Carbon gas exchange, primary production and litter decomposition of a restored fen on a former oil well-pad

by

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Author's Declaration

I hereby declare that I am the sole author of this thesis. This is a true copy of the thesis, including any required final revisions, as accepted by my examiners.

I understand that my thesis may be made electronically available to the public.

Abstract

Over 500,000 oil and gas wells have been drilled in Alberta. Recently updated peatland restoration criteria for well-pads creates incentive for peatland restoration, but little is known about functional outcomes of restoration methods. A valued primary function of peatlands is slower decomposition than production rates resulting in peat and carbon accumulation and net neutral or negative greenhouse gas balance. Three restoration techniques on an abandoned well-pad near Peace River, Alberta were measured for carbon dioxide (CO_2) and methane (CH_4) emissions during three growing seasons in 2014-2016 (May-Sept, inclusive), 2-4 years post-restoration. Net primary production (NPP), biomass and decay rates were also measured in the fourth year post-restoration. The peat replacement treatments (PRT) and restoration methods included burying the mineral-pad layer underneath the peat layer (clay), burying the mineral layers with some mixing with peat (mixed), and removing the mineral fill layer completely (peat). Seasonal measurements showed some variation between PRTs from year to year for mean gross ecosystem production and ecosystem respiration, while net ecosystem exchange and CH_4 fluxes were similar between all PRTs in all years, water table position (WT), soil temperature and vegetation cover explained variation in CO_2 exchange, while WT explained some variation in CH_4 flux. All PRTs increasingly developed peatland vegetation cover by the third year and had CO_2 and CH_4 fluxes comparable to the reference sites, despite having significantly different WT position to reference sites. It is likely, however that the reference results were non-representative of the ecosystem level as the similarity in CO_2 fluxes would likely not exist if not for the absence of tree photosynthesis captured in plot scale understory CO_2 measurements.

Decomposition, biomass and NPP also did not differ significantly between PRTs. When compared to natural sites, NPP and biomass, were lower on the restoration site, likely due to the

lack of tree establishment to date. Water table and soil temperature did not explain variation in NPP, or decomposition rates on site; however, ion supply rates (Al, Ca, Cu, Fe, K, Mg, Mn, NO_3, NH_4, P, S, Zn) were correlated to both in some cases. Overall, it is likely that the remnant mineral layer on the site altered peat chemistry, which was seen in abnormally high base cation supply rates. The high ionic availability combined with significantly shallower WT may explain greater decomposition rates compared to reference sites. It is expected that carbon cycling is not restored to that of natural peatlands, as supported by NPP and decay rates different from literature, as well as the drastically different site chemistry. All PRTs are recommended for future restorations as they do show promise for restoring peatland ecosystem functions, however, more research is needed to assess differences in PRTs carbon cycling and peat accumulation in restored peatland. Longer term restoration research should be continued until similar rates are found as on natural peatlands. Future research should involve single PRTs per well-pad for CO_2, CH_4, litter decomposition and production rates and incorporate the overstory carbon gas exchange of treed natural references.

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Preface

This thesis is presented in manuscript style. Both manuscripts were written with the intent of publication with the potential authorship of myself, Maria Strack, Bin Xu and Melanie Bird. I was primarily responsible for all data analysis and writing in both manuscripts as well as conducting the field work used to produce the results. For the first manuscript the in field data collection from 2014 and 2015 for greenhouse gas (GHG) fluxes were collected as part of an ongoing monitoring program on the site. I expanded the sampling program in the 2016 season adding an additional replicate for GHG fluxes in targeted treatments. For the second manuscript I made all measurements of biomass, net primary production and decomposition rates including all field and laboratory sample collection and processing.

Chapter 1: Literature review and objective

1.0 Introduction

The in-situ oil sands production industry has impacted over 800 km^2 of land in Alberta (Vitt et al., 2011; Pasher et al., 2013). In-situ mining is the term used for bitumen extraction methods deeper than 75 m underground (such as Cyclic Steam Stimulation and Steam Assisted Gravity Drainage) used to extract oil sands, that cannot be retrieved through open-pit mining. Attempts at peatland restoration in the oil-sands industry have mainly been focused on open pit mining (e.g. Borkenhagen and Cooper, 2016; Ketcheson et al., 2016; Nwaishi et al., 2016), rather than in-situ mining. Disturbance caused by the placement of mineral layers on peat during well-pad construction is not well studied despite Alberta's Environmental Protection and Enhancement Act (EPEA) that requires disturbed land to be restored to 'equivalent land capability' (Alberta Environment, 2017). Well-pad disturbance to peatland occurs primarily in Alberta, which has a relatively dry, continental climate, but still stores 48 Pg of carbon (C) as peat (Wind-Mulder and Vitt, 2000).

The presence of a mineral layer poses a unique obstacle to well-pad peatland restoration, setting it apart from other disturbance types. A 1-2 m thick layer of mineral fill (well-pad) is placed over the peat during the process of in-situ oil sands extraction to allow access to the production wells and drilling. Restoration methods aimed at returning the ecosystem function, disturbed by compaction of the well-pad, have included the partial removal of the well-pad (Vitt et al., 2011; Shunina, 2014). Additionally, species establishment has been successfully facilitated following partial pad removal by the application of the moss layer transfer technique (MLTT; Gauthier et al., 2017) as well as the transplanting of minerotrophic and woody species (Mowbray, 2010; Vitt et al., 2011; Shunina, 2014). There are currently no published studies on the effect of well-pad restoration on atmospheric-terrestrial C exchange or peat accumulation.

This goal of this study is to evaluate ecosystem functions related to C exchange on a restored peatland on a former oil sands well-pad.

1.2.1 Peatland carbon cycle

Peatlands are a globally important C sink, storing approximately one third of the planet's total soil C (Gorham 1991; Turunen et al. 2002). Peatlands in North America are defined as having at least 40 cm of partially-decomposed organic matter, or peat (Glaser, 1987; Vitt, 2013). Peatlands supplied with water from the surrounding area, which increases mineral inputs, are termed minerotrophic or fens, while bogs receive only water inputs from precipitation, causing ombrotrophic conditions (DuReitz, 1954). The majority of peat in a peatland is normally stored in the anoxic, permanently waterlogged catotelm. The catotelm increases in size and C content when the lowermost section of the overlying acrotelm eventually becomes saturated by the WT year-round (Ingram, 1978). The main way that C enters peatland systems is through litter inputs from plants. Vegetation takes up carbon dioxide (CO_2) from the atmosphere and fixes it to the plant material. This process of photosynthesis is fueled by and dependent on the energy from the sun. The remaining C not used for photosynthesis forms the plant structure, with most of the C making up the belowground biomass of the plant (Saarinen, 1996). After the plant dies, the majority of its C is deposited as litter on the surface of, and within, the acrotelm.

1.2.1 Controls on CO_2 exchange

Carbon dioxide uptake by plants is otherwise referred to as Gross Ecosystem Production (GEP). The contribution of both autotrophic (plant) and heterotrophic (microbial) respiration from plant roots and soil microorganisms to atmospheric CO_2 is known as Ecosystem

Respiration (ER; Clymo, 1984). Environmental conditions such as soil and air temperature, and water table position (WT) are strong controls on the Net Ecosystem Exchange (NEE), the balance between GEP and ER (Germino and Wraith,, 2003; Peichel et al., 2014; Strachan et al., 2015). Soil temperatures colder or warmer than an optimal range limit GEP of vegetation (Germino and Wraith, 2003; Harley et al. 1989). Likewise, GEP is decreased by high and low WT positions (Peichl et al. 2014). Ecosystem respiration can be strongly controlled by WT during dry conditions (Bubier et al., 2003), but has been found to be more dependent on soil temperature with higher rates under warmer conditions (Updegraff et al., 2001; Lafleur et al., 2005).

1.2.2 Controls on organic matter decomposition

Decomposition in peatlands refers to the partial mineralization of organic material by consumer organisms into H_2O, CO_2 and inorganic nutrients, primarily by bacteria and fungi (Wieder and Vitt, 2006). The decomposition process allows carbon and nutrients to cycle in the ecosystem. The key defining characteristic of peatlands lies within the balance of their production and decomposition rates. Accumulation of peat and organic matter is dependent on the annual net primary production (NPP) exceeding the decomposition rate (Clymo, 1984) resulting in organic matter accumulation. In general, conditions required for slow decomposition (e.g. low nutrient concentrations, high WT position) are in opposition with high productivity. Slow decomposition is, however, often considered a more important factor than high productivity for peat accumulation given the overall low productivity of peatlands (Clymo, 1965; Rochefort et al., 1990); however, several paleoecology studies have suggested that NPP is a more important than decomposition for determining long-term peat accumulation (Charman et al., 2013). High WT positions create anoxic conditions, slowing decomposition (Clymo, 1965). More acidic soil conditions are also associated with lower rates of decomposition (Williams et al., 2000). Low temperatures are likely the most important variable affecting rates of organic matter loss when moisture and oxygen availability do not limit decomposition rates (Brinson et al., 1981).

Nutrient status influences decomposition processes depending on the WT position and redox potential. Anoxic conditions cause reduction of terminal electron acceptor wherein several ions and compounds become more mobile such as such as manganese (Mn^{2+}) and ferrous iron (Fe^{2+}). Following these reactions, hydrogen sulfide (H_2S), and methane (CH_4) are produced (Rydin and Jeglum, 2013), with decomposition rates slowest under the most reducing conditions. The nutrient content of litter also affects decomposition rates as plants with high C/N ratios decompose more quickly (Malmer and Nihlgård, 1980). Phosphorus (P) concentration is also positively correlated to decomposition rates of organic material (Taylor, 1940). Substrate quality can also affect decomposition with lipids, crystalline cellulose and aromatic polymers having higher resistance to decomposition (Bohlin et al. 1989). Phenolic acids in *Sphagnum* species and lignin in trees make them the slowest to decompose in peatlands (Rydin & Jeglum, 2013). Sedges on the other hand, decompose more quickly (Thormann, 2001).

1.2.3 Controls on methane exchange

Methane production or methanogenesis occurs when organic matter is decomposed under highly reducing conditions and is controlled by a variety of factors including: WT, temperature, terminal electron acceptor availability, soil C quality, root exudates, plant type and salinity (Bridgham et al., 2013). The production of methane is a process carried out by heterotrophic microbes that use C as an electron donor for their metabolism and are therefore dependent on soil C quality (Yavitt and Lang, 1990; Updegraff et al., 1995). Complex polymers degraded by microbial exoenzymes become further degraded by fermenting bacteria (Drake et al., 2009) followed by a secondary fermentation to produce the end products of fermentation: acetate or H_2 and CO_2. In the final stage of methanogenesis, acetoclastic and hydrogenotrophic methenogens use the acetate or H_2 and CO_2 to produce methane (CH_4) with C as their terminal electron acceptor (TEA) (Bridgham, 2013). The fermentation end products can also be metabolised by microbial groups using inorganic TEAs (Megonigal et al., 2004). The microbes' competitiveness to use fermentation end products depends on the TEA's thermodynamic favourability, thus supressing CH_4 production until the most favorable TEAs are consumed first. Greater temperatures can also increase CH_4 fluxes by controlling methanogenesis pathways and methanogen community structure (Rooney-Varga et al., 2007).

In waterlogged soils of peatlands, predominantly the catotelm, the incomplete decomposition of organic matter creates CH_4. The methane that is not stored or oxidized can be emitted to the atmosphere (Frolking et al., 2011). It is estimated by the Intergovernmental Panel on Climate Change (IPCC) that CH_4 is 28 times more effective as a greenhouse gas than CO_2 at trapping heat in the atmosphere over a 100-year timescale (IPCC, 2013). The WT position explains the greatest amount of variation of CH_4 emissions due to methanogens' sensitivity to oxygen, causing them to only occur in anoxic zones (Williams and Crawford, 1984). When CH_4 passes through the peat profile it can be oxidized by methanotrophic bacteria, being converted to CO_2 (Anthony, 1986). Methanotrophic bacteria are dependent both on oxygen and CH_4, resulting in methane oxidation most often occurring just above the transition between the oxic and anoxic zones (Segers, 1998). Methane can bypass the oxic zones of a peat profile, however, through ebullition or through the aerenchymous tissues of specialized plants Bridgham, 2013). Aerenchyma can also allow plants to cause rhizospheric oxidation by channelling oxygen to CH_4 in the anoxic zone (Bridgham, 2013).

Methane flux increases with plant photosynthetic activity; observed in a study where CH_4 release decreased after plants' leaves were being clipped due to the lower photosynthetic area (Whiting and Chanton, 1993). Further, there is a positive correlation between the rate of CO_2 uptake and CH_4 emission on sites dominated by sedges (Waddington et al., 1996). Senesced plant litter in the soil provides substrate to derive the complex organic polymers used for methanogenesis. Root exudates from plants are believed to then fuel fast CH_4 production, enhancing peat decomposition (King et al., 2002).

1.3 Peatland restoration

Peat is used for many commercial purposes (e.g. horticultural growing medium and fuel) because of its high C content and water holding capacity, and often for agricultural use (CSPMA, 2017a,b). Following the majority of these disturbances the peatland is drained and most of the acrotelm is normally removed. The remaining layer exposed on the peatland, which was previously the catotelm, is comprised of much smaller pores and more decomposed peat, causing a reduced water-storage capacity, lower saturated hydraulic conductivity and higher water-retention capacity (McCarter and Price, 2013). In other words, although the peat can retain a high amount of water, the stored water available to vegetation is lower because the peat's capacity to transmit water is lower. Consequently, these conditions result in a much deeper WT during the growing season with more variability overall (Shantz and Price, 2006). The capillary flow forcing water up from greater depths does not meet evaporative demands at the peat surface, decreasing soil moisture and soil-water pressure (Price and Whitehead, 2001). These hydrological changes are especially detrimental to *Sphagnum* (and peatland mosses in general), which cannot withstand extended dry periods. A minimum soil-water pressure in the cutover peat is required for *Sphagnum* survival because below this level it can no longer generate the capillary forces needed to extract moisture from the cutover surface (Price and Whitehead, 2001). Restoring hydrological function post-disturbance is a crucial step towards restoration for both *Sphagnum* survival and peatland function on a whole. In peatland ecosystems where all vegetation as well as the seedbank is removed, The North American Approach is used for restoration, consisting of the Moss Layer Transfer Technique (MLTT), followed by fertilization (Quinty and Rochefort, 2003). This restoration method has proven to be a successful strategy for cutover peatlands (e.g. Rochefort and Lode, 2006; Waddington et al., 2010; Gonzalez and Rochefort, 2014). There is a body of research on a variety of disturbances to peatlands such as flooding, drainage for agriculture, forestry, and peat harvesting, and fires (e.g. Turetsky et al., 2002); however, research of peatland restoration on former oil well-pads is limited.

1.4 Study Site

The study site is an experimentally-restored well-pad on an industrially-disturbed fen in Northern Alberta, located northeast of the town of Peace River (56.397°N, 116.890° W). The decommissioned well-pad, which was never drilled, underwent three restoration peat replacement treatments (PRT) in November of 2011 in order to restore hydrological functionality and peat-accumulating characteristics (Fig. 1-1). For real industry applications of well-pad restoration, the drilling of well-pad would likely cause additional impacts that are not addressed in this study but would need to be considered for restoration. For instance, following well-head removal, the holes created from drilling would need to be filled and any hydrocarbon contamination would have to be removed.

Figure 1-1 Well-pad after peat replacement treatment (http://www.nait.ca/70709.htm).

During earthwork treatments (described in Sobze et al., 2012), careful attention was given to ensure water flow between the connection of the edge of the newly-uncovered peat and the surrounding natural peatland. All earthwork treatments involved the removal of the mineral well-pad, followed by a peat replacement treatment (PRT). The peat PRT consisted of the complete removal of the well-pad and underlying geotextile layer. The peat PRT was replicated on three sections of the site. On sections of the site where peat depth was less than 60 cm, excavators were used for ‘fluffing’ of the peat to increase its depth to achieve the target elevation of the surrounding landscape. The clay and mixed PRTs were performed in strips of 3-4 m by 110 m on sections of the site where the target elevation could not be reached after fluffing. Instead a mineral layer composed of the well-pad material was placed underneath the peat layer in order to achieve a peat depth of 40 cm and target elevation. A four-step process was used to invert the clay and peat layers in the clay PRT (Fig. 1-2).

