

Development of Ecohydrological Processes on a Partially Removed Well Pad Undergoing
Restoration to a Peatland on the Western Boreal Plain, Alberta, Canada

by

Murdoch Edward McKinnon

A thesis

presented to the University of Waterloo

in fulfillment of the

thesis requirement for the degree of

Doctor of Philosophy

in

Geography (Water)

Waterloo, Ontario, Canada, 2025

© Murdoch Edward McKinnon 2025

EXAMINING COMMITTEE MEMBERSHIP

The following served on the Examining Committee for this thesis. The decision of the Examining Committee is by majority vote.

External Examiner

Dr. Peter Whittington

Professor, Geography and Environment
Brandon University

Supervisors

Dr. Richard Petrone

Professor, Geography & Environmental Management
University of Waterloo

Dr. Felix Nwaishi

Associate Professor, Earth & Environmental Sciences
Mount Royal University

Adjunct Professor, Geography & Environmental Management
University of Waterloo

Internal Member

Dr. Scott Ketcheson

Associate Professor & Canada Research Chair in Hydrological Sustainability
Athabasca University

Adjunct Professor, Geography & Environmental Management
University of Waterloo

Internal-external Member

Dr. Rebecca Rooney

Associate Professor, Biology
University of Waterloo

Other Members

Dr. Bin Xu

NSERC Industrial Research Chair for Colleges in Peatland Restoration
Northern Alberta Institute of Technology (Centre for Boreal Research)

AUTHOR'S DECLARATION

This thesis consists of material all of which I authored or co-authored: see Statement of Contributions included in the thesis. This is a true copy of the thesis, including any required final revisions, as accepted by my examiners.

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

STATEMENT OF CONTRIBUTIONS

This thesis has been prepared in accordance with the manuscript format. Chapters two and three have been published as journal articles as detailed below. As such, the versions of Chapters two and three presented herein are inclusive of revisions made in response to feedback provided by anonymous peer reviewers. Chapter four has been submitted to a peer-reviewed journal. As the submitted manuscript will be subject to peer-review and associated revision, the final published version of Chapter four may differ from that presented herein. Chapter five has been prepared for publication, but has not yet been submitted.

Chapter two is published as:

McKinnon, M. E., Nwaishi, F. C., Xu, B., Ketcheson, S. J., Bird, M., & Petrone, R. M. (2024).
Hydrological dynamics following partial removal of an oil well pad undergoing restoration
to a boreal peatland. *Science of the Total Environment*, 957, 177750.

M. McKinnon wrote the original draft of the manuscript, contributed substantially to the conceptualization of the study and associated methods, and conducted formal investigation and analyses. All co-authors contributed to study conceptualization and provided suggestions on analyses and interpretation. All co-authors also provided editorial suggestions.

Chapter three is published as:

McKinnon, M. E., Nwaishi, F. C., Xu, B., Ketcheson, S. J., Bird, M., & Petrone, R. M. (2025).
Hydrologic assessment of mineral substrate suitability for true moss initiation in a boreal
peatland undergoing restoration. *Ecological Engineering*, 215, 107615.

M. McKinnon wrote the original draft of the manuscript, contributed substantially to the conceptualization of the study and associated methods, and conducted formal investigation and

analyses. All co-authors contributed to study conceptualization and provided suggestions on analyses and interpretation. All co-authors also provided editorial suggestions.

Chapter four is submitted as:

McKinnon, M. E., Nwaishi, F. C., Xu, B., & Petrone, R. M. Development of biogeochemical processes on a residual in-situ oil well pad undergoing restoration to a boreal peatland. Submitted to *Wetlands Ecology and Management* (manuscript number: 6dac158c-bbd1-48e8-ae6e-644737ab7ccb).

M. McKinnon wrote the original draft of the manuscript, contributed substantially to the conceptualization of the study and associated methods, and conducted formal investigation and analyses. All co-authors contributed to study conceptualization and provided suggestions on analyses and interpretation. All co-authors also provided editorial suggestions.

Chapter five has been prepared (but not yet submitted) as:

McKinnon, M. E., Nwaishi, F. C., Xu, B., Ketcheson, S. J., & Petrone, R. M. A comparison of hydrologic fluxes in a partially removed well pad undergoing fen restoration on the Western Boreal Plain, Canada.

M. McKinnon wrote the original draft of the manuscript, contributed substantially to the conceptualization of the study and associated methods, and conducted formal investigation and analyses. All co-authors contributed to study conceptualization and provided suggestions on analyses and interpretation. All co-authors also provided editorial suggestions.

By signing below, I indicate that I agree with the evaluation of individual contributions to each chapter of the thesis as described above.

Dr. Richard Petrone

Dr. Felix Nwaishi

Dr. Bin Xu

Dr. Scott Ketcheson

Melanie Bird

ABSTRACT

Peatlands on the Western Boreal Plain have been disturbed at a landscape scale by industrial developments including those associated with the oil and gas industry. Among these disturbances are in-situ well pads, which are constructed to provide a stable base for oil and gas drilling and extraction infrastructure. In the province of Alberta, Canada, well pads must legally be returned to a state of ‘equivalent land capability’ after decommissioning. For well pads constructed in peatlands, equivalent land capability has recently been defined as including the reestablishment of a self-sustaining and peat accumulating vegetation community. One method proposed to reintroduce peatland vegetation (including peatland mosses) onto decommissioned well pads involves the partial removal of the mineral fill used to construct a well pad. Termed the ‘Partial Removal Technique,’ this approach aligns the reprofiled surface elevation of a pad with that of the water table in the surrounding peatland. Peatland vegetation propagules are then introduced onto the residual mineral substrate using a modified version of the established Moss Layer Transfer Technique. However, considerable uncertainty has remained surrounding the efficacy of the technique as a form of peatland restoration, as it had not yet been applied at the scale of a full-size well pad.

Accordingly, a five-year ecohydrological study was undertaken following the first full-scale implementation of the Partial Removal Technique on a well pad. The subject well pad was located in a fen complex on the Western Boreal Plain near the town of Slave Lake, Alberta, Canada. A series of field studies were undertaken to assess the extent to which the residual mineral substrate would support environmental conditions requisite for the initiation and establishment of a peatland vegetation community. Specific objectives addressed included characterization of the hydrophysical properties of the residual mineral fill and their effect on hydrological connectivity with an adjacent fen, and assessment of whether hydrological connectivity was sufficient to maintain a near-surface water table and optimal moisture availability to mosses across the entire site. The role of additional water balance terms in supporting near-surface water tables and water availability was also assessed, including quantification of snowmelt, vertical groundwater exchange, and evapotranspiration. Additionally, monitoring of the development of biogeochemical

processes in the first five years post-partial removal was undertaken, including quantification of the rates of nutrient cycling and supply. The effects of microtopography and application of straw mulch and rock phosphate fertilizer on moisture and nutrient dynamics were also assessed.

Results indicate that hydrological connectivity between the residual well pad and the adjacent fen was limited by the low hydraulic conductivity of the mineral fill and the compacted peat underlying it. Combined with rapid drainage from the mineral fill into the underlying peat following rainfall, this resulted in the water table being poorly regulated across just over half of the pad's surface area. The deeper water tables observed in those areas were associated with non-optimal moisture availability to mosses (i.e., exceedance of literature desiccation thresholds), particularly in the late growing season when rainfall inputs were infrequent. Combined with high rates of water loss through evapotranspiration, it appears that much of the pad's surface area is likely to be favourable for the establishment of only those mosses with a high desiccation tolerance. The establishment of a vegetation community characteristic of swamps may thus occur over the long term in areas that are hydrologically disconnected from the fen. Nonetheless, hydrological connectivity with the adjacent fen was sufficient to maintain a water table within 6 cm of the surface in areas located within approximately 20 to 30 metres of the upgradient pad edges. This water table depth was associated with optimal water supply at the surface for moss survival and growth. As such, the establishment of a peatland true moss community is likely to be supported across just under half of the pad's surface area. Snowmelt may also have provided a large source of water in the early season, although additional study is required to determine the extent to which snowmelt may be lost from the pad as overland flow. Surface runoff from an upland feature constructed out of the excess mineral fill produced during the partial removal process did not constitute an appreciable source of water to the pad.

Nutrient cycling and availability demonstrated limited spatial variability across the residual well pad. Owing to the high cation content of the calcareous residual mineral fill, cation supply rates were sufficiently high to further increase the likelihood of peatland true moss establishment in areas with optimal substrate moisture availability. However, low rates of nitrogen production and a low ratio of nitrogen to

phosphorus supply rates indicate that productivity of the vegetation community on the residual pad may be nitrogen limited. This may change over time, as a layer of organic litter was observed to accumulate on the surface of the residual well pad during the study. This is likely to result in increased rates of decomposition, and thus also of nutrient mineralization over time.

Combined, the results of this thesis indicate that there is a need to increase horizontal hydrological connectivity with adjacent peatlands in future implementations of the Partial Removal Technique. This may improve the availability of moisture across a greater proportion of the surface area of residual well pads, while also ensuring the long-term development of anaerobic biogeochemical processes. Additional work is also required to reduce water losses in the form of both vertical drainage from residual mineral substrates and evapotranspiration from the surfaces of residual well pads. Overall, the Partial Removal Technique appears to have promise as a strategy to create favourable environmental conditions for the initiation and establishment of peatland mosses on decommissioned well pads.

ACKNOWLEDGEMENTS

The support I have received from my co-supervisors Dr. Felix Nwaishi and Dr. Richard Petrone has been invaluable. Thank you for your contributions to my academic and professional development. Thank you, Rich, for your patience, understanding, and encouragement, and for always including me as part of the Hydrometeorology Research Group despite my being so far away. Thank you, Felix, for all of your personal guidance over long conversations, for reminding me to keep the big picture in mind, and for always believing in me and supporting my growth as an independent researcher. Thanks also to my committee members Dr. Bin Xu, Dr. Scott Ketcheson, and Dr. Rebecca Rooney. Your feedback and advice made this process enjoyable, and I have felt supported at every step along the way.

To all those who supported me in the field and the laboratory: Thank you for your feedback, encouragement, and camaraderie. Working with each of you, in both good weather and bad, made this process truly enjoyable. Thanks to Traelyn Rachar, Abisola Allison, Evan Bishko, and Taylor Vodopija, who helped to get the project off the ground and who were never afraid to ask tough questions or to propose better ways of doing things. Thanks to Kaitlynn Bartlett, Mannah Brillantes, Rachel Davey, Mwitwa Chisela, Kim Renters, Tyler Bray, Justin Kentel, and Mathew Gauthier, who all made the impossible happen in the field. Thanks also to Sarah Darling, Joshua Chow, Tracy Kwan, Victoria Rosales, Lucia Walters, Tanmay Darak, and Marius Möller, who each provided assistance in the field or lab at one time or another.

I am also truly grateful to the members of the Hydrometeorology Research Group at the University of Waterloo. Your recommendations and support, and conversations over beers at conferences, have undoubtedly made this thesis stronger. An especial thanks goes to Dr. Matthew Elmes, Tim Trembath, Dr. Brandon van Huizen, Dr. Yi (Abby) Wang, Dr. Nataša Popović, Adam Green, Eric Murray, Jessica Williamson, and Dr. Myroslava Khomik, who each provided especially important advice or support along the way. Thanks also to Dr. Miranda Hunter for making my time as a geomorphology TA so positive.

The Northern Alberta Institute of Technology's Centre for Boreal Research in Peace River provided invaluable personal and financial support during my PhD journey. Thanks to the whole team there, who

catalyzed my interest in peatland mosses and who entrusted me with an independent contract research project. Thanks especially to Melanie Bird for the long conversations in the field and on the road, and for sharing your excitement and passion with the whole MRU team. Additional thanks go to Dr. Ba Thuong Le for providing an immense amount of personal and professional advice, and for reminding me that we don't just work today.

The Department of Earth and Environmental Sciences at Mount Royal University also provided an immense amount of support, both personal and in terms of laboratory and office space. Thanks especially to Glen Roberts, Shane Steininger, Dr. Mathew Swallow, Dr. Kevin Hayes, and Dr. Katherine Boggs for your support, encouragement, and advice. Thanks also to Dr. Gwen O'Sullivan for providing me with an invaluable opportunity to begin teaching, and to all of my students, from whom I've learned a great deal.

I could not have done this without the support of great friends and mentors. Thank you to Brendan Lazar, Zilin Qian, Byron Anderson, and Dalton Michelsen for the best conversations, and for your kind words, curiosity, and encouragement. Thanks to Fran Drummond for teaching me how to work outdoors during my summers at Twin Falls, and for instilling a strong work ethic in me. Thanks especially to Michael Wendlandt for the many great personal and scientific conversations shared over beers, in the field, or with a fishing rod in hand. Thanks also to Warren Lake for sending me down the path of environmental science all those years ago. Additional thanks go to the Kurila family for their support and hospitality.

Finally, I cannot thank my parents enough for their unwavering support and belief in my ability to do this. Knowing that I have your backing has given me tremendous confidence and allowed me to go so much farther than I ever thought possible. I am also forever grateful to my Grandpa Ed and Grandma Carol, whose curiosity, love, and prayers have given me motivation and strength during this journey, and to my Uncle Brian and Uncle Craig for their support. My memories of Grandma Melinda and Grandpa Bob have also given me strength, even though they could not be here to see the completion of this thesis. Most of all, my thanks go to Diana. Your patience, support, and love have made this all possible, and I could not have done it without you.

DEDICATION

For Diana, who encouraged me to make this dream come true and who supported me the whole way.

I could not have done this without you.

“I go where the wind is. That's my church.”

Neil Young

I respectfully acknowledge that this research was undertaken on the traditional lands of the Cree and Dene peoples in Treaty 8 territory, and in the territory of the Métis Nation of Alberta (Region 5).

TABLE OF CONTENTS

EXAMINING COMMITTEE MEMBERSHIP	ii
AUTHOR'S DECLARATION.....	iii
STATEMENT OF CONTRIBUTIONS.....	iv
ABSTRACT	vii
ACKNOWLEDGEMENTS	x
DEDICATION	xii
LIST OF FIGURES	xvii
LIST OF TABLES	xix
LIST OF ABBREVIATIONS.....	xx
LIST OF SYMBOLS	xxi
1 LITERATURE REVIEW.....	1
<i>1.1. Introduction.....</i>	<i>1</i>
<i>1.2. Peatland Formation and Development on the Western Boreal Plain.....</i>	<i>2</i>
<i>1.3. Disturbance of Peatlands on the Western Boreal Plain</i>	<i>5</i>
<i>1.4. Well Pad Restoration Techniques for Use in Peatlands</i>	<i>8</i>
<i>1.5. Partial Removal Technique</i>	<i>10</i>
<i>1.6. Thesis Objectives.....</i>	<i>12</i>
<i>1.7. Structure of the Thesis</i>	<i>13</i>
<i>1.8. Weather Conditions During the Thesis Study Period</i>	<i>14</i>
2 HYDROLOGICAL DYNAMICS FOLLOWING PARTIAL REMOVAL OF AN OIL WELL PAD UNDERGOING RESTORATION TO A BOREAL PEATLAND	17
<i>2.1. Introduction.....</i>	<i>17</i>
<i>2.2. Methods.....</i>	<i>20</i>
<i>2.3. Results</i>	<i>26</i>
<i>2.4. Discussion</i>	<i>37</i>
<i>2.5. Conclusion.....</i>	<i>45</i>
3 HYDROLOGIC ASSESSMENT OF MINERAL SUBSTRATE SUITABILITY FOR TRUE MOSS INITIATION IN A BOREAL PEATLAND UNDERGOING RESTORATION	47
<i>3.1. Introduction.....</i>	<i>47</i>

3.2. <i>Methods</i>	51
3.3. <i>Results</i>	57
3.4. <i>Discussion</i>	69
3.5. <i>Conclusion</i>	78
4 DEVELOPMENT OF BIOGEOCHEMICAL PROCESSES ON A RESIDUAL OIL WELL PAD UNDERGOING RESTORATION TO A BOREAL PEATLAND	79
4.1. <i>Introduction</i>	79
4.2. <i>Methods</i>	82
4.3. <i>Results</i>	88
4.4. <i>Discussion</i>	104
4.5. <i>Conclusion</i>	112
5 A COMPARISON OF HYDROLOGIC FLUXES IN A PARTIALLY REMOVED WELL PAD UNDERGOING FEN RESTORATION ON THE WESTERN BOREAL PLAIN, CANADA	114
5.1. <i>Introduction</i>	114
5.2. <i>Methods</i>	118
5.3. <i>Results</i>	128
5.4. <i>Discussion</i>	143
5.5. <i>Conclusion</i>	152
6. CONCLUSIONS AND RECOMMENDATIONS.....	153
6.1. <i>Key Findings</i>	153
6.2. <i>Potential Modifications to the Partial Removal Technique</i>	160
6.3. <i>Recommendations for Further Research</i>	164
6.4. <i>General Conclusion</i>	169
7. REFERENCES.....	170
APPENDIX A.....	189
APPENDIX B.....	193
APPENDIX C.....	195
APPENDIX D.....	203

LIST OF FIGURES

Figure 2.1: Map of the study site	21
Figure 2.2: Inverse distance weighted interpolation map of the residual pad surface elevation	27
Figure 2.3: Geometric mean K_h by depth below ground surface	30
Figure 2.4: Cross-sectional diagrams (vertical exaggeration = 10) of the residual pad and adjacent peatlands	31
Figure 2.5: Plots of site daily precipitation (P; gap filled for 2020) (a-d), depth to water table (DWT; e-h), horizontal hydraulic gradients (i-l), horizontal specific discharge (q) (m-p), and horizontal volumetric flow rates (across pad edges; Q) (q-r)	33
Figure 3.1: Location of the study site in the boreal region of western Canada	53
Figure 3.2: Daily precipitation (P) for the 2020 (15 June to 14 August) and 2021 (16 May to 14 September) study periods	58
Figure 3.3: Soil water retention datapoints and fitted soil water characteristic curves	59
Figure 3.4: Linear regressions between depth to water table (DWT) and soil water potential (Ψ) for low (a), intermediate (b), and high (c) microsites	61
Figure 3.5: Plot hydrological classifications based on mean manually-measured depth to water table	63
Figure 3.6: Depth to water table (DWT) duration curves for the 2020 (a) and 2021 (b) study periods	64
Figure 3.7: Comparison of daily site-measured precipitation (P) (a, c), depth to water table (DWT) referenced against low microsite heights (b, d), and soil volumetric water content (VWC) (e) and soil water potential (Ψ) (f) ..	65
Figure 3.8: Boxplots of VWC and Ψ at low and high microsites in plots along upgradient edges (classified as WO in 2020) and within interior areas (classified as NO in 2020) of the residual pad	68
Figure 4.1: Map of the study site including the location of nutrient monitoring points, piezometers, and mulched and fertilized plots	83
Figure 4.2: Comparison of monthly mean air temperature (a) and monthly total precipitation (b) between study years (2020-2024) and compared against 30-year climate normal data (monthly means for the years 1991-2020)....	89
Figure 4.3: Shallow groundwater electrical conductivity (EC; a-c), pH (d-f), and temperature (g-i) within the pad and the upgradient ('Up peat'), north ('North peat'), and downgradient ('Down peat') peatlands	92
Figure 4.4: Cation supply rates measured using PRS probes in the fen and in the hydrologically well optimized (WO) and non-optimized (NO) areas on the pad	95
Figure 4.5: Decomposition rate constants, k (a-b) and stabilization factors, S (c-d) on the pad and in the fen	97
Figure 4.6: Extractable NO_3^- pools (a-d) and daily net NO_3^- transformation rates (e-h) in the fen and in the hydrologically well optimized (WO) and non-optimized (NO) areas on the pad	98

Figure 4.7: Extractable NH_4^+ pools (a-d), daily net NH_4^+ transformation rates (e-h), and NH_4^+ supply rates (i-l) in the surrounding fen and in the hydrologically well optimized (WO) and non-optimized (NO) areas on the residual pad	100
Figure 4.8: Pore water nitrate (NO_3^- ; a-e), ammonium (NH_4^+ ; f-j), and phosphate (PO_4^{3-} ; k-o) concentrations in the fen and in well optimized (WO) and non-optimized (NO) areas on the pad	101
Figure 4.9: Extractable PO_4^{3-} pools (a-d), daily net P mineralization rates (e-h), and P supply rates (i-l) in the surrounding fen and in the hydrologically well optimized (WO) and non-optimized (NO) areas on the residual pad	103
Figure 5.1: Map of the study site including the surface elevations of the residual well pad, constructed upland feature, and surrounding peatland.....	119
Figure 5.2: Daily snow depth acquired from Environment and Climate Change Canada (Slave Lake station, black line) and snow depths transcribed from timelapse cameras installed on the research site in the winters of 2022-23 (a) and 2023-24 (b)	129
Figure 5.3: Comparison of manually measured snow depth (a-c), snow density (d-f), and snow water equivalent (SWE; g-i)	130
Figure 5.4: Comparison of the depth to the frost table (pad and upland) and seasonal ground ice (fen).....	132
Figure 5.5: Plots of daily rates of ET_a (a-b) and E_a (c-d), mean percentage surface ponding (e-f), and cover-weighted daily volumetric atmospheric water loss ($ET_{total} = ET_a + E_a$) (g-h)	135
Figure 5.6: Plots of manually measured volumetric water content.....	136
Figure 5.7: Plots of cumulative ET_{cont} from soil surfaces relative to cumulative rainfall (a-b), depth to water table (DWT; c-d), volumetric water content (VWC; e-f), and substrate temperature (g-h).....	138
Figure 5.8: Relationships between average daily plot ET_{cont} and depth to water table (DWT; a-b), volumetric water content (VWC; c-d), and vapour pressure deficit (VPD; e-f)	139
Figure 5.9: Plots of groundwater dynamics on the residual pad	141
Figure 6.1: Conceptual model of potential vegetation establishment outcomes in hydrologically well optimized (WO) and hydrologically non-optimized (NO) areas of the residual well pad.	158

LIST OF TABLES

Table 1.1: Summary of growing season total rainfall and mean air temperatures for all study periods included in this thesis. All values are compared to 30-year regional climate normal data for the period 1991 – 2020. Additional comparisons to 30-year regional climate normal data for the period 1981 – 2010 are provided for the 2020 and 2021 growing seasons to ensure consistency with Chapter 2 of the thesis.	15
Table 2.1: Summary of the hydrophysical properties of the residual mineral fill, near-surface peat collected from hollows in the adjacent peatlands, and the underlying basal mineral substrate	29
Table 3.1: Results summary for linear mixed effects models used to determine significance of main environmental and treatment effects and their interactions on volumetric water content (VWC) and soil water potential (Ψ) for each study period	60
Table 4.1: Mean residual mineral fill total micro and macronutrient concentrations and substrate nutrient ratios.....	90
Table 4.2: Residual mineral fill total cation exchange capacity (CEC) and exchangeable cation concentrations	90
Table 4.3: Median volumetric water content ($\text{m}^3 \text{m}^{-3}$; interquartile range in parentheses) for peat from the fen and the mineral substrate in the early season (ES) and late season (LS) each year	93
Table 4.4: Median temperatures ($^{\circ}\text{C}$; interquartile range in parentheses) for peat from the fen and the mineral substrate in the early season (ES) and late season (LS) each year	94
Table 5.1: Summary of mean monthly ET_{cont} , net radiation (Q^*), ground heat (Q_G ; day and night), latent heat (Q_E), sensible heat (Q_H), and the ratio of Q_H to Q_E	140
Table 5.2: Summary of hydrologic fluxes in the residual well pad.	142

LIST OF ABBREVIATIONS

a. s. l.	Above sea level	PRS	Plant Root Simulator (probes)
ANOVA	Analysis of variance	RTK	Real time kinematic
b. g. s.	Below ground surface	spp.	Species
Cen-Reg	Censored regression	SRP	Soluble reactive phosphorus
DWT	Depth to water table	SWCC	Soil water characteristic curve
EC	Electrical conductivity	SWE	Snow water equivalent
ICP-OES	Inductively Coupled Plasma Optical Emission Spectroscopy	TC	Total carbon
LME	Linear mixed effects model	TN	Total nitrogen
LOQ	Limit of quantitation	VIF	Variance inflation factor
MLTT	Moss Layer Transfer Technique	VPD	Vapour pressure deficit
NO	Non-optimized	VWC	Volumetric water content
NRAL	Natural Resources Analytical Laboratory	WBP	Western Boreal Plain
OLS-Reg	Ordinary least squares regression	WO	Well optimized
OM	Organic matter	WT	Water table

LIST OF SYMBOLS

Δ	Slope of the saturation vapour pressure temperature curve	q	Specific discharge
A	Flow face area	Q^*	Net radiation flux
C_{runoff}	Runoff coefficient	Q_E	Latent heat flux
dh/dl	Horizontal hydraulic gradient	Q_G	Ground heat flux
E_a	Actual evaporation	Q_H	Sensible heat flux
ET_a	Actual evapotranspiration	Q_h	Horizontal volumetric discharge
ET_{cont}	Continuous evapotranspiration	Q_v	Vertical volumetric discharge
ET_{total}	Total evapotranspiration	S	Stabilization factor
k	Decomposition rate constant	α	Priestley Taylor coefficient
K_h	Horizontal hydraulic conductivity	γ	Psychrometric constant
K_v	Vertical hydraulic conductivity	λ_v	Latent heat of vapourization
P	Precipitation	ρ_w	Density of water
PET_{eq}	Equilibrium potential evapotranspiration	σ	Standard deviation
$P_{ponding}$	Proportion of ponding	Ψ	Soil water potential

1 LITERATURE REVIEW

1.1. Introduction

Canada's Western Boreal Plain (WBP) extends from northern Alberta through central Saskatchewan and into southern Manitoba (Pojar, 1992). The WBP is underlain by sedimentary glacial till and is characterized by distinct landscape units including different upland forest stand types and different classes of peatland (Brandt et al., 2013; Pojar, 1992; Vitt and Chee, 1990). The region is subhumid, meaning that potential evapotranspiration frequently exceeds precipitation (Mwale et al., 2009), which makes water conservation an important peatland ecosystem function (Brown et al., 2010). Peatlands form the intersection between terrestrial and aquatic ecosystems (Brandt et al., 2013; Rezanezhad et al., 2016) and are often hydrologically coupled with local and regional-scale groundwater systems within the WBP (Volik et al., 2024). As such, peatlands play an important role in the regulation of water quantity and quality in both upland and aquatic ecosystems (Devito et al., 2012; Elmes and Price, 2019; Kimmel and Mander, 2010). Canadian peatlands also cover approximately 12 % of Canada's land mass (Tarnocai, 1984), store upwards of 150 Gt of carbon (Harris et al., 2021), and provide important habitats for a wide variety of wildlife including the threatened woodland caribou (Hill et al., 2021; Rettie and Messier, 2000). Furthermore, they are places with immense spiritual and cultural significance to Indigenous peoples including the Cree and the Dene (Bélisle and Asselin, 2021; Parlee et al., 2012; Speller and Forbes, 2022).

The different landscape units that comprise Canada's WBP can be described on the basis of both nutrient and moisture regimes in a system of 'ecosite classification' (Beckingham and Archibald, 1996). Under this system, ecosites are defined as locations sharing common nutrient and moisture regimes, which tend to support similar vegetation communities (Beckingham and Archibald, 1996). This system is based on the principle that moisture and nutrient availability determine the physical vegetation structure of ecosystems (Craine and Dybzinski, 2013), as they are the variables that define the fundamental niches of plants (Chapin et al., 2011). In turn, the physical vegetation structure of an ecosystem exerts considerable

control over energy, water, and nutrient cycling processes, and thus also over interspecific competitive dynamics and ecosystem succession (Chapin et al., 2011; Odum, 1969).

Ecosite classification has recently been used to inform the likely successional trajectories of restored and constructed boreal ecosystems (e.g., Farnden, 2023; Thiffault et al., 2017). In this context, it can be used to predict the successional trajectories of novel ecosystems that have been created through the processes of ecological restoration or reclamation (Audet et al., 2015; Gerwing et al., 2022; Nwaishi et al., 2015a). This is important, because novel ecosystems often differ from undisturbed natural ecosystems both in terms of their physical structure and the ecosystem functional processes occurring within them (D'Astous et al., 2013; Nwaishi et al., 2015a, 2015b; Popović et al., 2022). Comparison between the ecosite classifications of novel and natural ecosystems can provide a general indicator of which species a site is likely to support over the long-term based on their fundamental niches (Nwaishi et al., 2015a; Thiffault et al., 2017). This may be especially advantageous for the assessment of peatland restoration outcomes, given that peatlands can take hundreds to thousands of years to develop (Halsey et al., 1998; Kuhry et al., 1992; Ruppel et al., 2013; Rydin and Jeglum, 2006a).

1.2. Peatland Formation and Development on the Western Boreal Plain

Peat formation first began on the WBP approximately 8,000 to 9,000 years ago during the early Holocene (Brandt et al., 2013; Halsey et al., 1998) through the process of paludification (Nicholson and Vitt, 1989; Ruppel et al., 2013). Paludification is defined as the saturation of mineral soils that have either previously supported terrestrial vegetation or that have undergone pedogenic processes (Bauer et al., 2003; Kuhry and Turunen, 2006; Rydin and Jeglum, 2006a). Rates of peatland initiation then increased between about 6,000 and 7,400 years ago in response to the onset of cooler and wetter conditions during the mid-Holocene (Bauer et al., 2003; Ruppel et al., 2013; Zoltai and Vitt, 1990). These conditions were caused by a decrease in regional summer insolation (i.e., decreases in temperature and evapotranspiration) and increases in the

amount of moist air transported over the WBP from the Pacific Ocean (i.e., increases in precipitation) (Halsey et al., 1998).

Paludification continues to occur on the WBP in modern times, particularly along the margins between peatlands and upland forests. In these areas, increases in water table (WT) elevation within peatlands can result in flooding of terrestrial soils along the edges of uplands (Bauer et al., 2003; Ruppel et al., 2013). This, in turn, enables the lateral expansion of peatland vegetation communities, particularly on fine-textured soils that have a high water retention capacity (Kuhry and Turunen, 2006; Nicholson and Vitt, 1989). Increases in soil water content result in peat formation because saturation of the substrate limits the rate of decomposition to the point that it is exceeded by the rate of net primary productivity (Bauer et al., 2003; Vitt et al., 2009). Limitations on decomposition occur because microbial processes are less efficient in anoxic environments due to microbial use of alternate electron acceptors during anaerobic respiration (Nwaishi et al., 2015a). Because rates of primary production exceed those of decomposition, organic litter materials build up over time, forming organic soils (Vitt et al., 2009).

The conditions supporting paludification are often said to have occurred in response to the saturation of low-lying areas as a result of increased surface runoff from topographically high points (Bauer et al., 2003). However, it may also have occurred at least in part due to increases in groundwater discharge to low-lying areas (McCarter et al., 2023), given that low amounts of surface runoff are typically observed on the WBP in the present day (Devito et al., 2005; Redding and Devito, 2011). As such, it is likely that WT and soil moisture dynamics in recently paludified sites were partially dependent upon local and regional-scale groundwater systems, which dictate groundwater recharge and discharge dynamics in the WBP (Elmes and Price, 2019; Hokanson et al., 2019). Combined, this results in hydrologic conditions in recently paludified sites being more variable during the early stages of peatland succession (Kuhry et al., 1992; Zoltai and Vitt, 1990) than those observed in established peatlands today (Waddington et al., 2015). Thick peat soils in established peatlands can regulate water table dynamics through autogenic feedbacks, including reductions in surface elevation and subsurface flow in response to decreases in WT elevation

(Howie and Hebda, 2018; Waddington et al., 2015). These occur as a result of collapse of the pore structure, given that decreases in pore water pressure cause increases in effective stress within the peat profile (Price, 2003).

During the earliest stages of succession towards peatland communities, marsh vegetation communities are common and are primarily dominated by graminoids including *Typha latifolia* and *Carex* spp. (Kuhry et al., 1992). These early communities have high rates of primary productivity (Strack et al., 2006; Szumigalski and Bayley, 1996). This has been observed early during primary succession in a wide variety of ecosystem types (Odum, 1969) because plants with high growth rates are favoured by intense competition for nutrients, water, and light (Craine and Dybzinski, 2013). The fact that remnants of *Carex* species are particularly common at the base of paleoecological peat cores also indicates that water levels were highly variable for the establishment of peatland mosses (Kuhry et al., 1992), as they rely upon well-regulated, near-surface WTs to sustain growth (Ketcheson and Price, 2014; McCarter and Price, 2014). In contrast, the rooting systems of sedges allow them to stay in contact with fluctuating water tables (Chapin and Chapin, 1981; Strack et al., 2006).

Eventually, the establishment of peatland mosses does occur, including that of *Calliergon giganteum*, *Drepanocladus aduncus*, and *Campylium stellatum* (Kuhry et al., 1992), which are characteristic of rich fens (Vitt, 2014). This is supported by increases in WT elevation and increases in the water retention capacity of paludified mineral soils, which occurs when substrate organic matter content increases (Nwaishi et al., 2015a; Rawls et al., 2003; Rydin and Jeglum, 2006a). Increases in organic matter content also result in increased availability of organic substrates for decomposition, which supports greater amounts of nutrient mineralization (Nwaishi et al., 2015a). Combined, the establishment of true peatland mosses is supported by increases in the availability of both water and inorganic nutrients found in reduced forms (Nwaishi et al., 2015a; Vitt and Chee, 1990). Over time, conditions often become somewhat drier, with the ground layer shifting to a community dominated by species such as *Tomentypnum nitens* and *Aulacomnium palustre*, as well as *Sphagnum warnstorffii* and *Sphagnum fuscum* (Kuhry et al., 1992). These

later stages of peatland development also include a greater proportion of peatland shrubs, including *Rhododendron groenlandicum* and *Betula* spp., which are eventually followed by the tree *Larix laricina* (Bauer et al., 2003; Kuhry et al., 1992; Nicholson and Vitt, 1989).

The ‘true mosses’ characteristic of early peatland successional stages are defined as those belonging to the class Bryopsida, which are often termed ‘brown mosses’ based on the colour of many common species (Vitt, 2014). True moss-dominated ground layers are a defining characteristic of moderate and extreme-rich fens (Chee and Vitt, 1989). Moderate and extreme-rich fens are minerogenous, meaning they are also characterized by hydrological connectivity to groundwater (Elmes and Price, 2019). This groundwater has a considerable effect on fen biogeochemistry, as it maintains high concentrations of base cations in the pore water and results in higher pH (> 5.5) and electrical conductivity ($100 - 2,000 \mu\text{S cm}^{-1}$) than observed in bogs (Alberta Environment, 2015; Chee and Vitt, 1989). In contrast, poor fens and bogs are characterized by ground layers dominated by *Sphagnum* mosses (Vitt, 2014). Bogs are typically thought of as being a later-successional form of peatland, as they develop once the peat profile is sufficiently thick to isolate water at the surface of the peatland from inputs of groundwater (Rydin and Jeglum, 2006a). As such, the water inputs to bogs are primarily restricted to precipitation in the form of rainfall and snowmelt (Glaser et al., 1997). In addition to acidification of the peatland by *Sphagnum* mosses, this contributes to decreased pore water cation concentrations and lower pH (< 5.5) and electrical conductivity ($< 100 \mu\text{S cm}^{-1}$) compared to those in moderate and extreme-rich fens (Alberta Environment, 2015; Vitt et al., 1995).

1.3. Disturbance of Peatlands on the Western Boreal Plain

Prediction of the successional trajectories of novel peatland ecosystems in a restoration context is becoming increasingly important in Canada’s western boreal regions given the scale of industrial disturbance that has taken place in the area over the past several decades (Ficken et al., 2019; Nwaishi et al., 2015a; Pasher et al., 2013; Rooney et al., 2015; Schneider and Dyer, 2006; Volik et al., 2020). In the case of peatlands, much

of this disturbance is related to the oil and gas industry, as much of the WBP is underlain by geological formations that contain oil and gas deposits (Orbach, 2012). To access oil and gas reserves located within about 70 m of the ground surface, open-pit mining techniques are employed. This involves the complete removal of peatlands and upland forests from the land surface, and the subsequent excavation of deep open pits to extract bitumen-rich sand (Rooney et al., 2012). It is anticipated that open-pit oil sands mining will result in the loss of upwards of 29,500 ha of boreal peatland (Rooney et al., 2012), and only a fraction of this peatland area is likely to be reconstructed as part of the obligatory reclamation process (Borkenhagen et al., 2024; Nwaishi et al., 2015b; Rooney et al., 2015). Such reclamation efforts to date have been focused on the reconstruction of functional watersheds, with recent designs typically including mosaics of upland areas, low-lying constructed fens, and end-pit lakes (Gingras-Hill et al., 2018; Popović et al., 2022; Price et al., 2010; Zabel et al., 2024). Many of these constructed landscapes have also been found to support the autogenic development of ‘opportunistic wetlands’ in locations where the tailings sand used to create upland areas settles (Hawkes et al., 2025, 2020; Little-Devito et al., 2019; Wendlandt, 2023). Recent designs for these watersheds have included confining layers at depth within upland features, which are intended to promote the transmission of water to downslope lakes and wetlands (Ketcheson et al., 2017; Zabel et al., 2024). Surface runoff to these low-lying systems has also been observed during the spring freshet (Biagi and Carey, 2020; Ketcheson and Price, 2016).

In contrast, when oil and gas reserves are located at depths greater than 70 m below the ground surface, in-situ drilling is used to extract oil and gas (Orbach, 2012). This involves constructing stable platforms (termed ‘well pads’ or ‘well sites’) on the ground surface to support drilling infrastructure, pump jacks, and storage tanks (UNEP, 2022; Volik et al., 2020). Given the low trafficability of peat soils, the construction of well pads involves the removal of all trees and shrubs, and the subsequent placement of a thick layer of sand or clay (termed ‘mineral fill’) onto the peatland surface (Lemmer, 2022; Vitt et al., 2011). This results in compaction of the peat profile (Lemmer et al., 2020), which reduces its hydraulic conductivity (Elmes et al., 2021b; Pilon, 2015) and can disrupt subsurface flow through the adjacent

peatland (Osco et al., 2018; Saraswati et al., 2020b). Very little work has been done to evaluate the effects of well pads on biogeochemical dynamics and vegetation communities in surrounding peatlands, although it is possible their impact may be similar to that of roads given that they are also usually constructed out of mineral fill (Brawner and Tessier, 1969; Ducks Unlimited Canada, 2014).

The impacts of roads on peatland ecohydrological function are variable and are largely dependent upon road orientation relative to the direction of subsurface flow within the peat profile (Saraswati et al., 2020b; Williams-Mounsey et al., 2021). Specifically, when roads are oriented approximately perpendicular to the direction of subsurface flow, a ‘damming’ effect is observed, wherein the WT on the upgradient side is nearer to the surface than the WT on the downgradient side (Elmes et al., 2022; Osco et al., 2018; Saraswati et al., 2020b). When this occurs, the moisture regime downgradient of roads becomes drier, and supports increases in canopy cover and vascular plant density (Miller et al., 2015; Saraswati et al., 2020a; Willier et al., 2022). This also results in changes to ground-layer community composition. Specifically, shifts towards dry-adapted hummock and feather mosses occur due to the reduction in surface moisture availability, which also drives overall decreases in moss cover (Miller et al., 2015; Plach et al., 2017). On the upgradient side of roads constructed perpendicular to flow, decreases in the thickness of the unsaturated rooting zone tend to result in tree and shrub die-back (Bocking et al., 2017; Saraswati et al., 2020a; Willier et al., 2022).

The hydrological impacts of roads can also result in increased rates of microbial enzyme activity (and thus likely also decomposition) on their downgradient sides (Saraswati et al., 2019). Combined with increased methane emissions from flooded areas on the upgradient side, this contributes to the increases in carbon emissions that can be expected in the vicinity of many roads (Saraswati and Strack, 2019). The impacts of roads on peatland nutrient cycling are less well understood, although it is likely that increases in nutrient mineralization rates would occur on their downgradient side if the WT were to drop by more than 20 cm (Devito and Dillon, 1993; Plach et al., 2017). Notably, the impact of roads on peatland ecohydrological function appears to be muted in all cases when they are oriented parallel to groundwater

flow paths, as this results in minimal disruption to flow (Saraswati et al., 2020b). It is highly likely that all well pads will have at least some impact on hydrology in the peatlands that surround them, as these features are either square or rectangular in shape (Osco et al., 2018). Thus, groundwater flow paths will be bisected by at least one side of a well pad regardless of the direction of groundwater flow.

Within the direct footprint of oil and gas well pads, natural peatland ecosystem function is entirely lost given that the surface of the peatland is covered over with mineral fill materials. Once these sites are abandoned, they are often left in place and revegetated with graminoid communities typically dominated by grasses (Lemmer et al., 2022, 2020). However, there is a legal requirement in the Province of Alberta that these sites be reclaimed to a state of ‘equivalent land capability’ (Powter et al., 2012). Equivalent land capability is defined in Alberta as taking actions to ensure that “*the ability of the land to support various land uses after conservation and reclamation is similar to the ability that existed prior to an activity being conducted on the land*” (Alberta Environment and Parks, 2017). Provincial reclamation criteria are used to assess whether or not a site has reached a state of equivalent land capability (Powter et al., 2012). In the case of peatlands, these criteria require that a decommissioned well pad sustains a sufficiently wet moisture regime for the initiation and development of a peat-forming vegetation community (Alberta Environment and Parks, 2017). Efforts must also be made to reduce the extent to which reclaimed pads are covered with open water, non-vegetated areas, or upland ecosite types, and to minimize erosion and surface water flow (Alberta Environment and Parks, 2017).

1.4. Well Pad Restoration Techniques for Use in Peatlands

Given the recency of the release of the peatland-specific reclamation criteria, few if any well pads have been certified as having been reclaimed to a peatland to date (Tokay et al., 2019). Development of peatland restoration techniques for use on decommissioned oil and gas well pads is also in its infancy (Chimner et al., 2017; Volik et al., 2020). Nonetheless, several research trials have examined potential techniques to re-

establish peat forming vegetation communities on these sites. Two promising methods involve the removal of the mineral fill used to construct a well pad from the surface of a peatland. This material can then either be transported off-site for disposal (the ‘Complete Removal Technique’) or buried beneath the peat that it once sat on top of (the ‘Burial Under Peat Technique’) (Bird and Xu, 2020; Engering et al., 2021; Pouliot et al., 2021; Shunina et al., 2016; Xu et al., 2022). In both cases, an organic peat substrate is re-exposed and brought up to the surface of the site. These substrates are in some ways similar to those left on cutover peatlands after the extraction of peat for horticultural use (Rocheffort et al., 2016, 2003; Xu et al., 2022). Thus, they are typically actively revegetated by applying a modified version of the Moss Layer Transfer Technique (MLTT) originally developed for use on abandoned cutover peatlands in Eastern Canada (Bird and Xu, 2021a; M. E. Gauthier et al., 2018; Pouliot et al., 2021; Xu et al., 2022).

The original MLTT is a form of assisted succession, wherein viable propagules of peatland vascular species and mosses are spread across the surface of cutover peatlands. These ‘donor propagules’ are harvested from the acrotelm of either an undisturbed peatland, or a peatland that will be disturbed or extracted in the future (Quinty et al., 2020a; Rocheffort et al., 2003). They are then transported to the restoration site, where manure spreaders or other small equipment are used to distribute them across the surface in a 1:10 aerial ratio (Quinty et al., 2020b; Rocheffort et al., 2003). The donor propagules are then fertilized with phosphate at a rate of 150 kg ha⁻¹ and then covered with a thin layer of straw mulch (González and Rocheffort, 2014; Groeneveld et al., 2007; Quinty et al., 2020b; Rocheffort et al., 2003). The application of organic or rock phosphate fertilizer has been found to promote the establishment of the true moss *Polytrichum strictum* (Groeneveld et al., 2007; Pouliot et al., 2015). The establishment of *P. strictum* stabilizes the exposed peat substrates and reduces the risk that frost heaving will result in hydrological disconnection between the introduced mosses and the surface (González and Rocheffort, 2014; Groeneveld et al., 2007; Pouliot et al., 2015). Straw mulch has been found to reduce evaporative water losses (Price et al., 1998), which is important given that cutover peat substrates are often subject to a greater frequency of drying events than is observed in natural peatlands (Ketcheson and Price, 2014; Price, 2003; Price and

Whitehead, 2001). All of these steps are also normally undertaken when applying the MLTT on the peat substrates that are resurfaced from beneath well pads (Bird and Xu, 2021a, 2020).

Moss establishment following the implementation of the Complete Removal and Burial Under Peat techniques has been variable. Limited moss establishment following both natural revegetation and MLTT has been observed during some trials (Pouliot et al., 2021; Shunina et al., 2016), while the establishment of a promising mix of *Sphagnum* spp. and true mosses (including *Polytrichum strictum*, *Tomentypnum nitens*, and *Aulacomnium palustre*) has been observed during others (Xu et al., 2022). Notably, sedge cover exceeded the cover of peatland mosses when the technique was applied in a fen (Xu et al., 2022), although this may not be of concern given the high proportion of sedges in early stages of peatland development. However, the nutrient regimes on these exposed peat substrates can be variable (Pouliot et al., 2021; Shunina et al., 2016), and as such they may support establishment of both ombrotrophic communities characterized by *Sphagnum* mosses and minerotrophic communities characterized by true mosses (Xu et al., 2022).

1.5. Partial Removal Technique

Given that upwards of 63% of peatlands on the WBP are minerotrophic fens (Vitt et al., 2000), there is a need to develop techniques likely to promote primarily fen vegetation communities. As such, an alternative to the Complete Removal and Burial Under Peat techniques with the potential to support consistently high base cation availability has been proposed (Borkenhagen and Cooper, 2016; Vitt et al., 2012, 2011). This method involves recontouring the surface of a decommissioned pad through the removal of only the uppermost layers of mineral fill using heavy machinery (Bird & Xu, 2021). Termed the ‘Partial Removal Technique’, this strategy leaves a layer of cation-rich mineral fill in place at the surface (M. E. Gauthier et al., 2018; Lemmer et al., 2022; Vitt et al., 2011). The residual well pads created through this process can subsequently be planted with vascular species including sedges (Vitt et al., 2011), revegetated using the

MLTT (Borkenhagen and Cooper, 2016; M. E. Gauthier et al., 2018), or left to revegetate on their own through natural ingress of vegetation from surrounding peatlands (Lemmer et al., 2022, 2020). However, for reliable establishment of a moss-dominated community, it has been suggested that application of the MLTT on well pad-sized sites is required (Xu et al., 2022).

The Partial Removal Technique was inspired by the natural process of paludification (Borkenhagen and Cooper, 2016; Koropchak et al., 2012; Vitt et al., 2012, 2011), and is a form of assisted succession. That said, the technique could be considered physically analogous to the process of primary peat formation, given that the term ‘paludification’ is typically applied when the mineral soils that become saturated have undergone pedogenic processes and have supported terrestrial vegetation (Bauer et al., 2003; Kuhry and Turunen, 2006; Rydin and Jeglum, 2006a) as previously described. Primary peat formation is a similar process, wherein undeveloped mineral soils exposed by deglaciation or isostatic rebound become sufficiently wet for the establishment of wetland vegetation communities without an intermediate vegetated terrestrial stage (Ruppel et al., 2013; Rydin and Jeglum, 2006a). The mineral fill used to construct well pads is often collected from the substratum of boreal forests (Green et al., 1992; Macyk, 2006), and thus may not have previously undergone any form of pedogenesis. As such, from a physical perspective, the Partial Removal Technique may be more similar to primary peat formation than to paludification. Regardless, the process of peatland development following saturation of either formerly terrestrial mineral soils or undeveloped mineral soils is similar when considered over long timescales (Rydin and Jeglum, 2006a).

Various residual mineral surface elevations have been tested during implementation of the Partial Removal Technique. True moss establishment has been shown to be greatest when the elevation of residual pads is closely aligned with the average elevation of the water table in the surrounding peatland (i.e., within a few centimeters) (M. E. Gauthier et al., 2018; Lemmer et al., 2022). It is also critical that the donor vegetation community be harvested from a fen with similar pH, electrical conductivity, and calcium supply as the residual mineral fill (M. E. Gauthier et al., 2018). However, all previous research trials were conducted only along subsections of well pads or along narrow roads. As such, these trials were conducted

in locations directly adjacent to the surrounding peatlands. In these locations, subsurface hydrological connectivity was sufficient to maintain near-surface water tables when the reconfigured surface was closely aligned with the water table in the adjacent peatland, which in turn supported true moss initiation and establishment (M. E. Gauthier et al., 2018; Lemmer et al., 2022; Vitt et al., 2011).

As the technique has never been studied in a research context on a full-size well pad, there is considerable uncertainty regarding the potential for groundwater flow to maintain a near-surface water table across an entire residual well pad. Restrictions on subsurface flow are likely, given that the hydraulic conductivity of mineral substrates (Domenico and Schwartz, 1998) and the compacted peat beneath features constructed out of mineral fill (Elmes et al., 2021b) are lower than that in natural peatlands (Boelter, 1965). This could presumably result in limited water input to interior and central portions of residual well pads, which could in turn result in deep water tables. Previous studies have identified restrictions on the growth of mosses introduced to cutover peat substrates when the water table falls under approximately 20 cm below the surface (Ketcheson and Price, 2014; Price and Whitehead, 2001). This could also result in higher decomposition and nutrient mineralization rates within an oxic substrate (Haapalehto et al., 2014; Nwaishi et al., 2015a), which could in turn inhibit the accumulation of organic matter and support the availability of nutrients in oxidized, rather than reduced forms (Bayley et al., 2005; Bridgham et al., 1998; Updegraff et al., 1995). Given that such conditions would result in an ecosite dissimilar to regional reference peatlands, they would also likely support the establishment and succession of non-peatland vegetation communities (Beckingham and Archibald, 1996; Lemmer et al., 2022) possibly including undesirable ruderal species (Zedler and Kercher, 2004).

1.6. Thesis Objectives

The work described herein was conducted on the first full-scale residual well pad studied following the implementation of the Partial Removal Technique. The residual well pad was located in a moderate-rich

fen complex near the town of Slave Lake, Alberta, Canada. Given the uncertainty surrounding the hydrological and biogeochemical suitability of residual mineral substrates for the establishment of peatland true mosses, the objectives of this thesis are as follows:

1. Assess whether groundwater connectivity between a residual well pad and an adjacent fen is sufficient to maintain a near-surface and well-regulated water table,
2. Characterize the hydrological suitability of a residual mineral substrate to maintain water availability to an introduced moss community,
3. Characterize the biogeochemical suitability of a residual mineral substrate to maintain nutrient availability to an introduced moss community, and assess the role of substrate nutrient production and cycling in sustaining that availability, and,
4. Assess differences in the relative importance of major atmospheric and subsurface water fluxes to the maintenance of saturated (anoxic) substrate moisture conditions at different times during the growing season.

1.7. Structure of the Thesis

The contents of this thesis have been divided into six chapters to align with the manuscript format. The introductory chapter provides context regarding the impacts of industrial development on peatland ecohydrological function on the WBP, the process of peatland initiation and development following paludification, and the state of knowledge regarding peatland restoration techniques for use on well pads. The restoration techniques examined include the Partial Removal Technique, which aims to emulate paludification to set introduced vegetation communities on a successional trajectory towards peatland ecosystem function. The introductory chapter also outlines the purpose and overarching objectives of the thesis. The subsequent four chapters are individual manuscripts (based on empirical data) that each address one of the four objectives of this thesis.

Specifically, Chapter two addresses the first objective by means of a detailed field assessment of hydrological connectivity and water table regulation across a full-size partially removed oil well pad. Chapter three expands upon the research findings presented in Chapter two by means of a field assessment of soil moisture dynamics (including substrate water storage and availability) on the same pad, which makes comparisons against established literature-based moss water availability thresholds. Chapter four presents results from a field assessment of biogeochemical dynamics on the residual pad (including substrate decomposition, nutrient cycling, and nutrient supply). It then integrates the research findings presented in Chapter three to predict the successional trajectory of the introduced vegetation community based on a comparison between ecosite classifications on the site and in the broader WBP. Chapter five addresses the recommendations of Chapters two, three, and four by closing the residual well pad water balance over two growing seasons. It provides insight into the relative magnitudes of different water inputs to, and losses from, the pad. This is used to support the recommendation of modifications to the Partial Removal Technique that may shift the ecosite classification of future residual well pads closer to peatland ecosite types.

Chapter six provides a summary of the research findings presented in Chapters two through five, with specific emphasis on synthesizing the main contributions of the thesis to the peatland restoration literature. Four appendices are included after Chapter six, each of which presents additional data in support of the research findings and conclusions of Chapters two through five.

1.8. Weather Conditions During the Thesis Study Period

The empirical studies detailed in Section 1.7 were conducted during five growing seasons between the years 2020 and 2024. These seasons demonstrated considerable hydroclimatic variability, presenting an opportunity to compare the development of ecohydrological processes during drier-than-average, typical, and wetter-than-average growing seasons (Table 1.1). Specifically, Chapters 2 and 3 involved an initial

characterization of hydrological and substrate moisture dynamics in the 2020 and 2021 growing seasons, during which the Lesser Slave Lake region received 98 % and 84 % of the 30-year (1991 – 2020) climate normal growing season rainfall (May – September), respectively (Table 1.1). The growing seasons in 2020 and 2021 also had typical and warmer-than-average mean air temperatures, respectively (Table 1.1). Data collection in support of Chapter 5 was informed by the results presented in Chapters 2 and 3, and as such was undertaken during the subsequent two growing seasons (2022 – 2023). During these two growing seasons, the Lesser Slave Lake region received 62 % (2022) and 110 % (2023) of the 30-year (1991 – 2020) climate normal total seasonal rainfall (Table 1.1). Both 2022 and 2023 were characterized by warmer-than-average growing seasons (Table 1.1). As biogeochemical processes on the residual well pad were highly dynamic in the early years post-partial removal, data collection in support of Chapter 4 was undertaken during all five growing seasons. This included the 2024 growing season, which received 133 % of the 30-year (1991 – 2020) climate normal rainfall and which had warmer-than-average air temperatures (Table 1.1). A climatogram visualizing monthly mean air temperatures and total monthly rainfall against the respective 30-year climate normal values is provided in Section 4.3.1.

Table 1.1: Summary of growing season total rainfall and mean air temperatures for all study periods included in this thesis. All values are compared to 30-year regional climate normal data for the period 1991 – 2020. Additional comparisons to 30-year regional climate normal data for the period 1981 – 2010 are provided for the 2020 and 2021 growing seasons to ensure consistency with Chapter 2 of the thesis.

Year	Rainfall		Air Temperature		Season Classification	Manuscripts
	Season total (mm)	Percentage of 30-year mean	Season mean (°C)	Percentage of 30-year mean		
2020	286	91 (1981 - 2010) 98 (1991 - 2020)	13.1	99 (1981 - 2010) 101 (1991 - 2020)	Typical	Chapters 2, 3, 4
2021	244	77 (1981 - 2010) 84 (1991 - 2020)	14.3	108 (1991 - 2020) 110 (1991 - 2020)	Dry & warm	Chapters 2, 3, 4
2022	182	62 (1991 - 2020)	14.1	109 (1991 - 2020)	Dry & warm	Chapters 4, 5
2023	320	110 (1991 - 2020)	14.9	115 (1991 - 2020)	Wet & warm	Chapters 4, 5
2024	388	133 (1991 - 2020)	13.9	107 (1991 - 2020)	Wet & warm	Chapter 4

Of note to readers of this combined thesis, Chapter 2 was written and submitted for publication prior to the release of the 1991 – 2020 climate normals dataset by Environment and Climate Change Canada. As such, weather conditions detailed in Chapter 2 were compared to the older 1981 – 2010 climate normals dataset (Environment and Climate Change Canada, 2024a). All other manuscripts (i.e., Chapters 3 through 5) drew comparisons against the updated 1991 – 2020 climate normals dataset (Environment and Climate Change Canada, 2024b). The 30-year mean growing season (May – September) rainfall differed between the 1981 – 2010 (317 mm) and 1991 – 2020 (292 mm) climate normal datasets. However, this did not affect the classification of growing seasons as dry, typical, or wet between chapters (but note that the specific terminology used varies by chapter in response to peer-reviewer feedback).

Also of note to readers of this combined thesis is the fact that Chapters 2 through 4 draw comparisons between the 30-year climate normal dataset and regional growing season rainfall and/or air temperature data obtained from the Slave Lake airport weather station for the months of May through September. This was done to maintain consistency between years, as the tipping bucket used to measure rainfall was not installed on all days during the growing seasons of 2020 or 2021, and no site-level air temperature data was available for either of those years. Conversely, Chapter 5 draws comparisons between the 30-year climate normal dataset and site-level rainfall and air temperature data measured on-site for the months of May through August. This site-level comparison was possible because the site was instrumented with a tipping bucket and air temperature sensor for the full duration of those months in the 2022 and 2023 growing seasons.

2 HYDROLOGICAL DYNAMICS FOLLOWING PARTIAL REMOVAL OF AN OIL WELL PAD UNDERGOING RESTORATION TO A BOREAL PEATLAND

2.1. Introduction

Peatlands provide a variety of ecosystem services including regulation of watershed-scale hydrological connectivity (Volik et al., 2024) and sequestration of globally significant quantities of carbon (Yu, 2012). They cover much of Canada's boreal region, including on the Western Boreal Plain, where oil sands exploration and extraction have resulted in the construction of an extensive network of industrial features (Volik et al., 2020). As peatlands are characterized by near-surface water tables (WTs), features called well pads are constructed to provide an elevated base when in-situ oil drilling and extraction is undertaken in these environments. Well pads are constructed with mineral fill (clay or sand) which is deposited in layers of one to four metres thick either directly onto the peat surface or onto impermeable or semi-impermeable geotextile liners (M. E. Gauthier et al., 2018).

The peat underlying features constructed out of mineral fill (including well pads and roads) undergoes compaction (Lemmer et al., 2022; Xu et al., 2022), reducing hydraulic conductivity at depth relative to adjacent peatlands (Elmes et al., 2021b). Because the mineral fill itself also replaces the uppermost layer of the peat profile, near-surface hydrophysical properties are considerably altered (e.g., increased bulk density and reduced hydraulic conductivity), which creates a barrier to lateral flow (Saraswati et al., 2020b). This disruption to flow can disrupt ecohydrological function (Williams-Mounsey et al., 2021) and result in increases in peat bulk density on the downgradient side due to deformation of the peat structure (Elmes et al., 2021b). Notably, the ecohydrological effects of roads have been observed in peatland environments as varied as boreal bogs and fens (Bocking et al., 2017; Elmes et al., 2022; Saraswati et al., 2020a), blanket bogs (McKendrick-Smith, 2016), and mountain bogs and fens (Chimner et al., 2010; von Sengbusch, 2015). Little work has been done to specifically quantify the ecohydrological effects of well pads (which affect boreal peatlands), although they may be similar to those of roads (Volik et al., 2020).

In light of the potential negative impacts of well pads, there is a need for further detailed assessment of their impact on boreal peatland hydrology (Volik et al., 2020). This is especially critical in light of the fact that approximately 36,000 ha of well pads have been built in peatlands in western Canada as of 2019 (UNEP, 2022), with an estimated 650 ha of additional pads constructed in peatlands in the region annually (Drever et al., 2021). Further, oil and gas producers in the Canadian Province of Alberta are required by law to return well pads to a state of “equivalent land capability” following their decommissioning (Powter et al., 2012). This process was recently defined in provincial-level guidelines for sites within peatlands as the re-establishment of peatland ecosystem function within the footprint of decommissioned pads, including the establishment of a peatland vegetation community (Alberta Environment and Parks, 2017). However, well pads in peatlands have often been reclaimed as uplands due in part to a lack of proven peatland restoration techniques targeted for use on these features.

Recently, several novel techniques to reestablish peatland vegetation on sites formerly occupied by well pads have been tested. These include the use of a modified version of the Moss Layer Transfer Technique (MLTT) originally developed for use on cutover peatlands (Rochefort et al., 2003). One approach to using the MLTT on former well pads involves the introduction of peatland donor mosses onto organic substrates exposed through the complete removal or inversion of the mineral fill (Xu et al., 2022). Notably, this approach has been shown to improve subsurface flow through the surrounding peatland as a result of increases in near-surface hydraulic conductivity (Elmes et al., 2021b). However, complete removal or burying of the fill appears to be best optimized for the establishment of vegetation communities that are characteristic of ombrotrophic peatlands, given the low mineral content of the re-exposed organic substrates (Xu et al., 2022). Given that 63% of peatlands in western Canada are minerotrophic fens (Vitt et al., 2000), an alternative restoration technique has been proposed in which a well pad is only partially removed to lower its surface to the average elevation of the WT in the surrounding peatland. This leaves a residual clay or sand substrate with a high mineral cation content in contact with mosses introduced onto the surface using the MLTT (Vitt et al., 2011).

The partial removal approach is based on the natural process of paludification, whereby terrestrial mineral soils become saturated enough for the establishment of early fen successional species (Ruppel et al., 2013). Specifically, the technique aims to establish sufficient hydrological connectivity between a residual pad and adjacent peatlands for the maintenance of a near-surface WT and near-saturated substrate moisture conditions for most of the growing season. The maintenance of near-saturated substrate moisture levels is critical for fen moss initiation, as peatland mosses rely on the capillary connection between soil pores and the spaces formed between individual moss branches to access water (Goetz and Price, 2015; McCarter and Price, 2014). The strength of this capillary connection is dictated both by the physical properties of the establishing moss community (T. L. J. Gauthier et al., 2018; Ketcheson and Price, 2014) and the physical properties and water content of the underlying substrate (Graf and Rochefort, 2010; Price and Whitehead, 2001).

Initial trials of fen vegetation initiation on mineral fill materials yielded promising results at the mesocosm scale (Amon et al., 2005; Borkenhagen and Cooper, 2016) as well as on small partially-removed experimental sections along the edges of several well pads (M. E. Gauthier et al., 2018; Lemmer et al., 2022; Vitt et al., 2011). On sections of pads where the lowered surface was within a few centimetres of the average peatland WT, hydrological connectivity with adjacent peatlands resulted in well-regulated, near-surface WTs and high fill material moisture content (Lemmer et al., 2022). This supported the initiation of fen vegetation communities similar in composition to reference fens, whereas deeper WTs associated with higher reprofiled surface elevations (5 and 15 cm above the WT) hindered long-term peatland succession (M. E. Gauthier et al., 2018; Lemmer et al., 2022). Peatland moss establishment was also supported through the creation of microtopographic variability on residual mineral pads, as this resulted in the availability of microsites along a moisture gradient (Borkenhagen and Cooper, 2016; Lemmer et al., 2022). These trials demonstrate the feasibility of fen vegetation initiation on residual mineral fill materials at an experimental scale.

However, it has remained unclear whether hydrological connectivity with an adjacent peatland would be sufficient to maintain optimal substrate moisture levels following partial removal of a full-scale well pad. In particular, there is uncertainty about how much water interior areas on residual well pads would receive as a result of subsurface connectivity with adjacent peatlands, given the likelihood of reduced subsurface flow through the mineral fill left in place. It is also critical to understand the impact of well pads on the hydrology in the peatland ecosystems where they are constructed. Thus, the objectives of the present study are to:

- 1) characterize the hydrophysical properties of the residual mineral fill on a partially removed well pad and peat from the adjacent peatland; and
- 2) assess the relationship between substrate hydrophysical properties and hydrological connectivity through the surrounding peatland and across the residual well pad.

2.2. Methods

2.2.1. Study Site

The study was conducted on a 0.8 ha decommissioned well pad surrounded by a moderate-rich fen located to the northeast of Slave Lake, Alberta, Canada (55° 19' 11" N, 114° 28' 22" W). The fen was characterized by *Picea mariana*, *Salix* spp., *Pleurozium schreberi*, *Sphagnum angustifolium*, *Rhododendron groenlandicum*, *Rubus chamaemorus*, and *Carex aquatilis*. The regional climate is characterized by mean annual precipitation (rainfall + snowfall) of 462 mm, 68% of which is received as rainfall during the growing season (May-September) (Wabasca Station, 1981-2010; Environment and Climate Change Canada, 2024a). The mean daily air temperature during the growing season over the same 30-year period was 13.3°C, while the mean annual air temperature was 1.7°C (Environment and Climate Change Canada, 2024a).

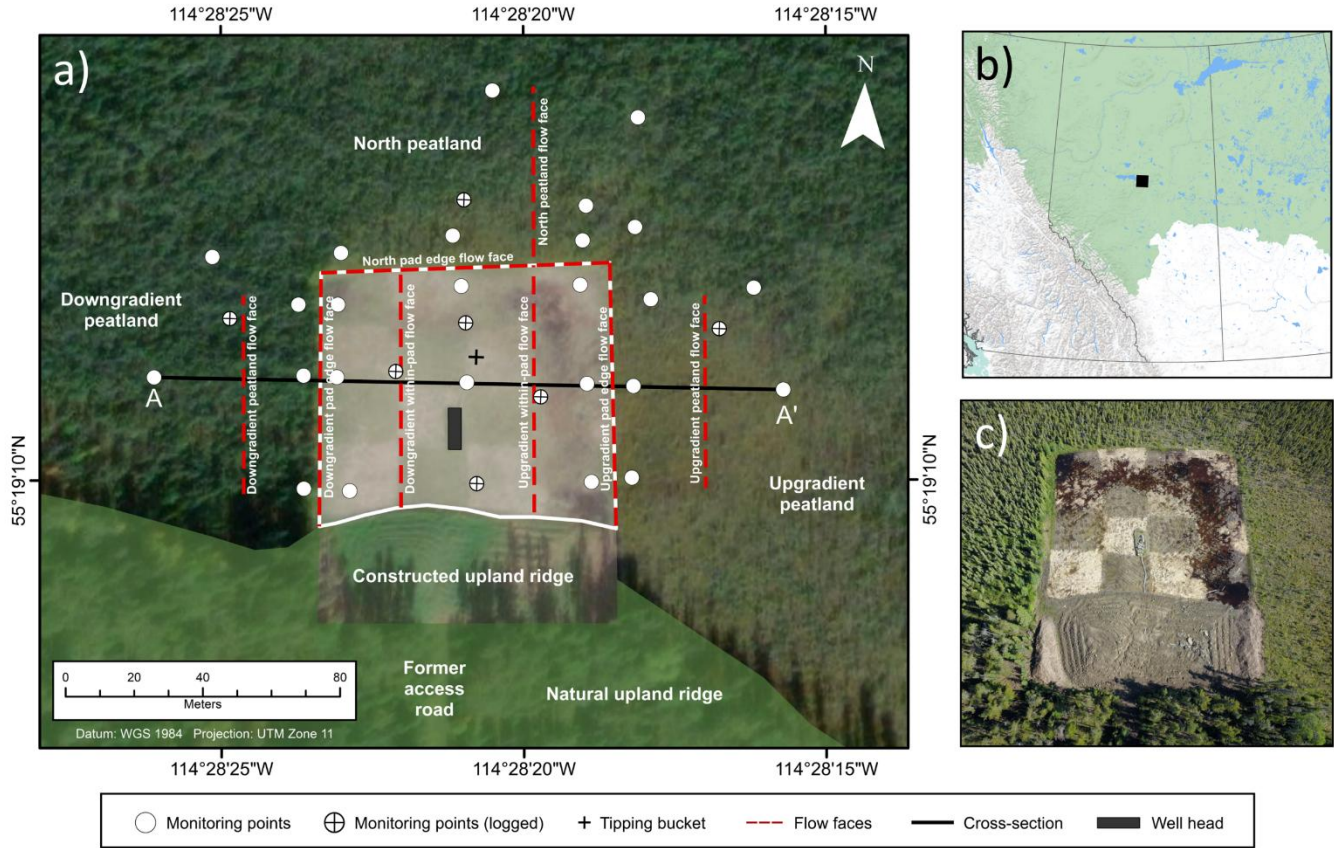


Figure 2.1: Map of the study site including boundaries of the residual pad and constructed and natural upland ridges, location of hydrological monitoring points, cross-section location, and flow faces used for quantification of horizontal gradients and lateral subsurface flow (a). Hollow circles denote locations with manual WT measurements, while crossed circles denote locations where WT loggers were installed. Location of the study site in the boreal region within the province of Alberta, Canada (b), and aerial image of the residual pad and constructed ridge three months following partial removal (June 2020) (c).

