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Hydrologic assessment of mineral substrate suitability for true moss initiation in a boreal peatland undergoing restoration

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ABSTRACT

Tens of thousands of oil and gas well pads have been constructed in peatlands on the North American Western Boreal Plain. The introduction of true mosses directly onto residual mineral substrates left following the partial removal of well pads may present a means of re-establishing peatland ecosystem function on these sites post-decommissioning. Accordingly, an assessment of mineral substrate moisture dynamics was undertaken on a residual well pad on the Western Boreal Plain to determine whether requisite conditions for the establishment of true mosses would be maintained throughout the growing season. The results indicate that substrate moisture conditions were most favourable for true moss establishment when the water table was maintained within 6 cm of the mineral surface of the residual well pad. Such conditions were most frequently observed along edges of the pad receiving direct groundwater inputs from an adjacent peatland, representing an area covering just under half of the pad. However, water table variation was high in interior areas of the pad which were hydrologically disconnected from the adjacent peatland. Here, substrate moisture dynamics were not optimized for true moss establishment late in the season. Mosses introduced to these areas faced a considerable risk of desiccation, which was not directly alleviated by the application of a straw mulch. These findings suggest that the partial removal technique has the potential to create requisite moisture conditions for true moss establishment, but there is a need to enhance subsurface hydrological connectivity across residual pads in future implementations.

1. Introduction

Peatlands are a prominent type of ecosystem on the Canadian Western Boreal Plains, 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 (Harris et al., 2021), 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., 2020a).

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 (Osko 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

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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., 2020a), especially given that over 36,000 ha 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 and 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 and 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., 2021; 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., 2021). 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 (Rochefort et al., 2003; Lemmer et al., 2022). 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 Boral Plain (Vitt, 2014) 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 (Osko et al., 2018; Volik et al., 2020a). 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, 2021a). 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 (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 m 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; Gauthier et al., 2018). However, the effect of this treatment remains unclear, as soil water availability was not limiting in the field-scale trial (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., 2000; 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.

2. Materials and methods

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; Fig. 1a). The 0.8 ha pad was originally constructed adjacent to an upland ridge in a moderate rich fen (pH = 6.4, electrical conductivity = 110 μS cm^{-1}) in 1991 through the placement of calcareous loamy sand mineral fill (Ca content >13,000 mg kg^{-1}) excavated from an upland borrow pit (Alberta Energy Regulator, 2022). The surrounding natural fen was characterized by *Picea mariana, Rhododendron groenlandicum, Rubus chamaemorus, Salix* spp., *Carex aquatilis, Pleurozium schreberi, Aulacomnium palustre,* 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 (Fig. 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 (Fig. 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 (Fig. 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 (Fig. 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

