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RESEARCH ARTICLE

Impact of drainage on soil CH_4 and N_2O fluxes in nutrient-rich hemiboreal peatland forests of Europe

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Abstract
Background and aims Peatland forests play a significant, yet highly variable, role in atmospheric methane (CH_4) and nitrous oxide (N_2O) emissions. Drainage reduces CH_4 but increases N_2O fluxes from nutrient-rich soils, creating complex climate trade-offs. To refine regional climate strategies, a clearer understanding of soil fluxes and influencing factors across forest types in the hemiboreal zone is needed.
Methods This study presents two years of soil flux and environmental data collected from 2020 to 2023 using manual chambers at 18 drained (DF) and 7 undrained (UF) sites, with different tree species, in Estonia, Latvia, and Lithuania.
Results Soil water-table depth was the primary control on annual CH_4 exchange; DF sites acted as CH_4 sinks (–4.6 kg ha^{-1} y^{-1}), whereas UF sites were net sources (38.2 kg ha^{-1} y^{-1}). In contrast, N_2O emissions occurred at all sites, averaging 5.9 kg ha^{-1} y^{-1} in DF and 1.4 kg ha^{-1} y^{-1} in UF soils. Within DF sites, Myrtillus-type pine stands exhibited the lowest N_2O fluxes, while alder stands showed the highest, followed by birch- and spruce-dominated stands, reflecting differences in hydrology, pH, and C/N ratios of the soil.
Conclusions Our findings reveal significantly lower CH_4 and moderately higher N_2O emissions from DF soils than the IPCC Tier 1 temperate zone emission factor (EF), which has thus far been used for the hemiboreal zone. This study can improve national GHG inventories using specific information. We recommend applying Baltic region-specific EFs, as the mean annual CH_4 and N_2O emissions showed no significant difference across the three countries.

Keywords Organic soil · Fen forest · GHG emission · Soil water content · Nitrogen · Temperate

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Introduction

Peatlands covering 4.23 million km^2 are predominantly located in the Northern hemisphere and tropical regions, holding about one-third (~550 gigatons [Gt]) of global soil carbon (FAO 2020; UNEP 2022; Xu et al. 2018). Over the long term, natural peatlands have played a significant role in global terrestrial greenhouse gas (GHG) budgets by sequestering atmospheric carbon dioxide (CO_2) and emitting methane (CH_4) under typical wet conditions. (FAO 2020; Turetsky et al. 2014). Additionally, peat soils can act as weak sinks or sources of nitrous oxide (N_2O) in natural or pristine conditions (Laine et al. 2019; Minkkinen et al. 2020; Mustamo et al. 2016). CH_4 and N_2O have global warming potentials 27 and 273 times higher than CO_2 over 100 years, respectively, according to the Intergovernmental Panel on Climate Change (IPCC). Their release into the atmosphere has a significant warming effect by increasing radiative forcing (IPCC 2023). Therefore, it is essential to examine their emissions from both natural and disturbed peatland soils for sustainable land management and to understand their response to regional climatic and environmental conditions, such as changes in soil water-table level (WTL) (Escobar et al. 2022; Laine et al. 2019; McDaniel et al. 2019).

Artificial drainage through ditches or pipe systems is a common management practice in peatlands to facilitate agriculture, forestry and peat extraction (Laine et al. 2009; Päivänen and Hånell 2012). This drainage lowers the WTL, accelerating soil organic matter decomposition and affecting GHG emissions (Eickenscheidt et al. 2014; Ojanen et al. 2013; Tiemeyer et al. 2016). Soil CH_4 emissions depend on the balance between production by methanogens in anoxic zones and oxidation by methanotrophs in the oxic zone, where CH_4 is converted to CO_2 (Bridgham et al. 2013; Lai 2009; Le Mer and Roger 2001). Lowering the WTL increases CH_4 consumption, allowing peat soil to function as a CH_4 sink (Abdalla et al. 2016; Evans et al. 2021; Yrjälä et al. 2011). On the other hand, a lower WTL can enhance N_2O emissions by increasing the availability of mineralized nitrogen (N) for nitrification and denitrification processes (Butterbach-Bahl et al. 2013; Leifeld 2018). Drainage has been shown to double N_2O emissions in nutrient-rich organic forest soils in the boreal zone compared to pristine soils (Blais et al. 2005; Minkkinen et al. 2020; Pearson et al. 2015). However, significant gaps still exist in the global peatland GHG data, and in understanding the environmental characteristics that control peatland GHG emissions (Mander et al. 2024; Zhao et al. 2023).

