From bare peat to *Sphagnum* cover: The success of *Sphagnum*-fragment and straw-mulch application for initiating peatland restoration

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SUMMARY

Restoring degraded peatlands begins with hydrological restoration, but this alone may not fully re-establish key plant species such as *Sphagnum* mosses. To address the need for improving restoration outcomes, this study investigates the short-term effects of active revegetation on bare peat using a modified Moss Layer Transfer Technique (MLTT). Two treatment types were tested: *Sphagnum* fragments with mulch, mulch-only, and untreated bare peat as a control. Experiments were conducted across three locations in Norway to assess the recovery of *Sphagnum* cover and the establishment of other plant functional types over three years. Results showed a significant increase in *Sphagnum* cover in the *Sphagnum* treatment plots, reaching 65.6 % (± 3.5 SE) on average compared to 4.6 % (± 1.4 %) in mulch-only plots and 2.0 % (± 0.6 %) in controls, demonstrating the efficacy of active revegetation. The straw mulch treatment alone did not significantly enhance *Sphagnum* re-establishment, though some plant functional types, particularly mosses other than *Sphagnum* and *Racomitrium* as well as evergreen *Ericaceae*, benefited slightly. *Sphagnum* re-establishment success varied among the four species used, with *S. angustifolium* showing the highest establishment rates. Overall, our findings emphasise that active revegetation accelerates *Sphagnum* recovery, which is crucial for reducing greenhouse gas emissions and stabilising peatland hydrology. The use of straw mulch alone is of limited benefit for rapid *Sphagnum* recovery but may provide a stabilising microclimate. These results underline the need for integrating active revegetation with hydrological restoration for effective peatland restoration.

KEY WORDS: boreal, MLTT, Norway, peatmoss, revegetation

INTRODUCTION

The UN Decade on Ecosystem Restoration (2021–2030) and the Kunming-Montreal Global Biodiversity Framework emphasise restoration as essential for tackling biodiversity loss, with a target to have 30 % of degraded ecosystems under effective restoration by 2030 (CBD 2022, UNEP/FAO 2023). As restoration efforts increase globally, ensuring effective and sustainable practices guided by clear goals and measurable indicators is crucial for the long-term recovery of ecosystems (Gann *et al.* 2019). Restoring degraded peatlands is increasingly recognised as a key conservation strategy (Leifeld & Menichetti 2018, Günther *et al.* 2020) and is also an important natural climate solution to meet the Paris Climate Agreement targets (Griscom *et al.* 2017, Cook-Patton *et al.* 2021, Strack *et al.* 2022). Peatlands cover 3–4 % of Earth's land surface and are unique ecosystems that support specialised biodiversity while storing one-third of global soil carbon (UNEP 2022). They are predominantly located in the boreal and temperate regions, where about 70 % of the world's peatland carbon is found (Hugelius *et al.* 2020, UNEP 2022). However, extensive destruction of peatlands due to human activities like drainage for agriculture and forestry, infrastructure development, and peat extraction has caused not only severe ecological disruption and habitat loss, but also significant carbon loss (estimated at 4 % of anthropogenic emissions (UNEP 2022)), contributing to climate change (Leifeld & Menichetti 2018, Loisel *et al.* 2021). Restoration efforts may reverse peatland degradation (Haapalehto *et al.* 2017), reduce greenhouse gas (GHG) emissions (Günther *et al.* 2020), and restore their role as effective carbon sinks (Nugent *et al.* 2018, Strack & Zuback 2013).

Sphagnum species are key ecosystem engineers in boreal peatlands and are responsible for forming approximately 50 % of the peat deposits in these regions (Gorham 1991, Turetsky 2003). They play a central role in CO_2 uptake, with their partially decomposed remains accumulating under water-saturated conditions to form peat, thereby facilitating long-term carbon sequestration (Pacheco-Cancino *et al.* 2024). Successful re-establishment of *Sphagnum* species strongly indicates effective restoration, signalling a return of suitable hydrological conditions and resumed peat accumulation (Couwenberg 2011, González *et al.* 2013, González & Rochefort 2014, Kyrkjeeide *et al.* 2024). The re-establishment of *Sphagnum* also plays a critical role in regulating the emissions of methane (CH_4). Unlike graminoid species, that promote CH_4 release through aerenchyma (air-filled tissues) from roots to stomata, *Sphagnum* supports methanotrophic bacteria that reduce CH_4 emissions by oxidising it in the aerobic peat layers (Larmola *et al.* 2010, Putkinen *et al.* 2018, Wen *et al.* 2018). Emissions of CH_4 are largely influenced by water-table depth, but when the water table is high the vegetation composition becomes the dominant regulating factor (Couwenberg 2011, Tiemeyer *et al.* 2020, Evans *et al.* 2021). Thus, ensuring the successful re-establishment of *Sphagnum* is not only a sign of ecological recovery but is also important for minimising greenhouse gas emissions during peatland restoration (Strack *et al.* 2022).

Peatland restoration primarily involves hydrological measures - rewetting - through reprofiling, blocking drainage ditches, and stopping water-pumping, to create suitable conditions for peatland specialist species (Rochefort & Lode 2006, Caporn *et al.* 2018). In afforested peatlands, restoration may also require tree removal (Andersen *et al.* 2017). However, restoring hydrology alone is often insufficient to fully recover ecosystem function and biodiversity, particularly in areas where vegetation has been removed or severely altered, as the natural recovery of *Sphagnum* remains a significant challenge (Girard *et al.* 2002, Karofeld *et al.* 2016, Zając *et al.* 2018, Zarzycki *et al.* 2022). In particular, bare peat provides harsh microclimatic conditions and is vulnerable to erosion, hindering the re-establishment process (Groeneveld & Rochefort 2002, Warburton 2003, Lavoie *et al.* 2005, Wilson *et al.* 2007).

Where hydrological restoration alone cannot achieve full ecosystem recovery, active revegetation measures become essential. One of these, the Moss Layer Transfer Technique (MLTT; Rochefort & Lode 2006), is frequently used alongside hydrological restoration to re-establish *Sphagnum* cover in North America (Waddington *et al.* 2010, González & Rochefort 2014, Gauthier *et al.* 2018, Lazcano *et al.* 2018, Hugron *et al.* 2020). The MLTT involves the collection of the uppermost layer (top 10 cm) of vegetation from a donor peatland, predominantly covered by *Sphagnum* species, which is then cut into smaller fragments and transferred onto the target peatland. This technique relies on the regenerative capacity of *Sphagnum*, which can establish from smaller fragments under suitable microclimatic conditions (Cronberg 1992, Campeau & Rochefort 1996). Covering *Sphagnum* fragments with straw mulch or other materials is commonly recommended (Quinty & Rochefort 2003) to maintain favourable growth conditions by stabilising moisture and temperature levels and reducing environmental stressors (Price *et al.* 1998, Petrone *et al.* 2004). However, its use in Europe, especially in northern regions, has been more limited (but see Karofeld *et al.* 2017 and Purre *et al.* 2020). The application has also been focused on former large-scale peat extraction sites. Other active revegetation measures for *Sphagnum* have been developed in the UK targeting degraded and eroded blanket bogs, with the reintroduction of *Sphagnum* encapsuled in beads, gel or clay, or as plant plugs or clumps, with mixed effects (Wittram *et al.* 2015, Caporn *et al.* 2018, Benson *et al.* 2021, Pilkington *et al.* 2021, Roberts *et al.* 2022).

In Norway, peatland restoration has mostly used a rewetting approach, while few revegetation methods have been applied (Kyrkjeeide *et al.* 2024). Most Norwegian peatland restoration sites differ from the large former peat extraction sites in North America and Central Europe and the vast areas of heathland and bare peat in the degraded blanket bogs of the UK in that they have mainly been drained for forestry, are often smaller and more varied, and consist of a mosaic of ditches, patches of bare peat and remaining vegetation. Whereas the distance to viable *Sphagnum* populations may be large in the former locations, populations of *Sphagnum* species may still be found in close proximity to the restoration sites in Norway. While natural revegetation of smaller bare peat areas can progress more quickly compared to fully stripped sites, it still tends to be a slow process even when propagules are available nearby (Campeau *et al.* 2004, Xu *et al.* 2022).

This study seeks to assess the efficacy of using principles of the MLTT for active revegetation measures to restore *Sphagnum* cover on patches of bare peat in Norway. We adapted the MLTT by selectively collecting specific *Sphagnum* species and cutting these into smaller fragments that were spread onto bare peat patches. Targeting specific species for revegetation can influence the speed and stability of *Sphagnum* re-establishment and the broader recovery of peatland functions (Rochefort *et al.* 2003, Chirino *et al.* 2006). *Sphagnum* species exhibit a range of ecological strategies; some are pioneers, colonising exposed areas rapidly, while others have specialised niches along the water-table gradient, thriving only under specific moisture conditions (Hayward & Clymo 1983, Økland 1989, Sundberg & Rydin 2002).

To assess whether improving microclimatic conditions alone could facilitate spontaneous *Sphagnum* recolonisation, we also included a straw-mulch-only treatment. Since remnant *Sphagnum* populations are often present near forestry-drained peatlands, propagules may be available for natural recolonisation, but their establishment can be limited by desiccation and surface instability. Applying straw mulch without *Sphagnum* fragments allowed us to test whether enhancing moisture retention and reducing environmental stressors could support passive recovery, providing insight into the necessity of direct *Sphagnum* introduction versus indirect facilitation of natural revegetation.

The aim of this study was to investigate the short-term effects (3 years post-implementation) of two restoration measures on the recovery of *Sphagnum* cover on bare peat patches: (1) by the use of active revegetation with the spreading of fragments of four locally abundant *Sphagnum* species with varied traits, covered with straw mulch; and (2) by utilising a more simplified revegetation method by covering bare peat with straw mulch only. Vegetation recovery was assessed using vegetation cover. Additionally, we explored whether the addition of straw mulch enhanced the cover of other plant functional types (PFTs) besides *Sphagnum*, potentially due to changes in soil nutrient availability and/or near-surface soil temperature. We also examined whether there were establishment differences between the *Sphagnum* species applied.

The questions asked were: 1) does the application of *Sphagnum* fragments speed up the recovery of a *Sphagnum* cover on bare peat compared to natural revegetation? and 2) will a cover of straw mulch initiate a faster recovery of *Sphagnum* than no measures?

METHODS

Study sites and experimental setup

Our revegetation experiment was established at three locations in Trøndelag, Norway: Vestersetermyra (VSM) in Namsos municipality, along with Stormyra (HM) and Bukkastakkmyra (BSM) within the Høydalsmoan Nature Reserve in Åfjord municipality. The sites had been disturbed by power mast installation (at VSM in 2018) and afforestation with ditching (at HM and BSM during the 1970s). At VSM the peat was temporarily removed and subsequently replaced after the installation was completed. Restoration involving logging of trees and rewetting was conducted in 2019 at HM and in 2020 at BSM. All locations had a heterogeneous coverage of vegetation with sections of bare peat at establishment of the experiment. Detailed site descriptions (peat depths, climate and water-table conditions) are provided in Table 1.

The revegetation experiment was set up in spring 2020 at HM and VSM, and in spring 2021 at BSM. Each experimental block (4 m²) contained the following treatments: (1) *Sphagnum* with mulch (2 m²); (2) mulch only (1 m²); and (3) control (bare peat, 1 m²). Four blocks made up a sublocation, with three sublocations at HM and BSM, and two at VSM (in total, 32 blocks; Figure 1). We collected and applied *Sphagnum* and mulch according to the Canadian standards (Campeau & Rochefort 1996, Quinty & Rochefort 2003). *Sphagnum* was collected by hand to no deeper than 10 cm, cut into fragments ~5–20 mm in size, and applied at a 1:10 donor-to-recipient-area ratio. Straw mulch was applied at ~2000–3000 kg ha^{-1} and secured with fruit netting for the first season (Figure 1 shows photo of the setup). We did not add phosphorus, as used in the MLTT, due to logistical challenges.

We reintroduced a mix of four *Sphagnum* species (*S. angustifolium*, *S. tenellum*, *S. capillifolium*, *S. papillosum*). The selection of species was based on three criteria: 1) locally abundant; 2) representative of a variety of *Sphagnum* subgenera (*S. angustifolium* and *S. tenellum* belonging to subgenus *Cuspidata*, *S. capillifolium* to subgenus *Acutifolia* and *S. papillosum* to subgenus *Sphagnum*); and 3) potentially able to grow on bare peat (Økland 1989, 1990; Rochefort *et al.* 2003, Tuittila *et al*. 2003, Waddington *et al.* 2003a).

Data collection

Vegetation analyses were conducted, on centrally placed 0.5 m × 0.5 m plots within each treatment unit to avoid edge effects, in May (before treatments were applied) in Year 0 and in September for the subsequent two years. We visually estimated the percentage cover of PFTs, mulch, litter and bare peat. Details on taxa included in PFTs are presented in Table A1 in the Appendix. Cover was evaluated separately for the field layer and the ground layer since bryophytes can be present beneath vascular plants like *Ericaceae* spp. and *Eriophorum* spp., resulting in total cover exceeding 100 %. Non-*Sphagnum* vegetation was summed up as “other vegetation”. To establish a reference point, we examined adjacent intact peatland, analysing two 10.5 m transects with five evenly spaced plots, using the same vegetation analyses as in the treatment plots. Futhermore, *Sphagnum* treatment plots were divided into 16 subplots of area 156.25 cm^2, with *Sphagnum* cover analysed in four subplots (primarily corner squares; see Figure 1, Block 3). All *Sphagnum*

Table 1. Detailed location information summarising peatland description, air temperature, precipitation and water table conditions (WT).

	Stormyra (HM)	Vestersetermyra (VSM)	Bukkastakkmyra (BSM)
Location information			
Coordinates	64°02'16"N, 10°25'56"E	64°16'26"N, 11°34'30"E	64°02'06"N, 10°26'30"E
Mire type	Flat ombrotrophic bog to poor fen, two mire massifs	Large mire complex with ombrotrophic edge and open poor fen	Sloping poor fen
Elevation (m AMSL)	107.5–111.0	118–120	109.5–118.5
Peat depth (m)	0.6–2.0	1.4–2.6 *	2.0–2.5
Study period	2020–2022	2020–2022	2021–2023
Climate - Normal (1991–2020)			
Annual mean temperature (°C)	6.2	5.0	6.2
Growing season mean temp. (°C)	11.4	10.6	11.4
Total annual precipitation (mm)	1700	1350	1700
Climate - Study Period			
Mean annual precipitation (mm)	1850	1590	1850
Mean monthly precipitation (growing season, mm)	141.4	96.5	141.4
Establishment Year Precipitation			
Days with ≥2 mm (growing season)	73	61	51
Days with ≥2 mm summer (June | July | August)	2 | 16 | 15	3 | 14 | 10	10 | 10 | 9
Max consecutive dry days (<2 mm) summer (June | July | August)	21 | 4 | 4	21 | 8 | 6	11 | 8 | 11
Water Table (WT)			
Min. | Mean | Max. WT (m)	-0.31 | -0.06 | 0.08	-0.61 | -0.22 | 0.08 **	-0.33 | -0.07 | 0.00 ***
Max consecutive days < -20 cm WT	3	153 **	4
Max total days < -20 cm WT (one season)	9	153 **	5

* With high uncertainties, as the peat had been mixed with deeper layers of mineral soils.

** Major differences within location, e.g. VSM2 had only max 35 consecutive days, and 37 in total.

*** Data was adjusted to set the highest WT at the surface, as the original data showed an unrealistically low peak levelling below the surface at 2 of 3 WT stations.