The pad was constructed in 1991 through the placement of mineral fill onto the peat surface without a geotextile liner. It was left intact following initial decommissioning in 1996 (Alberta Energy Regulator, 2024) and was partially removed in March 2020 by lowering its surface with heavy machinery (Supplementary Fig. A.1). In doing so, a layer of mineral fill approximately 50 cm thick was removed and was reprofiled into an extension of a natural upland ridge along the south edge of the pad to prevent the need for off-site disposal (Figure 2.1). The target elevation of the reprofiled pad surface was set at 20 cm below the elevation of the seasonal ground ice observed in the adjacent peatland at the time of field operations. The elevation of the seasonal ground ice in the peatland was used as a reference because it was

assumed to be representative of the position of the late-season water table prior to freezing. A depth of 20 cm below the late-season water table was selected with the intention of ensuring the reprofiled surface would remain saturated under late-season conditions, even in drier-than-average years. As the seasonal ground ice was observed to be 10 cm above the bottom of peatland hollows, the target reprofiled surface elevation corresponded with a depth of 10 cm below the hollows.

Following partial removal, surface microtopographic variability was created on the reprofiled surface using the teeth of a bucket excavator. This was followed by the introduction of true moss diaspores collected from a nearby rich fen using a modified version of the MLTT (Supplementary Figs. A.2 & A.3). Donor mosses were collected and applied at an aerial ratio of 1:7 (donor site area to well pad area), a rate that was slightly higher than that typically used on cutover peat surfaces (Rocheffort et al., 2003). As part of a related study, the residual pad was subsequently divided into 16 experimental plots, half of which were covered with a layer of straw mulch (Figure 2.1c; Supplementary Fig. A.3). In support of the assessment of site hydrological dynamics, the thickness of the residual mineral fill material left in place was determined with a ruled tape in three small holes augured adjacent to each hydrological monitoring point on the pad (n=36; Figure 2.1). The depth to the top of the basal mineral substrate underlying the peatland was determined using a graduated steel rod at all monitoring points. Surface elevations of low and high microsites on the residual pad and in adjacent peatlands were determined in August 2020 using a real-time kinematic GPS (RTK) (0.005 m mean vertical accuracy; S321+, Hemisphere GNSS, Scottsdale, USA).

2.2.2. Hydrophysical Characterization

The Bouyoucos hydrometer method (Bouyoucos, 1962) was used to determine the texture of disturbed samples of the residual mineral fill (n=12) and the basal mineral substrate underlying the peatland (n=9) in 2020. Additional samples of the residual mineral fill were collected from the pad using 250 cm³ soil rings with a length of 5 cm (n=12). These were also used to collect peat from hollows in the upgradient (n=3),

north (n=3), and downgradient (n=3) peatlands. Analyses for bulk density, total porosity (calculated from particle density and bulk density), and drainable porosity were performed on each core using standard methods (Freeze and Cherry, 1979). Particle density was determined using the volumetric flask method outlined by Blake and Hartge (1986). Specifically, 50 g samples of oven-dried mineral substrate or 10 g samples of oven-dried peat were placed in a 100 mL volumetric flask and weighed. Flasks were subsequently filled with 50 mL of deionized water and boiled to remove air from the substrate. Flasks were then cooled and topped up with deaired deionized water to 100 mL total volume (substrate + deionized water) and weighed. A glass thermometer was used to measure the temperature of each flask to permit temperature-specific volume correction. Particle densities were then calculated based on the weights of the empty (air-filled) flask, water-filled flask, flask containing only substrate, and water and substrate-filled flask (Blake and Hartge, 1986). Vertical saturated hydraulic conductivity (K_v) was determined for each core using a constant head permeameter for peat cores (n=9) and an automated falling head permeameter for mineral cores (n=12) (KSAT, Meter Group, Pullman, USA). Subsamples of each core (10 g) were oven dried for 24 hours at 105°C (mineral subsamples) or 80°C (peat subsamples) prior to the estimation of organic matter content through loss-on-ignition (550°C for 4 hours; Heiri et al., 2001). Additional horizontally-oriented cores of the same volume as vertical cores were taken to determine horizontal saturated hydraulic conductivity (K_h) using the falling head permeameter (n=12). The anisotropy of the mineral fill was calculated as K_h/K_v for each pair of cores (n=12).

The K_h was also determined in-situ at each monitoring point using bail tests performed on piezometers (2.5 cm i.d.) with 20 cm intake screens centred at depths of 25, 50, and 100 cm below ground surface. Screens were covered with filter sock (Nilex, Edmonton, Canada), and piezometers were developed prior to measurement (SurrIDGE et al., 2005). The hydrostatic time-lag method was used for analysis (Hvorslev, 1951), and the geometric mean of all K_h tests conducted at a given depth in each of four areas (the upgradient, north, and downgradient peatlands, and the pad; Figure 2.1) was taken for comparison. All bail tests with recovery <90% were excluded from the data analysis. Measurements of in-situ infiltration

rate were completed on the reprofiled pad (n=12) using an automated dual head infiltrometer (SATURO, Meter Group, Pullman, USA) during dry conditions in September 2021.

2.2.3. Hydrological Connectivity

Manual measurements of depth to water table (DWT) were taken monthly during the 2020 (June 15 to August 14) and 2021 (May 16 to September 14) study periods in fully slotted wells (5.1 cm i.d.), extending to a depth of 1 m below ground surface at each monitoring point. Half-hourly DWT was recorded using pressure transducers (HOBO U20L, Onset, Bourne, USA) during both seasons in a subset of four wells on the pad and one well in each peatland (Figure 2.1). Due to logger installation errors in 2021, DWT values exceeding between -37.5 and -75 cm (depending on the well) were not recorded on the pad. All DWT values were converted to absolute WT elevations using the elevation of well tops, determined in 2020 using the RTK (0.006 m mean vertical accuracy). WT elevations were used to calculate lateral hydraulic gradients and to develop maps of hydraulic head using inverse distance weighted interpolation (power = 4, search radius = 50 m) in ArcMap (ESRI, version 10.8.1). Vertical hydraulic gradients were calculated monthly using head values in all piezometers.

Subsurface flow was quantified across the east, north, and west pad edges, as well as across flow faces extending through the east and west halves of the pad and through each peatland area (Figure 2.1). When manual WT measurements were available, specific discharge (q) was calculated using Darcy's Law:

$$q = -K_h \frac{dh}{dl} \quad (2.1)$$

where dh/dl is the average horizontal hydraulic gradient across a flow face on a given day (Freeze and Cherry, 1979). To account for heterogeneity in the vertical distribution of hydrophysical properties within the soil profile (Quinton et al., 2008), an equivalent horizontal hydraulic conductivity was calculated for the soil profile at each flow face each day for use in Eq. 2.1:

$$K_h = \sum_{i=1}^n \frac{K_i d_i}{d} \quad (2.2)$$

where d is the height of the saturated soil profile (the mean distance between the WT and basal mineral substrate underlying the peatland), K_i is the hydraulic conductivity of a depth interval of soil within that profile, and d_i is the height of that depth interval, excluding any proportion of that interval which is above the WT (Freeze and Cherry, 1979). As a form of weighted averaging, this standard approach uses the proportion of the saturated profile made up of each depth interval as a weighting factor which is multiplied by the K_h of that layer or interval. For calculations across flow faces located within a given peatland, the K_h measured using bail tests in piezometers centred at 25 and 50 cm in that peatland were applied to the 0-40 and 40-75 cm depth intervals, respectively. The K_h measured in piezometers centred at 100 cm in a given peatland was applied to the interval between 75 cm and the basal mineral soil underlying that area. For calculations across flow faces located within and along the edges of the pad, the K_h of the mineral fill over the 0-8 cm depth was applied to the 0-10 cm interval. The interval from 10 cm to the base of the pad was assigned the K_h of the mineral fill measured using bail tests at 25 cm. The K_h of peat measured using piezometers centred at the 50 and 100 cm depths were applied to the intervals between the bottom of the pad and 75 cm, and between 75 cm and the basal mineral soil, respectively. The equivalent horizontal hydraulic conductivity of the soil profile within the pad footprint was used for fluxes across pad edges, as it was lower than that within all of the surrounding peatland areas, thus constraining the maximum rate of flow across any given pad edge.

Volumetric flow rates (Q) were calculated for flow faces located within and along the edges of the pad:

$$Q = q \cdot A \quad (2.3)$$

where A is the cross-sectional area of a given flow face (Freeze and Cherry, 1979) and q is the specific discharge (Eq. 1). The outer bounds of each flow face were defined as the length or width of the pad (Figure 2.1), while the height of the saturated profile was taken as the height of the flow face.

2.2.4. Rainfall

Total hourly rainfall was measured using a tipping bucket installed near the centre of the pad (ECRN-100, Decagon, Pullman, USA). Due to logger failure, site-level rainfall data was unavailable for 24 of the 53 days in the 2020 study period. A linear regression model (Allen et al., 1998) was developed between site data and data from the Slave Lake airport (~20 km away) for the period the logger was functional. The regression ($R^2 = 0.90$) was used to gap fill missing rainfall data for the site.

2.2.5. Data Analysis

Parametric t-tests or analysis of variance (ANOVA) were used to identify inequality between group means when data met the normality assumption, as assessed using Shapiro-Wilk tests and graphical inspection of residual distribution. Logarithmic transformations were applied when this assumption was not met. When transformations were ineffective, non-parametric Kruskal-Wallis tests were conducted. Post-hoc analyses were conducted using either Tukey's HSD (parametric) or Dunn's (non-parametric) tests. All analyses were completed using R (v4.3.1; R Core Team, 2022) with $\alpha = 0.05$ taken as the threshold for significance.

2.3. Results

2.3.1. Rainfall

The Lesser Slave Lake region received 286 and 244 mm of rainfall between May 1 and September 30 in 2020 and 2021, respectively, representing 91% and 77% of 30-year regional average rainfall over the same period (Slave Lake Station; Environment and Climate Change Canada, 2024b). Reflecting this pattern, regional mean daily rainfall was significantly lower in the second season than in the first (Kruskal-Wallis, $p < 0.001$), and 2020 and 2021 are accordingly referred to herein as 'typical' and 'dry' years, respectively. Despite the differing total rainfall amounts, most growing season rainfall was received before July 15 in

both years, with 70% and 64% received before this date in 2020 and 2021, respectively. During the drier 2021 study period, individual site-measured rainfall events were generally of low magnitude (median = 1 mm), short duration (median = 2 hrs), and low intensity (median = 0.5 mm hr⁻¹). However, infrequent large rainfall events delivered more than 10 mm day⁻¹ on several dates during both seasons.

2.3.2. Residual Pad Surface Elevation

As targeted, partial removal resulted in a relatively level reprofiled surface (σ of low microsites = 9 cm), with the mean low microsite elevation falling 9 and 11 cm lower than that of hollows in the upgradient and

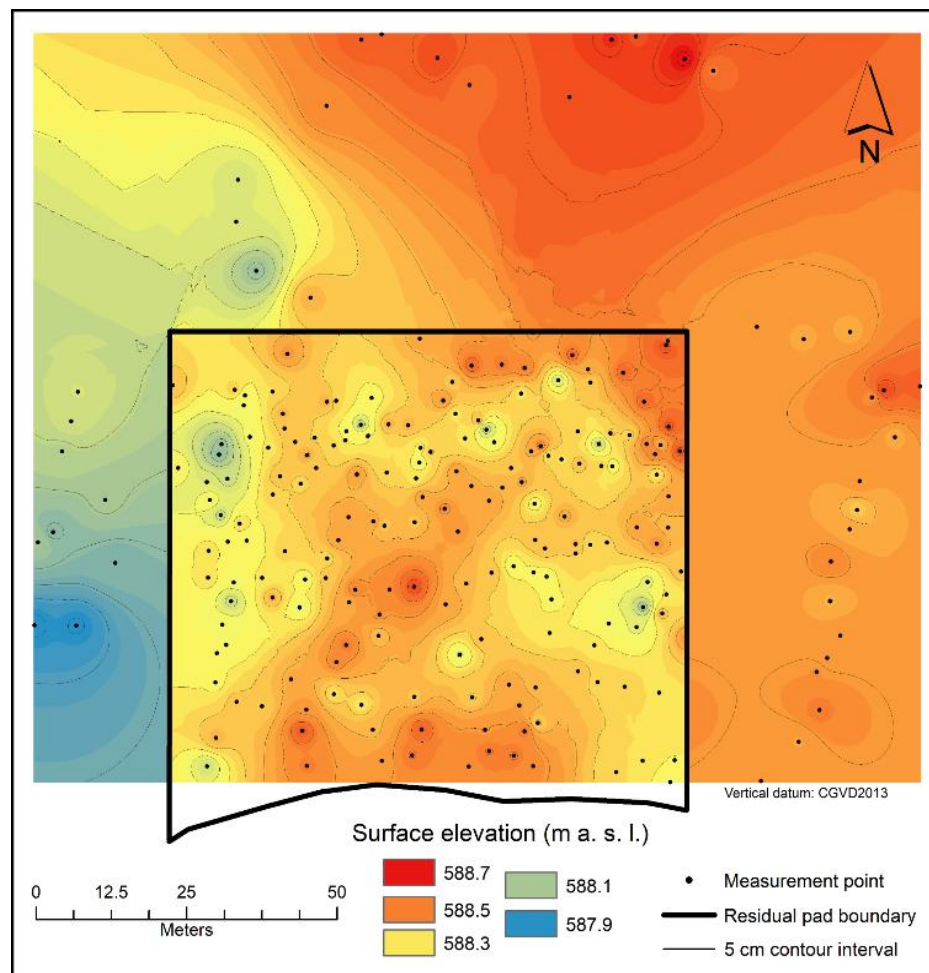


Figure 2.2: Inverse distance weighted interpolation map of the residual pad surface elevation following partial removal (metres above sea level; measured at low microsites) and peatland surface elevation (metres above sea level; measured in hollows).

north peatlands, respectively (Figure 2.2). Notably, however, the average surface elevation of hollows in the downgradient peatland (588.23 m above sea level; m a. s. l.) was slightly more than 30 cm lower than those in the upgradient (588.55 m a. s. l.) and north (588.57 m a. s. l.) peatlands. As a result, the mean elevation of low microsites on the pad was 23 cm higher than the mean surface elevation in hollows in the downgradient peatland (Figure 2.2), which falls considerably higher than the expected late-season WT position in that area. Surface roughing resulted in microsites with a mean amplitude of 19 cm. Notably, mean microsite amplitude was observed to have decreased by 6 cm by the end of the second growing season due to settling of the fill.

2.3.3. Substrate Hydrophysical Properties

2.3.3.1. General Characterization

The mean thickness of the residual pad was 37.5 cm, although it was spatially variable, ranging from 15 to 69 cm. The underlying peat had a mean thickness of 81 cm, which was again variable, ranging from 23 to 118 cm. The mean depth of the peat profile was 104, 100, and 113 cm in the upgradient, north, and downgradient peatlands, respectively. The basal mineral substrate underlying the peat profile was classified as clay loam, while the residual mineral fill was classified as loamy sand and had a low mean organic matter content (2.1%; Table 2.1). Porosity of the upper 5 cm of the mineral fill was considerably lower than that of peat from adjacent peatlands over the same interval (0.38 compared to 0.95-0.97), and it had a higher particle density (2.54 g cm⁻³ compared to 1.45-1.65 g cm⁻³; Table 2.1). Similarly, the mineral fill had a considerably higher bulk density (1.57 compared to 0.05-0.08 g cm⁻³) and lower drainable porosity (0.22 compared to 0.44-0.49; Table 2.1) relative to the peat substrate. Particle and bulk densities were significantly higher at 0-5 cm in the downgradient peatland than in the upgradient peatland (ANOVA, $p = 0.008$ and 0.01 , respectively), while the opposite was true for porosity ($p = 0.04$). However, it is important to note that the magnitudes of these differences were limited (Table 2.1).

Table 2.1: Summary of the hydrophysical properties of the residual mineral fill, near-surface peat collected from hollows in the adjacent peatlands, and the underlying basal mineral substrate. Arithmetic or geometric standard deviations are reported in brackets following arithmetic and geometric means, respectively.

Location/ substrate	Depth (cm)	Bulk density (g cm ⁻³)	Particle density (g cm ⁻³)	Total porosity	Drainable porosity	Organic matter content (%)	K _h (m s ⁻¹) *	K _v (m s ⁻¹) *	Infiltration rate (m s ⁻¹) *	Anisotropic ratio (K _h /K _v)	Sand (%)	Silt (%)	Clay (%)
Residual mineral fill	0-5	1.57 (0.10)	2.54 (0.04)	0.38 (0.04)	0.22 (0.04)	2.1 (0.7)	5.20 x 10 ⁻⁶ (1.91)	4.77 x 10 ⁻⁶ (1.59)	7.35 x 10 ⁻⁶ (2.41)	1.43 (1.24)	81.3 (3.9)	15.7 (3.8)	3.0 (1.7)
Upgradient peatland	0-5	0.05 (0.01)	1.45 (0.07)	0.97 (0.01)	0.46 (0.08)	95.3 (0.9)	-	2.41 x 10 ⁻³ (1.09)	-	-	-	-	-
North peatland	0-5	0.07 (0.02)	1.55 (0.05)	0.97 (0.01)	0.44 (0.06)	93.7 (0.8)	-	3.05 x 10 ⁻³ (1.18)	-	-	-	-	-
Downgradient peatland	0-5	0.08 (0.03)	1.65 (0.14)	0.95 (0.01)	0.49 (0.12)	90.6 (3.9)	-	2.35 x 10 ⁻³ (1.10)	-	-	-	-	-
Basal mineral substrate	150	-	-	-	-	-	-	-	-	-	40.6 (14.5)	31.7 (9.8)	27.8 (5.5)

*Geometric means and standard deviations (all others are arithmetic means and standard deviations)

2.3.3.2. Hydraulic Conductivity and Infiltration Rate

In all peatland areas, K_h decreased with depth below the surface (Figure 2.3). This was also observed for the residual pad and underlying peat, with the exception that K_h of the mineral fill over 0-5 cm was significantly lower than that measured over 15-35 cm (t-test, $p < 0.001$; Figure 2.3). As expected, the K_h of the mineral fill over the 15-35 cm depth was significantly lower than K_h in all peatlands over the same depth (Kruskal-Wallis, $p < 0.001$), while it did not vary significantly between peatlands. Over the 40-60 cm depth, the K_h of the peat underlying the pad and in the downgradient peatland did not significantly differ (Kruskal-Wallis, $p > 0.05$), but they were both significantly lower than K_h over that depth in the upgradient and north peatlands ($p < 0.001$). No significant difference in K_h was observed anywhere on the site over the 90-110 cm depth (Kruskal-Wallis, $p > 0.05$) (Figure 2.3; Supplementary Table A.1). Notably, the infiltration rate (geometric mean = $7.35 \times 10^{-6} \text{ m s}^{-1}$) and K_v (geometric mean = $4.77 \times 10^{-6} \text{ m s}^{-1}$) on the pad did not differ significantly (t-test; $p = 0.15$).

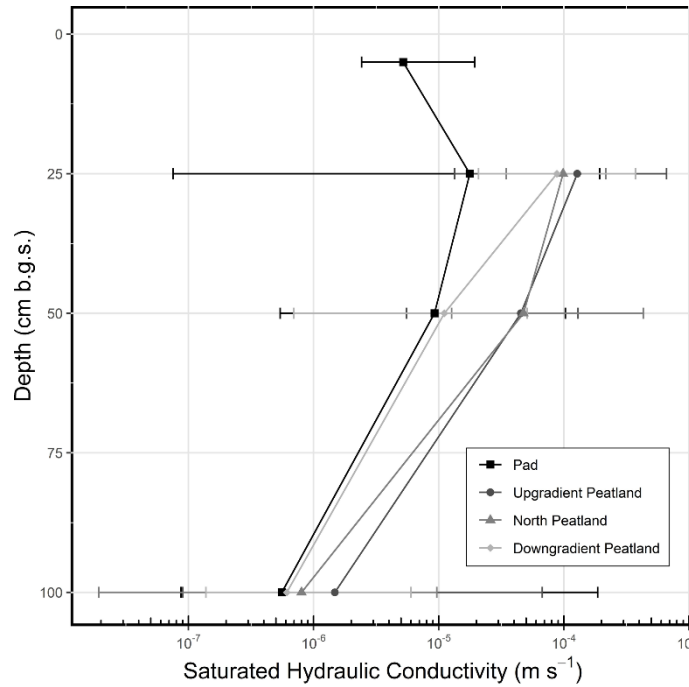


Figure 2.3: Geometric mean K_h by depth below ground surface. Horizontal bars illustrate the range of measured values, with outer bounds defined by minimum and maximum values. Pad values for 0-5 and 25 cm b.g.s. represent K_h of the residual mineral fill, while values at 50 and 100 cm b.g.s. represent K_h of the underlying peat.

2.3.4. Peatland Hydrological Dynamics

2.3.4.1. Peatland Hydraulic Gradients and Subsurface Flow

Horizontal hydraulic gradients in the peatland were generally directed from east to west (Figure 2.4b). The seasonal mean horizontal gradient through the upgradient and north peatlands in the 2020 season was -0.001, which corresponded with mean specific discharges of 4.7 and 4.5 mm day⁻¹, respectively (Figure 2.5i,m). In the downgradient peatland that year, a slightly larger seasonal mean horizontal gradient of -0.002 was unexpectedly associated with a lower mean specific discharge of only 3.9 mm day⁻¹ (Figure 2.5i,m). Patterns were similar in the drier 2021 season. Specifically, a seasonal mean horizontal gradient of -0.002 in both the upgradient and north peatlands was associated with mean specific discharge of 5.0 and 6.7 mm day⁻¹, respectively (Figure 2.5k,o). Notably, small reversals in the direction of the horizontal

gradient in the downgradient peatland late that season (from +0.002 to +0.004; Figure 2.5k) resulted in pad-directed specific discharge on several dates (mean = 0.73 mm day⁻¹; negative values on Figure 2.5o).

Interestingly, despite larger horizontal gradients observed across the upgradient pad edge (between -0.005 and -0.014) than through the north peatland (between -0.001 and -0.005) on all measurement dates (Figure 2.5i-l), specific discharge across that edge (1.1-3.5 mm day⁻¹) was always lower than through the north peatland (3.3-12.9 mm day⁻¹) (Figure 2.5m-p). Also of note, the peatland may have been hydrologically connected to groundwater in the basal mineral substrate. For instance, downward-directed vertical hydraulic gradients were usually observed in the upgradient (mean = -0.01) and north (mean = -0.04) peatlands, potentially indicating some degree of groundwater recharge. Conversely, vertical gradients downgradient of the pad were usually surface-directed (mean = 0.01), indicating some degree of groundwater discharge.

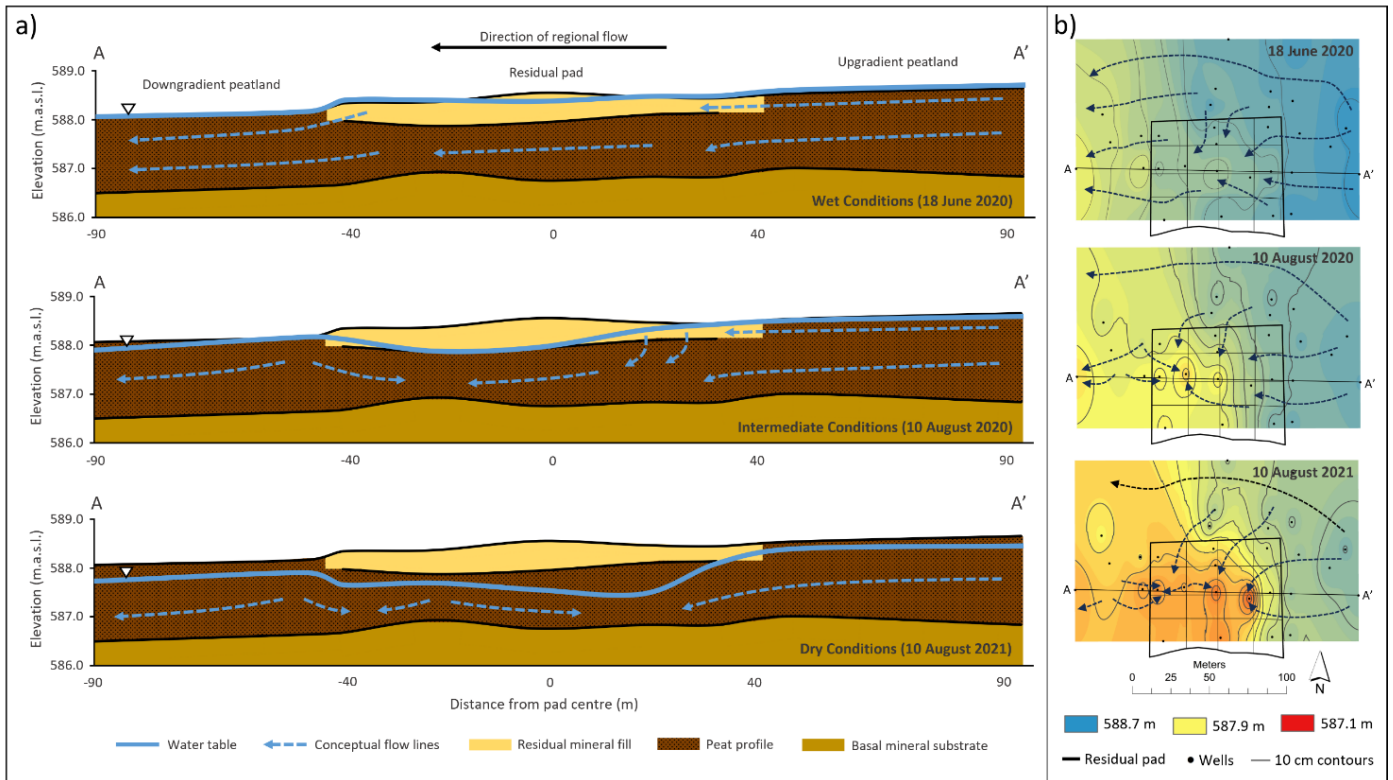


Figure 2.4: Cross-sectional diagrams (vertical exaggeration = 10) of the residual pad and adjacent peatlands along transect A-A' under representative wet (18 June 2020), intermediate (10 August 2020) and dry (10 August 2021) conditions, including water table elevation and conceptual (qualitative) flow lines (a). Interpolation maps of WT elevation (equipotential surface) for each date, including conceptual (semi-quantitative) flow lines (b). Regional flow direction is from right to left for both (a) and (b).

2.3.4.2. Peatland Water Table Dynamics

The WT in the peatland was well-regulated during the first, typical season (seasonal σ DWT < 0.04 m at logged wells; Supplementary Table A.2). The lowest WT was consistently observed in the downgradient peatland (Figure 2.4), where it remained just below the surface of peatland hollows for the full duration of the season (Figure 2.5e). Comparatively, peatland WT dynamics were more poorly regulated during the second, drier growing season (seasonal σ DWT > 0.10 m at logged wells) (Figure 2.5c,g). During this season, DWT was again greatest in the downgradient peatland, where it exceeded -0.50 m for a brief period in August (Figure 2.5g). Conversely, the WT in the upgradient peatland remained closer to the surface and reached a maximum depth of only -0.23 m (Figure 2.5g). Combined with the difference in surface elevation between the upgradient and downgradient peatlands, this resulted in considerable differences in absolute WT elevation of between 0.45 and 0.61 m between those two areas on dates when manual WT measurements were available (Figure 2.4b).

2.3.5. Residual Pad Hydrological Dynamics

2.3.5.1. Residual Pad Subsurface Fluxes: Wet Conditions

Under early season conditions in both years, the residual pad received water inputs from the peatland along its upgradient (east) and north sides and lost varying amounts of water to the downgradient (west) peatland (Figure 2.4b). Specifically, combined volumetric flow into the pad footprint (into both the mineral fill and underlying peat) from the upgradient and north peatlands was 0.28 and 0.30 m³ day⁻¹ on each of 18 June and 13 July in the first year (Figure 2.5q), while between 0.31 and 0.44 m³ day⁻¹ was lost across the downgradient edge (Figure 2.5q). At the same time, the mean volumetric flow across the interior of the pad footprint (through both the mineral fill and underlying peat along the upgradient and downgradient within-pad flow faces; Figure 2.1) was relatively small during this period (0.07-0.08 m³ day⁻¹) due to the small magnitude of within-pad horizontal hydraulic gradients (Figure 2.4).

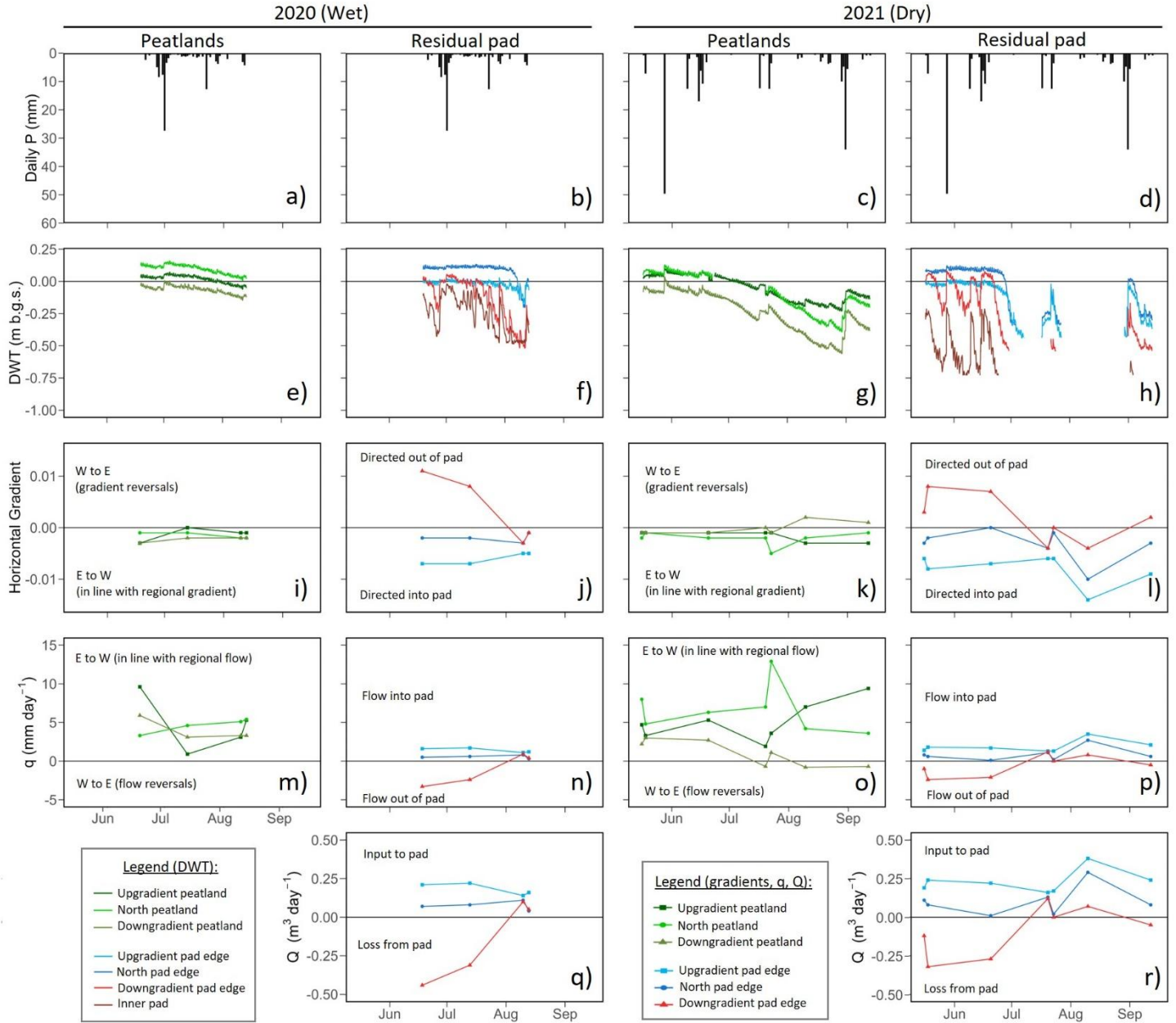


Figure 2.5: Plots of site daily precipitation (P; gap filled for 2020) (a-d), depth to water table (DWT; e-h), horizontal hydraulic gradients (i-l), horizontal specific discharge (q) (m-p), and horizontal volumetric flow rates (across pad edges; Q) (q-r), separated by growing season and location (adjacent peatlands or the residual pad). Negative horizontal gradients denote gradients either directed along regional flow paths (east to west; peatland fluxes) or directed into the pad (pad edge fluxes). Positive q and Q denote flow either in the direction of regional flow (east to west; peatland fluxes) or onto the pad (pad edge fluxes).

Similar patterns were initially observed towards the start of the second, drier season, with the pad footprint receiving a combined input of $0.30 \text{ m}^3 \text{ day}^{-1}$ from the upgradient and north peatlands on 16 May (Figure 2.5r). However, the loss to the downgradient peatland was, in this case, restricted to $0.12 \text{ m}^3 \text{ day}^{-1}$ as a result of the lower WT in the interior of the pad than that observed in the first season (Figure 2.5r). Following a 7 mm rainfall event on 18 May, water loss from the pad footprint to the downgradient peatland nearly tripled to $0.32 \text{ m}^3 \text{ day}^{-1}$ as a result of a rapid increase in WT elevation on the pad (Figure 2.5h,r). Conversely, inputs across the upgradient and north edges remained approximately the same following the rainfall event (Figure 2.5r).

2.3.5.2. Residual Pad Subsurface Fluxes: Intermediate Conditions

In contrast with early-season conditions, the residual pad footprint consistently received subsurface inputs from all sides under slightly drier late-season conditions in the first year (Figure 2.5q). Inputs across the downgradient (west) edge were associated with flow reversals (flow from west to east) across that edge which occurred when the WT within the pad footprint was lower than the WT in the downgradient peatland (Figure 2.4a). On 10 August of the first season, for example, a flow reversal across the downgradient pad edge (Figure 2.4) resulted in an input to the pad footprint of $0.10 \text{ m}^3 \text{ day}^{-1}$ from the downgradient peatland (positive values on Figure 2.5q). Highlighting the importance of rainfall in pad hydrological dynamics, an increase in WT elevation in the interior of the pad following a 4 mm rainfall event reduced the magnitude of the flow reversal from 0.10 to $0.05 \text{ m}^3 \text{ day}^{-1}$ (Figure 2.5b,f,q). Within the pad footprint under these slightly drier conditions, the difference in WT elevation between areas adjacent to the upgradient edge and in interior areas (Figure 2.4) resulted in considerably higher subsurface flow across the upgradient within-pad flow face towards the pad centre ($0.27 \text{ m}^3 \text{ day}^{-1}$) than in the early season. However, in contrast with wetter early-season conditions, much of this flow occurred in the peat underlying the residual mineral fill and not in the fill itself.

2.3.5.3. *Residual Pad Subsurface Fluxes: Dry Conditions*

The driest conditions observed within the pad footprint occurred through the mid- to late-season in the second year. Under these conditions, flow reversals across the downgradient pad edge again played an important role in the subsurface water balance. For example, a flow reversal was observed across the downgradient pad edge under dry conditions on 20 July that season (Figure 2.5l), resulting in an input of $0.12 \text{ m}^3 \text{ day}^{-1}$ from the downgradient peatland into the peat underlying the mineral fill (Figure 2.5r). On the same date, input across the upgradient edge ($0.16 \text{ m}^3 \text{ day}^{-1}$) was lower than that observed in the early season but was comparable to that observed under intermediate conditions late in the first season. Similar to observations made under intermediate conditions late in the first, typical season, relatively large volumetric flow rates were observed across the upgradient within-pad flow face under dry conditions. This was again a result of the large difference in WT elevation between the upgradient and inner portions of the pad footprint and resulted in subsurface flow of between 0.36 and $0.26 \text{ m}^3 \text{ day}^{-1}$ across the upgradient flow face.

Notably, the largest inputs from the upgradient ($0.38 \text{ m}^3 \text{ day}^{-1}$) and north ($0.29 \text{ m}^3 \text{ day}^{-1}$) peatlands were measured during the driest conditions observed (10 August 2021) when the WT within the residual pad footprint was at its lowest (Figure 2.4), and horizontal hydraulic gradients across the upgradient and north edges were accordingly large (Figure 2.5l). The subsurface flow associated with these inputs occurred almost exclusively between the adjacent peatlands and the peat underlying the mineral fill, with only minimal flow into the mineral fill itself occurring (Figure 2.4a). This was in contrast with the inputs observed during wet and intermediate conditions, when the upgradient and north edge flow faces were entirely saturated (Figure 2.4a). Concurrently, a flow reversal across the downgradient edge (Figs. 4 & 5l) resulted in an input from the downgradient peatland to the peat underlying the mineral fill of $0.07 \text{ m}^3 \text{ day}^{-1}$, and contributed to the largest net input observed during the study ($0.73 \text{ m}^3 \text{ day}^{-1}$; Figure 2.5r). In contrast, the low WT on the pad adjacent to the upgradient edge (Figure 2.4) reduced volumetric flow across the upgradient within-pad flow face to only $0.19 \text{ m}^3 \text{ day}^{-1}$, all of which was through the underlying peat.

2.3.5.4. *Residual Pad Water Table Dynamics*

In line with the consistent subsurface inputs received by areas of the pad footprint adjacent to the upgradient and north peatlands in the first, typical season, the WT along those edges remained at or just below the surface (and within the residual mineral fill) for the duration of that season (seasonal mean DWT = -0.02 and +0.09 m for logged wells) (Figure 2.5f; Supplementary Table A.2). At these locations, the WT was well regulated (seasonal σ DWT = 0.03 and 0.07 m) and did not demonstrate substantial responses to rainfall until late in the season (Figure 2.5f). Conversely, WTs in the downgradient and inner areas of the pad footprint were consistently deeper (seasonal mean DWT = -0.13 m and -0.26 m for logged wells) and were more poorly regulated for the duration of the season (seasonal σ DWT = 0.17 and 0.14 m; Figure 2.5f). Notably, the WT in these interior areas did not generally fall below the interface between the residual mineral fill and the underlying peat in the first season (Figure 2.4a).

Similar patterns were observed for downgradient and inner areas of the pad during the second, drier season. However, DWT was greater owing to the drier conditions (mean DWT = -0.19 and -0.51 m for logged wells), with the WT regularly falling below the mineral fill and into the underlying peat (Figure 2.4a). For example, DWT at the inner logged well notably reached -0.7 m as early as late May (Figure 2.5h). Although the WT in upgradient and north areas of the residual pad footprint remained closer to the surface (seasonal mean DWT = -0.13 and -0.02 m for logged wells; Figure 2.4v), it also became poorly regulated by late June following a period with no rainfall (Figure 2.5h). Notably, it also fell below the interface between the mineral fill and the underlying peat for a period in August (Figure 2.4a).

Further, responses of DWT to rainfall were of greater magnitude for all wells within the residual pad footprint under drier conditions, with the greatest responses occurring at the downgradient and inner wells late in the second season, where WT elevation increased by up to 0.5 m after rainfall (Figure 2.5h). Downward-directed vertical hydraulic gradients (between -0.005 and -0.17) were also observed under all conditions in both seasons between the 25 cm depth within the residual pad and the 100 cm depth in the underlying peat. This indicates that some degree of vertical flow out of the residual mineral fill and into the

underlying peat contributed to poor WT regulation whenever the WT was situated within the residual fill (Figure 2.4a).

2.4. Discussion

2.4.1. Substrate Hydrophysical Properties

The mean K_h of the residual mineral fill left in place following partial pad removal ($1.15 \times 10^{-5} \text{ m s}^{-1}$) was as much as an order of magnitude lower than that measured over the 15-35 cm depth in the adjacent surrounding peatland ($8.8 \times 10^{-5} - 1.28 \times 10^{-4} \text{ m s}^{-1}$). Notably, an even greater difference in K_h could be expected between the residual mineral fill and adjacent peat within the upper 15 cm of the soil profile, where peat K_h values greater than $1.0 \times 10^{-3} \text{ m s}^{-1}$ are commonly reported (Elmes and Price, 2019; Quinton et al., 2008; Waddington et al., 2015). The increase in K_h of the mineral fill with depth may have been a result of mixing between deeper fill and the underlying peat during pad construction as no geotextile liner was used or may have been a result of non-uniform settling of the peat and mineral fill over time (O’Kelly and Pichan, 2013). Either way, the hydrological implications of this increase with depth are likely to be limited, given that even the lower layers of fill had a significantly lower K_h than the adjacent peat at the same depth.

Similar to the residual mineral fill, the K_h of the peat underlying the pad over the 40-60 cm depth interval ($9.26 \times 10^{-6} \text{ m s}^{-1}$) was significantly lower than the K_h in the upgradient ($4.54 \times 10^{-5} \text{ m s}^{-1}$) and north ($4.76 \times 10^{-5} \text{ m s}^{-1}$) peatlands over that interval, indicating compaction due to the weight of the mineral fill. This is not unexpected, as peat compaction has been observed in peatlands impacted by the placement of mineral fill for the construction of well pads (Lemmer et al., 2022; Xu et al., 2022) and roads (Elmes et al., 2021b). The fact that no significant difference in K_h was observed between the underlying peat and adjacent peatlands at the 90-110 cm depth suggests compaction was primarily limited to the upper peat profile. This is likely because peat at depth is less porous (and thus less compressible) than near-surface peat due to its

advanced state of decomposition and natural compaction by the weight of the peat profile above (Price, 2003).

Peat bulk and particle densities were slightly but significantly higher in downgradient than upgradient peatland hollows, whereas porosity demonstrated the opposite trend. This may be related to peat deformation downgradient of the pad resulting from the low WT consistently observed there, which would have resulted in desaturation and collapse of soil pores and oxidation of peat near the surface (Price, 2003; Waddington et al., 2015). Notably, however, the bulk density (0.08 g cm^{-3}), particle density (1.65 g cm^{-3}), and porosity (0.95) values obtained in the downgradient peatland are similar to those reported for undisturbed fens in the region ($0.05\text{-}0.09 \text{ g cm}^{-3}$, $1.51\text{-}1.57 \text{ g cm}^{-3}$, $0.95\text{-}0.96$, respectively; Redding & Devito, 2006). A potential explanation for this is that the downgradient peatland may already have been functioning as a peatland margin swamp prior to pad construction, especially given the presence of a tall and well-established tree canopy, observed communities of dry-adapted feathermosses, and its location adjacent to the natural upland ridge (vegetation data not shown; Figure 2.1). Peatland margin swamps are naturally characterized by dense upper-story vegetation and a high density of feathermosses given their location along the transition from dry upland to wet fen ecosystems (Elmes et al., 2021a). In addition, long-term shifts in the vegetation community composition towards a denser upper-story and higher proportion of feathermosses might have occurred since construction of the pad 29 years prior to the study period. Shifts of this nature have been observed downgradient of roads built in peatlands (Miller et al., 2015; Willier et al., 2022). Either way, the high proportion of feathermosses (slow decomposition) and shrubs (provision of a woody lattice structure) in the downgradient peatland may have provided a buffer against more extensive peat deformation than that observed in the downgradient peatland in the present study (Moore et al., 2015).

2.4.2. Peatland Hydrological Connectivity

Preferential flow around the northeast corner of the pad occurred under all conditions observed, as evidenced by larger specific discharge through the north peatland (3.3-12.9 mm day⁻¹) than across the upgradient (east) pad edge (1.1-3.5 mm day⁻¹) on all measurement dates. This occurred despite steeper observed horizontal hydraulic gradients across the upgradient pad edge (between -0.005 and -0.014) relative to the north peatland (between -0.001 and -0.005), indicating preferential flow was a direct result of the low K_h of the residual mineral fill and underlying compacted peat. Of the two materials, the low K_h of the residual mineral fill likely limited subsurface flow to the greatest extent, given that K_h , and thus the magnitude of subsurface flow, is highest near the surface of undisturbed peatlands (Quinton et al., 2008). Interestingly, the observed preferential flow supported maintenance of a consistently high WT in the north peatland, resulting in pad-directed horizontal gradients and subsurface flow across the north pad edge on all measurement dates (0.1-2.7 mm day⁻¹). Thus, preferential flow around the pad appears to have contributed to the near-surface WTs observed along the north edge of the residual pad. Notably, however, the magnitude of preferential flow around the residual pad was likely compounded by the natural ridge to the south, which prevented flow around the southeast corner of the pad and which may have directed water flow towards the north peatland to some extent even prior to pad construction (Figure 2.1). Accordingly, the degree of preferential flow may be more limited around pads that are not connected to upland features. Limited flow through the pad and underlying peat are also likely responsible for the greater DWT in the downgradient peatland compared to that upgradient of the pad. Similar effects have been reported downgradient of roads built perpendicular to flow in peatlands due to the ‘damming’ effect of the mineral fill and underlying compacted peat (Bocking et al., 2017; Saraswati et al., 2020b).

The low WT in the downgradient peatland also likely explains the low K_h over the 40-60 cm depth there ($1.1 \times 10^{-5} \text{ m s}^{-1}$), which did not significantly differ from that of compacted peat beneath the pad. This is again likely due to the deformation of the pore structure associated with reductions in pore pressure, which reduces peat K_h (Price, 2003). Notably, reductions in K_h at depth have also been observed in a

peatland downgradient of a former road (Elmes et al., 2021b). Deformation of the pore structure is also apparent through the lowered surface elevation of the downgradient peatland, which was more than 30 cm lower than that in other peatland areas despite the limited distance between it and those areas (Figure 2.2). Importantly, the lower K_h in the downgradient peatland often resulted in lower specific discharge through that peatland relative to the other peatland areas despite having similar horizontal hydraulic gradients (Figure 2.5). This autogenic limitation on subsurface flow would have limited the rate of WT drawdown in the downgradient peatland (Elmes and Price, 2019; Waddington et al., 2015), in turn acting as a limitation on fluxes out of the pad by restricting the magnitude of peatland-directed gradients and fluxes across the downgradient pad edge. Further, this negative feedback on WT drawdown would have contributed to the flow reversals observed across the downgradient pad edge under dry conditions when WT in the downgradient peatland was higher than that in the downgradient portion of the pad. These in turn resulted in small-magnitude water inputs to the downgradient portion of the pad footprint ($0.4\text{--}1.2\text{ mm day}^{-1}$). Notably, similar gradient and flow reversals have been observed in a peatland downgradient of a road constructed out of mineral fill (Saraswati et al., 2020b).

The low downgradient peatland WT may also have been responsible for (but offset somewhat by) possible groundwater discharge driven by small surface-directed vertical gradients, which were observed under most conditions in this area (0.01-0.03). In groundwater-connected fens, low WT levels can result in groundwater pressures within the peat profile dropping below those of deeper groundwater, leading to surface-directed discharge (Elmes and Price, 2019). However, only limited connectivity with deeper groundwater can be expected in peatlands underlain by fine textured substrates (Ferone and Devito, 2004; Reeve et al., 2000), and it may therefore be of limited importance here. Instead, the observed upward gradients may be a result of localized gradient reversals occurring exclusively within the peat profile, which can occur under dry conditions even in the absence of connectivity with regional or intermediate groundwater flow systems (Devito et al., 1997). Installation of piezometers within the underlying basal mineral substrate would be required to estimate the magnitude of vertical groundwater exchange

confidently, and thus, no conclusions can be drawn about the impact of the pad on vertical groundwater flow at present.

2.4.3. Residual Pad Hydrological Connectivity

The well-regulated hydrological conditions observed along the upgradient and north edges of the pad under most meteorological conditions can be attributed to hydrological connection with adjacent peatlands. Combined volumetric flow across these edges was between 0.24 and 0.31 m³ day⁻¹ under typical wet early season conditions and generally increased to between 0.29 and 0.66 m³ day⁻¹ under the driest conditions observed. This was in response to the steepening of horizontal hydraulic gradients associated with reductions in WT elevation on the pad. This feedback response was sufficient to buffer the WT within approximately 20-40 m of the upgradient and north pad edges against substantial further decreases in elevation, as evidenced by its near-surface position under all but the driest conditions observed (Figs. 4 & 5). The high WT in the upgradient peatland (resulting from water impoundment upgradient of the residual pad) and north peatland (resulting from preferential flow around the residual pad) contributed to this buffering effect by enhancing the magnitude of pad-directed horizontal hydraulic gradients, and thus also that of lateral subsurface inputs.

By the end of both seasons, well-regulated, near-surface WTs on the pad were constrained to the pad edges, and were generally only observed within 20-40 m of the upgradient and north sides of the pad (Figure 2.4). Low WTs in interior areas and along the downgradient edge can be attributed in part to the low fluxes through the pad observed on nearly all measurement dates due to the low saturated profile K_h . Poor WT regulation in these areas was a result of rapid transmission of rainfall to the WT, as rainfall rates (generally < 10 mm hr⁻¹) were normally lower than the mean K_v and infiltration rate of the fill (both > 15 mm hr⁻¹), which had a low water retention capacity (data not shown) and high drainable porosity. Further, consistently downward-directed vertical hydraulic gradients and rapid WT drawdown following late-season

rainfall indicate that drainage to the underlying peat must have been largely unimpeded. Notably, no geotextile liner was present on this site to inhibit drainage. However, these results suggest liners may improve WT regulation within residual mineral pads during dry periods. Combined, the high early-season WTs observed in interior and downgradient areas of the residual pad were likely a result of snowmelt and the greater frequency of rainfall during this period as opposed to being supported by subsurface connectivity with adjacent peatlands.

Also, during the drier second season, the WT regularly dropped below the interface between the residual mineral fill and underlying peat. However, during the first season (which received close to long-term average rainfall), the lowest WT elevations observed closely matched the elevation of the interface (Figure 2.4a). This suggests that the lower specific yield of compressed peat (Elmes and Price, 2019) relative to the sandy mineral fill was an important factor in WT regulation and that pad thickness may be a good predictor of late-season DWT in interior and downgradient areas of residual pads. Conversely, WT elevation in interior areas was not aligned with the surface elevation profile of the residual pad in the late season (Figure 2.4a). Thus, reprofiled surface elevation may be less important for mediating DWT than pad thickness, at least on pads comprised of sandy mineral fill. Also, the volume of lateral subsurface inputs to the pad and underlying peat was generally greater than the volume of subsurface losses (by between 0.17 and 0.73 m³ day⁻¹), except under very wet early season conditions following the spring freshet. While this may be due in part to short-term increases in soil water storage, longer-term declining trends in soil water content (data not shown) indicate evapotranspiration is likely an important term in the pad water balance, which is expected given the regional subhumid climate (precipitation < potential evapotranspiration) (Brown et al., 2010). Evapotranspiration accordingly plays an important role in hydrological dynamics in natural (Petrone et al., 2007) and constructed (Scarlett et al., 2017) peatlands in the region. Also, the greatest net subsurface inputs to the pad notably occurred during the mid-season when regional evapotranspiration rates are generally highest (Volik et al., 2021).

2.4.4. Hydrological Implications for Fen Moss Establishment

Combined, it appears likely that the well-regulated and consistently near-surface WT within 20-40 metres of the upgradient and north edges will make an area covering just under half of the residual pad hydrologically suitable for the establishment of introduced fen mosses. This would be in line with previous studies conducted at an experimental scale on small, partially removed sections along the edges of well pads where well-regulated, near-surface WTs and associated desirable fen moss communities were observed within 30 m of pad edges (M. E. Gauthier et al., 2018; Lemmer et al., 2022; Vitt et al., 2011). Notably, however, peatland mosses are highly sensitive to even small changes in substrate water content (Graf and Rochefort, 2010; Schipperges and Rydin, 1998), and hydrological disconnection between mosses introduced using the MLTT and their underlying substrate has often been reported as a result of a capillary boundary effect (Goetz and Price, 2015; McCarter and Price, 2015). Thus, even in areas with reasonably well-optimized WT dynamics, further detailed assessment of soil moisture dynamics and long-term vegetation establishment outcomes will be required to confirm whether these conditions will promote the development of a desirable peatland moss community in these areas of the site. In contrast, the poorly regulated WT within the interior of the residual pad and towards its downgradient edge are unlikely to be optimized for survival and establishment of the fen mosses introduced there, given the high degree of sensitivity of peatland mosses to the regulation and position of the WT (Ketcheson and Price, 2014; Price and Whitehead, 2001). Areas on partially removed well pads with poorly regulated WTs, which are regularly more than a few centimetres below the surface, have previously been associated with graminoid-dominated vegetation communities lacking the bryophyte layer integral to peatland ecosystem function (M. E. Gauthier et al., 2018; Lemmer et al., 2022).

2.4.5 Study Limitations and Directions for Future Research

Given the poor hydrological regulation consistently observed in inner and downgradient portions of the residual pad through the mid to late season, it appears likely that modifications to the partial well pad removal technique and its application at the site level will be required during future implementations. Such modifications might include shallow trenching across pads to promote surface flow (e.g., Vitt et al., 2011) and more uniform distribution of water across the pad, or installation of subsurface flow conduits (e.g., Guérin, 2022) to improve groundwater connectivity. These options might result in improved water redistribution across residual mineral fill materials, thus possibly resulting in an improvement in the regulation of the WT in inner and downgradient areas and a decrease in the importance of rainfall for maintaining WT levels. Further, given the large difference in surface elevation between the downgradient peatland and that along the downgradient edge of the residual pad in the present study, there may be value in sloping the surface elevation of residual pads along the direction of flow. Sloping might better align the downgradient pad surface elevation with that of downgradient peatland hollows, thus possibly reducing DWT in downgradient areas on residual pads. However, steep sloping could also result in undesirable surface inundation and flooding in the early season, given the near-surface WTs observed in the present study in interior and downgradient areas following the spring freshet. Further work is also required in order to better define the optimal target reprofiled surface elevation on partially removed well pads, particularly given the high degree of variability in water table position between years and between different peatland settings.

Also, a major limitation of the present study is that it was conducted on a single residual well pad due to the high cost of replication. This pad was comprised of a coarse sandy mineral fill. These materials are characterized by poor water retention and a high specific yield compared to finer-textured materials (Jury and Horton, 2004a), both of which would have contributed to the poor regulation of the WT in interior and downgradient areas. However, many well pads on the Western Boreal Plain are constructed of finer-textured fill materials such as the clay loams common in the region (Volik et al., 2020). These fill materials

may support better-regulated WT dynamics than sandy mineral fill, given their good water retention characteristics and lower specific yield (Jury and Horton, 2004a). However, finer-textured fill materials may also pose additional limitations on subsurface volumetric flow, as they have considerably lower K_h values than coarse fill materials (Freeze and Cherry, 1979). Nonetheless, future trials of partial well pad removal should strive to incorporate numerous well pads, including those comprised of fill materials with a range of textures to better understand the implications of texture on hydrological connectivity and regulation. Furthermore, the present study was conducted on a well pad that was connected to an adjacent upland ridge. Further study on well pads surrounded on all sides by peatlands will be required to better understand the impact of well pads alone on peatland hydrology.

2.5. Conclusion

This study presents the results of a hydrological assessment completed following the implementation of the first full-scale trial of peatland restoration on a partially removed oil and gas well pad. The long-term impact of the well pad on soil hydrophysical properties and hydrological dynamics in the surrounding peatland was evident, and similar in nature to that reported for access roads constructed out of mineral fill in peatlands. However, further research is required to better disentangle the relative effects of the well pad itself and the adjacent natural upland ridge, which may have contributed to the observed differences in soil hydrophysical properties and WT position and regulation between sides of the pad. Nonetheless, the current study highlights the pressing need for further research to assess the impacts of well pads on peatland ecohydrological processes. Further, hydrological dynamics on the residual pad were likely only sufficiently well-regulated for the reliable establishment of introduced fen mosses in areas within 20-40 m of the upgradient and north pad edges, while areas within its interior and along the downgradient edge were often characterized by poorly regulated and flashy WT dynamics. As such, while well pad partial removal has the potential to support hydrological functions requisite for the establishment of peat-forming vegetation communities characterized by fen mosses, further development and refinement of the technique should be

undertaken. Specifically, modifications to the technique and its application at the site level should aim to better redistribute water across partially removed pads in an effort to improve hydrological regulation in interior areas away from upgradient pad edges.

3 HYDROLOGIC ASSESSMENT OF MINERAL SUBSTRATE SUITABILITY FOR TRUE MOSS INITIATION IN A BOREAL PEATLAND UNDERGOING RESTORATION

3.1. Introduction

Peatlands are a prominent type of ecosystem on the Canadian Western Boreal Plain, forming an integral part of an interconnected surface and groundwater system, which regulates the quality and quantity of downstream water resources (Elmes and Price, 2019; Ferone and Devito, 2004; Waddington et al., 2015). Peatlands also store immense amounts of carbon (Yu, 2012), and provide habitat for wildlife species including the culturally significant woodland caribou (Hill et al., 2021; Rettie and Messier, 2000). However, peatlands in this region are also often underlain by petroleum-bearing geologic formations, which support an established oil and gas industry with considerable potential to disrupt peatland ecosystem function (Volik et al., 2020).

Extraction of oil and gas on the Western Boreal Plain is either undertaken through open-pit mining when deposits are near the surface, or through the drilling of in-situ wells to access deeper deposits (Schneider and Dyer, 2006). Open pit mining results in the excavation of peatlands and their complete loss from the impacted area (Rooney et al., 2012). In contrast, the use of in-situ wells results in a less concentrated disturbance, wherein mineral features called well pads are constructed to support surface drilling equipment (Osco et al., 2018). Well pads are typically one or more hectares in size, and are connected to one another by access roads. Both pads and roads are constructed by placing sand or clay mineral fill either onto a geotextile liner or directly onto the peat surface (Partington et al., 2016). As such, well pads and roads result in the near-complete loss of native peatland vegetation from their direct footprint (Lemmer et al., 2022). Well pads and roads also often result in shifts in vegetation community composition in adjacent peatlands (Ficken et al., 2019; Miller et al., 2015; Saraswati et al., 2020a; Willier et al., 2022), which can occur due to reductions in shallow subsurface flow across these features (McKinnon et al., 2024;

Saraswati et al., 2020b). The cumulative impact of this disruption to natural ecosystem function is immense (Ficken et al., 2019; Volik et al., 2020), especially given that over 36,000 hectares of well pads (UNEP, 2022) and over 100,000 km of access roads (Pasher et al., 2013) have been constructed in peatlands across Canada's boreal regions during the past several decades.

Due to the direct and indirect impacts that well pads and access roads have on peatland ecosystems, there is a legal requirement in the Canadian Province of Alberta to return them to a state of 'equivalent land capability' post-decommissioning (Powter et al., 2012). This has recently been defined as including the reestablishment of hydrological conditions suitable for the initiation of peat-forming vegetation communities including foundational peatland mosses (Alberta Environment and Parks, 2017). Mosses are the primary drivers of natural boreal peatland ecosystem formation and function (Bauer et al., 2003; Vitt et al., 2009). Notably, however, peatland mosses are non-vascular, poikilohydric organisms, which means that their internal water content is in equilibrium with their surroundings (Proctor et al., 2007). As such, they have a relatively limited desiccation avoidance capacity (Price and Whitehead, 2001; Proctor et al., 2007). Indeed, the maintenance of consistently near-surface water tables has been identified as one of the primary drivers of successful moss restoration outcomes on cutover peatlands (González and Rochefort, 2014; Ketcheson and Price, 2014; Price et al., 2003), drained peatlands (Anderson et al., 2016; Lamers et al., 2015), and in constructed peatlands (Ketcheson et al., 2017; Price et al., 2010). For cutover peat substrates specifically, water supply to meet the physiological requirements of many establishing true mosses is only optimized when soil water potential (Ψ) in the substrate is between approximately -4 and -8 millibars (mb) (Graf & Rochefort, 2010; M. Graf, personal communication), a range that is associated with nearly saturated soils. Beyond this range, growth rates decline, resulting in less favourable moss establishment outcomes (Graf & Rochefort, 2010). For *Sphagnum* species, favourable establishment can be expected to occur when Ψ remains between 0 and -100 mb, with desiccation generally occurring once Ψ drops below approximately -100 mb (Price and Whitehead, 2001).

One technique with the potential to create sufficiently wet conditions for moss establishment on decommissioned well pads and roads involves the complete removal of the mineral fill (Elmes et al., 2021b; Lemmer et al., 2022), or its burial beneath the underlying peat (Pouliot et al., 2021; Xu et al., 2022). In either case, the mineral fill itself is removed from the upper peat profile, which results in an increase in near-surface groundwater flow (Elmes et al., 2021b). Peatland mosses can subsequently be reintroduced onto the re-exposed organic substrates that remain, often through the use of a modified version of the Moss Layer Transfer Technique (MLTT; Rochefort et al., 2003). The MLTT involves collection of donor mosses from an undisturbed peatland and their subsequent introduction to the surface of a restoration site in a thin layer. The introduction of donor mosses is intended to accelerate the pace of peatland development and succession given the long time periods required for natural ingress of peatland mosses onto large restoration sites (Lemmer et al., 2022; Rochefort et al., 2003). However, complete removal of the mineral fill tends to promote the initiation of *Sphagnum*-dominated moss communities due to the removal or burying of the cation-rich mineral fill (Pouliot et al., 2021; Xu et al., 2022). Accordingly, the rates of establishment of the true mosses (i.e., those belonging to the class Bryopsida) characteristic of minerotrophic fens on the Western Boreal Plain (Vitt, 2014; Vitt and Lüth, 2017) may be more limited, as these species rely on a high supply of base cations (Vitt and Chee, 1990).

Because 63% of peatlands on the Western Boreal Plain are fens (Vitt et al., 2000), vegetation communities dominated by true mosses are a desirable target outcome for many well pad and road restoration projects (Osco et al., 2018; Volik et al., 2020). Accordingly, an alternative approach to the complete removal or burial of the mineral fill has recently been tried. This technique involves the partial removal of a well pad or road through surface lowering, leaving a residual cation-rich mineral substrate at the ground surface (Vitt et al., 2011). Initiation of true mosses can then occur directly on the residual mineral substrate, either through natural ingress (Lemmer et al., 2022) or application of the MLTT (Bird and Xu, 2021b). The conceptualization of this method was informed by the natural process of paludification, which

occurs when saturation of formerly terrestrial mineral substrates supports the initiation of an early successional peat-forming vegetation community (Bauer et al., 2003; Ruppel et al., 2013).

The first trial of partial well pad removal lowered sections of a pad to elevations between 5 and 15 cm above the water table in the surrounding peatland, and results suggested that the remaining residual mineral substrates were sufficiently wet to support long-term establishment of a planted peatland graminoid community (M. E. Gauthier et al., 2018; Vitt et al., 2011). Additional work at the mesocosm scale demonstrated the feasibility of establishing true mosses directly on mineral substrates, and identified the importance of providing microsites to improve moss species diversity (Borkenhagen and Cooper, 2016). Subsequent field-scale trials were conducted on small, lowered sections along the edges of several well pads and on a short section of a road. These trials identified a high potential for the initiation of peatland mosses when the target reprofiled surface elevation of the mineral fill was closely matched (<5 cm difference) to that of the water table in the surrounding peatland (Guérin, 2022; Lemmer et al., 2022). In these cases, near-saturated conditions were sustained as a result of the water table remaining near the surface for most of the growing season (Lemmer et al., 2022).

However, when the partial removal technique was later scaled up and implemented on a full-size well pad in a study associated with the present assessment, near-surface water tables were only consistently observed in areas of the pad within 20-40 metres of the adjacent upgradient peatland (McKinnon et al., 2024). Hydrological connectivity through the residual mineral fill and compacted peat underlying it was limited as a result of the low hydraulic conductivity of those materials (McKinnon et al., 2024). As a result, the water table became poorly regulated in interior areas away from upgradient pad edges beginning in the mid- to late-growing season, and thus substrate moisture conditions may not have remained favourable for establishment of mosses in those areas (McKinnon et al., 2024). In anticipation of this problem, previous trials had incorporated the application of straw mulch onto the surface to reduce atmospheric water losses from mineral substrates (Borkenhagen and Cooper, 2016; M. E. Gauthier et al., 2018). However, the effect of this treatment remains unclear, as soil water availability was not limiting in the field-scale trial (M. E.

Gauthier et al., 2018). On unsaturated residual mineral substrates, it is possible straw mulch may improve moisture conditions by reducing near-surface temperatures and increasing near-surface humidity (Novak et al., 2000a; Price et al., 1998; Scarlett et al., 2017).

Given the poor water table regulation observed to occur in interior areas of a field-scale partially removed well pad, it has remained unclear whether the partial removal of full-size well pads would support adequate water availability for peatland moss establishment. Further, there is uncertainty surrounding the efficacy of straw mulch in regulating soil moisture dynamics under unsaturated conditions on residual well pads. Accordingly, the objectives of this study are to:

- 1) Evaluate the degree to which interactions among mineral fill hydrophysical properties, water table position, and hydrologic regulation contribute to the maintenance of requisite moisture conditions for the establishment of true mosses on a residual well pad;
- 2) Assess the degree to which microtopographic variability and application of straw mulch affect residual mineral substrate moisture dynamics; and
- 3) Understand the potential implications of observed mineral substrate moisture dynamics for long-term true moss establishment on residual mineral well pads.

