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Extending the SUSI peatland simulator to include dissolved organic carbon formation, transport and biodegradation - Proper water management reduces lateral carbon fluxes and improves carbon balance

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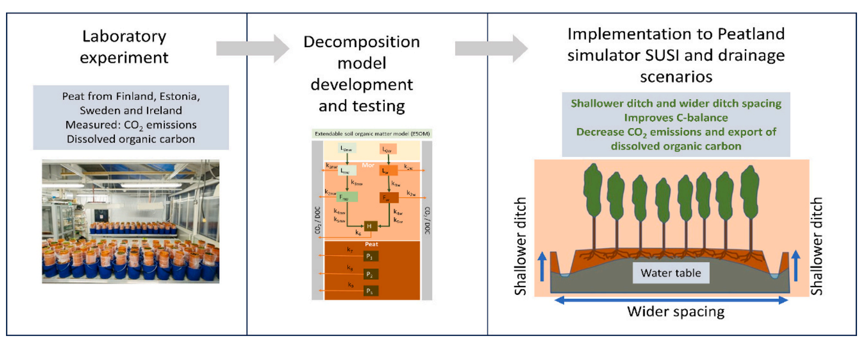
HIGHLIGHTS

- A new simulation model for complete C balance for peatland forests is presented.
- Shallower ditches or wider ditch spacing decreases dissolved organic C export.
- Labile dissolved organic C originates from a few meters distance from ditch.
- Slope changes dissolved organic C biodegradation and export.

GRAPHICAL ABSTRACT



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ABSTRACT

Drainage intensity and forest management in peatlands affect carbon dioxide (CO_2) emissions to the atmosphere and export of dissolved organic carbon (DOC) to water courses. The peatland carbon (C) balance results from a complex network of ecosystem processes from where lateral C fluxes have typically been ignored. Here, we present a new version of the SUSI Peatland simulator, the first advanced process-based ecosystem model that compiles a full C balance in drained forested peatland including DOC formation, transport and biodegradation. SUSI considers site, stand and terrain characteristics as well as the interactions and feedbacks between ecosystem processes and offers novel ways to evaluate and mitigate adverse environmental impacts with thorough management planning. Here, we extended SUSI by designing and parameterizing a mass-balance based decomposition module (ESOM) based on literature findings and tested the ESOM performance against an independent

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dataset measured in the laboratory using peat columns collected from Finland, Estonia, Sweden and Ireland. ESOM predicted the CO_2 emissions and changes in DOC concentrations with a reasonable accuracy for the peat columns. We applied the new SUSI for drained peatland sites and found that reducing the depth to which ditches are cleaned by 0.3 m decreased the annual DOC export by 34 (17 %), 29 (19 %) and 7 (5 %) kg ha^{-1} in Finland, Estonia and Sweden, respectively, using typical ditch spacing for these countries. Correspondingly, site annual C sink increased by 305, 409 and 32 kg ha^{-1} in Finland, Estonia and Sweden, respectively. Our results also indicated that terrain slope can markedly alter the water residence time and consequently DOC biodegradation and export to ditches. We conclude that DOC export can be decreased and site C sink increased by reducing the depth to which ditches are cleaned or by increasing the ditch spacing.

1. Introduction

The important role of lateral carbon (C) fluxes in the landscape C balance has been globally recognized (Juutinen et al., 2013; Wallin et al., 2013; Drake et al., 2018; Gómez-Gener et al., 2021). Lateral C fluxes and brownification of watercourses have increased throughout the northern hemisphere mainly due to climate change (Kritzberg et al., 2020). Increased flux of terrestrial dissolved organic carbon (DOC) to water courses causes a loss of a terrestrial C storage, deteriorates water quality, increases aquatic greenhouse gas emissions, weakens recreational value of water bodies and increases the costs of drinking water treatment (Minkkinen and Laine, 2006; Solomon et al., 2015; Kritzberg et al., 2020; Peacock et al., 2021; Tong et al., 2022; Härkönen et al., 2023).

Drained forested peatlands are large sources of DOC to watercourses (Kortelainen et al., 2006; Nieminen et al., 2015; Finér et al., 2021; Rosset et al., 2022). Lowering water table (WT) with drainage is needed to ensure forest growth on some peatlands, but at the same time the lowering WT also increases the DOC export and CO_2 emissions from peat (Nieminen et al., 2015; Finér et al., 2021; Härkönen et al., 2023). Control of WT with water management is essential in countries with high coverage of peatlands, such as in Finland, Sweden, Estonia and Ireland, where 25, 25, 35 and 44 % of the forests are located in peatlands, respectively. Thus, a planning tool that enables the ability to balance management practices mitigating DOC export and CO_2 emissions whilst allowing wood production on peatlands is of critical need in these countries.

In forested peatlands, lateral C fluxes result from a complex interaction between organic matter decomposition, advective transport with soil water and groundwater, and biodegradation of DOC to CO_2 during the transport (Kalbitz et al., 2003; Bernard-Jannin et al., 2018; Laurén et al., 2019; Payandi-Rolland et al., 2020; Rosset et al., 2020; Palviainen et al., 2022). Thus, the transportation time (residence time) has an important role in the DOC export to water courses (Worrall et al., 2006; Payandi-Rolland et al., 2020; Rosset et al., 2020). DOC is a mixture of organic molecules with different sizes and biodegradation rates (Mastný et al., 2018; Laurén et al., 2019; Palviainen et al., 2022; Prijac et al., 2022). Labile DOC contains more low-molecular weight compounds (LMW) such as sugars, while more decomposition resistant, or recalcitrant, DOC contains more high-molecular weight (HMW) compounds such as cellulose. Labile DOC biodegrades with a half-life in the range of days, whereas the half-life of recalcitrant DOC can be years (Kalbitz et al., 2003). Fresh organic matter releases more CO_2 and DOC, especially in labile fraction compared to more humified organic matter (Laurén et al., 2012). However, the share in which the organic matter decomposition produces CO_2 and DOC is not fully known. In forested peatlands the degree of decomposition and the quality and bulk density of peat changes with depth affecting DOC formation and quality. In a peat profile, the decomposition and subsequent CO_2 flux tends to increase when the WT lowers and the oxidized layer deepens (Laiho, 2006; Ojanen and Minkkinen, 2019). Therefore, forest management practices, such as drainage, ditch blocking or other water management, and forest harvesting that change WT and water flow paths along the transportation route also affect DOC formation and export (Wallage et al., 2006; Nieminen et al., 2015; Nieminen et al., 2018; Palviainen et al., 2022; Williamson et al., 2023).

Predicting DOC and C balance in drained forested peatlands, and their response to management and changing environmental conditions is difficult. This is because the forested peatland system is complex and characterized by multiple feedbacks between the forest stand, WT and water fluxes, litter production and quality, peat characteristics, forest management and drainage (Laurén et al., 2021). WT is affected especially by ditch depth, ditch spacing, evapotranspiration, slope and hydraulic characteristics of peat (Laurén et al., 2021). Development and application of advanced ecosystem models that consider site, stand and terrain characteristics as well as interactions and feedbacks between ecosystem processes, are required in planning of management that mitigates adverse environmental impacts in drained forested peatlands. The SUSI Peatland Simulator (Laurén et al., 2021) is among the first advanced process-based decision support tools developed to simulate peatland hydrology, forest growth, litter production, nutrient fluxes and C balance and can be used to compare different management options with respect to multiple simultaneous ecosystem service targets. In the first SUSI version organic matter decomposition was accounted for using a multivariate regression model, where WT, peat characteristics, temperature and stand dimensions were used as predictors (Ojanen et al., 2010). This regression model does not account for DOC, and CO_2 emissions were modeled based on a factor (per area and unit of time) instead of a mass balance of the decomposing material, and thus, did not include lateral C fluxes in water.

Our objective was to study the effects of drainage intensity on DOC export and site C balance. To achieve this, we developed an extended version of SUSI Peatland Simulator, that includes lateral C fluxes with a new mass-balance based decomposition module (Extended Soil Organic Matter decomposition model, ESOM) that includes DOC formation, transport and biodegradation. This is the first comprehensive C model for drained forested peatlands. We designed and parameterized ESOM using literature as well as tested the model performance against an independent dataset measured in the laboratory. The eight-month laboratory experiment used peat columns collected from study sites located in Finland, Estonia, Sweden and Ireland. SUSI was used to study the effects of drainage intensity on lateral C fluxes and C balance in the study sites in Finland, Estonia and Sweden. The study sites were selected based on the drainage strategy, i.e. a combination of depth, orientation and spacing of ditches, which vary between countries. This permitted a more comprehensive understanding on how drainage intensity of peatland forests affects C fluxes. We hypothesized that reducing the depth to which ditches are cleaned and widening the ditch spacing will reduce DOC export and improves C sink in peatland forests.

2. Material and methods

2.1. Layout of the study and study sites

Our study consisted of three parts: model development, model application and experiment supporting the modeling (Fig. 1). In the model development (Sections 2.4–2.6) we composed a mass-balance-based organic matter decomposition model called ESOM, tested it against an independent laboratory experiment (Section 2.6) and implemented ESOM in the SUSI Peatland Simulator (Section 2.7). We selected study sites from Finland, Estonia, Sweden and Ireland (Section 2.1) and extracted peat samples for 8-months laboratory incubation (Section 2.2) during which we measured CO_2 emissions and soil water DOC concentrations. The study sites represent typical drainage strategies for each country. In the model application, we used the updated SUSI Peatland Simulator to calculate the effects of different drainage intensities on lateral C fluxes and site C balance for the study sites in Finland, Estonia and Sweden.

Peat columns were extracted from six drained forested peatlands located in Finland, Estonia, Sweden and Ireland (Fig. 2, Table 1). Paroninkorpi study site is located in southern Finland (Palviainen et al., 2022). The site is minerotrophic herb-rich type (Laine, 1989) Norway spruce (*Picea abies* (L.) Karst) dominated forest with sedge-wood peat deposits. Estonian study sites Ullika in east Estonia and Ess-soo in south-east Estonia are Scots pine (*Pinus sylvestris* L.) dominated nutrient-poor ombrotrophic bogs with *Sphagnum* peat deposits (Burdun et al., 2021). Krycklan and Trollberget Experimental Areas are located in Northern Sweden approximately 50 km northwest of the city of Umeå (Laudon et al., 2013; Laudon et al., 2021). The stands represent Norway spruce-dominated bilberry horsetail type peatland with sedge-wood peat, and the vegetation consists of ericaceous shrubs, mostly bilberry (*Vaccinium myrtillus*). The Lough Atorick study site is close to the Lough Atorick lake in the Slieve Aughty Mountains, Co. Clare, Ireland and has been planted with Sitka spruce (*Picea sitchensis)* on blanket bog peat otherwise dominated by *Rhynchospora alba-Sphagnum cuspidatum*. The thickness of the *Sphagnum* peat deposits is 150–200 cm and the underlying soil is Old Red Sandstone. In the study sites, a different array of ecological monitoring has been conducted including measurements of WT and ditch water DOC concentrations (Laudon et al., 2013; Laudon et al., 2021; Veber et al., 2021; Palviainen et al., 2022; Mendes et al., 2023).

