The aftermath of novel peatland restoration following in situ oil and gas infrastructure disturbances

Thèse

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The aftermath of novel peatland restoration following *in situ* oil and gas infrastructure disturbances

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Résumé

Les tourbières boréales des régions des sables bitumineux de l'Alberta sont soumises à de vastes perturbations et à la pollution causée par l'industrie de l'extraction *in situ* du pétrole et du gaz. En même temps, les tourbières sont reconnues comme des écosystèmes avec d'énormes capacités à stocker du carbone (C) qui ont besoin d'être protégés en période d'augmentation des émissions de gaz à effet de serre et de réchauffement climatique continu d'origine anthropique. Depuis 2015, le gouvernement de l'Alberta a publié de nouvelles réglementations visant à conserver et à protéger les tourbières suite aux perturbations causées par des infrastructures pétrolières et gazières *in situ*, par une approche de la restauration d'une « capacité foncière équivalente ». Par conséquent, la restauration écologique réglementaire des tourbières vise à rétablir les fonctions primaires des tourbières, soient l'accumulation de tourbe et la séquestration de carbone.

Les premiers essais de restauration des tourbières dans ce contexte ont débuté au début des années 2000 et jusqu'à ce jour, très peu de recherches ont été menées à évaluer le succès des différentes techniques de restauration. L'objectif de cette thèse est donc l'évaluation de différentes techniques de restauration de tourbières à la suite de perturbations de plateformes de forage *in situ*, via l'évaluation des communautés végétales et des fonctions de la tourbière restaurée, en particulier le potentiel d'accumulation de tourbe et le retour de la séquestration du carbone. Trois sous-objectifs étaient axés sur le développement des espèces végétales caractéristiques des tourbières, la production et la décomposition de la matière organique végétale, la biogéochimie et la séquestration du carbone.

L'étude a eu lieu sept à dix ans après la restauration. Les sites de recherche étaient deux anciennes plateformes de forage *in situ*, situées dans les régions des sables bitumineux de Peace River et de Cold Lake, dans le nord de l'Alberta. Dans le cadre de cette étude, nous avons choisi cinq tourbières restaurées, une zone témoin non restaurée d'une ancienne plateforme et 28 milieux humides de référence non perturbés. Les techniques de restauration évaluées comprenaient l'enlèvement complet des matériaux de construction de la plateforme de forage *in situ* (CR), l'enlèvement partiel du remplissage minéral de la plateforme de forage jusqu'à 15 cm (PR15) au-dessus du niveau de la nappe phréatique (WTL), jusqu'à 5 cm au-dessus du WTL (PR5), et jusqu'à près du WTL de l'écosystème de tourbière non perturbé adjacent (PR0). La revégétalisation s'est faite soit spontanément par l'intermédiaire de la dispersion naturelle des plantes, soit par la plantation active d'espèces vasculaires, en particulier *Carex aquatilis*, *Larix laricina* et *Salix lutea*. Sur deux saisons de croissance, nous avons mesuré l'abondance, la diversité et la richesse des communautés végétales émergentes, la productivité primaire nette (PPN) et la décomposition, ainsi que l'échange brut de l'écosystème (EBE) via l'échange de dioxyde de carbone (CO_2) et les émissions de méthane (CH_4). En outre, nous avons mesuré les facteurs environnementaux, tels que les niveaux d'eau (WTL), la chimie du sol et de l'eau et les concentrations de nutriments.

Pour l'approche de restauration CR, une zone d'eau libre peu profonde s'est formée avec des espèces aquatiques flottantes migrant spontanément et une végétation de type marais sur le périmètre. On a observé que ces types de végétation étaient une source de C, où le CH_4 était libéré par les tissus d'aérenchyme bien développés de ces végétaux. La production de

biomasse et l'accumulation de tourbe ont été observées de façon marginale, sauf dans un tapis de mousse brune flottant. En conséquence, on a observé que l’approche de restauration par CR avait un potentiel de réchauffement global accru, en raison du bilan positif de C, où plus de C est libéré dans l'atmosphère qu'il n'est absorbé par la pédosphère.

PR15 et PR5, qui ont fait l'objet d'introduction d'espèces végétales, nous avons constaté que la diversité et la richesse des espèces étaient les plus faibles parmi les tourbières restaurées. Des conditions trop sèches, avec un niveau d’eau trop profond sous la surface du sol, ont transformé PR5 et PR15 en sources de carbone avec un potentiel de réchauffement global accru, en raison de la libération de CO_2 dans l'atmosphère. La forte production de biomasse a été neutralisée par un taux de décomposition tout aussi élevé et donc par un faible potentiel d'accumulation de tourbe. Le développement des arbustes a eu un effet positif sur l'absorption du carbone.

Nous avons observé que le PR0, qui a été spontanément revégétalisé par la migration naturelle des diaspores, a développé une végétation plus similaire aux tourbières de référence avec une plus grande diversité et richesse d'espèces végétales par rapport aux autres zones restaurées. Une couverture dominante de bryophytes ou une végétation arbustive ont contribué à de meilleurs taux d’accumulation de tourbe par rapport aux autres zones d'étude. Le WTL près de la surface était un facteur significatif pour le retour d'une fonction de puits de carbone dans cette zone restaurée (PR0).

Nous pensons que la restauration écologique de tourbières peut être réalisée avec l'élimination partielle du remplissage minéral. Les résultats suggèrent que la connectivité hydrologique avec les écosystèmes de tourbières adjacents non perturbés est le facteur limitant le plus important pour le développement de communautés végétales caractéristiques des tourbières ou la restauration des fonctions d’accumulation de tourbe et d'absorption de carbone. En outre, la proximité physique d’une banque de diaspores semble faciliter et accélérer la migration naturelle spontanée de diverses espèces végétales, même sur un sol minéral résiduel des anciennes plateformes de forage. L'introduction active de plantes ne s'est pas avérée avoir des effets significatifs sur la diversification et l'enrichissement des communautés végétales caractéristiques des tourbières.

Abstract

Boreal peatlands in the Oil Sands regions of Alberta are subject to vast disturbances and pollution caused by the *in situ* oil and gas extraction industry. At the same time, peatlands are recognized as enormous carbon (C) storing ecosystems that need protection during times of enhanced greenhouse gas emissions and ongoing anthropogenically-caused global warming. Starting in 2015, the Alberta Government released new regulations that aim at the conservation and protection of peatlands following disturbance by *in situ* oil and gas infrastructure via the restoration of an "equivalent land capability". The obligatory ecological restoration aims at the reestablishment of primary peatland functions, such as peat accumulation and C sequestration.

First trials to restore peatlands following *in situ* oil sands well pad disturbances started in the early 2000's and until this day little research on the success of the various restoration techniques has been done. The aim of this dissertation is therefore the evaluation of different peatland restoration techniques following *in situ* oil sands well pad disturbances, via the assessment of the restored peatland's vegetation communities and functions, in particular the peat accumulation potential and return of C sequestration. Three sub-objectives focussed on the development of peatland characteristic plant species, the plant organic matter production and decomposition, the biogeochemistry and carbon sequestration.

The study took place seven to 10 years post-restoration. Research sites were two decommissioned *in situ* oil sands well pads located in the Peace River and Cold Lake Oil Sands regions in northern Alberta. For this study, we selected five restored peatland areas, one unrestored control area of an *in situ* well pad, and 28 undisturbed reference wetlands. The evaluation of the restoration techniques included the complete removal of the *in situ* well pad's construction materials (CR), the partial removal of the well pad's mineral fill to 15 cm (PR15) above the water table level (WTL), to 5 cm above the WTL (PR5), and to near the WTL of the adjacent undisturbed fen ecosystem (PR0). Revegetation happened either spontaneously via natural ingress or was managed by active planting of vascular species, in particular *Carex aquatilis*, *Larix laricina*, and *Salix lutea*. Throughout the two-year study period, we measured the abundance, diversity, and richness of emerging plant communities, the net primary productivity (NPP) and litter decay, as well as net ecosystem exchange (NEE) via carbon dioxide (CO_2) exchange, and methane (CH_4) emissions. Furthermore, we measured environmental factors, such as WTL, soil and water chemistry and nutrient concentrations.

In CR, a shallow open water area had formed with mostly spontaneously colonizing floating aquatic species and marsh-like vegetation in the periphery. This type of vegetation was measured to be a C source, where CH_4 was released via aerenchyma. Biomass production and peat accumulation was observed marginal, except in a floating brown moss carpet. As a result, CR was observed to have an enhanced global warming potential, due to the positive C balance, where more C was released to the atmosphere than was taken up by the pedosphere.

At PR15 and PR5, which were subject to plant species introduction, we found the lowest species diversity and richness among restored peatlands. Too dry conditions, with low WTL below the surface, turned PR5 and PR15 into carbon sources with increased global warming

potential, due to the release of CO_2 to the atmosphere. High biomass production was neutralized by an equally high decay rate resulting in low peat accumulation potentials. There was a positive relationship between shrub cover and net carbon uptake.

We observed PR0, which was spontaneously revegetated by natural migration of diaspores, to develop fen characteristic vegetation with the highest plant species diversity and richness compared to other restored areas. Either dominant bryophyte cover or shrub vegetation helped contribute to the greatest peat accumulation potential compared to the other study areas. The WTL at the surface was a significant factor for returning a C sink function in the same restored area.

Results indicate that the benefit of the complete removal of a former *in situ* oil sands well pad is negligible, and that ecological peatland restoration can be achieved with the partial removal of the mineral fill. Also, hydrological connectivity to undisturbed adjacent fen ecosystems is the most important limiting factor for the development vegetation communities characteristic of peatlands and resume peat accumulation and C uptake. Furthermore, the physical proximity to the respective diaspore bank is believed to facilitate and accelerate spontaneous natural migration of diverse plant species even on a residual mineral soil. Active plant introduction did not prove to have significant effects on diversification and enrichment of peatland characteristic plant communities.

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List of abbreviations and acronyms

Abbreviation	Meaning
AG	Above-ground
BG	Below-ground
BOG	Wooded bog (study area consisting of grouped wetlands and individual study area)
BP	Before present
CA	Correspondence analysis
CR	Complete removal (of the mineral fill and underlying geotextile of an inactive and decommissioned *in situ* oil sands well pad, study area)
CR-D	CR, dry microform, floating moss carpet (study area)
CR-W	CR, wet microform, shallow open water area (study area)
DOC	Dissolved organic carbon
EC	Electrical conductivity
GEP	Gross ecosystem productivity
GPmax	Gross photosynthesis
GRF	Graminoid rich fen (study area consisting of grouped wetlands)
GWP	Global warming potential
M	Marsh (study area)
NEE	Net ecosystem exchange
NEP	Net ecosystem production
NONW	Non-wetland species
NPP	Net primary productivity
OTHW	Other wetland species
PAR	Photosynthetically active radiation
PCA	Principal component analysis
PEAT	Peatland species
PR	Partial removal (of the mineral fill of an *in situ* oil sands well pad)
PR0	PR to 0 cm above the water table level/level with adjacent peatland (study area)
PR0-D	PR0, dry microform (study area)
PR0-W	PR0, wet microform (study area)
PR0E	PR0, even lawn (study area)
PR15	Partial removal to 15 cm above the water table level (study area)
PR5	Partial removal to 4 to 6 cm above the water table level (study area)
RDA	Redundancy analysis
R_{eco}	Ecosystem respiration
REF	Reference peatland/ecosystem
RES	Restored area

sp.	Species (without any further identification to species level)
SPF	Shrubby poor fen (study area consisting of grouped wetlands)
spp.	More than one species
SRF	Shrubby rich fen (study area consisting of grouped wetlands and individual study area)
ssp.	Subspecies
ST5	Soil temperature at 5 cm depth
TRF	Treed rich fen (study area)
UNR	Unrestored (study area)
WIS	Wetland indicator status
WRF	Wooded rich fen (study area consisting of grouped wetlands)
WTL	Water table level

Chemical elements	
Al	Aluminum
C	Carbon
Ca	Calcium
CH_4	Methane
CO_2	Carbon dioxide
Fe^+	Iron
K	Potassium
Mg	Magnesium
Mn	Manganese
N	Nitrogen
$N\text{-}NH_4^+$	Ammonium
$N\text{-}NO_3^-$	Nitrate
Na	Sodium
NO_2	Nitrogen dioxide
$P\text{-}PO_4^{3-}$	Phosphate
$S\text{-}SO_4^{2-}$	Sulphate

In Gedenken an Paps

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Conflict of interest

Here I acknowledge that a conflict of interest may exist, because the founding for this research project was provided by the *National Sciences and Engineering Research Council Collaborative Research and Development Grant* (CRDPJ no. 501619-16) to LR, MS and BX in an industrial partnership collaboration with Imperial Oil Resources Limited.

Two industrial collaborators, Imperial Oil Resources Limited and Canadian Natural Resource Limited provided access to the research sites, personal protective equipment, on-site safety guidelines and measures. Unless there were safety concerns interfering with the researcher's plan of action, the collaborating companies did not contribute to any decision making regarding the research work (such as developing the research design, methods of measurements, data analysis, the preparation of the manuscripts). Neither I nor any of my co-authors have any shares in the collaborating companies, nor do we maintain any consulting contracts with them. All chapters were prepared expressing our free opinions and statements, and publications were done independently. Suggestions were made without any pressure or influence by the petrol companies.

Foreword

The research subject of this dissertation to obtain the degree Doctor of Philosophy, is the evaluation of peatland restoration following disturbances by the *in situ* ("in place") oil sands well pad infrastructures in Northern Alberta.

The complete doctorate thesis is written as a scientific article-based thesis, including a general introduction and conclusion. All chapters were prepared by me as fist author and always in cooperation with my thesis director, Professor Line Rochefort[1,2] and my co-director Professor Maria Strack[3]. LR and MS both supported me during all steps of each manuscript's preparation. Further support was given by Bin Xu[4] during the field measurements and the development of the research designs throughout 2017 and 2018 and in the preparation of the manuscript for chapter 1.

A general **introduction** and a literature review give a background of pertinent information and an overview about the restoration work done so far in this field of research.

Chapter 1 was published on April 28, 2022, in the scientific journal *Restoration Ecology*, as a contribution to the Joint Feature by the Society of Ecological Restoration and the British Ecological Society on the 'UN Decade of Ecosystem Restoration': Lemmer, Meike, Bin Xu, Maria Strack, and Line Rochefort (2021) Reestablishment of peatland vegetation following surface levelling of decommissioned *in situ* oil mining infrastructures. Restoration Ecology, doi: 10.1111/rec.13714. ML and BX developed the research design. LR made suggestions in the development of the sampling design, LR, MS, BX coordinated the funding, supervised the study, and reviewed the manuscript.

Chapter 2 was published in the scientific journal *Frontiers in Earth Science* on November 30, 2020, as a part of the research topic 'Wetland Ecology and Biogeochemistry Under Natural and Human Disturbance': Lemmer, Meike, Line Rochefort, and Maria Strack (2020) Greenhouse Gas Emissions Dynamics in Restored Fens After In-Situ Oil Sands Well Pad Disturbances of Canadian Boreal Peatlands. Frontiers in Earth Science (8): 1-21, doi: 10.3389/faert.2020.557943. LR and MS developed the research design, coordinated the funding, supervised the study, and reviewed the manuscript.

Chapter 3 consists of an unsubmitted/unpublished chapter about the organic matter productivity and decomposition in restored fens following *in situ* well pad disturbances. ML,

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BX, and LR developed the research design. LR, MS, BX coordinated the funding. LR and MS supervised the study.

A general **conclusion** closes the evaluation of the studied restoration techniques and provides suggestions for further ecological restoration work to be done following disturbances by *in situ* oil and gas extraction.

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- 9th World Conference on Ecological Restoration "A new global trajectory", 21-24 June 2021, oral presentation, Society of Ecological Restoration, virtual.
- Mer Bleue & Beyond Virtual Symposium 2021, 19-21 May 2021, oral presentation, McGill University, virtual.
- 2021 Virtual Annual General Meeting & Conference "Looking forward", 24-25 February 2021, oral presentation, Canadian Land Reclamation Association Alberta Chapter, virtual.
- CEN 2021 Annual Spring Symposium, 11-12 February 2021, poster, Centre d'études nordiques, Université Laval, virtual.
- SWS Virtual Meeting "Wetland connections over 40 years", 1-3 December 2020, oral flash presentation, Society of Wetland Scientists, virtual.
- 26th PERG Symposium & McGill Carbon team Annual Meeting, 21 February 2020, poster, Peatland Ecology Research Group and McGill Carbon Team, McGill University, Montreal, QC.
- CEN 2020 Annual Spring Symposium, 12 February 2020, poster and oral flash presentation, Centre d'études nordiques, Université du Québec à Montréal, Montreal, QC.

Introduction

Pertinence

The Canadian landmass is covered by roughly 12.5% peatlands, which make up 90% of the country's wetlands (Xu et al. 2018). Specifically, Canada's boreal ecoregion is characterized by approximately 1 136 million km^2 of peatlands within a vast mosaic of pristine conifer forests and wetlands (Tarnocai 2006). In the province of Alberta, these wetlands coincide with the world's third largest oil reserves situated within the Western Canadian Sedimentary Basin, called the Oil Sands regions of Alberta (Government of Canada 2019). Covering an area of more than 140 000 km^2 and containing an estimated total of 166.3 billion oil barrels, the Oil Sands regions are subject to extensive bitumen and natural gas mining operations to satisfy the high demand for petroleum and energy (Orbach 2012; ABMI 2018; Government of Alberta 2021b). Severe disturbances of natural ecosystems, particularly peatlands, are the consequence.

Open surface mining of the oil sands, a mix of sand, water and heavy, sticky oil called bitumen, is only feasible when deposits are located at surface-near depth up to 75 m and is accountable for the degradation of approximately 953 km^2 surface area in the Athabasca Oil Sands region as of 2019 (Government of Alberta 2019). A total of 4 890 km^2 is considered open surface mineable land area (ABMI 2018). *In situ* ("in place") extraction on the other hand, is used to extract bitumen from depths greater than 75 m, while being responsible for 97% (approximately 138,000 km^2) of the overall oil sands related disturbance in the region (Schneider & Dyer 2006; Orbach 2012; Government of Alberta 2019). Bitumen deposits in the Cold Lake Oil Sands on the border to Saskatchewan are located at 300-600 m depth, and in the Peace River Oil Sands in the northwest of Alberta at 300-770 m depth, making *in situ* mining the mandatory extraction method (Government of Alberta 2021b). *In situ* mining operations require a vast network of cutlines, access roads, mineral soil extraction pits, exploration pads, *in situ* oil sands well pads, pipelines, processing plants, and storage facilities for bitumen and crude oil. As a result, peatlands in the same regions are largely impacted by fragmentation, drainage, pollution, and exploitation (Turetsky & St. Louis 2006; Graf 2009; Brandt et al. 2013; Pasher et al. 2013; Strack et al. 2019).

Recognizing the serious magnitude of peatland degradation in the province, the Government of Alberta released in 2013 the Alberta Wetland Policy to improve peatland protection, conservation, management, and restoration in compliance with the Environmental Protection and Enhancement Act (Government of Alberta 2013; Province of Alberta 2021). The evaluation of emerging novel peatland restoration techniques following the disturbances caused by *in situ* bitumen mining infrastructure is the subject of this study.

Besides an extensive infrastructure of connecting roads and pipelines, the main disturbances caused by the *in situ* oil and gas industry are the construction of numerous *in situ* oil sands well pads scattered across the boreal forest. As of October 2021, there are more than 157 000 active oil wells in the Peace River and Cold Lake Oil Sands regions of Alberta (Figure 0.1; AER 2021). At the same time, there are an additional 172 000 inactive and abandoned wells (Government of Alberta 2021a). Because the swampy ground needs to become stable and firm to support the mining equipment, *in situ* well pads serve as a secure platform for installing well heads, pumping jacks, and processing facilities on top the saturated peat. During the construction of an *in situ* well pad, trees and larger shrubs are cut and left in place, then all remaining vegetation is covered with a geotextile before a mix of mineral fill containing loam, clay, sand, and gravel is placed and solidified. The thickness of the final compacted mineral fill platform varies between 1.5 and 4 m (Figure 0.2). The well pads' sizes depend on the number of well heads installed and range from a minimum of 1 ha to as large as 4 ha (Figure 0.1). The average lifespan of an *in situ* oil sands well pad is about 20 to 30 years, before decommissioning and reclamation (CAPP 2021).

Figure 0.1 *In situ oil sands well pad made of compacted mineral fill situated within a mosaic of boreal forest and wetland ecosystem. This well pad in the Cold Lake Oil Sands region is sized approximately 1.8 ha and supports 27 well heads, each one connected to an oil pump ("pumping jack").*

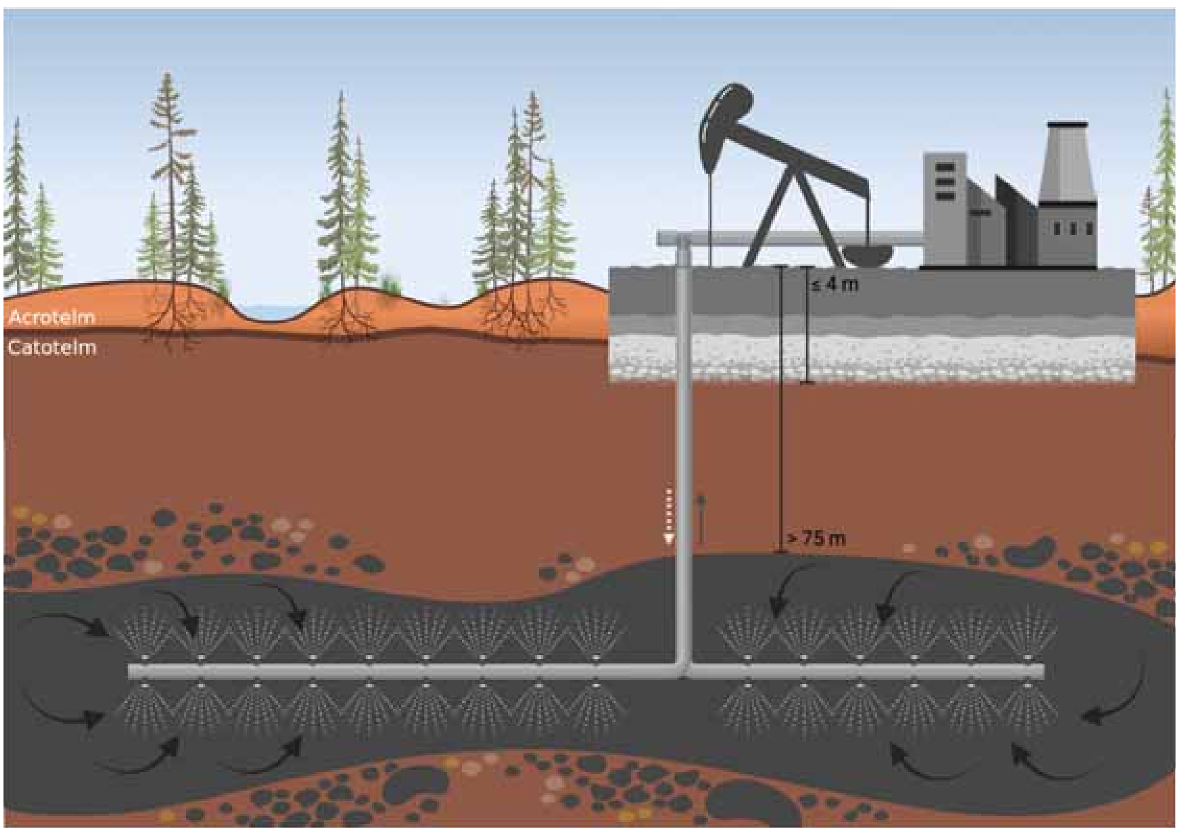

Figure 0.2 *Schematic cross section of an in situ oil sands well pad installed within a peatland ecosystem. The well pad is made up of a compacted mix of loam, clay, sand, and gravel. The thickness of the mineral soil varies between 1.5 m to 4 m in order to support the in situ bitumen extraction equipment despite the swampy ground of saturated peat. In situ oil sands well pads are used to extract bitumen from deposits located at depth greater than 75 m. In the Cold Lake Oil Sands bitumen deposits are situated about 300 to 600 m below-ground and in the Peace River Oil Sands at depth between 300-770 m (Government of Alberta 2021b).*

Definition and aim of ecological restoration

Ecological restoration is a rather new field of study, that has emerged during the last 40 years with a quick increase of importance, as global anthropogenic pressure on natural ecosystems intensifies (Charman 2002; Choi et al. 2008; Comín 2010). Ecological restoration has been defined as “the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed” (Clewell et al. 2004:3; Gann et al. 2019). The process includes the reestablishment of the ecosystem’s characteristic functions and attributes prior to degradation and the initiation of natural succession, to achieve complete self-sustainability and full functionality (Clewell et al. 2004).

In order to satisfy individual management plans and divergent restoration objectives, other comparable terms such as *regeneration*, *rehabilitation*, *reclamation*, and *creation* have been developed (Table 0.1 Comparison and meaning of terms used for restoration methods and respective contexts, their aim and outcome ([1]Clewell et al. 2004; [2]Rydin & Jeglum 2013; [3]Environment and Parks 2017; [4]Gerwing et al. 2021). Clewell et al. 2004; Gerwing et al. 2021). Considering the example of ecological restoration of peatlands, differences occur in the anticipated results and time frame for success of the various approaches (Figure 0.3). *Regeneration* aims at the power of nature to restore itself as soon as we cease the interruption and disturbance. This approach nurtures the development of some terrestrial ecosystem via natural evolution during the time post-disturbance but does not target peatlands or wetlands as the ultimate restoration outcome, although it can occur sporadically (Poulin et al. 2005). In contrast, *rehabilitation* aims at restoring functional wetlands including a more sophisticated and planned work effort in order to push the ecosystem succession in the direction towards becoming a wetland. Rehabilitation efforts, however, might favor a different, possibly non-native biota, and the reinstallation of a different wetland ecosystem without the specific peatland ecosystem functions (Gann et al. 2019). Nevertheless, under the right circumstanced, a rehabilitated wetland might someday become a peatland. In the context of restoration following the *in situ* oil mining activities in the Oil Sands regions, rehabilitation is applied, using here the term *reclamation* instead. While reclamation has been defined for restoration involving land-use change, reclamation in the context of oil sands disturbances aims at establishing an “*equivalent land capability”* (Bradshaw & Chadwick

1980; Government of Alberta 2013; Rydin & Jeglum 2013; Environment and Parks 2017). Equivalent land capability is defined as "the ability of the land to support various land uses after conservation and reclamation 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" but instead have the same value, purpose, and qualities (Environment and Parks 2017). Alternatively, *ecological restoration* precisely targets the development of a functional peatland ecosystem as a goal within several years, depending on the severity of the disturbance. An ecologically restored peatland is the product of an extensive restoration working plan, precisely focusing on the ecosystem's successional state prior disturbance. The goal is to revive peatland functions and processes, and abiotic and biotic characteristics within the years following the restoration (McDonald et al. 2016; Palmer et al. 2016).

Of the biota, only the flora will be considered in this research. Indigenous knowledge, historical reports, and data from regional reference ecosystems help to obtain data on typical indigenous vegetation species, plant functional groups and community structures (Temperton et al. 2004). Specifically targeted peatland functions include water filtering, hydrological connection, wildlife habitat, peat accumulation, and carbon sequestration (Rydin & Jeglum 2013). Throughout this dissertation, the simple englobing term *restoration* will be used as equivalent for ecological restoration and all its synonyms, which we consider to be interchangeable in this context (Clewell et al. 2004).

Table 0.1 Comparison and meaning of terms used for restoration methods and respective contexts, their aim and outcome ([1]Clewell et al. 2004; [2]Rydin & Jeglum 2013; [3]Environment and Parks 2017; [4]Gerwing et al. 2021).

Restoration method	Restoration aim	Restoration outcome	Context	Source
Ecological restoration	Reestablishment of the ecosystem's characteristic functions and attributes prior degradation	Peatland ecosystem	Degraded, damaged, or destroyed ecosystem	1, 2, 4
Reclamation	Establishment of an „equivalent land capability“	Various, non-identical but similar land uses and ecosystem functions	*In situ* oil sands disturbances	1, 3, 4
Regeneration	Self-healing of the ecosystem	Terrestrial functional ecosystem	Various	1, 2
Rehabilitation	Restoring functioning ecosystem succession at an early state	Wetland ecosystem (mineral or organic)	Various	1, 4
Creation	Establishment of the ecosystem's characteristic functions and attributes	Artificially constructed peatland	Open pit oil sands mining	1, 2

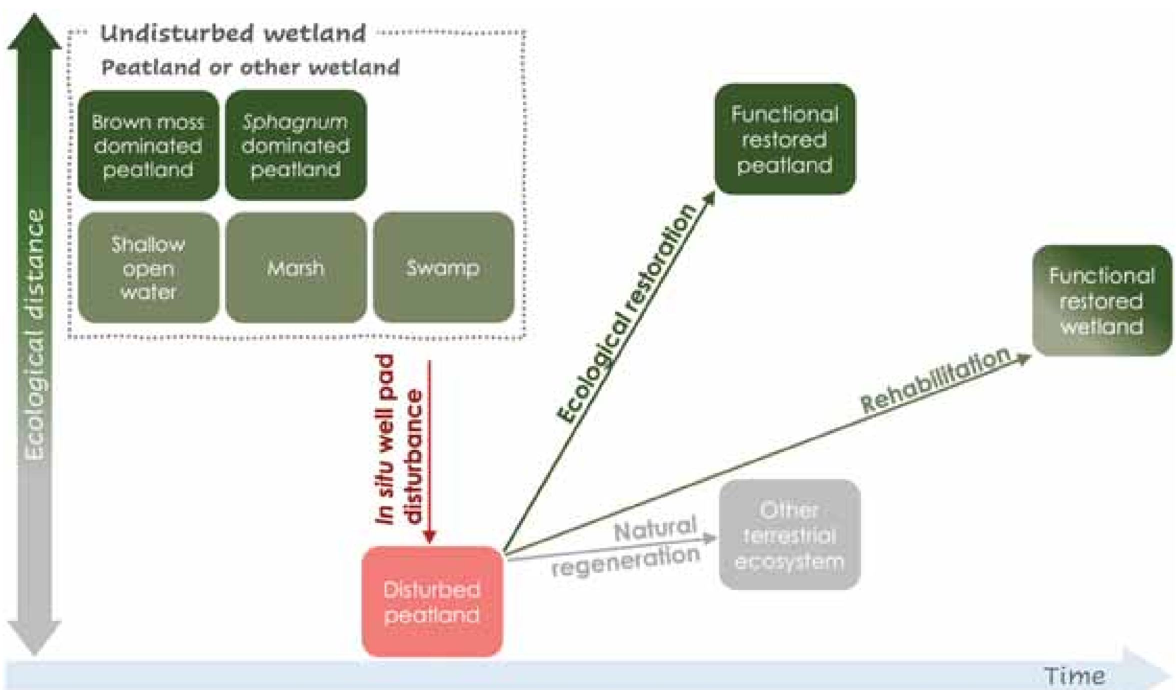

Figure 0.3 Conceptual schema of diverse pathways that the different management approaches "natural regeneration", "rehabilitation/reclamation", and "ecological restoration" may inflict on peatland ecosystem restoration following in situ oil sands well pads disturbances over time. An ecologically restored peatland's processes and functions are comparable to those found in undisturbed reference peatlands. Other management approaches may not lead to a full recovery of peatland functions and mineral wetlands form with unpredictable peat accumulation processes, i. e. in a marsh, a swamp, or shallow open water.

In order to evaluate restoration success, Stander and Ehrenfeld (2009) argue that precise definition of the anticipated restoration outcome(s) is needed, but too narrowly defined goals might not reflect the entire spectrum of wetland development. In general, it is very difficult to define the successional pathway and timeframe for wetland development and peatland formation since this process is known to last several hundreds to thousands of years (Kuhry et al. 1992, 1993; Halsey et al. 1998). In this context, the natural pathway also encompasses the different states along the natural successional pathway post recovery of a natural disturbance, such as fire, flooding, or drought. As a result, if the assessment period is too short, the restored ecosystem might be unable to meet restoration goals that are too complex and too narrowly defined (DeSteven et al. 2010). Accepting a variety of possible restoration outcomes allows the acknowledgement of unpredictable dynamics and their consideration in adjusting management plans, granting the ecosystem time to achieve its full potential.

Restoration approaches applied in the Alberta Oil Sands region

The peatland and wetland restoration work in the context of the Canadian oil and gas industry began in the late 2000's and is still developing and constantly improving. In order to learn from trial and error, several different restoration approaches were tested following the severe peatland disturbances by *in situ* oil and gas well pads, because solely the rewetting of disturbed peatlands does not always lead to effective restoration of characteristic peat properties and more reliable methods are needed for higher success rates (Kreyling et al. 2021). In accordance with the Environmental Protection and Enhancement Act and the Alberta Wetland Policy's focus on the conservation and restoration of disturbed peatlands, the oil and gas industry adjusted their practices and implemented peatland reclamation in their agenda (Alberta Environment 2008; CPP Environmental 2017). In this context, the aim of peatland reclamation is to ensure functional processes similar to those prior disturbances, including water storage/filtration, wildlife habitat, peat accumulation and carbon sequestration (Environment and Parks 2017) and can therefore be considered restoration as defined in Table 0.1.

One of the first studies of peatland restoration following *in situ* well pad disturbances in the Canadian Oil Sands regions aimed at the initiation of fen ecosystems on residual mineral fill.

Following the decommissioning in 2000, the former well pad in the Peace River Oil Sands had been seeded with *Melilotus albus* Medikus and *M. officinalis* (Linnaeus) Lamarck (

Figure **0.4**B)*,* while the peatland restoration starting in 2007 intended to mimic the processes of paludification on mineral soil comparable to the first peatland formations following the glacial era (Halsey et al. 1998; Vitt et al. 2011). The study included different restoration approaches, combining the reduction in thickness of the mineral fill and revegetation. In order to adjust the upper surface to the water table level of the adjacent bog and to reinstate characteristic hydrological conditions of minerotrophic peatlands, the mineral fill was partially removed and scraped down to 15 cm above the average water table level in one study area, and to 4-6 cm above the average water table level in another study area (Figure 0.4C; Vitt et al. 2011; Koropchak et al. 2012). In both areas, several different soil amendments, including the spread of a peat layer, were tested, in addition to plantings of *Carex aquatilis* Wahlenberg, *Larix laricina* (Du Roi) K. Koch, and *Salix lutea* Nuttall (Vitt et al. 2011; Koropchak et al. 2012). Three years post-restoration, the residual mineral fill that was left in the ground following the partial removal restoration treatment effectively supported the development of abundant *C. aquatilis* Wahlenberg communities, a representative species for rich fen ecosystems, whereas the other soil amendments, including peat, were observed to foster a high cover of undesired weeds (Koropchak et al. 2012).

In 2008, a study in the Cold Lake Oil Sands tested a restoration approach of aiming to rehabilitate pre-disturbance conditions, in a treed rich fen ecosystem (Imperial Oil Resources 2017, personal communication). The restoration approach consisted of the complete removal of the disturbance in combination with spontaneous revegetation via natural ingress from nearby diaspores (Imperial Oil Resources 2017, personal communication). However, unexpected severe compaction of the peat buried under the mineral fill created a depression below the average table and a shallow open water area quickly formed in the depression during ongoing restoration work (

Figure 0.4E). To avoid the development of shallow open water and instead support the fen restoration, restoration work continued to only partially remove the upper surface layers of the mineral fill to near the water table level. Hence, a residual mineral fill and the underlying geotextile were left in place and in the end, two restoration approaches were tested in this study, the complete and the partial removal of the mineral fill. Four years post-restoration, the shallow open water area contained floating and emergent aquatic vegetation and indicators predicted the succession towards a marsh ecosystem. Three years post-restoration, the partial removal area supported approximately 36 to 95% vegetation cover dominated by *Typha latifolia* L., *Equisetum* sp. L., *Carex* sp. L., and *Salix* sp. L..

In 2009, a study in the Peace River Oil Sands tested the restoration of a treed rich fen vegetation community and a shrubby rich fen vegetation community on a partially removed and remodeled former *in situ* oil sands well pad (Gauthier et al. 2018). Restoration work included the partial removal of the well pad's mineral fill following the technique applied by Vitt and colleagues two years earlier (2011). A residual mineral fill of 20 to 25 cm remained atop the underlying peat (Gauthier et al. 2018). Five substrate treatment were tested, including 1) clay loam (residual well pad mineral fill), 2) decompacted clay loam, 3) sawdust-clay mix, 4) decompacted sawdust-clay mix, and 5) peat. The revegetation was done following the moss layer transfer technique (MLLTT; Quinty & Rochefort 2003), using donor material from a shrubby rich fen and a treed rich fen (Gauthier et al. 2018). One growing season post-restoration, the vascular plants covered 3 to 12%, while bryophytes covered up to 58% of the study area. The bryophytes transferred from the shrubby rich fen were observed to cover almost twice as much (58%), compared to the bryophytes transferred from the treed rich fen (30%; Gauthier et al. 2018).

Starting in 2010, a study in the Cold Lake Oil Sands tested the restoration of a treed poor fen vegetation on partially removed mineral fill mixed with the underlying well decomposed peat, following *in situ* oil sands disturbances (Shunina et al. 2016). Several restoration and plant introduction approaches were tested. Ground treatments included the removal of the mineral fill except a layer of about 10 cm, which was then mixed with the underlying peat. Soil modeling included a smooth soil treatment with a relief of up to 15 cm, and a rough treatment with a relief of up to 1 m (Shunina et al. 2016). Revegetation treatments began in 2011 and incorporated first a modified MLTT (Quinty & Rochefort 2003), secondly a spontaneous revegetation via natural ingress, and thirdly in 2012 the active transplantation of *Picea mariana* (Miller) Britton, Sterns & Poggenburgh, *Rhododendron groenlandicum*

(Oeder) Kron & Judd, and various unidentified sedge seedlings, which later developed into abundant *C. aquatilis* Wahlenberg and *C. utriculata* Boott (Shunina et al. 2016). Two growing seasons post-restoration, the tree and shrub transplants survival was lower than 20%, while sedge transplants presented a survival rate of more than 98%. *Populus tremuloides* Michaux and *Beckmannia syzigachne* (Steudel) Fernald were observed as abundant species that had spontaneously migrated (Shunina et al. 2016).

In 2011, another study in the Peace River Oil Sands tested the restoration of a *Sphagnum* L.-dominated vegetation on resurfaced peat following *in situ* oil sands well pad disturbances. Restoration approaches incorporated in this study included the resurfacing of peat following the complete removal of the mineral fill, the inversion of the underlying peat with a residual mineral fill layer, and revegetation via the MLTT (Quinty & Rochefort 2003; Xu et al. 2021). Revegetation methods follow the successful practice of peatland restoration following cutover peatland disturbances in eastern Canada (Rochefort et al. 2003). Because the hydraulic conductivity, physical and chemical properties of peat change when buried underneath a solidified *in situ* mineral well pads (Daly et al. 2012), the leader of the project (L. Rochefort) incorporated active groundwork and the use of an excavator to decompress the exposed and resurfaced peat. One restoration approach had the former well pad's mineral fill completely removed, and the underlying peat exposed, mechanically decompressed, remodeled to the same surface elevation as the surrounding peatland ecosystem, and revegetated via the MLTT (Xu et al. 2021). In another restoration approach the upper layer of the mineral fill was used to compensate the elevation difference due to peat compaction, and the residual mineral fill was inverted with the underlying peat (i.e., it was buried under peat, exposing peat at the surface), before decompaction and revegetation was done (Figure 0.4D; Xu et al. 2021). Three years post-restoration, the total vegetation cover (>65%) was dominated by peatland characteristic vegetation (>63%), especially *Carex* spp. L., including 8% fen characteristic brown mosses, and more than 3% *Sphagnum* spp. L. (Xu et al. 2021).

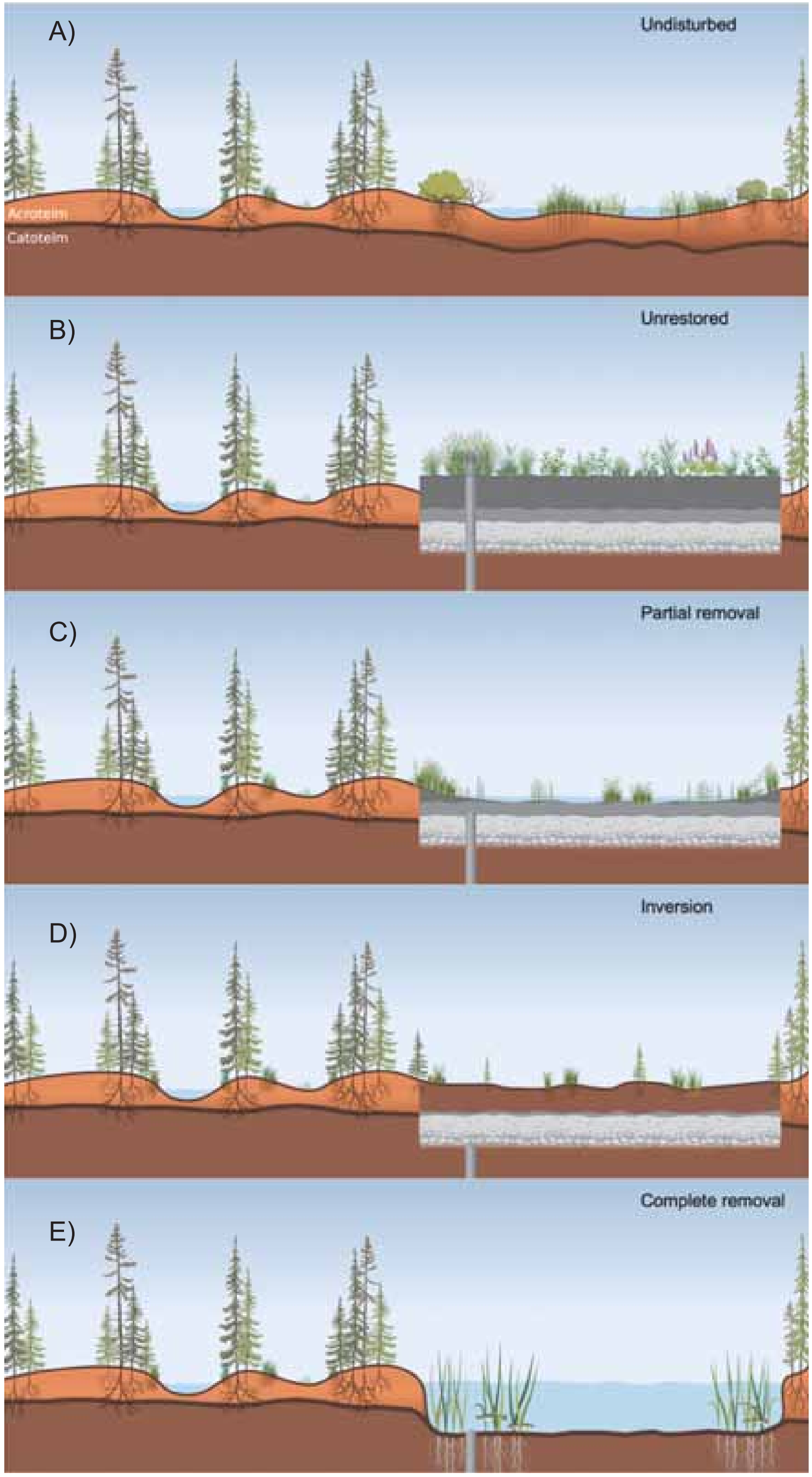

Figure 0.4 Schematic cross section of A) a pristine peatland before disturbance, in comparison to the B) unrestored peatland disturbed by an decommissioned/ abandoned in situ oil sands well pad, and peatlands restored via C) the partial removal of the mineral fill, via D) the inversion of the mineral fill and the underlying peat, and via E) the complete removal of the former in situ well pad, where a shallow open water area formed instead.

Peatland functions

Wetlands are defined as "land that is saturated with water long enough to promote wetland or aquatic processes as indicated by poorly drained soils, hydrophytic vegetation and various kinds of biological activity which are adapted to a wet environment" (National Wetlands Working Group 1997). Hydrology is the limiting factor for an ecosystem to become a wetland with characteristic functions and processes. Suitable hydrological conditions support a typical wildlife habitat for characteristic vegetation communities that provide the organic matter content for peat accumulation and subsequent carbon (C) sequestration (Kroetsch et al. 2011).

Differences occur between mineral and organic wetlands. Mineral wetlands, such as swamps, marshes, and shallow open water, are characterized by generally circumneutral to alkaline conditions, higher salinity and higher nutrient concentrations (National Wetlands Working Group 1997). Mineral wetlands do per definition not accumulate substantial layers of organic matter, although they can potentially accumulate peat layers thicker than 40 cm. On the contrary, organic wetlands also known as peatlands, such as fens and bogs (Table 0.2), must have a peat layer of at least 40 cm (National Wetlands Working Group 1997; Alberta Environment and Sustainable Resource Development 2015). Peatlands have been observed to accumulate significant peat layers of up to 10 m (National Wetlands Working Group 1997; Alberta Environment and Sustainable Resource Development 2015). Ongoing peat formation in waterlogged conditions is essential for the CO_2 uptake from the atmosphere via photosynthesis of peatland vegetation and continuous C sequestration within the peat layers. Hence, due to the different goals of the various restoration approaches that were discussed earlier, there is an importance to focus specifically on restoration efforts that revive peatland functions, particularly when the pre-disturbance ecosystem was a peatland.

Table 0.2 Parameters and rates defining Canadian peatland types, including the chemistry (pH and electrical conductivity "EC"), net primary productivity (NPP; above-ground "AG" and below-ground "BG" biomass), net ecosystem exchange (NEE), and methane (CH4) emissions (Sjörs 1950; Chee & Vitt 1989; Eggelsmann 1990; Zoltai & Vitt 1995; [a] Szumigalski & Bayley 1996a; [b] Saarinen 1996; [c] Campbell et al. 2000; Charman 2002; Sjörs & Gunnarsson 2002; [d] Strack et al. 2008; [e] Vitt et al. 2009; [f] Strack et al. 2014; Ducks Unlimited Canada 2011; Joosten 2016; [g] Bérubé & Rochefort 2018).