3.2. Methods

3.2.1. Study Site

The study was undertaken on a decommissioned oil well pad approximately 20 km northeast of the town of Slave Lake, Alberta, Canada (55° 19' 11" N, 114° 28' 22" W; Figure 3.1a). The 0.8 hectare pad was originally constructed adjacent to an upland ridge in a moderate-rich fen (pH = 6.4, electrical conductivity = 110 $\mu\text{S cm}^{-1}$) in 1991 through the placement of calcareous loamy sand mineral fill (Ca content > 13000 mg kg⁻¹) excavated from an upland borrow pit (Alberta Energy Regulator, 2024). The surrounding natural fen was characterized by *Picea mariana*, *Rhododendron groenlandicum*, *Rubus chamaemorus*, *Salix* spp.,

Carex aquatilis, *Drepanocladus aduncus*, *Aulacomnium palustre*, *Sphagnum warnstorffii* and *Sphagnum angustifolium*. The pad was included in a related study of shallow groundwater dynamics within and surrounding the pad footprint (McKinnon et al., 2024). That study found that shallow groundwater flow through the peatland was directed towards the northwest, and as such the pad consistently received water inputs across its eastern and northern edges (Figure 3.1b; McKinnon et al., 2024). During frozen ground conditions in March 2020, the well pad was partially removed using heavy machinery to scrape off the uppermost layers of mineral fill, with the exception of a small area surrounding the wellhead (Figure 3.1b). The target elevation of the lowered pad surface was 20 cm below the position of the late-season water table in the surrounding peatland, which in turn was inferred by measuring the elevation of seasonal ground ice in peatland hollows on the upgradient (east) side of the pad (Figure 3.1b). This target elevation was selected to maximize the likelihood the pad would remain wet under late-season conditions and to account for possible rebound of the pad surface elevation as a result of potential expansion of the underlying peat following the reduction in mineral fill weight.

The excess fill was reprofiled into an extension of the adjacent natural upland ridge, and a layer of residual mineral fill approximately 37.5 cm thick (McKinnon et al., 2024) was left in place across the remaining 75% of the pad footprint (Figure 3.1c). Microtopographic variability with a mean amplitude of 19 cm (McKinnon et al., 2024) was then created by roughing the surface of the fill with the bucket of an excavator (see Bird & Xu (2021b) for detailed methodology). Immediately following partial removal, donor moss propagules were collected from the upper 10 cm of a donor fen (pH = 6.1, electrical conductivity = 71 $\mu\text{S cm}^{-1}$) within 1 km of the site (Figure 3.1d) using an excavator. The donor mosses were transported to the residual pad (pH = 6.7, electrical conductivity = 698 $\mu\text{S cm}^{-1}$) using a trailer and spread across its surface using a skid steer at a 1:7 aerial ratio (see Bird & Xu (2021a) for detailed methodology). The donor moss community was comprised mainly of true moss species (including *Aulacomnium palustre*, *Tomentypnum nitens*, *Hamatocaulis vernicosus*, and *Helodium blandowii*) with some *Sphagnum* species (including *Sphagnum teres*, *Sphagnum warnstorffii*, and *Sphagnum magellanicum*), and was similar to that

observed in the rich fen surrounding the pad. Following donor moss introduction, the residual pad was subdivided into 16 study plots (each approximately 20 m by 20 m), eight of which were covered with straw mulch in an alternating pattern (Figure 3.1b). The mulch was applied by hand such that its coverage and thickness permitted light penetration to the underlying mosses while also potentially reducing evaporation from the pad surface and establishing moss communities (see Quinty et al. (2020) for guidelines on desirable mulch thickness and coverage following implementation of the MLTT). Application of mulch has

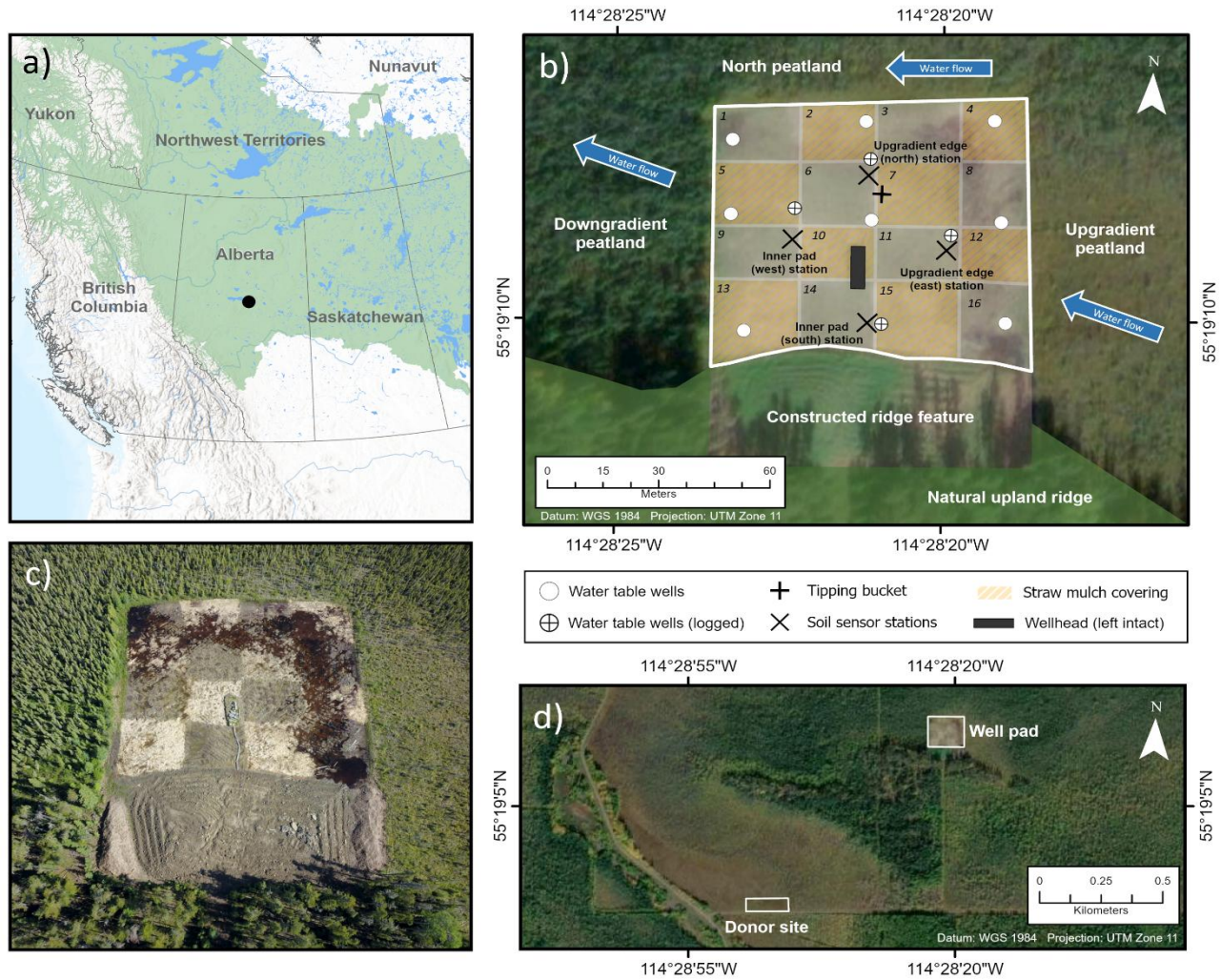


Figure 3.1: Location of the study site in the boreal region of western Canada (a) and detailed site diagram including the boundaries of the constructed ridge and residual well pad, treatment plot boundaries, plot numbers, and monitoring well and soil sensor locations (b). Blue arrows indicate the approximate direction of shallow groundwater flow through the surrounding peatland (see McKinnon et al., 2024a). An aerial photograph of the residual well pad taken three months post-partial removal (c) and map indicating the location of the moss propagule donor site (d) are provided for context.

been identified in previous trials as essential to reduce atmospheric water losses on peatland restoration sites (Price et al., 1998), especially as these sites may have more poorly regulated hydrological dynamics than natural peatlands (Price et al., 2003; Waddington et al., 2015).

3.2.2. Substrate Moisture Dynamics

The relationships between the hydrophysical properties of the residual mineral fill and its moisture retention characteristics were described using soil water characteristic curves (SWCCs). The SWCCs were fit to laboratory data obtained from five 250 cm³ cores collected from the upper 5 cm of the residual mineral fill on the well pad using stainless steel rings (8 cm internal diameter). Laboratory data was collected using a HYPROP 2 device (Meter Group, Pullman, USA), which implements the simplified evaporation method (Schindler et al., 2010). Model fitting was performed using the HYPROP-FIT software (Meter Group, version 4.2.2.0), and the final model was selected through a comparison of model performance as assessed using the Corrected Akaike Information Criterion. Care was also taken to ensure the model selected was not overparameterized by excluding all models with correlation between parameters ≥ 0.95 (Peters and Durner, 2015). Descriptions of all models tested are provided by Peters & Durner (2015).

Spatial variability in in-situ substrate water storage was assessed through monthly measurement of volumetric water content (VWC) in the upper 6 cm of the soil during both of two study periods (15 June to 14 August 2020, and 16 May to 14 September 2021) using a handheld probe (ML3, Delta-T Devices, Burwell, UK) at low, intermediate, and high microsites at three locations within each plot on the residual pad (n=48 for each height). Soil-specific calibrations were developed according to manufacturer guidelines (Delta-T Devices, 2017) using samples of the mineral fill collected from the site. Because the ML3 probe calculates VWC based on soil permittivity (which is dependent in part upon the electrical conductivity of the soil solution), calibration samples were saturated with pore water from the site to ensure their electrical conductivity was field-representative. During the second study period (2021), all manual VWC measurements were paired with additional measurements of soil water potential (Ψ) made over the 0.5-3.0

cm depth using portable tensiometers (QuickDraw 2900F1, Soilmoisture Equipment, Goleta, USA). All VWC and Ψ measurements were made under complete mulch cover in mulched plots. To account for temporal variability, hourly measurements of near-surface VWC and Ψ were made using sensors centred at 2.5 cm below ground surface (b.g.s.) at intermediate microsites between June and September in four non-mulched plots on the pad during the second study period (TEROS 10 & TEROS 21, Meter Group, Pullman, USA) (Figure 3.1). To further support the assessment of substrate moisture conditions, the percent coverage of surface water inundation (i.e., ponding) was estimated visually for each plot during field visits in 2021, and the depth of each pond covering more than 10% of any plot was estimated by averaging ten measurements.

3.2.3. Water Table Dynamics

To relate substrate moisture dynamics to hydrological dynamics on the residual well pad, manual measurements of depth to water table (DWT) were recorded at least once monthly during both study periods. These were made in fully slotted wells (i.e., perforated along the full length) constructed out of acrylonitrile butadiene styrene pipe (5.1 cm internal diameter), which extended to 1 m b.g.s. (Figure 3.1b). To prevent sedimentation, wells were covered with non-woven filter sock (Nilex, Edmonton, Canada) prior to installation. A subset of four wells were instrumented with pressure transducers (HOBO U20L, Onset, Bourne, USA) to enable measurement of DWT on a half-hourly basis throughout both study periods (Figure 3.1b). During the second study period, logger installation errors (loggers installed too high in the standpipe) precluded half-hourly measurement of DWT when it exceeded between -37.5 and -75 cm (depending on the well).

3.2.4. Precipitation

A tipping bucket (ECRN-100, Decagon, Pullman, USA) installed near the centre of the residual well pad was used to measure hourly precipitation (P) during both study periods. As described by McKinnon et al. (2024), a linear regression model ($R^2 = 0.90$) was used to gap-fill the dataset (Allen et al., 1998) using weather data from the Slave Lake Airport on 24 of the 53 days within the 2020 data collection period. This was necessary due to tipping bucket instrument failure on those dates.

3.2.5. Data Analysis

The significance of the effects of DWT, mulching, microsite, and their interactions on pad soil moisture were assessed by specifying them as fixed effects in separate linear mixed-effects models developed for VWC and Ψ . Treatment plot and month of measurement were set as random intercepts to control for non-independence of measurements made within common plots or months. To specifically assess spatial and temporal variability in soil moisture on the pad, subsequent linear mixed-effects models were fit for VWC and Ψ for each year with plot hydrological classification, microsite, and month of measurement specified as fixed effects. Treatment plot and mulching were specified as random intercepts. Hydrological classifications were assigned to individual treatment plots based on a comparison of seasonal mean plot DWT against a threshold DWT value. The threshold value was taken as the DWT approximately corresponding with Ψ values equal to the literature-based threshold for optimal moss-water availability (-10 mb; Graf and Rochefort, 2010), which was determined using a linear regression between manually measured DWT and Ψ values collected during the second study period. DWT values used for classification were the average of those manually measured in wells within or directly adjacent to a given plot.

Linear mixed-effects models were fit using the restricted maximum likelihood method using lme4 (Bates et al., 2015). Assumptions were checked through graphical inspection of residual distribution and logarithmic transformations were applied to the response variable when necessary. Post-hoc analyses were

conducted on estimated marginal means using Tukey's HSD p-value adjustment with Kenward-Roger approximations for degrees of freedom (Luke, 2017) using lmerTest (significance of main and interaction effects; Kuznetsova et al., 2017) or emmeans (significance of contrasts; Lenth, 2022). When transformations were applied, post-hoc analyses were conducted on back-transformed contrasts. All statistical analyses were performed using R (v4.3.1; R Core Team, 2022), and an α value of 0.05 was used to assess the significance of model outputs.

3.3. Results

3.3.1. Precipitation

Based on 30-year climate normal data for the Lesser Slave Lake region (1991-2020), mean annual precipitation is 422 mm, 292 mm of which is typically received as rainfall during the growing season (1 May to 30 September; Environment and Climate Change Canada, 2024a). Comparatively, cumulative growing season rainfall in the region during the 2020 and 2021 growing seasons was 286 and 244 mm, respectively (Environment and Climate Change Canada, 2024c), representing approximately 98% and 84% of the 30-year mean. Regional growing season cumulative precipitation was significantly lower in 2021 than in 2020 (Kruskal-Wallis, $p < 0.001$; McKinnon et al., 2024). As such, 2020 is referred to herein as having been characterized by a higher-precipitation growing season, while 2021 is referred to as having had a lower-precipitation growing season. Site-level measurements (including gap-filled data) indicate that the early growing season received more rainfall than the late growing season in both years. In general, individual rainfall events measured on-site were of low magnitude, with the exception of a small number of larger (i.e., > 10 mm) storm events (Figure 3.2). Of note, the second, lower-precipitation study period was characterized by extended dry periods between rainfall events, whereas the frequency of rainfall events was generally greater during the first, higher-precipitation study period (Figure 3.2).

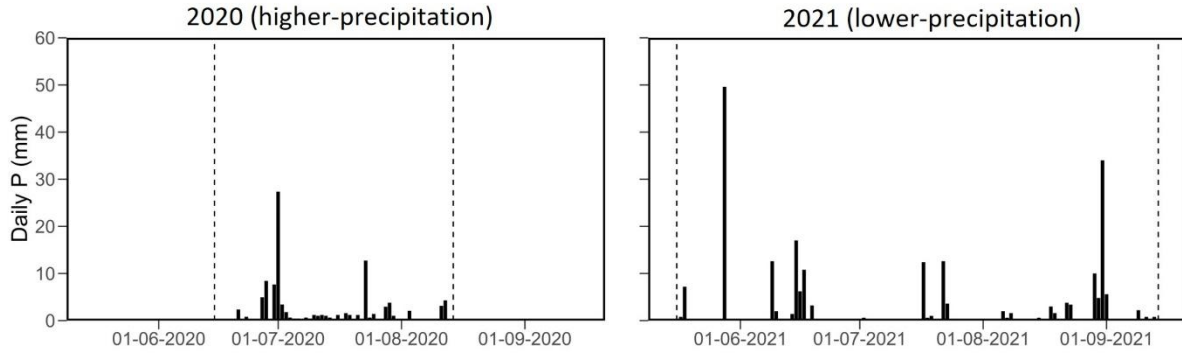


Figure 3.2: Daily precipitation (P) for the 2020 (15 June to 14 August) and 2021 (16 May to 14 September) study periods. Dashed lines indicate the start and end of the study period within each season (no site-level data was collected for dates outside of either study period).

3.3.2. Mineral substrate water retention characteristics

The soil water retention characteristics of the residual mineral fill were best described by a bimodal-constrained van Genuchten model (Durner, 1994), which illustrates a sharp decline in VWC with increasing $|\Psi|$ between 60 and 300 mb (Figure 3.3). Modelled VWC ranges for a given Ψ value indicate that optimal moss-water availability (i.e., Ψ approximately > -10 mb; Graf & Rochefort, 2010) is only likely to occur when VWC is $\geq 0.39 \text{ m}^3\text{m}^{-3}$ ($\pm 0.02 \text{ m}^3\text{m}^{-3}$; green shaded region in Figure 3.3). Notably, this is also the predicted VWC at saturation (Supplementary Table B.1), which indicates that optimal availability may only occur under conditions of complete saturation. In comparison, a risk of desiccation (i.e., $\Psi < -100$ mb; Price & Whitehead, 2001) is predicted to occur when the near-surface VWC drops below approximately $0.28 \text{ m}^3\text{m}^{-3}$, although this prediction was less precise ($\pm 0.07 \text{ m}^3\text{m}^{-3}$), with minimum and maximum modelled values of 0.20 and $0.34 \text{ m}^3\text{m}^{-3}$, respectively (Figure 3.3, Supplementary Table B.1). Fitted model parameters are summarized in Supplementary Table B.2.

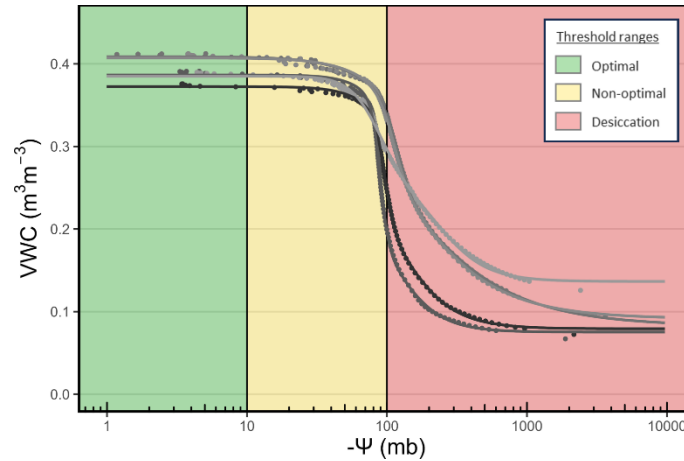


Figure 3.3: Soil water retention datapoints and fitted soil water characteristic curves (bimodal-constrained van Genuchten model). Vertical solid lines denote approximate soil water potential (Ψ) indicator thresholds (-10 and -100 mb; Graf and Rochefort, 2010; Price and Whitehead, 2001). When datapoints fall to the right of a given line, they are in exceedance of the associated threshold. Shading denotes ranges between the thresholds associated with optimal moss-water availability (green; 0 to -10 mb), non-optimal availability (yellow; -10 to -100 mb), and risk of desiccation (red; below -100 mb).

3.3.3. Environmental and treatment effects on substrate moisture dynamics

On the residual pad, both DWT and microsite position were found to have significant negative effects on VWC in both the higher and lower-precipitation years (rainfall over the study period was 6 and 48 mm less than the 30-year climate normal data in the higher and lower-precipitation years, respectively) and Ψ (in the lower-precipitation year only; no measurements were made in 2020) (Table 3.1). Specifically, post hoc analyses indicated that greater DWT values were associated with drier soil moisture conditions in general (lower VWC and more negative Ψ), while higher microsites were associated with drier conditions than lower ones. The interaction between DWT and microsite also had a significant effect on both soil moisture parameters in the dry year (Table 3.1), with post hoc analyses indicating that the negative effect of DWT on soil moisture was greater at higher microsites. Interestingly, the presence of a straw mulch was not found to have a direct significant effect on VWC or Ψ during either study period, although the interaction between mulching and microsite did have a significant effect on VWC in the dry year (Table 3.1). Specifically, post hoc analyses indicated that a slight positive effect of mulching on VWC was greater at lower microsites

Table 3.1: Results summary for linear mixed effects models used to determine significance of main environmental and treatment effects and their interactions on volumetric water content (VWC) and soil water potential (Ψ) for each study period. Statistically significant results ($\alpha = 0.05$) are provided in bold text.

Fixed Effect	2020		2021	
	(higher-precipitation season)		(lower-precipitation season)	
	F-value	p-value	F-value	p-value
<i>Volumetric water content</i>				
DWT	53.694 (1, 89)	< 0.001	39.051 (1, 201)	< 0.001
Mulching	0.155 (1, 16)	0.699	0.756 (1, 17)	0.397
Microsite	41.763 (2, 404)	< 0.001	71.821 (2, 654)	< 0.001
DWT x mulching	0.008 (1, 375)	0.931	0.852 (1, 668)	0.356
DWT x microsite	0.825 (2, 404)	0.439	10.421 (2, 654)	< 0.001
Mulching x microsite	0.174 (2, 404)	0.841	5.022 (2, 654)	0.007
DWT x mulching x microsite	2.174 (2, 404)	0.115	2.865 (2, 654)	0.058
<i>Soil water potential *</i>				
DWT	-	-	135.381 (1, 87)	< 0.001
Mulching	-	-	0.015 (1, 21)	0.903
Microsite	-	-	124.614 (2, 651)	< 0.001
DWT x mulching	-	-	0.670 (1, 609)	0.413
DWT x microsite	-	-	23.877 (2, 651)	< 0.001
Mulching x microsite	-	-	0.225 (2, 651)	0.798
DWT x mulching x microsite	-	-	0.143 (2, 651)	0.867

* Logarithmic transformation applied to satisfy model assumptions

than at higher microsites. Of note, the mulch cover was observed to have been partially blown off of high microsites by the wind during the first study period, resulting in a sparser mulch cover at high microsites than that at low microsites late in the first study period and for the duration of the second, lower-precipitation, study period. Also of note, the F-value for the interaction of DWT, microtopographic position, and mulching was relatively high for the VWC model in the dry year (indicating the potential importance of this interaction), although it was not found to be statistically significant (Table 3.1).

Linear regressions between DWT and manually measured Ψ indicate that moss water availability was generally only optimized (i.e., $\Psi > -10$ mb) at low microtopographic positions when the WT was within approximately 6 cm of the surface (Figure 3.7a). Of note, however, optimal moisture conditions were also

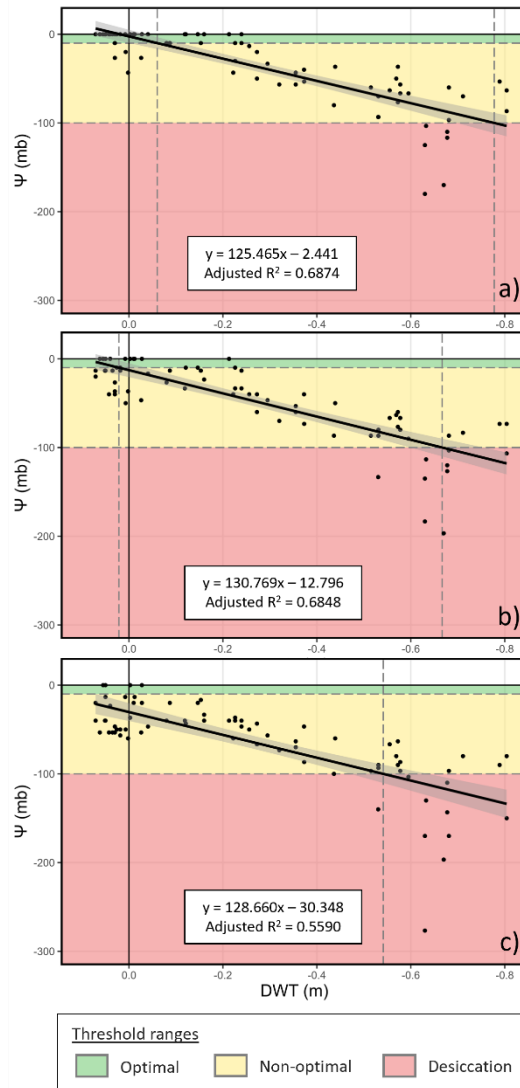


Figure 3.4: Linear regressions between depth to water table (DWT) and soil water potential (Ψ) for low (a), intermediate (b), and high (c) microsites. Shading denotes the Ψ ranges associated with optimal moisture availability (0 to -10 mb; green), non-optimal moisture availability (-10 to -100 mb; yellow), and risk of desiccation (less than -100 mb; red). Vertical dashed lines illustrate predicted DWT values corresponding with Ψ of -10 and -100 mb when applicable. All DWT values are referenced against the elevation of low microtopographic positions. Negative DWT values denote a WT below the ground surface, while positive DWT values denote a WT above the ground surface (i.e., surface inundation).

occasionally observed during periods when the WT was as deep as 25 cm b.g.s. (Figure 3.7a). In contrast, optimal moisture conditions for moss establishment at intermediate microsites were generally only observed when adjacent low-lying areas were inundated with approximately 2 cm of surface water (DWT values to the left of the y-axis on Figure 3.7b). Optimal moisture conditions were rarely observed at high

microsites, and again this generally only occurred when the adjacent low-lying areas were inundated (DWT values to the left of the y-axis on Figure 3.7c). Comparatively, exceedance of the -100 mb desiccation risk threshold is predicted to occur once the WT exceeds approximately 78, 67, and 54 cm b.g.s. for the low, intermediate, and high positions, respectively (DWT values measured against the ground surface at low positions; Figure 3.7a-c).

3.3.4. Water table dynamics

Based on the relationship between DWT and manually measured Ψ (Table 3.1 and Figure 3.4), treatment plots that had a seasonal mean manually-measured WT closer to the surface than 6 cm in a given year were classified as being hydrologically well-optimized (WO) for true moss establishment in that year (i.e., there was a high likelihood of Ψ values > -10 mb at low microsites in those plots on any given date). During the first, higher-precipitation study period, seven of the 16 plots on the pad were classified as being WO (see Supplementary Table B.3 for seasonal mean DWT values). Of note, all of these plots were located directly adjacent to the surrounding peatland along the upgradient and north pad edges, which received direct groundwater inputs from the peatland (Figure 3.5a; McKinnon et al., 2024). Furthermore, the small seasonal mean manually-measured DWT in these plots was reflected in the fact that the WT at logged wells along the north and upgradient (east) edges remained above 6 cm b.g.s. for between 91% and 94% of the study period that year (Figure 3.6a). Also of note, the surface of the pad was only consistently inundated in the area surrounding the upgradient east logged well (where inundation was observed for 90% of the study period; Figure 3.6a). As such, this was the only area where there was a high likelihood that moisture conditions would be optimal for true moss establishment at intermediate or high microsites.

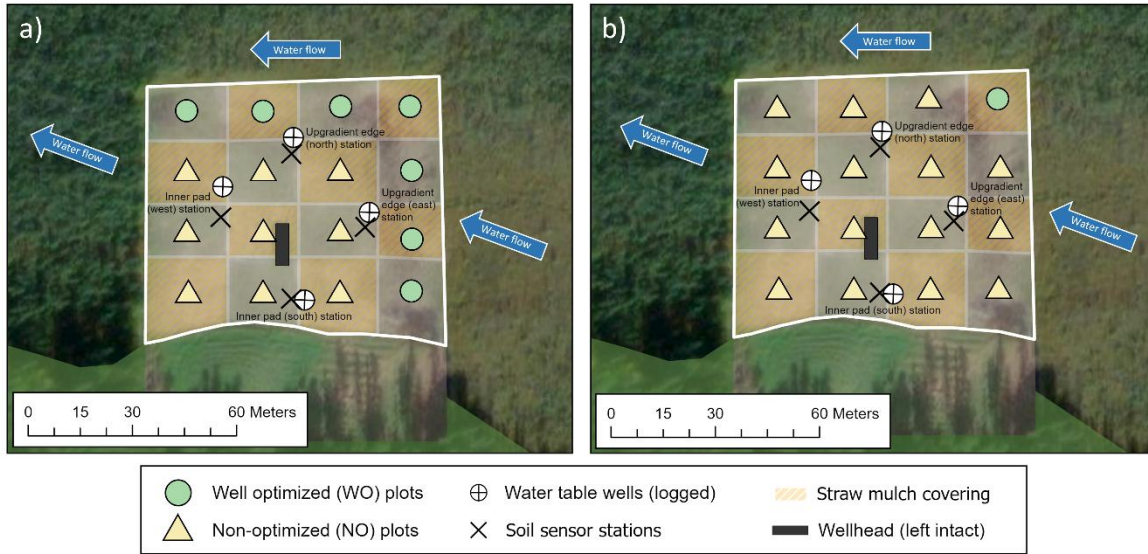


Figure 3.5: Plot hydrological classifications based on mean manually-measured depth to water table for the 2020 (higher-precipitation year; a) and 2021 (lower-precipitation year; b) study periods. Well optimized (WO) plots in a given year had a seasonal mean manually-measured water table closer to the surface than 0.06 m b.g.s., whereas non-optimized (NO) plots had a seasonal mean manually-measured water table position deeper than 0.06 m b.g.s. Blue arrows indicate the approximate direction of shallow groundwater flow through the surrounding peatland (as reported by McKinnon et al., 2024a).

Conversely, in the second, lower-precipitation study period, only one plot was classified as being WO for true moss establishment (Supplementary Table B.3). This plot was located in the northeast corner of the residual pad adjacent to both the upgradient and north pad edges (Figure 3.5b), while all other plots on the pad were classified as being NO that year (Supplementary Table B.3). This was primarily a result of the especially low WTs observed through the mid-to-late growing season in the second year, as the WT did, in fact, remain near the surface along the upgradient and north pad edges until late June (Figure 3.7d).

During the first, higher-precipitation year, nine plots had a seasonal mean WT deeper than 6 cm b.g.s. (Supplementary Table B.3), and as such were classified as being hydrologically non-optimized (NO) for true moss establishment (i.e., there was a high likelihood of Ψ values < -10 mb at low microsites in those plots on any given date). These NO plots extended through the interior of the pad and along its downgradient edge (Figure 3.5a). In contrast with the well-regulated WT along the upgradient and north edges, the WT in NO plots in the interior portions of the pad (inner west and south logged wells; Figure

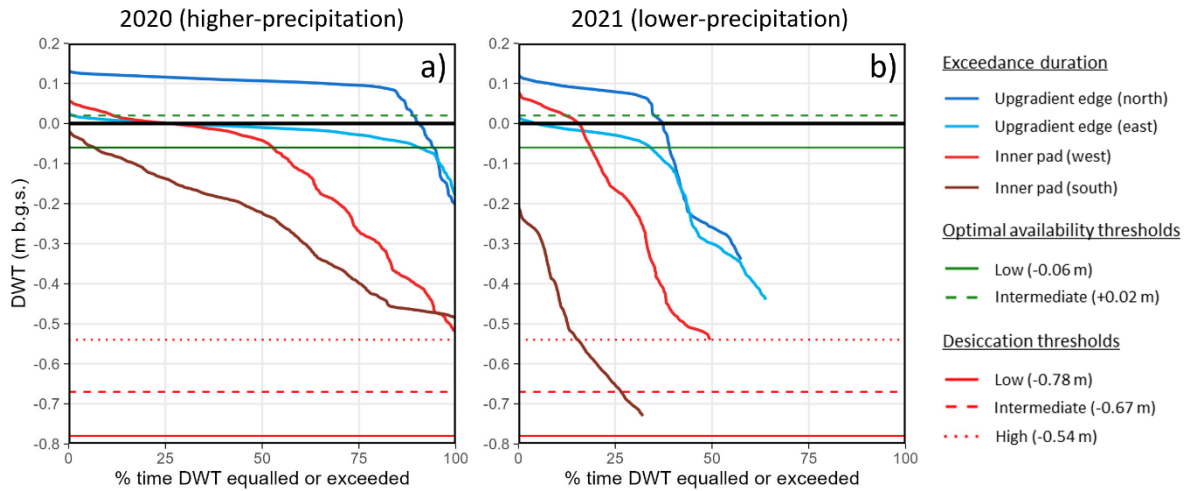


Figure 3.6: Depth to water table (DWT) duration curves for the 2020 (a) and 2021 (b) study periods. Plots illustrate the proportion of the study period that DWT exceeded (fell below) any given value on the y-axis. Note that the lengths of the study periods were not equal. For the 2021 study period, logger installation errors precluded development of complete DWT duration curves. Horizontal dashed lines represent y-axis values corresponding with the moss establishment DWT thresholds presented in Section 3.3.

3.5) was poorly regulated and frequently dropped below 6 cm b.g.s. even in the early season (Figure 3.7b). Reflecting this high degree of variability, the WT at the south and west logged wells in NO portions of the pad remained above 6 cm for only 7% and 53% of the first study period, respectively (Figure 3.6a). Despite this, the WT did not drop below 54 cm b.g.s. at either the interior or downgradient logged wells at any time (Figure 3.6a).

Reflecting the high proportion of plots classified as being NO in the second, lower-precipitation season, the WT was within 6 cm of the surface for only 34% and 39% of that study period (mainly during the early season) at the east and north logged wells, respectively (Figure 3.6b). At the downgradient logged well, the WT dropped below 6 cm b.g.s. for multiple periods of several days' duration in the early season, while it remained permanently below that level starting in late June (Figure 3.7d). As such, the WT was only within 6 cm of the surface at the downgradient logged well for 19% of the study period, while it remained below 6 cm b.g.s. for the entire study period at the inner logged well (Figure 3.6b). Furthermore, the WT at the inner and downgradient logged wells fell below 54 cm b.g.s. for 84% and 51% of the second

season, respectively (Figure 3.6b), while DWT exceeded -67 cm at the inner well for 73% of the study period (Figure 3.6b). Unfortunately, missing logger data precluded determination of the occurrence of exceedances of WT thresholds at the upgradient and north wells (Figure 3.6b). However, DWT was considerably greater at those wells for most of the second study period than observed during the first study period (Figure 3.7d).

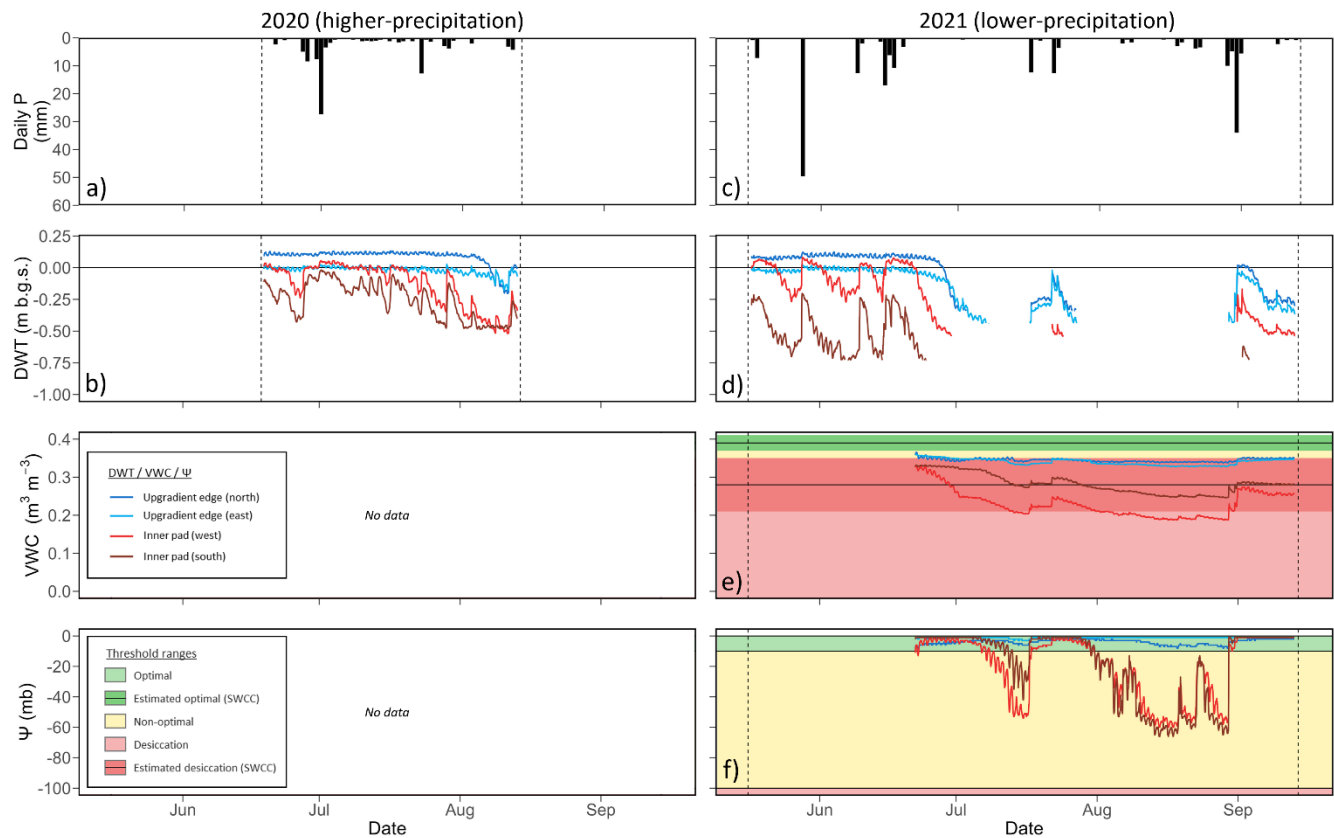


Figure 3.7: Comparison of daily site-measured precipitation (P) (a, c), depth to water table (DWT) referenced against low microsite heights (b, d), and soil volumetric water content (VWC) (e) and soil water potential (Ψ) (f) as measured at intermediate microsite heights on the residual pad for the lower-precipitation 2021 study period. On the VWC plot, horizontal lines denote the mean predicted VWC values associated with optimal moss-water availability ($0.38 \text{ m}^3 \text{ m}^{-3}$), and the mean predicted desiccation threshold ($0.27 \text{ m}^3 \text{ m}^{-3}$). Dark-shaded areas represent 95% confidence intervals about the means. On the Ψ plot, horizontal lines correspond to saturation (0 mb) and the optimal moss-water availability (-10 mb) and desiccation (-100 mb) indicator thresholds.

3.3.5. Mineral substrate moisture dynamics and inundation

As the second study period received significantly less rainfall than the 30-year climatic mean (see Section 3.1), the following analyses for both study periods are based on the plot hydrological classifications (i.e., WO vs. NO) assigned based solely on DWT data from the first, higher-precipitation study period (see Figure 3.5a). As such, the spatial comparisons detailed hereafter for both study periods were made between plots located adjacent to the upgradient (east) and north edges of the residual pad (which both received consistent groundwater inputs from the surrounding peatland, and which were classified as being WO in the first study period; Figure 3.5a) and those within the interior of the residual pad (which had limited hydrological connection to the surrounding peatland, and which were classified as being NO in the first study period; Figure 3.5a).

Owing to a shallow, stable WT in the 2020 (higher-precipitation) study period (Figure 3.7b), early season VWC did not vary significantly at any given microtopographic position between plots located along the pad edges (WO areas) and those located in its interior (NO areas) on either the June or July measurement dates (Figure 3.8a-b). Low microsites were significantly wetter than high microsites across the entire pad on those dates (Figure 3.8a-b). Accordingly, median VWC values at high microsites overlapped with the mean predicted desiccation threshold in June, whereas median VWC values at low microsites fell closer to the predicted optimal range (Figure 3.8a). Between the June and July measurement dates, the entire pad became significantly drier (with the exception of high positions in WO areas), bringing the median VWC values at low positions closer to or below the predicted mean desiccation threshold (Figure 3.8b).

In contrast, wetter conditions were observed along the upgradient and north pad edges early in the second study period compared to those in the first (Figure 3.8d-e). This resulted from the WT remaining at or above the surface in the early season (mean ponding coverage and depth of 70% and 10 cm, respectively), before the site became substantially drier in the late season (no surface inundation; Figure 3.7d). Thus, low microsites along the pad edges were significantly wetter than low microsites in interior areas of the pad (where mean ponding coverage and depth were 25% and 7 cm, respectively) on the May and June

measurement dates (Figure 3.8d-e). VWC values at low microsites generally fell within (or above) the predicted optimal range along the edges, while VWC values at high microsites were instead aligned with the mean desiccation threshold (Figure 3.8d-e). The same pattern was also observed for manually-measured Ψ early in the second study period, wherein water was more readily available to mosses (i.e., Ψ closer to zero) at low microsites than at high microsites across the full extent of the pad in May and June (Figure 3.8i-j). During this time, median Ψ values at low microsites across the pad fell within the range associated with optimal availability to mosses (> -10 mb), while Ψ values at high microsites normally fell below -10 mb, but above the -100 mb desiccation threshold (Figure 3.8i-j).

By the August measurement date in the first study period, both low and high microsites in interior areas of the pad were significantly drier than corresponding positions along the east and north edges (Figure 3.8c). Somewhat unexpectedly, this was due to significant increases in VWC at both high and low microsites along the east and north edges between the July and August measurement dates, whereas no significant change in VWC was observed at either microsite position in interior plots (Figure 3.8b-c). The wetting of the east and north edges in this case was likely associated with a late-season rainfall event, which occurred a few days prior to measurement (Figure 3.7a).

Similarly, long-term drying of the residual mineral fill was evident during the second, lower-precipitation study period. For example, half-hourly measurements of VWC and Ψ demonstrated a clear decreasing pattern between rainfall events (Figure 3.7e-f). Notably, however, half-hourly Ψ values recorded at the sensor stations located adjacent to the east and north pad edges remained within the predicted optimal range for true moss establishment (> -10 mb; Figure 3.7e). Half-hourly VWC values there fell below the optimal range predicted based on soil water characteristic data (although they remained above the mean predicted desiccation threshold at all times; Figure 3.7e-f). Conversely, at the sensor stations within the interior of the pad (south and west stations), half-hourly Ψ values frequently fell within the non-optimal range (between -10 and -100 mb) during periods characterised by little rainfall (Figure 3.7f), while half-hourly VWC values there frequently fell below the mean predicted desiccation threshold (Figure 3.7e).

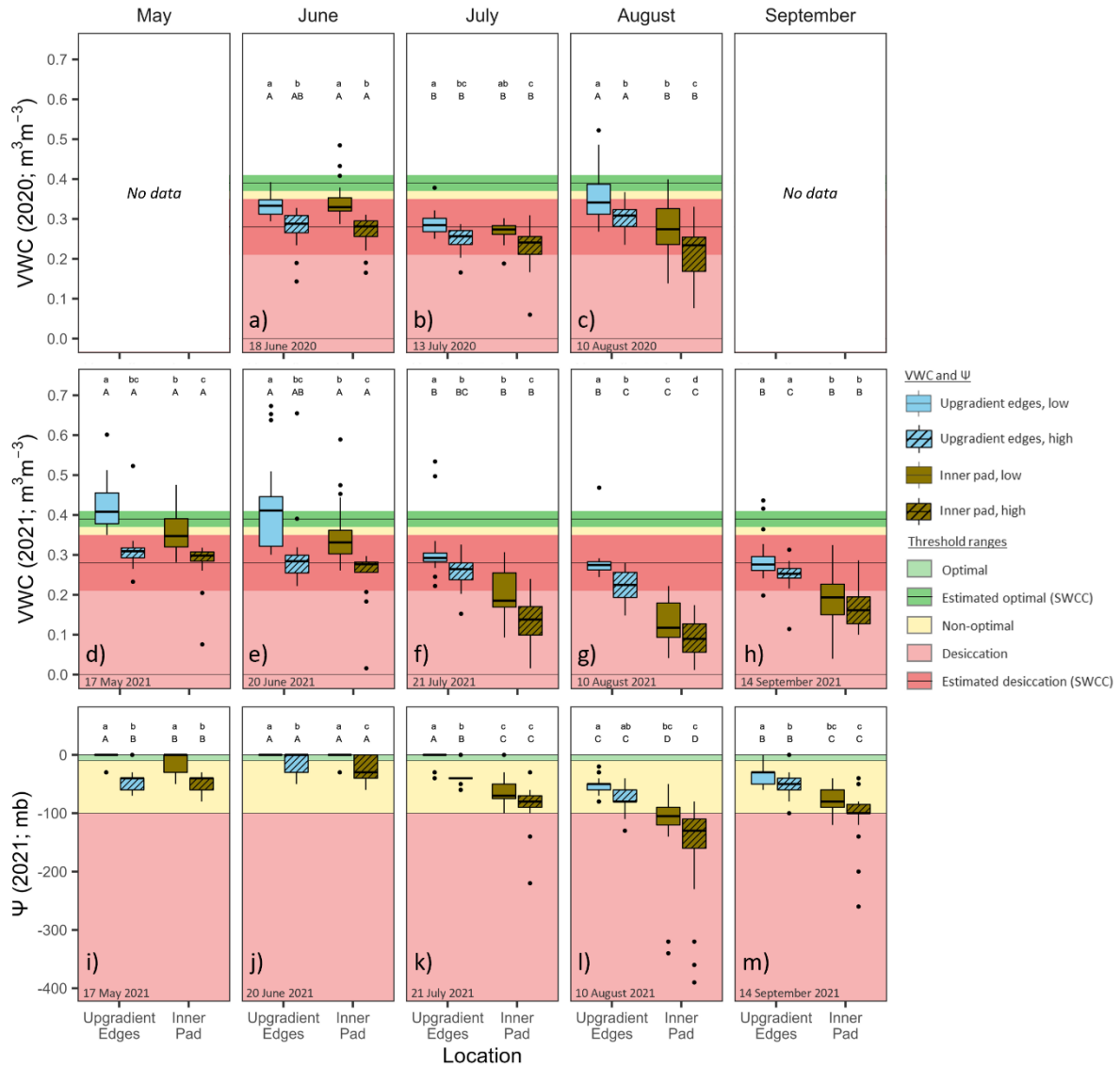


Figure 3.8: Boxplots of VWC and Ψ at low and high microsites in plots along upgradient edges (classified as WO in 2020) and within interior areas (classified as NO in 2020) of the residual pad. On VWC plots, horizontal lines denote the predicted mean VWC at Ψ values of -10 mb (optimal moss-water availability threshold; $0.38 \text{ m}^3\text{m}^{-3}$), and -100 mb (desiccation threshold; $0.27 \text{ m}^3\text{m}^{-3}$). Dark-shaded areas represent the 95% confidence interval about the mean. On Ψ plots, horizontal lines correspond to saturation (0 mb) and the optimal moss-water availability (-10 mb) and desiccation (-100 mb) indicator thresholds. Lowercase letters denote results of linear mixed-effects models comparing between microsite-plot type combinations within a given month. Uppercase letters denote results of linear mixed-effects models comparing between months for a given microsite-plot type combination. Statistically significant differences were observed between the least-squared means of groups not sharing common letters.

Considering the manual dataset, significant drying occurred at nearly all microsites between June and July during the second study period (Figure 3.8e-f). This caused median VWC values at both high and low microsite positions in the interior of the pad to fall below the lower limit of the predicted desiccation threshold range (Figure 3.8f). In contrast, VWC generally remained within the predicted desiccation threshold range along the east and north pad edges (Figure 3.8f). Decreases in VWC between June and July occurred simultaneously with significant decreases in manually-measured Ψ at both high and low positions in interior areas, and at high positions along the east and north edges (Figure 3.8k). Following these declines, Ψ fell between -10 and -100 mb across most of the pad, with the exception of low microsites along the edges where it generally remained above -10 mb (Figure 3.8k).

The interior of the residual pad continued to dry to a significant degree through the late season, with VWC values falling below the lower limit of the predicted desiccation threshold range (Figure 3.8g) and Ψ falling below -100 mb in interior areas (Figure 3.8l). Conversely, Ψ values recorded at both low and high microsites along the east and north edges generally remained above the -100 mb threshold value (Figure 3.8l). Following a series of large-magnitude late-season rainfall events, manually-measured VWC in interior areas of the pad and Ψ across the entire pad had significantly increased by the September measurement date within the second study period (Figure 3.8). This was accompanied by a sharp increase in both half-hourly VWC and Ψ values in interior areas of the pad (Figure 3.7e-f), although all manual measurements remained below their optimal ranges (Figure 3.8).

3.4. Discussion

3.4.1. Substrate water retention characteristics

The residual mineral substrate had a poor water retention capacity, as illustrated by the sharp decline in VWC observed at Ψ values between -60 and -300 mb (Figure 3.3). This can likely be attributed to the fact that the residual substrate had a loamy sand texture and low organic matter content (~2%; McKinnon et al.,

2024), as coarse-grained substrates have a high proportion of large, easily drained pores (Saxton et al., 1986) and the water retention capacity of sandy soils increases with increasing organic matter content (Rawls et al., 2003). Combined with the evident responsiveness of VWC and Ψ to changes in the position of the WT (Figure 3.7), it appears that retention of water within the residual sandy substrate was insufficient to maintain optimal conditions for moss establishment (i.e., $\Psi > -10$ mb) during periods when the WT was low. Notably, many well pads on the Western Boreal Plain are constructed out of clay substrates (Osisko et al., 2018). On well pads constructed out of these finer-textured materials, retention of water near the surface may be a more important control over water availability to mosses given their greater water retention capacity compared to sands (Jury and Horton, 2004a).

3.4.2. Substrate water availability assessment limitations

Some inconsistency was observed between the Ψ and VWC datasets with respect to moss-water availability thresholds (Figure 3.7 & 3.8). This can occur as a result of field variability in hydrophysical properties, which control the shape of soil water retention curves and in-situ VWC- Ψ relationships (Schelle et al., 2013). However, as tensiometers are generally more field-representative in mineral soils than VWC in terms of assessing water availability (assuming reasonable soil contact is achieved; Durner & Or, 2005), Ψ may be a more accurate indicator of in-field water availability. Measurements of Ψ were also made over the 0.5 – 3 cm depth, which may further contribute to them being more representative of moss-water availability than VWC, which was measured over the 0 – 6 cm depth. Furthermore, while the hydrological threshold of Price & Whitehead (2001) (*Sphagnum* desiccation at $\Psi < -100$ mb) and that based on the results of Graf & Rochefort (2010) (optimal true moss establishment at $\Psi > -10$ mb) are useful indicators, they may not be fully accurate in the present context.

Specifically, hydrological disconnection between introduced *Sphagnum* and *Tomentypnum* mosses and underlying cutover peat substrates has been reported due to a capillary boundary effect (Goetz and

Price, 2015; McCarter and Price, 2015). The extent to which this might occur between true mosses and mineral substrates, and thus the exact nature of the relationship between mineral fill Ψ and water supply to mosses, remains unknown (see Guérin, 2022). Furthermore, while many true mosses have a tomentum of rhizoids on their stems, which enable extracellular water retention (including *Tomentypnum nitens* and *Aulacomnium palustre*; Vitt, 2014), they lack the hyaline cells characteristic of the *Sphagnum* mosses for which the -100 mb desiccation threshold was developed (Price and Whitehead, 2001). Hyaline cells enable *Sphagnum* mosses to maintain high water contents for a prolonged period following the onset of drought conditions (Hájek and Vicherová, 2014). Thus, most true mosses have lower desiccation avoidance capacities than *Sphagnum* species (Vitt et al., 2014), and may therefore desiccate at Ψ values closer to zero than -100 mb. That said, many true mosses (including *T. nitens* and *A. palustre*) have similar desiccation tolerances as species in the *Sphagnum* genus (Vitt et al., 2014). Given that desiccation tolerance is an indicator of the ability of a moss to survive desiccation should it occur (Proctor et al., 2007; Vitt et al., 2014), the rates of true moss mortality following the onset of desiccation may be similar to those of *Sphagnum* species. As such, the *Sphagnum* desiccation threshold described above ($\Psi < -100$ mb) provides a reasonable (although potentially liberal) reference point for the assessment of potential limitations on the availability of water to true mosses in the absence of species-specific thresholds.

3.4.3. Substrate water availability and implications for true moss establishment

3.4.3.1. Well optimized areas

In general, substrate moisture conditions were optimized for true moss establishment at low microsites on the residual pad when the WT was within approximately 6 cm of the surface (referenced against the surface elevation at low microsites; Figure 3.4). The WT remained above this threshold value in plots directly adjacent to the east and north edges of the pad (classified as WO areas in 2020) for most of the early growing season (May and June) in both years (Figure 3.7). This can be attributed to a high degree of subsurface

hydrological connectivity between the adjacent peatland and those areas. Specifically, a previous associated study conducted on the same well pad examined in this study found that groundwater flow into the pad was sustained across the north and east pad edges under nearly all conditions measured (McKinnon et al., 2024). These subsurface inputs were sufficient to buffer the WT against substantial declines in elevation in the early to mid-growing season in both 2020 and 2021 (McKinnon et al., 2024). This occurred because lower WTs within the pad caused the magnitude of pad-directed hydraulic gradients, and thus that of subsurface water inputs, to increase (McKinnon et al., 2024).

Owing to the near-surface WT in WO areas in the early to mid-growing season, VWC at low microsites in the months of May and June tended to remain above the mean predicted desiccation threshold value ($0.28 \text{ m}^3\text{m}^3$) there and frequently overlapped with the predicted optimal range (0.37 to $0.41 \text{ m}^3\text{m}^3$; Figure 3.8). Direct measurements of Ψ at low microsites in those areas also tended to fall within the optimal range (0 to -10 mb) within the same measurement period in the second year. Taken together, these results suggest that subsurface hydrological connectivity with the adjacent peatland supported favourable early-season conditions for true moss establishment at low microsites within approximately 20 m of the upgradient (east) and north pad edges, which receive direct groundwater inputs from the adjacent peatland.

Moisture conditions generally remained suitable for some degree of true moss establishment at low microsites in WO areas through to the end of both study periods. However, drying of the substrate in those locations over time was evident, particularly in the second, lower-precipitation year. Specifically, measurements of VWC and Ψ at low microsites had both fallen below their respective optimal ranges in the latter half of the growing season in both years. Measurements of VWC frequently overlapped with the mean predicted desiccation threshold range, although directly-measured Ψ values at all times remained above the desiccation indicator threshold (-100 mb) at low microsites along those edges through to the end of the season (Figure 3.8). While these conditions indicate that reduced moss growth rates may occur in the late season relative to the early season, the combined early and late season results suggest that WO areas are likely suitable for establishment of true mosses adapted to wetter conditions. Previous studies have

reported establishment of species such as *Drepanocladus aduncus*, *Ptychostomum pseudotriquetrum*, and *Helodium blandowii* on mineral substrates with similar moisture conditions (Borkenhagen and Cooper, 2016; Guérin, 2022; Lemmer et al., 2022). At the same time, periods of complete saturation may limit moss establishment relative to conditions just below saturation due to inhibition of photosynthesis as a result of limitations on CO₂ diffusion or contamination of moss physical structures with bacteria and algae (Busby and Whitfield, 1978; Graf and Rochefort, 2010). These negative effects are less severe than those of desiccation (Schipperges and Rydin, 1998), however, and it is likely preferable to attain fully saturated conditions rather than drier conditions. Also of note, the aerial extent of surface ponding in WO areas (~70% in the early season) will likely result in the dominance of submergence-tolerant true mosses in low-lying areas there (e.g., *Hamatocaulis vernicosus*, *T. nitens*). Even short periods of inundation can allow these species to outcompete less tolerant species such as *A. palustre* (Borkenhagen and Cooper, 2018). Saturation with calcareous water can also lead to near-complete *Sphagnum* mortality (Borkenhagen and Cooper, 2018), suggesting that very limited establishment of *Sphagnum* mosses is likely in those areas.

In contrast to low microsites, VWC values at high microsites in WO areas regularly overlapped with the predicted desiccation range for the duration of both study periods. Directly measured Ψ remained above -100 mb (but usually below -10 mb) at this position during the lower-precipitation second season. Combined, this suggests that surface roughing in areas with a high WT resulted in microsites that will likely support the initiation of true mosses adapted to wide hydrological niches (including dry conditions) as well as those species which may be outcompeted in inundated areas. Previous studies have reported establishment of *T. nitens* and *A. palustre* over a wide range of moisture conditions on mineral substrates, including in drier areas (Borkenhagen and Cooper, 2016; Graf and Rochefort, 2010; Lemmer et al., 2022). Nonetheless, consideration could be given to reducing the target microtopographic amplitude in future trials, particularly as optimal water availability was only predicted to occur at high positions on the present site when adjacent low microsites were inundated with surface water (Figure 3.4). Preliminary vegetation surveys suggest the favourable hydrological conditions have indeed resulted in the early establishment of

true mosses in WO areas (data not shown), although a longer-term study is required to determine detailed community composition and development patterns.

3.4.3.2. Non-optimized areas

Interior and downgradient portions of the pad (NO areas) are likely best suited for the establishment of mosses with a high desiccation tolerance (see Vitt et al., 2014), given that those areas had limited subsurface hydrological connectivity with adjacent peatlands (McKinnon et al., 2024). This resulted in a highly variable WT, which in turn resulted in highly variable moisture conditions during the mid to late season. VWC and Ψ optimal availability indicator thresholds were regularly exceeded starting in the mid-season at all microtopographic positions in these areas, and VWC also regularly dropped below the desiccation indicator range. Directly measured Ψ also dropped below the -100 mb desiccation indicator threshold in NO areas under the driest conditions observed in the second season (Figure 3.8). Previous studies have suggested that true moss species adapted to drier hummocks or with wide hydrological niches (e.g., *T. nitens*, *A. palustre*, *Polytrichum strictum*) may be ideal candidates for mineral substrate restoration in drier areas (Borkenhagen and Cooper, 2016; Graf and Rochefort, 2010; Guérin, 2022), as they have high desiccation tolerances (Manukjanová et al., 2014) and can survive rapid declines in water content (Hájek and Vicharová, 2014). Similar studies have also identified some *Sphagnum* establishment in dry areas on residual mineral substrates (Guérin, 2022; Lemmer et al., 2022), although many dry-adapted hummock *Sphagnum* species (e.g., *Sphagnum warnstorffii*) require a hardening period over which desiccation tolerance develops (Hájek and Vicharová, 2014). Thus, *Sphagnum* species would likely be poorly suited for establishment in NO areas of the present site, where rapid declines in soil moisture can be expected. Additionally, surface inundation observed during the early season in NO areas (~25 % of the surface) may further limit *Sphagnum* establishment there (Borkenhagen and Cooper, 2018) as is likely in WO areas.

Notably, non-optimized hydrological dynamics may inhibit the establishment of all mosses regardless of desiccation tolerance. This is because growth ceases upon desiccation, and recovery times upon rewetting usually increase with longer periods of desiccation (Proctor et al., 2007). Therefore, while introduced desiccation-tolerant moss propagules may be capable of survival through the late season in NO areas of the residual pad, growth and community development are likely to be limited to the short early season when the WT is near the surface, as prolonged dry periods between rainfall events in the late season will limit growth rates (Figure 3.7). Instead, NO areas of the residual pad may be better suited for the establishment of sedges, including *Carex* and *Scirpus* species. These species have a high tolerance for water table fluctuations (Borkenhagen et al., 2024; Koropchak et al., 2012) and have been found to colonize unsaturated mineral substrates (Chapin and Chapin, 1981; Vitt et al., 2011).

Furthermore, it is likely that the second, lower-precipitation growing season may have been more representative of future climatic conditions on the Western Boreal Plain than the first, higher-precipitation year. Specifically, it is likely that there will be an increased prevalence of drought conditions in boreal regions under a future, warmer climate (Price et al., 2013; Schneider et al., 2016), which may be exacerbated in peatlands by their high rates of evapotranspiration compared to upland boreal forests (Helbig et al., 2020). Thus, it is possible that maintaining near-surface WTs on residual well pads may become more challenging in the future, and a greater proportion of the surface on these sites may support soil water availability similar to that observed in NO areas in the present study over time. As such, it is possible that residual well pads will be better suited for the initiation of graminoid species than mosses over the long-term.

3.4.4. Effect of straw mulch

Given the importance of atmospheric water losses in peatland water budgets under both current (Volik et al., 2021) and future climate scenarios (Helbig et al., 2020), reductions in evaporation rates on residual well pads may be an important restoration consideration. However, application of straw mulch in the present

study was not found to have a direct significant effect on either VWC or Ψ during either study period (Table 3.1). Notably, this finding is in alignment with the outcome of a trial of straw mulch application in saturated areas of a constructed peatland where soil moisture did not vary between mulched and non-mulched plots (Scarlett et al., 2017). However, for a considerable portion of both study periods, the mineral substrate across extensive areas on the residual pad was unsaturated. Under such conditions, the application of mulch typically improves substrate water content by reducing bare soil evaporation (Price et al., 1998; Scarlett et al., 2017). The lack of a significant effect of straw mulching on soil moisture is therefore an indicator that near-surface moisture dynamics on the well pad were primarily dictated by the poor water retention capacity of the coarse mineral fill (which permits rapid drainage; McKinnon et al., 2024) rather than evaporative drying. For example, the VWC measured in plots in interior areas of the pad was in the range of 10-20 m³m⁻³ for much of the late season, which aligns with the range for which stage II evaporation commonly occurs in loamy sands (Lehmann et al., 2018) such as the one the pad was constructed out of. Accordingly, evaporative demand was likely supplied only by low-magnitude capillary flow and vapour fluxes for much of the time (Dingman, 2015a). In contrast, a greater effect of straw mulch application might be observed on fine-textured residual clay pads which would retain water near the surface for longer and which could better maintain capillary flow from deeper WTs (Dingman, 2015a). In turn, this increased capillary flow would sustain greater evaporation rates, which might be reduced through the application of straw mulch.

3.4.5. Site hydrological trajectory and directions for future research

Regulation of hydrological dynamics in interior areas of the pad may improve over time. For example, the water retention capacity of the residual fill may improve as a result of the addition of organic matter in the form of below-ground root biomass and surface litter associated with the establishment of *Carex* and *Scirpus* sedge species, which supports the early stages of peatland succession following natural paludification (Rydin and Jeglum, 2013). Should the density of the moss community increase over time, improved capillary flow through a layer of partially decomposed moss litter at the surface into community

growth forms may also improve establishment (Goetz and Price, 2015). Similarly, enhanced water retention in dense community growth forms might increase desiccation avoidance capacity (McCarter and Price, 2014).

Given the high likelihood of observing only limited true moss establishment in NO areas of the pad, further consideration should be given to the development of modifications to the partial removal technique designed to optimize substrate moisture in areas with limited subsurface hydrological connection to adjacent peatlands. Treatments might include the application of soil amendments such as natural peat collected during the construction of new well pads near partially removed pads undergoing restoration. This might improve the water retention capacity of residual mineral substrates, while possibly improving the hydrological connection between introduced mosses and underlying substrates (M. E. Gauthier et al., 2018; Hugron et al., 2013). Consideration should also be given to enhancing subsurface hydrological connectivity between residual well pads and adjacent peatlands through the installation of either surface or subsurface flow conduits (Guérin, 2022; Vitt et al., 2011), or by sloping the surface of residual pads to make the elevation of interior and downgradient areas lower. This would better align the reprofiled surface with the position of the late season water table in those areas, thereby reducing DWT (McKinnon et al., 2024). These approaches are likely to be more important than reducing bare soil evaporation, given the likely greater control of the mineral fill hydrophysical properties (i.e., poor water retention capacity) and DWT on substrate moisture dynamics than that of evaporative water losses. Development of hydrologic thresholds (i.e., threshold Ψ values) corresponding with true moss desiccation limits should also be undertaken, similar to those already developed for *Sphagnum* species (Price and Whitehead, 2001). Based on preliminary observations of the establishing moss community on the site studied here (data not shown), it appears that true mosses can tolerate non-optimal conditions to at least some extent through a combination of tolerance and avoidance (Vitt et al., 2014). Quantification of these limits would provide a valuable tool for future restoration assessments on residual mineral well pads, which could effectively use measurements of Ψ to predict moss establishment outcomes. Further study of substrate water availability and the efficacy of straw

mulch on finer-textured residual well pads is also recommended, given that these fill materials are commonly used for pad construction (Osco et al., 2018).

3.5. Conclusion

This study assessed whether the partial removal of a well pad constructed out of sandy-textured mineral fill would result in the maintenance of requisite moisture conditions for the establishment of true mosses introduced onto it using a modified version of the MLTT. Optimal WT regulation and substrate moisture conditions were regularly observed in areas along pad edges receiving direct groundwater inputs from the adjacent peatland. The creation of microtopographic variability in those areas through roughing of the substrate surface resulted in the availability of microsites along a moisture gradient. Combined, it appears that these areas are likely to support the establishment of a diverse true moss community over time. However, unfavourable substrate moisture conditions (i.e., $\Psi < -10$ mb and $VWC < 0.27 \text{ m}^3\text{m}^{-3}$) were observed late in the growing season in areas away from pad edges, which were not hydrologically well-connected to the adjacent peatland. As such, mosses introduced to these areas faced considerable risk of desiccation late in the second, lower-precipitation study period, especially at high microtopographic positions. The application of a straw mulch was found to have little efficacy in the present study, highlighting the importance of targeting a high and stable water table via re-established hydrological connectivity with the adjacent peatland. Overall, the findings presented herein suggest that there is a need to enhance subsurface hydrological connectivity across partially removed well pads to reduce WT variability in interior areas away from pad edges. Nonetheless, the partial removal technique appears to have promise as a potential strategy to restore the tens of thousands of well pads that have been constructed in boreal peatlands to date.

4 DEVELOPMENT OF BIOGEOCHEMICAL PROCESSES ON A RESIDUAL OIL WELL PAD UNDERGOING RESTORATION TO A BOREAL PEATLAND

4.1. Introduction

The North American Western Boreal Plain (WBP) is characterized by a mosaic of distinct landscape units, each of which supports a unique combination of moisture and nutrient availability (Beckingham and Archibald, 1996). Spatial differences in moisture and nutrient regimes between landscape units drive heterogeneity in vegetation community structure (Craine and Dyzinski, 2013), which in turn has a major control over ecosystem functional processes including nutrient and water cycling (Wang et al., 2015; Wood et al., 2016), carbon sequestration (Harris et al., 2021), and provision of habitat for wildlife (Rettie and Messier, 2000). As such, a system of classification has been developed for use in Canada's boreal regions, wherein landscape units with comparable moisture and nutrient regimes are considered to belong to common 'ecosites' (Beckingham and Archibald, 1996). Ecosite classification has been used extensively in forest management (Tamminga et al., 2014; Van Rensen et al., 2015) and (more recently) in the assessment of restoration outcomes (Farnden, 2023; Thiffault et al., 2017). In the case of the latter, comparisons are drawn between ecosite classifications on restoration sites and in reference ecosystems (Lupardus et al., 2019; Nwaishi et al., 2015a). Consideration of abiotic factors (i.e., moisture and nutrient regimes) in addition to vegetation establishment outcomes can allow for an improved understanding of the successional trajectory a site may be following (Lupardus et al., 2019; Pinno and Hawkes, 2015). This enables informed prediction of whether the site is on a trajectory towards supporting similar vegetation and ecosystem functional processes as reference ecosites over time, even if differences exist in the present day.