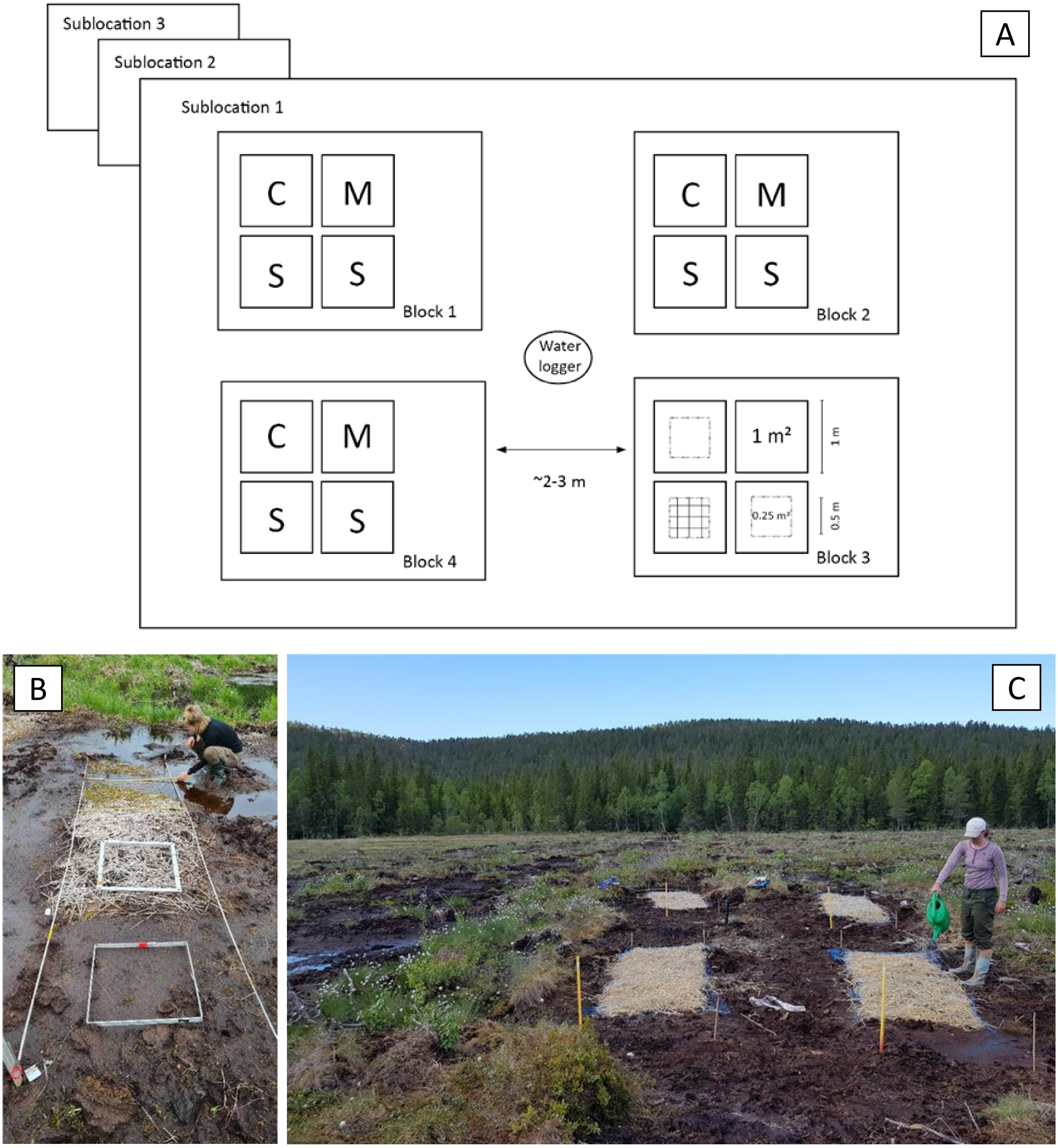

Figure 1. Overview of the experimental setup. A) A conceptual figure showing the experimental setup at each location. Within each location, three (BSM and HM) or two (VSM) sublocations were set up for the experiment. At each sublocation, four blocks consisted of four treatment units each*. In the middle of each sublocation, a water logger was continuously logging the water table level. Three treatments were included in each block: Control (C) with bare peat, mulch only (M), and *Sphagnum* fragments with mulch (M&S). Vegetation analyses were done for each treatment (0.5 × 0.5 m plots; illustrated in Block 3). *Sphagnum* measurements were done in subplots within plots, indicated as 16 smaller squares within the 0.5 × 0.5 m plot shown by the dotted outline within a treatment unit of $1 m^2$ in Block 3. B) A photo of vegetation analyses being taken on one block after two growing seasons. C) A photo showing the setup of a sublocation with four blocks. The mulch was watered to keep it in place during a hot spring day.

*One additional treatment with *Sphagnum* was included in the original design, for added phosphorous (P). This treatment was excluded from the final setup but the block was kept as a reserve against disruptions from natural causes, e.g. trampling.

species were identified, and their cover percentage was estimated.

Nutrient availability for 15 soil ions (see full overview in Figure A5) was quantified using Plant Root Simulator (PRS®) probes (Western AG Innovations Inc., Saskatoon, Canada). These were inserted into the topsoil when establishing the experiment, within the mulch and control plots at HM and VSM and all three treatments at BSM, and retrieved approximately 90 days later. During the first year, soil temperature was monitored in the control and mulch treatments with temperature loggers (HOBO® Pendant waterproof data logger UA-001-64; Onset Computer Corporation, MA, USA), which were buried in the topsoil to approximately 10 cm depth. Water-table height was logged hourly at each sublocation with a digital water logger (Levelogger 4 from Solinst®). The water logger was encapsuled within a perforated PVC pipe inserted into the peat.

Statistical analyses

Our analysis is divided based on two distinct modelling goals. First, we create inferential models for testing our hypotheses. Then we follow up with an exploratory analysis on some different but non-independent variables from the same experiment, the goal being to generate new hypotheses for future studies. All values are given as mean ± SE. All statistical analyses were performed using R (v4.4.1; R and RStudio v2024.09.0+375; Posit team 2024).

Hypothesis testing

We tested three competing hypotheses regarding *Sphagnum* revegetation:

- H1: *Sphagnum* cover increases with *Sphagnum* reintroduction (with mulch);
- H2: *Sphagnum* cover increases only due to added mulch, not reintroduced *Sphagnum*; and
- H3: *Sphagnum* cover is best explained by spatial variation (locations/sublocations) and hydrology, without treatment effects (a null model).

Our dataset includes observations from three time-steps, starting from Year 0. To simplify modelling and interpretation, we used data from only the final year, assuming *Sphagnum* cover development was roughly linear and initial conditions balanced across treatments. Exploratory figures confirmed this assumption, and systematic sampling helped to mitigate any initial differences.

Water-table data were summarised per growing season (01 May to 30 Sep) as mean, maximum and minimum water-table depth, total days below -20 cm, and longest period of consecutive days below -20 cm, as this is a threshold for optimal *Sphagnum* growth (Schouwenaars & Gosen 2007, Karofeld *et al.* 2020). Pearson's correlation and Variance Inflation Factor (car package; Fox & Weisberg 2018) analyses showed multicollinearity, so only mean water-table depth was included as a covariate. Missing water-table data (20 % of total) were interpolated using means from nearby wells.

We used Bayesian Generalized Linear Mixed Models (GLMMs) with the brms package (Bürkner *et al.* 2024), applying a beta family with logit and log links. Models were compared using posterior predictive checks (PPC), leave-one-out cross-validation (LOOCV) and Bayes R squared, employing the bayesplot (Gabry *et al.* 2024), loo (Vehtari *et al.* 2024) and brms packages, respectively. Each model ran for 3000 iterations per chain, excluding the first 1000 as warm-up, using four chains and no thinning. Default weakly informative priors were used, as narrower priors did not improve LOOCV results, suggesting the data predominantly shaped the posterior. The models were estimated using the No-U-Turn Sampler (NUTS) implemented in Stan, accessed through brms.

The model structure was:

$$\textit{Sphagnum cover} \sim \textit{Treatment} + \textit{Mean water-table} + (1 \mid \textit{Location} / \textit{Sublocation})$$

Here, (1 | *Location / Sublocation*) represents our hierarchical sampling design, with sublocations nested within locations. Blocks were excluded due to insufficient sample size. Hypotheses H1 and H2 were evaluated by comparing posterior distributions across the three treatments, assessing credible interval (CI) overlaps. H3 was tested by removing the treatment variable and comparing the simplified model to the full model. Effects were considered significant if their 95 % CI excluded zero, in line with standard Bayesian approaches to significance testing, and evaluated its magnitude and functional form using its marginal effect.

Exploratory analyses

First, we investigated whether adding straw mulch influenced soil nutrient availability and soil temperatures, as these factors could mediate potential treatment effects on *Sphagnum* and other PFTs. To investigate this, we used non-metric multi-dimensional scaling (NMDS) with the vegan package (Oksanen *et al.* 2024). The NMDS ordinations were run with Bray-Curtis distance measures since the data represent relative abundance.

Additionally, we used a further NMDS to examine the composition of PFTs across all treatments (including references) and locations, to see whether we could detect changes to the plant community after three growing seasons that would indicate recovery towards the reference community within any of the treatments. All values were log-transformed using a natural logarithm and subsequently min–max normalised to a range of 0–1 to standardise and normalise the data prior to analysis.

Lastly, we conducted a qualitative assessment of the variation in cover among the different *Sphagnum* species included in the *Sphagnum* mix. This evaluation was based on visual inspection of plots and summary statistics, without performing formal statistical analyses on species-specific cover or proportions.

RESULTS

Changes in *Sphagnum* cover

Before the treatments were applied, only four out of 120 plots had *Sphagnum* cover exceeding 1 % (Figure 2, Panel A). After three years, control plots had an average of 2.0 ± 0.6 % *Sphagnum* cover (max. 13 %) and mulch plots had an average of 4.6 ± 1.4 % (max. 30 %). On the other hand, the *Sphagnum* plots reached an average of 65.6 ± 3.5 % (max. 98 %), which is close to the levels of the reference plots' mean of 70.8 ± 5.0 % (Table A2).

The full GLMM model for treatment effects of *Sphagnum* (H1) or mulch only (H2) had an R^2 of 0.80 ± 0.03, indicating a generally strong model fit (Table A3, Figures A1 and A2 in the Appendix). In the model, *Sphagnum* treatment showed a substantial positive effect compared to the control, indicating higher values of *Sphagnum* cover by the end of Year 3 (Table A3, Figure A3). The confidence intervals (CIs) do not overlap with zero, indicating statistical significance in favour of H1 over H2. Although the mean cover in mulch plots was slightly higher than in controls, the overlapping CIs suggest no significant difference between mulch and no treatment, which does not support H2. Within the mulch treatments where *Sphagnum* was recorded before treatment, 5 out of 13 had a *Sphagnum* cover of >10 % after three years.

Initially, *Sphagnum* cover was similar across all locations, but after three growing seasons, treatment effects varied significantly between locations and sublocations (Table A4). The GLMM model for H3 (Table A3), which included only mean water-table and spatial variation, had a much lower R^2 of 0.05 ± 0.03 and a decreased expected log pointwise predictive density (ELPD) score based on LOOCV analysis. This allows us to reject H3, which suggested that *Sphagnum* re-establishment relies solely on water table and spatial variability. The variation

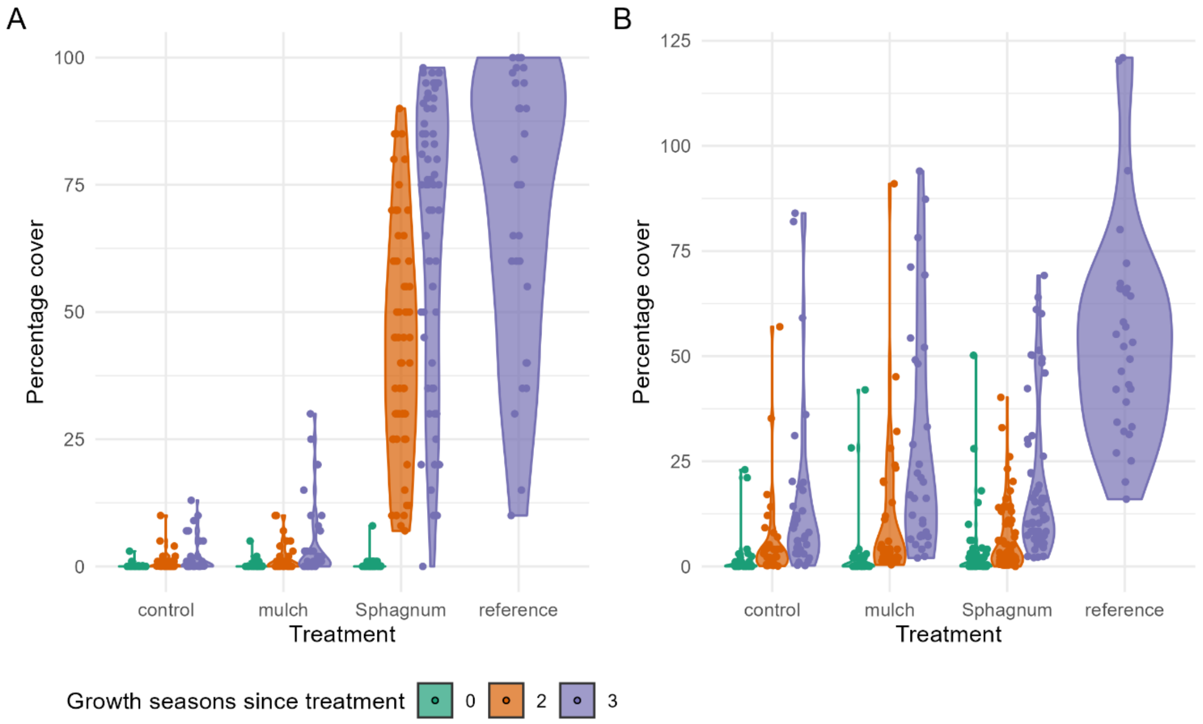

Figure 2. Violin plots showing the distribution of A) *Sphagnum* cover and B) total cover of vegetation other than *Sphagnum* across the treatment groups (control, mulch, *Sphagnum*, reference) and over growing seasons since treatment. Points inside the violins are individual measurements.

explained by location (sd = 1.28) and sublocation (sd = 0.65) in the full GLMM model indicate moderate variation of *Sphagnum* cover across these spatial levels. The posterior distributions do not overlap with zero, implying that spatial structure and water-table variation contribute to cover differences, though uncertainty remains due to wide CIs.

At BSM, *Sphagnum* cover ranged from 79.13 ± 2.22 % to 93.25 ± 2.08 %, exceeding reference values (56.60 ± 14.60 % and 84.00 ± 6.96 %). At HM, *Sphagnum* cover (51.25 ± 5.73 % to 87.50 ± 3.13 %) overlapped with but was generally lower than references (74.00 ± 12.19 % and 75.40 ± 10.92 %). At VSM, the sublocations were established in different mire types with distinct hydrological conditions, each paired with a corresponding reference transect, and the results reflect this variability. At VSM-1, *Sphagnum* treatment plots had a cover of 28.8 ± 4.4 %, compared to 48.0 ± 11.6 % in the reference transect, while VSM-2 plots exhibited 41.3 ± 10.7 %, markedly lower than the 86.6 ± 11.7 % in their reference transect.

Changes in cover of other plant functional types

The mean cover of vegetation other than *Sphagnum* was 2–3 % in all plots before treatment (Figure 2, Panel B). By Year 3, the cover of other vegetation had increased, particularly in plots with the mulch treatment (28.1 ± 4.8 %), but also in the control (15.8 ± 3.8 %) and *Sphagnum* treatment plots (19.4 ± 2.2 %). However, this was still considerably lower than in the reference sites, which had a mean cover of 54.7 ± 4.7 %. In the reference sites, the dominant PFTs other than *Sphagnum* were *Narthecium* (14.8 %), evergreen *Ericaceae* (10.4 %), and *Trichophorum* (9.1 %), while *Eriophorum* covered 3.3 % (Figure 3, Table A2). In contrast, *Eriophorum* showed slightly higher cover in all treatments after three years compared to the reference values, with 6.1 % in the control, 5.8 % in the mulch, and 6.8 % in the *Sphagnum* treatment, with maximum cover reaching 30–60 % in some plots. Notably, the maximum cover of *Eriophorum* never exceeded 12 % in the reference plots. Most other PFTs, such as *Narthecium* and *Trichophorum*, remained very low across all treatments (< 1 %).

In the mulch treatment, the vegetation was primarily composed of Other mosses (12.3 %), *Eriophorum* (5.8 %), Liverworts (3.5 %), evergreen *Ericaceae* (3.4 %) and *Polytrichum* (2.0 %). The only PFTs that showed any indication of a positive response to the mulch addition were evergreen *Ericaceae* and Other mosses, both still far below the reference values. Among these, Other mosses appeared to benefit most from the mulch, showing a clearly higher cover compared to the control (12.3 % vs. 5.3 %), while the increases in evergreen *Ericaceae* and Liverworts were more modest.

Treatment plots in the NMDS analysis with three dimensions based on PFT cover in Year 3 in all treatment types are primarily separated along the first NMDS axis (Figure 4, Panel A). Ellipses for control and mulch plots are broad and overlap considerably, not only with each other but also with the other two groups. The *Sphagnum* treatment ellipse is narrower and clustered closer to the references. The treatments are mainly separated on the first NMDS axis, which is primarily influenced by *Sphagnum* cover, with additional contributions in the same direction from *Trichophorum*, *Narthecium* and evergreen *Ericaceae*, a negative association with *Eriophorum*, and some with *Polytrichum* and Other mosses (Figure 4, Panel A).

Effect of straw mulch on soil temperature and nutrient availability

The mean temperatures during the growing season remained similar in both the control and the mulch treatment, while the mean maximum temperature was 1.2 °C lower in the mulch treatment and the mean minimum temperature was 0.5 °C higher. The maximum daily temperature fluctuations were 2.2 °C lower in the mulch treatment than in the controls (Figure A4). Thus, the mulch cover did seem to dampen the annual and daily temperature fluctuations of the peat through increased minimum and/or decreased maximum temperatures. This effect was only clearly present at BSM and VSM-1, while the temperature at HM and VSM-2 remained similar in both treatments.

Except for potassium (K), which was higher in the mulch treatment plots, no clear differences in soil nutrient levels were observed between the control and mulch treatments during the first growing season, indicating that the mulch did not provide an initial fertilisation effect (Figure A5). However, nutrient availability varied considerably between locations (Figure A6).

The NMDS ordination based on soil nutrient availability and soil temperatures showed no strong association between these two factors and the application of mulch (Figure 4, Panel B). Across all locations, a trend was observed where mulch-treated plots shifted slightly rightward along the first NMDS axis compared to controls within the same blocks. This shift was driven primarily by higher levels of NH_4-N, Fe and minimum temperature in the positive direction, while Mn and all three variables for temperature fluctuations influenced the negative direction. Location differences, by contrast, were clearly distinguished along both NMDS axes, with distinct clusters forming for each location (Figure 4, Panel B).