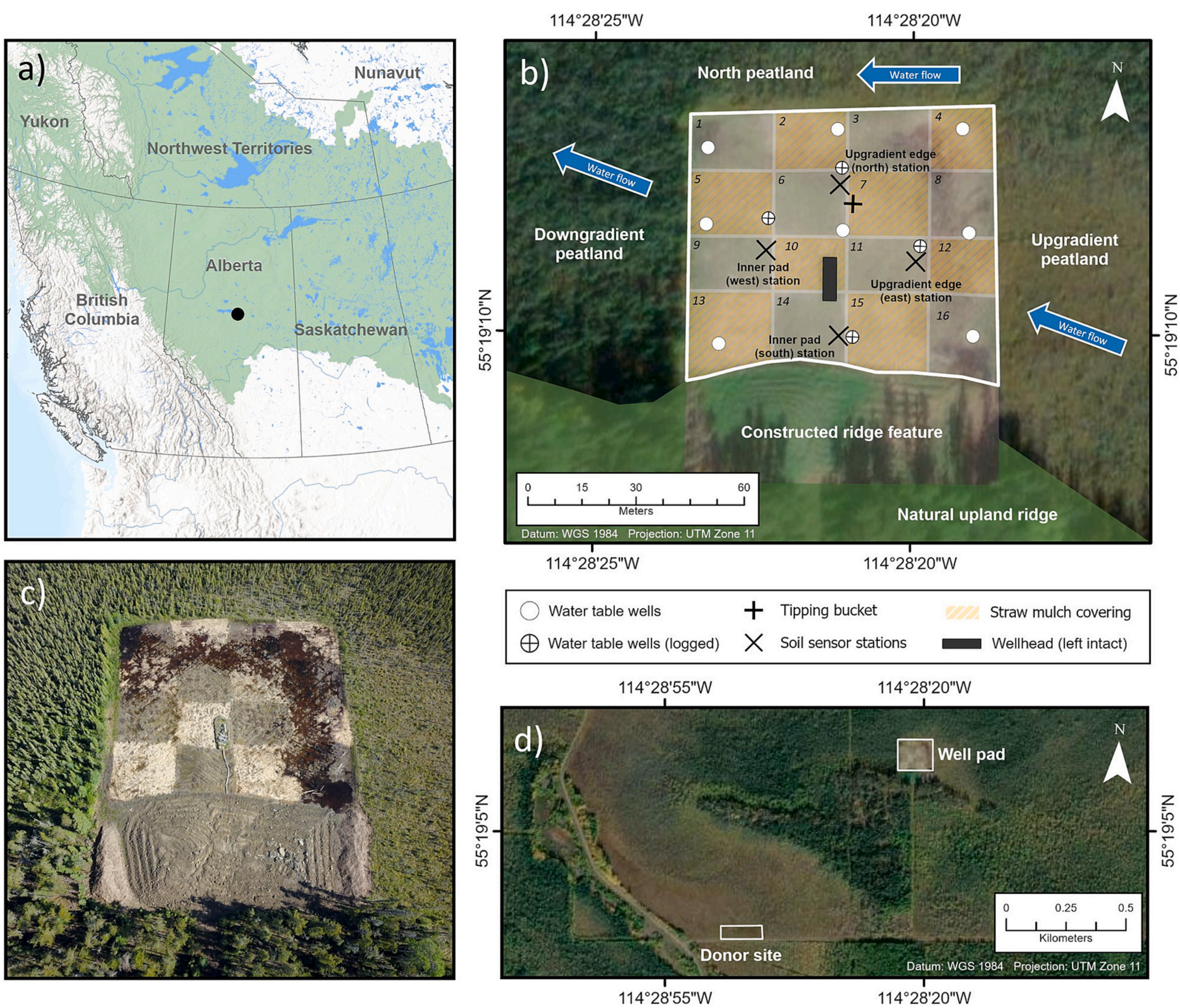

Fig. 1. Location of the study site in the boreal region of western Canada (a) and detailed site diagram including the boundaries of the constructed ridge and residual well pad, treatment plot boundaries, plot numbers, and monitoring well and soil sensor locations (b). Blue arrows indicate the approximate direction of shallow groundwater flow through the surrounding peatland (see McKinnon et al., 2024). 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. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

excavator (see Bird and 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 μS cm^{-1}) within 1 km of the site (Fig. 1d) using an excavator. The donor mosses were transported to the residual pad (pH = 6.7, electrical conductivity = 698 μS cm^{-1}) using a trailer and spread across its surface using a skid steer at a 1:7 aerial ratio (see Bird and Xu (2021c) 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 warnstorfii*, 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 (Fig. 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 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).

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 cm^3 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 Aikaike 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 and 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) (Fig. 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.

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. (Fig. 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 (Fig. 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).

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.

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. Results

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, 2024b), 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 (Fig. 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 (Fig. 2).

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 (Fig. 3). Modelled VWC ranges for a given Ψ value indicate that optimal moss-water availability (i.e., Ψ approximately > -10 mb; Graf and Rochefort, 2010) is only likely to occur when VWC is ≥ 0.39 m^3m^{-3} (+/− 0.02 m^3m^{-3}; green shaded region in Fig. 3). Notably, this is also the predicted VWC at saturation (Supplementary Table 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 and Whitehead, 2001) is predicted to occur when the near-surface VWC drops below approximately 0.28 m^3m^{-3}, although this prediction was less precise (+/− 0.07 m^3m^{-3}), with minimum and maximum modelled values of 0.20 and 0.34 m^3m^{-3}, respectively (Fig. 3, Supplementary Table 1). Fitted model parameters are summarized in Supplementary Table 2.