Given that peatland successional trajectories can extend over hundreds to thousands of years (Halsey et al., 1998; Ruppel et al., 2013), comparison of nutrient and moisture regimes on peatland restoration sites may prove to be a valuable tool to assess restoration outcomes. Peatland restoration is likely to occur on a large scale in the future, as peatlands in the WBP have been impacted by open-pit oil sands

mining (Rooney et al., 2015) and the construction of over 36,000 hectares of in-situ oil and gas well pads associated with the energy industry (UNEP, 2022). By law, each well pad must be restored to a state of ‘equivalent land capability’, which is defined for peatlands as the re-establishment of ecohydrological conditions sufficient to sustain a peat-accumulating vegetation community (Alberta Environment and Parks, 2017). However, the surfaces of well pads can be about one metre higher than that of surrounding peatlands, leading to limited substrate moisture availability and high rates of decomposition (Lemmer, 2022).

In the absence of ecological restoration, these conditions tend to support the growth of grasses, legumes, and ruderal species, and also inhibit peat accumulation (Lemmer et al., 2022). However, in natural boreal peatlands, *Sphagnum* and true mosses (i.e., those belonging to class Bryopsida) form the foundation of ecosystem function and are the primary contributors to peat formation (Bauer et al., 2003; Vitt et al., 2009). Accordingly, peatland restoration in Canada targets the initiation of peatland mosses (Borkenhagen and Cooper, 2016; González and Rochefort, 2014) that are re-introduced to a site using the Moss Layer Transfer Technique (MLTT; Rochefort et al., 2003). Originally developed for use on cutover bogs in Eastern Canada, the MLTT involves spreading donor moss propagules across the surface of a restoration site (Rochefort et al., 2003). Maintenance of a near surface water table is critical following use of the MLTT (González and Rochefort, 2014), as this permits adequate water supply to mosses on the surface (Ketcheson and Price, 2014) and supports the maintenance of anaerobic substrate conditions.

To ensure the maintenance of a shallow water table, the surface of a well pad must be reconfigured (i.e., lowered and de-compacted). Complete removal of the mineral fill appears to be best suited for the establishment of *Sphagnum*-dominated peatland vegetation communities (i.e., those characteristic of bogs and poor fens) and is also associated with a high risk of creating undesirable flooded conditions (Lemmer et al., 2022; Pouliot et al., 2021; Xu et al., 2022). Hence, the partial removal of well pads has been proposed as an alternative (Vitt et al., 2011). This method involves removing the uppermost layers of the mineral fill, whilst leaving a residual mineral substrate in place at the surface of the peatland (Vitt et al., 2011). Moss

propagules can then be spread directly onto the residual mineral substrate using a variation of the MLTT (Borkenhagen and Cooper, 2016; McKinnon et al., 2024). This approach can be considered as somewhat analogous to the natural process of primary peat formation, which results in the initiation of peatland graminoids and bryophytes on mineral soils (Ruppel et al., 2013).

Initial trials of the partial well pad removal technique were conducted on small subsections of pads lowered to levels between 15 cm above the water table and the same elevation as the water table in the surrounding peatlands. At that scale, residual mineral substrates that were closely aligned with the surrounding water table supported sufficient water availability for the establishment of peatland graminoids and bryophytes (M. E. Gauthier et al., 2018; Lemmer et al., 2022; Vitt et al., 2011), which was in agreement with earlier trials at the mesocosm scale (Borkenhagen and Cooper, 2016). The residual mineral substrates also had pH and electrical conductivity values and base cation contents that were similar to those reported for minerotrophic fens on the WBP (M. E. Gauthier et al., 2018). In contrast, when the partial well pad removal technique was scaled up and applied on a full-size well pad (~0.8 hectares) as part of a study associated with this one, the low hydraulic conductivity of the mineral fill limited hydrological connectivity across the pad (McKinnon et al., 2024). This resulted in variable hydrological conditions, with large water table fluctuations observed across more than half of the pad, leaving much of it unsaturated for extended periods of time (McKinnon et al., 2025). This could potentially inhibit or slow the development of wetland biogeochemical processes (Hartsock et al., 2020, 2016; Nwaishi et al., 2015a). Accordingly, there remains considerable uncertainty surrounding the potential of the partial well pad removal technique to create the requisite biogeochemical conditions for establishment of a self-sustaining true moss community.

There is also uncertainty surrounding the effectiveness of restoration treatments applied following partial removal, including fertilization with rock phosphate and the use of straw mulch surface coverings. The former is applied during restoration of cutover peatlands to encourage establishment of the moss *Polytrichum strictum* (González and Rochefort, 2014; Sottocornola et al., 2007). While phosphorus is not typically a limiting nutrient in peatlands in the WBP (Vitt et al., 1995), its use may nonetheless encourage

moss and graminoid establishment on poorly developed residual mineral substrates originally excavated from the substratum (Green et al., 1992). Furthermore, straw mulch surface coverings are often employed on both restored cutover (Price et al., 1998) and constructed (Scarlett et al., 2017) peatlands to improve soil moisture conditions. It has also been tested on residual mineral substrates (Borkenhagen and Cooper, 2016; M. E. Gauthier et al., 2018; McKinnon et al., 2025), although the literature is lacking any detailed study of its effect on biogeochemical dynamics in this context. Considering the uncertainties surrounding the potential of the partial well pad removal technique to support the establishment of peatland biogeochemical processes, the objectives of this study are to:

1. Characterize the physicochemical properties of a residual mineral substrate following the partial removal of a full-size well pad;
2. Compare the biogeochemical processes occurring within a residual well pad to those in an adjacent reference fen to assess the likely successional trajectory to conditions dominant in fens; and,
3. Assess the impact of restoration treatments (application of straw mulch and phosphorus fertilizer) on biogeochemical processes on the residual well pad.

4.2. Methods

4.2.1. Study Site

The study was conducted on a 0.8 ha residual pad located in a fen near Slave Lake, Alberta, Canada (55° 19' 11" N, 114° 28' 22" W; Figure 4.1b). The adjacent fen was characterized by *Drepanocladus aduncus*, *Aulacomnium palustre*, *Sphagnum warnstorffii*, *Sphagnum angustifolium*, *Carex aquatilis*, *Rubus chamaemorus*, *Salix* spp., and *Rhododendron groenlandicum*. Tree cover was primarily *Picea mariana*. The pad was originally constructed out of sandy textured mineral fill in 1991. In March 2020, the pad was partially removed under frozen conditions by using heavy machinery to remove a layer of the mineral fill (see McKinnon et al., 2024). Microtopographic variability with average relief of 19 cm was created by

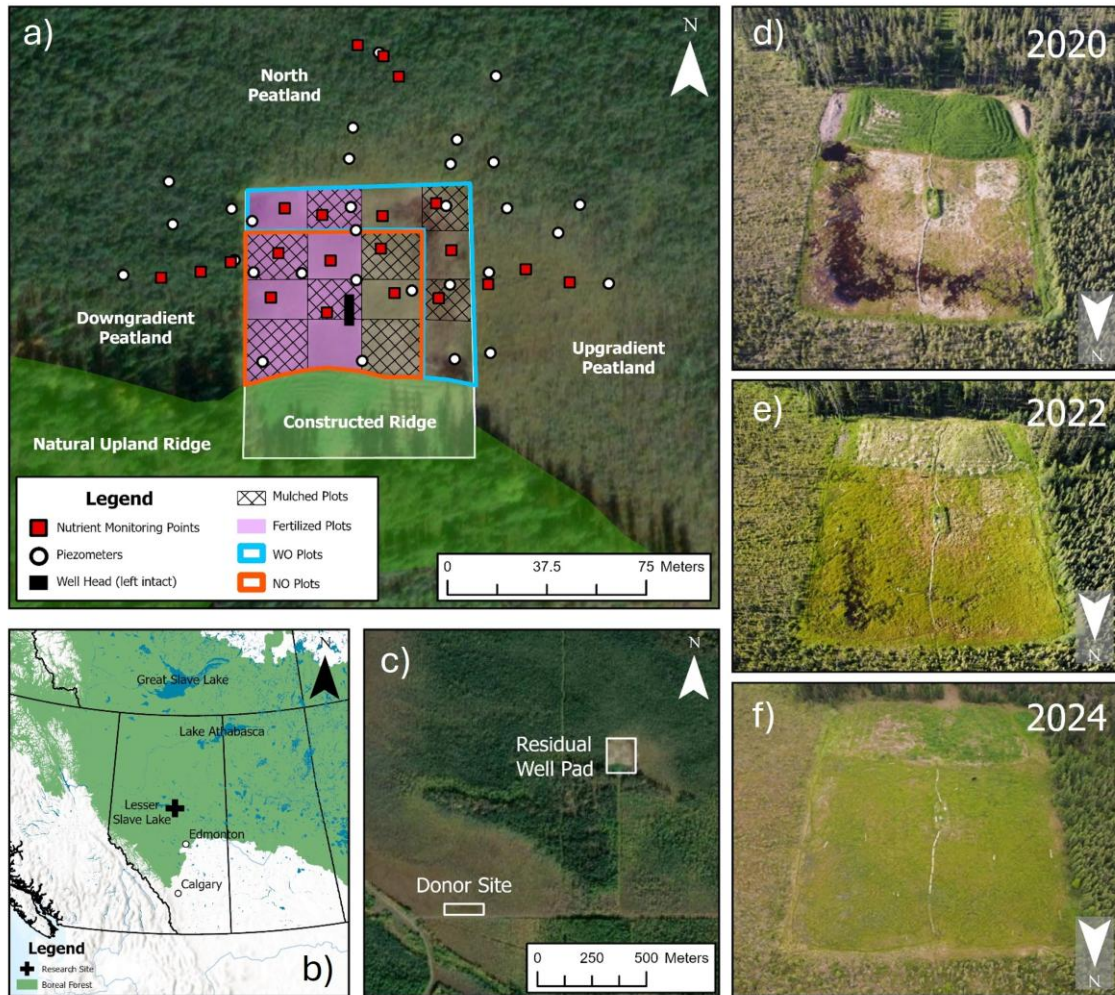


Figure 4.1: Map of the study site including the location of nutrient monitoring points, piezometers, and mulched and fertilized plots (a). Provided for context are the location of the study site in the Canadian western boreal region (b), location of the fen moss propagule donor site (c), and aerial photos of the residual pad and constructed ridge taken between 2020 and 2024 (d-f). Based on the findings of McKinnon et al. (2025, 2024), the classification of plots on the residual pad as being either hydrologically well optimized (*WO*) or hydrologically non-optimized (*NO*) for fen moss establishment was adopted.

scraping the surface of the fill using an excavator. Next, diaspores containing moss fragments, roots, seeds, and rhizomes were collected from a nearby donor fen (Figure 4.1c) and spread directly onto the residual mineral substrate using a skid steer. Based on previous trials of the use of the MLTT on residual pads, moss donors were introduced at an aerial ratio of 1:7 (donor fen to residual pad). The pad was then subdivided into 16 study plots (Figure 4.1a), half of which were covered by hand with a layer of straw mulch (Quinty et al., 2020b). Granular rock phosphate fertilizer (N-P-K of 2-13-0) was applied by hand to all treatment

plots on the western half of the pad (Figure 4.1a) at a rate of 150 kg ha⁻¹ in accordance with cutover peatland restoration guidelines (Quinty et al., 2020b).

4.2.2. Physicochemical Characterization

Measurements of groundwater pH, electrical conductivity (EC), and temperature were made using a water chemistry meter (HI9810-61, Hanna Instruments, USA) in polyvinyl chloride piezometers with 10 cm intake screens centred at a depth of 25 cm. Piezometers were covered with filter sock (Nilex, Canada) and installed within the pad (n = 12) and peatland (n = 18; Figure 4.1a). Monthly measurements were made during the study periods in 2020 (15 June – 14 August) and 2021 (16 May – 14 September). Additional measurements were made within the donor site in August (n = 10), including during the 2022 study period (14 May – 28 August). A third set of measurements was made on the pad during the 2023 study period (15 May – 11 September) to determine whether there was any change in pH or EC in response to vegetation establishment. All EC measurements were standardized to 20 °C and corrected for H⁺ ion activity using the method outlined by Sjörs (1950). Measurements of VWC were made approximately monthly during each study period including in 2024 (13 May – 30 September) using a moisture meter (ML3, Delta-T Devices, UK). Three measurements were made over the 0-6 cm depth at low and high microsites in each plot on the pad (n = 36 per microsite), and at each peatland monitoring point and piezometer (n = 81 per microsite). Soil-specific VWC calibrations were developed, and manual measurements of soil temperature were taken using a small portable thermometer in 2022 and 2023.

Analyses for substrate total micro and macronutrient concentrations were conducted by the University of Alberta Natural Resources Analytical Laboratory (NRAL) for samples of the mineral fill collected in March 2020 (n = 7). Analyses for total carbon (TC) and total nitrogen (TN) were conducted using dry combustion (FlashSmart Organic Elemental Analyzer; Thermo Fisher Scientific, Finland). Analyses for additional micro and macronutrients (Na, K, Mg, Ca, Zn, P, Fe, Mn, Cu, S, and Ni) were

undertaken by analyzing digested samples using inductively coupled plasma optical emission spectroscopy (ICP-OES) (iCAP6300 Duo, Thermo Fisher Scientific, Finland) according to Method 6010D (U.S. EPA, 2014). Soil organic matter (OM) content was estimated through loss-on-ignition. Specifically, 10 g samples of substrate were collected across the pad in June 2020 and June 2023 ($n = 12$ per year). Samples were dried at 105 °C prior to combustion at 550 °C for four hours (Heiri et al., 2001). Exchangeable cations (Na, K, Ca, Mn, Fe, Mg, and Al) and total cation exchange capacity (CEC) were quantified using the ammonium acetate method, with isopropanol and 1 M KCl used for measurement of total exchange capacity (Hendershot et al., 1993). Analyses were performed by NRAL on 10 g of mineral substrate collected from each pad monitoring point in 2020 ($n = 12$). Leachate concentrations of the index ion (ammonium acetate) and exchangeable cations were quantified using ICP-OES (iCAP6300 Duo, Thermo Fisher Scientific, Finland) in accordance with Method 6010D (U.S. EPA, 2014).

4.2.3. Nutrient Dynamics

Relative substrate decomposition rates were compared using the Tea Bag Index (Keuskamp et al., 2013). Briefly, paired green (EAN 8722700055525, Lipton, Netherlands) and rooibos (EAN 8711327514348, Lipton, Netherlands) teabags were pre-weighed and buried in 8 cm holes for 87 days between 17 May and 12 August 2021 and 79 days between 22 June and 9 September 2023. In 2021, three pairs of teabags were buried at an average microsite height at each monitoring point, while six pairs of tea bags were buried at low and high microsites at each monitoring point in 2023 (Figure 4.1). At the end of each period, teabags were collected and dried at 70 °C for 48 hours. Pre-burial and oven-dry weights were used to calculate stabilization factors (S) and decomposition rate constants (k). Values of S were used to compare the degree of stabilization of the labile organic fraction during decomposition, while values of k were used to compare relative labile organic matter decomposition rates (Keuskamp et al., 2013). Due to a change in teabag manufacturing, teabags buried in 2021 were composed of non-woven polypropylene, whereas teabags

buried in 2023 were composed of a tightly-woven polylactic acid mesh (Lipton, personal communication). As such, values of k and S were not directly comparable between years.

Paired samples of peat from the adjacent fen or mineral substrate from the pad were collected for determination of extractable (bio-available) nitrate (ext- NO_3^-), ammonium (ext- NH_4^+), and phosphate (ext- PO_4^{3-}), and net mineralization rates (quantified using the buried bag method of Eno, 1960). Samples were collected adjacent to one another over an interval of 0-10 cm below the surface and placed into polyethylene bags. Pre-incubation samples were stored at 4 °C prior to analyses within 24 hours. The second sample was buried for between 21 and 30 days (depending on the year) prior to collection and analyses. Extractions were performed by shaking samples at 100 rpm for 60 minutes in deionized water (for ext- PO_4^{3-}) or 2.0 M KCl (for ext- NO_3^- and ext- NH_4^+) at a gravimetric ratio of 1:5. Extracts were filtered through 2.5 μm cellulose filter paper (Whatman 42, Cytiva, UK) and frozen prior to analyses by NRAL (Gallery Plus Beermaster Autoanalyzer; Thermo Fisher Scientific, Finland). Pre- and post-incubation nutrient concentrations were normalized based on dry sample weights (Reddy et al., 2013). Net mineralization rates were calculated as the difference between post- and pre-incubation extractable nutrient concentrations divided by the incubation length.

Pore water was collected from the upper 10 cm at each monitoring point (Figure 4.1a) using samplers constructed out of perforated polyvinyl chloride pipe. Samples were stored at 4 °C prior to filtration within 24 hours through 2.5 μm filter paper (Whatman 42, Cytiva, UK) then frozen. Concentrations of soluble reactive P (SRP; PO_4^{3-}), NO_3^- , NH_4^+ , and Cl^- were determined using calorimetric methods by NRAL (Gallery Plus Beermaster Autoanalyzer; Thermo Fisher Scientific, Finland). Nutrient supply rates were quantified using Plant Root Simulator (PRS) probes installed for 72 hours at intermediate microsites at each monitoring point in 2020 and 2023 and at paired low and high microsites in 2021. A burial length of 72 hours was selected based on the recommendations of Hartsock et al. (2020) to ensure nutrient concentrations would be close to equilibrium. Upon retrieval, probes were cleaned with distilled water and shipped to Western Ag Innovations where they were eluted with 0.5 M HCl. Calorimetric methods

were used for NO_3^- and NH_4^+ (San++ Analyzer, Skalar, Netherlands) while analyses for PO_4^{3-} , Ca, Mg, K, Fe, Mn, and Al were conducted using ICP-OES (SpectroGreen Analyzer, Spectro Analytical Instruments, USA). Based on the method of Plach et al. (2017), PRS probe supply rates were used to calculate N:P ratios in 2023 that were used to inform potential nutrient supply limitations at that time.

4.2.4. Statistical Analyses

Statistical analyses were conducted using R (R Core Team, 2024) with $\alpha = 0.05$. When all observations of a given variable exceeded the analytical limit of quantitation (LOQ), ordinary least-squares regression (OLS-Reg) was used to assess the significance of environmental and treatment effects (VWC, temperature, mulching, and fertilization) in predicting nutrient variables on the pad. Log transformations were applied when required to meet statistical assumptions. When at least one, but $< 40\%$ of observations were censored ($< \text{LOQ}$), censored regression (Cen-Reg) models based on maximum likelihood estimation (Helsel, 2012) were employed. Censored regression accounts for the proportion of data below the LOQ without depending on an actual value (Helsel, 2012, 2005). OLS-Reg models were fit with the *lm* function of the *stats* package (R Core Team, 2024). Cen-Reg models were fit using the *cencorreg* function of the *NADA2* package (Julian and Helsel, 2024). Model fit was assessed through visual inspection of residual distribution using Q-Q plots (Zuur et al., 2009). Variance inflation factors (VIF) were computed to check for multicollinearity using the *vif* function of the *car* package (Fox and Weisberg, 2019). After running Cen-Reg models, partial Wald's tests were conducted within the *cencorreg* function to compute Wald's Z statistics and p-values for each predictor variable (Helsel, 2012). Based on the findings of Antweiler (2015), no statistical tests were performed when $> 40\%$ of observations in a dataset were censored.

For datasets without censored observations, spatial variability was assessed using Student's t-tests or analysis of variance (Tukey's HSD post hoc comparisons) with the *t.test* function of the *stats* package or the *Anova* function of the *car* package when the normality assumption was met (or met through log

transformation). When transformation was ineffective, Wilcoxon Rank-Sum tests or Kruskal-Wallis tests (Dunn's post hoc comparisons) were employed using the *wilcox.test* or *kruskal.test* functions of the *stats* package. For censored datasets with < 40 % censored observations, Peto-Peto tests with Benjamini-Hochberg post hoc comparisons were employed (Helsel, 2012) using the *cenlway* function of *NADA2*. Spatial comparisons between the surrounding peatland and locations on the pad were guided by the findings of a related study (McKinnon et al., 2025). That study found that plots located along the upgradient and north pad edges frequently had higher VWC than those within the interior of the pad. As such, plots along the upgradient and north edges were deemed to be hydrologically well-optimized (*WO*) for moss establishment based on moisture, while the interior plots were deemed to be hydrologically non-optimized (*NO*) (McKinnon et al., 2025; Figure 4.1a). Similar spatial patterns were identified in the present study (see Section 3.4), and spatial comparisons were thus made between the peatland, *WO* plots, and *NO* plots (Figure 4.1a). Data were plotted with *ggplot2* (Wickham, 2016). For censored datasets with < 40 % censored observations, plots were drawn by imputing values for censored observations using the robust ROS method (Helsel, 2012) with the *cenboxplot* function of the *NADA* package (Lee, 2020).

4.3. Results

4.3.1. Weather

Mean growing season (1 May to 30 September) air temperatures in the Slave Lake region (data from the Slave Lake airport) exceeded the 30-year climate normal (13.0 °C) in all years, ranging from 13.1 to 14.9 °C (Figure 4.2a; Supplementary Table C.1). As expected on the WBP, temperatures were greatest during the peak growing season (June to August), with cooler conditions observed through the spring and late summer (Figure 4.2a). Total rainfall in the region (data from the Slave Lake airport) exceeded the 30-year climate normal in the month of May in all years studied (Figure 4.2b). This pattern continued through much of the growing season in 2020, 2023, and 2024, which received 107 %, 110 %, and 133 % of the 30-year

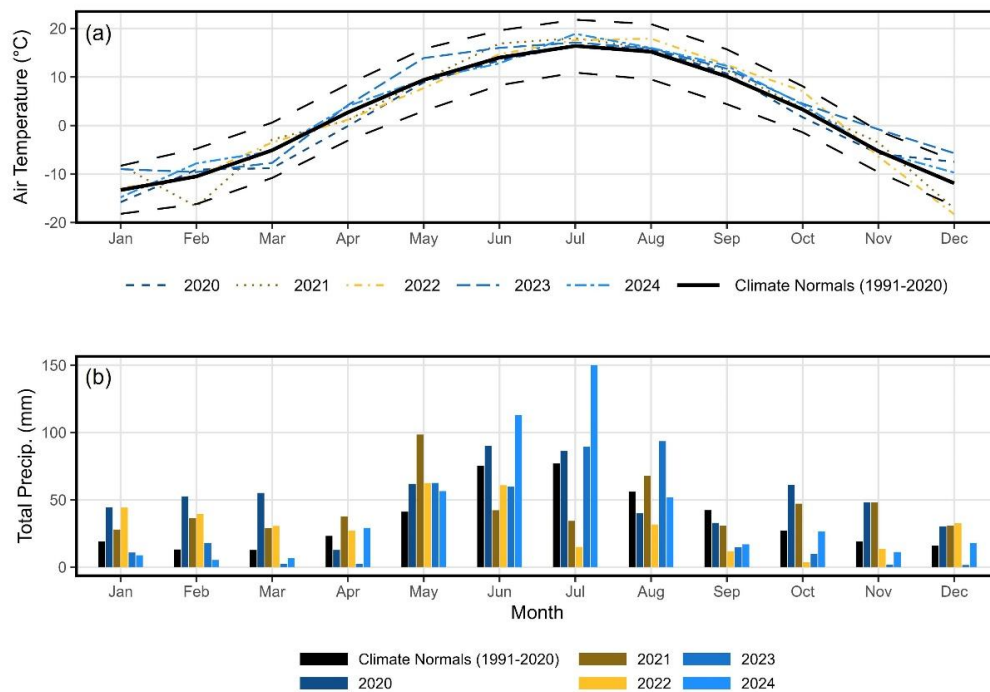


Figure 4.2: Comparison of monthly mean air temperature (a) and monthly total precipitation (b) between study years (2020-2024) and compared against 30-year climate normal data (monthly means for the years 1991-2020). All data was collected by Environment and Climate Change Canada at the Slave Lake airport (~20 km away from the study site).

growing season climate normal rainfall (292 mm), respectively (Figure 4.2b; Supplementary Table C.1). Nonetheless, there were still drier than average months during the growing season in each of those years, particularly during the late summer (Figure 4.2b). In contrast, the 2021 and 2022 growing seasons were drier than average, receiving only 94 % and 62 % of the 30-year climate normal rainfall, respectively (Figure 4.2b).

4.3.2. Substrate Physicochemical Characterization

The OM content of the mineral fill was relatively low immediately post-partial removal (mean = 2.1 ± 0.7 % in June 2020; McKinnon et al., 2024) and did not significantly increase during the subsequent three years (Wilcoxon rank-sum, $p = 0.1$; mean = 3.1 ± 2.8 % in June 2023). Nonetheless, the TC content of the fill (0.91 % w/w) was relatively high compared to other nutrients (Table 4.1), although this value also

Table 4.1: Mean residual mineral fill total micro and macronutrient concentrations and substrate nutrient ratios quantified for samples collected in 2020 (n = 7). Standard deviations are provided in parentheses below each mean value. Note that substrate total carbon, as opposed to substrate organic carbon, has been reported and used in the calculation of the reported nutrient ratios.

	Macronutrients (mg kg ⁻¹)							Micronutrients (mg kg ⁻¹)					Nutrient ratios			
	C	N	K	Ca	Mg	P	S	Fe	Mn	Zn	Cu	Ni	C:N	C:P	N:P	Fe:P
Mean	9121.9	270.9	914.3	13184.7	3021.3	270.6	63.4	8085.5	162.2	24.0	6.2	9.2	35:1	34:1	1:1	30:1
St. Dev.	(1629.0)	(49.6)	(106.5)	(4562.5)	(592.6)	(18.0)	(13.2)	(807.1)	(26.0)	(2.2)	(0.5)	(0.5)	(9.7)	(5.6)	(0.2)	(3.3)

includes inorganic (i.e., non-bioavailable) carbonates. The total contents of Ca, Fe, and Mg were the highest among other nutrients (Table 4.1). As such, Ca and Mg were the predominant cations on soil exchange sites (Table 4.2). Notably, exchangeable Fe was unexpectedly low (Table 4.2), suggesting that Fe may have been undergoing complexation with other ions. Exchangeable Mn and K were also comparatively low (Table 4.2), although this result was not unexpected given their lower total values (Table 4.1). In comparison, N, P, and S were the macronutrients found in the lowest quantities overall (Table 4.1). Combined with the high TC content, this resulted in high substrate C:N and C:P ratios (Table 4.1). The high total Fe content also resulted in a high substrate Fe:P ratio, while the quantities of total N and P were approximately equal (Table 4.1). Total (116.5 ± 10.4 mg kg⁻¹) and exchangeable Na were also low on the pad, as was exchangeable Al (Table 4.2). Total Al was not quantified for the residual pad, although the Al content of a similarly textured mineral fill collected from a nearby access road was high (mean Al = 6187.0 ± 1046.0 mg kg⁻¹; unpublished data). Combined, this suggests that Al might also have been undergoing complexation with other ions.

Table 4.2: Residual mineral fill total cation exchange capacity (CEC) and exchangeable cation concentrations quantified for samples collected in 2020 (n = 12). Standard deviations are provided in parentheses below each mean value.

	Exchangeable cations (cmol kg ⁻¹)							Total CEC* (cmol kg ⁻¹)
	K	Ca	Mg	Fe	Mn	Na	Al	
Mean	0.167	12.064	1.285	0.003	0.038	0.156	0.037	6.618
St. Dev.	(0.018)	(3.078)	(0.509)	(0.001)	(0.023)	(0.037)	(0.016)	(1.435)

*Total CEC is reported as the measured concentration of the index ion (ammonium acetate) as opposed to the sum of exchangeable cations.

4.3.3. Groundwater Chemistry

Median pore water Cl^- concentrations did not generally differ between the pad and fen, ranging from 1.1 to 6.6 mg L^{-1} (Supplementary Figure C.1). Nonetheless, the mean EC of shallow groundwater within the pad (616 to 771 $\mu\text{S cm}^{-1}$) was significantly higher than in the fen (110 to 113 $\mu\text{S cm}^{-1}$) in both 2020 ($p < 0.001$) and 2021 ($p < 0.001$) (the study periods with fen measurements; Figure 4.3a-b). On the pad, these values were aligned with regional extreme-rich fens, whereas values in the surrounding fen were aligned with moderate-rich fens (Vitt and Chee, 1990). In contrast, the EC in the moss donor site was aligned with poor fens, ranging from 43 to 101 $\mu\text{S cm}^{-1}$ (Supplementary Table C.2; Vitt & Chee, 1990). Notably, EC within the pad did not change significantly between 2020 and 2023 ($p = 0.07$). In the peatland, mean EC was significantly higher downgradient of the pad (160 $\mu\text{S cm}^{-1}$) than on its upgradient side (93 $\mu\text{S cm}^{-1}$) in 2020 ($p = 0.02$; Figure 4.3a), although no significant differences were observed in 2021 (Figure 4.3b).

Similarly, mean shallow groundwater pH was significantly higher within the pad (7.0) than in all surrounding peatland areas (6.4) in 2021 ($p < 0.001$; Figure 4.3e). However, mean pH did not significantly differ between the pad (6.9) and the downgradient peatland (6.8) in 2020 (Figure 4.3d). Correspondingly, mean pH in the downgradient peatland was significantly higher than in the upgradient (6.4; $p < 0.001$) and north (6.5; $p < 0.001$) peatlands that year (Figure 4.3d). In contrast with the stability of EC on the pad, mean pH there declined significantly between 2021 (7.0) and 2023 (6.2) ($p < 0.001$; Figure 4.3f). Notably, pH in both the pad and fen were typical of moderate-rich fens in the region, as was the pH in the moss donor site, which ranged from 5.9 to 6.3 (Supplementary Table C.2; Vitt & Chee, 1990). Shallow groundwater was also significantly warmer within the pad than in the peatland in both 2020 ($p < 0.001$) and 2021 ($p < 0.001$; Figure 4.3g-h).

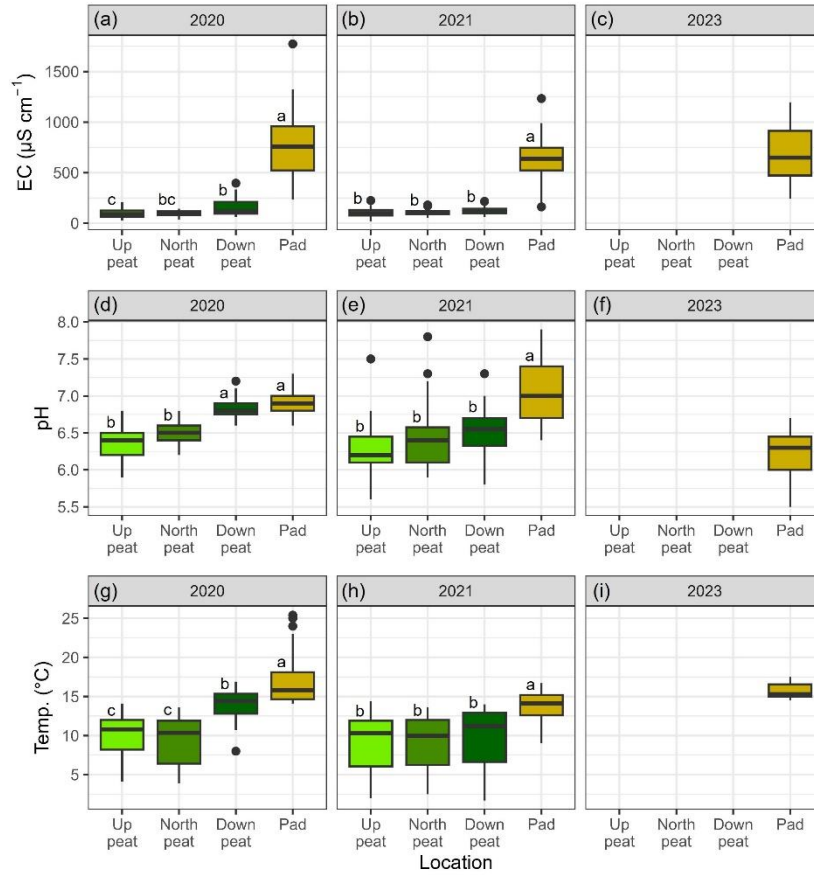


Figure 4.3: Shallow groundwater electrical conductivity (EC; a-c), pH (d-f), and temperature (g-i) within the pad and the upgradient ('Up peat'), north ('North peat'), and downgradient ('Down peat') peatlands. Measurements were not made in peatland areas in 2023. Groups not sharing common letters significantly differ based on either analysis of variance (all variables in 2020) or Kruskal-Wallis (all variables in 2021) tests.

4.3.4. Substrate Moisture and Temperature

On the residual pad, low microsites were significantly wetter in hydrologically *WO* areas than in hydrologically *NO* areas in the late season (July to September) in both 2021 and 2022 (the drier than average years; Figure 4.1 and Table 4.3). Notably, this difference was not observed in the early season (May to June) in those years, or at any time during the wetter than average growing seasons (Table 4.3). Low microsites in both *WO* and *NO* areas on the pad were generally drier than hollows in the peatland, likely reflecting the lower total porosity of the mineral substrate (0.38) as compared to the peat (0.95 to 0.97; McKinnon et al., 2024). Conversely, VWC at high microsites only significantly differed between *WO* and *NO* areas on the

Table 4.3: Median volumetric water content ($\text{m}^3 \text{m}^{-3}$; interquartile range in parentheses) for peat from the fen and the mineral substrate in the early season (ES) and late season (LS) each year. Medians for each location are grouped by microsite (hollow/hummock for the fen, low/high for the pad). When measurements were not available for a given location or season they are denoted by a ‘-’ symbol. Superscript letters denote the results of Kruskal-Wallis tests comparing medians between locations for a given microsite and time of season in a given year. Median values not sharing any common letters significantly differ from one another.

Microsite	Location	2020 ($\text{m}^3 \text{m}^{-3}$)		2021 ($\text{m}^3 \text{m}^{-3}$)		2022 ($\text{m}^3 \text{m}^{-3}$)		2023 ($\text{m}^3 \text{m}^{-3}$)		2024 ($\text{m}^3 \text{m}^{-3}$)	
		ES	LS	ES	LS	ES	LS	ES	LS	ES	LS
Hollow/ low	Peat (up)	0.91 (0.01) ^a	0.91 (0.01) ^a	0.88 (0.16) ^a	0.31 (0.24) ^a	-	-	0.51 (0.03) ^a	0.87 (0.08) ^a	-	-
	Peat (north)	0.76 (0.01) ^a	0.74 (0.04) ^{ab}	0.75 (0.18) ^{ab}	0.35 (0.10) ^a	-	-	0.45 (0.27) ^{ab}	0.93 (0.06) ^a	-	-
	Peat (down)	0.71 (0.01) ^{ab}	0.52 (0.07) ^{bc}	0.66 (0.29) ^{ab}	0.18 (0.06) ^b	-	-	0.40 (0.20) ^{ab}	0.85 (0.32) ^{ab}	-	-
	Pad (WO)	0.33 (0.01) ^b	0.32 (0.05) ^{cd}	0.40 (0.07) ^{bc}	0.28 (0.02) ^a	0.37 (0.02) ^a	0.38 (0.03) ^a	0.40 (0.06) ^{ab}	0.53 (0.06) ^{bc}	0.46 (0.05) ^a	-
	Pad (NO)	0.33 (0.01) ^b	0.27 (0.03) ^d	0.34 (0.06) ^c	0.17 (0.06) ^b	0.36 (0.03) ^a	0.29 (0.05) ^b	0.31 (0.08) ^b	0.43 (0.11) ^c	0.47 (0.05) ^a	-
Hummock/ high	Peat (up)	0.07 (0.02) ^{ab}	0.07 (0.04) ^b	0.09 (0.04) ^b	0.09 (0.02) ^{bc}	-	-	0.05 (0.02) ^b	0.08 (0.05) ^b	-	-
	Peat (north)	0.10 (0.17) ^{ab}	0.07 (0.04) ^b	0.06 (0.05) ^b	0.03 (0.03) ^d	-	-	0.03 (0.03) ^b	0.06 (0.04) ^b	-	-
	Peat (down)	0.05 (0.01) ^b	0.04 (0.04) ^b	0.05 (0.05) ^b	0.04 (0.01) ^{cd}	-	-	0.01 (0.02) ^b	0.06 (0.04) ^b	-	-
	Pad (WO)	0.28 (0.01) ^a	0.27 (0.05) ^a	0.31 (0.05) ^a	0.25 (0.05) ^a	0.31 (0.03) ^a	0.31 (0.02) ^a	0.28 (0.07) ^a	0.41 (0.07) ^a	0.46 (0.05) ^a	-
	Pad (NO)	0.27 (0.02) ^a	0.22 (0.05) ^a	0.28 (0.03) ^a	0.13 (0.07) ^b	0.30 (0.03) ^a	0.24 (0.06) ^b	0.26 (0.08) ^a	0.36 (0.10) ^a	0.34 (0.03) ^b	-

pad in the driest and wettest seasons studied (2022 and 2024, respectively), with no significant differences observed in years receiving rainfall close to the 30-year climate normal (Table 4.3). Also of note, VWC at high microsites on the pad was always either higher than, or comparable to VWC at hummocks in the peatland (Table 4.3).

The temperature of the mineral substrate did not significantly differ between *WO* and *NO* areas at any microsite position in either year for which temperature measurements were available (2022 and 2023; Table 4.4). Accordingly, median temperatures between the two areas fell within 1 °C of each other at all times (Table 4.4). In contrast, the temperature of the mineral substrate at low microsites was higher than that of peatland hollows late in the 2023 season (the year peatland temperature measurements were available; Table 4.4). At that time, the median temperatures between low microsites on the pad and hollows in the peatland differed by as much as 6.6 °C (Table 4.4). Some significant differences in temperature were also observed between low microsites on and off the pad in the early season that year, although the differences were of a lower magnitude (0.9 to 2.8 °C; Table 4.4). Patterns were similar for high microsites

in the early season (medians on and off the pad falling within 3.3 °C of each other), while no significant differences in temperature were observed between pad high microsites and peatland hummocks in the late season (Table 4.4).

4.3.5. Cation Supply

Cation supply on the residual pad was either comparable to, or higher than in the fen (Figure 4.4). The *WO* plots tended to have higher cation supply rates than *NO* plots at all microsite positions, although differences between plot types were often statistically non-significant (Figure 4.4). Cation supply also tended to be significantly higher at low microsites across the site than at high microsites (Figure 4.4). Reflecting the high total Ca, Mg, and Fe contents of the residual fill (Section 3.2), those cations had the greatest supply rates on the pad (Figure 4.4). Unsurprisingly, substrate temperature was not found to have a significant effect on cation supply (Supplementary Table C.3). However, VWC had significant positive associations with Fe and Mn supply rates in August 2020 (Fe: $t = 3.1$, $p = 0.014$; Mn: $t = 2.4$, $p = 0.041$) and June 2021

Table 4.4: Median temperatures (°C; interquartile range in parentheses) for peat from the fen and the mineral substrate in the early season (ES) and late season (LS) each year. Medians for each location are grouped by microsite (hollow/hummock for the fen, low/high for the pad). When measurements were not available for a given location they are denoted by a ‘-’ symbol. No temperature measurements were made in 2020, 2021, or 2024. Superscript letters denote the results of Kruskal-Wallis tests comparing medians between locations for a given microsite and time of season in a given year. Median values not sharing any common letters significantly differ from one another.

Microsite	Location	2022 (°C)		2023 (°C)	
		ES	LS	ES	LS
Hollow/ low	Peat (up)	-	-	17.4 (1.7) ^{ab}	10.0 (2.4) ^b
	Peat (north)	-	-	16.4 (3.0) ^b	10.6 (2.7) ^b
	Peat (down)	-	-	16.1 (1.9) ^b	10.9 (2.9) ^b
	Pad (WO)	20.0 (0.5) ^a	25.0 (1.5) ^a	18.3 (1.3) ^{ab}	16.6 (1.2) ^a
	Pad (NO)	19.3 (0.6) ^a	25.9 (1.2) ^a	18.9 (0.7) ^a	16.4 (1.2) ^a
Hummock/ high	Peat (up)	-	-	17.6 (0.8) ^b	19.5 (2.4) ^a
	Peat (north)	-	-	16.5 (2.5) ^b	19.5 (2.7) ^a
	Peat (down)	-	-	16.9 (1.5) ^b	17.5 (1.2) ^a
	Pad (WO)	21.1 (1.2) ^a	25.5 (2.1) ^a	18.8 (1.2) ^{ab}	17.6 (2.3) ^a
	Pad (NO)	20.5 (1.0) ^a	25.7 (1.7) ^a	19.8 (0.7) ^a	17.5 (2.8) ^a

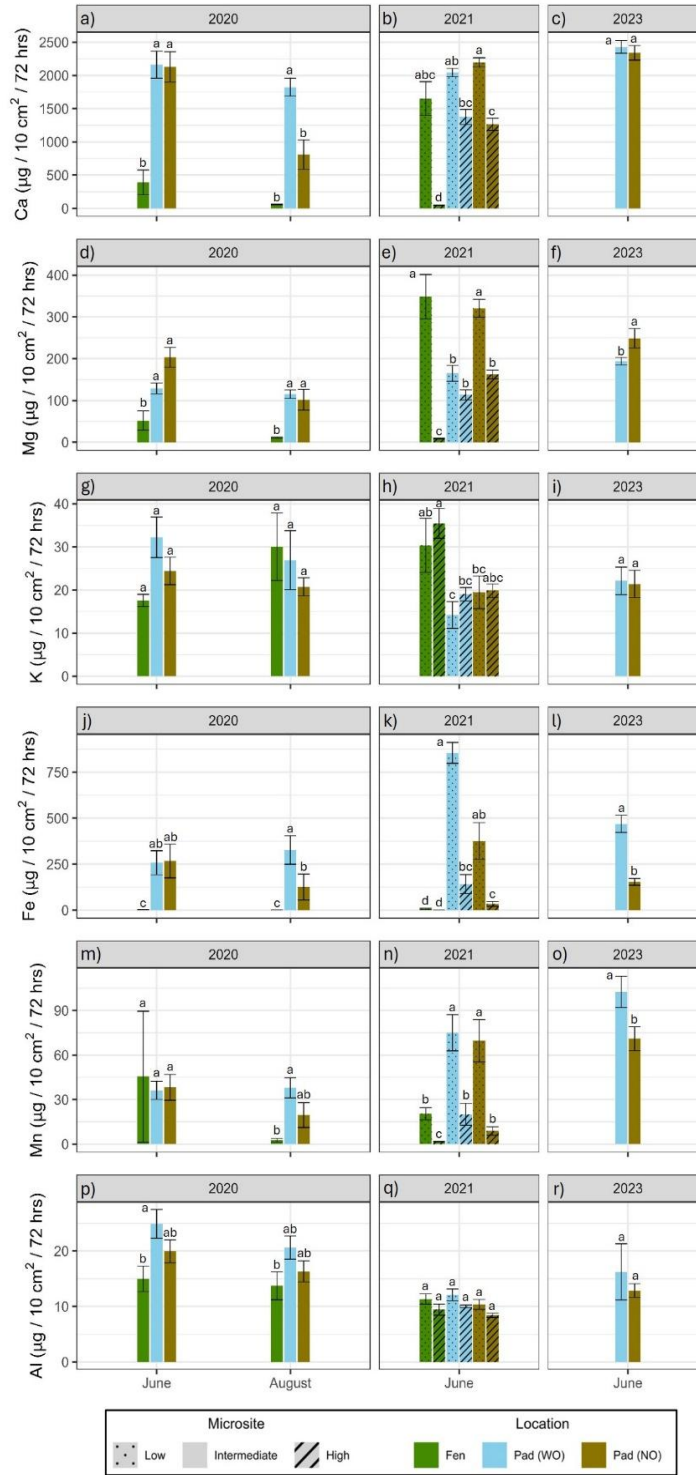


Figure 4.4: Cation supply rates measured using PRS probes in the fen and in the hydrologically well optimized (WO) and non-optimized (NO) areas on the pad (see Figure 1). Bars illustrate mean values ($\pm$ standard errors) for each group. Probes were installed at intermediate microsites in 2020 (June and August) and 2023 (June), and at low/hollow and high/hummock microsites in 2021 (June). PRS probes were not installed in the fen in 2023. The means of groups within a given year not sharing any common letters significantly differ based on analysis of variance.

(Fe: $t = 5.8$, $p < 0.001$; Mn: $t = 3.4$, $p = 0.003$), and with Ca supply rates in June 2021 ($t = 4.4$, $p < 0.001$). Unexpectedly, the supply of other cations was not significantly affected by VWC, although spatial differences in VWC in 2020 and 2023 were limited (Table 4.3).

Significant negative relationships were observed between fertilization and Fe ($t = -4.3$, $p = 0.002$) and Al ($t = -2.6$, $p = 0.031$) supply in August 2020. That month, PRS probes were installed for 72 hours immediately post-fertilization. Fertilization was not observed to have significant effects on the supply rates of any other cations in the first year, but did have significant positive relationships with Mg ($t = 2.7$, $p = 0.013$) and K ($t = 2.4$, $p = 0.024$) supply in June 2021. By 2023, no significant relationships were observed between cation supply and fertilization (Supplementary Table C.3). Mulching had a significant positive relationship with K supply in June 2020 ($t = 2.9$, $p = 0.017$) and June 2021 ($t = 2.1$, $p = 0.048$) and a significant negative relationship with Fe supply in August 2020 (post-fertilization; $t = -4.3$, $p = 0.003$). Similar to fertilization, mulching did not have a significant effect on cation supply by June 2023 (Supplementary Table C.3), by which time the mulch cover was substantially decomposed (visual observation).

4.3.6. Decomposition

Substrate VWC had a significant negative association with k values on the pad in 2021 ($t = -2.3$, $p = 0.049$) and 2023 ($t = -4.5$, $p < 0.001$), indicating that wetter conditions decreased decomposition rates. No significant relationships were observed between k values and fertilization, mulching, or substrate temperature in either year (Supplementary Table C.4). No spatial differences in k were observed between the peatland, *WO* plots, or *NO* plots at intermediate microsites in 2021 (Figure 4.5a). However, k did tend to be slightly lower in *WO* plots than elsewhere that year (Figure 4.5a). Reflecting this pattern, k was significantly lower in *WO* plots than in *NO* plots or the fen at low microsites in 2023 (Figure 4.5b). In

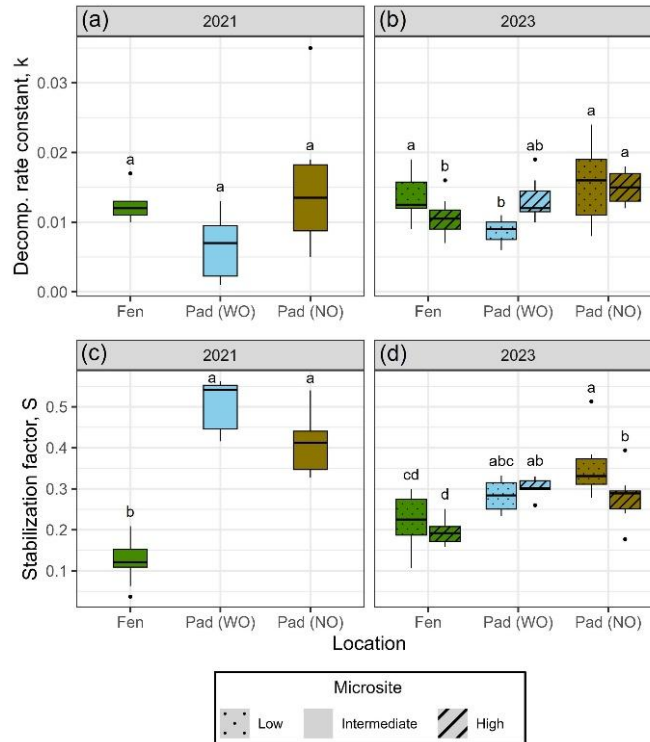


Figure 4.5: Decomposition rate constants, k (a-b) and stabilization factors, S (c-d) on the pad and in the fen. Groups within a given year not sharing any common letters significantly differ based on analysis of variance (k in 2023 and S in 2021 and 2023) or Kruskal-Wallis (k in 2021) tests. Due to differences in tea bag manufacturing processes, no direct comparisons were made between 2021 and 2023.

contrast, k was significantly lower for peatland hummocks than it was at high positions in *NO* plots that year (Figure 4.5b). Combined, values of k on the pad were either lower than, or not significantly different from those in the fen, with the exception of high microsites (where k was often higher on the pad than in fen hummocks). Unlike for k , VWC did not have a significant effect on values of S on the pad. However, fertilization was negatively associated with S in 2021 ($t = -4.2, p = 0.003$) and positively associated with S in 2023 ($t = 2.5, p = 0.019$). Substrate temperature was also negatively associated with S in 2023 ($t = -2.2, p = 0.039$). Spatially, S tended to be higher on the pad than in the fen in both 2021 and 2023 but did not vary between *WO* and *NO* plots on the pad in either year (Figure 4.5c-d).

4.3.7. Nitrogen Dynamics

Median concentrations of ext-NO₃⁻ (0.1 to 0.5 µg g⁻¹ dry soil) and ext-NH₄⁺ (0.1 to 2.6 µg g⁻¹ dry soil) in the pad were often significantly lower than in the fen in 2020 and 2021 (the years with fen measurements; Figs. 6a-b & 7a-b). They did not differ between *WO* and *NO* areas on the pad in any year (Figs. 6a-d & 7a-d). Interestingly, there was a considerable increase in concentration of ext-NO₃⁻ on the pad by June 2024 (12.3 to 13.8 µg g⁻¹ dry soil; Figure 4.6d). This was unexpected, as that month received 38 mm more rainfall than the 30-year mean, and similar increases in ext-NH₄⁺ or ext-PO₄³⁻ were not observed (Figs. 7d & 9d). Furthermore, large net negative NO₃⁻ transformation rates were observed on the pad in June 2024 (Figure 4.6h). Given that net NH₄⁺ production was observed at that time (Figure 4.7h), dissimilatory NO₃⁻ reduction to NH₄⁺ may have been occurring. However, because the magnitude of net NH₄⁺ production was smaller



Figure 4.6: Extractable NO₃⁻ pools (a-d) and daily net NO₃⁻ transformation rates (e-h) in the fen and in the hydrologically well optimized (WO) and non-optimized (NO) areas on the pad (see Figure 1). No data was collected in the fen in 2023 or 2024. Bars illustrate median values ± the 25th and 75th percentiles for each group (means and standard errors could not be computed due to the presence of censored observations in some groups). Medians of groups within a given year not sharing any common letters significantly differ based on Kruskal-Wallis, Wilcoxon Rank-Sum, or Peto-Peto tests.

than the net negative NO_3^- transformation rate, it is also possible that microbial uptake of NO_3^- was occurring. Prior to the increase in ext- NO_3^- , concentrations of it and ext- NH_4^+ were similar on the pad, whereas concentrations of ext- NH_4^+ tended to be higher than ext- NO_3^- in the fen (Figs. 6a-d & 7a-d).

Rates of NH_4^+ transformation on the pad were considerably smaller than in the fen in 2020 (Figure 4.7e). Net NH_4^+ production was observed in the fen in the early and late season that year, whereas net negative NH_4^+ transformation rates were observed on the pad in June (Figure 4.7e). Conversely, net NH_4^+ transformation rates did not differ between the pad and fen in 2021, although net negative NH_4^+ transformation was observed in the fen and *NO* areas as compared to net NH_4^+ production in *WO* areas (Figure 4.7f). In 2023 and 2024, net NH_4^+ transformation rates remained low and did not vary spatially across the pad, although they were negative in 2023 and positive in 2024 (Figure 4.7g-h). Comparatively, net NO_3^- transformation rates on the pad were negative in 2020 and 2024, but positive in July 2021 (Figure 4.6e-h). Low ext- NO_3^- values precluded calculation of transformation rates in June of 2021 and 2023. In the years with fen measurements, the magnitude and sign of net NO_3^- transformation rates did not differ between the fen and pad (Figure 4.6e-f). On the pad, there was a significant negative association between VWC and net NO_3^- ($t = -3.7$, $p = 0.002$) and NH_4^+ ($t = -4.3$, $p < 0.001$) transformation rates in 2020, although a similar effect was not seen in later years (Supplementary Table C.5). Although net N mineralization rates could not be quantified on the pad in any year (due to the combined proportion of censored ext- NO_3^- and ext- NH_4^+ observations exceeding 40%), this suggests that increased moisture may have adversely affected microbial efficiency in the first year post-partial removal. Interestingly, VWC was also positively associated with the ext- NO_3^- content of the mineral fill ($z = 2.5$, $p = 0.013$) in the same year (Supplementary Table C.6). Conversely, VWC was positively associated with ext- NH_4^+ on the pad by June 2023 ($z = 2.3$, $p = 0.021$).

Despite the lower ext- NO_3^- content on the pad than in the fen, no significant differences in pore water NO_3^- concentrations were observed between them in any year between 2021 to 2024 (Figure 4.8b-e). Similarly, few differences in pore water NO_3^- concentrations were observed between *WO* and *NO* areas

(Figure 4.8b-e). The 2020 growing season was an exception, as 94 % of pore water NO_3^- concentrations that year were $< \text{LOQ}$ ($13 \mu\text{g L}^{-1}$), precluding statistical analyses or data visualization. Observations of NO_3^- supply rates were also $< \text{LOQ}$ ($2 \mu\text{g } 10 \text{ cm}^{-2} \text{ 72 hours}^{-1}$) 72 % of the time across all years, which again precluded statistical analyses or data visualization. In comparison, median pore water NH_4^+ concentrations tended to be higher than those of NO_3^- in 2020 and 2021, falling between 12.3 and $211.7 \mu\text{g L}^{-1}$ in the fen,

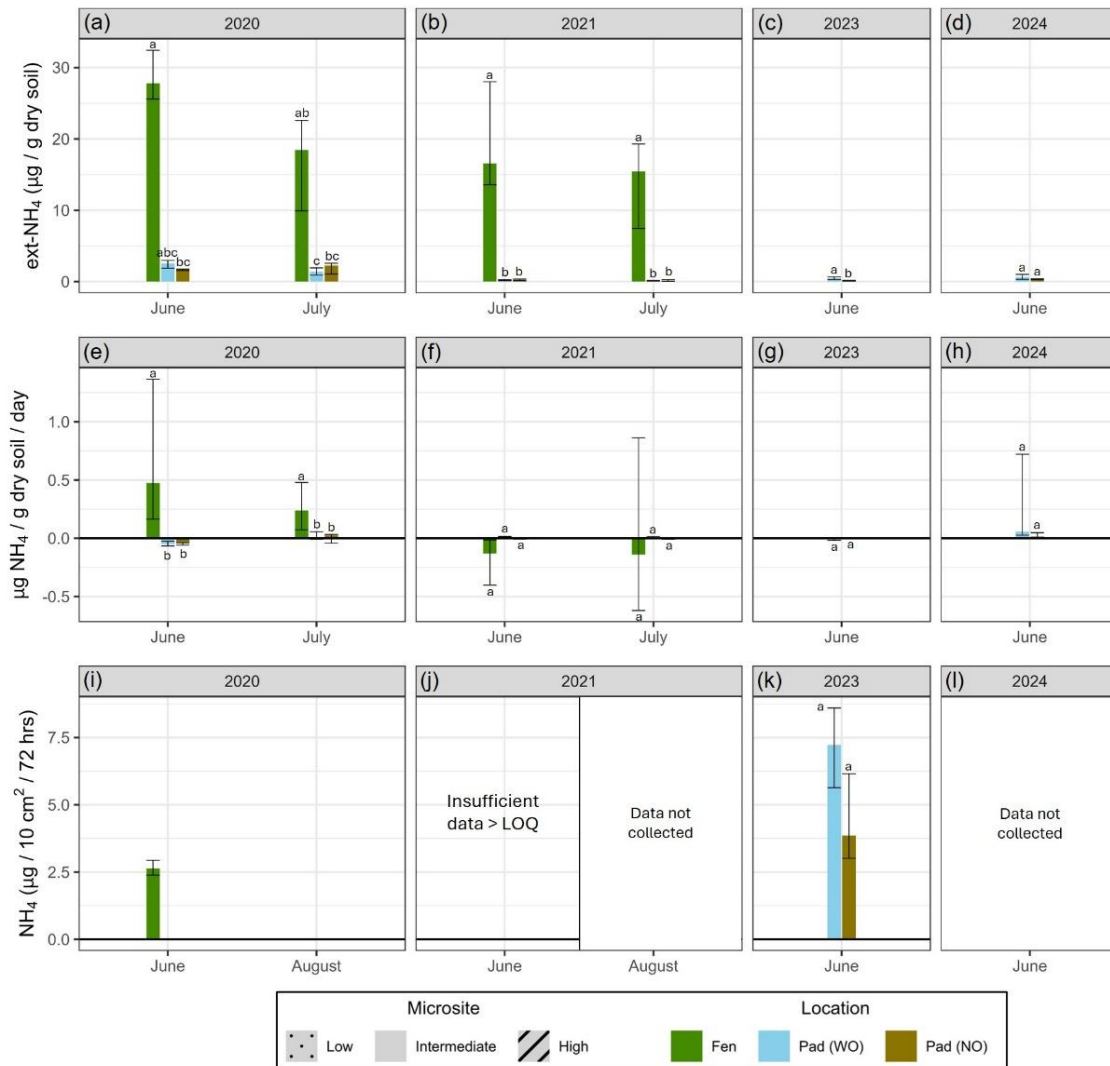


Figure 4.7: Extractable NH_4^+ pools (a-d), daily net NH_4^+ transformation rates (e-h), and NH_4^+ supply rates (i-l) in the surrounding fen and in the hydrologically well optimized (WO) and non-optimized (NO) areas on the residual pad (see Figure 1). No data was collected in the surrounding fen in 2023 or 2024. Bars illustrate median values $\pm$ the 25th and 75th percentiles for each group (means and standard errors could not be computed due to the presence of censored observations in some groups). Medians of groups within a given year not sharing any common letters significantly differ based on Kruskal-Wallis, Wilcoxon Rank-Sum, or Peto-Peto tests.

and between 12.6 and 300.1 $\mu\text{g L}^{-1}$ on the pad (Figure 4.7f-g). Despite relatively high pore water NH_4^+ concentrations on the pad, 75 % and 50 % of NH_4^+ supply rates there in 2020 and 2021, respectively, were $< \text{LOQ}$ ($2 \mu\text{g } 10 \text{ cm}^{-2} \text{ 72 hours}^{-1}$). By June 2023, however, NH_4^+ supply rates were above detectable limits, ranging between 3.9 and 7.2 $\mu\text{g } 10 \text{ cm}^{-2} \text{ 72 hours}^{-1}$ (Figure 4.7k). Notably, NH_4^+ supply rates did not vary between *WO* and *NO* plots at that time (Figure 4.7k). In terms of environmental and treatment effects, fertilization was positively associated with pore water NO_3^- concentrations in July 2021 ($t = 2.7, p = 0.027$;

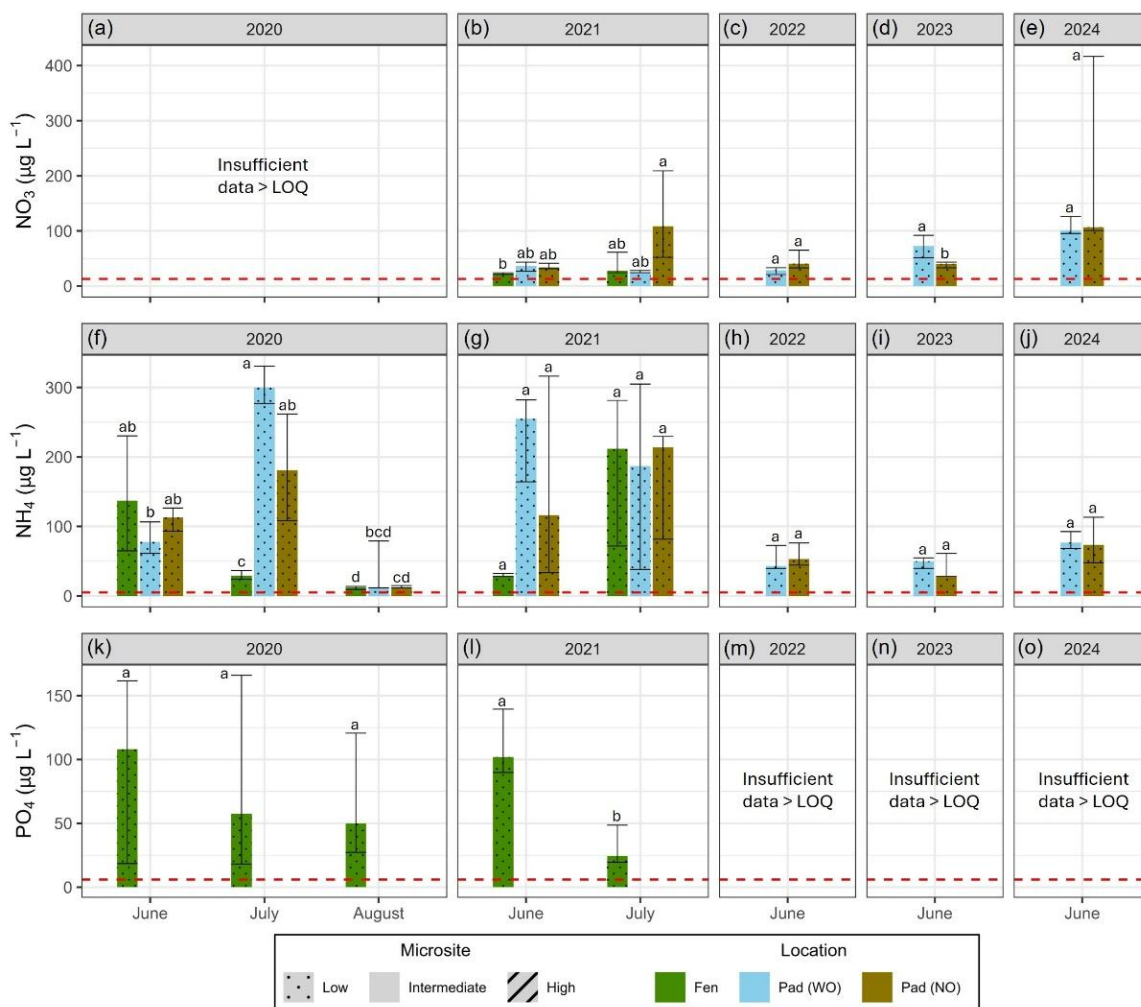


Figure 4.8: Pore water nitrate (NO_3^- ; a-e), ammonium (NH_4^+ ; f-j), and phosphate (PO_4^{3-} ; k-o) concentrations in the fen and in well optimized (WO) and non-optimized (NO) areas on the pad (see Figure 1). In the years 2022-2024, pore water was not collected in the fen. Bars illustrate median values $\pm$ the 25th and 75th percentiles for each group (means and standard errors could not be computed due to the presence of censored observations in some groups). Medians of groups within a given year not sharing any common letters significantly differ based on Kruskal-Wallis, Wilcoxon Rank-Sum, or Peto-Peto tests. Limits of quantitation are denoted by dashed red lines for each parameter.

Supplementary Table C.7). Mulching was associated with reductions in net NO_3^- transformation rates ($t = 2.4$, $p = 0.045$) in July 2020, and increases in ext-NH_4^+ ($z = 2.6$, $p = 0.010$) and pore water NH_4^+ concentrations in June 2021. Substrate VWC was negatively associated with pore water NH_4^+ concentrations in 2020 ($z = -2.8$, $p = 0.006$) and 2023 ($t = -3.3$, $p = 0.013$), whereas it was positively associated with NH_4^+ supply rates in 2023 ($t = 2.6$, $p = 0.036$).

4.3.8. Phosphorus Cycling and Supply

Concentrations of ext-PO_4^{3-} were $< \text{LOQ}$ ($18 \mu\text{g L}^{-1}$ for the liquid extract) 83% of the time on the pad, precluding statistical analyses or visual presentation of the data (Figure 4.9a-d). Notably, the median value of ext-PO_4^{3-} measurements on the pad that were $> \text{LOQ}$ was only $0.09 \mu\text{g g}^{-1}$ of dry soil. Comparatively, ext-PO_4^{3-} levels in the fen were considerably higher, at $20 \mu\text{g g}^{-1}$ of dry soil (Figure 4.9a-b). The low levels of ext-PO_4^{3-} on the pad were also reflected in the fact that pore water SRP was $< \text{LOQ}$ ($18 \mu\text{g L}^{-1}$) 97 % of the time there (Figure 4.7k-o). In the peatland, median pore water SRP usually ranged from 50 to $100 \mu\text{g L}^{-1}$, although the median in July 2021 was significantly lower ($24.4 \mu\text{g L}^{-1}$) than on other dates (Figure 4.7l). Nonetheless, median P supply rates on the pad tended to be comparable to (and in some cases significantly higher than) those in the fen in both 2020 and 2021 (the years peatland measurements were available; Figure 4.9i-j). Owing to the high proportion of ext-PO_4^{3-} values $< \text{LOQ}$, net P mineralization rates on the residual pad could not be reliably quantified there in most cases (Figure 4.9e-h). In the surrounding fen, net P mineralization was observed throughout the season in 2020, and in the late season in 2021 (Figure 4.9e-f). Conversely, net P immobilization was observed in the fen earlier in 2021 (Figure 4.9f).

Unexpectedly, application of rock phosphate fertilizer had a significant negative relationship with P supply rates in August 2020 ($t = -1.1$, $p = 0.014$), although it did not have any significant association with P supply in subsequent years (Supplementary Table C.3). In June 2021, for example, only VWC was found to have a significant (positive) relationship with P supply ($t = 4.7$, $p < 0.001$). However, P supply rates did not differ significantly between *WO* and *NO* plots at that time (Figure 4.9j) owing to the relatively consistent

VWC values observed across the pad that year (Table 4.3). No environmental variables were found to directly affect P supply rates in 2023 (Supplementary Table C.3), and no significant differences in P supply rates between plot types on the pad were observed that year (Figure 4.9k).