1. The clay well-pad and geo-textile was completely removed from the strip and used either to refill a borrow pit or fill an adjacent treatment.
2. Peat was removed and placed on an adjacent strip for later use. Temporary markers were used to keep track of clay and peat depth
3. Clay was replaced back into the cavity from an adjacent strip
4. Peat from the adjacent strip is replaced on top of the clay and lightly compressed to meet target elevation.

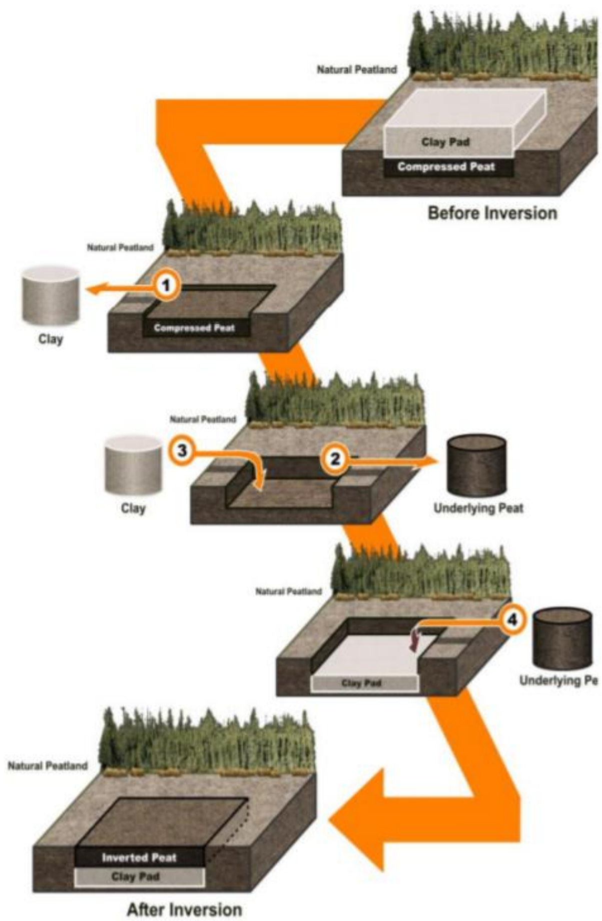


Figure 1-2 Clay PRT earthwork treatment process.
Sobze et al.. 2012.

In order to simplify and reduce time and cost for this lengthy process, a third PRT was used. The mixed PRT involved using an excavator to lift both the clay and peat layer, and flip it into an adjacent excavated strip of the site. However, breaking the geo-textile layer without mixing the clay from the well-pad and the peat layer was difficult to achieve and most often not successful. Once all earthwork was completed, the entire surface of the study site consisted of a bare peat substrate (Figure 1-1).

In July 2012, the site was revegetated using the MLTT with three nearby donor sites on cutlines for accessibility. An Argo (amphibious all-terrain vehicle) with a rototiller attachment was used to remove and fragment the top 10 cm of the moss carpet at the donor sites. After stock-piling to dry the material and reduce transport weight, a helicopter transported the collected material to the site. A manure spreader pulled by an Argo was used to distribute the material at a 1:10 ratio of moss to bare peat and to apply straw mulch afterwards. In East to West strips, the donor sites used were dominated in *Sphagnum* spp. and *Carex aquatilis* (south-most section), *Tomenthypnum nitens* and *Carex magellanica* (middle section), and *Polytrichum strictum*, *Sphagnum* spp, and *Calamagrostis canadensis* (mid-northmost section). One section of the site did not undergo MLTT and another did not undergo straw mulch application as controls. Finally, a 150 kg/ha of rock phosphate fertilizer (0-3-0) (Quinty and Rochefort, 2003) was applied across the entire site in order to promote *Polytrichum* moss growth.

1.5 Research objectives

Since natural peatlands are C sinks, restoration of well sites contributes to reducing greenhouse gases (GHG) in the atmosphere. Additionally, well-pad restoration is mandated by the Government of Alberta (Alberta Environment, 2017). According to EPEA, disturbed lands are required to be "*reclaimed and returned to an equivalent capability, which is the ability of the land to support various land uses after conservation and reclamation is similar to the ability that existed prior to an activity being conducted on the land, but that the individual land uses will not necessarily be identical"* (Alberta Environment, 2017). The most appropriate restoration method for the purposes of ecosystem function and GHG cycling similar to natural peatlands is unknown. It is known that with the replacement of peat level to the surrounding landscape, followed by planting or diaspore dispersion, peatland vegetation will regenerate (Gauthier et al., 2017; Mowbray, 2010; Vitt et al., 2011). However, it is unknown how these actions affect C gas fluxes and whether they will be comparable to reference sites. It is also unknown whether peat-accumulating functions, such as net primary production and decomposition rates, will become restored.

Therefore, the objectives of this study are to:

1. Quantify and compare CO_2 and CH_4 fluxes of PRTs to each other and to natural reference peatland,

2. Quantify and compare plant biomass, net primary productivity and litter decomposition rates of PRTs to each other and to natural reference peatlands, and

3. Provide recommendations for restoration of well pads in Alberta.

1.6 General Approach

This thesis is composed of two manuscripts which evaluate peat replacement treatments necessary for restoration of peatlands after well-pad use. Both manuscripts were written with the intent of publication. The first manuscript identified the differences of C gas fluxes between PRTs and compared them to reference peatlands. The second manuscript identified the differences of overall productivity and species-specific decomposition between PRTs and compared them to literature. I was primarily responsible for all data analysis and writing in both manuscripts as well as conducting the field work used to produce the results. Field data collection from 2014 and 2015 for GHG fluxes were collected as part of an ongoing monitoring program on the site. I expanded the sampling program in the 2016 season adding an additional replicate for C gas fluxes in targeted treatments. I also made all measurements of biomass, NPP and decomposition rates including all field and laboratory sample collection and processing. Together these manuscripts present the first complimentary C gas flux, plant production and litter decomposition study of peatland restoration on a former oil sands well-pad.

2.0 Carbon gas fluxes after oil-well pad restoration to peatland

2.1 Introduction

In-situ oil sands production has covered over 800 km² of peatland in Alberta (Vitt et al., 2011; Pasher et al., 2013). Alberta's Environmental Protection and Enhancement Act (EPEA) requires disturbed land to be restored to 'equivalent land capability', more specifically meaning the land can support similar activity that existed prior to the disturbance, but not necessarily identical (equivalent land capability; Alberta Environment, 2017). As peatlands were not included as a land type in the 2010 Reclamation Criteria for Well Sites, their land type could be changed to forested, grassland or cultivated, leading many to be restored to upland (Alberta Environment 2010). However, as there is now wellsite specific reclamation criteria for peatlands (Alberta Environment, 2017). The boreal forest region, where much of the oil and gas exploration and development has occurred in Canada, covers over half of Alberta and is dominated by peatlands, which can cover 50-100% of the land area (Gorham, 1991; Wieder and Vitt, 2006). Northern peatlands account for over half of the total peatland soil C stock, a globally significant amount of C (Page et al., 2011; Loisel et al., 2014). This C storage results from taking it up more CO_2 through gross ecosystem production (GEP) than is released as CO_2 through ecosystem respiration (ER), as methane (CH_4) or as waterborne C.

Infrastructure associated with the oil sands in-situ extraction such as access roads, seismic lines, power lines pipelines and well-pads have had serious impacts on wetlands in northern Alberta (Turchenek, 1990; Forest, 2001). A well-pad made of mineral fill, often over 1.5 m thick and 1-4 hectares in size, must be placed on in-situ extraction sites to provide a solid foundation (Graf, 2009). This practice causes major disturbance when placed on top of peatlands due to the compaction of underlying peat, and causes a hydrological disconnect from the surrounding area due to the elevation of the pad (Graf, 2009).

Due to ambiguity and overlap between the terms reclamation and restoration, the two terms are summarized as follows. Reclamation means to convert a land that has been rendered no longer valuable into a condition that is productive for human purposes, e.g. agriculture (Clewell and Aronson, 2013). The term is, however, sometimes used for the creation of wetlands in industrial context for the purposes of water storage and C credits and trading, as is the case with the Government of Alberta who describes reclamation objectives in the long term to be C sequestration, water storage/filtration and wildlife habitat (Alberta Environment, 2017). Similar to the understanding of reclamation, Clewell and Aronson, (2013, p.203) describe restoration of natural capital as "the replenishment of natural capital stocks in the interests of long-term human well-being and ecosystem health." Although it is acknowledged that the land change described in this study satisfies the definitions of both terms reclamation and restoration, the term restoration will be used due to the site's re-establishment as a peatland, including MLTT which aligns closely with the North American peatland restoration method described in Andersen et al. (2013).

While there has been little research on the restoration of well-pads to peatlands, there has been a large amount of research on restoration following other industrial peatland uses such as for peat extraction and agriculture (e.g., Lamers et al., 2014; Waddington et al., 2010). This research has shown that a practice called the Moss Layer Transfer Technique (MLTT) can lower CO_2 fluxes and can return restored peatlands to C sinks (e.g. Waddington et al., 2010; Strack et al., 2014), although rates of C exchange may remain dissimilar to undisturbed sites up to at least 15 years post-restoration, as observed in a study by Strack et al. (2016) that measured a peatland up to 15 years after restoration. The MLTT involves blocking drainage ditches, spreading diaspores (plant propagules) from donor peatlands, covering the area with straw mulch, and often fertilization with rock phosphate.

Overall, the main finding from restoration studies of peatlands on well-pads is that lowering the elevated pad to the surrounding water level and revegetating sites using minerotrophic communities is a viable restoration strategy (Gauthier et al., 2017; Shunina, 2014; Vitt et al., 2011). In another study, a restored well-pad was found to be a significantly greater sink of NEE CO_2 than natural plots which were sources of CO_2 (Strack et al., 2016).

The net CO_2 exchange of peatlands has been found to react strongly to environmental conditions, such as temperature and WT (Strachan et al., 2015; Peichel et al., 2014; Germino and Wraith, 2003), and these factors can thus help in understanding the effects of disturbance, and restoration on C cycling of restored peatlands on well-pads. Suboptimal and supraoptimal soil temperatures have both been shown to reduce GEP of vegetation (Harley et al. 1989; Germino and Wraith, 2003). Similarly, both shallow and deep WT position have been linked to a decrease in GEP (Peichl et al. 2014). While WT position has also been correlated to ER (Updegraff et al., 2001), most notably during dry seasons (Bubier et al., 2003), ER has largely been found to be more dependent on soil temperature (Updegraff et al., 2001; Lafleur et al., 2005). Water table levels that are too low also promote upland weedy species growth, while too high promotes invasion of marsh plants such as *Typha* spp. (cattails), outcompeting native species (Govermment of Alberta, 2017). Therefore, a WT level of 2-8 cm should be reached for restoration, as recommended by the Government of Alberta (2017), with annual fluctuation no greater than 30 cm as high fluctuations promotes decomposition. Despite the reality of environmental conditions changing from year to year, there are no studies tracking C exchange following well-pad restoration over more than one growing season.

In this study, three experimental well-pad removal strategies were undertaken on a well-pad situated within a forested fen with the intention of restoring C gas balance similar to undisturbed reference sites. The objectives of this study were to: 1) quantify plot-scale seasonal CO_2 and CH_4 exchange and compare between various well-pad restoration treatments and natural reference sites two to four years post- restoration, and 2) determine how moss and vascular plant cover, soil temperature, and WT position influence CO_2 and CH_4 fluxes, and how these controls vary between restored and reference sites. It is hypothesized that WT will be further below the surface on the restored well-pad, resulting in greater cover of vascular than non-vascular vegetation as well as higher T_5 than the reference sites. It is predicted that the site's controls on C exchange will become more similar to reference peatlands over time causing C gas exchange to decrease in amount of C emitted but will remain significantly different from undisturbed peatlands four year post-restoration.

2.2 Study sites and design

A decommissioned, never-drilled well-pad in Northern Alberta, located north-east of the town of Peace River (56.397°N, 116.890° W) underwent experimental restoration via several peat replacement treatments (PRT) in November 2011. At least 40 cm of peat was replaced as part of all restoration methods on sections of the 1.4 ha well-pad (Fig. 2-1). All treatments were leveled to 10 cm below adjacent hollows, either by placing mineral soil fill from the well-pad under peat or through peat de-compaction, with the idea that further expansion of the peat following restoration would result in surface level similar to the adjacent undisturbed fen. Using an excavator, the mineral well-pad and peat layer were individually excavated in two treatments. Only the peat layer was replaced in the peat PRT (peat) following de-compaction. In the second treatment, the pad was detached from its underlying geotextile layer, then a portion replaced under the peat to achieve the desired elevation (clay). In an attempt to increase efficiency of labour, a third treatment involved flipping the clay and peat layer in one motion, resulting in some mixing of the clay and geotextile layer with the peat at variable depths (mixed).

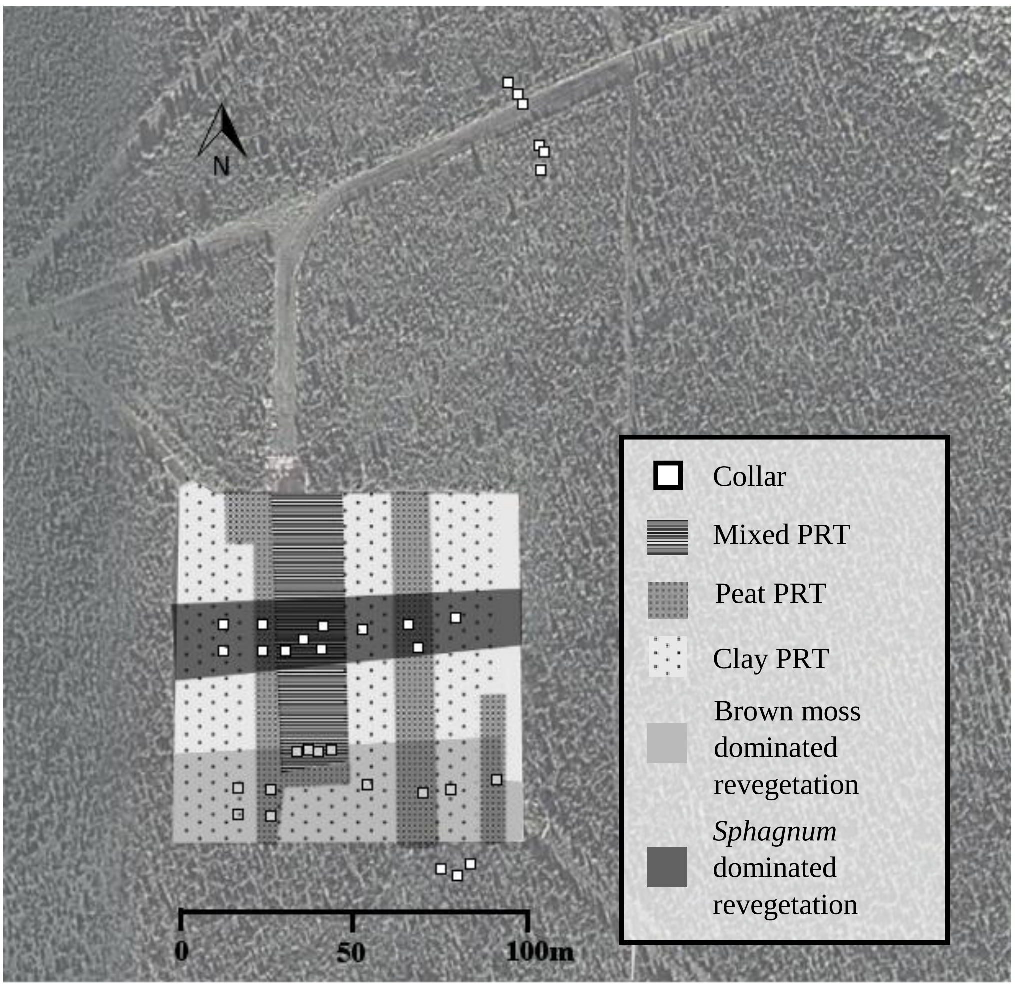


Figure 2-1 Site map of restored peatland following well-pad replacement treatments (PRT) and revegetation treatments (Google Earth, 2017).

