

Catchment-based approach for water table management with irrigation for cultivated peatlands

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ABSTRACT

Controlled drainage and subsurface irrigation have been proposed to enable shallow-drained agriculture in organic soils and mitigate greenhouse gas (GHG) emissions from intensively cultivated peatlands. However, the effects of current drainage practices on peatland water table depth (WTD) and the potential of using runoff from upstream catchment areas to adjust WTD in northern conditions are still poorly understood. To address these issues, WTD monitoring was initiated on 13 cultivated peatlands with different drainage systems in the flat western coastal region of Finland. Monitoring locations with old subsurface drainage, new subsurface drainage, and open ditch drainage had average WTD of 0.51 m, 0.82 m, and 0.95 m, respectively, during the two monitoring years (11/2022 – 10/2024). For each field, we estimated the size of the upper catchment, median summer total runoff, and mean 7-day summer low flow rate. The water required to reach a 0.3 m target WTD was estimated from peat specific yield. Each 0.1 m decrease in mean WTD was estimated to require of 13.2 mm of additional water. Median summer total runoff from the upper catchment was sufficient to reach any target WTD, but the summer low-flow rate did not fulfil the daily water demand. Most runoff is available during early summer, thus creating a timing challenge with water availability even in region with excess annual precipitation. This highlights the importance of catchment-scale management for GHG mitigation. In this study, we propose a generally applicable framework to link peatland GHG mitigation with water resources and catchment-scale management.

1. Introduction

Peatlands cover only 3 % of the global land surface but contain major terrestrial carbon storage (Joosten et al., 2016; Xu et al., 2018) and provide many ecosystem services, as well as food and resource security (Andersen et al., 2017). The utilization of peatlands to provide sustenance for the local population has a long history (Holden et al., 2004), including peat extraction for energy, peatland forestry, and agriculture. Today, 12 % of global peatlands are affected by drainage and are estimated to cause 4 % of the annual anthropogenic greenhouse gas (GHG) emissions (UNEP, 2022). Emissions of CO₂ and CH₄ are related to the average water table depth (WTD). Drainage exposes soil organic matter to the atmosphere and initiates decomposition processes that produce CO₂. Production of CH₄ is related to anoxic conditions and is very low under drained conditions. An exponential model for CH₄ emissions is found in multiple syntheses and commonly shows a rapid increase when the WTD is less than 0.25 m. The combined climate warming potential of

CO₂ and CH₄ is minimized with a WTD ranging from 0.01 – 0.1 m, depending on the global warming potential of CH₄ (Freeman et al., 2022), making complete rewetting of peatlands only known practice to stop the loss of soil carbon storage.

In areas where the local economy is highly dependent on biomass produced on drained peatlands, socially acceptable alternatives for rewetting are required to mitigate negative impacts and make the future transition to more sustainable land use easier. New management practices for cultivated peatlands, such as grass crops under wet conditions and wetland crops (paludiculture), are considered management options that allow agricultural production with a shallow WTD and potentially reduce annual GHG emissions (Freeman et al., 2022). Previous studies found contrasting evidence regarding the impact of water management on GHG emissions (Nijman et al., 2024; Offermanns et al., 2023; Tie-meyer et al., 2024; Weideveld et al., 2021). In these cases, water management was unsuccessful in decreasing WTD (Weideveld et al., 2021) or the relationship between annual WTD and total GHG emissions was

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unclear (Nijman et al., 2024). Tiemeyer et al. (2024) were successful in decreasing WTD; however, at the same time, GHG emissions increased. Another successful case, in which water management was able to decrease WTD, caused reduced carbon losses, but high N_2O emissions related to the renewal of the drainage system to allow water table management with irrigation (Offermanns et al., 2023). Successful GHG mitigation from cultivated peatlands appears to require consistent and long-term success in maintaining shallow water table depth.

Rewetting previously drained peatlands requires additional water, and a lack of available water resources can constrain restoration in some areas (Peng et al., 2024). Depending on the local conditions, gaining additional water to raise the water table in agricultural fields might be a challenge, especially at sites where the regional water table cannot be controlled or near water bodies such as lakes, ponds, and large rivers, or where the possibility of pumping deeper groundwater is lacking. Catchment-scale water management, particularly that using runoff from the upper catchment, could be a solution. However, demand depends on local drainage conditions and land-use practices together with the local hydrological regime. Thus, new frameworks and approaches are required to combine field-specific needs with catchment-scale water-management possibilities.

In this study, we quantified the potential of utilizing upstream surface runoff to increase the water table levels in cultivated peatlands in the western coastal lowlands of Finland. This region is characterized by a well-developed artificial drainage network and fractionated land ownership, where private landowners make land use and water management decisions. Catchments in this region have low slopes and small active catchment storage, with agriculture and forestry as the primary land use types. These limitations guide the practical implementation of water management for field-specific water table and irrigation. To explore the possibility of using upstream runoff to control water table

levels, the aims of this study were to i) Assess the long-term trends in summer (July – August) runoff in the west coast lowlands of Finland, ii) Study the impact of current drainage practices on WTD and soil moisture in cultivated peatlands and finally iii) Develop a micro-catchment-based framework to assess the hydrological feasibility of water table management in drained peatlands.

2. Materials and methods

2.1. Study region and target fields

The study fields and catchments were located on the western coast of Finland, in the mid-boreal vegetation zone, where 13 cultivated peatland fields were monitored (Fig. 1). The monitored fields were selected based on the following criteria: i) field sites were mainly organic soil; ii) volunteer farmers allowed us to install groundwater observation wells and water level sensors; and iii) sites representing different types of drainage conditions, including subsurface, open ditch, and controlled drainage. Farmers with the possibility of adjusting the drainage depth in their fields were instructed to continue operating the controlled drainage system as previously used.

The most common forms of land use in the region are forestry and agriculture. Up to one-third of all fields are in drained organic soils (Kekkonen et al., 2019). The main crops in organic fields are cereals and grass used for animal fodder. Regional drainage channels are designed for spring floods and provide the necessary depth for outlets from the field drainage. Agricultural fields are drained using subsurface or open ditch drainage. The subsurface-drained fields had ditch spacings between 10 and 15 m. If the drainage system has a control structure at the outlet, the drainage depth measured at the groundwater observation well can be adjusted between 0.34 m and 1.58 m, depending on the