Reflecting the small size of the bioavailable P pool on the pad, the N:P ratio calculated using PRS probe supply rates there in June 2023 was 5:1 (Supplementary Table C.8). Notably, this value was somewhat higher than that calculated using substrate total N and P values (1:1; Table 4.1). Supply rate-based N:P

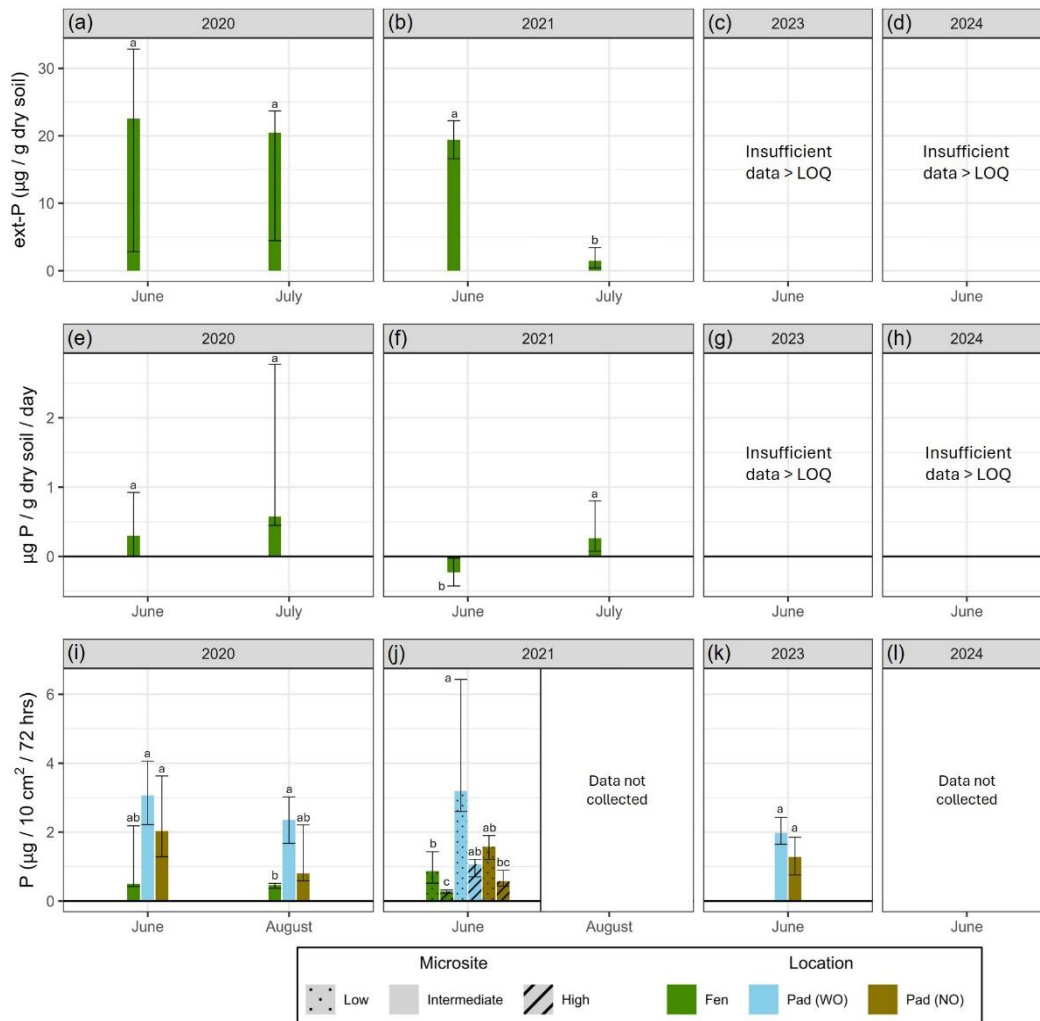


Figure 4.9: Extractable PO_4^{3-} pools (a-d), daily net P mineralization rates (e-h), and P supply rates (i-l) in the surrounding fen and in the hydrologically well optimized (WO) and non-optimized (NO) areas on the residual pad (see Figure 1). No data was collected in the surrounding fen in 2023 or 2024. Bars illustrate median values $\pm$ the 25th and 75th percentiles for each group (means and standard errors could not be computed due to the presence of censored observations in some groups). Medians of groups within a given year not sharing any common letters significantly differ based on Kruskal-Wallis, Wilcoxon Rank-Sum, or Peto-Peto tests.

could not be calculated prior to June 2023 due to the low NO_3^- and NH_4^+ supply rates on the pad in 2020 and 2021 (Figs. 6 & 7), although supply rate-based N:P in the fen ranged from 6:1 to 8:1 between 2020 and 2023 (Supplementary Table 4.8). Interestingly, the supply rate-based Fe:P ratio on the pad ranged between 91:1 and 195:1 between 2020 and 2023, which was considerably higher than both the Fe:P ratio calculated using substrate total values (30:1; Table 4.1) and that of peat in the fen (4:1 to 7:1; Supplementary Table C.8). The supply rate-based Al:P ratio on the pad was comparatively lower, ranging from 10:1 to 13:1 between 2020 and 2023.

4.4. Discussion

4.4.1. Physicochemical Characterization

The fact that the residual mineral fill was calcareous (Table 4.1) was likely a result of its original excavation from the substratum of an upland boreal forest soil (Green et al., 1992). Owing to the calcareous nature of the fill, the pH and EC values of shallow groundwater across the pad were aligned with those typical of moderate and extreme-rich fens on the WBP, respectively (Figure 4.3). Previous trials of partial well pad removal have identified the importance of ensuring that moss propagules are collected from donor fens with similar pH, EC, and Ca contents as the residual pads they are introduced onto (M. E. Gauthier et al., 2018). Thus, these results suggest that the residual pad should provide suitable conditions for the initiation of fen true mosses, at least in areas where moss water availability is optimized (M. E. Gauthier et al., 2018; Lemmer et al., 2022). However, the fact that cation supply rates and groundwater pH and EC were frequently higher on the pad than in the adjacent moderate-rich fen suggests that the site may be best suited for development of an extreme-rich fen vegetation community, rather than those of less minerogenous peatlands (Vitt and Chee, 1990). Notably, the EC of shallow groundwater in the donor site used in the present study was aligned with poor fens on the WBP despite its close proximity to the pad (Figure 4.1c), highlighting the need for detailed physicochemical characterization to be undertaken prior to selection of moss donor sites.

The calcareous nature of the mineral fill may also explain the often elevated pH and EC of shallow groundwater in the fen downgradient of the pad. This difference between upgradient and downgradient sides of the pad is contrary to the findings of Vitt et al. (2011), who identified no significant effect of two residual pads on pH and EC in a surrounding bog. However, that study did not assess the degree of hydrological connectivity between the pads and the bog in detail. Conversely, the results of McKinnon et al. (2024) indicate that the residual pad examined here was hydrologically connected to the fen. As such, the downgradient fen frequently received water inputs from the pad (McKinnon et al., 2024). While low sample sizes precluded detailed comparison of cation supply rates between sides of the pad, the flow from the pad could presumably have resulted in export of base cations to the downgradient fen, potentially increasing the EC of shallow groundwater there (Vitt and Chee, 1990).

4.4.2. Organic Matter and Decomposition

The fact that values of k on the pad were either comparable to, or lower than those in the fen may indicate that the mineral substrate was sufficiently wet to limit decomposition rates to a similar extent as seen in natural fens on the WBP (Vitt et al., 2009). The negative association between VWC and k supports this possibility, although it is also possible that microbial decomposition may instead have been inhibited by the low OM content on the pad (2.1% to 3.1%) as compared to the fen (91% to 95%; McKinnon et al., 2024). Even though the ratios of substrate TC to total N (35:1) and P (34:1) were high, the TC content of calcareous mineral substrates is not fully representative of bioavailable carbon as it is inclusive of the inorganic carbonate (i.e., non-bioavailable) fraction. Levels of dissolved organic carbon have also been found to be lower on residual pads than in surrounding peatlands (Lemmer et al., 2020), which may further limit the availability of organic carbon. Such limitations may explain why high substrate temperatures did not result in higher values of k as was expected (Keuskamp et al., 2013), despite the pad being several degrees warmer than the adjacent fen.

Either way, the low values of k on the pad are an indicator that carbon accumulation can be expected, at least in the short term, which would align with the results of previous trials that found residual mineral substrates to be net carbon sinks eight to ten years post-partial removal (Lemmer, 2022; Lemmer et al., 2020). Contrary to values of k , values of S were generally higher on the residual pad than in the fen (Figure 4.5). This could again have been a result of potentially limited microbial activity on the pad, although further study of microbial activity would be required to confirm this. Given that greater amounts of stabilization result in an increased fraction of the soil OM being inaccessible for microbial decomposition (Keuskamp et al., 2013), the high S values on the pad may further contribute to the accumulation of OM over time. Notably, however, decomposition rates could increase as OM builds up if current values of k are in fact limited by the present availability of OM.

4.4.3. N and P Dynamics

Lower concentrations of ext-NO_3^- and ext-NH_4^+ in the residual pad compared to the fen suggest that the bio-available N pool on the pad was small compared to natural peatlands in the region. This is likely a result of the limited OM content of the fill, as the low rates of decomposition on the pad would have limited the degree to which N mineralization could have occurred. As such, the large increase in ext-NO_3^- by June 2024 suggests that there must have been an external source of N, at least in the later years of the study. Preliminary vegetation surveys across the site found considerable establishment of *Melilotus officinalis*, which can fix N from the atmosphere (data not shown). Combined with the potential for increased microbial decomposition of an increasingly thick sedge litter layer by that growing season (data not shown), it appears that increases in the availability of N over time can be expected. This is further supported by the increase in NH_4^+ supply rates observed by 2023 (prior to that date, both NH_4^+ and NO_3^- supply rates were < LOQ). Nonetheless, the fact that the increase in ext-NO_3^- was associated with large net negative NO_3^- transformation rates, but not large net positive NH_4^+ transformation rates, suggests that the available N at that time was rapidly immobilized by the microbial community. This may indicate that the microbial

community was N limited, which is further supported by the low supply rate-based N:P ratio calculated for the mineral fill in June 2023 (5:1). Productivity in natural peatlands can generally be considered to be N limited when N:P < 15:1 (Nwaishi et al., 2022; Wang et al., 2015).

The low supply rate-based N:P ratio was also due in part to the fact that P supply rates on the residual pad were either comparable to, or higher than those in the surrounding fen. These high supply rates were unexpected, as the lower ext- PO_4^{3-} and pore water SRP concentrations observed on the pad as compared to the fen were likely a result of P fixation within the mineral substrate. Fixation of P can occur readily in mineral substrates, wherein inorganic P becomes non-bioavailable through complexation with Fe and Al (Dunne and Reddy, 2005; Lucas and Davis, 1961). Notably, Fe was likely the dominant driver of P fixation, given the high Fe:P ratios calculated using total substrate nutrient contents (30:1) and nutrient supply rates (between 91:1 and 195:1). Combined with the low rates of decomposition (and thus, P mineralization; Figure 4.9e-h) on the pad, this may account for the lower ext- PO_4^{3-} and pore water SRP concentrations observed on the pad as compared to the fen. Notably, fens on the WBP have naturally high levels of P (Plach et al., 2016; Vitt et al., 1995).

Indeed, the solubility of P, and consequently the stability of P-Fe complexes, is dictated by redox conditions (Reddy et al., 2023). Across much of the residual pad, large fluctuations in the water table (McKinnon et al., 2024) resulted in rapid changes in VWC in response to rainfall events (McKinnon et al., 2025). As such, rapid changes in redox conditions would likely also have occurred, which could have temporarily mobilized P and elevated P supply rates. However, consistency in the rates of P supply on the pad (including in the wetter early season and drier late season; Figure 4.9i-l) suggest that fluctuations in redox conditions may not have been the dominant control over P availability. This may be due in part to the fact that formation of complexes between Ca and ferric Fe can occur under oxidizing conditions (Reddy et al., 2023), particularly in calcareous substrates such as the residual mineral fill. Thus, should P have been solubilized through dissolution of P-Fe precipitates during drier conditions, it may have been rapidly immobilized again through complexation with Ca.

Interestingly, there is some evidence to suggest that the presence of humic acids, which are found in high levels in natural peatlands (Rydin and Jeglum, 2006b), may result in the dissolution of Al and Fe phosphates through chelation (Lobartini et al., 1998) and inhibit the subsequent formation of Ca phosphates (Ge et al., 2020). Given that the residual pad received shallow groundwater inputs from the fen across its upgradient (east) and north edges (McKinnon et al., 2024), it is possible that at least some degree of humic acid input may have occurred. Visual observations of a sheen on standing water across the pad in later study years also support this hypothesis (data not shown). Regardless of the mechanism of P solubilization, the low bio-available pool of P on the pad, when combined with the high P supply rates, suggest that rapid microbial immobilization or plant uptake of any soluble P must have been occurring.

4.4.4. Impact of Fertilization

Application of rock phosphate fertilizer (N-P-K of 2-13-0 at a rate of 150 kg ha⁻¹) was unexpectedly associated with an immediate reduction in the supply of P at the surface of the residual pad. However, fertilization was also associated with immediate reductions in the supply rates of Fe and Al, suggesting that inorganic P added through fertilization became non-bioavailable as a result of rapid complexation with Fe and Al (Dunne and Reddy, 2005; Lucas and Davis, 1961). By 2021, no significant relationship was observed between fertilization and P, Fe, or Al supply rates, indicating that the impact of fertilization on P supply was limited to the first growing season. Conversely, fertilization was associated with increased Mg and K supply rates, increased pore water NO₃⁻ concentrations, and decreased *S* values by 2021. The increase in pore water NO₃⁻ in fertilized plots was likely a result of the fertilizer containing 2% N on a gravimetric basis. The increase in Mg and K supply rates may have been related to changes in soil pH, which can occur following fertilization (Barrow et al., 2022). The negative association between fertilization and *S* may have been caused by a temporary increase in microbial biomass, given that the addition of inorganic P can alleviate microbial P limitations (Cleveland et al., 2002; Fisk et al., 2015). However, because values of *k* were not significantly affected by fertilization and because N may have been the primary limiting nutrient, decreases

in the stabilization of substrate OM could instead have been associated with desorption of OM from soil mineral surfaces (Spohn and Schleuss, 2019). This can occur when inorganic P exchanges with sorbed organic compounds, making those compounds less stable and more available to the microbial community for decomposition (Spohn et al., 2022).

Conversely, fertilized plots were associated with increased *S* values by the 2023 growing season. This contrasting result may indicate that fertilization with P actually enhanced the formation of stable organo-mineral complexes over the longer-term. This could have occurred as a result of partial decomposition of new moss and sedge litter inputs and any organic compounds desorbed from mineral surfaces following initial fertilization. Specifically, the combined effects of C loss during decomposition (as a result of microbial respiration) and the build-up of N and P rich microbial biomass can result in enrichment of soil OM with N and P (Xu et al., 2013). Increases in the N and P content of OM has been found to increase its affinity to adsorb to soil minerals (due to the high charge densities and number of functional groups associated with organic N and P), thereby increasing the resistance of OM to decomposition (Spohn, 2024). Thus, fertilization may have slightly increased the carbon sequestration capacity of the mineral fill. As such, there is a need for further detailed study of microbial dynamics on residual well pads, given the potential for microbial-mediated decomposition and mineralization to control nutrient availability and organic matter stability on these sites.

Nonetheless, it appears that fertilization of residual mineral substrates with P has limited efficacy for improvement of P availability. In the context of peatland construction (Nwaishi et al., 2016) and cutover peatland restoration (Meilleur et al., 2022; Sottocornola et al., 2007), P fertilization has been identified as an essential step to enhance moss productivity (Groeneveld et al., 2007). However, many of these sites have low available P relative to reference ecosystems (Wind-Mulder and Vitt, 2000), or have high N:P ratios (Nwaishi et al., 2016). While ext- PO_4^{3-} was indeed lower on the residual pad than in the surrounding peatland in the present study, the fact that P supply rates tended to be significantly higher on the pad than in the fen suggests that P addition may not be necessary on residual well pads.

4.4.5. Impact of Straw Mulch

Application of straw mulch onto the surface of the residual mineral substrate during the winter of 2020 was positively associated with K supply rates early in the growing season in both 2020 and 2021. This was likely a direct result of decomposition of the straw mulch itself, which has been shown to increase available K in a range of agricultural soils (Akhtar et al., 2019; Fan et al., 2021). The implications of this association may be limited, however, as K is not usually a limiting nutrient for fen vegetation growth (Wang et al., 2015). Mulched plots were also associated with decreases in Fe supply in August 2020, and increased rates of production of both NO_3^- and NH_4^+ in June 2021.

The decrease in Fe supply was observed at the same time as the decrease associated with fertilizer application, although low sample sizes precluded statistical analysis of the interaction term between the two variables. Nonetheless, mulching may have increased the quantity of insoluble Fe oxides somewhat, which can occur due to the formation of stable complexes between Fe oxides and organic matter derived from the mulch (Li et al., 2023; Xue et al., 2022). Increased rates of production of NO_3^- and NH_4^+ may also have been the result of decomposition of the straw mulch, which may have provided a source of organic N for microbial-mediated mineralization (Scheller and Joergensen, 2008). However, no significant associations with N dynamics were observed after the second growing season, and it can thus be concluded that the potential biogeochemical effects of mulching are short-term in nature. Combined with the limited effect of straw mulch in improving moisture dynamics on the same residual pad examined here (McKinnon et al., 2025), this suggests that the application of straw mulch is likely not a necessary restoration treatment on coarse-textured (i.e., sandy) residual well pads. However, as straw mulch may be more effective in improving moisture levels on fine-textured residual pads (McKinnon et al., 2025), additional study of its biogeochemical effects on clay pads is recommended.

4.4.6. Spatial Variability on the Pad and Potential Successional Trajectory

While significant differences in the physicochemical and nutrient variables assessed in this study were often observed between the residual pad and the fen, many did not vary (or did not vary consistently) between *WO* and *NO* areas on the pad itself. This is surprising, given that an associated study found that the water table remained near the surface for most of the growing season in *WO* areas (owing to a high degree of hydrological connectivity with the fen) but frequently fell far below the surface in *NO* areas (McKinnon et al., 2024). These spatial differences in hydrological regulation were found to result in consistent water availability to mosses in *WO* plots, but not in *NO* plots (McKinnon et al., 2025). As such, it appears that biogeochemical dynamics on the residual pad may have been less sensitive to hydrological regulation than moss water availability was, thus creating a similar nutrient regime across the site. While the bio-available N and P pools were small across the pad, the high supply of base cations suggests that the site had a rich to very-rich nutrient regime (Beckingham and Archibald, 1996).

The consistency of the nutrient regime suggests that the entire site may support similar vegetation communities over the long-term, although differences in moisture may result in some degree of successional divergence. Specifically, the moisture regimes in *NO* and *WO* areas can best be described as ranging from sub-hygic to hygric and sub-hydric to hydric, respectively (Beckingham and Archibald, 1996; McKinnon et al., 2025). Nutrient-rich ecosites in the boreal forest with sub-hygic to hygric soils similar to *NO* areas on the pad are commonly associated with *Cornus sericea*, *Rhododendron groenlandicum*, *Ribes* spp., *Rubus* spp., *Equisetum* spp., *Populus* spp., *Picea glauca*, and various ferns (Beckingham and Archibald, 1996). In comparison, nutrient-rich ecosites with sub-hydric to hydric soils like those found in *WO* areas on the pad are typically characterized by a greater number of wetland-typical species, including *Rhododendron groenlandicum*, *Vaccinium oxycoccos*, *Typha latifolia*, *Carex* spp., *Scirpus* spp., *Picea mariana*, *Larix laricina*, and brown mosses (Beckingham and Archibald, 1996). As such, it appears likely that *WO* areas on the pad will support a greater establishment of peatland-typical species than *NO* areas, despite the similar nutrient regime between the two. Nonetheless, further study will be required to determine actual long-term

vegetation establishment outcomes and to capture the effects of any change in the physicochemical properties of the mineral substrate over time. Such changes, including changes in the availability of N and P, can have a considerable impact on the successional trajectories of boreal ecosystems undergoing restoration (Audet et al., 2015; Pinno and Hawkes, 2015).

4.5. Conclusion

The similarity between the pH, EC, and cation supply rates measured on the residual well pad and in moderate to extreme-rich fens on the WBP suggests that the establishment of a fen true moss community should be supported where water availability to mosses is optimized. However, the relative availabilities of N and P suggest that vegetation productivity may be N limited, at least in the early stages of initiation. This may change over time as the establishing vegetation community provides a source of labile organic matter for microbial-mediated decomposition. Nutrient regimes were relatively consistent across the site. As such, any differences in vegetation establishment between *WO* and *NO* areas are likely to be driven by differences in moisture regime. Specifically, the well regulated water tables in *WO* areas are more likely to support succession towards a peatland vegetation community than the poorly regulated water tables in *NO* areas, where succession towards a vegetation community dissimilar to the surrounding reference fen may occur. Application of rock phosphate fertilizer had a negative effect on P supply rates in the short-term, and no discernable effect on P supply in the long-term owing to geochemical reactions between P and the cation-rich mineral substrate. Thus, application of rock phosphate fertilizer to residual well pads may not be necessary. Application of straw mulch had a similarly limited biogeochemical effect. When combined with its limited efficacy in improving moisture levels on coarse-textured residual pads, this suggests it may also be an unnecessary treatment, although further study of its use on fine-textured residual pads is recommended. Combined, this study contributes to the growing body of scientific literature indicating that the partial removal of well pads has potential as a strategy to set the tens of thousands of well pads

constructed in boreal peatlands to date on a successional trajectory towards supporting peatland ecosystem function.

5 A COMPARISON OF HYDROLOGIC FLUXES IN A PARTIALLY REMOVED WELL PAD UNDERGOING FEN RESTORATION ON THE WESTERN BOREAL PLAIN, CANADA

5.1. Introduction

The Western Boreal Plain (WBP) exists in a subhumid climate and is characterized by a mosaic of hydrologically connected upland forests and low-lying peatlands (Devito et al., 2005; Thompson et al., 2015; Volik et al., 2020). The latter are characterized by a ground layer of mosses that sequester immense amounts of carbon (Harris et al., 2021) and regulate various hydrological processes including evapotranspiration (Brown et al., 2010). However, the region hosts numerous industries including upstream oil and gas, which has considerably altered ecosystem function (Ficken et al., 2019; Volik et al., 2020). In total, open-pit oil sands mining is likely to result in the loss of over 29,500 ha of peatland (Rooney et al., 2012), while well pads constructed for in-situ drilling activities have resulted in a further loss of up to 36,000 ha of peatland to date (UNEP, 2022). Although much attention has been given to the reconstruction of peatlands lost to open-pit mining (Borkenhagen et al., 2024; Ketcheson et al., 2017; Nwaishi et al., 2015a; Popović et al., 2022), comparatively little work has been done to develop techniques to rebuild peatland ecosystems on decommissioned well pads (Chimner et al., 2017).

One method that appears to have promise as a way to reestablish fen vegetation on well pads involves the partial removal of the mineral fill (sand or clay) used to construct a pad. This involves the use of heavy machinery to ‘shave’ the uppermost layers of the fill back, leaving a residual layer of mineral fill in place at the surface (Bird and Xu, 2021b). A modified version of the Moss Layer Transfer Technique (MLTT) is then applied (Rocheffort et al., 2003). In brief, the MLTT involves using an excavator to collect mosses and fragments of vascular peatland species from the upper 10 cm of a donor fen. These are then transported to the restoration site. There, a manure spreader or compact skid steer can be used to spread the donor material over the residual mineral substrate in a thin layer (Bird and Xu, 2021a).

As long as the water table (WT) across coarse-textured residual well pads remains within approximately 6 cm of the surface, optimal water availability to the introduced mosses is maintained (McKinnon et al., 2024). Indeed, diverse communities of true mosses and sedges have been observed to develop over the eight to ten years following partial removal in areas where high soil moisture is observed (M. E. Gauthier et al., 2018; Lemmer et al., 2022; Vitt et al., 2011). However, the low horizontal hydraulic conductivity (K_h) of residual mineral substrates restricts hydrological connectivity across residual pads (McKinnon et al., 2024). This can lead to poorly regulated WTs that can result in non-optimal soil moisture availability to mosses, or exceedance of moss desiccation thresholds (McKinnon et al., 2025).

In natural peatlands, near-surface WTs are maintained by interactions between precipitation and evapotranspiration (Van Huizen et al., 2020; Volik et al., 2021), groundwater fluxes (Elmes and Price, 2019; Ferone and Devito, 2004), and feedbacks that restrict the depth of the WT (Howie and Hebda, 2018; Waddington et al., 2015). Water conservation and storage in peatlands on the WBP are especially important due to the region's subhumid climate, meaning that potential evapotranspiration often exceeds precipitation (Brown et al., 2010). In peatlands underlain by coarse-textured till, groundwater can provide a source of water (Elmes and Price, 2019), while the high K_h of near-surface peat permits rapid lateral distribution of water when WTs are high (Waddington et al., 2015). Conversely, lower K_h values at depth serve to restrict water losses when WTs are low (Quinton et al., 2008; Waddington et al., 2015). When WTs do drop, capillary flow is often maintained by surface elevation adjustment, wherein deformation of the peat pore structure upon desaturation results in decreases in the peatland surface elevation (Howie and Hebda, 2018). The living moss layer can also regulate evaporative losses, whereby increases in albedo and changes in moss growth forms in response to dry conditions can reduce available energy and capillary flow, respectively (Brown et al., 2010; McCarter and Price, 2014).

Residual well pads lack many of the controls on water table elevation found in natural peatlands. For instance, the rigid pore structure of mineral fill means that the surface elevation of residual pads remains static. This allows the WT to fall farther below the surface than in natural peatlands (McKinnon et al.,

2024), likely resulting in more frequent disconnection of the capillary fringe from establishing mosses (i.e., stage III evaporation; Dingman, 2015). Moss layers on the surface of residual well pads are also often discontinuous, particularly during the early stages of vegetation establishment (M. E. Gauthier et al., 2018; Lemmer et al., 2022). Thus, controls over available energy and capillary flow within moss growth forms will not be consistent across the surfaces of these features.

Notably, any limitations on soil moisture availability at the surface of residual pads are likely to have a large impact on the initiation and establishment of peatland mosses, given that they lack both vascular and rooting systems (Proctor et al., 2007). As such, there is concern that residual well pads may be better suited for the establishment of peatland vascular species including sedges and rushes (see Thesis Sections 3 and 4), which can access water at depth (Chapin and Chapin, 1981; Koropchak et al., 2012). As mosses are the foundation of peatland ecosystem function (Vitt et al., 2009) and moss-dominated communities have been identified as a desirable reclamation target (Alberta Environment and Parks, 2017), there is an evident need to better understand the components of the water balance of residual well pads. In particular, there is a need to maximize water inputs and to minimize water losses from them in order to maintain the WT within the upper 6 cm (McKinnon et al., 2025, 2024). This will be of critical importance for moss establishment on restored well pads given the increased frequency of droughts projected for the WBP as a result of climate change (Price et al., 2013; Thompson et al., 2017).

One peatland restoration strategy commonly employed to increase substrate moisture is the placement of straw mulch on the surface (González and Rochefort, 2014; Price et al., 1998). Mulch can create favourable near-surface microclimatic conditions, while also reducing available energy by reflecting incoming radiation (Novak et al., 2000b, 2000a; Petrone et al., 2004; Price et al., 1998). The effect of mulch during peatland restoration appears to be greatest during stage II evaporation (i.e., when evaporation is soil-water limited). Specifically, no direct effects on residual mineral substrate moisture content have been observed under saturated conditions (M. E. Gauthier et al., 2018) or on dry residual mineral substrates when evaporative demand was likely supplied by vapour fluxes (McKinnon et al., 2025). However, no work to

date has tested the effect of straw mulch on evapotranspiration from residual mineral well pads. Thus, it is unknown to what extent mulch may contribute to reducing overall evaporative losses from these sites.

Previous assessments of hydrological dynamics on residual well pads have also identified the likely importance of snowmelt in sustaining near-surface WTs early in the growing season, but did not directly quantify snowmelt inputs (McKinnon et al., 2024). As such, there is a need to better understand the role of snowmelt and ground frost in regulating spring ecohydrological dynamics on residual well pads. Another consideration is that the mineral fill removed during partial removal is often used to construct upland features on a portion of the original pad footprint (Drozdowski et al., 2020). This mitigates disturbance offsite and results in adjacent residual pad-constructed upland systems. However, it remains unclear whether these two systems will be hydrologically coupled. Surface runoff from natural upland forests to peatlands tends to be limited on the WBP due to the high storage and transmission capacities of upland boreal mineral soils (Devito et al., 2005; Redding and Devito, 2011), although they can be connected through groundwater fluxes (Elmes and Price, 2019). Nonetheless, constructed upland features are frequently included in large-scale watershed reconstruction projects following open-pit oil sands mining, in part because they can be specifically engineered to promote groundwater recharge and surface runoff (Ketcheson et al., 2017; Zabel et al., 2024). To date, however, it remains unknown what role small-scale uplands may play in supporting water availability on residual well pads, particularly given that they lack engineered confining layers (Drozdowski et al., 2020).

Given the low degree of horizontal hydrological connectivity within residual mineral well pads and the high degree of sensitivity of establishing peatland mosses to substrate water limitations, there is a need to modify the Partial Removal Technique to maximize substrate water availability. As such, the objectives of this study were to compare the relative magnitudes of hydrologic input and loss terms on a residual well pad, and to use those comparisons to inform potential restoration treatments or modifications to the Partial Removal Technique that could be used to maximize the water table elevation and substrate water storage. Specific consideration was given to direct rainfall and snowmelt inputs, surface runoff from a constructed

upland feature, horizontal and vertical groundwater fluxes, and evapotranspiration. Overland flow from the adjacent fen was not considered in detail due to difficulties in measuring it at a remote site with limited topographic relief.

5.2. Methods

5.2.1. Study Site

The study was undertaken in the Lesser Slave Lake region of northern Alberta, Canada (55° 19' 11" N, 114° 28' 22" W). Based on 30-year climate normal data, the region typically receives 422 mm of precipitation annually, 338 mm of which is typically received as rainfall (Environment and Climate Change Canada, 2024b). Of this, approximately 254 mm of rainfall is typically received during the growing season (May 1 to August 31). Snowmelt in the region typically occurs during a short period of time in the months of March and April (Biagi and Carey, 2020; Ketcheson and Price, 2016) when mean monthly air temperatures begin to rise above 0 °C (Environment and Climate Change Canada, 2024b).

The study examined a residual well pad that was originally constructed in 1991 in a moderate-rich fen (pH $\approx$ 6.5; electrical conductivity $\approx$ 110 $\mu\text{S cm}^{-1}$) that was characterized by a canopy of *Picea mariana* and a shrub layer of *Salix* spp. and *Rhododendron groenlandicum*. The herb layer in the fen was predominantly *Carex aquatilis* and *Rubus chamaemorus*, while *Drepanocladus aduncus*, *Aulacomnium palustre*, *Sphagnum warnstorffii*, and *Sphagnum angustifolium* were characteristic of the ground layer. Partial removal of the well pad was undertaken using heavy machinery during March 2020 when the ground was frozen. The residual mineral substrate had a loamy sand texture and a low organic matter content (2.1 %) and was roughened to create surface microtopographic variability (McKinnon et al., 2024).

Donor mosses collected from a fen within 2 km of the study site were then introduced onto the mineral substrate at an aerial ratio of 1:7 (donor site to pad) using the MLTT. A layer of straw mulch was

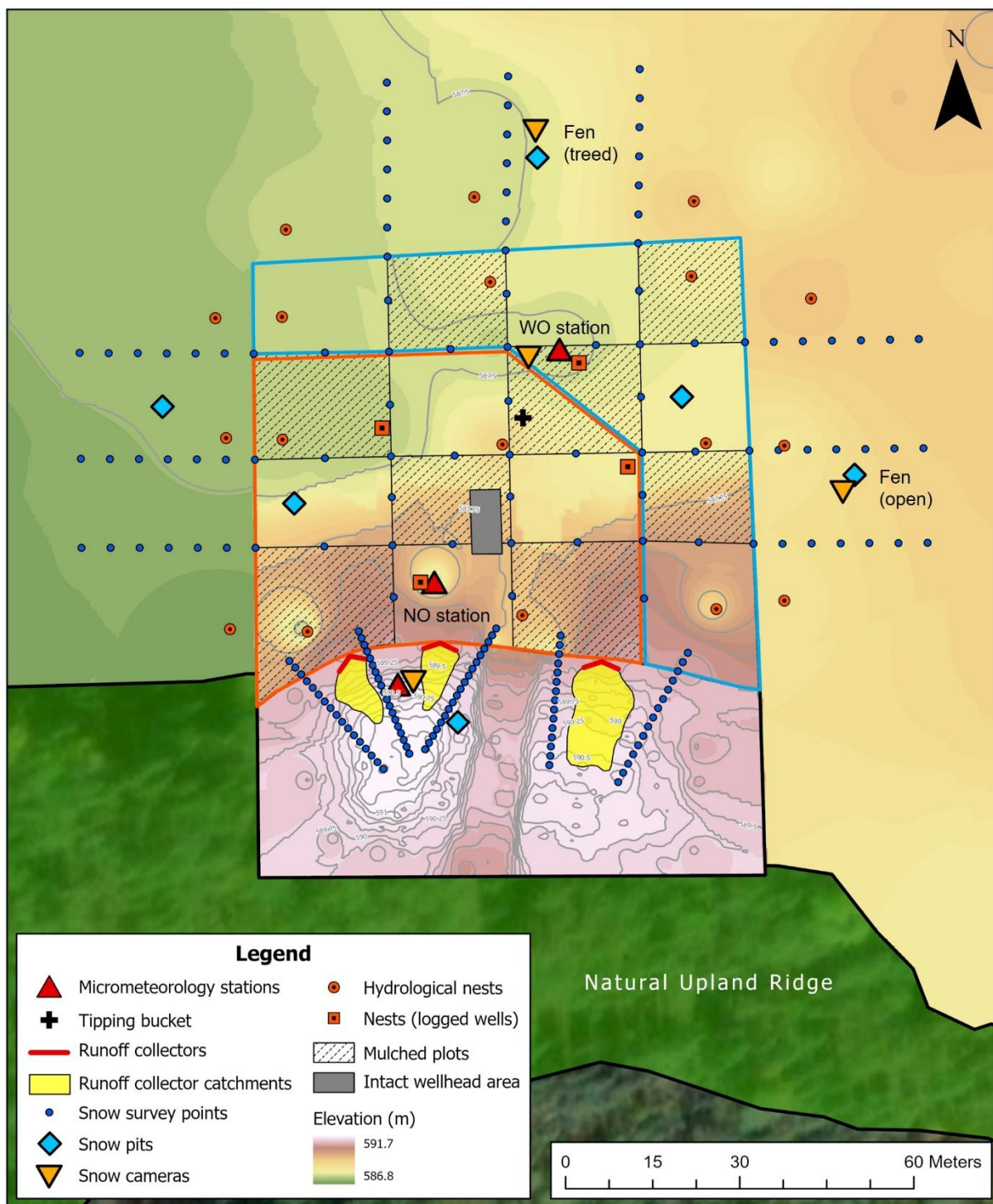


Figure 5.1: Map of the study site including the surface elevations of the residual well pad, constructed upland feature, and surrounding peatland. Hydrological nests included wells and piezometers. All logged hydrological nests included water table loggers. Hydrologically well-optimized (WO) areas are outlined in blue. Hydrologically non-optimized (NO) areas are highlighted in orange. Hydrological classifications (WO and NO) are based on the results of McKinnon et al. (2024). No surface elevation measurements were available for the forested natural upland ridge.

then applied by hand over top of the introduced mosses across half of the pad (Figure 5.1). The excess mineral fill removed during partial removal was used to construct an upland feature on the southern portion of the site, which was located adjacent to a natural upland forest (Figure 5.1). To reduce erosion of the exposed sandy substrate, the slopes of the constructed upland were contoured to create a series of narrow swales oriented perpendicular to the slope (Supplementary Figure D.1). These were designed to mitigate the risk that mosses introduced onto the pad would be buried under eroded material. The swales were separated by narrow ‘berms’ that also ran perpendicular to the slope. Based on the findings of McKinnon et al. (2025, 2024), a portion of the pad was classified as being hydrologically well-optimized (WO) for peatland vegetation establishment, while a portion was classified as being hydrologically non-optimized (NO) for vegetation (Figure 5.1). In the WO area, the WT was reported to remain stable and near the surface throughout multiple growing seasons, whereas the WT in the NO area frequently became poorly regulated. Thus, occasional exceedances of moss desiccation thresholds were observed in NO areas, but not WO areas (McKinnon et al., 2025). The residual pad was covered with a dense sedge community primarily comprised of *Carex aquatilis* and *Scirpus* spp. by the time of this study, which shaded a discontinuous layer of mosses growing directly on the residual mineral substrate (Supplementary Fig. D.2).

5.2.2. Rainfall

Total daily rainfall (P) was measured during the 2022 and 2023 growing seasons using a tipping bucket (ECRN-100, Decagon, Pullman, USA) installed at a height of 0.5 m above the ground near the centre of the pad (Figure 5.1). Due to datalogger failure, rainfall data was gap-filled for six days in 2022 and 45 days in 2023. Gap-filling was completed using a linear regression model (Allen et al., 1998) developed between available site-measured data and data reported by Environment and Climate Change Canada for the nearby Slave Lake Airport ($R^2 = 0.39$ in 2022; $R^2 = 0.31$ in 2023).

5.2.3. *Snow and Ground Ice*

Snow depth (cm) and density (g cm^{-3}) were measured along transects extending through the fen and across the pad (Figure 5.1). Manual surveys were undertaken in 2023 (March 10 and April 6), and 2024 (March 9 and April 6). In the fen, depth was measured every 5 m, while density was measured every 10 m. On the pad, depth and density were measured every 10 and 20 m, respectively. Snow depth and density measurements were also made along five transects extending from the toeslope of the constructed upland to its crest (Figure 5.1). Alternating measurements along slope transects were taken between swales and the high berms separating them. Snow depth was measured using a graduated avalanche probe, while snow density was measured using a snow tube (ESC30, GeoScientific, Vancouver, Canada) that was weighed with a hanging scale (Adams and Barr, 1974). Snow depth and density were used to calculate the snow water equivalent (SWE; cm) of the snowpack (Dingman, 2015b).

In addition to manual surveys, daily snow depth was automatically recorded at one location on the constructed upland, one location on the pad, and two locations in the fen (Figure 5.1). Snow depth was manually transcribed from daily timelapse images taken of snow rulers (Kinar and Pomeroy, 2015) using a combination of research-grade (PC900, Reconyx, Holmen, USA) and recreational-grade (Outfitter Gen 4, Cabela's, Sidney, USA) wildlife cameras. Due to later installation and periodic failure of the recreational-grade cameras, daily snow depth was not available at all locations on all dates. Snow rulers were installed on lawn microtopographic features in the fen to avoid biases associated with installation in hollows (i.e., biased towards high snow depth) or on hummocks (i.e., biased towards low snow depth). Snow pits were completed in March each year (Figure 5.1) to characterize snowpack physical and thermal properties and to validate snow tube density measurements (Adams and Barr, 1974). Once the snowpack was sufficiently thin, measurements of depth to ground ice were made along each transect using a steel rod (Woo and Xia, 1996) on an approximately monthly basis until no ground ice was encountered.

5.2.4. Surface Runoff

Surface runoff from the constructed upland was quantified in the spring of 2023 and 2024 and during the 2022 and 2023 growing seasons using runoff collectors modified from Biagi & Carey (2020). Collectors were constructed out of metallic angle flashing installed in trenches at a depth of 5 cm. The angle flashing prevented infiltration into the bottom and downslope walls of the trenches (Supplementary Figure D.3). Trenches extended slightly upslope for 1.8 m from either side of a central point in a “V” shape (Figure 5.1). The collectors were cemented to a concrete block at the central point to reduce frost heaving and drained into a bucket with a v-notch weir. Pressure transducers (HOBO U20L, Onset, Bourne, USA) were installed in each bucket, and were placed in latex sheaths filled with ethylene glycol to prevent freezing of the transducers.

The water in each bucket was continuously aerated with a small pump (Whisper Aquarium, Tetra, Blacksburg, USA) powered by a 100-Watt solar panel and a 300-Watt inverter in an effort to minimize freezing. The contributing catchment areas of each flume (Figure 5.1) were estimated based on a raster layer of surface elevation created using ArcGIS Pro (Hydrology Toolset; v3.2.0). The raster was created using inverse distance weighted interpolation applied to surface elevations measured using a real-time kinematic GPS (RTK) with vertical accuracy of 0.005 m (S321+, Hemisphere GNSS, Scottsdale, USA). Spring runoff coefficients (C_{runoff}) were calculated for each flume catchment during the snowmelt period each year as:

$$C_{runoff} = \frac{\text{Total runoff volume}}{\text{Catchment SWE}} \quad (5.1)$$

where runoff volumes were taken as the total flume discharge (m^3) recorded during the entire spring freshet. Values of catchment SWE were the mean of manual SWE measurements taken along the two transects adjacent to a given catchment (Figure 5.1) at the approximate time of maximum snowpack depth. These were multiplied by the catchment area (m^2). To support the assessment of runoff during the growing season,

ten 250 cm³ cores of mineral fill from the constructed upland were run on a falling head permeameter (KSAT, Meter Group, Pullman, USA) to determine their vertical hydraulic conductivity (K_v).

5.2.5. Groundwater Fluxes

Manual measurements of depth to water table (DWT) were recorded in fully slotted wells extending to a depth of 1 m on the pad ($n = 14$) and in the fen ($n = 9$) during approximately monthly field visits in the 2022 and 2023 growing seasons (Figure 5.1). The DWT was also measured half-hourly in a subset of wells within the residual pad (Figure 5.1) using pressure transducers (HOBO U20L, Onset, Bourne, USA). Depth to water table was converted to absolute water table (WT) elevation (in metres above sea level) using the elevation of well tops measured using the RTK in August 2022. Horizontal groundwater fluxes across the edges of the pad footprint (i.e., into or out of both the residual mineral fill and underlying peat) were calculated using depth-weighted horizontal hydraulic conductivity (K_h ; m s⁻¹) values determined for the soil profile as part of a related study (see McKinnon et al. (2024) for detailed methodology). Briefly, manual measurements of absolute WT elevation were used to calculate horizontal specific discharge (q_h ; m s⁻¹) across pad edges according to Darcy's Law:

$$q_h = -K_h \frac{dh}{dl} \quad (5.2)$$

where dh/dl is the mean horizontal hydraulic gradient across a given pad edge (Freeze and Cherry, 1979).

Horizontal volumetric flow rates (Q_h ; m³ s⁻¹) across each pad edge were subsequently calculated as:

$$Q_h = q_h \cdot A \quad (5.3)$$

where A represents the cross-sectional area of the flow face along a given pad edge (Freeze and Cherry, 1979). Values of A (m²) were defined by the length of pad edges and depth of the peat profile. As Q_h data were only available on a small number of dates, the mean of Q_h values calculated for all dates with manual WT measurements in a given study period was multiplied by the number of days in that study period to

estimate total seasonal Q_h . Values of Q_h were subsequently normalized over the surface area of the residual pad ($\sim 6058 \text{ m}^2$) to permit direct comparison to other hydrologic fluxes.

Vertical hydraulic gradients were calculated using manual hydraulic head measurements made in nested piezometers at all monitoring points on several dates each season (Figure 5.1). Piezometers had 20 cm intake screens centred at 25, 50, or 100 cm below the ground surface. Within the pad footprint, 25 cm screens were centred within the mineral fill, while the 50 and 100 cm screens were centred in the compacted peat beneath the fill. At a subset of nine monitoring points, additional piezometers were installed with intake screens centred at 150 cm below the surface (in the basal mineral substrate underlying the peat profile). Vertical gradients were calculated between the 25 and 100 cm depths at all nests, while additional vertical gradients were calculated between the 50 and 150 cm depths at nests with 150 cm piezometers.

Net changes in groundwater storage (ΔS) within the footprint of the residual pad over the duration of each study period were estimated as:

$$\Delta S = S_y \cdot \frac{\Delta h}{\Delta t} \quad (5.4)$$

where S_y is the specific yield of the substrate and $\Delta h/\Delta t$ is the difference in mean daily water table between the first and last dates of a given study period (Freeze and Cherry, 1979). The mean S_y (0.23) of the mineral fill (0.22) and underlying peat (0.24) (McKinnon et al., 2024) was used in all calculations because the WT fluctuated between those substrates. As the total number of dates for which vertical hydraulic gradient measurements were available was limited, vertical groundwater fluxes (Q_v) were estimated as the residual of other terms in the pad water balance for each study period by rearranging Equation 5.5:

$$\Delta S = P + Q_h + Q_v - ET_{cont} \quad (5.5)$$

where Q_h and Q_v represent net hydrologic fluxes over the full duration of each study period and P and ET_{cont} represent cumulative rainfall and evapotranspiration, respectively, during each study period (modified from Scanlon et al., 2002). Overland flow was excluded from the water balance as no direct measurements of it

were available and only limited surface flow was observed during site visits. Error ranges for each term were estimated based on the error ranges proposed by Healy et al. (2007) for similar water balance analyses. Because the magnitudes of Q_v fluxes estimated using the water balance residual approach were large compared to the magnitudes of other hydrologic fluxes (potentially resulting in a large degree of error; see Winter, 1981), comparisons were also drawn against Q_v calculated using Darcy's Law for each season. To ensure that the comparative Q_v calculations were representative of the full range of conditions possible on the site, they were performed using the mean vertical hydraulic gradient (25 – 100 cm b.g.s.) observed across the pad footprint on all manual measurement dates across 2022 and 2023. As such, these values can be thought of as representing 'average' conditions, but do not capture daily-scale variability in Q_v . The vertical hydraulic conductivity (K_v) of the mineral fill ($4.77 \times 10^{-6} \text{ m s}^{-1}$) was used for the comparative calculations as it was lower than the K_v of the underlying peat ($2.08 \times 10^{-5} \text{ m s}^{-1}$), and thus would have constrained the maximum rate of flow (Freeze & Cherry, 1979).

5.2.6. Evapotranspiration (physical measurements)

Rates of actual evaporation (E_a ; mm day^{-1}) from open water surfaces on the pad were manually measured daily using pan lysimeters ($n = 4$) during field visits (approximately one week per month) in the 2022 and 2023 growing seasons. The pan lysimeters were constructed out of transparent acrylic to limit heating effects on E_a (Supplementary Figure D.4). The positioning of the pans was regularly adjusted during measurement periods to ensure that their tops were closely aligned with the surface of the surrounding water (but never overtopped). Rates of actual evapotranspiration (ET_a ; mm day^{-1}) from soil and vegetated surfaces were manually measured using weighing lysimeters installed in each treatment plot on the pad ($n = 16$; Figure 5.1). Lysimeters were modified from the design of Scarlett et al. (2017), and were constructed out of nested white plastic buckets (7.5 L, 23 cm diameter). An intact soil monolith collected from the pad was placed into each upper bucket. Care was taken to ensure the cover of straw mulch and type and density of vegetation growing in lysimeters was representative of the surrounding area (Supplementary Figure D.4).

The bottoms of the upper buckets were perforated to permit drainage, and the upper and lower buckets were weighed separately using a hanging scale (KHS-200, Kilotech, Lachine, Canada). The surfaces of lysimeters were closely aligned with the ground surface, and water was periodically added to ensure their VWC was within 5 % of the VWC adjacent to each lysimeter.

Total evapotranspiration from both open water and non-inundated surfaces (ET_{total} ; mm day⁻¹) was calculated for each treatment plot on the pad for dates with manual measurements as:

$$ET_{total} = (P_{ponding} \cdot E_a) + ((1 - P_{ponding}) \cdot ET_a) \quad (5.6)$$

where $P_{ponding}$ is the proportion of surface water coverage within a given plot. Values of $P_{ponding}$ were estimated visually for each plot on a monthly basis during the growing seasons of 2022 and 2023. Values of ET_{total} were then weighted based on the surface area of each plot to yield site-wide ET_{total} values for each date with manual measurements. To support interpretation of spatial differences in ET_{total} , manual measurements of volumetric water content (VWC) were made over the 0-6 cm depth at average microtopographic positions at three locations per plot each month using a calibrated ML3 probe (Delta-T Devices, Burwell, UK).

5.2.7. Evapotranspiration (energy balance)

Components of the pad energy balance were measured on a continuous basis at two micrometeorological stations in 2022 and 2023 (one in a WO area, and one in a NO area; Figure 5.1). Both stations were located in areas with minimal mulch cover (at the WO station, mulch had largely been washed away in previous growing seasons) and therefore were representative of non-mulched areas (Figure 5.1). Both were also located in areas that were not permanently flooded, although the WO station was located on a high point surrounded by a pond that persisted for much of the growing season (Figure 5.1). Net radiation (Q^* ; W m⁻²) was measured at a height of 1 m (NRLite, Kipp and Zonen, Delft, Netherlands). Substrate VWC was measured at depths of 5, 12.5, and 22.5 cm (m³ m⁻³; HydraProbe, Stevens, Portland, USA), while substrate

temperature was measured at depths of 2.5, 5, 7.5, 12.5, 17.5, 22.5, and 27.5 cm using thermocouples constructed out of type-T thermocouple wire (Omega Engineering, Norwalk, USA). Values of Q^* , VWC, and substrate temperature were recorded at 60-second intervals using CR1000 dataloggers (Campbell Scientific, Logan, USA) and averaged over 30-minute intervals. Ground heat fluxes (Q_g ; $W m^{-2}$) were calculated using the calorimetric method (Halliwell and Rouse, 1987; Sauer and Horton, 2005). Substrate bulk heat capacities were calculated using the heat capacities of individual soil components obtained from Oke (1987) and depth-profile VWC measurements. Relative humidity (%) and air temperature ($^{\circ}C$) were measured at a height of 1 m using an internally-logging sensor housed in a radiation shield at each micrometeorological station (HOBO U23 Pro v2, Onset, Bourne, USA).

Measurements of energy balance components were used to calculate daily equilibrium potential evapotranspiration (PET_{eq} ; $mm day^{-1}$) on a continuous basis each growing season. This was done using the Priestley-Taylor equation (Priestley and Taylor, 1972), which is a radiation-based method that calculates PET_{eq} as:

$$PET_{eq} = \frac{\Delta \cdot (Q^* - Q_g)}{\rho_w \cdot \lambda_v \cdot (\Delta + \gamma)} \quad (5.7)$$

where Δ is slope of the saturation vapour pressure-temperature curve ($kPa ^{\circ}C^{-1}$), Q^* is the net radiation flux, Q_g is the ground heat flux, ρ_w is the density of water ($kg m^{-3}$), λ_v is the latent heat of vapourization ($MJ kg^{-1}$), and γ is the psychrometric constant ($0.00662 kPa ^{\circ}C^{-1}$ at $20 ^{\circ}C$) (S. L. Dingman, 2015). Because equilibrium conditions (i.e., uniform saturated surfaces) are uncommon, ET_a and ET_{eq} were related to one another by calculating Priestley-Taylor coefficients (α) for each study period using:

$$\alpha = \frac{ET_a}{PET_{eq}} \quad (5.8)$$

where ET_a and PET_{eq} are measured over the same period of time under the same conditions (Priestley and Taylor, 1972). Values of α were thus taken as the slope of the regression between daily values of ET_a and

ET_{eq} obtained during site visits. All values of PET_{eq} in a given study period were multiplied by the relevant α value to yield continuous daily estimates of actual evapotranspiration (ET_{cont} ; mm day⁻¹).

5.2.8. Data Analysis

The effects of location or time of measurement on environmental variables were assessed using linear mixed effects (LME) models. Relationships between variables were assessed for significance using linear regression (LR) models. Comparisons between group means were drawn using analysis of variance (ANOVA) with Tukey's HSD post hoc tests. The assumption of normality of residuals was assessed for LME, LR, and ANOVA models through visual inspection of Q-Q plots (Zuur et al., 2009). All statistical analyses were conducted with α set to 0.05 within R (v4.4.2; R Core Team, 2024).

5.3. Results

5.3.1. Weather

The site received 227 and 354 mm of rainfall during the 2022 and 2023 growing seasons, respectively, from May 1 to August 31 (data from the tipping bucket with gap-filling). In 2022, this was 11 % less than the 30-year climatic normal for the region over the same months (254 mm), while the site received 39 % more rainfall than the long-term average in 2023 (Environment and Climate Change Canada, 2024b). Mean air temperatures during the 2022 (14.4 °C) and 2023 (15.6 °C) growing seasons were both higher than the 30-year growing season climate normal (13.8 °C). As such, the 2022 growing season can be described as having been drier and warmer than average, while the 2023 growing season was wetter and warmer than average. Average air temperatures in the months of March and April (the typical snowmelt period) were lower than the 30-year climate normal (-1.2 °C) in 2023 (-1.9 °C), but slightly higher than the long-term average in

2022 (-1 °C) and 2024 (-0.6 °C). Thus, the 2023 and 2024 snowmelt periods, examined in detail herein, can be described as having been slightly colder and slightly warmer than average, respectively.

5.3.2. Snowpack

Patterns of snow accumulation differed considerably between the winters of 2022-23 and 2023-24 (Figure 5.2a-b). Specifically, colder air temperatures from December 2022 to March 2023 (mean = -11.3 °C) permitted a greater accumulation of snow across the site than during the winter of 2023-24 (Figure 5.2a-b), which was characterized by warmer air temperatures (mean = -8.4 °C) and several mid-winter melt events (Figure 5.2b). A mid-winter melt event did occur during the winter of 2022-23 as well, although it did not

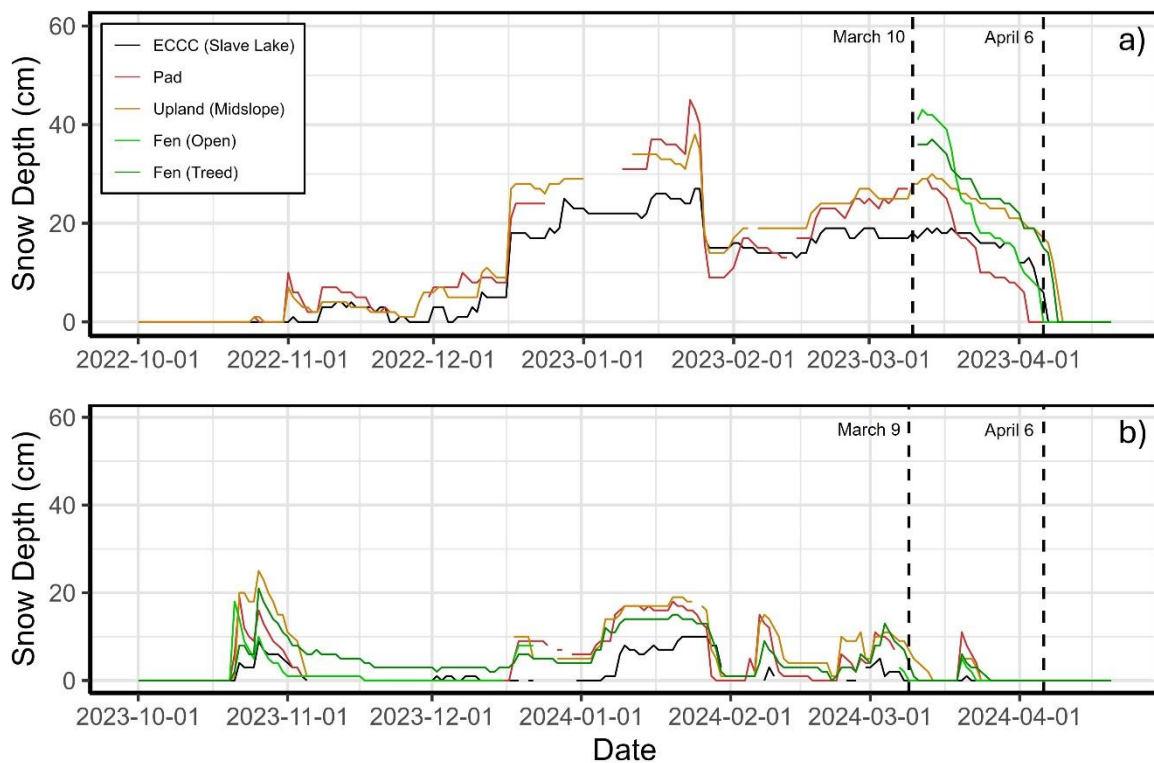


Figure 5.2: Daily snow depth acquired from Environment and Climate Change Canada (Slave Lake station, black line) and snow depths transcribed from timelapse cameras installed on the research site in the winters of 2022-23 (a) and 2023-24 (b). Cameras were added to the fen locations on March 10, 2023. Due to camera failure, snow depth was not available at all locations on all dates. Vertical dashed lines denote the dates of manual snow surveys. See Figure 1 for camera locations.

result in complete loss of the snowpack like in the winter of 2023-24 because of the greater snowpack depths that year (Figure 5.2a). Combined, the depth of the snowpack at camera locations at peak SWE in 2023 (29 cm on the pad on March 13th) was considerably greater than that in 2024 (11 cm on the pad on March 2nd) (Figs. 5.2a-b).

Similarly, the average depth of the snowpack on the manual survey date closest to maximum SWE was significantly lower on the pad (35.7 cm in 2023 and 7.8 cm in 2024) than in the fen (49.9 cm in 2023 and 14.5 cm in 2024) in both years (Figure 5.3a-c). Despite this, average snowpack density was comparable between the pad and fen on all measurement dates (0.18 g cm^{-3} in 2023 and 0.23 g cm^{-3} in 2024) (Figure

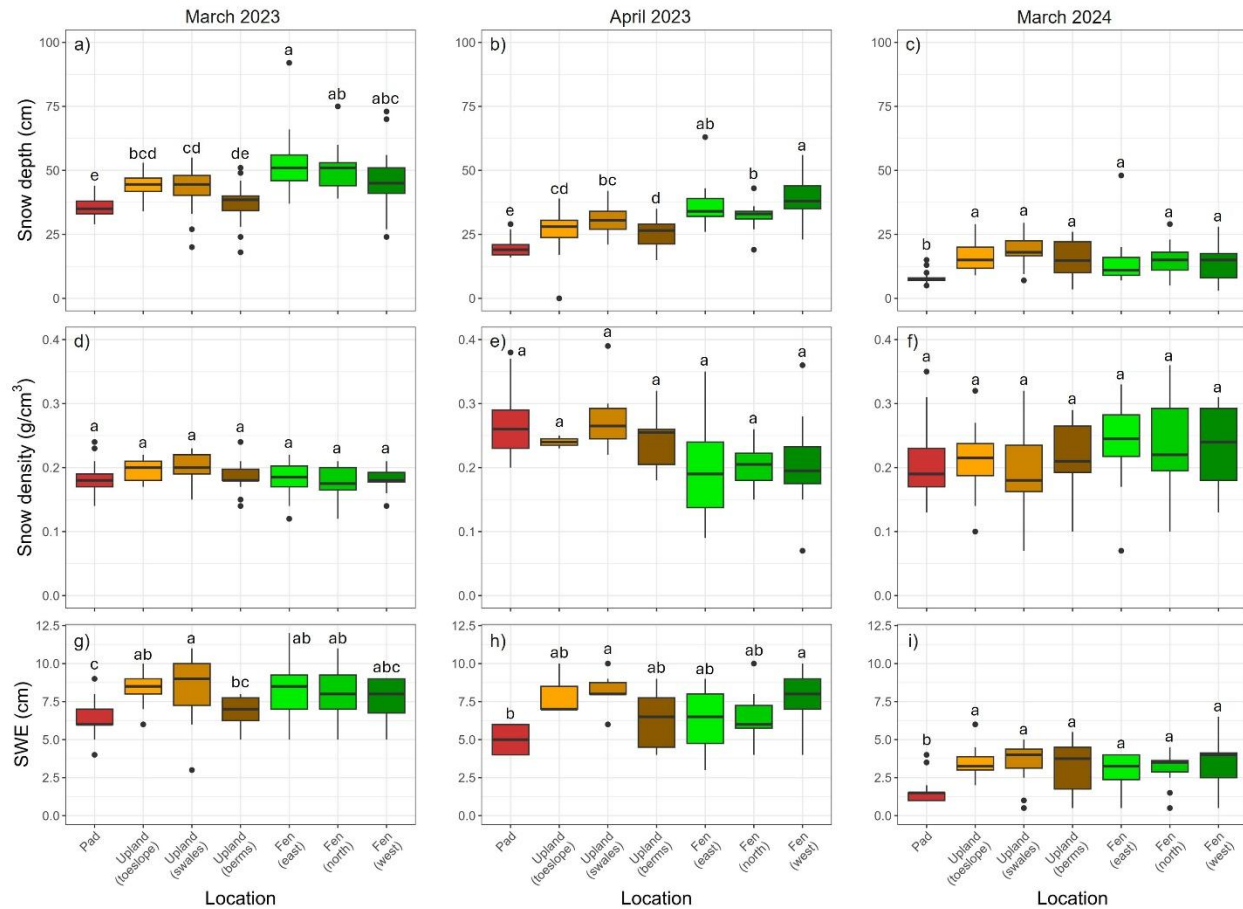


Figure 5.3: Comparison of manually measured snow depth (a-c), snow density (d-f), and snow water equivalent (SWE; g-i) on the residual pad, constructed upland (toe slope, swales, and berms), and in the fen (east, west, and north of the pad). No snowpack was present by the April 6, 2024 measurement date. Groups within a plot not sharing any common letters significantly differ based on analysis of variance with Tukey HSD post hoc tests.

5.3d-f). Similarities in density were also observed in the snow pit dataset. Nonetheless, the lower depth of snow within the pad footprint meant that the pad snowpack often had a significantly lower SWE (6.5 cm in 2023 and 1.6 cm in 2024) than the fen snowpack (8.1 cm in 2023 and 3.2 cm in 2024) (Figure 5.3g-i). Snow depth, density, and SWE rarely differed significantly between the fen areas to the east, north, and west of the pad (Figure 5.3), despite differing degrees of tree cover in those areas (Supplementary Figure D.1). Also, the average snowpack depth on the constructed upland was frequently significantly greater (41.6 cm in 2023 and 16.9 cm in 2024) than on the pad (Figure 5.3a-c). Somewhat unexpectedly, differences in snow depth between the swale, berm, and toe slope microsites on the upland tended to be limited, as were differences in density (mean = 0.19 g cm^{-3} in 2023 and 0.21 g cm^{-3} in 2024) and SWE (mean = 7.9 cm in 2023 and 3.4 cm in 2024) between those microsites in a given year (Figure 5.3). When significant differences were observed, the swale microsites had slightly greater snow depth or SWE than the berms separating them (Figure 5.3).

Owing to the colder air temperatures observed during the snowmelt period in 2023, the onset of snowmelt was later that year (approximately March 13th) than in 2024 (approximately March 2nd) (Figure 5.2). Due to the higher air temperatures in 2024, the average rate of snow ablation within the footprint of the pad that year (1.8 cm day^{-1}) was somewhat greater than that observed in 2023 (1.3 cm day^{-1}). Ablation rates on the pad were slightly higher than those on the constructed ridge (1.1 cm day^{-1} in 2023 and 2024) in both years but were comparable to those in the fen (1.4 cm day^{-1} in 2023 and 1.6 cm day^{-1} in 2024) in a given year. Notably, snowmelt normally began approximately 1-2 days later in the fen than within the footprint of the pad (Figure 5.2). The combination of an earlier start to melt, greater ablation rates, and a lower snowpack depth in 2024 meant that the residual pad was entirely snow-free by March 11th that year, although a large spring snowfall event on March 20th, 2024 resulted in a second, rapid (2.2 cm day^{-1}) melt event (Figure 5.2b). No manual measurements of depth, density, or SWE were collected during this event. Conversely, the snowpack persisted within the pad footprint until April 3rd during the colder 2023 snowmelt period (almost one month longer; Figure 5.2). Owing to the deeper snowpack in the fen, complete melt was

not observed there until April 9th in 2023 (Figure 5.2a). In both years, swales on the constructed upland became snow-free a few days later than the pad, with complete melt of the snowpack there occurring by April 10th, 2023 and March 13th, 2024 (Figure 5.2).

5.3.3. Ground Ice Dynamics

The depth of the frost table within the footprint of the residual pad was either shallower than, or similar to the depth of seasonal ground ice in the fen early during the 2023 melt period (Figure 5.4a,e). Estimated rates of ground thaw were slightly higher on the pad (0.3 cm day^{-1}) than in the fen (0.2 cm day^{-1}) between April 6th and May 16th that year (Figure 5.4a-b). Consequently, the average frost table depth was slightly

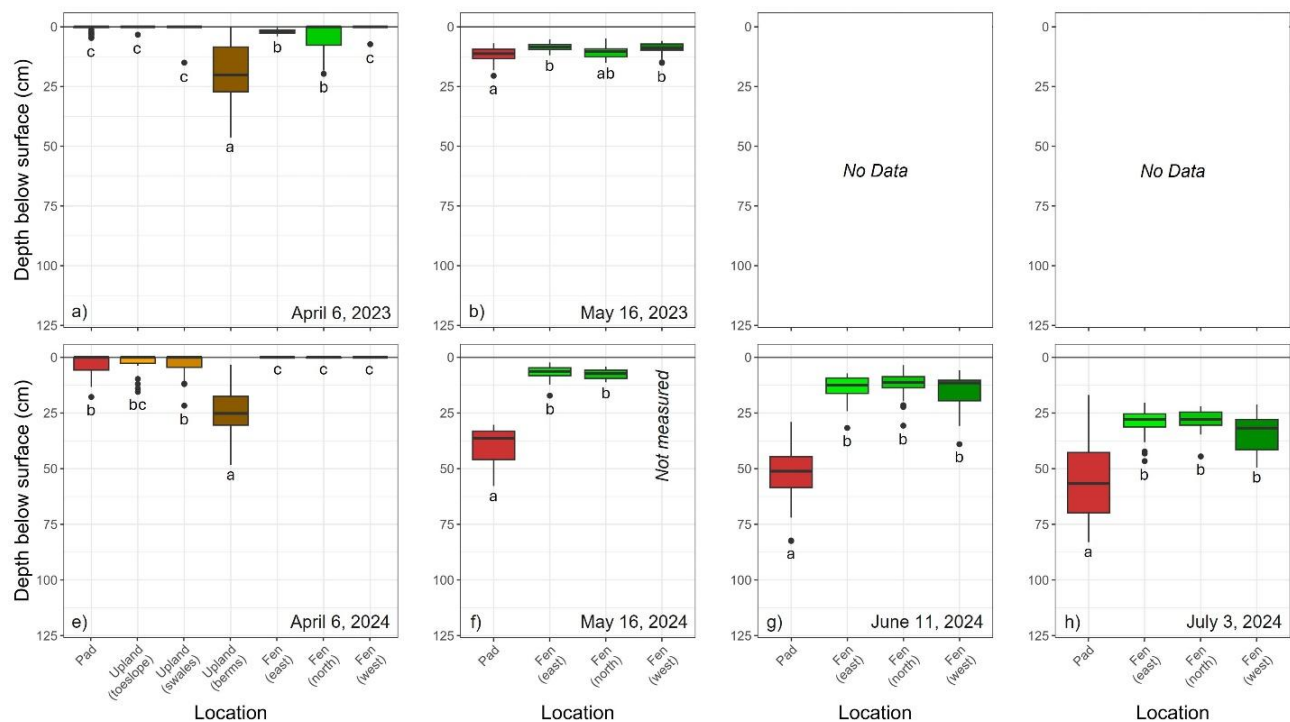


Figure 5.4: Comparison of the depth to the frost table (pad and upland) and seasonal ground ice (fen) on the residual pad, the constructed upland (toeslope, swales, and berms), and the adjacent fen (east, west, and north of the pad). No frost or seasonal ground ice was observed by June 27, 2023 or July 13, 2024. Measurements were only made on the constructed upland feature in April of each year. Groups within a plot not sharing any common letters significantly differ based on analysis of variance with Tukey HSD post hoc tests.

deeper on the pad (11.7 cm b.g.s.) than in the surrounding fen (9.4 cm b.g.s.) by May 16th, when continuous ground ice was still observed across the entire site (Figure 5.4b). Although no measurements were available in late May or early June, no ground ice was observed by June 27th. This indicates that complete melt had occurred prior to the peak growing season in 2023. In contrast, discontinuous ground ice was observed until the July 3rd measurement date in 2024 (the year with considerably thinner winter snowpack depths).