In the Nordic and Baltic countries, drained peatlands are widespread (IPCC 2021; Tubiello et al. 2016), with forestry being the most common use (FAO 2020; Paavilainen and Päivänen 1995). In Baltic countries, drained peatland forests cover approximately 0.3 to 0.4 million hectares in each state, with a total organic forest soil area of 0.63 Mha in Estonia, 0.60 Mha in Latvia, and 0.43 Mha in Lithuania, as reported in National Inventory Submissions to the UNFCCC (Barthelmes et al. 2015; Paal and Leibak 2011; UNFCCC 2023; Valatka et al. 2018). Drained organic forest soil is generally identified as a source of GHG emissions, particularly in the temperate zone (IPCC 2014; Jauhiainen et al. 2023), while little information is available for undrained organic forest soils (Butlers et al. 2023; Minkkinen et al. 2020).

The default IPCC emission factors (EFs) (Tier 1 EFs; Good Practice Guidance 2006, Wetland supplement 2014) for drained organic forest soils are average values collated from peer-reviewed studies, stratified by broad climatic zones (boreal and temperate) and soil nutrient status classes (nutrient-rich and nutrient-poor) (Aitova et al. 2023; Jauhiainen et al. 2023). Baltic countries are expected to use Tier 1 EFs estimated for temperate regions, while data facilitating higher Tiers are lacking (Jauhiainen et al. 2023, 2019). The IPCC EFs are limited by data availability and thus exhibit wide uncertainty ranges. Moreover, they involve an imbalanced geographical distribution of measurement sites, with no data from the Baltic region, which has distinct geology, climate, and ecological features. The Baltic countries, located in the hemiboreal region, are characterized by a mix of boreal coniferous and temperate deciduous forests. The forest vegetation in this region shares many similarities with the boreal zone, but experiences slightly warmer climatic conditions associated with the temperate zone (Ahti et al. 1968). European countries are committed to using Tier 2 and advancing towards Tier 3 methods for reporting national GHG emissions/removals of large carbon (C) pools, including organic soils, under the UNFCC by 2027 and 2030 (UNFCCC 2023). Given the large share of organic soils in the Baltic states, improving the accuracy and reducing the uncertainty of organic forest soil GHG emission estimates is crucial (IPCC 2021). Additionally, enhancing knowledge of GHG emissions from undrained organic forest soils is important for a better understanding of the share of anthropogenic GHG emissions in the total emissions from drained soils, as well as for evaluating their responses to changing environmental conditions.

Soil nutrient status and water content are key edaphic factors regulating microbial processes that control CH_4 and N_2O fluxes in organic soils (Duan et al. 2022; Lin et al. 2022; Turetsky et al. 2014). Peatland site-types, differ in nutrient availability, hydrology, and redox conditions, which regulate anaerobic and aerobic production pathways and thus make the site-types relatively stable proxies for soil GHG emissions (Jauhiainen et al. 2023; Ojanen et al. 2013; Westman and Laiho 2003). Tree-stand characteristics, such as species composition and vegetation structure, primarily affect biomass C dynamics and plant–microbial interactions (Mundra et al. 2022; Truu et al. 2020). Dominant tree species strongly influence litter chemical and physical composition, litter inputs and root exudation, altering the substrate quality and availability for microbial GHG production (Cornwell et al. 2008; von Arnold et al. 2005a, 2005b). Although site-types and tree species as explanatory factors interact, they differ mechanistically and temporally, and treating site-type and tree-specie as distinct drivers enhances the precision of emission factors, GHG inventories, and mitigation strategies aimed at emission reductions in boreal peatlands (Alm et al. 2023; Couwenberg et al. 2011; IPCC 2006). However, most of the information on site type or tree-specie level variation in fluxes is reported from the boreal zone (Jauhiainen et al. 2023). Therefore, this study fills knowledge gaps concerning the Baltic region by evaluating CH_4 and N_2O fluxes across nutrient-rich peatland forests, considering site type, dominant tree species, soil drainage regime (drained vs undrained), and soil physiochemical properties, WTL, soil temperature, moisture and water chemistry. We carried out a two-year measurement campaign at multiple drained and undrained hemiboreal forest sites in Estonia, Latvia, and Lithuania. The specific objectives were: (1) to study the seasonal dynamics and apparent drivers of the hourly CH_4 and N_2O fluxes in drained and undrained organic forest soils, (2) to determine the annual soil CH_4 and N_2O exchange rates and (3) their relations to factors directly or indirectly influencing soil processes, such as, dominant tree species, site type, soil and soil water properties, and WTL, (4) to estimate EFs for the drained nutrient-rich organic forest soils in the hemiboreal region. Nutrient-rich sites were targeted since they generally show higher emissions than nutrient-poor sites (Jauhiainen et al. 2023; Mander et al. 2024).

