THESIS

APPROACHES FOR CREATING SUSTAINABLE BIOMASS PRODUCTION IN A RECLAIMED FEN IN THE ALBERTA OIL SANDS REGION, CANADA

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Lewis Erwin Messner

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Advisor: David J. Cooper

Mark Paschke
Cynthia Brown

ABSTRACT

APPROACHES FOR CREATING SUSTAINABLE BIOMASS PRODUCTION IN A RECLAIMED FEN IN THE ALBERTA OIL SANDS REGION, CANADA

Oil sands surface mining in the boreal region of Alberta, Canada partially alters the natural hydrologic processes, vegetation, and geochemistry of affected ecosystems. Peat accumulating bogs and fens cover approximately 30% of the oil sands region and function as long-term carbon sinks. The government of Alberta has legislated that disturbed areas be reclaimed to "*equivalent land capacity*." However, no guidelines exist for reclaiming peatlands in post-mined landscapes. A pilot fen was constructed on Suncor's Millennium oil sands mine in 2013 and I analyzed the effects on annual biomass growth by introducing plants as seeds, seedlings, and from locally harvested propagules. Total above-ground biomass (AGB) in year five was 460 ± 30.7 g m^{-2} (*n* 56) and was comparable to natural fens in the region. Total live below-ground biomass (BGB) averaged 1640 ± 99.9 g m^{-2} (*n* 56) by year five, falling slightly below ranges for regional fens. When averaged across all treatments, *C. aquatilis* produced the greatest AGB (404 ± 32.8 g m^{-2}, *n* 56) in year five and represented over 70% (*se* 1.520) of the site total biomass. AGB of *C. aquatilis* and *J. balticus* and total AGB and BGB were positively correlated with water table depth. Total BGB was positively correlated with electrical conductivity. *Typha latifolia* AGB was significantly affected by removal treatments when averaged across sampling years. Plant derived carbon inputs to reclaimed peatlands and long-term storage are characterized in part by the effects of abiotic variables on vascular plant biomass. Results from this study provide guidance for evaluating reclaimed post-mined fens in Alberta, Canada.

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DEDICATION

"*If I have seen further it is by standing on the shoulders of Giants*."

~ Sir Isaac Newton, 1675.

I dedicate the completion of my masters to my life-partner, Claire.

I see further because of you.

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INTRODUCTION

The oil sands region of Alberta, Canada, contains the Earth's third largest petroleum deposit, which underlies 142,200 km^2 of the western boreal forest (Government of Canada, 2015). Peatlands occupy 30% of the region's land cover and are predominantly groundwater-driven fens (Vitt et al., 2000). Approximately 3% (4800 km^2) of oil-sand deposits can be mined from the ground surface and development can permanently change peat-forming processes by removing overburden materials. Alberta law requires mining companies to replace disturbed habitat to, "*equivalent land capability*" (RSA, 2016), yet previous reclamation efforts have largely targeted upland and marsh habitat resulting in a net loss of peatland (Rooney, Bayley, & Schindler, 2012). Understanding the effects of post-mined conditions on natural long-term vegetation biomass production and potential carbon-storage processes is key for successfully reclaiming peatlands in the oil sands region.

In North America, peatlands have formed primarily in high latitude landscapes with low topographic relief and where precipitation exceeds evapotranspiration (Sjors, 1959; Walter, 1977; Ovenden, 1990; Riley & Michaud, 1994; Botch et al., 1995; Kuhry & Turunen, 2006; Wieder & Vitt, 2006). These ecosystems function as long-term sinks of carbon (C) fixed through plant primary production. In Alberta, paludification is the main process leading to peatland formation where saturated peat and its vegetation spreads to cover inorganic soils (Wieder & Vitt, 2006). Plant growth responds to surrounding precipitation and hydrogeochemical landscape characteristics, which drive anaerobic conditions that limit decomposition rates (Clymo, 1984; Wallén, Falkengren-Grerup, & Malmer, 1988; Vitt & Chee, 1990). Poor fens and bogs have low dissolved mineral concentrations supplied by precipitation or groundwater inputs and are highly acidic, resulting in low species diversity (National Wetlands Working Group, 1997). In contrast, rich fens have higher species diversity due to the high concentrations of dissolved minerals and slightly acid to neutral pH (National Wetlands Working Group, 1997). Perennially saturated soils have slow decomposition rates, facilitating the accumulation of partially decomposed plant material that results in peat accumulation.

Fens are groundwater-fed peatlands and in northwestern Canada have more than 40 cm of peat, are dominated by moss and graminoid vegetation, and maintain a high annual water table (National Wetlands Working Group, 1997). Annual total above-ground biomass averages 395 g m^{-2} in poor-fens, 238 g m^{-2} in bogs, 210 g m^{-2} in rich-fens, and 380 g m^{-2} in moderately-rich open fens (Gorham & Somers, 1973; Szumigalski & Bayley, 1997; Thormann & Bayley, 1997). The majority of C may be fixed by bryophytes due to low concentrations of soil nitrogen and recalcitrant cell properties (Thormann, Bayley, & Currah, 2001). High turnover of above-ground vascular vegetation leads to sedge roots providing the primary vascular plant input to peat-formation (Saarinen, 1996; Chimner, Cooper, & Parton, 2002). Below-ground production in northwestern Canada accounts for over 90% of total annual biomass in peatlands when fine roots (< 2 mm) are included (Bernard & Fiala, 1986; Wallén, 1986; Backeus, 1990; Shaver & Chapin III, 1991). Annual below-ground biomass in boreal fens ranges from 500 to 2700 g m^{-2} (Bernard, Solander, & Kvet, 1988; Saarinen, 1996). Morphological characteristics of dominant peatland plants largely influence annual biomass inputs to peatlands through resource competition with smaller species.

Plant species identity and shoot density have contrasting effects on annual above and below-ground production by altering soil-surface temperature, evaporation rates, and light competition. In sedge-dominated bogs and fens, intense competition for light availability can preclude smaller vascular and bryophytic species due to robust rooting structure, large leaf area, and higher vertical reach (Kotowski & van Diggelen, 2004; Cooper, Wolf, & Gage, 2006). In contrast, canopy shade by vascular plants and their litter may facilitate bryophyte growth by reducing soil-surface temperature and evaporation (Bergamini et al., 2001). The effect of plant density on annual production varies across spatial scales and defining thresholds for maximizing growth is limited due to complex biotic and abiotic interactions (Waide, Willig, & Steiner, 1999).

Climate and landscape variables create the hydrological processes resulting in peatland formation. On average, water table depth for boreal peatlands ranges from -10 to -75 cm below the soil surface in a growing season (Rochefort et al., 2012). Water table and electrical conductivity (EC) of pore water affects osmotic potential of soils and osmotic stress in vascular plants, which subsequently impacts species richness (Burchill & Kenkel, 1991). Electrical conductivity in natural saline wetlands ranges from 0.6 to > 30 mS cm^{-1} (Hammer & Heseltine, 1988; Trites & Bayley, 2009a; Wells & Price, 2015), and 0.5 – 4.0 mS cm^{-1} in oil sands affected wetlands (Trites & Bayley, 2009b; Glaeser, Vitt, & Ebbs, 2016). Concentrations of sodium [Na^+] in regional rich fens range from 100 mg kg^{-1} to 1000 mg kg^{-1} (Vitt & Chee, 1990). Electrical conductivity ranges from 40 μS cm^{-1} to 2700 μS cm^{-1} in natural boreal wetlands (Trites & Bayley, 2009b), 28 to 560 μS cm^{-1} in boreal plains fens (Zoltai & Johnson, 1987; Vitt & Chee, 1990; Turetsky & Ripley, 2005), and from 1900 to 3900 μS cm^{-1} for surface water EC in a brackish fen near Fort McMurray (Wells & Price, 2015). Salt concentrations exceeding 500 mg L^{-1} in peat create negative effects on photosynthetic performance of plants (Pouliot, Rochefort, & Graf, 2013; Glaeser, Vitt, & Ebbs, 2016). Habitat suitability thresholds for vascular plants and bryophytes are characterized by water table depth and electrical conductivity ranges and are useful tools for achieving peat accumulation similar to that of regional fens.

The Nikanotee fen (the Fen) constructed on the Suncor Millennium mine in 2013 pioneered creating and revegetating a groundwater supported fen in a post-mined landscape. Several studies have identified peatland plant species able to withstand post-mined site conditions (Cooper & MacDonald, 2000; Mollard et al., 2012), and methods for introducing these and other species must be tested. The vascular plant species *Carex aquatilis* and *Juncus balticus* were chosen for introduction to the Fen as they are rhizomatous and can spread rapidly to create a vegetation cover, have high annual biomass production, and are tolerant of elevated salinity and variable water table depths (Cooper & MacDonald, 2000; Mollard et al., 2012; Montemayor, Price, & Rochefort, 2015; Wells & Price, 2015; Glaeser, Vitt, & Ebbs, 2016). *Carex aquatilis* is a dominant sedge species in peatlands across western Canada (Gignac et al., 2004), occurs where the water table ranges from 0 to 7 cm below the ground in Alberta (Cooper, Wolf, & Gage, 2006), fresh to slightly brackish water (Thunhorst, 1993) with EC ranging from 0.1 – 20.1 mS cm^{-1} (Kantrud, Krapu, & Swanson, 1989), and can tolerate water with electrical conductivity ranging from 0.3 – 3.8 mS cm^{-1} (Kantrud, Krapu, & Swanson, 1989). *Juncus balticus* tolerates even a broader range of water table depth including inundation up to 15 cm deep (Thunhorst, 1993) and can persists through periods of seasonal drought (Stevens & Hoag, 2000). *Juncus balticus* dominates natural saline fens in the oil sands region and facilitates the growth of bryophytes and low-growing vascular plants (Montemayor, Price, & Rochefort, 2015; Wells & Price, 2015; Borkenhagen & Cooper, 2016). The tolerance of a wide range of environmental conditions make *C. aquatilis* and *J. balticus* prime candidates for creating a C-accumulating peatland in post-mined conditions.

The study was conducted on the Nikanotee fen (the Fen) constructed in 2013 on Suncor's Millennium mine during the 3rd, 4th, and 5th years after planting. In this study, the term "biomass" refers to living above ground biomass (AGB) or below-ground biomass (BGB) collected during the study. In addition to planting treatments, the effects of *Typha latifolia* removal were tested during the first three years of this project but discontinued in 2016. Quantifying the effects of planting treatments, water table depth, and salinity on biomass production can provide guidance for reclaiming peat-forming ecosystems in the future. To test the suitability of *C. aquatilis* and *J. balticus* as initial colonizers of post-mined oil sand reclamation, I examined the effects of plant reintroduction methods, water table depth and water electrical conductivity on annual above and below-ground biomass.

In this thesis I address the following questions:

1. What effects did planting treatment have on AGB or BGB for planted species?
2. Does planting density influence AGB or BGB of *C. aquatilis* and *J. balticus* 3 to 4 years after planting?
3. What are the temporal effects of *T. latifolia* removal on *T. latifolia* AGB?
4. What effects does depth to water (DTW) and electrical conductivity (EC) have on biomass?
5. Are total AGB and BGB by years three and four similar to natural fens in the region?

MATERIALS AND METHODS

STUDY AREA

The study area is located on the Suncor Millennium Mine, 40 km north of Fort McMurray, Alberta (56° 57' 22" N, 111° 22' 44" W, elevation: 310 m), within the northern boreal natural region. The landscape is characterized by cool summers and mild winters (Ecological Stratification Working Group, 1995; Downing & Pettapiece, 2006). Mean annual temperature and total precipitation for the region are 1 °C and 419 mm, and daily mean temperature and total precipitation during the May 1st – August 31st growing season are 14 °C and 248 mm (averages from 1981 to 2010). Mean monthly temperature and total precipitation during the May 1st – August 31st growing season were 15.1 °C and 165 mm for 2015, and 16.0 °C and 272 mm for 2016, and 15.9 °C and 35 mm in 2017 (‘Government of Canada’, 2016).

The central mixedwood subregion of the greater northern boreal forest natural region is dominated by wetland ecosystems, predominantly fen peatlands (Downing & Pettapiece, 2006; Wieder & Vitt, 2006; Vitt et al., 2009). Black spruce (*Picea mariana* [Mill.] BSP.) is the most common peatland tree species and is often accompanied by white spruce (*Picea glauca* [Moench] Voss) in nutrient poor sites, and tamarack (*Larix laricina* [Du Roi] K. Koch) in rich fens (Downing & Pettapiece, 2006). Bryophytes dominate the understory, and open peatlands are often dominated by species of *Carex* (Downing & Pettapiece, 2006).

THE NIKANOTEE FEN

The Nikanotee fen is bordered on the east and west by previously reclaimed upland slopes, a natural forest remnant on the south, and an unplanted berm on the north (Price & Rudolph, 2007; Price, McLaren, & Rudolph, 2010) (Figure 1). Groundwater inputs to the fen are from a 7.7-ha upland watershed constructed from tailings material and contain residual salt and naphthenic acids (NAs) from the mining process (Allen, 2008). The 2.9-ha fen basin contains 2 m of locally sourced peat from a dewatered rich fen and overlies a 0.5-m thick petroleum coke layer that facilitates hydraulic connection between the upland watershed and fen (Nwaishi, Petrone, Price, & Andersen, 2015). Donor peat was stored for 2 years, and prior to construction, the top 30 cm was removed to exclude naturally recruited plants and seeds (Nwaishi, Petrone, Price, Ketcheson, et al., 2015). Construction was completed and the peat body was planted in 2013 (Price & Rudolph, 2007; Price, McLaren, & Rudolph, 2010).