2.2. Laboratory experiment

We studied the effect of WT on CO_2 emission and pore water DOC concentration by incubating peat columns (diameter 0.16 m height 0.5 m) in a laboratory for eight months (Table 1). WT was set to −0.2 m and − 0.4 m distance from the column upper end. This results in slightly different WTs because peat depth did not always fill the whole column. During the incubation, the air temperature in the laboratory ranged between 18 and 34 °C. CO_2 emissions and DOC concentrations were measured at the same time in monthly intervals from all columns starting after a two-month stabilization period. Similar stabilization period during which the labile C is consumed, has commonly been used in soil incubation experiments (Conant et al., 2011; Hamdi et al., 2013; Laurén et al., 2019).

Water samples were collected using Rhizon soil water samplers (Rhizosphere Research Products) inserted into the peat column at 10 cm height from the bottom. Water samples were filtered through syringe filters (0.45 μm) and stored at −21 °C. Water samples were treated with phosphoric acid and DOC concentrations were analyzed with thermal oxidation coupled with infrared detection (Multi N/C 2100, Analytik). Change in the DOC concentration (ΔDOC) during the experiment was obtained by subtracting the DOC concentration at the end of the experiment from the initial DOC concentration.

CO_2 emissions were measured from the headspace of the column with dark chambers (h = 0.21 m and Ø = 0.16 m). Thus, the measured CO_2 flux includes both the heterotrophic respiration from the organic matter decomposition and the maintenance respiration of the possible vegetation growing on the column. A small fan was set inside the chamber to ensure air circulation, and the CO_2 concentration was measured with a nondispersive infrared CO_2 probe (GMP343, Vaisala Oyj). Relative humidity and temperature were measured using a sensor (HMP75, Vaisala Oyj). The sensor measured at 15 s intervals for 5 min

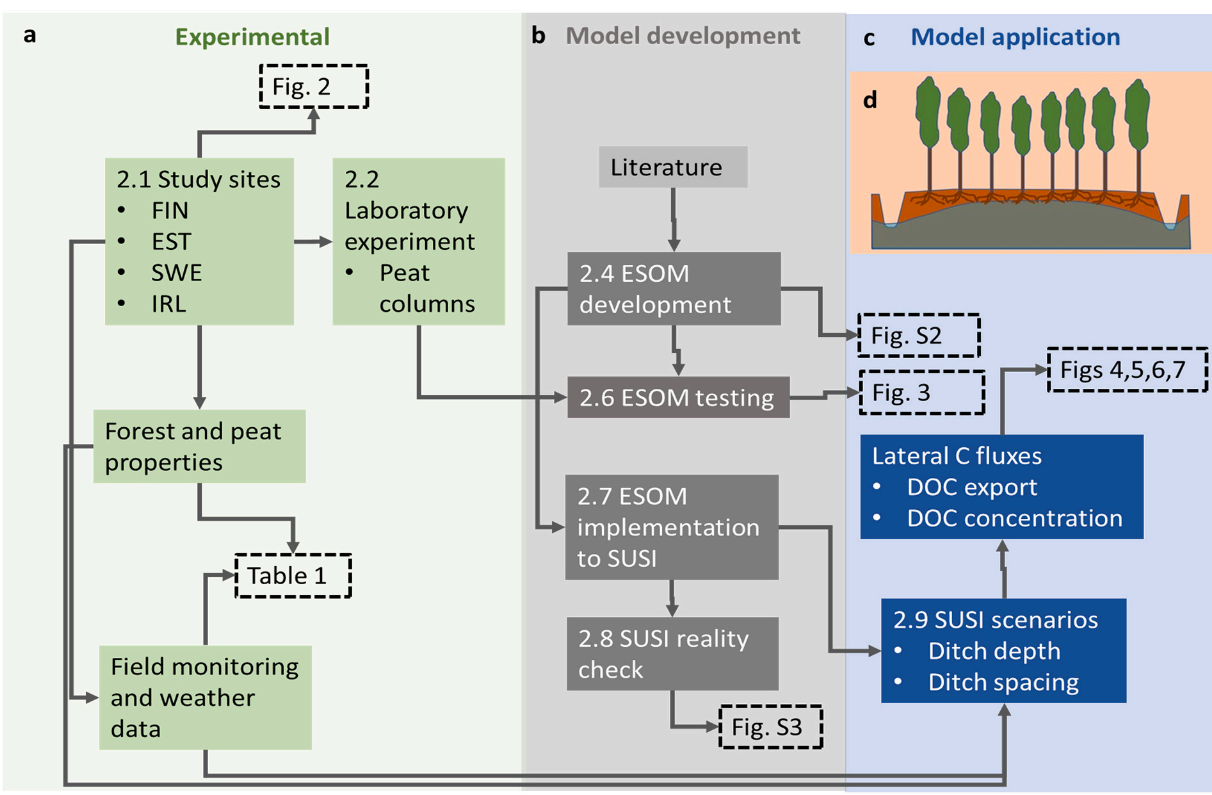

Fig. 1. The study consists of a) Experimental, b) Model development and c) Model application parts. Numbers in the solid boxes represent the sections, arrows show the data flow, and the dashed boxes show the outputs in tables and figures. Extended Soil Organic Matter model ESOM (b) was developed using literature and was tested against independent laboratory experiments (a). ESOM was embedded into the SUSI Peatland Simulator which describes peatland as a 2D cross section located between adjacent ditches (d). Updated SUSI was applied to simulate effects of drainage intensity on lateral carbon fluxes and site carbon balance.

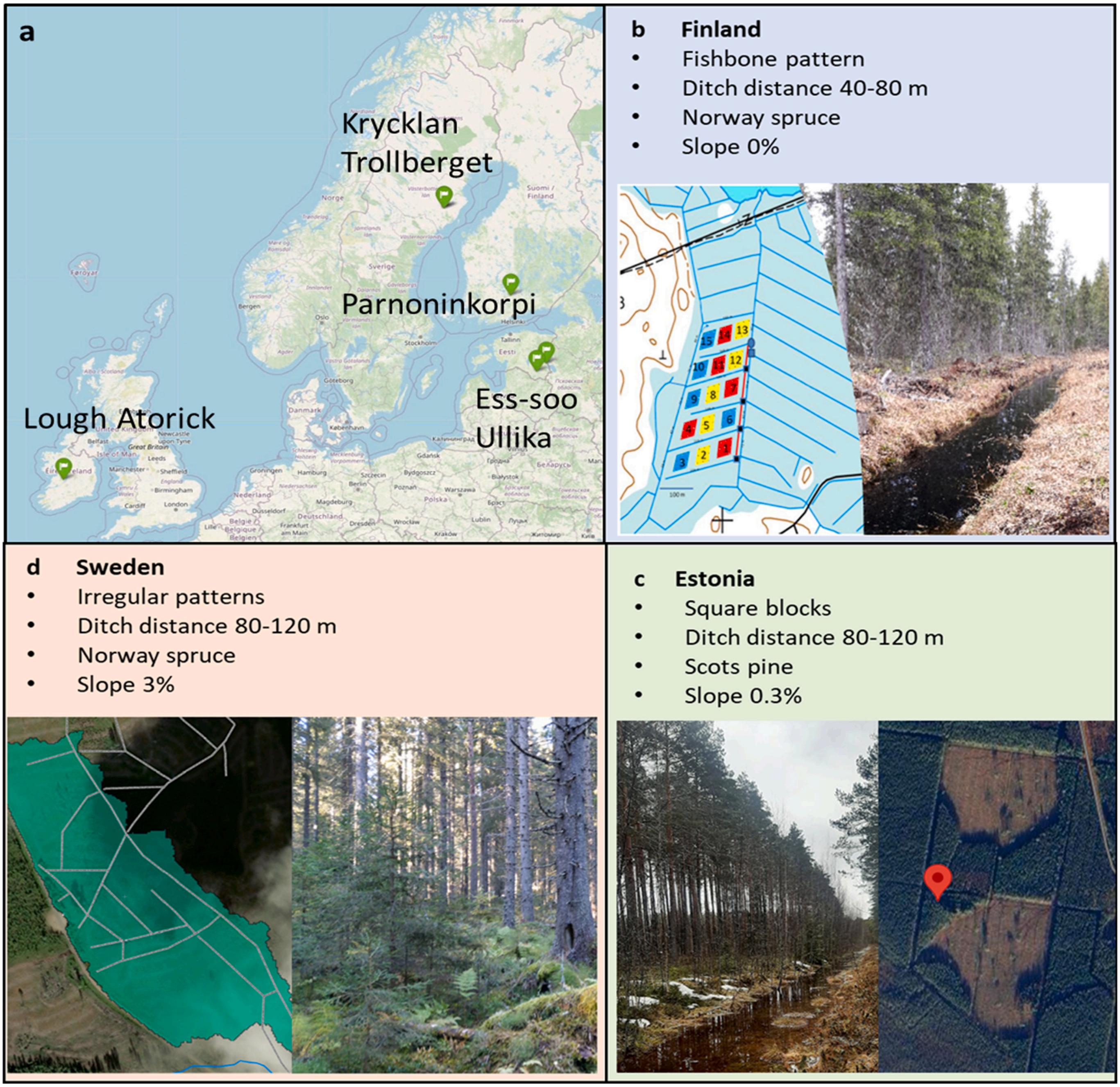

Fig. 2. (a) Peat columns were collected from Finland (Paroninkorpi), Estonia (Ess-soo and Ullika), Sweden (Kryckland and Trollberget) and Ireland (Lough Atorick). Peatland simulator SUSI was applied to study sites in Finland (b), Estonia (c) and Sweden (d). Drainage strategy, main tree species and slope were different in these sites.

periods. The CO_2 flux was calculated as described in Aaltonen et al. (2022). The measured CO_2 emissions were converted to daily values, and the linearly interpolated CO_2 emission values were summed throughout the experiment time to obtain the cumulative CO_2 emissions.