Species distribution in subplots

The four evenly distributed *Sphagnum* species showed clear differences in re-establishment success, with *S. angustifolium* being the most successful, reaching a mean of 17 % cover in Year 2 and 29 % in Year 3 (Figure 5). In contrast, *S. capillifolium*, *S. papillosum* and *S. tenellum* had more moderate growth, each reaching around 6.5 % after two seasons and about 7–9 % after three. *Sphagnum angustifolium* accounted for 43 % of the total *Sphagnum* cover in Year 2 and 52 % by Year 3. *Sphagnum capillifolium* and *S. tenellum* each had a share of about 20 % by Year 2, but these proportions went down to about 16 % by Year 3. *Sphagnum papillosum* contributed 12 % and 13 % over the same period. This difference between species varied somewhat between locations and sublocations, with *S. angustifolium* being very dominating at HM and VSM, while not as pronounced at BSM (Figure 5, Table A5). *Sphagnum tenellum* had the lowest

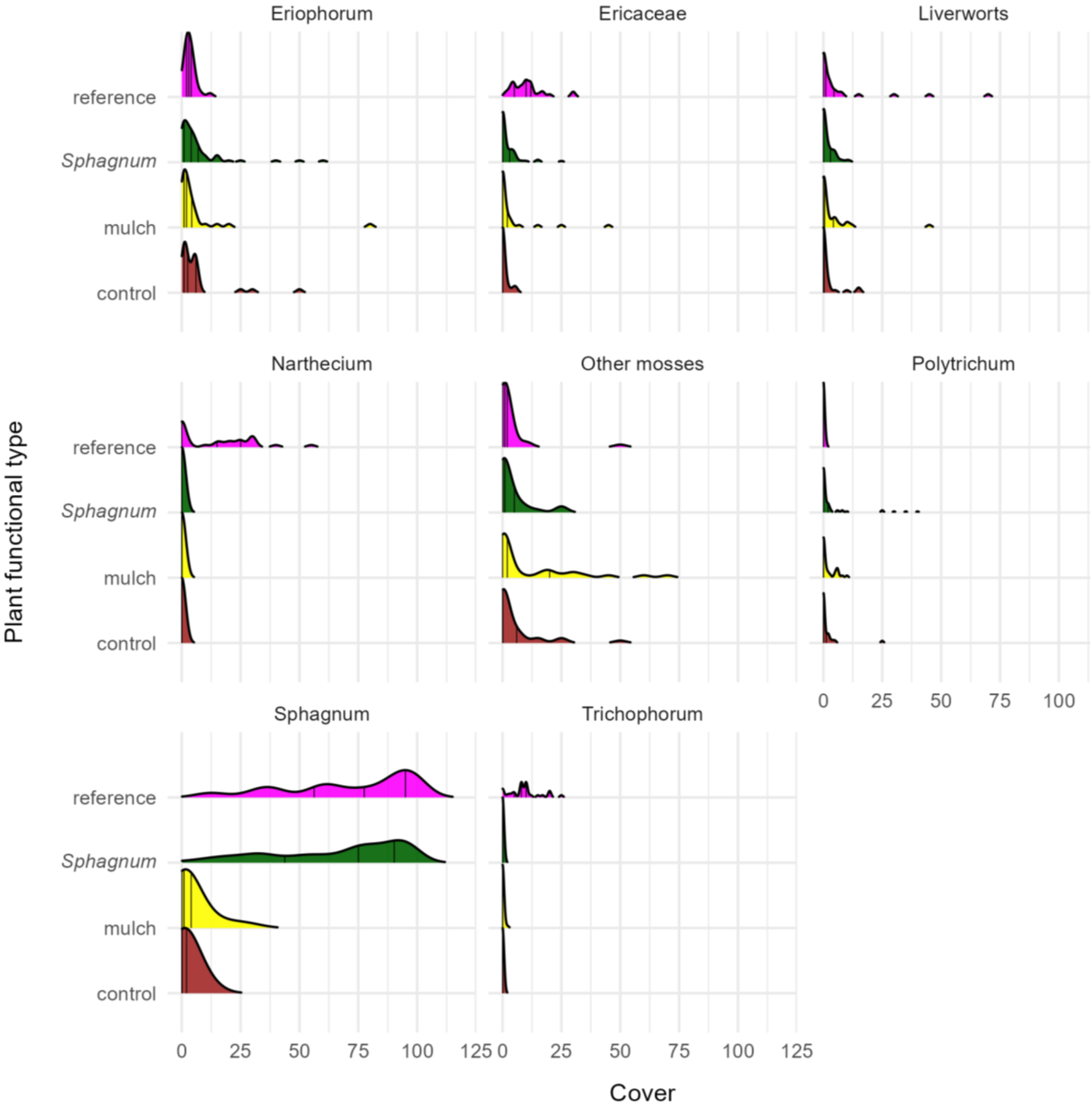

Figure 3. Density ridge plot of the percentage cover of the most prominent plant functional types within all treatments (mulch cover only, *Sphagnum* with mulch, controls, references) after three growing seasons.

proportion (only 7 %) at VSM but 16 % at HM and 24 % at BSM. *Sphagnum capillifolium* had similar low proportions at VSM (10 %) and HM (about 14 %), with 23 % at BSM.

Six additional *Sphagnum* species (*S. balticum*, *S. compactum*, *S. divinum*, *S. majus*, *S. riparium* and *S. russowii*) appeared in the plots, along with some unidentified *Sphagnum* species. Although the average cover of these extra species was low (< 0.5 %), they could make up as much as 10 % in individual subplots.

DISCUSSION

Peatland restoration is increasingly recognised as a key conservation strategy, both for mitigating greenhouse gas (GHG) emissions and protecting the specialised biodiversity of peatlands (Fraixedas *et al.* 2017, Leifeld & Menichetti 2018, Günther *et al.* 2020). Hydrological restoration is the cornerstone of peatland recovery, but additional measures can accelerate revegetation and ecosystem stabilisation (Allan *et al.* 2023). This study provides the first experimental assessment of active revegetation techniques for peatland restoration in Norway, demonstrating how the addition of *Sphagnum* fragments and straw mulch influences early recovery trajectories.

Active revegetation with *Sphagnum* fragments

The addition of *Sphagnum* fragments significantly accelerated the recovery of *Sphagnum* in our study plots. After three growing seasons, *Sphagnum* cover in the *Sphagnum* treatment at BSM matched that in the reference plots, while HM had about 65 % cover, close to the 75 % seen in reference plots. These outcomes exceed the general success rates reported

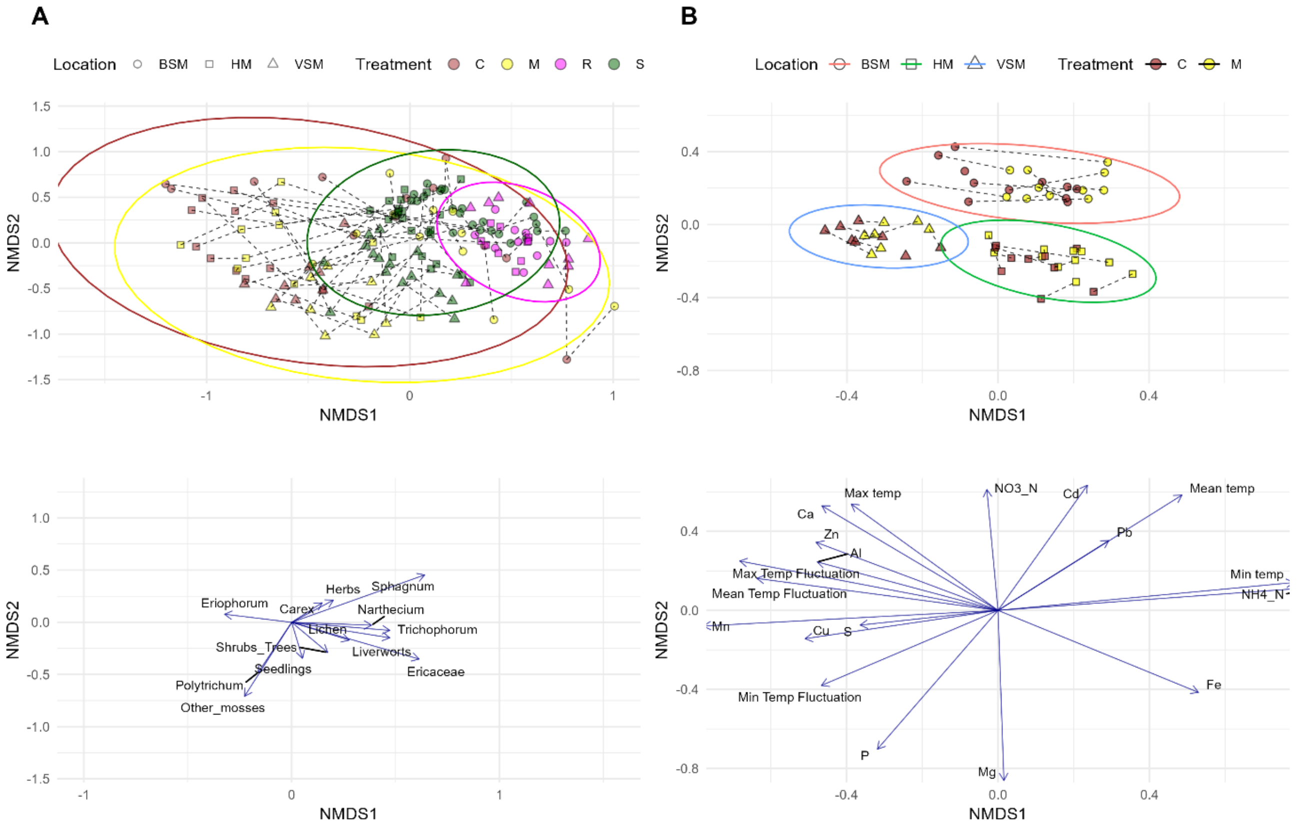

Figure 4. Two non-metric multidimensional scaling (NMDS) analyses, with site scores in the top row and the vectors for the species scores and environmental variables in the bottom row. The analysis in Column A includes cover of PFTs within all treatments. The analysis in Column B spreads the sites based on soil temperature and nutrient level variables during first growing season within mulch treatment (M) and controls (C). Dotted lines connect datapoints from the same block. The ellipses represent the 95 % confidence interval. Only vectors from significant environmental variables and PFTs are shown ($p < 0.05$). C = control, M = mulch, S = *Sphagnum*, R = reference plots, BSM = Bukkastadmyra, HM = Stormyra, VSM = Vestersetermyra. NMDS Panel A shows a model fit with $R^2 = 0.98$ and stress = 0.16, while NMDS Panel B shows a model fit with $R^2 = 0.98$ and stress = 0.12.

in other studies using the MLTT (Rochefort *et al.* 2003, Chirino *et al.* 2006, Sottocornola *et al.* 2007, González & Rochefort 2014) but are comparable to those reported by Rochefort *et al.* (2013), with *Sphagnum* cover of 65 % after eight years compared to 82–84 % cover in intact peatlands.

The faster establishment in our study suggests that factors beyond the MLTT itself may contribute to success rates. One key factor may be the manual collection and application method, which cuts rather than crushes the *Sphagnum* shoots and gives a more even distribution of fragments compared to the mechanical approaches commonly used in large-scale restoration projects. Previous studies have shown that manually collected and cut fragments establish more successfully than mechanically processed ones, regardless of fragment size (Rochefort *et al.* 2003, Campeau *et al.* 2004, Sottocornola *et al.* 2007).

It is likely that climatic conditions also played a role. While all sites experienced frequent precipitation, it varied in both total amount and the length of dry periods. HM and VSM had a prolonged drying spell in June of the establishment year, followed by above-average wet conditions in July and August, whereas BSM had more evenly distributed precipitation throughout the first summer (Table 1). Frequent effective rainfall events (≥ 2 mm) during the growing season are particularly important for maintaining adequate moisture at the peat surface, supporting *Sphagnum* expansion (Strack & Price 2009). Water-table levels and precipitation were generally similar at BSM and HM throughout the experiment. However, there was an early summer drought in the establishment year at HM. This period may have been particularly important for establishment rates, as the *Sphagnum* cover at HM was lower than at BSM (Table A4).

At VSM, the consistently low water tables at VSM-1 could not be attributed to climatic conditions. Water-table levels at this sublocation remained below -20 cm, with values between -30 and -40 cm during

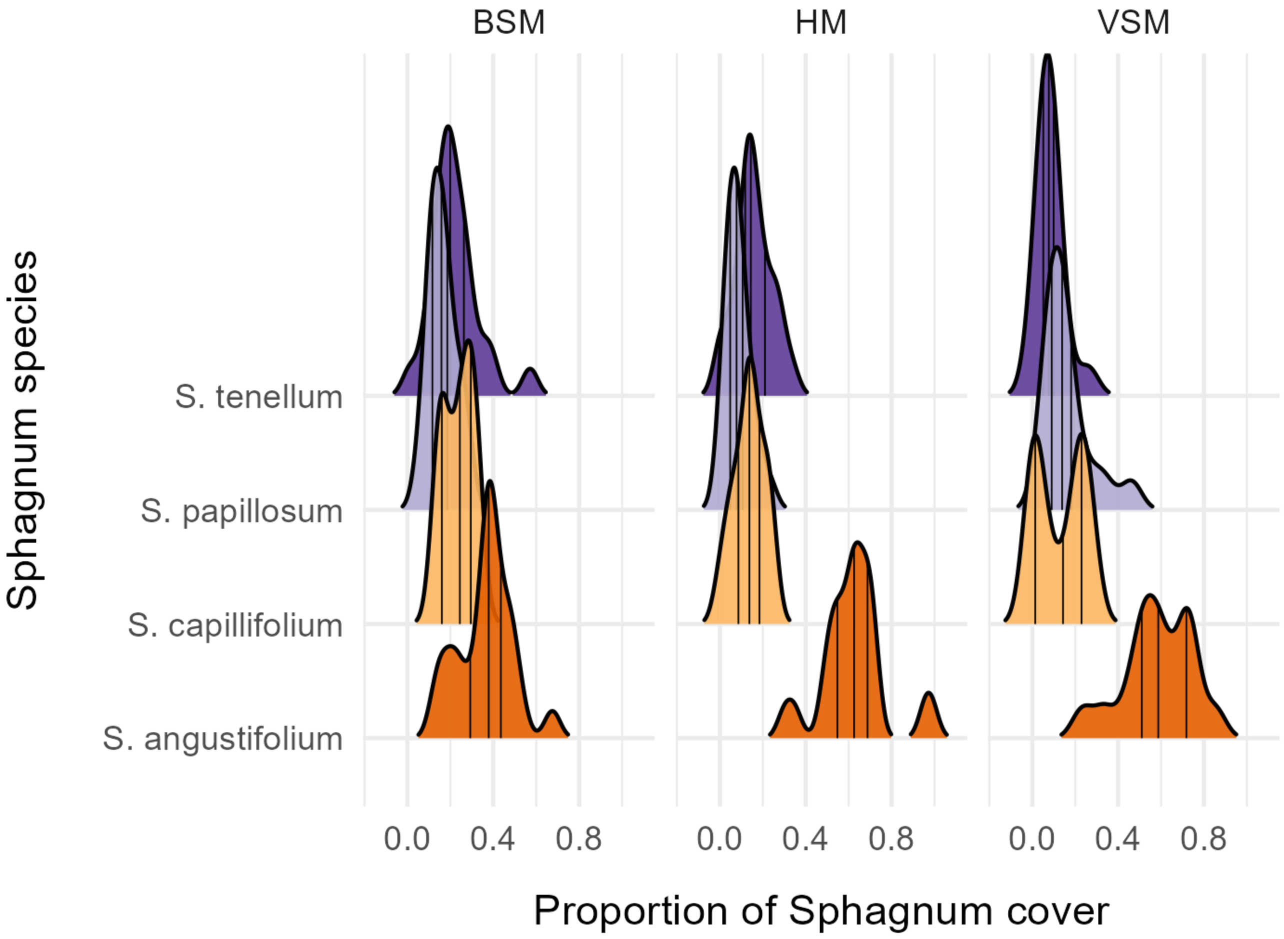

Figure 5. Density ridge plot of the proportion of species-specific *Sphagnum* cover within the subplots of *Sphagnum* treatment at each of the locations BSM (Bukkastakkmyra), HM (Stormyra), and VSM (Vestersetermyra) after three growing seasons.

most of the growing season, creating dry conditions that are likely to have impeded early *Sphagnum* establishment (Figure A7). These results reinforce the importance of maintaining a stable, high water-table level. While a water table above -40 cm is often considered sufficient for re-establishment (Schouwenaars 1988, Price & Whitehead 2001, Lavoie *et al.* 2003), recent studies suggest a threshold above -20 cm for robust cover development (Schouwenaars & Gosen 2007, Karofeld *et al*. 2020), as this level is necessary for efficient gross photosynthetic activity (Pacheco-Cancino *et al.* 2024). However, these suggestions are for already-established *Sphagnum* carpets. Quadra *et al.* (2025) found that even a moderate change in water table fluctuations, from 0 to -10 cm (stable) to -10 to -30 cm (fluctuating) during the growing season, significantly reduced *Sphagnum* cover and height growth in the first year. While these fluctuations are within known survival thresholds for *Sphagnum*, they could still hinder early establishment, especially for fragments that have not yet formed a carpet structure - where soil moisture might be of higher importance than water-table levels directly (Chirino *et al*. 2006).

The effect of straw mulch

Sphagnum fragments without mulch cover were not included in our study, as literature consistently demonstrates that establishment success improves with some form of protective cover (Price *et al.* 1998, Rochefort *et al.* 2003, Corson & Campbell 2013, Gaudig *et al.* 2014).