EXPERIMENTAL DESIGN

The Fen was planted in a two-factor randomized block split-plot design (*n* 7) replicated 9 times with each block controlling for hydrologic variation. Each block contained seven whole-plot factors of planting treatments (PT) in 17-m x 18-m plots; moss layer transfer (MLT) treatment; MLT planted with three-month-old *Carex aquatilis* or *J. balticus* seedlings (MLT + SLc and MLT + SLj, respectively); and seedling treatments (SLc and SLj) planted with *Carex aquatilis* and *J. balticus* seedlings. The remaining unplanted (UP) and seeded (SD) treatments occupied 8-m x 18-m plots. Each whole-plot was divided into 4 split-plots (4-m x 8.5-m) for *Typha latifolia* removal treatments (TR); removal (r), mulching (m), removal + mulching (rm), and untreated control (nt). *Carex aquatilis* and *J. balticus* seedling planted in MLT + SL and SL plots were randomly assigned treatments of high (*h*) (*C. aquatilis*, 3 – 4 seedlings m^{-2}; *J. balticus* 6 seedlings m^{-2}), and low (*l*) densities (*C. aquatilis*, 1 – 2 seedlings m^{-2}; *J. balticus* 3 seedlings m^{-2}). Planting occurred from mid-June to mid-July of 2013 (Borkenhagen, 2014).

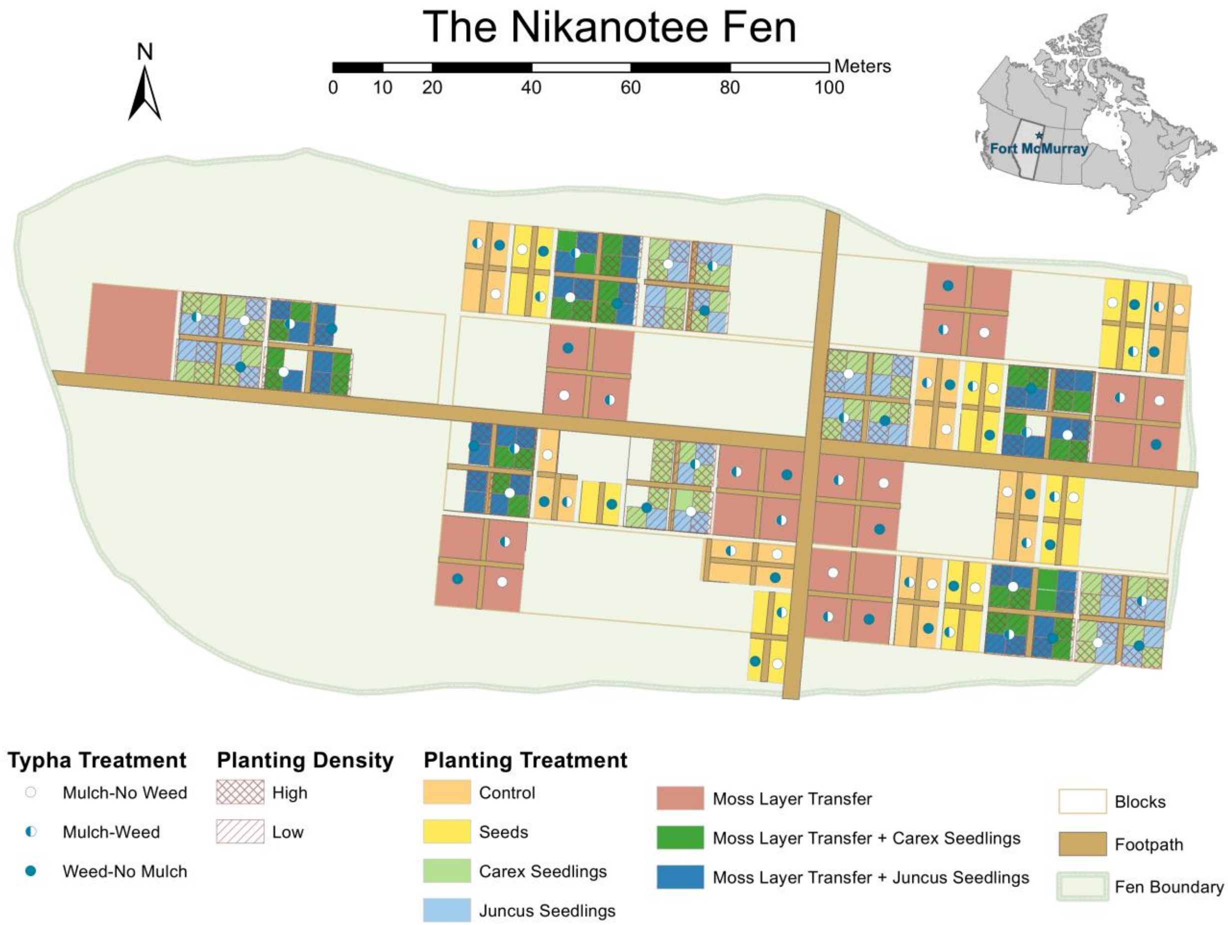

Figure 1 – Location and planting design on the Nikanotee reclamation fen north of Fort McMurray, Alberta (upper right). Unused plots are represented by block outlines.

TREATMENTS

Material used in the moss layer transfer (MLT) treatments was harvested from a rich fen located 12-km west of the Fen (56° 56' 34" N, 111° 33' 9" W) dominated by regional vascular and bryophytic plant communities (Chee & Vitt, 1989). Native species identified in previous studies to withstand post-mined conditions (Rochefort et al., 2003; Borkenhagen & Cooper, 2016) included *Tomentypnum nitens, Aulacomnium palustre, Sphagnum warnstorfii, Sphagnum angustifolium, Betula pumila*, and *Carex aquatilis*. An excavator-mounted rototiller was used to remove 5 cm – 10 cm surface material from the donor site in mid-June 2013 and was stockpiled at the Fen. The MLT material was hand spread on the Fen's bare peat surface from late-June to mid-July 2013 in a 1:10 ratio of harvested-to-covered area (Rochefort et al., 2003; Borkenhagen, 2014). Three-month-old *C. aquatilis* and *J. balticus* seedlings used in SL and MLT + SL treatments were grown from locally collected seed at the Smokey Lake Forest Nursery in Alberta, Canada.

Locally sourced seeds from a nearby rich and saline fen were used in the seeded (SD) treatments and seed viability was tested using the Tetrazolium method. Prior to planting, seeds requiring cold-stratification were mixed in ratios of sand and seed and stratified by Golder Associates Ltd. (Table 1). Seeds were hand-mixed and dispersed using a walk-behind spreader. The mixture included the following species: *C. aquatilis*, *Betula glandulosa* Michx., *Sarracenia purpurea* L., *Vaccinium vitis-idaea* L., and *Oxycoccus microcarpus* Turcz. Ex Rupr. Seeds collected from a saline fen included: *J. balticus*, *Calamagrostis stricta* (Timm.) Koeler, and *Triglochin maritima* L. No plant material was introduced to unplanted (UP) control treatments.

Table 1 – Seed species, stratification duration in hours (hrs), days (D), temperature (°C), percent viability (Tetrazolium Method), sand and seed ratio, and final quantity (seeds m^{-2}) used in seeded treatments.

Plant Species	Cold-Stratification Temperature (°C)	Percent Viability	Sand:Seed	Quantity Used in Treatment (seeds m^{-2})
Carex aquatilis	(30°C, 24 hrs) (5°C, 65 D) (30°C, 60 D)	52.7	8:1	375
Oxycoccus microcarpus	(30°C, 24 hrs) (5°C, 45 D) (21°C, 50 D)	76.5	8:1	21
Sarracenia purpurea	(30°C, 24 hrs) (5°C, 25 D) (21°C, 55 D)	78.5	16:1	224
Triglochin maritima	(30°C, 24 hrs) (5°C, 35 D) (21°C 40 D)	64.0	8:1	109
Betula pumila	(30°C, 0 hrs) (5°C, 0 D) (30°C, 30 D)	9.50	8:1	59
Calamagrostis inexpansa	(30°C, 0 hrs) (5°C, 0 D) (30°C, 30 D)	58.0	48:1	227
Juncus balticus	(30°C, 0 hrs) (5°C, 0 D) (30°C, 30 D)	75.0	8:1	85

Typha latifolia removal treatments were applied during 2013 – 2015 and randomly assigned split-plots and *T. latifolia* was unmanaged in control (nt) treatments. Manual removal of seed-heads, prior to dehiscence, and above-ground vegetation of *T. latifolia* were clipped at the surface in (r) and removal + mulched (rm) treatments.

Wood-strand mulch treatments were applied at approximately 0.8 kg m^{-2}, creating 90% visual vertical obstruction of the peat surface to hinder germination and growth of undesired species, while regulating moisture and temperature at the surface for bryophytes (Price, Rochefort, & Quinty, 1998; Athy, Keiffer, & Stevens, 2006; Chimner, 2011; González & Rochefort, 2014).

SAMPLING METHODS

Due to persistent ponding and construction errors, sampling was restricted to seven usable blocks. In years three and four samples were collected from early to mid-Aug (*n* 141). To minimize disturbance and due to persistent ponding, sampling was subset to include only mulched (m) and control (nt) treatments in the *T. latifolia* removal group (*n* 58) in the fifth year.

Above-ground biomass collected in 2014 was clipped at ground-level from four 0.25-m^2 randomly placed quadrats in each sub-plot at the end of August. The size of the sampling quadrat was modified in years three through five to a randomly placed 20 x 50 cm quadrat. Living vascular plant material was separated from litter by shaking the clipped vegetation over a bucket. Incidental living vegetation was removed from the bucket by hand and included with live vegetation. Senesced leaves still attached to living shoots were also included with living vegetation. Plant material was separated on site into litter, and by species. Non-dominant species (NDS) representing less than 2% of the total above-ground biomass were pooled for analysis, including *Calamagrostis stricta*, *Carex diandra* Schrank, *Carex atherodes* Spreng., *Carex prairea* Dewey ex Alph., *Carex utriculata* Booth, *Juncus nodosus* L., *Salix spp., Triglochin maritima* L., and *Triglochin palustris* L. Samples were dried to a constant mass in an oven at 55

°C for 48 hours, tillers were counted for all species except *T. maritima* and *T. palustris* and weighed.

Below-ground biomass was sampled in each plot to a depth of 50 cm using a corer with an inside diameter of 10 cm. A serrated blade attached to the corer base allowed woody and herbaceous roots to be severed, reducing compression of the peat profile to < 4 cm. An access channel was dug adjacent to the corer to alleviate suction to retain the bottom portion of the sample during removal. Below-ground material was rinsed in 2 mm grated sieves with a mixture of sodium hexametaphosphate (100 g L^{-1}) to remove soil from plant roots, sorted according to species, dried at 55 °C for 48 hours, and weighed (Franks & Goings, 1998). To stratify the loss of fine roots (< 0.5 mm) equally across all samples, below-ground material was rinsed for no longer than 10 minutes per sample.

Plot-level abiotic variables were collected from the holes left after extracting the below-ground biomass cores. The water in each hole was bailed prior to measurement. Electrical conductivity (μS cm^{-2}), and DTW (cm) were collected after allowing the water to settle for no more than 8 hours using an Oakton Pttester 35 Waterproof Multimeter™, and depth to water was measured to the nearest centimeter. Temperature, EC and pH probes were calibrated according to the product instruction manual.

STATISTICAL ANALYSIS

Treatment effects on above-ground biomass were analyzed by species, and total above and total below-ground biomass using a linear mixed model in R version 3.4.0 (R Development Core Team, 2015). The intent was to quantify the effect size and not survivability. To achieve this, analyses were performed on species planted in each sample (i.e. all zero biomass values were removed for unplanted species). Equality of variance (Levene, 1960) and residual normality for the above-ground biomass response variables were examined using "*homog.test*" and "*nor.test*" functions in R (Dag, Dolgun, & Konar, 2018). Violations of the normality assumption were corrected using a log (x + 1), square root, or cube root-transformations.

Fixed effects included planting treatments (PM), *T. latifolia* removal (TR), density (Den) and year, and continuous predictor variables included were mid-summer depth to water (DTW) (cm) and electrical conductivity (EC) (µS cm^{-2}). A likelihood ratio test was used to examine the random and fixed effect of replicate and was retained as a random variable in all models to account for variations between replicates (Table 2). Two-way linear mixed models with a random replicate term were conducted to examine the interaction between planting method and year, depth to water, and electrical conductivity. Fixed factors and interactions in the final models were selected by standard backward stepwise selection using the "*step*" function from the "MASS," package (Venables & Ripley, 2002). Final linear mixed effect models were conducted using the "*lmer*" function from the "*lme4*" package in R (Bates et al., 2015). A Kenward-Roger *F*-statistic modification was applied to both one-way and two-way linear mixed model to adjust for small-sample bias and account for incompleteness of the sampling design (Kenward & Roger, 1997). Post-hoc Tukey-adjusted least squared means analysis were conducted on the fixed factors using the "*lsmeans*" package in R to determine statistical significance (alpha = 0.05) (Lenth, 2016).

Pearson's correlation coefficients were computed between DTW and EC for above-ground biomass (AGB) of *C. aquatilis*, *J. balticus*, *T. latifolia*, non-dominant species (NDS), total above-ground biomass (TAGB), and total below-ground biomass (TBGB) for all years individually and combined. Zero values for all species were removed from correlation analysis to accurately examine the effect size of DTW and EC on above and below-ground biomass for present species.

Outliers and associated leverages were identified by the Cook's Distance descriptive plots of the multi-factor linear regression. Below-ground biomass outliers with values of zero were identified as errors in post-processing and removed. Removal did not influence the analysis results. High variance is an intrinsic characteristic of plant biomass studies, and outliers with extremely high values were retained for AGB and BGB.