To separate the heterotrophic respiration from the autotrophic respiration, the columns were grouped based on the presence of vegetation. In non-vegetated columns, the measured CO_2 emission represents solely the heterotrophic respiration. The contribution of the autotrophic respiration for the columns containing vegetation was estimated based on the biomass change of mosses and vascular plants during the experiment. We assumed that the moss biomass is totally aboveground and that the vascular plants (mainly sedges) contain 80 % of their biomass belowground (Saarinen, 1996). The total mass of the vegetation was assumed to be a result of the cumulative plant net primary production during the experiment. The autotrophic maintenance respiration typically is approximately equal to the net primary production, and therefore, we assumed further that the cumulative autotrophic respiration is equal to the biomass change. To obtain the cumulative soil respiration we subtracted the cumulative autotrophic respiration from the measured cumulative emissions.

After the experiment, the peat columns were horizontally cut in three equal length pieces (upper, middle and lower section) and cylindrical subsamples (diameter 30 mm, length 30 mm) were extracted from the middle of the sections for dry bulk density (ρ_b) measurement. The bulk density was obtained by dividing the dry mass (measured after 4 days of drying in 105 °C) with the volume of the subsample.

Table 1

Weather conditions, site, stand and peat characteristics and water table (WT) depth in the laboratory experiment and in the field in the study sites in Finland, Estonia, Sweden and Ireland. Simulations were carried out for Paroninkorpi, Ullika and Krycklan study sites. Stand and site characteristics and drainage scenarios used in the parameterization of SUSI are presented in the table. In the Estonian site, the stand characteristics varied and the model domain was divided into three equal-sized sections with different stand volume, basal area and number of stems (separated with semicolons). The ditch depths separated by semicolons indicate the depths of the left and the right ditch (see Modeling domain in Fig. 3). Note the asymmetric ditch depths in the Estonian site.

	Finland Paroninkorpi	Estonia Ullika	Estonia Ess-soo	Sweden Krycklan	Sweden Trollberget	Ireland Lough Atorick
Mean annual temperature,°C	4.7	5.9	5.9	1.8	2.1	9.4
Mean annual precipitation, mm	638	661	661	623	713	1186
Peat thickness (cm)	150	20–157	>300	30–40	10–21	150–200
Underlying soil	Glacial Till	Glacial Till	Glacial Till	Glacial Till	Glacial Till	Sandstone
Main tree species	*Picea abies*	*Pinus sylvestris*	*Pinus sylvestris*	*Picea abies*	*Picea abies*	*Picea sitchensis*
Stand volume, m^3 ha^{-1}	122–203	20–238	14	200	275	384
Number of peat columns	48	12	8	10	10	20
Peat bulk density, kg m^3 layer 1	106 ± 40	101 ± 45	101 ± 45	144 ± 23	144 ± 23	163 ± 33
Peat bulk density, kg m^3 layer 2	158 ± 24	120 ± 46	120 ± 46	185 ± 25	185 ± 25	185 ± 25
Peat bulk density, kg m^3 layer 3	159 ± 18	146 ± 84	146 ± 84	201 ± 19	201 ± 19	105 ± 17
WT (cm) in the laboratory, vegetation	0.25 ± 0.10	0.24 ± 0.11	0.24 ± 0.11	0.17 ± 0.04	0.17 ± 0.04	0.25 ± 0.09
WT (cm) in the laboratory, no vegetation	0.29 ± 0.11	0.29 ± 0.10	0.29 ± 0.10	0.21 ± 0.10	0.21 ± 0.10	0.29 ± 0.09
June–August WT (cm) in the field	50–99	23–79	15–46	33–55	52–92	17–44
Input values in the simulations:						
Stand volume, m^3 ha^{-1}	225	175; 80; 30		250		
Basal area, m^2 ha^{-1}	25	20; 13; 8		24		
Number of stems ha^{-1}	1350	1000; 800; 720		1200		
Slope, %	0	0.3		3		
Ditch spacing scenarios, m	40; 60; 80	80; 100; 120		80; 100; 120		
Ditch depth scenarios, cm	30; 30	30; 10		30; 30		
	60; 60	60; 30		60; 60		
	90; 90	90; 90		90; 90		

2.3. SUSI peatland simulator

The SUSI Peatland Simulator describes hydrology, biogeochemical processes and stand growth along a 2D cross-section between two parallel ditches (Laurén et al., 2021, Fig. 1, Fig. S1). The cross section located between the ditches was discretized into two-meter-wide vertical columns. All inputs and outputs were given individually for each column. The solution of the daily WT follows a quasi-2D approach. The calculation of vertical water fluxes was simplified by assuming that a change in water storage in the peat column immediately affects WT, and that the water content above WT achieves hydraulic equilibrium instantly. The horizontal water movement is computed using the implicit solution of a groundwater equation (Laurén et al., 2021). We introduced new model components for simulating organic matter decomposition, C balance, and lateral C fluxes (Fig. S1).

2.4. ESOM - extendable soil organic matter decomposition module

We developed a new spatially Extendable Soil Organic Matter decomposition model (ESOM) to replace the previously used statistical empirical model (Ojanen et al., 2010) in the SUSI Peatland Simulator (Laurén et al., 2021). A commonly used first-order decay approach was selected (Manzoni and Porporato, 2009) because it supports maintaining the soil mass balance, is computationally efficient and enables the computation of a grid of locations at the same time. ESOM facilitates modeling of DOC formation and biodegradation, and therefore is a prerequisite for quantifying lateral C fluxes. A new, computationally efficient, process-based model structure was also needed to support calculation of spatially heterogeneous managed peatland ecosystems, where dynamic and spatial variation of WT is a fundamental characteristic and affected by drainage. ESOM keeps the mass balance, and therefore also supports simulation of different harvesting regimes. ESOM takes new organic matter, including tree and ground vegetation litter, tree mortality and logging residues as an input. During decomposition, organic matter moves from the decomposition stage to the next, the mass decreases and CO_2, DOC and nutrients are released. Mass balance implies that the mass or nutrient storage change in soil equals the difference between inputs and mass or nutrient release.

Drainage changes the dynamics of the stand, ground vegetation, litter production and decomposition of organic matter (Laiho, 1997; Minkkinen et al., 1999; Sarkkola, 2006). As a result of these changes, a raw humus layer develops over the original peat (Päivänen and Hånell, 2012). In ESOM all the new litter is located in the raw humus. Raw humus has similar morphological structure to mor layers in upland soils. This layer is divided into three successional storage compartments following the degree of decomposition: litter (L), partly decomposed fermentation (F), and highly decomposed humus (H) storage compartment. Organic matter input is derived from the stand litter and mortality, ground vegetation litter and logging residues. All the incoming litter is separated into woody and non-woody material. The woody litter follows its own decomposition succession (L_w, F_w, H) and the non-woody litter its own succession (L_{nw}, F_{nw}, H). The decomposition of L, F and H storage compartments are described using the SOMM model (Chertov and Komarov, 1997). Similarly to SOMM, the rate of decomposition depends on soil temperature, water content, pH, and litter ash, lignin and nitrogen (N) content. Daily raw humus temperature is calculated with the peat temperature module of SUSI (Laurén et al., 2021). Water content relative to field capacity is used to scale the decomposition rate of L and F storage compartments (see ROMUL manual). Litter ash content parameter does not change during the simulation. For the raw humus, the release of CO_2 and the transition of the organic matter from one successional stage to the next is controlled by parameters (k_{1nw}, k_{1w}, k_{2nw}, k_{2w}, k_{3nw}, k_{3w}, k_{4nw}, k_{4w}, k_{5nw}, k_{5w}, k_6; see Fig. S2) and their dependency on environmental conditions and properties of the decomposing material are described in Chertov and Komarov (1997) and Chertov et al. (2001).

ESOM extends the SOMM model including the decomposition of peat under the raw humus layer. Peat is divided into three layers (P_1 0 m…-0.3 m; P_2 0.3 m … -0.6 m, and P_3 below −0.6 m), which decompose according to the same principles as the H storage in the SOMM model (Fig. S2, Eq. (1)). Parameters (k_7, k_8, k_9) were derived from the literature. Initial soil pH depends on site fertility class and it is given as an input value for the simulation.

ESOM includes matter storage compartments accounted for in the mass vector $B_{x,y}$:

$$\begin{bmatrix} L_{0nw} \\ L_{0w} \\ L_{nw} \\ L_w \\ F_{nw} \\ F_w \\ H \\ P_1 \\ P_2 \\ P_3 \\ out \end{bmatrix}$$

where L_{0nw} and L_{0w} are the non-woody and woody litter inputs from the stand and ground vegetation, and L_{nw}, L_w, F_{nw}, F_w and H are successional stages of the raw humus layer following Chertov and Komarov (1997) and Chertov et al. (2001), and P_1, P_2 and P_3 are the peat storage compartments, and *out* represents cumulative mass loss from all the mass storage compartments. The unit is kg m^{-2}. The mass loss and the shift of the remaining mass from one storage to another was calculated in daily time steps using the transition matrix $A_{x,y}$:

$$\begin{bmatrix} 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 \\ 1 & 0 & 1-(k_1+k_3)_{nw} & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & 1 & 0 & 1-(k_1+k_3)_w & 0 & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & k_3 & 0 & 1-(k_2+k_4+k_5)_{nw} & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & k_{3w} & 0 & 1-(k_2+k_4+k_5)_w & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & (k_4+k_5)_{nw} & (k_4+k_5)_w & 1-k_6 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & 0 & 0 & 1-k_7 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 1-k_8 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 1-k_9 & 0 \\ 0 & 0 & k_1 & k_{1w} & k_2 & k_{2w} & k_6 & k_7 & k_8 & k_9 & 1 \end{bmatrix}$$

In the transition matrix $A_{x,y}$, *k*-parameters regulate the mass loss and the transition of the organic matter from one successional stage to the next. Values of the raw humus *k*-parameters (k_1…k_6) depend on soil temperature, soil moisture and litter quality as described in Chertov and Komarov (1997) and Chertov et al. (2001), and the woody litter has adjustments for the lignin content (ROMUL manual).

For each daily update, the transition matrices ($A_{x,y}$) for each grid point (x,y) were located in a single, block-diagonal matrix *A*, and the grid-wise mass vectors ($B_{x,y}$) were located in a single mass vector B. The values of the mass for the whole modeling domain for time *t* were obtained with a single matrix multiplication (Eq. (1)).

$$B_t = A_t B_{t-1} \tag{1}$$

To facilitate tracking of the development of the soil organic matter, the mass vector *B* was stored as a three-dimensional array (x, y, 11), where x and y are coordinates of the soil column and 11 is the length of mass vector $B_{x,y}$. A separate instance of the ESOM model was initiated to simulate the dynamics of mass, N, phosphorus (P) and potassium (K). These substances are moved across the successional stages using the transition matrix $A_{x,y}$ and the same rate parameters adjusted by the temperature, soil moisture, ash contents, pH and the N content of litter.

