from the first through third growing season in all but the spreading treatment. The agronomic species richness remained consistent across the three sampling seasons (Figure 4)

The total species richness between treatments produced an absolute annual gain for all areas; however, the variability displayed in the standard errors is considerably wider than in the facultative species richness (Figure 4).

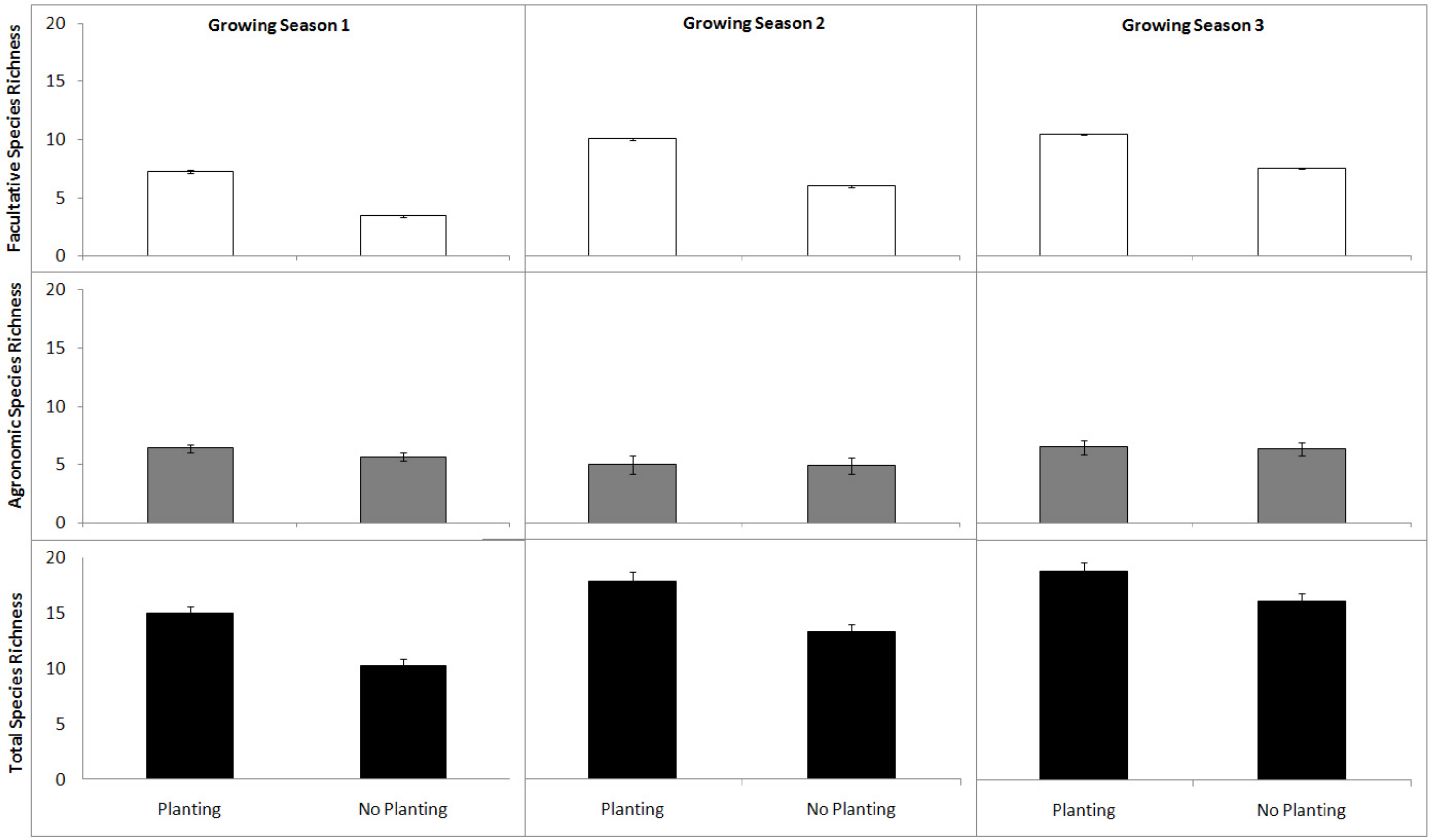

Figure 4: Estimated mean species richness sectioned into associated functional groups relative to growing season and target response type. Treatments include spreading and fertilization (n=10), no spreading and no fertilization (n= 8), fertilizer only application (n= 9), and spreading only, (n=8). Treatments were regrouped to show the effects of planting (see table 3) (Significant interaction found in 2007 for both facultative and total species groupings). Error bars represent standard error.

Precipitation and Ambient Temperature

The growing season precipitation over the research period was considerably lower than the eigthteen year average from the Entwhistle meteorlogical station. The mean precipitation value during June, July and August in the study period were approximately half of the 18-year average, in keeping with a region-wide drying trend. July experienced the greatest disparity with a 50 mm shortfall while September was not significantly different from the 18-year average. The precipitation shortfall was of particular note for 2 weeks post spreading on the experimental units as there was limited precipitation (<5 mm). The bulk of the 50 mm precipitation came in large volume from several short duration storms late during the growing season.

Mean ambient temperature was not significantly different from the 18-year average.

Peat Chemistry and Water Table

Electrical conductivity was significantly different and highest in replicates 1 through 3 with the values ranging between 0.2 and 1.3 µS cm^{-1} (Table 5 row 1) though this is unlikely to be biologically meaningful. Although not significant in treatments, nitrate (NO_3^-) concentrations were significantly higher in replicates 4 through 6 ranging in values from values below detection limit to 7.31 mg l^{-1} (Table 5 row 2). Calcium (Ca+) displayed significantly different and higher values in replicates 1 through 3 ranging from 11.61 to 203.58 mg l^{-1} (Table 5row 5). Following this trend, magnesium (Mg) ranged from 5 through 64.39 mg l^{-1} (Table 5 row 7). Sodium (Na) tracks the same pattern as replicates 1 through 3, having the highest concentrations ranging from 11.45 to 32.5 mg l^{-1} (Table 5 row 9). Parameters not displaying significant treatment effects included pH, copper, potassium, molybdenum, silicon, zinc, nickel, and ammonium (pH, Cu, K, Mo, Si, Zn, Ni, and NH_4). Appendix 1 outlines the physical location of each experimental unit, replicate and treatment type.

Water table depth ranged from 68 cm below the surface to 37 cm above the surface across the greater research site. The mean water table level across the site was 10.13 (+/-28.39) cm below the surface post peat damming. The post damming water table depth was elevated from mean depth recorded by M. Graf during a reconnaissance sampling trip in 2005 (Graf *et al.* 2009) where the depth was recorded at 80.17 (+/- 53.68) cm below the surface.

Table 5. Means and Standard deviations of peat chemistry and water table for the 6 replicates of the wet meadow research project in the Evansburg North Site. Significant parameters are indicated as "R" for significance in replicates and "T" for significance between treatments. n_T=35. Non-significant parameters analyzed include: pH, Cu, K, Mo, Si, Zn, Ni and NH_4.

		1 (n=6)	2 (n=6)	3 (n=6)	4 (n=6)	5 (n=6)	6 (n=5)	Sig
1	**EC (µS.cm^{-1})**	0.50 ± 0.30	0.54 ± 0.26	1.10 ± 0.20	0.10 ± 0.05	0.09 ± 0.03	0.09 ± 0.03	R
2	**NO_3 (mg.l-1)**	0.91 ± 1.47	1.27 ± 0.95	0.55 ± 0.50	3.46 ± 3.85	2.47 ± 1.31	3.95 ± 1.67	R
3	**Al (mg.l-1)**	0.09 ± 0.05	0.08 ± 0.08	0.19 ± 0.21	0.15 ± 0.10	0.22 ± 0.07	0.30 ± 0.13	R
4	**B (mg.l-1)**	0.09 ± 0.03	0.13 ± 0.03	0.24 ± 0.08	0.03 ± 0.02	0.04 ± 0.03	0.05 ± 0.05	R
5	**Ca (mg.l-1)**	58.48 ± 46.84	62.92 ± 33.47	162.92 ± 40.66	4.32 ± 4.29	4.04 ± 2.6	4.53 ± 2.85	R
6	**Fe (mg.l-1)**	0.03 ± 0.03	0.05 ± 0.07	0.17 ± 0.21	0.07 ± 0.05	0.20 ± 0.14	0.29 ± 0.19	R
7	**Mg (mg.l-1)**	19.09 ± 14.09	21.39 ± 11.23	50.88 ± 13.51	2.11 ± 2.57	1.47 ± 1.1	1.66 ± 1.07	R
8	**Mn (mg.l-1)**	0.19 ± 0.28	0.31 ± 0.08	1.89 ± 1.79	0.02 ± 0.01	0.01 ± 0.01	0.02 ± 0.01	R
9	**Na (mg.l-1)**	18.46 ± 7.01	21.20 ± 7.51	27.90 ± 4.60	11.70 ± 3.21	11.38 ± 3.15	11.37 ± 0.7	R
10	**P (mg.l-1)**	1.40 ± 1.38	1.06 ± 0.96	0.16 ± 0.03	1.73 ± 1.36	1.44 ± 1.58	0.67 ± 0.31	T
11	**Cl (mg.l-1)**	9.19 ± 0.51	8.85 ± 0.42	8.60 ± 0.22	9.09 ± 1.00	8.48 ± 0.32	8.32 ± 0.19	T
12	**SO_4 (mg.l-1)**	126.78 ± 95.31	136.25 ± 68.06	347.52 ± 100.53	18.75 ± 6.81	14.45 ± 2.86	12.74 ± 1.53	R
13	**Mean H_2O Table (cm)**	5.48 (+/- 29.85)	16.98 (+/-4.96)	22.70 (+/-13.56)	-31.22 (+/-17.71)	-34.04 (+/-12.01)	-46.82 (+/-11.30)	R
14	**Max Water Table (cm)**	37.40	34.20	33.00	-9.20	-7.80	-29.90	R
15	**Min Water Table (cm)**	-41.47	-13.70	-14.80	-56.80	-53.10	-68.15	R