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-

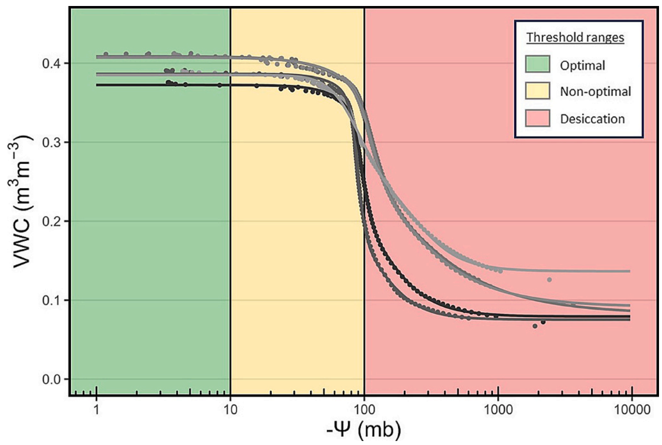

Fig. 3. Soil water retention datapoints and fitted soil water characteristic curves (bimodal-constrained van Genuchten model). Vertical solid lines denote approximate soil water potential (Ψ) indicator thresholds (−10 and − 100 mb; Graf and Rochefort, 2010; Price and Whitehead, 2001). When datapoints 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). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

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 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 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 1). Specifically, post hoc analyses indicated that a slight positive effect of mulching on VWC was greater at lower microsites 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 1).

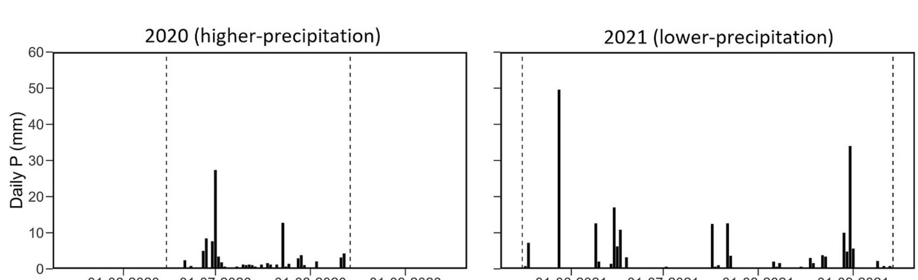

Fig. 2. Daily precipitation (*P*) for the 2020 (15 June to 14 August) and 2021 (16 May to 14 September) study periods. Dashed lines indicate the start and end of the study period within each season (no site-level data was collected for dates outside of either study period).

Table 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 (α = 0.05) are provided in bold text.

Fixed Effect	2020 (higher-precipitation season)		2021 (lower-precipitation season)	
	F-value	p-value	F-value	p-value
Volumetric water content				
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
Soil water potential *				
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.

Linear regressions between DWT and manually measured Ψ indicate that moss water availability was generally only optimized (i.e., Ψ > −10 mb) at low microtopographic positions when the WT was within approximately 0.06 m of the surface (Fig. 7a). Of note, however, optimal moisture conditions were also occasionally observed during periods when the WT was as deep as 0.25 m b.g.s. (Fig. 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 0.02 m of surface water (DWT values to the left of the y-axis on Fig. 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 Fig. 7c). Comparatively, exceedance of the −100 mb desiccation risk threshold is predicted to occur once the WT exceeds approximately 0.78, 0.67, and 0.54 m b.g.s. for the low, intermediate, and high positions, respectively (DWT values measured against the ground surface at low positions; Fig. 7a-c).

3.4. Water table dynamics

Based on the relationship between DWT and manually measured Ψ (Table 1 and Fig. 4), treatment plots that had a seasonal mean manually-measured WT closer to the surface than 0.06 m 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 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 (Fig. 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 0.06 m b.g.s. for between 91 % and 94 % of the study