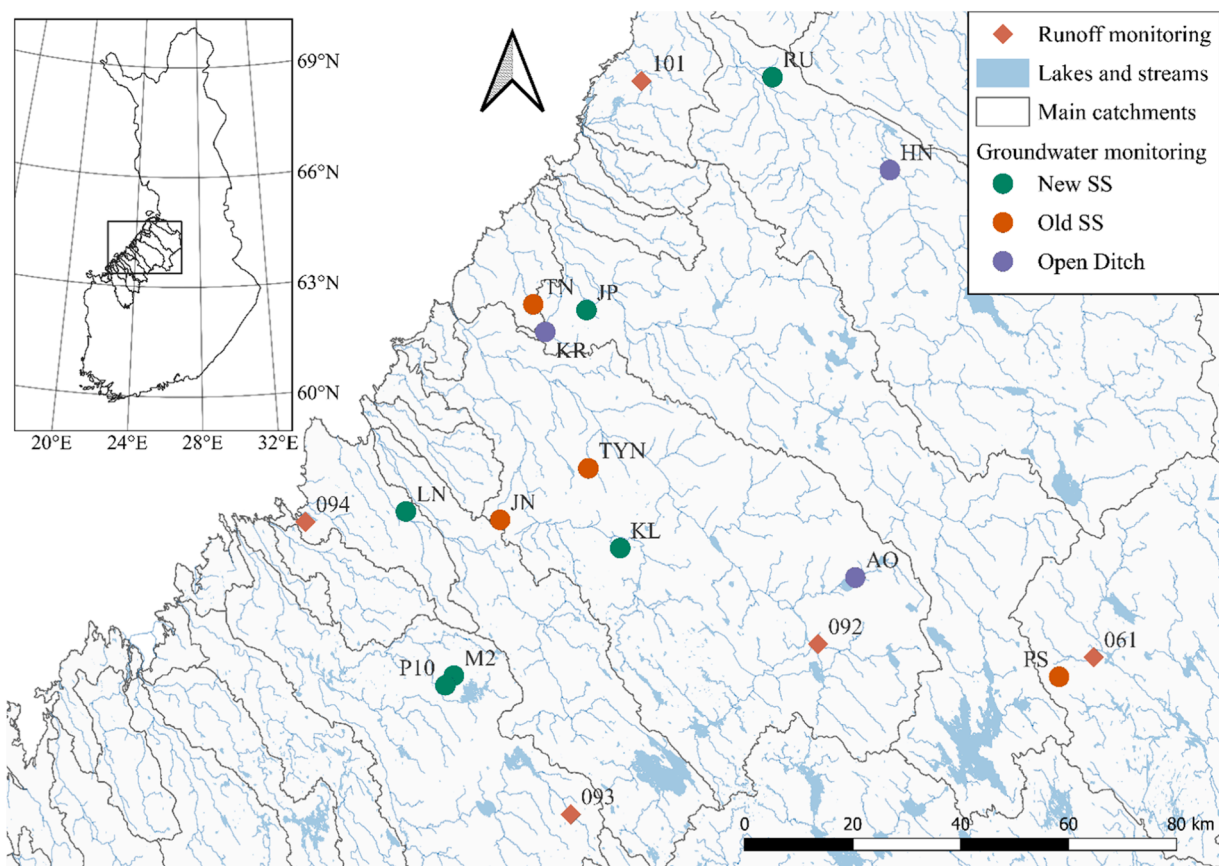


Fig. 1. Map of the study region showing the locations of gauged catchments with long-term runoff monitoring and target fields with groundwater level measurements. Fields are coloured according to the drainage type: recently renewed subsurface drains (SS), old subsurface drains, and open ditch drains.

slope of the field and the distance between drain outlet and groundwater observation well. In the open ditch drained fields drainage depth, measured as vertical distance between ditch bottom and soil surface at the midpoint between ditches varied from 0.98 m to 1.94 m and ditch spacing between 25 – 35 m (Table 1).

Topography is low, and geologically, soils are young former sea bottom sediments that have been raised after the ice age (Tikkanen and Oksanen, 2002). The peatlands in the region are typically less than a few thousand years old and have an average depth of 1.1 m (Virtanen et al., 2003). The annual average temperature and precipitation (1991–2020) were 2 – 3 °C and 550 – 600 mm, respectively, and the elevation varied between 0 and 150 m above sea level (MASL) (Jokinen et al., 2021). Typically, one third of the annual precipitation during the winter months is snow (Meriö et al., 2019).

For each field, the upper catchment area was determined using the ArcGIS Online tool created by the Finnish Forest Centre (FFC, 2025). The tool uses a flow accumulation raster created from a 2-meter digital elevation model (DEM). For each catchment, the potential land uses for water storage areas were calculated from land uses that are already wet or do not currently have high economic productivity, such as poorly growing forests, peat extraction sites, inland marshes, mineral extraction sites, and water bodies. Poorly growing forest are defined as forest areas where annual tree growth is less than 0.1 m³ ha⁻¹ and acquired from forest stand records provided by FFC. The total area of peat extraction sites, inland marshes, and mineral extraction areas was calculated from the Corine Land Cover (2018) database. The total area of agricultural land was calculated using a field parcel register (Finnish Food Authority, 2020). Land cover and field parcel data were obtained from the Paituli Database (CSC, 2025). The surface areas of the lakes and other surface waters were calculated from the Finnish Environment Institute (SYKE) dataset, which includes a natural stream network and larger drainage channels (Finnish Environment Institute, 2021).

Specific runoff observations [$\text{l s}^{-1} \text{ km}^{-2}$] during years 1981–2023 from the Syke small catchment observation network (Seuna, 1983) were used to estimate the regional hydrological characteristics, and hydrological drought indices. Runoff data were downloaded from the Hertta portal (Syke, 2025). Daily meteorological data, precipitation and reference evaporation ET_0 between 1981 and 2023 were obtained from gridded daily meteorological data created for Finland (Aalto et al., 2016; Pirinen et al., 2022) and accessed from Finnish meteorological institute cloud storage (FMI, 2025).

2.2. Field sampling and hydrological monitoring

For water level monitoring, a total number of 16 relative pressure sensors (Caproc LoraWan, Finland) were installed to selected peatlands in between 6/2022 and 08/2022. The observation wells were 3-meter-long PEH pipes with an inner diameter of 41 mm installed at a 2-meter depth. The wells had a 1 m long perforated section with 0.3 mm wide slits. The perforated section was covered with filter sock to prevent sediment infiltration. Water levels were measured hourly using a piezoresistive pressure transmitter, PR-26D (Keller AG) and sent to the data storage cloud over the LoraWAN network. The water depth from the soil surface was manually calibrated by measuring the WTD in each monitoring well four to ten times (Supplementary Figure S1). Soil volumetric moisture [$\text{cm}^3 \text{ cm}^{-3}$] and temperature were measured using TOMST-TMS4 data loggers (Wild et al., 2019). The volumetric water content for sensor calibration (Supplementary Figure S2) was determined by sampling the soil near all dataloggers with a 140 cm³ (height 6 cm, diameter 5.5 cm) sample cylinder and drying the sample until a constant mass was reached at 65 °C. A second-degree polynomial was fitted to the raw sensor data and laboratory determinations of the volumetric water content. The same equation was applied to all raw soil moisture data to calculate the volumetric water content at the soil surface.

For soil sampling (Supplementary Figure S3), hammer-driven window probes with diameters of 10 and 7 cm were used to sample soil from depths of 0–100 cm and 100–200 cm. The sample surface was cleaned prior to sampling. Each soil core was divided into 10 cm subsections in the field. Subsections were sampled from the window probe using a 47 cm³ or 30 cm³ sample cylinder and transferred to a resealable plastic bag. The remainder of the subsection was placed in a plastic bag (Walter et al., 2016). The samples were stored in cold storage (< 4 °C) until further analyses. Small cylinder samples were used for bulk density (BD) and loss on ignition (LOI) analyses. After overnight drying at 65 °C, the samples were weighed again. The dried soil remaining from the bulk density determination was used for the LOI and total nitrogen determination. LOI was measured in duplicate by burning the samples in a laboratory furnace at 550 °C for 120 min. Total Kjeldahl nitrogen (TKN) was used as a measure of amount of nitrogen in the dried soil samples. Modified Kjeldahl method (CEN, 2001) was used to determine total nitrogen in the dry soil. The peat type for each sample was determined by sampling a 10 cm³ subsample. The plant remains in the sample were identified under a microscope as Bryophyta mosses (excluding Sphagnum-mosses), Carex sedges, Sphagnum-mosses, tree-like plants, or shrubs. Peat humification was determined using the Von Post Scale. (von Post, 1922)

Table 1

Distribution of water table and soil sampling locations within each field. Groundwater observation points, catchments, and field drainage information Monitoring locations sharing the same catchment were grouped by catchment for the following analysis. Drain spacing was estimated from the drainage plans or aerial pictures.

Location (n = 17)	Field area [ha]	Elevation [MASL]	Catchment (n = 13)	Drainage system	Drain spacing [m]	Drainage depth [m]	Distance to closest field drain [m]	Distance to drainage channel [m]
AO	7.3	114.0	AO	Open Ditch	35	0.98	16.5	-
KL4	21	83.4	KL	New	10	1.22	5.25	157
KL6	21	83.3	KL	New	10	1.58	3.4	153
KL7	21	82.7	KL	New	10	0.87	2.65	36
TYN	16.6	72.7	TYN	Old	12	n.d	n.a	32
JP	6.2	44.6	JP	New	10	0.84	3.6	55
TN1	2.44	33.0	TN1	Old	n.d	n.d	n.a	59
KR	4.5	40.0	KR	Open Ditch	25	1.74	13	-
JN	6.3	89.2	JN	Old	n.d	n.d	n.a	45.1
LN1	19.6	30.4	LN	New	15	0.34	6	60.1
LN2	19.6	30.8	LN	New	14	0.55	n.a	38.7
M	1.3	117.5	M	New	15	1	7.3	17.5
P	4.3	113.5	P10	New	12	0.60	5.1	39
PS	12.2	144.6	PS	Old	15	n.d	n.a	37
HN1	15.6	53.5	HN	Open Ditch	30	1.65	15	-
HN2	15.6	53.5	HN	Open Ditch	30	1.94	15	-
RU ¹	25.7	47.0	RU	New	14	1.1	7	220

¹ Detailed description of the field in (Yli-Halla et al., 2022)

2.3. Data processing and analysis

Catchment delineation and land use data were handled for each catchment using ArcMap 10.7.1, and exported to R version 4.4.0 (Core Team, 2023), where rest of the analysis with the soil physical and chemical properties, groundwater level measurements, specific runoff observations, and weather data was conducted.