Straw mulch alone did not significantly enhance natural *Sphagnum* re-establishment. Mulch reduced temperature fluctuations at BSM and VSM-1, consistent with its reported benefits in mitigating desiccation and moderating temperature extremes (Rochefort *et al.* 2003, Petrone *et al.* 2004, Groeneveld *et al.* 2007). In contrast, minimal effects were observed at HM and VSM-2, which may be due to inconsistencies in temperature measurements. Frost heaving caused some temperature loggers to be displaced, potentially recording data from the surface rather than within the soil, which could have influenced the observed results. We did not investigate the microclimatic effects of mulch in detail, as this was not our primary goal, and it has been documented thoroughly elsewhere (Price *et al.* 1998, Waddington *et al.* 2003b). NMDS ordination showed that mulch-treated plots exhibited a subtle but consistent shift along the first NMDS axis, suggesting a small but detectable influence of mulch on soil conditions. This shift was mainly associated with reduced temperature fluctuations, as all three temperature fluctuation variables were negatively associated with NMDS1 whereas minimum temperature was positively associated. This suggests that mulch contributed to more stable temperature conditions, potentially moderating extreme variations. Few of the other PFTs in our study benefited from mulch, except ‘Other mosses’ and ‘Evergreen *Ericaceae*’, which had higher cover in mulch than in control plots. Amongst the ‘Other mosses’ there are many opportunistic pioneers not necessarily associated with natural peatland vegetation. Likewise, Triisberg-Uljas *et al.* (2018) observed that mulch alone primarily promoted non-peatland-specific species like *Betula* sp. and mosses other than *Sphagnum*. *Eriophorum* spp. established across all treatments, reaching higher cover levels than in reference plots, consistent with their observed role as pioneer species on bare peat (Lavoie *et al.* 2003, Lavoie *et al.* 2005). No other PFTs re-established a substantial cover during the monitoring period. Overall, we found no clear evidence that mulch directly facilitated recovery of *Sphagnum* or of beneficial species for *Sphagnum*. A longer monitoring period may be necessary to assess whether such effects can be observed over time.

Nutrient availability measurements showed minimal alterations due to mulch, apart from slightly elevated potassium (K), iron (Fe) and ammonium-nitrogen (NH_4) levels. To our knowledge, no studies have explicitly examined the fertilisation effects of mulch application in peatland restoration. However, Waddington *et al.* (2003b) demonstrated that straw mulch decomposition leads to substantial carbon release, which could also influence nutrient dynamics by affecting microbial activity and nutrient mineralisation. The absence of significant enrichment in this study is positive, as excessive nutrients could promote unwanted vegetation shifts. Substantial nutrient differences were observed between locations, such as VSM's low ammonium-nitrogen (NH_4) but relatively high phosphorus (P), calcium (Ca) and magnesium (Mg). It is likely that these differences reflect the sites’ classification as different mire types and their original vegetation in the adjacent reference areas, suggesting site-specific nutrient availability.

Species-specific treatment adjustments

In our experiment, *S. angustifolium* exhibited substantially better establishment success than the other three species tested. This result is consistent with its attributes, given its close relationship to *S. fallax* which is a known rapid grower under humid conditions and a generalist pioneer often recommended for reintroduction (Grosvernier *et al.* 1997, Buttler *et al.* 1998, Caporn *et al.* 2018). The consistently high water tables across our study sites are likely to have provided particularly favourable conditions for *S. angustifolium*, which has been shown to establish under a variety of conditions and may establish as well as - or better than - hummock-forming species until a critical drought threshold is reached (Campeau & Rochefort 1996, Chirino *et al*. 2006). Tuittila *et al.* (2003) observed high establishment success of *S. angustifolium* in a restored peatland in Finland, despite significantly lower annual precipitation (~700 mm) and lower mean water-table levels than in our study. *Sphagnum angustifolium* also outperformed the other species at VSM, despite lower and more variable water-table levels than at the other locations during the establishment year. We initially expected the hummock-forming species *S. capillifolium* to be more competitive in these conditions, given its association with drier microhabitats, yet *S. angustifolium* remained dominant. This may reflect differences in species-specific responses at an early stage of establishment. Quadra *et al.* (2025) found that hummock species had lower survival rates than lawn species when introduced as single shoots, with both stable high and lower, more fluctuating, water tables. This could be because the desiccation tolerance of hummock species is linked primarily to their ability to avoid drying out rather than to recover from it. They rely on a high water-retention capacity based on dense capitulum structure, tightly clustered branches, and compact layering (Hájek & Beckett 2008, Robroek *et al.* 2009, Hájek 2014). When fragmented, they may struggle to retain moisture effectively. Hollow species, on the other hand, experience drying more frequently and for longer periods, but often exhibit higher survival rates after short desiccation events (Wagner & Titus 1984, Schipperges & Rydin 1998, Hájek & Vicherová 2014). This ability to withstand and recover from drying may explain the greater success of hollow species in early-stage revegetation.

Sphagnum tenellum had very low establishment success at VSM compared to the other locations. This aligns with its ecological traits, as *S. tenellum* is primarily restricted to hollows and lower lawns (Økland 1989). *Sphagnum tenellum* is rarely mentioned in large-scale revegetation projects, probably due to its narrow regional distribution (mainly oceanic and hyperoceanic areas in Europe) and relatively low competitive ability (Rydin 1993, 1997; Økland 1990). Despite these limitations, *S. capillifolium* and *S. tenellum* performed similarly across all three of our study locations, suggesting a better initial competitive ability than expected. As a known pioneer on bare peat, with a strong presence in early successional stages and a high regenerative capacity (Økland 1989, 1990; Flatberg 2013), it could play a valuable role in the initial revegetation of bare peat surfaces in wet climates.

Among the four *Sphagnum* species tested, *S. papillosum* had the lowest cover; although the differences between this species, *S. tenellum* and *S. capillifolium* were minimal. This aligns with the mixed results for re-establishment success on bare peat reported for *S. papillosum* and the related *S. medium / S. divinum* (Campeau & Rochefort 1996, Robert *et al.* 1999, Chirino *et al.* 2006, Purre *et al.* 2020). While species of subgenus *Acutifolia* are generally considered to be better at colonising bare peat than species from subgenus *Sphagnum* (Rochefort 2000), species like *S. papillosum* will be important for developing peat-forming *Sphagnum* lawns and lower hummocks, underscoring their importance in long-term peatland restoration (Smolders *et al.* 2003). Despite its dominance in natural bogs and poor fens in Norway, establishment of *S. papillosum* may require different environmental cues or longer timeframes to reflect its typical dominance. We predict that its cover will expand gradually and persistently over time.

The presence of additional *Sphagnum* species, although at very low cover levels, suggests early signs of natural colonisation or unintentional introduction alongside the targeted species. Early recolonisation is often influenced by stochastic dispersal events rather than strictly deterministic environmental factors (Soro *et al.* 1999). This emerging diversity indicates the potential for forming a more diverse *Sphagnum* assemblage, which could support peatland recovery and enhance ecosystem resilience over time.

Implications for management

This study marks the first experiment testing active revegetation in peatland restoration in Norway, demonstrating the potential of this technique to complement hydrological restoration by accelerating *Sphagnum* establishment. The success of active revegetation underscores its value as a complementary technique to rewetting, applicable not only to sites with extensive bare peat but also to those where the residual vegetation lacks *Sphagnum*.

We did not add any fertiliser. Although phosphorus (P) supplementation seems to be common practice in many places, it does not necessarily boost *Sphagnum* re-establishment directly (Triisberg *et al.* 2013, Gaudig *et al.* 2014, Gauthier *et al.* 2018, Purre *et al.* 2020). Its primary purpose is to support the re-establishment of *Polytrichum strictum*, which is considered a nurse plant that facilitates *Sphagnum* development (Quinty & Rochefort 2003, Groeneveld *et al.* 2007, Sottocornola *et al.* 2007). Although there is a clear positive effect on *Sphagnum* development at bogs and poor fens in Canada (Liu *et al.* 2024), sourcing P supplementation may be challenging in Norway, and it is a limited resource with associated environmental risks. Here we demonstrated that *Sphagnum* can clearly recover without it.

Our study also explored the use of straw mulch application as a restoration measure. Our results indicate that, under near-optimal environmental conditions, i.e. high water table and sufficient precipitation, mulch alone plays a limited role in promoting the natural recolonisation of peatland plant species. While it may help to buffer microclimatic fluctuations, it appears insufficient for rapid recovery of *Sphagnum*-dominated vegetation without additional interventions.

Matching *Sphagnum* species to the specific microhabitats and hydrological conditions of the restoration site is likely to be the most effective strategy for achieving rapid recovery (González & Rochefort 2014, Caporn *et al.* 2018, Purre *et al.* 2020). When sourcing specific species is impractical, a diverse species mix may provide long-term benefits by enhancing resilience to environmental changes and ensuring that at least some species thrive (Swindles *et al.* 2016, Hugron & Rochefort 2018, Laine *et al.* 2021), ultimately supporting the recovery of peatland ecosystem functions. The initial success of our four *Sphagnum* species varied, with *S. angustifolium* emerging as the most competitive and successful pioneer. However, extended monitoring may reveal shifts in this trajectory as species interactions and environmental conditions change over time. Since *Sphagnum* species are unlikely to fully outcompete each other, leading to a very slow-developing community (Rydin 1993, Rydin *et al.* 2006), a diverse mix may promote diversity. Incorporating locally abundant, well-adapted species is a practical approach to restoration, as these species are better suited to regional conditions and support local biodiversity. Sourcing large quantities of *Sphagnum* species emphasises the need for scalable collection methods or cultivation approaches to effectively meet restoration needs.

To broaden the applicability of this approach, future studies should replicate and scale it across various peatland types and geographical regions. While the Moss Layer Transfer Technique (MLTT) has been widely used for large-scale restoration in Canada since the 1990s, our findings demonstrate that its principles can be effectively adapted to accelerate revegetation in smaller-scale applications, such as localised patches of bare peat. Incorporating active revegetation measures into restoration planning, alongside hydrological measures, should be part of the ‘best practice’ approach in peatland restoration. Future research should explore both the short-term and the long-term effects of active revegetation on carbon dynamics, peatland hydrology and vegetation community development, ensuring these methods are adapted to regional conditions while contributing to global restoration standards.

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AUTHOR CONTRIBUTIONS

Marte Fandrem: Conceptualisation (equal), Methodology (lead), Investigation (equal), Formal analysis (lead), Data Curation (lead), Visualisation (lead), Writing - Original Draft (lead), Writing - Review & Editing (lead). **Magni Olsen Kyrkjeeide:** Funding acquisition (lead), Conceptualisation (equal), Methodology (supporting), Investigation (equal), Writing - Review & Editing (equal). **Kristian Hassel:** Writing - Review & Editing (equal). **James D.M. Speed:** Formal analysis (supporting)**,** Writing - Review & Editing (equal). **Anders Lorentzen Kolstad:** Formal analysis (supporting), Writing - Review & Editing (equal).

REFERENCES

Allan, J.M., Guêné-Nanchen, M., Rochefort, L., Douglas, D.J.T., Axmacher, J.C. (2023) Meta-analysis reveals that enhanced practices accelerate vegetation recovery during peatland restoration. *Restoration Ecology*, 32(3), e14015, 13 pp. https://doi.org/10.1111/rec.14015

Andersen, R., Farrell, C., Graf, M., Muller, F., Calvar, E., Frankard, P., Caporn, S., Anderson, P. (2017) An overview of the progress and challenges of peatland restoration in Western Europe. *Restoration Ecology*, 25(2), 271–282.

https://doi.org/10.1111/rec.12415

Benson, J.L., Crouch, T., Chandler, D. (2021) *Monitoring Single-Species* Sphagnum *Plug Growth on Blanket Bog*. Report, Moors for the Future Partnership, Edale, UK, 24 pp. Online at: https://www.iucn-uk-peatlandprogramme.org/sites/default/files/2021-05/Single-Species-Sphagnum-Report-MoorLIFE-2020-2021.pdf, accessed 02 Mar 2025.

Bürkner, P.-C., Gabry, J., Weber, S., Johnson, A., Modrak, M., Badr, H.S., Weber, F., Vehtari, A., Ben-Shachar, M.S., Rabel, H., Mills, S.C., Wild, S., Popov, V. (2024) brms: Bayesian Regression Models using 'Stan'. Online at: https://cran.r-project.org/web/packages/brms/index.html, accessed 02 Mar 2025.

Buttler, A., Grosvernier, P., Matthey, Y. (1998) Development of *Sphagnum fallax* diaspores on bare peat with implications for the restoration of cut-over bogs. *Journal of Applied Ecology*, 35(5), 800–810. https://doi.org/10.1046/j.1365-2664.1998.355351.x

Campeau, S., Rochefort, L. (1996) *Sphagnum* regeneration on bare peat surfaces: Field and greenhouse experiments. *Journal of Applied Ecology*, 33(3), 599–608. https://doi.org/10.2307/2404988

Campeau, S., Rochefort, L., Price, J.S. (2004) On the use of shallow basins to restore cutover peatlands: Plant establishment. *Restoration Ecology*, 12(4), 471–482. https://doi.org/10.1111/j.1061-2971.2004.00302.x

Caporn, S.J.M., Rosenburgh, A.E., Keightley, A.T., Hinde, S.L., Riggs, J.L., Buckler M., Wright N.A. (2018) *Sphagnum* restoration on degraded blanket and raised bogs in the UK using micropropagated source material: a review of progress. *Mires and Peat*, 20, 09, 17 pp. https://doi.org/10.19189/MaP.2017.OMB.306

CBD (2022) *Kunming-Montreal Global Biodiversity Framework*. CBD/COP/DEC/15/4, Decision adopted by the Conference of the Parties to the Convention on Biological Diversity (CBD), Montreal, 15 pp. Online at: https://www.cbd.int/doc/decisions/cop-15/cop-15-dec-04-en.pdf, accessed 02 Mar 2025.

Chirino, C., Campeau, S., Rochefort, L. (2006) *Sphagnum* establishment on bare peat: The importance of climatic variability and *Sphagnum* species richness. *Applied Vegetation Science*, 9(2), 285–294. https://doi.org/10.1111/j.1654-109X.2006.tb00678.x

Cook-Patton, S.C., Drever, C.R., Griscom, B.W., Hamrick, K., Hardman, H., Kroeger, T., Pacheco, P., Raghav, S., Stevenson, M., Webb, C., Yeo, S., Ellis, P.W. (2021) Protect, manage and then restore lands for climate mitigation. *Nature Climate Change*, 11, 1027–1034. https://doi.org/10.1038/s41558-021-01198-0

Corson, A., Campbell, D. (2013) Testing protocols to restore disturbed *Sphagnum*-dominated peatlands in the Hudson Bay Lowland. *Wetlands*, 33(2), 291–299. https://doi.org/10.1007/s13157-013-0383-3

Couwenberg, J. (2011) Greenhouse gas emissions from managed peat soils: is the IPCC reporting guidance realistic? *Mires and Peat*, 8, 02, 10 pp.

Cronberg, N. (1992) Reproductive biology of *Sphagnum*. *Lindbergia*, 17(4/6), 69–82.

Evans, C.D., Peacock, M., Baird, A.J., Artz, R.R.E. and 27 others (2021) Overriding water table control on managed peatland greenhouse gas emissions. *Nature*, 593, 548–552. https://doi.org/10.1038/s41586-021-03523-1

Flatberg, K.I. (2013) *Norges torvmoser* (*Norway's Peat Bogs*). Akademika forlag, Oslo, 307 pp.

Fox, J., Weisberg, S. (2018) *An R Companion to Applied Regression*. SAGE Publications Inc., USA, 608 pp.

Fraixedas, S., Lindén, A., Meller, K., Lindström, Å., Keišs, O., Kålås, J.A., Husby, M., Leivits, A., Leivits, M., Lehikoinen, A. (2017) Substantial decline of Northern European peatland bird populations: Consequences of drainage. *Biological Conservation*, 214, 223–322. https://doi.org/10.1016/j.biocon.2017.08.025

Gabry, J., Mahr, T., Bürkner, P.-C., Modrák, M., Barrett, M., Weber, F., Sroka, E.C., Sailynoja, T., Vehtari, A. (2024) bayesplot: Plotting for Bayesian Models. Online at: https://cran.r-project.org/web/packages/bayesplot/index.html, accessed 02 Mar 2025.

Gann, G.D., McDonald, T., Walder, B., Aronson, J., Nelson, C.R., Jonson, J., Hallett, J.G., Eisenberg, C., Guariguata, M.R., Liu, J., Hua, F., Echeverría, C., Gonzales, E., Shaw, N., Decleer, K., Dixon K.W. (2019) International principles and standards for the practice of ecological restoration. Second edition. *Restoration Ecology*, 27(S1), S1–S46. https://doi.org/10.1111/rec.13035

Gaudig, G., Fengler, F., Krebs, M., Prager, A., Wichmann, S., Joosten, H. (2014) Sphagnum farming in Germany - a review of progress. *Mires and Peat*, 13, 08, 11 pp.

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. *Wetlands Ecology and Management*, 26, 475–487. https://doi.org/10.1007/s11273-017-9532-4

Girard, M., Lavoie, C., Thériault, M. (2002) The regeneration of a highly disturbed ecosystem: A mined peatland in Southern Québec. *Ecosystems*, 5(3), 274–288. https://doi.org/10.1007/s10021-001-0071-7

González, E., Rochefort, L. (2014) Drivers of success in 53 cutover bogs restored by a moss layer transfer technique. *Ecological Engineering*, 68, 279–290. https://doi.org/10.1016/j.ecoleng.2014.03.051.

González, E., Rochefort, L., Boudreau, S., Hugron, S., Poulin, M. (2013) Can indicator species predict restoration outcomes early in the monitoring process? a case study with peatlands. *Ecological Indicators*, 32, 232–238. https://doi.org/10.1016/j.ecolind.2013.03.019

Gorham, E. (1991) Northern peatlands: Role in the carbon cycle and probable responses to climatic warming. *Ecological Applications*, 1(2), 182–195. https://doi.org/10.2307/1941811

Griscom, B.W., Adams, J., Ellis, P.W., Houghton, R.A. and 28 others (2017) Natural climate solutions. *Proceedings of the National Academy of Sciences (PBAS)*, 114(44), 11645–11650. https://doi.org/10.1073/pnas.1710465114

Groeneveld, E.V.G., Rochefort, L. (2002) Nursing plants in peatland restoration: on their potential use to alleviate frost heaving problems. *Suo*, 53(3–4), 73–85.