Donor Site Response

Although monitoring was limited on the donor area, site recovery began during the first growing season. During the second and third growing season it became more difficult to distinguish harvested from unharvested areas. A higher proportion of *Carex* sp. were present on the donor site than on the restoration site.

DISCUSSION

This study assessed a general wet meadow community type rather than a specific species composition; however, a native species composition was preferred over an invasive weedy species community following abandonment. The development of a native wet meadow as a transition towards a productive peatland was successful based on the presence of mid and late seral facultative and obligate wetland species. Fertilization did not significantly contribute to the establishment of native wet meadow species in this study.

The species collected and dispersed from the donor area are assumed to be adapted to periods of inundation and drought as they persisted through the unblocked drainage and active peat extraction years. Several other factors influence the expression, presence and composition of vegetation on the site; including continued seasonal inundation, recreational activity (all-terrain vehicles and equestrian use), historic and adjacent industrial activity, selective herbivory by moose (*Alces alces*) and deer (*Odocoileus* sp), neighbouring invasive species presence, fire history, nutrient type, and nutrient level.

Vegetation Cover

The overall condition of the restoration site was improved with respect to suitable wetland vegetation composition and cover. There was a significant composition of wetland specialized vegetation that encompassed wet meadow species as outlined by the Stewart and Kantrud classification system (Stewart and Kantrud 1971). Movement away from a flora dominated by pioneer species to a more appropriate mid-seral plant community was supported by a slight decline in the rate of change in species composition. This pattern follows closely with the hypothesised successional trajectory outlined in Figure 1. where the ruderal and bare community structure is replaced with mid-seral plant species as a wet meadow develops. Hammersmark *et al.* (2009) suggest early and mid-seral meadow species could adjust to an altered hydrologic regime within a six-year timeline in California, USA. The change from short-lived rapid colonizers to longer-lived constituents within a community suggests a positive response to the restoration treatments on site. After three growing seasons, many of the primary successional vegetation species found on the experimental units occurred at mean cover values lower than 1%, while longer-lived facultative species represent a larger presence in the local community. The mid- and late-seral species were evidence of community stability based on 4 years of establishment time while the longer-lived species were adjusting to the post-restoration conditions. As site environmental parameters are maintained and improved, potential exists for the wet meadow plant species to nurse additional fen vegetation, particularly bryophytes, and restart the peat accumulation dynamics onsite.

Though not anticipated, the consistent increase in the bryophyte community over the three growing seasons sampled was noteworthy. Graf *et al.* (2008) discuss the success of vascular plants over bryophytes following vacuum peat extraction and suggested the need for implementing the *Sphagnum* moss transfer method to increase bryophyte abundance. The vascular community can reduce surface erosion and oxidation while simultaneously enhancing the relative humidity of the peat-air interface. These favourable conditions combined with the spreading of diaspores allowed ground cover establishment by bryophytes similar to the findings of Rochefort *et al.* (2003). Although the bryophytes are currently dominated by early succession species of mosses including *Pohlia* and *Ceratodon* species *Aulocomnium sp., Drepanocladus sp.* and *Polytrichum sp.* are beginning to establish along several of the wetter fringes. Following the development of the peat substrate into an active acrotelm and catotelm system linked to a change in nutrient availability, establishment of *Sphagnum sp.* will likely occur. If *Sphagnum* establishes, an active peatland replacement can begin in earnest under an acidifying and insulating moss layer.

Cooper and MacDonald (2000) suggest restoration methods that transfer rhizomatous materials rather than just seeds provide higher *Carex* establishment potential; hence, the use of a modified *Sphagnum* moss layer transfer method in this project. *Carex* was a dominant genus on the donor area but provided limited cover to the experimental units. This limited cover and lack of dominance in the first few years was not surprising based on studies conducted by Schultz (2000) and Leck and Schultz (2005) suggesting the need for a seed sources and a suitable transfer volume of seed materials to restoration sites for Carex establishment (Leck and Schutz 2005). Since water is the main dispersal method for most wetland carices, the unintentional flooding in the second growing season may have provided a means for additional input of perigyna or the movement of buried *Carex* perigyna to a suitable level for establishment. Durable and resistant seeds of *Carex* may span a longer and species-specific set of establishment conditions. Leck and Schultz (2005) noted seed sources >130 yrs old sprouting upon the creation of favourable conditions; hence, the limited short term carices establishment may be linked to lack of suitable microsites, rather than seed absence on site. Additional concerns regarding the limited initial Carex establishment is linked to the considerably warmer and drier conditions immediately after planting which may have negatively impacted the rhizomatous materials leaving perigyna as the primary propagation method. Dry root zones are a management challenge and usually lead to poor vegetation establishment success (Bakker and Wolff 1995; Wind-Mulder *et al*. 1996; van det Hoek and Braakhekke 1998).

Mulch Cover

The cover of *Hordeum vulgare* (barley) from the straw mulch provided benefits to establishment of the overall vegetation community. The benefit of the temporary overstory is comparable to the benefits noted from cotton-grass (Eriophorum sp.) tussocks mentioned in several papers (Grosvernier *et al.* 1995; Lavoie *et al.* 2003 and Lavoie *et al.* 2005). The presence of barley as a vascular nurse species allowed for a suitable overstory that adjusted the microclimate along the peat-air interface to facilitate vegetation establishment (Grosvernier *et al.* 1995; Ferland and Rochefort 1997) and provide a second wave of mulch in the form of standing dead annual stalks after the first growing season. The second round of mulch assisted in maintaining moisture at the peat-air interface through the growing season and provided a means of snow capture on the open field during the winter months. The snow likely insulated and protected the delicate growing tissues which helped the establishment of the facultative species in subsequent years. Although the barley first-year dominance was an unintentional success story, ruderal introductions near active peat-harvesting operations are not acceptable because their seeds can spread and contaminate commercial peat products. Consequently, this restoration treatment should be applied in closed or contained peat extracted areas where moisture is suitable for target species establishment rather than drier peat extracted areas. Mulch improves suitable growing conditions at the peat surface for bryophytes (Quinty and Rochefort, 1997) and vascular species alike. Trial areas on the Evansburg site showed almost no establishment when mulch was absent.

Fertilization and Flooding

Although no statistical significance was noted with fertilization, Rapid establishment and development on fertilized units as compared with non-fertilized experimental units was perceived. There was an observable difference in vertical structure between the spreading-with-fertilization and all other experimental units early in the first growing season. Functional groups in the non-spreading experimental units did not change over the duration of the experiment.

The higher nutrient levels associated with the fen peat combined with the potential influx of nutrients through flooding may have masked differences between fertilized and unfertilized treatments. Although rock phosphate fertilization is a way to increases vascular plant cover in bog restorations, there is little information about fertilization success or failure in abandoned fen peats (Silva and Pfadenhauer, 1999; Rochefort *et al*. 2003; Groeneveld *et al.* 2007). Graf and Rochefort (2008) suggest that further research is required involving timing and level of fertilization on a larger scale in fen type substrates.

Spreading treatments responded to experimental manipulation by developing wet meadow plant cover. The use of a modified Sphagnum transfer method allowed the capture of both seed and rhizomatous materials to assist in the establishment of vegetation, while the 'fertilized only' and 'control sites' relied primarily on seed movement. The lack of statistical distinction between the fertilized and non-fertilized sites was unexpected. There was no significant difference with the addition of fertilizer to the overall cover on each experimental unit as treatments were applied once during the first growing season. A second application of fertilizer was not conducted due to flooded conditions during the second growing season. The lack of distinction was linked to second season flooding events that mobilized fertilizer and propagule materials across several experimental units. Moreover, the observable difference between sites was reduced with time and represented a convergence in establishment status.