RESULTS

PLANTING TREATMENT EFFECTS ON ABOVE AND BELOW-GROUND BIOMASS

Carex aquatilis AGB averaged 405 ± 12.7 g m^{-2} (*n* 349) across planting treatments and years, providing 76.5% (*se* 1.50, *n* 349) of the total AGB on the Fen (Figure 2). Planting treatments explained 58.1% of variance in *C. aquatilis* AGB between treatments ($F_{1,6}$ = 41.6, P < 0.001). *C. aquatilis* AGB was highest in MLT, MLT + SLc, and SLc treatments compared to MLT + SLj and SLj treatments (Figure 3). Planting treatment SLj contained the lowest *C. aquatilis* AGB, averaging 182 ± 24.2 g m^{-2} (*n* 35) across all years and was significantly lower than all other planting treatments (P < 0.001). The interaction between year and planting treatment was not significant ($F_{1,8}$ = 0.95, ns). When averaged across planting treatments, *C. aquatilis* AGB increased significantly ($F_{3,6}$ = 5.07, P < 0.05) by 50% (*se* 8.88) between years two and five (Figure 2). *Carex aquatilis* produced greater AGB by year four in MLT + SLj (mean 255 ± 29.7, *n* 38) and SLj (mean 185 ± 24.4, *n* 37) treatments compared to year three, and in MLT + SLc (*M* 536 ± 45.0, *n* 19) and SLc (mean 461 ± 48.4, *n* 21) between years two and three (Figure 3).

Juncus balticus AGB averaged 191 ± 13.0 g m^{-2} (*n* 212) and contributed 43.2% ± 2.55 (*n* 212) of total AGB across all planting treatments and years (Figure 2). Planting treatment significantly affected mean *J. balticus* AGB ($F_{1,6}$ = 34.1, P < .001) and was greater in treatment SLj (342 ± 25.3 g m^{-2}, n = 68) compared to SLc (8.99 ± 1.91 g m^{-2}, n = 9) (Figure 3). Sample year was a significant predictor of average above-ground *J. balticus* biomass ($F_{1,3}$ = 24.3, P < .001) and declined by 53.5% (*se* = 10.6) between years three and four (Figure 2). The interaction between sampling year and planting treatment was not significant for *J. balticus* AGB ($F_{1,11}$ = 1.47, ns). Average *J. balticus* AGB decreased between years three and four in SD, MLT + SLj, and SLj treatments and was present in MLT + SLc planting treatments (162 ± 88.2 g m^{-2}, n = 4) despite not being planted in these plots.

Year and planting treatment were significant predictors of non-dominant species' AGB (year: $F_{1,2} = 6.07$, $P < 0.001$, planting treatments: $F_{1,6} = 4.28$, $P < 0.001$) that averaged 67.1 ± 7.24 g m^{-2} (n 157) across year and planting treatment and decreased from the fourth to fifth year by 65% (*se* 6.56) (Figure 4). Although only significantly different from MLT + SLj ($P < 0.015$), mean NDS AGB was highest in year three for MLT treatments (233 ± 95.4 g m^{-2}, n 4) and in seeded treatments by year five (130 ± 54.3 g m^{-2}, n 7). All planting treatments had lower AGB for NDS between years three and five.

Planting treatment was a significant explanatory variable for *T. latifolia* AGB when averaged across years ($F_{1,6} = 3.29$, ns) and produced significantly more AGB in UP treatments (273 ± 66.1 g m^{-2}, n 17) compared to MLT + SLj (64.0 ± 25.3 g m^{-2}, n 10) and SLj (40.9 ± 11.7 g m^{-2}, n 11).

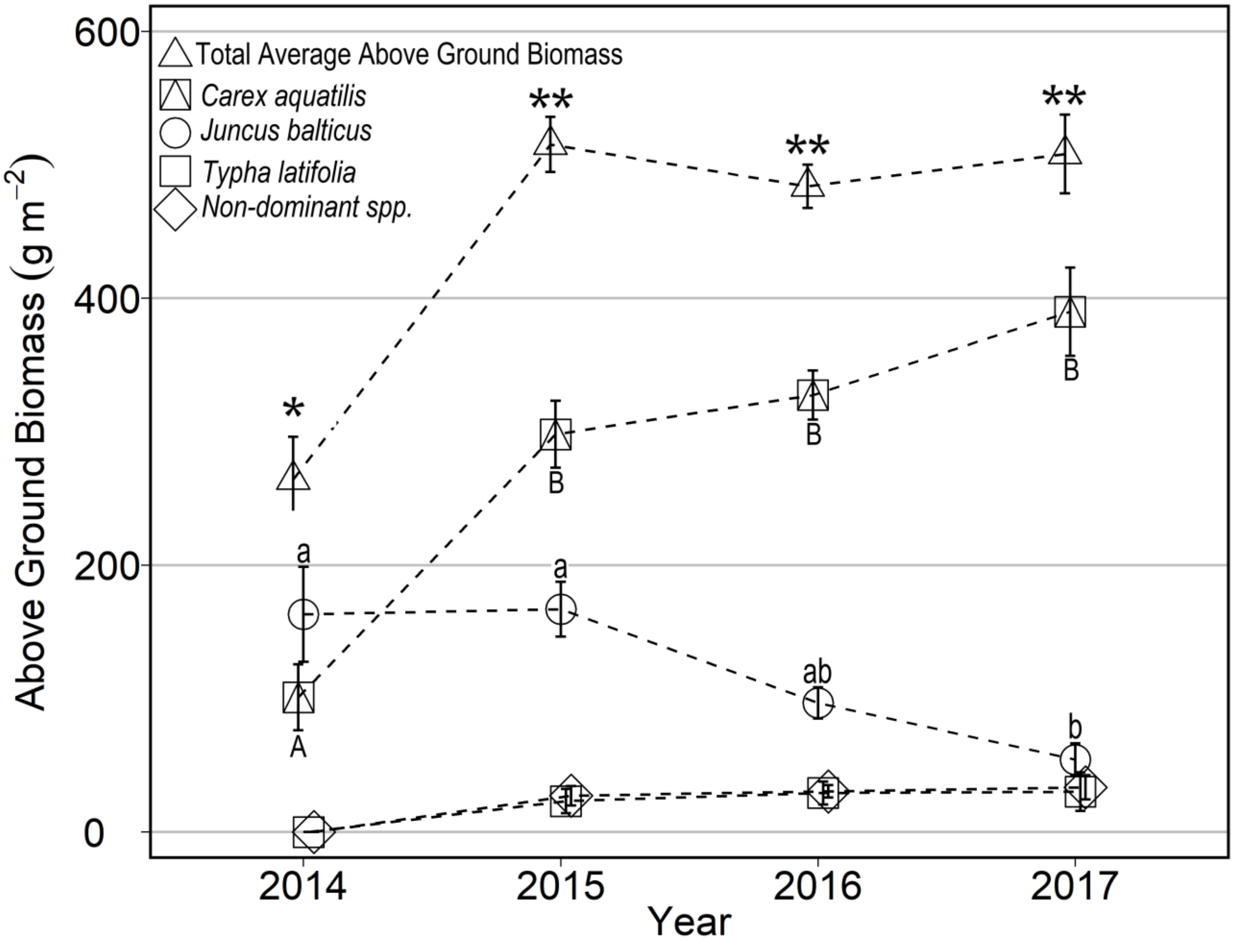

Figure 2 – Total above-ground biomass averaged across all samples for all vascular plant species (AGB), above-ground biomass of *Carex aquatilis*, *Juncus balticus, Typha latifolia*, and non-dominant species (NDS) averaged across all planting treatments for each sampling year. Asterisks and letters indicate significant differences between years.

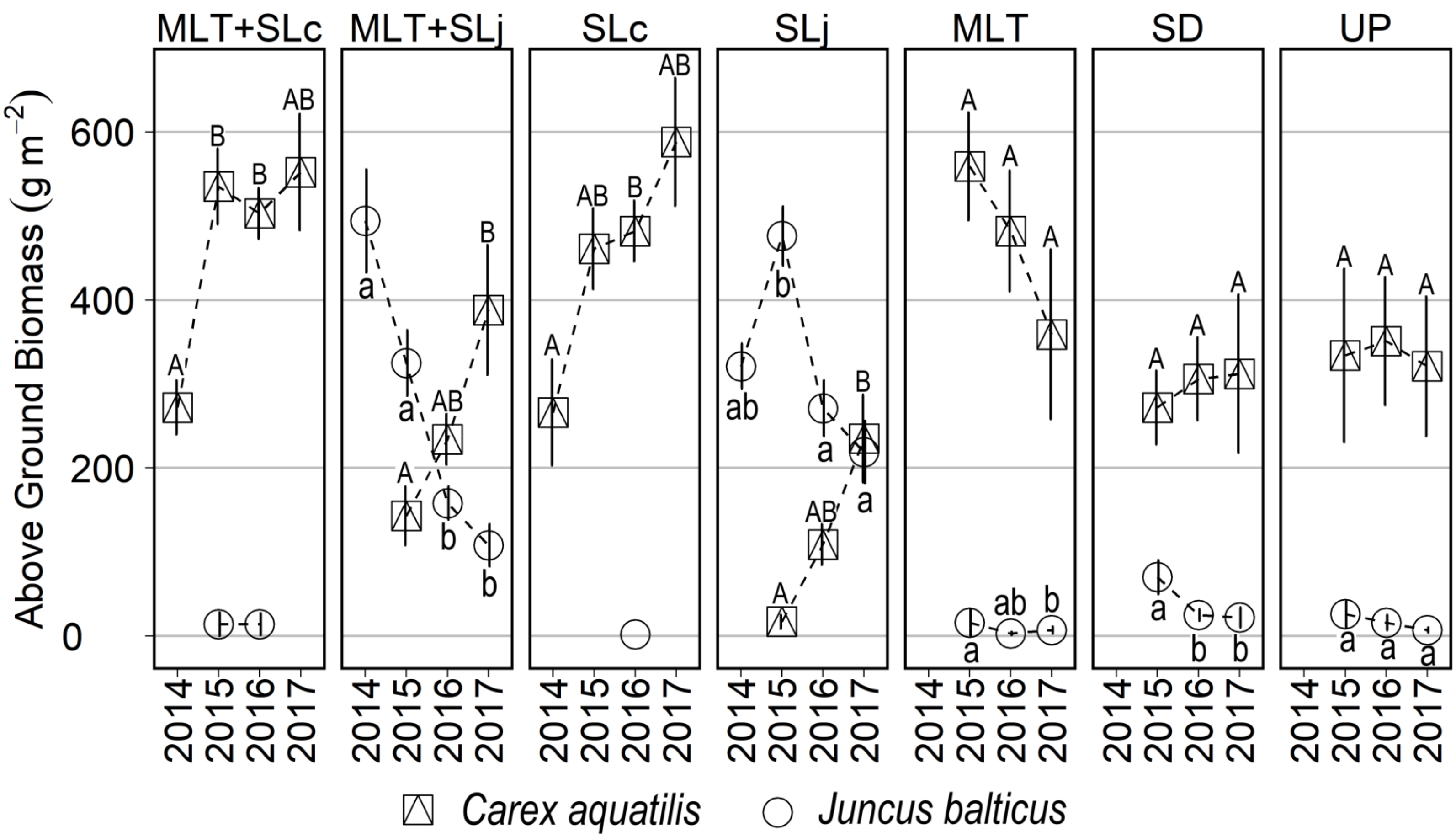

Figure 3 – Mean annual above-ground biomass for *Carex aquatilis* and *Juncus balticus* in treatments for each year. Letters indicate significant differences between years. Treatments include moss layer transfer (MLT), *Carex aquatilis* seedlings (SLc), *Juncus balticus* seedlings (SLj), seeded (SD), and unplanted (UP).

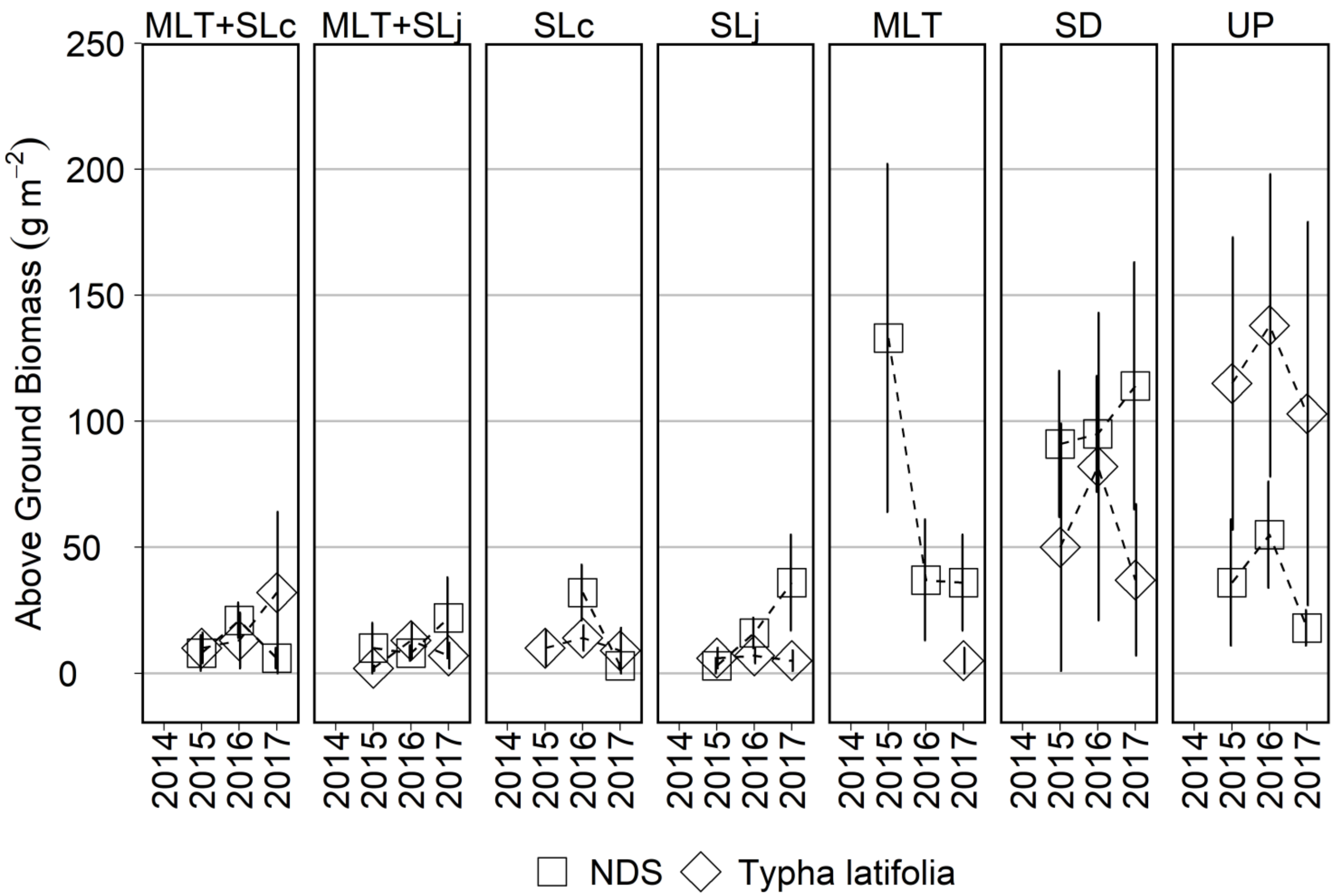

Figure 4 – Mean annual above-ground biomass for *Typha latifolia* and non-dominant (NDS) vascular species in planting treatments for each year. No significant differences occurred between years for either species or species group. Treatments include moss layer transfer (MLT), *Carex aquatilis* seedlings (SLc), *Juncus balticus* seedlings (SLj), seeded (SD), and unplanted (UP).