Groeneveld, E.V.G., Massé, A., Rochefort, L. (2007) *Polytrichum strictum* as a nurse-plant in peatland restoration. *Restoration Ecology*, 15(4), 709–719. https://doi.org/10.1111/j.1526-100X.2007.00283.x

Grosvernier, P., Matthey, Y., Buttler, A. (1997) Growth potential of three *Sphagnum* species in relation to water table level and peat properties with implications for their restoration in cut-over bogs. *Journal of Applied Ecology*, 34(2), 471–483. https://doi.org/10.2307/2404891

Günther, A., Barthelmes, A., Huth, V., Joosten, H., Jurasinski, G., Koebsch, F., Couwenberg, J. (2020) Prompt rewetting of drained peatlands reduces climate warming despite methane emissions. *Nature Communications*, 11(1), 1644, 5 pp. https://doi.org/10.1038/s41467-020-15499-z

Haapalehto, T., Juutinen, R., Kareksela, S., Kuitunen, M., Tahvanainen, T., Vuori, H., Kotiaho, J.S. (2017) Recovery of plant communities after ecological restoration of forestry-drained peatlands. *Ecology and Evolution*, 7(19), 7848–7858. https://doi.org/ 10.1002/ece3.3243

Hájek, T. (2014) Physiological ecology of peatland bryophytes. In: Hanson, D.T., Rice, S.K. (eds.) *Photosynthesis in Bryophytes and Early Land Plants*, Advances in Photosynthesis and Respiration 37, Springer, Dordrecht, 233–252. https://doi.org/10.1007/978-94-007-6988-5_13

Hájek, T., Beckett, R.P. (2008) Effect of water content components on desiccation and recovery in *Sphagnum* mosses. *Annals of Botany*, 101(1), 165–173. https://doi.org/10.1093/aob/mcm287

Hájek, T., Vicherová, E. (2014) Desiccation tolerance of *Sphagnum* revisited: a puzzle resolved. *Plant Biology*, 16(4), 765–773. https://doi.org/10.1111/plb.12126

Hayward, P.M., Clymo, R.S. (1983) The growth of *Sphagnum*: Experiments on, and simulation of, some effects of light flux and water-table depth. *Journal of Ecology*, 71(3), 845–863. https://doi.org/10.2307/2259597

Hugelius, G., Loisel, J., Chadburn, S., Jackson, R.B., Jones, M., MacDonald, G., Marushchak, M., Olefeldt, D., Packalen, M., Siewert, M.B., Treat, C., Turetsky, M., Voigt, C., Yu, Z. (2020) Large stocks of peatland carbon and nitrogen are vulnerable to permafrost thaw. *Proceedings of the National Academy of Sciences (PNAS)*, 117(34), 20438–20446. https://doi.org/10.1073/pnas.1916387117

Hugron, S., Rochefort, L. (2018) *Sphagnum* mosses cultivated in outdoor nurseries yield efficient plant material for peatland restoration. *Mires and Peat*, 20, 11, 6 pp. https://doi.org/10.19189/MaP.2018.OMB.358

Hugron, S., Guêné-Nanchen, M., Roux, N., LeBlanc, M.-C., Rochefort, L. (2020) Plant reintroduction in restored peatlands: 80% successfully transferred - Does the remaining 20% matter? *Global Ecology and Conservation*, 22, e01000, 11 pp. https://doi.org/10.1016/j.gecco.2020.e01000

Karofeld, E., Müür, M., Vellak, K. (2016) Factors affecting re-vegetation dynamics of experimentally restored extracted peatland in Estonia. *Environmental Science and Pollution Research*, 23(14), 13706–13717. https://doi.org/10.1007/s11356-015-5396-4

Karofeld, E., Jarašius, L., Priede, A., Sendžikaitė, J., (2017) On the after-use and restoration of abandoned extracted peatlands in the Baltic countries. *Restoration Ecology*, 25(2), 293–300. https://doi.org/10.1111/rec.12436

Karofeld, E., Kaasik, A., Vellak, K. (2020) Growth characteristics of three *Sphagnum* species in restored extracted peatland. *Restoration Ecology*, 28(6), 1574–1583. https://doi.org/10.1111/rec.13245

Kyrkjeeide, M.O., Jokerud, M., Mehlhoop, A.C., Foldnes Lunde, L.M., Fandrem, M., Lyngstad, A. (2024) Peatland restoration in Norway - evaluation of ongoing monitoring and identification of plant indicators of restoration success. *Nordic Journal of Botany*, 2024(4), e03988, 11 pp. https://doi.org/

10.1111/njb.03988

Laine, A.M., Lindholm, T., Nilsson, M., Kutznetsov, O., Jassey, V.E.J., Tuittila E.-S. (2021) Functional diversity and trait composition of vascular plant and *Sphagnum* moss communities during peatland succession across land uplift regions. *Journal of Ecology*, 109(4), 1774–1789. https://doi.org/10.1111/1365-2745.13601

Larmola, T., Tuittila, E.-S., Tiirola, M., Nykänen, H., Martikainen, P.J., Yrjälä, K., Tuomivirta, T., Fritze H. (2010) The role of *Sphagnum* mosses in the methane cycling of a boreal mire. *Ecology*, 91(8), 2356–2365. https://doi.org/10.1890/09-1343.1

Lavoie, C., Grosvernier, P., Girard, M., Marcoux, K. (2003) Spontaneous revegetation of mined peatlands: An useful restoration tool? *Wetlands Ecology and Management*, 11, 97–107.

Lavoie, C., Saint-Louis, A., Lachance, D. (2005) Vegetation dynamics on an abandoned vacuum-mined peatland: 5 years of monitoring. *Wetlands Ecology and Management*, 13(6), 621–633. https://doi.org/10.1007/s11273-005-0126-1

Lazcano, C., Robinson, C., Hassanpour, G., Strack, M. (2018) Short-term effects of fen peatland restoration through the moss layer transfer technique on the soil CO_2 and CH_4 efflux. *Ecological Engineering*, 125, 149–158. https://doi.org/10.1016/j.ecoleng.2018.10.018

Leifeld, J., Menichetti, L. (2018) The under-appreciated potential of peatlands in global climate change mitigation strategies. *Nature Communications*, 9(1), 1071, 7 pp. https://doi.org/10.1038/s41467-018-03406-6

Liu, C., Guêné-Nanchen, M., Rochefort, L. (2024) Improving restoration outcomes of boreal *Sphagnum*-dominated peatlands after peat-extraction: The key role of phosphorus fertilization. *Biological Conservation*, 298, 110770, 11 pp. https://doi.org/10.1016/j.biocon.2024.110770

Loisel, J., Gallego-Sala, A.V., Amesbury, M.J., Magnan, G. and 66 others (2021) Expert assessment of future vulnerability of the global peatland carbon sink. *Nature Climate Change*, 11(1), 70–77. https://doi.org/10.1038/s41558-020-00944-0

Nugent, K.A., I.B. Strachan, M. Strack, N.T. Roulet, Rochefort, L. (2018) Multi-year net ecosystem carbon balance of a restored peatland reveals a return to carbon sink. *Global Change Biology*, 24(12), 5751–5768. https://doi.org/10.1111/gcb.14449

Økland, R.H. (1989) *A Phytoecological Study of the Mire Northern Kisselbergmosen, SE. Norway. I. Introduction, Flora, Vegetation and Ecological Conditions*. Sommerfeltia 8(1), Botanical Garden and Museum, University of Oslo, 172 pp. https://doi.org/10.2478/som-1989-0001

Økland, R.H. (1990) A phytoecological study of the mire Northern Kisselbergmosen, SE Norway. III. Diversity and habitat niche relationships. *Nordic Journal of Botany*, 10(2), 191–220. https://doi.org/10.1111/j.1756-1051.1990.tb01766.x

Oksanen, J., Simpson, G.L., Blanchet, F.G., Kindt, R. and 30 others (2024) vegan: Community Ecology Package. Online at: https://cran.r-project.org/web/packages/vegan/index.html, accessed 03 Mar 2025.

Pacheco-Cancino, P.A., Carrillo-López, R.F., Sepulveda-Jauregui, A., Somos-Valenzuela, M.A. (2024) *Sphagnum* mosses, the impact of disturbances and anthropogenic management actions on their ecological role in CO_2 fluxes generated in peatland ecosystems. *Global Change Biology*, 30(1), e16972, 22 pp. https://doi.org/10.1111/gcb.16972

Petrone, R.M., Price, J.S., Waddington, J.M., Von Waldow, H. (2004) Surface moisture and energy exchange from a restored peatland, Québec, Canada. *Journal of Hydrology*, 295(1–4), 198–210. https://doi.org/10.1016/j.jhydrol.2004.03.009

Pilkington, M., Walker, J., Fry, C., Eades, P., Meade, R., Pollett, N., Rogers, T., Helliwell, T., Chandler, D., Fawcett, E., Keatley, T. (2021) Diversification of *Molinia*-dominated blanket bogs using *Sphagnum* propagules. *Ecological Solutions and Evidence*, 2(4), e12113, 19 pp. https://doi.org/10.1002/2688-8319.12113

Posit team (2024) RStudio: Integrated development for R (Posit PBC); URL http://www.rstudio.com/. Posit Software, Boston MA, http://www.posit.co/

Price, J.S., Whitehead, G.S. (2001) Developing hydrologic thresholds for *Sphagnum* recolonization on an abandoned cutover bog. *Wetlands*, 21(1), 32–40. https://doi.org/10.1672/0277-5212(2001)021[0032:DHTFSR]2.0.CO;2

Price, J.S., Rochefort, L., Quinty, F. (1998) Energy and moisture considerations on cutover peatlands: surface microtopography, mulch cover and *Sphagnum* regeneration. *Ecological Engineering*, 10(4), 293–312. https://doi.org/10.1016/S0925-8574(98)00046-9

Purre, A., Ilomets, M., Truus, L., Pajula, R., Sepp, K. (2020) The effect of different treatments of moss layer transfer technique on plant functional types' biomass in revegetated milled peatlands. *Restoration Ecology*, 28(6), 1584–1595. https://doi.org/10.1111/rec.13246

Putkinen, A., Tuittila, E.-S., Siljanen, H.M.P., Bodrossy, L., Fritze, H. (2018) Recovery of methane turnover and the associated microbial communities in restored cutover peatlands is strongly linked with increasing *Sphagnum* abundance. *Soil Biology and Biochemistry*, 116, 110–119. https://doi.org/10.1016/j.soilbio.2017.10.005

Quadra, G.R., Käärmelahti, S., van Dijk, G., Gaudig, G., Krebs, M., Prager, A., Koks, A.H.W., Vroom, R.J.E., Liu, W., Temmink, R.J.M., Fritz, C. (2025) The effect of water table fluctuation on 12 *Sphagnum* species during establishment: implications for peatland restoration and paludiculture. *Restoration Ecology* (in press), e14384, 11 pp. https://doi.org/10.1111/rec.14384

Quinty, F., Rochefort, L. (2003) *Peatland Restoration Guide: Second Edition*. Canadian Sphagnum Peat Moss Association, St Albert AB and New Brunswick Department of Natural Resources and Energy, Fredericton NB, 106 pp.

Robert, É.C., Rochefort, L., Garneau, M. (1999) Natural revegetation of two block-cut mined peatlands in eastern Canada. *Canadian Journal of Botany*, 77(3), 447–459.

Roberts, G., Fry, C., Jackson, D. (2022) *Sphagnum Practitioners' Guide: A Toolkit for Sphagnum Reintroduction into Blanket Bog*. Moors for the Future Partnership, Edale, UK. Online at: https://www.moorsforthefuture.org.uk/__data/assets/pdf_file/0028/96094/Sphagnum-Practitioners-Guide-2022.pdf, accessed 03 Mar 2025.

Robroek, B.J.M., van Ruijven, J., Schouten, M.G.C., Breeuwer, A., Crushell, P.H., Berendse, F., Limpens, J. (2009) *Sphagnum* re-introduction in degraded peatlands: The effects of aggregation, species identity and water table. *Basic and Applied Ecology*, 10(8), 697–706. https://doi.org/10.1016/j.baae.2009.04.005

Rochefort, L. (2000) *Sphagnum* —A keystone genus in habitat restoration. *The Bryologist*, 103(3), 503–508. https://doi.org/10.1639/0007-2745(2000)103[0503:SAKGIH]2.0.CO;2

Rochefort, L., Lode, E. (2006) Restoration of degraded boreal peatlands. In: Kelman Wieder, R., Vitt, D.H. (eds.) *Boreal Peatland Ecosystems*, Ecological Studies 188, Springer, Berlin-Heidelberg, 381–423. https://doi.org/10.1007/978-3-540-31913-9_17

Rochefort, L., Quinty, F., Campeau, S., Johnson, K., Malterer, T. (2003) North American approach to the restoration of *Sphagnum* dominated peatlands. *Wetlands Ecology and Management*, 11, 3–20.

Rochefort, L., Isselin-Nondedeu, F., Boudreau, S., Poulin, M. (2013) Comparing survey methods for monitoring vegetation change through time in a restored peatland. *Wetlands Ecology and Management*, 21(1), 71–85. https://doi.org/10.1007/s11273-012-9280-4

Rydin, H. (1993) Interspecific competition between *Sphagnum* mosses on a raised bog. *Oikos*, 66(3), 413–423. https://doi.org/10.2307/3544935

Rydin, H. (1997) Competition between *Sphagnum* species under controlled conditions. *The Bryologist*, 100(3), 302–307. https://doi.org/10.2307/3244500

Rydin, H., Gunnarsson, U., Sundberg. S. (2006) The role of *Sphagnum* in peatland development and persistence. In: Kelman Wieder, R., Vitt, D.H. (eds.) *Boreal Peatland Ecosystems*, Ecological Studies 188, Springer, Berlin-Heidelberg, 47–65. https://doi.org/10.1007/978-3-540-31913-9_4

Schipperges, B., Rydin, H. (1998) Response of photosynthesis of *Sphagnum* species from contrasting microhabitats to tissue water content and repeated desiccation. *New Phytologist*, 140(4), 677–684. https://doi.org/10.1046/j.1469-8137.1998.00311.x

Schouwenaars, J.M. (1988) The impact of water management upon groundwater fluctuations in a disturbed bog relict. *Agricultural Water Management*, 14, 439–449. https://doi.org/10.1016/0378-3774(88)90096-0

Schouwenaars, J.M., Gosen, A.M. (2007) The sensitivity of *Sphagnum* to surface layer conditions in a re-wetted bog: a simulation study of water stress. *Mires and Peat*, 2, 02, 19 pp.

Smolders, A.J.P., Tomassen, H.B.M., van Mullekom, M., Lamers, L.P.M., Roelofs, J.G.M. (2003) Mechanisms involved in the re-establishment of *Sphagnum*-dominated vegetation in rewetted bog remnants. *Wetlands Ecology and Management*, 11(6), 403–418. https://doi.org/10.1023/B:WETL.0000007195.25180.94

Soro, A., Sundberg, S., Rydin, H. (1999) Species diversity, niche metrics and species associations in harvested and undisturbed bogs. *Journal of Vegetation Science*, 10(4), 549–560. https://doi.org/10.2307/3237189

Sottocornola, M., Boudreau, S., Rochefort. L. (2007) Peat bog restoration: Effect of phosphorus on plant re-establishment. *Ecological Engineering*, 31(1), 29–40. https://doi.org/10.1016/j.ecoleng.2007.05.001

Strack, M., Price, J.S. (2009) Moisture controls on carbon dioxide dynamics of peat-*Sphagnum* monoliths. *Ecohydrology*, 2(1), 34–41. https://doi.org/10.1002/eco.36

Strack, M., Zuback, Y.C.A. (2013) Annual carbon balance of a peatland 10 yr following restoration. *Biogeosciences*, 10(5), 2885–2896. https://doi.org/10.5194/bg-10-2885-2013

Strack, M., Davidson, S.J., Hirano, T., Dunn, C.

(2022) The potential of peatlands as nature-based climate solutions. *Current Climate Change Reports*, 8(3), 71–82. https://doi.org/10.1007/s40641-022-00183-9

Sundberg, S., Rydin, H. (2002) Habitat requirements for establishment of *Sphagnum* from spores. *Journal of Ecology*, 90(2), 268–278. https://doi.org/10.1046/j.1365-2745.2001.00653.x

Swindles, G.T., Morris, P.J., Wheeler, J., Smith, M.W., Bacon, K.L., Turner, T.E., Headley, A., Galloway, J.M. (2016) Resilience of peatland ecosystem services over millennial timescales: evidence from a degraded British bog. *Journal of Ecology*, 104(3), 621–636. https://doi.org/10.1111/1365-2745.12565

Tiemeyer, B., Freibauer, A., Borraz, E.A., Augustin, J. and 18 others (2020) A new methodology for organic soils in national greenhouse gas inventories: Data synthesis, derivation and application. *Ecological Indicators*, 109, 105838, 14 pp. https://doi.org/10.1016/j.ecolind.2019.105838

Triisberg, T., Karofeld, E., Paal, J. (2013) Factors affecting the re-vegetation of abandoned extracted peatlands in Estonia: a synthesis from field and greenhouse studies. *Estonian Journal of Ecology*, 62(3), 192–211. https://doi.org/10.3176/eco.2013.3.02

Triisberg-Uljas, T., Vellak, K., Karofeld, E. (2018) Application of oil-shale ash and straw mulch promotes the revegetation of extracted peatlands. *Ecological Engineering*, 110, 99–106. https://doi.org/10.1016/j.ecoleng.2017.10.006

Tuittila, E.-S., Vasander, H., Laine, J. (2003) Success of re-introduced *Sphagnum* in a cut-away peatland. *Boreal Environment Research*, 8(3), 245–250.

Turetsky, M.R. (2003) The role of bryophytes in carbon and nitrogen cycling. *The Bryologist*, 106(3), 395–409.