Wetland class	**Peatland type**	**Nutrient status**	**Chemistry**	**NPP ($g\ m^{-2}\ yr^{-1}$)**	**NEE ($g\ CO_2\ m^{-2}\ yr^{-1}$)**	**CH_4 emissions ($g\ CH_4\ m^{-2}\ yr^{-1}$)**
Bog		oligotrophic	acidic pH 3.7 to 4.2 EC 10 to 30 µS/cm	AG 42 to 1 118 [a, c] BG 70 to 1 461 [c]	-7 to -411 [d]	4 to 6 [d, f]
Fen	Poor fen	minerotrophic to oligotrophic	acidic pH 3.8 to 5.0 EC 16 to 45 µS/cm	AG 114 to 310 [c] BG 396 [c]	-359 [d]	3 to 23 [d]
		mesotrophic to minerotrophic	acidic-neutral pH 4.8 to 5.7 EC <100 µS/cm	AG 310 to 325 [c] BG 129 to 297 [c]		4 to 18 [d]
	Moderate-rich fen	Mesotrophic	neutral pH 5.2 to 6.4 EC 100 to 250 µS/cm	AG 176 to 214 [b, c, e] BG <1 248 [b, g]		
		eutrophic to mesotrophic	neutral-alkaline pH 5.8 to 7.0 EC 48 to 374 µS/cm	AG 316 to 360 [a, e] BG 290 to 1 248 [g]	-207 to -282 [d]	3 to 40 [d]
	Extreme-rich fen	eutrophic	alkaline pH 7.0 to 8.4 EC 250 to 2 000 µS/cm	AG 245 to 341 [a, e]	-34.5 to -153.5 [d]	

Habitat

Peatland ecosystems provide habitat for characteristic plant species, specifically hydrophytes, species adjusted to wet conditions that either merely tolerate or have well adapted to the extreme conditions found in wetlands (Rydin & Jeglum 2013). In fact, the hydrology is considered to be the base for the ecological structures in peatlands and their respective functions and processes (Richardson et al. 2016). A peatland's soil quality, plant species composition, and vegetation productivity are impacted by the climatic factors such as precipitation rate and temperature (Bernard et al. 1988; Breeuwer et al. 2008; Jassey et al. 2013; Churchill et al. 2015; Mäkiranta et al. 2018) and are limited by the availability and composition of nutrients (Craft & Richardson 1997). Compared to bogs, which are habitat to few characteristic species, fens have in general higher and more variable water tables, higher minerotrophic concentrations and therefore their vegetation is composed of a wide variety of communities with variable vascular and bryophytes species (Succow & Joosten 2001; Warner & Asada 2006). The scientific names of plant species mentioned in this thesis follow the Integrated Taxonomic Information System (IT IS 2021) and the Consortium of North American Lichen Herbaria (CNALH 2021).

Bogs are known to be ombrogenous, oligotrophic and rather acidic ecosystems. Ombrogenous peatlands are disconnected from the upwelling or lateral below-ground water flow and depend on precipitation for water supply and nutrient input from the atmosphere. A continental bogs' vegetation composition is dominated by various bryophyte species, ericaceous shrubs, cottongrass (*Eriophorum* ssp. L.), and coniferous trees, predominantly black spruce (*Picea mariana* (Miller) Britton, Stern & Poggenburgh). Dominant bryophytes found in bogs are abundant peat moss species, like *Sphagnum capillifolium* (Ehrh.) Hedw., *S. medium* Limpr., and *S. fuscum* (Schimp.) H. Klinggr., and few brown moss species, such as *Hylocomium splendens* (Hedw.) Schimp., *Pleurozium schreberi* (Willd. ex Brid.) Mitt., *Pohlia nutans* (Hedw.) Lindb., and *Polytrichum strictum* Menzies ex. Brid. (Vitt & Lüth 2017). Prominent ericaceous shrubs are *Rhododendron groenlandicum* (Oeder) Krohn & Judd, *Chamaedaphne calyculata* (Linnaeus) Moench, *Empetrum nigrum* L., *Kalmia polifolia* Wangenheim, and *Vaccinium* spp. L. (Zoltai et al. 1988).

In contrast, fens are minerogenous ecosystems that are connected to a gradual, slow moving lateral and surface or upwelling groundwater flow (Alberta Environment and Sustainable Resource Development 2015). The transport and exchange of nutrients and minerals from neighboring ecosystems and the underlying geology is the reason for the chemical variability of fens from oligotrophic to eutrophic (Ducks Unlimited Canada 2011). As a result, fens support much more diverse plant species communities compared to bogs. Dominant vascular species include sedges like *Scirpus* ssp. L., *Carex* ssp. L. and *Eleocharis* ssp. R. Brown, forbs like *Caltha palustris* L., *Comarum palustre* L., and *Menyanthes trifoliata* L., as well as trees and shrubs, such as *L. laricina* (Du Roi) K. Koch, *Betula pumila* L., *Myrica gale* L., and *Salix* spp. (Zoltai et al. 1988; Alberta Environment and Sustainable Resource Development 2015). Characteristic bryophyte species include mostly brown mosses, such as *Drepanocladus* spp. (Müll. Hal.) G. Roth, *Scorpidium scorpioides* (Hedw.) Limpr. or *S. cossonii* (Schimp.) Hedenäs, *Campylium stellatum* (Hedw.) C.E.O. Jensen, *Calliergon* spp. (Sull.) Kindb., and *Tomentypnum nitens* (Hedw.) Loeske (Vitt & Lüth 2017). We focus in particular on the characteristic fen vegetation composition and the dominant fen plant species as a key component of peatland ecosystems, to evaluate the outcome of the peatland restoration approaches in the course of this research.

Peat accumulation

Peat is a histosol and is defined as a soil with more than 30% organic matter content (Lindsay & Andersen 2018). The high organic matter content builds up due to reduced decomposition of the plant litter in waterlogged, constraining conditions. Peat can undergo varying degrees of decomposition from fibric, low degree of decay with visible residual plants or parts of plants, to hemic, medium degree of decomposition with only recalcitrant plants and parts of plants still visible, and sapric, high degree of decay without visible differentiation of plants possible (Clymo 1983). Considering the simplest model, a peatland's peat body can be separated into two functional layers, the upper, occasionally aerated acrotelm and the lower, constantly waterlogged catotelm (Ivanov 1953; Ingram 1978; Morris et al. 2011). The acrotelm is characterized by less humified fibric to hemic peat and is situated in the perimeter of the changing water table level. The catotelm remains continuously inundated and contains mostly hemic to sapric peat due to ongoing slow decomposition, which is hampered by oxygen deprivation and low temperatures. Waterlogged conditions delay the rate of decay and enable peat accumulation, but there is no linearity between peat accumulation and soil moisture or rather water table level (Gallego-Sala et al. 2018). Instead, the water table is known to be limiting the migration, composition, and survival of peatland characteristic plant species which are the essential organic matter input for peat accumulation (Murphy & Moore 2010; Wagg et al. 2017; Mäkiranta et al. 2018; Bengtsson et al. 2021). In fact, the peat accumulation potential of wetlands with high water table levels and distinctive water table fluctuations, i.e., shallow open water and marshes, is influenced by many factors, such as quality of plant litter, aeration, or nutrient access for microbes (Thormann et al. 1999; Kreyling et al. 2021).

A peatland's net primary productivity (NPP; the difference between the gross primary productivity and plant respiration) of characteristic vegetation communities dictates the biomass input, which in turn is needed for peat accumulation. Besides the coverage of the characteristic peatland plant functional types, like trees, shrubs, herbs, and bryophytes, differences in peat accumulation rate occur among peatland types. Peat deposits in undisturbed bogs have been observed to be especially rich in *Sphagnum*-litter (about 50%), while the remaining portion of organic litter (necromass) is added by all other plant functional types combined (Turetsky 2003). Fine root production of vascular bog plants has been observed to be as much as 38% in hummocks and 59% in lawns of the total vascular primary production (Backéus 1990). The recalcitrancy of the perennial species characteristic to bogs has been observed to serve a higher peat accumulation, than the higher decay rate of necromass from herbaceous plants and brown mosses in rich fens (Thormann et al. 1999; Vitt & Wieder 2008; Konings et al. 2019). In fens, most of the tissue found in peat samples has been observed to be from highly productive herbaceous fen plants, especially sedges, as well as brown mosses, woody species, and subsurface roots (Campbell et al. 2000; Turetsky 2003). The large contribution of herbaceous species is due to the annual die back of the entire aerial biomass at the end of a vegetation period in addition to the considerable below-ground root necromass (Saarinen 1996), while in contrast the perennial shrubs and bryophytes aerial biomass contributes only the annual die back of leaves and inflorescence, and the subsurface fine root necromass (Bona et al. 2018; Bérubé & Rochefort 2018; Schwieger et al. 2020).

The decomposition process in peatlands is retarded due to recalcitrant plant organic matter, cool, acidic and anoxic conditions because of surface-near water table levels (National Wetlands Working Group 1997; Mitsch & Gosselink 2015; Joosten 2016). Decomposition is limited by the quality of plant litter, where the lower the litter quality, with high contents of cellulose, lignin and tannin, as it is found for instance in recalcitrant ericaceous shrub species or *Sphagnum* sp. L., the slower it will decompose (Hájek 2009; Rydin & Jeglum 2013). Hence, peat accumulation is higher in the ombrotrophic, acidic bogs, with litter input from slow decomposing plant species including litter from ericaceous shrubs and *Sphagnum* mosses (Turunen & Moore 2003; Andersen et al. 2013). Decomposition rates of boreal peatlands have been observed to vary from 17% in bogs, and 31% in treed moderate-rich fens during the first year of study (Bayley et al. 2005). Gorham and colleagues (2012) estimated that North American peatlands accumulate peat at a yearly average rate of 0.43 mm. The difference between NPP and mass loss via decomposition allows us to estimate the amount of carbon accumulating within the peat layers. As the peat accumulation and carbon sequestration are key factors of functional peatland ecosystems, a part of this research focuses on the organic matter accumulation.

Carbon cycle and sequestration

Peatlands are known to be enormous carbon (C) sinks if left undisturbed. Canadian peatlands alone are estimated to store 154 Gt* C, which is equivalent to 60% more C stored in peatlands than in forests (Tarnocai 1998; Roulet 2000). The Canadian peatlands of the boreal region alone are estimated to store approximately 99 Gt C (Tarnocai 2006). In spite of this, if these ecosystems are disturbed and their characteristic functions interrupted, peatlands may become carbon sources, releasing more carbon to the atmosphere than is being stored. Degraded peatlands may cover only approximately 0.3% of the global land area, but the ongoing degradation proceeds at a loss rate of 5 000 km^2/yr (Joosten 2016). Disturbed peatlands are responsible for 5% of the global anthropogenically induced carbon dioxide (CO_2) emissions (Joosten 2009). With the goal to reduce emissions of greenhouse gases, specifically the potent CO_2 and methane (CH_4) and to mitigate global warming, the conserva-

*One gigaton (Gt) is equal to one petagram (Pg) or 10^{15} gram (g) and will be used interchangeably in this dissertation.

tion and ecological restoration of peatlands have been identified to be effective tools to increase the peatlands' health and their C sink function (IPCC 2014).

The total C uptake rate, the net ecosystem production (NEP), of a natural northern bog is on average 20 to 30 g C m^{-2} yr^{-1} (Strack 2008). The C accumulation rate differs across the earth's eras, being highest during the time following the glacial era, when peatlands first started forming in the Western Canadian region approximately 8 000 to 9 000 BP at a rate of 38 g C m^{-2} and lowest 2 000 to 3 000 BP at a rate of 5.6 g C m^{-2} (Yu 2012). During the last millennium, the C accumulation rate was approximately 10.4 g C m^{-2} (Yu 2012). However, long-term studies during current times, contradict the last millennia's uptake rate and reveal an average C uptake rate of 32.3 g C m^{-2} (Yu 2012). These large variations illustrate the importance of long-term studies to fully understand the ecological processes within peatlands, as large seasonal, annual, and even millennial differences are evident.

The C cycle depicts the net ecosystem exchange (NEE) of C uptake and release through greenhouse gas exchange, such as CO_2 and CH_4, between the atmosphere and the biosphere, and loss via ground water flow in form of dissolved organic carbon (DOC; Figure 0.5). The biosphere includes the vegetation and microorganisms that live in the pedosphere and make up the organic matter of the soil through decomposition. Relative C fluxes of a northern peatland may vary each year, depending on climatic factors, environmental conditions and also vegetation response. We use the meteorological sign convention, where C uptake values are negative since the flow is from the atmosphere to the biosphere. Several studies demonstrate that over a long-term observation period, an undisturbed, functional northern peatland is capable of long-term C accumulation where the C sink is represented by a negative NEE and despite occasional years of C being released, represented in a positive NEE (Moore et al. 2002; Bubier et al. 2003; Ward et al. 2009). NEE is the sum of gross ecosystem productivity (GEP) and ecosystem respiration (R_{eco}). GEP is the C flow from the atmosphere to the biosphere happens in form of CO_2 uptake by plants via photosynthesis. Ecosystem respiration is the release of CO_2 and CH_4 via decomposition, respiration, oxidation, and ebullition. Functional northern peatlands may take up more than 220 g CO_2 m^{-2} yr^{-1} and release up to 310 g CO_2 m^{-2} yr^{-1} (Strack et al. 2008).

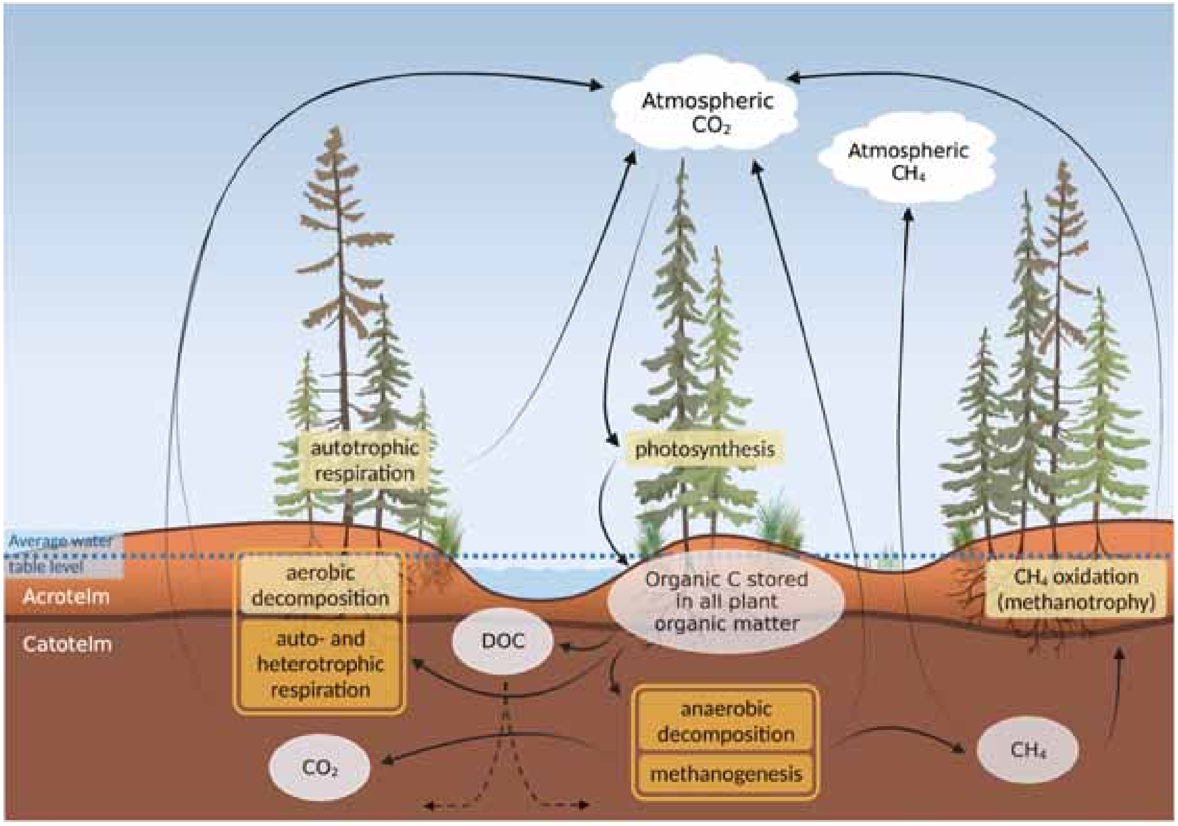

Figure 0.5 Schema of a northern peatland's carbon (C) cycle. C sequestration is the sum of C uptake and release, in form of the major greenhouse gases carbon dioxide (CO_2) and methane (CH_4), known as net ecosystem exchange (NEE). C uptake, the gross ecosystem productivity (GEP), happens via photosynthesis, where plants absorb CO_2 from the atmosphere. While CO_2 is released to the atmosphere, via ecosystem respiration, respiration of plants and microorganisms, CH_4 is produced in the anaerobic soil and released via oxidation or ebullition. CH_4 emissions are observed to increase in inundated conditions and are much affected by temperature, precipitation, and land management. Some carbon is lost through the export of dissolved organic carbon (DOC) via the ground water flow. Rates of nitrous oxide (N_2O) are not considered in this study.

For the purpose of climate reporting, the global warming potential (GWP) is an index designed to compare the greenhouse gases' potential to contribute to the warming of the earth climate (Allen et al. 2016). CO_2 is the reference for the GWP of greenhouse gases, therefore it has the GWP coefficient of 1. CH_4 on the other hand does not stay in the atmosphere as long as CO_2, but it has a much higher energy absorption, which is represented in a GWP coefficient of 28-36 over a 100-year time period, depending on the prediction scenario (Strack et al. 2008; Allen et al. 2016). In this thesis the cautious GWP coefficient of 28 is considered for calculations of the ecosystem GWP including data on CH_4 emissions.

Research objectives

This research study came to life at a time when the Canadian oil and gas industry and ecological scientists and biologists became partners to find practical solutions for feasible peatland restoration in the wake of a new law to conserve and protect wetlands in the province of Alberta. The main objective of this dissertation is to evaluate a variety of tested fen restoration methods that aimed to return characteristic peatland functions, such as habitat for peatland plants, peat accumulation, and C sequestration, following *in situ* oil sands well pad disturbances. This study aims to draw helpful conclusions from the past ecological restoration of peatlands impacted by *in situ* oil sands well pads, and to develop suggestions for improving the outcome and viability of future ecological peatland restoration works to be accomplished by the oil and gas industry.

Two study sites had been selected in the Peace River and Cold Lake Oil Sands regions, where the complete removal of a former *in situ* well pad had been tested, as well as different adaptations of partial removal of the mineral fill, and different revegetation methods were applied. Three sub-objectives help to evaluate the overall restoration outcome of each restoration method by assessing the functionality of the vegetation communities, the carbon sequestration, and the peat accumulation, impacted by the site-specific biochemistry and hydrological connection following the corresponding mineral fill removal.

The first sub-objective aims to evaluate the success of ecological peatland restoration by comparing the vegetation communities and the biochemical conditions of the restored peatlands to an unrestored decommissioned well pad and to undisturbed reference wetlands (Figure 0.6). The vegetation development was considered to be influenced by the mineral fill removal which induced the hydrological conditions, which in turn induced the biochemical conditions. Vegetation surveys and water (or peat) sampling were done during two growing seasons. The first sub-objective is addressed in Chapter 1.

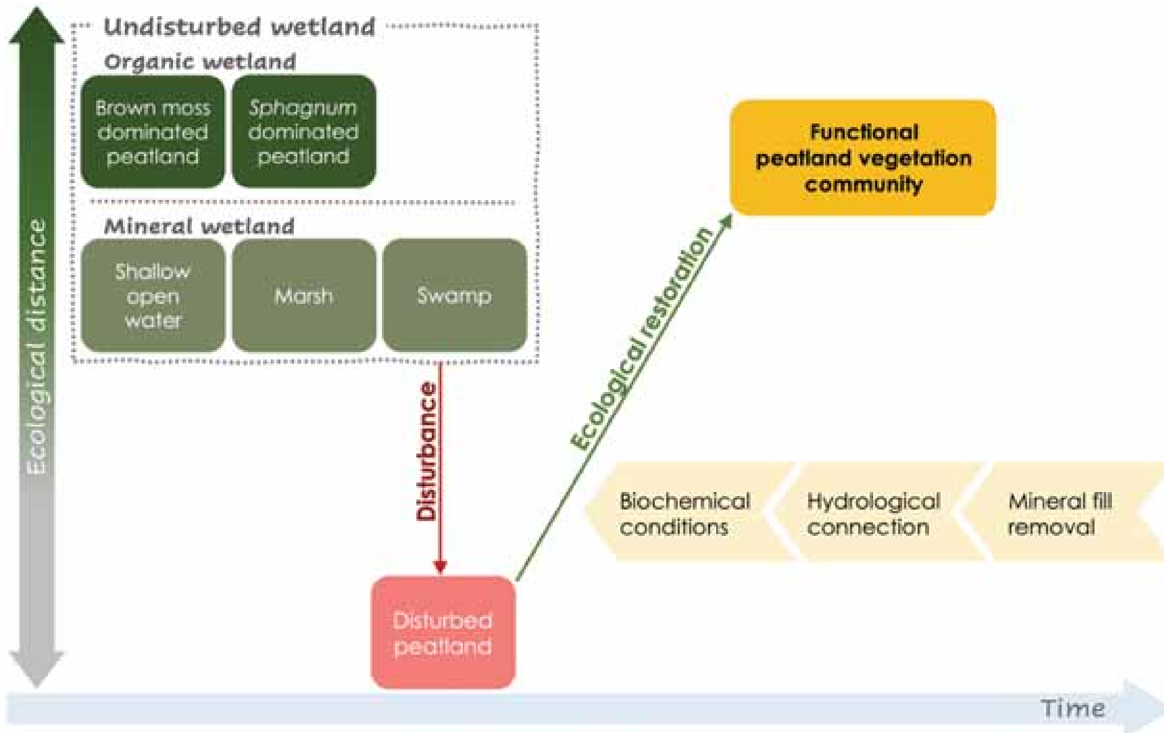

Figure 0.6 Conceptual schema of the first sub-objective concentrates on the development of vegetation communities in ecologically restored areas following in situ oil sands well pad disturbances. Independent factors influencing the vegetation development is first of all the management of the mineral fill removal inducing the hydrological conditions and ultimately the biochemical conditions,

The second sub-objective aims to evaluate the ecological restoration success by assessing the carbon (C) sequestration function and the resulting global warming potential of the restored peatlands as compared to the unrestored former well pad and undisturbed reference peatlands (Figure 0.7). C sequestration was calculated according to the rates of methane (CH_4) emissions and net ecosystem exchange (NEE) in the respective vegetation communities of the study sites. NEE was measured via carbon dioxide (CO_2) uptake and release during two growing seasons. Vegetation communities were considered to be influenced by the mineral fill removal inducing the hydrological and biochemical conditions. The second sub-objective is addressed in Chapter 2.

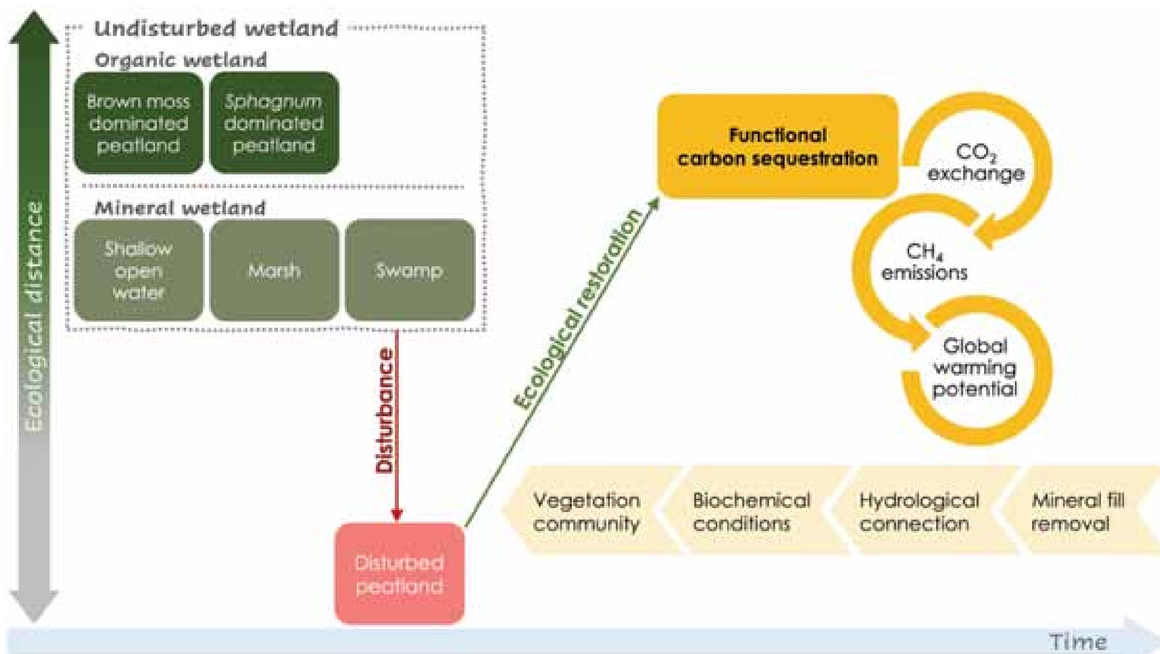

Figure 0.7 Conceptual schema of the second sub-objective, which focuses on the carbon (C) sequestration function of restored areas, which was assessed via measurements of carbon dioxide (CO_2) exchange and methane (CH_4) emissions in restored areas, which then allowed us to calculate the ecosystem GWP of the restored areas. Influencing factors are the mineral fill removal approach, affecting the hydrological connection, the development of specific biochemical conditions, and the development of characteristic vegetation communities in each study area

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The third sub-objective aims to evaluate the success of the ecological restoration to reinstate functional peat accumulation in restored peatlands, compared to an unrestored former well pad and undisturbed reference peatlands (Figure 0.8). The peat accumulation potential was calculated according to the estimates of above-ground and below-ground net primary production (NPP) and plant litter decay in the study areas, during a two-year study period. The mineral fill removal was considered to induce the hydrological and biochemical conditions, that are stimulating the respective plant communities. The third sub-objective is addressed in Chapter 3.

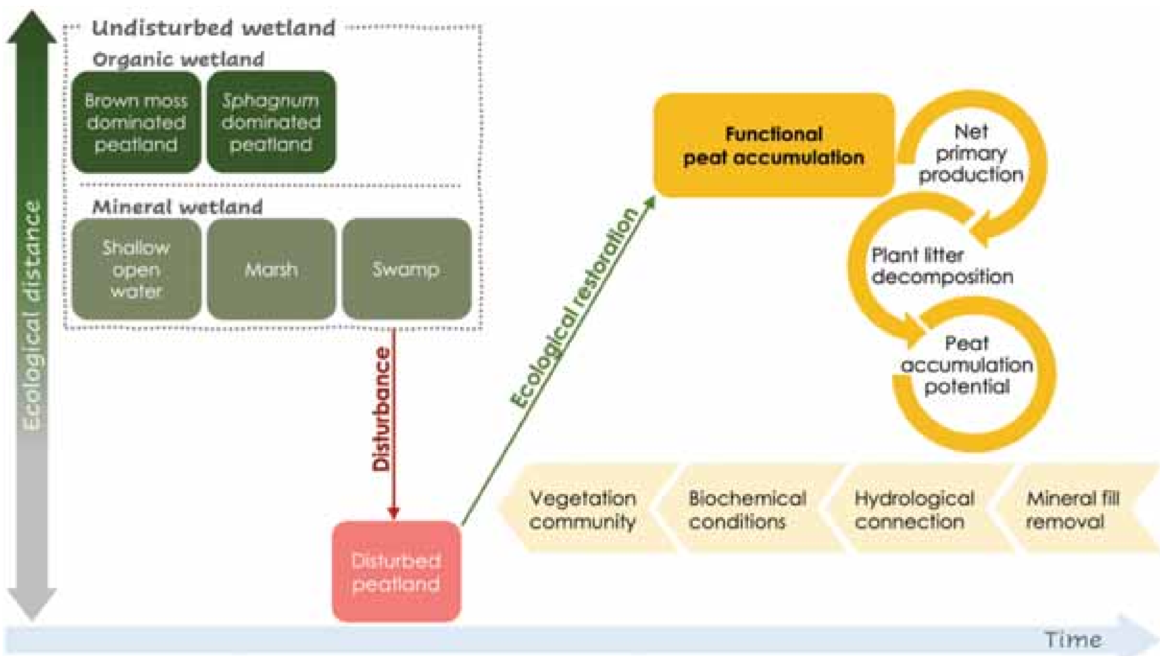

Figure 0.8 Conceptual schema of the third sub-objective that addresses the peat accumulation function of restored peatlands, which is the product of net primary production and plant litter decomposition. Influencing factors are the mineral fill removal approach, affecting the hydrological connection, the development of specific biochemical conditions and the development of characteristic vegetation communities in each study area.

The general Conclusion presents main findings across all three studies and aims to give recommendations for future fen restoration practices on *in situ* oil sands well pads.

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Chapter 1 Reestablishment of peatland vegetation following surface levelling of decommissioned *in situ* oil mining infrastructures

Meike Lemmer, Bin Xu, Maria Strack, Line Rochefort

1.1 Résumé

La restauration des tourbières, suite aux activités d'extraction du pétrole dans les régions des sables bitumineux de l'Alberta, vise à rétablir les fonctions cruciales des tourbières comme l'habitat de la faune et de la flore. Des conditions anoxiques, caractéristiques des milieux humides, sont nécessaires pour soutenir l'établissement et la croissance de la végétation. Dans ce contexte, nous avons évalué l'efficacité de quatre techniques de restauration de tourbières, et évalué la richesse et la diversité de la composition des espèces végétales, la qualité biochimique du substrat et l'écohydrologie. Les résultats ont été comparés à une zone non restaurée ainsi qu'à 28 milieux humides de référence non perturbées (REF). Dix ans après la restauration, la couverture végétale totale moyenne est de 57% dans les tourbières restaurées, contre 68% dans les REF, avec une contribution des espèces caractéristiques des tourbières de 61 et 100% respectivement. Dans les zones restaurées, en moyenne 35 espèces de plantes vasculaires et de bryophytes ont été enregistrées contre 64 espèces dans le REF. L'enlèvement complète d'une ancienne plateforme de forage a entraîné la formation d'une zone d'eau libre peu profonde alors que l'enlèvement partielle du remblai minéral et le raccordement de la surface aux tourbières adjacentes non perturbées ont permis d'obtenir une nappe phréatique proche de la surface et la plus grande diversité d'espèces végétales de tourbières.

1.2 Abstract

Peatland ecosystem restoration following oil mining activities in Alberta, Canada, aims at re-establishing crucial peatland functions, such as wildlife habitat, water storage and filtration, peat accumulation and carbon sequestration. To reinstate peatland functions, characteristic hydrological conditions are necessary to support the establishment and growth of characteristic wetland vegetation. Following *in situ* oil sands well pad disturbances in the Peace River and Cold Lake Oil Sands regions in Alberta, we evaluated the efficiency of peatland restoration approaches including different groundwork and revegetation techniques. Groundwork techniques included the complete removal (CR) or partial removal (PR) of the former *in situ* well pads' mineral fill and revegetation included the spontaneous revegetation via natural ingress of diaspores from nearby peatlands, or managed revegetation via planting of *Carex aquatilis, Larix laricina*, and *Salix lutea*. We assessed the plant species composition, biochemical and hydrological properties of all study areas, including restored peatland areas, an unrestored area and reference areas (REF) for comparison. Ten years post-restoration, in the restored areas the mean total plant cover was 57% with an average of 35 vascular plant and bryophyte species, while in REF 68% mean total plant cover and an average of 64 plant species were recorded. Respectively, characteristic peatland species contributed with 61 and 100% to the species composition. PR and hydrological connection to the adjacent peatland resulted in surface-near water table and highest peatland plant species diversity, while the CR promoted the formation of a shallow open water area.

1.3 Introduction

The Canadian province of Alberta is characterized by 20% of extensive, pristine wetland ecosystem mosaics, 90% of which are peatlands that cover approximately 119 000 km^2 surface area (Government of Alberta 2013). Peatlands store large amounts of soil organic carbon and are therefore recognized for their carbon sequestration function and climate regulating capacity, highlighting the importance for their conservation and restoration to help our fight against global warming (Joosten et al. 2012; Reed et al. 2014; Harenda et al. 2018).

In northern Alberta, peatland ecosystems have been disturbed by anthropogenic development, predominantly the *in situ* oil and gas industry (Israel et al. 2020; Rooney et al. 2012). The operation of *in situ* oil and gas well pads is essential for the bitumen extraction from deposits located at more than 75 m below the ground. The well pads are sized 1 to 4 ha and require vast interconnected infrastructures of access roads, pipelines, processing and storage facilities. As of 2016, the human footprint of the oil and gas sector adds up to approximately 39 000 km^2 (ABMI 2018). Further development is to be expected, as a total land area of 79 000 km^2 have been leased to *in situ* oil sands operators as of November 2021, and the *in situ* mineable area covers nearly 140 000 km^2 in total (Moorhouse et al. 2010; ABMI 2018). The *in situ* well pad construction in a peatland requires the cutting of trees and tall shrubs, covering of the remaining ground vegetation with a geotextile, and the introduction of a solid 1.5 to 4 m thick platform of mineral fill material mixed of clay, gravel, sand, and loam. The construction may cause nutrient enrichment of surrounding areas and compaction of the ground surface beneath the well pad, resulting in pollution and changes in the peatland ecohydrology and soil properties (Graf 2009; Wood et al. 2016). Following an average operation period of about 20 years, the well pad reclamation and peatland restoration is obligatory (Environment and Parks 2017). The Alberta Government released a policy in 2013 to guide the conservation, protection, management, and restoration of peatlands, acknowledging their ecological importance (Government of Alberta 2013).

The first peatland restoration approaches following *in situ* oil sands well pads disturbances were inspired by the natural peatland initiation during following the glacial retreat during the early Holocene (Yu et al. 2003; Vitt et al. 2011; Environment and Parks 2017). The peatland restoration approaches aim at repeating the processes of paludification and primary succession of peatland characteristic vegetation communities. Holocene peatland initiation began via terrestrialization, the accumulation of organic matter and peat formation in shallow waterbodies, and paludification, the formation and further development of peat layers in terrestrial ecosystems taking place mostly on basal clay material, the reminiscent, sedimental mineral soil from glacial transportation and retreated glacial rivers (Halsey et al. 1998; Joosten & Clarke 2002; Gorham et al. 2007). The early establishment of vegetation on new substrates and subsequent dynamics, known as primary succession, and species assembly, are influenced by and feedbacks on the peatland's hydrological regime (Vitt 1994; Large et al. 2007; Waddington et al. 2015). Vegetation communities, their specific structure of plant functional types and characteristic species composition, are key factors for assessing the restoration success (Del Moral et al. 2007; Hobbs et al. 2007). Limited field trials have shown successful establishment of peatland vegetation on decommissioned well pads in a process similar to paludification (Vitt et al. 2011; Gauthier 2014). However, the efficacy of well pads restoration in peatland remains understudied and poorly documented (Graf 2009).

Our study aims to fill this gap in knowledge using former *in situ* oil sands well pads that were subject to different peatland restoration approaches in the recent past (Vitt et al. 2011; Lemmer et al. 2020). We assessed three restoration approaches that tested different treatments of groundwork, either the complete removal (CR) or the partial removal (PR) of the mineral fill, and revegetation, either spontaneous revegetation or active planting of fen plant species (Vitt et al. 2011; Lemmer et al. 2020). Three objectives were defined to evaluate the restoration approaches: O1) To assess if the active reintroduction of characteristic plant species necessary to facilitate the establishment of peatland vegetation communities; O2) To assess which level of the mineral fill surface and distance to the water table best promote the recovery of fen characteristic plant species; O3) To assess if a residual mineral fill negatively affects the biochemistry and inhibits the recovery of peatland characteristic plant species composition. Following these three objectives, we aim to answer the following two research questions: Q1) How do the vegetation communities differ between the study areas? Q2) How do the environmental conditions differ across the study areas and which environmental factors contribute to the development of peatland vegetation? To answer these questions, we compared the restored areas, with reference areas and an unrestored control area, regarding plant species composition, the species' richness, and environmental variables.

1.4 Methods

1.4.1 Study sites

The study sites included two decommissioned *in situ* oil sands well pads and several larger reference wetland complexes in the Peace River and the Cold Lake Oil Sands regions in Alberta (**Figure 1.1**; Lemmer et al. 2020). If different reference wetland sites were located in one vast wetland complex but differed in terms of peatland typology, they were considered different reference wetland areas.

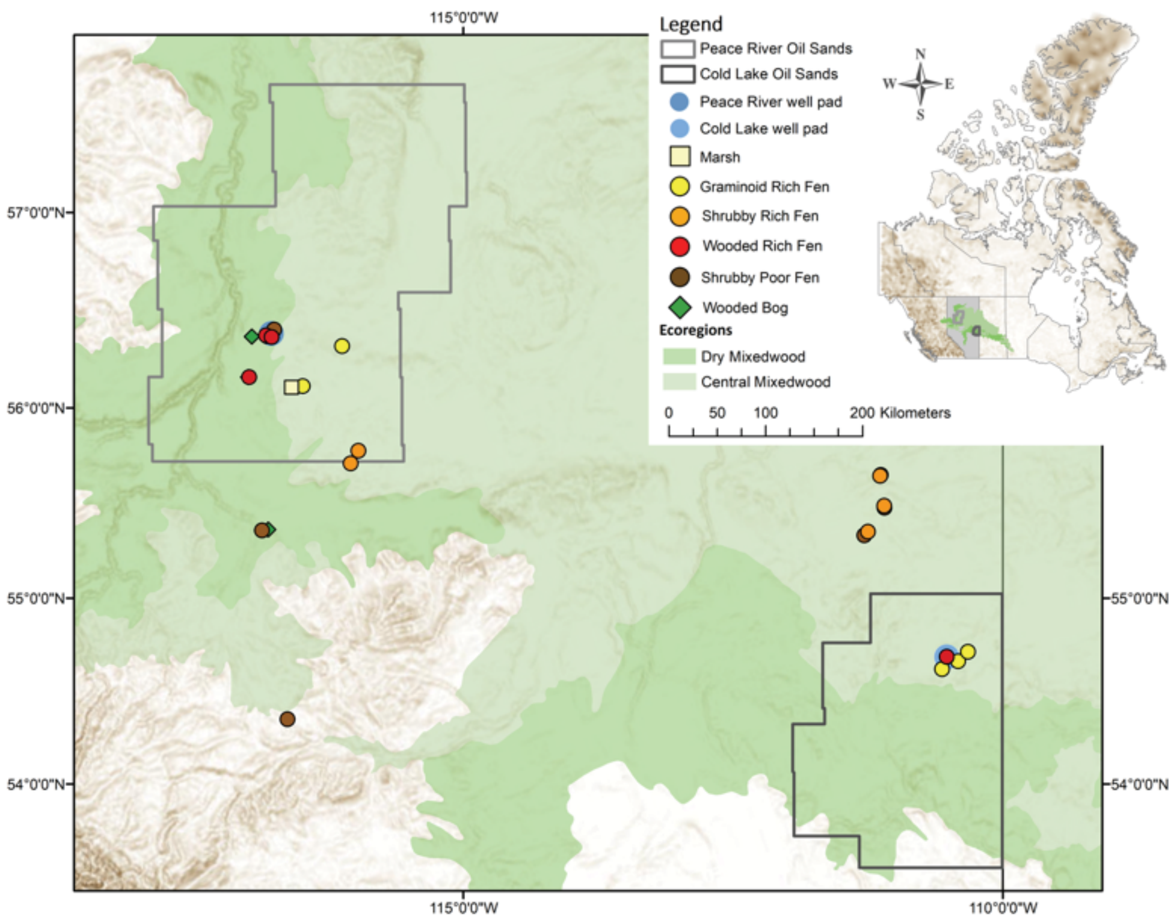
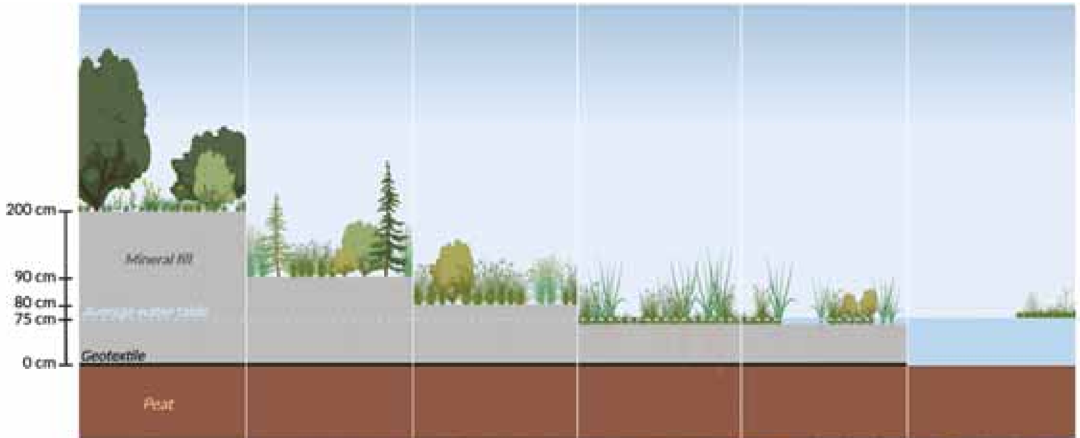


Figure 1.1 Location of all study areas within the dry and central mixedwood ecoregions of the boreal forest in Alberta, Canada. All study areas are located within the Peace River and Cold Lake Oil Sands regions. The restored areas and unrestored control area are situated at two decommissioned in situ oil sands well pads, which were subject to trials of different peatland restoration approaches. Six designated comprehensive reference areas combine 28 reference area sites, which were located within several vast peatland complexes.

Restored study areas

The study areas where restoration measures have been applied more than eight years prior this study are still in the process of restoration but are here labelled “restored areas”. In 2017, ten years post restoration, we selected two restored study areas on a decommissioned *in situ* well pad in the Peace River region: partial removal of well pad’s mineral fill to 15 cm above the water table (*PR15*) and partial removal to 5 cm above the water table (*PR5*; **Figure 1.2**; Table S1; Lemmer et al. 2020). Vitt et al. (2011) and Koropchak et al. (2012) describe the restoration and revegetation method, where the mineral fill was graded down to near the water table and active planting of *Carex aquatilis*, *Larix laricina* and *Salix lutea* reintroduced characteristic fen plant species. PR15 was characterized by abundant *Calamagrostis inexpansa* and *C. aquatilis*, with a low cover of *L. laricina*, *S. planifolia* and *S. pyrifolia,* while the average water table level was at 16 cm below the surface. PR5 was dominated by *C. aquatilis*, while some *S. planifolia* and *S. exigua* were present, and the average water table level was at 1 cm above the surface.



Wetland class	None	Mineral wetland	Mineral wetland	Mineral wetland	Mineral wetland	Mineral wetland
Site ID	UNR	PR15	PR5	PR0E	PR0	CR
Status	Unrestored	Restored	Restored	Restored	Restored	Restored
Years since restoration	n.a.	10	10	8	8	9
Restoration treatment - groundwork	No restoration	Partial removal (PR) of well pad mineral fill, to near the surface to 15 cm above the water table level	PR of well pad mineral fill, to near the surface to 5 cm above the water table level	PR of well pad mineral fill and levelling to the adjacent peatland (0 cm), even ground without microforms	PR of well pad mineral fill and levelling to the adjacent peatland (0 cm), with microforms	Complete removal (CR) of mineral fill and underlying geotextile with shallow open water and floating moss carpet
Residual mineral fill	200 cm	90 cm	80 cm	75 cm	75 cm	0 cm
Distance to WTL	+125 cm	+15 cm	+5 cm	±0 cm	±0 cm	-75 cm
Restoration treatment - revegetation	Spontaneous revegetation	Managed revegetation via introduction of plant species (*Carex aquatilis, Larix laricina, Salix lutea*)		Spontaneous revegetation	Spontaneous revegetation	Spontaneous revegetation
Dominant plant functional type	Shrubs, herbs, brown mosses	Sedges, shrubs, herbs	Sedges, shrubs, herbs	Herbs, sedges, brown mosses	Sedges, shrubs	Aquatics, floating brown moss carpet
Dominant plant species	*Campylium stellatum, Hamatocaulis vernicosus, Poa palustris, Trifolium hybridum*	*Calamagrostis inexpansa, C. aquatilis, Equisetum pratense, L. laricina, Parnassia palustris, Salix pyrifolia*	*C. inexpansa, C. aquatilis, E. arvense, Ptychostomum pseudotriquetrum, Salix* spp.	*Aulacomnium palustre, Carex* spp., *Equisetum* spp., *P. pseudotriquetrum, Sarmentypnum tundrae, T. latifolia*	*Carex* spp., *Equisetum* spp., *Salix* spp., *Typha latifolia*	*Drepanocladus aduncus, Sarmentypnum exannulatum, Warnstorfia fluitans*
Environmental parameters	ST5 20.3 °C WTL -27 cm pH 3.6 EC 605 µS/cm	ST5 20.2 °C WTL -16 cm pH 3.4 EC 2 456 µS/cm	ST5 19.4 °C WTL +1 cm pH 3.6 EC 2 573 µS/cm	ST5 17.7 °C WTL -2 cm pH 7.3 EC 476 µS/cm	ST5 21.2 °C WTL +17.8 cm pH 7.5 EC 342 µS/cm	ST5 18.7 °C WTL +47 cm pH 6 EC 246 µS/cm
Year of construction	2002	~1980	~1980	2002	2002	2002
Year of decommissioning	2003	2000	2000	2003	2003	2003
Year of restoration	n.a.	2007	2007	2009	2009	2008
Years of compaction	16	20	20	7	7	6

Figure 1.2 Schematic cross section and traits of five restored areas and the unrestored control area (UNR). Restoration approaches tested the partial removal (PR) and complete removal (CR) of an in situ well pad's mineral fill. Restoration approaches shown are PR15: PR to 15 cm above the water table level (WTL); PR5: PR to 5 cm above the WTL; PR0E: PR to the same level (0 cm) as the WTL (even, without microforms); PR0: PR to the same level (0 cm) as the WTL (differences occur between dry and wet microforms); CR: CR of the mineral fill and the underlying geotextile (differences occur between the shallow open water area, and the floating brown moss carpet).

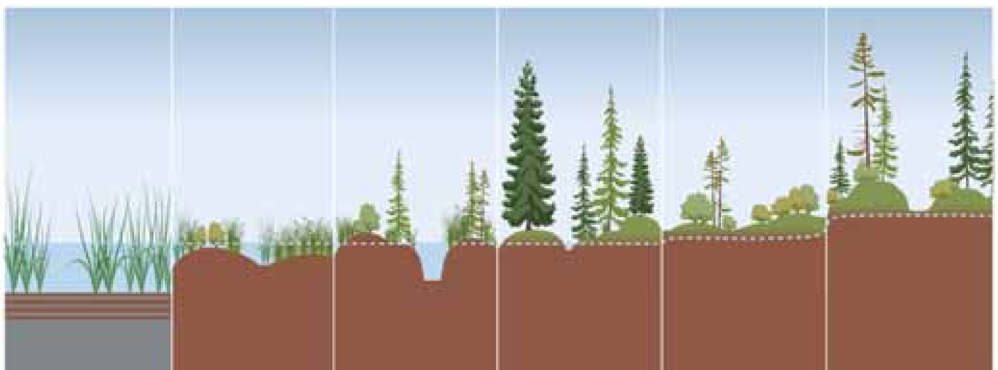
Also in 2017, three restored study areas were selected on a decommissioned *in situ* well pad in the Cold Lake region, eight to nine years post restoration: complete removal of the well pad's mineral fill and the underlying geotextile (*CR*), partial removal of the mineral fill and levelling with the adjacent peatland to connect to its water table level (0 cm) with wet and dry microforms (*PR0*), and partial removal and levelling to 0 cm with even ground (*PR0E*; Figure 1.2; Appendix 1.1; Lemmer et al. 2020). CR is characterized by a shallow open water area with a water table level of 47 cm above the surface where mostly floating aquatic vegetation and some *T. latifolia* grow at the water's edge. In CR, a floating brown moss carpet with a water table level at the surface and very few small shrubs and sedges are present. *In PR0* the water table varied between the microforms, but averaged at 18 cm above the surface, and abundant stands of *Salix* sp., *T. latifolia, C. aquatilis,* and *Eleocharis palustris* had developed. In PR0E a dense brown moss ground cover and abundant *T. latifolia*, *C. aquatilis* and *Equisetum* spp. dominated the vegetation, while the water table level was at 2 cm below surface.