Table 2 – Model parameters for performing analysis of covariance on annual above-ground biomass for *Carex aquatilis*, *Juncus balticus*, *Typha latifolia*, non-dominant species (NDS), total above-ground (AGB), and total below-ground biomass (BGB). Main effects include; year (Yr), planting treatment (PT), *Typha latifolia* removal treatment (TR), and seedling density treatments (Den). Covariate factors include; depth to water (DTW) and electrical conductivity (EC). Replicate was set to random to account for hydrologic variations in the model (1 | replicate).

Response	Model Predictors	Transformation	$r^2_{adj.}$	AIC	*N*
Carex aquatilis	PT : Yr : DTW + Yr : DTW + Yr : AGB + (1 | replicate)	Cube Root	0.69	902	298
Juncus balticus	PT : EC + Yr + Den + AGB + (1 | replicate)	Log (x + 1)	0.77	351	148
Typha latifolia	PM + TR + (1 | replicate)	Log (x + 1)	0.56	218	66
NDS.	PT + Yr	Log (x + 1)	0.22	563	158
AGB	Yr + DTW + Yr : DTW + (1 | replicate)	Square Root	0.31	2110	354
BGB	PT + Yr + Den + TR + (1 | replicate)	Square Root	0.51	1740	252

Table 3 – Linear regression for fixed-effects, covariates, and interactions between species above-ground biomass, total above-ground biomass, and total below-ground biomass. Abbreviations: Yr, years (2015, 2016, 2017); planting treatments (PT); *Typha latifolia* removal treatments (TR); density treatments (Den); depth to water in centimeters (DTW); electrical conductivity (EC); total above-ground biomass (AGB); total below-ground biomass (BGB); random blocking variable (1 | replicate).

		Carex aquatilis		*Juncus balticus*		*Typha latifolia*		Non-Dominant spp.	
Factor	d.f.	F-value	*p*-value	F-value	*p*-value	F-value	*p*-value	F-value	*p*-value
Yr	2	6.95	< 0.001	25.1	< 0.001	0.04	0.96	6.23	< 0.01
PT	6	21.4	< 0.001	28.2	< 0.001	3.29	< 0.01	4.28	< 0.01
TR	3	0.521	0.668	0.53	0.663	6.04	< 0.01	0.98	0.403
Den	1	5.04	< 0.05	4.75	0.031	0.08	0.778	0.56	0.458
DTW	1	2.30	0.133	1.19	0.277	6.32	< 0.05	0.12	0.733
EC	1	0.349	0.555	1.03	0.312	0.23	0.635	3.30	0.071
AGB	1	248	< 0.001	12.0	< 0.001	--	--	--	--
BGB	1	7.54	< 0.001	6.45	0.012	--	--	--	--
PT × Yr	10	1.01	0.441	0.452	0.888*	0.67	0.771	1.22	0.276
TR × Yr	4	1.60	0.174*	0.242	0.9141*	0.58	0.675	0.26	0.902
Den × Yr	2	2.94	0.0551	4.17	< 0.05	1.44	0.257	0.57	0.57
DTW × Yr	2	5.12	< 0.001	0.20	0.820	3.01	0.056	1.27	0.284
EC × Yr	2	1.61	0.202	0.27	0.767	0.23	0.793	0.82	0.444
AGB × Yr	2	3.35	< 0.05	1.66	0.194	--	--	--	--
BGB × Yr	2	3.13	< 0.05	0.803	0.450	--	--	--	--
PT × DTW	6	1.28	0.265	1.86	0.089	0.58	0.676	0.26	0.902
TR × DTW	3	0.938	0.423	0.951	0.417	1.44	0.257	0.57	0.57
Den × DTW	2	0.028	0.867	1.01	0.317	3.01	0.056	1.27	0.284
EC × DTW	1	0.229	0.632	4.49	< 0.05	--	1	--	0.054
AGB × DTW	1	0.119	0.730	0.564	0.454	--	--	--	--
BGB × DTW	1	1.53	0.217	0.829	0.364	--	--	--	--
PT × EC	6	1.64	0.137	3.14	< 0.05	0.58	0.675	--	--
TR × EC	3	2.37	0.071	1.27	0.284	1.44	0.257	--	--
Den × EC	1	0.072	0.788	0.254	0.615	3.01	0.056	--	--
AGB × EC	1	0.288	0.592	0.072	0.788	--	--	--	--
BGB × EC	1	0.017	0.896	2.08	0.151	--	--	--	--
(1 | replicate)	2								

* Indicates where sub setting occurred to satisfy rank deficiencies.

DENSITY EFFECTS ON BIOMASS

Mean *Carex aquatilis* AGB was not significantly different ($F_{1,1}$ = 2.10, ns) in high-density (504 ± 23.8 g m^{-2}, *n* 72) compared to low-density treatments (444 ± 27.7 g m^{-2}, *n* 59) when averaged across and within planting treatments or years. Sampling year was a significant predictor of *Carex aquatilis* AGB ($F_{1,3}$ = 0.627, P < .01), which increased in low-density MLT + SLc and SLc treatments between year two and three (Figure 5 and 6). Low-density planting treatments were removed from sampling in 2017 because the treatments did not affect AGB.

Juncus balticus AGB averaged 320 ± 23.1 g m^{-2} (*n* 63) in high-density and 264 ± 21.8 g m^{-2} (*n* 88) in low-density treatments across years and was not significantly affected by planting density ($F_{1,1}$ = 1.37, ns). However, year was a significant predictor of *J. balticus* AGB ($F_{1,3}$ = 12.5, P < 0.01) and decreased by 45% and 61% between years three and four in high and low-density MLT + SLj and SLj treatments, respectively. No significant differences were detected ($F_{1,3}$ = 1.00, ns) for *Juncus balticus* AGB in high or low-density MLT + SLc and SLc planting treatments.

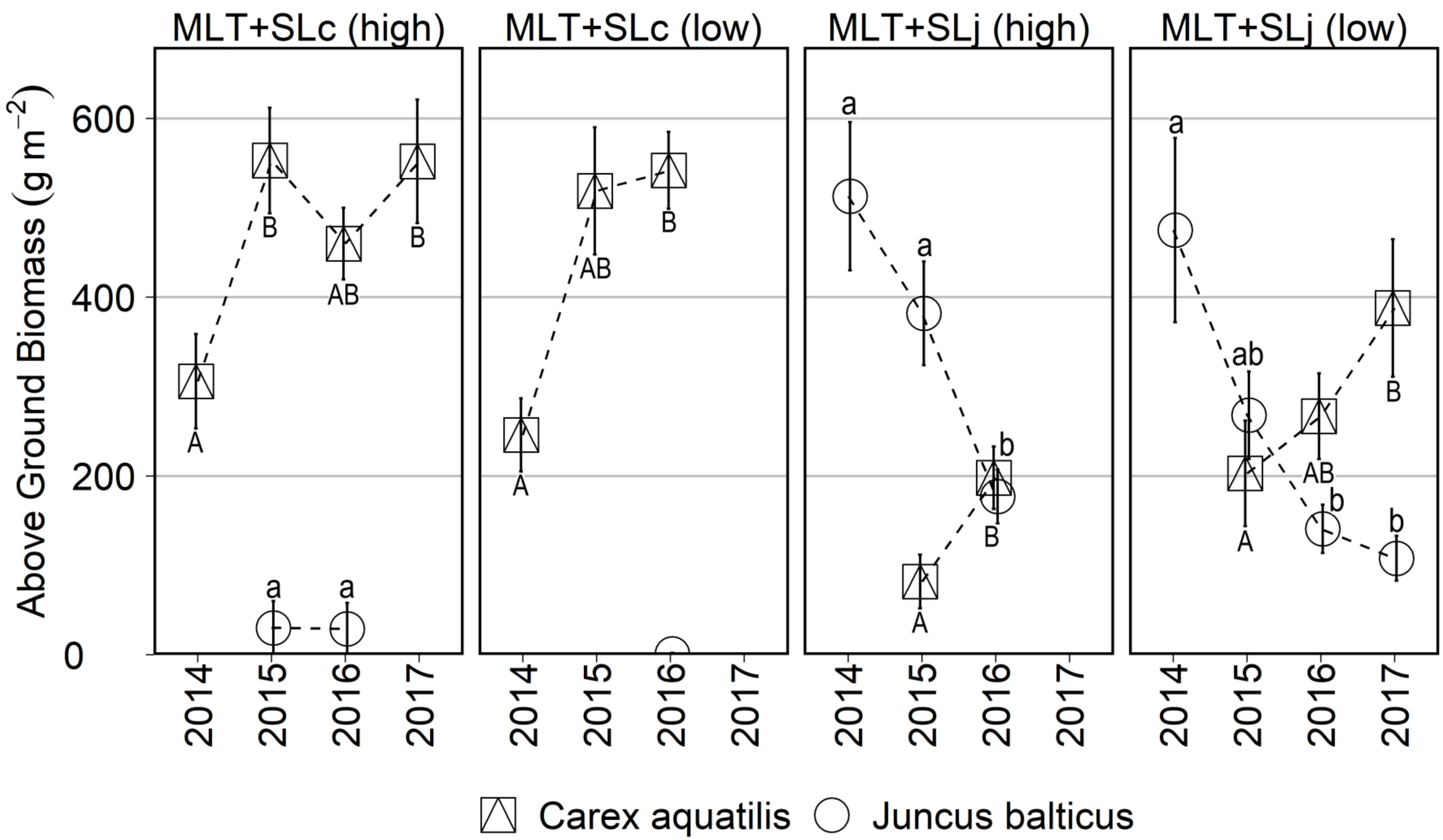

Figure 5 – Mean annual above-ground biomass for *Carex aquatilis* and *Juncus balticus* in high and low-density treatments for each year. Letters indicate significant changes between years. Planting treatments include; moss layer transfer (MLT), *Carex aquatilis* seedlings (SLc), *Juncus balticus* seedlings (SLj), high (high) and low (low) densities.

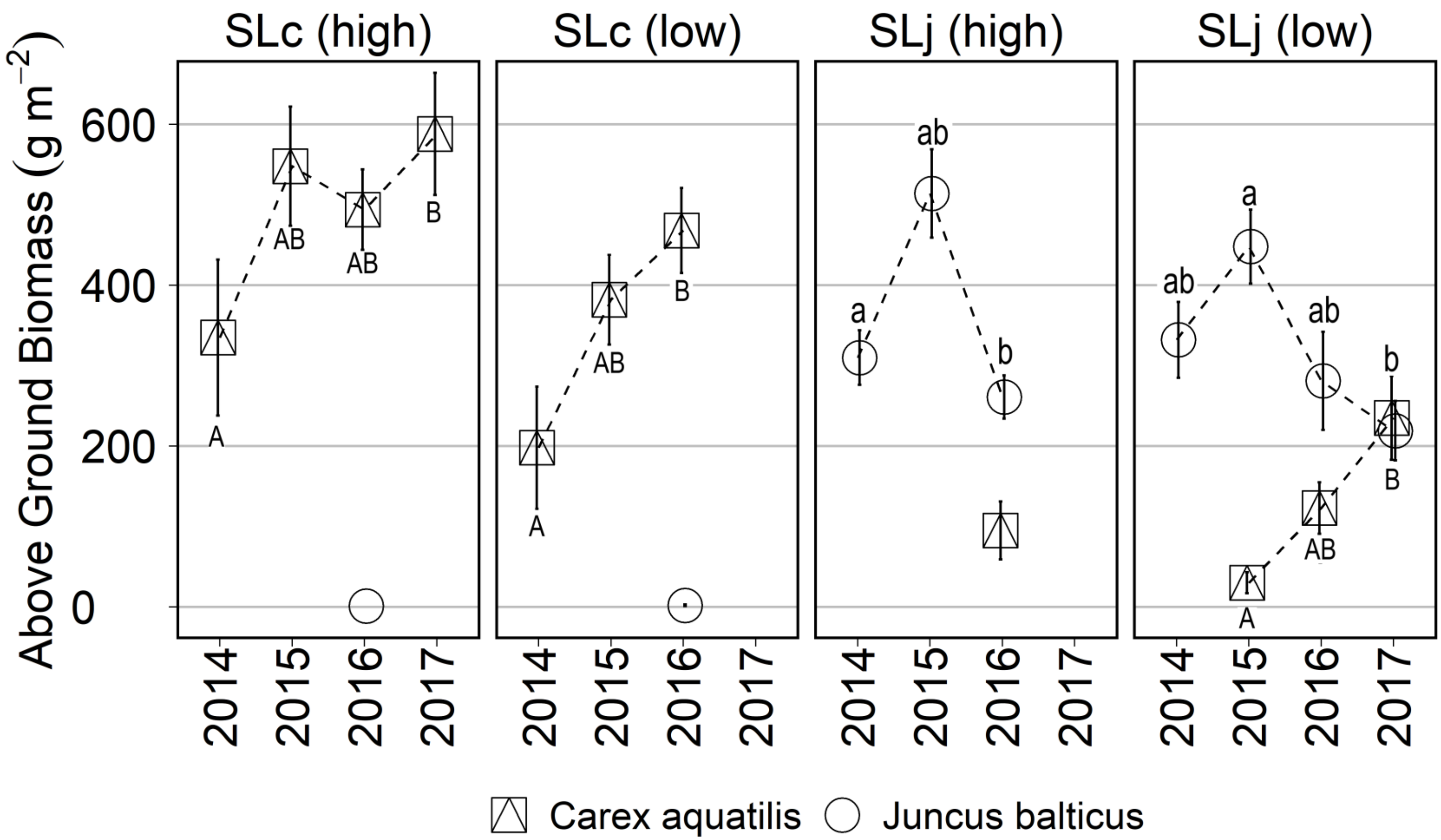

Figure 6 – Mean annual above-ground biomass for Carex aquatilis and Juncus balticus in seedling (SL) planting treatments averaged across sampling years. Letters indicate significant differences between years. High and low densities are indicated in parentheses next to planting treatment. Where error bars are missing, two or fewer samples were taken.