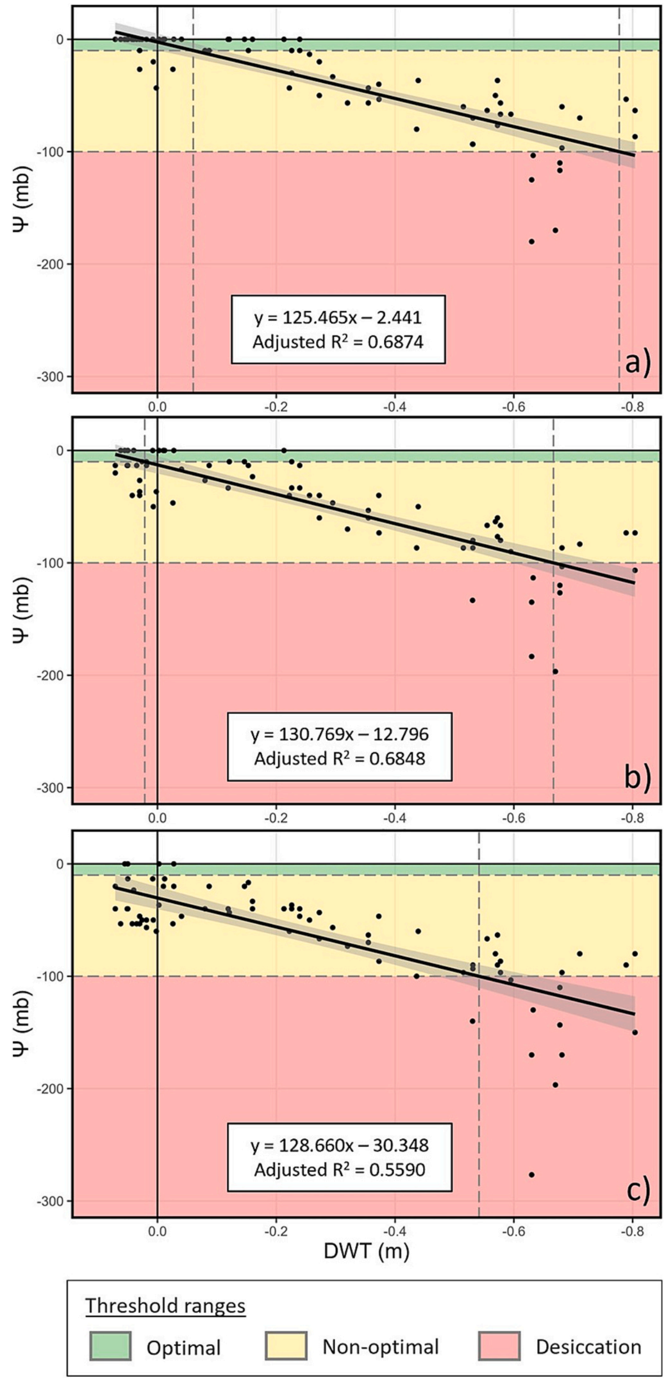

Fig. 4. Linear regressions between depth to water table (DWT) and soil water potential (Ψ) for low (a), intermediate (b), and high (c) microsites. Shading denotes the Ψ ranges associated with optimal moisture availability (0 to −10 mb; green), non-optimal moisture availability (−10 to −100 mb; yellow), 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). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

period that year (Fig. 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; Fig. 6a). As such, this was the only area where there was a high