Unrestored study area

The here designated "unrestored" control area (*UNR*) was left for natural regeneration and underwent no restoration or reclamation treatment following the decommissioning. UNR was selected in 2017 at the decommissioned Cold Lake well pad (Appendix 1.1; Lemmer et al. 2020). UNR was dominated by upland species including *Poa* ssp., *Trifolium hybridum, Campylium stellatum*, and *Hamatocaulis vernicosus,* and the average water table level was 27 cm below the surface (Figure 1.2).

Reference wetland study areas

In 2018, we surveyed 10 reference sites, while 18 supplementary reference sites had been surveyed during two previous studies done in 2011 and 2016 (Gauthier 2014; Guêné-Nanchen 2018; Appendix 1.1). All 28 surveyed reference sites were part of larger peatland complexes located in the same geographical ecoregions as either Peace River or Cold Lake (Figure 1.1, Appendix 1.1). Both ecoregions were subject to extensive anthropogenic development. Given the constraints of field safety and logistical considerations, areas with an active peat layer and reasonable access were selected, excluding areas with visible major human impact. The reference sites were grouped into the six broad classes, based on water table level (WTL; Fig. S1), soil and water chemistry (Appendix **1.3** & Appendix **1.4**), that served in this study as designated reference areas (*REF*): marsh (*M*), graminoid rich fen (*GRM*), shrubby rich fen (*SRF*), wooded rich fen (*WRF*), shrubby poor fen (*SPF*), wooded bog (*BOG*; **Figure 1.3**). The groups of REF were formed using the '*cascadeKM*' function for K-means clustering with the Calinski-Appendix 1.1an optimal number of groups, including data on environmental factors and plant species abundance (Calinski & Harabasz 1974; Oksanen et al. 2019; Figure 1.2). To avoid an emphasis on abundant species and a subsequent skewed Euclidean distance (Borcard et al. 2018), a Hellinger transformation was applied on the complete vegetation datasets, using the '*decostand*' function. Reference area groups were verified with the respective definitions by the Alberta Wetland Classification System (Alberta Environment and Sustainable Resource Development 2015).



Wetland class	Marsh	Graminoid rich fen	Shrubby rich fen	Wooded rich fen	Shrubby poor fen	Wooded bog
Site ID	M	GRF	SRF	WRF	SPF	BOG
Status	Reference ecosystem	Reference ecosystem	Reference ecosystem	Reference ecosystem	Reference ecosystem	Reference ecosystem
Successional age	young					old
Characteristic plant functional types	>25% herbs and sedges, aquatics	Herbs, sedges, mosses, shrubs, <25% trees	>25% shrubs, herbs, sedges, brown mosses, <25% trees	>25% coniferous trees, brown mosses	<25% trees, >25% shrubs, *Sphagnum* sp.	*Sphagnum* sp., >25% coniferous trees
Dominant plant species	*Calamagrostis* sp., *Caltha palustris*, *Carex* sp., *Eleocharis* sp., *Lemna minor*, *Phalaris arundinaceae*, *Rumex occidentalis*, *T. latifolia*, *Utricularia vulgaris*	*Calla palustris*, *Calliergon giganteum*, *Campylium stellatum*, *C. aquatilis*, *C. diandra*, *C. trisperma*, *C. limosa*, *Comarum palustre*, *D. aduncus*, *Hamatocaulis* sp., *Menyanthes trifoliata*, *Scorpidium scorpioides*, *Tomentypnum nitens*, *Triglochin* sp.	*Betula* sp., *C. giganteum*, *C. stellatum*, *Chamaedaphne calyculata*, *Carex* sp., *C. palustre*, *D. aduncus*, *Maianthemum trifolium*, *Meesia triquetra*, *Rhododendron groenlandicum*, *Salix* sp.	*Aulacomium palustre*, *Betula* sp., *Carex* sp., *C. palustre*, *H. vernicosus*, *Larix laricina*, *Picea mariana*, *R. groenlandicum*, *Salix* sp., *Sphagnum* sp., *Triglochin* sp., *T. nitens*	*Betula* sp., *C. lasiocarpa*, *C. calyculata*, *Eriophorum vaginatum*, *L. laricina*, *R. groenlandicum*, *Salix* sp., *Sarmentypnum exannulatum* *Sphagnum* sp.,	*C. calyculata*, *E. vaginatum*, *Hylocomium splendens*, *L. laricina*, *P. mariana*, *Pleurozium schreberii*, *R. groenlandicum*, *S. fuscum*, *S. magellanicum*, *Vaccinium vitis-idaea*
Environmental parameters	ST5 19.3 °C WTL +115 cm pH 5.6 EC 218 µS/cm	ST5 19.8 °C WTL +15 cm pH 6.8 EC 191 µS/cm	ST5 18.9 °C WTL n.a. pH 6.9 EC 343 µS/cm	ST5 15.4 °C WTL +2 cm pH 6 EC 255 µS/cm	ST5 21.9 °C WTL n.a. pH 6.1 EC 505 µS/cm	ST5 17.7 °C WTL -2 cm pH 4.7 EC 156 µS/cm

Figure 1.3 Schematic cross section and traits of six reference areas by means of plant species composition, soil and water chemistry following Alberta Environment and Sustainable Resource Development (2015). Reference areas shown are M: Marsh; GRF: Graminoid rich fen; SRF: Shrubby rich fen; WRF: Wooded rich fen; SPF: Shrubby poor fen; BOG: Wooded bog.

1.4.2 Experimental design

In each study area, five survey plots were randomly selected by aimlessly tossing a survey frame. At each survey plot we estimated vascular plant cover in one 1 m^2 survey frame (n=5) and bryophyte cover in four nested 25 × 25 cm survey frames (n=20; Appendix 1.1). In the unrestored and restored areas one water sample (n=3) and one soil grab sample (n=3) were taken at three measurement plots that had been already installed for related studies described in Lemmer et al. (2020). In the REF, water and soil samples were taken at survey plot 1, 3, 5. At the same sample plots, we measured water table level (WTL) and soil temperature at 5 cm depth (ST5). The experimental design, and the sampling and analysis methods applied in the previous studies including the supplementary reference sites, are described in Gauthier (2014) and Guêné-Nanchen (2018).

1.4.3 Vegetation survey

Vegetation surveys in the unrestored and restored areas were performed during the peak of the vegetation period between July and August 2017 and 2018. The surveys in the REF were done between June and August 2018. Cover in absolute percent was estimated for the total vegetation, the individual plant functional types (trees, ericaceous shrubs (*Ericaceae*), other shrubs, sedges (*Cyperaceae*), other herbs, peat mosses (*Sphagnaceae*), other bryophytes, lichens), and the individual species (Wullschleger et al. 2014). Plant taxonomy and nomenclature followed the Integrated Taxonomy Information System for vascular plants (2021) and the Consortium of North American Lichens Herbaria for bryophytes and lichens (2021).

1.4.4 Environmental variables

In the unrestored and restored areas WTL was measured between May and September in 2017 and 2018 as part of a related study (Lemmer et al. 2020). In the REF areas, if the water table level (WTL) was inaccessible and immeasurable at or above the surface, holes were dug prior the vegetation survey and WTL was allowed to equilibrate until the initial measurement at the end of the vegetation survey. Additionally, at each REF area, an automated WTL logger and barologger (Solinst Canada Ltd., Georgetown, ON) was installed to record the WTL for one year following the vegetation survey in 2018. Before analysis, WTL values were corrected for atmospheric pressure variations and expressed relative to the surface level at the study area.

Measurements of additional environmental variables included the pH, the electric conductivity, and the concentrations of major elements and micronutrients (calcium (Ca), magnesium (Mg), potassium (K), sodium (Na), ammonium nitrogen ($N\text{-}NH_4^+$), nitrate nitrogen ($N\text{-}NO_3^-$), phosphate ($P\text{-}PO_4^{3-}$), sulfate ($S\text{-}SO_4^{2-}$), aluminum (Al), iron (Fe^+), manganese (Mn)) of water and soil. A pair of two water and soil samples each were taken at three samples plots at each study area, at the same time of the vegetation survey. The two water samples of 200 ml each were collected from water in the respective well pipe or hole used for measuring the WTL. The two soil samples were each composed of four grab samples of approximately 100 ml each, collected within the first 10 cm of the soil, within a designated sample plot area of about 60 × 60 cm. All water samples were filtered through a 0.45 µm cellulose filter (Thermo Fisher Scientific Inc., Chelmsford, MA, USA) before further respective treatment. Of the two elements, one sample of each pair was kept cool at 4 °C until analysis of pH and EC the next or the following day. The respective soil samples were liquified in a 1:10 mixture with deionized water, to create a water sample suited for analysis of pH and EC with an Orion Versastar Advanced Electrochemistry Meter (Thermo Fisher Scientific Inc., Chelmsford, MA, USA). The second respective sample was frozen at -20 °C and shipped to the Centre d'Étude de la Forêt (Université Laval, QC) for analysis of the major elements and micronutrients. Prior analysis, the thawed respective soil samples were cleaned from roots and other organic litter, dried at 70 °C to constant weight, and ground. Concentrations of Na, $N\text{-}NO_3^-$, $P\text{-}PO_4^{3-}$, $S\text{-}SO_4^{2-}$ were analyzed via a FIA Quikchem 8500 Series 2 (Lachat Instruments, Milwaukee, WI, USA). Concentrations of Al, Ca, Fe^+, K, Mg, Mn, and $N\text{-}NH_4^+$ were analyzed via an ICP Agilent 5110 SVDV (Agilent Technologies Inc., Santa Clara, CA, USA).

Soil temperature at 5 cm (ST5) was measured biweekly at different times of the day at three measurement plots in the unrestored, restored, and some reference areas as described in Lemmer et al. (2020). At the remaining reference areas, ST5 was measured simultaneously with WTL.

The climate averages of 1981-2010 measured at the meteorological stations of Peace River and Cold Lake served as reference for total precipitation during a 120-day vegetation period (May-August) and daily temperatures (Government of Canada 2019).

1.4.5 Data analysis

All statistical and multivariate analysis were computed in R version 3.6.0 (R Core Team 2019). Utilized packages were the '*agricolae*' package for statistical procedures for agricultural research, the community ecology package '*vegan'*, and the multivariate exploratory data analysis and data mining package '*FactoMineR*', and the data visualizations package '*ggplot2*' (DeMendiburu 2019, Oksanen et al. 2019, Husson et al. 2020, Wickham et al. 2020). Additional figures were created using BioRender (2021) and ArcGIS version 10.4.1 (ESRI 2015).

Analysis of vegetation data

To understand how different the plant species composition is among all study areas, we assigned each species a specific wetland indicator status and compared the species richness and diversity, and the abundance of plant functional types. Because lichens were rare in all study areas and the low frequency of lichens was a factor for non-normal data distribution, the lichen data was omitted during analysis. A Hellinger transformation was applied on the remaining vegetation dataset to avoid the double zero problem and improve normality (Borcard et al. 2018).

Wetland indicator species are characteristic hydrophytic plant species that are considered specialists colonizing specific wetland habitats and ecosystems (Tiner 1993; Seppelt et al. 2008; Lichvar et al. 2009). All plant species were appointed a wetland indicator status (*WIS*) specifically defined for this study, with inspiration from Tiner (1993), Gillrich & Bowman (2010), Payette & Rochefort (2013), and USDA, NRCS (2021). The WIS categories for this study are: 1) Peatland species (*PEAT*): essential for peatland communities in Alberta, preferentially bogs and fens; 2) Other wetland species (*OTHW)*: obligate and facultative wetland species; 3) Non-wetland species (*NONW)*: upland species and generalists (Table S2). Each species was appointed to one category only. In the end, we identified 75 PEAT species, 98 OTHW species and 58 NONW species (Table S2). We used an ANOVA with a Tukey's Honest Significant Difference (HSD) test to identify significant differences ($p<0.05$) in species richness among WIS groups in all study areas.

Species are also assigned to wetland classes (B=bog, F=fen, M=marsh, S=swamp, W=shallow open water) followed the Alberta Wetland Classification System (Alberta Environment and Sustainable Resource Development 2015) that helped to decide the exclusive affiliation for the WIS category. Fen characteristic species were defined as peatland species colonizing preferentially fens, and species that have been associated in other studies with early successional fens (Yu et al. 2003; Seppelt et al. 2008; Bérubé & Rochefort 2018).

Species richness, the number of species that are present within a specific biological community, and α-diversity indices (Shannon's *H*), the species diversity of an area-specific biological community, were calculated in R using the '*specnumber*' and the '*diversity*' functions. We considered the complete plant species dataset for all study areas. An ANOVA in combination with a Tukey's HSD test was used to identify significant differences ($p<0.05$) in species diversity and richness between all study areas. The restored areas are not replicated at other sites or studies, therefor the study plots within the restored areas served as replications in the ANOVA and multiple linear regressions.

Analysis of environmental parameters

To understand the environmental factors driving the vegetation composition, a redundancy analysis (RDA) was performed to explain the ecological variation between unrestored, restored and the reference areas using the '*cca*' function. The model's constraining variables included biochemical and environmental factors. The corresponding response matrix included the vegetation cover of species with at least 5% mean abundance if present. A Hellinger transformation was applied to the vegetation dataset, in order to avoid the double-zero problem (Borcard et al. 2018). Permutation tests (n=999) involving a stepwise forward model selection was used to retain significant explanatory biochemical and environmental variables. Eigenvalues were proportionally scaled (scaling 2), while study areas remained unscaled with a weighted, equal dispersion on all dimensions.

To understand the effect of significant environmental variables on plant functional types, we ran multiple linear regressions following the RDA. Regression models were run via the '*lm*' function and were considered significant at the 95% confidence level ($p<0.05$).

1.5 Results

1.5.1 Vegetation development

A total of 231 plant species were identified in all study areas combined (Appendix 1.1). Vascular plants accounted for 157 species (31 of which were sedges), bryophytes accounted for 66 species (including 11 *Sphagnum* sp.), and lichens accounted for 7 species (Table 1.1). Overall species richness was highest in the REF (BOG: 59, SPF: 68, GRF: 70, SRF: 85, WRF: 88), except in the marsh where only eight species were observed. Highest species richness among the restored areas was measured in PR0E and PR0 (50 and 48 species respectively). while the lowest species richness was in PR5 (18 species). The study areas' α-diversity was estimated via the mean Shannon's diversity index H, which is higher with increasing species richness and evenness (Borcard et al. 2018). Eight to ten years post-restoration, the restored areas appear as diverse as undisturbed REF (**Figure 1.4**). Among all study areas, the restored PR0E and PR0 and the undisturbed WRF had the highest α-diversity (H=1.9, H=1.8, H=1.9 respectively; Figure 1.4). The marsh study area had the lowest α-diversity among all study areas (H=0.8; **Figure 1.4**).

Table 1.1 Species richness at all study areas, showing the total numbers of plant species sorted by vascular and bryophyte species, and by wetland indicator status (WIS; PEAT=peatland species: essential for peatland community in Alberta, preferentially bogs and fens; OTHW=other wetland species: obligate and facultative wetland species; NONW=non-wetland species: upland species and generalists). Meaning of status and study area codes according to Appendix 1.1.

Status	Study area	Total plant species	Vascular plant species	Bryophyte species	Wetland indicator status		
					PEAT	OTHW	NONW
UNR	UNR	46	28	18	10	17	19
RES	PR15	29	25	4	2	13	14
	PR5	19	14	5	2	13	4
	PR0E	50	37	13	12	30	8
	PR0	48	42	6	14	29	5
	CR	32	21	11	11	18	3
REF	M	9	8	1	2	7	0
	GRF	70	50	20	30	32	8
	SRF	85	58	27	34	40	11
	WRF	91	52	39	52	26	13
	SPF	68	40	28	39	15	14
	BOG	62	32	30	45	9	8
Total	---	231	157	66	75	98	58

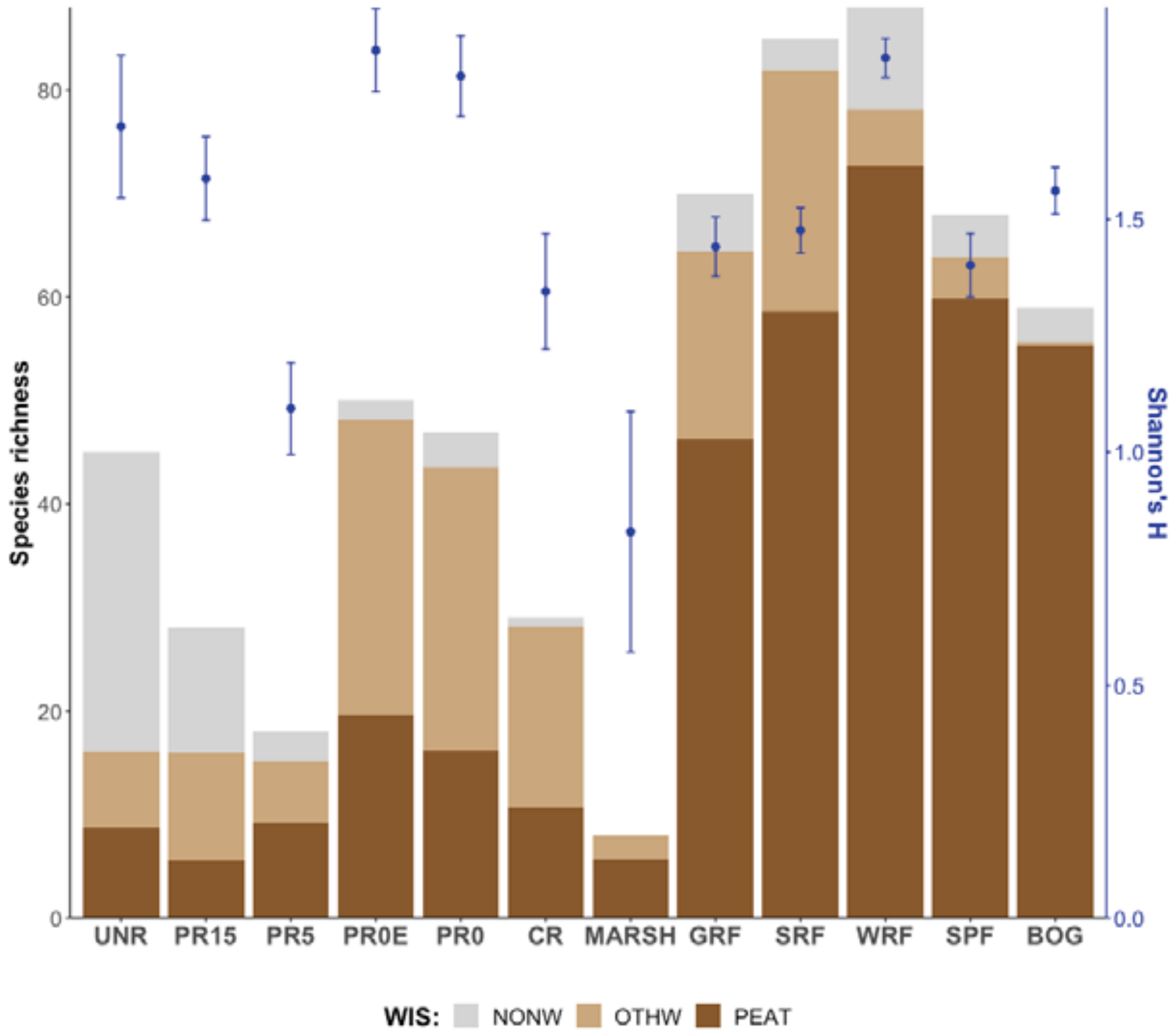


Figure 1.4 Species richness (bars) and α-diversity index Shannon's H (points; ANOVA between classes $F_{(11, 289)} = 8.52$, $p<0.001$, adj. $r^2 = 0.22$) calculated for the unrestored, restored and reference areas (code for study areas according to Method section and Figure 1.2Figure 1.3; see also Appendix 1.1). Species richness is represented according to the species' natural habitat and wetland indicator status WIS (meaning of WIS codes according to Table 1.1). Each species was appointed to one category only.

Regarding plant functional types, brown mosses and sedges had the highest abundance within restored areas eight to ten years post-restoration (particularly *Drepanocladus aduncus* and *Ptychostomum pseudotriquetrum,* and *C. aquatilis* and *E. palustris*; **Figure 1.5**; see also Appendix **1.5**). Simultaneously, peat mosses and ericaceous shrubs are largely absent from all restored areas, except for very low occurrence (2% cover each) in the dry microform of PR0 (**Figure 1.5**). The covers of small trees and shrubs like *L. laricina, S. exigua,* and *S. glauca*, were highest in PR15 (>18%) and lowest in PR0E (5%) and CR (1%). In contrast, the covers of characteristic fen plants were highest in PR0E (52%) and the lowest in PR15 (10%). Including the floating moss carpet, the shallow open water area CR had a cover of 29% fen characteristic plant species.

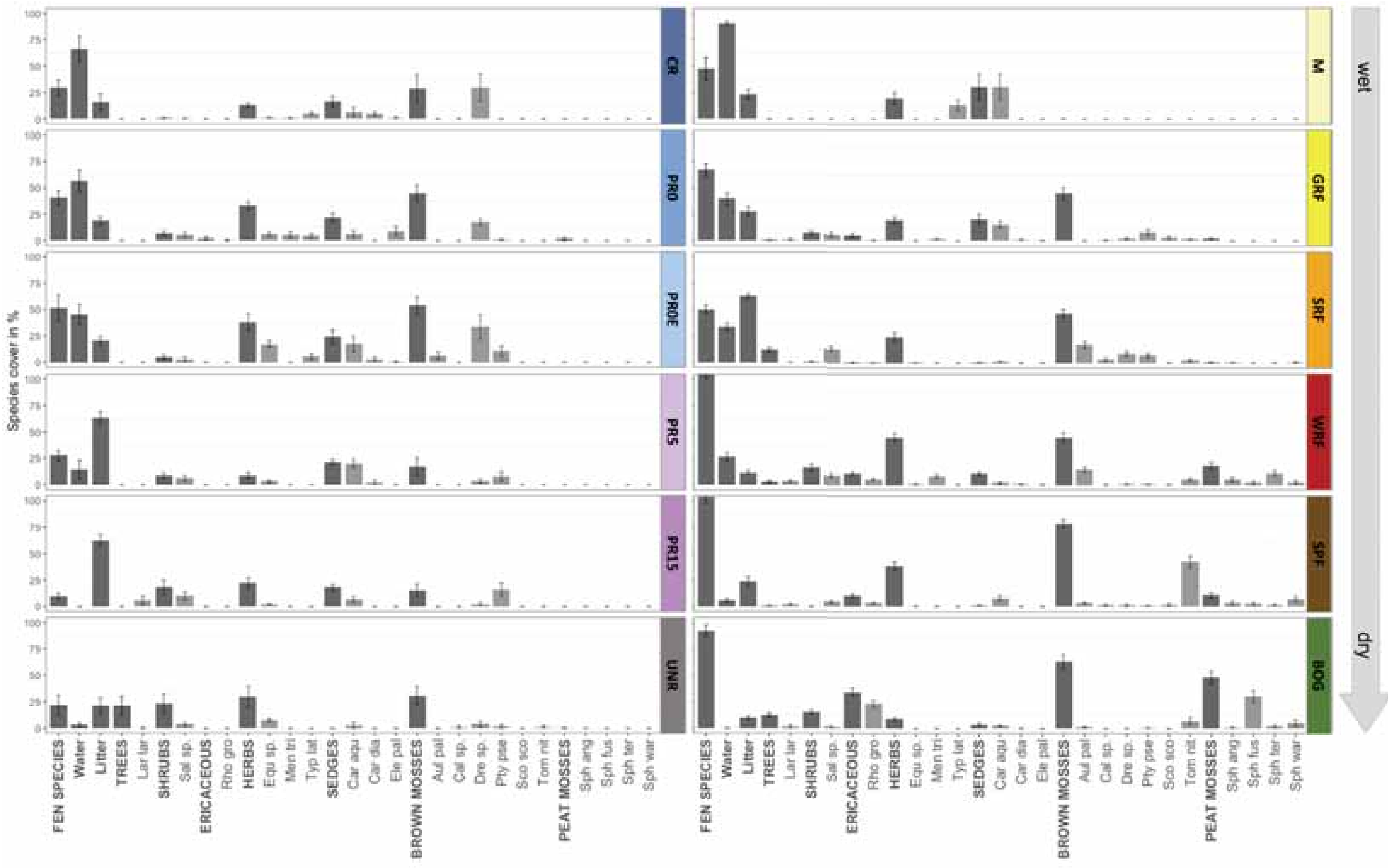


Figure 1.5 *Mean species cover (%) in all unrestored, restored and reference study areas (see also Table S5. In dark grey presented are fen characteristic plant species (fen species), water, plant litter, and plant functional types (trees, shrubs, ericaceous shrubs, herbs, sedges, brown mosses, peat mosses). Shown in light grey are particularly abundant plant species of the respective plant functional type. Code for study areas according to Method section and Figure 1.2Figure 1.3 (see also Appendix 1.1). Code for plant species as in Appendix 1.2. Restored and unrestored areas (left), and reference areas (right) are arranged from the wettest areas at the top, to the driest areas at the bottom.*

The highest PEAT species count among restored areas was observed in PR0 (14 species; Table 1.1). The highest PEAT species cover was measured in CR (34%). In PR0E, PEAT species were twice as diverse, according to the Shannon's *H* index than in CR and almost four times more diverse than in PR15 (Figure 1.4). OTHW species contributed more than 60% to the species richness of PR0E, PR0, and PR5. The NONW species richness in the restored areas was highest in PR15 (12.1) with almost 48% of the vegetation being NONW species, and lowest in CR (0.8). UNR is dominated by 19 NONW species, which represents approximately 41% of the species in this area (Figure 1.4). Among REF, the highest PEAT species count was observed in WRF (52 species, making up 57% of the area-specific species composition) followed by BOG (45 species, >72% of area-specific species).

Sonchus arvensis was the only observed invasive species that was recorded in the restored areas PR15 and PR5 only during the 2017 survey.

1.5.2 Effects of environmental conditions on the vegetation community

The 2017 growing season had a warm, dry summer with intense precipitation events, while the 2018 season had a cooler and consistently wetter summer. The diverging weather between the two study seasons is well represented in (Appendix **1.7**a & b). During the first year of the study, the WTL of the drier study areas UNR, PR15, PR5, and BOG, were on average approximately 20 cm below the surface, and the WTL of the wetter study areas PR0, PR0E, CR, GRF and WRF were at or 20 cm above the surface. During the second year of the study the WTL of all study areas were close to or above the surface.

Among all study areas, PR5 and PR15 had highest values for water EC (2574 and 2456 μS/cm) and soil EC (1309 and 1327 μS/cm), water extractable concentrations of Ca (273 and 208 mg/L), Mg (109 and 99 mg/L), S-SO_4^{2-} (298 and 268 mg/L) and soil S-SO_4^{2-} (1638 and 5608 mg/L; Appendix 1.3Appendix 1.4).

The concentrations of soil Mn, water extractable concentration of P-PO_4^{3-}, soil temperature at 5 cm depth (ST5) and WTL, explained approximately 70% of the variability of plant species cover on the first two axis ($F_{(4,11)}$=2.1, p=0.001, r^2=0.43; **Figure 1.**6). The restored and graminoid rich fen areas are clearly separated along the first axis from the bog, the shrubby rich fen and the wooded rich fen by ST5 and the concentration of phosphate in water

(**Figure 1.**6). On the second axis, WTL and soil manganese separate the shrubby and wooded rich fens from the restored areas (**Figure 1.**6).

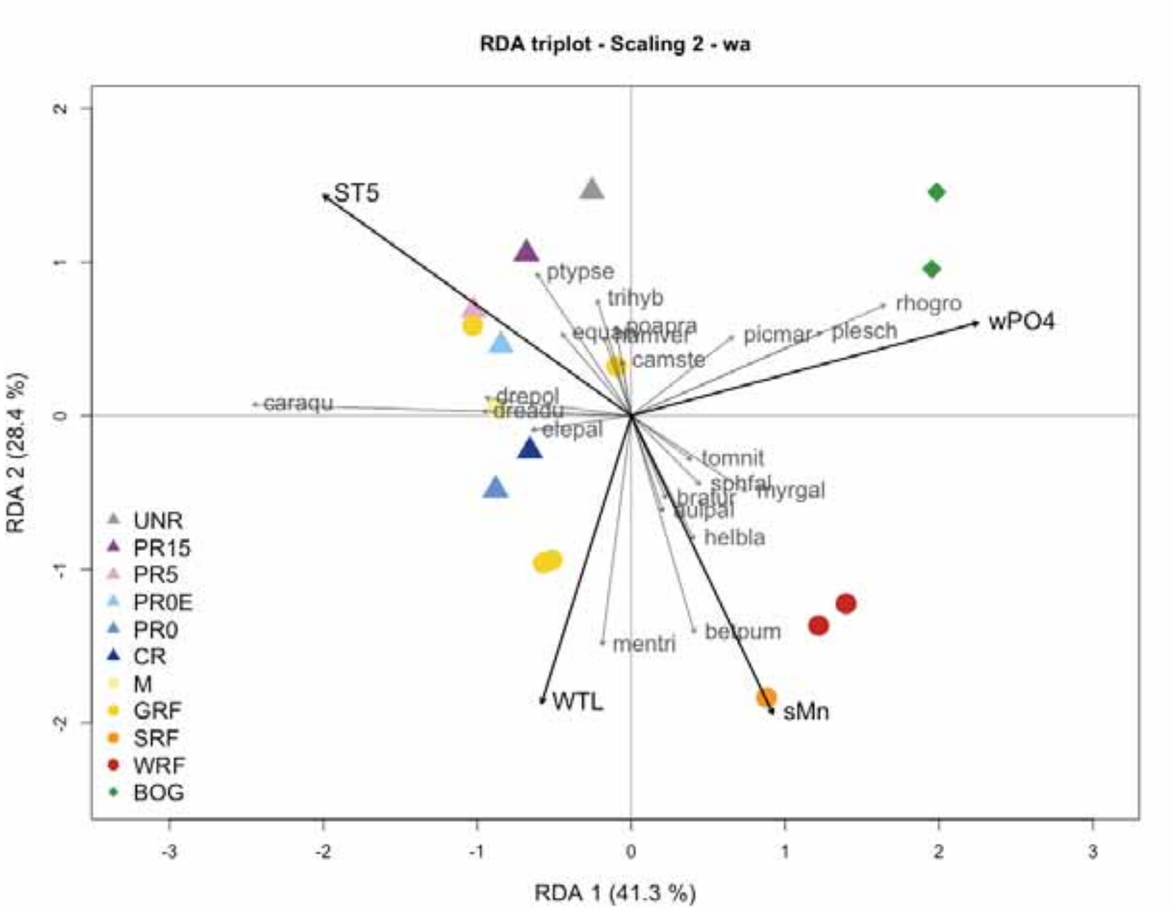


Figure 1.6 Redundancy analysis (RDA) triplot (scaling 2) with forward selection of environmental and chemical variables (ST5=soil temperature at 5 cm; wPO4=water extracted P-PO_4^{3-}; sMn=soil Mn; WTL=water table level) constraining the plant species cover. The reference area shrubby poor fen (SPF) does not appear, due to missing biochemical and environmental data. Code for study areas according to Method section and Figure 1.2Figure 1.3 (see also Appendix 1.1). Code for plant species in Appendix 1.2. For better visualization, not all plant species are shown.

The environmental variables soil Mn, water PO_4 and ST5 explained 69 to 96% of variation in the cover of plant functional types, except for herbs and sedges (Appendix 1.6). Brown moss cover was significantly higher with shallower WTL (p<0.0). The concentration of soil Mn was positively related to the cover of shrubs (p<0.0) and peat mosses (p<0.0), while being negatively related to the cover of other bryophytes (p<0.01). Significant positive regression relations were found between water P-PO_4^{3-} and the cover of the total vegetation, representing all plant functional types combined (p<0.04), the cover of ericaceous shrubs

($p<0.0$), of bryophytes ($p<0.02$), and of peat mosses ($p<0.0$). Soil temperature (ST5) had a significant negative regression relationship to the cover of trees ($p<0.0$), shrubs ($p<0.05$), and the peat mosses ($p<0.0$). No significant relationship was found between the environmental variables and cover of herbs or sedges.

1.6 Discussion

1.6.1 Vegetation development

In regard of our first research question, the investigation of the plant communities in all restored areas indicated that active reintroduction of characteristic plant species does not seem crucial to establish peatland characteristic vegetation. Instead, we suggest that revegetation alone is not sufficient for a restored area to recover, but site-specific hydrological conditions and location within proximity to a nearby diverse diaspore pool are the more important factors to create the best chance for desired vegetation reestablishment and peatland recovery.

Eight to ten years post-restoration, the vegetation composition in the restored areas PR0, PR0E and the floating moss carpet in CR is comparable to reference fens. In all three restored areas, spontaneous revegetation followed the mineral fill removal. While the species composition of CR's floating moss carpet compares to the vegetation of graminoid rich fens (GRF), the moss carpet is not representative of the entire restored area, which is dominated by shallow open water. We suspect the proximity of CR-D, PR0 and PR0E to adjacent undisturbed fen ecosystems and the meticulous reconnection of hydrology to have had positive impact on the return of characteristic biochemical processes and opened the newly reclaimed wetland to nearby natural diaspore pools (Campbell et al. 2003; Price et al. 2010; Rochefort et al. 2016).

The prolonged, excessive flooding of previously disturbed and restored wetlands has been perceived to promote plant communities other than fen vegetation (Caners & Lieffers 2014; Kreyling et al. 2021). The same effect was observed in CR, where a marsh-like vegetation developed in the shallow open water area. The marsh-like vegetation may take decades to centuries to transform into a functional peatland, if this occurs at all (Kuhry et al. 1993;

Caners & Lieffers 2014; Kreyling et al. 2021). In contrast, the floating moss carpet has a high potential to accumulate great amounts of peat and drive the peatland succession via terrestrialization (Joosten & Clarke 2002 ; Asada et al. 2005). However, the success of terrestrialization via the expansion of the floating moss carpet and subsequent peat formation may prove difficult in large-scale restoration with large areas of turbulent waters (Bechstein et al. 2010; Caners & Lieffers 2014). Furthermore, unlike the aim of ecological restoration, this scenario is impossible to achieve in the near future (Gann et al. 2019). Other research shows, that the CR approach has been effective for peatland restoration (Cooper et al. 2017; Xu et al. 2021). Many questions remain to be answered for the significant differences between the CR approaches of the different studies that should be tested in large-scale trials for practicality.

In terms of plant species composition, PR0 and PR0E are comparable to the graminoid and shrubby rich fens (SRF). While both areas PR0 and PR0E show a high species diversity and very high species evenness, they are up to 43% less species rich than REF. The species composition is dominated by more than 85% PEAT and OTHW species combined, demonstrating the driving force of the water table level (WTL) and chemical conditions for the natural ingress of a diverse, large spectrum of peatland species (Keddy 1999; Hedwall et al. 2017; Wagg et al. 2017). Additionally, microtopography (differences in elevation of the ground surface) has been observed to increase the successional recovery of fen characteristic plant species in restored cutover peatlands and following oil sands exploration (Pouliot et al. 2011; Caners & Lieffers 2014). This effect is represented in the dry and wet microforms within PR0 that resemble the dry, moss dominated hummocks and the wet, herb dominated hollows found in shrubby rich fens and graminoid rich fens (SRF, GRF).

Initially, we had expected the species composition of PR5 and PR15 to compare to reference fens (GRF, SRF, WRF), since both restored areas received active introduction of characteristic fen plant seedlings. On the contrary, the species composition in both areas, PR5 and PR15, was even less peatland species rich than the unrestored area (UNR). Fattorini and Halle (2004) and Moreno-Mateos et al. (2012) argue that active revegetation, as done at PR5 and PR15, hinders natural ingress from nearby diaspores. The same effect was observed in PR5, where the lowest species richness and diversity was measured, despite the dominance of planted *C. aquatilis*, a PEAT and characteristic fen species. The naturally emerging brown mosses observed in PR5 and PR15 in the beginning of our first study season, struggled to survive the two seasons of the entire study. During the first season, the bryophytes desiccated and dies due to drought and a low water table level (Kuhry et al. 1993; Churchill et al. 2015). Other studies in restored peatlands following *in situ* oil sands exploration disturbances show that especially during periods of drought, emerging bryophytes are suppressed by abundant, untargeted graminoid herb species, such as *C. inexpansa* and *Poa* sp., and very dense *Cyperaceae*-communities with abundant *C. aquatilis* (Caners & Lieffers 2014; Churchill et al. 2015). This succession towards a vegetation community dominated by undesired, non-wetland graminoid herb species was also observed in PR15 that experienced continuously dry conditions in 2017 due to its higher elevation relative to water table level. The few bryophytes that had persisted through the drought of 2017, eventually died during the 2018 season with extended periods of inundation that are also known to hinder moss establishment (Caners & Lieffers 2014; Granath et al. 2010).

1.6.2 Effects of environmental parameters on the vegetation

In regard to our second research question, we found a residual mineral fill a suitable base layer supporting the development of peatland characteristic vegetation, despite enrichment in base cations. We consider the partial removal to the water table level (PR0 and PR0E) and subsequent biochemical and environmental conditions to be the most successful restoration approach assessed in this study, based on the area's abundant peatland characteristic plant species compositions. The return of characteristic physical soil properties and biochemistry are important for a successful peatland restoration (Davidson et al. 2020; Saraswati et al. 2020*a*; 2020*b*). The complete removal (CR) approach assessed in our study did not result in transforming a disturbed peatland into a functional peatland ecosystem comparable to the reference models, at least not within a decadal timeframe (Gann et al. 2019). The peat was unable to rebound to the level prior disturbance, although the mineral fill was removed, and the peat resurfaced. Long-time compression and subsequent inundation meant that there was no suitable substrate for potential peatland vegetation to establish on, and instead mostly undesired aquatic species flourished and at the edges of the open water area, a marsh-like community with dominant *T. latifolia* was able to colonize.

At PR5 and PR15 environmental variables did not compare to characteristic peatland conditions as observed in the surrounding bog or the other reference areas, ten years post-restoration. Characteristic peatland plant species migrating from the adjacent bog are less likely to survive in the environmental conditions found in PR5 and PR15, where a residual mineral fill remains in place without neither steady hydrological connection, nor much influence from well pad surface runoff. The mineral fill appears to impede hydrologic and hydraulic reconnection to the adjacent bog's relatively low ground and surface water table level, and to alter the biochemistry of the restored areas to less suitable conditions for peatland vegetation. The low and acidic pH 3.6 and 3.4 that we observed in PR5 and PR15 respectively, might be caused by sulfuric acid created under high S-SO_4^{2-} concentrations (297 and 268 mg/L). In the water we observed extremely high EC (2574 and 2456 µS/cm), and high base cation concentrations of Ca (273 and 208 mg/L) and Mg (109 and 99 mg/L). While the abundant sedge vegetation's high root density might contribute to enhanced sulfate reduction (Altor 2016), the elevated Mg and S-SO_4^{2-} concentrations at PR5 and PR15 are likely a result of the residual mineral fill, which is known to be rich in salts in the Peace River region (Alberta Environment 2001). Given the combination of high salt concentrations and little hydrological exchange, the dominance of the planted *C. aquatilis* and *Salix* sp. with little bryophyte ingress on PR5 and PR15 is no surprise. This agrees with other studies of constructed peatlands in the oil sands regions with emerging communities of dense graminoid and shrubby rich fen vegetation species, that tolerate saline conditions, including *C. aquatilis, Typha* sp., *L. laricina*, and *Salix* sp. (Biagi et al. 2019; Hartsock et al. 2021*a;* 2021*b*).

In contrast, in the residual mineral fill affected restored areas PR0 and PR0E, biochemical conditions were found similar to extreme-rich fens, eight years post-restoration. The residual mineral fill certainly influenced soil and water chemistry but did not seem to hinder ingress of fen characteristic plant species and the development of robust vegetation communities, as the biochemistry remains within the natural range of variation (Vitt et al. 1995). This indicates that paludification can be instigated and develop a healthy mire if hydrological reconnection is optimized and diaspore sources are nearby (Kuhry et al. 1992; Vitt 1994; Kroetsch et al. 2011).

Soil temperature at 5 cm depth (ST5) had a significant impact on the vegetation cover, particularly of trees, ericaceous shrubs, and peat mosses. Various studies have found soil warming a serious threat to the health of peatland vegetation communities by altering the plant species composition and the associated quality of plant organic matter input that is important for ongoing peat formation (Moore 2002; Juszczak et al. 2013; De Long et al. 2016). Furthermore, warmer soil temperatures lead to accelerated desiccation and decomposition processes that triggered enhanced greenhouse gas emissions (Hedwall et al. 2017; Gong et al. 2018; Lemmer et al. 2020). In our study, we found the colder the soil temperature the higher the total vegetation cover was. In the exposed restored areas without trees and shrubs spending cooling shade, and without a cooling hydrological connection, as in PR5 and PR15, the warmer ST5 may have been the benefiting factor for non-wetland species to dominate the vegetation community, instead of characteristic peatland species.

The unrestored area (UNR) was dominated by non-wetland species, without comparable traits to reference areas (REF), ten years post-restoration. The large ecological distance between UNR and REF highlights the earnest need for ecological peatland restoration intervention following *in situ* oil sands disturbances. To evaluate future wetland restoration trials and success, we need a network of regional reference peatlands. Reference peatlands serve as important benchmarks, to know which conditions and functions to target under which circumstances. With this study we have started to build a comprehensive database of regional reference peatlands. To continue this process now is especially important considering the fast growing *in situ* infrastructure network and the threat of losing potential reference peatlands to the ongoing expansion.

Our findings indicate that given a proper hydrological connection to a surface-near water table and an adjacent peatland ecosystem, a residual mineral fill following the partial removal of an *in situ* oil sands well pad can support the ingress and development of characteristic peatland vegetation, particularly rich fen species. It became clear that the development of the vegetation communities is dependent on the site-specific biochemistry. We observed large variation in the biochemistry and environmental variables between study areas that was most likely driven by the combination of local environmental factors and the chemistry of the respective mineral fill. We stress the importance of careful groundwork and site management suited to local conditions and ecosystem mosaics, and the vital need for a nearby fen as diaspore source for a desired ingress of diverse characteristic plant species. Hence, the need for further studies on replicated peatland restoration trials to better evaluate the effect of a residual mineral fill on the peatland's biochemistry and plant species composition, as well as the factors leading to the shallow open water formation and how to avoid it. First, we recommend further trials of the complete removal groundwork as peatland restoration approach, to understand the peat structure and the peatland's response to compression and rewetting following the complete removal of a well pad's mineral fill. Secondly, we strongly suggest further large-scale trials (entire *in situ* well pad) testing the partial removal approach, to evaluate the development of biochemistry, environmental drivers, and vegetation communities at operational scales.

1.7 Implications for Practice

- When restoring a fen following *in situ* oil sands well pad disturbances, the surface-near hydrology should be re-established by adjusting the soil surface at the summer average water table level of adjacent peatlands, while the development of shallow open water should be avoided at all costs.
- Partial removal of mineral fill can support fen vegetation growth, while complete removal may lead to shallow open water with mostly aquatic species.
- Characteristic peatland vegetation resembling graminoid rich fens can successfully emerge on residual mineral fill, if a surface-near hydrology is reinstated and a diaspore source is in close range (adjacent fen).
- Careful characterization of the pre-disturbance peatland type and the adjacent peatland can help to determine the restoration target, and methods to promote natural ingress and succession of characteristic peatland species.

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1.10 Appendices3

Appendix 1.1 Characteristics of the studied unrestored control area (UNR), all restored areas (RES) and the reference wetlands (REF). The five restored areas are defined by the individual restoration method applied, especially the management of mineral fill removal, such as the partial removal of mineral soil to 15 cm above the water table level (PR15), the partial removal of mineral soil to 5 cm above the (PR5), the partial removal of mineral soil to 0 cm even (PR0E), the partial removal of mineral to 0 cm with dry and wet microsites (PR0), and the complete removal of the well pad and all construction materials (CR). The reference wetlands include a marsh (M), graminoid rich fens (GRF), shrubby rich fens (SRF), wooded rich fens (WRF), shrubby poor fens (SPF), and wooded bogs (BOG). Wetland classification was done according to k-means clustering. All sites are located in the oil sands regions of Cold Lake (CL) and Peace River (PR), Alberta. The unrestored and restored areas were each considered as an individual class, in order to permit comparison of restoration treatments with REF.