Based on field experiments, it is known that N is released from the decomposing organic matter at a slower rate than the mass loss, whereas the release of K is considerably faster than the mass loss (Palviainen et al., 2004). This was described by introducing parameter *u* to $A_{x,y}$ in a similar way to what Chertov et al. (2001) used to restrict the N release rate in the advanced version of SOMM. In the transition matrix $A_{x,y}$ parameter k_1 was adjusted by u_{k1} = (0.1, 1.1,1.5) for N, P and K, respectively. Parameter k_2 was adjusted by u_{k2} = (0.1, 0.45, 1.5) for N, P and K, and parameters k_6, k_7, k_8 and k_9 were adjusted by u_{kn} = (0.125, 0.3, 1.5) for N, P and K, respectively.

In ESOM the peat layers P_1, P_2 and P_3 decompose according to first-order kinetics (Eq. (1)). Decomposition constants were gained from literature from controlled laboratory experiments that have been conducted for corresponding peat types that exist in our study sites (Lappalainen et al., 2018; Laurén et al., 2019). In first-order kinetics, the decomposition is controlled by the mass of the decaying material and the decay constant (k_p, kg kg^{-1} day^{-1}). In peat, the oxygen availability and the prevailing temperature regulate the rate of decomposition (Moore and Basiliko, 2006) (Eq. (2)). During a drought event, peat can occasionally become too dry for the optimal decomposition and therefore k_p should be scaled down when the WT lowers.

$$k_p = k_{ref}{}^* f(afp)^* f(T)^* f(WT) \tag{2}$$

where k_p is the decomposition rate for peat layer P_1, P_2 or P_3 (kg kg^{-1} day^{-1}), k_{ref} is the decomposition rate at reference conditions (temperature 10 °C) [kg kg^{-1} day^{-1}], $f(afp)$ is the function restricting the decomposition rate on the basis of oxygen availability, $f(T)$ on the basis of soil temperature, and $f(WT)$ on the basis of water content. Oxygen supply to the peat volume above the WT depends on the architecture and connectivity of air-filled pores (Kiuru et al., 2022), and consequently, only a part of the peat volume is accessible to the aerobic decomposition. We assumed that the fraction of peat mass accessible to the aerobic decomposition is directly proportional to the air-filled porosity:

$$f(afp) = \frac{\theta s - \theta}{\theta s} \tag{3}$$

where θ_s is the water content at full saturation [m^3 m^{-3}], θ is the prevailing water content [m^3 m^{-3}]. For the temperature dependency, we applied the Q_{10} approach:

$$f(T) = Q_{10}^{\frac{T - T_{ref}}{10}} \tag{4}$$

where Q_{10} is the temperature sensitivity parameter (Q_{10} value 2.0).

The values for k_{ref} were obtained from the literature from controlled laboratory experiments that have been conducted for corresponding peat types that exist in our study sites (Lappalainen et al., 2018; Laurén et al., 2019). The reported decomposition rates were first converted to unit kg kg^{-1} day^{-1} of organic matter and then the values were standardized by dividing them with the air-filled porosity, and finally projected to the reference temperature of 10 °C using Eq. (4). The mean k_{ref} values in Lappalainen et al. (2018) and Laurén et al. (2019) were 0.00045 kg kg^{-1} day^{-1} and 0.0001 kg kg^{-1} day^{-1}, respectively. The higher value was used to describe the decomposition in the topmost P_1 layer (k_7 = 0.00045 kg kg^{-1} day^{-1}) and the lower value for P_2 and P_3 layers ($k_8 = k_9$ = 0.0001 kg kg^{-1} day^{-1}). Following the evapotranspiration moisture modifier composed by Koivusalo et al. (2008), the value of $f(WT)$ decreased linearly from 1 to 0.5 when the WT lowered from −0.5 m to −1.2 m; and thereafter the value of $f(WT)$ further decreased from 0.5 to 0 when the WT lowers below −1.2 m, and the consequent water potential approaches the wilting point. However, it is important to point out that in the peat hydrology model, the vertical distribution of water follows the hydrological equilibrium assumption, which likely to overestimates the surface water content under extensive drought.

Organic matter decomposition produces CO_2, labile low molecular weight DOC (LMW-DOC) and recalcitrant high-molecular weight DOC (HMW-DOC) in a proportion as experimentally described by Laurén et al. (2019). Laurén et al. (2019) considered DOC as an intermediate product of the decomposition and noticed that the share between the CO_2 and DOC formation is temperature dependent. At low temperatures (0–5 °C) DOC formation can be c.a. 7 % of the CO_2 production, whereas at higher temperatures (20–25 °C) the decomposition is more complete and the share of DOC formation decreases to 1–2 % of the CO_2 production. Following the different temperature sensitivity of CO_2 production and DOC formation presented in Laurén et al. (2019) the share of DOC formation from the CO_2 production was obtained as:

$$DOC/CO2 = 0.066^{*}exp(-0.061^{*}T) \tag{5}$$

where DOC/CO2 is the share of DOC production from the calculated CO_2 emission, and T is temperature in peat layer P_1, P_2 or P_3 (°C).

2.5. Advection, residence time and biodegradation of DOC

DOC disintegrates into CO_2 in a biodegradation process (Kalbitz et al., 2003; Palviainen et al., 2022) following a combined negative exponential decay function for labile and recalcitrant DOC.

$$DOC_{bd} = LMWexp(-k_{LMW}{}^{*}t_r) + HMWexp(-k_{HMW}{}^{*}t_r) \tag{6}$$

where DOC_{bd} is the amount of DOC transformed to CO_2 (kg ha^{-1} day^{-1}), LMW is the storage of low molecular weight DOC (kg ha^{-1}), HMW is the storage of high molecular weight DOC (kg ha^{-1}), k_{LMW} is the biodegradation rate for LMW (value 0.15 day^{-1}), k_{HMW} is the biodegradation rate for HMW (value 0.0004 day^{-1}), and t_r is the residence time (days). Parameters for the DOC biodegradation were adopted from Kalbitz et al. (2003), which were supported by our measurements from the study site in Finland (Palviainen et al., 2022). The longer the residence time of water in the soil, the higher the proportion of DOC that disintegrates before reaching the ditch.

We calculated the residence time (t_r, days), i.e. the time required for water transport from each computation node (n) to the ditch node (d) using the mean annual WT gradient $\Delta H/\Delta x$ at each computation node, the mean hydraulic conductivity (K_{sat} m s^{-1}) and the mean porosity (θ_s, m^3 m^{-3}) of the peat. The total residence time was obtained by summing up the node-wise residence times from the column n to the ditch node d.

$$t_r = \Sigma_n^d \frac{\Delta x}{\frac{K_{sat}}{\theta_s}\frac{\Delta H}{\Delta x}} \tag{7}$$

where H is the hydraulic head (m above a fixed datum), ΔH is the difference of H between the adjacent nodes (m), Δx is the horizontal distance between the adjacent nodes.

2.6. Testing ESOM against experimental data

We tested the performance of ESOM against the laboratory experiment (see Section 2.2). In the laboratory experiment, air temperature was recorded in conjunction with the CO_2 emission measurements, and daily temperatures between the CO_2 measurements were linearly interpolated. The daily interpolated temperature data was used as an input in the ESOM simulations. In the simulations, WT in each peat column was set to the same level as in the experiment and the peat column was divided into 0–0.15 m, 0.15–0.40 and 0.40–0.50 m layers. The measured ρ_b was used to calculate water retention characteristics according to the equations presented by Päivänen (1973). Water content and air-filled porosity were estimated using water retention characteristics separately for 0.05 m thick layers of the peat column. We assumed that the water potential (ψ, m H_2O) in the peat column stays in a hydrologic equilibrium, which means that the prevailing water potential in a certain peat layer equals the vertical distance from the layer centerpoint to the WT (in m H_2O). This assumption allows the application of the water retention characteristics and the derivation of the air-filled porosity in the sample. We recorded the calculated cumulative heterotrophic respiration and converted it to CO_2 emission and derived the DOC release using Eq. (5) and the biodegradation using Eq. (6). The simulated cumulative DOC release subtracted by the biodegradation (mg) was divided by the water volume (unit L) of the sample to obtain the DOC concentration change (ΔDOC_{sim}) in the peat column during the experiment. The mean absolute error was calculated between measured and simulated values.

2.7. Incorporating the ESOM into the SUSI peatland simulator to include the C balance with lateral C fluxes

We incorporated the ESOM decomposition model into the SUSI Peatland Simulator, where it replaced an empirical regression model for peat CO_2 emissions (Ojanen et al., 2010). ESOM receives daily WT, soil temperature and air-filled porosity in the depths of the ESOM layer midpoints and using a daily time step, calculates the organic matter decomposition and release of CO_2, HMW-DOC, LMW-DOC, and nutrients (N, P, K), as well as new organic layer mass and nutrient contents. The woody and non-woody litter inputs were calculated using an annual timestep (September 1st). Woody litters included remnants of branches, stems, stumps and coarse roots (called “woody litter”), but also had separate categories for logging residues or trees that died (called “tree mortality”). Non-woody litter included residues from leaves, fine roots and ground vegetation. Annual stand mass balance for organic matter and C was updated with an annual time step:

$$S_{bal} = (\Delta B_s + \Delta B_{gv} + L_{ws} + L_{nws} + L_{wlr} + L_{nwlr} + L_{gv} + L_{wm} + L_{nwm} - O_{atm})f_{mtoC} - O_{HMW} - O_{LMW} - O_{CH4C} \tag{8}$$

where S_{bal} is the stand annual C balance (kg ha^{-1} yr^{-1}), ΔB_s is the stand mass change (kg ha^{-1} yr^{-1}), ΔB_{gv} ground vegetation mass change (kg ha^{-1} yr^{-1}), L_{ws} is woody litter from the stand (kg ha^{-1} yr^{-1}), L_{nws} is non-woody litter from the stand (kg ha^{-1} yr^{-1}), L_{wlr} is woody logging residues (kg ha^{-1} yr^{-1}), L_{nwlr} is non-woody logging residues (kg ha^{-1} yr^{-1}), L_{gv} is non-woody litter from ground vegetation (kg ha^{-1} yr^{-1}), L_{wm} is woody litter from tree mortality (kg ha^{-1} yr^{-1}), L_{nwm} is non-woody litter from tree mortality (kg ha^{-1} yr^{-1}), O_{atm} is the mass loss to the atmosphere due to decomposition (kg ha^{-1} yr^{-1}), $f_{m\ to\ C}$ is a conversion factor from organic mass to C (0.5 kg C kg^{-1}), O_{HMW} is C loss in decomposition in form of HMW-DOC (kg C ha^{-1} yr^{-1}), O_{LMW} is C loss in decomposition in form of LMW-DOC (kg C ha^{-1} yr^{-1}) and O_{CH4C} is C loss in form of methane outflux (kg C ha^{-1} yr^{-1}). Note that CH_4 is expressed as C flux and not converted to CO_2 equivalent. Soil C balance was calculated after omitting ΔB_s and ΔB_{gv} from the equation.