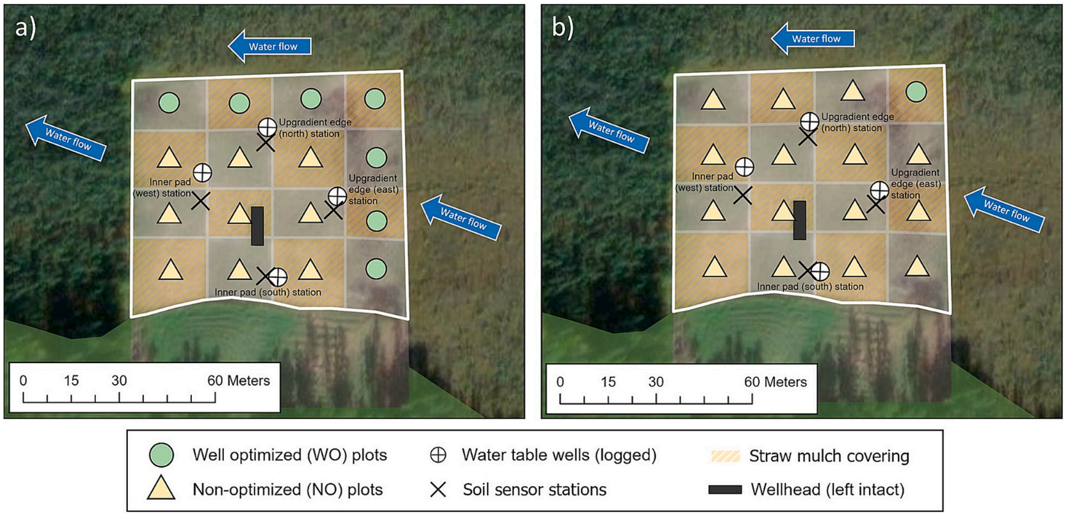

Fig. 5. Plot hydrological classifications based on mean manually-measured depth to water table for the 2020 (higher-precipitation year; a) and 2021 (lower-precipitation year; b) study periods. Well optimized (WO) plots in a given year had a seasonal mean manually-measured water table closer to the surface 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., 2024). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

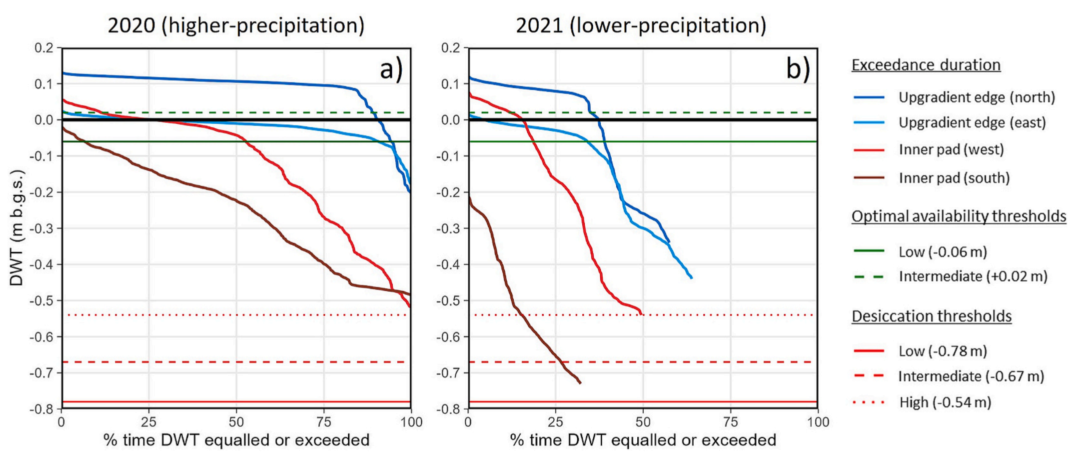

Fig. 6. Depth to water table (DWT) duration curves for the 2020 (a) and 2021 (b) study periods. Plots illustrate the proportion of the study period that DWT exceeded (fell below) any given value on the y-axis. Note that the lengths of the study periods were not equal. For the 2021 study period, logger 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.

likelihood that moisture conditions would be optimal for true moss establishment at intermediate or high microsites.

Conversely, in the second, lower-precipitation study period, only one plot was classified as being WO for true moss establishment (Supplementary Table 3). This plot was located in the northeast corner of the residual pad adjacent to both the upgradient and north pad edges (Fig. 5b), while all other plots on the pad were classified as being NO that year (Supplementary Table 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 (Fig. 7d).

During the first, higher-precipitation year, nine plots had a seasonal mean WT deeper than 0.06 m b.g.s. (Supplementary Table 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 (Fig. 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; Fig. 5) was poorly regulated and frequently dropped below 0.06 m b.g.s. even in the early season (Fig. 7b). Reflecting this high degree of variability, the WT at the south and west logged wells in NO portions of the pad remained above 0.06 m for only 7 % and 53 % of the first study period, respectively (Fig. 6a). Despite this, the WT did not drop below 0.54 m b.g.s. at either the interior or downgradient logged wells at any time (Fig. 6a).

Reflecting the high proportion of plots classified as being NO in the second, lower-precipitation season, the WT was within 0.06 m 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 (Fig. 6b). At the downgradient logged well, the WT dropped below 0.06 m 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 (Fig. 7d). As such, the WT was only within 0.06 m of the surface at the downgradient logged well for 19 % of the study period, while it remained below 0.06 m b.g.s. for the entire study period at the inner logged well (Fig. 6b).