The standardized precipitation and evaporation index (SPEI) and standardized runoff index (SRI) were used to assess the probability of drought conditions during the summer (Beguería et al., 2014). Log-logistic and gamma distributions were used for the SPEI and SRI, respectively (Ahopelto et al., 2023). The index values were calculated for every August between 1981 and 2023 using a three-month averaging period to produce a single index value and to characterize summer weather (SPEI3) and runoff conditions (SRI3). Index value -1 was used as the threshold for summer drought conditions, meaning that all summers with index values less than -1 were considered drought summers.

Water table depth and soil moisture observations from the fields during hydrological years 11/2022 – 10/2024 were used for this analysis. The average annual temperature was 3.7 °C and 3.4 °C and the precipitation was 613 mm and 356 mm for the first and second hydrological year respectively. The daily mean was calculated from hourly observations and 15-min interval observations of the groundwater and soil moisture. Observations were grouped according to the drainage method as old subsurface drainage, new subsurface drain, and open ditch drained fields. The oldest controlled drainage system was installed in 2010 and was used as the cut-off year between the new and old subsurface drains. For each monitoring point, the annual mean, minimum, maximum, and skewness were calculated, along with the summertime mean WTD between June and August. The shape of the annual hydrographs is summarized by g_1 skewness (Tiemeyer et al., 2016).

The soil organic carbon (SOC) fraction for each 10 cm layer depth was calculated from the loss on ignition (LOI) with conversion factors of 0.578 and 0.518 for vascular peat (with plant remains from sedges or shrubs) and amorphous peat, respectively (Klingensfuß et al., 2014). Peat was determined to be Amorphous if Von Post humification was higher than or equal to 8. The specific yield (S_y) [unitless], bulk density (BD) [g cm^{-3}], LOI, peat humification, total nitrogen, and carbon content were averaged for each sampling location. Nitrogen and carbon stocks were calculated from the average values of all peat samples. The specific water yield of the peat samples was estimated from the bulk density and sample depth (d) [cm], using Eq. 1 and the coefficients from Menberu et al. (2021).

$$\ln(S_y) = -0.697 \times \ln(BD) - 0.104 \times \ln(d) - 2.640 \quad (1)$$

Daily specific runoff in units of $1 \text{ s}^{-1} \text{ km}^{-2}$ was converted to mm day^{-1} . The total summer runoff [mm] and summer 7-day low flow rate [mm day^{-1}] were calculated for each gauged catchment for years with complete daily observations during the summer months (July – August). The median summer runoff was used as a runoff signature for the average summer $q_{sum,avg}$ [mm summer^{-1}], and runoff at the first decile was used as a signature for dry summer $q_{sum,dry}$ [mm summer^{-1}]. The mean summer low flow rate q_{low} [mm day^{-1}] was used to estimate the minimum daily runoff available for irrigation. Summer low-flow rate has been connected to catchment storage in Finnish catchments (Meriö et al., 2019), and in this article we use catchment storage as a potential target for catchment scale interventions.

Each runoff signature q_i was converted to the target field irrigation potential I_{pot} by using the catchment area ratio (CAR) (Eq. 2) and runoff signatures from the closest gauged catchment. CAR is the area of the target catchment A_c [ha] divided by the area of the target field A_f [ha], A_c/A_f . Average distance between the target field and gauged catchments was 34 km ranging from 6 km to 51 km, which creates uncertainties to the water availability calculations for individual fields.

$$I_{pot} = CAR \times q_i \quad (2)$$

The additional water demand I_{req} for the target summertime WTD was calculated for each field by multiplying the distance between the mean summertime WTD to target WTD by the average specific yield of the peat samples in the peat profile (Eq. 3). A target depth of 0.3 m was used in this study for a conservative estimate of the required water resources for rewetting for individual fields.

$$I_{req} = S_y \times (WTD_{mean} - WTD_{target}) \quad (3)$$

Eqs. 2 and 3 were combined to analyse the water balance related to the rewetting potential RW_{pot} of the upper catchment runoff with three different runoff signatures and a target summer water table depth of 0.3 m (Eq. 4). The evaluated cases were i) Rewetting potential during average summer using $q_{sum,avg}$, ii) rewetting potential during dry summer with $q_{sum,dry}$, and iii) rewetting potential during summer low-flow using q_{low} . If $RW_{pot} < 0$, for any of the three cases, the catchment is not fully sufficient to fill the additional water required for rewetting.

$$RW_{pot} = I_{pot} - I_{req} \quad (4)$$

3. Results

3.1. Peat layer properties

The average peat depth in the sampled profiles was 77 cm ranging from 30 cm to 150 cm, peat bulk density had average of 0.29 g cm^{-3} ranging between 0.16 g cm^{-3} and 0.55 g cm^{-3} , and the average humification of sampled peat horizons was between 5 and 9 (Table 2). Most humified peats were in locations with the shallowest peat horizon and highest bulk densities. The average estimated soil carbon (C) stock in the fields was 825 t C ha^{-1} , ranging from 1309 t C ha^{-1} to 337 t C ha^{-1} . Nitrogen (N) stock in the dry peat was on average 33.5 t N ha^{-1} with a range from 15 t N ha^{-1} to 57 t N ha^{-1} .

The C:N ratio ($p = 0.003$) and SOC stock ($p < 0.001$) showed positive linear associations with organic horizon depth. Average humification ($p = 0.001$) had a negative linear association with the depth of the organic horizon. The total Kjeldahl nitrogen stock (TKN) did not show a statistically significant linear association with the depth of the organic horizon (Supplementary Figure S4).

3.2. Water table observations from cultivated peatlands

For this study, 16 new water table monitoring wells were established

Table 2

Average peat layer properties at the water table monitoring locations for each field, bulk density, specific yield, and degree of humification of peat layers based on one sampling location. Soil organic carbon (SOC) stock, Total Kjeldahl nitrogen stock (TKN), and Carbon to Nitrogen Ratio.

Field	Peat depth [cm]	Bulk density [g cm^{-3}]	Specific yield [-]	Von post	SOC stock [t C ha^{-1}]	TKN stock [t N ha^{-1}]	C: N
HN ^a	110	0.27	0.13	6	1 309	55	24
JN	150	0.16	0.19	6	1 163	31	38
AO	90	0.39	0.10	8	1 148	57	20
KR	90	0.25	0.13	7	1 062	31	34
M	120	0.17	0.17	6	1 012	30	33
LN ^a	100	0.17	0.18	5	925	37	25
JP	60	0.27	0.13	8	751	39	19
KL ^b	77	0.24	0.15	7	750	32	24
PS	70	0.20	0.18	8	599	29	21
TYN	30	0.55	0.08	9	432	23	19
TN	30	0.49	0.09	9	412	23	18
P	30	0.24	0.15	9	337	15	22
RU	40	0.37	0.11	n.d	n.d	n.d	n.d

^a Average of two sampling locations.

^b Average of three sampling locations.

to drained cultivated peatlands and monitored for two years. The monitoring wells were placed at a range of distances to the nearest open drainage channel, closest subsurface drainage, depth of organic layer, and elevation that could affect the water table measurements. Overall, none of these parameters exhibited a clear impact on the 2-year average water table depth (Supplementary Figure S5). The acquired water table data are taken to represent drainage conditions in each field and in Table 3 we present summary statistics from the field observations made in the study.