Vegetation Species Richness

There was significantly different species richness between treatments on planted sites. The highest absolute levels of richness were associated with reintroduction treatments while the lowest were in unplanted controls. There was a distinct shift in composition on all the planted experimental units. These experimental units underwent a transformation from mostly ruderal, pioneer or early successional species to mid-seral, longer-lived facultative vascular species. The agronomic species showed consistent, yet, lower, species richness values. The facultative species display a slight; yet, consistently upward trend in species richness across all years in most treatments. This increase is a result of the spreading treatments and an indefinite or ambiguous result of the water retention on site.

At the study onset, the site had little vegetation outside of wet ditches and oxidation cracks. Dead and decaying plant materials were obvious throughout the area and growing conditions appeared particularly harsh. The species richness was low and tended toward scattered weedy or early successional species. The condition on site in year 4 showed very different conditions where vegetation was growing and there was a shift away from early successional vegetation species dominating the system to mid and late seral species. The use of a modified *Sphagnum* moss layer transfer method for the establishment of wet meadow species, combined with blocking the main drainage canal has developed favourable conditions for the first step towards a functional wetland ecosystem.

Donor Site Response

Potential existed for a loss or change of function on the site from which scraped donor material was extracted. The donor area recovered well after the original extraction of transfer materials and will likely be ready for a second harvest of propagules 4-5 years after the first harvest.

Plate 2. Immediately post-roto-tilling conditions along donor area immediately preceding diaspore collection, Evansburg North, Alberta, Canada (2006).

Plate 2. Ameliorated diaspore piles at donor site immediately after scraping diaspores for transport and spreading, Evansburg North, Alberta Canada (2006).

Plate 3. Vegetation recovery within one week of roto-tilling the donor site, Evansburg North, Alberta Canada (2006).

Plate 4. Vegetation recovery one month after roto-tilling the donor site, Evansburg North, Alberta Canada (2006).

Chemical/Nutrient analysis

Based on a mean substrate pH of 4.8, this site falls between a transitional fen and a minerotrophic peatland in chemistry rather than an ombrotrophic peatland system (Moore and Belamy 1974; Vitt *et al.* 1995; Wind-Mulder *et al.* 1996). The associated nutrient conditions are generically classified as a mesotrophic system (Moore and Belamy 1974) that is capable of providing nutrients for a wet meadow system. Wind-Mulder *et al.* (1996) describes the post-extracted peatland condition of a nearby peatland site as a moderate-rich fen. The elevated nitrate levels in replicates 4 through 6 follow with the relative assessment of moisture level, previous research from Wind-Mulder *et al.* (1996), and laboratory studies by Koerselma *et al.* (1993). The dry conditions precluded the direct release of ammonia and allowed the redox potential electron acceptor to move from oxygen gas to nitrates. Although Wind-Mulder *et al.* (1996) acknowledge a different condition, in Meade's (1992) research, this study along with Koerselma *et al.* (1993) support a connection between dry conditions and higher nitrate levels. Further studies on the research area are anticipated to find an elevated level of ammonia in the wettest areas as the anaerobic conditions will favour ammonification and denitrification over nitrification. Vitt *et al.* (1995) found a decrease in ammonia and nitrates along a gradient of bogs to extreme rich fens and attributes this to nutrient uptake and storage by vegetation on the sites. Long-term study of the nitrogen levels on similar sites will likely lead to a better in situ understanding of the decomposition and production relationship. This may facilitate a local origin point for peat accumulation as decomposition in an anaerobic area will be slower than the adjacent oxygenated areas.

The wettest replicates (1 through 3) were linked to higher levels of electrical conductivity, calcium and magnesium whose concentrations were typically greater than undisturbed peatlands. The influx of soluble nutrients through post-extracted flooding combined with the excavated exposure of fen peat is likely responsible for the elevated calcium and magnesium level. The elevated ionic salt level increases the electrical conductivity of the peat substrate. Higher sodium levels in the lowest/wettest sites may have acted as a sodium sink for the highly mobile salt molecules from surrounding areas.

Post-study follow up

Based on qualitative site observations made during 2009 (a fourth growing season without extensive sampling), the site is moving towards a healthy level of cover and the influence of the vegetated areas was spreading into adjacent unplanted buffer zones between the experimental units. In contrast to this condition, the bryophytes had high variability in cover percent over each experimental unit and were establishing more slowly as would be expected based on different reproductive strategies. The bryophyte data is presented as functional groupings rather than individual species because of the high number of protonemal forms which hampered identification until the second season.

Based on the production of different species on site, the hay transfer method (Patzelt *et al.* 2001; Tallowin and Smith, 2001; Norbert and Otte, 2003; Graf and Rochefort, 2008) is not the best method for Alberta cut-away peatlands; a modified *Sphagnum* moss layer transfer method will provide enhancement and more consistent results (Quinty and Rochefort 1997; 2003; Rochefort *et al.* 2003; Graf and Rochefort 2008). The overall trend associated with the treatments suggests that fertilization was not effective at this dosage, timing, and composition.

Although there is a similar cover value between the spreading with fertilization and the spreading alone, further research may be required to evaluate the biomass production and vertical structure to understand the true complexities of the treatments.

Management Direction and Future Research Potential

Provincial regulators, consultants and industrial groups are seeking restoration and reclamation guidelines to help with replacing cut-away peatlands after peat extraction. Actual success must follow clear definitions and timelines for restoration and reclamation while balancing the clarity and efficacy of the goals. Future mitigation of cut-away peatlands depends on understanding the differences between bog and fen restoration and actively targeting landscape-level mosaic development to allow a spectrum of plants and wetland processes during the reclamation and future restoration process. Although this project was focused on vascular plant establishment, the use of a modified *Sphagnum* moss layer transfer method provided the opportunity to observe the successful movement of bryophytes on a larger scale project within fen peat conditions.

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Chapter III General Observations and Management Options

Global Distribution

Peatlands are recognized as major global features; specifically in the northern hemisphere where they range in coverage from 2.4-4.1 x 10^6 km^2 (Gorham 1991; Mitsch and Gosselink 2000; Charman 2002). The range in peatland coverage is likely resultant of the classification differences in peatland organic matter requirements. The International Mire Conservation Group (IMCG) and the province of Quebec regard peatlands as areas of land with greater than 30 cm of peat depth where most Canadian inventories separate peatlands as areas with greater than 40 cm of peat. This coverage equates to approximately 3% of the global land-base (Gunnarsson 2005). Politically, the former Soviet Union and Canada represent the greatest shareholders in the peatland resources of the North. With this recognition, combined with an understanding of the potential impact on carbon cycling and sequestration, several approaches to inventorying the wetland communities have been undertaken in Canada. The National Wetland Working Goup (1988) outlined a peatland land base coverage of 1.27 x 10^6 km^2. In contrast, Tarnocai et al. (2005) estimated the total land coverage of Canadian peatlands to be 1.136 x 10^6 km^2. This estimation follows the NWWG definition of peatlands as an area with 40 cm or more peat accumulation. It may be assumed that the Tarnocai (2005) value is representative of the current Canadian peatland land base occupation as it is more recent.

Although there is discrepancy in estimates of the exact coverage of peatlands from Canadian literature there is broad agreement about the relative scale of coverage and the importance of these wetlands at regional, national and international levels.

Provincial Distribution

The current and generally accepted value for peat land coverage in Alberta was derived from work conducted by Vitt *et al*. (1990). Regionally, Alberta manages 1.03 x 10^5 km^2 of peatland area. This area is attributed to 16.3% of the total Alberta landscape. In contrast, non-peat forming wetlands of Alberta represent approximately 1% of the landscape.

Nicholson *et al*. (1992) outlines seven different peatland community structures based on water source, overstory composition and cover as well as the forms of wetland complexes. Within the summary map the predominant form of peatlands in Alberta are minerotrophic with few small patches of true ombrotrophic peatland areas (Nicholson *et al*. 1992). This unique assemblage of peatlands in Alberta presents several challenges to restoration that are different from eastern Canadian situations.

Differences in Eastern and Western Restoration Approaches

Although there are obvious discrepancies between eastern and western Canadian climatic conditions, the overall approach to restoration is fundamentally the same and information sharing should be pursued as a normal practice rather than an exception. The experience in cut-over peatland restoration by various researchers within the Peatland Ecology Research Group (PERG) should be utilized in Alberta. Although Alberta has several cut-away peatlands, the fundamental process to restoration is similar; the species composition and restoration targets may be slightly different. Success in restoration will involve clear goal setting and reasonable evaluation timelines will need to be addressed as well as a willingness to pursue an adaptive management strategy when processes do not work as intended. If minerotrophic peatland restoration is a goal in Alberta management practices, consideration of the unique chemical and biological needs of the species must be evaluated prior to pursuing restoration. Deeper historic peat deposits that have been removed were typically lower in pH and markedly elevated from groundwater flow events. As peat removal occurs on sites for sale or altered land use needs, the distance to water, nutrient availability and pH is significantly altered in comparison to the original pre disturbed state. Peatland restoration within Alberta, may require a slightly different approach than eastern Canada due to the starting conditions of the post abandoned peatland sites. Wetlands that are precursors to bogs should be considered as alternative jumping off points for site specific restoration projects. Wet Meadows, fen meadows and fens may all be useful options to target as a means to belay further degradation and initiate a vector of development that strives towards peat accumulation in the future. Peatlands along the agricultural white zone fringe in Alberta face additional challenges related to adjacent land use: particularly agricultural fertilization and agricultural clearing. Many

European groups have assessed the impacts of fen restoration in relation to agricultural activities, this data will be instrumental in development of appropriate responses to land use challenges in Alberta (Klimkowska *et al*. 2010a; Klimkowska *et al*. 2010b; Malson *et al*. 2010; Hedberg *et al*. 2012; Kolos and Banaszuk, 2013; Hedberg *et al*. 2013; Schrautzer *et al.* 2013) The afore mentioned projects repeatedly state that combining multiple approaches to restoration will be critical for success in restoration of these degraded fen systems. As such the concepts arising out of the PERG group specific to ombrotrophic peatlands will continue to be very useful with slight modifications in operational practices and management goals.