All PRTs measured in this study were systematically revegetated following the MLTT (Quinty and Rochefort, 2003) with phosphorus fertilization, using diaspores collected from two nearby peatland donor sites. The material collected from a donor site dominated in brown moss was spread over the south end of the site, and material collected from a donor site dominated in *Sphagnum* moss was spread over the middle section of the site. Both donor sites had species other than their dominant vegetation. Additionally, the middle of the site was revegetated with material collected from a donor site dominated in *Polytrichum* mosses but was not included in the present study.

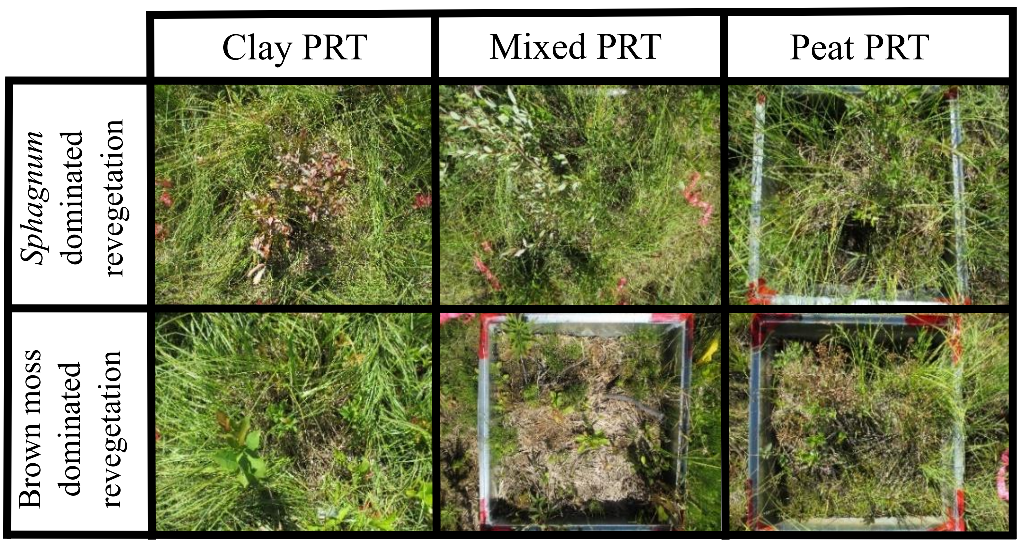


Figure 2-2 Restored well-pad vegetation in 2016. Columns show vegetation at collars at each peat replacement treatment (PRT) and rows are of the same collars' revegetation treatments.

Two reference sites on natural peatlands were chosen (natural) for comparison of both hummocks and hollows. In 2014 and 2015, six plots were measured in a reference site roughly 50 meters north east of the well pad, in a treed fen. Directly south of the well pad a reference site in a poor treed fen was chosen and measured in 2016. Three plots were placed on every combination of revegetation and peat replacement treatment with an additional (Figure 2-2) replicate added for the 2016 field season for a total of 24 collars in 2014-2015 and 30 in 2016 .

All collars were measured for CO_2 and CH_4 from May to September every 1 to 2 weeks. Here we focus on evaluating the PRTs and group measurements from the revegetation treatments together. We tested for an effect of different donor sites on differences in vegetation cover and CO_2 exchange but no differences were found. Therefore, comparison of the two revegetation treatments were not focused on in this study in order to narrow scope[MS1].

2.3 Methods

2.3.1 Carbon Exchange

Using the closed chamber technique, CO_2 fluxes were measured in a 60 cm x 60 cm x 30 cm chamber using a portable infrared gas analyzer (EGM-4 PP Systems, Massachusetts, USA) every other week at random times between 9 a.m. and 4 p.m.. Chambers were placed on 60 cm x 60 cm collars installed at each plot that had a groove that was filled with water to create a gas-tight seal. A fan located inside the chamber powered by a battery was used to mix the air in the headspace. The change of concentration of CO_2 in the chamber over 1 minute and 45 seconds, recorded every 15 seconds indicated the flux that was calculated from the linear change in concentration over time. The CO_2 concentration measured was corrected for the volume of the chamber and ambient temperatures, measured with a thermocouple. Photosynthetically active radiation (PAR) was measured using a sensor located in the chamber. A series of shades was used to alter PAR and measurements repeated, with an opaque cover used to block all PAR for the measurement of ecosystem respiration (ER). Net ecosystem exchange flux measurements of PAR>800 µmol m^{-2} s^{-1} were used to calculate GEP (NEE minus ER) under full light conditions.

Negative CO_2 measurements indicated an uptake of C by the peatland, while positive measurements indicated a release of CO_2 to the atmosphere.

Methane was measured in a similar manner to CO_2 but with opaque, reflective chambers to keep temperatures from getting too high. The headspace was extracted from the chamber at 7, 15, 25 and 35 minutes after closure using a syringe. Each 20 mL gas sample was transferred into pre-evacuated exetainer (Labco Ltd., UK) and analyzed on a gas chromatograph (Shimadzu GC2014) equipped with a flame ionization detector. Ambient gas samples were also taken (used as time 0 min) and used to determine CH_4 flux (mg CH_4 m^{-2} d^{-1}) from linear change in concentration over time. Concentration changes over the closure time below the range of precision of the GC were considered zero (concentration changes < 10% of the first value). Non-linear and erratic changes in concentration and were also omitted such as fluxes $\leq$ -5.5 with an r^2 value <0.8 resulting in a loss of <20% of data (Murray et al., 2017a).

2.3.3 Environmental conditions

Water table position (WT) and soil temperature were measured at each collar during each CO_2 flux measurement. Manual measurements were taken during all chamber flux measurements of WT position in a well installed adjacent to each plot and soil temperature (T_5) at 5 cm (thermocouple). Vascular and non-vascular vegetation cover was estimated visually during a vegetation survey conducted on all of the collars in August of 2014 and 2016, as well as late July of 2015. Soil samples at 2 cm depth, roughly 100 cm^2 were collected by hand, in a grid pattern over the site, four times over the 2015 and 2016 growing season. A slurry of one part soil and three parts deionized water was made from each sample and measured for pH and electric conductivity (EC) (Thermo Scientific Orion pH meter and Orion Versa Star Advanced Electrochemistry meter). Samples were collected from the natural reference site for pH and EC in 2016 only.

Soil temperature and PAR on the site were measured at 1 hour intervals from June - September and logged on a Decagon Devices EM50 logger by a GS3 soil probe and PYR probe, respectively, placed in the centre of the well-pad. The PAR data from the logger was regressed with the sensor used during flux measurements to calibrate with the PAR recorded in the chambers. Soil temperature used for CO_2 balance modeling of the natural site was not measured; soil temperature measured on the peat PRT was regressed with manual measurements recorded at the natural site to generate a half-hourly temperature record for the natural site.

2.3.2 CO_2 balance modelling

The seasonal GEP for each PRT in 2016 (May to October) was calculated according to the equation by Thornley & Johnson, 1990).

$$GEP = \frac{PAR \times Q \times GPmax}{PAR \times Q + GPmax}$$

where Q is the quantum efficiency and represents the slope of the rectangular hyperbola, and GPmax is a theoretical maximum rate of GEP and represents the asymptote of the hyperbola. Ecosystem respiration was calculated according to an equation by Taylor and Lloyd (1994) where Rref is the CO_2 release (g m^2 d^2) from respiration at the natural site temperature Tref (283.5 K), E0 is the activation energy (K), T0 is the temperature when biological activity ceases

(237.48 K) and T is the temperature during the measurements at 5 cm depth during the measurement.

$$ER = Rref \times e^{\left(Eo \times \left(\frac{1}{Tref - To} - \frac{1}{T - To}\right)\right)}$$

Net ecosystem exchange was calculated by adding modelled GEP and ER. Model errors were calculated according to Adkinson et al., (2011) based on differences between modeled and observed NEE values. Seasonal CH_4 emissions were calculated by multiplying the mean CH_4 emission per treatment per day by the number of days in the growing season along with standard error.

2.3.4 Data analysis

All statistical analyses were conducted using R software (Version 2.6.1; R Development Core Team 2006). All data was tested for normal distribution using quantile-quantile plots and histograms. To evaluate controls on C fluxes, the nlme package (Pinheiro et al. 2011) was used to perform linear mixed effects modelling with plot as a random factor to account for repeated measures. The best model for GEP, NEE, ER and CH_4 was chosen by means of a method similar to stepwise selection. Starting at complicated models involving all fixed factors and two-way interactions (WT, T_5, Year and PRT), the factor with the highest, non-significant p-values were removed for the new model. This process was repeated until no interaction term had a significant p-value. Only significant main effects and interaction terms were included in results, with the exception of non-significant fixed effects that must remain in the model if they are part of a significant interaction term. Significance was indicated by a p-value < 0.05 with the exception of when more than one treatment was considered in which it was adjusted for number of treatments using the Bonferroni correction. The amount of variance described by each model as R^2GLMM, as defined by Nakagawa & Schielzeth (2013) was determined using the package MuMIn (Barton 2015). Tukey's HSD was used to compare fluxes, T_5 and WT between site types. Plant cover was measured one time at the end of the season and was therefore not included in a regression with collar means of observed NEE.

2.4 Results

2.4.1 General site conditions

Seasonal temperature from June to September at the site was slightly warmer in 2014 than 2015 and 2016 (15.1, 13.6 and 13.4 °C). The years 2014-2016 are the second, third and fourth year following restoration and will hereafter be referred to this way. The long term 30 year normal from a weather station located in Peace River from 1981 to 2010 showed a mean temperature of 13.8 °C and mean precipitation of 214.4 mm (Environment and Natural Resources, 2015) for the same June – September period. On the site, during the first study year, total precipitation was much lower than the normal but increased annually (87.1, 141.4 and 182.6 mm). WT position varied between years as well as treatments, becoming closer to the surface and more consistent across the site in the fourth year than in previous years (Table 1). In all years, the WT position at the natural site was significantly closer to the surface than any PRT (Table 1). In the second and third year, the mixed WT position was significantly further from the surface than the clay and peat PRTs. In the fourth year, mixed was only significantly further from the surface than clay. Peat and clay PRTs were always statistically similar; however, the WT in the peat PRT was always slightly shallower than clay (Table 1).

Table 2-1 Average (standard error). Different letters indicate statistical significance. Electrical conductivity and pH were measured 2 cm below the surface.

Year post reclamation	Site	WT (cm)	T_5 (°C)	EC (µS cm^{-1})	pH	Vascular plant cover (%)	Moss cover (%)
Year 2	Clay	-15.5 (1.6) a	18.9 (0.5)	-	-	17 (4) a	12 (3) a
	Mixed	-23.8 (2.1) b	19.9 (0.8)	-	-	8 (3) a	8 (3) a
	Peat	-17.4 (1.8) a	18.1 (0.5)	-	-	16 (3) a	6 (2) a
	Natural	-8.0 (0.9) c	15.2 (1.0)	-	-	42 (2) b	93 (2) b
Year 3	Clay	-30.0 (4.4) a	21.2 (1.1) a	1515.0 (1022.8) b	6.4 (1.2)	84 (10)	9 (1)
	Mixed	-41.5 (2.4) b	22.8 (0.9) a	1596.4 (522.0) b	6.5 (1.5)	47 (15)	8 (1)
	Peat	-32.1 (2.3) a	22.4 (0.9) a	891.4 (701.7) a	6.4 (1.5)	63 (14)	12 (3)
	Natural	-14.8 (1.0) c	17.3 (1.0) b	-	-	-	-
Year 4	Clay	-7.4 (1.0) a	14.0 (0.7) a	346.3 (703.2)	5.6 (0.6)	73 (8) a	45 (7)
	Mixed	-12.1 (1.4) b	14.9 (0.8) a	626.6 (462.7)	5.6 (0.6)	53 (1) a	35 (7)
	Peat	-9.8 (1.6) a b	14.5 (0.8) a	384.1 (271.2)	5.5 (0.6)	59 (9) a	39 (14)
	Natural	-1.9 (1.0) c	10.1 (1.2) b	170.6 (315)	5.8 (0.9)	42 (6) b	98 (2)

The mean pH of all three PRTs fell between 6.3 and 6.5 in the third year (Table 1). In the fourth year, the pH dropped on the restored site to fall between 5.5 and 5.8, similar to the natural site at 5.5. Standard deviation of pH was ±0.6 for all PRTs and 0.9 for the natural site in the fourth year. In the third year, standard deviation was ±1.5 for mixed and peat PRTs and ±1.2 for clay. Average EC on the restored pad in the third year was 1515.0 ± 1022.8, 1596.4 ± 522.0 and 891.4 ±701.7 µS cm^{-1}, respectively for clay, mixed and peat PRTs. The peat PRT had significantly lower EC than mixed and clay PRTs despite a high standard deviation (Table 1). In the fourth year, overall the EC on the site was lower across all treatments (Table 2-1). The EC at the natural site was 170 µS ± 315 µS cm^{-1}, considerably lower than on the restored well-pad, but not significantly lower due to the high standard deviation at the natural site.

Vascular vegetation cover generally increased from year three to four (Table 2-1) and was not collected in the third year at the natural site. In the second year, the natural site had a significantly greater vascular plant cover than all restoration treatments, but by year four the cover on the restored site increased enough to surpass the natural site and was no longer significantly different. Non-vascular vegetation (bryophyte) cover was significantly higher at the natural site than on the restored site for all years (Table 1). Total moss cover on the restored site was similar among treatments and below 20% in the second and third year, increasing to over 30% in the fourth year.

2.4.2 Carbon dioxide exchange

In general, both the restored and natural site took up greater amounts of net CO_2 in the fourth year than in the second year (Fig. 2-3). Between treatments, the rates of ER, GEP and NEE were similar in all years except in the fourth year when the mixed PRT took up significantly more CO_2 (Fig. 2-3). The ER ranged from 1.3 - 16.2 g CO_2 m^{-2} d^{-1} on the PRTs and between 0.3 - 69.4 g CO_2 m^{-2} d^{-1} on the natural site between the second and fourth year. Ecosystem respiration was similar between PRTs and natural sites, emitting slightly less CO_2 in the fourth year than previous years. In the third year, the clay and peat PRTs took up significantly more CO_2 as NEE than the natural site, while in the fourth year the NEE was similar at all locations. Net ecosystem exchange ranged from -38.3 – 9.3 CO_2 m^{-2} d^{-1} on the PRTs and between -35.1 and 30.0 g CO_2 m^{-2} d^{-1} on the natural site between the second and fourth year (greater negative indicating greater CO_2 sink). Both clay and peat PRTs took up significantly less CO_2 as GEP than the mixed PRT and natural sites (mean of -16.9, -12.1, -35.9 and -28.7 g CO_2 m^{-2} d^{-1}, respectively) . In the fourth year, despite there being more vascular vegetation, the rate that the site took up CO_2 was lower than in the previous year (Table 2-1 and Fig. 2-3).