The average rate of ground thaw in the residual pad was 0.6 cm day⁻¹ between April 6th, 2024 and July 3rd, 2024 (Figure 5.4e-h). The average rates of thaw in the fen on the east (0.3 cm day⁻¹), north (0.3 cm day⁻¹), and west (0.4 cm day⁻¹) sides of the pad were somewhat lower over the same period (Figure 5.4e-h). As a result, the frost table was significantly deeper within the pad footprint than in the fen on all measurement dates, reaching an average of 55 cm below the pad surface before no ground ice was observed (Figure 5.4e-h). Comparatively, the maximum average depth of the seasonal ground ice observed in the fen was only 34 cm below the surface (Figure 5.4h).

Interestingly, the high ‘berms’ separating the swales on the constructed upland had thawed to a greater depth than the swales themselves or the residual pad on the first measurement dates in April of both years (Figure 5.4a,e). This may have been a result of the berms becoming snow-free earlier than either the swales or the pad. Although this was not captured by the manual snow surveys, the berms were observed to be snow-free earlier than all other locations in timelapse images taken on the midslope (Supplementary Figure D.5), which would have permitted ground thaw to begin on an earlier date. No measurements of frost table depth on the constructed upland were made on subsequent dates in either year due to time constraints.

5.3.4. Surface Runoff from the Constructed Upland

Despite continuous aeration of collected runoff, all of the flume collection buckets froze for portions of the spring freshet in both years. However, none of the flume collection buckets had reached capacity between

field visits (i.e., they had not filled to the bottom of the v-notch weir). Thus, runoff volumes were estimated by removing the ice and melting it in a separate bucket. Notably, values of C_{runoff} calculated for the constructed upland were $< 1\%$ during the spring freshet in both 2023 and 2024 (Supplementary Tables D.1 & D.2). Scaling runoff volumes measured using the flumes to the full area of the north-facing slopes of the constructed upland (approximately 977 m^2), only 0.11 and 0.14 m^3 are estimated to have reached the pad during the spring freshet in 2023 and 2024, respectively. Owing to the fact that the K_v of the mineral fill comprising the constructed upland (115.5 mm hr^{-1}) was considerably higher than the maximum rainfall rate observed (9.4 mm hr^{-1}), no surface runoff was observed at any time during either growing season.

5.3.6. Evapotranspiration

During the growing seasons of 2022 and 2023, neither plot location (LME; $p = 0.271$ in 2022; $p = 0.903$ in 2023) nor mulching (LME; $p = 0.267$ in 2022; $p = 0.177$ in 2023) had significant effects on lysimeter-measured ET_a across the pad. This was also reflected in spatial comparisons of ET_a in both years. Specifically, ET_a only differed significantly between WO and NO plots in May 2022, when it was somewhat higher in WO plots (mean = 1.6 mm day^{-1}) than in NO plots (mean = 1.0 mm day^{-1}) (Figure 5.5a). Owing to the higher position of the WT in WO plots (Figure 5.6b), VWC was slightly (but not significantly) higher in WO plots on that date (Figure 5.6a). Similarly, ET_a only differed significantly between mulched and non-mulched plots in July of 2022, when mulched plots unexpectedly lost slightly more water (mean = 1.2 mm day^{-1}) than non-mulched plots (mean = 1.0 mm day^{-1}) (Figure 5.5a). Nonetheless, daily values of energy-balance based ET_{cont} were consistently higher in WO plots than in NO plots, resulting in cumulative ET_{cont} of 55 mm by August 31st of that year at the WO micrometeorology station, as compared to only 118 mm at the NO station (Figure 5.7a). Of note, cumulative ET_{cont} was less than total cumulative rainfall by August 31st that year (188 mm), despite the fact that rates of ET_{cont} in both WO and NO plots exceeded the rate of rainfall in the early season (Figure 5.7a). In comparison, rates of rainfall exceeded those of ET_{cont} for nearly all of the wetter 2023 season, during which cumulative ET_{cont} reached 118 mm at the WO station by August

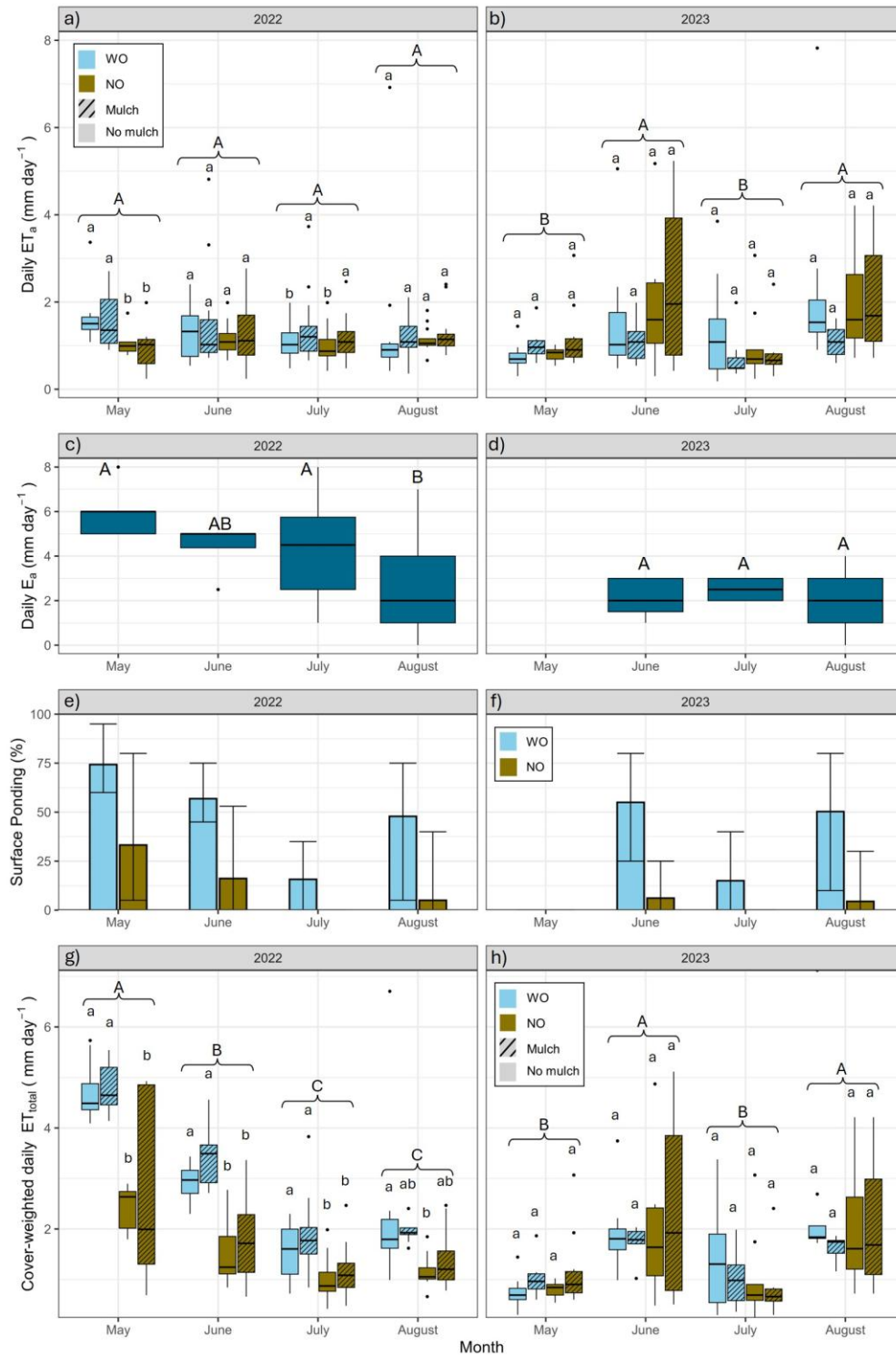


Figure 5.5: Plots of daily rates of ET_a (a-b) and E_a (c-d), mean percentage surface ponding (e-f), and cover-weighted daily volumetric atmospheric water loss ($ET_{total} = ET_a + E_a$) (g-h). Values of ET_a and E_a were recorded every 24 hours during a one-week period each month. No surface ponding was present anywhere on the site during the May 2023 study period. Upper case letters denote the results of statistical comparisons of observations across the entire pad between months. Lower case letters denote the results of statistical comparisons of observations between plot types within a given month. Groups not sharing any common letters significantly differ based on analysis of variance with Tukey HSD post hoc tests.

18th, but only 91 mm at the NO station (Figure 5.7b). Cumulative ET_{cont} was again less than cumulative rainfall that season (309 mm) (Figure 5.7a-b). This occurred despite the fact that values of α were at or above unity (≥ 1) for the duration of the growing season in both years (Supplementary Table D.3).

On all dates when surface ponding was observed, the aerial coverage of ponding was higher in WO plots than in NO plots (Figure 5.5e-f). This meant that E_a made up a greater component of calculated rates of cover-weighted ET_{total} in WO plots than in NO plots in all cases, except in May 2024 when no ponding was observed (Figure 5.5f). Given that average rates of E_a were higher (2.7 to 5.9 mm day⁻¹) than those of ET_a (1.1 to 1.3 mm day⁻¹), this resulted in significantly higher rates of cover-weighted ET_{total} in WO plots (1.7 to 4.7 mm day⁻¹) than in NO plots (1.1 to 2.6 mm day⁻¹) in most cases in 2022 (Figure 5.5g).

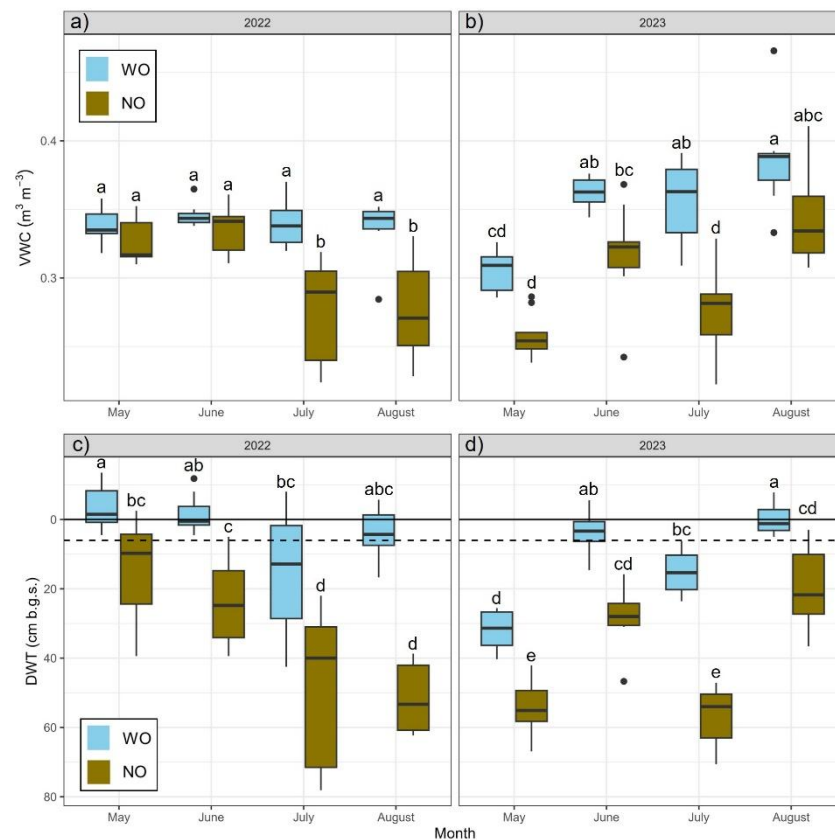


Figure 5.6: Plots of manually measured volumetric water content (VWC; a-b) and depth to water table (DWT; c-d). Groups within a plot not sharing any common letters significantly differ based on analysis of variance with Tukey HSD post hoc tests. The solid and dashed horizontal lines on DWT plots denote the ground surface and the optimal moss-establishment threshold (6 cm b.g.s.) proposed by McKinnon et al. (2024), respectively.

Interestingly, no significant differences in rates of ET_{total} were observed between WO and NO plots on any date in the 2023 season, when average ET_{total} ranged from 1.0 to 2.1 mm day⁻¹ (Figure 5.5h).

Pad-wide average cover-weighted rates of ET_{total} declined significantly between May and July 2022, when the proportion of ponding across the pad also declined (Figure 5.5e,g). Over the same period, VWC generally decreased (Figure 5.6a), although this did not result in significant decreases in pad-wide rates of ET_a (Figure 5.5a). Despite an increase in the proportion of pad-wide ponding between July and August 2022 (Figure 5.5e), no significant differences in ET_a (Figure 5.5a) or ET_{total} (Figure 5.5g) were observed. As such, the significant decrease in E_a between July and August was likely responsible for the maintenance of the similar rates of ET_{total} between July and August that year (Figure 5.5c). Rates of ET_{cont} also tended to be lower from June to August as compared to the early season in 2022 (Figure 5.7a), a pattern that was also observed in 2023 (Figure 5.7b).

In contrast, ET_{total} was significantly higher in June and August than in May and July in 2023 (Figure 5.5h). Pad-wide average rates of ET_a were also higher in June (1.9 mm day⁻¹) and August (2.0 mm day⁻¹) than in May (1.0 mm day⁻¹) and July (0.9 mm day⁻¹) (Figure 5.5b). Of note, VWC was also significantly higher in June and August in both WO and NO plots (Figure 5.6b). Reflecting the patterns between VWC and increased rates of lysimeter-measured ET_a , a significant positive linear relationship was observed between ET_a and VWC ($p = 0.001$). However, a low R^2 value (0.06) indicates that this relationship was weak, and thus VWC may not have been the main control on ET_a . Similarly, significant positive linear relationships were observed at both micrometeorological stations between ET_{cont} and VWC ($p = 0.036$ at WO; $p < 0.001$ at NO) and VPD ($p < 0.001$ at WO; $p = 0.002$ at NO), although R^2 values again tended to be low (Figure 5.8c-f). A significant negative relationship between ET_{cont} and DWT was only observed at the NO station ($p = 0.006$; Figure 5.8a-b), where the WT was more poorly regulated (Figure 5.7a-b).

The energy balance at both the WO and NO micrometeorological stations was dominated by latent heat fluxes in the early season in both 2022 and 2023 (Table 5.1). However, from the month of July onwards in 2022 (the drier and warmer season), the energy balance was dominated by the sensible heat flux at both

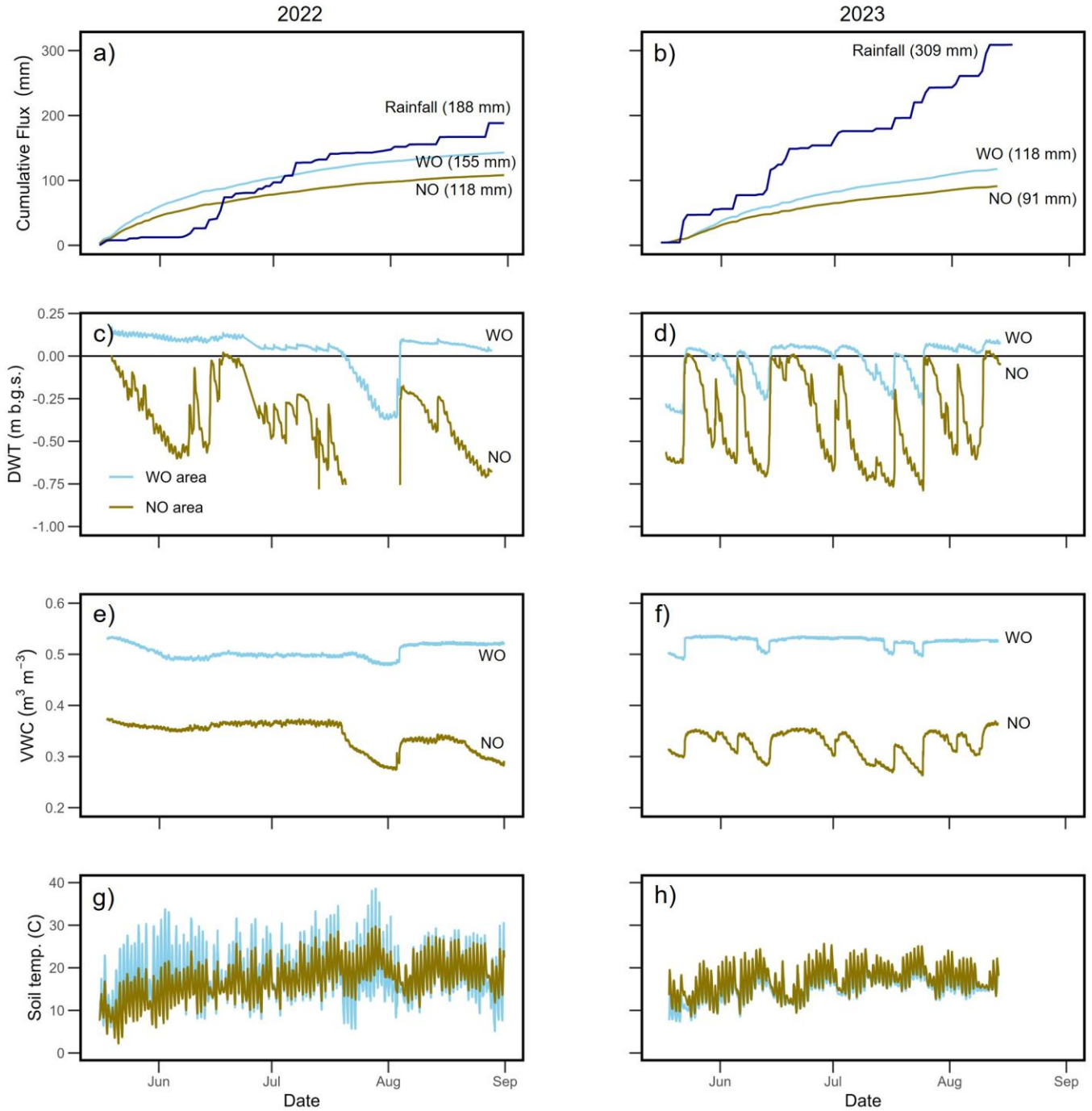


Figure 5.7: Plots of cumulative ET_{cont} from soil surfaces relative to cumulative rainfall (a-b), depth to water table (DWT; c-d), volumetric water content (VWC; e-f), and substrate temperature (g-h). The VWC measurements were recorded at 5 cm below the ground surface. The substrate temperature measurements were recorded at 2.5 cm below the ground surface. All data were collected at the micrometeorological stations located in WO and NO areas of the residual pad in 2022 and 2023. The solid horizontal line on DWT plots denotes the ground surface. The logged well at the WO station was adjacent to the elevated (non-inundated) location that the station itself was located on.

stations (Table 5.1). A similar seasonal shift was not observed in the wetter and warmer-than-average 2023 season, when the energy balance remained dominated by latent heat fluxes until the end of the season at both stations (Table 5.1). With the exception of WO areas in 2023, the ground heat flux comprised a relatively small energy balance component (Table 5.1). Notably, the mean overnight mean ground heat flux frequently remained positive (i.e., directed into the ground) (Table 5.1).

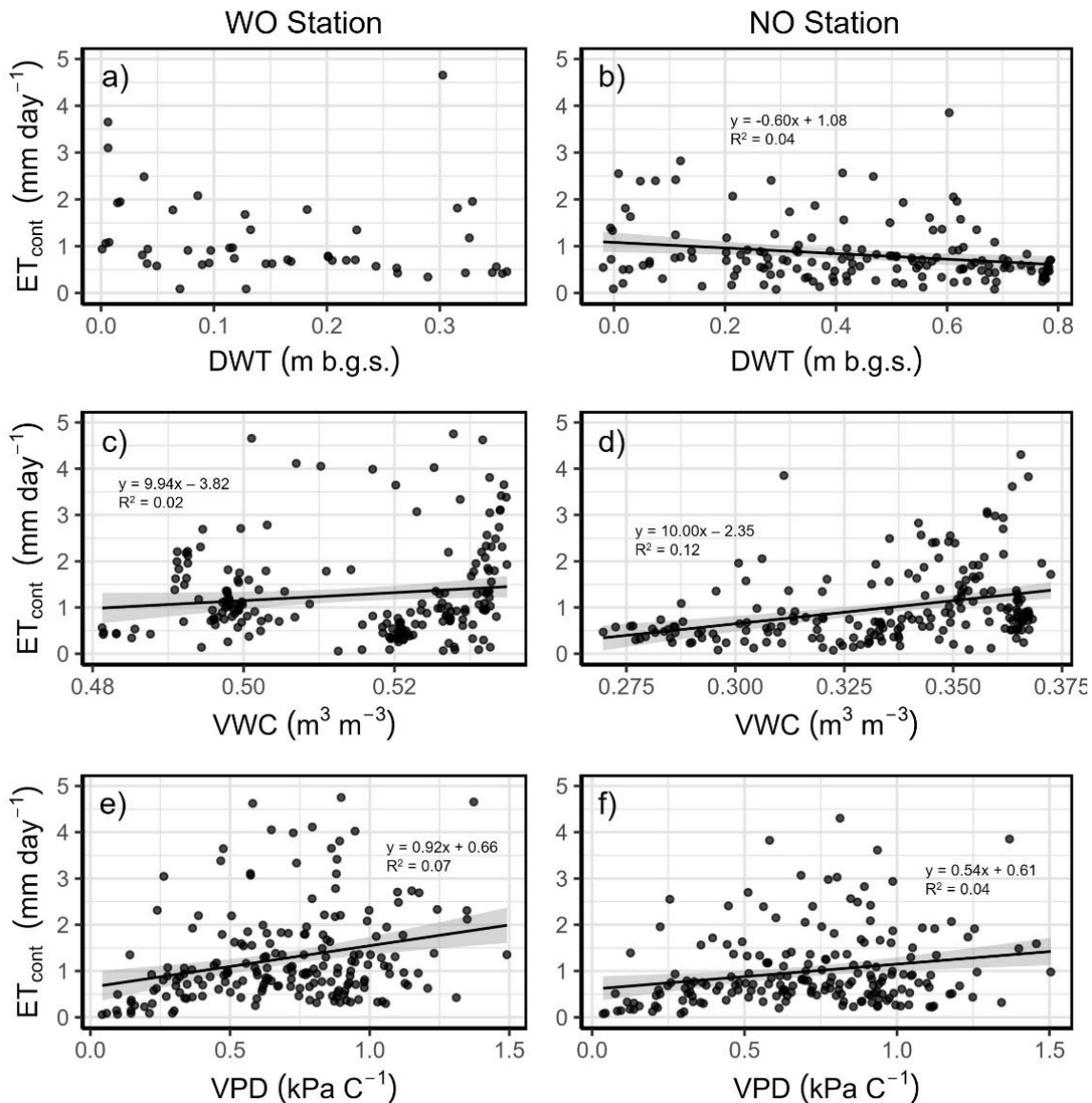


Figure 5.8: Relationships between average daily plot ET_{cont} and depth to water table (DWT; a-b), volumetric water content (VWC; c-d), and vapour pressure deficit (VPD; e-f) as measured at the WO (left plots) and NO (right plots) micrometeorological stations. Data from 2022 and 2023 were combined for regression analyses. Regression lines and equations are provided for significant relationships. Note that the x-axis scales differ between regressions for the WO and NO stations.

Table 5.1: Summary of mean monthly ET_{cont} , net radiation (Q^*), ground heat (Q_G ; day and night), latent heat (Q_E), sensible heat (Q_H), and the ratio of Q_H to Q_E at the micrometeorological stations located in WO and NO plots for 2022 and 2023.

		ET_{cont} (mm day ⁻¹)	Q^* (W m ⁻²)	Q_G (W m ⁻²)		Q_E (W m ⁻²)	Q_H (W m ⁻²)	Q_H / Q_E (unitless)
				7:00 – 20:30 h	21:00 – 6:30 h			
2022								
WO	May	3.6	137	6	5	110	23	0.21
	June	1.5	131	3	-3	80	51	0.64
	July	0.8	145	8	-8	69	77	1.12
	August	0.5	113	8	-11	49	67	1.37
NO	May	2.7	117	14	1	83	26	0.31
	June	1.1	112	14	2	60	43	0.72
	July	0.6	125	15	2	52	64	1.23
	August	0.4	98	9	-1	33	59	1.77
2023								
WO	May	2.4	105	59	44	41	14	0.34
	June	1.6	127	35	35	63	36	0.57
	July	0.9	120	26	26	64	36	0.56
	August	0.7	109	20	20	65	30	0.46
NO	May	2.1	94	12	6	61	24	0.39
	June	1.2	116	15	7	56	48	0.86
	July	0.7	109	10	5	53	47	0.89
	August	0.5	100	8	3	53	42	0.79

5.3.5. Groundwater Fluxes

Due to consistent groundwater inputs across the east and north pad edges (denoted as positive q_h on Figure 5.9g-h), the WT generally remained at or near the surface of the pad in areas along those edges (i.e., WO areas) (Figure 5.9c-d & Figure 5.9c-d). Consistent groundwater inputs across the east and north edges occurred as a result of pad-directed horizontal hydraulic gradients (-0.001 to -0.01) on all measurement dates (Figure 5.9e-f). In interior portions of the residual pad (i.e., NO areas), the WT was more poorly regulated between rainfall events (in some cases falling below 80 cm b.g.s.) than in WO areas (Figure 5.9c-d & Figure 5.9c-d). Values of q_h across the downgradient edge were often negative (Figure 5.9g-h), indicating that the pad frequently lost water to the downgradient fen, particularly in the early season.

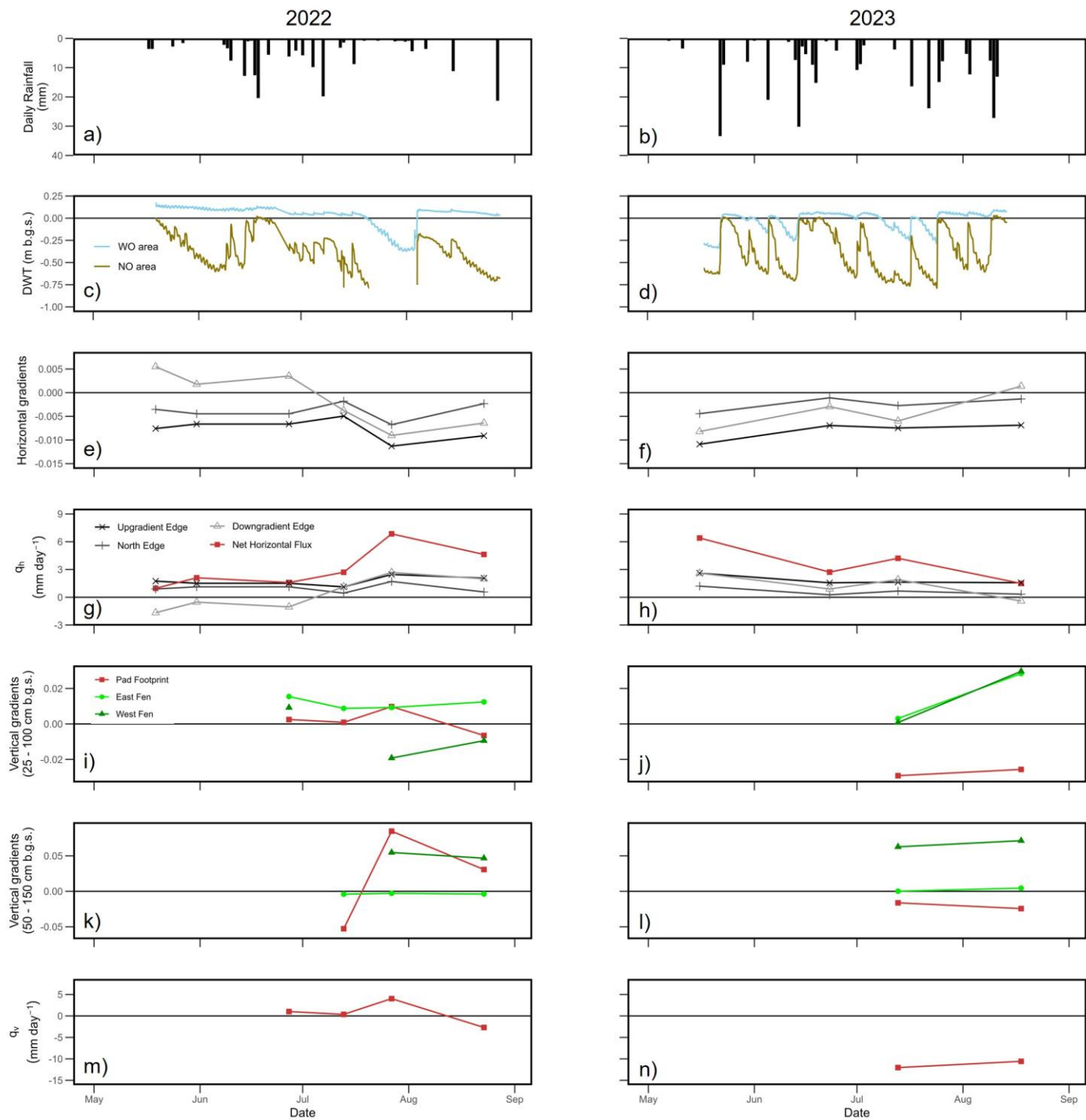


Figure 5.9: Plots of groundwater dynamics on the residual pad. Included are daily rainfall (a-b), depth to water table (DWT; c-d), horizontal hydraulic gradients (e-f), horizontal specific discharge (q_h ; g-h), vertical hydraulic gradients over the 25 – 100 cm (i-j) and 50 – 150 cm (k-l) depths, and vertical specific discharge within the pad footprint (q_v ; m-n). Negative gradients are either directed into the pad (horizontal) or into the ground (vertical). Positive volumetric fluxes are either directed into the pad (horizontal) or towards the surface (vertical).

However, flow reversals across the downgradient pad edge occurred on several dates, resulting in flows from the fen eastward back into the pad (Figure 5.9g-h). Combined, net q_h was positive on all measurement dates (Figure 5.9g-h), resulting in the pad receiving net groundwater inputs of 5.5 and 6.4 mm from the fen during the full duration of the 2022 and 2023 study periods, respectively (normalized over the surface area of the residual pad for comparison to other hydrologic fluxes; Table 5.2). Compared to rainfall inputs (188 – 309 mm), horizontal flow from the adjacent fen thus constituted a relatively small input term in the pad water balance (Table 5.2).

Due to the high degree of WT fluctuation observed across much of the residual pad and hydroclimatic variability between years, ΔS varied considerably between study periods, measuring -81.7 mm in 2022 and +98.6 mm in 2023 (Table 5.2). This reflected an overall decline in WT elevation in the NO portion of the pad in 2022, and an overall increase in WT elevation during the 2023 study period (Figure 5.5c-d). The increase in WT position in 2023 was driven by late-season rainfall events that year (Figure 5.5b). Calculated as the residual of the other pad water balance terms according to Equation 5.5, large net negative vertical groundwater fluxes (i.e., drainage out of the pad) were observed in both 2022 (-140.4 mm $\pm$ 26.6) and 2023 (-113.5 mm $\pm$ 36.6) (Table 5.2). However, these values of Q_v should be interpreted with

Table 5.2: Summary of hydrologic fluxes in the residual well pad. Values of SWE represent the approximate maximum water input from the snowpack within the residual pad footprint (not accounting for sublimation). Values of spring runoff represent the total estimated input of runoff during the spring freshet (March – April). All other values are estimated totals for the growing season study periods in 2022 (May 16 – August 31) and 2023 (May 18 – August 13). Estimates of Q_v were calculated as the residual of the other growing season water balance terms (P , net Q_h , ET_{cont} , and ΔS), and are thus highly uncertain estimates that should be interpreted with caution. Spring freshet water balance terms (SWE and surface runoff) were excluded from the water balance residual calculation as their measurement periods did not overlap with measurement of the growing season water balance terms.

Study period	Spring freshet		Growing season				
	SWE (mm)	Surface runoff (mm)	P (mm)	Net Q_h (mm)	ET_{cont} (mm)	ΔS (mm)	Net Q_v (mm)
2022	-	-	188 ($\pm$ 19)	5.6 ($\pm$ 2.2)	-134.9 ($\pm$ 16.9)	-81.7 ($\pm$ 8.2)	-140.4 ($\pm$ 26.6)
2023	64.5	0.02	309 ($\pm$ 31)	6.4 ($\pm$ 2.6)	-103.3 ($\pm$ 12.9)	98.6 ($\pm$ 9.9)	-113.5 ($\pm$ 36.6)
2024	16.2	0.02	-	-	-	-	-

caution given that they are inclusive of both the actual magnitude of Q_v and the error associated with the estimation of P (~10 %), Q_h (~40 %), ET_{cont} (~12.5 %), and ΔS (~10%) (error percentages estimated based on Healy et al., 2007). Considering the errors of individual components, the error term of the residual (taken as the estimate of Q_v) ranges from 19 to 32 % (Table 5.2). Furthermore, potential small-magnitude overland flow was not included as a measured or estimated variable in the water balance residual calculation, and may thus have constituted an unknown portion of the residual. Finally, vertical hydraulic gradients within the footprint of the residual pad ranged between being positive (surface-directed) and negative (directed into the ground) during the two study periods (Figure 5.9i-j). This resulted in either inputs or losses being observed depending on the measurement date (Figure 5.9m-n), indicating that the estimated net Q_v for each study period does not capture daily-scale variability that could have important hydrological implications over shorter (sub-seasonal) time scales. Nonetheless, values of Q_v calculated using the average of all manually measured vertical hydraulic gradients measured between 2022 and 2023 (-0.008) ranged between -354 mm and -288 mm for the 2022 and 2023 study periods, respectively. This further indicates that Q_v is highly likely to be a net loss term in the pad water balance, with this loss being offset somewhat by at least occasional groundwater discharge towards the surface. Of note, the average of all vertical hydraulic gradients measured within the residual pad between 2020 and 2023 was -0.024 (vertical gradient measurements for the 2020 and 2021 study periods were reported by McKinnon et al., 2024).

5.4. Discussion

Peatland water storage and availability are controlled in large part by the relative magnitudes of the input and loss terms of the ecosystem water balance (Fraser et al., 2001; Wu et al., 2010). Given the subhumid climate on the WBP, characterization of hydrological fluxes on residual well pads is therefore essential to inform potential restoration treatments or modifications to the Partial Removal Technique that may increase water storage and availability to mosses establishing on those sites (M. E. Gauthier et al., 2018; Lemmer et al., 2022; McKinnon et al., 2025, 2024). This may contribute to improved peatland vegetation establishment

outcomes, which tend to occur when the water table remains near the surface, and hydrological conditions are stable (Gonzalez & Rochefort, 2014). As such, the following discussion considers the relative magnitudes of different hydrologic fluxes observed on the residual well pad and their likely importance at different times of the growing season to inform potential modifications to the Partial Removal Technique.

5.4.1. Snow and Ground Frost

The maximum snowpack SWE values reported for the pad in 2023 (65 mm) and 2024 (16 mm) were both lower than the regional average, which has been reported as 91 mm for the nearby Utikuma Region Study Area (URSA) for the period between 2000 and 2011 (Ketcheson and Price, 2016). The URSA is located within a 100 km radius of the residual pad. Notably, SWE in the fen was closer to the regional average in 2023 (81 mm), but not in 2024 (32 mm). Regardless, snowmelt within the footprint of the pad would have provided a large potential input of water in March or early April (Table 5.2). The timing of this input is typical for reclaimed sites on the WBP (Biagi and Carey, 2020; Ketcheson and Price, 2016), although it did correspond with the presence of shallow ground frost within the pad in both years (Figure 5.4). Combined with the high hydraulic conductivity of near-surface peat (Quinton et al., 2008), the presence of continuous layers of seasonal ground ice in natural peatlands can result in rapid lateral distribution of snowmelt during the spring freshet (Price and Fitzgibbon, 1987; Woo and Winter, 1993). A similar effect can be observed on upland hillslopes when spring melt events and refreezing permit the development of concrete frost at the mineral soil surface (Redding and Devito, 2011). As such, it is possible that much of the snowmelt within the pad may have been lost as overland flow to the surrounding fen rather than having infiltrated into the mineral fill. The extent to which this might have occurred would depend upon the antecedent moisture conditions of the fill, which were not quantified as part of the present study.

The fact that snow depth and SWE tended to be lower within the footprint of the pad than in the fen is somewhat unexpected, given that both wide seismic lines (Weiland et al., 2024) and forest gaps

(Bouchard et al., 2022; Golding and Swanson, 1978) have frequently been found to accumulate more snow than surrounding forested areas. This is due to lower amounts of interception in areas without a canopy, as canopy interception can reduce the depth and SWE of the snowpack (Golding and Swanson, 1978). However, the large size of the pad may have allowed snow redistribution from the pad by the wind. This possibility is supported by the fact that a narrow drift of snow was observed along the boundary between the forested downgradient fen and the western pad edge. Along that edge, snow depth was up to twice as deep as the snowpack across the pad itself (data not shown). Wind redistribution can also facilitate considerable sublimation of windblown snow particles (Pomeroy and Gray, 1995), with rates of sublimation potentially exceeding those of snowmelt (Pomeroy and Essery, 1999). Furthermore, the large size of the pad would have permitted greater amounts of incident solar radiation relative to the surrounding treed fen (Williams and Quinton, 2013), which can result in increased rates of sublimation (Sexstone et al., 2016) and therefore also reduced snow depth and SWE (Weiland et al., 2024).

As the pad was primarily vegetated with low sedges and mosses, physical barriers to wind redistribution were limited. However, as a shrub and tree community develops over time, it is possible that wind redistribution may become somewhat reduced, given that shrub growth forms can trap snow (Pomeroy et al., 2006). This could potentially permit a greater accumulation of snow within the pad footprint in the future, thus potentially increasing the water input at the time of snowmelt. However, increased absorption of shortwave radiation and associated heating of seedlings and shrubs can sustain high rates of sublimation and melt (Pomeroy et al., 2006), therefore possibly resulting in earlier snow-free conditions (Ketcheson and Price, 2016). This could mean that a greater proportion of the snowmelt input could be lost prior to the growing season, given that rates of evapotranspiration in WBP peatlands tend to occur at near potential rates in the spring (Van Huizen et al., 2020).

Combined, it appears likely that direct snowmelt from the snowpack located within the footprint of the pad may not actually provide a substantial source of water for the early growing season as expected. However, the ground frost melted at a greater rate within the pad than in the fen in the second season. This

resulted in ground frost in the pad being upwards of 20 cm deeper than in the fen on several dates, likely as a result of the greater thermal conductivity of mineral soils (0.3 to $2.20 \text{ W m}^{-1} \text{ }^{\circ}\text{C}^{-1}$) than peat (0.06 to $0.5 \text{ W m}^{-1} \text{ }^{\circ}\text{C}^{-1}$) (Oke, 1987). This finding also aligns with studies of well pads constructed in permafrost landscapes, where large changes in soil thermal regimes can result in accelerated permafrost melt (Miner et al., 2024). The greater depth to ground frost on the pad could have allowed it to act as a sink for water flowing laterally across the top of the seasonal ground ice in the fen. Particularly in years when ground frost persists into the growing season, this could mean that snowmelt from the fen might support near-surface WTs within the pad itself. However, further detailed study of spring ecohydrological dynamics with a greater frequency of measurements would be required to ascertain the importance of this potential mechanism, and it would likely occur to a lesser extent in years with earlier ice-free conditions. Quantification of overland flow rates off of residual pads during snowmelt is also recommended to ascertain the extent to which snowmelt is retained prior to ground frost melt.

5.4.2. Runoff from the Constructed Upland

Despite the fact that the SWE of the snowpack on the north-facing slopes of the constructed upland was typically greater than that of the pad snowpack, melt of the upland snowpack did not constitute an appreciable term in the pad water balance. The very limited quantity of surface runoff from the constructed upland during the spring freshet (0.11 to 0.14 m^3 ; Table 5.2) and its low spring runoff coefficients ($< 1 \%$ in both years) is not unexpected given that natural boreal upland forests tend to have low runoff coefficients. For example, Redding and Devito (2011) reported a median runoff coefficient of 7% for a range of hillslopes at the URSA, with values as low as 1% being observed in some cases. As previously discussed, substantial runoff from natural hillslopes is generally only observed when concrete frost forms following the refreezing of meltwater from mid-winter melt events (Redding and Devito, 2011). While mid-winter melt events were observed in both years (Figure 5.2), the high K_v of the mineral fill ($3.2 \times 10^{-5} \text{ m s}^{-1}$) indicates that it likely had a very low autumn antecedent moisture content, and thus particularly high

quantities of meltwater might be required for concrete frost development. The fact that the upland slopes were north-facing may also have reduced the effects of mid-winter melt events as is typical on north-facing natural boreal hillslopes due to lower levels of incident solar radiation (Redding and Devito, 2011).

Combined, the total runoff observed during the spring freshet constituted a far smaller input than the typical daily Q_h or rainfall inputs observed during the growing season (Table 5.2). This is in contrast with the upland features included in landscape scale reconstruction projects following open-pit oil sands mining (Biagi and Carey, 2020; Ketcheson and Price, 2016), although this is not unexpected as those uplands are engineered to include confining layers that enhance runoff and subsurface flow (Ketcheson et al., 2017). Indeed, uplands constructed on residual mineral well pads without confining layers cannot be expected to play a major hydrological role on these sites. Nonetheless, further study should be undertaken to determine whether constructed uplands might contribute to increased groundwater discharge to residual pads, given that natural boreal uplands and peatlands are often hydrologically coupled through groundwater exchange (Elmes and Price, 2019).

5.4.3. Groundwater Fluxes and Rainfall

The magnitude of Q_h was small compared to that of P , Q_v , and ET_{cont} (Table 5.2). The small magnitude of seasonal Q_h was a direct result of the small size of the flow faces along the edges of the pad through which Q_h occurs (approximately 80 m² per side) as compared to the surface area of the pad (approximately 6058 m²) over which P , ET_{cont} , and Q_v occur. Nonetheless, a previous study on the same site identified the likely importance of consistent net Q_h inputs in maintaining near-surface WTs in areas adjacent to the upgradient and north edges of the pad (i.e., WO areas) (McKinnon et al., 2024). That study found that net inputs in the form of Q_h increased in response to lower WTs in WO areas, as lower WTs resulted in increases in the magnitude of pad-directed horizontal hydraulic gradients (McKinnon et al., 2024). Similar temporal patterns were also observed in the present study (Figure 5.9e-h), confirming the likely importance of the Q_h term. Previous trials of partial removal have recommended or incorporated the excavation of trenches (Vitt

et al., 2011) or the installation of subsurface flow conduits (Guérin, 2022) to increase horizontal hydrological connectivity with surrounding peatlands. The present findings suggest that these efforts may be especially valuable.

In comparison, the absolute magnitude of Q_v exceeded that of Q_h on all measurement dates (Table 5.2). The fact that large net losses to the underlying peat were observed under a wide range of conditions (and particularly following rainfall; Figure 5.9m-n) agrees with the findings of McKinnon et al. (2024), who identified downward-directed vertical hydraulic gradients on the same residual well pad under most conditions. That said, surface-directed vertical hydraulic gradients and associated surface-directed Q_v fluxes were observed within the footprint of the residual pad on several dates (Figure 5.9i-n). This indicates that groundwater discharge did constitute a component of the residual pad water balance, and that this discharge served to offset losses through drainage to a certain extent. However, the low number of dates for which manual measurements of vertical hydraulic gradients were available precludes a more detailed analysis of the relative contributions of groundwater recharge and discharge to the pad water balance. Furthermore, the estimates of seasonal net Q_v fluxes presented herein are uncertain as they were calculated as the residual of other growing season water balance terms (Table 5.2). This means the values presented are inclusive of both the actual Q_v fluxes occurring within the footprint of the pad and the error associated with estimation of P , Q_h , ET_{cont} , and ΔS (Winter, 1981), as well as possible small-magnitude overland flow. However, the fact that long-term average vertical hydraulic gradients measured between 2020 and 2023 were negative (i.e., directed into the ground) suggests that it is reasonable to assume that groundwater recharge (i.e., downward flow out of the residual mineral fill) constitutes one of the most important water loss terms in the pad water balance.

While vertical groundwater exchange does occur in natural peatlands (Devito et al., 1997; Elmes and Price, 2019; Ferone and Devito, 2004), the likely importance of Q_v in the pad water balance presents a major concern in terms of substrate water conservation. As such, efforts to either increase the water retention capacity of residual fill materials (e.g., by increasing organic matter content; Hugron et al., 2013) or to

reduce vertical drainage (e.g., by focusing partial removal efforts on pads underlain by impermeable or semi-impermeable geotextiles) should be explored. Notably, water inputs in the form of groundwater recharge (i.e., surface-directed flow) could be impeded or cut off by the presence of a geotextile liner between a residual well pad and underlying compacted peat. This suggests the need to assess the magnitude of Q_v fluxes on pads underlain by geotextile liners, particularly given that they are commonly used in well pad construction (Osco et al., 2018). The fact that vertical hydraulic gradients between the residual mineral fill and underlying peat, and between the underlying peat and basal mineral substrate changed directions periodically (Figure 5.5i-l) suggests that the well pad may have been situated over a local groundwater flow system, and was potentially hydrologically connected to local upland features (Elmes & Price, 2019).

5.4.4. Energy Balance and Evapotranspiration

The energy balance on the residual pad was dominated by Q_E (i.e., $Q_E > Q_H$ and Q_G) at both micrometeorological stations in the early growing season of 2022 and for the full duration of the 2023 season (Table 5.1). This is in alignment with both the regionally high evaporative demand and the energy balance at a constructed fen in the open-pit oil sands mining region near Fort McMurray, Alberta (Popović et al., 2023; Scarlett et al., 2017). However, Q_E in that constructed fen was greater than Q_H under nearly all conditions over several years (Popović et al., 2023). In contrast, Q_H became the dominant component of the energy balance in both the WO and NO areas on the residual pad in July and August 2022 (Table 5.1). While the dominance of Q_E over the energy balance of natural and constructed peatlands is normally observed during periods of ample water supply (Brown et al., 2010; Scarlett et al., 2017), the dominance of Q_H over the energy balance can occur during dry conditions that limit water supply to the surface (Petrone et al., 2007). Indeed, declines in VWC were observed at the NO station (Figure 5.7c) and across the entire NO area (Figure 5.6a) in July and August 2022 when rainfall was limited (Figure 5.5a). These declines in VWC were thus likely responsible for the high proportion of available energy partitioned as Q_H in those

months, highlighting the importance of VWC and DWT as controls over evaporative losses, especially in NO areas (Figure 5.8).

However, significant declines in VWC were not observed across the WO area (Figure 5.6a) or at the WO station (Figure 5.7c) in the late season that year, despite the similar increase in the importance of Q_H there (Table 5.1). That said, diurnal fluctuations in substrate temperature were larger at the WO station than at the NO station in 2022, which frequently resulted in higher daily maximum substrate surface temperatures at the WO station (Figure 5.7g). These high substrate surface temperatures could have resulted in greater losses of heat to the atmosphere in the form of Q_H by creating a greater temperature gradient. The greater diurnal variability is somewhat unexpected, given that a unit temperature increase in wetter soils like those at the WO station requires more energy than the same increase in drier soils like those at the NO station (Jury and Horton, 2004b). Thus, the greater substrate temperatures at the WO station are likely a result of the fact that monthly mean Q^* tended to be somewhat higher at the WO station than at the NO station (Table 5.1). The higher Q^* at the WO station may have been a result of its greater distance from the natural forested upland to the south of the site (Figure 5.1), as a tall stand of spruce and pine there was observed to have shaded the NO station slightly earlier in the late afternoon than the WO station.

Due to the predominance of Q_E over the energy balance under most conditions, rates of ET_{total} and ET_{cont} were sufficiently high to make evapotranspiration a significant negative hydrologic flux. Specifically, rates of ET_{total} were frequently sufficient to offset the combined inputs of pad-directed Q_h and surface-directed Q_v (Table 5.2). Cumulative losses through ET_{cont} were also sufficient to offset between 29 % (i.e., NO area in the high-rainfall 2023 season) and 82 % (i.e., WO areas in the low-rainfall 2022 season) of rainfall inputs (Figure 5.7a-b). The fact that rates of ET_{total} exceeded the daily average rainfall rate in the early season in 2022 was a result of extensive surface ponding across both the WO and NO areas (Figure 5.5e). This permitted a greater fraction of ET_{total} to be made up of E_a (Table 5.2), the rates of which were higher than those of ET_a at that time (Figure 5.5). This is in alignment with natural ecosystems on the WBP, wherein rates of evapotranspiration are greater from open water systems than from peatlands (Petrone et

al., 2007). Similarly high rates of evapotranspiration (4.4 mm day^{-1}) have also been observed in ponded areas of a constructed fen on the WBP (Scarlett et al., 2017). Combined with the fact that ponding can result in extensive colonization of *Typha latifolia* and erosion of established moss carpets (Lemmer et al., 2022, 2020), this indicates that there is a need to limit the spatial extent of ponding on residual pads.

The lack of a direct significant effect of mulching on ET_a in either year and the limited spatial differences in ET_a between mulched and non-mulched plots align with the findings of McKinnon et al. (2024), which found that mulch application did not have a direct significant effect on volumetric water content or soil water potential (i.e., water availability) in the sandy mineral substrate. This was attributed to the water table falling sufficiently far below the surface to cut off capillary flow from the water table to the surface in NO areas, and to soil water supply to the surface being non-limiting in WO areas. However, it is possible that the high degree of spatial and temporal variability in the physical measurements of ET_a may have masked any subtle differences between mulched and non-mulched plots, should they exist. Specifically, significant differences in ET_a were also rarely identified between the WO and NO areas, despite the fact that cumulative ET_{cont} (calculated daily using the energy balance) was higher at the WO station than the NO station for the duration of both seasons (Figure 5.7a-b). As such, further study is recommended to determine whether mulching has an effect on rates of atmospheric water loss over longer timescales than those captured by the lysimeter measurements presented in this study. Further study of the effects of mulching should also be undertaken on residual well pads consisting of finer-textured fill materials, given that clays have higher water retention capacities than sandy substrates (Jury and Horton, 2004a). This might restrict drainage from finer-textured pads, thus allowing greater rates of ET_a under a wider range of moisture conditions and, consequently, increasing the efficacy of straw mulch.

5.5. Conclusion

The maintenance of high substrate moisture levels on residual well pads undergoing fen restoration is of paramount importance given the susceptibility of peatland mosses to desiccation under dry conditions. This study confirms that horizontal hydrological connectivity with adjacent peatlands is insufficient to maintain near-saturated substrate moisture levels across the full extent of residual pads. Instead, the primary growing season water input was rainfall, which was largely offset by a combination of evapotranspiration and rapid vertical drainage. Manually measured open-water evaporation rates were often higher than both manual and half-hourly energy-balance-based measurements of evapotranspiration, highlighting the importance of minimizing the extent of surface ponding on residual pads. Snowmelt may have provided an important early-season water input, but further study is required to determine how much of that input may be rapidly lost to the surrounding fen as a result of overland flow. Similar to natural boreal uplands, surface runoff from upland features constructed out of the excess fill removed during partial removal cannot be expected to provide a sustainable source of water to adjacent residual pads. However, the degree of groundwater connection between upland features and residual pads should be examined in further detail. Overall, the key findings of this study suggest that future trials of partial well pad removal should seek to increase the magnitude of horizontal hydrological connectivity across residual pads to reduce reliance on precipitation. Additional study on finer-textured residual pads and those underlain by geotextile liners is also required, given the likely control of these factors on vertical groundwater exchange, and, thus on evapotranspiration and substrate water conservation.

6. CONCLUSIONS AND RECOMMENDATIONS

This thesis presented the results of a five-year ecohydrological study conducted following the first full-scale implementation of the Partial Removal Technique. The goal of the thesis was to assess the suitability of environmental conditions on a residual well pad for the initiation of peatland vegetation introduced using the MLTT. To do so, individual studies were conducted to assess the extent to which the residual pad was hydrologically connected to the adjacent fen (Chapters 2 & 5), the degree to which moisture and biogeochemical dynamics can be expected to support the establishment of true mosses (Chapters 3 & 4), and to quantify the relative magnitudes of additional hydrological fluxes (Chapter 5). The following section provides a summary of the key findings of the thesis. It subsequently integrates the findings of each chapter to propose modifications to the Partial Removal Technique that may contribute to improved conditions for fen vegetation initiation and establishment. Finally, it expands upon the recommendations made in each individual chapter to highlight key areas of priority for future research.

6.1. Key Findings

6.1.1. Hydrological Connectivity and the Residual Pad Water Balance

The low horizontal hydraulic conductivity of the residual loamy sand and the compacted peat underlying it presented a limitation on hydrological connectivity between the pad and the fen (Chapter 2). This was reflected in the low magnitude of horizontal groundwater fluxes observed across the pad edges compared to the magnitudes of other hydrologic fluxes (Chapter 5). Nonetheless, groundwater inputs from the fen were identified as being essential to maintain near-surface WTs within 20-30 metres of the upgradient and north pad edges (i.e., WO areas; Chapters 2 & 5). The low groundwater fluxes across the pad edges also resulted in preferential flow through the peatland around the northeast corner of the pad (Chapter 2). This indicates that partial removal did not alleviate impacts to ecohydrological function in the fen.

Furthermore, the low magnitude of horizontal groundwater fluxes across the interior of the residual pad meant that areas away from the upgradient edges did not have a high degree of hydrological connection to the adjacent fen (Chapter 2). The WT in those areas (i.e., NO areas) was instead highly responsive to rainfall events and vertical drainage of water into the underlying peat (Chapter 5). This resulted in a poorly regulated WT, which had considerable implications for substrate water availability (Chapter 3). Flow reversals from the downgradient peatland into the residual well pad were insufficient to offset the rapid vertical drainage of the residual mineral fill in those areas (Chapters 2 & 5). This is a novel finding, given that the Partial Removal Technique has only been applied on either narrow roads (Guérin, 2022) or the edges of well pads to date (M. E. Gauthier et al., 2018; Lemmer et al., 2022, 2020; Vitt et al., 2011), where the WT has generally remained near the surface. It also suggests that there is a need to enhance horizontal hydrological connectivity across residual well pads (i.e., through the creation of flow conduits) in order to improve the distribution of water (Chapters 2 & 5). There is also a clear need to reduce the magnitude of vertical drainage from coarse-textured residual mineral substrates by testing restoration treatments that might increase their water retention capacity, such as the addition of organic matter amendments or mixing of mineral fill with underlying peat through ripping of the pad surface (Chapter 5).

Evapotranspiration was a large loss term in the pad water balance, with the greatest rates of atmospheric water loss frequently being observed when a considerable portion of the pad's surface area was inundated with surface water (Chapter 5). Combined with the potential for ponding to encourage the establishment of *Typha latifolia* and to prevent the formation of moss carpets (Lemmer et al., 2020, 2022), this indicates that there is a need to reduce the extent of surface ponding across residual well pads by refining the reprofiled surface elevation. However, daily rates of evapotranspiration from non-inundated surfaces were also sufficiently high to offset large proportions of the water inputs to the residual pad (Chapter 5), suggesting that efforts should also be taken to reduce bare soil evaporation. That said, the straw mulch surface covering applied in the present study had a limited effect on evapotranspiration rates (Chapter

5). Additional study of the effects of mulch surface coverings at a higher temporal resolution is thus required.

Finally, the exact magnitude of snowmelt inputs could not be determined with confidence, given that much of the initial snowmelt input during the spring freshet may have been lost to the surrounding fen owing to potential formation of concrete frost within the pad footprint (Chapter 5). Thus, additional detailed study of spring ecohydrological dynamics on residual well pads is also required. What is clear is that hillslopes constructed out of the excess mineral fill removed during civil earthworks cannot be expected to provide an appreciable source of snowmelt or surface runoff to residual well pads (Chapter 5), further confirming the importance of direct rainfall within the pad footprint as a source of water during the growing season.

6.1.2. Residual Mineral Substrate Moisture Dynamics

The well regulated WT observed within 20-30 metres of the upgradient pad edges resulted in the maintenance of moisture conditions that were optimized for the establishment of true mosses for much of the growing season (Chapter 3). In those areas, the creation of microtopography through roughing of the residual mineral substrate (Chapter 2) provided a gradient of microhabitats that was likely to support the establishment of a range of moss species with different hydrological tolerances (Chapter 3). In contrast, the poorly regulated WT in NO areas of the residual pad was associated with more variable substrate moisture conditions for true moss establishment. Specifically, literature-based desiccation thresholds were occasionally exceeded in those areas, and the WT frequently fell below 6 cm depth, which was identified as the threshold separating optimal from non-optimal moisture availability to mosses (Chapter 3). Combined, the NO areas of the residual pad can be said to support a mesic to hygric moisture regime, while the WO areas can be said to support a sub-hydric to hydric moisture regime (Chapter 4). The application of a straw mulch surface covering had no clear effect on substrate water content or availability to mosses,

which may have been a result of the poor water retention capacity of the residual loamy sand substrate (Chapters 2 & 3).

6.1.3. Residual Mineral Substrate Nutrient Biogeochemical Dynamics

Nutrient regimes were relatively consistent across the residual well pad (Chapter 4), indicating that short-term vegetation establishment outcomes are likely to be controlled primarily by spatial differences in moisture regime (Chapter 3), as well as changes in moisture availability over time (Chapter 4). However, the availability of nitrogen was limited for the duration of the study, and both phosphorus and nitrogen were rapidly immobilized or taken up by vegetation in all years (Chapter 4). This indicates that vegetation productivity may be limited by nutrient availability across the full site, at least until an increase in the availability of organic substrates for decomposition occurs as a result of litter build-up (Chapter 4). Regardless of nitrogen and phosphorus availability, the similarity between the base cation supply rates, pH, and EC observed within the residual well pad and those reported for regional moderate and extreme-rich fens suggests that true moss establishment can be expected in areas with optimal moisture conditions (Chapter 4). This would be in alignment with previous vegetation establishment patterns on residual mineral substrates (M. E. Gauthier et al., 2018; Lemmer et al., 2022, 2020). Notably, application of rock phosphate fertilizer had a very limited effect on mineral fill nutrient biogeochemical dynamics, as did the application of the straw mulch surface covering (Chapter 4). Combined with the limited efficacy of straw mulch in improving substrate moisture dynamics (Chapter 3) and reducing atmospheric water losses (Chapter 5), it appears that these treatments may not be effective when applied onto coarse-textured residual well pads (Chapter 4).

6.1.4. Potential Vegetation Successional Trajectories

The high base cation content of the residual mineral fill resulted in the site being characterized by a rich to very rich (minerotrophic) nutrient regime (Chapter 4; Beckingham and Archibald, 1996). Furthermore, increases in the availability of nitrogen and phosphorus over the medium to long-term are likely, owing to the eventual decomposition of the litter layer observed to be developing on the surface of the pad (Chapter 4). Combined, this indicates that partial removal of the well pad resulted in conditions that were likely favourable for the establishment of vegetation characteristic of nutrient-rich ecosites on the WBP (Chapter 4), as also found in previous studies (M. E. Gauthier et al., 2018). However, considerable variation is common between natural nutrient-rich boreal ecosites based on differences in site-specific moisture regimes (Beckingham and Archibald, 1996). As such, the WO and NO areas of the residual pad are likely to follow differing successional trajectories, and will likely be characterised by different vegetation community structures in the years to come (Chapters 3 & 4).

Specifically, the consistently wet conditions in WO areas (sub-hydric to hydric moisture regime) were identified in Chapter 3 as being potentially favourable for the establishment of peatland mosses including *Tomentypnum nitens*, *Drepanocladus aduncus*, *Helodium blandowii*, and *Ptychostomum pseudotriquetrum* (Figure 6.1). High and low microsites might also be colonized by submergence-intolerant *Aulacomnium palustre* and submergence-tolerant *Hamatocaulis vernicosus*, respectively (Chapter 3; Figure 6.1). This assessment was based on the fact that those species have been identified in previous trials of moss initiation on cation-rich residual mineral substrates at various scales in areas with well regulated hydrological dynamics (Borkenhagen and Cooper, 2016; M. E. Gauthier et al., 2018; Guérin, 2022; Lemmer et al., 2022, 2020). Many of them also tend to be found in natural rich fens on the WBP (Vitt, 2014).

Notably, the WO areas can also be expected to support the establishment of a graminoid community characterized by *Carex* and *Scirpus* spp. (Figure 6.1) given their dominance in previous trials of partial well pad removal (M. E. Gauthier et al., 2018; Lemmer et al., 2022, 2020; Vitt et al., 2011). Areas with nearly-permanent surface inundation might also support establishment of *Typha latifolia* (Figure 6.1; Lemmer et

al., 2022, 2020), given how common this species is in regional mineral open water wetlands (Alberta Environment, 2015). Furthermore, natural peatland ecosites on the WBP characterized by similar moisture and nutrient regimes also commonly support robust communities of *Rhododendron groenlandicum*,

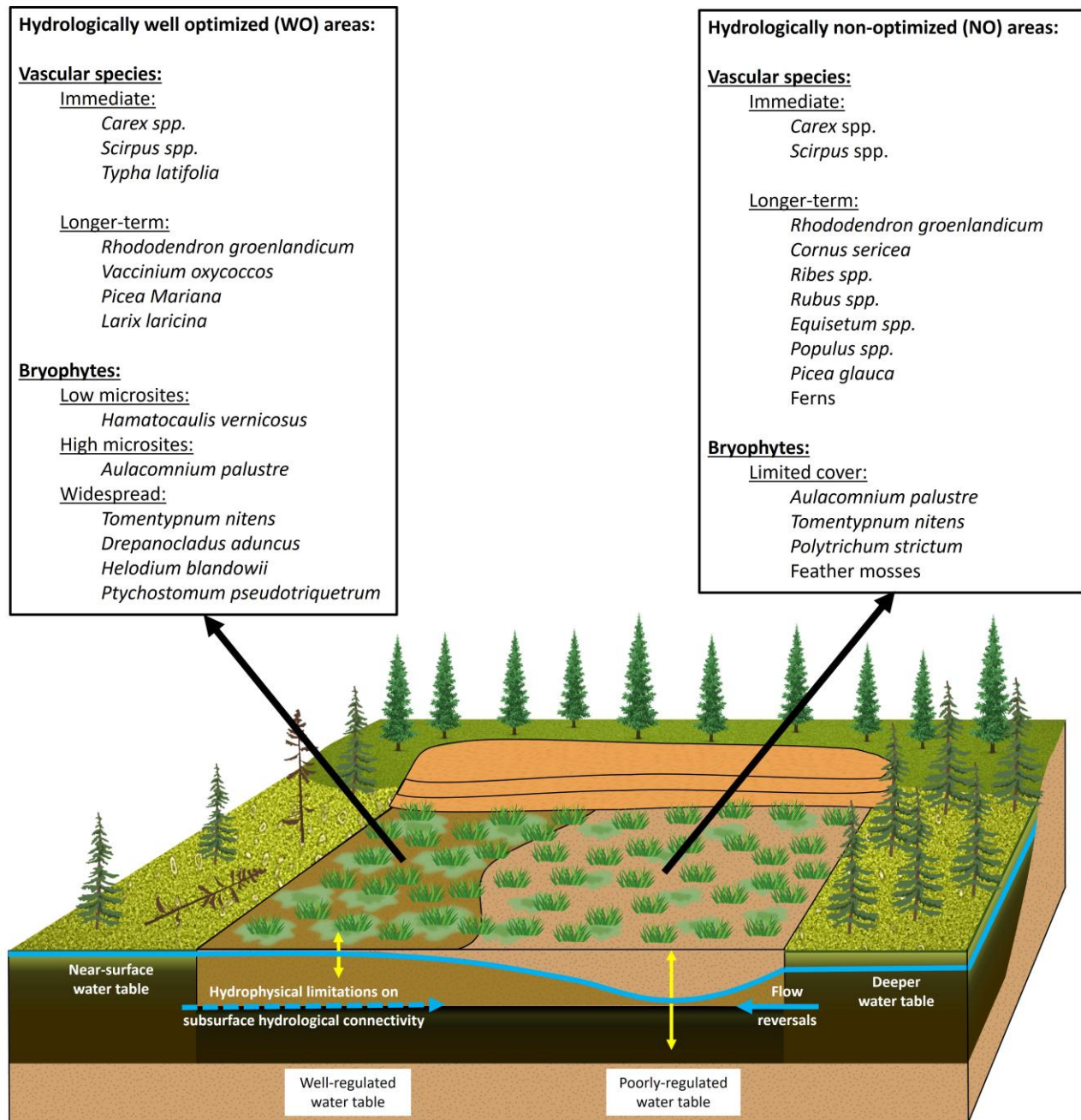


Figure created with the Health Canada Conceptual Site Model Builder tool and graphics from BioRender.com (used under license).

Figure 6.1: Conceptual model of potential vegetation establishment outcomes in hydrologically well optimized (WO) and hydrologically non-optimized (NO) areas of the residual well pad. Potential vegetation establishment outcomes were informed by previous trials of partial well pad removal (references in-text) and current environmental conditions on the site.

Vaccinium oxycoccos, and canopies of *Larix laricina* and *Picea mariana* (Beckingham and Archibald, 1996). While the site currently lacks a layer of peat, this does suggest that WO areas on the residual well pad might support the establishment of typical fen vegetation communities over the long-term (Chapter 4; Figure 6.1). That said, it may take a considerable amount of time for these sites to develop layers of peat given the slow rates of peat accumulation observed in natural fens on the WBP (Halsey et al., 1998; Kuhry et al., 1992). As the development of such a peat layer will be essential for ecosystem functional processes to resemble those in reference peatlands (Nwaishi et al., 2015a), this means the establishing ecosystem may remain vulnerable to shifts towards divergent successional trajectories in response to disturbances like drought for some time.

Conversely, the poorly regulated hydrological dynamics observed across NO areas of the residual pad resulted in a moisture regime best described as mesic to hygric for much of the growing season (Chapters 3 & 4). Chapter 3 identified the potential of these areas to support robust communities of *Carex* and *Scirpus* spp. (Figure 6.1) given their extensive rooting systems and ability to access water at depth (Borkenhagen et al., 2024; Chapin and Chapin, 1981; Koropchak et al., 2012). However, it is likely that the highly variable substrate moisture conditions observed in those areas may restrict the establishment of extensive moss carpets, and may only favour desiccation tolerant moss species (Chapter 3). In turn, this might result in only discontinuous cover of particularly hardy mosses like *Tomentypnum nitens*, *Aulacomnium palustre*, and *Polytrichum strictum* (Borkenhagen and Cooper, 2016; M. E. Gauthier et al., 2018; Lemmer et al., 2022, 2020), as well as non-peatland feather mosses that have been found to colonize dry areas in disturbed peatlands on the WBP (Miller et al., 2015) (Figure 6.1).