2.8. Reality check for the updated peatland simulator SUSI

The updated SUSI was run against a large experimental dataset which was also used in the development of the previous version (Laurén et al., 2021, Fig. S3). This reality check was done because WT affects organic matter decomposition, nutrient release and stand growth, and the stand, in turn, has a feedback to WT and litter production.

After the modification, SUSI was still producing the WT, as well as the tree biomass and stand volume growths with good accuracy (Fig. S3). ESOM predicted, on average, similar CO_2 emissions than the empirical model (Ojanen et al., 2010) used in the previous version of SUSI. The model by Ojanen et al. (2010) is based on average WT during the growing season but ESOM can take into account spatial and temporal variation in WT, and therefore, predict higher CO_2 emissions close to the ditches and lower CO_2 emission in the midway between the ditches.

2.9. Parameterization and application of the updated SUSI peatland simulator and calculation of lateral C fluxes

In Finland, the ditch drains (contour ditches) are usually located in a fishbone pattern around the main ditch and the distance between the contour ditches is typically 40 m - 80 m (Fig. 2 b). The fishbone

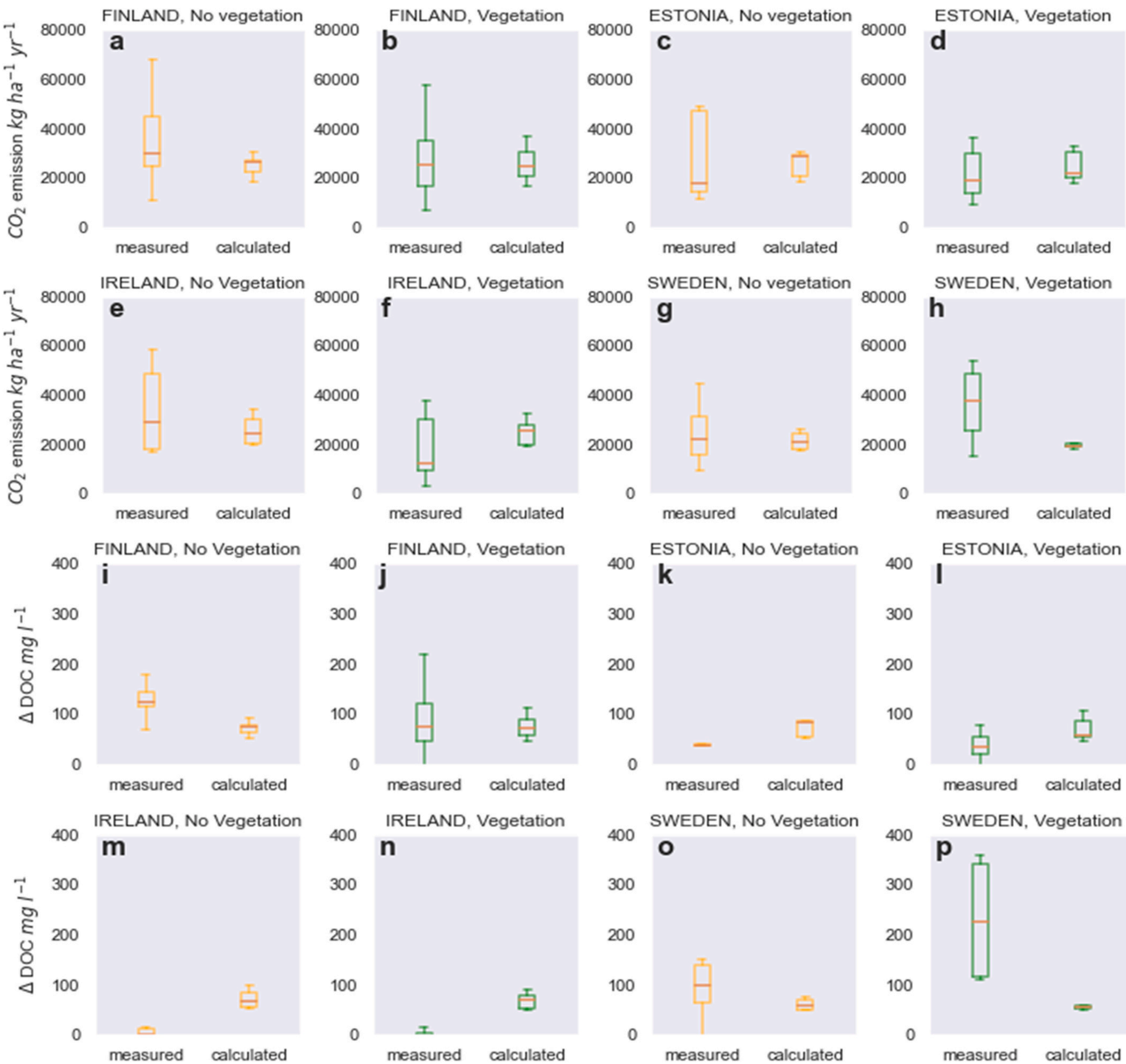

Fig. 3. Measured and calculated soil CO_2 emissions (a-h) and change in dissolved organic carbon (DOC) concentration from peat columns (i-p) collected from Finland (a, b, i, j), Estonia (c, d, k, l), Ireland (e, f, m, n) and Sweden (g, h, o, p). Peat columns were grouped based on the presence of vegetation. Calculated values CO_2 were obtained by cumulating daily CO_2 emissions predicted by the ESOM model. Note that the cumulative CO_2 fluxes during the laboratory experiments have been converted to annual basis and hectare basis. The first water sample was extracted after two months and the last one was extracted eight months after the onset of the experiment. The concentration change (ΔDOC) was calculated as a concentration difference between these two time points. Calculated values were obtained by DOC accumulation (DOC release subtracted by biodegradation) and dividing the mass of accumulated DOC by the sample water volume. Horizontal line indicates the median and the box extends from 25 to 75-percentiles and the whiskers show the range of the data.

arrangement of the ditches leads to intensive drainage and allows effective removal of water from the area through a larger main ditch (Päivänen and Hånell, 2012). While in Estonia, marginal ditches surround the peatland, and within the dominantly flat area, the ditch drains delineate regular square blocks, where the shortest distance between the ditches varies between 80 m - 120 m (Fig. 2 c). In this arrangement the wide main ditch is missing and, therefore, compared to the fishbone pattern, the square-block arrangement is less effective in removing water from the area. The Swedish study site represents a typical drained peatland in Sweden, where the ditch network is irregular and the distance between the ditches is longer than in Finland (Fig. 2 d). This results in less intensive drainage, however, for our study site the slope improves the drainage.

In the ditch excavation the target depth has been 90 cm in ditch drains, whereas the main ditches can be deeper (Päivänen and Hånell, 2012). After the excavation, the ditch depth starts to decrease due to sedimentation and ingrowth of vegetation (Hökkä et al., 2020). Thus, the ditch depth typically varies from 30 cm to 90 cm depending on the time elapsed from the ditching. Consequently, the drainage intensity gradually decreases. In ditch network maintenance (DNM), the sediments and vegetation are removed from the ditches using an excavator, often in conjunction with harvesting operations or about 20–40 years intervals. DNM increases the stand growth by 10–15 m^3 ha^{-1} over 20 years (Ahtikoski et al., 2008), but it also increases the export load of sediments and dissolved nutrients into water courses (Joensuu et al., 2001; Nieminen et al., 2018; Finér et al., 2021). As a response to the concern of the adverse environmental effects of drainage, there is an increasing need to raise WT in forested peatlands using water management. The impacts of lowering WT with DNM and raising of WT by reducing the depth to which ditches are cleaned in countries with high peatland coverage has been scarcely studied.

To study the effects of drainage intensity on C balance and lateral C fluxes in boreal forested peatlands, we compiled simulation domains that correspond to our study sites, and thereafter, varied the drainage intensity by changing the distance between the ditches in 20 m intervals and ditch depth in 30 cm intervals (Table 1). Simulation sites were selected so that they represented typical drainage strategies in forestry in different countries. We left out the sites that were poorly drained and had low stand volume or very thin peat layer (Table 1). Ireland was not included in the simulations, because currently SUSI does not account for Sitka spruce development.

For each of the three locations, we compiled a 10-year daily meteorological dataset including air temperature, precipitation, water vapor pressure and global radiation. The ten-year period included varying weather conditions in terms of temperatures and precipitation. The simulation domain representing a two-dimensional cross-section of peat and forest between two adjacent ditches was constructed based on the field data (Table 1). Measured peat bulk density (ρ_b) (Table 1) was used in the parameterization of the soil hydrology and ESOM. The bulk density of the lowermost subsample was used for layers below 50 cm.

The annual release of LMW-DOC and HMW-DOC, biodegradation during the transport and the resulting export load of DOC to ditches (kg ha^{-1} yr^{-1}) were extracted from the simulation results. The simulated mean annual concentration of DOC in the ditch water was obtained by dividing the annual export load with the annual volume of the runoff. The simulated DOC concentrations were compared to DOC concentrations in ditch water observed from the study sites.

3. Results

3.1. Laboratory experiment and ESOM development

Measured and simulated CO_2 emissions from peat columns in the laboratory conditions had large variation both for samples with and without vegetation (Fig. 3). ESOM predicted the mean CO_2 emissions for Finnish (mean absolute error 2550 kg CO_2 ha^{-1} yr^{-1}) and Estonian (mean absolute error 2448 kg CO_2 ha^{-1} yr^{-1}) columns reasonably well. Mean absolute errors for Swedish (9669 kg CO_2 ha^{-1} yr^{-1}) and Irish (7046 kg CO_2 ha^{-1} yr^{-1}) columns were higher.