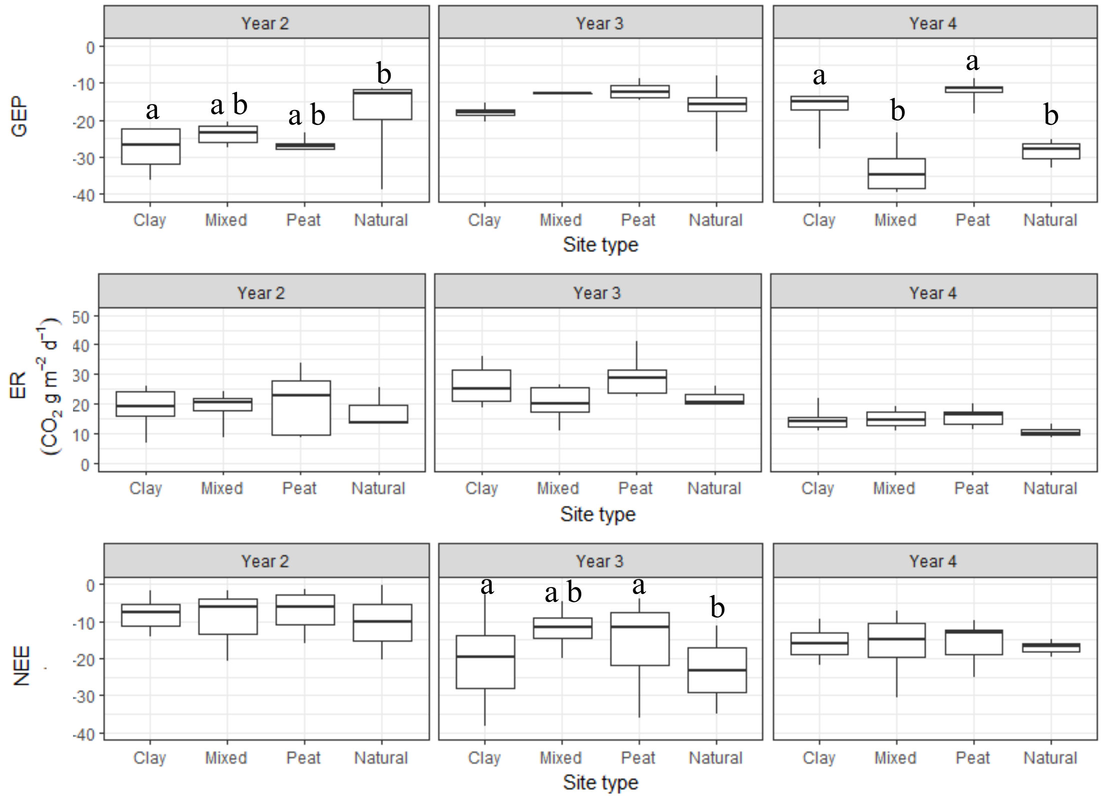


Figure 2-3 Comparison of seasonal mean GEP, ER and NEE between restoration types and reference sites for 2-4 years post restoration. Different letters indicate significant difference between site types within he respective year. No letters indicates no significant difference.

Table 2-2 Seasonal C model fluxes of 2016 (g m^{-2})

	ER	**GEP**	**NEE**	**CH_4**
Clay	487.28 ±32.32	-442.75 ±51.34	44.54 ±14.34	8.32 ±2.82
Mixed	500.91 ±31.50	-427.18 ±47.96	73.73 ±89.82	14.31 ±7.05
Peat	461.39 ±32.02	-409.92 ±50.57	51.47 ±379.83	19.25 ±5.64
Natural	457.11 ±21.30	-402.89 ±47.62	54.23 ±21.31	6.13 ±2.82

The results of the seasonal CO_2 balance modelled in the fourth year showed that the restored and natural sites were both net sources of CO_2 (Table 2-2). The natural site and peat PRT had the most similar ER, GEP and NEE, (457.11, -402.89, 54.23 g C m^{-2} and 461.39, -409.92, 51.47 g C m^{-2}) with slightly higher NEE than the clay PRT (44.54 g of C m^{-2}). The highest NEE was at the mixed PRT (73.73 g of C m^{-2}), which had similar rates of GEP (-427.18 g m^{-2} C m^{-2}) but higher ER than the clay and peat PRTs, as well as the natural site (500.91 g C m^{-2}). Among all treatments, the seasonal predicted GEP varied more than ER.

2.4.3 Methane flux

The rate of CH_4 emissions was higher, but not significantly different, on the restored site than in natural sites for all three years, and was similar among treatments (Fig. 2-4). Overall, CH_4 emissions dropped from the second year to fourth year. The lowest seasonal estimate was at the natural site (6.13 g C m^{-2}) followed by the clay (8.32 g C m^{-2}), mixed (14.31 g C m^{-2}) and peat (19.25 g C m^{-2}) treatments.

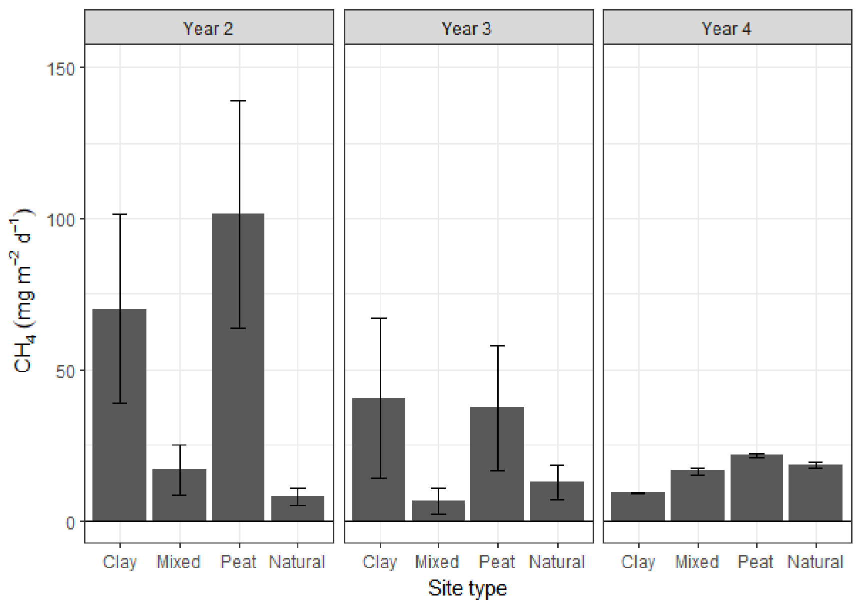


Figure 2-4 Comparison of methane fluxes between reclamation states of the third, fourth and fifth growing seasons post reclamation.

2.4.4 Controls on rate of CO_2 and CH_4 exchange

There was a negative correlation between CO_2 flux as NEE and vascular vegetation cover for all years (i.e., higher vegetation cover resulted in greater CO_2 uptake; Fig. 2-5). The rate of CO_2 uptake increased after the second year. In the fourth year, despite there being more vascular vegetation, the rate that the site took up CO_2 was lower than in the previous year with less vegetation cover (Fig. 2-3, 2-5).

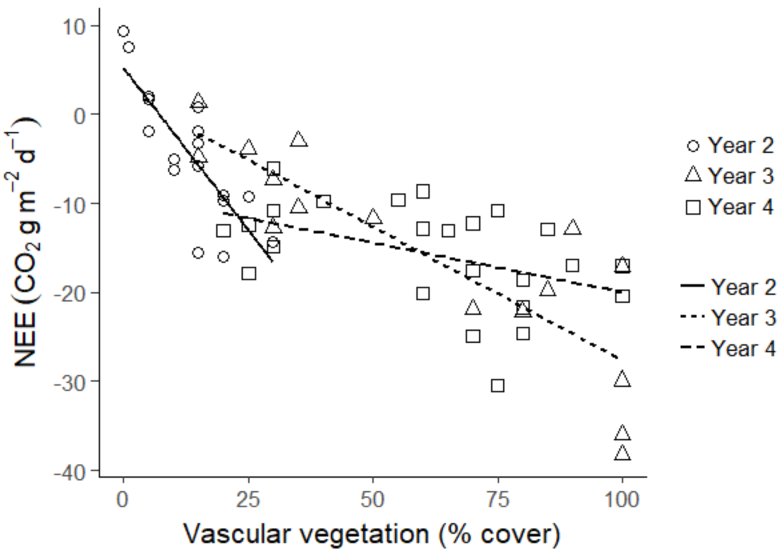


Figure 2-5 Regression of NEE and vascular vegetation cover from third, fourth and fifth year post reclamation. Regression equations are A) Year 2: y=-0.9278x+9.1242, r^2 =0.65, B) Year 3: y=-2.4437x+21.7123, r^2=0.71, C) Year 4: y=-1.9275x+31.677.

Using a linear mixed effects model with the fixed effects of WT, T_5, year and PRT, results varied between GEP, ER, NEE and CH_4 (Table 2-3). The fixed effect of site type (restored vs. natural) was initially included in the model; however, since there were no significant differences between the restored site and natural sites, only the restored site was included in subsequent modelling. On both the restored and natural sites, T_5 had a significant positive effect on ER. At only the restored site there was a significant three-way interaction with PRT-WT-Year as well as a significant interaction of PRTs with WT and T_5 separately. At the restored site, the mixed PRT had higher ER with increasing temperatures. The ER was also higher at mixed when WT position was closer to the surface. This relationship occurred at a significantly lower rate than at clay and peat PRTs. From the second to fourth year, the PRT type influenced the effect of WT position on ER. The mixed PRT emitted significantly less CO_2 at

WT position closer to the surface in the second year than the other PRTs, while the peat PRT emitted significantly more in the third year. In the fourth year, however, there was no significant difference in the WT-ER relationships between the PRTs.

Table 2-3 Statistical results of linear mixed effects models on restored well-pad

Flux Component	Effect	F	p-value	R^2 GLMM
ER	PRT	$F_{2,33} = 2.80$	0.0756	0.55
	T_5	$F_{1,442} = 31.44$	<.0001	
	WT	$F_{1,442} = 0.59$	0.4444	
	Year	$F_{2,442} = 3.55$	0.0295	
	PRT x WT	$F_{2,442} = 2.87$	0.058	
	PRT x T_5	$F_{2,442} = 7.95$	0.0004	
	PRT x Year	$F_{4,442} = 2.24$	0.0636	
	WT x Year	$F_{2,442} = 0.53$	0.5884	
	Year x PRT x WT	$F_{4,442} = 3.55$	0.0073	
	Intercept	$F_{1,442} = 0.35$	0.5561	
GEP	PRT	$F_{2,19} = 4.37$	0.0275	0.58
	T_5	$F_{1,338} = 63.04$	<.0001	
	WT	$F_{1,338} = 29.84$	<.0001	
	Year	$F_{1,338} = 5.93$	0.0154	
	Year x PRT	$F_{2,338} = 4.37$	0.0134	
	Intercept	$F_{1,338} = 5.91$	0.0155	
NEE	PRT	$F_{2,33} = 2.30$	0.1158	0.56
	T_5	$F_{1,456} = 84.76$	<.0001	
	WT	$F_{1,456} = 33.14$	<.0001	
	Year	$F_{1,456} = 16.89$	<.0001	
	T_5 x PRT	$F_{2,456} = 8.97$	0.0002	
	Intercept	$F_{1,456} = 16.91$	<.0001	
CH_4	Year	$F_{1,226} = 20.57$	<.0001	0.34
	WT	$F_{1,226} = 35.76$	<.0001	
	WT x Year	$F_{1,226} = 30.57$	<.0001	
	Intercept	$F_{1,226} = 30.47$	<.0001	

Restoration state was included in lme but not included in this table as it was not statistically significant

On the restored site T_5, WT and the PRT-T_5 interaction were all significant factors explaining variation in NEE. The mixed PRT emitted CO_2 with increasing temperature at a significantly lower rate than at the clay and peat PRTs. Water table position further below the ground surface resulted in more CO_2 emitted. Neither WT nor T_5 on their own were significant factors for NEE at the natural site but the interaction between the two was. At WT positions further below the surface more CO_2 was emitted with increasing temperatures, whereas at positions closer to the surface, less CO_2 was emitted with increasing temperatures.

WT position and had a significant positive effect, and T_5 negative on GEP in the restored site plots. The year also had a significant effect on uptake and this interacted with the PRT. Clay and peat PRT uptake of CO_2 decreased over the years, while mixed PRT uptake was much greater in the fourth year than the second year. At the natural site, only T_5 was a significant factor with greater CO_2 uptake at higher temperatures.

The fixed effects WT and year were included in models to evaluate variation in CH_4 fluxes on the restored and natural site, with separate models for each site type. Both WT and year were significant factors explaining variation in CH_4 emissions on the restored site. At the natural sites, year was the only significant factor explaining variation in CH_4 flux (Table 2-3). This result is confounded; however, due to a different location of natural site from year three to four.

2.5 Discussion

In the second to fourth year, CO_2 and CH_4 flux was quantified on three different well-pad to peatland restoration treatments and compared to a natural site. All restoration treatments (PRTs) re-established vegetation dominated by sedges, increasing both vascular vegetation cover and productivity with time. We observed no significant difference in the C gas exchange between the three peat replacement treatments used. Similar to *Wetlands Supplement* (IPCC, 2014) and in contrast to Strack et al. (2016), most restoration treatments had statistically similar

CO_2 fluxes to those from a natural site. The same was true for CH_4; CH_4 flux at the natural site was lower but not statistical different between PRTs and natural sites in any year of study. In contrast, Strack et al. (2016) reported higher CH_4 emissions from natural sites compared to restored peatlands.

Under full light conditions, mean GEP across all years ranged from -16 to -25 g CO_2 m^{-2} d^{-1}, falling within the lower half of the range of those observed in another restored boreal peatland, also restored using the MLTT (around -3 to -50 g CO_2 m^{-2} d^{-1}; Strack et al., 2014). The mean NEE for the clay, mixed and peat PRTs was -9 to -14 CO_2 m^{-2} d^{-1}, again in the lower range of those observed by Strack et al. (2014). The CH_4 flux during the sampling season ranged from 9.4 and 150.0 mg CH_4 m^{-2} d^{-1}, spanning less than half the lower range of those seen at wet and dry restored sites from Strack et al. (2014) (−1.77 and 394.68 mg CH_4 m^{-2} d^{-1}).

WT position was significantly deeper at the natural site than the restored site in every year, with the exception of the clay PRT being similar to the natural site in the fourth year only. The remnant clay material likely prevented water from percolating to the peat below it in the fourth year, which had more precipitation than the previous two years. A similar pattern would be expected also at the mixed PRT; however, differences may be expected due to the construction of the treatment. The mineral-fill layer was broken up rather than flipped or replaced in one piece like the clay and peat PRTs. This inconsistency in the clay layer is likely to allow percolation through the remnant fill. Moreover, unlike the other PRTs, mixed only occurs in the centre of the pad due to logistical constraints during the restoration work and it is therefore difficult to separate conditions related to the PRT itself from location on the pad. The centre of the site is likely drier and at slightly higher elevation than the edges of the pad, which were leveled to surrounding hollows. Testing the mixed PRT on larger areas and near the edges of pads in the future is advised to better evaluate the effectiveness of this method.

An increase of WT roughly 20 cm closer to the surface was seen in the fourth year, while less change was seen between the second and third year. Although a WT position less than 40 cm beneath the surface has been recognized as a restoration target due to the higher success rate of *Sphagnum* moss reestablishment under these conditions (Price and Whitehead, 2001), the increase seen between the third and fourth year is likely due to the greater amount of precipitation in the fourth year, which was 37.6 mm more than the previous year. The moss cover on the site jumped from an average of 9% in the third year to 40% in the fourth year, likely responding to this increase in WT from precipitation. Little change in peat and clay PRTs' C uptake was observed between the third and fourth year, but corresponding with this increase in WT at all sites was a considerable increase in uptake as GEP by the mixed PRT. This finding is concurrent with Sulman et al., (2010) which has found WT positions closer to the surface to result greater peatland GEP across several natural peatlands.

The greater precipitation and subsequent rise in WT position closer to the surface in the fourth year likely caused the drop in EC observed in this year. The hydrogen ions and organic acids produced during the second and third, drier years, likely also caused a reduction in pH in the fourth year when the soil became saturated from the precipitation. The expected cause for the discrepancies in EC can be seen most clearly in the third year, where the two PRTs containing residual clay (clay and mixed PRTs) have much higher EC than peat (Table 2-1). The two PRTs with the highest moss cover (peat and clay PRTs), had only slightly lower EC than mixed in the fourth year (Table 1). The mixed PRT had the highest EC and lowest moss cover in both the third and fourth year aligning with research that has found lower EC in bogs dominated with

Sphagnum species (Vitt et al., 1995). In the third year, the peat PRT had both the highest moss cover and the lowest EC suggesting it may represent the optimal conditions for *Sphagnum* establishment.