BIOMASS RESPONSE TO *TYPHA LATIFOLIA* TREATMENTS

Typha latifolia AGB in year three was highest in untreated plots (243 ± 89.4 g m^{-2}, *n* 9), and lowest in removal + mulch treatment plots 45.6 ± 15.2 g m^{-2} (*n* 2). When averaged across years, removal treatment was a significant predictor of *Typha latifolia* AGB ($F_{1,3}$ = 4.39, P < 0.01) and was greater in untreated (206 ± 43.1 g m^{-2}, *n* 33) compared to removal + mulch treatments (26.1 ± 5.49 g m^{-2}, *n* 6) (Figure 7). Year was not a significant predictor of *T. latifolia* AGB ($F_{1,2}$ = 0.484, ns) and no changes were detected after treatments stopped in year four (Figure 7).

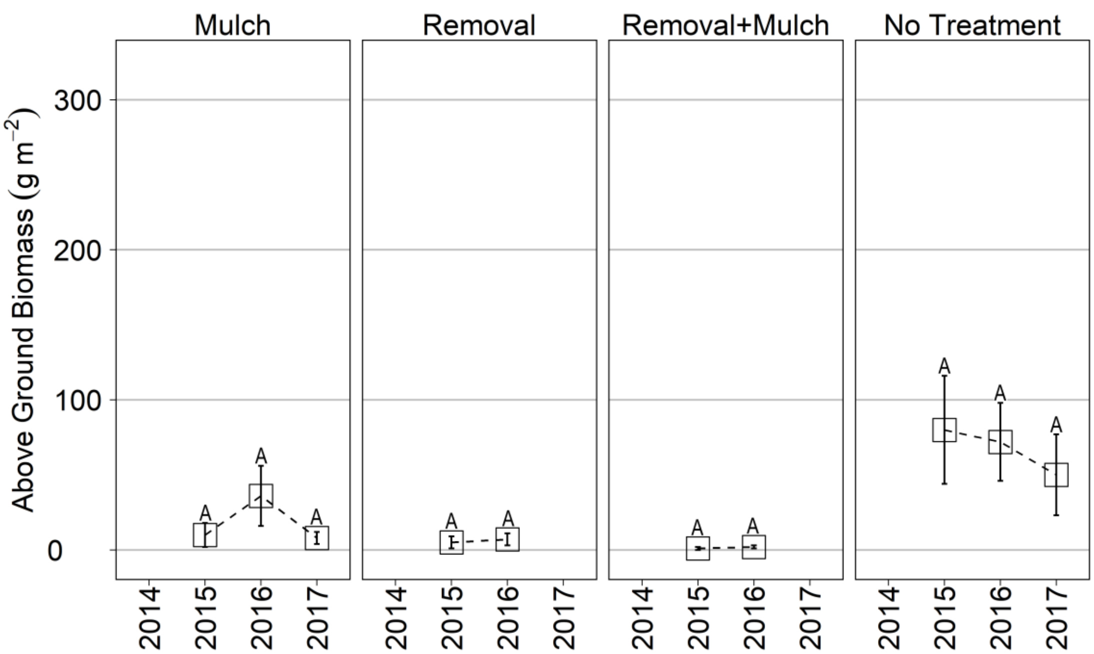

Figure 7 – Average *Typha latifolia* above-ground biomass in removal treatments for sampled years. Removal treatments include: mulched, removal only, removal+mulch, and untreated (no treatment).

EFFECTS OF WATER TABLE AND ELECTRICAL CONDUCTIVITY

Depth to water at the time of sampling ranged from -44 cm to +7 cm in years three through five. Total AGB and total BGB exhibited a weak positive correlation with water table depth, r^2(*n* 330) = 0.26, P < 0.01 and r^2(*n* 334) = 0.13, P < 0.05, respectively. *Carex aquatilis* and *J. balticus* AGB was weakly and positively correlated with water table depths when averaged across years, r^2(*n* 285) = 0.19, P < 0.01 and r^2(*n* 198) = 0.15, P < 0.10, respectively (Figure 8).

Electrical conductivity (EC) ranged from 0.03 to 6.77 mS cm^{-1} in sampling years three through five and did not differ significantly between years. Non-dominant species AGB was weak and negatively correlation with EC, while total BGB was weak and positively correlated, r^2(n 157) = 0.16, P < 0.05 and r^2(n 334) = 0.13, P < 0.01, respectively (Figure 9).

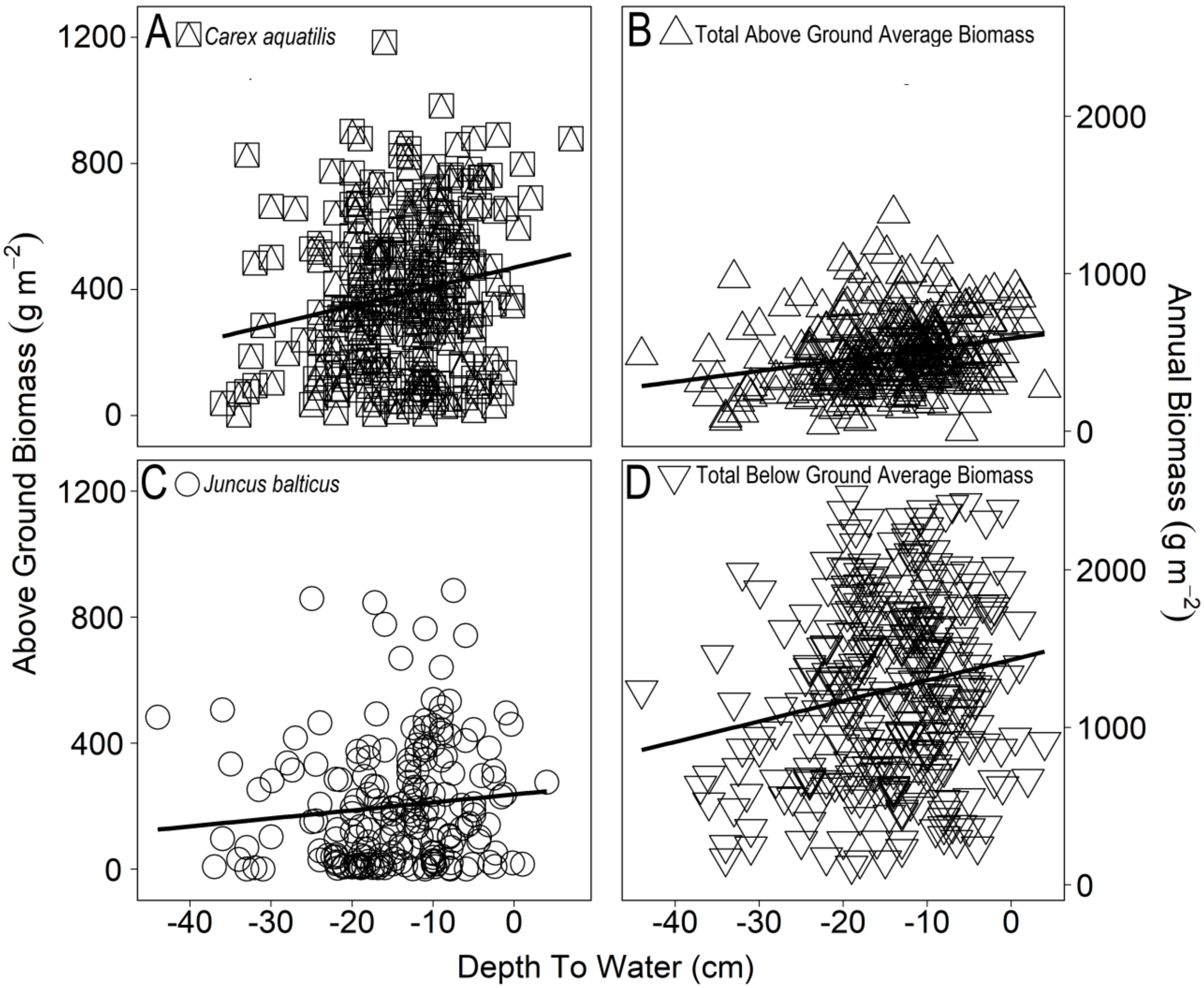

Figure 8 – Scatterplots depicting the Pearson's Product-Moment correlation between annual plant biomass (g m^{-2}) and water table depth (cm) of *Carex aquatilis* (A), r^2(n 285) = 0.19, P < 0.01, and *Juncus balticus* (C), r^2(n 198) = 0.15, P < 0.10, above-ground biomass, total above-ground biomass (B), r^2(n 330) = 0.26, P < 0.01, and total below-ground biomass (D), r^2(n 334) = 0.13, P < 0.05. Zero values were removed for the analysis.

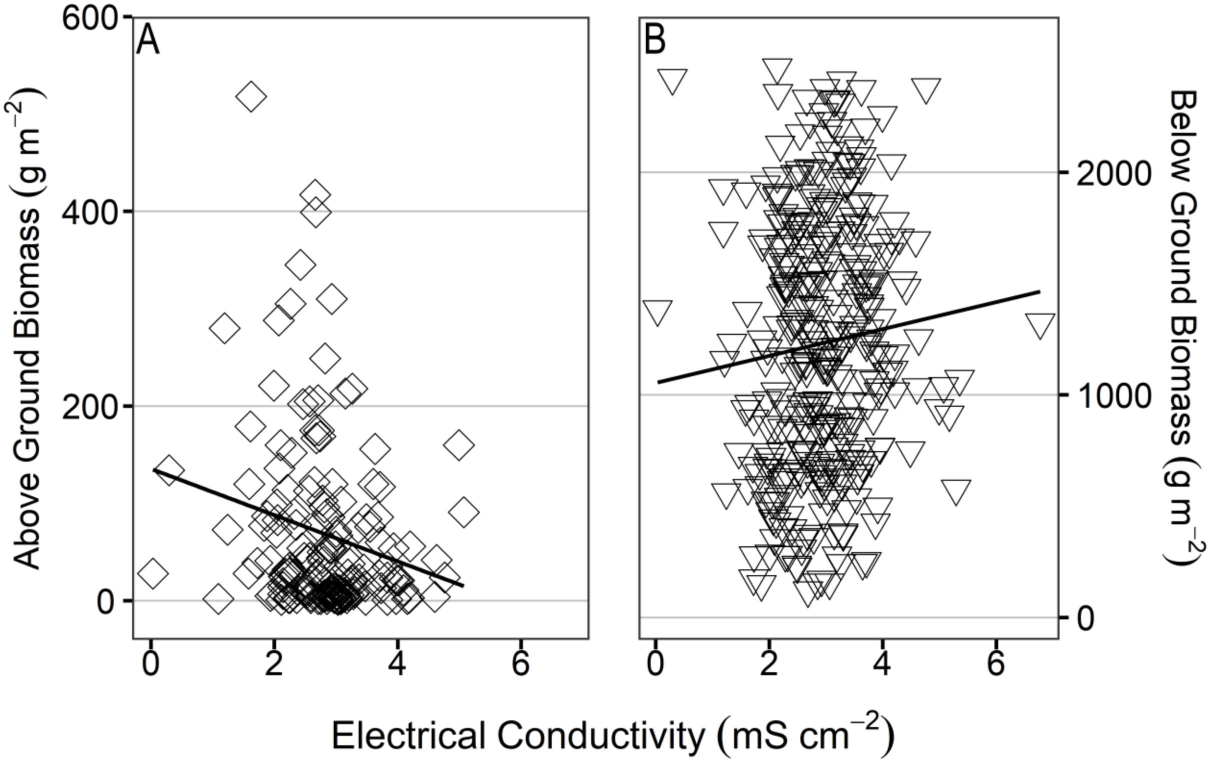

Figure 9 – Pearson's Product-Moment correlations between water table, electrical conductivity, (EC) mS cm^{-1}, and above-ground biomass of non-dominant vascular plant species (A) and total below-ground biomass (B). Points represent sampled locations in years three through five were biomass was collected (i.e., zero values were removed). Non-dominant species AGB demonstrated a weak and negative correlation, r^2(n 157) = 0.16, P < 0.05, with EC, whereas total below-ground biomass was weak and positively correlated, r^2(n 334) = 0.13, P < 0.01, with electrical conductivity.