Status	Study area	Site ID	Coordinates		Region	Survey date	Total area surveyed	
							vascular	bryophytes
UNR	Unrestored	UNR	54°41'10.13" N	110°30'55.68" W	CL	25.07.2018	5 × 1 m^2	20 × 0.06 m^2
RES	Partial removal 15 cm	PR15	56°23'0.02" N	116°46'42.19" W	PR	06.08.2018	5 × 1 m^2	20 × 0.06 m^2
RES	Partial removal 5 cm	PR5	56°23'1.65" N	116°46'40.19" W	PR	06.08.2018	5 × 1 m^2	20 × 0.06 m^2
RES	Partial removal 0 cm (even)	PR0E	54°41'10.73" N	110°31'3.62" W	CL	25.07.2018	5 × 1 m^2	20 × 0.06 m^2
RES	Partial removal 0 cm (microsites)	PR0	54°41'12.81" N	110°31'0.61" W	CL	25.07.2018	5 × 1 m^2	20 × 0.06 m^2
RES	Complete removal	CR	54°41'11.66" N	110°30'55.43" W	CL	25.07.2018	5 × 1 m^2	20 × 0.06 m^2
REF	Freshwater marsh	HVMARSH	56° 6'24.15" N	116°35'32.17" W	PR	22.08.2018	5 × 1 m^2	20 × 0.06 m^2
REF	Graminoid rich fen	LUBICON	56°19'5.65" N	116° 7'14.92" W	PR	03.08.2011	10 × 1 m^2	10 × 1 m^2
REF	Graminoid rich fen	NABIYEN	54°42'48.84" N	110°19'17.28" W	CL	13.08.2018	5 × 1 m^2	20 × 0.06 m^2
REF	Graminoid rich fen	NABIYES	54°39'47.50" N	110°24'40.72" W	CL	09.08.2018	5 × 1 m^2	20 × 0.06 m^2
REF	Graminoid rich fen	HVEAST	56° 6'45.49" N	116°29'11.09" W	PR	22.08.2018	5 × 1 m^2	20 × 0.06 m^2
REF	Graminoid rich fen	MAHIHKAN	54°37'13.62" N	110°33'36.12" W	CL	13.08.2018	5 × 1 m^2	20 × 0.06 m^2
REF	Graminoid rich fen	SKEGFEN	56°22'56.60" N	116°46'41.40" W	PR	08.08.2011	10 × 1 m^2	10 ×1 m^2
REF	Shrubby rich fen	CONKLINN1	55°38'60.00" N	111° 7'46.00" W	CL	08.07.2016	n.a.	6 × 0.39 m^2
REF	Shrubby rich fen	CONKLINN2	55°38'42.00" N	111° 8'2.00" W	CL	08.07.2016	n.a.	6 × 0.39 m^2
REF	Shrubby rich fen	CONKLINS1	55°29'18.00" N	111° 5'51.00" W	PR	07.07.2016	n.a.	6 × 0.39 m^2
REF	Shrubby rich fen	UTIKUMA1	55°46'43.00" N	115°58'16.00" W	PR	14.07.2016	n.a.	12 × 0.39 m^2
REF	Shrubby rich fen	UTIKUMA2	55°42'36.00" N	116° 2'30.00" W	PR	14.07.2016	n.a.	12 × 0.39 m^2
REF	Shrubby rich fen	WIAU2	55°21'7.00" N	111°14'54.00" W	PR	08.07.2016	n.a.	6 × 0.39 m^2

REF	Wooded rich fen	CCEAST	56°22’7.90” N	116°48’58.80” W	PR	09.08.2011	10 × 1 m^2	10 × 1 m^2
REF	Wooded rich fen	CCWEST	56°22’23.70” N	116°49’31.90” W	PR	08.08.2011	10 × 1 m^2	10 × 1 m^2
REF	Wooded rich fen	CNRLS	56°21’53.51” N	116°46’29.79” W	PR	24.08.2011	10 × 1 m^2	10 × 1 m^2
REF	Wooded rich fen	H38N	54°41’15.07” N	110°31’0.09” W	CL	25.07.2018	5 × 1 m^2	20 × 0.06 m^2
REF	Wooded rich fen	H38W	54°41’8.63” N	110°31’4.48” W	CL	25.07.2018	5 × 1 m^2	20 × 0.06 m^2
REF	Wooded rich fen	HVFEN	56° 9’32.15” N	116°58’57.46” W	PR	09.08.2018	5 × 1 m^2	20 × 0.06 m^2
REF	Shrubby poor fen	CONKLINS2	55°28’44.00” N	111° 4’41.00” W	PR	08.07.2016	n.a.	6 × 0.39 m^2
REF	Shrubby poor fen	CNRLN	56°24’10.70” N	116°45’5.80” W	PR	05.08.2011	10 × 1 m^2	10 × 1 m^2
REF	Shrubby poor fen	FOXCREEK	54°21’3.21” N	116°37’40.92” W	PR	04.08.2011	10 × 1 m^2	10 × 1 m^2
REF	Shrubby poor fen	HPWEST	55°21’30.91” N	116°51’49.70” W	PR	10.08.2011	10 × 1 m^2	10 × 1 m^2
REF	Shrubby poor fen	WIAU1	55°20’0.00” N	111°16'53.00" W	PR	07.07.2016	n.a.	6 × 0.39 m^2
REF	Wooded bog	HPEAST	55°21'53.39" N	116°48'27.41" W	PR	23.08.2011	11 × 1 m^2	11 × 1 m^2
REF	Wooded bog	HVBOG	56° 9'35.79" N	116°59'40.74" W	PR	09.08.2018	5 × 1 m^2	20 × 0.06 m^2
REF	Wooded bog	SKEGBOG	56°22'58.73"N	116°46'38.22"W	PR	06.08.2018	5 × 1 m^2	20 × 0.06 m^2
REF	Wooded bog	TCREEKS	56°22'3.60"N	116°57'30.80"W	PR	25.08.2011	10 × 1 m^2	10 × 1 m^2

Appendix 1.2 Complete plant species list, indicating the wetland indicator status WIS (PEAT=peatland species: essential for peatland community in Alberta, preferentially bogs and fens; OTHW=other wetland species: obligate and facultative wetland species; NONW=non-wetland species/upland species/generalists), inspired by Gillrich and Bowman (2010), Payette & Rochefort (2013), NRCS (2021), and Tiner (1993). The wetland class indicates the representative habitat (B=bog, F=fen, M=marsh, S=swamp, W=shallow open water) following (Alberta Environment and Sustainable Resource Development 2015). To be considered as fen characteristic plant species (FCPS), species need to belong into the WIS class PEAT or OTHW, while preferential habitat is fen. Nomenclature and taxonomy follow CNALH (2021) and ITIS (2021). Following the (Alberta Invasive Species Council 2021), only one invasive species (Sonchus arvensis) was observed but is not specifically indicated.

Life form	Code	Latin name	WIS	Wetland class	FCPS
Tree	betpap	*Betula papyrifera*	NONW	S	
Tree	larlar	*Larix laricina*	PEAT	F, S	
Tree	pic_sp	*Picea* sp.	NONW		
Tree	picmar	*Picea mariana*	PEAT	B, F, S	
Tree	popbal	*Populus balsamifera*	OTHW	S	
Tree	poptre	*Populus tremuloides*	NONW		
Shrub	alnrug	*Alnus incana* ssp. *rugosa*	PEAT		
Shrub	alnten	*Alnus incana* ssp. *tenuiflora*	OTHW	S	
Shrub	alnaln	*Alnus viridis* ssp. *crispa*	OTHW		
Shrub	bet_sp	*Betula* sp.	NONW		
Shrub	betgla	*Betula glandulosa*	OTHW	F	
Shrub	betpum	*Betula pumila* var. *glandulifera*	PEAT	B, F, S	FCPS
Shrub	sal_sp	*Salix* sp.	NONW		
Shrub	salbra	*Salix brachycarpa*	OTHW		
Shrub	salcan	*Salix candida*	OTHW	F, S	FCPS
Shrub	salexi	*Salix exigua*	OTHW	F, S	
Shrub	salgla	*Salix glauca*	OTHW	F, S	
Shrub	sallas	*Salix lasiandra*	OTHW		
Shrub	sallut	*Salix lutea*	OTHW		
Shrub	salped	*Salix pedicellaris*	PEAT	F, S	FCPS
Shrub	salpla	*Salix planifolia*	OTHW	F, S	FCPS
Shrub	salpro	*Salix prolixa*	OTHW		
Shrub	sarpur	*Salix purpurea*	OTHW		
Shrub	salpyr	*Salix pyrifolia*	OTHW	F, S	FCPS
Shrub	dasfru	*Dasiphora fruticosa*	OTHW		
Shrub	junhor	*Juniperus horizontalis*	NONW		
Shrub	myrgal	*Myrica gale*	PEAT	F, S	FCPS

Shrub	rib_sp	*Ribes* sp.	NONW		
Shrub	rub_sp	*Rubus* sp.	NONW		
Ericaceous	andpol	*Andromeda polifolia*	PEAT	B, F, S	FCPS
Ericaceous	chacal	*Chamaedaphne calyculata*	PEAT	B, F	FCPS
Ericaceous	empnig	*Empetrum nigrum*	PEAT	B, F	
Ericaceous	kalpol	*Kalmia polifolia*	PEAT	B, F, S	
Ericaceous	rhogro	*Rhododendron groenlandicum*	PEAT	B, F, S	FCPS
Ericaceous	vacces	*Vaccinium cespitosum*	OTHW		
Ericaceous	vacmic	*Vaccinium microcarpum*	PEAT	B, F, S	FCPS
Ericaceous	vacvit	*Vaccinium vitis-idaea*	PEAT	B, F, M, S	
Herb	achmil	*Achillea millefolium*	NONW	M, S	
Herb	agrsca	*Agrostis scabra*	NONW	F, M, S	
Herb	aloaeq	*Alopecurus aequalis*	OTHW	M	
Herb	ast_sp	*Aster* sp.	NONW		
Herb	becsyz	*Beckmannia syzigachne*	OTHW	M	
Herb	calcan	*Calamagrostis canadensis*	PEAT	F, M, S	
Herb	caline	*Calamagrostis stricta* ssp. *inexpansa*	OTHW	F, M, S	
Herb	calstr	*Calamagrostis stricta* ssp. *stricta*	OTHW	F, M, S	
Herb	callpal	*Calla palustris*	OTHW	F, M, S	FCPS
Herb	caltpal	*Caltha palustris*	OTHW	F, M, S	FCPS
Herb	chaang	*Chamaenerion angustifolium* ssp. *angustifolium*	NONW	F, M, S	
Herb	chalat	*Chamaenerion latifolium*	OTHW	M, F	
Herb	cicbul	*Cicuta bulbifera*	OTHW	F, M, S	
Herb	compal	*Comarum palustre*	PEAT	B, F, M, S	FCPS
Herb	desces	*Deschampsia cespitosa*	OTHW	B, F, M	
Herb	drorot	*Drosera rotundifolia*	PEAT	B, F, S	FCPS
Herb	elygla	*Elymus glaucus*	NONW		
Herb	elytra	*Elymus trachycaulus*	NONW	M	
Herb	epipal	*Epilobium palustre*	OTHW	F, M, S	FCPS
Herb	equarv	*Equisetum arvense*	OTHW	B, F, M, S	
Herb	equhye	*Equisetum hyemale*	OTHW	M	
Herb	equpra	*Equisetum pratense*	OTHW	F, M, S	
Herb	fesovi	*Festuca ovina*	NONW		
Herb	fessax	*Festuca saximontana* var. *saximontana*	NONW		
Herb	fravir	*Fragaria virginiana*	NONW	M	
Herb	galbor	*Galium boreale*	NONW	B, S	

Herb	gallab	*Galium labradoricum*	PEAT	B, F, M, S	FCPS
Herb	galtri	*Galium trifidum*	OTHW	B, F, M, S	
Herb	gauhis	*Gaultheria hispidula*	NONW		
Herb	geo_sp	*Geocaulon* sp.	PEAT		
Herb	geoliv	*Geocaulon lividum*	PEAT	B, F, S	
Herb	gly_sp	*Glyceria* sp.	OTHW		
Herb	horjub	*Hordeum jubatum*	NONW	M	
Herb	juneff	*Juncus effusus* ssp. *effusus*	OTHW		
Herb	lobkal	*Lobelia kalmii*	OTHW	M	
Herb	lysthy	*Lysimachia thyrsiflora*	PEAT	B, F, M, S	FCPS
Herb	maican	*Maianthemum canadense*	NONW	S	
Herb	maitri	*Maianthemum trifolium*	PEAT	B, F, M, S	FCPS
Herb	melalb	*Melilotus albus*	NONW		
Herb	meloff	*Melilotus officinalis*	NONW	M	
Herb	mentri	*Menyanthes trifoliata*	PEAT	F, S	FCPS
Herb	mitnud	*Mitella nuda*	OTHW	B, F, M, S	
Herb	parpal	*Parnassia palustris*	OTHW	B, F, M, S	FCPS
Herb	pet_sp	*Petasites* sp.	OTHW		
Herb	petsag	*Petasites frigidus* var. *sagittatus*	OTHW	F, M, S	
Herb	phaaru	*Phalaris arundinaceae*	OTHW	M	
Herb	phlpra	*Phleum pratense*	NONW	M	
Herb	pinvul	*Pinguicula vulgaris*	OTHW		
Herb	plamaj	*Plantago major*	NONW		
Herb	plahyp	*Platanthera hyperborea*	OTHW	F, M, S	
Herb	plaobt	*Platanthera obtusata* ssp. *obtusata*	OTHW	F, S	
Herb	poa_sp	*Poa* sp.	NONW		
Herb	poapal	*Poa palustris*	NONW	F, M, S	
Herb	poapra	*Poa pratensis*	NONW	M	
Herb	pyr_sp	*Pyrola* sp.	NONW	F, S	
Herb	ranaqu	*Ranunculus aquatilis*	OTHW	M, W	
Herb	rangme	*Ranunculus gmelinii*	OTHW	B, F, M, S	
Herb	rhibor	*Rhinanthus minor* ssp. *groenlandicus*	NONW		
Herb	rubaca	*Rubus arcticus* ssp. *acaulis*	PEAT	B, F, M, S	FCPS
Herb	rubcha	*Rubus chamaemorus*	PEAT	B, F, S	FCPS
Herb	rubpub	*Rubus pubescens*	OTHW	B, F, M, S	
Herb	rum_sp.	*Rumex* sp.	NONW		

Herb	rumbri	*Rumex britannica*	OTHW	M, S	
Herb	rumocc	*Rumex occidentalis*	OTHW		
Herb	sol_sp	*Solidago* sp.	NONW		
Herb	solgig	*Solidago gigantea*	NONW	M	
Herb	sonarv	*Sonchus arvensis*	NONW	M, S	
Herb	sonasp	*Sonchus asper*	NONW	M	
Herb	spaang	*Sparganium angustifolium*	OTHW	F, M, W	FCPS
Herb	spaeur	*Sparganium eurycarpum*	OTHW	M	
Herb	spirom	*Spiranthes romanzoffiana*	OTHW	B, F, M, S	FCPS
Herb	stelon	*Stellaria longifolia*	OTHW	F, M, S	
Herb	symlan	*Symphyotrichum lanceolatum* ssp. *hesperium*	OTHW	M	
Herb	taroff	*Taraxacum officinale*	NONW	M, S	
Herb	triglu	*Triantha glutinosa*	OTHW	M, F, S	
Herb	trices	*Trichophorum cespitosum*	PEAT	B, F, M	FCPS
Herb	trihyb	*Trifolium hybridum*	NONW	M	
Herb	tripra	*Trifolium pratense*	NONW		
Herb	trirep	*Trifolium repens*	NONW	M	
Herb	trimar	*Triglochin maritima*	PEAT	F, M	FCPS
Herb	typlat	*Typha latifolia*	OTHW	F, M	FCPS
Herb	val_sp	*Valeriana* sp.	NONW		
Herb	vicame	*Vicia americana*	NONW	F, M, S	
Herb	vio_sp	*Viola* sp.	NONW		
Sedge	car_sp	*Carex* sp.	OTHW		
Sedge	caraqu	*Carex aquatilis*	PEAT	F, M, S	FCPS
Sedge	cararc	*Carex arcta*	OTHW		
Sedge	carath	*Carex atherodes*	OTHW	F, M, S	FCPS
Sedge	caraur	*Carex aurea*	OTHW	B, F, M, S	
Sedge	carbeb	*Carex bebbii*	OTHW	F, M, S	FCPS
Sedge	carcan	*Carex canescens*	PEAT	B, F, M, S	FCPS
Sedge	carcap	*Carex capillaris*	OTHW	B, F, M, S	
Sedge	carcho	*Carex chordorrhiza*	PEAT	B, F, M, S	FCPS
Sedge	cardew	*Carex deweyana*	NONW	S	
Sedge	cardia	*Carex diandra*	PEAT	F, M, S	FCPS
Sedge	cardis	*Carex disperma*	OTHW	B, F, S	
Sedge	cargyn	*Carex gynocrates*	PEAT	B, F, M, S	FCPS
Sedge	carint	*Carex interior*	OTHW	F, M, S	FCPS

Sedge	carlas	*Carex lasiocarpa*	PEAT	B, F, M, S	FCPS
Sedge	carlim	*Carex limosa*	PEAT	F, M, S	FCPS
Sedge	carmag	*Carex magellanica*	PEAT	B, F, S	FCPS
Sedge	carirr	*Carex magellanica* ssp. *irrigua*	PEAT	B, F, S	FCPS
Sedge	carros	*Carex rostrata*	PEAT	F, S	FCPS
Sedge	carsar	*Carex sartwellii*	OTHW	M	
Sedge	carten	*Carex tenuiflora*	PEAT	B, M, S	FCPS
Sedge	cartri	*Carex trisperma*	PEAT	B, F, M, S	FCPS
Sedge	carutr	*Carex utriculata*	OTHW	B, F, M	FCPS
Sedge	carvag	*Carex vaginata*	PEAT	B, F, M, S	FCPS
Sedge	elepal	*Eleocharis palustris*	OTHW	M	
Sedge	eri_sp	*Eriophorum* sp.	OTHW		
Sedge	eriang	*Eriophorum angustifolium*	PEAT	B, F, M, S	FCPS
Sedge	erivir	*Eriophorum viridicarinatum*	PEAT		FCPS
Sedge	sci_sp	*Scirpus* sp.	OTHW	M	
Sedge	scicyp	*Scirpus cyperinus*	OTHW	M	
Sedge	trialp	*Trichophorum alpinum*	OTHW	F, M	FCPS
Aquatic	hipvul	*Hippuris vulgaris*	OTHW	F, M	
Aquatic	lemmin	*Lemna minor*	OTHW	M	
Aquatic	myrsib	*Myriophyllum sibiricum*	OTHW	M, W	
Aquatic	potgra	*Potamogeton gramineus*	OTHW	M, W	
Aquatic	stufil	*Stuckenia filiformis*	OTHW	M	
Brown moss	aulpal	*Aulacomnium palustre*	PEAT	B, F, S	FCPS
Brown moss	braacu	*Brachythecium acutum*	OTHW	F, S	
Brown moss	bratur	*Brachythecium turgidum*	OTHW	B, F, S	
Brown moss	calcor	*Calliergon cordifolium*	OTHW	F	FCPS
Brown moss	calgig	*Calliergon giganteum*	OTHW	F	FCPS
Brown moss	calric	*Calliergon richardsonii*	OTHW	F, S	FCPS
Brown moss	cal_sp	*Calliergon* sp.	OTHW		
Brown moss	calcus	*Calliergonella cuspidata*	OTHW	F	FCPS
Brown moss	callin	*Calliergonella lindbergii*	OTHW	B, F	
Brown moss	calasp	*Calliergonella* sp.	NONW		
Brown moss	camste	*Campylium stellatum*	PEAT	F	FCPS
Brown moss	catnig	*Catoscopium nigritum*	OTHW	F	
Brown moss	cerpur	*Ceratodon purpureus*	NONW	F, S	
Brown moss	cinsty	*Cinclidium stygium*	PEAT	B, F, S	FCPS

Brown moss	cliden	*Climacium dendroides*	OTHW	F, S	FCPS
Brown moss	crafil	*Cratoneuron filicinum*	OTHW		
Brown moss	dicund	*Dicranum undulatum*	PEAT	B, F	FCPS
Brown moss	dreadu	*Drepanocladus aduncus*	OTHW	F, M, S	
Brown moss	drepol	*Drepanocladus polygamus*	OTHW	F	FCPS
Brown moss	dresor	*Drepanocladus sordidus*	OTHW	F	
Brown moss	hamver	*Hamatocaulis vernicosus*	PEAT	F	FCPS
Brown moss	helbla	*Helodium blandowii*	PEAT	B, F, S	FCPS
Brown moss	hylspl	*Hylocomium splendens*	NONW	B, F, S	
Brown moss	hyppra	*Hypnum pratense*	PEAT	F, S	FCPS
Brown moss	leppyr	*Leptobryum pyriforme*	NONW	B, F, S	
Brown moss	meetri	*Meesia triquetra*	PEAT	F, S	FCPS
Brown moss	micrec	*Microbryum rectum*	NONW		
Brown moss	palsqu	*Paludella squarrosa*	PEAT	F	FCPS
Brown moss	pla_sp	*Plagiomnium* sp.	NONW		
Brown moss	plaell	*Plagiomnium ellipticum*	PEAT	B, F, S	FCPS
Brown moss	plesch	*Pleurozium schreberi*	PEAT	B, F, S	
Brown moss	pohnut	*Pohlia nutans*	PEAT	B, F, S	
Brown moss	polstr	*Polytrichum strictum*	PEAT	B, F, S	FCPS
Brown moss	psecin	*Pseudobryum cinclidioides*	OTHW	B, F, S	FCPS
Brown moss	pticri	*Ptilium crista-castrensis*	OTHW	B, F, S	
Brown moss	ptycre	*Ptychostomum creberrimum*	NONW		
Brown moss	ptypal	*Ptychostomum pallens*			
Brown moss	ptypse	*Ptychostomum pseudotriquetrum*	OTHW	F	
Brown moss	ptywei	*Ptychostomum weigelii*	OTHW		
Brown moss	ptyrub	*Rosulabryum rubens*	NONW		
Brown moss	sanunc	Sanionia uncinata	OTHW	F	
Brown moss	sarexa	*Sarmentypnum exannulatum*	PEAT	B, F, S	FCPS
Brown moss	sartun	*Sarmentypnum tundrae*	PEAT	F	FCPS
Brown moss	scocos	Scorpidium cossonii	OTHW	F	FCPS
Brown moss	scosco	*Scorpidium scorpioides*	PEAT	F	FCPS
Brown moss	spllut	*Splachnum luteum*	NONW		
Brown moss	tomfal	*Tomentypnum falcifolium*	PEAT	B, F	FCPS
Brown moss	tomnit	*Tomentypnum nitens*	PEAT	F	FCPS
Brown moss	torlan	*Tortula lanceolata*	NONW		
Brown moss	warflu	Warnstorfia fluitans	PEAT	B, F, S	FCPS

Liverwort	anepin	*Aneura pinguis*	OTHW	F, S	
Liverwort	calsph	*Calypogeia sphagnicola*	PEAT	F, S	FCPS
Liverwort	lopven	*Lophozia ventricosa*	NONW	F, S	
Liverwort	marpol	*Marchantia polymorpha*	OTHW	B, F	
Liverwort	mylano	*Mylia anomala*	PEAT	B, F, S	FCPS
Peat moss	sphang	*Sphagnum angustifolium*	PEAT	B, F, S	FCPS
Peat moss	sphcap	*Sphagnum capillifolium*	PEAT	B, F, S	FCPS
Peat moss	sphfal	*Sphagnum fallax*	PEAT	B, F	FCPS
Peat moss	sphfim	*Sphagnum fimbriatum*	PEAT	B, F	FCPS
Peat moss	sphfus	*Sphagnum fuscum*	PEAT	B, F, S	FCPS
Peat moss	sphdiv	*Sphagnum divinum*	PEAT	B, F	FCPS
Peat moss	sphrub	*Sphagnum rubellum*	PEAT	B, F	FCPS
Peat moss	sphrus	*Sphagnum russowii*	PEAT	F	FCPS
Peat moss	sphsqu	*Sphagnum squarrosum*	PEAT	F, S	FCPS
Peat moss	sphter	*Sphagnum teres*	PEAT	F, S	FCPS
Peat moss	sphwar	*Sphagnum warnstorfii*	PEAT	F, S	FCPS
Lichen	Lichen_sp	Lichen sp.	NONW		
Lichen	clabot	*Cladonia botrytes*	NONW	B	
Lichen	clacor	*Cladonia cornuta*	NONW	B	
Lichen	claran	*Cladonia rangiferina*	PEAT	B	
Lichen	claste	*Cladonia stellaris*	PEAT	B	
Lichen	clasul	*Cladonia sulphurina*	NONW	B	
Lichen	pelaph	*Peltigera aphthosa*	OTHW		
Lichen	pelneo	*Peltigera neopolydactyla*	OTHW		

Appendix 1.3 Means ± SE of ***soil/peat*** *environmental factors, presenting the soil temperature at 5 cm depth (ST5), pH and electric conductivity (EC) and extractable major elements and the micronutrients. Meaning of study areas according to Appendix 1.1.*

Study area	n	ST5 (°C)	pH	EC (µS/cm)	Ca (mg/L)	Mg (mg/L)	K (mg/L)	Na (mg/L)	$N\text{-}NH_4^+$ (mg/L)	$N\text{-}NO_3^-$ (mg/L)	$P\text{-}PO_4^{3-}$ (mg/L)	$S\text{-}SO_4^{2-}$ (mg/L)	Al (mg/L)	Fe^+ (mg/L)	Mn (mg/L)
UNR	3	20.3 ± 0.1	8 ± 0.1	289.9 ± 15.9	2416.9 ± 257.1	721.2 ± 111.4	150.8 ± 11.2	58.5 ± 6.8	14.4 ± 0.4	4 ± 1.8	44.4 ± 3.2	24.8 ± 17.8	0.7 ± 0.5	0	5.6 ± 0.6
PR15	3	20.2 ± 1.5	4.9 ± 0.1	1326.8 ± 344.9	4651.9 ± 529.4	1921.6 ± 58.6	211.8 ± 19.2	128.6 ± 11.1	33.5 ± 6.8	1.4 ± 0.1	85.6 ± 1.9	5608.1 ± 256.3	0.5 ± 0.1	0	0.5 ± 0.1
PR5	3	19.4 ± 0.4	4.2 ± 0.1	1308.8 ± 339.5	3776 ± 498.2	1005.2 ± 326.7	198.5 ± 9.2	73 ± 13.3	20.9 ± 2.5	5.1 ± 2.7	113.7 ± 12.8	1638.1 ± 394.2	0.4 ± 0	0	3.6 ± 0.8
PR0	5	21.2 ± 0.7	3.2 ± 0.2	367.2 ± 11.9	2710.5 ± 196.1	466.4 ± 28.6	125.9 ± 12.7	55.1 ± 3.9	20.5 ± 4.2	2.4 ± 0.1	67.9 ± 7.6	52.4 ± 37.4	1.5 ± 0.9	0.5 ± 0.5	51.2 ± 23.2
PR0E	3	17.7 ± 0.5	5.0 ± 1.2	348.7 ± 6.6	2534 ± 113.1	464.9 ± 89.3	159.3 ± 39.8	45.4 ± 4.4	10.9 ± 1.4	7.1 ± 4.7	76.4 ± 11.5	9.7 ± 3.9	0.5 ± 0	0	3.5 ± 1
CR	3	18.7 ± 1.6	7 ± 0.9	128.7 ± 8.4	9575.5 ± 1235.6	2359.5 ± 164.5	869.5 ± 360.8	247.2 ± 26.4	168.6 ± 45.2	8.1 ± 1.8	267.5 ± 79.9	32.5 ± 8.3	1.7 ± 0	0.6 ± 0.1	55.9 ± 21.1
M	3	19.3 ± 0.2	4.6 ± 0.1	117.1 ± 6.4	9005.2 ± 701.6	1079.7 ± 70.8	689.2 ± 341.8	202.8 ± 47.5	768.2 ± 105.9	7.2 ± 0.5	228.9 ± 47.2	265.9 ± 95	3.4 ± 0.6	33.5 ± 9.7	84.6 ± 14
GRF	12	19.8 ± 0.1	4.2 ± 0.1	94.9 ± 10.3	11025.1 ± 1158.7	2131.3 ± 215.3	1342.0 ± 252.6	242.6 ± 30.8	190.6 ± 14.8	10.2 ± 0.9	127.2 ± 23.9	62.9 ± 6.4	6.9 ± 2.2	24.9 ± 6.3	264.7 ± 63
SRF	10	18.9 ± 1.2	6.2 ± 0.4	526.6 ± 136.6	13870.2 ± 713.7	2257.9 ± 197.9	1555.2 ± 173.7	451.4 ± 65.2	201 ± 45.7	5.9 ± 0.7	221.4 ± 41.5	82.9 ± 5.4	1.4 ± 0	6.2 ± 1.1	466.5 ± 225.9
WRF	8	15.4 ± 1.3	6.3 ± 0.5	84.3 ± 9.3	1566.1 ± 969.4	3671.2 ± 300.8	1286.3 ± 459.8	301 ± 66.2	296.2 ± 31.2	8.9 ± 0.4	248.5 ± 29.4	77.4 ± 7.5	1.7 ± 0.1	0.9 ± 0.3	402.9 ± 192.7
SPF	3	21.9 ± 0.8	7.1 ± 0.2	443 ± 33	11830 ± 4241.1	1916.4 ± 572.1	1008.3 ± 451.2	555.1 ± 81	58.1 ± 20.9	5.7 ± 5.7	65.5 ± 5.9	n.a.	n.a.	6 ± 1	136.2 ± 121.5
BOG	8	17.7 ± 0.8	5.2 ± 0.5	61.4 ± 9.2	8508.0 ± 1779.6	2062.6 ± 300.6	638.2 ± 88.6	219.3 ± 54	113.9 ± 7.9	8.2 ± 1.4	113.6 ± 13.7	109.7 ± 6.4	28.7 ± 11.1	50.9 ± 21.5	43.0 ± 5.3

Appendix 1.4 Means ± SE of the **water** environmental factors, presenting pH, electric conductivity (EC), and water extractable major elements and micronutrients. Meaning of study areas according to Appendix 1.1.

Study areas	n	pH	EC (µS/cm)	Ca (mg/L)	Mg (mg/L)	K (mg/L)	Na (mg/L)	$N\text{-}NH_4^+$ (mg/L)	$N\text{-}NO_3^-$ (mg/g)	$P\text{-}PO_4^{3-}$ (mg/g)	$S\text{-}SO_4^{2-}$ (mg/L)	Al (mg/g)	Fe^+ (mg/L)	Mn (mg/L)
UNR	3	3.6 ± 0.1	604.7 ± 22.5	19.8 ± 1.7	15.8 ± 0.6	0.9 ± 0.1	0.5 ± 0	0.7 ± 0.1	0	0	0.4 ± 0	0	0	0.1 ± 0.1
PR15	3	3.4 ± 2.9	2456 ± 2541.2	208.3 ± 39.6	99.4 ± 33.1	1.2 ± 0.2	0.8 ± 0.2	0.2 ± 0	0	0	267.9 ± 88.2	0	0	1.9 ± 0.5
PR5	3	3.6 ± 1.4	2573.7 ± 4643.4	273.2 ± 17.3	109.1 ± 35.7	1.8 ± 0.5	1.1 ± 0.2	0.9 ± 0.3	0	0	296.5 ± 33.9	0	0	2.1 ± 0.4
PR0	6	7.4 ± 0	344 ± 5.8	23.1 ± 2.7	10.8 ± 0.9	0.7 ± 0.1	0.3 ± 0	0.1 ± 0	0	0	1.2 ± 0.1	0	0.1 ± 0	0.3 ± 0.1
PR0E	3	7.3 ± 0.1	476.3 ± 134.4	21.1 ± 1.8	12.3 ± 3.2	0.3 ± 0.1	0.3 ± 0.1	0.1 ± 0	0	0	0.7 ± 0.2	0	0	0
CR	6	5.9 ± 0.1	289.9 ± 7.7	18.0 ± 1.1	9.1 ± 0.6	0.6 ± 0.1	0.5 ± 0	0.2 ± 0	0	0	1.5 ± 0	0	0.1 ± 0	0.1 ± 0
M	3	5.6 ± 2	218.2 ± 35.2	16.0 ± 1	4.0 ± 0.4	1 ± 0.2	0.1 ± 0	0.1 ± 0	0	0	1.8 ± 0.1	0	0.1 ± 0	0.1 ± 0
GRF	12	6.8 ± 0.5	191.1 ± 14.4	11.6 ± 0.7	6.3 ± 0.6	0.8 ± 0.1	0.3 ± 0	0.2 ± 0	0	0	2.1 ± 0.4	0.1 ± 0	0.5 ± 0.2	0.1 ± 0
SRF	5	6.9 ± 0.2	343.4 ± 37.5	15.7 ± 1.8	6.5 ± 0.8	0.5 ± 0.1	0.2 ± 0	0.1 ± 0	0	0	1.6 ± 0.2	0	0.2 ± 0	0.3 ± 0.2
WRF	8	6 ± 0.2	254.9 ± 24.9	13.7 ± 2.3	6.8 ± 1.1	1.9 ± 0.7	0.3 ± 0	0.1 ± 0	0	0	2.1 ± 0.5	0	0.1 ± 0	0.1 ± 0
SPF	3	6.1 ± 0.1	505 ± 250	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.
BOG	8	4.7 ± 0.5	155.9 ± 26.5	4.5 ± 0.9	2.2 ± 0.3	1.3 ± 0.2	0.4 ± 0.1	0.2 ± 0	0.09 ± 0.01	0.01 ± 0.01	5.2 ± 0.3	0.04 ± 0.02	0.3 ± 0.1	0

Appendix 1.5 Mean ± SE cover (%) of plant functional types (PFT) and specifically abundant species within respective PFT, open water (water at the surface), plant litter, and total peatland characteristic plant species (across PFT). Physically overlapping species within different strata may cause the total vegetation cover to exceed 100%. Codes for study areas according to Appendix 1.1. Plant species codes as in Appendix 1.2.

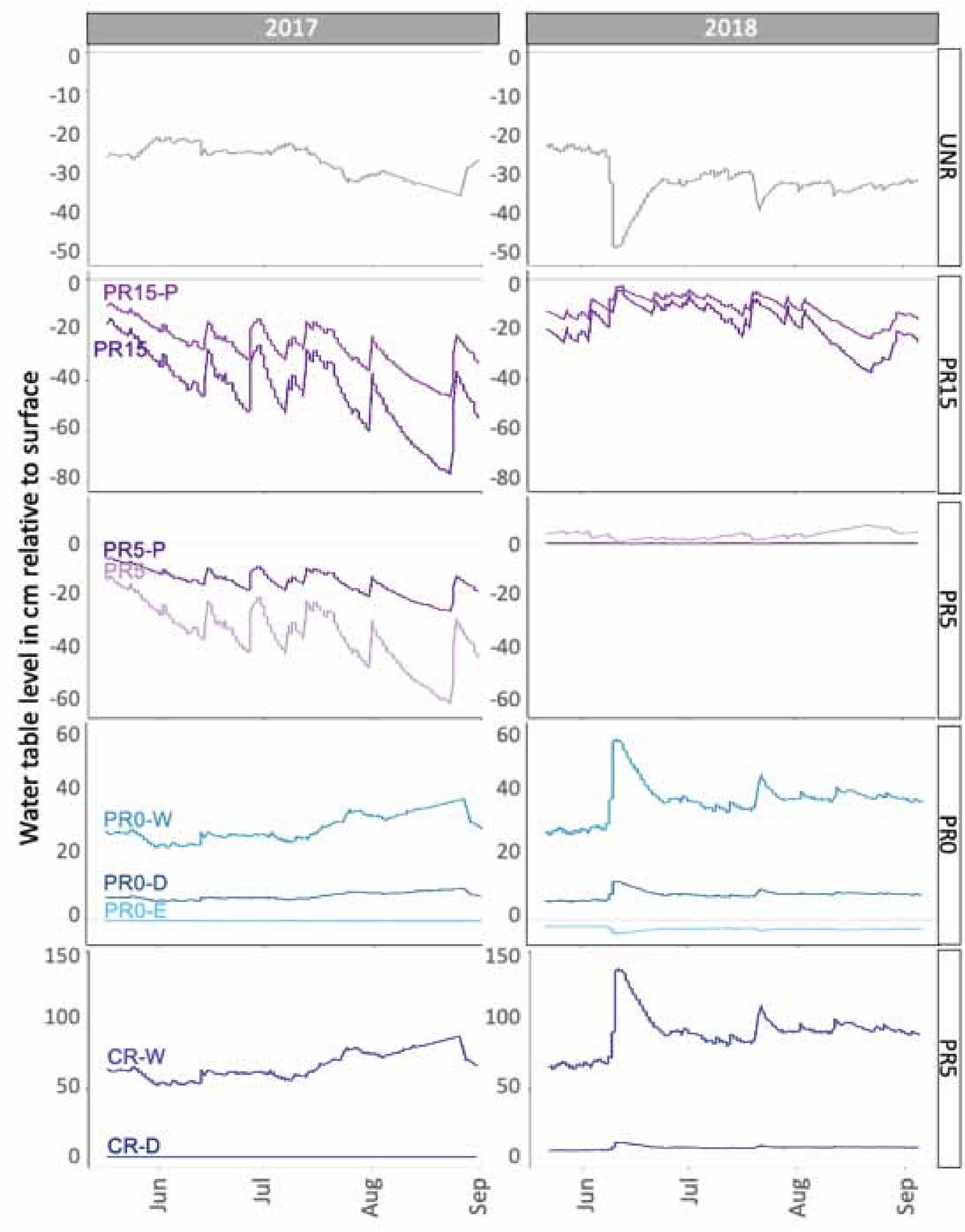
Study area / **Plant functional type** / **Species**	**UNR n = 11**	**PR15 n = 11**	**PR5 n = 11**	**PR0 n = 13**	**PR0E n = 10**	**CR n = 10**	**M n = 5**	**GRF n = 40**	**SRF n = 62**	**WRF n = 46**	**SPF n = 46**	**BOG n = 36**
Open water	3 ± 2	0	14 ± 9	57 ± 10	45 ± 10	67 ± 12	91 ± 2	40 ± 6	34 ± 4	27 ± 4	6 ± 1	0
Plant litter	21 ± 8	63 ± 5	63 ± 6	19 ± 4	21 ± 4	16 ± 8	23 ± 5	28 ± 4	63 ± 3	12 ± 2	24 ± 4	10 ± 2
Peatland characteristic species	22 ± 10	10 ± 3	27 ± 5	41 ± 7	52 ± 13	29 ± 7	48 ± 10	67 ± 6	64 ± 5	105 ± 7	103 ± 6	92 ± 6
Total vegetation	68 ± 6	51 ± 6	37 ± 6	70 ± 5	80 ± 8	48 ± 11	50 ± 10	54 ± 4	80 ± 4	68 ± 4	61 ± 5	64 ± 5
Trees	21 ± 9	0	0	0	0	0	0	1 ± 0	12 ± 2	3 ± 1	1 ± 0	12 ± 2
Lar lar	0	6 ± 4	0	0	0	0	0	1 ± 1	0	4 ± 1	2 ± 1	2 ± 1
Shrubs	24 ± 9	18 ± 7	8 ± 3	7 ± 2	5 ± 2	1 ± 1	0	7 ± 2	1 ± 0	16 ± 3	0	15 ± 2
Sal sp.	4 ± 2	10 ± 4	6 ± 3	6 ± 2	3 ± 2	0	0	6 ± 2	13 ± 2	9 ± 2	5 ± 1	2 ± 1
Ericaceous	0	0	0	2 ± 6	0	0	0	5 ± 2	1 ± 1	13 ± 2	10 ± 2	34 ± 4
Rho gro	0	0	0	0	0	0	0	1 ± 0	0	7 ± 1	4 ± 1	23 ± 3
Herbs	30 ± 10	22 ± 5	9 ± 3	33 ± 4	38 ± 8	13 ± 2	20 ± 6	20 ± 3	25 ± 4	45 ± 4	38 ± 4	9 ± 1
Equ sp.	8 ± 1	2 ± 1	3 ± 1	6 ± 2	17 ± 3	1 ± 1	0	0	0	1 ± 0	0	0
Men tri	0	0	0	6 ± 3	0	1 ± 1	0	2 ± 1	0	8 ± 2	0	0
Sedges	0	18 ± 3	22 ± 3	22 ± 4	24 ± 6	16 ± 5	31 ± 12	21 ± 5	1 ± 0	11 ± 2	1 ± 1	4 ± 1
Car aqu	3 ± 3	7 ± 3	20 ± 4	6 ± 3	18 ± 7	7 ± 4	31 ± 12	16 ± 3	1 ± 0	2 ± 1	8 ± 3	3 ± 1
Car dia	0	0	2 ± 2	0	3 ± 2	5 ± 2	0	1 ± 1	0	1	0	0
Ele pal	0	0	0	9 ± 4	1 ± 1	1 ± 1	0	0	0	0	0	0
Brown mosses	31 ± 9	16 ± 6	17 ± 8	45 ± 8	54 ± 8	29 ± 14	0	45 ± 6	47 ± 4	45 ± 5	79 ± 4	63 ± 7
Aul pal	0	0	0	0	6 ± 3	0	0	0	17 ± 3	14 ± 3	4 ± 1	1 ± 1
Cal sp.	1 ± 1	0	0	0	0	0	0	1 ± 0	3 ± 1	0	1 ± 1	0
Dre sp.	4 ± 3	3 ± 1	4 ± 2	18 ± 3	33 ± 12	30 ± 13	0	2 ± 1	9 ± 2	1 ±	1 ± 1	0
Pty pse	2 ± 2	16 ± 6	8 ± 4	1 ± 0	11 ± 5	0	0	8 ± 3	7 ± 2	1 ± 0	1 ± 0	1 ± 0
Tom nit	1 ± 1	0	0	0	0	0	0	2 ± 1	2 ± 1	5 ± 1	42 ± 6	7 ± 3
Peat mosses	0	0	0	2 ± 2	0	0	0	2 ± 1	1 ± 0	18 ± 3	11 ± 3	48 ± 6
Sph ang	0	0	0	0	0	0	0	0	0	5 ± 2	3 ± 2	1 ± 1
Sph fus	0	0	0	0	0	0	0	0	0	2 ± 1	3 ± 2	30 ± 6
Sph ter	0	0	0	0	0	0	0	0	0	11 ± 2	2 ± 1	2 ± 1
Sph wari	0	0	0	0	0	0	0	0	1 ± 0	2 ± 1	7 ± 2	5 ± 3

Appendix 1.6 Statistical results of multiple linear regression models performed with the cover of plant functional types of all study areas (dependant variables). Independent model variables are concentrations of soil Mn (sMn) and water extracted P-PO_4^{3-} (wPO4), water table level (WTL), and soil temperature at 5 cm depth (ST5). Significant results (p<0.05) are presented in bold.

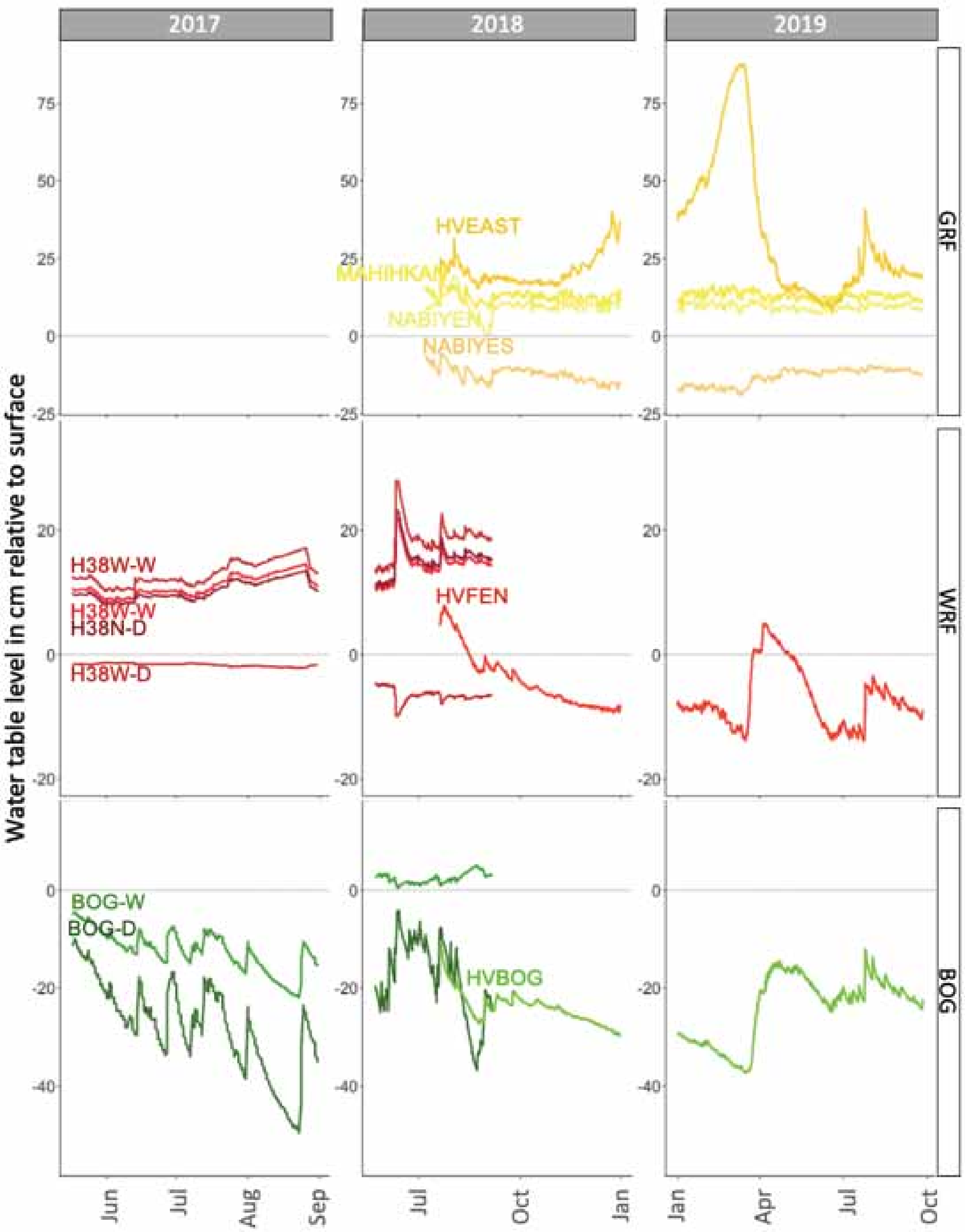
	Total vegetation			Trees			Shrubs			Ericaceous			Herbs			Sedges			Brown mosses			Peat mosses		
	F	***p***	**adj. r^2**	**F**	***p***	**adj. r^2**	**F**	***p***	**adj. r^2**	**F**	***p***	**adj. r^2**	**F**	***p***	**adj. r^2**	**F**	***p***	**adj. r^2**	**F**	***p***	**adj. r^2**	**F**	***p***	**adj. r^2**
Overall model	4.48	0.08	0.72	35.92	**0.00**	0.96	4.04	0.09	0.69	18.58	**0.01**	0.93	0.77	0.67	0.21	1.02	0.54	0.02	13.37	**0.01**	0.90	32.77	**0.00**	0.96
sMn	2.69	0.18		1.07	0.36		32.79	**0.00**		0.77	0.43		0.80	0.42		0.27	0.63		29.19	**0.01**		62.50	**0.00**	
wPO4	9.18	**0.04**		1.22	0.33		0.63	0.47		132.36	**0.00**		0.97	0.38		2.09	0.22		16.24	**0.02**		76.60	**0.00**	
ST5	6.78	0.06		154.11	**0.00**		1.35	0.31		11.11	**0.03**		0.23	0.65		0.52	0.51		1.85	0.25		45.58	**0.00**	
sMn*wPO4	0.01	0.91		114.82	**0.00**		0.35	0.59		13.24	**0.02**		0.15	0.71		0.73	0.44		0.89	0.40		0.18	0.70	
sMn*WTL	1.43	0.30		2.74	0.17		5.59	0.08		0.92	0.39		0.14	0.73		0.94	0.39		6.67	0.06		42.00	**0.00**	
sMn*ST5	0.05	0.83		116.37	**0.00**		0.64	0.47		1.86	0.24		1.36	0.31		0.13	0.74		0.12	0.74		8.79	**0.04**	
wPO4*WTL	3.11	0.15		2.32	0.20		0.01	0.92		16.00	**0.02**		0.36	0.58		0.10	0.76		4.16	0.11		17.71	**0.01**	
wPO4*ST5	0.03	0.86		1.07	0.36		0.03	0.86		3.09	0.15		0.74	0.44		0.15	0.71		26.41	**0.01**		34.13	**0.00**	
WTL*ST5	10.17	**0.03**		1.20	0.33		0.13	0.73		0.02	0.90		2.50	0.19		0.03	0.88		36.88	**0.00**		0.02	0.88	
sMn*wPO4*WTL*ST5	3.95	0.12		0.14	0.73		1.07	0.36		0.17	0.70		0.19	0.69		5.38	**0.08**		21.57	**0.01**		0.24	0.65	

Appendix 1.7 Water table level relative to the surface in the study areas: (a) restored and unrestored areas, (b) undisturbed REF adjacent to the restored areas and in the greater Peace River and Cold Lake oil sands regions. Meaning of study areas according to Appendix 1.1.

a)



b)



2017
2018
2019
GRF
WRF
BOG
Water table level in cm relative to surface
HVEAST
MAHIHKAN
NABIYEN
NABIYES
H38W-W
H38W-W
H38N-D
H38W-D
HVFEN
BOG-W
BOG-D
HVBOG
75
50
25
0
-25
20
0
-20
0
-20
-40
Jun
Jul
Aug
Sep
Jul
Oct
Jan
Jan
Apr
Jul
Oct

Chapter 2 Greenhouse gas emissions dynamics in restored fens after *in situ* oil sands well pad disturbances of Canadian boreal peatlands

Meike Lemmer, Line Rochefort, Maria Strack

2.1 Résumé

La restauration des tourbières, suite aux perturbations par l'extraction *in situ* du pétrole, vise à rétablir les fonctions essentielles des tourbières telles que l'accumulation de tourbe et la séquestration du carbone (C). Dans ce contexte, nous avons évalué les conditions biogéochimiques, les bilans C saisonniers via l'échange net des écosystèmes (ENE), les émissions de méthane (CH_4) ainsi que le potentiel de réchauffement global de quatre méthodes de restauration des tourbières. Des comparaisons ont été effectuées avec des écosystèmes de référence régionaux (REF). Après l'enlèvement complet de tous les matériaux de construction, une zone d'eau libre peu profonde s'est formée et est devenue une source nette de carbone dans l'atmosphère, avec un potentiel de réchauffement global élevé en raison d'émissions élevées de CH_4. L'introduction active d'espèces végétales ne semble pas nécessaire pour ramener des espèces végétales bénéfiques pour la séquestration du carbone. Les traitements de restauration qui ont entraîné le nivellement du remplissage minéral par rapport au REF environnant et au niveau de la nappe phréatique ont présenté le bilan saisonnier de C le plus similaire au REF. De plus, le rétablissement d'espèces d'arbustes et de mousses brunes a amélioré de manière significative l'absorption de C.