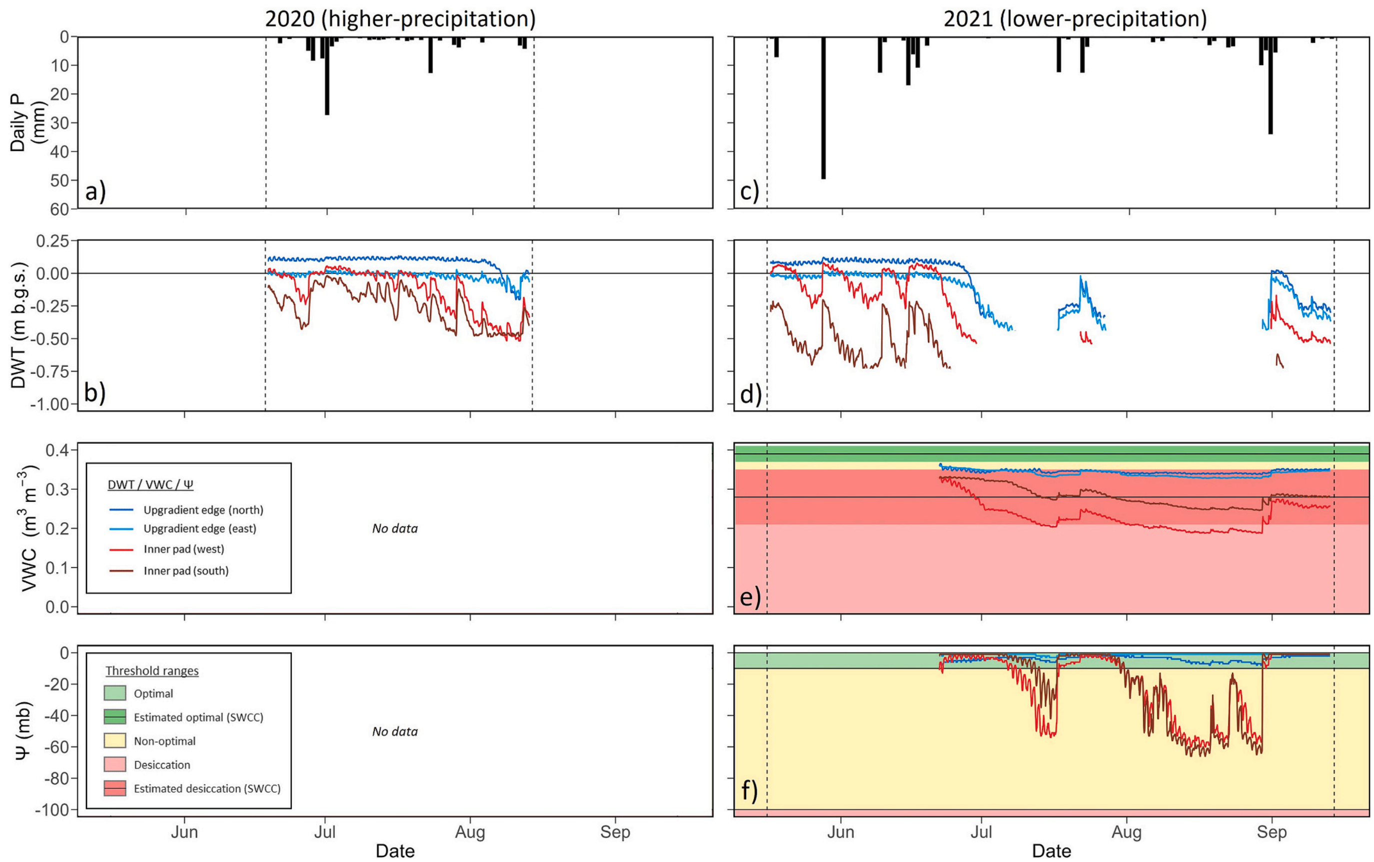

Fig. 7. Comparison of daily site-measured precipitation (P) (a, c), depth to water table (DWT) referenced against low microsite heights (b, d), and soil volumetric water content (VWC) (e) and soil water potential (Ψ) (f) as measured at intermediate microsite heights on the residual pad for the lower-precipitation 2021 study period. On the VWC plot, horizontal lines denote the mean predicted VWC values associated with optimal moss-water availability (0.38 m^3m^3), and the mean predicted desiccation threshold (0.27 m^3m^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.