TOTAL ABOVE AND BELOW-GROUND BIOMASS ON THE RECLAIMED FEN

Total AGB averaged 449 ± 11.0 g m^{-2} (*n* 428) across all treatments and years, which accounted for 27.2% ± 0.7 of the total annual biomass contributed by vascular plants (Figure 10). Planting treatment was a significant predictor ($F_{1,6}$ = 2.44, P < 0.05) of total AGB when analyzed in a one-way ANOVA. Despite an insignificant main effect of sampling year ($F_{1,3}$ = 0.11, ns), the interaction between planting treatment and sampling year was significant ($F_{2,9}$ = 1.91, P < 0.05) and explained 20.4% of the variance for total above-ground annual growth. *Typha latifolia* removal treatments had no significant effect on total AGB ($F_{1,3}$ = 1.04, ns). Water table depth and the interaction between sampling year was also significant in predicting total AGB ($F_{1,1}$ = 15.7, P < 0.01 and $F_{1,2}$ = 4.25, P < 0.05, respectively) and explained 17.9% of the variance for AGB between years four and five.

Total BGB averaged 1330 ± 35.8 g m^{-2} (*n* 334) across all treatments and years and accounted for 72.5% ± 0.7 of the total annual biomass from vascular plants (Figure 10). Planting treatments MLT + SLc (1501 ± 155 g m^{-2}, *n* 50) and SLc (1260 ± 82.4 g m^{-2}, *n* 65) produced 39.9% more BGB ($F_{1,6}$ = 4.64, P < 0.001) compared to UP treatments (978 ± 111 g m^{-2}, *n* 51) when averaged across years. Sampling year was a significant predictor of total BGB ($F_{1,2}$ = 84.8, P < 0.001), produced more BGB in years four and five compared to year three, and explained 34.6% of variance between years. The interaction between planting treatment and sampling year was not significant ($F_{2,8}$ = 0.594, ns). *Typha latifolia* removal treatments explained 6.96% of mean variance ($F_{1,3}$ = 3.25, P < 0.05) for total BGB. However, differences between treatments were not significant in the tukey-adjusted post-hoc analysis. The effect-level of water table depth on total BGB was significant ($F_{1,3}$ = 5.13, P < 0.05) when combined with sampling year and the resulting interaction ($F_{1,3}$ = 3.44, P < 0.05) explained 34.8% of BGB variance between years. The one-way effect of electrical conductivity was also a significant predictor ($F_{1,1}$ = 5.76, P < 0.05) of total BGB.

Table 4 – One-way and two-way ANCOVA results for fixed and continuous predictor variables on total above-ground (AGB) and total below-ground (BGB) vascular biomass. Main effects include; year (Yr), planting treatment (PT), *Typha latifolia* removal treatment (TR), and seedling density treatments (Den). Covariate factors include; depth to water (DTW) and electrical conductivity (EC). Replicate was set to random to account for hydrologic variations in the model (1 | replicate).

		AGB		BGB	
Factor	d.f.	*F*	*P*	*F*	*P*
Yr	2	0.110	0.898	84.76	< 0.001
PT	6	2.44	< 0.05	4.63	< 0.001
TR	3	2.79	< 0.05	3.25	< 0.05
Den	1	0.39	0.533	2.75	0.099
DTW	1	20.5	< 0.001	5.97	< 0.05
EC	1	0.01	0.939	5.76	< 0.05
PT * Yr	12	1.80	< 0.05	0.59	0.847
TR * Yr	4	1.71	0.146	0.35	0.841
Den * Yr	2	2.62	0.075	2.69	0.070
DTW * Yr	2	7.77	< 0.001	4.00	< 0.015
EC * Yr	2	1.21	0.30	0.94	0.392
(1 | replicate)	1	--	< 0.001	--	< 0.001

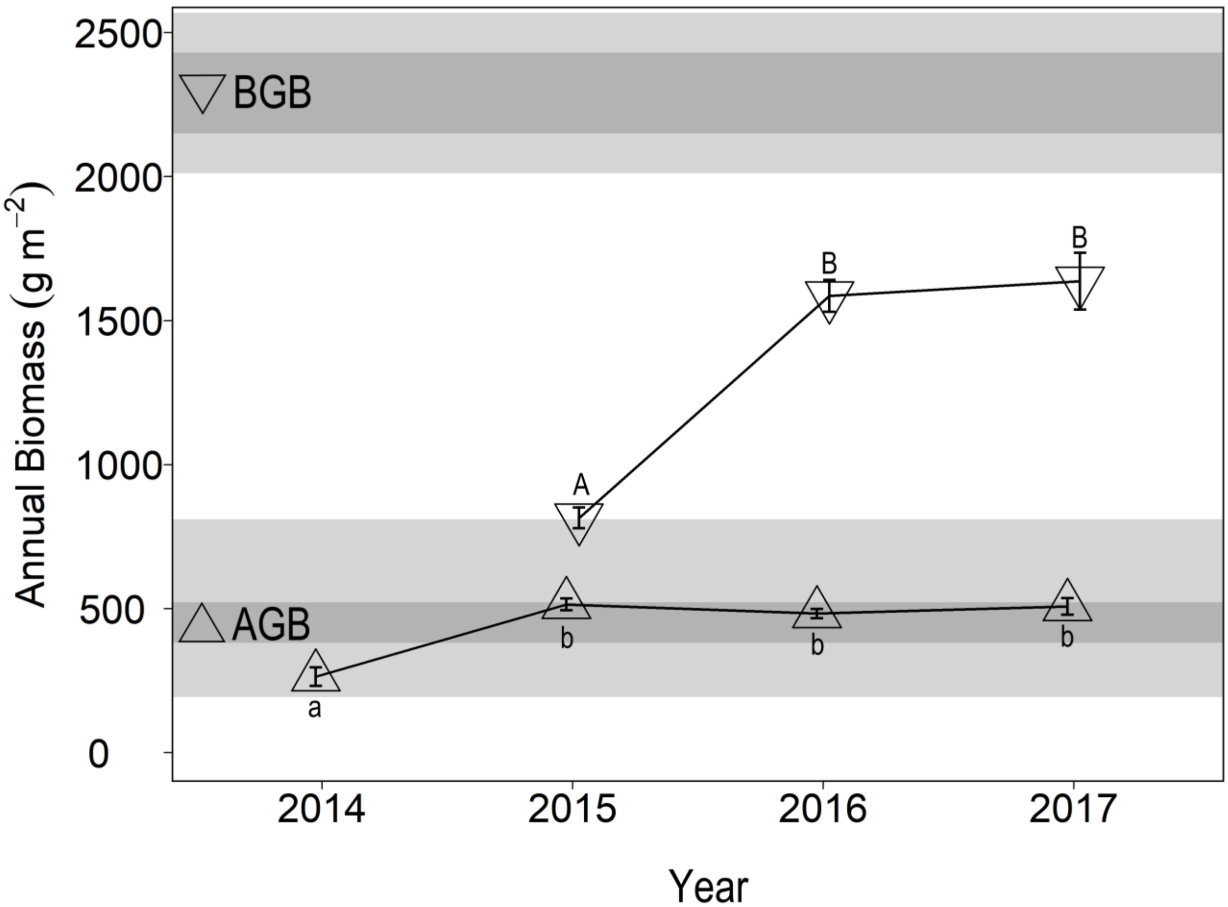

Figure 10 – Total above-ground (AGB) and below-ground (BGB) vascular biomass averaged across planting treatments for sampling years two (2014) through five (2017) on the reclaimed fen. Shaded area represents the range (light) and mean ± sd (dark) for above-ground and below-ground annual biomass found in sedge-dominated peatlands of western Canada (Bernard, Solander, & Kvet, 1988; Szumigalski, 1995; Saarinen, 1996; Szumigalski & Bayley, 1997; Thormann & Bayley, 1997).

DISCUSSION

I demonstrate the effects of introducing vascular plant species as seedlings, seeds, and propagules on annual above-ground biomass production in post-mined fen peatlands. Three years after planting mean annual vascular plant AGB was comparable to natural sedge-dominated peatlands in western Canada. However, average BGB in the upper 50 cm of soil was below the range and mean for regional peatlands after five years. Treatments planted with *C. aquatilis* seedlings produced the most AGB when averaged across years and were surpassed by *J. balticus* AGB only in SLj and MLT + SLj plots. *Juncus balticus* remained the dominant species through year four where it was planted as seedlings without the donor material. Seedling planting densities did not influence *J. balticus* persistence and there were no differences in *C. aquatilis* AGB between high and low-density treatments in any year. Water table depth and pore-water electrical conductivity were relatively uniform in the Fen and below the range occurring in regional natural saline fens where *J. balticus* is the dominant vascular plant. The planting of *C. aquatilis* seedlings, seeds, and propagules from the MLT treatments and suitable water table and electrical conductivity ranges have facilitated the tremendous growth of *C. aquatilis* that has reduced *J. balticus* presence and production throughout this experiment.

CAREX AQUATILIS ABOVE-GROUND BIOMASS ON THE FEN

By year three, annual *C. aquatilis* AGB on the Fen was similar to published ranges for temperate zone wetlands 500 and 1050 g m^{-2} (Bernard, Solander, & Kvet, 1988), a montane fen in west-central Alberta, 193 and 810 g m^{-2} (Gorham & Somers, 1973), and 277 to 757 g m^{-2} yr^{-1} along a marsh-bog-fen gradient (Thormann & Bayley, 1997). Annual production on the Fen was slightly higher than the pooled mean AGB reported for open fens in Alberta (Campbell et al., 2000) and lower than a boreal mesotrophic fen in Finland (2280 ± 249 g m^{-2}, Saarinen, 1996).

Similar biomass occurred in a reclaimed wet meadow in Alberta's oil sands region (Raab and Bayley 2013).

The transfer of residual *C. aquatilis* seeds and fragments to MLT treatments from the donor site likely increased *C. aquatilis* density and presence across the Fen. Natural recruitment on reclaimed sites is often limited if the sites are disconnected from natural seed and propagule sources (Houlahan et al., 2006; Aronson & Galatowitsch, 2008), so collecting donor material from ecologically similar sites, including residual seeds and living plant fragments, often occurs in reclamation to achieve plant species assemblages similar to natural communities (Cooper & MacDonald, 2000; Cobbaert, Rochefort, & Price, 2004; Galatowitsch, 2006). This revegetation method is effective for introducing native species (Mitsch & Gosselink, 2000; Rochefort et al., 2003; Trites & Bayley, 2009b); however, the inclusion of residual *C. aquatilis* seeds and fragments in donor material can ultimately serve as a hindrance for increasing plant diversity on reclaimed post-mined fen peatlands.

JUNCUS BALTICUS ABOVE-GROUND BIOMASS ON THE FEN

Juncus balticus produced the most AGB where water levels averaged 10 to 20 cm below the ground surface and EC ranging from 3 to 4 mS cm^{-2}. This is comparable to a natural saline fen located ~ 10 km south of Fort McMurray where *J. balticus* occurs in dense stands with an average water table depth of -10.9 cm and EC ranging from 10.6 to 20 mS cm^{-1} (Wells & Price, 2015; Murray, Barlow, & Strack, 2017). In natural wetlands, *J. balticus* occurs as a co-dominant in seasonally to permanently saturated conditions (up to 15 cm) and in electrical conductivity ranging from 0.1 to 20.1 mS cm^{-1} across North America (Kantrud, Krapu, & Swanson, 1989; Cooper, Wolf, & Gage, 2006). Water availability is an important factor affecting seedling establishment and restoration success (Roth et al., 1999) and greater success for introducing fen species as transplants was achieved at near-surface water levels (Cooper & MacDonald, 2000). Consistent near-surface water levels occurred during the first and second growing seasons across the Fen and increased to approximately -20 cm in year three (Kessel, 2016).

Successful revegetation of *Juncus* spp. as rhizome transplants and seeds was relatively low compared to *C. aquatilis* where seasonal water levels ranged from 20 to 50 cm below the surface on a peat mined site in Colorado (Cooper & MacDonald, 2000). Consistent waterlogging adversely affected the survival of *Juncus balticus* seedlings planted on a reclaimed bog (Montemayor, Price, & Rochefort, 2015).

The seedling planting treatment containing the most above-ground *J. balticus* biomass and the seedlings have persisted as the dominant species through year five. Success of plant species on reclaimed and restored sites is dependent on both their tolerance thresholds to site conditions and their life-stage when introduced (Callaway & Walker, 1997; Zedler, Morzaria-Luna, & Ward, 2003). The decline of *J. balticus* on the Fen indicates that the long-term persistence of *J. balticus* requires its introduction as seedlings, especially when planted in the vicinity of *C. aquatilis*. Plant community development can vary depending on whether species are introduced as rhizome fragments, seeds, or seedlings (Parmenter & Macmahon, 1983; Palmer & Poff, 1997). An example of this is the seeded treatment that contained the third highest *J. balticus* above-ground biomass and had the highest AGB of non-dominant species. In addition, *J. balticus* recruitment in UP, MLT, and MLT + SLc suggests that seed dispersal of *J. balticus* from adjacent plots was successful despite the negative competitive interaction with *C. aquatilis*. Above-ground biomass in *J. balticus* seedling treatments indicated that its co-dominance with *C. aquatilis* is more likely to occur when introduced as a high-density planting treatment.

PRESENCE OF *TYPHA LATIFOLIA* AND NON-DOMINANT VASCULAR SPECIES

The highest mean above-ground *T. latifolia* biomass occurred where planting treatments were not applied due to excessively deep standing water. The accumulation of *T. latifolia* has been positively correlated with live plant biomass and nutrient availability (Olde Venterink, Van Der Vliet, & Wassen, 2001). However, litter accumulation reduces seed germination and dense stands of *T. latifolia* in natural, restored and reclaimed wetlands have been shown to reduce plant diversity (Vaccaro, 2005; Frieswyk & Zedler, 2006; Tulbure, Johnston, & Auger, 2007).