2.2 Abstract

Peatland restoration following the *in situ* oil extraction aims at reestablishing crucial peatland functions, such as peat accumulation and carbon (C) sequestration. In this context, we assessed the biogeochemical conditions, the seasonal carbon balances via net ecosystem exchange (NEE) and methane (CH_4) emissions and addressed the global warming potential of four peatland restoration methods. Restoration work involved the partial or complete removal of a former well pad's mineral fill, and spontaneous revegetation or active reintroduction of typical fen plant species such as *Larix laricina*, *Salix lutea* and *Carex aquatilis*. Comparisons were done with regional reference ecosystems (REF). A shallow open water area, following the complete removal of all construction materials, became a net C source to the atmosphere with elevated global warming potential, due to highest CH_4 emissions. The active introduction of plant species does not seem necessary to return beneficial plant species for C sequestration. Restoration treatments that resulted in the levelling of the mineral fill to the surrounding REF and the water table level showed the most similar seasonal C balance to REF. Furthermore, the reestablishment of shrub and brown moss species significantly improved the C uptake.

2.3 Introduction

In the boreal biomes of the northern countries, industrial activities are constantly increasing within the last decades, including the area of oil sands mining, bitumen and gas extraction. The location of the development of this fast-growing industry coincides with the main distribution of the world's peatlands. Here we investigate the carbon (C) dynamics of disturbed northern peatlands impacted by oil and gas extraction infrastructure, following restoration with a variety of different techniques after in-place (*in situ*) bitumen extraction has ended. The goal is to evaluate the impact of fen restoration on different ecosystem attributes, such as greenhouse gas emissions and the return of the carbon sequestration function, compared with conditions prior to disturbances.

Undisturbed peatlands are recognized as the most effective C storing ecosystems on earth, which, globally, cover an area of more than 3 million km^2 and store an estimated 644 to 1 105 Pg C (Leifeld & Menichetti 2018; Nichols & Peteet 2019). At the same time, they continuously take up approximately 0.37 Pg carbon dioxide (CO_2) from the atmosphere per year (IUCN 2017), making them a substantial ally in the fight to reverse global warming. Nevertheless, approximately 1.91 Pg CO_2-e are emitted annually by drained and degrading peatlands (Leifeld & Menichetti 2018). Restoration and rewetting of disturbed peatlands is therefore recognized as a natural climate solution and allows countries to improve their C emission balance according to the national climate action plan under the United Nations Framework Convention on Climate Change (UNFCCC 2009).

Peatland disturbances by the oil and gas industry in the boreal region of northern Alberta are caused by open-pit oil sands mining activities up to 75 m depth (3% of the deposits), and the deep drilling *in situ* bitumen extraction infrastructure for oil deposits at approximately 200 m depth (97% of the deposits; Government of Alberta 2020). The *in situ* oil sands extraction process involves the construction of thousands of oil extraction well pads scattered across the landscape, associated steam, power, and water treatment plants, processing and storage facilities, and exploration and access roads. While an average oil sands well pad is approximately 1 ha in size, the total area disturbed, including more than 180,000 well pads and associated facilities installed to date, added up to more than 149 000 km^2 by 2009 (Lee

& Cheng 2009; Natural Resources Canada 2015). These developments occur in the boreal region's vast mosaic of forests and wetlands, and affect the ecosystems' hydrology, biodiversity, and biogeochemistry by ground compaction and introduction of foreign mineral substrates (Price et al. 2003; Graf 2009). In order to stabilize the oil pumps and other processing facilities within a peatland ecosystem, an *in situ* oil sands well pad needs to be well-drained and firm. The construction process involves the clearing of larger trees and shrubs if necessary and placing of a geotextile over the then levelled original peatland surface, followed by the installation of a 1 to 2 m thick layer of compacted mineral substrate, prior the installation of pumping equipment and related oil extraction infrastructure. When the oil reserves are exhausted and the well pad will no longer be used, oil sands operators are required to reclaim these disturbed peatlands according to the Alberta Environmental Protection and Enhancement Act (Alberta Queen's Printer 1994). Specifically targeted peatland restoration outcomes anticipating an "equivalent land capability" were defined in 2015, where criteria for restoration assessment are based on the vegetation species composition of bryophytes and vascular plants, biogeochemical soil conditions, such as nutrient supply rate, hydrology and soil organic matter content, as well as landscape quality (Environment and Parks 2017). Environment and Parks (2017) defined the long-term goals of peatland restoration after well pad disturbances to be the return of the interdependent ecosystem functions present prior to disturbance, including water storage and filtration, wildlife habitat, peat accumulation, and carbon sequestration.

Restoring peatland functions after *in situ* oil sands well site disturbances in Alberta is a fairly new process that has started in 2007. All available trials have stressed the importance of restoring hydrological conditions (Vitt et al. 2011; Sobze et al. 2012; Vitt et al. 2012a; Caners & Lieffers 2014). Although few studies have investigated ecological functions returning to restored peatlands after oil sands well site disturbances, the importance of restoring proper hydrologic conditions in peatlands affected by drainage and peat extraction has been broadly studied (Price et al. 2003; Large et al. 2007; Price et al. 2010; Cooper et al. 2017; Ahmad et al. 2020; Saraswati et al. 2020). If oxygen levels rise in the upper peat layer called "acrotelm", which is periodically saturated and aerated according to the changing water table level (WTL), microbial activity and aerobic oxidation are enhanced (Price et al. 2003). In the case of disturbed peatlands due to peat extraction, vascular plant cover is often higher following restoration than in comparable undisturbed peatlands (Strack et al. 2016). The higher the vascular plant cover, the higher the ecosystem respiration (R_{eco}), but the vegetation takes up significant amounts of CO_2 at the same time, generally leading to net CO_2 storage (Strack et al. 2006; Nwaishi et al. 2016; Strack et al. 2016; Nugent et al. 2018). Vascular plant species of boreal peatlands in Northern Alberta include shrub species, such as *Betula* sp., *Larix laricina*, *Salix* sp., *Picea mariana*, and ericaceous shrubs like *Rhododendron groenlandicum* and *Vaccinium* sp., as well as herbaceous species, namely *Caltha palustris*, *Comarum palustre*, *Equisetum* sp., *Maianthemum* sp., and a large variety of sedges, such as *Carex aquatilis*, *C. diandra, C. bebbii, C. lasiocarpa, C. utriculata, Eleocharis* sp., *Eriophorum* sp. (Alberta Environment and Sustainable Resource Development 2015). However, vascular peatland plant species, in particular graminoid species such as sedges, rushes, and grasses, are considered to enhance methane (CH_4) emissions due to their large aerenchyma (Green & Baird 2012; Lazcano et al. 2018), while Strack et al. (2017) have found brown mosses to effectively decrease CH_4 emissions. Following a hydrological restoration after peat extraction for example, an increase in the WTL and vascular plant and moss cover result in the return to uptake of CO_2, while CH_4 emissions rise due to enhanced methanogenesis, but do not reach the emission rates of natural peatlands (Sundh et al. 1995; Evans et al. 2016; Strack et al. 2016; Hemes et al. 2018).

As mentioned above, very few well pad to peatland restoration projects have been attempted to date. In 2012, the moss layer transfer technique (Quinty & Rochefort 2003) was successfully applied on a restored well pad within a wooded bog, in the Carmon Creek division of the Peace River, Alberta. The inversion of the mineral pad and underlying peat layers proved to be a successful base for the introduction of bog moss propagules (Sobze et al. 2012). Shunina et al. (2016) conclude from their bog restoration trial in the Cold Lake region, Alberta that different microtopographic conditions prove to be favorable for different vascular plant and moss species, while they observed a higher resilience towards interannual moisture variation due to changes in WTL. On the other hand, to restore fens on former *in situ* oil sands well pads, only few attempts have been made during the last 12 years, and our understanding of fen restoration method’s abilities to return ecosystem functions remains limited. In this study, we evaluate the effect of different fen restoration techniques on two research sites, located in the Oil Sands regions of Peace River and Cold Lake, Alberta where a series of different restoration strategies were tested, including the complete and partial removal of the former well pad's mineral soil and clay layers, as well as re-introduction of specific plant species or natural re-vegetation. While the complete removal of a well pad favors the development of shallow open water areas with mostly aquatic vegetation, the partial removal of the mineral soil promised to achieve a well-adjusted levelling of the residual well pad with the surrounding fen ecosystems and obtain an optimal WTL to stimulate natural fen re-vegetation (Imperial Oil Resources 2017, personal communication). In 2007, Vitt et al. (2011) attempted to imitate fen initiation via paludification, by restoring peatlands directly on the mineral substrate of the well pad. Pioneer plant species were introduced and the WTL was well managed, in order to promote plant succession for the development of organic matter accumulation over time (Vitt et al. 2011; Koropchak et al. 2012). The introduction of sedge species known to colonize early stage fens proved successful, if hydrological conditions were maintained (Wieder & Vitt 2006; Vitt et al. 2011; Koropchak et al. 2012; Vitt et al. 2012b). Another peatland initiation technique was tested in 2009, focusing on the transfer of moss propagules (*Sphagnum* sp.) in addition to the introduction of vascular plant species (Gauthier 2014; Gauthier et al. 2018). This study clearly illustrated the importance of choosing characteristic fen moss species over bog moss species for mineral wetland restoration after well pad disturbances (Gauthier 2014).

The investigated fen restoration techniques in this study represent some first trials to restore Canadian *in situ* oil sands well pads in the boreal region, hence, the outcomes have not been studied before and no best practice has been established to restore former *in situ* oil sands well sites. Since the restoration of characteristic peatland functions such as C sequestration and peat accumulation is targeted, our aim was to evaluate the impact of fen restoration techniques on different ecosystem attributes, such as greenhouse gas emission rates and primary production. In this paper we will focus especially on the net ecosystem exchange (NEE) of CO_2 and CH_4 flux dynamics of spontaneously emerged vegetation communities and of communities with intentionally re-introduced species, in restored fens impacted by mineral substrate. Comparisons will be made to regional peatland reference ecosystems (REF). We hypothesize that the net C uptake will be most similar to the rate of REF when 1) characteristic fen vegetation species are present, and also when 2) biogeochemical conditions, such as nutrient concentrations, pH and electric conductivity, are most similar to

REF. We further hypothesize that 3) CH_4 emissions are enhanced through the complete well pad removal, as in the process created depressions and permanently saturated conditions enhance methanogenesis.

2.4 Methods

2.4.1 Study sites

This study was conducted in two different research areas in the Peace River and Cold Lake Oil Sands regions in northern Alberta, Canada (Appendix 2.1). Two decommissioned *in situ* oil sands well pads and three adjacent REF served as study sites. Both well pads were about 100 × 100 m in size and were constructed in the following manner: first, trees and taller shrubs were cut, then a geotextile was placed over the remaining vegetation upon the original peatland, and then 1 to 2 m compacted clay was laid down on the top of it to stabilize the "swampy" ground before the oil extraction equipment and infrastructure were installed.

Well pad in the Peace River Oil Sands

Located 35 km northeast of the city of Peace River one well pad was constructed within a wooded bog ecosystem (56°23'0.95" N, 116°46'43.43" W). In the adjacent bog, a natural undisturbed area was chosen as one REF for this study. Peace River, the well pad's name from here on, is located in the Dry Mixedwood natural region of Alberta's boreal region (Beckingham & Archibald 1996), with 70% of its annual precipitation falling between April and August, while the annual average precipitation reaches 386 mm (Government of Canada 2019). The average frost free period is 112 days with average daily temperatures of 13 °C between May and September (Government of Canada 2019). Wooded and shrubby fens dominate in this region, while sedge fens and bogs are rather seldomly encountered (Natural Regions Committee 2006).

Peace River was decommissioned in 2000, 20 years post construction. The unaltered original well pad was reclaimed with herb seedings of *Melilotus alba* and *M. officinalis*, which were spread atop of the compacted clay surface. In 2007, the site was offered to a research group for a restoration experiment based on the principles of ecological restoration, to assist the return of a peatland. On the east side, a band of the well pad (30 × 100 m) was used to trial treatments to initiate fen development on mineral soil (Vitt et al. 2011; Koropchak et al. 2012). The experimental area was divided into two study areas (Figure 2.1). Within one area, the clay fill was partially removed in order to create a surface profile sitting on average 4 to 6 cm above the WTL of the adjacent bog. In this study, we refer to this treatment as partial removal with water table at 5 cm (PR5). Within the second area, less clay was removed so that the grading would create a surface profile being on average 15 cm above the bog's water level (PR15). In both study areas, different soil and fertilization amendments were then applied (see Vitt et al. 2011 for details), while *L. laricina*, *S. lutea* and *C. aquatilis* were planted. Ten years following this study, we observe in PR5 some *S. planifolia* and *S. exigua* present among the very dominant *C. aquatilis*, while in PR15 *L. laricina*, *S. planifolia* and *S. pyrifolia* were well developed among the mix of dominating *Calamagrostis inexpansa* and *C. aquatilis.* To minimize possible effects of the type of soil amendments on the greenhouse gas dynamics, the measurement plots installed in the present study were chosen within amendments as natural as possible, such as commercial peat, slough hay and control plots without any amendment (Vitt et al. 2011; Koropchak et al. 2012).

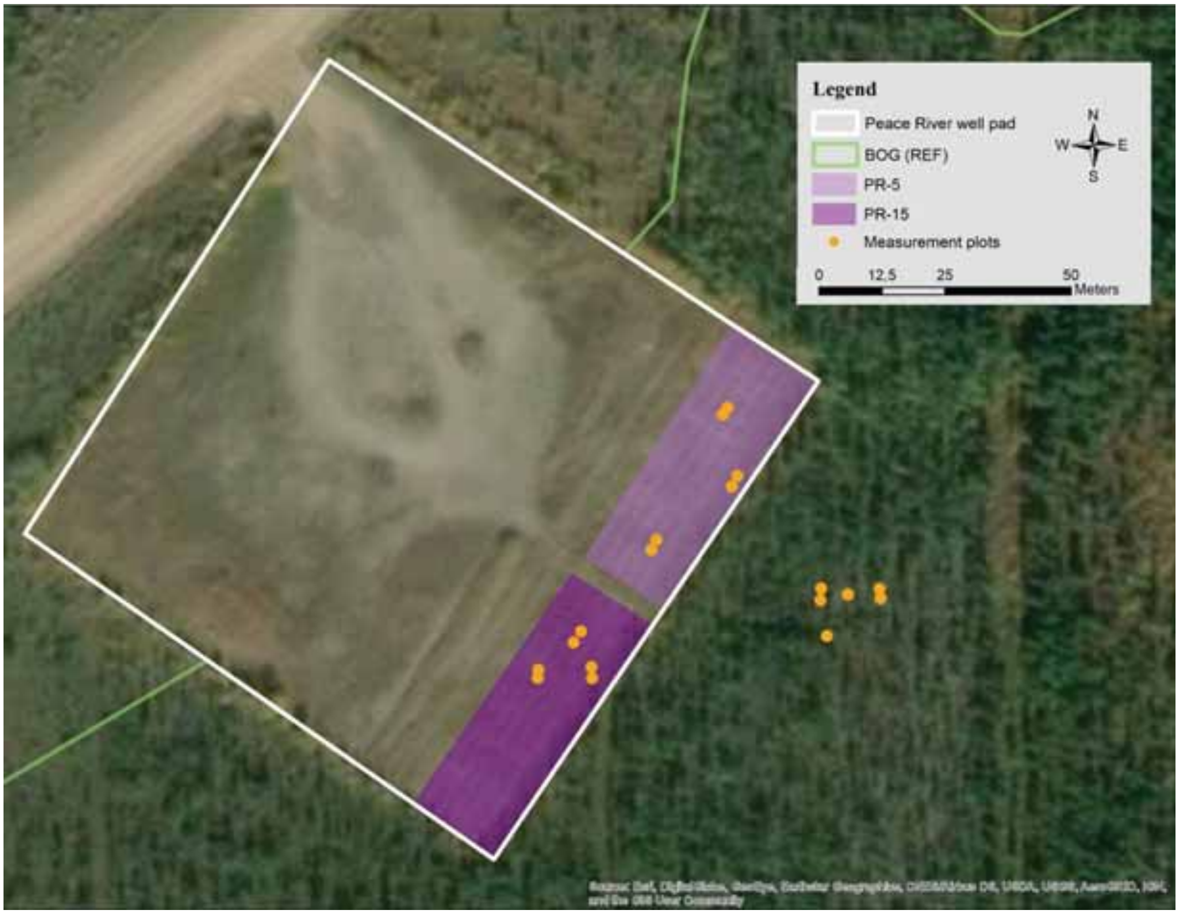


Figure 2.1 The Peace River well pad (white outline) is located within a wooded bog ecosystem, which serves as reference ecosystem (BOG) in this study. In two restored sectors of the well pad, the mineral soil was partially removed (PR) to 15 cm (PR-15) and to 5 cm (PR-5) above the water table of the adjacent peatland. Yellow dots indicate measurement plots.

Well pad in the Cold Lake Oil Sands

The second well pad in this study is located 33 km northwest of the city of Cold Lake (54°41'10.82" N, 110°30'59.75" W). Cold Lake, what this second well pad will be named from here on, was partially constructed on upland, partially in a wooded rich fen characterized by tall trees and partially in a wooded extreme-rich fen, characterized by shrub-sized tree species. Within each of these two fens adjacent to the former well pad, at least 10 m away from any disturbance, an area was chosen as REF: 1. treed rich fen (*TRF*) and 2. shrubby extreme-rich fen (*SRF*). Cold Lake lies in the moist Central Mixedwood ecoregion of boreal Alberta (Beckingham & Archibald 1996), with an annual average of 421 mm precipitation, an average frost free period of 116 days and a daily average temperature of 13.9 °C during the summer months (Government of Canada 2019).

Cold Lake was decommissioned in 2003, only one year after its construction, due to drilling problems caused by underlying shale. The well pad was subject to different restoration techniques between 2008 and 2009. The central part of the former well pad was kept intact in order to continue operating a monitoring well (unrestored study area, UNR hereafter, Figure 2.2). For another part, a complete removal of all introduced building materials (mineral substrate and geotextile) was achieved in spring 2008 (complete removal, CR). In this complete removal area, a shallow open water area established due to compaction of the underlying peat by the weight of the mineral material. For a third part of the pad, a partial removal of the mineral substrate was carried out successively during 2008 and 2009, as done at Peace River. The goal was to obtain a surface elevation similar to that of the surrounding fens, where water table is close to surface elevation (PR0). All restored areas of Cold Lake were left to re-vegetate spontaneously, with the reasoning that the surrounding undisturbed wetlands could provide a natural source of diaspores by dispersion. By 2017, emergent aquatic vegetation and a floating moss carpet with sedges and emergent *Salix* sp. and *Betula* sp. had formed in the CR area. We separated the PR0 area into two areas considering diverse ground relief, where different marsh-like vegetation communities had established. One area is characterized by an uneven relief forming drier and wetter microforms, where dominant *Typha latifolia*, *Salix* spp., and sedge communities formed (named PR0-D/W, D for dry and

W for wet). The other area's ground is even and covered by diverse bryophytes that formed between abundant *Equisetum* spp. and *T. latifolia* (named PR0-E, E standing for even).

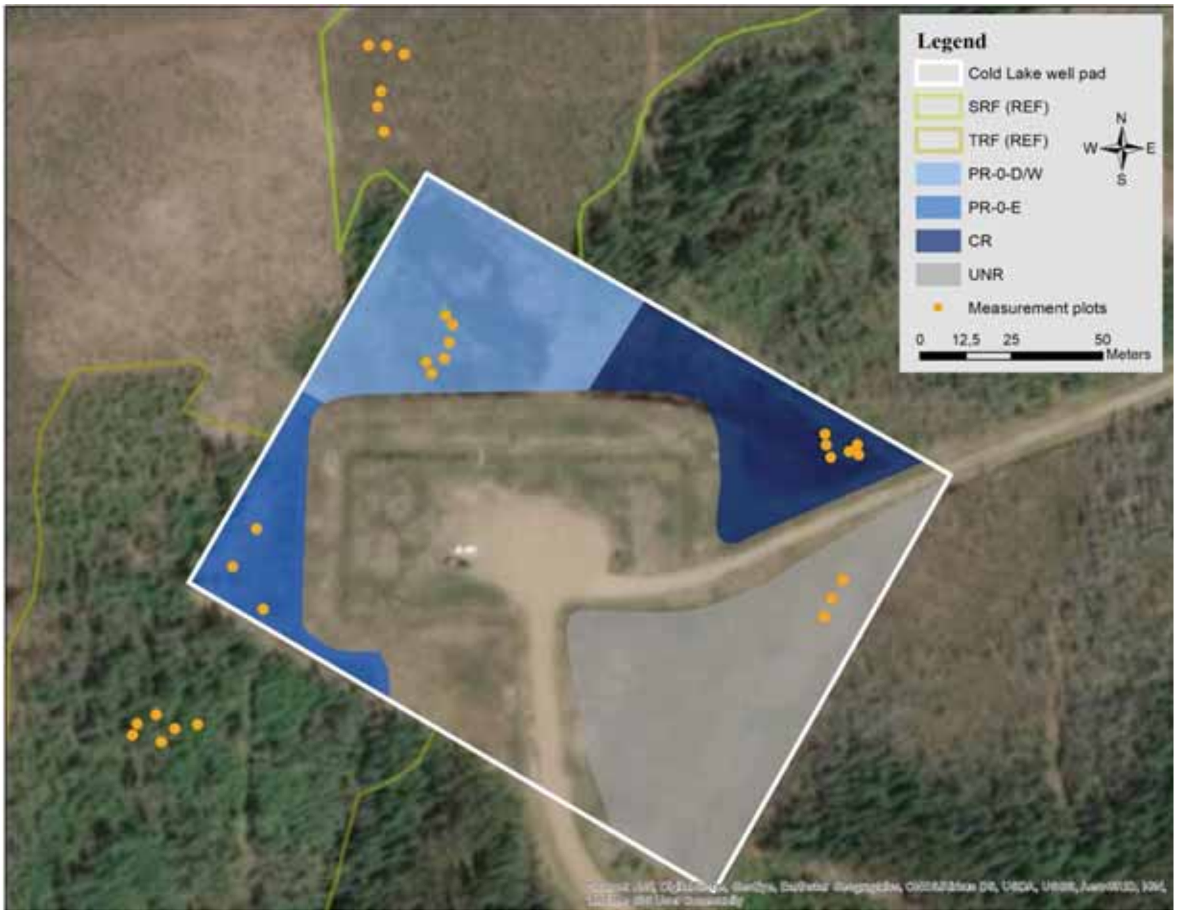


Figure 2.2 The Cold Lake well pad (white outline) is located within a mosaic of uplands and wetlands, which serve as reference ecosystems (TRF=treed rich fen, SRF=shrubby extreme-rich fen). Blue shaded sectors are restored areas where different restoration techniques of complete and partial mineral soil removal were tested (CR=complete removal, PR-0-D/W and PR-0-E=partial removal of mineral soil to near the adjacent fen ecosystems, with high/dry (D) and low/wet (W) microforms, and with even ground (E)). The grey shaded unrestored sector (UNR) serves as a control sector on the former well pad's residual mineral soil. Yellow dots indicate measurement plots.

Reference sites

Three peatland REF served as monitoring sites for comparison: *BOG* (56°22'59.50" N, 116°46'38.60" W; Figure 2.1), a wooded bog, had a characteristic tree and shrub vegetation composition of *P. mariana*, *R. groenlandicum*, *Chamaedaphne calyculata* and *Vaccinium vitis-idaea*, as well as a dense moss layer with *S. fuscum*, *S. rubellum* and *P. pseudotriquetrum*. *TRF* (54°41'8.88" N, 110°31'4.06" W; Figure 2.2), a treed rich fen, had a distinct tree layer with *P. mariana* and *L. laricina*, a shrub layer with *R. groenlandicum* and a ground layer with *E. hyemale*, *M. trifoliata* and moss species, such as *Aulacomnium palustre*, *Helodium blandowii* and *T. nitens. SRF* (54°41'14.80" N, 110°31'0.54" W; Figure 2.2) was a wooded extreme-rich fen with abundant *B. pumila*, *Salix* sp. and *L. laricina* that formed a shrub mosaic with abundant herbaceous vegetation like *Equisetum* sp., *M. trifoliata, Triglochin maritima*, and sedges, such as *C. lasiocarpa, C. interior, and C. sartwellii*.

2.4.2 Measurement plots

Within all study areas (Appendix 2.2), monitoring plots were selected according to the restoration technique applied and according to the most representative natural state of the REF. Special focus was placed on the vegetation development resulting from differing microforms (Table 2.1). Microforms were considered either for different elevation, such as hummock/hollow (i.e., in BOG), or for different moisture gradients, such as dry/wet/even (i.e., high lawn with shrubs/low lawn with sedges and mosses/even lawn with mosses). Triplicate measurement plots were selected in each microform within each of the eight study areas (n=48) where measurements of CO_2, CH_4, and abiotic data took place (Figure 2.1 Figure 2.2). Each measurement plot was defined by a metal collar of 60 × 60 × 20 cm size that was inserted approximately 17 cm deep into the ground, and which served as a base for the gas flux chamber. All plots were accessible via boardwalks in order to mitigate ground disturbance around the installed collars, during measurements. At each plot, data was collected biweekly during the regional vegetation period from May to September of both monitoring years, 2017 and 2018. The large distance between the sites, as well as weather and industry related constraints made a higher sampling frequency impossible to achieve. This data collection corresponded to 10-11 years post-restoration for Peace River and 8 to 10 years for Cold Lake.

Table 2.1 Site characteristics for the unrestored and restored areas on two former in situ oil sands well sites in Peace River and Cold Lake and three reference sites (SRF, TRF, BOG). Monitoring sectors were selected according to the representativity of the natural state, restoration, and re-vegetation technique.

Monitoring sector		**Restoration**			**Microform/ moisture gradient**	**Revegetation**	
Microform	**Site**	**Substrate removal**	**Surface elevation**	**Year**		**Technique**	**Dominant plant group**
UNR	Cold Lake	Unrestored	2 to 4 m above water table	2002	Lawn	Spontaneous	Mosses, herbaceous
PR15	Peace River	Partial removal	15 cm above water table	2007	High, dry lawn	Spontaneous	Shrubs, sedges
PR15-P	Peace River	Partial removal	15 cm above water table	2007	High, dry lawn	Re-introduced	Shrubs, sedges
PR5	Peace River	Partial removal	4 to 6 cm above water table	2007	Wet, low lawn	Spontaneous	Sedges
PR5-P	Peace River	Partial removal	4 to 6 cm above water table	2007	Wet, low lawn	Re-introduced	Sedges
PR0-D	Cold Lake	Partial removal	Adjusted to surrounding reference ecosystem (0 cm)	2009	Dry, high	Spontaneous	Shrubs, herbaceous, mosses
PR0-W	Cold Lake	Partial removal	Adjusted to surrounding natural ecosystem (0 cm)	2009	Wet, low depression	Spontaneous	Sedges, aquatics
PR0-E	Cold Lake	Partial removal	Equal to surrounding natural ecosystem (0 cm)	2009	Even lawn	Spontaneous	Herbaceous, mosses
CR-D	Cold Lake	Complete removal	Below surrounding reference ecosystems	2008	Floating moss carpet	Spontaneous	Mosses, herbaceous, small shrubs
CR-W	Cold Lake	Complete removal	Below surrounding reference ecosystems	2008	Shallow open water	Spontaneous	Floating aquatics
SRF-D	Shrubby extreme-rich fen	n.a. (undisturbed)	n.a.	n.a.	Dry, high hummock	n.a.	Shrubs, mosses
SRF-W	Shrubby extreme-rich fen	n.a. (undisturbed)	n.a.	n.a.	Wet, low depression	n.a.	Herbs
TRF-D	Treed rich fen	n.a. (undisturbed)	n.a.	n.a.	Dry, high hummock	n.a.	Mosses, ericaceous
TRF-W	Treed rich fen	n.a. (undisturbed)	n.a.	n.a.	Wet, low depression	n.a.	Herbs

BOG-D	Wooded Bog	n.a. (undisturbed)	n.a.		n.a.	Dry, high hummock	n.a.	Mosses, ericaceous
BOG-W	Wooded Bog	n.a. (undisturbed)	n.a.		n.a.	Wet, low depression	n.a.	Ericaceous, sedges

2.4.3 CO_2 exchange

Measurements of CO_2 fluxes were assessed between 17 May and 28 September 2017 and between 14 May and 14 August 2018. Fluxes were measured using a dynamic closed chamber technique with a portable infrared gas analyzer (EGM-4) fitted with a PAR Quantum sensor (both PP Systems, Amesbury, MA, USA) that was placed on top of the chamber during measurements. The 60 × 60 × 30 cm large clear polyethylene chamber was equipped with two standard computer fans connected to an external 12V battery for air circulation, a thermocouple wire to connect to an external type K thermometer (Sper scientific, Scottsdale, AZ, USA), and two tube adapters to connect the IRGA, in order to exchange the sampled air in a circular flow. During measurements, the chamber was fit into the collar's u-profile rim, which we then filled with water in order to create an airtight seal. In sample plots, where WTL were too high to install collars (CR-W) or collars were submerged at certain times (PR0-W), the chamber was fitted with a Styrofoam collar, enabling it to float on the surface. In plots, where shrubby vegetation was too large to fit in the 30 cm high chamber, a 60 cm tall extension made likewise from clear polyethylene, with a u-profile collar at the upper edge, was stacked under the chamber. To account for the enlarged chamber volume, calculations for the flux analysis were accordingly adjusted. Readings of CO_2 concentration (ppm), photosynthetically active radiation (PAR; µmol m^{-1} s^{-1}) and temperature in the chamber (℃) were recorded in a 15 second interval for 105-120 seconds. Measurements of net ecosystem exchange (NEE) were repeated under full light conditions and imitating different light conditions through shading of the chamber and the PAR sensor with a mesh material. One mesh cover created 25% shading and a second cover imitated up to 48% covered conditions. Ecosystem respiration (R_{eco}) was determined 8 to 10 minutes after full light conditions were captured, by blocking all incoming PAR with an opaque tarp covering both the chamber and PAR sensor. Between each measurement imitating different light conditions, the chamber headspace was vented to adjust to ambient conditions.

2.4.4 CH_4 emissions

CH_4 fluxes were measured bi-weekly eight times each year between 17 May and 28 September 2017 and between 14 May and 24 August 2018. CH_4 concentration was determined using a closed static chamber technique with opaque polyethylene chambers of the same dimensions as for CO_2 measurements. Chambers were darkened with standard spray-paint, aluminum-colored to reduce heating during the flux measurement. Chamber equipment included one standard computer fan for air circulation connected to an external 12 V battery, a thermocouple connected to an external thermometer and one tube with a three-way-valve in order to extract gas samples. All wires and tubes exited the chamber via a rubber plug that fills a hole (5 cm diameter) in the chamber top. At the same time, the rubber plug served as a regulator for possible build-up of air pressure inside the chamber when fitting the chamber to the collar. Again, water poured into the u-profile rim of the collar created an airtight seal of the chamber headspace. Gas samples were taken at 7, 15, 25, and 35 minutes after chamber closure using a standard 20 ml disposable syringe connected to the three-way-valve. A 20 ml gas sample from the chamber headspace was stored in a 12 ml round bottom Exetainer vial with a septum lid (Labco Limited, Lampeter, Wales, UK). The created overpressure was necessary in order to prevent any ambient air leaking in. Also, septum lids were discarded after the third use, in order to prevent any leakage due to repeated piercing. Gas samples were sent to the Wetland Soils and Greenhouse Gas Exchange Laboratory at the University of Waterloo, ON for analysis. Analysis for CH_4 concentrations was done with a Shimadzu GC2014 gas chromatograph equipped with a flame ionization detector (Shimadzu Scientific Instruments, Kyoto, Japan).

2.4.5 Environmental parameters, soil and water chemistry

At each plot, manual measurements of WTL and soil temperature were taken biweekly at the same time as flux measurements. Water level was measured at each spot in perforated pipes, serving as well-tubes that were covered with nylon mesh to prevent silting. These were inserted about 50 to 100 cm deep into the ground. Soil temperature was measured at 2, 5, 10, 15, 20, 25, and 30 cm with a thermocouple probe temperature sensor and reader (Digital thermometer, VWR, Radnor, PA, USA). Water chemistry (pH and electrical conductivity) was measured in August of both 2017 and 2018 with an Orion Versastar Advanced Electrochemistry Meter (Thermo Fisher Scientific Inc., Chelmsford, MA, USA).

Soil samples were collected in August 2018 at each measurement spot and were analyzed for plant available nutrient concentrations of ammonium (N-NH_4^+), iron (Fe), phosphate (P-

PO_4^{3-}), and sulphate ($S\text{-}SO_4^{2-}$) as well as for DOC, as these elements were considered indicators of nutrient status and redox state. Analysis for $P\text{-}PO_4^{3-}$, and $S\text{-}SO_4^{2-}$ were done using a FIA Quikchem 8500 Series 2 (Lachat Instruments, Milwaukee, WI, USA). Fe and $N\text{-}NH_4^+$ were analyzed via an ICP Agilent 5110 SVDV (Agilent Technologies Inc., Santa Clara, CA, USA). Solutions for DOC analysis were produced via 1:3 soil to water mixtures from soil samples taken within the top 10 cm at each plot. Filtration was done the following day through a 0.45 µm glass fiber filter and stored at 4 °C until being analyzed at the Physical Geography Laboratory at the University of Calgary with a TOC-L analyzer (Shimadzu Scientific Instruments Inc., Columbia, MD, USA).

Environmental data of soil temperature (GS3 sensor) and water temperature (CTD-10 sensor) was continuously recorded via EM 50 data loggers (Meter Group Inc., Pullman, WA, USA) from May to September 2017 and May to August 2018. Water tables were also continuously measured with a levelogger (Solinst Canada Ltd., Georgetown, ON) inside a well tube, while the atmospheric pressure for calculating corrected WTL was measured with barologgers (Solinst Canada Ltd., Georgetown, ON) at each site. Additionally, continuous meteorological data of air temperature, precipitation (ECRN-100 sensor), and PAR (PYR sensor) was recorded. At Cold Lake, the data recording station was installed directly on the well pad, whereas at Peace River, only environmental data was recorded on site, while the weather station was set up at a nearby restored well pad (approximately 7 km distance).

2.4.6 Vegetation

The vegetation survey was done within the different study areas during the peak of the vegetation period, in August of both monitoring years. The cover percentage of each vegetation stratum, as well as the percentage of open water, bare peat and litter present, were surveyed in all survey quadrats. For each stratum, an additional focus was put on important plant groups, i.e., *Ericaceae* in the shrub stratum, sedges in the herbaceous stratum, and *Sphagnum* sp. in the moss stratum. Within each study area, five vascular plant surveys were done using a 1 m^2 survey quadrat and 20 Bryophyte surveys were done using a 25 × 25 cm survey quadrat. According to the survey type, all plant species were identified to species level while their respective percentage cover and height (in cm) were noted. We determined if a given species, including vascular and bryophyte species, likewise, was characteristic of the different fen types found in Alberta, following Environment and Parks (2017). From here on we refer to these species as fen typical plant species (FTP).

2.4.7 Data Analysis

Instantaneous CO_2 and CH_4 fluxes

NEE (g CO_2 m^{-2} d^{-1}) was calculated according to the linear change of measured CO_2 concentration over time, considering collar surface area, chamber volume and air temperature. Gross ecosystem productivity (GEP) was then calculated according to Eq. 1, considering measured NEE and R_{eco},

$$GEP = NEE - R_{eco}. \qquad \text{[Eq. 1]}$$

Following the atmospheric sign convention, we use negative values to indicate uptake by the ecosystem from the atmosphere, while positive values indicate the release of CO_2 and CH_4. During data cleaning 1% of the data was discarded, including fluxes without a linear change in concentration over time and negative values of ecosystem respiration.

CH_4 fluxes (mg CH_4 m^{-2} d^{-1}) were calculated according to the linear change of CH_4 concentration over time, in consideration of collar surface area, chamber volume and air temperature inside the chamber. When concentration was low (<3 ppm) and changed less than 0.5 ppm (precision of the sampling and analysis method), respective fluxes were set to 0. In cases where no obvious trend was recognized and where no linearity was achievable, the flux data was rejected. Rejection occurred also in cases of a negative curve following high starting values larger than 5 ppm. These fluxes are considered as evidence of ebullition caused by ground disturbance during chamber placement and do not represent regular CH_4 fluxes. Following this data cleaning procedure, 5% of the data were discarded.

Statistical analyses were done in R 3.6.0 (R Core Team 2019). The package ‘ggplot2‘ was used to create figures (Wickham 2016). Analysis of histograms, residuals and the Shapiro-Wilk test for normality indicated that data was not normally distributed in all cases, but no transformations were able to improve the distribution to normality. Despite the non-normally distributed data, we are confident in reporting on one-way ANOVA results, because of the ANOVA’s robustness, where we fit linear models with ‘microform’ as fixed factor and ‘plot’ as random factor. We achieved additional validation by comparison with the results of a non-parametric Kruskal-Wallis test. The Kruskal-Wallis test was validated with a Conover-Iman post-hoc test after Bonferroni adjustment using the package ‘conover.test’ (Dinno 2017), and results were consistent with the ANOVA output. Furthermore, we performed pairwise comparisons using the ‘emmeans’ package and a Tukey Honest Significant Differences (HSD) post hoc analysis with 95% confidence interval (Lenth 2019). We further performed multiple comparisons of treatments between groups with the ‘agricolae’ package (DeMendiburu 2019), in order to complete figures with letters for groups with statistically significant differences. All ANOVAs for NEE and GEP included PAR values larger than 1 000 µmol m^{-2} s^{-1} to obtain rates for CO_2 uptake that were not limited by light availability (Bubier et al. 2003). A statistical significance was accepted when p less than 0.05.

Environmental influence on greenhouse gas fluxes

Linear regressions were used to further investigate the effect of vascular plant and moss cover, especially of FTP, and open water present in the measurement plots, using the lme function (linear mixed effects) in the ‘nlme’ package (Pinheiro et al. 2019). Regressions were calculated between mean seasonal fluxes of CO_2 and CH_4, and nine independent variables including water table level (WTL), soil temperature at 5 cm depth (ST_5), cover of FTP, and of all plant functional types (trees, shrubs, ericaceous, herbaceous, sedges, mosses, Sphagnum) and open water, litter and bare soil, while plot functioned as a random factor to account for repeated measures over the two years. Package ‘MuMIn’ was used to determine the marginal and conditional r^2 (Bartoń 2020). We used a principal component analysis (PCA) to examine the variation among measurement plots considering biotic and abiotic data, and furthermore to explain the variables’ contribution to the observed differences among the plots.

Seasonal carbon balance

The seasonal carbon balance was estimated according to Eq. 2, where the seasonal GEP (g CO_2 m^{-2} h^{-1}) modelling was done following Baird et al. (2019), fitting a two-parameter model with PAR values in a non-linear regression:

$$GEP = \frac{\alpha \times PAR \times GP_{max}}{(\alpha \times PAR + GP_{max})}. \quad \text{[Eq. 2]}$$

The model parameters are given physical meaning as α represents the slope of a rectangular hyperbola, *Q*, and gross photosynthesis GP_{max} is the asymptotic limit theoretical maximum (Baird et al. 2019).

Modelling ecosystem respiration (g CO_2 m^{-2} h^{-1}) was done following Renou-Wilson et al. (2014), who considered WTL and ST_5 in order to factor in the different microforms' diverse moisture regimes, using the Eq. 3:

$$R_{eco} = (a + (b \times WTL)) \times \left[\exp\left(c \times \left(\frac{1}{T_{Ref} - T_0} - \frac{1}{ST_5 - T_0}\right)\right)\right], \quad \text{[Eq. 3]}$$

where *a*, *b*, and *c* are model parameters, T_{Ref} is the reference temperature of 283.15 K, and T_0 is the temperature at biological activation (227.13 K; Lloyd & Taylor 1994). Model parameters *a*, *b*, and *c* for the seasonal carbon models of GEP and ecosystem respiration were calculated using the nonlinear least squares (nls) function in R. GEP and ecosystem respiration were estimated in half-hourly intervals, averaged and summed for the growing season between 17 May to 31 August 2017 and between 22 May to 5 September 2018.

The models' form was evaluated considering statistically significant parameters and the highest possible correlation coefficient between measured and modelled values.

For each collar we fitted an individual model per year, after exploration of the data suggested that models' fits were improved by dividing the data. Due to smaller GEP values caused by springtime conditions such as low temperatures and less photosynthesizing vegetation cover in the early season in 2018, we additionally divided the season into early season (22 May until 5 June 2018) and late season measurements (6 June to 6 September 2018) for each study area and fit separate models for each collar per season.

Seasonal CH_4 fluxes were estimated by multiplying the measured mean flux values with the number of days for each growing season, following Baird et al. (2019).

2.5 Results

2.5.1 Environmental parameters, soil and water chemistry

The climatic conditions observed during the time of the study were well within the range of the Dry Mixedwood ecoregion's dry climate in Peace River, and the Moist Mixedwood ecoregion's humid climate in Cold Lake (Figure 2.3). During both years of data recording, Cold Lake was characterized by about twice as many rain days as the region of Peace River and even shows consistently higher precipitation data as compared to the 1981-2010 climate normal average of about 213 mm and 246 mm (Environment and Climate Change Canada 2019). In both regions, the year 2018 was characterized by much wetter conditions than 2017. Due to the higher precipitation in 2018's field season, WTL in the restored areas and REF were accordingly higher especially in the Peace River region (Table 2.2). We observed an average rise in WTL of 10.4 cm in the restored areas, compared to an average rise of 5.6 cm in the REF.

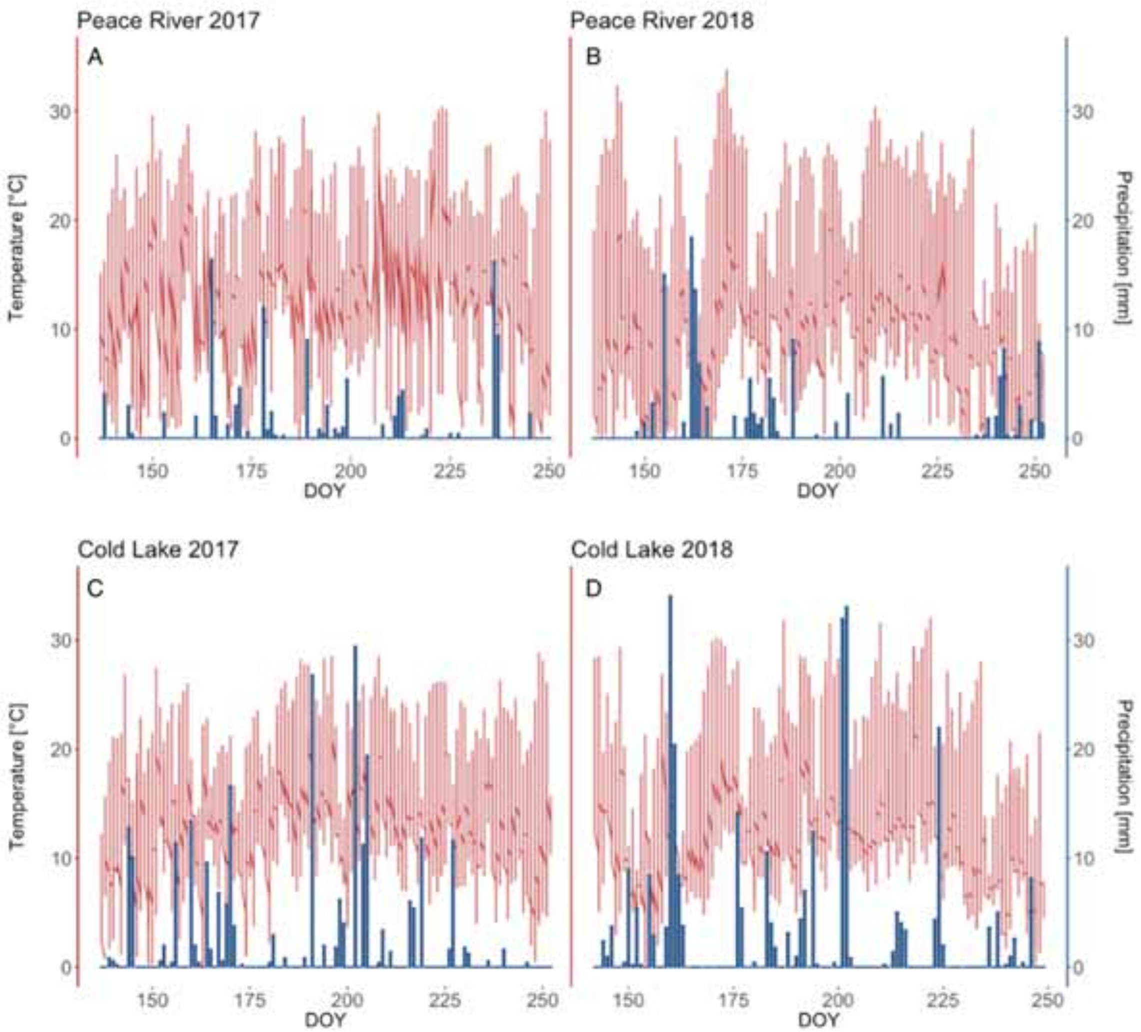


Figure 2.3 Precipitation (mm) and air temperature (°C) measured at a meteorological station on restored in situ oil sands well pads in the Peace River and Cold Lake Oil Sands, during the monitoring period (May 17 until September 9) in the years 2017 (A + C) and 2018 (B + D).