Furthermore, the WT at the inner and downgradient logged wells fell below 0.54 m b.g.s. for 84 % and 51 % of the second season, respectively (Fig. 6b), while DWT exceeded −0.67 m at the inner well for 73 % of the study period (Fig. 6b). Unfortunately, missing logger data precluded determination of the occurrence of exceedances of WT thresholds at the upgradient and north wells (Fig. 6b). However, DWT was considerably greater at those wells for most of the second study period than observed during the first study period (Fig. 7d).

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 Fig. 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; Fig. 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; Fig. 5a).

Owing to a shallow, stable WT in the 2020 (higher-precipitation) study period (Fig. 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 (Fig. 8a-b). Low microsites were significantly wetter than high microsites across the entire pad on those dates (Fig. 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 (Fig. 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 (Fig. 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 (Fig. 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; Fig. 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 (Fig. 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 (Fig. 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 (Fig. 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

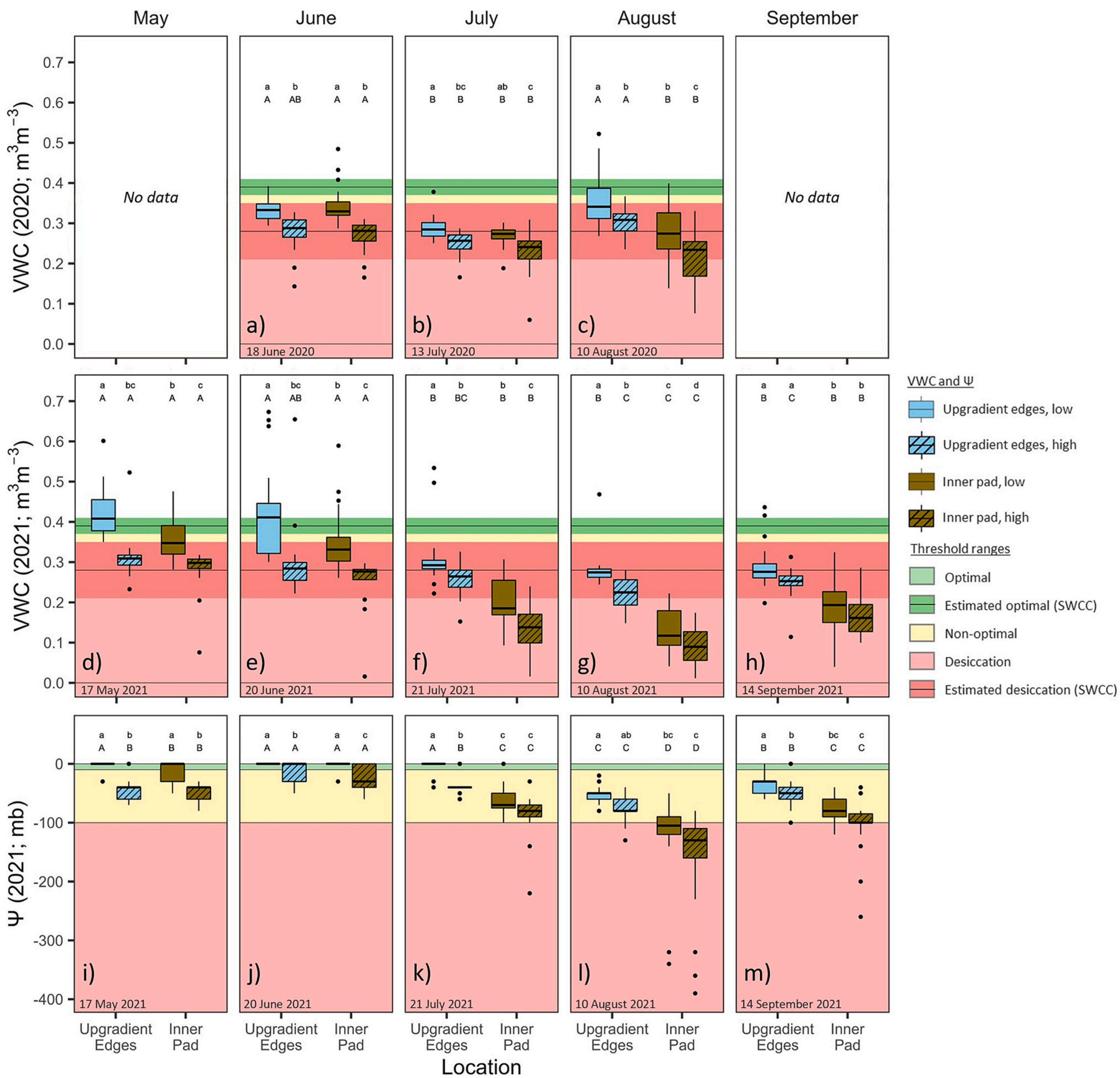

Fig. 8. Boxplots of VWC and Ψ at low and high microsites in plots along upgradient edges (classified as WO in 2020) and within interior areas (classified as NO in 2020) of the residual pad. On VWC plots, horizontal lines denote the predicted mean VWC at Ψ values of −10 mb (optimal moss-water availability threshold; 0.38 m^3m^{-3}), and − 100 mb (desiccation threshold; 0.27 m^3m^{-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.

mb), while Ψ values at high microsites normally fell below −10 mb, but above the −100 mb desiccation threshold (Fig. 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 (Fig. 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 (Fig. 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 (Fig. 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 (Fig. 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; Fig. 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; Fig. 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 characterized by little rainfall (Fig. 7f), while half-hourly VWC values there frequently fell below the mean predicted desiccation threshold (Fig. 7e).

Considering the manual dataset, significant drying occurred at nearly all microsites between June and July during the second study period (Fig. 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 (Fig. 8f). In contrast, VWC generally remained within the predicted desiccation threshold range along the east and north pad edges (Fig. 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 (Fig. 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 (Fig. 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 (Fig. 8g) and Ψ falling below −100 mb in interior areas (Fig. 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 (Fig. 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 (Fig. 8). This was accompanied by a sharp increase in both half-hourly VWC and Ψ values in interior areas of the pad (Fig. 7e-f), although all manual measurements remained below their optimal ranges (Fig. 8).

4. Discussion