Management Tools and Key Research Areas for western Canada

Much of the ground breaking work from eastern Canada is transferable to Alberta; however there are several techniques that still require consideration, evaluation and adaptation for Alberta.

Non-Interference Option (do-nothing option)

Lease holders may appear negligent if they do nothing, but a practical case may be made for the non-interference option. By not undertaking restoration, the drainage ditches may remain open, facilitating a consistent drawdown in water levels and the potential for a change in end use of the land. Agricultural activity, rural and urban development, and recreational access may be enhanced by this strategy. If wetland replacement is desired, efforts involving one or more of the following known techniques should be attempted to establish a functional ecosystem on the post extraction area.

Tree and Shrub Removal

As noted in the higher elevation natural regeneration areas on the Evansburg research site, sapling and shrub establishment is extensive. The presence of trees and shrubs increase the site potential evapotranspiration and functionally lowers the available water table; resulting in drier surface conditions than needed for peatland restoration (Malson *et al*. 2010). Careful consideration of overstory removal must be conducted to support the restoration targets or the land-owner goals. Additional research into the benefits of local tree sapling and shrub removal will be required as there are conflicting research results on this topic. Graf and Rochefort

(2010) discuss significant fen bryophyte regeneration under shade nets (greenhouse) and *Scirpus* plants (Field). Although some of this is attributed to the suitable microclimatic conditions present under the overstory (stable microsites, and elevated substrate moisture) not all of the benefits are easily distinguishable. Hedberg *et al*. (2012) consistently displayed benefits to fen species regeneration in clear cut and clear cut combined with rewetting treatments. Specifically, Hedberg *et al*. (2012) addressed the benefit of coverage for five distinct groupings including: *Sphagnum*, wetland bryophytes, wetland vascular plants, grasses and sedges over a ten year sampling period. Malson *et al*. 2010 report the disappearance of characteristic rich fen species when tree and shrub cover was greater than 50%. Not only does the presence of large vascular systems (Trees and shrubs) decrease the availability of water to the surface level of the restoration site (Malson *et al*. 2010), it will limit the incident light required by many fen species. A considerable portion of natural fen communities exhibit open canopies and full light regimes are required for the graminoid and shorter functional groups.

In support of the observations by Graf and Rochefort (2010) the sampling timeline was significantly shorter, at two years in the field and six months in the greenhouse, than the Hedberg *et al*. (2012) ten year sampling period. This may prove extremely useful for operational decision making practices as there is a functional difference in a restoration site two years after planting and at ten years after planting. Additionally, the fen restoration projects discussed by Hedberg *et al*. (2012) see a magnitude of difference occurring after 4 years of observation; previously the clear cut areas were more successful in absolute terms but much closer in relative relationship than the final 6 years of observations.

Propagules and transplanted materials are well documented to be prone to desiccation earlier in their establishment. Operationally, overstory management strategies may be implemented to take advantage of the shading and microclimatic conditions present on some sites earlier in the restoration process. The author suggests 2-4 years post planting overstory removal may be an ideal option that promotes further enhancement of meadow and fen conditions. Removal may be achieved through manual clipping, digging, mowing or chemical means providing resources and support are available for this operation. As an initial response to excessive overstory development on abandoned peatlands, the surface scraping and removal promoted by Rochefort *et al*. (2003) is extremely functional; specifically, for wet meadow and fen restoration as outlined below.

Sowing specific plants

A common method of establishing vegetation in agricultural fields is by sowing or spreading the seeds of species on the ground. Seed collection and sowing differs slightly from a hay transfer by not including mulch; yet may also transfer an unknown composition of species. Wild seed is rarely available for peatland habitats. In areas where suitable seed mixes are commercially available, they are expensive and in limited supply (Pfadenhauer and Grootjans 1999). Seeding with wild seeds is an intensive approach using greenhouse seedlings. Seed material may represent limited genetic diversity and the relatively sensitive seedling stage of the young plants (Holzel, 2005). Seed viability is dependent on species and variety yet it will likely remain low for slow establishing and longer-lived species characteristic of wet meadows. Restoration requires cover by native and indigenous species and this is an unlikely approach on highly degraded peatland sites.

Rewetting strategies

Rewetting is a critical component of peatland restoration and entails considerable planning requirements for success on individual sites yet maintains low input cost and has possible net gain in habitat quality. Rewetting is a necessary step to reduce organic soil degradation and facilitate a movement towards natural ecological functioning. Klimkowska *et al*. (2007) claim this as the oldest method of restoration for drained fen meadows. The use of the term “restoration” in this case may be inappropriate as there is limited research to show the actual convergence of a restored site to that of habitat with similar pre-disturbance quality when rewetting was applied. Van Bodegom *et al*. (2006) agree that the rewetted species composition fluctuates unpredictably and there is limited success over short-term timelines. Rewetting may create conditions where anoxia prevails; preventing the initial establishment and germination of some target and rare species. In cases where rewetting is the sole restoration measure, suitable propagules are required. In heavily disturbed areas, like those of an extracted peatland, the original seed source is lacking or has been overwhelmed by a high proportion of ruderal and invasive species from adjacent lands. To achieve the goals of restoration set out by SER (2002) the ruderal and invasive species may further hinder the process by reducing the available microsites for native species re-establishment.

Establishing natural hydrology on site will increase the ecosystem services of nutrient binding and reduce the mobilization potential of nitrogen and phosphorus as nutrient sinks return to balance (Zak *et al*. 2010). Essentially, rewetting is a crucial step towards nutrient binding through creation of anaerobic substrate conditions that promote slower decomposition that would otherwise result in mobilizing inorganic compounds (Aldous *et al*. 2005).

Extracted sites tend to be manicured in such a manner to remove water rapidly off the peat surface into drainage ditches for removal from site. Individual peat fields will be contoured to facilitate this water movement; as such, re-contouring and potentially terracing of sites will be required to manage the rewetting process. Graf *et al*. (2008) clearly state the need for active management of drainage systems for restoration success. The Evansburg site was blocked with a single point wet peat dam on the main drainage ditch. This provided exceptional access to water at points closest to the dam. However, There was noticeable drying as distance from the dam increased. Quinty and Rochefort (2003) suggest for complete site restoration ditches should be blocked every 75m as general rule of thumb. This will effectively raise the rewet levels to promote reasonably uniform access across the Evansburg site. Klimkowska *et al*. (2010a) comment that historic fen meadow restoration in eastern European countries often only use rewetting although the species richness is dramatically reduced when contrasted with natural sites.

Water source selection for provincial white zone fringe wetlands may provide future challenge as many feeder watersheds have agricultural activity which could promote allocthonous nutrient loading and acidification of the site. I suggest the inclusion of a groundwater well that can be used in conjunction with a sprinkler system to rewet the site initially and saturate the substrate with groundwater based mineral water rather than poorer quality agricultural fed surface water. Although this was not tested to date, the author maintains that there is tremendous potential for promoting fen species and reducing the acidification of the site and potential alteration of chemistry. In areas bordering on ombrotrophic and minerotrophic peat post abandonment there is potential to reduce the overlain peat depth further and bring the growing surface closer to the water table level. Doing this will promote fen and meadow species water and nutrient access.

Top Soil Removal

This is a costly restoration approach that will not provide suitable restoration success if used alone. Topsoil removal is a component of restoration that may aid success associated with further restoration measures. Top soil removal has several potential beneficial attributes: decreased substrate level in relation to the water table, removal of the highly competitive ruderal species composition of the site and moderation of nutrient inputs from previous management regimes prior to restoration efforts.

Topsoil removal may be used to lower the growing substrate to a level that is appropriate for the target vegetation's access to the water table. Patzelt *et al*. (2001) suggest that wet fen meadow restoration requires the water table to reach the growing surface at least 50% of the time and should always be within 80 cm of the surface. Variable depths may be useful on a larger site to increase the potential diversity and improve microclimatic conditions. Variable soil depths may stimulate colonization by different species and enhance the resilience of the site to disturbance and climatic changes. Although there are other techniques available to block water loss and raise the water table, topsoil removal may work well in areas with variable water availability or areas with known water deficits in western Canada.