While the focus of the present study was C exchange, there are several other indicators besides CH_4 and CO_2 uptake that can be used to assess peatland function (e.g., Nwaishi et al., 2015). In particular, vegetation is a widely used indicator of peatland restoration outcome. The increase in CO_2 uptake from the second to fourth year at the restored site can be explained by the increase in vegetation cover. Leppälä et al., (2008) found that a shift in chronosequence from younger, more sedge- and herb-dominated, to older sedge- and *Sphagnum*-dominated peatlands, caused lower seasonal variation of CO_2 flux. The non-vascular vegetation cover on the restored site has increased in the fourth year by roughly three times since the second year. This vegetation switch, particularly to *Sphagnum*-dominated communities is not only important for restoring peatland CO_2 flux, but also for restoring C-accumulating abilities by switching to peatland plant groups which have slower decomposition rates (e.g. Johnson & Damman, 1993; Hogg, 1993). The vegetation cover in the second year on the site was slightly greater than but comparable to a study by Gauthier et al. (2017) who found up to 14% vascular cover and 6% bryophyte cover 12 months after treatment installation. On the site there was much greater cover three years post restoration; vascular plants and moss covered 62 and 40%. The González and Rochefort (2014) study on Eastern Canada sites restored with MLTT 3-4 years post restoration which found between 35 and 40% cover of bryophytes. This suggests that the PRTs used, in combination with MLTT, are effective at restoring peatland plant cover on former oil well-pads. The reliability of plant cover estimates, however, is dependent on the consistency of visual estimates which were collected by different researchers every year. In order to increase reliability, it is recommended to take the average of multiple visual assessments of the same collar and to have a consistent researcher collect the data.

Overall on the restored site, despite increasing WT position through the years, CH_4 emissions decreased from the second to fourth year, an unexpected result considering CH_4 production occurs under anoxic conditions (Conrad, 1989). This occurrence was also highlighted in relationship between the seasonal CH_4 in the fourth year and the WT position. Seasonal CH_4 was highest at the peat PRT followed by mixed and clay PRTs, while mean WT was closest to the surface at the clay PRT, followed by mixed and peat. It is possible that a greater rate of CH_4 oxidation could have occurred in the fourth year. The non-vascular cover increased from the second to fourth year allowing for a greater oxic zone following an increase of *Sphagnum* and size of acrotelm, which can oxidize CH_4 (Zhao et al., 2016). Similarly, the increasing vascular plant community dominated by sedges may have increased the aeration of the root zone, also increasing CH_4 oxidation. The decline in EC in the fourth year is another possible explanation for the decrease in CH_4 flux. Electron acceptors could have been produced during drier conditions in the first and second year, and later reduced prior to CH_4 production in the fourth year. Higher EC in the third year may be evidence of the availability of these electron acceptors in the fourth year, but further research of water chemistry is recommended. Additionally, the decrease in CH_4 flux in the fourth year could have been partially caused by the drop in pH, which has been found to inhibit CH_4 production (Dunfield et al., 1993).

It is likely that the microbial controls occurring on CH_4 production and oxidation on the site were not able to be simplified by using WT and/or T_5 as an indicator considering neither of these two factors were significant controls at the natural site. A study by Wang et al. (2016) has shown that the presence of mineral soil layers in peatlands caused different C mineralization among different soil types, including sedge peat over moss peat profiles, and that recalcitrant C, labile C, bacteria:fungi, and microbial physiological stress were greatest in peat above mineral sediments. This may be evidence that remnant clay material provides additional terminal electron acceptors, altering pathways of C exchange from those in the natural site. Additional research on chemical and microbial controls on C exchange in the presence of mineral soil fill in peatlands is recommended to better understand return of C cycling on restored peatland well-pads. Despite these complications, there is some evidence that deeper WT resulted in lower CH_4 emissions. For example, in the second and third year when the mixed PRT had the greatest difference in WT from peat and clay PRTs, there was also lower CH_4 emissions recorded.

The clay PRT had greatest ER and lowest GEP possibly due to having the most vascular cover. This also resulted in lowest NEE among PRTs. Given the restoration objective of lowering CO_2 emissions, it would make sense to target vascular vegetation cover; however, this is not congruent with the processes occurring within the natural site where there is strong uptake of CO_2 as NEE with a lower vascular vegetation cover than any other treatment. It is recommended that future studies include not only vegetation cover but also consider species when evaluating restoration and comparing to undisturbed sites as Shunina (2014) found a Jaccard index and Bray Curtis similarity of <10% and 5% respectively at another restored well-site three years after revegetation, indicating differences in plant community at restored sites, that likely function differently than natural peatlands.

The CO_2 seasonal balance in the fourth year showed there was less GEP CO_2 uptake at the mixed PRT and natural site while the clay PRT had the greatest seasonal GEP. This may be due to the clay PRT having the greatest vascular and non-vascular vegetation cover. Neither the natural site or the restored site fell within the range of seasonal NEE of those recorded on a subarctic fen which was a net sink of CO_2 (daily mean -1.05 g C m^{-2} d^{-1} compared to 0.38 g C m^{-2} d^{-1} in the present study; Aurela et al., 2002). It is likely the natural site measurements in this study do not provide a fair representation of the entire ecosystem's CO_2 exchange and that given different sampling methods, the natural site would likely act as a CO_2 sink. At both natural sites, trees and large shrubs are present and were likely included in chamber measurements through below ground root respiration which can account for around 9 g CO_2 m^{-2} d^{-1} (Munir et al., 2017). Meanwhile the plant productivity from these species are not captured in chamber measurements. The closed chamber technique used, measures at plot scale and is not able to capture overstory CO_2 exchange which contributes greatly to ecosystem CO_2 exchange in treed peatlands (e.g. Wieder et al., 2009; Munir et al., 2017). The inclusion of overstory root respiration in natural site plots would also explain the high measured ER at these sites despite having a significantly deeper WT than the restored site. At a series of plots within 200 m of the natural sites, Strack et al. (2017) estimated overstory productivity of 32-81 g C m^{-2}. Adding this to the modelled NEE from the fourth year would result in the natural site having a near neutral C balance. It is also likely that the respiration processes on the restored vs. natural sites are different despite having similar values. It is expected that heterotrophic respiration plays a larger role at the restored site from higher microbial activity than at the natural site.

It is important to acknowledge the other forms of C that would change the total C balance if included, such as particulate organic C and dissolved organic C. It is also important to note the limitations of a sampling campaign that occurred only during the growing season. Wintertime CO_2 balance, although small, can still play an important role in the annual C balance (Aurela et al., 2002). Despite these limitations, the present study provides valuable information regarding the return of C and greenhouse gas exchange rates on a restored peatland well-pad and indicate that the tested PRTs show promise for restoring peatland ecosystem functions quickly post-restoration.

2.6 Conclusions and Recommendations

Two to four years following the PRT restoration of a peatland after use as an oil well-pad, vegetation cover has increased, leading to greater uptake of CO_2 as NEE and GEP. The WT, T_5 and vegetation cover explained CO_2 fluxes with different interacting variables, while WT explained CH_4 fluxes. Overall, the restored well-pad treatments cycled CO_2 and CH_4 at similar rates as natural sites for two out of the three years studied. It is suspected, however, that the closed chamber method does not adequately capture CO_2 flux at natural sites with higher tree density suggesting that the restored pad takes up less C than the natural site once trees are included. Methane emissions have dropped every year despite the WT rising closer to the surface and vascular plants increasing coverage.

In order to re-establish GEP on the restored site, it is recommended to focus on tree regeneration. Water table levels that are too deep also promote upland weedy species growth, while too shallow promotes invasion of marsh plants such as *Typha* spp., outcompeting native species (Govermment of Alberta, 2017). Therefore, the Government of Alberta (2017) recommends a WT level of 2-8 cm to be reached for restoration, with annual fluctuation no greater than 30 cm, as high fluctuations promote decomposition. The ability for trees to regenerate, namely the most dominant species in the area, *Picea marina*, should also be considered when choosing WT positions, as positions too shallow may hinder tree growth (Dang and Lieffers; 1989; Pepin et al., 2002).

All tested peat replacement restoration treatments generally had similar C exchange rates; however, in the fourth year when WT was much closer to the surface than the second and third year, peat and clay PRTs took up significantly less CO_2 as GEP than the mixed PRT and the natural site. All PRTs show promise as restoration techniques for well pads in order to quickly recover peatland vegetation and C uptake. However, due to the isolated area of the mixed PRT on the site, the lower CO_2 uptake results are confounded. When considering treatment options for future restoration projects the clay PRT appeared to be a slightly better carbon sink due to its lowest modelled CO_2 balance and WT positions closer to the surface; however, it is likely that this effect may be due to the inconsistency in site leveling during the PRTs' construction rather than due to the nature of the PRT. Acknowledging that cost and resources are a factor in restoration treatment choices, the mixed PRT is also recommended. The mixed PRT is recommended due to its ability to maintain similar CO_2 cycling function to the natural site despite its higher overall net seasonal CO_2 flux and despite environmental conditions such as deeper WT position which are likely a result of relatively higher PRT elevation. It is expected that given time, forest cover will regenerate, contributing to overstory GEP and site biomass, however long term monitoring is needed to ensure this outcome.

Chapter 3: Net-primary production, biomass and decay rates of a restored peatland following well-pad removal

3.1 Introduction

The oil-sands industry has caused major disturbance to a substantial area of peatlands ecosystems in Alberta (Vitt et al., 1996). Although research of oil-sands peatland restoration is limited, there is a body of research on a variety of other disturbances to peatlands such as flooding, drainage for agriculture, forestry, and peat harvesting, and fires (e.g. Turetsky et al., 2002). Peatland disturbances alter ecohydrological conditions in a number of ways including lower soil moisture retention, changes in vegetation composition, and change in WT position (Cagampan and Waddington, 2008; Quinty and Rochefort, 2003). When vegetation is removed by the disturbance there is little or no further input of organic matter, but if oxidation and decomposition of surface layers continues, as is often the case, sites become C sources rather than sinks (Waddington et al., 2002). In order to regain previous C cycling patterns, peatland restoration, involving regaining hydrological function and peatland plant species cover is crucial. This study aims to evaluate the success of several restoration methods following disturbance caused by the placement of mineral layers on peat during well-pad construction for in situ oil sand extraction by assessing plant production and decomposition rates.

Unlike other peatland disturbances, the amendment of a mineral layer is an additional obstacle for well-pad restoration. The process of in-situ oil sands extraction requires organic soil to be covered with 1-2 m of mineral fill (well-pad) over the peat in order to allow access to the production wells and drilling during its lifetime. The well-pad occurs at higher elevation than the surrounding peatland, cutting off hydrological flow and causing much dryer conditions (Graff, 2009). For several years of oil production, the well-pad becomes compacted and incapable of supporting wetland species establishment. Restoration methods aimed at returning the ecosystem function of natural peatlands have included shaving the fill down to the level of the surrounding peatland (Vitt et al., 2011; Shunina et al., 2014.

Similar to other disturbances, which cause the loss vegetation and seedbanks, the mineral layer application eliminates plant life on the well-pad footprint. Historically in peatland restoration spontaneous colonization seldom results in adequate vegetation re-establishment (Poulin et al., 2005; Graf et al., 2008) . The Moss Layer Transfer Technique (MLTT) (Quinty and Rochefort, 2003) has proven to be a successful revegetation strategy for cutover (e.g. González and Rochefort, 2013) and well-pad restoration alike (Gauthier et al., 2017), providing the conditions necessary for C to be captured in vegetation and stored due to conditions created by re-established *Sphagnum* species. A reclamation strategy from the Reclamation Criteria for Wellsites and Associated Facilities is to introduce plant species with high polyphenol content on organic substrate (Alberta Environment, 2015) to encourage peat accumulation. A strategy for sourcing this organic substrate, or peat, while ensuring the site's elevation is at the level to maintain connectivity with the surrounding landscape has been to resurface the underlying peat, and replace it over the pad, or to remove the pad entirely followed by peat de-compaction (Vitt et al., 2011; Gauthier et al., 2017).

Productive plants and low rates of decomposition are required to achieve the long-term objective of peatland restoration: the establishment of peat accumulation. Plant growth and decomposition rates are in turn controlled by hydrology, substrate and microbial activity. The influence of the mineral layer on nutrient availability, hydrology, and peat structure are expected to change decomposition rate and productivity in the restored peatland. Nutrients introduced from the pad material are likely to increase microbial activity and thus decomposition with newly exposed peat becoming oxidized. Peatland productivity and decomposition rates in natural peatlands are well-researched (e.g. Thormann et al., 1999; Vitt et al., 2000, 2009), but only three studies, to our knowledge exist on restored peatlands (Graf and Rochefort, 2009; Andersen et al., 2013; Touchette, 2017) none being on restored peatland well-pads.

In this study, three experimental well-pad removal strategies (described in section 3-2) were undertaken in a fen with the intention of restoring peatland function, including rates of carbon (C) cycling similar to natural peatlands in the region. The objectives of this study were to:

A) Quantify and compare plant biomass and productivity between restoration treatments.

B) Quantify and compare decomposition rates of the dominant plant species between restoration treatments.

C) Synthesize published data on productivity, biomass and decomposition on restored and natural peatlands in Canada to evaluate ability of well-pad restoration techniques to return peat accumulation function.

3.2 Study sites and design

A decommissioned, never-drilled well-pad in northern Alberta, located northeast of the town of Peace River (56.397°N, 116.890° W) underwent experimental restoration peat replacement treatments (PRT) in November 2011. At least 40 cm of peat was replaced as part of all restoration methods on sections of the 1.4 ha well-pad (Fig. 3-1). All treatments were leveled to 10 cm below adjacent hollows, either by placing mineral soil fill from the well-pad under peat or through peat de-compaction, with the idea that further expansion of the peat following restoration would result in surface level similar to the adjacent undisturbed fen.

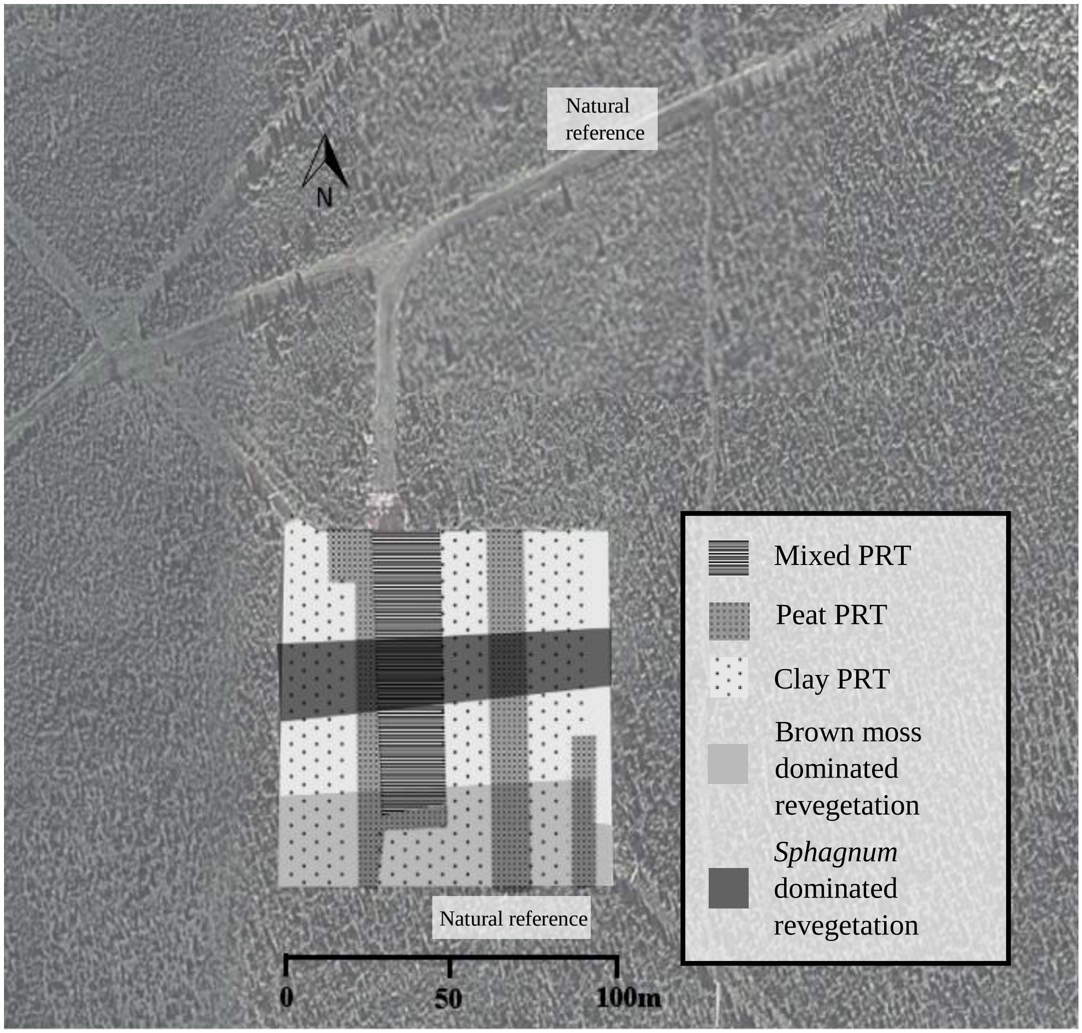


Figure 3-1 Site map of restored peatland following well-pad replacement treatments (PRT) and revegetation treatments (Google Earth, 2017).