Non-dominant vascular plant species contributed a small proportion of above-ground biomass across all treatments (Figure 4). This suggests that the methods used are insufficient for introducing a diverse plant community in the presence of *Carex aquatilis*. Seeded and moss layer transfer treatments were applied as means to test the introduction of vascular plants and bryophytes to areas lacking a viable seed bank.

EFFECTS OF WATER TABLE AND ELECTRICAL CONDUCTIVITY ON PLANT GROWTH

The moss layer transfer approach may have effectively mitigated the negative effects of Na^+ in oil sands wetlands during the initial stages of plant growth (Vitt et al., 2011). Electrical conductivities range from 28 to 560 $\mu S\ cm^{-1}$ in boreal plains fens (Zoltai & Johnson, 1987; Vitt & Chee, 1990; Turetsky & Ripley, 2005) and near-surface EC ranging from 19 to 39 $mS\ cm^{-1}$ in a brackish fen near Fort McMurray (Wells & Price, 2015). Trites and Bayley (2009b, 2009a) reported electrical conductivity ranging from 500 to 3300 $\mu S\ cm^{-1}$ in oil sands wetlands, compared to ranges from 30 to 6770 $\mu S\ cm^{-1}$ on the Fen (Kessel, 2016). Concentrations of Na^+ within the peat on alkaline fens range from 100 to ~ 1000 mg kg^{-1} (Vitt & Chee, 1990), compared to 202 to 428 mg L^{-1} on the Fen (Kessel, 2016). Salinity and naphthenic acid content in oil sands tailings have been shown to reduce plant height and leaf elongation in *C. aquatilis* (Mollard et al., 2012) and decrease below-ground growth (Roy, Mollard, & Foote, 2014). This suggests that salinity and/or naphthenic acid may have long-term effects on biomass accumulation on the Fen. Evaporative properties in natural and reclaimed systems cause Na^+ to accumulate at the peat surface (Rowell, 1994; Qadir, Ghafoor, & Murtaza, 2000; Montemayor et al., 2008), negatively affecting above and below-ground plant production (Montemayor et al., 2008; Rezanezhad et al., 2012). Mollard et al. (2012) demonstrated that despite reductions to above-ground structure, the physiological performance (e.g. water use efficiency, photosynthetic rate, and stomatal conductance) of *C. aquatilis* was minimally affected when exposed to 13.7 ± 5.1 μmol $^{-1}$ dry weight Na^+ concentrations.

FUTURE PEATLAND RECLAMATION IN ALBERTA, CANADA

The planting treatments MLT and MLT + SLc led to the highest total AGB when averaged across sampling years, likely due to the dominance and high production of *C. aquatilis*. Total BGB increased dramatically between the third and fourth years yet remains below that observed in regional fens. Introducing plant species resilient to increasing salinity is imperative to the success of reclaiming fen peatlands on post-mined oil sands landscapes. While the transfer of moss, seeds, and propagules can be effective for introducing native plant communities and mitigating the accumulation of Na^+ on the peat surface, viable *C. aquatilis* seeds and plant fragments present in the moss layer transfer material may have minimized the effects of seedling planting. Reducing or eliminating viable *C. aquatilis* seeds is necessary to increase plant diversity on reclaimed post-mined fen peatlands.

Hydrologic gradients have a strong effect on plant species composition and can alter the biota, moisture, and oxygen dynamics of litter decomposition (Wisheu and Keddy, 1992; Battle and Golladay, 2001). Homogenous water levels on the Fen reduce competition dynamics between plant species, but ultimately hinder the development of microtopography. Introducing hydrologic variation to future reclaimed fens may reduce the competitive advantage of *C. aquatilis* while increasing *J. balticus* persistence. The collective influence of hydrologic fluctuations, seed dispersal and viability, litter accumulation, and biogeochemistry must be considered to successfully reclaim fen peatlands on the post-mined oil sands landscape.

LITERATURE CITED

Allen, E. W. (2008). Process Water Treatment in Canada's Oil Sands Industry: I. Target Pollutants and Treatment Objectives. *Journal of Environmental Engineering and Science*, 7(2), 123–138. Retrieved from http://www.icevirtuallibrary.com/doi/10.1139/S07-038

Aronson, M. F. J. & Galatowitsch, S. (2008). Long-Term Vegetation Development of Restored Prairie Pothole Wetlands. *Wetlands*, 28(4), 883–895. Retrieved from http://link.springer.com/10.1672/08-142.1

Athy, E. R., Keiffer, C. H. & Stevens, M. H. (2006). Effects of Mulch on Seedlings and Soil on a Closed Landfill Effects of Mulch on Seedlings and Soil on a Closed Landfill. *Restoration*, 14(2), 233–241.

Backeus, I. (1990). Production and Depth Distribution of Fine Roots in a Boreal Open Bog. *Annales Botanici Fennici*, 27(3), 261–265. Retrieved from https://www.jstor.org/stable/23725366

Bates, D., Mächler, M., Bolker, B. M. & Walker, S. (2015). Fitting Linear Mixed-Effects Models Using Lme4. *Journal of Statistical Software*, 67(1).

Bergamini, A., Pauli, D., Peintinger, M. & Schmid, B. (2001). Relationship between Productivity, Numer of Shoots and Number of Species in Bryophytes and Vascular Plants. *Journal of Ecology*, 89(6), 920–929.

Bernard, J. M. & Fiala, K. (1986). Distribution and Standing Crop of Living and Dead Roots in Three Wetland Carex Species. *Bulletin of the Torrey Botanical Club*, 113(1), 1–5. Retrieved from http://www.jstor.org/stable/2996226

Bernard, J. M., Solander, D. & Kvet, J. (1988). Production and Nutrient Dynamics in Carex Wetlands. *Aquatic Botany*, 30, 125–147.

Borkenhagen, A. K. (2014). Recreating Peatland Initiation Conditions: Methods for Reclaiming Peatlands in Alberta's Oil Sands Region. Retrieved July 2, 2017, from https://dspace.library.colostate.edu/bitstream/handle/10217/82582/Borkenhagen_colostate_0053N_12330.pdf?sequence=1&isAllowed=y

Borkenhagen, A. K. & Cooper, D. J. (2016). Creating Fen Initiation Conditions: A New Approach for Peatland Reclamation in the Oil Sands Region of Alberta. *Journal of Applied Ecology*, 53(2), 550–558.

Botch, M. S., Kobak, K. I., Vinson, T. S. & Kolchugina, T. P. (1995). Carbon Pools and Accumulation in Peatlands of the Former Soviet Union. *Global Biogeochemical Cycles*, 9(1), 37–46.

Burchill, C. A. & Kenkel, N. C. (1991). Vegetation–environment Relationships of an Inland Boreal Salt Pan. *Canadian Journal of Botany*, 69(4), 722–732.

Callaway, R. M. & Walker, L. R. (1997). Competition and Facilitation : A Synthetic Approach to Interactions in Plant Communities. *Ecology*, 78(7), 1958–1965. Retrieved from http://www.jstor.org/stable/2265936

Campbell, C., Vitt, D. H., Halsey, L. A., Campbell, I. D., Thormann, M. N. & Bayley, S. E. (2000). *Net Primary Production and Standing Biomass in Northern Continental Wetlands.*

Chee, W. L. & Vitt, D. H. (1989). The Relationships of Vegetation to Surface Water Chemistry and Peat Chemistry in Fens of Alberta, Canada. *Vegetatio*, 9(2), 227–261.

Chimner, R. A. (2011). Restoring Sedges and Mosses into Frost Heaving Iron Fens, San Juan Mountains, Colorado. *Mires and Peat*, 8(January 2011), Article 07, 1-9.

Chimner, R. A., Cooper, D. J. & Parton, W. J. (2002). Modeling Carbon Accumulation in Rocky Mountain Fens. *Wetlands*, 22(1), 100–110. Retrieved from http://resolver.scholarsportal.info/resolve/02775212/v22i0001/100_mcairmf

Clymo, R. S. (1984). The Limits to Peat Bog Growth. *Philosophical Transactions of the Royal*

Society of London. Series B, Biological Sciences Phil. Trans. R. Soc. Lond. B, 303(303), 605–654. Retrieved January 25, 2017, from http://www.jstor.org/stable/2396128
Cobbaert, D., Rochefort, L. & Price, J. S. (2004). Experimental Restoration of a Fen Plant Community after Peat Mining. *Applied Vegetation Science*, 7(2), 209–220. Retrieved from http://www.jstor.org/stable/4620414
Cooper, D. J. & MacDonald, L. H. (2000). Restoring the Vegetation of Mined Peatlands in the Southern Rocky Mountains of Colorado, U.S.A. *Restoration Ecology*, 8(2), 103–111. Retrieved from http://onlinelibrary.wiley.com/doi/10.1046/j.1526-100x.2000.80016.x/full
Cooper, D. J., Wolf, E. C. & Gage, E. (2006). Plant Establishment for Wetland Reclamation: A Review of Plant Establishment Techniques, and Species Tolerances for Water Level and Salinity. *Cumulative Environmental Management Association*, 4(780), 1–115.
Dag, O., Dolgun, A. & Konar, N. M. (2018). Onewaytests: An R Package for One-Way Tests in Independent Groups Designs. *The R Journal*, 10(July), 175–199. Retrieved from https://journal.r-project.org/archive/2018/RJ-2018-022/index.html
Downing, D. J. & Pettapiece, W. W. (2006). *Natural Regions and Subregions of Alberta*. Retrieved from http://www.cd.gov.ab.ca/preserving/parks/anhic/Natural_region_report.asp
Ecological Stratification Working Group. (1995). *A National Ecological Framework for Canada*.
Franks, C. D. & Goings, K. A. (1998). Separating Roots from the Soil by Hand Sieving. *USDA NRCS publication*, 2–3. Retrieved March 27, 2017, from https://www.nrcs.usda.gov/Internet/FSE_DOCUMENTS/nrcs142p2_050967.pdf
Frieswyk, C. B. & Zedler, J. B. (2006). Do Seed Banks Confer Resilience to Coastal Wetlands Invaded by Typha × Glauca ? *Canadian Journal of Botany*, 84(12), 1882–1893. Retrieved from http://www.nrcresearchpress.com/doi/10.1139/b06-100
Galatowitsch, S. M. (2006). Restoring Prairie Pothole Wetlands: Does the Species Pool Concept Offer Decision-Making Guidence for Re-Vegetation. *Applied Vegetation Science*, 9(2), 261–270.
Gignac, L. D., Gauthier, R., Rochefort, L. & Bubier, J. (2004). Distribution and Habitat Niches of 37 Peatland Cyperaceae Species across a Broad Geographic Range in Canada. *Canadian Journal of Botany*, 82(9), 1292–1313. Retrieved from http://www.nrcresearchpress.com/doi/10.1139/b04-081
Glaeser, L. C., Vitt, D. H. & Ebbs, S. (2016). Responses of the Wetland Grass, Beckmannia Syzigachne, to Salinity and Soil Wetness: Consequences for Wetland Reclamation in the Oil Sands Area of Alberta, Canada. *Ecological Engineering*, 86, 24–30. Retrieved from http://dx.doi.org/10.1016/j.ecoleng.2015.10.009
González, E. & Rochefort, L. (2014). Drivers of Success in 53 Cutover Bogs Restored by a Moss Layer Transfer Technique. *Ecological Engineering*, 68, 279–290. Retrieved March 21, 2017, from http://dx.doi.org/10.1016/j.ecoleng.2014.03.051
Gorham, E. & Somers, M. G. (1973). Seasonal Changes in the Standing Crop of Two Montane Sedges. *Canadian Journal of Botany*, 51, 1097–1108.
Government of Canada. (2015). Calculation Information for 1981 to 2010 Canadian Normals Data. Retrieved February 5, 2017, from http://climate.weather.gc.ca/climate_normals/results_1981_2010_e.html?stnID=2519
Hammer, U. T. & Heseltine, J. M. (1988). Aquatic Macrophytes in Saline Lakes of the Canadian Prairies. *Hydrobiologia*, 158, 101–116.
Historical Climate Data, 1981 - 2010. (2016). Government of Canada. Retrieved October 3, 2016, from climate.weather.gc.ca/climate_normals/index_e.html.
Houlahan, J. E., Keddy, P. A., Makkay, K. & Findlay, C. S. (2006). Effects of Adjacent Land Use on Wet Sp Richness and Community Composition. *Wetlands*, 26(1), 79–96.
Kantrud, H. A., Krapu, G. L. & Swanson, G. A. (1989). *Prairie Basin Wetlands of the Dakotas a Community Profile*. Retrieved from https://pubs.er.usgs.gov/publication/2000127
Kenward, M. G. & Roger, J. H. (1997). Small Sample Inference for Fixed Effects from Restricted

Maximum Likelihood. *Biometrics*, 53(3), 983–997. Retrieved from https://www.jstor.org/stable/2533558

Kessel, E. (2016). The Hydrogeochemistry of a Constructed Fen Peatland in a Post-Mined Landscape in The.

Kotowski, W. & van Diggelen, R. (2004). Light as an Environmental Filter in Fen Vegetation. *Journal of Vegetation Science*, 15, 583–594.

Kuhry, P. & Turunen, J. (2006). The Postglacial Development of Boreal and Subarctic Peatlands, in: Caldwell, L., Heldmaier, M., Jackson, D., Lange, W., Mooney, S., Schulze, J., and Sommer, K. (Eds.), *Ecological Studies*, (pp. 1–431). Springer-Verlag Berlin Heidelberg.

Lenth, R. V. (2016). Least-Squares Means: The R Package Lsmeans. *Journal of Statistical Software*, 69(1). Retrieved from http://www.jstatsoft.org/v69/i01/

Levene, H. (1960). Robust Tests for Equality of Variances, in: *Contributions to probability and statistics; essays in honor of Harold Hotelling.*, (pp. 278–292). Palo Alto: Standor, CA : Stanford University Press.