Famous *et al*. (1991) found cut-away peatlands tend to have rapid colonization and revegetation when contrasted to cut-over peatlands. Graf *et al.* (2008) found this to be true; however, the target *Carex* and bryophyte colonizers tend to be lacking with natural revegetation. In untreated areas, residual and highly competitive ruderal species occupy the restoration site and reduce recruitment gaps which in turn present a challenge to establishing target vegetation (Holzel 2005). Removing vegetation promotes the presence of water on site rather than being lost atmospherically through evapotranspiration. The surface scraping is a retrogression technique that not only removes productive vegetation and increases the water table access; but, may remove the historic seedbank and recent incident spore and seed rain on site to further promote establishment of target species.

One of the final and likely more significant benefits to surface scraping is the inadvertent alteration of site substrate chemistry. By positioning the initial restoration planting materials on a lower substratum of peat, there is a potential for enhanced nutrient access and a pH shift towards supporting minerotrophic over ombrotrophic adapted species. The surface scraping functionally facilitates opportunities for establishment of meadow and fen adapted target species and parallels reforestation concepts of free-to-grow conditions in Alberta with minimal competition during initial growth stages.

With an agricultural management regime there is typically fertilization and species selection on large tracts of land. Topsoil removal as a treatment has succeeded on various grasslands, floodplains, and wet meadow areas where a reduction in soil fertility was targeted (Tallowin and Smith 2001; Patzelt *et al*. 2001; Rasran *et al*, 2007). Tallowin and Smith (2001) found the greatest success in establishing Cirsio-Molinietum communities, on nutrient-poor exposed substrate, with 15-20 cm top soil removal due to the dramatic reduction in total soil phosphorus. Deeper removal of topsoil in fen meadow restoration was beneficial when combined with hay transfers (Klimkowska *et al*. 2010b)

Fertilization

Potash fertilization treatments have been addressed to some extent in southeastern Canada by Sottocornola *et al.* (2007) and there is evidence to suggest that vascular plants are limited by nitrogen and phosphorus in many peatlands (Cobbaert 2003; Cobbaert *et al*. 2004); as such, appropriate blends of fertilizer require testing with fen specific species involved. Care should be taken to ensure elevated background levels are not persistent from adjacent land use practices. Kaplova and Edwards (2011) observed the impact of elevated nutrient inputs maintaining eutrophic states and non-target species for their Czech Republic field study. Further field testing of modified techniques in western Canada is required and should focus on the measureable impacts of fertilization. Fertilization timing, mixture, intensity and type are all components to the questions revolving around the efficacy of fertilization. Additionally, testing the benefits of multiple fertilization events will also be required. Each fertilization trial should be tailored to the propagule introduction techniques and the type of restoration goals pursued. Graf and Rochefort (2008) clearly state the need to address bryophyte fen restoration projects as a means to increase species richness and potential biodiversity production (Vitt 2000). The fertilization regime will ideally augment rather than impede success of the bryophyte structural layer in future projects.

Hay Transfer

A common technique employed in European restoration projects is the application of hay. The relatively easy access and low costs for this technique makes it appealing as a restoration method (Rasran *et al*. 2007). The mulch materials used with the transfer are more suitable when considering local development and conditioning as compared to the use of expensive and hard-to-acquire commercial seed mixes (Pfadenhauer and Grootjans 1999). Hay is collected at a time when the seeds are fully developed to maximize the potential regrowth from the transferred materials. Although this approach has limited impact on the donor area's long term survival, only a portion of the site biodiversity is captured and transferred to the restoration area. Rooting structures are left behind and many low growing species will not be included in the mixture used to propagate the target site. Rasan *et al*. (2006) noted a high rate of germination and establishment in the Carex sp. which is in direct contrast to result presented by Shultz (2000). The successful transfer of a suite of desirable species, from donor area to experimental trial, noted by Rasran et al. (2007) was unclear because of the difficulty in clearly separating transfers from seed rain off adjacent sources. This lack of separation is important to consider; however, there appears to be significant restoration benefit with the use of hay transfers from multiple donor sites (Klimkowska *et al*. 2010b). The transfers should be timed to coincide with the seed developmental stage and consideration should be given to which species are being captured at each transfer area (Rasran *et al*. 2006; Klimkowska *et al*. 2010b). If single transfer events occur from specific donor areas, species composition may be restricted to those species suited to dispersal but may miss a considerable compliment of early maturing species. With any hay transfer restoration plan, care should be taken to reduce the potential selection of species that would otherwise overwhelm the restoration site and subsequently decrease the prospective species establishment to a handful of dominant competitors. Hay transfer methods may not be suited to western Canadian restoration projects as a sole propagule transfer technique. The hay transfer may be an excellent additive prescription to complement other restoration measures. Multiple applications of the technique may be advisable because diaspore materials are typically collected during frozen periods and a secondary collection during vascular seed set could further enhance vascular species.

Bryophyte Introduction

Bryophytes are a crucial component to natural fen function and tend to be left out in traditional European fen restoration hay transfer methods (Malson and Rydin 2007; Graf and Rochefort 2008; Klimkowska *et al*. 2010b) which have unique end goals when contrasted to North America. Access to suitable donor sites and using a modified *Sphagnum* transfer technique with particular attention directed towards brown moss establishment should be tested in Alberta on small to medium scale projects. The introduction of fen and fen meadow bryophytes should be evaluated to ensure success in a wide expanse of substrate conditions. Care should be taken to ensure the bryophytes tested are brown moss and fen species rather than the widely tested bog species. Fen bryophyte testing must also include hydrology, fertilization and various mulch covers.

Adjacent Land Use Management

Success of restoration includes not only a changed vector of retrogression towards a restored community; it includes the impacts and integration of immediate and adjacent land use practices to ensure long-term viability of each project. With respect to rewetting; the concept of lowering the peat substrate further to capitalize on minerotrophic conditions for restoration also reduces the need to further raise the water table with terraces or drainage dams. By skipping the need for significantly elevated water tables, adjacent land use will not experience the same level of impediment that could arise upstream from downstream flooding. Adjacent land may have been altered and adapted based on historic conditions post drainage of the active peat extracting area. In cases where this occurred, joint ventures may be the ideal approach to achieve success for both stakeholders. Furthermore, land transfers and sale may assist each land owner in achieving a suitable balance of ecological, economic and social challenges.

Considerable potential arises with the restoration of healthy peatland communities as they relate to nutrient and water capture and storage. Open discussions between landowners and management agencies may facilitate a greater understanding of the watershed or regional level benefits and cumulative impacts that may be attained through wise use and cooperative management strategies. Peatlands are well documented to be nutrient and water sources and sinks at various times in their development and seasonal status. Creative solutions will be required in many cases where adjacent land use concerns appear at odds.

An additional challenge to address in relation to adjacent land use is the impact of seed rain onto and off of the restoration site. Large scale restoration projects may require completing activities on adjoining peat fields prior to restoration. Fens and wet meadows can be extremely productive systems with high seed and flower production. Kaplova and Edwards (2011) cite several projects in the Czech Republic where wet meadow/grassland dominated communities have as high as 4000 g dry weight $m^{-2}yr^{-1}$ productivities. When combining the potential biomass production of these systems with reproductive success; production peat fields may experience seed rain that could degrade the quality of the product as a direct result of the success of a restoration project.

In situ impacts of restoration on product quality are one aspect of the seed rain; however, a classic consideration would be the input of seeds and spores to a pre-restoration site that reduces the efficacy of treatments on site. Ruderal and invasive species will produce large amounts of seeds and spores for dispersal locally and regionally (Sundberg 2013). This large seed production will initially increase the sites biodiversity with non-target species. Numerous studies have displayed the low dispersal and success of target facultative and obligate fen and meadow species (Middleton *et al*. 2006; Klimkowska *et al*. 2010a; Sundberg 2013) while invasive and weedy species are prolific.

Management Tools Summary

Mulch cover has been demonstrated to provide success in many cut-over peatlands in Eastern Canada by Rochefort *et al*. (2003) and several peatland Ecology Research Group projects. In cut-away peatlands, I gained insights into the restoration potential by contrasting areas with and without mulch cover and concluded mulching to be a beneficial practice to enhance not only the potential bryophyte coverage but also the vascular floral component. On the Evansburg site mulch was consistently applied across all experimental units; however a few smaller pilot areas outside of the experimental units were left without mulch to test the importance of the mulch application. I observed that in areas where there was mulch application vegetation developed with planting. Additionally, in areas where there was no mulch application *Equisetum arvense* was the only floral development on the substrate.

Whichever vegetation cover is utilized, local donor materials are required to ensure the natural genetic variability of the donor materials will be adapted to the local climatic conditions of the site. The use of local materials promotes the assertion that natural dormancy cycles of reproductive materials will be aligned to the restoration efforts of the site. Materials should be collected and spread in the spring to optimize the natural dormancy cycles of the donor materials. Fall prescriptions may be appropriate if there is limited concern about natural dormancy issues and donor site access. This may be the case where there is limited need for Carex sp. establishment.