Using an excavator, the mineral well-pad and peat layer were individually excavated in two treatments. Only the peat layer was replaced in the peat PRT (peat) following de-compaction. In the second treatment, the pad was detached from its underlying geotextile layer, then a portion replaced under the peat to achieve the desired elevation (clay). In an attempt to increase efficiency of labour, a third treatment involved flipping the clay and peat layer in one motion, resulting in some mixing of the clay and geotextile layer with the peat at variable depths (mixed). All treatments were systematically revegetated following the MLTT (Quinty and

Rochefort, 2003) using diaspores collected from two nearby peatland donor sites. The material collected from three nearby donor sites dominated with brown moss (*Tomenthypnum nitens* and *Polytrichum strictum*) was spread over the south end of the site, and material collected from a donor site dominated in *Sphagnum* moss was spread over the middle section of the site. Additionally, the middle of the site was revegetated with material collected from a donor site dominated in *Polytrichum* mosses but was not included in the present study. To promote *Polytrichum* moss growth150 kg/ha of phosphate fertilizer (0-3-0) (Quinty and Rochefort, 2003) was applied across the entire site in order. Two reference sites on natural peatlands were chosen (natural) for comparison: one roughly 50 m north east of the well pad, in a treed moderately-rich fen, the other directly south of the well pad in a poor treed fen.

3.3 Methods

3.3.1 Biomass and Net Primary Production (NPP)

Above- and belowground biomass was collected from the site in August 2016. Aboveground biomass was collected by clipping all plants in a 25 x 25 cm quadrat down to the restored peat surface. Cores were taken for belowground biomass to 40 cm depth using PVC pipe (8 cm diameter). Triplicates per revegetation and PRT treatment combination were taken for aboveground, belowground and moss biomass samples. Cores were frozen and shipped to University of Waterloo where roots and rhizomes were extracted from peat and cleaned before drying, along with moss and aboveground biomass, at 80 ºC for 48 hours and weighed to the nearest 0.1 g. The NPP was calculated by dividing biomass by the number of years since peatland restoration (4 years) with the exception of aboveground vascular plant biomass which had <6% woody material and was therefore simplified to be equal to NPP in the study year.

3.3.2 Decomposition

Above- and belowground biomass of the four most common vascular species: *Salix bebbiana, Equisetum pratense, Carex canescence, Carex aquatilis,* and two most common non-vascular species on the restored well-pad: *Polytrichum strictum* and *Sphagnum angustifolium*, were collected from the site in July 2016, cleaned, and dried at 80 ºC for 48 hours. The plant material was weighed to the nearest 0.0001 g, placed in 1 mm nylon mesh bags, sewed shut and weighed again. Litter bags were roughly 8 x 8 cm with 1-2 g of plant material placed inside. In the instance of plant material that could not be contained by 1 mm mesh (*C. canescence* aboveground, *P. strictum* and *S. angustifolium*), Nitex® screening (250 µm mesh size) was used (Graf and Rochefort, 2009). Careful attention was taken so as not to rustle bags to avoid loss of material. Bags were tied to a fishing line and flag, buried at 5 cm depth in August 2016 and retrieved one year later. Three replicates of litter bags per PRT and revegetation treatment were made for each species' above- and belowground litter or moss litter. Upon retrieval, the bags were carefully cleaned and foreign objects were removed (i.e., root ingrowth, loose peat), then dried and weighed again. Drying of material before and after burial was to maintain consistency by eliminating water weight.

The linear decay rate (k') was calculated for each litter type using the following equation (Reader and Stewart, 1972).

$$k' = [(X_0 - X)/X_0)]$$

Where X_0 represents the initial dried litter mass (g) and X, the dried mass (g) after burial. The exponential decay coefficient (k) was also calculated using the following equation (Brinson, Lugo & Brown, 1981).

$$k = \ln(X_0/X)/t$$

where *t* is the time in years.

Of the 216 litter bags deployed, 171 were retrieved. Of those retrieved, data from 17 litter bags was discarded due to having a perceived mass gained rather than loss from inability to remove small roots.

3.3.3 Nutrient availability

Plant root simulators (PRS®) are probes with ion exchange resin membranes of 17.5 cm^2 surface area used to determine differences in nutrient availability (i.e., soil chemistry) in several peatland studies (e.g., (Wood et al., 2016; Munir et al., 2017; Murray et al., 2017a;) by attracting and adsorbing ions through electrostatic attraction. Each probe is 15 x 3 x 0.5 cm with the 5 x 1.75 cm resin area placed 3 cm from the tip of the probe. One set of PRS® probes (four anion and four cation probes) were buried at four locations at each natural site, as well as on each PRT and revegetation treatment combination, with a set distributed within a 1m × 1m quadrat. The probes were inserted upright with top of the membrane on each probe buried 8 cm below the surface and were kept cool before and after installation. After 20 days (July 11^{th}- 31^{st}, 2016), the probes were removed, thoroughly cleaned and scrubbed with deionized water and sent to Western Ag Innovations Inc., (Saskatoon, Ontario) for analysis. At the lab, all ionic species that were adsorbed were measured using analytical instruments depending on the ion (e.g., ammonium ion analysis occurred colorimetrically with an automated flow injection analysis system, while sulfur, iron, and manganese were analyzed via inductively-coupled plasma spectrometry (PerkinElmer Optima 3000-DV, PerkinElmer Inc., Shelton, CT)).

3.3.4 Environmental conditions

WT position (WT) and soil temperature (T_5) at 5 cm (thermocouple), were measured manually at four well locations on each combination of PRT and revegetation treatment bi-monthly from June to September 2016. Soil samples of roughly 100 cm^2 were collected by hand of the top two centimeters of soil in a grid pattern over the site, four times over the 2016 growing season. A slurry of soil and deionized water was made from each sample and measured for pH and electric conductivity (EC) (Thermo Scientific Orion pH meter and Orion Versa Star Advanced Electrochemistry meter). A solution of one part peat and two parts de-ionized water was measured with the meter, which was rinsed with de-ionized water between every sample.

3.3.5 Statistical Analysis

Seasonal means of WT, T_5, EC and pH were calculated to compare environmental conditions between PRTs. Mean decay rates of PRTs were calculated from the means of the above- and belowground decay rates of each species at each PRT and revegetation treatment in order to have an equal weight of each species type at each PRT despite missing data. ANOVA followed by pairwise comparisons with Tukey's HSD was used to determine significant differences between PRTs, indicated by a $p<0.05$. All statistical analyses were conducted using R software (Version 2.6.1; R Development Core Team 2006).

Net primary production and decay rates were regressed with all PRS® ion availabilities with the exception of ions with supply rates lower than the detection limit. These ions with low supply rates are included in Table 3-1. Each data point in the regressions was derived from the mean of each PRT and revegetation combination in order to associate nutrient availability to NPP and decomposition data sampled at different locations within the same treatments (n=6 for each PRT). To determine significance of these regressions $p<0.10$ was used due to the low statistical power from the small sample size and thus this analysis should be considered exploratory in nature.

3.4 Results

3.4.1 Environmental control and soil chemistry

Most environmental conditions were similar between PRTs across the site (Table 3-1). Mean T_5 in 2016 was similar across treatments with the highest temperature at the mixed and clay PRTs (14.9 and 14.0°C) followed by the peat PRT (10.1°C). Mean WT varied between treatments with the deepest (i.e., closest to the surface) occurring at the clay PRT (-7.4 cm), significantly higher than mixed (-12.1 cm). The peat PRT was not significantly different than either PRT at -9.8 cm. The electrical conductivity (EC) was statistically similar across treatments but was almost double in the mixed PRT (626.6 µS cm^{-1}) compared to 346.3 and 384.1 µS cm^{-1}, at peat and clay PRTs, respectively. The pH was 5.5 to 5.6 across the sites and similar at all PRTs (Table 3-1).

Table 3-1 Mean environmental variables. ± standard error. Different letters indicate statistical difference.

PRT	WT (cm)	T_5 (°C)	EC (µS cm^{-1})	pH
Clay	-7.4 ±1.0 a	14.0 ±0.7 a	346.3 ±703.2	5.6 ±0.6
Mixed	-12.1 ±1.6 b	14.9 ±0.8 a	626.6 ±462.7	5.6 ±0.6
Peat	-9.8 ±1.5 a b	10.1 ±0.8 a	384.1 ±271.2	5.5 ±0.6
Natural	-1.9 ±1 c	10.1 ±1.2 b	170.6 ±315	5.8 ±0.9

Plant root simulators showed similar supply rate of all ions to plants among the PRTs with the exception of sulfur (Table 3-2). The mixed PRT had significantly higher supply rate of S than clay or peat. The ions NO_3^-, NH_4^+ and Cu all had supply rates below the detectable limit. On the PRTs, calcium had the highest supply rate followed by Fe, Mn, Al, K, Zn and P. The natural site has significantly lower supply rates of Ca and Mg than all PRTs. The supply rate of S at the natural site was similar to the peat PRT, significantly lower than the mixed PRT, and lower than the clay PRT, although not quite significantly different (p=0.06). The natural site also had a lower supply rate of NO_3^- and Al and a higher rate of K and Fe but not significantly so. There was a significantly higher supply rate of P at the natural site than all PRTs.

Table 3-2 Mean ion supply rates (µg $10cm^{-2}$ 20 $days^{-1}$) ± standard error, *below detectable limits.

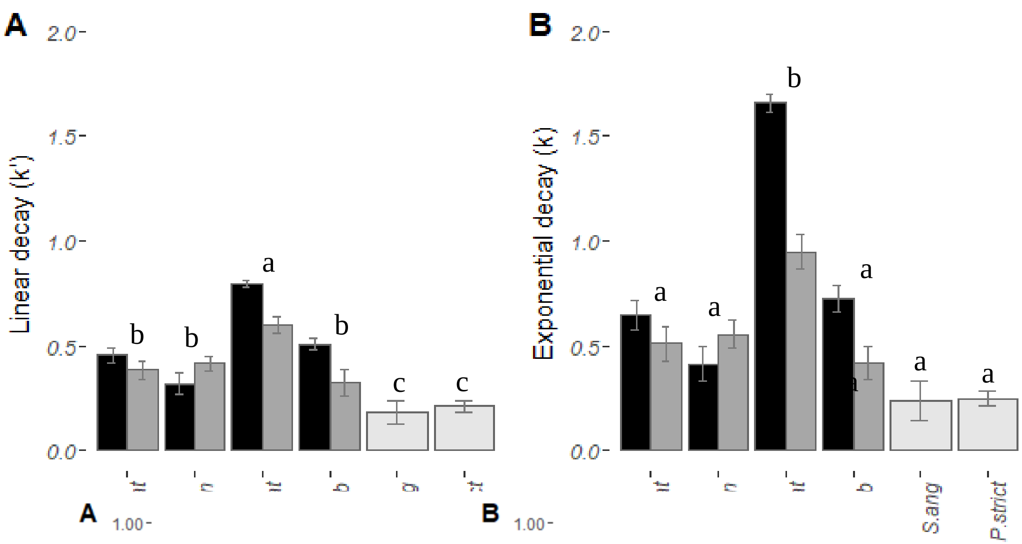
PRT	NO_3-*	NH_4+*	Ca	Mg	K	P	Fe	Mn	Cu*	Zn	S	Al
Clay	0.4	2.1	2349 a ±50	373 a ±26	17 a ±4	1 a ±0	214 ±74	27 ±5	0.1	6.0 ±2	856 a ±184	10.5 ±2
Mixed	0.8	1.5	2350 a ±112	395 a ±24	12 a ±1	1 a ±0	93 ±24	18 ±3	0.1	3.1 ±1	1335 b ±74	12.1 ±3
Peat	0.3	2.0	2299 a ±49	385 a ±27	18 a ±6	2 a ±0	253 ±74	31 ±7	0.0	8.0 ±48	768 a ±182	10.8 ±2
Natural	0.1	2.9	1287 b ±302	275 b ±61	75 b ±36	5 b ±2	303 ±118	26 ±7	0.0	7.3 ±5	358 a ±92	8.2 ±1

3.4.2 Decomposition

One year after burial, k' was significantly different between vascular and non-vascular species, where all vascular species decayed at a significantly faster rate than the mosses. The species with the lowest k' were *Sphagnum angustifolium* and *Polytrichum strictum* (Fig. 3-2). The species with the highest k', significantly higher than all other species was *Equisetum pratense*, followed by *Carex aquatilis*, *Salix bebbiana* and *Carex canescence.*

Similarly, exponential decay varied among vascular and moss species on the restored site. *Equisetum pratense* had significantly higher exponential decay rate than *C. aquatilis, C. canescence, S. bebbiana, S. angustifolium* and *P. strictum* (Fig. 3-2). The aboveground litter of *Equisetum* decayed at a faster rate than belowground. The aboveground litter had a higher k and k' than the below ground biomass for all species except *C. canescence*.

The rate of k' and k was similar between PRTs with the lowest loss at the peat PRT, followed by clay and mixed PRTs (Fig. 3-3). WT position and soil temperature had no significant effect on decay rates (Fig. 3-4).



Figur ... *escence,*
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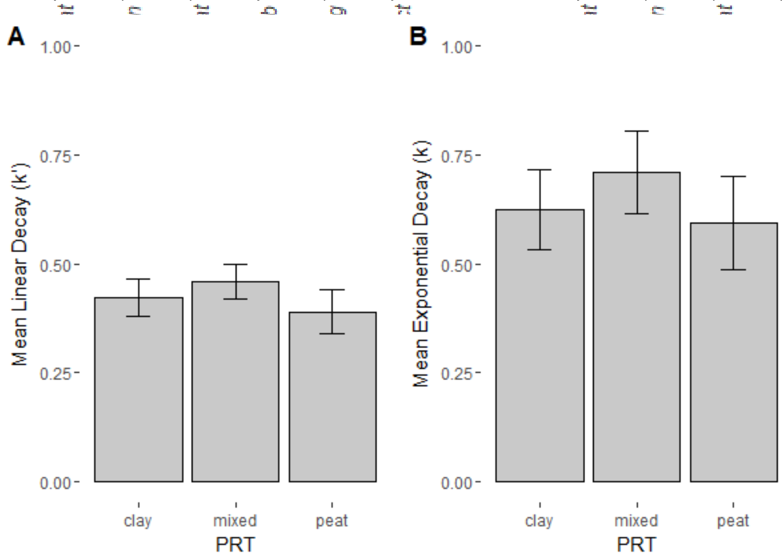

Figure 3-3 Mean decay rates at PRTs after one year. A: linear decay (k'). B:exponential decay (k)

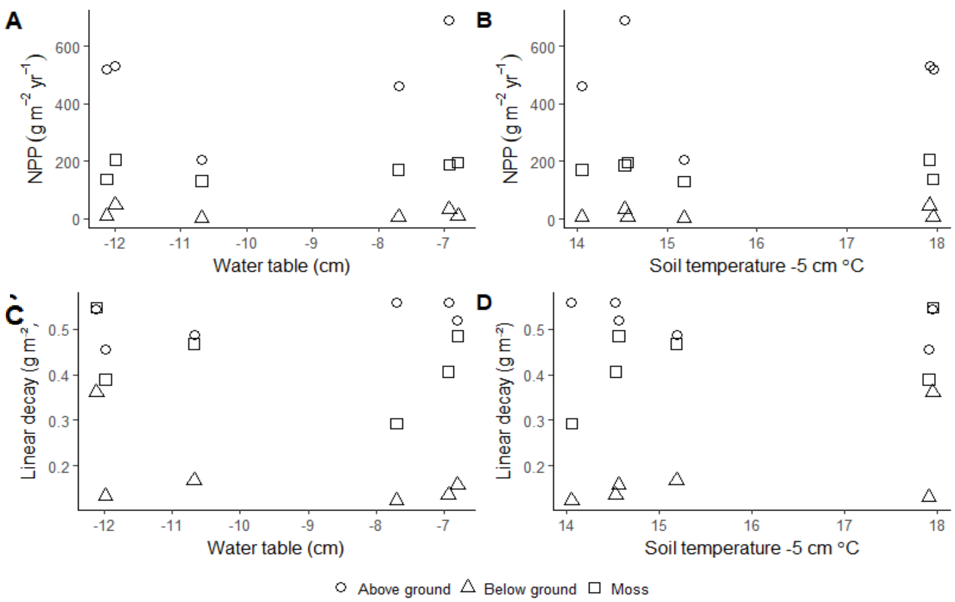

Figure 3-4 Plot of environmental controls on NPP and Linear decay. A: WT position and NPP. B: Soil temperature and NPP. C: WT position and linear decay. C: Soil temperature and linear decay.