Table 2.2 Mean ± SD of the water table level (WTL in cm), pH and electric conductivity (EC in µS/cm) in all different restoration sectors on two former well pads in Peace River and Cold Lake and three reference ecosystems. Negative values signify a water table below the soil surface, and positive values signify a water table above the soil surface. Monitoring sectors according to Table 2.1.

Sector	2017			2018		
	WTL	$pH_{(W)}$	$EC_{(W)}$	WTL	$pH_{(W)}$	$EC_{(W)}$
UNR	-26.2 ± 17	6.7 ± 0	1015.3 ± 194.9	-27.4 ± 11.6	3.6 ± 0.2	604.7 ± 39
PR15	-36.9 ± 19.2	5.8 ± 0.2	2456 ± 282.4	-15.2 ± 9.5	3.4 ± 0.3	461.3 ± 793
PR5	-24.1 ± 20.1	6 ± 0.5	1452.6 ± 1329.9	1.4 ± 5.2	3.6 ± 0.2	2573.7 ± 515.9
PR0-D/W	18.4 ± 10.9	7.3 ± 0.4	458.7 ± 55	20.2 ± 11.3	7.4 ± 0.1	341.5 ± 13.2
PR0-E	-1.7 ± 3.8	6.9 ± 0.4	687 ± 450.6	-1.9 ± 2.6	7.3 ± 0.1	476.3 ± 232.7
CR	29.2 ± 19.7	6.7 ± 0.2	305 ± 9.5	41.6 ± 38.4	6 ± 0.2	245.3 ± 110.2
SRF	16 ± 3.9	7.1 ± 0.4	452.3 ± 18.7	15.1 ± 4.7	7.3 ± 0.3	284.9 ± 23.6
TRF	3.1 ± 9.3	6.8 ± 0.2	478.8 ± 9.5	5.5 ± 10.6	5.3 ± 0.1	329.8 ± 4.9
BOG	-25.7 ± 13.3	4.8 ± 0.3	512.9 ± 1058	-10.4 ± 9.6	3.5 ± 1.4	102.1 ± 54.5

The restored areas PR5 and PR15 at Peace River, that were restored by partial removal of the former well pad mineral soil, showed higher values of water pH and electric conductivity in 2017, compared to the water chemistry conditions of wooded fens at SRF and TRF adjacent to Cold Lake, which had lightly more acidic but brackish milieu, with moderate salinity and elevated electric conductivity (Table 2.2). In 2018, the pH of PR5 and PR15 were lower and comparable to the conditions of the adjacent acidic BOG. Here, the values were consistently characteristic for ombrotrophic bogs throughout both years, with highly acidic and oligotrophic conditions (Alberta Environment and Sustainable Resource Development 2015). In all other restored areas, PR0 and CR at Cold Lake, fen conditions were maintained throughout both years, where the electric conductivity of water declined in the second year on average by 129 µS/cm compared to the previous year, and pH stayed quite stable throughout both years. The same trend in the drop of electric conductivity and water pH can be seen in UNR. TRF and SRF water chemistry remained quite consistent during both study seasons, with high pH and slightly brackish conditions typical for fens.

Acidic soil conditions were maintained in all restored areas and BOG throughout 2018 (Table 2.3). The only outlier with a slight alkaline pH 8 was observed at UNR. Electric conductivity of the soil remained notably higher in all unrestored and restored areas, compared to REF. DOC in the REF was on average 19 mg/L higher than in the restored areas. Comparisons of the average plant available nutrient concentrations of iron, ammonium, phosphorus and sulfur reveal mean concentrations found in the restored areas were below values found in REF. Only the shallow open water area CR-D showed REF-equivalent concentrations. In PR15 and PR5 we noted exceptionally high sulfur concentrations (5 608 and 1 638 mg/L respectively). Highest sulfur concentration found in the REF, was measured in BOG-D.

*Table 2.3 Mean ± SD **soil** pH and electric conductivity (EC in µS/cm), as well as mean dissolved organic carbon (DOC in mg/L) and plant available soil nutrient supply rates (in mg/L) of ammonium ($N\text{-}NH_4^+$), iron (Fe), phosphorus ($P\text{-}PO_4$) and sulfur ($S\text{-}SO_4^{2-}$), in all monitoring sectors in 2018. BDL stands for values below detection limit (detection limit for Fe=0.12 µg/L). Monitoring sectors according to Table 2.1.*

Sector	$pH_{(S)}$	$EC_{(S)}$	DOC	$N\text{-}NH_4^+$	$P\text{-}PO_4$	$S\text{-}SO_4^{2-}$	Fe
UNR	8 ± 0.2	290 ± 27.6	n.a.	14.4	44.5	24.8	BDL
PR15	4.8 ± 0.2	1 109.8 ± 870.8	5	33.6	85.6	5 608.2	BDL
PR5	4.2 ± 0.3	1 442 ± 753.3	6.3	21	113.7	1 638.2	BDL
PR0-D	3.2 ± 0.4	378.7 ± 17.2	5.8	15.9	58.2	12.2	2.43
PR0-W	3.3 ± 0.4	344.3 ± 26.9	6.5	28.8	88.6	80.8	BDL
PR0-E	5 ± 2.1	348.7 ± 11.5	10.8	11	76.4	9.7	BDL
CR-D	4.8 ± 0.3	106.3 ± 13.3	8.6	168.6	267.5	32.5	0.55
SRF-D	4.9 ± 0.3	122.5 ± 27.8	24.5	364.9	330.6	95.7	5.43
SRF-W	4.6 ± 0.2	76.1 ± 32.9	15.5	332.7	296.5	69.2	2.97
TRF-D	5 ± 0.1	86.7 ± 58.5	n.a.	213	182.2	88.9	1.76
TRF-W	4.9 ± 0.2	68.7 ± 20.2	n.a.	356.6	355.7	81.5	1.47
BOG-D	4 ± 0.2	34.7 ± 10.9	35	109	117.1	100.2	2.19
BOG-W	2.9 ± 0.6	38.9 ± 3.6	28.8	126.8	85.2	95.2	0.90

According to the principal component analysis (PCA), the first two components explain 48,9% of the variation in biogeochemistry among study plots (Figure 2.4). Chemical differences are represented in PC1, while PC2 depicts mostly the impact of vegetation and hydrology. The Kaiser-Meyer-Olkin test for sampling adequacy of variables reached 0.57 (Dziuban & Shirkey 1974). Six of the 17 components showed an eigenvalue larger than 1, accounting for more than 81% of the variation. The two variables with the highest loadings are DOC and Fe concentration. A strong clustering according to microforms can be seen, especially distinctive between the restored areas and REF (Figure 2.4). Restored areas and REF are clearly separated along PC1 with REF sites having higher $N\text{-}NH_4^+$, and lower $S\text{-}SO_4^{2-}$ and electric conductivity. REF sites were further separated along PC2 depending on differences in vegetation cover, hydrology, and pH. Restored areas where the mineral fill was removed or scraped to near surface level (PR0) were closer to the REF along PC1 than those areas with thicker mineral fill remaining. CR-W was separated from other sites along PC2 due to deep inundation.

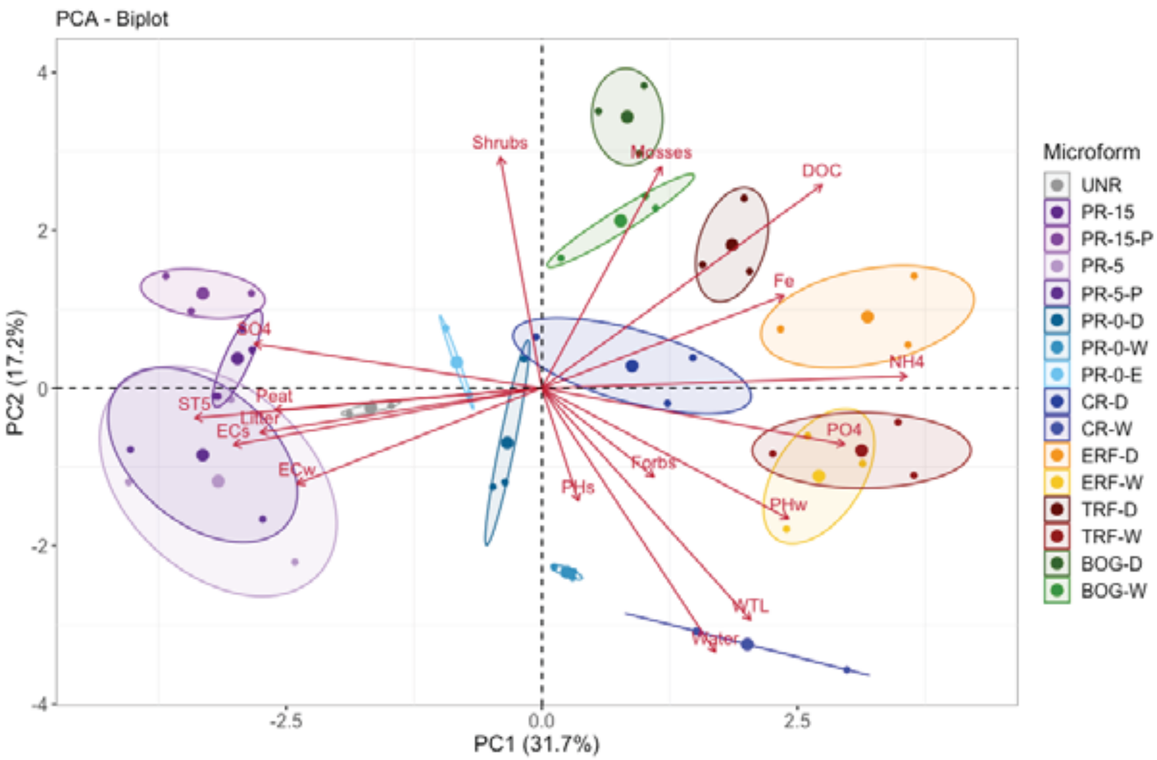


Figure 2.4 Principle component analysis (PCA) of microforms, according to environmental controls of water table level (WTL), soil temperature at 5 cm depth (ST_5), soil pH (pH_S) and electric conductivity (EC_S), water pH (pH_W) and electric conductivity (EC_W), dissolved organic carbon (DOC), vegetation survey strata (shrubs, forbs, mosses, water, litter, peat), and plant available soil nutrient supply rates (Fe^+, NH_4^+, PO_4^-, SO_4^{2-}). Strong clustering of monitoring sectors can be observed. Monitoring sectors according to Table 2.1.

2.5.2 Vegetation

The total vegetation cover in restored areas ranges from 37% (PR5) to 80% (PR0-E), compared to 68% to 95% in REF (Table 2.4). The high cover in PR0-E is due to bryophytes covering 54%, which represents the highest moss cover in all study areas. The other restored areas had similar mix of forbs and bryophytes, while no particularly dominant vegetation stratum developed, ranging from a lowest 9% forb-cover in PR5 to 45% bryophyte-cover in PR-0-D/W. The shrub-cover of 16% in PR15 was nearly three times higher than in the neighboring area PR5 (4%), while shrub species were planted in both areas equally during the restoration process. Yet PR5 is characterized by open water of 40% cover, while PR15 remains dry. On the other hand, an even higher rate of flooding of 93% water cover in PR-0-

D/W does not seem to prevent a natural establishment of plants (67% total vegetation cover) in this study area, of which 7% were shrubs. Considering the fen characteristic plant cover, undisturbed fens TRF (74%) and SRF (77%) have more than double the cover compared to the restored areas PR-15 (31%), PR5 (34%) and PR0-D/W (31%). Only for PR0-E, we can report a fen characteristic plant cover (64%) comparable to REF.

Table 2.4 *Mean cover ± SD in % for plant functional types, open water, plant litter and bare soil (mineral soil or peat), as surveyed in August 2017 and 2018 in all monitoring sectors. Monitoring sectors according to Table 2.1.*

Sector	Total	Fen typical plant species	Vascular plants					Bryophytes		Open water	Plant litter	Bare soil
			Ligneous			Herbaceous						
			Trees	Shrubs	Ericaceous	Forbs	Sedges	Brown mosses	Sphagnum			
UNR	68 ± 19	31 ± 46	4 ± 6	4 ± 6	n.a.	51 ± 24	5 ± 12	16 ± 12	n.a.	n.a.	45	7
PR15	50 ± 19	32 ± 26	16 ± 24	16 ± 24	n.a.	22 ± 17	18 ± 10	16 ± 19	n.a.	n.a.	63	21
PR5	37 ± 20	35 ± 22	6 ± 9	6 ± 9	n.a.	9 ± 9	22 ± 9	17 ± 28	n.a.	14	63	29
PR0-D/W	70 ± 18	35 ± 25	7 ± 9	7 ± 8	2 ± 6	33 ± 14	22 ± 14	45 ± 27	2 ± 6	57	19	8
PR0-E	80 ± 24	64 ± 30	3 ± 6	1 ± 3	n.a.	38 ± 24	24 ± 20	54 ± 27	n.a.	45	21	8
CR	48 ± 35	32 ± 34	1 ± 1	1 ± 2	n.a.	13 ± 6	16 ± 15	29 ± 43	n.a.	67	16	10
SRF	68 ± 25	70 ± 37	32 ± 21	28 ± 20	4 ± 8	31 ± 14	16 ± 8	21 ± 22	n.a.	50	8	12
TRF	79 ± 29	71 ± 44	75 ± 14	16 ± 17	7 ± 13	31 ± 22	19 ± 22	33 ± 40	15 ± 31	42	5	n.a.
BOG	95 ± 3	80 ± 32	15 ± 9	10 ± 10	51 ± 18	7 ± 7	2 ± 2	12 ± 12	47 ± 34	n.a.	17	4

2.5.3 Greenhouse gas exchange and seasonal carbon balance

In 2017, fluxes of gross ecosystem productivity (GEP) were between -104.4 g CO_2 m^{-2} d^{-1} at PR15-P, and -0.1 g CO_2 m^{-2} d^{-1} in SRF-D (Figure 2.5A). In 2018 on the other hand, lowest GEP fluxes were measured in SRF-W (-89.6 g CO_2 m^{-2} d^{-1}), and highest fluxes with -0.2 g CO_2 m^{-2} d^{-1} in CR-W (Figure 2.5D). Net ecosystem exchange (NEE) in 2017 ranged from -55.9 g CO_2 m^{-2} d^{-1} in PR15-P to 34.6 g CO_2 m^{-2} d^{-1} in SRF-W (Figure 2.5B), and in 2018 from -56.6 g CO_2 m^{-2} d^{-1} in SRF-W to 20.6 g CO_2 m^{-2} d^{-1} in CR-W (Figure 2.5E). ANOVA results reveal a significant effect of ST_5 on GEP and ecosystem respiration in 2017, which is repeated only for ecosystem respiration in 2018 (Table 2.5). Microform on the other hand has a consistent significant effect on all fluxes, whereas WTL only has a significant effect on all fluxes 2018. We notice that the wetter the microform, the higher the ecosystem respiration fluxes (Figure 2.5C & F) and CH_4 fluxes.

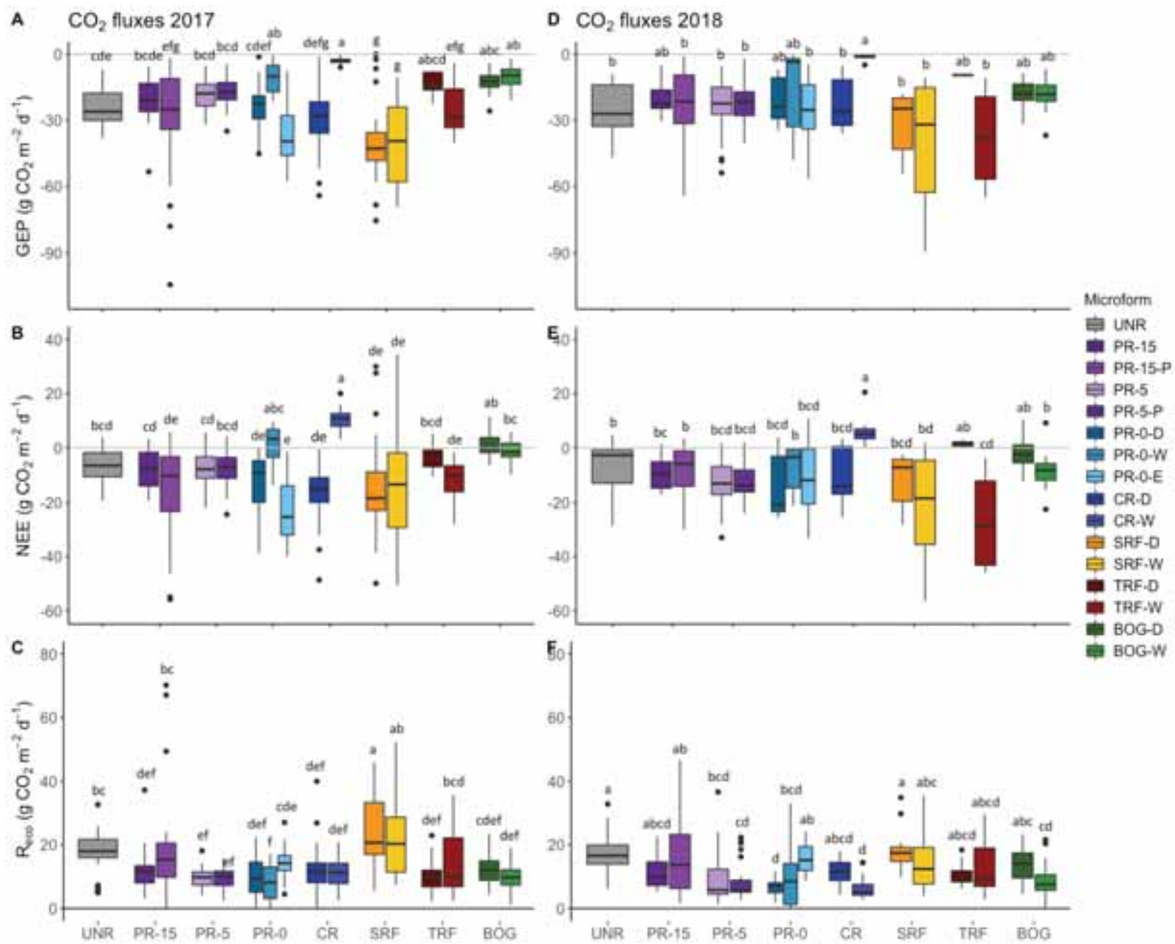


Figure 2.5 Measurements in all monitoring sectors during the monitoring season of 2017 (A-C) and the season of 2018 (D-F) show the mean gross ecosystem production (GEP in g CO_2 m^{-2} d^{-1}) at a photosynthetically active radiation (PAR) photon flux density ≥1000 µmol m^{-2} s^{-1} (A & D), the mean net ecosystem exchange (NEE in g CO_2 m^{-2} d^{-1}; B & E), and the mean ecosystem respiration (R_{eco} in g CO_2 m^{-2} d^{-1}; C & F). Groups with the same letters are not significantly different. Statistical results of the ANOVAs for 2017 show GEP: $F_{17, 359}=16.43$, $p<0.001$, adj. $r^2=0.41$; NEE: $F_{17, 376}=14$, $p<0.001$, adj. $r^2=0.36$; R_{eco}: $F_{17, 550}=19.16$, $p<0.001$, adj. $r^2=0.35$. For flux recordings in 2018 the ANOVA results reveal GEP: $F_{17, 212}=4.38$, $p<0.001$, adj. $r^2=0.2$; NEE: $F_{17, 220}=7.29$, $p<0.001$, adj. $r^2=0.31$; R_{eco}: $F_{17, 329}=6.77$, $p<0.001$, adj. $r^2=0.22$. Monitoring sectors according to Table 2.1

Table 2.5 Statistical results of ANOVAs for fluxes of carbon dioxide (CO_2), presented for gross ecosystem productivity (GEP), net ecosystem exchange (NEE), and ecosystem respiration (R_{eco}), and methane (CH_4) for 2017 and 2018.

Year	Component	Effect	F statistics	*p*-value	Adjusted r^2
2017	GEP		$F_{17,359} = 16.43$	<0.001	0.41
		Microform	$F_{15,359} = 17$	<0.001	
		Soil temperature 5 cm	$F_{1,359} = 24.18$	0.000	
		Water table level	$F_{1,359} = 0.21$	0.65	
	R_{eco}		$F_{17,550} = 19.16$	<0.001	0.35
		Microform	$F_{15,550} = 15.62$	<0.001	
		Soil temperature 5 cm	$F_{1,550} = 91.15$	<0.001	
		Water table level	$F_{1,550} = 0.23$	0.63	
	NEE		$F_{17,376} = 14$	<0.001	0.36
		Microform	$F_{15,376} = 15.7$	<0.001	
		Soil temperature 5 cm	$F_{1,376} = 2.41$	0.12	
		Water table level	$F_{1,376} = 0.01$	0.91	
	CH_4	Microform	$F_{15,292} = 6.75$	<0.001	0.22
2018	GEP		$F_{17,212} = 4.38$	<0.001	0.2
		Microform	$F_{15,212} = 4.05$	0.000	
		Soil temperature 5 cm	$F_{1,212} = 2.04$	0.155	
		Water table level	$F_{1,212} = 11.75$	0.001	
	R_{eco}		$F_{17,329} = 6.77$	<0.001	0.22
		Microform	$F_{15,329} = 6.41$	<0.001	
		Soil temperature 5 cm	$F_{1,329} = 15.52$	<0.001	
		Water table level	$F_{1,329} = 3.47$	0.063	
	NEE		$F_{17,220} = 7.29$	<0.001	0.31
		Microform	$F_{15,220} = 6.86$	0.000	
		Soil temperature 5 cm	$F_{1,220} = 0.26$	0.609	
		Water table level	$F_{1,220} = 20.74$	0.000	
	CH_4	Microform	$F_{15,283} = 7.54$	<0.001	0.25

CH_4 emissions were especially high at all submerged and regularly flooded measurement plots that occurred at CR, SRF, TRF-W (Figure 2.6). The same trend can be noted for extremely wet plots close to the WTL, like PR0 and PR5. Microform showed a significant effect on CH_4 fluxes (Table 2.5). Microforms with highest WTL showed significant effects on CH_4 fluxes, where highest averaged CH_4 fluxes of 417 ± 476 mg CH_4 m^{-2} d^{-1}) in TRF-W, 398 ± 711 mg CH_4 m^{-2} d^{-1} in CR-W, 354 ± 608 mg CH_4 m^{-2} d^{-1} in SRF-W, and 199 ± 294 mg CH_4 m^{-2} d^{-1} in SRF-D were observed. The lowest fluxes on the other hand were recorded at the driest sites with 7 ± 20 mg CH_4 m^{-2} d^{-1} in BOG-D, 1 ± 5 mg CH_4 m^{-2} d^{-1} in BOG-W, 2 ± 8 mg CH_4 m^{-2} d^{-1} in PR15-P, and 7 ± 19 mg CH_4 m^{-2} d^{-1} in PR5-P.

In particular, CR-W was not comparable with the other restored areas and behaved in a completely opposite manner, where specifically NEE and GEP in CR-W differed significantly from all other plots. CH_4 fluxes in CR-W and the reference fens showed similarities, whereas all other restored and unrestored areas were comparable to BOG.

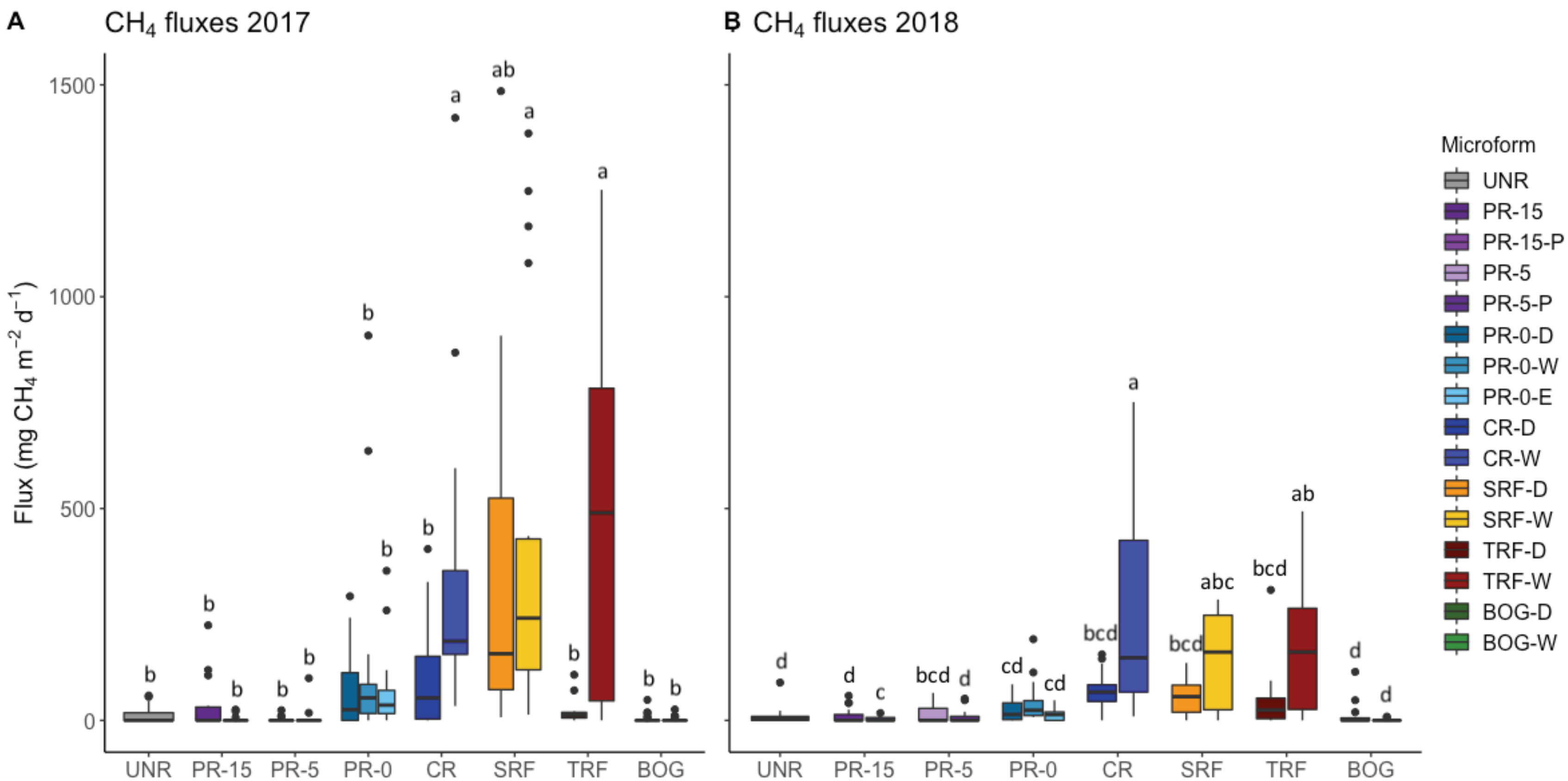


Figure 2.6 Measurements of mean methane (CO_4) fluxes (mg CH_4 m^{-2} d^{-1}) during the 107-days-monitoring seasons in 2017 A) and in 2018 B). Groups with the same letters (in separate years) are not significantly different. Microform showed a significant effect on CH_4 fluxes in both monitoring years, 2017 ($F_{15, 292}$=6.75, p<0.001, adj. r^2=0.2) and 2018 ($F_{15, 283}$=7.54, p<0.001, adj. r^2=0.3). Monitoring sectors according to Table 2.1.

Both linear regression analyses to evaluate spatial variation in NEE and CH_4 fluxes based on environmental variables, such as vegetation strata, WTL and ST_5 were found to be significant (Table 2.6). While for NEE the significant variables were ST_5, shrub cover, and forb cover, the significant variables to predict CH_4 fluxes were ST_5, WTL, cover of ericaceous, and cover of sedges. The higher the cover of these plant functional types, the higher the emission rates observed in the respective study area. Despite wetter and colder weather conditions in 2018, we did not observe a significant effect of climate on CH_4 emissions. Compared to the first year, CH_4 fluxes actually decreased by almost 45% in 2018. Furthermore, we cannot confirm any significant relation between NEE or CH_4 fluxes and FTP.

Table 2.6 Statistical results of multiple linear regressions to predict net ecosystem exchange (NEE) and methane (CH_4) fluxes in 2017 and 2018, based on soil temperature (at 5 cm depth), cover of vegetation strata, and water table level. Marginal $r^2_{(m)}$ shows the proportion of variance explained by the fixed factors alone, while the conditional $r^2_{(c)}$ describes the proportion of variance explained by fixed factors and the random factor 'plot'.

Component	**Effect**	**F-value**	***p*-value**	**SE**	$r^2_{(m)}$	$r^2_{(c)}$
NEE	Intercept	$F_{1,46}$=69.64	<0.0001	5.84	0.22	0.67
	Soil temperature	$F_{1,42}$=2.24	0.142	0.29		
	Shrub cover	$F_{1,42}$=15.46	<0.001	0.09		
	Forb cover	$F_{1,42}$=8.64	0.005	0.06		
CH_4	Intercept	$F_{1, 47}$=56.26	<0.0001	108.83	0.44	0.46
	Soil temperature	$F_{1, 44}$=24.47	<0.0001	6.84		
	Water table level	$F_{1,42}$=39.22	<0.0001	0.58		
	Ericaceous cover	$F_{1,42}$=6.69	0.013	1.07		
	Sedge cover	$F_{1,42}$=4.74	0.035	1.58		

Modelled seasonal NEE revealed restored areas and REF to be greater C sources in 2017 than in 2018 (Table 2.7). A cumulative two-year total C balance shows that C sinks have established in the restored areas closest to the WTL, in PR5, PR0-D and PR0-E, and CR-D, ranging from -625 to -67 g C m^{-2} (Table 2.8). On the contrary, very wet areas with inundated conditions, such as PR0-W, CR-W, and SRF acted as C sources with C emission up to 1039 g C m^{-2}. Very dry areas, such as PR15 and UNR also act as C sources.

Table 2.7 Cumulative seasonal carbon fluxes of methane (CH_4), and net ecosystem exchange (NEE) as a product of gross ecosystem production (GEP) and ecosystem respiration (R_{eco}), for all monitoring sectors in 2017 and 2018. Both seasonal calculations were done for a period of 107 days (17 May 2017 to 31 August 2017 and 22 May 2018 to 5 September 2018). Monitoring sectors according to Table 2.1.

Sector Microform	May – September 2017					May – August 2018				
	GEP	R_{eco}	NEE	CH_4	$Total_{2017}$	GEP	R_{eco}	NEE	CH_4	$Total_{2018}$
	(g C m^{-2})					(g C m^{-2})				
UNR	-394	506	112	0.7	113	-487	574	87	0.8	88
PR15	-322	368	47	1.4	48	-348	368	21	0.7	22
PR15-P	-525	559	34	0.2	34	-512	499	-13	0.2	-13
PR5	-339	285	-54	3.5	-51	-416	346	-70	6.1	-64
PR5-P	-329	282	-47	0.6	-46	-342	224	-118	0.7	-117
PR0-D	-417	314	-103	3.9	-99	-323	110	-213	1.8	-211
PR0-W	-160	206	46	6.6	53	-216	580	365	3.3	368
PR0-E	-421	420	-2	3.3	1	-443	409	-34	1.1	-33
CR-D	-475	348	-127	7.2	-120	-375	284	-91	5.8	-85
CR-W	-39	333	294	31.9	326	-27	225	198	21.7	220
SRF-D	-606	715	109	16	125	-407	638	231	4.6	236
SRF-W	-652	671	19	28.4	47	-606	561	-45	11.9	-33
TRF-D	-219	299	81	2.7	84	-150	334	184	3.7	188
TRF-W	-578	418	-160	33.4	-127	-454	283	-171	14.3	-157
BOG-D	-204	358	155	0.6	156	-283	377	94	0.8	95
BOG-W	-184	262	77	0.1	77	-337	248	-88	0	-88

Table 2.8 Cumulative two-year total carbon (C) balance and global warming potential (GWP) for two 107-days-research seasons in two consecutive years (17 May 2017 to 31 August 2017 and 22 May 2018 to 5 September 2018). Calculations of the total C balance include C fluxes of methane (CH_4), and net ecosystem exchange (NEE) as a sum of gross ecosystem production (GEP) and ecosystem respiration (R_{eco}). Monitoring sectors according to Table 2.1.

Status	Sector	Total C balance ($g C m^{-2}$)	GWP ($g CO_2$-e)
Well pad	UNR	201	735
Restored 2009	PR15	70	257
	PR15-P	21	78
Restored 2009	PR5	-114	-419
	PR5-P	-164	-600
Restored 2008	PR0-D	-310	-1138
	PR0-W	421	1543
	PR0-E	-32	-116
Restored 2007	CR-D	-205	-752
	CR-W	546	2001
REF	SRF-D	361	1322
	SRF-W	14	52
REF	TRF-D	271	995
	TRF-W	-283	-1039
REF	BOG-D	250	918
	BOG-W	-11	-40

2.6 Discussion

Reestablishing hydrological conditions has become the main goal in peatland restoration (Bonn et al. 2016). However, several studies report an increase of CH_4 rates during the first years of rewetting of sites previously drained and used for peat extraction, due to higher methanogenesis activity (Jordan et al. 2016; Nugent 2019). Reported greenhouse gas fluxes of restored peatlands range from -90 to -30 g CO_2-C m^{-2} d^{-1} and 3.7 to 4.2 mg CH_4-C m^{-2} d^{-1} (Strack et al. 2014; Abdalla et al. 2016; Nugent et al. 2018). On the contrary, rates of NEE and CH_4 for undisturbed peatlands in the Mixedwood region of the Canadian boreal forest range from -7.6 to -3.1 g CO_2 m^{-2} d^{-1} and 3 to 65.8 mg CH_4 m^{-2} d^{-1} (Webster et al. 2018).

Since complete removal of introduced *in situ* well pad construction materials creates open water areas that are not representative of pre-disturbance conditions, we hypothesize that this technique would not be beneficial for peatland restoration efforts and would likely increase C emission rates. We further hypothesized that the emergence and a high abundance of fen characteristic plant species post-restoration, as well as biochemical attributes comparable to REF, would enable the return of net C uptake in restored areas at rates similar to nearby REF.

2.6.1 Fen typical vegetation not improving C uptake, whilst need for shrub species

Biochemical conditions, specifically nutrient concentrations, differed greatly between REF and restored areas. REF were characterized by high ammonium, phosphate and iron concentrations and did not compare to any restored area. The sulfur concentration of only one restored area, PR0-W, was comparable to the REF fens, while sulfur concentrations of all remaining restored areas were very different than REF. It is likely that the extremely high sulfur concentrations in PR15 and PR5 to sulfur compounds in the well-pad materials leaching through the root zone and being held in place by the clay layer (Himes 1998), used for infilling the well pad, which does not allow an exchange between the mineral clay and the peat layer underneath. This effect might be enhanced through decomposing organic matter at the surface, considering the high cover percentage of plant litter (63%) and the low WTL in both areas. These processes would also explain the low pH in the same study areas despite the residual mineral soil (pH 3.4 in PR15, pH 3.6 in PR5) in the second drier year

2018. Indeed, during drier conditions, H^+ ions might have been produced and dissolved, acidifying the milieu (DeVries & Breeuwsma 1987).

Soil chemistry post-restoration is comparable to characteristic poor fen (pH lower than 5.5 and electric conductivity lower than 100 $\mu S\ cm^{-1}$) and moderate-rich fen conditions (pH 5.5 to 7 and electric conductivity 100 to 250 $\mu S\ cm^{-1}$) that come with the acidic and moist environment (Alberta Environment and Sustainable Resource Development 2015). The exceptional high values of electric conductivity of the soil in PR5 and PR15 might be due to the remnants of various soil amendments, i.e., the application of commercial peat, field peat, slough hay, or woodchips, applied during the restoration work in 2007 (Vitt et al. 2011). We note that the chemical conditions depicted for CR depicts conditions found in CR-D and no represent well CR-W, because soil sampling was not possible in deep water.

Despite the chemical differences found in the study sites, vegetation re-established among all restored areas. We observed a fen-like vegetation recovery in areas that were most closely leveled with the WTL of the surrounding ecosystem. This effect was especially true of the moss layer. However, we could not confirm a higher abundance of fen characteristic plant species the closer the restored areas are levelled to the WTL. We had expected to observe a gradient of low abundance of fen characteristic plant species in the areas with a highest distance to the WTL, PR15, to a high abundance of fen typical species in the areas with the shortest distance and most even level to the WTL, PR0-E. Indeed, the highest cover of fen typical species in restored areas (64%) was observed in PR0-E, but among the remaining areas with partially and completely removed mineral soil, and even UNR, there was no difference in the fen typical species cover. WTL is an important driver of vegetation re-establishment on residual mineral soil (Vitt et al. 2011; Howie et al. 2016), but it does not appear to be a crucial factor for fen characteristic plant species to distribute. While Halsey et al. (1998) consider the mineral soil's larger grain size (i.e., glaciofluvial and eolian deposits) with high hydraulic conductivity to be an important driver for peatland development, our findings indicate that achieving an elevation similar to the surrounding peatland is likely sufficient for the establishment of typical fen plant communities.

Furthermore, we cannot confirm that the cover of fen typical plants improved the C sink function considerably. In fact, we found no significant relationship between the cover of fen typical plants and either CO_2 or CH_4 exchange. While the species composition in PR0-E was most comparable to a REF and the C sink function seems to have returned to this area, CH_4 fluxes were not drastically decreased. This might be due to the high cover of sedges (24%), such as *Carex* spp. and *T. latifolia,* with their large aerenchymatic systems and abundant root biomass that are known to promote CO_2 emissions and are more likely to enhance C emissions to the atmosphere rather than C uptake (Bellisario et al. 1999; Strack et al. 2016; Rupp et al. 2019). These species were also well-established in the areas with partially removed mineral soil, PR15, PR5 and PR0. Overall, it appeared that the presence of plant cover in general and the position of the WTL were the drivers of the net C balance observed in each restoration treatment.

On the contrary, other studies show that species identity, and especially the presence of *C. aquatilis* was important for C sequestration at restored fens, specifically in combination with bryophyte species (Hassanpour Fard et al. 2019; Murray et al. 2019). The floating moss carpet CR-D displays very well the positive effect of this combination of fen typical vegetation on C accumulation and greenhouse gas fluxes, where the total C emission rates of the emerging floating moss carpet in CR-D (>7.2 g C m^{-2} d^{-1}) are almost 85% lower than in the open water areas CR-W. The development of the floating moss carpet in this study area was favored by proximity to the adjacent peatland with bryophytes able to grow out from the edge of the open water area, but this result cannot be applied in case of the removal of an entire *in situ* well pad of 1ha where much of the area would be far from the edge. In that case, the development of vast shallow open water will cause wave development and too rough water movements for a moss carpet to establish (Blievernicht et al. 2007; Gaudig et al. 2013). At the same time the vegetation that developed in CR-D accommodated the second highest cover rate of fen characteristic plant species (7%), including species such as *C. aquatilis*, *C. diandra*, *Drepanocladus aduncus*, *D. polygamus*, and *Menyanthes trifoliata*.

The highest C sink was found in PR0-D, an area defined by a vegetation community including shrub and sedge species. Fen characteristic plant species in this study area included *Andromeda polifolia*, *B. pumila*, *C. limosa*, *C. utriculata*, *C. palustre*, *P. pseudotriquetrum*.

Opposed to herbaceous species, shrub species store C in their woody structures and in combination with their smaller root structure, emit lower rates of CO_2 to the atmosphere (Rupp et al. 2019). Furthermore, because of their overwintering photosynthesizing leaves, shrub and bryophyte species have the advantage to begin C uptake early in the growing season before herbaceous plant species (Arndal et al. 2009). The regression analysis also indicated that shrubby plant species had an effect on NEE and CH_4 fluxes. In PR0-D, CR-D, and PR5-P, we observed the same trend of highest C uptake, compared to the other restored areas that act as a C source. Especially in these three study areas similar plant species composition had established, with high cover of *C. aquatilis* in combination with various shrub (PR0-D) and bryophyte (CR-D) species. However, we note that both REF fens TRF and SRF, despite a high cover of shrub species (31% and 32% shrub cover, respectively) in combination with high WTL (32% and 37% open water respectively), were carbon sources. We note, however, that the tall overstory trees of both wooded TRF and BOG were not included in our C flux measurements, which would contribute additional C uptake to the system. Furthermore, tree root respiration rates in TRF and BOG were not considered in our measurements and likely contributed to the ecosystem respiration measured indicating that actual NEE needs to be adjusted according to previous research. Munir et al. (2017; 2015) report on tree root respiration rates in an ombrotrophic bog in Alberta ranging from 2 ± 0 g CO_2-C m^{-2} in ambient hollows to 70 ± 6 g CO_2-C m^{-2} in warmed hummocks. Several studies describe a strong relation between tree productivity and hydrology, where belowground production and root respiration rates were enhanced in drained conditions (Hanson et al. 2000; Hermle et al. 2010; Munir et al. 2017).

Despite the apparent importance of shrubs, the introduction of plant species in some areas, even including *Salix* spp. species, does not seem to have specifically improved the C sequestration at the respective areas. Considering the high percentage of bare soil in both PR5 (22%) and PR-15 (16%), as compared to a maximum of 3% bare soil in PR0-D/W, the revegetation effort does not seem to be more successful than natural and spontaneous revegetation (Prach & Hobbs 2008), which is favorable if landscape factors, such as WTL and a nearby abundant species pool, are considered during the restoration process (Konvalinková & Prach 2014). This effect can be seen in the mean cover of FTP, such as *B. pumila*, *C. aquatilis*, *C. diandra*, *C. palustre*, *Campylium stellatum*, *S. warnstorfii*, *T. nitens*, being almost twice as high in PR-0 (43%) where no plants were introduced compared to PR5 (24%) and PR15 (22%) where planting occurred. The combination of shrub species and a stable WTL at the soil surface, as can be found in PR0-D, seems to provide best results in returning a C sink function to restored areas, as well as establishing biochemical conditions comparable with REF fens. This effect seems to have greater impact on C exchange, rather than the variable climatic conditions, since despite wetter and colder conditions during the second year, GEP rates were higher than in the previous year with lower precipitation rates and warmer temperatures (Strack et al. 2016).

2.6.2 Enhanced CH_4 emissions after complete well pad removal

Our third hypothesis that a complete *in situ* well pad removal would enhance CH_4 emissions was confirmed through highest emission rates observed in the open water area CR-W (21.7 to 31.9 g C m^{-2} d^{-1}). PR0-W, TRF-W and SRF, with WTL above ground surface, show comparable emission rates of 11.9 to 33.4 g C m^{-2} d^{-1}. While such high CH_4 emission rates are normal in undisturbed fens with high water table (Waddington & Roulet 2000; Strack et al. 2006; Bienida et al. 2020), adjusted management practices are available to avoid unnecessary high emission rates in restored wetlands (Strack et al. 2014), just as it has been done in the drier managed microforms PR0 (<3.9 g C m^{-2} d^{-1}), PR5 (<6.1 g C m^{-2} d^{-1}), PR15 (<1.4 g C m^{-2} d^{-1}), which low emission rates endorse this argument. Considering the enormous global warming potential (GWP; Myhre et al. 2013) of the wet areas of up to 3807 g CO_2-e in CR-W (Table 2.8), the drier managed sites ranging between -556 g CO_2-e and 326 g CO_2-e prove to be a powerful argument to pay attention to proper site management and hydrological adjustments. Furthermore, Günther et al. (2020) show that rewetting of drained peatlands should not be feared or avoided, because the positive effects of rewetting for restoration purposes outweighs the rather short-lived radiative forcing of CH_4, as opposed to the long-lived impacts of CO_2 emissions, when sites remain too dry, and the long-term benefits of recreating net C accumulating ecosystem through restoration.

2.7 Conclusion

After eight to eleven years following restoration, the rates of C exchange in the restored areas were comparable with the rates of long-time established REF, although the soil and water chemistry remained quite different due to the residual mineral soil layer. Regarding our results, we conclude that the C uptake in restored areas is most similar to reference peatlands, when vegetation has established and particularly when regional wetland typical shrub species have colonized. The crucial base for vegetation communities to emerge is the proper hydrological management of the site, where the surface elevation should be evenly adjusted to the surrounding ecosystem and as close to the landscape surrounding WTL as possible. It appears that in order to have appropriate chemical fen conditions establish post-restoration, available plant organic matter needs to remain in anaerobic conditions and therefore the WTL is crucial to remain near the surface. However, the development of open water areas is to be avoided at all times. As was observed in flooded conditions in PR0-W where the WTL is higher than 30 cm above surface, and in the shallow open water areas CR-W, with a WTL larger than 77 cm above surface, hydrological conditions appeared to be less beneficial to fen plant establishment while driving C emissions, resulting in high rates of ecosystem respiration and CH_4 emissions. It is therefore crucial during future restoration work, to level any residual layers of remaining mineral soil with the adjacent peatland ecosystem, in order to obtain a seamless connection and create optimal hydrological conditions in the restored area. Reintroduction of (shrub) species can be neglected, if appropriate hydrological conditions are achieved and a source of peatland plant species is available in adjacent wetland ecosystems.

The assessment of the biogeochemical and biochemical conditions in the restored areas should continue on a regular, long-term basis in order to monitor the effect of the developing fen vegetation and associated peat accumulation on the soil chemistry. Long-term monitoring is important to maintain in the context of peatland development, as these ecosystems may take several decades to centuries to establish. We again stress the importance of individual management and soil adjustment for a hydrological connection of restored peatlands following *in situ* oil mining well pad disturbances.