Average annual WTD was 0.78 m varying between 0.39 m to 1.27 m. Grouping by drainage type produced differences in annual average WTD, summer average WTD, maximum WTD, and skewness of the annual groundwater hydrograph (Table 3). Monitoring wells within fields with new subsurface drains and open ditch drains tended to show highest WTDs. Fields with old subsurface drains had annual average WTD 0.31 m shallower than fields with new subsurface drainage. One open ditch drained field was exceptionally dry (Fig. 2.b) which contributed to the differences seen in open ditch drained fields and new subsurface drained fields. Water table skewness, that is, the shape of the annual hydrograph, is related to drainage conditions at the field. Fields with new subsurface drains and open ditches had more negatively skewed WTDs, which means that water levels in fields with better functioning drainage tended to stay closer to the annual maximum, and monitoring locations with old subsurface drains fluctuated more symmetrically around the average WTD.

The observed soil moisture [$\text{cm}^3 \text{cm}^{-3}$] ranged from 0.30 to 0.76 and similar range was found in all drainage types. Average soil volumetric moisture was 0.51 and varied from 0.40 to 0.67. Differences in the soil moisture were smaller between different drainage types, but measurement points in the open ditch drained fields tended to be lower than measurements in subsurface drained fields. Relatively large range in water table depths aren't reflected in soil moisture measurements as summer maximum WTD in open ditch drained fields had a range of 0.93 meters, but the minimum soil moisture had range between 0.30–0.37.

The WTD in the fields with old subsurface drains was greater than 0.3 m 80 % of the time (Fig. 2. a). The WTD in fields with open ditch

Table 3

Summary of water table and soil moisture measurements from 17 groundwater level observation wells during the first two hydrological years between 11/2022 – 10/2024. Each monitoring well contributed two samples to the table and ranges shown include variation over two monitoring years. Each cell shows the mean value, and the range is shown in parentheses.

Characteristic	Overall, N = 34	New, N = 18	Old, N = 8	Open Ditch, N = 8
Water table depth [m]				
Summer mean [m]	0.89 (0.43, 1.44)	0.92 (0.61, 1.14)	0.64 (0.43, 0.79)	1.09 (0.46, 1.44)
Annual mean [m]	0.78 (0.39, 1.27)	0.82 (0.55, 1.02)	0.51 (0.39, 0.64)	0.95 (0.42, 1.27)
Minimum [m]	0.12 (−0.22, 0.61)	0.19 (−0.04, 0.61)	0.02 (−0.18, 0.22)	0.06 (−0.22, 0.17)
Maximum [m]	1.10 (0.65, 1.67)	1.13 (0.93, 1.49)	0.87 (0.65, 1.07)	1.27 (0.74, 1.67)
Water table skewness	−1.03 (−2.41, 0.28)	−1.07 (−2.31, −0.03)	−0.46 (−0.91, 0.28)	−1.50 (−2.41, −0.34)
Soil moisture [$\text{cm}^3 \text{cm}^{-3}$]				
Mean soil moisture	0.51 (0.40, 0.67)	0.53 (0.43, 0.65)	0.51 (0.41, 0.67)	0.46 (0.40, 0.55)
Minimum soil moisture	0.37 (0.30, 0.50)	0.40 (0.33, 0.50)	0.35 (0.31, 0.44)	0.33 (0.30, 0.37)
Maximum soil moisture	0.67 (0.44, 0.76)	0.67 (0.44, 0.76)	0.71 (0.60, 0.75)	0.62 (0.49, 0.72)

drainage was more than 0.3 m for 90 % of the time. The water table depth in the new subsurface drained fields was greater than 0.3 m more than 95 % of the time. Two of the observation points with new subsurface drainage had a water table below 0.3 m 100 % of the time (Fig. 2. a).

The hydrographs for the groundwater observations followed a pattern similar to that of the regional hydrology (Fig. 2.b). Summer and winter were dry seasons, with similar maximum water table depths due to precipitation accumulating as snow during winter and low precipitation and high evapotranspiration during summer. Differences in water table depths for drainage systems increased during the dry seasons. The winter dry season ends with spring, releasing large amounts of water from the melting snow. Differences in the WTD between drainage systems decreased during wet seasons as the soils became saturated with water. January 2023 had a warm period that melted some of the snow and caused a decrease in WTD. The winter WTD maxima occurred around April before snowmelt for all drainage types, and the summer WTD maxima occurred at the end of August. (Fig. 2. b).

3.3. Catchment properties and availability of runoff for irrigation

The total area of the studied catchments varied from 10 ha to 970 ha (Supplementary Table S1), with an average catchment area of 270 ha. The area of the fields had a range of 1.3 ha to 25.7 ha, with an average field size of 11 ha. The catchment area ratios ranged from 2.4 to 155.3. The mean and median catchment area ratios are 31 and 16.2, respectively. The most common land-use forms in the study catchments were forestry and agriculture, with an average of 57 % and 29 %, respectively. One catchment (JP) also has ongoing peat-extraction activities. Two catchments had inland marshes (JP and KR) and one catchment had mineral extraction activities (P). The most common potential area for establishing water storage was a poorly growing forest, which was found in 9 of the 13 catchments.

The results of catchment based rewetting potential calculations for individual target fields are presented in Table 4. Additional water requirements for the 0.3 m target summertime (June – August) WTD varied from 21 mm to 150 mm or from 0.2 mm d^{-1} to 1.6 mm d^{-1} . Most of the water is required for open-drained fields and fields with renewed subsurface drainages. The median summer runoff was sufficient to fulfil the estimated additional water demand for all fields. The dry summer runoff was insufficient for the three fields (M, KR, and AO). The mean summer low flow rate was not sufficient for daily water demand in 7 of the 13 fields.

3.4. Drought indices

During the runoff monitoring period from 1981 to 2023, the five gauged catchments had 215 observational years and 31 meteorological drought summers (Table 5). Based on this analysis, 14.4 % of the summers experienced meteorological drought. Complete runoff observations were made over 201 years, and there was a total of 20 hydrological drought summers; 10 % of the summers had hydrological drought. No significant trends were observed in the summertime SRI or SPEI droughts (Supplementary Table S2). Summer precipitation and potential evapotranspiration are related to hydrological droughts (Fig. 3.b), but the SPEI does not solely determine hydrological drought conditions, as not all summers with SPEI droughts have SRI droughts (Fig. 3.a).

In the five gauged catchments (Table 5) median summertime runoff was on average 33.6 mm with a range from 26 mm to 45 mm. The average of the first deciles of the summer runoff observations was 11 mm, ranging from 7 to 14 mm. The summer mean 7-day low flow rate was, on average, 0.048 mm day^{-1} and varied from 0.037 mm day^{-1} to 0.064 mm day^{-1} .