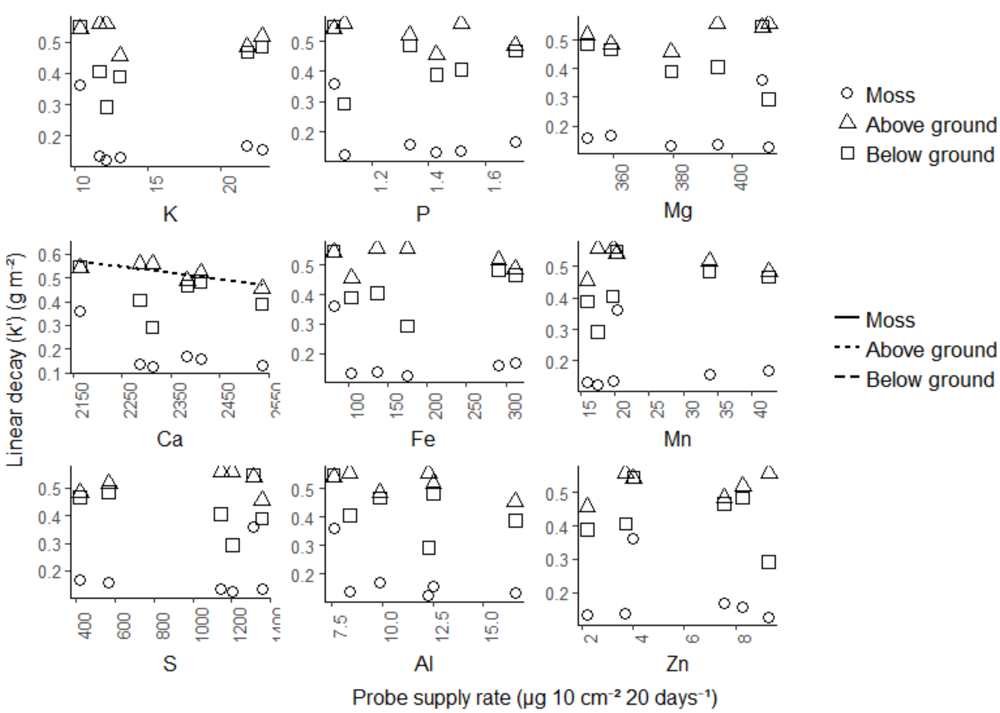

Figure 3-5 Plant Root Simulator (PRS) supply rate *versus* linear decay (k'). Regression lines are plotted for each vegetation type when statistically significant. Regression equations are: Ca: y=-0.0005x- p=0.07647, r^2=0.21

Linear decay of above-belowground vascular plant litter and mosses was regressed with supply rates of K, P, Fe, Mn, Mg, Ca, S, Al and Zn (Fig. 3-5). Calcium had a significant negative correlation with moss k'. None of the other ions had a significant relationship with decay rates.

3.4.3 Biomass and NPP

Net primary production and biomass varied between moss, and above- and belowground vascular plant tissues. Moss had significantly lower biomass than both above- and belowground biomass of vascular plants (Fig. 3-6, A and B). Annual growth of aboveground vascular plant biomass was significantly higher than belowground and moss NPP (493, 162 and 18 g m^{-2} yr^{-1}). Biomass was highest belowground followed by aboveground vascular plants and moss (683.9, 493.4 and 85.3 g m^{-2}). Both NPP and biomass were similar at all PRTs (Fig. 3-6, C and D), with the highest NPP occurring in the peat PRT, followed by mixed and clay PRTs (767.9, 560.0 and 421.1 g m^{-2} yr^{-1}). Biomass was highest at the peat PRT as well, followed by the mixed and clay PRTs (1237.7, 1163.8, 1001.2 g m^{-2}).

The measured supply rates of P, Mg, Ca, S, Al and Zn did not have a correlation to NPP or biomass (Fig. 3-7). The supply rates of K, Fe and Mn, however, had significant negative relationships with aboveground NPP. Zinc had a significant negative relationship with moss NPP. For sake of brevity, regressions with biomass and ion supply rates are not displayed; however, these relationships and lack thereof would hold equally true to aboveground vascular plant NPP, which is equal to biomass, as well as belowground and moss biomass which are exactly four times NPP. Water table position and soil temperature had no significant effect on NPP (Fig. 3-4).

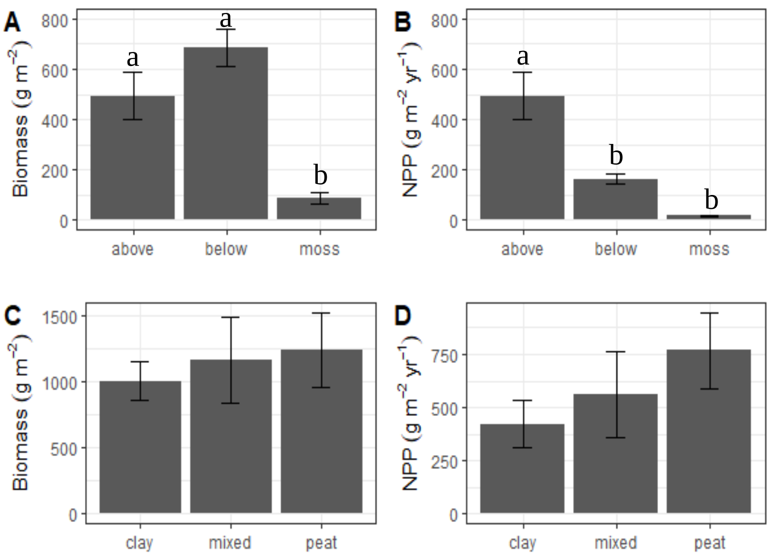

Figure 3-6 A: Moss, above- and belowground biomass. B: Moss, above- and belowground NPP. C: Mean biomass at PRTs D: Mean NPP at PRTs.

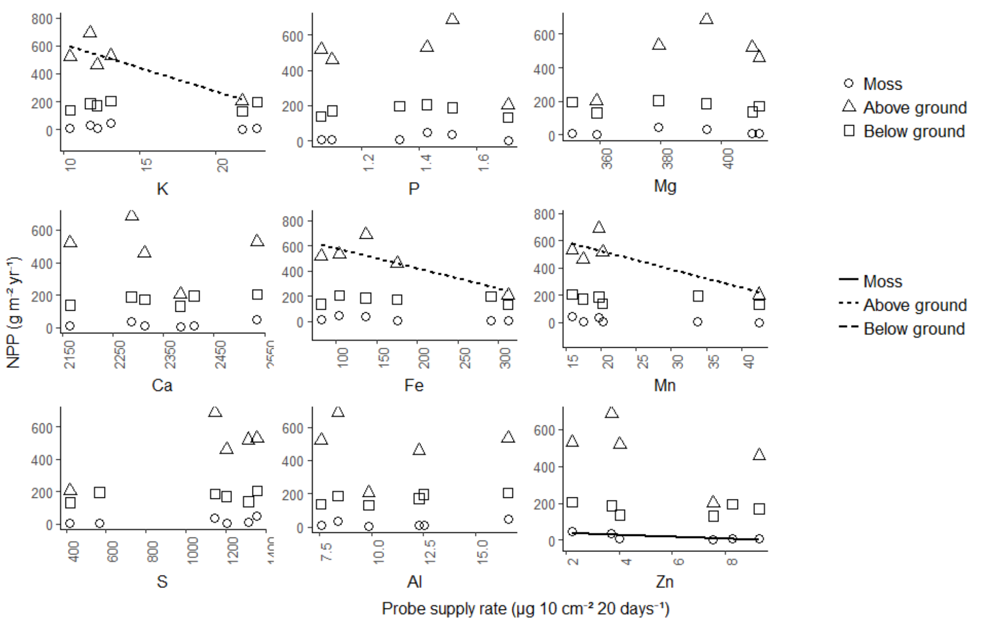

Figure 3-7 Plant Root Simulator (PRS) supply rate *versus* NPP. Regression lines are plotted for each vegetation type when statistically significant. Regression equations are: Aboveground NPP with K: y=-33.57x-605.20, p=0.0575, r^2= 0.6673. Aboveground NPP with Fe: y=-1.62x+740.46, p=0.08414, r^2= 0.58. Aboveground NPP with Mn: y=-13.40x+790.83, p=0.0794, r^2=0.59. Moss NPP with Zn: y=-0.13x+8.13, r^2=0.55.

3.5 Discussion

The primary objective of peatland restoration is to transform a disturbed ecosystem that has become a source of C, back to become a C sink with similar function to natural peatlands (Waddington et al., 2010). Four years after PRTs and applying the Moss Layer Transfer Technique for peatland restoration (outlined by *Rochefort et al.* [2003]) resulted in NPP rates similar to non-treed peatland and drastically lower than treed natural references (Table 3-3). Decay rates were higher than natural reference sites (Table 3-4). All NPP results presented are analogous to biomass because all vegetation started growing following PRT, therefore aboveground vascular plant NPP is equal to biomass, while belowground and moss NPP are a consistent fraction of biomass.

3.5.1 Reestablishment of biomass and NPP on the restored well-pad

The total NPP across PRTs ranged from 421-768 g m^{-2} yr^{-1}, greater than total NPP found in some wooded and non-wooded natural peatlands (Thormann et al., 1999) and much lower than the sum of understory and tree biomass in another wooded natural peatland NPP (Miller et al., 2015), not including below-ground (Table 3-3). Many of these studies also do not include overstory production, which is a substantial contributor to NPP in Alberta, considering many Alberta peatlands are forested (Vitt et al., 2000). Similar results were found for the biomass, which had greater mass when comparing the restored site in this study to the understory of most non-wooded natural peatlands but much lower mass than the overstory of wooded peatlands (Dimitrov et al., 2014).

Above-ground biomass on this restored well-pad site greatly outweighs NPP; however, these relative proportions are the result of the young age of the site- only four years post restoration. It will take many years until trees currently present on the site will be able to contribute to a greater amount of NPP in their trunk and limbs to be able to have similar NPP as natural peatlands. In a study of a peatland disturbed for collection of plant material for restoration similarly found tree colonization uncommon 2-4 years post-disturbance (Murray et al., 2017b) as woody plants require a longer period for establishment compared to herbs (González et al., 2013). After wildfire disturbance in Alberta bogs, it has been found that the recovery time for *Picea mariana* stands took decades, with maximum C accumulation rates occurring at 34 years for fine root biomass and 74 years for aboveground and coarse root biomass (Wieder et al., 2009). When compared to a well-pad without PRTs or MLTT, NPP at that site was much smaller (20 g m^{-2} yr^{-1}; Engering, unpublished) than the present study, with vegetation composition comprising exclusively of non-peatland species.

Table 3-3 Biomass and NPP of fens in the boreal region. * indicates subtotals of available data rather than exhaustive net totals.

	NPP ($g\ m^{-2}\ yr^{-1}$)					Biomass ($g\ m^{-2}$)					Notes	reference
	Over-story	Under-story	below-ground	moss	total	Over-story	Under-story	below-ground	moss	total		
Natural sites	361	211		143	715*		164			164*	2 Year mean wooded bog	Szumigalski, 1995
	218	206		43	466*		52			52*	2 Year mean open bog	Szumigalski, 1995
		294		41	335*						rich fen	Bartsch and Moore, 1985
				154	266					122	2 year mean wooded rich fen	Thormann et al., 1999
				123	198					95	2 year mean bog	Thormann et al., 1999
				122	225					110	2 year mean poor fen	Thormann et al., 1999
			140		140*						Mean of 3 fens	Vitt et al., 2009
				132	132*						4 year mean	Rochefort, Vitt and Bayley, 1990
						14	170		156	230*	poor fen	Glenn et al., 2006
						<1	81		76	157*	extreme-rich fen	Glenn et al., 2006
						6600				6600*	black spruce forest	Dimitrov et al., 2014
						1822.2				1822*	forest transition to fen	Dimitrov et al., 2014
						800				800*	black spruce fen	Dimitrov et al., 2014
	3054				3045*						treed fen	Miller et al., 2015
	6456				6456*						treed fen	Miller et al., 2015
	3751				2751*						treed fen	Miller et al., 2015
	5999				5999*						treed fen	Miller et al., 2015
	1692				1692*						treed fen	Miller et al., 2015
	218				218*						shrubby fen	Miller et al., 2015
	868				868*						shrubby fen	Miller et al., 2015
			290		290*						wooded fen	Vitt et al., 2009
			60		60*						shrubby fen	Vitt et al., 2009
			50		50*						open fen	Vitt et al., 2009
				141	141*						4 year mean bog	Rochefort, Vitt and Bayley, 1990
				132	132*						4 year mean fen	Rochefort, Vitt and Bayley, 1990
Restored sites		460	183	8	421		460	733	38	1001	clay PRT	this study
		367	168	25	560		367	672	125	1164	mixed PRT	this study
		616	139	19	768		616	647	94	1238	peat PRT	this study
					54.09						*Polytricum* carpet	Anderson, Pouliot and Rochefort, 2013
					134.13						*Sphagnum* carpet	Anderson, Pouliot and Rochefort, 2013
					126.92						*Sphagnum* and shrub carpet	Anderson, Pouliot and Rochefort, 2013

The restored site had much greater understory vascular NPP than natural peatlands (Bartsch and Moore,1985; Szumigalski, 1995) (Table 3-3). Vascular vegetation cover is an important factor contributing to CO_2 uptake several years following restoration and therefore also significant to overall carbon sequestration (Yli-Petäys et al., 2007); however, as measured in the present study, vascular vegetation also decomposes faster than mosses (Thormann et al., 1999) and therefore may contribute less to peat accumulation.

The moss biomass was similar to natural peatlands. Although individual species' biomass was not recorded, the majority of moss biomass comprised of *Sphagnum* species. Establishment of *Sphagnum* is crucial to peatland restoration because of its ability to restore hydrological function in the surface of disturbed peatlands, which otherwise have altered porosity and water retention (Lucchese et al., 2010; McCarter and Price, 2013).

3.5.2 Reestablishment of decomposition rates on the restored well-pad

When comparing decay rates to other studies it is important to note the many differences in litter bag methods and site conditions will impact measured decay rates. The time of year the litter was harvested can lead to different nutrient storage location in the storage organs of plant species, many of which store the majority of their nutrients in the roots and rhizomes in late fall rendering their roots and rhizomes more labile and their above-ground tissues more recalcitrant during this time (Berendse and Jonasson, 1992)**.** Conversely, the opposite is true in late spring to early fall (Berendse and Jonasson, 1992). Site conditions such as soil chemistry, temperature and water level can influence decay rates (Wieder and Vitt, 2006). Species with more recalcitrant properties such as woody species would also decay more slowly than those found at this site (Taylor et al., 1989).