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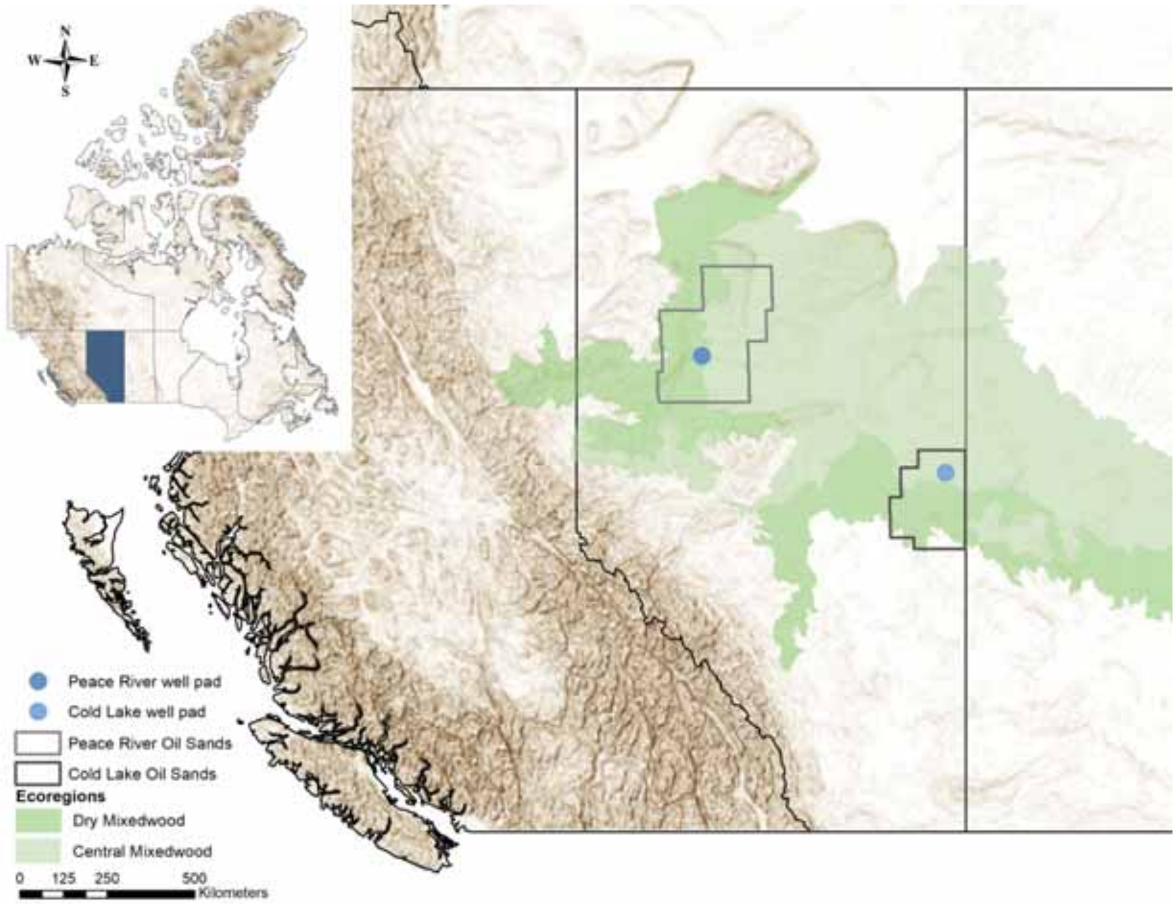
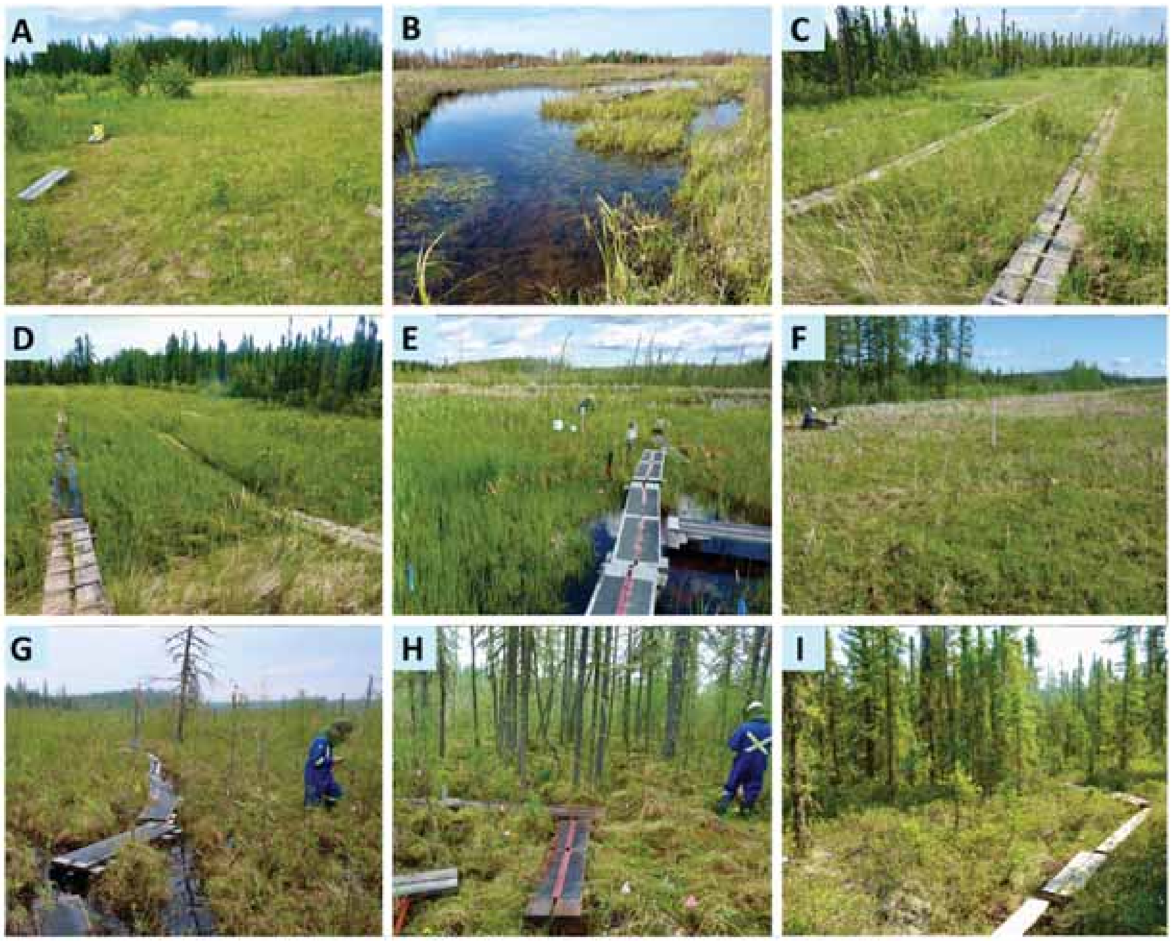
2.10 Appendices

Appendix 2.1 The study sites, two restored in situ oil sands well pads, situated within the Oil Sands regions of Peace River and Cold Lake. Both sites are located in the Boreal Mixedwood ecoregion of boreal Northern Alberta.



Appendix 2.2 All study sectors in the Peace River and Cold Lake Oil Sands regions. A) An unrestored sector (UNR) on the Cold Lake well pad served as a control, where measurements took place on the remaining well pad's mineral soil (MS). Restored sectors are B) CR: Complete removal of MS with a floating moss carpet, C) PR15: Partial removal of MS to 15 cm above seasonal water table, D) PR5: Partial removal of MS to 4 to 6 cm above seasonal water table, E) PR0-D/W: Partial removal of MS to surface elevation of surrounding fen reference ecosystem (uneven ground relief), F. PR0E: Partial removal of MS to same surface elevation of surrounding fen reference ecosystem (even ground relief). Reference ecosystems (REF) were G) SRF: a shrubby extreme-rich fen, H) TRF, a wooded rich fen, and I. BOG: a wooded bog.



Chapter 3 Organic matter production and decomposition in restored fens following *in situ* well pad disturbances

3.1 Résumé

La restauration écologique vise à rétablir et à initier le développement successif de fonctions et de processus caractéristiques d'un écosystème. Dans le cas des tourbières perturbées c'est souvent la fonction d'accumulation de tourbe. Dans cette étude, nous avons documenté le potentiel d'accumulation de tourbe de cinq zones restaurées à la suite des perturbations causées par des infrastructures de plateforme de forage *in situ*, en comparaison avec trois tourbières de référence non perturbées et une zone non restaurée. L'accumulation de tourbe a été calculée au moyen de la productivité primaire nette et des taux de décomposition. La production de biomasse aérienne a été estimée en utilisant le pic de récolte sur pied à la fin de deux périodes de végétation. La production de racines souterraines a été mesurée à l'aide de sacs de croissance incubés sur une période de deux ans. La décomposition a été estimée en utilisant la méthode des sacs de litière végétale avec une incubation sur une période de 23 mois. Les résultats montrent que l'accumulation de tourbe revient dans les tourbières restaurées malgré le remplissage minéral résiduel, en particulier dans les zones de développement des bryophytes. Nous avons constaté que l'élimination partielle du remblai minéral favorise l'accumulation de tourbe et la séquestration du carbone, car les conditions hydrologiques et le développement de la végétation fournissent un apport de biomasse et des conditions anoxiques pour une décomposition lente. L'enlèvement complète d'une ancienne plateforme de forage a créé une zone d'eau libre peu profonde, avec des conditions semblables à celles des marais, qui inhibe l'accumulation de tourbe en raison de la faible production de matière organique végétale et de la décomposition rapide.

3.2 Abstract

Ecological restoration aims at returning and initiating successional development of characteristic functions and processes, like peat accumulation, to disturbed peatlands. In this study, we documented the peat accumulation potential of five restored areas following disturbances caused by *in situ* well pad infrastructures, compared to three undisturbed reference peatlands and an unrestored area. Peat accumulation was calculated by means of net primary productivity and decay rates. The above-ground biomass production was estimated using the peak standing crop at the end of two vegetation periods. Below-ground root production was measured with ingrowth bags incubated over a two-year period. Decomposition was estimated using the plant litter bag method, with incubation for a period of 23 months. Results show that peat accumulation returns to restored peatlands despite residual mineral fill, especially in areas with bryophyte development. The complete removal of a former well pad resulted in a shallow open water area, with marsh-like conditions, which inhibits peat accumulation because of little plant organic matter production and fast decomposition. We found the partial removal of the mineral fill to support peat accumulation and carbon sequestration because hydrological conditions and vegetation development provide biomass input and anoxic conditions for slow decomposition.

3.3 Introduction

The Canadian landscape is characterized by about 1.1 million km^2 of peatland ecosystems, which are particularly abundant (64% of Canadian peatlands) within the boreal forest (Tarnocai et al. 2011; WCS Canada 2021). However, these vast boreal peatland regions are vulnerable to disturbance by the oil and gas industry within the extensive oil sand regions of northern Alberta, especially to *in situ* ("in place") bitumen extraction activities. As of 2016, in the Oil Sands regions of Alberta, a land area of about 39 000 km^2 have been under operation by the oil and gas industry, while the designated total *in situ* mineable area equals approximately 142 000 km^2 (ABMI 2018). *In situ* oil extraction requires a vast network of exploration sites and roads, bitumen storage facilities and processing plants, *in situ* oil sands well pads supporting the bitumen well heads, connecting access roads, and pipelines. As of October 2021, there are combined more than 329 000 *in situ* wells (157 000 active and 172 000 inactive/abandoned) in Alberta that will require restoration work in the future (AER 2021; Government of Alberta 2021). For the construction of an *in situ* well pad, the vegetation is cut down, especially larger trees and shrubs, and covered with a geotextile before a compacted mineral soil mix of sand, loam, clay, a gravel is put in place. The decommissioning of an *in situ* oil sands well pad after an average lifespan of about 20 to 30 years includes the sealing of the wells and dismounting of oil extraction equipment such as well heads and pumping jacks (CAPP 2021). Since 2015, peatland restoration is obligate and aims at returning pre-disturbance functions such as wildlife habitat, peat accumulation, and carbon sequestration (Government of Alberta 2013; Environment and Parks 2017).

Peatland restoration following *in situ* oil sands disturbances started in the 2000's, with different restoration techniques being tested. Restoration trials included 1) the complete removal of all well pad construction materials following one of the restoration ecology principles to first eliminate any further disturbance ("CR" hereafter), 2) the inversion of underlying peat layers with the upper former *in situ* well pad's mineral fill, and 3) the partial removal of the mineral fill to near the water table ("PR" hereafter). The CR technique aimed at exposing the peat that was buried and compacted underneath the mineral fill and reinstating suitable substrate with respective biochemical conditions to support characteristic peatland vegetation (Imperial Oil Resources 2017, personal communication; Xu et al. 2021). In one restoration trial, the buried peat was decompressed and then revegetated following the steps of the moss layer transfer technique, where fragments of peatland characteristic vegetation, specifically mosses such as *Sphagnum* spp. L. and *Polytrichum strictum* Menzies ex. Brid., were introduced (Quinty & Rochefort 2003; Xu et al. 2021). In another trial, the depression that was created during the process of CR filled up quickly with inflowing water from surrounding wetland ecosystems and a shallow open water area formed instead (Imperial Oil Resources 2017, personal communication). The inversion of the buried peat and the upper compacted mineral fill resulted in underlying peat to resurface and provide peat substrate for introduced peatland bryophyte species (Xu et al. 2021). The PR technique on the other hand, was inspired by the way peatlands initiated their formations on mineral soil via paludification following the glacial era (Halsey et al. 1998; Vitt et al. 2011). PR aimed at reinstating early wetland characteristic hydrological conditions to initiate the development and succession of fen ecosystems by stripping the upper soil layers of the well pad down to near the water table level (Vitt et al. 2011).

Functional, undisturbed peatlands are characterized by ongoing carbon (C) sequestration during the peat accumulation process. Peat accumulation is only possible if C uptake via gross ecosystem productivity and plant biomass production exceeds the C loss via ecosystem respiration and organic matter decomposition. In functional peatlands, the process of decomposition is delayed due to low temperatures and oxygen (O_2) deprivation reducing most microbial activity except of adapted microorganisms including those that produce methane (CH_4) as a consequence of anaerobic respiration. Due to the preservation of organic matter in the anoxic layers of water-logged peat, undisturbed peatlands are efficient C storing ecosystems with a C density of 1 048 to 1 441 million g C ha^{-1} (Tarnocai et al. 2011). Specifically, the continental western Canadian peatland carbon stock has been estimated at 48 billion g C (Vitt et al. 2000). The rate of the ongoing soil carbon uptake in Canadian peatlands is estimated at 190 000 g C ha^{-1} yr^{-1} (19.4 g C m^{-2} yr^{-1} for western Canadian peatlands; Vitt et al. 2000; Rydin & Jeglum 2013). C uptake via net primary productivity (NPP) and loss through decomposition largely depend on the interaction of biotic and abiotic factors, in particular hydrology, nutrient availability, biochemistry, vegetation community composition, and the activity and composition of decomposing microbial communities (Laiho 2006; Vitt 2007; Mitsch & Gosselink 2015).

In western Canadian bogs, above-ground NPP of 198 to 674 g m^{-2} yr^{-1} and a below-ground root production of about 26 to 60% of the total NPP have been observed (Thormann et al. 1999; Weltzin et al. 2000; Vitt et al. 2001). Research shows that more than 50% of the litterfall is returned to the soil following initial decomposition (Wardle et al. 2004), but no linear relationship has been found between NPP and soil organic matter accumulation (Jackson et al. 2017). Mass loss of 33 to 46% after one year has been observed in west Canadian bogs, indicating a slow decomposition rate (Thormann & Bayley 1997b). Organic matter accumulation in bogs has been observed to happen at a rather low rate of 59 to 270 g m^{-2} yr^{-1} due to recalcitrant plant litter quality that slows decomposition (Thormann et al. 1999; Frolking et al. 2001; Turetsky et al. 2007). Straková and colleagues (2011) argue that the lower decomposition rates of organic matter found in bogs are due to the low activity of extracellular enzymes produced by microbes that are found in litters of characteristic bog plant species, in particular *Sphagnum* spp. L. and ericaceous plants, compared to the activity found in vascular plants' litter. Despite the low productivity the C accumulation rate in bogs is higher, compared to highly productive fen ecosystems due to reduced decomposition rates in the former (Turunen et al. 2002; Laiho 2006). C accumulation rates in bogs have been estimated at 33.4 to 95 g m^{-2} yr^{-1} (Thormann et al. 1999).

Rich fens, on the other hand, have higher water table levels and mineral content compared to bogs, and provide habitat for high productive vegetation communities. Numerous vascular herbaceous and sedge species growing in fens provide above-ground biomass at NPP rates of 214 to 1 050 g m^{-2} yr^{-1} (Thormann & Bayley 1997a; Vitt et al. 2001, 2009). The root productivity in fens, especially of productive sedge species such as *Carex* spp. L. and *Eriophorum* spp. L. has been estimated at 55 to 86% of the total NPP (Weltzin et al. 2000). Fens have a much higher decomposition rate than bogs in west Canada, where organic mass loss ranges from approximately 45 to 83% during the first year (Thormann & Bayley 1997b). Hence, organic matter accumulation rates in west Canadian rich fens have been estimated at 126 to 254 g m^{-2} yr^{-1} (Thormann et al. 1999; Vitt et al. 2009). C accumulation rates in rich fens have been estimated at 14 to 122 g m^{-2} yr^{-1} (Thormann & Bayley 1997a).

The aim of our study was to evaluate the peat accumulation potential of restored peatlands following disturbances caused by *in situ* oil and gas infrastructures via the net primary productivity (NPP) and organic matter decay. The results will help to understand corresponding data from earlier studies on C sequestration in the same study areas. Five study areas were selected for this research on two former *in situ* oil sands well pads in the Peace River and Cold Lake Oil Sands regions of Alberta where peatland restoration had been achieved via the complete (CR) and partial removal (PR) technique. Three study areas (CR and PR treatments) had been left for natural ingress of migrating diaspores, while two study areas (PR treatment) were actively revegetated through planting of characteristic peatland species. We assessed the plant biomass produced above- and below-ground and the remaining plant matter following decomposition during two study years in the five restored peatland study areas. Results were compared to an unrestored area on the former *in situ* well pad and three undisturbed reference peatlands. Our research questions were: 1) Which technique of mineral fill removal is necessary to support peat accumulation? 2) Is revegetation of peatland plant species necessary to return peat accumulation potential? 3) Do vegetation communities return to restored peatlands that have potential peat accumulation rates comparable to undisturbed reference peatlands?

3.4 Materials and Methods

3.4.1 Study sites

The research was carried out on two decommissioned *in situ* well pads situated in the Oil Sands regions of Peace River (56°23'0.95" N, 116°46'43.43" W) and Cold Lake (54°41'10.82" N, 110°30'59.75" W) in the boreal forest of northern Alberta (Appendix 2.1). Three undisturbed peatland ecosystems adjacent to the former well pads served as reference sites. A total of nine designated study areas included five restored areas, three reference areas, and one unrestored control area.

Peace River well pad research areas

The former *in situ* well pad in the Peace River Oil Sands is situated in the dry mixedwood ecoregion. The ecoregion is characterized by daily temperatures of 13 °C and an average 112 days frost-free period between May and September. 70% of the approximately 390 mm annual precipitation fall during the average 112 days frost-free period between May and September (Table 3.1; Government of Canada 2019).

Table 3.1 Climate data for all three years of the study with special focus on the five-month-long vegetation season between May to September (CRIM 2021).

Climate data	**Study region**	**Study year**		
		2017	**2018**	**2019**
Mean min. temperature May-Sep (°C)	Cold Lake	9	8	8
	Peace River	7	6	6
Mean max. temperature May-Sep (°C)	Cold Lake	21	20	19
	Peace River	21	20	19
Cumulative precipitation May-Sep (mm)	Cold Lake	300	347	284
	Peace River	194	209	256
Cumulative precipitation total year (mm)	Cold Lake	495	501	403
	Peace River	321	313	334

Restoration done at the Peace River well pad consisted of the partial removal of the well pad's mineral fill, where the surface mineral soil layer was scraped down to 15 cm above the average water table level (restoration area PR15; **Figure 3.1**A), and to 5 cm above the average water table (restoration area PR5; **Figure 3.1**B). Both restoration areas were subject to vegetation reintroduction by planting of *Carex aquatilis* Wahlenberg, *Larix laricina* (Du Roi) K. Koch and *Salix lutea* Nuttall. An undisturbed wooded bog (BOG; **Figure 3.1**C), adjacent to the restoration areas, served as a reference area. Detailed description of the restoration treatments and on the selection of the study sites and research areas can be found in Vitt and colleagues (2011) and Lemmer and colleagues (2020), respectively.

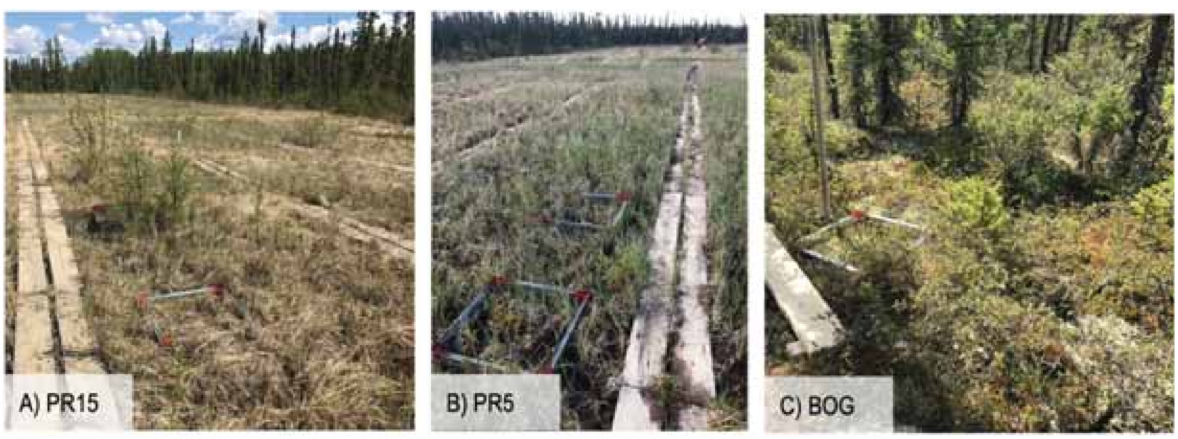


*Figure 3.1 Study areas at the former Peace River in situ oil sands well pad **A)** PR15: Partial removal of mineral soil (MS) to 15 cm above seasonal water table; **B)** PR5: Partial removal of MS to 4 to 6 cm above seasonal water table; **C)** BOG: Treed bog with characteristic hummocks (dry microsite) and hollows (wet microsite).*

Cold Lake well pad research areas

The Cold Lake Oil Sands are part of the moist mixedwood ecoregion, with typically average daily temperatures of about 14 °C during a 116 days frost-free period, and an average precipitation rate of 421 mm per year (Table 3.1; Government of Canada 2019).

Restoration techniques tested at the Cold Lake well pad consisted, besides the partial removal of the mineral fill, also of the complete removal of all introduced well pad construction materials. The partial removal that was applied, levelled the surface of the former well pad to the water table (mineral fill scraped down to 0 cm). Different microforms developed in the partial removal area (restoration area PR0), where the soil surface is either above the average water table (dry microform PR0-D; **Figure 3.2**B), or below the average water table (wet microform PR0-W; **Figure 3.2**C). Another restoration area, where partial removal to 0 cm was applied but no microforms developed, remained very even at the water table (restoration area PR0E; **Figure 3.2**D). The restoration area where the former well pad's mineral fill and the underlying geotextile were completely removed (CR) is characterized by a shallow open water area with a water table level of more than 80 cm above surface (restoration area CR-W; **Figure 3.2**F) in which a brown moss carpet (CR-D; **Figure 3.2**E) floated. All the restoration areas at the Cold Lake well pad study site have been left for spontaneous revegetation by natural ingress from nearby diaspore sources. As a control, an unrestored but spontaneously revegetated area on the former well pad (unrestored control area UNR; **Figure 3.2**A) was used. A treed rich fen (TRF; **Figure 3.2**H) and a shrubby rich fen (SRF; **Figure 3.2**G) adjacent to the restored wetlands serve as two additional reference study sites. Detailed descriptions of the research areas can be found in Lemmer and colleagues (2020).

Figure 3.2 Study areas at the former Cold Lake in situ oil sands well pad: A) UNR: Unrestored; B) PR0-D: Partial removal of mineral soil (MS) to surface elevation of surrounding fen reference ecosystem, dry microform; C) PR0-W: Partial removal of MS to surface elevation of surrounding fen reference ecosystem, wet microform; D) PR0E: Partial removal of MS to same surface elevation of surrounding fen reference ecosystem, even ground relief without microform; E) CR-D: Complete removal of MS with a floating moss carpet as dry microform; F) CR-W: Complete removal of MS with shallow open water as wet microform; G) SRF: Shrubby extreme-rich fen (REF); H) TRF, a treed rich fen (REF). Both REF had characteristic hummocks (dry microsite) and hollows (wet microsite).

3.4.2 Sampling design

Sampling of the restoration approaches applied in each study area included net primary productivity (NPP), the decay rate *k*, and environmental parameters during two study seasons. NPP included the above-ground (AG) and below-ground (BG) organic matter produced. Both above-ground and below-ground biomass were collected at three sample spots in each of the study areas (n=39) at the peak of two corresponding vegetation periods. Decomposition was measured over a period of two years for vascular plants (n=260) and bryophytes (n=260) separately in each study area. Data on environmental parameters were measured for related studies at three study plots in each microform of the study areas, except in PR5 and PR15, where six study plots had been installed (n=45; Lemmer et al. 2020; Chapter 1).

3.4.3 Environmental parameters

Environmental constraints on plant productivity are water table level (WTL), temperature, and biochemistry (Brinson et al. 1981; Jackson et al. 2017; Mäkiranta et al. 2018). We measured WTL in respect to the surface (including the groundlayer vegetation, such as bryophyte cover) and soil temperature at 5 cm depth (ST5) biweekly for related research at three study plots in each microform of the study areas during the vegetation period of 2017 and 2018 (Lemmer et al. 2020).

3.4.4 Biomass and productivity

An ecosystem's productivity can be measured as a rate of biomass production during a specific period.

Above-ground biomass

To assess above-ground biomass, the aerial live portion of plant biomass was collected at peak standing crop in August 2017 and at the end of September 2019. Three 50 × 50 cm sample frames were randomly placed in the study areas and with respect to present microforms (dry/wet; n=39). In each sample frame, all living plant biomass that had been produced in the respective vegetation season, was cut at the shoot/root transition zone just above the surface of the vegetation covering the ground, such as bryophytes. The ground layer of bryophytes and lichens was considered acrotelmic above-ground living plant canopy

(Vitt 2007) and was therefore included in the above-ground biomass. We were careful not to include any root biomass. No large trees were present in the unrestored or restored areas.

Samples were cleaned of all organic litter (necromass). If the biomass expressed a decomposition state of more than 75% it was considered necromass and excluded from the biomass used for productivity calculation of that year. All biomass was sorted into corresponding life forms (shrubs, ericaceous, herbs, sedges, brown mosses, peat mosses, lichens). Tree species such as *Picea mariana* (Miller) Britton, Stern & Poggenburgh and *L. laricina* (Du Roi) K. Koch that had only grown to shrub size since restoration were treated as shrubs and added to the respective sample. Considering perennial plant species (woody shrubs and trees), we carefully selected only the plant tissue which had been produced during the respective growing season and excluded plant tissue from previous years. Samples were then dried at 70 °C to constant weight (g/m^2).

Below-ground biomass

Below-ground root growth was measured with root ingrowth bags (Finér & Laine 2000). In each microsite (dry and wet) of all restored and reference wetlands, three replicate cylindrical cores were extracted from peat (10 × 30 cm) or of mineral soil (5 × 20 cm) in the respective study areas in August 2017 (n=48). The cores were cleaned of all living and senescent plant litter and roots and filled into respectively sized root ingrowth bags made of 1 mm fibreglass mesh material. The bags were individually labelled and closed with fishing line. Ingrowth bags were stored frozen until reinsertion in their corresponding sample spots at the end of October 2017 at the Cold Lake study site and in beginning of November 2017 at the Peace River study site.

Ingrowth bags were retrieved after an incubation period of 692 days at the Peace River study site and 700 days at the Cold Lake study site. Before retrieving the bags, we made sure to cut all roots growing in or out of the bag, to not disturb the total root biomass collected. A total of 40 bags were retrieved. In five instances, ingrowth bags had been lost or damaged, while in CR-W all the incubated ingrowth bags were categorically destroyed by wildlife. The total of eight bags were lost, which accounts for a loss of 17%. Peat cores were kept frozen until analysis in the lab. Analysis consisted of careful manual cleaning the bags from any necromass and live aerial plant organic matter, other than roots. In order to retrieve principal and fine roots without problems, mineral soil cores were washed in a water bath and roots were manually extracted after sedimentation allowed a clear view. The loose peat structure was much less dense than mineral soil cores and allowed for careful manual extraction of the roots under a binocular. The extracted roots were then dried for at least 48 hours at 70 °C until constant weight.

3.4.5 Decomposition

To measure above-ground decomposition, we applied the litter bag technique (Falconer et al. 1933; Lunt 1933; Bocock & Gilbert 1957), which simulates the decay of above-ground organic plant matter after mortality. The technique measures the remaining mass of a certain aerial plant biomass after a given period of time. Since the focus of this study was the decomposition of a representative plant community in the restored areas, and not of a specific plant species, we used mixed above-ground biomass to fill the litter bags (Ward et al. 2010). Vascular plants and bryophytes were incubated separately in different bags.

Live organic plant matter was collected during August 2017. The biomass collection was done in each microform of the study areas in three randomly placed 50 × 50 cm quadrats. All species collected in each microform are shown in (Appendix 3.1). The live biomass was cleaned of litter, cut, mixed, and dried at 40 °C until constant weight. Litter bags for vascular plants, sized 10 × 10 cm, were made of 1 mm fibreglass mesh material (Appendix 3.2). Litter bags for bryophytes were made of 1 mm nylon mesh material and sized 6 × 6 cm (Appendix 3.2). The bags were filled with 1-1.5 g dried mixed litter, tagged, and closed with fishing line. To avoid ongoing decomposition, all prepared litter bags were kept frozen until incubation on 30 October 2017 in the Cold Lake study areas, and 2 November 2017 in the Peace River study areas.

To meet adequate repetition (Berg 2014), we prepared 20 litter bags of each vegetation group for each microform of the study areas. Exceptions were made for PR15 and PR5, where no living mosses were found and accordingly no bryophyte litter bags were laid out in these study areas. Thus, a total of 280 vascular litter bags and 240 bryophyte litter bags were incubated. All litter bags were rewetted with surface available water before incubation. To mimic the natural senescence process of vascular litter at the surface, we followed Straková et al. (2012) and placed vascular litter bags horizontally on top of the surface. A natural senescence of bryophytes was assumed to start belowground, following Graf (2008). Consequently, bryophyte litter bags were buried horizontally at 5 cm depth. Litter bags were retrieved after an incubation period of 692 days at the Peace River study site, and after 701 days at the Cold Lake study site. From an originally buried total of 520 litter bags, we were able to retrieve 382 litter bags; 27% of the bags were lost or had been damaged by wildlife. Bags were carefully cleaned of any ingrowing living vegetation (Appendix 3.3). The remaining biomass was dried for a minimum of 48 hours at 70 °C until constant weight.

3.4.6 Data analysis

The statistical analysis was done using R version 3.6.0 (R Core Team 2019). All figures were generated with the package '*ggplot2*' (Wickham et al. 2020).

Productivity

Productivity P (g m^{-2} yr^{-1}) was calculated as the plant organic matter (g) produced per unit area (m^2) over a one-year period (yr). To evaluate an effect of ST5 and WTL on the above-ground and below-ground biomass production and rates of productivity among study areas, we used one-way analysis of variance (ANOVA) with multiple variables and a post-hoc Tukey's HSD pairwise comparison. The level of significance was accepted at $p \leq 0.05$. Logarithmic transformation was applied to obtain normality. To confirm the ANOVA result's consistency despite the non-normal distribution of the data, a non-parametric Kruskal-Wallis test was performed. Linear regressions were then performed to investigate the effect of the environmental parameters WTL and ST5 on above-ground and below-ground biomass, using the 'lm' function for linear models with a confidence level of 95%.

Decomposition

Biomass loss is expressed with the decay rate constant k, which is usually a negative value since the decomposition continues to decrease the remaining mass (Berg 2014). The single exponential decay rate constant k (yr^{-1}) was calculated as in Eq. 4 (Jenny et al. 1949; Olson 1963; Brinson et al. 1981; Wieder & Lang 1982; Berg 2014), where M_R and M_O represent the remaining and original mass respectively, during a specific time t (in years):

$$-k = \frac{\ln(M_R/M_O)}{t} \qquad \text{[Eq. 4]}$$

Furthermore, we calculated the percent mass remaining (MR) in a single exponential model as in Eq. 5 (Wieder & Lang 1982; Szumigalski & Bayley 1996; Thormann & Bayley 1997b):

$$MR = 100 - \frac{M_O - M_R}{X_O} \times 100. \qquad \text{[Eq. 5]}$$

Weight gain instead of loss has been observed in 13 decomposition bags after the two-year incubation period and omitted before statistical analysis to avoid data error. Non-normal data were transformed logarithmically. To explore the effect of ST5 and WTL on the decomposition, we conducted a one-way ANOVA with multiple variables and a 95% confidence interval. The results of the ANOVA were confirmed with a non-parametric Kruskal-Wallis test. Linear regressions were used to visualize the effect of the environmental parameters WTL and ST5 the decay rate *k*, using the linear models function 'lm'.

Peat accumulation potential

The rate of potential peat accumulation (g/m^2) in each study area was calculated in two different ways: First we calculated the quotient of the annual total biomass production ($g\ m^{-2}\ yr^{-1}$) and the total mass loss ($g\ m^{-2}\ yr^{-1}$) according to Thormann and colleagues (1999), then secondly we calculated the asymptotic peat accumulation limit (ρ_a/α_a in g/m^2) according to the acrotelm model by Clymo (1984), considering organic matter production ($g\ m^{-2}\ yr^{-1}$) and decay rate k (yr^{-1}). Results from previously reported measurements of net ecosystem exchange and respective C storage or release in the same study areas were used for comparisons (Lemmer et al. 2020).

3.5 Results

3.5.1 Biomass and productivity

Highest mean sum of above-ground (AG) and below-ground (BG) biomass productivity in the restored areas was observed in PR0E (627 $g\ m^2\ yr^{-1}$) and PR0D (516 $g\ m^2\ yr^{-1}$; Table 3.2). This is similar to the total biomass production seen in the dry microform of the undisturbed shrubby rich fen (SRF-D: 542 $g\ m^2\ yr^{-1}$; Table 3.2), where the highest productivity of the reference areas was observed. Soil temperature at 5 cm depth (ST5) was found a significant driver (positive relation) on the above-ground biomass productivity ($F_{3,294}$=4.93, p<0.01, adj. r^2=0.04; Appendix 3.4A). Some study areas, especially PR5, PR0E, CR-D, showed substantial differences in the total above-ground biomass productivity for the two individual years of collection, 2017 and 2019 (Figure 3.3). At the same time, slight changes in the weather have been observed, where mean minimum and maximum temperatures were observed 1 to 2 °C colder in 2019, but precipitation remained comparable during both years (Table 3.1). Study areas that had less biomass production in 2019 versus 2017 had a dominant bryophyte cover, while study areas with more biomass production in 2019 as compared to 2017 were dominated by herbaceous and graminoid vegetation with little or no bryophyte cover (Chapter 1). Furthermore, large differences occur in the biomass production which are related to spatial variability within each study area, especially within the restored areas, due to selection of different measurement plots in each study year.

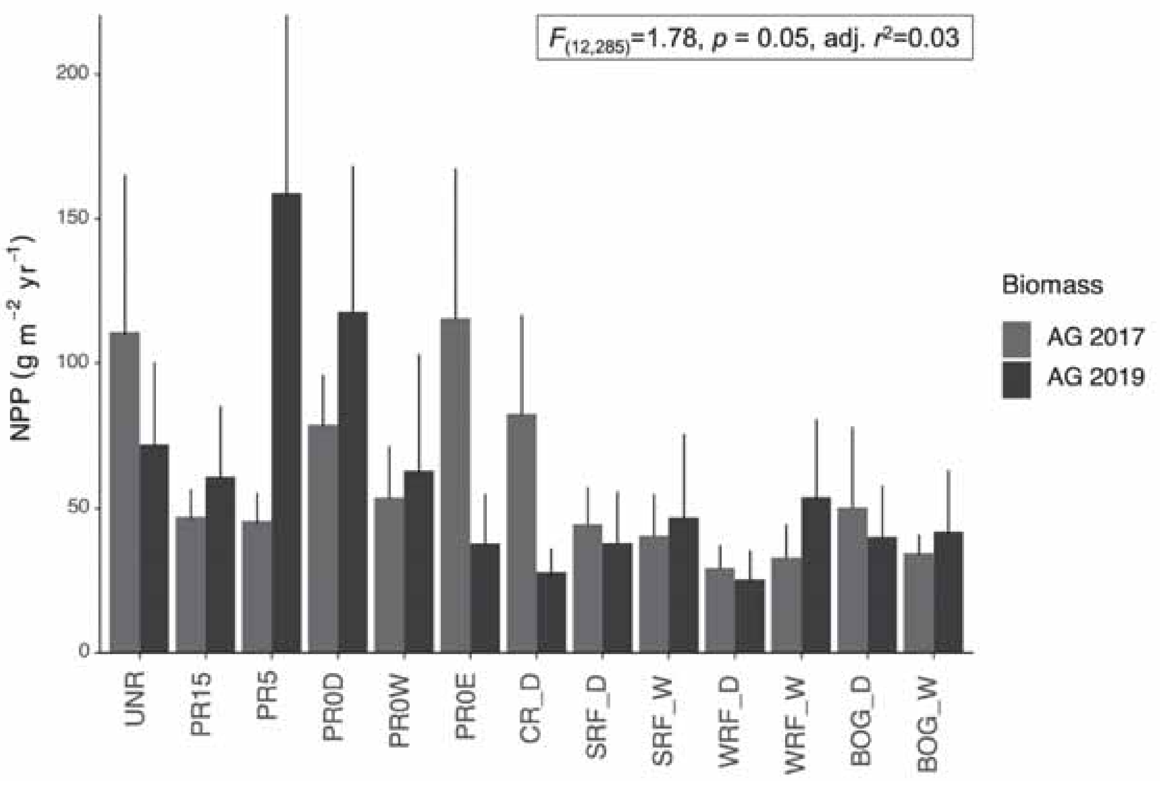


Figure 3.3 Differences between the two study seasons of mean above-ground (AG) biomass collected in each microsite (D=dry; W=wet; E=even) of all study areas (UNR=unrestored; PR15=partial removal (PR) to 15 cm above the average water table level (WTL); PR5=PR to 5 cm above the WTL; PR0=PR to (0 cm) the WTL with microforms D=dry/W=wet/E=even without microforms; CR=complete removal D=dry/floating moss carpet; SRF=shrubby rich fen with microform D=dry/W=wet, TRF=treed rich fen with microform D=dry/W=wet; BOG=wooded bog with microform D=dry/W=wet).

Table 3.2 Mean ±SD estimates for net primary productivity (NPP in g m^2 yr^{-1}) of above-ground (AG) biomass by plant functional types, and below-ground (BG) root biomass, and the overall total biomass. AG biomass was collected in August of 2017 and September 2019. Root ingrowth bags were incubated from October 2017 to September 2019. Study area codes as in Figure 3.3.

	Shrubs	n	Ericaceous	n	Herbs	n	Sedges	n	Brown mosses	n	Peat mosses	n	Lichen	n	Roots	n	Total
UNR	1 ± 1	4	n.a.		222 ± 143	6	88	1	25 ± 15	6	n.a.		n.a.		164 ± 56	3	499
PR15	9 ± 13	7	n.a.		71 ± 51	9	83 ± 46	7	16 ± 14	3	n.a.		n.a.		141 ± 112	6	321
PR5	5 ± 7	6	n.a.		77 ± 86	7	125 ± 86	8	2	1	n.a.		n.a.		205 ± 129	6	414
PR0-D	6 ± 12	4	n.a.		146 ± 90	4	167 ± 38	5	7 ± 7	3	70	1	n.a.		110 ± 7	3	516
PR0-W	n.a.		n.a.		59 ± 64	6	73 ± 48	3	2	1	n.a.		n.a.		38	1	172
PR0-E	5 ± 6	4	n.a.		172 ± 197	5	99 ± 57	4	24 ± 15	6	n.a.		n.a.		328 ± 280	3	627
CR-D	2 ± 1	3	<1	1	90 ± 139	6	76 ± 77	6	43 ± 9	5	n.a.		<1	1	37 ± 34	3	248
SRF-D	41 ± 34	6	4 ± 2	6	41 ± 35	3	97 ± 97	6	46 ± 30	4	n.a.		<1 ± <1	3	314 ± 517	3	542
SRF-W	7 ± 2	4	1 ± 2	5	54 ± 45	2	129 ± 77	5	14 ± 20	3	n.a.		n.a.		80 ± 74	2	284
TRF-D	19 ± 31	5	4 ± 4	5	58 ± 52	6	14 ± 10	5	38 ± 41	5	71 ± 6	3	1 ± 1	5	42 ± 46	3	246
TRF-W	1 ± 2	3	1 ± 1	2	92 ± 58	6	17 ± 12	3	37 ± 32	4	39	1	<1 ± <1	2	4 ± 3	2	192
BOG-D	14 ± 15	6	20 ± 15	5	3 ± 2	4	8 ± 4	4	9 ± 11	2	227 ± 126	5	11 ± 23	6	65 ± 53	3	357
BOG-W	11 ± 11	4	16 ± 18	4	168 ± 226	2	18 ± 18	4	45 ± 26	6	23 ± 32	5	47 ± 22	6	24 ± 24	2	351

Of the plant functional types, herbs and sedges were observed to contribute the most to the organic mass production in the study areas (Figure 3.4). This was confirmed through the highest overall NPP rates estimated in PR0D, PR0E, SRF-D, which were all dominated by fen characteristic plant species. In previous studies, the vegetation composition in the same study areas has been observed to be characterized especially by a high cover percentage of herbs like *Equisetum* sp. L. and *Menyanthes trifoliata* L., and sedges like *Carex aquatilis* Wahlenberg, *C. diandra* Schrank, and *Eleocharis palustris* (Linnaeus) Roemer & Schultes (Chapter 1).

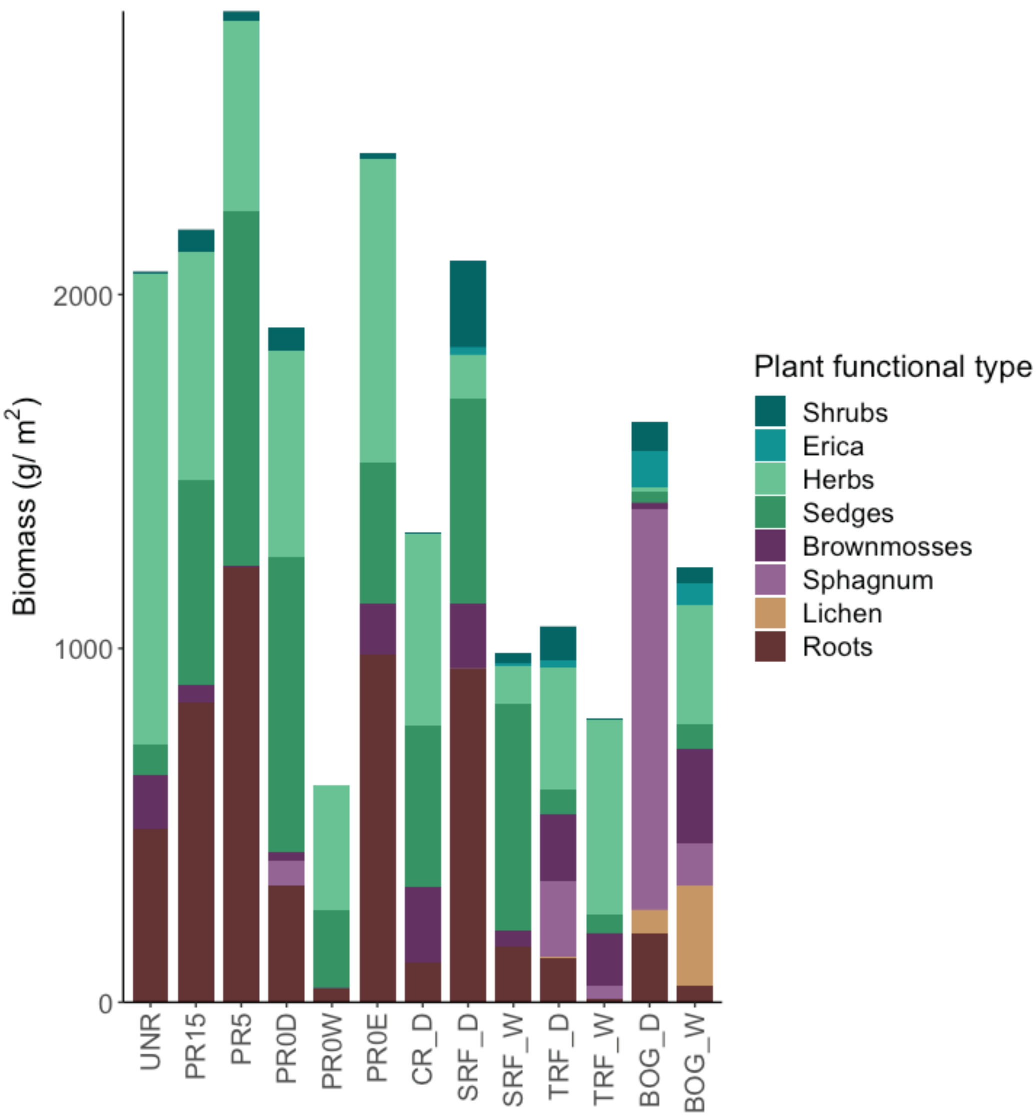


Figure 3.4 Contribution to the total biomass sorted by plant functional type (above-ground biomass) and roots (below-ground biomass) over a two-year study period. Characteristic plant groups of the respective plant functional types were specifically noted, such as ericaceous shrubs (shrubs), sedges (herbs), Sphagnum sp. (mosses). Codes for study areas as in Figure 3.3.

The same study areas (PR0E, SRF-D, PR5) with the highest above-ground productivity in their respective group (restored/reference) exhibited the highest root production below-ground (Figure 3.4 & Figure 3.5). The root productivity in restored areas was approximately 51% of the total biomass production, while in REF roots made up only about 23% of the total biomass produced (Figure 3.5). ST5 was found a significant driver on the below-ground biomass production ($F_{3,36}$=4.04, p=0.01, adj. r^2=0.19; Appendix 3.4A), where a positive relation was observed between temperature and biomass production. Although WTL is not a significant driver explaining below-ground biomass production, we noted the lowest root productivity among the restored peatlands in the inundated study areas PR0W (38 g m^2 yr^{-1}) and CR-D (37±34 g m^2 yr^{-1}) and likewise in the wet microforms of the reference wetlands SRF-W (80±74 g m^2 yr^{-1}), TRF-W (4±3 g m^2 yr^{-1}), BOG-W (24±24 g m^2 yr^{-1}).