Methods

Study sites

The study was conducted at 25 peatland forest sites in the Baltic region, seven undrained (UF) sites and 18 drained (DF) sites. Five sites were in southern Estonia, nine in northern and central Latvia, and five in southern Lithuania (Fig. 1A). The sites were chosen to represent the peatland forests typical of each country under local conditions, and their characteristics are described in Tables S1 and S2. All sites were located in the temperate cold and moist climate region according to the IPCC major climatic zones (IPCC 2006), and more specifically in the hemiboreal vegetation zone (Ahti et al. 1968). Weather data specific to each country were obtained from the nearest meteorological stations to the sites: Tõravere station (58°16’ N, 26°28’ E) in Estonia, Zilani station (56°31’ N, 25°55’ E) in Latvia, and Vilnius station (54°40’ N, 25°06’ E) in Lithuania, respectively. The 30-year (1991–2020) average air temperatures (T_{air}) were 6.3 °C, 6.8 °C and 7.3°C in Estonia, Latvia and Lithuania, and the annual precipitation sums were 683 mm, 686 mm and 695 mm, respectively.

Weather conditions during the measurement period (2021–2022 or 2022–2023) differed slightly from the long-term averages and varied between the years. The average T_{air} in 2022 was higher than in 2021 across nearly all countries (Table S2). Annual T_{air} (°C) was above the long-term average in Estonia, Latvia and Lithuania, respectively 6.6, 7.1 and 8.4 °C. The highest T_{air} was observed in Lithuania, the southernmost of the Baltic countries. Annual precipitation (mm) was lower in Estonia (534 mm) than in Latvia (650 mm) and Lithuania (664 mm). Generally, the highest precipitation occurred during the growing season, with maximums observed from June to