DOC concentration increased by 40–130 mg L^{-1} in Finnish and Estonian columns during the incubation, and the measured and simulated concentration changes were in the same order of magnitude (Fig. 3 i-l). For Finnish columns, the mean absolute error was 28 mg L^{-1} and for Estonian columns 34 mg L^{-1}. In columns originating from Ireland (Fig. 3 m, n), the measured DOC concentration changes were very small while ESOM predicted considerably higher increases in DOC concentration during the experiment (mean absolute error 86 mg L^{-1}). In Swedish columns, ESOM underestimated the DOC concentration increase (Fig. 3 o, p), and the mean absolute error was 134 mg L^{-1}. Swedish peat columns were rather thin and WT was close to the sample surface (Table 1).

3.2. Effect of drainage intensity on DOC export and site C balance

The simulated DOC export (Fig. 4 a-c) and concentration (Fig. 4 d-f) increased with deeper ditches and denser ditch spacing at all study sites. Overall, the simulated annual DOC exports were in a range of 113–227 kg ha^{-1} and DOC concentrations in a range of 20–219 mg L^{-1}. The simulated DOC export load and concentrations were highest for the Finnish study sites. DOC concentrations were the highest in the ditch depth scenario of 0.9 m, which was probably due to a very low WT (Fig. 5 a).

Overall, the simulated and observed DOC concentrations were similar. The DOC concentrations in ditch water were measured in all the study sites at approximately in monthly intervals during the snow-free

a Mean annual DOC export kg ha^{-1} ± SD

FIN	Ditch depth m		
Ditch spacing m	0.3	0.6	0.9
40	153±14	195±17	227±19
60	135±16	169±21	198±26
80	120±16	148±22	171±27

b Mean annual DOC export kg ha-1 ± SD

EST	Ditch depth m		
Ditch spacing m	0.3; 0.1	0.6; 0.3	0.9
80	129±21	161±24	198±28
100	121±20	150±22	187±28
120	113±18	140±21	175±26

c Mean annual DOC export kg ha-1 ± SD

SWE	Ditch depth m		
Ditch spacing m	0.3	0.6	0.9
80	150±13	159±15	163±18
100	144±14	151±15	154±18
120	139±14	145±15	148±17

d Mean annual DOC concentration mg L^{-1} ± SD

FIN	Ditch depth m		
Ditch spacing m	0.3	0.6	0.9
40	83±55	132±90	219±164
60	56±35	93±62	155±17
80	48±33	70±49	101±73

e Mean annual DOC concentration mg L^{-1} ± SD

EST	Ditch depth m		
Ditch spacing m	0.3; 0.1	0.6; 0.3	0.9
80	25±7	40±14	79±50
100	22±6	36±13	69±40
120	20±5	31±11	61±32

f Mean annual DOC concentration mg L^{-1} ± SD

SWE	Ditch depth m		
Ditch spacing m	0.3	0.6	0.9
80	28±10	31±12	34±13
100	26±9	27±10	30±11
120	24±9	26±10	27±10

Fig. 4. Simulated mean (± standard deviation) of annual export (a-c) and concentration (d-f) of dissolved organic carbon (DOC) including high and low molecular weight fractions in the study sites of Finland (a, d), Estonia (b, e) and Sweden (c, f) with different ditch spacing and ditch depth.

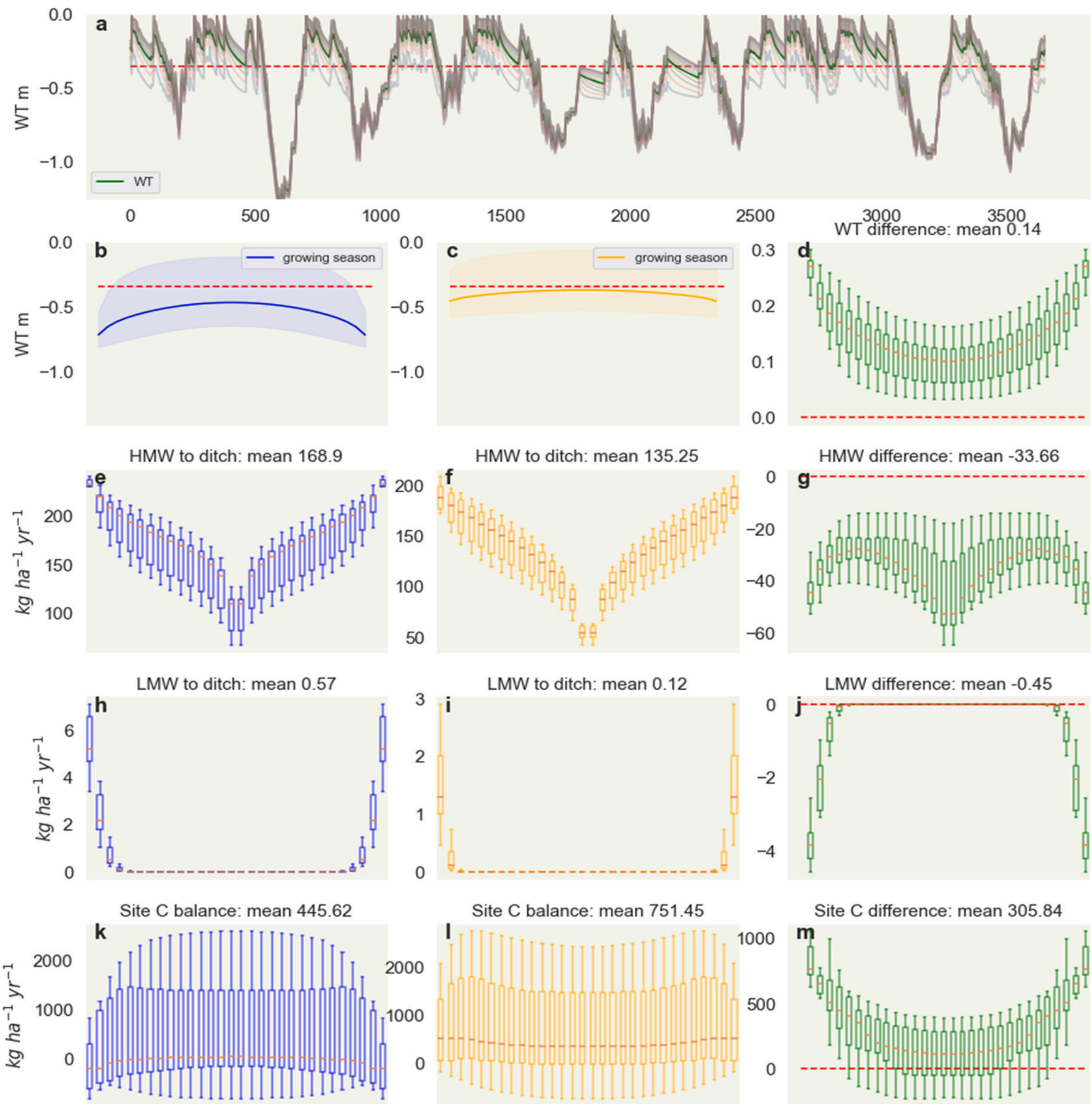

Fig. 5. Paroninkorpi study site in Finland (ditch spacing 60 m). Daily time series of water table (WT) during the 10- year simulation period (a), mean growing season WT between two adjacent ditches (b, c), high-molecular-weight dissolved organic carbon (HMW-DOC) export to ditches (e, f), low-molecular-weight dissolved organic carbon (LMW-DOC) export to ditches (h, i), and site carbon (C) balance (k, l, positive indicates C sink, negative C source). Blue figures (b, e, h, k) represent ditch depth scenario 60 cm, orange (c, f, i, l) represents ditch depth 30 cm and the green (d, g, j, m) figures represent the difference between these scenarios. The horizontal red line in b and c represent the optimal WT. In Figures d, g, j and m the horizontal red lines indicate the zero-difference between the ditch depth scenarios. Horizontal axes in Figures b-m represent the space between two adjacent ditches in 2-m intervals. Box plots show the variation between years and the whiskers show the range, box upper and lower shoulders show the Q1 and Q3 quartiles, respectively, and the red horizontal lines indicate the median.

period. At the Finnish study site, the mean observed DOC concentration was 72 ± 32 mg L^{-1}, which matched simulations of 60 m ditch spacing and ditch depths of 0.3 m - 0.6 m (Fig. 4 d). In Estonia, where the ditch depths in the field were 0.6 m in the downslope ditch and 0.3 m in the upslope ditch, the measured DOC concentration was 63 ± 17 mg L^{-1}, being slightly higher than what was estimated based on SUSI simulations (Fig. 4 e). In Sweden, the mean measured DOC concentration had a large variation, 32 ± 30 mg L^{-1}, but the simulated DOC concentrations were within this range (Fig. 4 f).

In all simulations, WT frequently fell below 0.35 m during the growing season (Figs. 5a, 6a, 7a). In Finland, where the drainage was most intensive and the tree stand consisted of mature Norway spruce with high interception, evaporation and transpiration capacity, the WT fell below 1 m during summers indicating occasional over-drainage (Fig. 5 a). This result was also consistent with the field measurements (Table 1). Areas close to ditches were modeled to be the main sources of DOC (Figs. 5–7 e, f). The area located in the midway between the adjacent ditches produced less DOC. This was partially due to the higher WT (Figs. 5–7 b, c), which slowed down the rate of decomposition and thus the release of DOC. Furthermore, longer residence time required for the transport of DOC to the ditch, in turn, allowed more DOC biodegradation into CO_2. The biodegradation rate of LMW-DOC is high and therefore, the LMW-DOC reaching the ditch is formed within a few meters distance from the ditch (Figs. 5–7 h, i).