During the process of field sampling, a suitable donor site with vascular and non-vascular fen vegetation was observed on the Sungro lease. This site is due west of the cut-away peatland in a reasonably undisturbed fen (Zone: 11 Easting: 622187 Northing: 5945345). Access to this donor area is provided with a cutline and will require frozen conditions for travel. In future development and peatland openings, pre-determination of donor site locations should be conducted to ensure enough volume associated to suitable species is available for restoration efforts. Not only is the location of principle concern, but the protection of natural function during the peat extraction period is instrumental to future restoration success.

Vision for Alberta Peatland Restoration

Alberta peatlands are challenged on numerous development fronts; recreational, industrial, agricultural and urban. With the challenge comes the need for effective management of these resources and critical assessment of long-term goals and benchmarks for success. Eastern Canadian peatland restoration has undergone numerous research projects on varying scales dedicated heavily towards cut-over peatlands. Although there are some direct linkages with respect to techniques from Eastern Canada, direct assessment of these techniques on fens and wet meadows is required. Modification and alteration based on site conditions and species composition will be required to fully integrate these techniques and provide sound management tools. In conjunction with these tools, clearly establishing long term goals will be paramount to the success of restoration efforts in Western Canada. I recommend the establishment of early peatland successional communities as a critical link in development of minerotrophic peatlands. The idea of preventing further degradation should be given more credence than immediate and full restoration as a stopgap measure for altering the vector of restoration success. Wet meadows, fen meadows and wet grasslands are all suitable primary communities that can develop with time and appropriate management into full peatlands. Although this should not be prescribed as a blanket measure to restoration, the techniques are ideally suited for restoration in heavily degraded peatlands and significantly cut-away areas. Initiating an ombrotrophic peatland restoration program in conditions that are not ideal based on pH, temperature, moisture and nutrients will prove to be a costly venture. The techniques described and tested during this project will increase success rates and restoration potential for a number of highly degraded western peatlands.

Continued use of the adaptive management strategies often implemented in restoration initiatives is highly recommended as continued learning will provide additional tools for management on various peatland sites. Testing additional techniques and variations on tried and tested methodologies will be essential as climatic changes and managerial agendas shift over time. Low-cost high return long-term planning initiatives that support the spirit of the Wise use of Mires and Peatlands (Joosten and Clarke 2002) will provide the greatest benefit to restoration success in Alberta peatlands. Cost effectiveness and efficacy will include pre planning of extraction efforts combined with donor selection areas. With the inclusion of donor areas in peatland opening processes, overall project costs will be reduced as restoration success can be increased using geographically localized planting materials and minimal transportation of propagules.

Linking various partners in the peatland restoration may prove to be a successful approach for western Canada. Furthermore, the development of a demonstration site that has ready accessibility to academic researchers, public and industry with the potential for numerous treatment techniques within a reasonable small area will be a positive step towards show casing various successes to a wide breadth of audiences. The Evansburg north site would be a potential candidate for such a venture and could include funding opportunities from oilsands, Conventional oil, forestry, agricultural agencies and organizations. Interdisciplinary models to restoration are an appropriate means to achieve success and are quite feasible in the economic and social climate of Western Canada.

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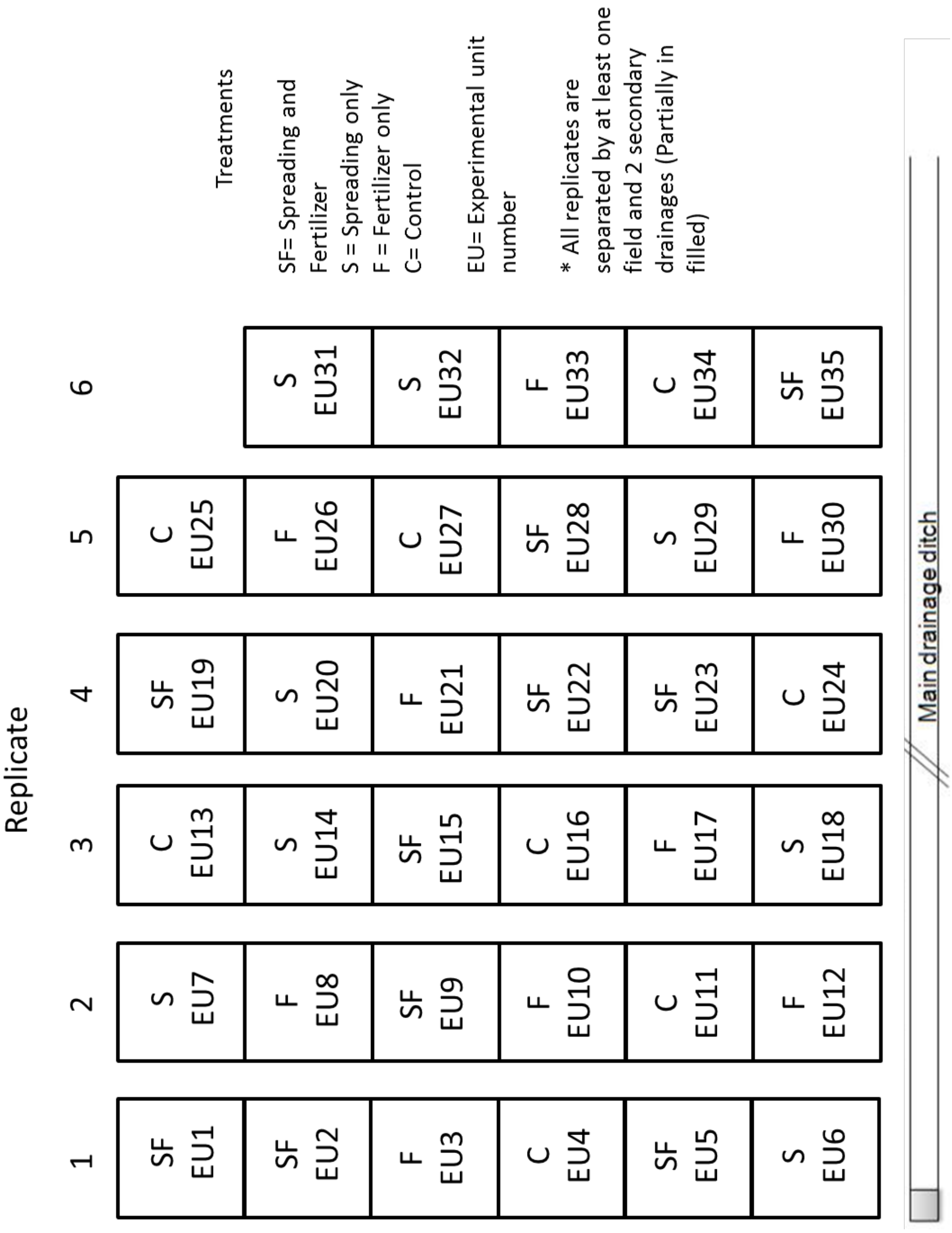
APPENDIX

Appendix 1. Experimental layout and treatment position for research project



Appendix 2. Mean cover class organized by functional association (2006).

	Mean Cover Value								
Exp unit	Obligate	Facultative	Agronomic	Bryophyte	Bare Peat	Mulch	Water	UNID	Total
1	0.0	23.3	25.0	0.0	4.5	47.8	0.0	1.5	102.5
2	0.0	34.1	32.3	0.0	4.0	26.9	0.0	2.1	99.3
3	0.0	1.7	19.7	0.0	6.0	72.4	0.0	1.7	101.5
4	0.0	3.6	13.2	0.0	6.9	73.6	0.0	1.6	98.8
5	0.0	51.6	27.9	0.0	3.8	17.3	0.0	2.5	103.1
6	0.0	4.0	7.0	0.0	24.1	66.1	0.0	2.5	103.7
7	0.0	2.8	37.9	0.0	3.2	57.2	0.0	0.2	101.5
8	0.0	4.1	9.7	0.0	3.4	83.0	0.0	0.9	101.1
9	0.0	6.9	31.6	0.0	2.6	61.4	0.0	1.6	104.4
10	0.0	2.0	13.0	0.0	5.9	70.8	0.0	1.1	92.8
11	0.0	7.8	46.3	0.0	1.3	37.7	0.0	1.8	94.9
12	0.0	0.9	5.1	0.0	2.5	84.6	0.0	0.0	93.6
13	0.0	4.7	11.7	0.0	9.0	73.9	0.0	0.0	99.3
14	0.0	16.4	9.9	0.0	4.5	73.6	0.0	0.8	105.1
15	0.0	44.7	9.3	0.0	2.9	63.9	0.0	0.4	121.2
16	0.0	14.2	5.0	0.0	2.3	59.6	0.0	0.2	81.3
17	0.0	9.8	15.5	0.0	1.9	64.9	0.0	0.2	92.5
18	0.0	26.0	9.9	0.0	8.7	53.6	0.0	0.8	99.0
19	0.5	4.2	32.5	0.0	11.2	46.1	0.0	0.6	94.9
20	0.0	82.9	37.6	0.0	0.8	7.1	0.0	2.6	131.0
21	0.0	2.9	12.3	0.0	7.1	70.5	0.0	1.3	94.1
22	0.0	37.8	76.3	0.0	1.2	10.1	0.0	1.4	126.8
23	0.0	49.5	61.2	0.0	3.1	19.3	0.0	0.9	134.0
24	0.0	0.9	15.4	0.0	65.5	36.2	0.0	0.8	118.8
25	0.0	2.0	32.1	0.0	0.8	63.3	0.0	0.0	98.2
26	0.0	2.4	66.5	0.0	5.0	34.5	0.0	0.4	108.8
27	0.0	0.0	16.2	0.0	0.8	75.2	0.0	0.8	92.8
28	0.0	11.3	66.3	0.0	1.4	41.4	0.0	0.8	121.2
29	0.0	1.5	13.1	0.0	3.1	81.4	0.0	0.4	99.5
30	0.0	1.4	28.9	0.0	12.8	66.3	0.0	0.4	109.8
31	0.0	31.0	16.2	0.0	1.0	70.5	0.0	0.6	119.3
32	0.0	11.2	43.6	0.0	0.0	59.6	0.0	0.6	114.8
33	0.0	1.1	46.3	0.0	0.2	66.1	0.0	0.4	114.1
34	0.0	0.2	29.1	0.0	0.2	75.2	0.0	0.0	104.6
35	0.0	17.6	43.3	0.0	0.8	63.9	0.0	0.9	126.5