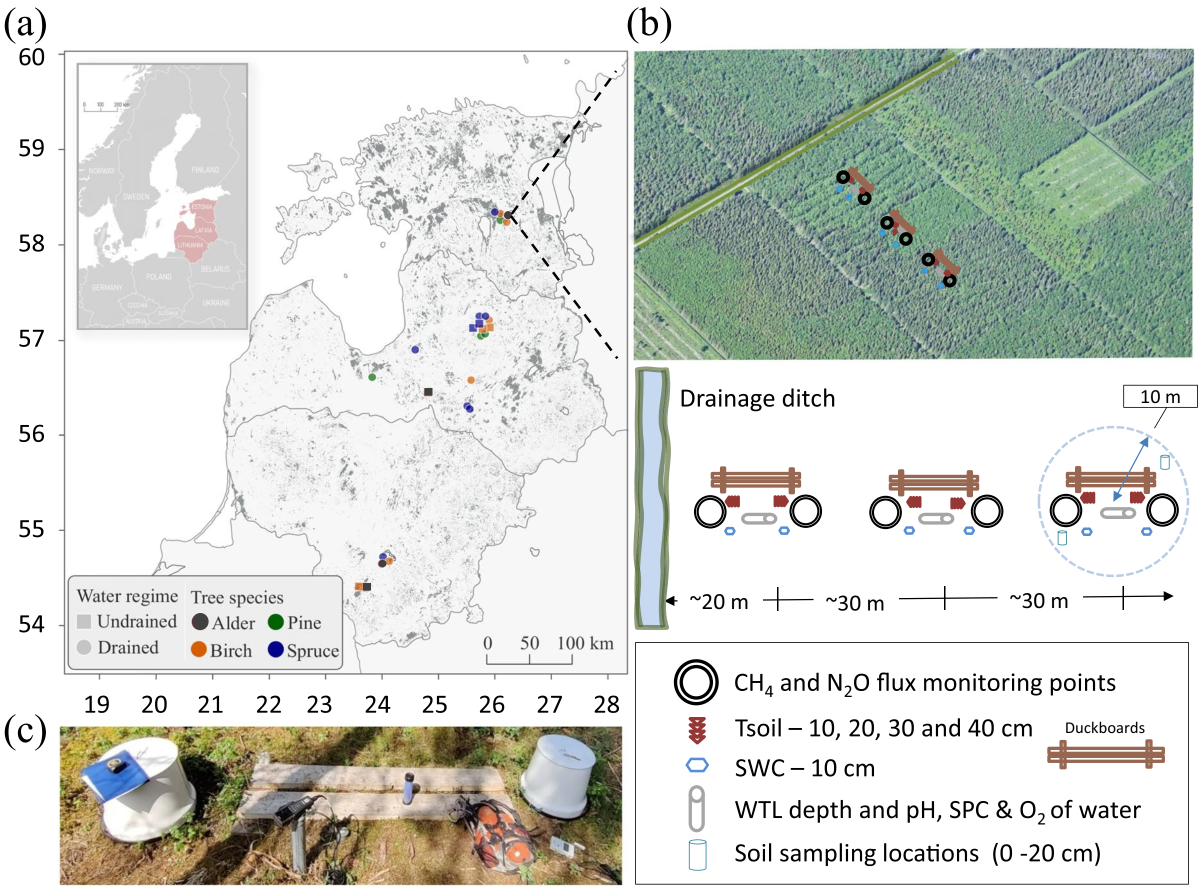

Fig. 1 Study area and sites: (**a**) Map of the Baltic states showing peatlands with organic soil as grey shading, and the locations of the study sites in Estonia, Latvia and Lithuania. Site symbols represent the drainage, and colour indicates the dominant tree species. (**b**) Aerial photograph showing the layout of sampling plots located on a transect perpendicular to a drainage ditch for an example stand in Estonia (Site EE-S, Table S3); and (**c**) Schematic presentation and photograph showing the details of the experimental design of a site with a pair of gas monitoring points as well as setup and locations of soil variable measurements

August across all countries. In this study, the growing season is defined according to local conditions as extending from May, when vegetation becomes established and actively growing, to the end of October, when dormancy or unfavourable environmental conditions end plant growth (Tiner 2009). The non-growing season thus spans from November to April. The average monthly T_{air} was below 5 °C during non-growing (Nov-Apr) and reached its maximum of 14 to 15 °C in July and August.

The study sites had organic soils (IPCC 2006), ranging from shallow (27 cm) with > 12% organic carbon (C) in the top 20 cm to deep peat deposits over two meters thick (Table S4). All sites represented peatland forests resulting from natural succession, i.e., the UF sites represented inherently forested peatland site-types, and the DF sites had been drained naturally regenerated forested. The DF sites had been shaped to varying degrees by secondary succession following ditching, as well as forest management, as reflected in their ground vegetation and tree stand composition. The DF sites were distinguished by ditches within or along the borders of 50 ± 25 ha forest compartments, while the UF sites had no ditches. Although the exact timing of drainage for each DF site is unknown, it was generally initiated in the mid-nineteenth century and resumed in the mid-twentieth century. Ditches were 0.8–1.2 m deep, spaced 100–400 m apart, and maintained to remain functional.

Undrained and drained forests were classified based on dominant tree species and ground vegetation types. The tree stand had one of the four typical native tree species as dominant: black alder (*Alnus glutinosa* (L.) Gartner.), and downy birch (*Betula pubescens* Ehrh.) or silver birch (*Betula pendula* Roth.), Norway spruce (*Picea abies* (L.) H. Karsten.), or Scots pine (*Pinus sylvestris*, (L.) (Table S1). In certain birch sites, silver birch (*Betula pendula* Roth.) was the dominant species, and both birch species were pooled as there was no evidence to expect different impacts between them. The sites were categorised into different site-types according to regionally shared classifications based on ground vegetation and tree stand productivity, which are known to reflect soil nutrient status (Laine 1989; Lõhmus 1981). We harmonised the site-type names between countries to follow the classification described by (Lõhmus 2004, 1981). Drained sites were represented by Oxalis, Alnus and Myrtillus site-types, and the undrained sites were grouped into Dryopteris site type (Table S1, 3). These classes all represent nutrient-rich soils (IPCC 2014). In the Oxalis site type, which has relatively the highest nutrient status and species richness, the dominant ground vegetation species are tall ferns (*Athyrium filix-femina*) and herbs (*Oxalis acetosella*). The Alnus site type features tall shrubs (*Frangula alnus, Ribes nigrum*). The Myrtillus site type, which represents the most moderate nutrient availability in the nutrient-rich category, is characterised by dwarf shrubs (*Vaccinium* spp). The Dryopteris site type includes a variety of herbaceous species (e.g., *Dryopteris* spp., *Ribes* spp., and *Carex* spp.) and shrub species (e.g., *Prunus padus, Salix, and Lonicera* spp.). More detailed site type classifications, soil characteristics, tree species, and ground vegetation descriptions are provided in the supplementary (Table S1).