Regardless of species composition, the coverage of peatland mosses is likely to be more limited in NO than in WO areas due to the greater frequency of non-optimal moisture conditions there (Chapter 3). Furthermore, natural nutrient-rich ecosites on the WBP with similar moisture regimes (and which lack substantial peat accumulation) are also characterized by a range of shrubs and trees, only some of which are characteristic of peatlands (Beckingham and Archibald, 1996). This might result in the establishment of

a community including plants like *Cornus sericea*, *Ribes* spp., *Rubus* spp., *Equisetum* spp., and ferns (Chapter 4; Figure 6.1). Of note, it might also support the establishment of trees like *Picea glauca* or *Populus* spp. (Chapter 4; Figure 6.1), which may contribute to shifting the vegetation community in NO areas to one more typical of swamps on the WBP (Alberta Environment, 2015; Beckingham and Archibald, 1996; Elmes et al., 2021a). This further highlights the need to modify the Partial Removal Technique to support better-regulated hydrological dynamics if fen vegetation communities are to be established across their full extent. Thus, the next section integrates the findings of the individual chapters of this thesis to identify potential modifications of research and practical interest.

6.2. Potential Modifications to the Partial Removal Technique

6.2.1. Refinement of the Target Reprofiled Surface Elevation

The surface elevation of the fen on the downgradient side of the residual well pad examined herein was considerably lower than that of the fen on its upgradient and north sides (Chapter 2). However, the target reprofiled surface elevation of the pad itself was based on a pre-removal survey of ground ice elevation in the upgradient fen only, as no ground ice elevation measurements were made in the downgradient fen due to logistical and time constraints. This survey was subsequently used to identify a single target reprofiled surface elevation for use during civil earthworks, meaning that the residual pad had an approximately even elevation across its full extent (i.e., it was not intentionally sloped). Combined, this meant that the reprofiled pad surface was considerably higher than that of the peat surface and the WT in the fen on the downgradient side of the pad at all times (Chapter 2). As such, this thesis recommended sloping the reprofiled surfaces of future residual pads to better align them with the WT elevation in the peatland on all sides of a given pad. This would require determination of the elevation of the seasonal ground ice adjacent to all sides of a well pad prior to the commencement of civil earthworks. Notably, sloping of residual well pads would be more challenging to implement in the field than targeting a single reprofiled elevation, especially when the extent

of surface ponding is to be minimized. Nonetheless, the findings of this thesis suggest that this modification should be considered in future trials, given its potential to improve substrate moisture dynamics in areas of residual pads not receiving consistent groundwater inputs (Chapters 2 & 3).

6.2.2. Improvement of Horizontal Hydrological Connectivity

This thesis identified the importance of horizontal hydrological connectivity for the maintenance of near-surface WTs in portions of the residual pad (Chapters 2 & 5). However, because Q_h represented a small term in the pad water balance (Chapter 5), horizontal hydrological connectivity with the surrounding fen was insufficient to maintain near-surface WTs and optimal substrate moisture conditions across a large extent of the pad's surface (Chapters 2, 3, & 5). The need to consider the creation of either surface or subsurface flow conduits across future residual well pads was therefore identified. Surface flow conduits (i.e., trenches) were first tested by one of the earliest trials of partial well pad removal (Vitt et al., 2011). While the open trenches created in that case resulted in water distribution into the residual well pad (Vitt et al., 2011), they also promoted undesirable establishment of *T. latifolia* (B. Xu, personal communication). As such, a more recent trial of partial road removal tested the creation of subsurface flow conduits. These were constructed by first excavating shallow trenches across the residual mineral substrate and filling them with bales of straw that were subsequently buried under a layer of mineral fill (Guérin, 2022). However, it remains unclear whether straw-filled subsurface flow conduits would be feasible at the scale of a full-size residual well pad. Also of concern is the fact that the straw bales could become compressed or clogged with peat from the adjacent peatland, thus potentially reducing their efficiency.

An alternative form of subsurface flow conduit could include backfilling trenches with coarse gravel, as it has a high effective porosity (Domenico and Schwartz, 1998) and a rigid structure. Subsurface flow conduits constructed in this way could be covered with a permeable geotextile material before placing mineral fill over it to provide a spatially consistent mineral substrate for vegetation establishment.

Alternatively, it could be possible to leave gravel at the surface of such flow conduits, given that fen mosses have been observed to establish directly on saturated gravel substrates in past trials (Amon et al., 2005). Another potential option would be to combine the Complete Removal and Partial Removal techniques, wherein sections of residual mineral substrate could be interspersed with elongated sections within which the mineral fill could be completely removed. While this would be operationally complex, a high degree of subsurface flow has been observed through the peat substrates re-exposed following the complete removal of both roads (Elmes et al., 2021b) and well pads (Lemmer et al., 2022; Xu et al., 2022). This could permit better distribution of water into the patches of residual mineral fill that are left in place by increasing the proportional surface area of the flow faces along their edges.

6.2.3. Application of Soil Amendments

Given that the poorly regulated WT across much of the residual pad resulted in rapid fluctuations in substrate water availability to establishing vegetation, the application of organic soil amendments may be a useful restoration treatment following partial removal (Chapters 3 & 4). This could both increase substrate water retention (Rawls et al., 2003) and increase the availability of organic substrates for microbially-mediated decomposition processes (Nwaishi et al., 2015a), thus potentially increasing the availability of both water and nutrients (Chapters 3 & 4). Soil amendments have been employed in the restoration of borrow pits (Hugron et al., 2013) and disturbed upland sites (Hemstock et al., 2010) in boreal regions, although trials on residual mineral substrates have been limited to date (M. E. Gauthier et al., 2018). Notably, it may be especially useful to target the use of locally-sourced peat amendments on residual well pads (Chapter 4). These would be less likely to result in the introduction of undesirable ruderal species, and would likely contain humic acids that might limit phosphorus precipitation with iron, aluminum, and calcium contained within the mineral fill (Ge et al., 2020; Lobartini et al., 1998).

6.2.4. Refinement of Microtopography

The fact that the provision of microtopography was associated with the creation of microhabitats that were generally suitable for the establishment of a wide range of true moss species suggests that this is an important restoration treatment (Chapter 3). This is also in alignment with previous trials of moss establishment on mineral substrates (Borkenhagen and Cooper, 2018, 2016). However, high microsites in NO areas examined in the present study (which had a mean height of between 0.13 and 0.19 m above low microsites) were associated with more frequent exceedance of moss desiccation thresholds than low microsites (Chapter 3). Thus, it may be beneficial to reduce the amplitude of the microsites created during future implementations of the Partial Removal Technique. This would enable the creation of a range of microsites, but would also reduce the risk of desiccation at high positions during especially dry conditions. Notably, the effects of microtopography may differ on residual well pads constructed out of fine-textured mineral substrates, and further study of moisture dynamics on clay pads is thus also recommended (see Section 6.3.3).

6.2.5. Modification of the Moss Introduction Ratio

The moss community establishing on the residual well pad examined in this thesis was observed to be spatially discontinuous, with extensive areas of bare mineral substrate remaining at the surface (Chapter 5). In natural peatlands, the physical structure of continuous moss carpets plays a major role in regulating evaporative water losses by controlling the magnitude of capillary flow to the surface and decreasing available energy during dry conditions (Brown et al., 2010; McCarter and Price, 2014). Therefore, moss community growth forms are unlikely to have had a substantial control over evaporative water losses on the residual well pad examined here (Chapter 5), and it may thus be useful to increase the density of the introduced moss community (i.e., increasing the ratio of the area of the restoration site to that of the harvested donor site). While this would result in greater disturbance to an offsite donor peatland, it may be

a desirable option in cases where the construction of new well pads or other industrial facilities will result in the loss of peatland vegetation anyway. In such instances, it might be possible to collect mosses from a peatland prior to the construction of a well pad or other facility for use in the restoration of a nearby decommissioned well pad (see Chapter 3).

6.3. Recommendations for Further Research

6.3.1. Long-term Vegetation Establishment Monitoring

While the results presented in this thesis confirm that the Partial Removal Technique has promise for the reestablishment of peatland vegetation communities on decommissioned well pads, long-term monitoring of actual vegetation establishment outcomes will be required. Specifically, although near-saturated moisture conditions on residual mineral substrates have supported the establishment of favourable peatland vegetation communities in past trials (Borkenhagen and Cooper, 2016; M. E. Gauthier et al., 2018; Lemmer et al., 2022, 2020; Vitt et al., 2011), it is likely that areas with a more variable hydrological regime may follow divergent successional trajectories (see Section 6.1.3). Reestablishing vegetation communities can also be set on divergent successional trajectories in response to changes in environmental conditions (e.g., decreased moisture availability due to drought), making long-term establishment outcomes difficult to predict (Audet et al., 2015; González and Rochefort, 2014; Pinno and Hawkes, 2015). As such, continued monitoring of vegetation establishment on this and other residual well pads will likely need to occur on a multi-decadal scale. Of note, similar long-term monitoring has also been found to be necessary during the assessment of restoration techniques applied to cutover and disturbed peatlands (Chimner et al., 2017; González and Rochefort, 2014). Recent results also indicate that multi-decadal monitoring is likely required to fully assess the development of ecosystem function following the construction of peatlands in post-mining landscapes (Borkenhagen et al., 2024; Popović et al., 2023, 2022). These long timescales reflect the

long successional trajectories associated with natural peatland development (Halsey et al., 1998; Kuhry et al., 1992).

6.3.2. Long-term Environmental Monitoring

Similar to the need for long-term vegetation establishment monitoring, there is also a need for ongoing monitoring of substrate moisture conditions and the development of biogeochemical processes on this and other residual well pads (Chapters 3 & 4). This will be especially important given that the development of an organic litter layer is likely to occur over time (Chapter 4), as this has been observed to affect ecosystem energy balances and moisture dynamics (Popović et al., 2023; Scarlett et al., 2017) as well as nutrient cycling and availability (Popović et al., 2022). Furthermore, increases in organic matter availability can be expected to result in increases in the water retention capacity of mineral substrates (Rawls et al., 2003), which may decrease the magnitude of vertical drainage from the residual mineral fill over time (Chapters 2 & 5). That said, increases in substrate water retention capacity could also increase evaporative losses, given that the supply of water at the surface would be increased, thereby potentially sustaining stage I evaporation for longer (Chapter 5). The combined effects of these changes on the water balance of residual well pads remains unknown, as does the rate at which these changes may occur and their effects on both moisture and nutrient availability to establishing vegetation.

6.3.3. Trial of Partial Removal on Fine-Textured Residual Well Pads

As the present study was conducted on a residual well pad constructed out of loamy sand, it is unlikely that the results will be fully applicable to fine-textured residual well pads. Specifically, there is a need to assess whether WT dynamics would be better regulated on fine-textured sites pads as a result of their greater water retention capacity and lower K_v than sandy fill materials (Chapter 2). This could reduce the magnitude of drainage to the underlying compacted peat, thus potentially changing the relative importance of all water

balance components (Chapter 5). Furthermore, finer-textured residual well pads might restrict horizontal hydrological connectivity with adjacent peatlands to an even greater extent than the mineral substrate examined here (Chapter 2). Thus, additional assessment of horizontal groundwater fluxes on fine-textured pads will be essential to understand hydrological function on those sites. There is also a clear need to assess what impact finer-textured mineral substrates might have on water availability to establishing mosses (Chapter 3) and their impacts on nutrient cycling and availability, including the effects of fertilization and application of straw mulch surface coverings (Chapter 4). Finally, finer-textured residual well pads will likely sustain different rates of evapotranspiration than coarse-textured pads like that observed herein, and as such detailed assessment of atmospheric water losses from fine-textured residual well pads is recommended (Chapter 5).

6.3.4. Trial of Partial Removal on Well Pads Underlain by Geotextile Liners

Vertical groundwater fluxes were an especially important hydrological flux on the residual pad examined herein, which lacked a geotextile liner (Chapter 5). As such, the presence of impermeable or semi-impermeable geotextile liners between residual mineral fill materials and the compacted peat beneath them could have a considerable effect on the overall water balance of residual well pads (Chapters 2 & 5). Thus, future trials should seek to apply the Partial Removal Technique on well pads underlain by geotextile liners of varying permeability. These studies should specifically assess the relative magnitudes of horizontal and vertical groundwater fluxes into the residual mineral fill left at the surface, and draw comparisons against the water balance of residual pads not underlain by these materials.

6.3.5. Additional Assessment of Mulch Surface Coverings

Further assessment of the effects of mulch coverings applied onto the surface of residual well pads should also be undertaken. Specifically, the assessment of the effects of straw mulch on rates of evapotranspiration

presented herein was limited in its temporal scope, being based solely on physical lysimeter measurements representative of only one week per month, and conducted on the scale of approximately 24 hours per measurement (Chapter 5). An assessment with greater temporal resolution could be undertaken by implementing an energy-balance approach to the quantification of evapotranspiration rates in both mulched and non-mulched plots, allowing for direct comparison of atmospheric losses between the two. Such an approach would be similar to that applied in previous assessments conducted in both mulched and non-mulched plots in a constructed fen in the Athabasca Oil Sands region (Scarlett et al., 2017) and on cutover peatlands undergoing restoration (Price et al., 1998). However, this would be resource-intensive, requiring multiple micrometeorological stations to quantify the energy balance in mulched and non-mulched plots in both WO and NO areas on a residual pad. Additional study of the effect of straw mulch surface coverings on soil moisture and rates of evapotranspiration should also be undertaken on fine-textured residual well pads (Chapters 3 & 5), given the high likelihood that those substrates would sustain greater rates of evaporation under dry conditions than the loamy sand substrate examined herein.

6.3.6. Assessment of Microbial Dynamics within Residual Mineral Substrates

Given the uncertainty surrounding the controls on rates of organic matter decomposition and stabilization on the residual pad examined herein (Chapter 4), there is a clear need to conduct a detailed assessment of substrate microbial community dynamics following the partial removal of well pads. Similar assessments have been conducted in restored cutover peatlands (Andersen et al., 2013, 2006) and in road-bisected peatlands (Saraswati et al., 2019), which has improved our understanding of nutrient cycling processes and greenhouse gas dynamics in those sites. A detailed understanding of microbial dynamics on residual well pads would greatly improve our understanding of the controls and drivers of early biogeochemical function (Chapter 4), and thus also of the biological processes supporting nutrient availability to establishing peatland vegetation.

6.3.7. Assessment of Fertilization in Combination with Soil Amendments

Owing to rapid geochemical interactions with the mineral fill, the application of rock phosphate fertilizer was observed to have limited biogeochemical effects on the residual well pad examined in this study (Chapter 4). However, if organic soil amendments were to be added to residual well pads in future trials, this might reduce the extent to which geochemical interactions would make phosphorus non-bioavailable. This might make fertilization a more effective restoration treatment (Chapter 4), and additional study of rock phosphate application in combination with organic matter amendments is therefore recommended.

6.3.8. Development of Water Availability Thresholds for Common True Mosses

Given the sensitivity of peatland mosses to changes in substrate water content (Graf and Rochefort, 2010; Ketcheson and Price, 2014; Price and Whitehead, 2001; Vitt et al., 2014) and the high degree of spatial and temporal variability in moisture dynamics across the residual pad examined here (Chapter 3), there is a clear need to better understand the desiccation tolerances of peatland true mosses. This would complement existing thresholds used to predict the optimal range of moisture levels for true moss survival and growth (Graf and Rochefort, 2010) by allowing for improved prediction of the lower range of substrate water contents suitable for true moss colonization and establishment (see Guérin, 2022). This would also reduce reliance upon desiccation thresholds developed specifically for *Sphagnum* mosses, like the one cautiously used in support of the interpretations herein (Price and Whitehead, 2001) (Chapter 3). Such an assessment of true moss desiccation tolerances should ideally include a combination of field (e.g., Price and Whitehead, 2001) and greenhouse experiments (e.g., Guérin, 2022) to control confounding variables while also ensuring that the results are field-representative (e.g., Graf and Rochefort, 2010).

6.4. General Conclusion

The construction of tens of thousands of oil and gas well pads in boreal peatlands to date presents a landscape-scale environmental management challenge. Furthermore, techniques to reestablish peatland ecosystem function and self-sustaining peatland vegetation communities on these features are still in their infancy. Thus, the Partial Removal Technique has received considerable interest from researchers, regulators, and practitioners alike. While the residual mineral substrate left following the partial removal of a well pad in support of the present study did impede hydrological connectivity with the surrounding peatland, the well-regulated and near-surface water tables observed in areas adjacent to the fen suggest that the technique has promise. Nonetheless, additional research is required to confirm the generalizability of the findings presented herein, and to refine the technique itself and its application at the site level. Combined, while the Partial Removal Technique is likely to be vulnerable to divergent successional trajectories, it does appear that refinements may make it a viable strategy to better meet equivalent land capability at a large scale.

7. REFERENCES

- Adams, W.P., Barr, D.R., 1974. Techniques and equipment for measurement of snow cover including stratigraphy, in: *Measurement in Physical Geography: Laboratory and Field Equipment Built at Trent University* (Trent University Department of Geography Occasional Paper 3). pp. 11–26.
- Akhtar, K., Wang, W., Ren, G., Khan, A., Feng, Y., Yang, G., Wang, H., 2019. Integrated use of straw mulch with nitrogen fertilizer improves soil functionality and soybean production. *Environ Int* 132. <https://doi.org/10.1016/j.envint.2019.105092>
- Alberta Energy Regulator, 2024. Well summary report for 00/02-13-073-04W5/0. Calgary, AB.
- Alberta Environment, 2015. Alberta wetland classification system. Alberta Environment and Sustainable Resource Development, Edmonton, AB.
- Alberta Environment and Parks, 2017. Reclamation criteria for wellsites and associated facilities for peatlands. Edmonton, AB.
- Allen, R.G., Pereira, L.S., Raes, D., Smith, M., 1998. Crop evapotranspiration: Guidelines for computing crop water requirements (FAO irrigation and drainage paper 56).
- Amon, J.P., Jacobson, C.S., Shelley, M.L., 2005. Construction of fens with and without hydric soils. *Ecol Eng* 24, 341–357. <https://doi.org/10.1016/j.ecoleng.2004.11.011>
- Andersen, R., Francez, A.J., Rochefort, L., 2006. The physicochemical and microbiological status of a restored bog in Québec: Identification of relevant criteria to monitor success. *Soil Biol Biochem* 38, 1375–1387. <https://doi.org/10.1016/j.soilbio.2005.10.012>
- Andersen, R., Wells, C., Macrae, M., Price, J., 2013. Nutrient mineralisation and microbial functional diversity in a restored bog approach natural conditions 10 years post restoration. *Soil Biol Biochem* 64, 37–47. <https://doi.org/10.1016/j.soilbio.2013.04.004>
- Anderson, R., Vasander, H., Geddes, N., Laine, A., Tolvanen, A., O’Sullivan, A., Aapala, K., 2016. Afforested and forestry-drained peatland restoration. *Peatland Restoration and Ecosystem Services: Science, Policy and Practice* 213–233. <https://doi.org/10.1017/CBO9781139177788.013>
- Antweiler, R.C., 2015. Evaluation of Statistical Treatments of Left-Censored Environmental Data Using Coincident Uncensored Data Sets. II. Group Comparisons. *Environ Sci Technol* 49, 13439–13446. <https://doi.org/10.1021/acs.est.5b02385>
- Audet, P., Pinno, B.D., Thiffault, E., 2015. Reclamation of boreal forest after oil sands mining: Anticipating novel challenges in novel environments. *Canadian Journal of Forest Research* 45, 364–371. <https://doi.org/10.1139/cjfr-2014-0330>
- Barrow, N.J., Debnath, A., Sen, A., 2022. Effect of phosphate sorption on soil pH. *Eur J Soil Sci* 73. <https://doi.org/10.1111/ejss.13172>
- Bates, D., Mächler, M., Bolker, B.M., Walker, S.C., 2015. Fitting linear mixed-effects models using lme4. *J Stat Softw* 67. <https://doi.org/10.18637/jss.v067.i01>
- Bauer, I.E., Gignac, L.D., Vitt, D.H., 2003. Development of a peatland complex in boreal western Canada: Lateral site expansion and local variability in vegetation succession and long-term peat accumulation. *Canadian Journal of Botany* 81, 833–847. <https://doi.org/10.1139/b03-076>

- Bayley, S.E., Thormann, M.N., Szumigalski, A.R., 2005. Nitrogen mineralization and decomposition in western boreal bog and fen peat 12, 455–465.
- Beckingham, J.D., Archibald, J.H., 1996. Field Guide to Ecosites of Northern Alberta. Canadian Forest Service (Northern Forestry Centre), Edmonton, AB.
- Bélisle, A.C., Asselin, H., 2021. A collaborative typology of boreal indigenous landscapes. *Canadian Journal of Forest Research* 51, 1253–1262. <https://doi.org/10.1139/cjfr-2020-0369>
- Biagi, K.M., Carey, S.K., 2020. The role of snow processes and hillslopes on runoff generation in present and future climates in a recently constructed watershed in the Athabasca oil sands region. *Hydrol Process* 34, 3635–3655. <https://doi.org/10.1002/hyp.13836>
- Bird, M., Xu, B., 2021a. Donor Moss Transfer: How and When to Use in Peatland Restoration. Peace River, AB.
- Bird, M., Xu, B., 2021b. Initiating wetland and peat formation on residual mineral substrates (Technical Note #43). Peace River, AB.
- Bird, M., Xu, B., 2021c. Hydrology and Microtopography: Importance for Wetland Reclamation (Technical Note #44). Peace River, AB.
- Bird, M., Xu, B., 2020. Peatland restoration by burying: An effective option (Technical Note #37). Peace River, AB.
- Blake, G.R., Hartge, K.H., 1986. Particle density, in: Klute, A. (Ed.), *Methods of Soil Analysis, Part 1: Physical and Mineralogical Methods*. Soil Science Society of America, Madison, WI, pp. 377–382. <https://doi.org/10.2136/sssabookser5.1.2ed>
- Bocking, E., Cooper, D.J., Price, J., 2017. Using tree ring analysis to determine impacts of a road on a boreal peatland. *For Ecol Manage* 404, 24–30. <https://doi.org/10.1016/j.foreco.2017.08.007>
- Boelter, D.H., 1965. Hydraulic conductivity of peats. *Soil Sci*. <https://doi.org/10.1097/00010694-196510000-00001>
- Borkenhagen, A., Cooper, D.J., 2018. Tolerance of fen mosses to submergence, and the influence on moss community composition and ecosystem resilience. *Journal of Vegetation Science* 29, 127–135. <https://doi.org/10.1111/jvs.12610>
- Borkenhagen, A., Cooper, D.J., 2016. Creating fen initiation conditions: A new approach for peatland reclamation in the oil sands region of Alberta. *Journal of Applied Ecology* 53, 550–558. <https://doi.org/10.1111/1365-2664.12555>
- Borkenhagen, A., Cooper, D.J., House, M., Vitt, D.H., 2024. Establishing peat-forming plant communities: A comparison of wetland reclamation methods in Alberta’s oil sands region. *Ecological Applications* 34. <https://doi.org/10.1002/eap.2929>
- Bouchard, B., Nadeau, D.F., Domine, F., 2022. Comparison of snowpack structure in gaps and under the canopy in a humid boreal forest. *Hydrol Process* 36. <https://doi.org/10.1002/hyp.14681>
- Bouyoucos, G.J., 1962. Hydrometer Method Improved for Making Particle Size Analyses of Soils. *Agron J* 54, 464–465. <https://doi.org/10.2134/agronj1962.00021962005400050028x>

- Brandt, J.P., Flannigan, M.D., Maynard, D.G., Thompson, I.D., Volney, W.J.A., 2013. An introduction to Canada's boreal zone: Ecosystem processes, health, sustainability, and environmental issues1. *Environmental Reviews* 21, 207–226. <https://doi.org/10.1139/er-2013-0040>
- Brawner, C.O., Tessier, G., 1969. Road and Railway Construction, in: MacFarlane, I.C. (Ed.), *Muskeg Engineering Handbook*. University of Toronto Press, Toronto, pp. 150–208.
- Bridgman, S.D., Updegraff, K., Pastor, J., 1998. Carbon, nitrogen, and phosphorus mineralization in northern wetlands. *Ecology* 79, 1545–1561. [https://doi.org/10.1890/0012-9658\(1998\)079\[1545:CNAPMI\]2.0.CO;2](https://doi.org/10.1890/0012-9658(1998)079[1545:CNAPMI]2.0.CO;2)
- Brown, S.M., Petrone, R.M., Mendoza, C., Devito, K.J., 2010. Surface vegetation controls on evapotranspiration from a sub-humid Western Boreal Plain wetland. *Hydrol Process* 24, 1072–1085. <https://doi.org/10.1002/hyp.7569>
- Busby, J.R., Whitfield, D.W.A., 1978. Water potential, water content, and net assimilation of some boreal forest mosses. *Canadian Journal of Botany* 56, 1551–1558. <https://doi.org/10.1139/b78-184>
- Chapin, F.S., Chapin, M.C., 1981. Ecotypic Differentiation of Growth Processes in *Carex Aquatilis* along Latitudinal and Local Gradients. *Ecology* 62, 1000–1009.
- Chapin, F.S., Matson, P.A., Vitousek, P.M., 2011. *Principles of Terrestrial Ecosystem Ecology*. Springer.
- Chee, W.L., Vitt, D.H., 1989. The vegetation, surface water chemistry and peat chemistry of moderate-rich fens in central Alberta, Canada. *Wetlands* 9, 227–261. <https://doi.org/10.1007/BF03160747>
- Chimner, R.A., Cooper, D.J., Wurster, F.C., Rochefort, L., 2017. An overview of peatland restoration in North America: where are we after 25 years? *Restor Ecol* 25, 283–292. <https://doi.org/10.1111/rec.12434>
- Chimner, R.A., Lemly, J.M., Cooper, D.J., 2010. Mountain fen distribution, types and restoration priorities, San Juan Mountains, Colorado, USA. *Wetlands* 30, 763–771. <https://doi.org/10.1007/s13157-010-0039-5>
- Cleveland, C.C., Townsend, A.R., Schmidt, S.K., 2002. Phosphorus limitation of microbial processes in moist tropical forests: Evidence from short-term laboratory incubations and field studies. *Ecosystems* 5, 680–691. <https://doi.org/10.1007/s10021-002-0202-9>
- Craine, J.M., Dybzinski, R., 2013. Mechanisms of plant competition for nutrients, water and light. *Funct Ecol* 27, 833–840. <https://doi.org/10.1111/1365-2435.12081>
- D'Astous, A., Poulin, M., Aubin, I., Rochefort, L., 2013. Using functional diversity as an indicator of restoration success of a cut-over bog. *Ecol Eng* 61, 519–526. <https://doi.org/10.1016/j.ecoleng.2012.09.002>
- Delta-T Devices, 2017. User Manual for the ML3 ThetaProbe. Burwell, UK.
- Devito, K., Mendoza, C., Qualizza, C., 2012. Conceptualizing water movement in the Boreal Plains: Implications for watershed reconstruction (synthesis report prepared for the Canadian Oil Sands Network for Research and Development).

- Devito, K.J., Creed, I.F., Fraser, C.J.D., 2005. Controls on runoff from a partially harvested aspen-forested headwater catchment, Boreal Plain, Canada. *Hydrol Process* 19, 3–25. <https://doi.org/10.1002/hyp.5776>
- Devito, K.J., Dillon, P.J., 1993. The influence of hydrologic conditions and peat oxia on the phosphorus and nitrogen dynamics of a conifer swamp. *Water Resour Res* 29, 2675–2685. <https://doi.org/10.1029/93WR00622>
- Devito, K.J., Waddington, J.M., Branfireun, B.A., 1997. Flow reversals in peatlands influenced by local groundwater systems. *Hydrol Process* 11, 103–110. [https://doi.org/10.1002/\(sici\)1099-1085\(199701\)11:1<103::aid-hyp417>3.0.co;2-e](https://doi.org/10.1002/(sici)1099-1085(199701)11:1<103::aid-hyp417>3.0.co;2-e)
- Dingman, S.L., 2015a. Evapotranspiration, in: *Physical Hydrology*. Waveland Press, Long Grove, USA, pp. 253–310.
- Dingman, S.L., 2015b. Snow and Snowmelt, in: *Physical Hydrology*. Waveland Press, Long Grove, Illinois, pp. 203–252.
- Domenico, P.A., Schwartz, F.W., 1998. Ground Water Movement, in: *Physical and Chemical Hydrogeology*. John Wiley & Sons, Inc., pp. 33–44. <https://doi.org/10.1002/j.1551-8833.1968.tb03545.x>
- Drever, C.R., Cook-Patton, S.C., Akhter, F., Badiou, P.H., Chmura, G.L., Davidson, S.J., Desjardins, R.L., Dyk, A., Fargione, J.E., Fellows, M., Filewod, B., Hessing-Lewis, M., Jayasundara, S., Keeton, W.S., Kroeger, T., Lark, T.J., Le, E., Leavitt, S.M., LeClerc, M.E., Lemprière, T.C., Metsaranta, J., McConkey, B., Neilson, E., St-Laurent, G.P., Puric-Mladenovic, D., Rodrigue, S., Soolanayakanahally, R.Y., Spawn, S.A., Strack, M., Smyth, C., Thevathasan, N., Voicu, M., Williams, C.A., Woodbury, P.B., Worth, D.E., Xu, Z., Yeo, S., Kurz, W.A., 2021. Natural climate solutions for Canada. *Sci Adv* 7. <https://doi.org/10.1126/sciadv.abd6034>
- Drozdowski, B., Powter, C., Tokay, H., Mackenzie, D., Renkema, K., Xu, B., 2020. Certification of mineral soil pads in the boreal region – decision framework and support tools.
- Ducks Unlimited Canada, 2014. Operational Guide: Forest Road Wetland Crossings.
- Dunne, E.J., Reddy, K.R., 2005. Phosphorus biogeochemistry of wetlands in agricultural watersheds, in: Dunne, E.J., Reddy, K.R., Carton, O.T. (Eds.), *Nutrient Management in Agricultural Watershed: A Wetlands Solution*. Wageningen Academic, Wageningen, Netherlands, pp. 105–119. <https://doi.org/10.3920/978-90-8686-558-1>
- Durner, W., 1994. Hydraulic conductivity estimation for soils with heterogeneous pore structure. *Water Resour Res* 30, 211–223. <https://doi.org/10.1029/93WR02676>
- Durner, W., Or, D., 2005. 73: Soil Water Potential Measurement, in: Anderson, M.G. (Ed.), *Encyclopedia of Hydrological Sciences*. Wiley.
- Elmes, M.C., Davidson, S.J., Price, J.S., 2021a. Ecohydrological interactions in a boreal fen–swamp complex, Alberta, Canada. *Ecohydrology* 14. <https://doi.org/10.1002/eco.2335>
- Elmes, M.C., Kessel, E., Wells, C.M., Sutherland, G., Price, J.S., Macrae, M.L., Petrone, R.M., 2021b. Evaluating the hydrological response of a boreal fen following the removal of a temporary access road. *J Hydrol (Amst)* 594. <https://doi.org/10.1016/j.jhydrol.2020.125928>

- Elmes, M.C., Petrone, R.M., Volik, O., Price, J.S., 2022. Changes to the hydrology of a boreal fen following the placement of an access road and below ground pipeline. *J Hydrol Reg Stud* 40. <https://doi.org/10.1016/j.ejrh.2022.101031>
- Elmes, M.C., Price, J.S., 2019. Hydrologic function of a moderate-rich fen watershed in the Athabasca Oil Sands Region of the Western Boreal Plain, northern Alberta. *J Hydrol (Amst)* 570, 692–704. <https://doi.org/10.1016/j.jhydrol.2018.12.043>
- Engering, A., Davidson, S.J., Xu, B., Bird, M., Rochefort, L., Strack, M., 2021. Restoration of a boreal peatland impacted by an in-situ oil sands well-pad 2: Greenhouse gas exchange dynamics. *Restor Ecol*. <https://doi.org/10.1111/rec.13508>
- Eno, C.F., 1960. Nitrate production in the field by incubating the soil in polyethylene bags. *Soil Science Society of America Journal* 24, 277–279.
- Environment and Climate Change Canada, 2024a. Canadian climate normals 1981–2010.
- Environment and Climate Change Canada, 2024b. 1991-2020 Climate Normals & Averages [WWW Document]. URL https://climate.weather.gc.ca/climate_normals/index_e.html (accessed 1.14.24).
- Environment and Climate Change Canada, 2024c. Historical Data [WWW Document]. URL https://climate.weather.gc.ca/historical_data/search_historic_data_e.html
- Fan, Y., Gao, J., Sun, J., Liu, J., Su, Z., Wang, Z., Yu, X., Hu, S., 2021. Effects of straw returning and potassium fertilizer application on root characteristics and yield of spring maize in China inner Mongolia. *Agron J* 113, 4369–4385. <https://doi.org/10.1002/agj2.20742>
- Farnden, C., 2023. Results from a pilot ecosystem mapping project on Syncrude’s reclaimed oil sands mine sites, in: *British Columbia Mine Reclamation Symposium*.
- Ferone, J.M., Devito, K.J., 2004. Shallow groundwater-surface water interactions in pond-peatland complexes along a Boreal Plains topographic gradient. *J Hydrol (Amst)* 292, 75–95. <https://doi.org/10.1016/j.jhydrol.2003.12.032>
- Ficken, C.D., Cobbaert, D., Rooney, R.C., 2019. Low extent but high impact of human land use on wetland flora across the boreal oil sands region. *Science of the Total Environment* 693, 133647. <https://doi.org/10.1016/j.scitotenv.2019.133647>
- Fisk, M., Santangelo, S., Minick, K., 2015. Carbon mineralization is promoted by phosphorus and reduced by nitrogen addition in the organic horizon of northern hardwood forests. *Soil Biol Biochem* 81, 212–218. <https://doi.org/10.1016/j.soilbio.2014.11.022>
- Fox, J., Weisberg, S., 2019. *An R Companion to Applied Regression*, 3rd ed. Sage, Thousand Oaks, CA.
- Fraser, C.J.D., Roulet, N.T., Lafleur, M., 2001. Groundwater flow patterns in a large peatland. *J Hydrol (Amst)* 246, 142–154.
- Freeze, R.A., Cherry, J.A., 1979. *Groundwater*. Prentice-Hall, Englewood Cliffs, NJ.
- Gauthier, M.E., Rochefort, L., Nadeau, L., Hugron, S., Xu, B., 2018. Testing the moss layer transfer technique on mineral well pads constructed in peatlands. *Wetl Ecol Manag* 26, 475–487. <https://doi.org/10.1007/s11273-017-9532-4>

- Gauthier, T.L.J., McCarter, C.P.R., Price, J.S., 2018. The effect of compression on Sphagnum hydrophysical properties: Implications for increasing hydrological connectivity in restored cutover peatlands. *Ecohydrology* 11. <https://doi.org/10.1002/eco.2020>
- Ge, X., Wang, L., Zhang, W., Putnis, C. V., 2020. Molecular Understanding of Humic Acid-Limited Phosphate Precipitation and Transformation. *Environ Sci Technol* 54, 207–215. <https://doi.org/10.1021/acs.est.9b05145>
- Gerwing, T.G., Hawkes, V.C., Gann, G.D., Murphy, S.D., 2022. Restoration, reclamation, and rehabilitation: on the need for, and positing a definition of, ecological reclamation. *Restor Ecol* 30. <https://doi.org/10.1111/rec.13461>
- Gingras-Hill, T., Nwaishi, F.C., Macrae, M.L., Price, J.S., Petrone, R.M., 2018. Ecohydrological functioning of an upland undergoing reclamation on post-mining landscape of the Athabasca oil sands region, Canada. *Ecohydrology* 11, 1–12. <https://doi.org/10.1002/eco.1941>
- Glaser, P.H., Siegel, D.I., Romanowicz, E.A., Shen, Y.P., 1997. Regional linkages between raised bogs and the climate, groundwater, and landscape of north-western Minnesota. *Journal of Ecology* 85, 3–16.
- Goetz, J.D., Price, J.S., 2015. Role of morphological structure and layering of Sphagnum and Tomenthypnum mosses on moss productivity and evaporation rates. *Can J Soil Sci* 95, 109–124. <https://doi.org/10.4141/CJSS-2014-092>
- Golding, D.L., Swanson, R.H., 1978. Snow accumulation and melt in small forest openings in Alberta. *Canadian Journal of Forest Research* 8. <https://doi.org/10.1139/x78-057>
- González, E., Rochefort, L., 2014. Drivers of success in 53 cutover bogs restored by a moss layer transfer technique. *Ecol Eng* 68, 279–290. <https://doi.org/10.1016/j.ecoleng.2014.03.051>
- Graf, M.D., Rochefort, L., 2010. Moss regeneration for fen restoration: Field and greenhouse experiments. *Restor Ecol* 18, 121–130. <https://doi.org/10.1111/j.1526-100X.2008.00437.x>
- Green, J.E., Van Egmond, T.D., Wylie, C., Jones, I., Knapik, L., Paterson, L.R., 1992. A User Guide to Pit & Quarry Reclamation in Alberta. Calgary, Alberta.
- Groeneveld, E.V.G., Massé, A., Rochefort, L., 2007. Polytrichum strictum as a nurse-plant in peatland restoration. *Restor Ecol* 15, 709–719. <https://doi.org/10.1111/j.1526-100X.2007.00283.x>
- Guérin, P., 2022. Restauration d’une communauté végétale avec potentiel d’accumulation de tourbe sur substrat minéral d’une tourbière perturbée par une route. Université Laval, Quebec, Canada.
- Haapalehto, T., Kotiaho, J.S., Matilainen, R., Tahvanainen, T., 2014. The effects of long-term drainage and subsequent restoration on water table level and pore water chemistry in boreal peatlands. *J Hydrol (Amst)* 519, 1493–1505. <https://doi.org/10.1016/j.jhydrol.2014.09.013>
- Hájek, T., Vicherová, E., 2014. Desiccation tolerance of Sphagnum revisited: A puzzle resolved. *Plant Biol* 16, 765–773. <https://doi.org/10.1111/plb.12126>
- Halliwell, D.H., Rouse, W.R., 1987. Soil heat flux in permafrost: Characteristics and accuracy of measurement. *Journal of Climatology* 7, 571–584. <https://doi.org/10.1002/joc.3370070605>

- Halsey, L.A., Vitt, D.H., Bauer, I.E., 1998. Peatland initiation during the Holocene in continental western Canada. *Clim Change* 40, 315–342.
- Harris, L.I., Richardson, K., Bona, K.A., Davidson, S.J., Finkelstein, S.A., Garneau, M., McLaughlin, J., Nwaishi, F., Olefeldt, D., Packalen, M., Roulet, N.T., Southee, F.M., Strack, M., Webster, K.L., Wilkinson, S.L., Ray, J.C., 2021. The essential carbon service provided by northern peatlands. *Front Ecol Environ* 1–9. <https://doi.org/10.1002/fee.2437>
- Hartsock, J.A., Bremer, E., Vitt, D.H., 2020. Nutrient supply patterns across the Sandhill Fen reclamation watershed and regional reference fens in Alberta, Canada: An ion exchange membrane study. *Ecohydrology* 13, 1–15. <https://doi.org/10.1002/eco.2188>
- Hartsock, J.A., House, M., Vitt, D.H., 2016. Net nitrogen mineralization in boreal fens: A potential performance indicator for peatland reclamation. *Botany* 94, 1027–1040. <https://doi.org/10.1139/cjb-2015-0263>
- Hawkes, V.C., Hornung, J., Novoa, J., Miller, M.T., Kohlsmith, S., 2025. Wetlands happen: the delineation and classification of opportunistic wetlands in the Athabasca oil sands region of Canada. *Wetl Ecol Manag* 33, 19. <https://doi.org/10.1007/s11273-025-10034-7>
- Hawkes, V.C., Miller, M.T., Novoa, J., Ibeke, E., Martin, J.P., 2020. Opportunistic wetland formation, characterization, and quantification on landforms reclaimed to upland ecosites in the Athabasca Oil Sands Region. *Wetl Ecol Manag* 28, 953–970. <https://doi.org/10.1007/s11273-020-09760-x>
- Healy, R.W., Winter, T.C., LaBaugh, J.W., Franke, O.L., 2007. Water Budgets: Foundations for Effective Water-Resources and Environmental Management (U. S. Geological Survey Circular 1308).
- Heiri, O., Lotter, A.F., Lemcke, G., 2001. Loss on ignition as a method for estimating organic and carbonate content in sediments: Reproducibility and comparability of results. *J Paleolimnol* 25, 101–110. <https://doi.org/10.1023/A:1008119611481>
- Helbig, M., Waddington, J.M., Alekseychik, P., Amiro, B.D., Aurela, M., Barr, A.G., Black, T.A., Blanken, P.D., Carey, S.K., Chen, J., Chi, J., Desai, A.R., Dunn, A., Euskirchen, E.S., Flanagan, L.B., Forbrich, I., Friberg, T., Grelle, A., Harder, S., Heliasz, M., Humphreys, E.R., Ikawa, H., Isabelle, P.E., Iwata, H., Jassal, R., Korkiakoski, M., Kurbatova, J., Kutzbach, L., Lindroth, A., Löfvenius, M.O., Lohila, A., Mammarella, I., Marsh, P., Maximov, T., Melton, J.R., Moore, P.A., Nadeau, D.F., Nicholls, E.M., Nilsson, M.B., Ohta, T., Peichl, M., Petrone, R.M., Petrov, R., Prokushkin, A., Quinton, W.L., Reed, D.E., Roulet, N.T., Runkle, B.R.K., Sonnentag, O., Strachan, I.B., Taillardat, P., Tuittila, E.S., Tuovinen, J.P., Turner, J., Ueyama, M., Varlagin, A., Wilmking, M., Wofsy, S.C., Zyryanov, V., 2020. Increasing contribution of peatlands to boreal evapotranspiration in a warming climate. *Nat Clim Chang* 10, 555–560. <https://doi.org/10.1038/s41558-020-0763-7>
- Helsel, D., 2012. Statistics for censored environmental data using Minitab and R. Wiley.
- Helsel, D., 2005. More than obvious: Better methods for interpreting nondetect data. *Environ Sci Technol* 39, 419A–423A. <https://doi.org/10.1021/es053368a>
- Hemstock, S.S., Quideau, S.A., Chanasyk, D.S., 2010. Nitrogen availability from peat amendments used in boreal oil sands reclamation. *Can J Soil Sci* 90, 165–175. <https://doi.org/10.4141/CJSS09021>
- Hendershot, W.H., Lalande, H., Duquette, M., 1993. Ion Exchange and Exchangeable Cations, in: Carter, M.R. (Ed.), *Soil Sampling and Methods of Analysis*. Lewis Publishers, Boca Raton, pp. 167–176.

- Hill, D., Simpson-Marran, M., Gould, L., Nason, S., 2021. Status of Boreal Woodland Caribou Conservation in Canada A summary of range planning, restoration, and opportunities to win on caribou and climate. Calgary, Alberta.
- Hokanson, K.J., Mendoza, C.A., Devito, K.J., 2019. Interactions Between Regional Climate, Surficial Geology, and Topography: Characterizing Shallow Groundwater Systems in Subhumid, Low-Relief Landscapes. *Water Resour Res* 55, 284–297. <https://doi.org/10.1029/2018WR023934>
- Howie, S.A., Hebda, R.J., 2018. Bog surface oscillation (mire breathing): A useful measure in raised bog restoration. *Hydrol Process* 32, 1518–1530. <https://doi.org/10.1002/hyp.11622>
- Hugron, S., Poulin, M., Rochefort, L., 2013. Organic matter amendment enhances establishment of reintroduced bryophytes and lichens in borrow pits located in boreal forest highlands. *Boreal Environment Research* 18, 317–328.
- Hvorslev, M.J., 1951. Time lag and soil permeability in ground-water observations. Vicksburg, MS.
- Julian, P., Helsel, D., 2024. NADA2: Data analysis for censored environmental data.
- Jury, W.A., Horton, R., 2004a. Water Retention in Soil, in: *Soil Physics*. Wiley, pp. 37–73.
- Jury, W.A., Horton, R., 2004b. Soil Thermal Regime, in: *Soil Physics*. Wiley, pp. 161–200.
- Ketcheson, S.J., Price, J.S., 2016. Snow hydrology of a constructed watershed in the Athabasca oil sands region, Alberta, Canada. *Hydrol Process* 30, 2546–2561. <https://doi.org/10.1002/hyp.10813>
- Ketcheson, S.J., Price, J.S., 2014. Characterization of the fluxes and stores of water within newly formed Sphagnum moss cushions and their environment. *Ecohydrology* 7, 771–782. <https://doi.org/10.1002/eco.1399>
- Ketcheson, S.J., Price, J.S., Sutton, O., Sutherland, G., Kessel, E., Petrone, R.M., 2017. The hydrological functioning of a constructed fen wetland watershed. *Science of the Total Environment* 603–604, 593–605. <https://doi.org/10.1016/j.scitotenv.2017.06.101>
- Keuskamp, J.A., Dingemans, B.J.J., Lehtinen, T., Sarneel, J.M., Hefting, M.M., 2013. Tea Bag Index: A novel approach to collect uniform decomposition data across ecosystems. *Methods Ecol Evol* 4, 1070–1075. <https://doi.org/10.1111/2041-210X.12097>
- Kimmel, K., Mander, Ü., 2010. Ecosystem services of peatlands: Implications for restoration. *Prog Phys Geogr* 34, 491–514. <https://doi.org/10.1177/0309133310365595>
- Kinar, N.J., Pomeroy, J.W., 2015. Measurement of the physical properties of the snowpack. *Reviews of Geophysics* 53, 481–544. <https://doi.org/10.1002/2015RG000481>
- Koropchak, S., Vitt, D.H., Bloise, R., Wieder, R.K., 2012. Fundamental paradigms, foundation species selection, and early plant responses to peatland initiation on mineral soils, in: Vitt, D., Bhatti, J. (Eds.), *Restoration and Reclamation of Boreal Ecosystems: Attaining Sustainable Development*. Cambridge University Press, Cambridge.
- Kuhry, P., Halsey, L.A., Bayley, S.E., Vitt, D.H., 1992. Peatland development in relation to Holocene climatic change in Manitoba and Saskatchewan (Canada), *J. Earth Sci.*
- Kuhry, P., Turunen, J., 2006. The Postglacial Development of Boreal and Subarctic Peatlands, in: Wieder, R.K., Vitt, D.H. (Eds.), *Boreal Peatland Ecosystems*. Springer, pp. 25–46.

- Kuznetsova, A., Brockhoff, P.B., Christensen, R.H.B., 2017. lmerTest package: Tests in linear mixed effects models. *J Stat Softw* 82, 1–26. <https://doi.org/10.18637/JSS.V082.I13>
- Lamers, L.P.M., Vile, M.A., Grootjans, A.P., Acreman, M.C., van Diggelen, R., Evans, M.G., Richardson, C.J., Rochefort, L., Kooijman, A.M., Roelofs, J.G.M., Smolders, A.J.P., 2015. Ecological restoration of rich fens in Europe and North America: from trial and error to an evidence-based approach. *Biol Rev Camb Philos Soc* 90, 182–203. <https://doi.org/10.1111/brv.12102>
- Lee, L., 2020. NADA: Nondetects and Data Analysis for Environmental Data.
- Lehmann, P., Merlin, O., Gentine, P., Or, D., 2018. Soil texture effects on surface resistance to bare-soil evaporation. *Geophys Res Lett* 45, 10398–10405. <https://doi.org/10.1029/2018GL078803>
- Lemmer, M., 2022. The aftermath of novel peatland restoration following in situ oil and gas infrastructure disturbances. Université Laval, Québec.
- Lemmer, M., Rochefort, L., Strack, M., 2020. Greenhouse Gas Emissions Dynamics in Restored Fens After In-Situ Oil Sands Well Pad Disturbances of Canadian Boreal Peatlands. *Front Earth Sci (Lausanne)* 8. <https://doi.org/10.3389/feart.2020.557943>
- Lemmer, M., Xu, B., Strack, M., Rochefort, L., 2022. Reestablishment of peatland vegetation following surface leveling of decommissioned in situ oil mining infrastructures. *Restor Ecol.* <https://doi.org/10.1111/rec.13714>
- Lenth, R., 2022. emmeans: Estimated Marginal Means.
- Li, Q., Hu, W., Li, L., Li, Y., 2023. Interactions between organic matter and Fe oxides at soil micro-interfaces: Quantification, associations, and influencing factors. *Science of the Total Environment* 855. <https://doi.org/10.1016/j.scitotenv.2022.158710>
- Little-Devito, M., Mendoza, C.A., Chasmer, L., Kettridge, N., Devito, K.J., 2019. Opportunistic wetland formation on reconstructed landforms in a sub-humid climate: influence of site and landscape-scale factors. *Wetl Ecol Manag* 27, 587–608. <https://doi.org/10.1007/s11273-019-09679-y>
- Lobartini, J.C., Tan, K.H., Pape, C., 1998. Dissolution of aluminum and iron phosphate by humic acids. *Commun Soil Sci Plant Anal* 29, 535–544. <https://doi.org/10.1080/00103629809369965>
- Lucas, R.E., Davis, J.F., 1961. Relationships between pH values of organic soils and availabilities of 12 plant nutrients. *Soil Sci* 92, 177–182. <https://doi.org/10.1097/00010694-196109000-00005>
- Luke, S.G., 2017. Evaluating significance in linear mixed-effects models in R. *Behav Res Methods* 49, 1494–1502. <https://doi.org/10.3758/s13428-016-0809-y>
- Lupardus, R.C., McIntosh, A.C.S., Janz, A., Farr, D., 2019. Succession after reclamation: Identifying and assessing ecological indicators of forest recovery on reclaimed oil and natural gas well pads. *Ecol Indic* 106. <https://doi.org/10.1016/j.ecolind.2019.105515>
- Macyk, T.M., 2006. Tailings sand and natural soil quality at the Syncrude Aurora, Albion Sands, CNRL, and Suncor Mines. Edmonton, AB.
- Manukjanová, A., Štechová, T., Kučera, J., 2014. Drought survival test of eight fen moss species. *Cryptogam Bryol* 35, 397–403. <https://doi.org/10.7872/cryb.v35.iss4.2014.397>

- McCarter, C.P.R., Price, J.S., 2015. The hydrology of the Bois-des-Bel peatland restoration: Hydrophysical properties limiting connectivity between regenerated Sphagnum and remnant vacuum harvested peat deposit. *Ecohydrology* 8, 173–187. <https://doi.org/10.1002/eco.1498>
- McCarter, C.P.R., Price, J.S., 2014. Ecohydrology of Sphagnum moss hummocks: Mechanisms of capitula water supply and simulated effects of evaporation. *Ecohydrology* 7, 33–44. <https://doi.org/10.1002/eco.1313>
- McCarter, C.P.R., Quinton, W.L., Price, J.S., 2023. Groundwater Controls on Peatland Form and Function, in: *Groundwater in Peat and Peatlands. The Groundwater Project*, pp. 2–12.
- McKendrick-Smith, K.A., 2016. *The Impact of Tracks on Blanket Peat Ecohydrology*. University of Leeds.
- McKinnon, M.E., Nwaishi, F.C., Xu, B., Ketcheson, S.J., Bird, M., Petrone, R.M., 2025. Hydrologic assessment of mineral substrate suitability for true moss initiation in a boreal peatland undergoing restoration. *Ecol Eng* 215. <https://doi.org/10.1016/j.ecoleng.2025.107615>
- McKinnon, M.E., Nwaishi, F.C., Xu, B., Ketcheson, S.J., Bird, M., Petrone, R.M., 2024. Hydrological dynamics following partial removal of an oil well pad undergoing restoration to a boreal peatland. *Science of the Total Environment* 957, 177750. <https://doi.org/10.1016/j.scitotenv.2024.177750>
- Meilleur, S., Guêné-Nanchen, M., Hugron, S., Fenton, N.J., Rochefort, L., 2022. Towards the regeneration of brown mosses for fen restoration. *Bryologist* 125, 23–35. <https://doi.org/10.1639/0007-2745-125.1.023>
- Miller, C.A., Benscoter, B.W., Turetsky, M.R., 2015. The effect of long-term drying associated with experimental drainage and road construction on vegetation composition and productivity in boreal fens. *Wetl Ecol Manag* 23, 845–854. <https://doi.org/10.1007/s11273-015-9423-5>
- Miner, K., Baskaran, L., Gay, B., Sousa, D., Miller, C., 2024. Frozen no more, a case study of Arctic permafrost impacts of oil and gas withdrawal. *Sci Rep* 14. <https://doi.org/10.1038/s41598-024-76292-2>
- Moore, P.A., Morris, P.J., Waddington, J.M., 2015. Multi-decadal water table manipulation alters peatland hydraulic structure and moisture retention. *Hydrol Process* 29, 2970–2982. <https://doi.org/10.1002/hyp.10416>
- Mwale, D., Gan, T.Y., Devito, K., Mendoza, C., Silins, U., Petrone, R., 2009. Precipitation variability and its relationship to hydrologic variability in Alberta. *Hydrol Process* 23, 3040–3056. <https://doi.org/10.1002/hyp.7415>
- Nicholson, B.J., Vitt, D.H., 1989. The paleoecology of a peatland complex in continental western Canada. *Canadian Journal of Botany* 68, 121–138. <https://doi.org/10.1139/b90-017>
- Novak, M.D., Chen, W., Orchansky, A.L., Ketler, R., 2000a. Turbulent exchange processes within and above a straw mulch. Part II: Thermal and moisture regimes. *Agric For Meteorol* 102, 155–171. [https://doi.org/10.1016/S0168-1923\(00\)00097-6](https://doi.org/10.1016/S0168-1923(00)00097-6)
- Novak, M.D., Chen, W., Orchansky, A.L., Ketler, R., 2000b. Turbulent exchange processes within and above a straw mulch. Part I: Mean wind speed and turbulent statistics. *Agric For Meteorol* 102.

- Nwaishi, F., Morison, M., Plach, J., Macrae, M., Petrone, R., 2022. Carbon and Nutrient Stoichiometric Relationships in the Soil–Plant Systems of Disturbed Boreal Forest Peatlands within Athabasca Oil Sands Region, Canada. *Forests* 13. <https://doi.org/10.3390/f13060865>
- Nwaishi, F., Petrone, R.M., Macrae, M.L., Price, J.S., Strack, M., Slawson, R., Andersen, R., 2016. Above and below-ground nutrient cycling: A criteria for assessing the biogeochemical functioning of a constructed fen. *Applied Soil Ecology* 98, 177–194. <https://doi.org/10.1016/j.apsoil.2015.10.015>
- Nwaishi, F., Petrone, R.M., Price, J.S., Andersen, R., 2015a. Towards Developing a Functional-Based Approach for Constructed Peatlands Evaluation in the Alberta Oil Sands Region, Canada. *Wetlands* 35, 211–225. <https://doi.org/10.1007/s13157-014-0623-1>
- Nwaishi, F., Petrone, R.M., Price, J.S., Ketcheson, S.J., Slawson, R., Andersen, R., 2015b. Impacts of donor-peat management practices on the functional characteristics of a constructed fen. *Ecol Eng* 81, 471–480. <https://doi.org/10.1016/j.ecoleng.2015.04.038>
- Odum, E.P., 1969. The Strategy of Ecosystem Development. *Science* (1979) 164, 262–270.
- Oke, T.R., 1987. *Boundary Layer Climates*, 2nd ed. Routledge, London, UK.
- O’Kelly, B.C., Pichan, S.P., 2013. Effects of decomposition on the compressibility of fibrous peat - A review. *Geomechanics and Geoengineering* 8, 286–296. <https://doi.org/10.1080/17486025.2013.804210>
- Orbach, R.L., 2012. Energy production: A global perspective. *Developments in Environmental Science* 11, 1–18. <https://doi.org/10.1016/B978-0-08-097760-7.00001-9>
- Osko, T., Gillies, C., Pyper, M., 2018. COSIA in-situ oil sands shared practices for working in and around wetlands. Calgary, AB.
- Parlee, B.L., Geertsema, K., Willier, A., 2012. Social-ecological thresholds in a changing boreal landscape: Insights from Cree knowledge of the Lesser Slave Lake region of Alberta, Canada. *Ecology and Society* 17. <https://doi.org/10.5751/ES-04410-170220>
- Partington, M., Gillies, C., Gingras, B., Smith, C., Morissette, J., 2016. Resource roads and wetlands: A guide for planning, construction and maintenance (special publication SP-530E). FPInnovations, Point-Claire, QC.
- Pasher, J., Seed, E., Duffe, J., 2013. Development of boreal ecosystem anthropogenic disturbance layers for Canada based on 2008 to 2010 Landsat imagery. *Canadian Journal of Remote Sensing* 39, 42–58. <https://doi.org/10.5589/m13-007>
- Peters, A., Durner, W., 2015. SHYPFIT 2.0 User’s Manual. Berlin, Germany.
- Petrone, R.M., Price, J.S., Waddington, J.M., Von Waldow, H., 2004. Surface moisture and energy exchange from a restored peatland, Québec, Canada. *J Hydrol (Amst)* 295, 198–210. <https://doi.org/10.1016/j.jhydrol.2004.03.009>
- Petrone, R.M., Silins, U., Devito, K.J., 2007. Dynamics of evapotranspiration from a riparian pond complex in the Western Boreal forest, Alberta, Canada. *Hydrol Process* 21, 1391–1401. <https://doi.org/10.1002/hyp.6298>

- Pilon, J.K., 2015. Characterization of the physical and hydraulic properties of peat impacted by a temporary access road. University of Waterloo.
- Pinno, B.D., Hawkes, V.C., 2015. Temporal trends of ecosystem development on different site types in reclaimed boreal forests. *Forests* 6, 2109–2124. <https://doi.org/10.3390/f6062109>
- Plach, J.M., Ferone, J.M., Gibbons, Z., Smerdon, B.D., Mertens, A., Mendoza, C.A., Petrone, R.M., Devito, K.J., 2016. Influence of glacial landform hydrology on phosphorus budgets of shallow lakes on the Boreal Plain, Canada. *J Hydrol (Amst)* 535, 191–203. <https://doi.org/10.1016/j.jhydrol.2016.01.041>
- Plach, J.M., Wood, M.E., Macrae, M.L., Osko, T.J., Petrone, R.M., 2017. Effect of a semi-permanent road on N, P, and CO₂ dynamics in a poor fen on the Western Boreal Plain, Canada. *Ecohydrology* 10, 1–15. <https://doi.org/10.1002/eco.1874>
- Pojar, J., 1992. Environment and biogeography of the western boreal forest. *The Forestry Chronicle* 72, 51–58. <https://doi.org/10.5558/tfc72051-1>
- Pomeroy, J.W., Bewley, D.S., Essery, R.L.H., Hedstrom, N.R., Link, T., Granger, R.J., Sicart, J.E., Ellis, C.R., Janowicz, J.R., 2006. Shrub tundra snowmelt. *Hydrol Process* 20, 923–941. <https://doi.org/10.1002/hyp.6124>
- Pomeroy, J.W., Essery, R.L.H., 1999. Turbulent fluxes during blowing snow: Field tests of model sublimation predictions. *Hydrol Process* 13, 2963–2975. [https://doi.org/10.1002/\(SICI\)1099-1085\(19991230\)13:18<2963::AID-HYP11>3.0.CO;2-9](https://doi.org/10.1002/(SICI)1099-1085(19991230)13:18<2963::AID-HYP11>3.0.CO;2-9)
- Pomeroy, J.W., Gray, D.M., 1995. Snowcover: Accumulation, Relocation, and Management (National Hydrology Research Institute Science Report No. 7). National Hydrology Research Institute, Saskatoon, Saskatchewan.
- Popović, N., Petrone, R.M., Green, A., Khomik, M., Price, J.S., 2023. Evolution of ecosystem-scale surface energy fluxes of a newly constructed boreal upland-fen watershed. *Ecol Eng* 194. <https://doi.org/10.1016/j.ecoleng.2023.107059>
- Popović, N., Petrone, R.M., Green, A., Khomik, M., Price, J.S., 2022. A temporal snapshot of ecosystem functionality during the initial stages of reclamation of an upland-fen complex. *J Hydrol Reg Stud* 41. <https://doi.org/10.1016/j.ejrh.2022.101078>
- Pouliot, K., Rochefort, L., LeBlanc, M.C., Guêné-Nanchen, M., Beauchemin, A., 2021. The Burial Under Peat Technique: An Innovative Method to Restore Sphagnum Peatlands Impacted by Mineral Linear Disturbances. *Front Earth Sci (Lausanne)* 9, 1–10. <https://doi.org/10.3389/feart.2021.658470>
- Pouliot, R., Hugron, S., Rochefort, L., Godbout, S., Palacios, J.H., Groeneveld, E., Jarry, I., 2015. Manure derived biochar can successfully replace phosphate rock amendment in peatland restoration. *J Environ Manage* 157, 118–126. <https://doi.org/10.1016/j.jenvman.2015.04.019>
- Powter, C., Chymko, N., Dinwoodie, G., Howat, D., Janz, A., Puhlmann, R., Richens, T., Watson, D., Sinton, H., Ball, K., Etmanski, A., Patterson, B., Brocke, L., Dyer, R., 2012. Regulatory history of Alberta's industrial land conservation and reclamation program. *Can J Soil Sci* 92, 39–51. <https://doi.org/10.4141/CJSS2010-033>

- Price, D.T., Alfaro, R.I., Brown, K.J., Flannigan, M.D., Fleming, R.A., Hogg, E.H., Girardin, M.P., Lakusta, T., Johnston, M., McKenney, D.W., Pedlar, J.H., Stratton, T., Sturrock, R.N., Thompson, I.D., Trofymow, J.A., Venier, L.A., 2013. Anticipating the consequences of climate change for Canada's boreal forest ecosystems. *Environmental Reviews* 21, 322–365. <https://doi.org/10.1139/er-2013-0042>
- Price, J., 2003. Role and character of seasonal peat soil deformation on the hydrology of undisturbed and cutover peatlands. *Water Resour Res* 39, 1–10. <https://doi.org/10.1029/2002WR001302>
- Price, J., Heathwaite, A., Baird, A., 2003. Hydrological processes in abandoned and restored peatlands. *Wetlands and Ecological Management* 11, 65–83.
- Price, J., Rochefort, L., Quinty, F., 1998. Energy and moisture considerations on cutover peatlands: surface microtopography, mulch cover and Sphagnum regeneration. *Ecol Eng* 10, 293–312. [https://doi.org/10.1016/S0925-8574\(98\)00046-9](https://doi.org/10.1016/S0925-8574(98)00046-9)
- Price, J., Whitehead, G.S., 2001. Developing hydrologic thresholds for Sphagnum recolonization on an abandoned cutover bog. *Wetlands* 21, 32–40. [https://doi.org/10.1672/0277-5212\(2001\)021\[0032:DHTFSR\]2.0.CO;2](https://doi.org/10.1672/0277-5212(2001)021[0032:DHTFSR]2.0.CO;2)
- Price, J.S., Fitzgibbon, J.E., 1987. Groundwater storage-streamflow relations during winter in a subarctic wetland, Saskatchewan. *Canadian Journal of Earth Science* 24.
- Price, J.S., McLaren, R.G., Rudolph, D.L., 2010. Landscape restoration after oil sands mining: Conceptual design and hydrological modelling for fen reconstruction. *Int J Min Reclam Environ* 24, 109–123. <https://doi.org/10.1080/17480930902955724>
- Priestley, C.H.B., Taylor, R.J., 1972. On the Assessment of Surface Heat Flux and Evaporation Using Large-Scale Parameters. *Mon Weather Rev* 100, 81–92. [https://doi.org/10.1175/1520-0493\(1972\)100<0081:otaosh>2.3.co;2](https://doi.org/10.1175/1520-0493(1972)100<0081:otaosh>2.3.co;2)
- Proctor, M.C.F., Oliver, M.J., Wood, A.J., Alpert, P., Stark, L.R., Cleavitt, N.L., Mishler, B.D., 2007. Desiccation-tolerance in bryophytes: A review. *Bryologist* 110, 595–621. [https://doi.org/10.1639/0007-2745\(2007\)110\[595:DIBAR\]2.0.CO;2](https://doi.org/10.1639/0007-2745(2007)110[595:DIBAR]2.0.CO;2)
- Quinton, W.L., Hayashi, M., Carey, S.K., 2008. Peat hydraulic conductivity in cold regions and its relation to pore size and geometry. *Hydrol Process* 22, 2829–2837. <https://doi.org/10.1002/hyp.7027>
- Quinty, F., Leblanc, M.-C., Rochefort, L., 2020a. Peatland Restoration Guide: Plant Material Collecting and Donor Site Management.
- Quinty, F., Leblanc, M.-C., Rochefort, L., 2020b. Peatland Restoration Guide: Spreading Plant Material, Mulch and Fertilizer.
- R Core Team, 2024. R: A language and environment for statistical computing.
- Rawls, W.J., Pachepsky, Y.A., Ritchie, J.C., Sobecki, T.M., Bloodworth, H., 2003. Effect of soil organic carbon on soil water retention. *Geoderma* 116, 61–76. [https://doi.org/10.1016/S0016-7061\(03\)00094-6](https://doi.org/10.1016/S0016-7061(03)00094-6)
- Redding, T., Devito, K., 2011. Aspect and soil textural controls on snowmelt runoff on forested Boreal Plain hillslopes. *Hydrology Research* 42, 250–267. <https://doi.org/10.2166/nh.2011.162>

- Redding, T.E., Devito, K.J., 2006. Particle densities of wetland soils in northern Alberta, Canada. *Can J Soil Sci* 86, 57–60. <https://doi.org/10.4141/S05-061>
- Reddy, K.R., Clark, M.W., DeLaune, R.D., Kongchum, M., 2013. Physicochemical Characterization of Wetland Soils, in: DeLaune, R.D., Reddy, K.R., Richardson, C.J., Megonigal, J.P. (Eds.), *Methods in Biogeochemistry of Wetlands*. Soil Science Society of America, Madison, WI, pp. 41–54.
- Reddy, K.R., DeLaune, R.D., Inglett, P.W., 2023. Phosphorus, in: *Biogeochemistry of Wetlands*. Taylor and Francis.
- Reeve, A.S., Siegel, D.I., Glaser, P.H., 2000. Simulating vertical flow in large peatlands. *J Hydrol (Amst)* 227. [https://doi.org/10.1016/S0022-1694\(99\)00183-3](https://doi.org/10.1016/S0022-1694(99)00183-3)
- Rettie, W.J., Messier, F., 2000. Hierarchical habitat selection by woodland caribou: Its relationship to limiting factors. *Ecography* 23, 466–478. <https://doi.org/10.1111/j.1600-0587.2000.tb00303.x>
- Rezanezhad, F., Price, J.S., Quinton, W.L., Lennartz, B., Milojevic, T., Van Cappellen, P., 2016. Structure of peat soils and implications for water storage, flow and solute transport: A review update for geochemists. *Chem Geol* 429, 75–84. <https://doi.org/10.1016/j.chemgeo.2016.03.010>
- Rocheftort, L., LeBlanc, M.C., Bérubé, V., Hugron, S., Boudreau, S., Pouliot, R., 2016. Reintroduction of fen plant communities on a degraded minerotrophic peatland. *Botany* 94, 1041–1051. <https://doi.org/10.1139/cjb-2016-0023>
- Rocheftort, L., Quinty, F., Campeau, S., Johnson, K., Malterer, T., 2003. North American approach to the restoration of wetlands. *Wetl Ecol Manag* 11, 3–20. <https://doi.org/10.1023/A:1022011027946>
- Rooney, R.C., Bayley, S.E., Schindler, D.W., 2012. Oil sands mining and reclamation cause massive loss of peatland and stored carbon. *Proc Natl Acad Sci U S A* 109, 4933–4937. <https://doi.org/10.1073/pnas.1117693108>
- Rooney, R.C., Robinson, D.T., Petrone, R., 2015. Megaproject reclamation and climate change. *Nat Clim Chang* 5, 963–966. <https://doi.org/10.1038/nclimate2719>
- Ruppel, M., Väliiranta, M., Virtanen, T., Korhola, A., 2013. Postglacial spatiotemporal peatland initiation and lateral expansion dynamics in North America and northern Europe. *Holocene* 23, 1596–1606. <https://doi.org/10.1177/0959683613499053>
- Rydin, H., Jeglum, J.K., 2013. Peatland Succession and Development, in: *The Biology of Peatlands*. Oxford University Press, pp. 127–147.
- Rydin, H., Jeglum, J.K., 2006a. Peatland succession and development, *The Biology of Peatlands*. Oxford University Press. <https://doi.org/10.1093/acprof:osobl/9780199602995.001.0001>
- Rydin, H., Jeglum, J.K., 2006b. Peat and organic soil, in: *Biology of Peatlands*. Oxford University Press, pp. 77–100.
- Saraswati, S., Bhusal, Y., Trant, A.J., Strack, M., 2020a. Roads impact tree and shrub productivity in adjacent boreal peatlands. *Forests* 11. <https://doi.org/10.3390/F11050594>
- Saraswati, S., Parsons, C.T., Strack, M., 2019. Access roads impact enzyme activities in boreal forested peatlands. *Science of the Total Environment* 651, 1405–1415. <https://doi.org/10.1016/j.scitotenv.2018.09.280>

- Saraswati, S., Petrone, R.M., Rahman, M.M., McDermid, G.J., Xu, B., Strack, M., 2020b. Hydrological effects of resource-access road crossings on boreal forested peatlands. *J Hydrol (Amst)* 584. <https://doi.org/10.1016/j.jhydrol.2020.124748>
- Saraswati, S., Strack, M., 2019. Road crossings increase methane emissions from adjacent peatland. *J Geophys Res Biogeosci* 124, 3588–3599. <https://doi.org/10.1029/2019JG005246>
- Sauer, T.J., Horton, R., 2005. Soil Heat Flux, in: Hatfield, J.L., Baker, J.M. (Eds.), *Micrometeorology in Agricultural Systems*. American Society of Agronomy, Madison, WI, pp. 131–154.
- Saxton, K.E., Rawls, W.J., Romberger, J.S., Papendick, R.I., 1986. Estimating Generalized Soil-water Characteristics from Texture. *Soil Science Society of America Journal* 50, 1031–1036.
- Scanlon, B.R., Healy, R.W., Cook, P.G., 2002. Choosing appropriate techniques for quantifying groundwater recharge. *Hydrogeol J* 10, 18–39. <https://doi.org/10.1007/s10040-0010176-2>
- Scarlett, S.J., Petrone, R.M., Price, J.S., 2017. Controls on plot-scale evapotranspiration from a constructed fen in the Athabasca Oil Sands Region, Alberta. *Ecol Eng* 100, 199–210. <https://doi.org/10.1016/j.ecoleng.2016.12.020>
- Schelle, H., Heise, L., Jänicke, K., Durner, W., 2013. Water retention characteristics of soils over the whole moisture range: A comparison of laboratory methods. *Eur J Soil Sci* 64, 814–821. <https://doi.org/10.1111/ejss.12108>
- Scheller, E., Joergensen, R.G., 2008. Decomposition of wheat straw differing in nitrogen content in soils under conventional and organic farming management. *Journal of Plant Nutrition and Soil Science* 171, 886–892. <https://doi.org/10.1002/jpln.200700272>
- Schindler, U., Durner, W., von Unold, G., Müller, L., 2010. Evaporation method for measuring unsaturated hydraulic properties of soils: Extending the measurement range. *Soil Science Society of America Journal* 74, 1071–1083. <https://doi.org/10.2136/sssaj2008.0358>
- Schipperges, B., Rydin, H., 1998. Response of photosynthesis of Sphagnum species from contrasting microhabitats to tissue water content and repeated desiccation. *New Phytologist* 140, 677–684. <https://doi.org/10.1046/j.1469-8137.1998.00311.x>
- Schneider, R., Dyer, S., 2006. Death by a thousand cuts: impacts of in situ oil sands development on Alberta's boreal forest, Canadian Parks and Wilderness Society and the Pembina Institute.
- Schneider, R.R., Devito, K., Kettridge, N., Bayne, E., 2016. Moving beyond bioclimatic envelope models: integrating upland forest and peatland processes to predict ecosystem transitions under climate change in the western Canadian boreal plain. *Ecohydrology* 9, 899–908. <https://doi.org/10.1002/eco.1707>
- Sexstone, G.A., Clow, D.W., Stannard, D.I., Fassnacht, S.R., 2016. Comparison of methods for quantifying surface sublimation over seasonally snow-covered terrain. *Hydrol Process* 30, 3373–3389. <https://doi.org/10.1002/hyp.10864>
- Shunina, A., Osko, T.J., Foote, L., Bork, E.W., 2016. Comparison of site preparation and revegetation strategies within a sphagnum-dominated peatland following removal of an oil well pad. *Ecological Restoration* 34, 225–235. <https://doi.org/10.3368/er.34.3.225>

- Sjörs, H., 1950. On the relation between vegetation and electrolytes in North Swedish mire waters. *Oikos* 2, 241–258.
- Sottocornola, M., Boudreau, S., Rochefort, L., 2007. Peat bog restoration: Effect of phosphorus on plant re-establishment. *Ecol Eng* 31, 29–40. <https://doi.org/10.1016/j.ecoleng.2007.05.001>
- Speller, J., Forbes, V., 2022. On the role of peat bogs as components of Indigenous cultural landscapes in Northern North America. *Arct Antarct Alp Res* 54, 96–110. <https://doi.org/10.1080/15230430.2022.2049957>
- Spohn, M., 2024. Preferential adsorption of nitrogen- and phosphorus-containing organic compounds to minerals in soils: A review. *Soil Biol Biochem* 194. <https://doi.org/10.1016/j.soilbio.2024.109428>
- Spohn, M., Diáková, K., Aburto, F., Doetterl, S., Borovec, J., 2022. Sorption and desorption of organic matter in soils as affected by phosphate. *Geoderma* 405. <https://doi.org/10.1016/j.geoderma.2021.115377>
- Spohn, M., Schleuss, P.M., 2019. Addition of inorganic phosphorus to soil leads to desorption of organic compounds and thus to increased soil respiration. *Soil Biol Biochem* 130, 220–226. <https://doi.org/10.1016/j.soilbio.2018.12.018>
- Strack, M., Waller, M.F., Waddington, J.M., 2006. Sedge succession and peatland methane dynamics: A potential feedback to climate change. *Ecosystems* 9, 278–287. <https://doi.org/10.1007/s10021-005-0070-1>
- Surridge, B.W.J., Baird, A.J., Heathwaite, A.L., 2005. Evaluating the quality of hydraulic conductivity estimates from piezometer slug tests in peat. *Hydrol Process* 19, 1227–1244. <https://doi.org/10.1002/hyp.5653>
- Szumigalski, A.R., Bayley, S.E., 1996. Net above-ground primary production along a bog-rich fen gradient in central Alberta, Canada. *Wetlands* 16, 467–476.
- Tamminga, A., Scott, N.A., Treitz, P., Woods, M., 2014. A biogeochemical examination of Ontario's boreal forest ecosite classification system. *Forests* 5, 325–346. <https://doi.org/10.3390/f5020325>
- Tarnocai, C., 1984. Peat Resources of Canada.
- Thiffault, E., Webster, K., Lafleur, B., Wilson, S., Mansuy, N., 2017. Biophysical indicators based on spatial hierarchy for informing land reclamation: The case of the Lower Athabasca River (Alberta, Canada). *Ecol Indic* 72, 173–184. <https://doi.org/10.1016/j.ecolind.2016.08.020>
- Thompson, C., Mendoza, C.A., Devito, K.J., 2017. Potential influence of climate change on ecosystems within the Boreal Plains of Alberta. *Hydrol Process* 31, 2110–2124. <https://doi.org/10.1002/hyp.11183>
- Thompson, C., Mendoza, C.A., Devito, K.J., Petrone, R.M., 2015. Climatic controls on groundwater-surface water interactions within the Boreal Plains of Alberta: Field observations and numerical simulations. *J Hydrol (Amst)* 527, 734–746. <https://doi.org/10.1016/j.jhydrol.2015.05.027>
- UNEP, 2022. Global Peatlands Assessment - The State of the World's Peatlands: Evidence for action toward the conservation, restoration, and sustainable management of peatlands. Nairobi, Kenya.