Appendix 3. Mean cover class organized by functional association (2007).

	Mean Cover Value								
Exp unit	Obligate	Facultative	Agronomic	Bryophyte	Bare Peat	Mulch	Water	UNID	Total
1	0.0	62.4	64.3	37.3	0.0	37.3	0.0	2.3	204.4
2	1.9	130.1	19.6	52.9	0.0	48.5	0.0	0.0	253.1
3	0.0	41.8	3.4	11.6	0.0	54.8	0.0	0.0	111.6
4	1.3	23.2	0.4	17.1	10.8	37.8	20.8	0.4	111.7
5	1.9	112.7	7.3	38.3	0.0	26.6	29.6	0.4	216.6
6	0.9	56.1	1.9	10.2	8.8	46.0	35.8	0.4	160.1
7	0.0	66.6	18.3	29.9	0.0	35.3	4.4	0.0	156.6
8	9.0	26.1	0.4	21.0	1.9	66.1	26.3	0.0	150.8
9	8.8	26.0	0.0	2.1	8.8	58.3	72.9	0.0	177.3
10	5.3	24.6	0.0	12.1	3.9	79.9	89.3	0.0	214.9
11	5.2	38.3	0.0	8.7	1.9	69.8	54.4	0.0	178.3
12	3.9	32.4	7.0	27.9	1.9	57.3	0.0	0.0	131.6
13	0.0	29.8	10.3	5.1	5.8	57.9	0.0	0.0	109.3
14	7.4	64.2	1.9	21.4	22.1	13.3	14.2	0.0	144.4
15	12.7	96.7	0.9	23.6	0.0	26.5	25.2	0.0	185.6
16	6.4	16.1	0.0	0.8	4.4	69.3	86.1	0.0	183.1
17	4.1	26.5	0.4	0.5	0.6	73.5	73.0	0.0	178.8
18	5.5	130.4	1.9	33.3	0.0	30.9	8.8	0.0	210.9
19	0.0	29.0	37.4	53.2	8.8	48.6	0.0	0.0	177.0
20	0.0	114.0	51.6	62.1	0.0	48.6	0.0	0.0	276.3
21	0.0	9.6	37.1	35.8	13.3	52.9	0.0	0.0	148.7
22	1.9	115.8	44.8	47.2	0.0	52.9	0.0	0.0	262.6
23	0.0	99.9	29.0	50.5	0.9	70.5	0.0	0.0	251.8
24	0.0	8.4	16.2	12.2	25.8	44.1	0.0	0.0	108.5
25	0.0	11.9	10.2	41.7	6.4	70.5	0.0	0.0	140.7
26	0.0	28.5	60.6	55.9	0.0	61.8	0.0	0.0	206.7
27	0.0	2.1	27.4	18.5	9.7	70.5	0.0	0.0	128.2
28	0.0	57.4	51.8	64.0	0.8	48.6	0.0	0.0	222.6
29	0.0	13.8	18.1	9.4	17.0	70.5	0.0	0.4	129.6
30	0.9	24.1	39.0	12.1	7.7	48.9	0.0	0.4	133.4
31	0.0	49.4	9.2	5.9	16.2	70.5	0.0	0.0	152.4
32	0.0	48.8	22.8	16.0	5.2	70.5	0.0	0.8	164.1
33	0.0	25.4	24.1	3.6	5.5	61.7	0.0	6.2	127.3
34	0.0	14.1	15.9	3.1	7.0	69.4	0.0	0.4	110.0
35	0.0	50.9	22.4	2.6	10.2	61.7	0.0	0.0	148.7

Appendix 4. Mean cover class organized by functional association (2008).

	Mean Cover Value								
Exp unit	**Obligate**	**Facultative**	**Agronomic**	**Bryophyte**	**Bare Peat**	**Mulch**	**Water**	**UNID**	**Total**
1	0.0	77.9	21.1	30.5	16.1	43.6	0.0	0.0	190.1
2	1.9	70.2	20.7	33.0	3.8	48.6	0.0	0.0	178.2
3	8.8	18.7	28.4	24.5	20.0	41.8	0.0	0.0	142.1
4	4.8	21.6	19.4	33.9	19.6	36.4	0.0	0.0	135.6
5	2.6	61.6	17.1	38.3	3.2	44.3	0.0	0.0	167.1
6	4.1	35.4	17.9	18.4	36.6	21.1	0.0	0.0	133.4
7	0.0	58.1	17.9	35.5	2.1	56.8	0.0	0.0	172.2
8	8.3	34.4	4.3	16.5	3.8	66.1	0.0	0.0	133.4
9	5.3	37.4	7.8	31.3	20.4	43.6	0.0	0.0	146.2
10	14.9	28.8	6.4	13.5	4.8	66.1	0.0	0.0	134.6
11	16.9	40.6	6.6	29.3	11.2	40.6	0.0	0.0	145.2
12	4.1	49.5	14.3	36.4	5.8	61.8	0.0	0.0	173.1
13	0.0	8.9	13.2	8.3	8.4	57.4	0.0	0.0	96.6
14	0.0	38.3	19.4	2.6	12.9	26.3	0.0	0.0	99.4
15	0.4	52.5	4.8	4.6	3.9	64.9	0.0	0.0	131.0
16	0.0	27.4	4.8	11.1	40.9	30.9	0.0	0.0	115.0
17	1.9	41.3	2.3	4.0	6.1	65.5	0.0	0.0	122.0
18	0.0	45.8	18.1	11.4	10.3	61.8	0.0	0.0	147.3
19	0.0	35.8	22.6	61.8	13.7	25.4	0.0	0.0	159.3
20	0.0	69.8	38.9	57.8	16.2	46.1	0.0	0.0	228.7
21	0.0	11.2	13.3	28.4	41.3	37.6	0.0	0.0	131.8
22	0.0	48.4	24.1	32.9	13.7	50.5	0.0	0.0	169.6
23	0.0	58.9	10.6	57.4	12.8	39.8	0.0	0.0	180.4
24	0.0	42.5	22.5	32.9	2.8	73.6	0.0	0.0	176.3
25	0.0	15.5	10.8	48.9	2.6	61.8	0.0	0.0	139.6
26	0.0	15.8	22.9	62.9	1.9	57.4	0.0	0.0	160.8
27	0.0	7.5	28.7	53.4	5.8	63.6	0.0	0.0	158.9
28	0.0	17.7	29.8	66.0	9.1	30.4	0.0	0.0	152.9
29	0.0	15.6	17.8	28.0	8.1	45.5	0.0	0.0	115.4
30	0.0	17.3	18.9	21.6	16.3	57.7	0.0	0.0	132.1
31	0.0	34.1	8.5	8.6	5.2	70.5	0.0	0.0	128.8
32	0.0	35.9	16.8	8.4	9.1	57.0	0.0	0.0	127.2
33	0.0	20.7	24.8	8.9	3.8	73.6	0.0	0.0	132.2
34	0.0	13.9	9.4	11.5	2.1	70.5	0.0	0.0	107.3
35	0.0	32.8	15.1	8.6	7.1	53.0	0.0	0.0	117.4

Appendix 5. Substrate nutrient analysis and mean water table depth on experimental units.