The terrain in the Finnish study site was flat, whereas the Estonian and Swedish sites had 0.3 % and 3.0 % slopes, respectively. Consequently, in Finland, the water divide between the ditches located in the middle of the peat strip and WT was symmetrical at both ends of the strip (Fig. 5 b, c). In Estonia, there was a slight slope and the downslope ditch was deeper than the upslope ditch (Fig. 6 b, c). Furthermore, the stand

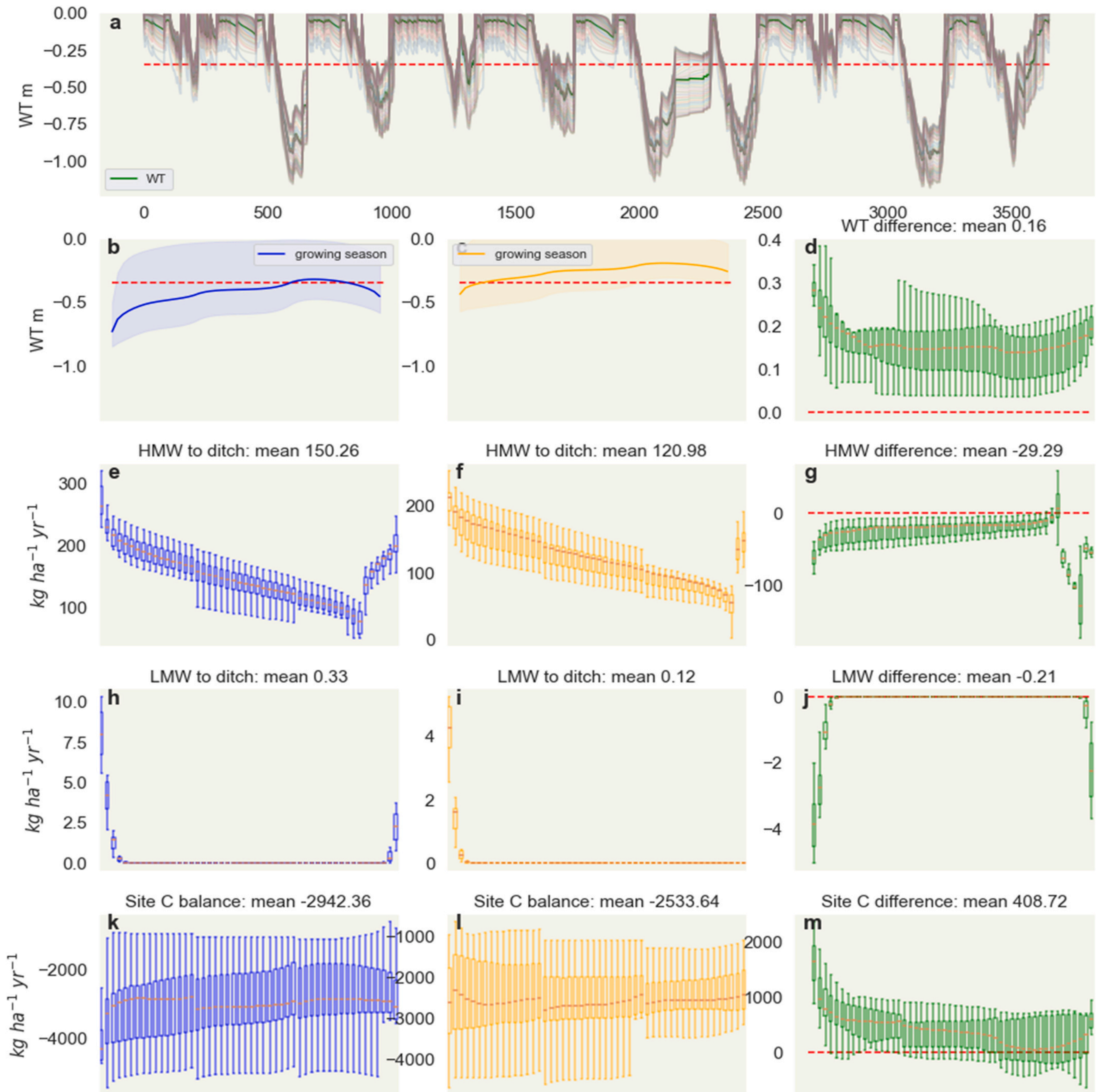

Fig. 6. Ullika study site in Estonia (ditch spacing 100 m). See the panel explanations from Fig. 5. Blue figures (b, e, h, k) represent a ditch depth scenario with 60 cm in the left ditch and 30 cm in the right ditch. Orange (c, f, i, l) figures represent a ditch depth scenario with 30 cm in the left ditch and 10 cm in the right ditch, and the green (d, g, j, m) figures represent the difference between these scenarios.

volume was considerably higher close to the downslope ditch causing lower WTs and consequently water flow from right to left (Fig. 6 b, c). This resulted in an asymmetrical arrangement of export of the HMW-DOC, because the water divide was located rather close to the upslope ditch (Fig. 6 c, f). The resulting long distance from the water divide to the downslope ditch extended the residence time and, thus, increased the biodegradation of DOC to CO_2. In the Swedish study site (Fig. 7 e, f), the 3 % slope moved the water divide very close to the upslope ditch, and consequently, practically all DOC had to be transported across the strip until the downslope ditch. Adjusting drainage intensity caused changes in WT and horizontal fluxes of water and DOC. The differences between the ditch depth scenarios were highest in Finland with flat terrain and denser ditch spacing (Fig. 5 d, g, j). In contrast, the lowering of ditch depth had only a small influence on the WT and DOC fluxes in Sweden, where the slope was higher and the peat layer was thin (Fig. 7 d, g, j). Lowering of ditch depth improved site C balance, especially in intensively drained area in Finland, where the consecutive decrease in lateral C fluxes accounted for 10 % (Fig. 5 g) of the increased C sink (Fig. 5 m).

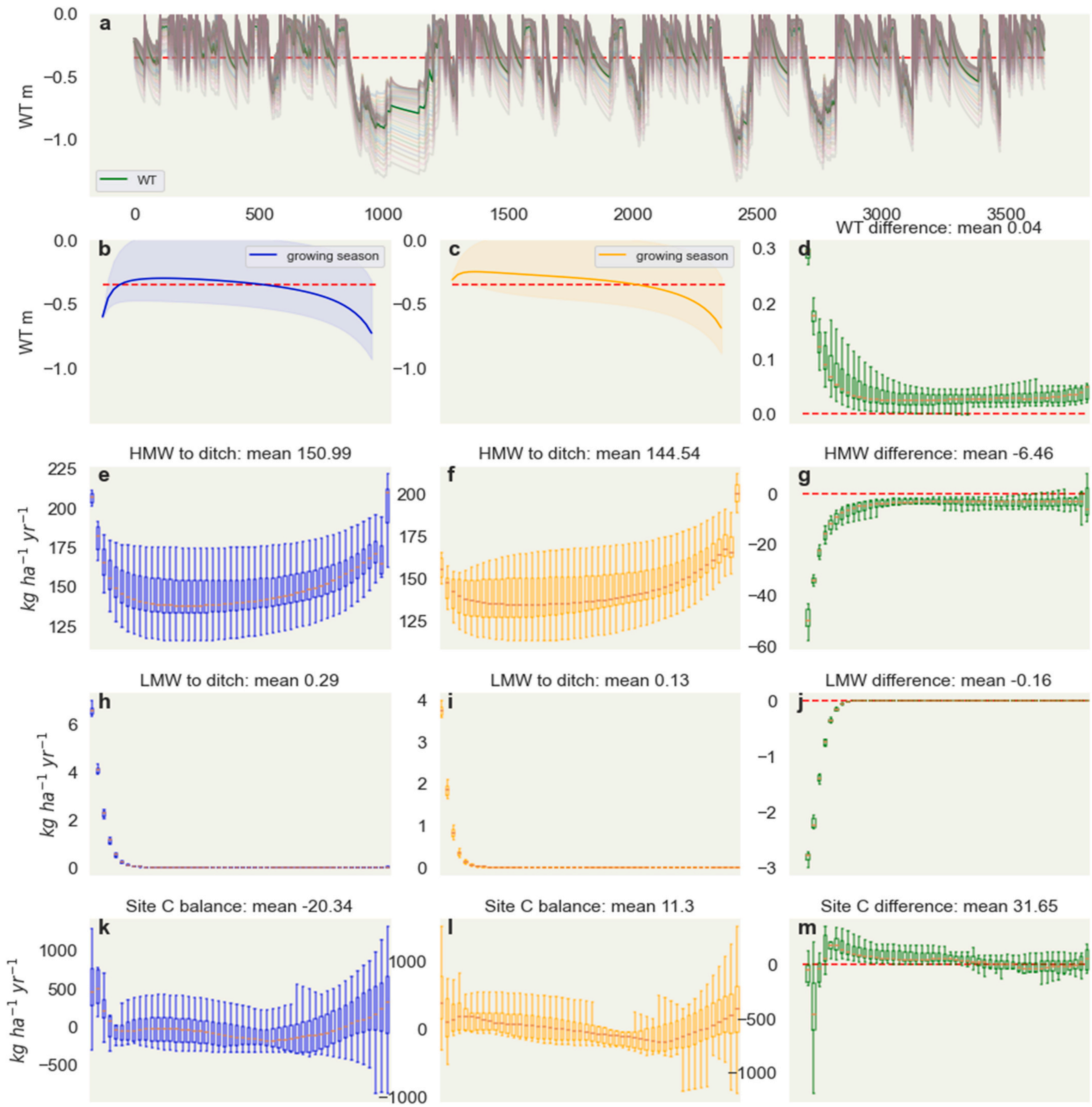

Fig. 7. Krycklan study site in Sweden (ditch spacing 100 m). See the panel explanations from Fig. 5. Blue figures (b, e, h, k) represent ditch depth scenario 60 cm, orange (c, f, i, l) represents ditch depth 30 cm and the green (d, g, j, m) figures represent the difference between these scenarios.

4. Discussion

4.1. Laboratory experiment and ESOM development

Mass-balance based organic matter dynamics models are required for long-term simulations that aim to capture the effects of drainage and forest management. In our study, the development and parameterization of the ESOM were based on literature. Further, we tested ESOM performance against an independent dataset from laboratory experiments. We found that ESOM predicted the mean CO_2 emissions rather well for peat columns extracted from a range of different drained peatland ecosystems (Fig. 3 a-h). The change in DOC concentration was also reasonably predicted both for the Finnish and Estonian peat column samples. However, ESOM overestimated the DOC change in the Irish peat column samples and underestimated the change in the Swedish peat column samples (Fig. 3 m, n, o, p). This suggests that the DOC concentration change was partly due to such factors that were not included in the model. Irish peat samples were thick, well-decomposed with high bulk density and low macroporosity. Because of high WT and the organic matter quality, decomposition reactions take place mainly close to the column surface. The dense structure, low macroporosity and high WT restrict oxygen diffusion, peat decomposition and DOC formation in the deeper layers, which was likely the case for the Irish samples. Furthermore, the DOC released in the active decomposition layer was not necessarily transported to the level where Rhizon samplers were located. Swedish samples, in turn, were loose structured woody peat and the thin samples were exposed to very high WT. In these, almost submerged conditions, especially the vegetated peat samples released a considerable amount of DOC (Fig. 3 o, p). When dismantling the experiment, these samples were smelled like sulfur suggesting the presence of sulfur compounds and reductive conditions. Therefore, alternative terminal electron acceptors may have facilitated organic matter decomposition and DOC formation in anoxic conditions (see e.g. Boothroyd et al., 2021). Especially in peatlands, iron reduction/oxidation cycles can have important role in DOC mobilization (Knorr, 2013). Overall, although the ESOM is relatively simple, it predicts CO_2 emissions and DOC concentrations with rather good accuracy, considering that decomposition is a very complex physical, chemical and biological process.