UNEP (2022) *Global Peatlands Assessment: The State of the World's Peatlands. Evidence for Action Toward the Conservation, Restoration, and Sustainable Management of Peatlands. Main Report*. Global Peatlands Initiative, United Nations Environment Programme (UNEP), Nairobi, 418 pp. https://doi.org/10.59117/20.500.11822/41222

UNEP/FAO (2023) *Action Plan for the UN Decade on Ecosystem Restoration, 2021–2030*. Version April 2023, United Nations Environment Programme (UNEP) and Food and Agriculture Organization of the United Nations (FAO), 134 pp. https://wedocs.unep.org/20.500.11822/42095

Vehtari, A., Gabry, J., Magnusson, M., Yao, Y., Bürkner, P.-C., Paananen, T., Gelman, A., Goodrich, B., Piironen, J., Nicenboim, B., Lindgren, L. (2024) loo: Efficient Leave-One-Out Cross-Validation and WAIC for Bayesian Models. Online at: https://cran.r-project.org/web/packages/loo/index.html

Waddington, J.M., Rochefort, L., Campeau, S. (2003a) *Sphagnum* production and decomposition in a restored cutover peatland. *Wetlands Ecology and Management*, 11, 85–95.

Waddington, J.M., Greenwood, M.J., Petrone, R.M., Price, J.S. (2003b) Mulch decomposition impedes recovery of net carbon sink function in a restored peatland. *Ecological Engineering*, 20(3), 199–210. https://doi.org/10.1016/S0925-8574(03)00025-9

Waddington, J.M., Strack, M., Greenwood, M.J. (2010) Toward restoring the net carbon sink function of degraded peatlands: Short-term response in CO_2 exchange to ecosystem-scale restoration. *Journal of Geophysical Research: Biogeosciences*,115(G1), G01008, 13 pp. https://doi.org/10.1029/2009JG001090

Wagner, D.J., Titus, J.E. (1984) Comparative desiccation tolerance of two *Sphagnum* mosses. *Oecologia*, 62(2), 182–187. https://doi.org/10.1007/BF00379011

Warburton, J. (2003) Wind-splash erosion of bare peat on UK upland moorlands. *CATENA*, 52(3–4), 191–207. https://doi.org/10.1016/S0341-8162(03)00014-6

Wen, X., Unger, V., Jurasinski, G., Koebsch, F., Horn, F., Rehder, G., Sachs, T., Zak, D., Lischeid, G., Knorr, K.-H., Böttcher, M.E., Winkel, M., Bodelier, P.L.E., Liebner, S. (2018) Predominance of methanogens over methanotrophs in rewetted fens characterized by high methane emissions. *Biogeosciences*, 15(21), 6519–6536. https://doi.org/10.5194/bg-15-6519-2018

Wilson, D., Tuittila, E.-S., Alm, J., Laine, J., Farrell, E.P., Byrne, K.A. (2007) Carbon dioxide dynamics of a restored maritime peatland. *Ecoscience*, 14(1), 71–80. https://doi.org/10.2980/1195-6860(2007)14[71:CDDOAR]2.0.CO;2

Wittram, B.W., Roberts, G., Buckler, M., King, L., Walker, J.S. (2015) *A Practitioners Guide to Sphagnum Reintroduction*. Moors for the Future Partnership, Edale, UK, 39 pp. Online at: https://www.moorsforthefuture.org.uk/__data/assets/pdf_file/0023/85406/Practitioners-guide-to-the-reintroductin-of-sphagnum.pdf, accessed 03 Mar 2025.

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. *Restoration Ecology*, 30(3), e13514, 12 pp. https://doi.org/10.1111/rec.13514

Zając, E., Zarzycki, J., Ryczek, M. (2018) Substrate quality and spontaneous revegetation of extracted peatland: case study of an abandoned Polish mountain bog. *Mires and Peat*, 21, 12, 14 pp. https://doi.org/10.19189/MaP.2017.OMB.310

Zarzycki, J., Zając, E., Vončina, G. (2022) Bryophytes and vascular plants on peat extraction sites - which factors influence their growth? *Journal for Nature Conservation*, 70, 126287, 9 pp. https://doi.org/10.1016/j.jnc.2022.126287

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Appendix

Climate and weather data

All weather data were downloaded from seklima.met.no. The closest weather stations for our study sites were Åfjord II (precipitation and temperatures) for HM and BSM, whereas Overhalla-Unnset (precipitation) and Namsos lufthavn (temperatures) were the closest weather stations for VSM. While the weather station near HM and BSM provided directly relatable data due to proximity and similar elevation, those nearest to VSM were located at a considerable distance, meaning their data required adjustments. These adjustments accounted for dissimilarities in elevation and surrounding topography, and for proximity to the sea. Precipitation data from Overhalla-Unnset were adjusted by +10 %, based on long-term precipitation data from an older weather station near VSM. Additionally, temperatures from Namsos lufthavn were adjusted by -0.6 °C to accommodate the 100 m altitude difference between the weather station and Vestersetermyra. The maximum daily temperature throughout the growing season during our study period was around 30 °C each year. Most years had more than 50 % of the days within the growing season with temperatures reaching >15 °C.

Table A1. Functional trait groups and cover components, with species lists. Nomenclature follows Artsdatabanken's Nortaxa for vascular plants and bryophytes ([Nortaxa - Norsk taksonomisk register (artsdatabanken.no)](https://nortaxa.artsdatabanken.no)), and the classification of PFTs follows Hagen *et al*. (2013).

Plant functional type (PFT)/ component	Description	Species included
Sphagnum	Living *Sphagnum* species.	*S. angustifolium,* *S. capillifolium,* *S. tenellum,* *S. medium,* *S. papillosum,* *S. divinum,* *S. balticum,* *S. compactum,* *S. majus,* *S. riparium,* *S. russowii,* *S. fuscum,* *S. rubellum,* *S. pulchrum*
Dead *Sphagnum*	Visibly dead and decomposing *Sphagnum.*	
Polytrichum	*Polytrichum* species.	*P. strictum, P. commune*
Racomitrium	*Racomitrium* species.	*R. lanuginosum*
Other bryophytes	Includes bryophytes from all other genera.	*Dicranum* sp., *Pogonatum dentatum,* *Pleurozium schreberi,* *Rhytiadelphus* sp., *Aulacomnium palustre,* *Hypnum* sp., *Ptilidium ciliare,* *Straminergon stramineum,* *Scorpidium* sp., *Hylocomium splendens,* *Plagiothecium undulatum,* *Warnstorfia fluitans*

Plant functional type (PFT)/ component	Description	Species included
Liverworts	Liverworts (*Marchantiophyta*).	*Mylia anomala*, *Pellia* sp., Other small thread-like species e.g. *Cephalozia* sp., *Gynocolea inflata*, *Calypogeia* sp., *Lophozia s.l.* sp.
Lichen	Lichens.	*Cladonia stellaris,* *Cladonia arbuscula,* *Cladonia rangiferina,* *Cladonia* sp.,
Carex	*Carex* species.	*Rhynchospora alba,* *Carex rostrata,* *Carex pauciflora*
Trichophorum	*Trichophorum* species.	*T. cespitosum*
Eriophorum	*Eriophorum* species.	*E. vaginatum, E. angustifolium*
Rushes	*Juncaceae* species.	
Grasses	*Poaceae* species.	*Molinia caerulea,*
Evergreen ericoids	Evergreen dwarf shrubs.	*Calluna vulgaris,* *Vaccinium vitis-idaea,* *Andromeda polifolia,* *Empetrum nigrum,* *Erica tetralix*
Deciduous *Vaccinium*	Leaf-shedding dwarf shrubs.	*Vaccinium myrtillus,* *Vaccinium uliginosum*
Herbs	Herbs/forbs.	*Oxycoccus palustris,* *Oxycoccus microcarpus,* *Drosera rotundifolia,* *Drosera anglica,* *Rubus chamaemorus*
Narthecium		*Narthecium ossifragum*
Seedlings	Tiny seedlings difficult to identify to species.	
Shrubs and trees	All species that will eventually become a tree or a shrub of > 0.5 m height.	*Pinus sylvestris,* *Picea abies,* *Betula pubescens,* *Salix caprea,* *Salix* sp.
Ferns	Ferns and horsetails.	*Equisetum fluviatile*
Bare peat	Bare soil.	
Mulch	The straw mulch applied by us.	
Litter	Withered leaves and stems, thin branches/twigs.	
Dead wood	Larger pieces of broken branches.	
Temporary water	Temporary standing water Not possible to identify other PFTs underneath.	
Other	E.g. faeces of moose.	

Table A2. Percentage (%) cover of plant functional types (PFTs) with mean, median and max values; with SE of all variables evaluated in field within the treatments C (control), M (mulch only), S (*Sphagnum* and mulch), and R (references).

		Year 0				Year 2				Year 3			
PFTs and other groups	Treatment	mean	median	max	SE	mean	median	max	SE	mean	median	max	SE
Bare peat	C	90.78	95.00	100.00	1.83	90.94	95.00	100.00	2.44	80.94	90.00	99.00	3.84
	M	89.13	95.00	100.00	2.29	24.91	20.00	60.00	2.85	38.28	34.00	85.00	4.40
	S	88.66	90.00	100.00	1.53	6.19	3.00	50.00	1.04	9.67	5.00	45.00	1.42
	R									0.03	0.00	1.00	0.03
Deciduous *Vaccinium*	C	0.00	0.00	0.00	0.00	0.01	0.00	0.10	0.00	0.00	0.00	0.10	0.00
	M	0.00	0.00	0.00	0.00	0.01	0.00	0.10	0.00	0.01	0.00	0.10	0.00
	S	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.04	0.00	2.00	0.03
	R									0.00	0.00	0.00	0.00
Carex	C	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.10	0.00
	M	0.00	0.00	0.00	0.00	0.00	0.00	0.10	0.00	0.03	0.00	1.00	0.03
	S	0.00	0.00	0.00	0.00	0.00	0.00	0.10	0.00	0.05	0.00	1.00	0.03
	R									1.17	0.00	11.00	0.47
Dead *Sphagnum*	C	2.61	0.00	35.00	1.22	0.57	0.00	15.00	0.47	0.69	0.00	17.00	0.55
	M	1.82	0.00	15.00	0.76	0.03	0.00	1.00	0.03	0.22	0.00	4.00	0.14
	S	1.43	0.00	10.00	0.35	0.70	0.00	10.00	0.26	1.19	0.00	45.00	0.72
	R									0.10	0.00	1.00	0.06
Dead wood	C	0.73	0.00	6.00	0.25	1.07	0.00	10.00	0.41	0.78	0.00	10.00	0.36
	M	0.47	0.00	5.00	0.19	0.41	0.00	3.00	0.17	0.28	0.00	3.00	0.13
	S	0.48	0.00	5.00	0.15	0.07	0.00	2.00	0.04	0.25	0.00	3.00	0.08
	R									0.00	0.00	0.00	0.00

		Year 0				Year 2				Year 3			
PFTs and other groups	Treatment	mean	median	max	SE	mean	median	max	SE	mean	median	max	SE
Eriophorum	C	0.08	0.00	1.00	0.04	2.46	1.00	15.00	0.69	5.79	2.50	50.00	1.73
	M	0.67	0.00	20.00	0.62	3.45	0.10	80.00	2.49	4.17	2.00	30.00	1.14
	S	0.23	0.00	10.00	0.18	1.98	0.55	30.00	0.58	6.83	4.00	60.00	1.38
	R									3.33	3.00	12.00	0.45
Evergreen ericoids	C	0.05	0.00	1.00	0.03	0.37	0.00	6.00	0.20	0.73	0.00	6.00	0.29
	M	0.10	0.00	1.00	0.05	1.03	0.00	15.00	0.54	3.40	0.10	45.00	1.62
	S	0.03	0.00	1.00	0.02	0.72	0.10	10.00	0.22	2.28	0.10	25.00	0.54
	R									10.37	10.00	30.00	1.29
Grasses	C	0.00	0.00	0.10	0.00	0.04	0.00	1.00	0.03	0.29	0.00	4.00	0.15
	M	0.01	0.00	0.10	0.00	0.01	0.00	0.10	0.01	0.13	0.00	2.00	0.07
	S	0.01	0.00	0.10	0.00	0.02	0.00	1.00	0.02	0.23	0.00	4.00	0.09
	R									0.00	0.00	0.00	0.00
Herbs	C	0.00	0.00	0.10	0.00	0.23	0.00	6.00	0.19	0.20	0.00	3.00	0.11
	M	0.00	0.00	0.10	0.00	0.18	0.00	2.00	0.09	0.30	0.00	3.00	0.12
	S	0.00	0.00	0.10	0.00	0.07	0.00	2.00	0.03	0.18	0.00	2.00	0.05
	R									1.29	1.00	11.00	0.39
Lichen	C	0.00	0.00	0.00	0.00	0.00	0.00	0.10	0.00	0.03	0.00	1.00	0.03
	M	0.00	0.00	0.10	0.00	0.02	0.00	0.10	0.01	0.04	0.00	1.00	0.03
	S	0.00	0.00	0.00	0.00	0.01	0.00	0.10	0.00	0.01	0.00	0.10	0.00
	R									2.83	0.00	35.00	1.33
Litter	C	4.58	3.00	20.00	0.79	0.54	0.00	4.00	0.18	1.03	0.10	7.00	0.34
	M	5.88	4.00	40.00	1.37	0.33	0.00	5.00	0.18	0.51	0.00	5.00	0.20

		Year 0				Year 2				Year 3			
PFTs and other groups	Treatment	mean	median	max	SE	mean	median	max	SE	mean	median	max	SE
Litter *(continued)*	S	6.47	4.00	50.00	1.09	0.57	0.00	20.00	0.33	0.69	0.10	7.00	0.18
	R									14.20	10.00	45.00	2.16
Liverworts	C	0.00	0.00	0.00	0.00	0.01	0.00	0.10	0.00	1.63	0.00	15.00	0.71
	M	0.00	0.00	0.00	0.00	0.04	0.00	1.00	0.03	3.53	0.55	45.00	1.46
	S	0.01	0.00	0.10	0.00	0.33	0.00	5.00	0.10	1.67	0.10	11.00	0.32
	R									6.74	1.00	70.00	2.81
Mulch	C	0.00	0.00	0.00	0.00	0.05	0.00	1.00	0.03	0.00	0.00	0.00	0.00
	M	0.00	0.00	0.00	0.00	62.19	62.50	90.00	2.80	29.32	30.00	70.00	3.13
	S	0.00	0.00	0.00	0.00	47.42	50.00	75.00	2.22	11.03	8.00	35.00	1.21
	R									0.00	0.00	0.00	0.00
Narthecium	C	0.00	0.00	0.00	0.00	0.00	0.00	0.10	0.00	0.01	0.00	0.10	0.01
	M	0.00	0.00	0.00	0.00	0.01	0.00	0.10	0.00	0.03	0.00	0.10	0.01
	S	0.00	0.00	0.00	0.00	0.01	0.00	0.10	0.00	0.03	0.00	0.10	0.01
	R									14.80	15.00	55.00	2.75
Other bryophytes	C	1.69	0.10	20.00	0.86	2.99	0.10	40.00	1.39	5.23	0.10	51.00	1.89
	M	1.85	0.00	25.00	0.97	6.14	1.00	25.00	1.54	12.29	2.00	70.00	3.30
	S	2.54	0.10	50.00	0.99	1.92	1.00	15.10	0.39	4.12	1.00	25.00	0.88
	R									3.25	1.00	50.00	1.67
Polytrichum	C	0.20	0.00	2.00	0.09	0.45	0.10	5.00	0.18	1.46	0.05	25.00	0.79
	M	0.30	0.00	3.00	0.13	0.71	0.05	5.00	0.22	1.96	0.10	10.00	0.51
	S	0.30	0.00	4.00	0.10	0.98	0.10	12.00	0.28	3.45	0.10	40.00	1.06
	R									0.07	0.00	1.00	0.05

		Year 0				Year 2				Year 3			
PFTs and other groups	Treatment	mean	median	max	SE	mean	median	max	SE	mean	median	max	SE
Rushes	C	0.00	0.00	0.10	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
	M	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
	S	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
	R									0.00	0.00	0.00	0.00
Seedlings	C	0.02	0.00	0.10	0.01	0.04	0.00	1.00	0.03	0.13	0.00	3.00	0.09
	M	0.06	0.00	1.00	0.03	0.02	0.00	0.10	0.01	0.16	0.00	2.00	0.07
	S	0.02	0.00	0.10	0.01	0.03	0.00	1.00	0.02	0.08	0.00	1.00	0.03
	R									0.03	0.00	0.10	0.01
Shrubs and trees	C	0.00	0.00	0.00	0.00	0.01	0.00	0.10	0.01	0.00	0.00	0.00	0.00
	M	0.00	0.00	0.00	0.00	0.19	0.00	2.00	0.08	0.35	0.00	6.00	0.21
	S	0.05	0.00	2.00	0.04	0.54	0.00	20.00	0.32	0.41	0.00	15.00	0.25
	R									1.64	0.00	27.00	0.97
Sphagnum	C	0.15	0.00	3.00	0.09	1.06	0.10	10.00	0.36	1.97	0.10	13.00	0.61
	M	0.34	0.00	5.00	0.17	1.71	0.10	10.00	0.50	4.61	1.00	30.00	1.36
	S	0.33	0.00	8.00	0.15	43.30	45.00	90.00	2.86	65.64	75.00	98.00	3.50
	R									70.77	77.50	100.00	5.01
Trichophorum	C	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
	M	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.06	0.00	2.00	0.06
	S	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
	R									9.13	8.00	25.00	1.18

Table A3. Results of the Bayesian regression models. All values are logit-transformed. Draws were obtained using the No-U-Turn Sampler (NUTS). For each parameter, Bulk_ESS and Tail_ESS indicate the effective sample size, while Rhat is the potential scale reduction factor for split chains, with Rhat equal to 1 indicating convergence.