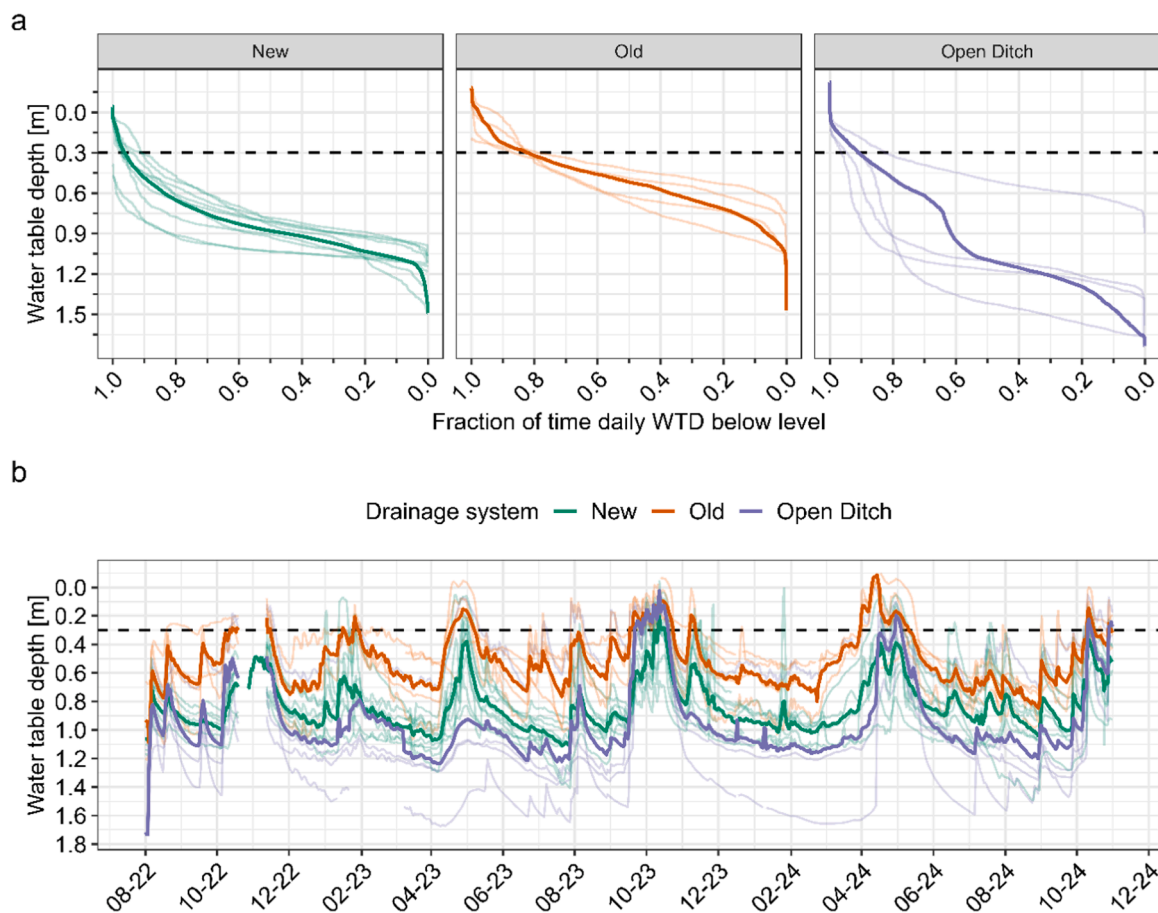


Fig. 2. a) Groundwater level duration curves for all observation points used in the study and mean values for different drainage systems; b) time series of groundwater level observations between 08/22 – 11/24. The reference line was drawn at 0.3 m, which is the boundary between the deep- and shallow-drained organic soils in the IPCC wetland supplement (IPCC, 2014).

3.5. Regional application of rewetting water balance

For the application of Eq. 3 to the study region, a model to calculate the additional water required as a function of the decrease in average WTD during summer (June–August) was required. A linear model was sufficient to model the additional water required fields included in the study (Fig. 4. a). According to this function, each 0.1 m reduction in summertime average WTD required 13.2 mm or 132 m³/ha of additional water, when compared to the baseline. For example, reducing the mean summertime WTD from 1 m to 0.3 m required in the study region about 97 mm or 970 m³/ha of additional water during summer months. This calculation considers the difference in the total water balance. Additional water can come from reduced drainage depth or from increasing groundwater recharge, for example, through subsurface irrigation. Eq. 3 was used to calculate required additional water, for any required WTD adjustment, which is shown on the x-axis in Fig. 5.

For the application of Eq. 2 to the study region, we used average values for the median summer total runoff, dry summer total runoff, and mean summer 7-day low flow rate from the five gauged catchments (Table 5). Eq. 2, with the regional runoff signatures, was used to calculate the potentially available water for each Catchment Area Ratio which are shown on the y-axis in Fig. 5. The rewetting water balance in Eq. 4 was calculated for each point in the plane defined by the axes. Fig. 5 shows the combined results of the water balance equation for the three runoff signatures. Supplementary materials (Figure S7) show similar plots for individual runoff signatures. In region 1 (R1 in the Fig. 5), the rewetting water balance was always positive, meaning that there is enough runoff available for the required rewetting during the

typical summer, dry summer, and during summer low flow. In region 2 (R2 in the Fig. 5), the water balance with dry summer runoff was negative, and in region 3 (R3 in Fig. 5), the water balance was negative during summer low-flow. The difference in the daily required additional water and the available summer low-flow rate was used in the equation shown in Fig. 4. b to calculate the required increase in catchment storage for the low flow rate to match the additional water demand of field. Contours in the low-flow limited region R3 show the required increase in catchment storage as millimetres (mm) of water depth over the catchment area.

With the average summer WTD and catchment area ratios each target peatland in the study can be placed to the Fig. 5, allowing comparison of the individual target field based calculation presented in Table 4 and the regional application of the rewetting potential equations. Nine of the 13 study fields were placed in the same water availability regions in the Fig. 5 and the Table 4. Field AO was placed in R2 in the figure, but in Table 4, the total summertime runoff was sufficient to meet the water requirements. Fields TN and RU are both located in region R3, whereas in Table 4, they do not have any water availability limitations. Field PS should be placed at the boundary between R3 and R1 according to Table 4, but it is located well inside R1 in Fig. 5. For a cultivated peatland in the study region without information about the actual summer WTD, we estimate that fields with Catchment area ratio of at least 20 would be sufficient for rewetting. This is shown in the Fig. 5 with arrow starting from 65 cm decrease in summer WTD and reaching the border between regions R3 and R1.

Table 4
Water required and estimated availability from catchment runoff for a targeted mean summer water table depth of 0.3 m. The catchment area ratio is the relative size of the upper catchment to the target field. The numbers in parentheses indicate the fraction of the field of the entire catchment. Fields are grouped by the type of drainage system and, in descending order, by the catchment area ratio within a drainage system.

Drainage	Catchment			Water required		Runoff available during summer		
	Target field	Catchment area ratio (%-of catchment area)	Potential Storage Areas [ha]	Summer [mm summer ⁻¹]	Daily [mm day ⁻¹]	Median [mm summer ⁻¹]	Dry [mm summer ⁻¹]	Mean 7-day low-flow [mm day ⁻¹]
Old subsurface	JN	55.1 (1.8 %)	107.1	40.0	0.4	1 422.5	588.4	2.6
	TYN	35.4 (2.8 %)	22.5	31.4	0.3	913.5	377.9	1.7
	TN	13.3 (7.5 %)	0.0	44.0	0.5	447.9	179.5	0.8
New subsurface	PS	16.2 (6.2 %)	17.4	50.8	0.6	731.0	178.9	0.6
	JP	155.3 (0.6 %)	30.8	93.0	1.0	5 244.6	2 101.9	9.9
	P	62.3 (1.6 %)	16.8	112.1	1.2	2 013.1	456.0	2.6
	RU	20.0 (5.0 %)	0.6	94.7	1.0	675.6	270.8	1.3
Open ditch	LN	17.0 (5.9 %)	39.2	96.0	1.0	438.5	181.4	0.8
	M	8.2 (12.2 %)	0.0	124.5	1.4	265.6	60.2	0.3
	KL	6.5 (15.4 %)	1.8	72.8	0.8	181.7	87.2	0.3
	KR	7.0 (14.3 %)	21.9	150.0	1.6	235.4	94.3	0.4
	HN	4.4 (22.7 %)	0.0	115.5	1.3	147.2	59.0	0.3
	AO	2.4 (41.7 %)	0.0	21.3	0.2	66.9	32.1	0.1

4. Discussion

The requirements for water management practices to mitigate GHG emissions from cultivated peatlands depend on the climate, hydrological regimes, and land-use practices. Previous work has demonstrated water management in tropical peatlands (Fawzi et al., 2024) and the Netherlands (Nijman et al., 2024), where regional water authorities can control water levels in large regions. Land use and water management practices in the study region are shaped by strong rights for private landowners, a mosaic of land use, and snowmelt-driven hydrology, thus requiring different water management solutions to mitigate emissions. The micro-catchment-based approach suggested in this study and field-scale hydrological information can provide crucial information for targeting GHG mitigation efforts and practical applications under similar northern conditions.