DOC dynamics in drained forested peatlands depend on several factors such as vegetation characteristics, organic matter quality, soil heterogeneity, the thickness of raw humus layer, soil microbial communities, WT, water transportation routes and the availability of oxygen and other terminal electron acceptors (Marschner and Kalbitz, 2003; Nieminen et al., 2015; Bernard-Jannin et al., 2018; Lappalainen et al., 2018; Nieminen et al., 2018; Zhong et al., 2020; Boothroyd et al., 2021). DOC is also an intermediate product in decomposition which means that two processes, both DOC formation and DOC biodegradation simultaneously affect the observable concentration (Laurén et al., 2019; Payandi-Rolland et al., 2020; Palviainen et al., 2022). The model outcome also depends on the temperature sensitivity of DOC release and biodegradation processes, which so far have not thoroughly been studied experimentally.

Our laboratory experiment indicated that DOC release corresponded to 3–8 % of the gaseous CO_2-C emissions (Fig. 3), which is consistent with the findings of previous incubation studies (Lappalainen et al., 2018; Laurén et al., 2019). Even though DOC release is a small flux compared to direct gaseous CO_2-C emissions, it is not negligible in the C balance of drained forested peatlands (Juutinen et al., 2013; Wallin et al., 2013; Drake et al., 2018). Further, even small increases in DOC concentrations may affect water quality and cause significant water brownification in receiving watersheds (Solomon et al., 2015; Kritzberg et al., 2020; Härkönen et al., 2023). This highlights that lateral C fluxes should be considered in planning of water and forest management in drained peatlands.

4.2. Effect of drainage intensity on DOC export and site C balance

Our simulation results suggested that higher drainage intensity, i.e. use of deeper ditches or denser ditch spacing, increases the concentrations and export loads of DOC from drained forested peatlands (Figs. 5–7). In SUSI simulations, lower WT results in increased DOC concentration and export because a thicker aerobic peat layer decomposes and releases more DOC. Our result is consistent with the previous findings that organic C concentrations (Marttila et al., 2018; Nieminen et al., 2021) and export loads (Finér et al., 2021) are higher in drained than in undrained boreal peatlands. On the other hand, it has been found in previous studies that rising WT after forest harvesting or peatland restoration has increased DOC export from boreal peatlands (Nieminen et al., 2015; Kaila et al., 2016; Härkönen et al., 2023). DOC concentrations have been found to decrease when ditch depth has been increased in ditch network maintenance (Nieminen et al., 2018), whereas ditch blocking has lowered DOC concentrations in boreal and temperate peatlands (Holl et al., 2009; Turner et al., 2013; Menberu et al., 2017) DOC export is a complex interplay between the decomposing material, DOC formation and transport, and therefore WT or its fluctuation alone are not strong predictors of DOC concentration nor export (Evans et al., 2018; Zhong et al., 2020; Härkönen et al., 2023). Many seemingly contradictory observations in literature may be due to the time discrepancy between the DOC formation, transport and biodegradation. Fresh litter, raw humus layer and top peat are easily decomposable materials and release more DOC than the deeper peat layers (Laurén et al., 2012; Lappalainen et al., 2018; Laurén et al., 2019). The decomposition rate is also higher when WT is low (Ojanen et al., 2010; Ojanen and Minkkinen, 2019), but at the same time the transport is smaller due to small hydraulic gradients. During high WT the released DOC is captured by the moving water and because the surface layers also have high hydraulic conductivity (Koivusalo et al., 2008), DOC is rapidly transported to ditches and less DOC biodegradation happens during the transportation. The intensity of DOC release and transport are also altered over time, which may mask the causality between the DOC processes and forest and water management.

Our SUSI Peatland Simulator applies a simplified version of DOC transport, where the cumulative annual DOC release in each computation node is moved to the receiving ditch at the end of each computation year. Biodegradation is subtracted from the released DOC according to the residence time estimated from the mean annual hydraulic gradient between the computation nodes and summed along the water flow path to the ditch. Consequently, the initial DOC storage in soil water, which is known only for intensively monitored study sites, is not needed as an input for the model. Therefore, this simplified version markedly improves the usability of the model for practical forestry.

In our simulations, annual DOC export in the drainage scenarios varied from 113 kg ha^{-1} to 227 kg ha^{-1} (Fig. 4). For comparison, the annual DOC export from peatland-dominated catchments (the proportion of peatlands 50–75 %) has been reported to range from 80 kg ha^{-1} to 227 kg ha^{-1} in the boreal conditions (Kortelainen et al., 2006; Aaltonen et al., 2021; Finér et al., 2021). Furthermore, our simulated DOC export values are the same order of magnitude than the mean annual DOC export values reported for boreal (137 kg ha^{-1}) and temperate (242 kg ha^{-1}) peatlands in the review study by Rosset et al. (2022).

The SUSI Peatland Simulator with our upgraded Extendable Soil Organic Matter decomposition model (ESOM) markedly improves C balance calculations because it accounts for several processes that are critical in drained peatlands. In drained peatlands, the WT depends on ditch depth, ditch spacing, slope, peat properties, stand characteristics and weather conditions (Koivusalo et al., 2008; Sarkkola et al., 2010; Päivänen and Hånell, 2012; Waddington et al., 2015; Leppä et al., 2020). SUSI also accounts for the most important spatial dimension, i.e. the space between the adjacent ditches. This allows differentiation of litter input, prevailing WT, CO_2 emissions, DOC formation, biodegradation and transport along the strip between the ditches. For the first time, HMW and LMW-DOC fluxes are explicitly a part of complete site C balance. Our simulations indicated that annual DOC exports are rather small compared to C that is sequestered by forest stand and emitted from soil. However, in some drained peatlands, net C balance can be close to zero and, in these cases, lateral C fluxes can determine whether the site is a C source or sink (Figs. 5–7). We found that slope and drainage were closely connected to lateral C fluxes. Even a small slope together with different ditch depths changed the location of the water divide between the ditches (Figs. 6, 7) and altered the residence time and biodegradation of DOC. In these cases, changing the drainage dimensions only had a small effect on the DOC export. Slope angle is known to influence the depth of WT, the route of runoff, and the water residence time and further to DOC production and DOC quality (Evans et al., 2005; Holden, 2005; Kothawala et al., 2014; Williamson et al., 2023).

The improved SUSI Peatland Simulator now offers a novel decision support tool for planning and implementing smart water and forest management strategies in drained forested peatlands for climate change mitigation and adaptation. According to Sarkkola et al. (2010) lowering of July–August WT below 35 cm level does not improve the stand growth but increases adverse environmental effects of peatland forestry. The simulations and field observations revealed that the summer-time WT in the Finnish study site was frequently below 80 cm or even below 1 m indicating a clear over-drainage (Table 1, Fig. 5). Both mitigation of climate change and adaptation to climate change require modification of currently applied water management schemes. Decreasing drainage intensity by using either shallower ditches and/or by using wider ditch spacing (blocking some of the ditches) may be required under changing climate in areas, where increasing droughts may decline forest growth and where the drought-induced high soil C emissions have to be tackled (Gong et al., 2012; Venäläinen et al., 2020). The extent to which the drainage intensity needs to be decreased depends also on the amount and seasonal distribution of rainfall under changing climate. Annual ditch network maintenance areas have decreased dramatically in Finland during the past ten years (Vaahtera et al., 2023) which results in gradual filling in of ditches (Hökkä et al., 2020). After 20 years, the ditches have been found to be 20–30 cm shallower than immediately after the digging (Hökkä et al., 2020). This range of reduction in ditch depth does not likely interfere with stand growth (Hökkä et al., 2021) but according to our study, it likely results in improved C balance and reduced DOC export.

5. Conclusions

Our ESOM mass-balance based decomposition model predicted the average CO_2 emissions and changes in DOC concentrations with reasonable accuracy for peat columns extracted from a range of different drained peatland ecosystems. After incorporating the ESOM into the SUSI Peatland Simulator, we were able to compile a full site C balance that includes lateral C fluxes divided in HMW- and LMW-DOC fractions. Annual DOC exports were rather small (3–8 %) compared to soil CO_2 emissions, but in some areas, DOC export can determine whether the site is a C source or sink. Our simulation results supported the hypothesis that DOC export can be decreased and site C sink can be increased by reducing drainage intensity, through reducing the depth to which ditches are cleaned or increasing the ditch spacing. However, slope can also markedly alter the water residence time and consequently DOC biodegradation and export to ditches.

The complexity of the management challenge in drained forested peatlands calls for profound planning of water management to mitigate the adverse environmental impacts of forest management. We recommend using process-based ecosystem models considering site, stand and terrain characteristics, as well as interactions and feedbacks between ecosystem processes to support planning and implementation of climate smart and responsible management of peatland forest resources.

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CRediT authorship contribution statement

Marjo Palviainen: Writing – review & editing, Writing – original draft, Visualization, Validation, Supervision, Software, Resources, Project administration, Methodology, Investigation, Funding acquisition, Formal analysis, Data curation, Conceptualization. **Jukka Pumpanen:** Writing – review & editing, Supervision, Resources, Project administration, Funding acquisition, Data curation. **Virginia Mosquera:** Writing – review & editing, Data curation. **Eliza Maher Hasselquist:** Writing – review & editing. **Hjalmar Laudon:** Writing – review & editing, Supervision, Resources, Funding acquisition. **Ivika Ostonen:** Writing – review & editing, Supervision, Resources, Project administration, Data curation. **Ain Kull:** Writing – review & editing, Data curation. **Florence Renou Wilson:** Writing – review & editing, Supervision, Resources, Project administration, Funding acquisition, Data curation. **Elina Peltomaa:** Writing – review & editing, Data curation. **Mari Könönen:** Writing – review & editing. **Samuli Launiainen:** Writing – review & editing. **Heli Peltola:** Writing – review & editing. **Anne Ojala:** Writing – review & editing, Funding acquisition. **Annamari Laurén:** Writing – review & editing, Writing – original draft, Visualization, Validation, Supervision, Software, Resources, Project administration, Methodology, Investigation, Funding acquisition, Formal analysis, Data curation, Conceptualization.

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.

Data availability

Data will be made available on request.

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