Table 3-4 Poor fen decay rates of dominant species from this study

Species	Plant type	linear decay (k')	exponential decay (k) ($g\ m^{-2}\ yr^{-1}$)	Site	Reference
Carex aquatilis	leaves	-0.254		natural	Thormann et al., 2001
	rhizomes	-0.172		natural	Thormann et al., 2001
	above	-0.456	0.64	4 years restored	this study
	roots and rhizomes	-0.384	0.51	4 years restored	this study
Carex canescence	above	-0.319	0.41	4 years restored	this study
	below	-0.415	0.56	4 years restored	this study
	litter		0.77	4 years restored	Touchette, 2017
Carex spp.	above	-0.58	-0.87	natural	Thorman et al., 1999
Equisetum pratense	above	-0.80	1.66	4 years restored	this study
	roots and rhizomes	-0.60	0.95	4 years restored	this study
Polytrichum strictum	moss	-0.21	0.25	4 years restored	this study
Sphagnum angustifolium	moss	-0.16	0.17	natural	Thorman et al., 1999
	moss	-0.18	0.24	4 years restored	this study
Sphagnum spp.	moss	-0.11	0.61	8 years spontaneous recolonization	Graf and Rochefort, 2009
Salix bebbiana	above	-0.51	0.73	4 years restored	this study
	below	-0.33	0.42	4 years restored	this study
Salix spp.	leaves	-0.26		natural	Thormann et al., 2001
	roots	-0.36		natural	Thormann et al., 2001

On the restored well-pad, the decay rates of species differed after one year. The significantly greatest k and k' was of the species *Equisetum pratense.* There is no literature on the decay rates of *Equisetum* spp. for more than 5 months; however, it has been found that *Equisetum* spp. decomposed almost twice as fast as *Carex* spp. in a wetland (Danell and Sjöberg, 1979). The decay rates for remaining vascular species *C. aquatilis, C. canescence* and *S. bebbiana* were all similar. The decay rates of *C. aquatilis* were slightly greater than those of *Carex* spp. found in natural peatlands with the exception of those found in a moderate-rich fen by Thormann et al. (1999) (Table 3-4). The exponential decay rate of *C. aquatilis* was lower than that found in another restored peatland (Thromann et al., 2001). The decay rates of *Salix bebbiana* were also similar but greater than those of *Salix* spp. found at a natural peatland (Thormann et al., 2001;Table 3-4).

The two moss species, *S. angustifolium* and *P. strictum,* decayed significantly slower than vascular plant litter. These low decay rates were expected in *Sphagnum*, which typically has an extremely low mass loss in the catotelm of peatlands (0.1 % to 0.001 % per year) due to the polyphenolic network of polymers and a lipid surface that provide bonding to the cell wells (van Breeman, 1995). The decay rates of *S. angustifolium* were only slightly higher than at other peatlands (Table 3-4). To our knowledge there are no other studies to measure the decay rates of *Polytricum*, which has a similarly low decay rate to *S. angustifolium* in the present study.

3.5.3 The effect of remnant mineral fill on restored site chemistry

Ion supply rates were measured on the site to compare the effect of the remnant mineral fill on soil chemistry between the site's treatments and between the restored and natural site. There is evidence of greater base cation supply rates found across all PRTs from the remnant mineral fill that may explain differences in NPP, biomass, and decay rates between this study and natural peatlands. It was expected that there would be greater amounts of NO_3 and NH_4 than typically found in natural peatlands due to the greater amount of ions in mineral/clay soils used for well-pad construction (Graf 2009) as pad material was suspected to be the cause of higher than normal pore water NO_3 concentrations in a natural peatland situated on a Steam Assisted Gravity Drainage oil sands site (Wood et al., 2016). Instead, NO_3 and NH_4 supply rates were extremely low and below the detectable limit. We believe that the supply rate in the case of NO_3 and NH_4 may not be reflective of the concentration in the soil, as plants may be outcompeting the PRS® probes for these nutrients (Western Ag, 2008). Despite the low levels, there is some evidence that NO_3 availability is higher at the mixed PRT (Table 3-2), possibly indicating a source of N from the mineral soil. Ammonium supply rates found at a poor fen in Alberta had greater supply rates that ranged between 7 and 20 μg10 cm^{-2} 14 d^{-1}, for a slightly shorter burial time (Murray et al., 2017a). Nitrate was much higher at a rich fen, bog and poor fen, ranging between 20 and 120 μg 10 cm^{-2} μg 21 d^{-1} (Wood et al., 2016), but not much higher at the poor fen previously mentioned which ranged between <1 and 3 μg10 cm^{-2} 14 d^{-1} (Murray et al., 2017a).

At a poor fen in Alberta, sulfur supply rates varied from 5-14 μg 10 cm^{-2} 20 d^{-1}, much lower than in this study (Murray, unpublished); however, values measured on the well-pad were within the range found at a rich fen in Alberta (~70-900 μg 10 cm^{-2} 21 d^{-1}; (Wood et al., 2016). Calcium supply rates were also much greater in this study than these natural fens at means of 592 μg 10 cm^{-2} 20 d^{-1} and 955 μg 10 cm^{-2} 21 d^{-1} in previous studies compared to over 2000 μg 10 cm^{-2} 20 d^{-1} across PRTs. The phosphate rock fertilizer used on the restored site, which is Ca based, likely caused significantly higher rates of Ca on the restored site than the natural site.

A much smaller supply rate of K was found at PRTs compared to the poor fen (mean of 40 μg 10 cm^{-2} 20 d^{-1}) and rich fen (mean of 64 μg 10 cm^{-2} 21 d^{-1}). Iron and manganese supply rates were both an order of magnitude larger on PRTs compared with the rich fen (mean of 20 and 4 μg 10 cm^{-2} 21 d^{-1}, respectively). In summary, the restored well-site has high, to abnormally high supply rates of all comparable PRS probe type data available at natural peatlands, with the exception of K.

While the supply rates of ions were similar across PRTs on the site, the exception was S that had significantly higher availability at the mixed PRT. This could be explained by dissimilatory sulfate reduction occurring more in clay and peat PRTs that had a deeper water level. In the oxic and anoxic zone, sulfur in the form of SO_4 can be adsorbed onto soil particles or assimilated by plants and microbes, while dissimilatory sulfate reduction only occurs under anoxic conditions (Fauque, 1995). It is possible that S supply rate in clay and peat PRTs is lower because it has been reduced. However, the pH observed on the site (5.5-5.6) is at the lower end of the range that sulfate reducing bacteria tolerates (pH > 5.5), as they grow better under slightly alkaline conditions (Fauque, 1995).

Lower supply rates of Fe at the mixed PRT may also be indicative of oxidation. The ions Fe, Mn, Cu, and Zn mainly become adsorbed by PRS® anion probes rather than cation probes due to addition of ethylenediaminetetraacetate on anion probes and their low mobility as cations (Western Ag, n.d.). High Fe and Mn levels (> ~ 20 µg cm^{-2} burial $period^{-1}$) are indicative of anaerobic and/or acidic conditions. The Fe supply rates at all PRTs were well over 20 µg cm^{-2}; however, the mixed PRT was at least 120 µg cm^{-2}, lower than clay and peat. The Mn supply rates were lower than 20 µg cm^{-2} at only the mixed PRT suggesting more aerated conditions at the mixed PRT than clay and peat considering the similarity in acidity between the treatments. The significantly shallower mean water level at mixed PRT supports this finding.

3.5.4 The effect of chemistry from remnant mineral fill on restored plant production and decay rates

Ion supply rates were regressed with NPP and decay rates to determine the controls of soil chemistry on moss, above- and below-ground plant material (Fig. 3-5 and 3-7). The above-ground NPP rates were negatively correlated with K, Fe and Mn. Instead, we expected to find a positive correlation between NPP and nutrient availability as high K and P availability increase biomass of graminoids and forbs (Bowman et al., 1993). The low availability of NO_3 and NH_4, despite their highly likely presence in greater concentrations, supports the explanation that competition with plants may have also decreased K and P supply rates, thereby resulting in the observed negative correlation. As Fe and Mn are both redox reactive with a more soluble reduced form, they are more likely to show up on PRS® probes under more reducing conditions. It is therefore likely that Fe, and Mn rates are indicating anoxic conditions in which vascular plant growth is reduced.

A negative relationship was found between moss NPP and zinc supply rates. An increase of zinc in soil solution under more acidic conditions and the reduction of zinc availability with greater content of organic matter or clay was found by Rutkowska et al., (2015). The greatest zinc supply rates were measured at the peat PRTs (which lack a clay-mineral layer), while the lowest were measured at mixed and clay PRTs. The decrease in Zn with more moss productivity can be explained by the zinc becoming less available at treatments with more clay rather than those with greater moss NPP. Moss is likely not to grow as well on mineral substrate as it is on peat. The clay PRT, for example, had the lowest moss cover (Table 3-3); however, the mixed PRT had higher moss coverage than peat PRT.

We also expected to find a negative correlation with S and moss NPP because HSO_{3-} and SO_4^{2-} inhibit photosynthesis, growth and survival of *Sphagnum* species (Ferguson et al. 1978; Ferguson and Lee 1979,1980, 1983). Although the reduced form of S was not measured, it is expected that at least some S would be reduced. It would have been likely to find the least moss in the mixed PRT due to its significantly higher S supply rate and significantly lower WT. However, there was no correlation found. It is possible that the lowest amount of vascular plants of all PRTs, found at mixed, allowed for more moss establishment, while moss may have been outcompeted at peat and clay PRTs (Table 3-3).

Regressions between linear decay of moss, above- and belowground with ion availability showed a significant negative relationship with only Ca and moss decay. Most of the moss on site was *Sphagnum* spp., a group which is known to prefer low Ca concentrations (Wheeler and Proctor, 2000). Under a larger sample size, it would have also been expected to see a negative correlation between redox reactive ions (Fe, Mn, Zn, Al) and decay rates. Greater levels of Fe, Mn, Zn and Al would indicate more anoxic conditions, lowering decomposition rates. The regression of WT and k' does not support this assumption; however, the mean was taken of litter bags mass loss and wells per treatment, each of which was replicated at multiple locations across the site. It is possible that the WT in each of those replicates is controlled more by the local microtopography resulting from inconsistencies in the leveling of peat rather than any treatment specific-controls such as the clay layer impeding water percolation affecting WT depth.

Considering all six dominant species together, the decay rates of all three PRTs were not statistically different (Fig. 3-3 A, B); however, we expected higher decay rates at mixed compared to the other PRTs for several reasons. The significantly lower water level would have increased decomposition in the larger aerobic zone. Additionally, the greater presence of clay material mixed into the peat could have increased availability of nutrients and electron acceptors, making soil more labile (Border et al., 2012). Both of these notions are supported by the PRS® results which suggested greater oxidation or less reducing occurring in the mixed PRT (see section 3.5.3). Unfortunately, it is not clear whether differences in the mixed PRT are due to the PRT itself or the spatial placement of the treatment on the restored site, which occurs only in the centre of the site and is not replicated elsewhere. Further research would be needed to determine whether fragmentation of the mineral pad is causing this lower water level and increased redox potential. It is likely that the surface elevation of the mixed PRT was not properly leveled with the surrounding hollows as well as the other treatments, causing, or contributing, to a more shallow water level. It is therefore recommended that restoration practitioners ensure consistent elevation at the centre of the site. Testing the mixed PRT across a full well-pad is also recommended.

Considering all ion relationships with NPP and decay rates it is possible that there is little to no, or undetectable influence (by PRS probes buried at 5 cm depth) from the clay well-pad layer due to the depth disparity between the clay layer and surface processes. The clay layer was buried at a minimum of 40 cm below peat, which may have caused a decoupling from the vegetation dynamics at the surface and decomposition processes at 5 cm below the surface, both of which are likely more driven by surface processes.

3.6 Conclusion

Productivity and decomposition processes on a restored fen following well-pad removal were not completely restored to the state of a natural peatland four years after restoration but the amount of productivity was similar to some peatlands. The major limiting factor of the site's NPP was the lack of tree growth, which is known to take decades to establish. Remnant fill was likely the source of abnormally high supply rates of Ca, Mg, and S that likely contributed to overall rapid establishment of biomass due to the increase in nutrient availability from the remnant fill. This may be a positive outcome from a carbon exchange standpoint but may be setting the site's ecosystem trajectory to stray from a peatland ecosystem. The minerotrophic state, along with shallower WT levels is also likely the reason for higher decay rates than in natural peatlands. (Aurela et al., 2002). Despite these limitations, the present study provides valuable information regarding the return of productivity and slow decomposition rates on a restored peatland well-pad and indicate that the tested PRTs show promise for restoring peatland ecosystem functions quickly post-restoration.

Chapter 4.0: Conclusions and Recommendations

Despite difficulties in comparing C fluxes on the open restored well-pad to forested natural sites, it was determined that CO_2 and CH_4 fluxes on the PRTs studied on the site were restored to similar capability as reference sites. The treed overstory of reference peatlands posed a challenge in using these sites as references for CO_2 and NPP comparison. The amount of biomass and GPP of the trees at the reference sites make an unrealistic comparison for a newly restored site. Trees also pose a challenge in the measurement of CO_2 at reference sites, as they do not fit in chambers used for plot scale measurement. It would be recommended to use eddy covariance towers, however this method would not likely be feasible. The tower would not be able to capture only undisturbed peatlands in its footprint area due to the amount of nearby industrial disturbance. Additionally, eddy covariance towers would not be able to capture the small scale nature of PRTs or even an entire well-pad, often only 1 ha in size within a matrix of forested peatland. Plot-based measurements are needed to differentiate PRT fluxes from one another. Instead it is recommended to choose additional non-treed reference peatlands despite distance from the restored site and differences in equivalency to the site's pre-disturbed ecosystem. Development of a database that includes values for C exchange, NPP and litter decomposition for a wide range of peatland types in Alberta would be a valuable asset for assessing restoration outcomes. The collection of this information has been started in this thesis but recent expansion of restoration studies, provincial monitoring efforts (e.g. Alberta Biodiversity Monitoring) and industrial reports likely include additional information not easily searchable in literature. This information would be a valuable resource, if compiled.

Site chemistry revealed probable influence on the restored well-pad from the remnant mineral fill on the site. It is possible that this mineral fill influence has greater adverse effects on the mixed PRT than others, suspected due to significantly greater sulfur supply rates. The mixed PRT also has significantly lower WT position than clay and peat PRT, however; it is not certain whether or not these differences are due to its central location on the site, which may not have been adequately leveled to surrounding natural hollows. Due to the mixed PRT's confounded results and its potential to be the most cost and labour effective restoration treatment, further research is recommended to determine the effectiveness of this treatment, particularly as it is likely to be the most economically viable.

Finally, decomposition rates were not as low as those found at natural reference sites. Aside from more labile plant species dominating the restored site compared to most peatland species being the cause, the increased ionic activity and lower WT could also be enhancing mineralization rates although no significant correlation was observed in the present study. The nature of this study was more exploratory in nature and more focused research is recommended to determine whether higher rates of decomposition exist at other restored well-pads, persist over time, and if so, their specific biogeochemical causes.

Between treatments, all PRTs had similar CO_2 and CH_4 emissions. Overall, all the PRT methods have resulted in an ecosystem on a trajectory towards restoration for a site only four years after undergoing restoration treatments and are therefore all recommended as viable options for restoration. Considering industry interests of economic and time-effective solutions, the mixed PRT is recommended as the most cost effective and quickest restoration treatment. Effort should be focused on leveling treatments more accurately and consistently across the site to the surrounding natural peatlands during restoration treatments.

4.1 Future Research

To improve our understanding of well-pad restoration to peatlands, additional PRT experiments, and additional aspects of study on them, should be carried out. It is recommended to conduct a study performing just one PRT per well-pad. This would allow spatial variation on the site to be taken into account and would allow for a greater isolation of controlling variables by eliminating effects other PRTs may have on an adjacent PRT such as on hydrology and chemistry. However, consistent leveling of well-pads during civil-earthwork treatments for the creation of PRTs may still be an issue for comparison between different PRTs/well-sites.

Fields of study other than, and expanding on, carbon exchange should be undertaken and compared to natural sites. Dissolved organic carbon (DOC) and hydrology after restoration should be studied to understand hydrological fluxes and DOC transport off of the site. The hydrological conductivity of different PRTs would also help explain water availability to vegetation. More detailed site chemistry studies should investigate the cycling of nutrients other than C.

The studies done on well-pad restoration to peatlands have looked mainly at revegetation success, and diversity. These past studies as well as those undertaken in this thesis, should continue many years in the future until restoration has similar function (including but not exclusively C exchange rates) as natural peatlands. This will not only allow for determination of the time needed to restore a variety of ecosystem functions, but also evaluate whether short term studies are representative of longer term restoration outcomes.

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Appendix 1 Seasonal GEP model perameters

Treatment	**Q**	**GP_{max}**
Clay	0.08419	43.42763
Peat	0.07567	44.58261
Mixed	0.07558	41.27548
Natural	0.07178	41.82515