4.1. Drainage systems influence the average WTD and shape of the hydrograph

Our monitoring showed that annual WTD in cultivated peatlands in the studied region can vary between 1.27 m to 0.39 m. All locations were sufficiently drained to indicate soil-based CO₂ emissions (Tiemeyer et al., 2016), but grouping by the existing drainage system shows differences in the average WTD and soil moisture, which are considered the main drivers of soil-based emissions from drained peatlands (Boonman et al., 2022; Evans et al., 2021). Observations from fields with old subsurface drainage systems and one field with open drainage ditches consistently showed a shallower WTD. These WTDs are in a region with reduced emissions according to the nonlinear model used for national GHG inventories in Germany (Tiemeyer et al., 2020). However, the Finnish GHG inventory uses a Tier 2 approach with nationally derived emission factors (EFs) for deep-drained croplands and grasslands in organic soils, which cannot recognize the variability in the WTDs. When using these EFs, it is assumed that all cultivated peatlands are well-drained, but it is likely that the total area of well-drained organic soils is smaller than the total area of cultivated peatlands (Tuohy et al., 2023). Developing an estimate of the current drainage status of organic soils can provide a baseline for targeting emission mitigation activities and potential reductions in reported emissions when linked to annual GHG inventories (Koch et al., 2023). The classification used in this study can be used as a target for more extensive hydrological and GHG emission monitoring of drained agricultural peatlands to better capture the already existing variability and potentially derive a more representative EF classification for Finnish cultivated peatlands.

Monitoring locations within fields with recent subsurface drains (2010 and later) and open ditch drains had the highest WTD, and water management measures are necessary to mitigate GHG emissions. Fields with new drainage systems have also controlled drainage systems with the potential to adjust the drainage outlet level and reduce drainage runoff from the fields (Carstensen et al., 2020; de Wit et al., 2022). Controlled drainage has been considered a tool to maintain shallower WTDs while allowing traditional agricultural production, but our monitoring results do not support the current application of this practice as a climate change mitigation tool. Lack of incentives and guidance for farmers to use of controlled drains for their full potential can explain this finding, but recent studies examining the controlled drainage in cultivated peatlands have shown limited reduction to average WTD from 0.21 m to 0.01 m (Salla et al., 2024; Tähtikarhu et al., 2025). It seems that the runoff reduction from controlled drainage is not fully converted to a reduction in WTD. If improved drainage systems are necessary, guidance and incentives for farmers to implement new drainage management practices are also necessary. The guidance could include providing some required drainage depth during the off-season or providing information about the proper timing for increasing drainage depth.

New subsurface drainage systems allow water input to the drainage

Table 5

Count of drought summers, total runoff amounts for median and dry summers and mean 7-day low flow rate for each of the runoff donor catchments in the region between years 1981–2023. Maximum number of monitoring years for each catchment was 43. Last column shows the receiving target field for each gauged catchment.

Gauged catchment number	Drought summers SPEI	Drought summers SRI (count of years)	Median summer total runoff [mm]	Dry summer total runoff [mm]	Mean 7-day low flow rate [mm day ⁻¹]	Receiving target field
061	7	6 (42)	45	11	0.037	PS
092	6	2 (40)	28	13	0.049	AO, KL
093	5	5 (43)	32	7	0.041	P, M
094	7	4 (43)	26	11	0.047	LN, JN, TYN
101	6	3 (33)	37	14	0.064	TN, HN, JP, KR, RU

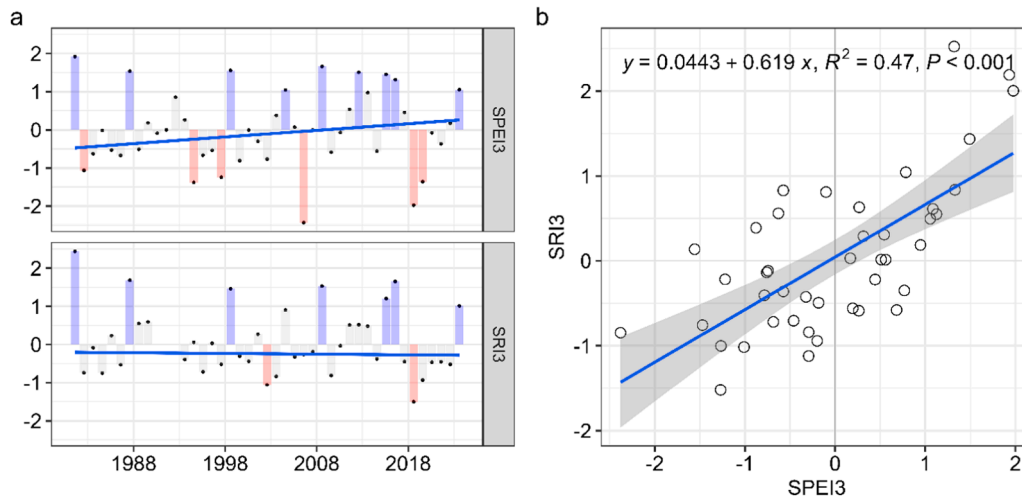


Fig. 3. Drought index example from gauged catchment 94 (see Fig. 1) a) Time series of August hydrological drought index SRI3 and meteorological drought index SPEI3 values. b) Scatterplot of August SRI3 and SPEI3 with R^2 and linear fit. Similar figures for all gauged catchments in the study can be found in the [Supplementary materials](#) (Fig. S6).

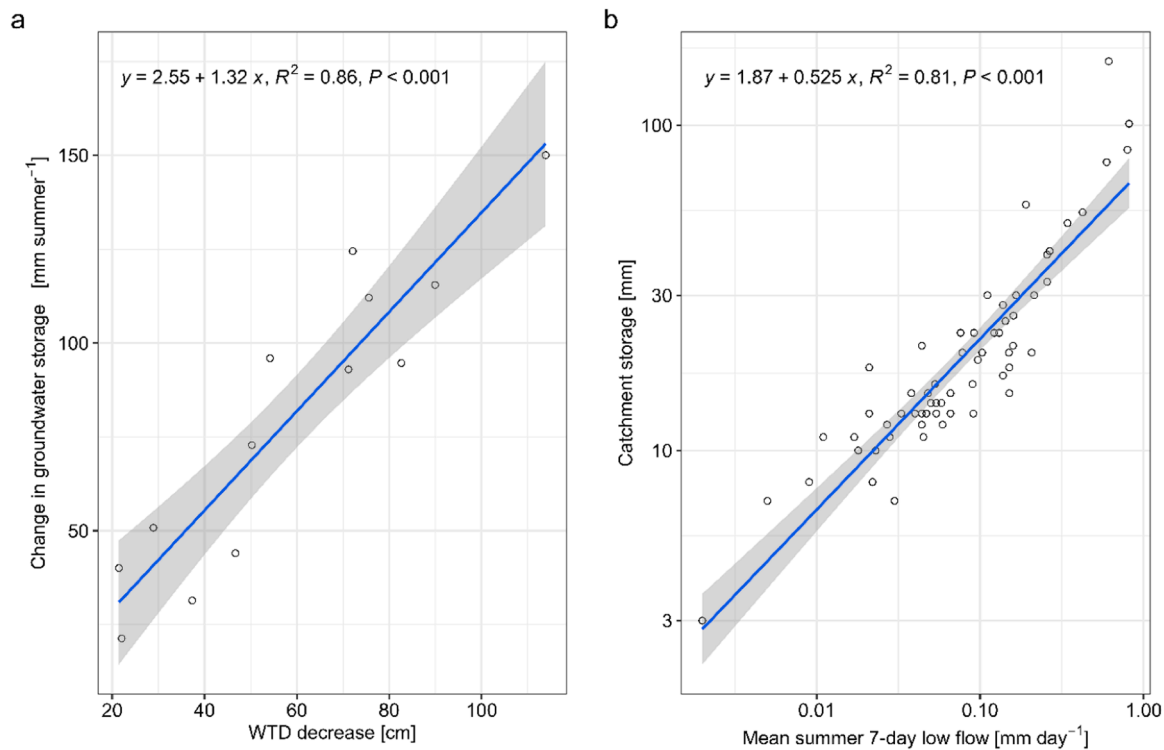


Fig. 4. Relationships for field water storage changes and catchment storage. a) Required increase in field groundwater storage to decrease mean summer WTD. Each point in a graph represents one target field of this study and required change in storage is calculated with Eq. 3 with target WTD of 0.3 m b) Relationship between catchment storage and mean summer 7-day low flow rates in Finnish catchments. Both axes are logarithmic (Data from Meriö et al. (2019)).