Model 1
Family: beta
Links: mu = logit; phi = identity
Formula: Sphagnum_beta/100 ~ Treatment + WT_mean + (1 | Location/Sublocation)
Data: H1.Sph.revWT.imp (Number of observations: 128)
Draws: 4 chains, each with iter = 4000; warmup = 1000; thin = 1;
total post-warmup draws = 12000

Model 2
Family: beta
Links: mu = logit; phi = identity
Formula: Sphagnum_beta/100 ~ WT_mean + (1 | Location/Sublocation)
Data: H1.Sph.revWT.imp (Number of observations: 128)
Draws: 4 chains, each with iter = 4000; warmup = 1000; thin = 1;
total post-warmup draws = 12000

Multilevel Hyperparameters:

	Model 1							Model 2						
~Location (Number of levels: 3)														
	Estimate	Est. Error	l-95% CI	u-95% CI	Rhat	Bulk ESS	Tail ESS	Estimate	Est. Error	l-95% CI	u-95% CI	Rhat	Bulk ESS	Tail ESS
sd (Intercept)	1.28	1.02	0.09	3.94	1.00	3707	4350	0.76	0.76	0.03	2.88	1.00	3634	4848
~Location:Sublocation (Number of levels: 8)														
	Estimate	Est. Error	l-95% CI	u-95% CI	Rhat	Bulk ESS	Tail ESS	Estimate	Est. Error	l-95% CI	u-95% CI	Rhat	Bulk ESS	Tail ESS
sd (Intercept)	0.65	0.37	0.18	1.55	1.00	3150	4509	0.37	0.29	0.02	1.08	1.00	3083	4597
Regression Coefficients:														
	Estimate	Est. Error	l-95% CI	u-95% CI	Rhat	Bulk ESS	Tail ESS	Estimate	Est. Error	l-95% CI	u-95% CI	Rhat	Bulk ESS	Tail ESS
Intercept	-3.17	0.91	-5.10	-1.33	1.00	6066	6058	-0.85	0.62	-2.08	0.49	1.00	5129	5035
TreatmentM	0.34	0.26	-0.17	0.86	1.00	12010	8597							
TreatmentS	3.67	0.29	3.08	4.24	1.00	9432	8520							
WT_mean	-0.58	3.75	-7.75	7.20	1.00	5832	6481	-0.28	2.46	-5.06	4.63	1.00	6992	5703
Further Distributional Parameters:														
	Estimate	Est. Error	l-95% CI	u-95% CI	Rhat	Bulk ESS	Tail ESS	Estimate	Est. Error	l-95% CI	u-95% CI	Rhat	Bulk ESS	Tail ESS
phi	3.58	0.59	2.53	4.88	1.00	10328	3.58	0.76	0.08	0.61	0.94	1.00	12543	8594

Table A4. Summary of mean percentage cover with SE of *Sphagnum* at plot level for sublocation within location.

LOCATION	SUBLOCATION	C	M	S	R1	R2
BSM	1	0.05 ± 0.03	1.03 ± 0.70	79.13 ± 2.22	-	-
	2	3.78 ± 3.08	5.75 ± 1.93	93.25 ± 2.08	-	-
	3	6.00 ± 1.68	15.00 ± 5.40	88.38 ± 2.99	-	-
	Reference	-	-	-	56.60 ± 14.60	84.00 ± 6.96
HM	1	0.03 ± 0.03	0.03 ± 0.03	55.63 ± 9.23	-	-
	2	0.03 ± 0.03	7.80 ± 7.40	87.50 ± 3.13	-	-
	3	0.53 ± 0.49	0.78 ± 0.47	51.25 ± 5.73	-	-
	Reference	-	-	-	74.00 ± 12.19	75.40 ± 10.92
VSM	1	0.55 ± 0.26	1.50 ± 0.50	28.75 ± 4.41	-	-
	2	5.28 ± 1.91	5.00 ± 2.20	41.25 ± 10.68	-	-
	Reference	-	-	-	86.60 ± 11.69	48.00 ± 11.58

Table A5. Summary of mean cover and proportion with SE of all *Sphagnum* species within locations.

Location	Species	Cover (%)	Proportion
BSM	*S. angustifolium*	23.63 ± 1.68	0.37 ± 0.02
	S. capillifolium	14.91 ± 0.86	0.23 ± 0.01
	S. papillosum	9.74 ± 0.78	0.15 ± 0.01
	S. tenellum	15.25 ± 1.37	0.24 ± 0.01
	S. balticum	0.54 ± 0.16	0.01 ± 0.00
	S. compactum	0.47 ± 0.15	0.01 ± 0.00
	S. divinum	0.03 ± 0.02	0.00 ± 0.00
	S. majus	0.00 ± 0.00	0.00 ± 0.00
	S. riparium	0.03 ± 0.03	0.00 ± 0.00
	S. russowi	0.00 ± 0.00	0.00 ± 0.00
	Unidentified *Sphagnum*	0.05 ± 0.05	0.00 ± 0.00
HM	*S. angustifolium*	38.20 ± 2.56	0.63 ± 0.02
	S. capillifolium	8.23 ± 0.72	0.14 ± 0.01
	S. papillosum	4.95 ± 0.60	0.08 ± 0.01
	S. tenellum	9.65 ± 1.17	0.16 ± 0.01
	S. balticum	0.10 ± 0.10	0.00 ± 0.00
	S. compactum	0.00 ± 0.00	0.00 ± 0.00
	S. divinum	0.03 ± 0.03	0.00 ± 0.00
	S. majus	0.00 ± 0.00	0.00 ± 0.00
	S. riparium	0.00 ± 0.00	0.00 ± 0.00
	S. russowi	0.00 ± 0.00	0.00 ± 0.00
	Unidentified *Sphagnum*	0.00 ± 0.00	0.00 ± 0.00
VSM	*S. angustifolium*	22.53 ± 2.70	0.63 ± 0.03
	S. capillifolium	3.50 ± 0.58	0.10 ± 0.02
	S. papillosum	6.20 ± 1.43	0.17 ± 0.02
	S. tenellum	2.45 ± 0.31	0.07 ± 0.01
	S. balticum	0.00 ± 0.00	0.00 ± 0.00
	S. compactum	0.22 ± 0.11	0.01 ± 0.00
	S. divinum	0.05 ± 0.03	0.00 ± 0.00
	S. majus	0.16 ± 0.16	0.00 ± 0.00
	S. riparium	0.00 ± 0.00	0.00 ± 0.00
	S. russowi	0.34 ± 0.14	0.01 ± 0.01
	Unidentified *Sphagnum*	0.11 ± 0.08	0.00 ± 0.02

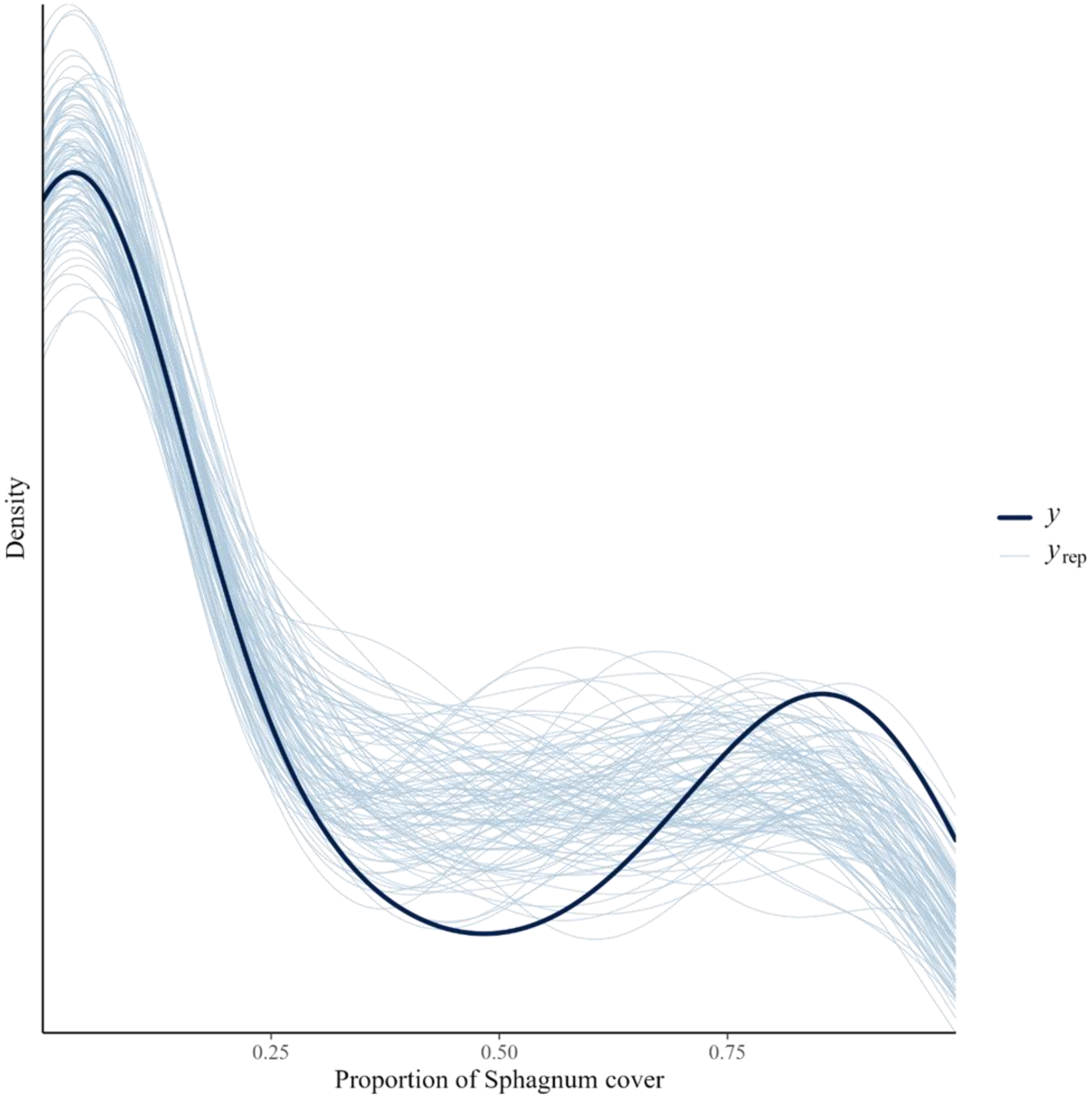

Figure A1. Posterior predictive check (PPC) displayed as a density overlay plot comparing the observed data (solid dark line) of *Sphagnum* cover with 100 simulated datasets (y_{rep}, light blue lines) from the posterior predictive distribution of the Bayesian regression model H1.

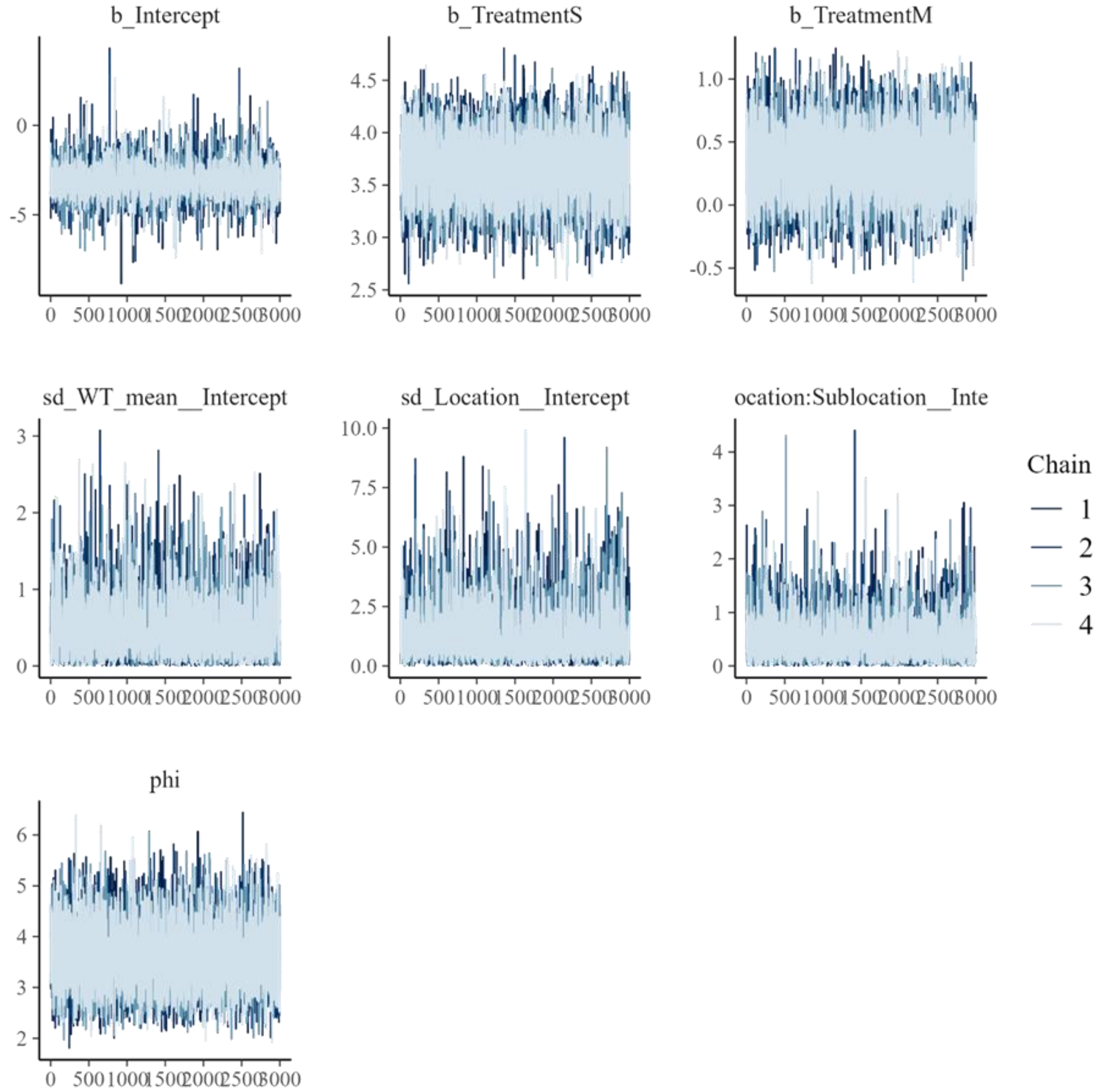

Figure A2. Trace plots showing the sampling behaviour of the main parameters from model H1, including the intercept (b_Intercept), treatment effects (b_TreatmentS and b_TreatmentM), random intercept standard deviations for water table (sd_WT_mean__Intercept), location (sd_Location__Intercept), and sublocation (sd_Location), along with the dispersion parameter (phi). Each trace plot illustrates the convergence and mixing of the MCMC chains across iterations, with the iteration number on the x-axis and the sampled parameter value on the y-axis.

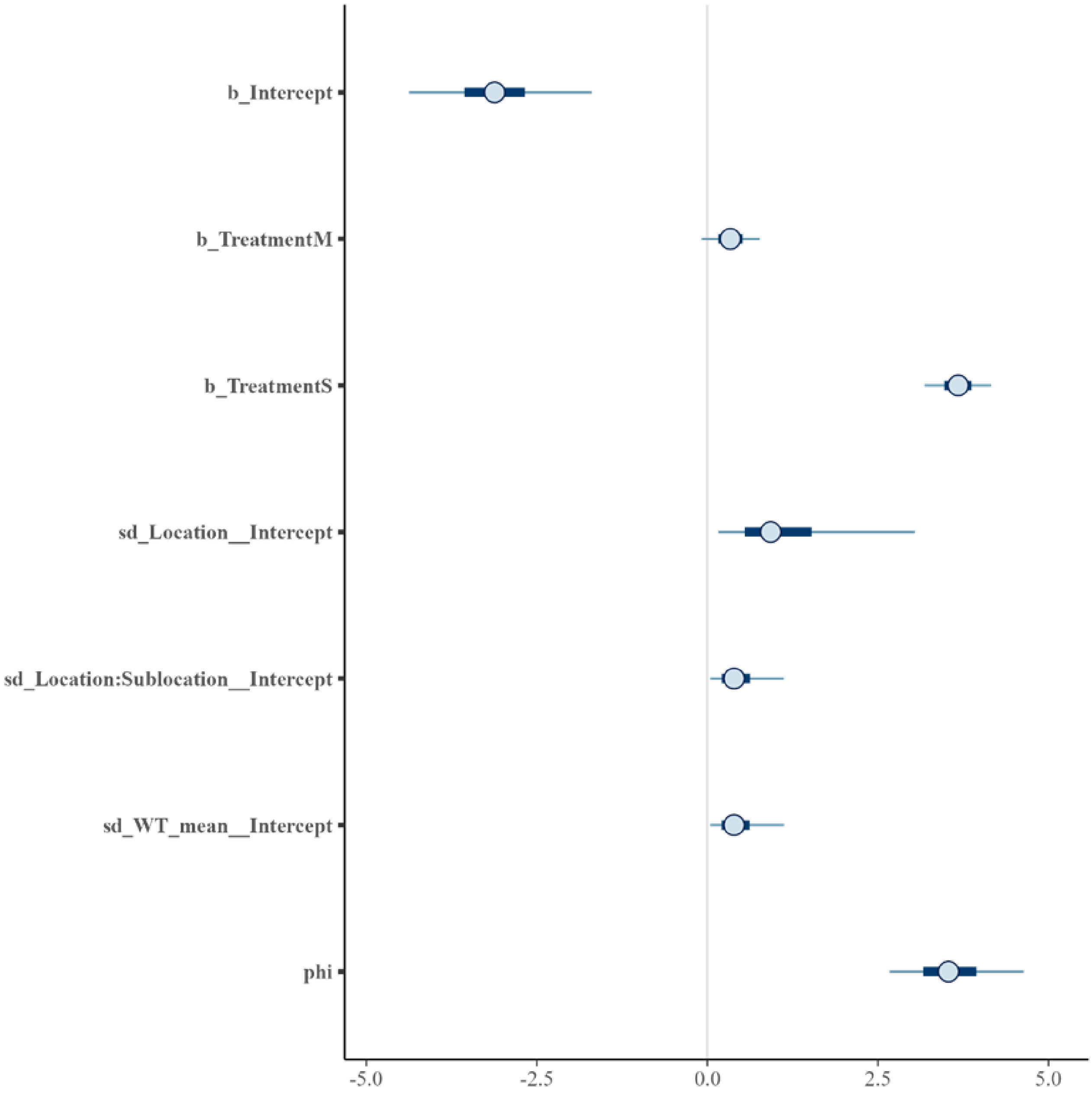

Figure A3. Posterior distributions of the main parameters from the Bayesian regression model H1: intercept (b_Intercept), treatment effects (b_TreatmentS and b_TreatmentM), random intercept standard deviations for water-table (sd_WT_mean__Intercept), location (sd_Location__Intercept), and sublocation (sd_Location:Sublocation__Intercept), as well as the dispersion parameter (phi). The box plots illustrate the uncertainty in the parameter estimates, highlighting the credible intervals derived from the posterior distributions.