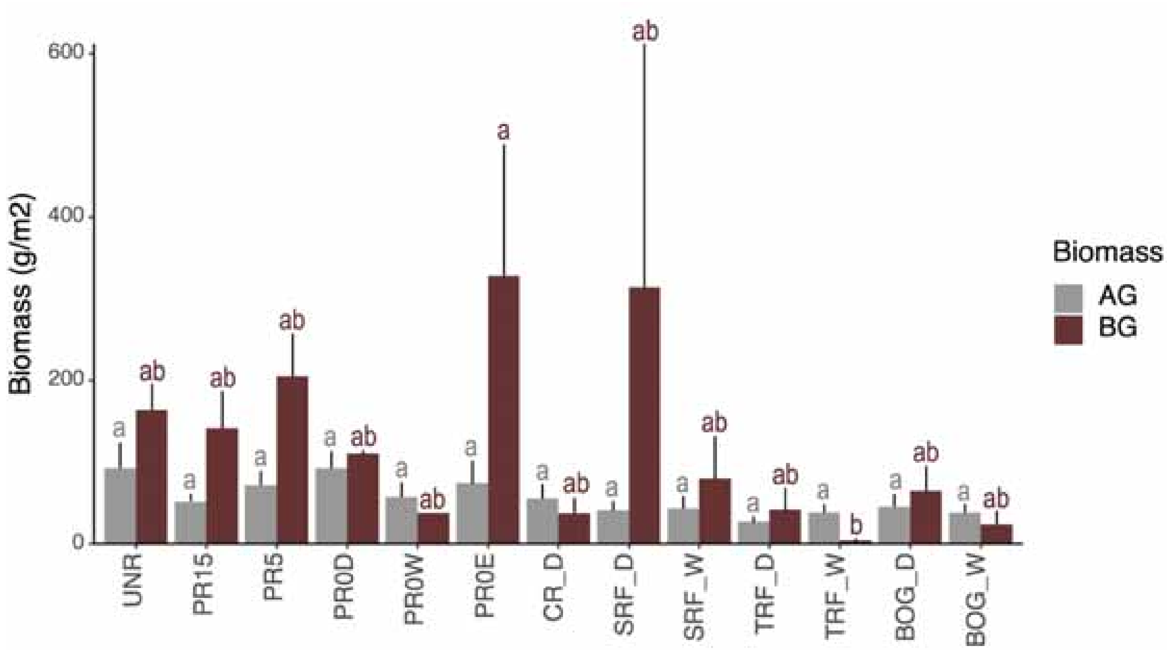


Figure 3.5 Mean organic plant matter produced above-ground (AG: $F_{12,285}$=1.78, p=0.05, adj. r^2=0.03) and below-ground (BG: $F_{12,27}$=2.27, p=0.04, adj. r^2=0.28) in the study areas during a two-year study period. Groups with the same letters are not significantly different. Study area codes as in Figure 3.3.

3.5.2 Decomposition

After about two years of incubation, the remaining biomass of bryophytes was in general higher than the remaining mass of vascular plants (

Figure 3.6). At the same time, less organic matter remained in restored areas than in REF. A decay gradient became visible among the microforms, where larger *k*-values were observed in the wet microforms. In restored areas, the decay rate (*k*) of vascular plants ranges from -1.26/yr in CR-D to -0.48/yr in PR15, compared to the slower decay rate of bryophytes ranging from -0.14/yr in PR0E to -0.26/yr in PR0D (Table 3.2). For comparison, *k*-values in REF ranged from as low as -0.04/yr for bryophyte organic matter decomposition in the wet microforms of the treed rich fen (TRF-D) and the treed bog (BOG-D), to -0.93/yr for vascular plant decomposition in the wet microform of the shrubby rich fen (SRF-W). ST5 was found to significantly affect the decay rate *k* ($F_{1,364}$=18.6, p<0.01, adj. r^2<0.01; Appendix 3.5A). The higher the soil temperature, the higher was the decomposition rate.

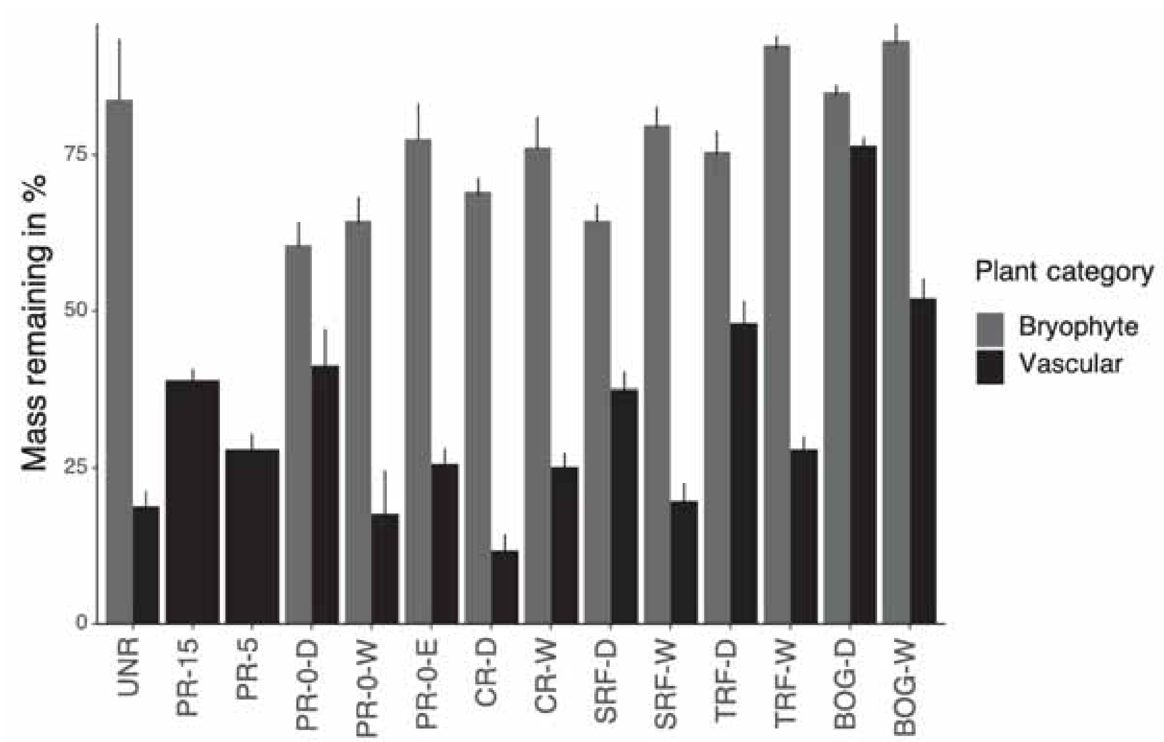


Figure 3.6 Remaining mass (in %) of bryophyte and vascular plant litter in each microsite of the restoration treatments, the unrestored and reference peatlands, after a two-year incubation period. No bryophytes were present at PR15 and PR5 in the begin of the study and thus not incubated in those locations. Study area codes as in Figure 3.3.

Table 3.3 *Overview of primary productivity and decomposition estimates by plant category (above-ground vascular and moss biomass) observed in the study areas. Bryophyte decomposition was not measured in PR15 and PR5 due to absence of mosses in 2017. Therefore, the bryophyte production observed during the 2019 biomass collection was neglected for the calculation of the peat accumulation potential (otherwise shown in parentheses are the skewed values that apply if bryophyte production is considered). Peat accumulation potential shown as production-decomposition-quotient (Thormann et al. 1999) and asymptotic limit according to the acrotelm model (Clymo 1984). Study area codes as in Figure 3.3.*

Study area	Vascular plants				Bryophytes				Peat accumulation potential	
	Biomass production ($g\ m^{-2}\ yr^{-1}$)	Exponential decay coefficient *k* (yr^{-1})	Mass lost ($g\ m^{-2}\ yr^{-1}$)	Mass remaining (%)	Biomass production ($g\ m^{-2}\ yr^{-1}$)	Exponential decay coefficient *k* (yr^{-1})	Mass lost ($g\ m^{-2}\ yr^{-1}$)	Mass remaining (%)	Production-decomposition quotient	Asymptotic limit ρ_a/α_a (g/m^2)
UNR	170	-0.94	-160	19	168	-0.14	-23	84	1.8	359
PR15	114	-0.48	-55	39	(81)	n.a.	n.a.	n.a.	(3.5) 2.1	(402) 234
PR5	127	-0.68	-86	28	0	n.a.	n.a.	n.a.	1.5	187
PR0D	217	-0.49	-106	41	28	-0.26	-7	61	2.2	500
PR0W	66	-1.15	-76	18	10	-0.22	-2	64	1	66
PR0E	154	-0.72	-111	26	79	-0.14	-11	78	1.9	468
CRD	94	-1.26	-118	12	177	-0.19	-33	69	1.8	1 421
CRW	217	-0.73	-158	25	28	-0.15	-4	76	1.5	336
SRF-D	262	-0.53	-139	38	185	-0.22	-41	64	2.5	1 063
SRF-W	129	-0.93	-120	20	63	-0.12	-7	80	1.5	205
TRF-D	118	-0.39	-46	48	415	-0.15	-62	76	5	3 553
TRF-W	65	-0.65	-42	28	267	-0.04	-11	93	6.3	510
BOG-D	137	-0.14	-19	77	933	-0.08	-76	85	11.4	7 639
BOG-W	188	-0.34	-64	52	255	-0.04	-10	93	6	2 282

3.5.3 Peat accumulation potential

The peat accumulation potential according to the production-decomposition-quotient (PDQ) is on average 1.9, indicating plant organic mass production exceeds biomass decay. The asymptotic limit ρ_a/α_a in restored peatlands is not always congruent with the PDQ. The lowest peat accumulation potential was observed in PR0W (PDQ 1, ρ_a/α_a 66 g/m^2) and the highest in PR0D (PDQ 2.2, ρ_a/α_a 500 g/m^2) and in CR-D (PDQ 1.8, ρ_a/α_a 1 421 g/m^2; Table 3.3). In contrast, the mean peat accumulation in reference peatlands (REF) is PDQ 5.5, with the lowest quotient estimated for the wet microform of the shrubby rich fen (SRF-W, 1.5, ρ_a/α_a 374 g/m^2) and the highest quotient estimated for the dry microform of the ombrotrophic (BOG-D, 11.4, ρ_a/α_a 9 598 g/m^2). Restored peatlands with a peat accumulation potential larger than 2 were observed to have a considerable cover of shrubs (Chapter 1), in particular PR0D (PDQ 2.2 and ρ_a/α_a 682 g/m^2) and PR15 (PDQ 2.1 and ρ_a/α_a g/m^2). Restored peatlands without shrubby vegetation but likewise high peat accumulation potential larger than 1.5 were defined by a dominant bryophyte cover like CR-D and PR0E (PDQ 1.9 and ρ_a/α_a 468 g/m^2).

The variation in the peat accumulation potential observed in this study did not always align with measurements of growing season C exchange observed at the same study areas. The cumulative total carbon (C) balance for a two-year study period, as seen in Chapter 2 (Lemmer et al. 2020), identifies PR0D as the largest C sink among the restored areas (-310 g C m^2), followed by CR-D (-205 g C m^2), PR5 (average of -139 g C m^2) and PR0E (-32 g C m^2; Table 2.8). Negative values indicate C uptake by the ecozone, positive values indicate C release to the atmosphere. Despite the effective C storing shrub vegetation, PR15 developed as a C source (on average 45.5 g C m^2). In two restored peatlands with inundated conditions, we observed simultaneously the highest C emissions (421 g C m^2 in PR0W and 546 g C m^2 in CR-W; Table 2.8) and the lowest peat accumulation potential (peat accumulation potential quotient 1 in PR0W and 1.5 in CR-W; Table 3.3).

3.6 Discussion

The aim of this study was to assess peat accumulation potential, via the net primary productivity and decomposition, of restored peatlands following *in situ* oil extraction disturbances. In particular, we considered if the mineral fill removal technique and the method of revegetation was of importance to return the peat accumulation function to the restored peatlands. Results show higher peat accumulation and C sequestration potential in restored areas that are defined by cooler soil conditions, as observed in restored peatlands with a water table level (WTL) at the surface. Such conditions were found in the restored peatlands that were treated with a partial removal of the mineral fill to near the WTL. The complete removal resulted in a shallow open water area with mostly floating aquatic vegetation where *Typha latifolia* L.-dominated communities developed in the margins of the shallow open water area (CR-W). Results generated for the floating moss carpet (CR-D) are not representative for the whole restored area but add interesting data on floating peatland bryophyte lawns. Many incubated samples were lost in the shallow open water area (CR-W) and data for this area could not be generated and integrated in analysis. However, the marsh-like conditions allow for comparison with other studies. The active revegetation via the planting of peatland vegetation did not prove to be necessary to return a peat accumulation and carbon sequestration function to restored peatlands. On the other hand, the introduction of shrubs can accelerate the C uptake and positively contribute to peat accumulation.

Even though we observed net primary productivity (NPP) and decomposition significantly affected only by soil temperature at 5 cm depth (ST5) and not by WTL, ST5 and WTL are known to be interdependent environmental factors in peatland ecosystems (Nichols 1998). A high WTL in peatlands is responsible for steady, cool temperatures, while a WTL drawdown is known to result in warmer soils (Weiss et al. 2006). Other research shows that with soil warming, root production increases, in particular for shrubby species (Mäkiranta et al. 2018; Malhotra et al. 2020). The same effect has been observed in our study, where higher mean root production of 143 g m^{-2} yr^{-1} of understory plant functional types was noted in the restored peatland's residual mineral soil, compared to the same plant functional type's mean root production of 88 g m^{-2} yr^{-1} in peat of REF. The contribution of decomposing roots to peat accumulation has not been considered in this study, which was observed to have significant contribution to peat accumulation below the ground (Scheffer & Aerts 2000; Graf & Rochefort 2009).

Conversely, a significant positive effect of WTL drawdown on root NPP has been observed in previous studies (Murphy & Moore 2010; Malhotra et al. 2020), which we could not find in our study. This study however is limited in the interpretation of the effect of environmental factors on NPP, since WTL and ST5 have been measured throughout the vegetation periods of 2017 and 2018, while the biomass was collected in 2017 and 2019 only. Although the weather conditions of 2018 and 2019 growing seasons are comparable in terms of temperature and precipitation, the results might yet be skewed. However, we noted a trend of lower below-ground biomass productivity within the wet microforms of REF (SRF-W, TRF-W, BOG-W), compared to the dry microforms, and likewise in the extremely wet, inundated restored microforms (PR0W and CR-D). Our findings support the fact that biomass production decreases with water table levels high above the surface, as the highest below-ground productivity was observed in the restored peatlands with a steady the WTL at the surface, as seen in PR0E and in PR5. The same restored peatlands, where partial removal of the former *in situ* oil sands well pad was done, supported highest cover of herbaceous communities (Chapter 1), in particular sedge vegetation communities dominated by *C. aquatilis* Wahlenberg and *T. latifolia* L., as was observed in other studies (Finér & Laine 2000; Weltzin et al. 2000).

A shortcoming of this study in this context is the neglected tree biomass productivity in the REF for completeness. Eight to 10 years post restoration, large trees have yet to develop in the restored peatlands, and the few growing tree species (*L. laricina* (Du Roi) K. Koch and *Picea mariana* (Miller) Britton, Sterns & Poggenburgh) that had grown to shrub-size were integrated with the shrub layer. Values provided in this study for tree biomass productivity in the respective REF (the treed rich fen "TRF" and the treed bog "BOG") therefore need to be considered with caution and are compared to other published literature. Campbell and colleagues (2000) and Vitt and colleagues (2001) provide values for general tree NPP observed in continental treed fens (44 g m^{-2} yr^{-1}) and continental treed bogs (27 to 106 g m^{-2} yr^{-1}). Above-ground NPP in particular for *P. mariana* (Miller) Britton, Sterns & Poggenburgh ranges from 27 to 310 g m^{-2} yr^{-1} in continental non-permafrost bogs (Grigal et al. 1985; Thormann 1995), and 6 g m^{-2} yr^{-1} in a continental moderate rich fen (Szumigalski 1995). Above-ground NPP in particular for *L. laricina* (Du Roi) K. Koch was noted at 38 g m^{-2} yr^{-1} in a continental moderate rich fen (Szumigalski & Bayley 1997). These values may provide an indication of the ecological distance between restored and reference areas, until the restored areas have transformed in the targeted fen ecosystem with fully developed vegetation. At this moment, the comparison of the restored areas to the reference areas excluding the tree biomass but including all shrub-sized tree and shrub species should suffice.

In our study, we observe generally high mass loss in all study areas, without WTL drawdown. Decomposition rates with an annual mass loss of up to 40%, have been observed to increase due to WTL drawdown enhancing soil temperature, aeration, and enzyme activities due to a shift from nitrogen (N) and phosphorus (P) acquisition to C uptake (Straková et al. 2011; Dieleman et al. 2016). In our study, in the undisturbed REF, mass loss ranged from an average 18% for bryophyte biomass to 56% for vascular plant matter. In contrast, in the restored peatlands, we found an average mass loss of 73% for vascular plants and more than 30% for bryophytes. Highest mass loss of 88% of vascular plant matter was found in the restored CR-D where the former well pad was completely removed, and a floating moss carpet developed. Great variation of short-term decomposition rates among sites during the first 100 days of decomposition have been observed (Thormann & Bayley 1997b), but the same study shows that long-term decomposition rates over 456 days are rather constant, which gives us confidence for our results reported for a two-year decomposition period.

Easily decomposable vascular plant litter was perceived to support higher enzyme activities, in contrast to recalcitrant bryophyte, especially *Sphagnum* spp. L., litter which is considered to contribute significantly to peat accumulation (Turetsky et al. 2000; Vitt et al. 2001; Straková et al. 2020). De Long and colleagues (2016) found mosses to drive decomposition, especially beneath the moss layer where the hydrological regime and moisture retention offer perfect conditions for decomposers. Litter which decomposes on top of the moss layer was found to dry faster and inhibit decomposition (De Long et al. 2016). Although we incubated the decomposition bags at the surface, imitating decaying above-ground biomass that falls to the ground (Straková et al. 2012), we observed the same trend in the moss dominated restored peatlands CR-D and PR0E, noting the highest decay coefficients k of $-1.26\ yr^{-1}$ and $-0.72\ yr^{-1}$ with a mass loss of 88% and 74%, respectively.

The phenomenon of some decomposition bags gaining weight instead of losing mass has been observed in other studies, which assumed C and N depositions as a result of growing microflora in decomposing plant tissue, as well as increased presence of decomposers to be the reason for the weight gain (Thormann & Bayley 1997b; De Long et al. 2016). It is known that decomposition continues throughout all layers of the peat profile, although at different rates. Above-ground plant organic matter naturally begins the decomposition cycle at the surface of the peatland and over time, will continue its way into deeper layers of the peat body.

There is little research on decomposition rates in rewetted residual mineral fill. Further studies comparing the decomposition rates at different depths and under different surface vegetation in restored peatlands following *in situ* well pad disturbances could help to gain more insight in the peat accumulation dynamics of these early successional peatlands.

3.7 Conclusion

This study indicates that peatland restoration on residual mineral fill of *in situ* oil sands well pads is on the trajectory to return peat accumulation and carbon sequestration functions comparable to reference peatlands (REF). In fact, in this study, the partial removal of the mineral fill has been more effective for supporting peat accumulation and carbon sequestration than the complete removal of the entire well pad construction materials. Decay coefficients were generally higher in restored peatlands than in REF. Because the peat accumulation depends to a large part on the dominant plant species composition, we observe the importance of restoration work to provide suitable hydrological conditions for the development of peatland characteristic plant species, especially bryophytes. More research and especially the implementation of large-scale studies (i.e., operational well pad scale) about peatland restoration following the partial and complete removal of *in situ* well pad is needed to increase the success and efficiency of the peatland restoration work done in the oil sands regions.

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3.9 Appendices

Appendix 3.1 List of plant species enclosed in biomass fill used for the litter bags. Nomenclature and taxonomy follow CNALH (2021) and ITIS (2021).

Life form	Latin name	UNR	PR15	PR5	PR0		PR0E	CR		SRF		TRF		BOG	
					D	W		D	W	D	W	D	W	D	W
Tree	*Larix laricina*		×								×			×	
Shrub	*Betula pumila* var. *glandulifera*									×	×	×			
Shrub	*Salix bebbiana*									×					
Shrub	*Salix brachycarpa*									×					
Shrub	*Salix exigua*							×							
Shrub	*Salix glauca*														×
Shrub	*Salix lasiandra*			×	×		×	×							
Shrub	*Salix lucida*														×
Shrub	*Salix lutea*							×							
Shrub	*Salix pedicellaris*				×						×				
Shrub	*Salix petiolaris*			×				×							
Shrub	*Salix planifolia*			×								×			
Shrub	*Salix pyrifolia*	×													
Ericaceous	*Andromeda polifolia*										×	×	×		
Ericaceous	*Chamaedaphne calyculata*													×	×
Ericaceous	*Empetrum nigrum*													×	×
Ericaceous	*Rhododendron groenlandicum*													×	×
Ericaceous	*Vaccinium vitis-idaea*											×		×	
Herb	*Achillea millefolium*	×													
Herb	*Agrostis scabra*									×					
Herb	*Symphyotrichum lanceolatum* ssp. *hesperium*	×					×								
Herb	*Calamagrostis canadensis*		×	×											
Herb	*Calamagrostis stricta* ssp. *inexpansa*											×			
Herb	*Caltha palustris*						×								
Herb	*Ceratophyllum demersum*					×									
Herb	*Cicuta bulbifera*						×								
Herb	*Comarum palustre*									×			×		
Herb	*Deschampsia cespitosa*	×													

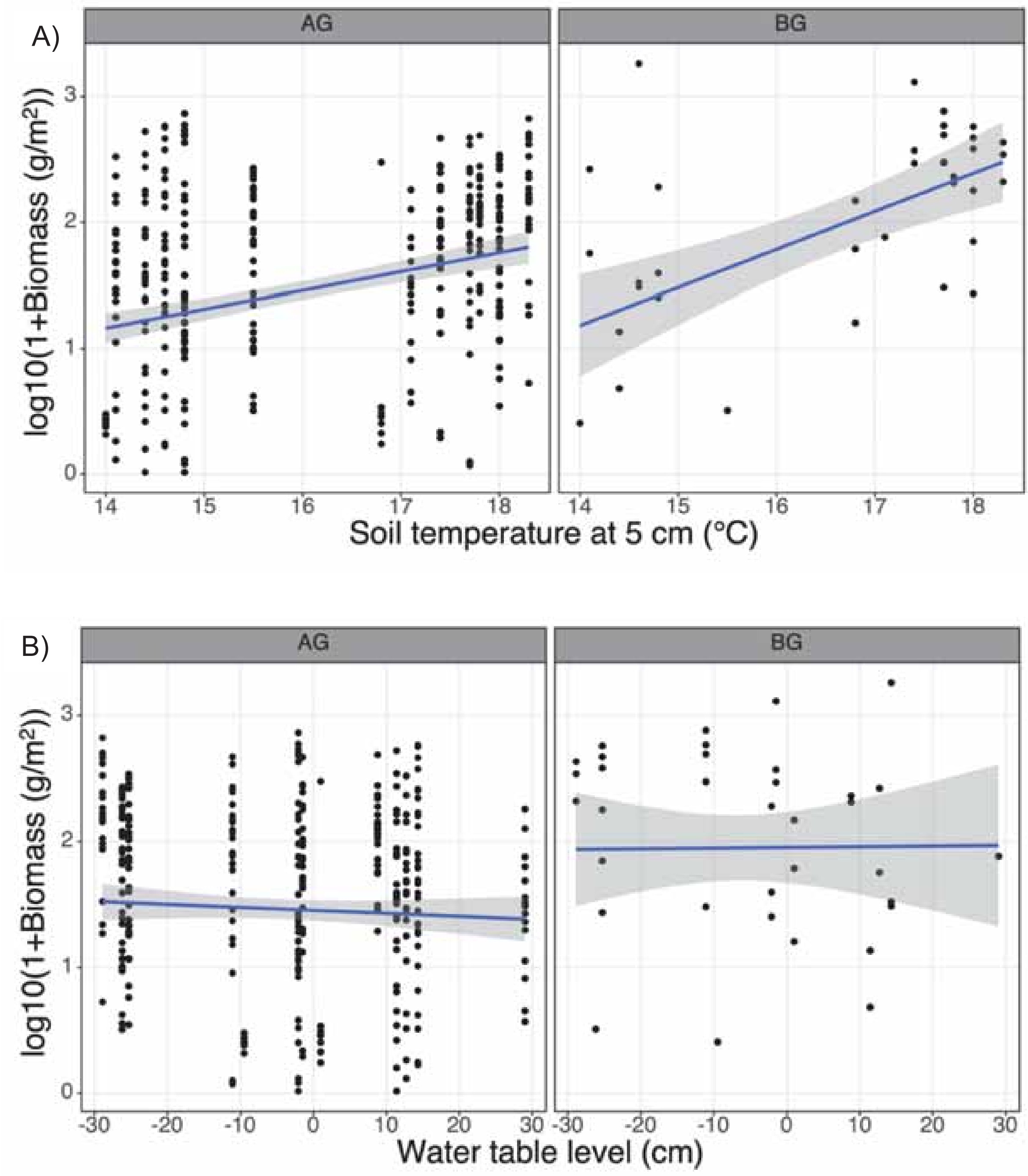
Life form	Latin name	UNR	PR15	PR5	PR0		PR0E	CR		SRF		TRF		BOG	
					D	W		D	W	D	W	D	W	D	W
Herb	*Drosera rotundifolia*													×	
Herb	*Equisetum arvense*	×		×	×		×	×							
Herb	*Equisetum hyemale*	×				×	×	×		×	×	×	×		
Herb	*Galium labradoricum*												×		
Herb	*Galium trifidum*									×	×				
Herb	*Juncus nodosus*							×							
Herb	*Menyanthes trifoliata*							×		×	×		×		
Herb	*Plantago major*			×											
Herb	*Poa palustris*	×					×	×							
Herb	*Poa pratensis*	×													
Herb	*Rubus chamaemorus*													×	
Herb	*Rumex occidentalis*				×										
Herb	*Spiranthes romanzoffiana*		×												
Herb	*Stellaria longifolia*							×				×			
Herb	*Trifolium hybridum*	×	×	×											
Herb	*Triglochin maritima*				×					×	×				
Herb	*Typha latifolia*				×	×	×	×							
Sedge	*Carex aquatilis*		×	×				×							
Sedge	*Carex chordorrhiza*									×					
Sedge	*Carex diandra*									×	×	×	×		
Sedge	*Carex disperma*							×				×	×		×
Sedge	*Eleocharis palustris*				×	×		×							
Sedge	*Eriophorum angustifolium*							×				×	×		×
Aquatic	*Lemna minor*					×									
Aquatic	*Potamogeton gramineus*					×							×		
Brown moss	*Aulacomnium palustre*									×		×			
Brown moss	*Calliergon* sp.							×							
Brown moss	*Calliergonella cuspidata*											×			
Brown moss	*Dicranum undulatum*											×			
Brown moss	*Drepanocladus aduncus*							×							
Brown moss	*Hamatocaulis vernicosus*							×							
Brown moss	*Hylocomium splendens*											×			

Life form	Latin name	UNR	PR15	PR5	PR0		PR0E	CR		SRF		TRF		BOG	
					D	W		D	W	D	W	D	W	D	W
Brown moss	*Plagiomnium ellipticum*														
Brown moss	*Pleurozium schreberi*											×			
Brown moss	*Pohlia nutans*											×			
Brown moss	*Polytrichum strictum*													x	
Brown moss	*Ptilium crista-castrensis*									x		x	x		
Brown moss	*Tomentypnum falcifolium*														x
Brown moss	*Tomentypnum nitens*											x	x		
Peat moss	*Sphagnum fuscum*													x	
Peat moss	*Sphagnum warnstorfii*											x	x		
Lichen	*Cladonia rangiferina*													x	x
Lichen	*Hypogymnia physodes*												x		
Lichen	*Vulpicida pinastri*												x		

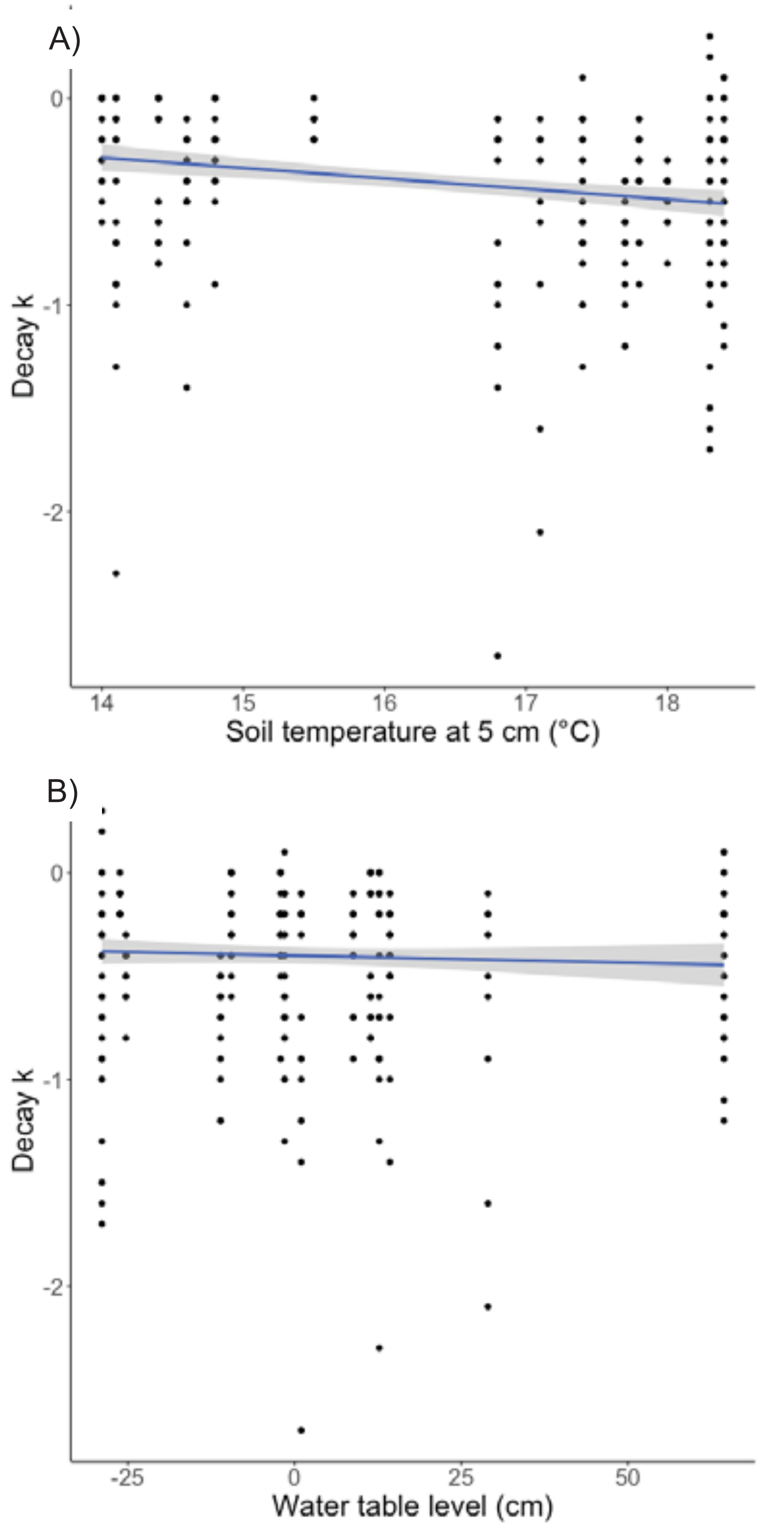
Appendix 3.2 Litter bags made of 1 mm fiberglass mesh material for vascular plants (left) and 1 mm nylon material for bryophytes (right) before filling.

Appendix 3.3 The litter biomass material in a retrieved bryophyte litter bag became the new base for living biomass on top, demonstrating the need for the retrieved bag to be cleaned before further analysis of the remaining mass inside the bag. Weight gain is possible due to plant ingrowth,

Appendix 3.4 Linear regressions visualizing A) the effect of soil temperature at 5 cm depth (ST5) on logarithmic transformed above-ground biomass (AG: $F_{1,296}$=12.7, p<0.01, adj. r^2=0.04) and below-ground biomass (BG: $F_{1,38}$=12.75, p<0.01, adj. r^2=0.23), as well as B) the effect of water table level relative to the surface (WTL) on above-ground and below-ground biomass (AG: $F_{1,296}$<0.01, p=0.96, adj. r^2<0; BG: $F_{1,38}$=2.28, p=0.14, adj. r^2=0.03).



Appendix 3.5 Linear regression visualizing the effects of A) soil temperature at 5 cm depth (ST5) ($F_{1,364}$=18.6, $p<0.01$, adj. $r^2<0.01$), and of B) water table level (WTL) relative to the surface ($F_{3,364}$=0.81, p=0.37, adj. $r^2<0$) on the decay rate k.



Appendix 3.6 Mean water table level (WTL) and soil temperature at 5 cm depth (ST5) for 2017 and 2018, as measured in 2018 for related research (Lemmer et al. 2020; Chapter 1).

Study area	ST5		WTL	
	2017	**2018**	**2017**	**2018**
UNR	18 ± 5	20±3	-29 ± 16	-28 ±13
PR15	16 ± 3	20±8	-36 ± 18	-13 ±8
PR5	16 ± 3	19+5	-23 ± 19	2± 5
PR0-D	17 ± 3	20+5	9 ± 5	9+ 4
PR0-W	15 ± 3	22+6	29 ± 5	29± 8
PR0-E	17 ± 5	18±5	-1 ± 5	-3± 4
CR-D	16 ± 5	18+5	0 ± 4	3± 5
SRF	17 ± 6	14+4	14 ± 12	14+ 5
SRF	14 ± 3	14+4	13 ± 9	13+ 5
TRF	16 ± 3	12+6	-1 ± 8	-5±7
TRF	15 ± 3	12+5	11 ± 7	11+8
BOG	14 ± 4	18±7	-36 ± 8	-17±5
BOG	12 ± 4	16±14	-18 ± 9	-1±4

Conclusion

This research is among the first to evaluate novel peatland restoration approaches following the *in situ* oil sands well pads disturbances in the Oil Sands regions of northern Alberta. The assessed peatland restoration had been done in the early 2000's and included the partial or complete removal of the decommissioned well pad's mineral soil. Since already thousands of inactive and abandoned *in situ* well pads are located in the fragmented boreal forest and thousands more will be constructed for ongoing bitumen extraction in the upcoming years, an increasing number of valuable Carbon storing peatlands will be disturbed. In this context, restoration is crucial to return necessary ecosystem functions, such as the ongoing peat accumulation and Carbon sequestration. The comprehensive results of this research will be discussed in the following sections. The principles of ecological restoration have been applied using different methods for peatland restoration to favor the recovery of disturbed peatlands towards their former ecosystem functions (Figure 0.9).

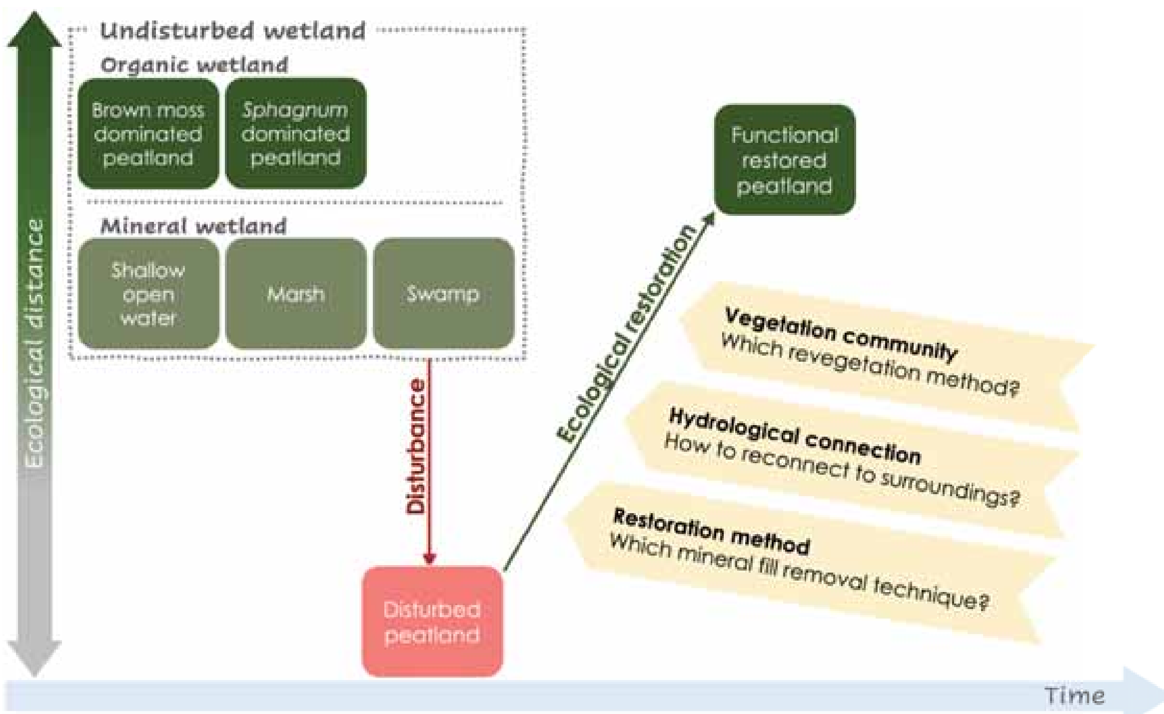


Figure 0.9 Concept of ecological peatland restoration, including focal points to be considered during an ecological pre-assessment and implemented in the following restoration management plan.

Chapter 1 compared the vegetation communities that developed in the restored peatlands. We observed the development of marsh-like vegetation in the shallow open water area that formed in the depression created due to compressed underlying peat following the complete removal of an *in situ* well pad. In contrast, we found emerging dominant fen characteristic species and vegetation communities in the near-surface rewetted areas where partial removal of the mineral material allowed the reestablishment of peatland-like hydrology.

In chapter 2 we evaluated the net ecosystem exchange, methane emissions, and global warming potential of the restored peatlands. Our study shows greenhouse gas emissions, especially methane (CH_4) emissions, are enhanced in the shallow open water area following the complete removal restoration technique. In the areas treated with the partial removal of the mineral fill that have a near-surface water table, carbon uptake from the atmosphere is comparable to reference peatlands.

In chapter 3 we assessed the peat accumulation potential via net primary productivity and decay of vascular and bryophyte litter. High peat accumulation potential was observed for a bryophyte carpet floating in the shallow open water area prompted by the complete removal technique and in the partial removal areas scraped down to the water table level revegetated with peatland characteristic plants, especially bryophytes and shrubs.

The extensive restoration via the complete removal of the disturbance was compared to the restricted method via the partial removal of an *in situ* well pad in combination with active and passive revegetation methods. In our study, the complete removal of a former well pad resulted in the development of a shallow open water area, with mostly floating aquatic vegetation and marginal marsh-like vegetation. However, marshes are known to remain mineral wetlands for a long time and accumulate peat only very slowly if at all (Rydin & Jeglum 2013; Mitsch & Gosselink 2015). In this case, the succession towards a peatland ecosystem within a relatively short time frame is not certain and the ecological restoration is not as effective as can be. In contrast, our study shows that the partial removal technique is effective for returning peatland characteristic vegetation, which is the base for carbon uptake and storage in the biomass that is accumulating and slowly decomposing to peat. The partial removal of the former well pad was the more successful the closer the mineral fill was scraped down and adjusted to the surface and the respective water table level of the adjacent undisturbed peatland ecosystem. The hydrological connection was observed to impact the soil temperature, while both environmental factors are known drivers of greenhouse gas emissions and vegetation development (Lloyd & Taylor 1994; Bubier et al. 2003).

In terms of peatland characteristic vegetation, we observed spontaneously developed vegetation composition to be more abundant and richer in peatland species, compared to the actively revegetated community composition. According to studies on natural species dispersal (Palmer et al. 1997; Campbell et al. 2003), we conclude that the active revegetation via planting of peatland characteristic sedges (*Carex aquatilis* Wahlenberg), shrubs (*Salix lutea* Nuttall) and trees (*Larix laricina* (Du Roi) K. Koch) is not necessary to return peatland characteristic vegetation, if the area to be restored is small in size (<1 ha) and species rich diaspore sources comparable to the target peatland ecosystem are nearby to allow for natural ingress. In particular, trees have been found to have difficulties developing in restored peatlands post oil and gas disturbances (Shunina 2015; Saraswati et al. 2020; Bork et al. 2021). In case of the restoration of treed peatlands, the planting of typical tree species may accelerate their development and the overall restoration success (Caners & Lieffers 2014; Dohong et al. 2018; Murray et al. 2021). An ecological assessment of the peatland prior disturbance will confirm whether the active tree species introduction should be included in the restoration management plan.

Our results about the inefficiency of the complete removal restoration techniques to return peatland characteristic vegetation and functions are in contrast to the study by Xu and colleagues (2021), who had a positive feedback of a complete removal restoration trial at a small scale. Among other tested restoration approaches, the complete removal did not develop a shallow open water area, and therefore, in combination with the applied moss layer transfer technique (Quinty & Rochefort 2003), presented an adequate technique to restore peatlands and characteristic *Sphagnum*-dominated vegetation following *in situ* oil sands well pad disturbances (Xu et al. 2021). A major difference of the two studies is the adjacent or surrounding type of peatland ecosystem and its respective hydrology. Xu and colleagues (2021) work was done within a treed bog complex, where the water table is normally below the surface and the peat substrate is expected to be less degraded and more robust allowing for active decompression and fluffing up to reinstate vertical and horizontal hydrological connectivity. Our study of the complete removal techniques took place in a rich fen ecosystem, which is characterized by a water table at or above the surface and a more strongly decomposed peat substrate. Once established, a shallow open water area is not certain to take the successional route to a peatland ecosystem (Volik et al. 2018; Ketcheson et al. 2016; Kreyling et al. 2021).

On the other hand, the research areas for this study treated with the partial removal of the mineral fill were located within a rich fen complex (PR0) and within a bog (PR15 and PR5) but showed very different vegetation development. The difference of revegetation success between the study of Xu and colleagues (2021) and our study also lies within the revegetation techniques applied. While in our study, three vascular fen species were actively and selectively introduced, while in contrast, the MLTT applied in the study of Xu et al. (2021) comprehensively transfers a wide variety of the upper ecological crust of an undisturbed reference peatland, including seeds and rhizomes of vascular plants, spores of bryophytes and lichens, enzymes, microbes, virus, and bacteria (Hugron et al. 2020).

Despite active introduction of three fen species (*C. aquatilis* Wahlenberg, *L. laricina* (Du Roi) K. Koch and *S. lutea* Nuttall; PR15 and PR5), the restored peatlands developed less diverse and less rich species communities being surrounded by a bog, compared to the peatland species rich vegetation communities in the restored peatlands located in within the rich fen complex (PR0). However, to support natural species dispersal and ingress, we believe the proximity of a suitable natural diaspore source for the targeted restored peatland ecosystem to be of utmost importance. We found the proximity to the water table level the most important factor contributing to effective peatland restoration and stress the importance of meticulous levelling of the residual mineral fill's surface. We again stress the importance for comprehensive ecological assessments of the individual peatlands prior to disturbance in order to develop suitable management plans and a greater success rate of ecological peatland restoration. We conclude, following the decision tree in (Figure 0.10), the ecological pre-assessment dictates whether to choose a complete or partial mineral fill removal of a former *in situ* well pad. The restoration target should reflect the initially disturbed peatland type, which is characterized by the hydrological conditions and it4s vegetation composition.

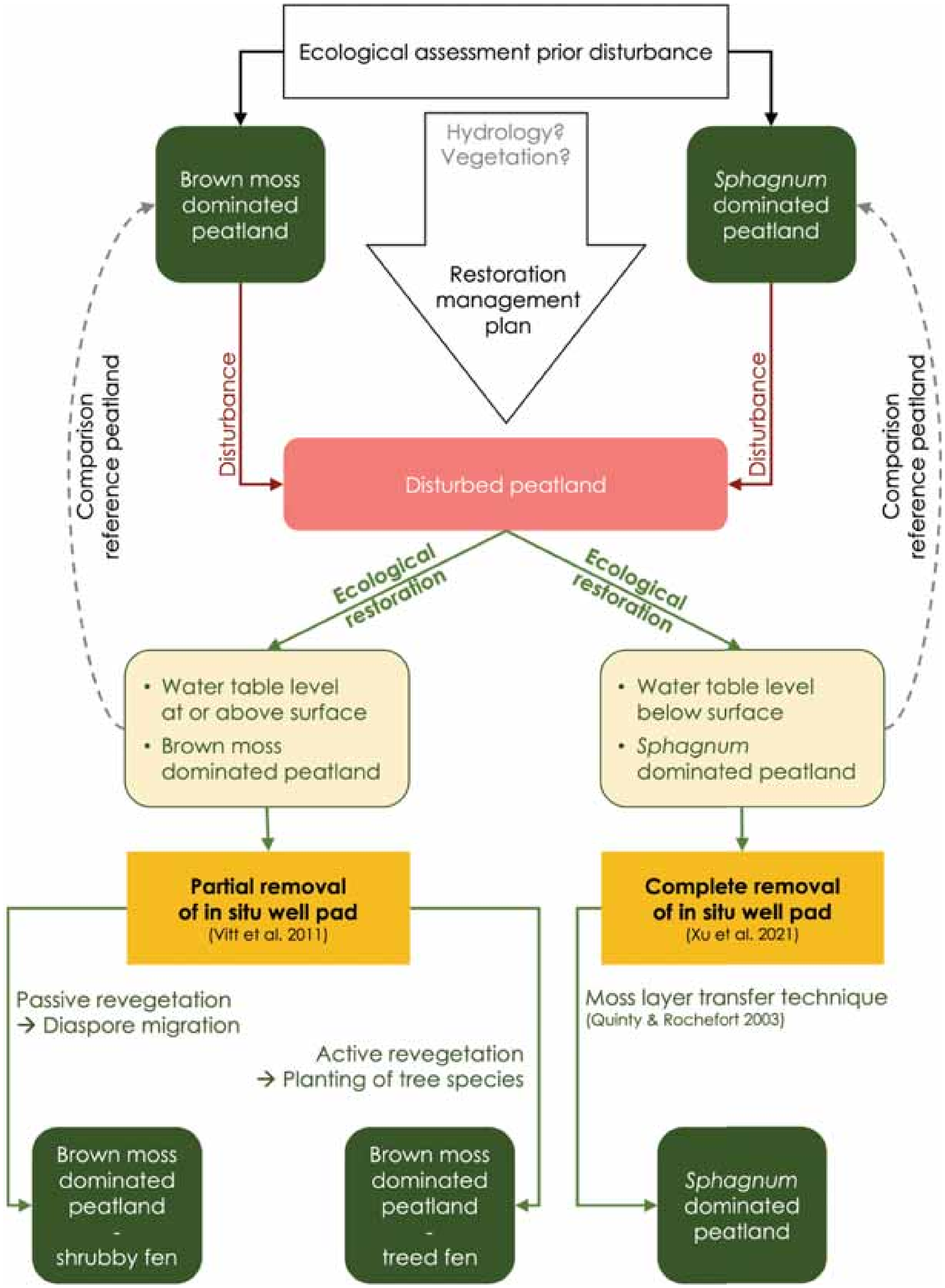


Figure 0.10 Decision tree for practitioners to decide on ecological restoration techniques, regarding methods for mineral fill removal and revegetation, for peatland restoration following in situ oil sands well pad disturbances.

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