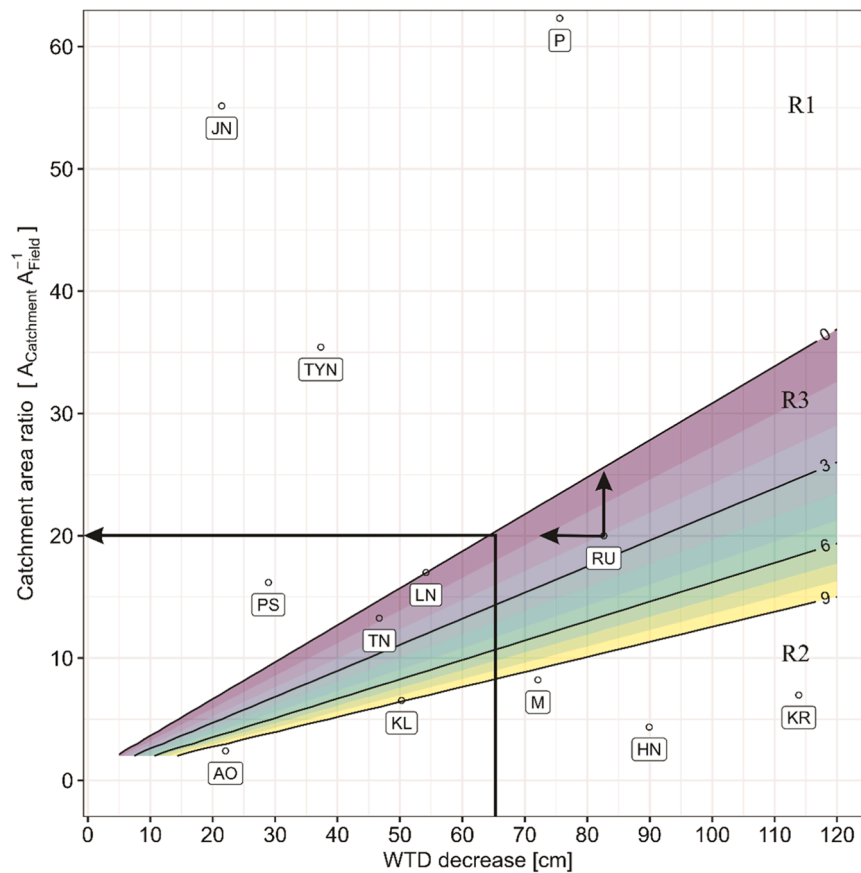


Fig. 5. The WTD decrease and the catchment area ratio plane for the study region with locations of each target field are shown. Field JP is outside the plotting area. Figure area is separated to three distinct regions. Region 1 (R1) has fields without water limitations. Region 3 (R3) with contours shows field-catchment combinations where summer low-flow limits WTD adjustment. Contours show the required increase of catchment storage in mm of water depth to match required summer low-flow rate. Region 2 (R2) has combinations with lack of total summer runoff during dry summer. Arrows from RU shows how the location of the point would move according to different interventions applied to the field or to the catchment. Arrow starting from 65 cm decrease in WTD shows estimate of minimum catchment area ratio for successful rewetting for fields belonging to study region.

systems to irrigate fields, a water management practice called subsurface irrigation. Additional water is beneficial for drought management (Peltonen-Sainio et al., 2021), reducing the impact of acid-sulfate soils (Österholm et al., 2015), and possibly mitigating GHG emissions from cultivated peatlands. However, using an existing controlled drainage field for subsurface irrigation might pose a challenge related to the optimal design. Ditch spacing in the subsurface drained fields in this study varied from 10 m to 15 m. Recent studies demonstrating subsurface irrigation have reported drain spacings ranging from 4 m to 6 m (Offermanns et al., 2023; Tiemeyer et al., 2024; Weideveld et al., 2021). If effective subsurface irrigation requires a smaller ditch spacing than that recommended by drainage guidelines, all existing drainage systems in Finland are not optimised for subsurface irrigation. The effectiveness of subsurface irrigation system in maintaining a shallow WTD with a field drained using current drainage standards needs to be evaluated.

4.2. Irrigation as a climate change mitigation tool for high productivity boreal peatlands

The target fields and their upper catchments exhibited large variations in irrigation potential as the catchment area ratio varied between 2.4 and 155. The potential runoff available for irrigation varied between 67 mm and 5245 mm, highlighting the variability in the potential to apply runoff from the upper catchment as a GHG mitigation practice. The catchment area ratio, which is the relative size of the catchment supplying water to the target field, can be used as a simple parameter to assess the potential of runoff from the upper catchment to maintain

shallow WTD during summer in cultivated peatlands or as a guide for rewetting degraded peatlands.

The main results summarised in Fig. 5 can assist this plot-by-plot planning by providing simple criteria to determine whether additional water from the catchment is a suitable water management option for the field. The boundary between R3 and R1 shows the minimum catchment size for a given target WTD increase where the runoff could be directly routed to the field area without additional water storage. For a field in R3, this gives three options to achieve the target summertime WTD: i) reducing drainage depth will shift a point horizontally in the plot area, ii) redirecting water from a larger catchment will shift a point vertically, making more summer low-flow available, and iii) increasing catchment storage, which can have positive influence on summer low-flows (Meriö et al., 2019). Increased catchment storage would move the point towards the boundary between R1 and R3 with some curved trajectory, but it is not obvious what the trajectory would look like.

Peat depth is a simple parameter that can be used to target GHG emission mitigation efforts (Kekkonen et al., 2019). Catchment area ratio presented here is a simple to estimate parameter that can provide additional detail to the potential to restore peatlands. This study included five sampling locations with sampled carbon stocks above 1000 t C ha⁻¹ and peat depths above 90 cm (HN, JN, AO, KR, and M), which would be ideal locations to retain as much soil carbon by rewetting. Based on the catchment area ratios and regional runoff signatures, only one field (JN) had access to sufficient water resources for complete rewetting. Field HN did not have access to sufficient runoff, even during typical summer, and fields AO, KR, and M were limited by

dry summer runoff. For these fields, the rewetting potential is dependent on the effectiveness of the reduced drainage depth in restoring the shallow WTD. If the reduced drainage depth is not sufficient, a further decrease in WTD by irrigation would require the construction of water reservoirs or water detention structures near fields or close by in the upper catchment, water transfer from a larger catchment, or recirculation of drainage water to be used to irrigate the fields, making rewetting expensive.

Additional water required to maintain an average summer WTD of 0.3 m was estimated to be between 21 mm and 150 mm based on the average WTD over two summers. This estimate was based on the specific yield of peat estimated from bulk density and was generalized to a linear model where one meter decrease in WTD would require 132 mm of additional water. This approach ignores the dynamic relationship between water balance components and should not be interpreted as the required irrigation input. Instead, the calculated estimate reflects the change required for the groundwater water balance. Groundwater balance includes inputs from recharge from precipitation and groundwater inflow from the surrounding catchment. Losses from groundwater storage include capillary rise, drainage, and groundwater seepage into surrounding ditches or deeper groundwater are thus not considered in our calculation.

We did not find comparable studies from northern European conditions, but a more detailed model developed for fens in Northeast Germany decreasing WTD from 1 m to 0.3 m required 175 mm a⁻¹ or 210 mm a⁻¹, depending on the peat type (Renger et al., 2002). The difference between their model estimates is that they include changes in capillary rise and seepage related to changes in the WTD and assume that water can be added to the field without loss. Few experiments have reported the actual water used to maintain a shallow WTD with subsurface irrigation. Experiments on peatlands in Northwest Germany used an average of 300 mm during the growing season to maintain high WT levels in the fields (Kalinsky et al., 2021). Two experiments on sandy soils in the Netherlands required 400 mm a⁻¹ or 900 mm a⁻¹ to increase groundwater levels by one meter (de Wit et al., 2022). The application of subsurface irrigation to maintain WT levels during the growing season likely requires large amounts of water, and the availability of easily sourced water for irrigation can limit the potential for GHG mitigation in peatland-rich regions. For this reason, our approach, in which upper catchment runoff water is used for controlling WT, can provide a reasonable solution for some cultivated peatland cases.