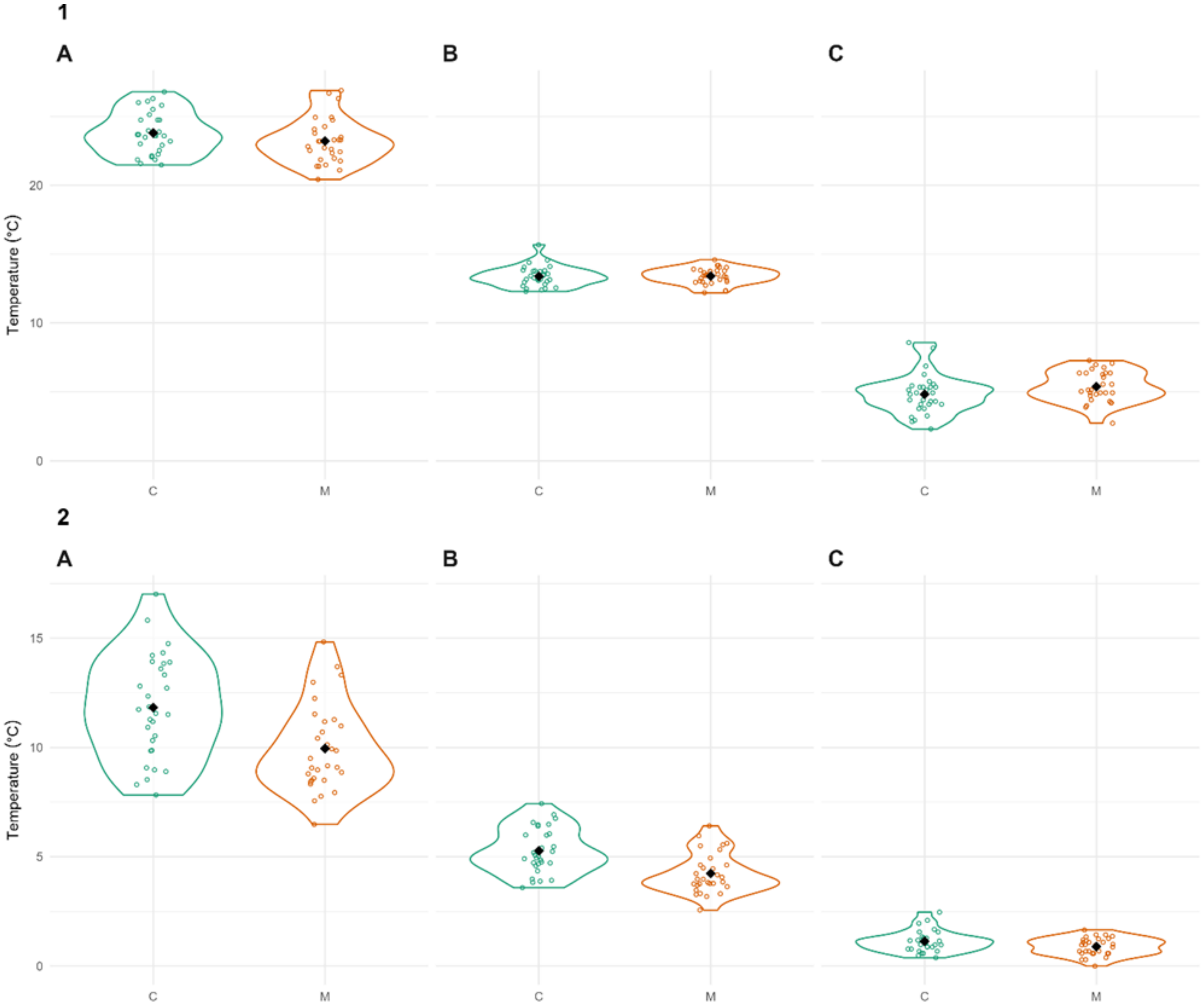

Figure A4. Violin plots of (1) averaged daily soil temperatures and (2) averaged daily soil temperature fluctuations during the first growing season for control (C) and mulch (M) treatments. Panels show A) maximum, B) mean, and C) minimum values.

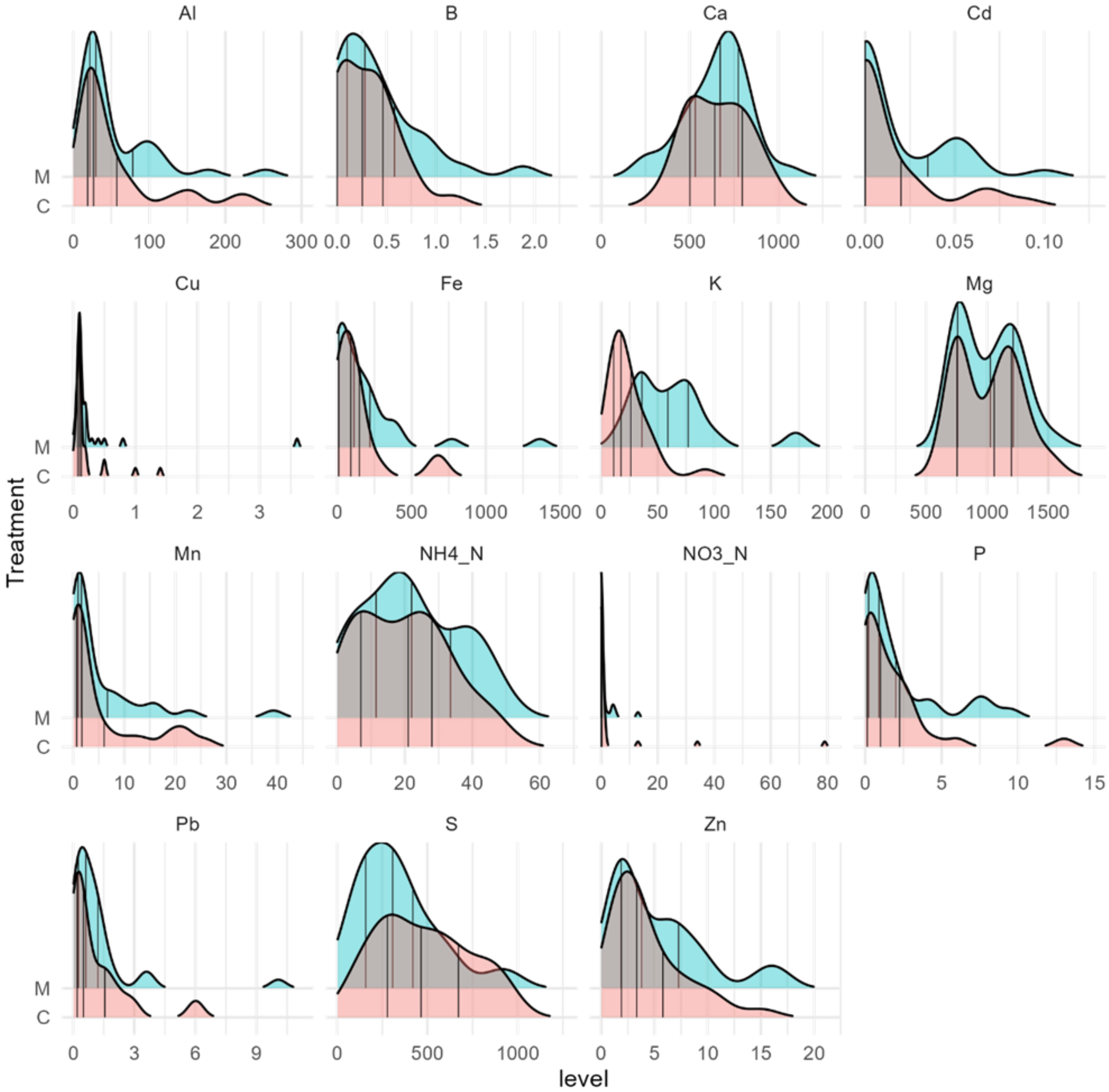

Figure A5. Density ridgeline plot showing the distribution of nutrient levels given as μg nutrient/10 cm² ion-exchange membrane surface area over a time period of 15 weeks across the treatments C (control) and M (mulch) at Year 0 (before the experiment was initiated). The plot includes quantile lines (0.25, 0.5, 0.75).

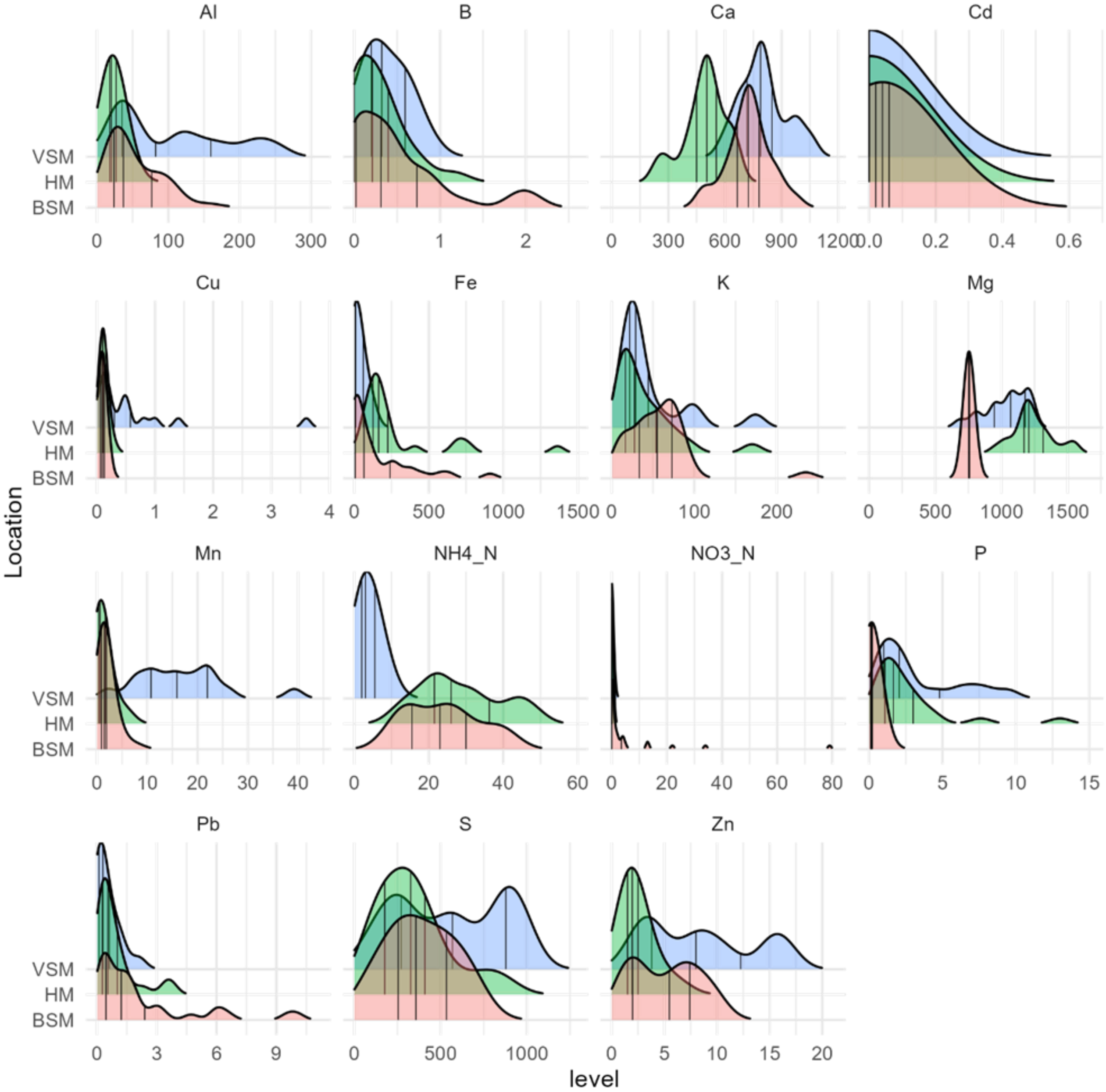

Figure A6. Density ridgeline plot showing the distribution of nutrient levels given as µg nutrient/10 cm² ion-exchange membrane surface area over a time period of 15 weeks across the various locations (VSM: Vestersetermyra, HM: Stormyra, BSM: Bukkastakkmyra) at Year 0. The Figure combines data from the control and mulch treatments. The plot includes quantile lines (0.25, 0.50, 0.75).

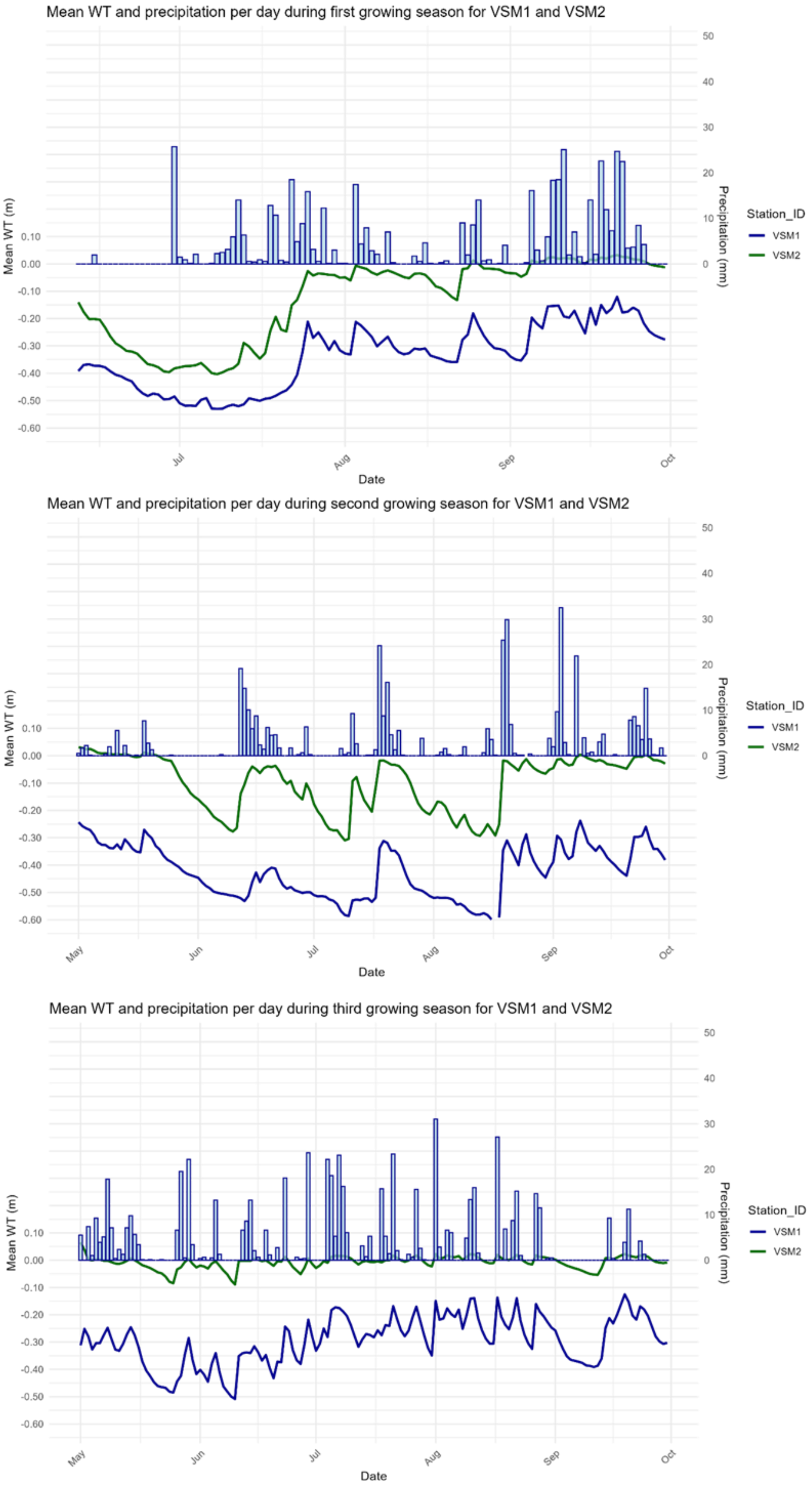

Figure A7. Daily mean water table (WT; solid line, left y-axis) and total precipitation (bars, right y-axis) at the sublocations VSM1 and VSM2. The water-table fluctuations mainly reflect precipitation, with dry periods drawing the water table down.