Mitsch, W. J. & Gosselink, J. G. (2000). The Value of Wetlands: Importance of Scale and Landscape Setting. *Ecological Economics*, 35, 25–33. Retrieved January 28, 2017, from www.elsevier.com/locate/ecolecon

Mollard, F. P. O., Roy, M. C., Frederick, K. & Foote, A. L. (2012). Growth of the Dominant Macrophyte Carex Aquatilis Is Inhibited in Oil Sands Affected Wetlands in Northern Alberta, Canada. *Ecological Engineering*, 38(1), 11–19.

Montemayor, M. B., Price, J. S. & Rochefort, L. (2015). The Importance of PH and Sand Substrate in the Revegetation of Saline Non-Waterlogged Peat Fields. *Journal of Environmental Management*, 163, 87–97. Retrieved from http://dx.doi.org/10.1016/j.jenvman.2015.07.052

Montemayor, M. B., Price, J. S., Rochefort, L. & Boudreau, S. (2008). Temporal Variations and Spatial Patterns in Saline and Waterlogged Peat Fields. 1. Survival and Growth of Salt Marsh Graminoids. *Environmental and Experimental Botany*, 62(3), 333–342.

Murray, K. R., Barlow, N. & Strack, M. (2017). Methane Emissions Dynamics from a Constructed Fen and Reference Sites in the Athabasca Oil Sands Region, Alberta. *Science of the Total Environment*, 583, 369–381. Retrieved from http://dx.doi.org/10.1016/j.scitotenv.2017.01.076

National Wetlands Working Group. (1997). The Canadian Wetland Classification System (B. G. Warner and C. D. A. Rubec, Eds.). , 68.

Nwaishi, F., Petrone, R. M., Price, J. S. & Andersen, R. (2015). Towards Developing a Functional-Based Approach for Constructed Peatlands Evaluation in the Alberta Oil Sands Region, Canada. *Wetlands*, 35(2), 211–225.

Nwaishi, F., Petrone, R. M., Price, J. S., Ketcheson, S. J., Slawson, R. & Andersen, R. (2015). Impacts of Donor-Peat Management Practices on the Functional Characteristics of a Constructed Fen. *Ecological Engineering*, 81, 471–480. Retrieved from http://dx.doi.org/10.1016/j.ecoleng.2015.04.038

Olde Venterink, H., Van Der Vliet, R. E. & Wassen, M. J. (2001). Nutrient Limitation along a Productivity Gradient in Wet Meadows. *Plant and Soil*, 234(2), 171–179.

Ovenden, L. (1990). Peat Accumulation in Northern Wetlands. *Quaternary Research*, 33(3), 377–386.

Palmer, M. A. & Poff, L. N. (1997). The Influence of Environmental Heterogeneity on Patterns and Processes in Streams. *Journal of the North American Benthological Society*, 16(1), 169–173.

Parmenter, R. R. & Macmahon, J. A. (1983). International Association for Ecology Factors Determining the Abundance and Distribution of Rodents in a Shrub-Steppe Ecosystem: The Role of Shrubs. *Oecological*, 59(2/3), 145–156.

Pouliot, R., Rochefort, L. & Graf, M. D. (2013). Fen Mosses Can Tolerate Some Saline

Conditions Found in Oil Sands Process Water. *Environmental and Experimental Botany*, 89, 44–50. Retrieved from http://dx.doi.org/10.1016/j.envexpbot.2013.01.004
Price, J. S., McLaren, R. G. & Rudolph, D. L. (2010). Landscape Restoration after Oil Sands Mining: Conceptual Design and Hydrological Modeling for Fen Reconstruction. *International Journal of Mining, Reclamation and Environment*, 24(2), 109–123. Retrieved from http://dx.doi.org/10.1080/17480930902955724
Price, J. S., Rochefort, L. & Quinty, F. (1998). Energy and Moisture Considerations on Cutover Peatlands: Surface Microtopography, Mulch Cover and Sphagnum Regeneration. *Ecological Engineering*, 10, 293–312.
Price, J. S. & Rudolph, D. L. (2007). Creating a Fen Peatland on a Post-Mined Oilsands Landscape: A Feasibility Modelling Study (Phase 2). *Cumulative Environmental Management Association*, 4(780), 1–37.
Qadir, M., Ghafoor, A. & Murtaza, G. (2000). Amelioration Strategies for Saline Soils: A Review. *Land Degradation & Development*, 11, 501–521.
R Development Core Team. (2015). R Internals. *R Development Core Team*, 1, 63. Retrieved from https://cran.r-project.org/doc/manuals/r-release/R-ints.html
Raab, D. & Bayley, S. E. (2013). A Carex Species-Dominated Marsh Community Represents the Best Short-Term Target for Reclaiming Wet Meadow Habitat Following Oil Sands Mining in Alberta, Canada. *Ecological Engineering*, 54, 97–106. Retrieved from http://dx.doi.org/10.1016/j.ecoleng.2013.01.024
Rezanezhad, F., Andersen, R., Pouliot, R., Price, J. S., Rochefort, L. & Graf, M. D. (2012). How Fen Vegetation Structure Affects the Transport of Oil Sands Process-Affected Waters. *Wetlands*, 32(3), 557–570.
Riley, J. L. & Michaud, L. (1994). Ontario Peatland Inventory: Field-Work Methods. *Ontario Geological Survey*, 62.
Rochefort, L., Quinty, F., Campeau, S., Johnson, K. & Malterer, T. (2003). North American Approach to the Restoration of Sphagnum Dominated Peatlands. *Wetlands Ecology and Management*, 11, 3–20.
Rochefort, L., Strack, M., Poulin, M., Price, J. S., Graf, M. D., Desrochers, A. & Lavoie, C. (2012). Northern Peatlands, in: Batzer, D. P. and Baldwin, A. (Eds.), *Wetland Habitats of North America: Ecology and Conservation Concerns*, (p. 408). Los Angeles: University oc California Press. Retrieved from http://www.ucpress.edu/book.php?isbn=9780520271647
Rooney, R. C., Bayley, S. E. & Schindler, D. W. (2012). Oil Sands Mining and Reclamation Cause Massive Loss of Peatland and Stored Carbon. *Proceedings of the National Academy of Sciences*, 109(13), 4933–4937. Retrieved from http://www.pnas.org/cgi/doi/10.1073/pnas.1117693108
Roth, S., Seeger, T., Poschlod, P., Pfadenhauer, J. & Succow, M. (1999). Establishment of Helophytes in the Course of Fen Restoration Source : Applied Vegetation Science , Vol . 2 , No . 1 , From Basic to Applied Ecology : Vegetation Science for Nature Conservation (May , 1999), Pp . 131-136 Published by : Wiley Stable URL : *Applied Vegetation Science*, 2(1), 131–136.
Rowell, D. L. (1994). Soil Science: Methods and Applications. *Journal of the Science of Food and Agriculture*, 66, 573–576.
Roy, M.-C., Mollard, F. P. O. & Foote, A. L. (2014). Do Peat Amendments to Oil Sands Wet Sediments Affect Carex Aquatilis Biomass for Reclamation Success? *Journal of Environmental Management*, 139, 154–163. Retrieved from http://dx.doi.org/10.1016/j.jenvman.2014.03.003
RSA. (2016). *Environmental Protection and Enhancement Act*. AR 115/93.
Saarinen, T. (1996). Biomass and Production of Two Vascular Plants in a Boreal Mesotrophic Fen. *Canadian Journal of Botany*, 74(6), 934–938.
Shaver, G. R. & Chapin III, F. S. (1991). Production : Biomass Relationships and Element

Cycling in Contrasting Arctic Vegetation Types. *Ecological Monographs*, 61(1), 1–31. Retrieved from http://www.jstor.org/stable/1942997
Sjors, H. (1959). Bogs and Fens in the Hudson Bay Lowlands. *Arctic*, 12, 2–19.
Stevens, M. & Hoag, C. (2000). USDA NRCS Plant Fact Sheet for Baltic Rush (Juncus Balticus Willd.). *USDA NRCS publication*, 2. Retrieved from http://plants.usda.gov/plantguide/pdf/cs_juba.pdf
Szumigalski, A. R. (1995). Production and Decomposition of Vegetation along a Wetland Gradient in Central Alberta.
Szumigalski, A. R. & Bayley, S. E. (1997). Net Aboveground Primary Production along a Peatland Gradient in Central Alberta in Relation to Environmental Factors. *Ecoscience*, 4(3), 385–393. Retrieved from http://www.jstor.org/stable/42900869
Thormann, M. N. & Bayley, S. E. (1997). Aboveground Net Primary Production along a Bog-Fen-Marsh Gradient in Southern Boreal Alberta, Canada. *Ecoscience*, 4(3), 374–384. Retrieved from http://www.jstor.org/stable/42900868
Thormann, M. N., Bayley, S. E. & Currah, R. S. (2001). Comparison of Decomposition of Belowground and Aboveground Plant Litters in Peatlands of Boreal Alberta, Canada. *Canadian Journal of Botany*, 79(1), 9–22. Retrieved from http://www.nrcresearchpress.com/doi/10.1139/b00-138
Thunhorst, G. A. (1993). *Wetland Planting Guide for the Northeastern United States: Plants for Wetland Creation, Restoration, and Enhancement*. St. Michaels, Md. : Environmental Concern.
Trites, M. & Bayley, S. E. (2009a). Organic Matter Accumulation in Western Boreal Saline Wetlands: A Comparison of Undisturbed and Oil Sands Wetlands. *Ecological Engineering*, 35(12), 1734–1742.
Trites, M. & Bayley, S. E. (2009b). Vegetation Communities in Continental Boreal Wetlands along a Salinity Gradient: Implications for Oil Sands Mining Reclamation. *Aquatic Botany*, 91(1), 27–39.
Tulbure, M. G., Johnston, C. A. & Auger, D. L. (2007). Rapid Invasion of a Great Lakes Coastal Wetland by Non-Native Phragmites Australis and Typha. *Journal of Great Lakes Research*, 33(3), 269–279. Retrieved from http://dx.doi.org/10.3394/0380-1330(2007)33[46:HNSILE]2.0.CO;2
Turetsky, M. R. & Ripley, S. (2005). Decomposition in Extreme-Rich Fens of Boreal Alberta, Canada. *Soil Science Society of America Journal*, 69(6), 1856.
Vaccaro, L. E. (2005). Patterns , Mechanisms , and Ecological Implications of Cattail (Typha Spp .) Dominance in Great Lakes Wetlands. *Wetlands*, (August).
Venables, W. N. & Ripley, B. D. (2002). *Modern Applied Statistics with S-Plus*. New York: Springer. Retrieved from http://www.stats.ox.ac.uk/pub/MASS4
Vitt, D. H. & Chee, W. (1990). The Relationships of Vegetation to Surface Water Chemistry and Peat Chemistry in Fens of Alberta, Canada. *Vegetatio*, 89(2), 87–106.
Vitt, D. H., Halsey, L. A., Bauer, I. E. & Campbell, C. (2000). Spatial and Temporal Trends in Carbon Storage of Peatlands of Continental Western Canada through the Holocene. *Canadian Journal of Earth Sciences*, 37(5), 683–693.
Vitt, D. H., Wieder, R. K., Scott, K. D. & Faller, S. (2009). Decomposition and Peat Accumulation in Rich Fens of Boreal Alberta, Canada. *Ecosystems*, 12(3), 360–373.
Vitt, D. H., Wieder, R. K., Xu, B., Kaskie, M. & Koropchak, S. (2011). Peatland Establishment on Mineral Soils: Effects of Water Level, Amendments, and Species after Two Growing Seasons. *Ecological Engineering*, 37(2), 354–363. Retrieved from http://dx.doi.org/10.1016/j.ecoleng.2010.11.029
Waide, R. B., Willig, M. R. & Steiner, C. F. (1999). The Relationship Between Productivity and Species Richness. *Annual Review of Ecology and Systematics*, 30, 257–300.
Wallén, B. (1986). Above and below Ground Dry Mass of the Three Main Vascular Plants on

Hummocks on ASubarctic Peat Bog. *Oikos*, 46(1), 51–56. Retrieved from http://www.jstor.org/stable/3565379
Wallén, B., Falkengren-Grerup, U. & Malmer, N. (1988). Biomass , Productivity and Relative Rate of Photosynthesis of Sphagnum at Different Water Levels on a South Swedish Peat Bog. *Holarctic Ecology*, 11(1), 70–76. Retrieved from http://www.jstor.org/stable/3682591
Walter, H. (1977). The Oligotrophic Peatlands of Western Siberia-the Largest Peino-Helobiome in the World. *Vegetatio*, 34(3), 167–178.
Wells, C. M. & Price, J. S. (2015). A Hydrologic Assessment of a Saline-Spring Fen in the Athabasca Oil Sands Region, Alberta, Canada - a Potential Analogue for Oil Sands Reclamation. *Hydrological Processes*, 29(20), 4533–4548.
Wieder, R. K. & Vitt, D. H. (2006). Functional Characteristics and Indicators of Boreal Peatlands.Pdf, in: Wieder, R. K. and Vitt, D. H. (Eds.), *Boreal Peatland Ecosystems*. Springer-Verlag Berlin Heidelberg.
Zedler, J. B., Morzaria-Luna, H. & Ward, K. (2003). The Challenge of Restoring Vegetation on Tidal, Hypersaline Substrates. *Plant and Soil*, 253(1), 259–273.
Zoltai, S. C. & Johnson, J. D. (1987). Relationships Between Nutrients and Vegetaiton in Peatlands of the Prairie Provinces, in: Rubec, C. D. A. and Overend, R. P. (Eds.), *Wetlands/Peatlands Symposium ’87*, (pp. 535–543). Edmonton: Environment Canada, Lands Directorate. Retrieved from http://cfs.nrcan.gc.ca/publications?id=19222