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 (Fig. 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 (Fig. 7), it appears that retention of water within the residual sandy substrate was insufficient to maintain optimal conditions for moss establishment (i.e., Ψ > −10 mb) during periods when the WT was low. Notably, many well pads on the Western Boreal Plain are constructed out of clay substrates (Osko 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, 2004).

4.2. Substrate water availability assessment limitations

Some inconsistency was observed between the Ψ and VWC datasets with respect to moss-water availability thresholds (Figs. 7 & 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 and Or, 2005), Ψ may be a more accurate indicator of in-field water availability. Furthermore, while the hydrological threshold of Price and Whitehead (2001) (*Sphagnum* desiccation at Ψ < −100 mb) and that based on the results of Graf and Rochefort (2010) (optimal true moss establishment at Ψ > −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 (Ψ < −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.

4.3. Substrate water availability and implications for true moss establishment

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 0.06 m of the surface (referenced against the surface elevation at low microsites; Fig. 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 (Fig. 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 m^3m^3) there and frequently overlapped with the predicted optimal range (0.37 to 0.41 m^3m^3; Fig. 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 (Fig. 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 CO_2 diffusion or contamination of moss physical structures with bacteria and algae (Busby and Whitfield, 1977; 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 (Fig. 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.

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 (Fig. 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 Vicherová, 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 warnstorfii*) require a hardening period over which desiccation tolerance develops (Hájek and Vicherová, 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 (Fig. 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.

4.4. Effect of straw mulch

Given the importance of atmospheric water losses in peatland water budgets under both current (Volik et al., 2020b) 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 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 m^3m^{-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, 2015). 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, 2015). In turn, this increased capillary flow would sustain greater evaporation rates, which might be reduced through the application of straw mulch.

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 (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 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 (Osko et al., 2018).

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 m^3m^{-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.

CRediT authorship contribution statement

MEM, RMP, FCN, BX, SJK, and MB contributed to study conceptualization, methodology, and investigation. RMP and FCN provided project supervision. FCN and BX acquired funding and provided project administration. MEM curated and conducted formal analysis on study data and wrote the original draft of the manuscript. RMP, FCN, BX, and SJK provided field and laboratory resources. RMP, FCN, BX, SJK, and MB contributed to review and editing of the manuscript.

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Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecoleng.2025.107615.

Data availability

The data used to support the findings contained herein are available from the corresponding author upon reasonable request.

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