4.3. Drought risks and climate change mitigation potential of drained peatlands

Three target fields were estimated to be limited by the runoff available for water table management during the dry summers. On average, summer (June-August) meteorological water balance ($P - ET_0$) is negative (Supplementary Figure S8). With a precipitation deficit, there is limited potential to decrease WTD by reducing the drainage depth alone, and very dry summers make the use of surface runoff more attractive for all agriculture.

Dry summers are part of the climate in the region, and hydrological observations in this region already show summers with very low total summer runoff (Supplementary Figure S9). Future projections indicate an increased risk of summer droughts and drought-related damage in Southwest Finland (Ahopelto et al., 2023). Only 3 % of the agricultural areas in Finland have irrigation equipment, and they are commonly located on the southwest coast of Finland (Official Statistics of Finland (OSF), 2022). If water table management with irrigation is proven to be an effective method for controlling GHG emissions, the area of irrigated farmland can increase, and water used to rewetting cultivated peatlands can amplify droughts. To address water availability issues related to dry summers, potential areas with low economic productivity (for example, poorly growing forests or natural wetlands) that could be used for construction of additional water storage were mapped from the target

catchments (Table 4). Nine of the 13 catchments had potential locations where the water-holding structures could be engineered. Using the entire catchment in an integrated manner for GHG mitigation from cultivated peatlands can also provide benefits related to water quality from peatland forestry (Härkönen et al., 2023). In addition to drought preparedness, retention structures in the upper catchment can provide flood control (Roberts et al., 2023) and other services offered through holistic catchment-scale planning and management.

For the best results, the integration of the needs of all water consumers at different catchment scales is required for optimal water management options for GHG mitigation. This study was constrained to assess water availability for one field, and we assumed that all runoff from the upper catchment could be used for water table management. For catchment application interaction of multiple rewetted peatlands needs to be addressed as well. Information about the distribution of cultivated peatlands and their WTDs is necessary for catchment wide application. If this information is available, the rewetting water balance presented here can be applied at catchment scale in an iterative fashion where fields without any rewetted fields in upper catchment create the first iteration layer. Water required to rewet the first layer of fields is removed from the available water for fields downstream and the iteration would continue until the available water is allocated to all rewetted fields.

4.4. A framework for assessing hydrological limitations for climate change mitigation

Surface runoff is the easiest water source for mitigating GHG emissions from drained boreal peatlands, especially if no other surface water bodies or major GW aquifers are located nearby. The micro-catchment analysis presented in this study outlines a framework for relating the available water resources to the total or partial rewetting of cultivated peatlands (Fig. 6). The catchment hydrology-related information used in this framework was the average summer total runoff, dry summer total runoff, and average summer low-flow. Runoff information combined with the catchment area ratio and new estimates of water required to adjust the WTD were sufficient to determine whether runoff from the catchment could be a sufficient for GHG mitigation in cultivated or rewetted peatlands.

The surface runoff used to decrease the WTD is removed from downstream users, making GHG mitigation a water resource management problem with wider catchment management implications. Focusing on cultivated peatlands and their micro-catchments can be used to create a bottom-up approach to identifying interventions that can have positive catchment-scale benefits in addition to GHG mitigation from cultivated peatlands. In peat-dominated regions, water quality issues are linked to peatland drainage and carbon storage is major catchment function. Method presented here can also be applied in larger catchment scale to integrate soil carbon storage preservation as a target for interventions in catchment scale planning frameworks such as Catchment Systems Engineering (CSE) presented by Hewett et al. (2020).

5. Conclusion

The potential to mitigate GHG emissions from drained peatlands depends on the current drainage status of the peatlands and the success of integrating new water management systems into agricultural and catchment management practices. The initial findings from WTD monitoring show that the construction of new drainage systems is not sufficient to guarantee agriculture with wet organic soils, but also a previously unaccounted variability in WTDs explained by drainage system characteristics. A micro-catchment-based approach for assessing water-related limitations to the partial or full rewetting of agricultural peat soils provides a new framework for assessing the potential to mitigate GHG emissions. For a single peatland in the study region, the

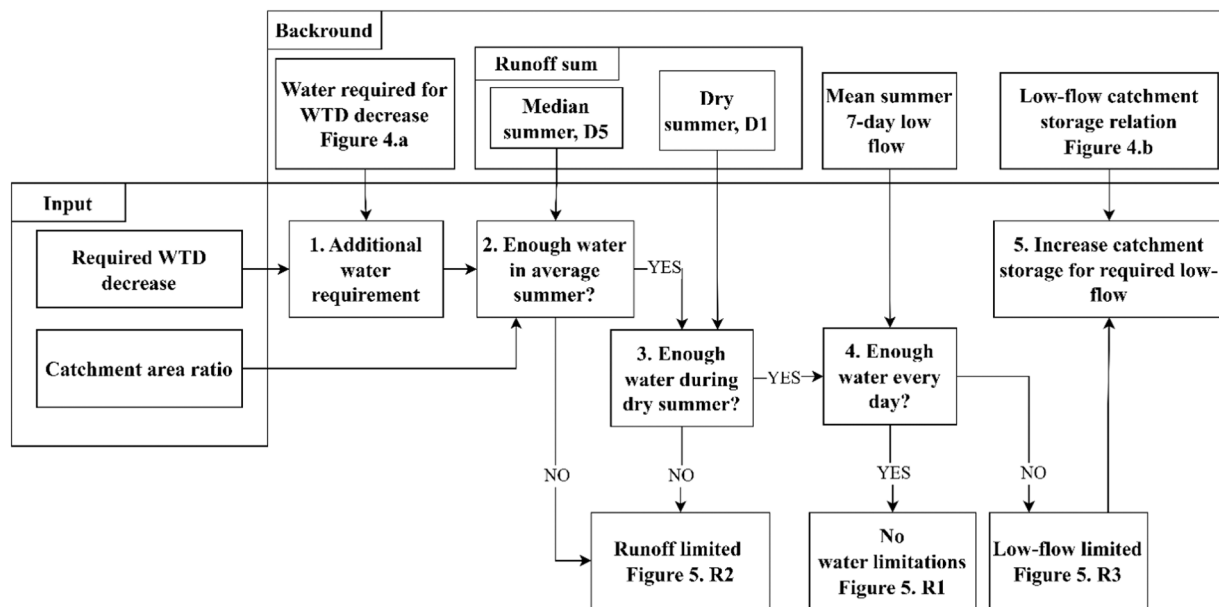


Fig. 6. Framework for assessing the hydrological limitations related to the WTD decrease in cultivated peatlands applied in this study. Median runoff sum (5th decile, D5) was used as a typical summer conditions and total summer runoff at 1st decile (D1) was used to indicate dry summer conditions.

total summer runoff was almost always sufficient to achieve any target WTD. However, the low summer flow is likely to limit rewetting. Studies addressing the availability of water in catchments shared by multiple peatlands and the effectiveness of catchment interventions to increase water availability during dry periods are necessary to better understand the potential to mitigate GHG emissions. This approach highlights that sustainable peatland management is not only an agricultural problem but also a water resource and catchment management problem.

CRedit authorship contribution statement

Liimatainen Maarit: Writing – review & editing, Supervision, Funding acquisition, Conceptualization. **Kløve Bjørn:** Writing – review & editing, Supervision, Conceptualization. **Lämpikivi Miika:** Writing – original draft, Visualization, Investigation, Conceptualization. **Marttila Hannu:** Writing – review & editing, Supervision, Funding acquisition, Conceptualization.

Declaration of Generative AI and AI-assisted technologies in the writing process

During the preparation of this work the authors used Paperpal in order to improve grammar and readability of text. After using this tool, the authors reviewed and edited the content as needed and take full responsibility for the content of the publication.

Declaration of Competing Interest

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

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Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at [doi:10.1016/j.agwat.2025.109427](https://doi.org/10.1016/j.agwat.2025.109427).

Data Availability

Data will be shared in open access IDA-prepository (<https://ida.fairdata.fi/login>) with DOI number after acceptance of the article to journal.

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