Experimental Unit	Field (Replicate)	Mean Water Depth (mm)	pH	Electrical Conductivity	NH_4	NO_3	Al	B	Ca	Cu	Fe	K	Mg	Mn	Mo	Na	P	Si	Zn	Cl	SO_4	Ni
1	1	-31.3	4.60	0.10	1.18	3.80	0.1056	0.0657	5.6354	0.0235	0.0847	7.6082	1.7816	0.0418	0.0243	7.8530	3.7451	1.1658	0.0222	8.9600	16.1100	0.0291
2	1	49.7	4.65	0.35	1.41	0.90	0.1400	0.1002	33.5165	0.0074	0.0258	6.2368	12.3837	0.0367	0.0045	16.3459	1.5259	1.6409	0.0038	9.5600	78.0000	0.0117
3	1	29	4.51	0.43	0.70	0.00	0.1016	0.0576	46.4728	0.0000	0.0165	9.2068	14.6075	0.0727	0.0000	14.1892	0.5049	1.0173	0.0030	9.0900	99.6500	0.0034
4	1	82	4.54	0.97	1.05	0.00	0.0553	0.1145	137.6489	0.0000	0.0118	9.0766	41.7218	0.7448	0.0002	24.6284	0.3283	0.6159	0.0056	8.8900	286.2800	0.0031
5	1	164	4.81	0.43	2.34	0.75	0.0101	0.0670	40.5078	0.0000	0.0078	9.0887	15.0119	0.0491	0.0000	21.2886	2.1238	1.3765	0.0003	10.0100	94.2300	0.0058
6	1	35.5	4.48	0.71	1.05	0.00	0.1029	0.1247	87.1110	0.0000	0.0280	7.3388	29.0500	0.2034	0.0000	26.4737	0.1547	0.9466	0.0001	8.6300	186.3800	0.0085
7	2	89.5	4.24	0.42	0.00	2.42	0.0940	0.1188	48.9963	0.0000	0.0398	4.0085	15.2651	0.3576	0.0017	14.9562	0.6538	0.4908	0.0010	8.8000	98.3000	0.0040
8	2	182	4.57	0.73	0.00	0.27	0.0046	0.1534	94.4914	0.0000	0.0059	6.6606	30.1535	0.3588	0.0000	22.0103	0.0588	0.3601	0.0043	9.1700	192.6900	0.0007
9	2	246	5.19	0.77	5.19	1.49	0.0547	0.1404	87.3291	0.0000	0.0279	8.1804	30.5267	0.3685	0.0024	30.5187	2.2961	1.8959	0.0012	9.4900	196.4200	0.0145
10	2	217	5.20	0.69	1.87	0.00	0.0607	0.1090	80.7145	0.0000	0.0120	8.6437	27.2114	0.3326	0.0037	26.3860	0.4951	1.0780	0.0011	8.7700	173.2400	0.0042
11	2	220	4.73	0.56	0.00	1.50	0.0198	0.0885	62.2400	0.0000	0.0073	5.6434	23.6507	0.2699	0.0010	23.2655	0.5946	0.7066	0.0017	8.4700	136.9700	0.0084
12	2	64.5	5.09	0.08	0.00	1.96	0.2180	0.1690	3.7658	0.2050	0.1901	4.9682	1.5565	0.1744	0.1610	10.0334	2.2486	1.6242	0.2005	8.3800	19.8600	0.2136
13	3	-38.5	3.66	1.09	3.04	0.00	0.5532	0.2198	142.9911	0.0000	0.2602	6.9361	51.6534	2.5610	0.0105	23.8756	0.1310	1.3614	0.1142	8.2500	315.1400	0.0093
14	3	244	4.61	1.10	2.50	0.48	0.0499	0.1755	172.8296	0.0007	0.0220	8.2380	43.0143	0.6710	0.0063	23.9960	0.1399	0.9954	0.0205	8.7500	337.2100	0.0115
15	3	297	5.05	0.89	0.73	0.00	0.0000	0.1735	122.8783	0.0000	0.0058	8.4015	38.4471	0.2947	0.0018	25.5864	0.1655	0.9593	0.0079	8.7700	246.2900	0.0056
16	3	263	4.68	1.31	0.00	1.26	0.0745	0.3513	198.5116	0.0000	0.0626	6.0067	67.6934	2.9303	0.0000	35.3976	0.1642	1.1779	0.0080	8.6200	448.3500	0.0083
17	3	280	4.53	1.34	0.18	0.68	0.1609	0.3213	219.1308	0.0000	0.5481	7.7959	66.5197	4.6589	0.0008	31.4187	0.2059	1.5850	0.0217	8.8000	487.4600	0.0000
18	3	318	4.85	0.88	0.00	0.88	0.2972	0.1814	121.2062	0.0000	0.1040	5.7160	37.9339	0.2067	0.0013	27.1544	0.1425	1.6382	0.0079	8.4300	250.6900	0.0108
19	4	-442	4.92	0.07	0.00	3.19	0.3141	0.0156	2.1121	0.0092	0.1340	8.4317	0.7728	0.0094	0.0000	11.3436	2.8394	1.1664	0.0009	8.7900	13.4300	0.0191
20	4	-350	4.81	0.06	0.00	1.27	0.0666	0.0204	1.3105	0.0079	0.0082	4.3960	0.2692	0.0077	0.0026	10.3939	1.3498	0.6072	0.0000	10.9400	16.7600	0.0082
21	4	-258	4.45	0.07	0.00	0.72	0.0519	0.0216	3.2641	0.0000	0.0505	4.6866	1.8097	0.0172	0.0006	7.8746	0.3029	0.3723	0.0001	8.2100	17.5700	0.0072
22	4	-281	4.64	0.11	0.00	1.83	0.1482	0.0657	3.9823	0.0161	0.1237	8.7685	1.5863	0.0336	0.0127	13.8273	2.1474	0.6430	0.0161	8.9900	19.7900	0.0065
23	4	-260	4.51	0.08	0.00	2.65	0.1249	0.0240	2.3884	0.0149	0.0782	7.5915	0.9878	0.0087	0.0000	9.8722	3.5299	0.6571	0.0017	9.3600	13.2600	0.0070
24	4	-283	4.59	0.19	0.79	11.10	0.1946	0.0274	12.8813	0.0090	0.0397	4.3540	7.2334	0.0364	0.0032	16.8988	0.1914	0.6556	0.0034	8.2700	31.6600	0.0087
25	5	-388	5.14	0.10	0.00	1.73	0.1482	0.0853	3.2888	0.0185	0.1347	6.4660	1.0056	0.0086	0.0030	13.1874	0.1030	1.4269	0.0034	8.4200	16.8100	0.0074
26	5	-341	4.71	0.06	0.00	3.60	0.1820	0.0000	1.1536	0.0086	0.0513	6.2027	0.6000	0.0063	0.0009	7.0355	2.3627	0.6087	0.0008	8.1500	15.1700	0.0131
27	5	-332	5.20	0.07	0.00	0.88	0.2475	0.0430	2.4659	0.0090	0.2705	7.2345	0.5736	0.0085	0.0047	9.6500	0.8637	0.8820	0.0013	8.4800	11.4700	0.0075
28	5	-409	4.58	0.11	0.00	3.44	0.2232	0.0514	4.2011	0.0161	0.2656	10.3268	1.4505	0.0292	0.0045	13.6363	4.3420	0.9414	0.0015	9.0100	14.0000	0.0297
29	5	-268	5.17	0.09	0.00	1.43	0.3409	0.0627	6.9652	0.0089	0.4547	11.6097	2.4308	0.0209	0.0000	9.8454	0.9081	1.1709	0.0010	8.2300	12.3800	0.0007
30	5	-305	4.94	0.14	0.65	3.98	0.2575	0.0512	7.8294	0.0173	0.1232	7.6551	3.3241	0.0121	0.0022	15.5106	0.5073	1.3922	0.0020	8.7000	19.1800	0.0109
31	6	-429	4.96	0.07	0.19	2.23	0.1561	0.0083	2.3655	0.0106	0.1066	5.5703	0.8992	0.0035	0.0000	10.7811	0.9630	1.0386	0.0000	8.4000	12.1200	0.0086
32	6	-493	5.07	0.08	0.00	3.68	0.4292	0.0620	3.6931	0.0360	0.3684	6.8260	1.1241	0.0266	0.0201	11.5428	0.6022	1.4185	0.0107	8.0900	11.5800	0.0038
33	6	-416	4.79	0.14	0.34	6.29	0.4333	0.1274	9.3034	0.0054	0.5244	9.1195	3.4249	0.0277	0.0000	12.3209	0.2375	1.4938	0.0058	8.1800	15.4300	0.0141
34	6	-361	4.96	0.09	1.29	4.90	0.2782	0.0326	4.8358	0.0176	0.3588	14.1301	1.8894	0.0278	0.0029	11.6067	0.9915	1.0377	0.0036	8.3800	12.3600	0.0000
35	6	-642	5.00	0.07	0.44	2.64	0.1823	0.0276	2.4572	0.0121	0.0832	4.4115	0.9485	0.0070	0.0034	10.5930	0.5730	1.1019	0.0064	8.5600	12.1900	0.0146