- Updegraff, K., Pastor, J., Bridgham, S.D., Johnston, C.A., 1995. Environmental and substrate controls over carbon and nitrogen mineralization in northern wetlands. *Ecological Applications* 5, 151–163.
- U.S. EPA, 2014. Method 6010D (SW-846): Inductively Coupled Plasma-Atomic Emission Spectrometry (Revision 4). Washington, D. C.
- Van Huizen, B., Petrone, R.M., Price, J.S., Quinton, W.L., Pomeroy, J.W., 2020. Seasonal ground ice impacts on spring ecohydrological conditions in a western boreal plains peatland. *Hydrol Process* 34, 765–779. <https://doi.org/10.1002/hyp.13626>
- Van Rensen, C.K., Nielsen, S.E., White, B., Vinge, T., Lieffers, V.J., 2015. Natural regeneration of forest vegetation on legacy seismic lines in boreal habitats in Alberta’s oil sands region. *Biol Conserv* 184, 127–135. <https://doi.org/10.1016/j.biocon.2015.01.020>
- Vitt, D.H., 2014. A Key and Review of Bryophytes Common in North American Peatlands. *Evansia* 31, 121–158. <https://doi.org/10.1639/079.031.0402>
- Vitt, D.H., Bayley, S.E., Tai-Long Jin, 1995. Seasonal variation in water chemistry over a bog-rich fen gradient in continental western Canada. *Canadian Journal of Fisheries and Aquatic Sciences* 52, 587–606. <https://doi.org/10.1139/f95-059>
- Vitt, D.H., Chee, W.L., 1990. The relationships of vegetation to surface water chemistry and peat chemistry in fens of Alberta, Canada. *Vegetatio* 89, 87–106. <https://doi.org/10.1007/BF00032163>
- Vitt, D.H., Crandall-Stotler, B., Wood, A., 2014. Bryophytes: Survival in a dry world through tolerance and avoidance, in: Rajakaruna, N., Boyd, R., Harris, T. (Eds.), *Plant Ecology and Evolution in Harsh Environments*. Nova Publishers, pp. 267–295.
- Vitt, D.H., Halsey, L.A., Bauer, I.E., Campbell, C., 2000. Spatial and temporal trends in carbon storage of peatlands of continental western Canada through the Holocene. *Can J Earth Sci* 37, 683–693. <https://doi.org/10.1139/e99-097>
- Vitt, D.H., Hayes, C., Wieder, K., 2012. Reclamation of decommissioned oil and gas pads initially constructed in boreal peatlands, Canadian Reclamation.
- Vitt, D.H., Lüth, M., 2017. A Guide to Mosses and Liverworts of Alberta Peatlands, *Evansia*. Northern Alberta Institute of Technology Boreal Research Institute, Edmonton, Alberta.
- Vitt, D.H., Wieder, R.K., Scott, K.D., Faller, S., 2009. Decomposition and peat accumulation in rich fens of boreal Alberta, Canada. *Ecosystems* 12, 360–373. <https://doi.org/10.1007/s10021-009-9228-6>
- Vitt, D.H., Wieder, R.K., Xu, B., Kaskie, M., Koropchak, S., 2011. Peatland establishment on mineral soils: Effects of water level, amendments, and species after two growing seasons. *Ecol Eng* 37, 354–363. <https://doi.org/10.1016/j.ecoleng.2010.11.029>
- Volik, O., Elmes, M., Petrone, R., Kessel, E., Green, A., Cobbaert, D., Price, J., 2020. Wetlands in the Athabasca Oil Sands Region: The nexus between wetland hydrological function and resource extraction. *Environmental Reviews* 28, 246–261. <https://doi.org/10.1139/er-2019-0040>
- Volik, O., Kessel, E., Green, A., Petrone, R., Price, J., 2021. Growing season evapotranspiration in boreal fens in the Athabasca Oil Sands Region: Variability and environmental controls. *Hydrol Process* 35, 1–17. <https://doi.org/10.1002/hyp.14020>

- Volik, O., Petrone, R., Price, J., 2024. Wetlands as integral parts of surface water–groundwater interactions in the Athabasca Oil Sands Area: Review and synthesis. *Environmental Reviews*. <https://doi.org/10.1139/er-2023-0064>
- von Sengbusch, P., 2015. Enhanced sensitivity of a mountain bog to climate change as a delayed effect of road construction. *Mires and Peat* 15.
- Waddington, J.M., Morris, P.J., Kettridge, N., Granath, G., Thompson, D.K., Moore, P.A., 2015. Hydrological feedbacks in northern peatlands. *Ecohydrology* 8, 113–127. <https://doi.org/10.1002/eco.1493>
- Wang, M., Moore, T.R., Talbot, J., Riley, J.L., 2015. The stoichiometry of carbon and nutrients in peat formation. *Global Biogeochem Cycles* 29, 113–121. <https://doi.org/10.1002/2014GB005000>
- Weiland, L., Ketcheson, S., Strack, M., McDermid, G.J., 2024. The Influence of Seismic Lines on Local Hydrology and Snow Accumulation in the Boreal Region of Northern Alberta. *Hydrol Process* 38. <https://doi.org/10.1002/hyp.70032>
- Wendlandt, M.E., 2023. Characterizing Hydrological Balance and Hydrochemistry of a Chronosequence of Newly Formed Wetlands within the Athabasca Oil Sands Region using Isotopic and Geochemical Tracers. University of Calgary, Calgary, AB.
- Wickham, H., 2016. *ggplot2: Elegant Graphics for Data Analysis*. Springer-Verlag, New York, NY. <https://doi.org/10.1007/978-0-387-98141-3>
- Williams, T.J., Quinton, W.L., 2013. Modelling incoming radiation on a linear disturbance and its impact on the ground thermal regime in discontinuous permafrost. *Hydrol Process* 27, 1854–1865. <https://doi.org/10.1002/hyp.9792>
- Williams-Mounsey, J., Grayson, R., Crowle, A., Holden, J., 2021. A review of the effects of vehicular access roads on peatland ecohydrological processes. *Earth Sci Rev* 214, 103528. <https://doi.org/10.1016/j.earscirev.2021.103528>
- Willier, C.N., Dennett, J.M., Devito, K.J., Bater, C.W., Nielsen, S.E., 2022. The extent and magnitude of edge effects on woody vegetation in road-bisected treed peatlands in boreal Alberta, Canada. *Ecohydrology* 15. <https://doi.org/10.1002/eco.2455>
- Wind-Mulder, H.L., Vitt, D.H., 2000. Comparisons of water and peat chemistries of a post-harvested and undisturbed peatland with relevance to restoration. *Wetlands* 20, 616–628. [https://doi.org/10.1672/0277-5212\(2000\)020\[0616:COWAPC\]2.0.CO;2](https://doi.org/10.1672/0277-5212(2000)020[0616:COWAPC]2.0.CO;2)
- Winter, T.C., 1981. Uncertainties in estimating the water balances of lakes. *J Am Water Resour Assoc* 17, 82–115. <https://doi.org/10.1111/j.1752-1688.1981.tb02593.x>
- Woo, M.-K., Winter, T.C., 1993. The role of permafrost and seasonal frost in the hydrology of northern wetlands in North America. *J Hydrol (Amst)* 141, 5–31.
- Woo, M.K., Xia, Z., 1996. Effects of hydrology on the thermal conditions of the active layer. *Nordic Hydrology* 27, 129–142. <https://doi.org/10.2166/nh.1996.0024>
- Wood, M.E., Macrae, M.L., Strack, M., Price, J.S., Osko, T.J., Petrone, R.M., 2016. Spatial variation in nutrient dynamics among five different peatland types in the Alberta oil sands region. *Ecohydrology* 9, 688–699. <https://doi.org/10.1002/eco.1667>

- Wu, J., Kutzbach, L., Jager, D., Wille, C., Wilmking, M., 2010. Evapotranspiration dynamics in a boreal peatland and its impact on the water and energy balance. *J Geophys Res Biogeosci* 115. <https://doi.org/10.1029/2009JG001075>
- Xu, B., Rochefort, L., Bird, M., Khadka, B., Strack, M., 2022. Restoration of boreal peatland impacted by an in-situ oil sands well-pad 1: Vegetation response. *Restor Ecol* 30. <https://doi.org/10.1111/rec.13514>
- Xu, X., Thornton, P.E., Post, W.M., 2013. A global analysis of soil microbial biomass carbon, nitrogen and phosphorus in terrestrial ecosystems. *Global Ecology and Biogeography* 22, 737–749. <https://doi.org/10.1111/geb.12029>
- Xue, B., Huang, L., Li, X., Lu, J., Gao, R., Kamran, M., 2022. Straw residue incorporation and potassium fertilization enhances soil aggregate stability by altering soil content of iron oxide and organic carbon in a rice–rape cropping system. *Land Degrad Dev* 33, 2567–2584. <https://doi.org/10.1002/ldr.4333>
- Yu, Z.C., 2012. Northern peatland carbon stocks and dynamics: A review. *Biogeosciences* 9, 4071–4085. <https://doi.org/10.5194/bg-9-4071-2012>
- Zabel, A., Ketcheson, S.J., Petrone, R.M., 2024. Climatic controls on the water balance of a pilot-scale oil sands mining pit lake in the Athabasca oil sands region, Canada. *Int J Min Reclam Environ* 38, 306–323. <https://doi.org/10.1080/17480930.2023.2270301>
- Zedler, J.B., Kercher, S., 2004. Causes and consequences of invasive plants in wetlands: Opportunities, opportunists, and outcomes. *CRC Crit Rev Plant Sci* 23, 431–452. <https://doi.org/10.1080/07352680490514673>
- Zoltai, S.C., Vitt, D.H., 1990. Holocene climatic change and the distribution of peatlands in western interior Canada. *Quat Res* 33, 231–240. [https://doi.org/10.1016/0033-5894\(90\)90021-C](https://doi.org/10.1016/0033-5894(90)90021-C)
- Zuur, A.F., Ieno, E.N., Walker, N.J., Saveliev, A.A., Smith, G.M., 2009. *Mixed Effects Models and Extensions in Ecology with R*, 1st ed. Springer, New York, NY. <https://doi.org/10.1007/978-0-387-87458-6>

APPENDIX A

Supplementary Figures and Tables for Chapter 2

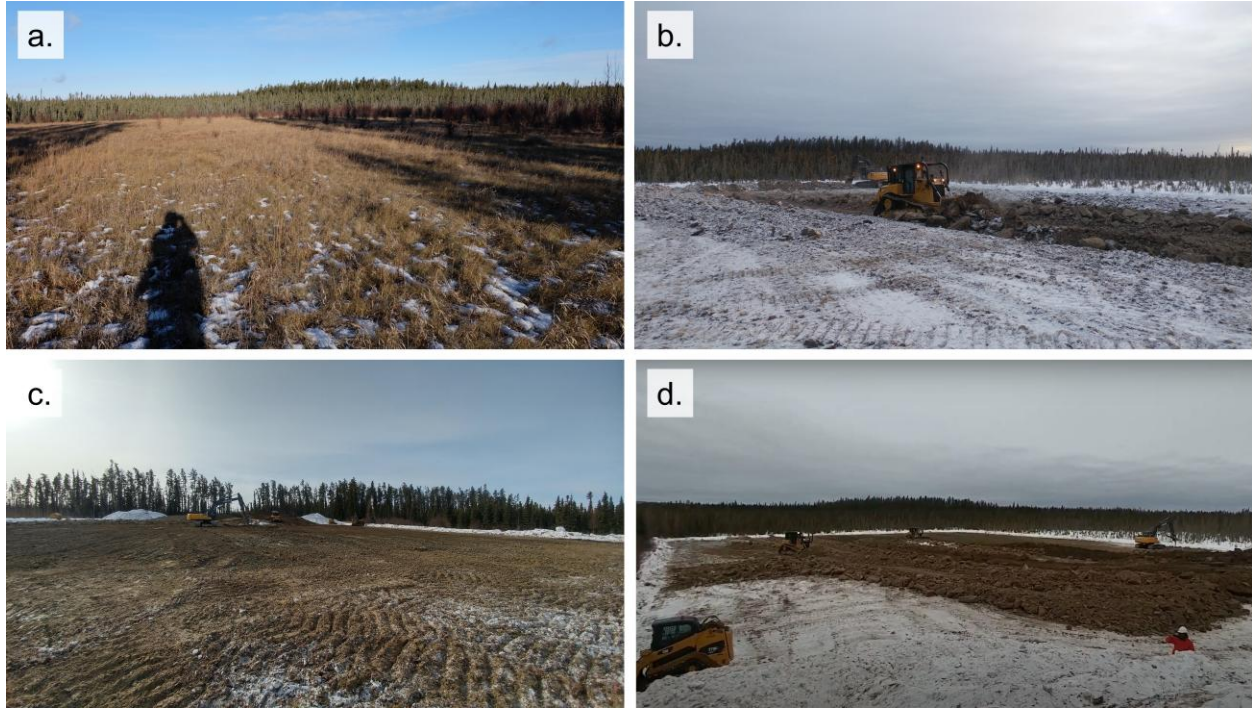


Figure A.1: Photos of the intact well pad taken prior to partial removal (a) and photos of the partial removal process taken during earthworks in March 2020 (b-d). Photo (a) was provided by the Northern Alberta Institute of Technology (Centre for Boreal Research).



Figure A.2: Photos of the vegetation propagule collection process (a), vegetation propagules prior to introduction onto the site (b), spreading of the vegetation propagules using a skid-steer (c), and propagules introduced directly onto the roughened residual mineral substrate (d). All photos were provided by the Northern Alberta Institute of Technology (Centre for Boreal Research).

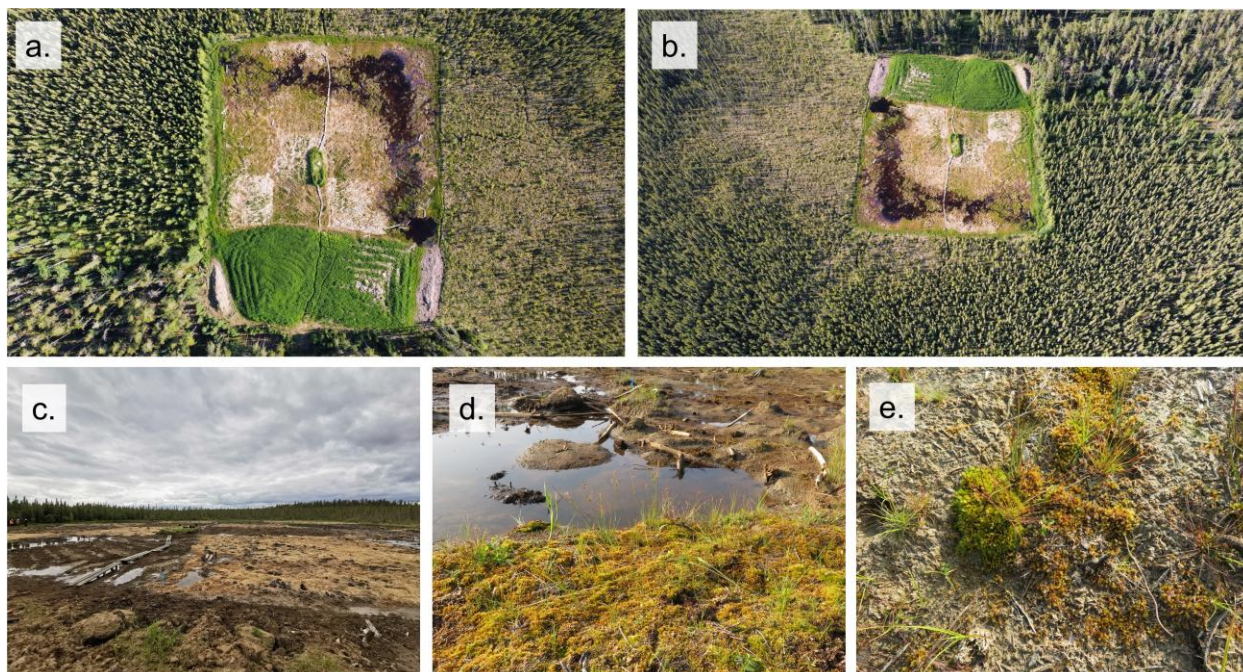


Figure A.3: Aerial photos of the residual well pad taken using a drone during wet conditions (a-b), overview photo of the mulch treatment (c), and detailed photos of recently introduced moss propagules (d-e). All photos were taken in June 2020. All photos were provided by the Northern Alberta Institute of Technology (Centre for Boreal Research).

Table A.1: Geometric mean horizontal saturated hydraulic conductivity by depth and location/substrate type, including sample size (number of laboratory cores or in-situ bail tests used for calculation). Geometric standard deviations (unitless) are provided in brackets.

Location (substrate type)	Sampling depth (cm b.g.s.)	Geometric mean K_h ($m\ s^{-1}$)	n
Residual pad	0-5	5.20×10^{-6} (1.91) ^a	12
(mineral fill)	25	1.77×10^{-5} (3.90)	38
Residual pad	50	9.29×10^{-6} (3.25)	33
(underlying peat)	100	5.81×10^{-8} (5.37)	26
Upgradient peatland	25	1.28×10^{-4} (3.02)	21
(peat)	50	4.54×10^{-5} (2.42)	27
	100	1.48×10^{-6} (7.18)	22
Downgradient peatland	25	8.80×10^{-5} (2.18)	15
(peat)	50	1.11×10^{-5} (3.53)	14
	100	6.11×10^{-7} (3.39)	12
North peatland	25	9.86×10^{-5} (1.65)	22
(peat)	50	4.76×10^{-5} (2.22)	27
	100	8.01×10^{-7} (4.82)	22

^a Laboratory-based estimate, all other values are in-situ estimates

Table A.2: Summary of seasonal mean half-hourly depth to water table (DWT) for logged wells on the residual pad and in adjacent peatlands. Negative values denote mean WTs below the ground surface. Standard deviations are provided in brackets.

Well location	Mean DWT (m)	
	2020 (Wet)	2021 (Dry)
Upgradient pad	-0.02 (0.03)	-0.13 (0.14)
North pad	0.09 (0.07)	-0.02 (0.15)
Inner pad	-0.26 (0.14)	-0.51 (0.16)
Downgradient pad	-0.13 (0.17)	-0.19 (0.21)
Upgradient peatland	0.02 (0.03)	-0.05 (0.10)
North peatland	0.09 (0.03)	-0.09 (0.15)
Downgradient peatland	-0.06 (0.03)	-0.24 (0.16)

APPENDIX B

Supplementary Tables for Chapter 3

Table B.1: Summary of VWC at total saturation ($\Psi = 0$) and the approximate threshold values associated with optimal moss-water availability (> -10 mb) and desiccation risk (< -100 mb) modelled using the constrained bimodal van Genuchten equations fit for each of five soil cores collected from the residual mineral pad. The 95% confidence intervals about the means are reported in brackets.

Ψ (mb)	Mean modelled VWC (m^3m^{-3})	Maximum modelled VWC (m^3m^{-3})	Minimum modelled VWC (m^3m^{-3})
0	0.39 (0.02)	0.41	0.37
-10	0.39 (0.02)	0.41	0.37
-100	0.28 (0.07)	0.34	0.20

Table B.2: Summary of bimodal-constrained van Genuchten soil water retention model parameters for each of five soil cores collected from plots on the residual mineral pad.

Core	α_1 (cm^{-1})	n_1	w_1	Residual water content ($\text{cm}^3 \text{cm}^{-3}$)	Saturated water content ($\text{cm}^3 \text{cm}^{-3}$)	α_2 (cm^{-1})	n_2	w_2
1	0.00846	7.728	0.349	0.082	0.409	0.00806	1.899	0.651
2	0.00931	3.359	0.457	0.075	0.387	0.011	15.000	0.543
3	0.0103	12.671	0.433	0.079	0.372	0.00861	3.021	0.567
4	0.00423	3.212	0.404	0.137	0.385	0.0117	4.143	0.596
5	0.00766	2.102	0.662	0.091	0.407	0.00885	7.808	0.338

Table B.3: Summary of seasonal mean manually measured depth to water table values on the residual pad for 2020 and 2021 by plot and associated hydrological classifications (well-optimized and non-optimized classifications denoted as WO and NO, respectively). Negative depth to water table values denote mean water table positions below the ground surface, while positive depth to water table values denote mean water table positions above the ground surface. Standard deviations are provided in brackets.

Plot	Mean depth to water table (m)			Classification (2020)	Classification (2021)
	2020	2021	Multi-year		
1	-0.04 (0.13)	-0.18 (0.20)	-0.13 (0.19)	WO	NO
2	-0.03 (0.10)	-0.15 (0.22)	-0.11 (0.20)	WO	NO
3	-0.03 (0.10)	-0.15 (0.22)	-0.11 (0.20)	WO	NO
4	0.03 (0.05)	-0.06 (0.12)	-0.03 (0.11)	WO	WO
5	-0.18 (0.19)	-0.35 (0.26)	-0.29 (0.25)	NO	NO
6	-0.18 (0.17)	-0.31 (0.26)	-0.25 (0.24)	NO	NO
7	-0.14 (0.12)	-0.25 (0.22)	-0.18 (0.19)	NO	NO
8	-0.02 (0.05)	-0.12 (0.13)	-0.07 (0.11)	WO	NO
9	-0.19 (0.19)	-0.35 (0.26)	-0.29 (0.25)	NO	NO
10	-0.28 (0.19)	-0.38 (0.28)	-0.33 (0.25)	NO	NO
11	-0.36 (0.16)	-0.38 (0.28)	-0.37 (0.24)	NO	NO
12	-0.02 (0.05)	-0.12 (0.13)	-0.07 (0.11)	WO	NO
13	-0.18 (0.15)	-0.29 (0.26)	-0.24 (0.22)	NO	NO
14	-0.24 (0.16)	-0.46 (0.28)	-0.32 (0.24)	NO	NO
15	-0.31 (0.17)	-0.47 (0.28)	-0.41 (0.26)	NO	NO
16	0.00 (0.02)	-0.11 (0.11)	-0.07 (0.10)	WO	NO

APPENDIX C

Supplementary Figures and Tables for Chapter 4

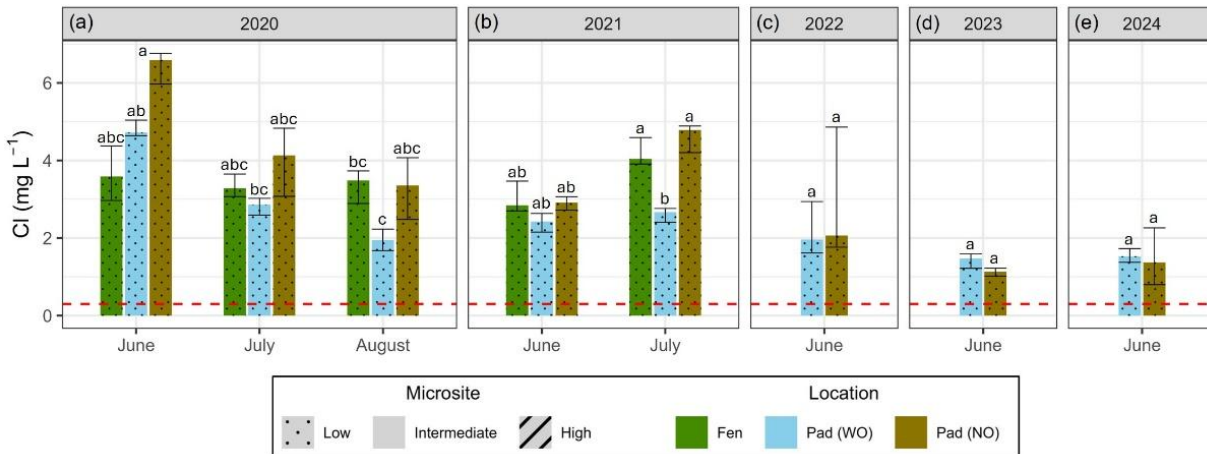


Figure C.1: Pore water chloride (Cl) concentrations in the surrounding fen and in the hydrologically well optimized (*WO*) and non-optimized (*NO*) areas on the residual pad. In the years 2022-2024, pore water was not collected in the surrounding fen. Bars illustrate median values $\pm$ the 25th and 75th percentiles for each group. Medians of groups within a given year not sharing any common letters significantly differ based on Kruskal-Wallis, Wilcoxon Rank-Sum, or Peto-Peto tests. The limit of quantitation is denoted by a dashed red line.

Table C.1: Mean annual and growing season (May to September) air temperatures and total precipitation reported for all study years, and comparison to 30-year climate normals (1991-2020). All data were recorded at the Slave Lake airport (55°17'35.000" N, 114°46'38.000" W) approximately 20 km away from the study site.

Year	Air temperature (°C)		Total precipitation (mm)	
	Annual mean	Growing season mean	Annual sum	Growing season sum
30-year climate normal (1991-2020)	2.1	13.0	422.1	292.0
2020	1.7	13.1	615.6	311.3
2021	2.3	14.3	530.7	273.7
2022	2.3	14.1	372.8	181.6
2023	4.2	14.9	366.9	320.0
2024	4.1	13.9	474.6	388.1

Table C.2: Mean ($\pm$ standard deviation) shallow groundwater pH, electrical conductivity (EC), and temperature values for the moss propagule donor site. Donor site measurements ($n = 10$ per year) were made in the month of August in each year between 2020 and 2022.

Year	pH	EC ($\mu\text{S cm}^{-1}$)	Temp. ($^{\circ}\text{C}$)
2020	6.3 ± 0.1	70.2 ± 5.4	13.1 ± 0.2
2021	5.9 ± 0.1	43.2 ± 6.7	12.7 ± 0.6
2022	6.0 ± 0.1	100.8 ± 5.4	<i>Not measured</i>

Table C.3: Results of statistical analyses performed on nutrient supply rates measured using PRS probes installed on the pad and in the fen. Statistically significant results are provided in bold typeface. The type of model used (OLS or OLS-Log) is provided for each statistical test.

Parameter	Effect	2020 (June; pre-fertilization)				2020 (August; post-fertilization)				2021 (June)				2023 (June)			
		Model	Estimate	t-value	p-value	Model	Estimate	t-value	p-value	Model	Estimate	t-value	p-value	Model	Estimate	t-value	p-value
NH ₄	VWC		-	-	-		-	-	-		-	-	-		15.055	2.588	0.036
	Fertilization	Insufficient data > LOD	-	-	-	Insufficient data > LOD	-	-	-	Insufficient data > LOD	-	-	-	OLS-Log	-0.099	-0.371	0.722
	Mulching		-	-	-		-	-	-		-	-	-		0.118	0.504	0.630
	Soil temp.		-	-	-		-	-	-		-	-	-		0.072	0.579	0.581
P	VWC		-6.944	-0.193	0.851		5.827	1.099	0.304		7.167	4.701	<0.001		21.711	2.120	0.072
	Fertilization	OLS	-	-	-	OLS-Log	-1.147	-3.140	0.014	OLS-Log	0.339	1.388	0.180	OLS	-0.189	-0.403	0.699
	Mulching		-0.019	-0.019	0.985		-0.355	-1.164	0.278		0.415	1.712	0.102		-0.519	-1.259	0.248
	Soil temp.		-	-	-		-	-	-		-	-	-		-0.126	-0.581	0.580
Ca	VWC		-0.420	-0.088	0.932		9559.400	2.231	0.056		4049.600	4.357	<0.001		0.719	0.518	0.620
	Fertilization	OLS-Log	-	-	-	OLS	-587.700	-1.991	0.082		278.100	1.866	0.077	OLS-Log	-0.038	-0.596	0.570
	Mulching		0.218	1.694	0.125		-124.400	-0.505	0.628		-157.300	-1.065	0.299		-0.035	-0.631	0.548
	Soil temp.		-	-	-		-	-	-		-	-	-		-0.053	-1.787	0.117
Mg	VWC		-160.160	-0.117	0.909		0.717	0.184	0.858		1.163	1.119	0.276		-4.189	-1.609	0.152
	Fertilization	OLS	-	-	-	OLS-Log	-0.588	-2.192	0.060		0.457	2.740	0.013	OLS-Log	-0.191	-1.606	0.152
	Mulching		42.940	1.170	0.272		-0.060	-0.269	0.795		-0.226	-1.368	0.186		-0.186	-1.775	0.119
	Soil temp.		-	-	-		-	-	-		-	-	-		0.034	0.607	0.563
K	VWC		-333.930	-1.892	0.091		11.350	0.087	0.933		-1.628	-2.146	0.044		-1.388	-0.302	0.772
	Fertilization	OLS	-	-	-	OLS	-2.912	-0.323	0.755		0.297	2.442	0.024	OLS-Log	0.227	1.078	0.317
	Mulching		13.845	2.916	0.017		12.613	1.678	0.132		0.254	2.108	0.048		0.005	0.028	0.978
	Soil temp.		-	-	-		-	-	-		-	-	-		0.098	1.006	0.348
Fe	VWC		569.900	0.124	0.904		2318.650	3.109	0.014		3321.750	5.789	<0.001		18.358	2.320	0.053
	Fertilization	OLS	-	-	-	OLS	-223.090	-4.341	0.002		-90.450	-0.983	0.337	OLS-Log	-0.054	-0.150	0.885
	Mulching		-45.110	-0.364	0.724		-183.700	-4.282	0.003		-56.720	-0.622	0.541		0.077	0.241	0.816
	Soil temp.		-	-	-		-	-	-		-	-	-		-0.092	-0.550	0.600
Mn	VWC		-257.610	-0.646	0.535		24.269	2.438	0.041		288.170	3.361	0.003		494.876	1.420	0.199
	Fertilization	OLS	-	-	-	OLS-Log	-0.943	-1.376	0.206		11.430	0.831	0.416	OLS	30.228	1.895	0.100
	Mulching		13.990	1.304	0.225		-0.016	-0.029	0.978		-16.940	-1.244	0.228		-17.296	-1.232	0.258
	Soil temp.		-	-	-		-	-	-		-	-	-		-9.759	-1.320	0.228
S	VWC		21.857	0.943	0.370		6.007	0.844	0.423		2.032	1.241	0.229		10.726	1.620	0.149
	Fertilization	OLS-Log	-	-	-	OLS-Log	0.267	0.544	0.601		0.492	1.875	0.075	OLS-Log	-0.060	-0.197	0.849
	Mulching		0.123	0.198	0.848		0.414	1.011	0.342		-0.102	-0.392	0.699		-0.151	-0.566	0.589
	Soil temp.		-	-	-		-	-	-		-	-	-		-0.021	-0.152	0.883
Al	VWC		-171.421	-1.294	0.228		-1.169	-0.469	0.651		0.644	1.190	0.248		-1.434	-0.205	0.843
	Fertilization	OLS	-	-	-	OLS-Log	-0.448	-2.613	0.031		0.007	0.079	0.938	OLS-Log	0.090	0.280	0.787
	Mulching		4.655	1.306	0.224		0.178	1.243	0.249		0.003	0.033	0.974		-0.077	-0.273	0.793
	Soil temp.		-	-	-		-	-	-		-	-	-		0.134	0.907	0.394

OLS = ordinary least squares regression (untransformed data); OLS-Log = ordinary least square regression (log-transformed response variable)

Table C.4: Results of statistical analyses performed on values of k and S determined using the Tea Bag Index for substrates within the pad and fen. Statistically significant results are provided in bold typeface. The type of model used (OLS or OLS-Log) is provided for each statistical test.

Parameter	Effect	2021				2023			
		Model	Estimate	t-value	p-value	Model	Estimate	t-value	p-value
k	VWC		-0.132	-2.291	0.049		-3.775	-4.484	<0.001
	Fertilization	OLS	0.006	1.310	0.227	OLS-Log	0.101	1.023	0.316
	Mulching		0.004	1.003	0.345		-0.074	-0.794	0.434
	Soil temp.		-	-	-		-0.090	-1.727	0.096
S	VWC		0.533	1.391	0.202		0.227	0.401	0.692
	Fertilization	OLS	-0.122	-4.169	0.003	OLS-Log	0.165	2.485	0.019
	Mulching		-0.041	-1.515	0.168		0.009	0.146	0.885
	Soil temp.		-	-	-		-0.076	-2.168	0.039

*OLS = ordinary least squares regression (untransformed data); OLS-Log = ordinary least square regression (log-transformed response variable)

Table C.5: Results of statistical analyses performed on nutrient transformation rates quantified for substrates on the pad and in the fen. Statistically significant results are provided in bold typeface. The type of model used (OLS or OLS-Log) is provided for each statistical test.

Parameter	Effect	2020				2021				2023				2024			
		Model	Estimate	t-value	p-value	Model	Estimate	t-value	p-value	Model	Estimate	t-value	p-value	Model	Estimate	t-value	p-value
NO ₃	VWC		-0.213	-3.747	0.002		0.023	1.442	0.180		-	-	-		0.951	0.280	0.789
	Fertilization	OLS*	-	-	-	OLS*	0.006	2.027	0.070	Insufficient data > LOD	-	-	-	OLS*	-0.014	-0.025	0.981
	Mulching		-0.002	-0.460	0.652		-0.001	-0.551	0.594		-	-	-		-0.335	-0.686	0.518
	Soil temp.		-	-	-		-	-	-		-	-	-		-	-	-
NH ₄	VWC		-1.280	-4.260	<0.001		-	-	-		-	-	-		-0.117	-0.074	0.943
	Fertilization	OLS	-	-	-	Insufficient data > LOD	-	-	-	Insufficient data > LOD	-	-	-	OLS	-0.399	-1.440	0.188
	Mulching		0.016	0.946	0.355		-	-	-		-	-	-		0.091	0.327	0.752
	Soil temp.		-	-	-		-	-	-		-	-	-		-	-	-

*OLS = ordinary least squares regression (untransformed data)

Table C.6: Results of statistical analyses performed on substrate extractable nutrient contents quantified for substrates on the pad and in the fen. Statistically significant results are provided in bold typeface. The type of model used (OLS, OLS-Log, or MLE-Cen) is provided for each statistical test. For OLS and OLS-Log models, t-values are reported. For MLE-Cen models, Wald's Z-values are reported.

Parameter	Effect	2020				2021				2023				2024			
		Model	Estimate	Test statistic	p-value	Model	Estimate	Test statistic	p-value	Model	Estimate	Test statistic	p-value	Model	Estimate	Test statistic	p-value
ext-NO ₃	VWC		17.156	2.470	0.013		-6.787	-2.890	0.004		-	-	-		-0.530	-0.198	0.848
	Fertilization	MLE-Cen	-	-	-	MLE-Cen	-0.507	-1.300	0.193	Insufficient data > LOD	-	-	-	OLS-Log	0.133	0.283	0.785
	Mulching		0.441	1.150	0.250		0.253	0.680	0.496		-	-	-		0.281	0.594	0.569
	Soil temp.		-	-	-		-	-	-		-	-	-		-	-	-
ext-NH ₄	VWC		8.076	1.810	0.085		-0.280	-0.060	0.951		37.278	2.310	0.021		-0.054	-0.014	0.989
	Fertilization	OLS-Log	-	-	-	MLE-Cen	-1.202	-1.640	0.101	MLE-Cen	-1.152	-1.620	0.106	OLS-Log	-0.694	-1.022	0.337
	Mulching		0.188	0.763	0.454		1.306	1.830	0.067		1.157	1.800	0.071		0.865	1.275	0.238
	Soil temp.		-	-	-		-	-	-		0.539	1.640	0.101		-	-	-

*OLS = ordinary least squares regression (untransformed data); OLS-Log = ordinary least squares regression (log-transformed response variable); MLE-Cen = censored regression (maximum likelihood; log-transformed response variable)

* Insufficient data > LOD to perform statistical analyses on extractable PO₄ concentrations

* F-values are presented for OLS and OLS-Log models, while z-values are presented for MLE-Cen models.

Table C.7: Results of statistical analyses performed on nutrient concentrations in pore water collected from the pad and the fen between 2020 and 2024. Statistically significant results are provided in bold typeface. The type of model used (OLS, OLS-Log, or MLE-Cen) is provided for each statistical test. For OLS and OLS-Log models, t-values are reported. For MLE-Cen models, Wald's Z-values are reported.

Parameter	Effect	2020 (June)				2020 (July)				2020 (August, pre-fertilization)				2020 (August, post-fertilization)			
		Model	Estimate	Test statistic	p-value	Model	Estimate	Test statistic	p-value	Model	Estimate	Test statistic	p-value	Model	Estimate	Test statistic	p-value
NO ₃	VWC		-	-	-		-	-	-		-	-	-		-	-	-
	Fertilization	Insufficient data > LOD	-	-	-	Insufficient data > LOD	-	-	-	Insufficient data > LOD	-	-	-	Insufficient data > LOD	-	-	-
	Mulching		-	-	-		-	-	-		-	-	-		-	-	-
	Soil temp.		-	-	-		-	-	-		-	-	-		-	-	-
NH ₄	VWC		-2.570	-0.091	0.929		15.546	1.771	0.110		-1.735	-0.200	0.840		-1.221	-0.120	0.900
	Fertilization	OLS-Log	-	-	-	OLS-Log	-	-	-	MLE-Cen	-	-	-	MLE-Cen	-0.470	-0.560	0.580
	Mulching		-0.641	-1.172	0.271		0.045	0.125	0.903		-0.380	-0.520	0.610		-0.486	-0.680	0.500
	Soil temp.		-	-	-		-	-	-		-	-	-		-	-	-
TON	VWC		-	-	-		-	-	-		-	-	-		-	-	-
	Fertilization	Insufficient data > LOD	-	-	-	Insufficient data > LOD	-	-	-	Insufficient data > LOD	-	-	-	Insufficient data > LOD	-	-	-
	Mulching		-	-	-		-	-	-		-	-	-		-	-	-
	Soil temp.		-	-	-		-	-	-		-	-	-		-	-	-
Cl	VWC		8.094	0.957	0.363		-2.521	-0.369	0.721		-0.899	-0.378	0.715		-1.124	-0.098	0.924
	Fertilization	OLS-Log	-	-	-	OLS-Log	-	-	-	OLS-Log	-	-	-	OLS	-0.074	-0.075	0.942
	Mulching		0.012	0.070	0.946		0.071	0.253	0.806		0.132	0.644	0.537		-0.088	-0.107	0.917
	Soil temp.		-	-	-		-	-	-		-	-	-		-	-	-

*OLS = ordinary least squares regression (untransformed data); OLS-Log = ordinary least squares regression (log-transformed response variable); MLE-Cen = censored regression (maximum likelihood; log-transformed response variable)

* Insufficient data > LOD to perform statistical analyses on pore water concentrations of NO₂, PO₄, and SO₄

* F-values are presented for OLS and OLS-Log models, while z-values are presented for MLE-Cen models.

Table C.7 Continued

Parameter	Effect	2021 (June)				2021 (July)				2022 (June)				2023 (June)				2024 (June)			
		Model	Estimate	Test statistic	p-value	Model	Estimate	Test statistic	p-value	Model	Estimate	Test statistic	p-value	Model	Estimate	Test statistic	p-value	Model	Estimate	Test statistic	p-value
NO ₃	VWC		-0.104	-0.053	0.959		1623.600	1.547	0.160		34.037	1.770	0.077		24.986	0.071	0.945		-7.062	-1.035	0.331
	Fertilization	OLS-Log	-0.322	-1.196	0.266	OLS	406.000	2.695	0.027	MLE-Cen	-0.432	-1.030	0.304	OLS	25.592	1.132	0.295	OLS-Log	0.654	1.323	0.222
	Mulching		0.236	0.892	0.398		222.600	1.764	0.116		-0.085	-0.180	0.858		-16.668	-0.807	0.446		-0.501	-0.983	0.354
	Soil temp.		-	-	-		-	-	-		-0.333	-0.720	0.471		-5.675	-0.466	0.656		-	-	-
NH ₄	VWC		9.615	2.101	0.069		-19.790	-0.022	0.983		10.027	0.359	0.730		-19.427	-3.326	0.013		-5.095	-1.868	0.099
	Fertilization	OLS-Log	0.142	0.223	0.829	OLS	17.600	0.137	0.894	OLS-Log	0.160	0.254	0.807	OLS-Log	-0.253	-0.675	0.521	OLS-Log	-0.034	-0.171	0.869
	Mulching		1.937	3.109	0.014		42.370	0.394	0.704		0.485	0.672	0.523		-0.503	-1.467	0.186		-0.215	-1.052	0.323
	Soil temp.		-	-	-		-	-	-		0.158	0.225	0.828		-0.137	-0.676	0.521		-	-	-
TON	VWC		-0.181	-0.086	0.934		3.213	0.731	0.485		34.482	1.780	0.075		16.836	0.047	0.964		-7.086	-1.041	0.328
	Fertilization	OLS-Log	-0.272	-0.931	0.379	OLS-Log	2.123	3.367	0.010	MLE-Cen	-0.429	-1.010	0.311	OLS	26.413	1.149	0.288	OLS-Log	0.652	1.322	0.223
	Mulching		0.279	0.972	0.360		0.828	1.568	0.156		-0.100	-0.210	0.835		-16.617	-0.792	0.455		-0.506	-0.994	0.349
	Soil temp.		-	-	-		-	-	-		-0.337	-0.720	0.469		-5.752	-0.464	0.657		-	-	-
Cl	VWC		-1.819	-0.802	0.446		-5.275	-1.088	0.308		22.368	1.010	0.346		-1.629	-0.534	0.610		-8.986	-1.909	0.093
	Fertilization	OLS-Log	0.259	0.824	0.434	OLS	1.241	1.783	0.113	OLS-Log	-0.044	-0.089	0.932	OLS-Log	0.074	0.376	0.718	OLS	-0.061	-0.180	0.862
	Mulching		0.096	0.310	0.764		-0.157	-0.270	0.794		-0.679	-1.185	0.275		-0.204	-1.139	0.292		-0.091	-0.259	0.802
	Soil temp.		-	-	-		-	-	-		-0.194	-0.349	0.737		-0.029	-0.275	0.791		-	-	-

*OLS = ordinary least squares regression (untransformed data); OLS-Log = ordinary least squares regression (log-transformed response variable); MLE-Cen = censored regression (maximum likelihood; log-transformed response variable)

* Insufficient data > LOD to perform statistical analyses on pore water concentrations of NO₂, PO₄, and SO₄

* F-values are presented for OLS and OLS-Log models, while z-values are presented for MLE-Cen models.

Table C.8: Mean ($\pm$ standard deviation) nutrient supply ratios (N:P, Fe:P, and Al:P) for peat from the surrounding fen and mineral fill from the residual pad. Ratios were calculated using PRS probe supply rates. The supply of total N is the sum of the supplies of NO_3^- and NH_4^+ . When data were not available (due to nutrient supply rates $<$ LOQ), they are denoted by *NA*.

Year	Month	Substrate	N:P	Fe:P	Al:P
2020	June	Peat	8:1 (5.6)	4:1 (1.9)	24:1 (16.6)
2020	June	Mineral fill	<i>NA</i>	93:1 (78.4)	10:1 (5.4)
2020	August	Peat	8:1 (2.9)	4:1 (2.9)	30:1 (10.2)
2020	August	Mineral fill	<i>NA</i>	91:1 (75.1)	13:1 (7.2)
2021	June	Peat	6:1 (3.6)	7:1 (9.7)	23:1 (12.3)
2021	June	Mineral fill	<i>NA</i>	195:1 (237.2)	10:1 (6.9)
2023	June	Mineral fill	5:1 (2.9)	195:1 (81.0)	13:1 (13.1)

APPENDIX D

Supplementary Figures and Tables for Chapter 5

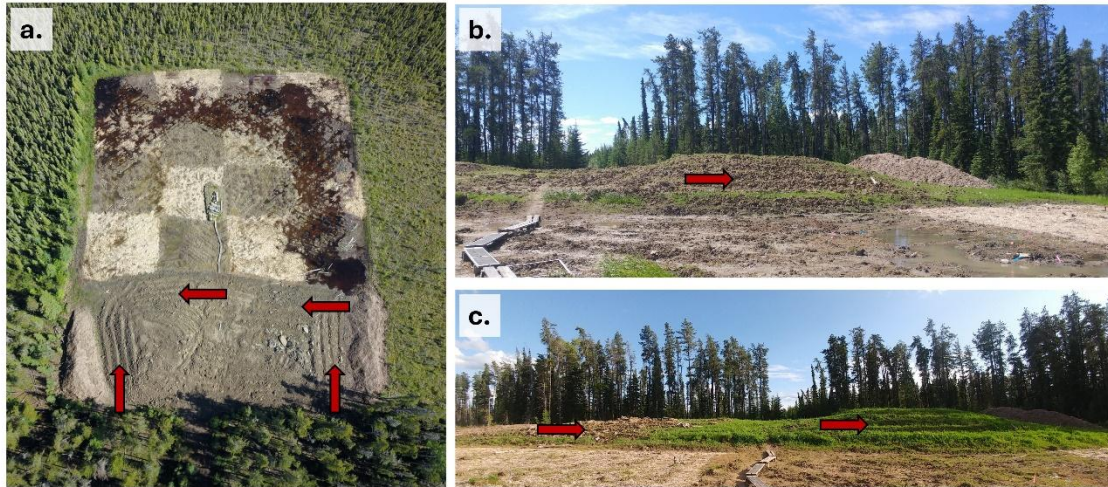


Figure D.1: Photographs of the constructed upland including an aerial overview image taken in June 2020 using a drone (a) and detail images taken from the residual pad (looking south towards the ridge) in June (b) and August (c) 2020. Red arrows highlight examples of swales. Swales were configured such that they ran perpendicularly across the constructed upland slopes. The location of runoff collectors is not shown. Note that the upland slopes were fully vegetated with *Melilotus officinalis* (all areas) and *Scirpus* spp. (moist swales at toeslope positions) by the study periods detailed in this paper (2022 to 2024).

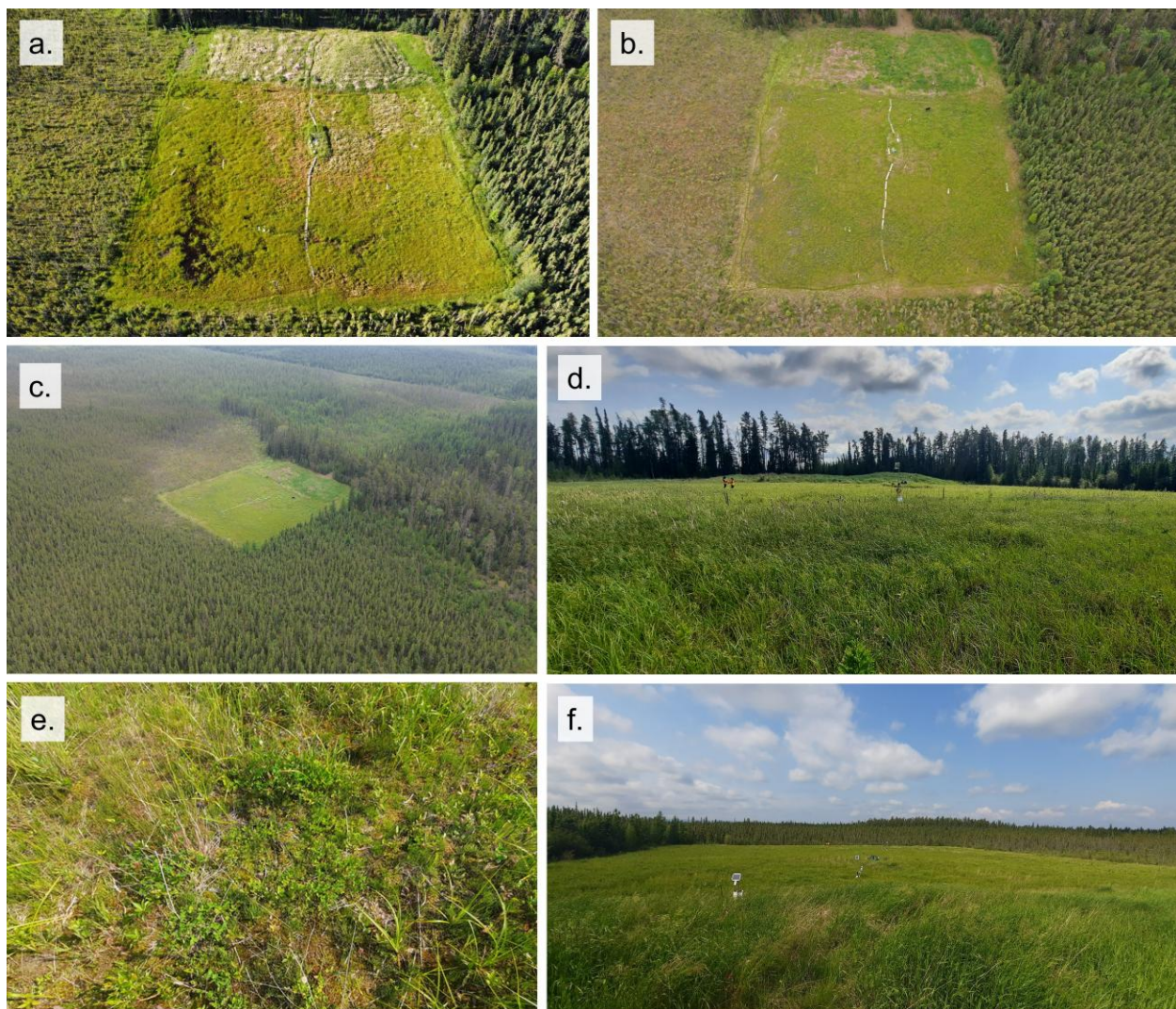


Figure D.2: Aerial photos of the residual pad taken using a drone during wet conditions in 2022 (a) and dry conditions in 2024 (b-c), and detailed ground-level photos of the sedge and moss community present across the residual pad taken in July 2024 (d-f).

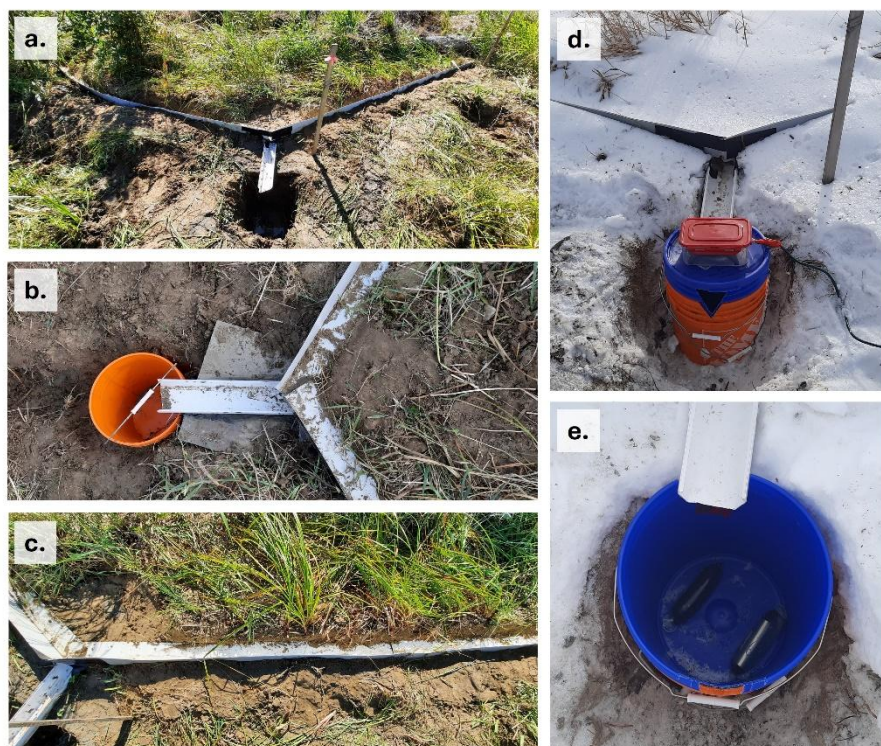


Figure D.3: Photographs of the runoff collectors installed on the constructed upland including an overview of the collection trench layout (a), detail of the eavestrough connecting collection trenches to buckets including the concrete base (b), detail of a collection trench lined with metal angle flashing (c), detail of the air pump location and bucket insulation technique (d), and detail of pressure transducers protected from freezing using latex sheaths filled with ethylene glycol (e). A second pressure transducer was added to each bucket in the first year as a backup in case of freezing, but not in the second season.

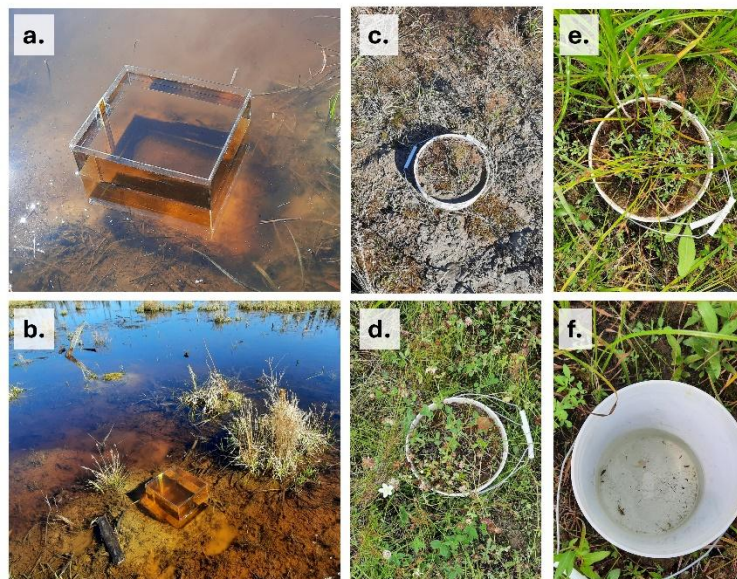


Figure D.4: Photographs of an example of the square acrylic evaporation pans used to quantify actual open-water evaporation rates (a-b), examples of weighing lysimeters used to quantify actual evapotranspiration rates in sparsely vegetated (c) and well vegetated (d-e) areas, and an example of a drainage bucket used to quantify drainage losses from the lysimeters (f). Note that evaporation pans were adjusted prior to each measurement period to ensure that the surface of the pan was closely aligned with the surface of the surrounding water (images shown in panels a and b were taken prior to adjustment).

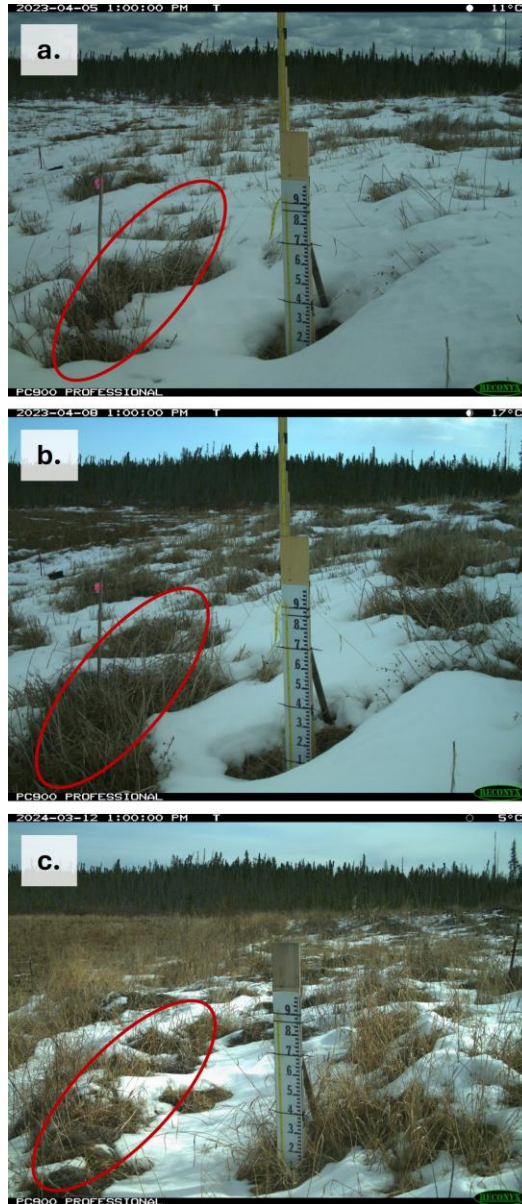


Figure D.5: Timelapse photographs of the snow ruler on the midslope of the constructed upland feature taken on April 5, 2023 (a), April 8, 2023 (b), and March 12, 2024 (c). The approximate location of a high berm feature separating the swales is identified with a red oval in each photo. The berm microtopographic features become snow-free one to four days before the swales.

Table D.1: Summary of surface runoff from the constructed upland feature during the 2023 spring freshet.

Spring 2023	West flume	Centre flume	East flume
Approximate catchment area (m ²)	56.47	47.83	154.88
Flume catchment SWE (m)	0.079	0.080	0.073
Flume catchment SWE (m ³)	4.45	3.83	11.23
Total spring runoff (m ³)	0.011	0.005	0.014
Runoff coefficient (ratio)	0.0025	0.0013	0.0012
Runoff coefficient (%)	0.25	0.13	0.12

Table D.2: Summary of surface runoff from the constructed upland feature during the 2024 spring freshet.

Spring 2024	West flume	Centre flume	East flume
Approximate catchment area (m ²)	56.47	47.83	154.88
Flume catchment SWE (m)	0.043	0.035	0.025
Flume catchment SWE (m ³)	2.45	1.66	3.81
Total spring runoff (m ³)	0.018	0.014	0.005
Runoff coefficient (ratio)	0.0074	0.0082	0.0014
Runoff coefficient (%)	0.74	0.82	0.14

Table D.3: Priestley-Taylor α coefficients developed for each plot type (WO and NO) in 2022 and 2023. The R^2 and p values for the linear regressions between ET_a and ET_{eq} used to develop each α value are also presented.

	2022			2023		
	α	R^2	p	α	R^2	p
WO	1.06	0.50	< 0.001	1.51	0.50	< 0.001
NO	0.99	0.51	< 0.001	1.15	0.68	< 0.001