We aimed to characterise general patterns within drained and undrained sites, rather than focusing on pairwise comparisons between individual UF and DF sites. Such comparisons are not warranted due to natural variation in site characteristics among individual sites within specific site type groups (e.g., (Laiho and Pearson 2016). The mean characteristics of site type groups, in contrast, typically differ in ways that make group-level comparisons meaningful.

Data collection setup

The field studies followed a consistent setup with harmonised protocols and equipment. Each site included three sampling plots spaced at least 30 m apart along a transect perpendicular to the drainage ditch in DF sites or to the stand border in UF sites (Fig. 1B). The aim was to cover most of the forest area topography and to include the possible WTL gradient of the site. The flux monitoring points were established in late autumn 2020 by inserting rounded collars ($h = 20$ cm, $\varnothing = 50$ cm) at each study site, a total of six per site, except at some sites, where five collars were installed. These collars, each with a groove on the top vertical edge, were inserted to a depth of 10 cm with minimal disturbance to the surface soil and ground vegetation. Duckboards were set close to the collars to limit soil disturbance and to provide stability during sampling. The field campaigns spanned over two full years at most sites, from January 2021 to December 2022 in Estonia and Latvia, and from November 2021 to October 2023 in Lithuania.

GHG sampling and analysis

Soil gas exchange and auxiliary measurements were done once every two weeks in Estonia and monthly in Latvia and Lithuania to cover year-round fluxes. The manual closed chamber technique was used for collecting data on GHG flux exchange between the soil surface (including ground vegetation) and the atmosphere (Soosaar et al. 2011). During gas sampling, the collar was covered with opaque chamber ($V = 65$ L, $\varnothing = 50$ cm, $h = 40$ cm) and the collar-groove was filled with water to seal. Four gas samples were collected in pre-evacuated (0.3 mbar) glass vials (50 ml) with a syringe connected to a tube and valve. Chamber closure time was either 60 min (Estonia and Lithuania) or 30 min (Latvia), in which subsequent gas samples were collected after 20- or 10-min intervals, respectively (Butlers et al. 2025). All gas samples were then transported to the Geography Department laboratory at the University of Tartu for analysis with a gas chromatograph (Shimadzu GC-2014, Shimadzu Corporation, Kyōto, Japan). Equipped with a flame ionisation detector (FID) for measuring CH_4 concentration and with an electron capture detector (ECD) for N_2O concentration.

Flux quality control and calculation of annual emissions

The gas concentration values were first validated by fitting a linear regression on CO_2 emission data (Butlers et al. 2025), measured from the same gas samples to assess potential leakage from the chamber or issues with the vials during sampling. A regression coefficient (R^2) greater than 0.90 was set as the criterion for including the CH_4 and N_2O fluxes (note that CO_2 flux data are not reported in this paper). Fluxes with $R^2 < 0.90$ were still accepted if the change in gas concentration was < 20% of the initial concentration. Low fluxes typically yield lower R^2-values, which should not disqualify a measurement (Alm et al. 2007b; Ernfors et al. 2011). Therefore, a nonsignificant amount of data, less than 4% and 2%, was discarded from individual flux observations of CH_4 and N_2O, respectively.

An ideal gas law function was applied to convert gas concentration into masses (Eq. 1). The rate of hourly CH_4 and N_2O flux (μg m^{-2} h^{-1}) was calculated from the fitted slope of the linear function, i.e., change of trace gas concentration (four gas samples) over time since the chamber was closed.

$$F\left[\mu g m^{-2} \mathrm{h}^{-1}\right] = \frac{\mathrm{M} \times \mathrm{P} \times \mathrm{V} \times \sigma\mathrm{v}}{R \times Tair \times A}, \tag{1}$$

where F is the rate of hourly flux for CH_4 or N_2O, M = molecular mass of gas (g mol^{-1}) (16 for CH_4 and 44 for N_2O), P = assumption of air pressure inside the chamber (101.300 Pa), V = chamber volume (0.0655 m^3), σv = the slope of regression, describing the gas concentration change over time in the chamber (ppm per hour), R = gas constant (8.314 m^3 Pa K^{-1} mol^{-1}), T_{air} = air temperature (K), A = the soil surface area covered by the chamber (0.196 m^2).

The arithmetic means of the 5–6 flux monitoring points (collars) on individual sampling events were calculated as hourly mean flux values per site (μg m^{-2} h^{-1}). Then, it was multiplied by 24 h and the number of days between sampling events using trapezoidal interpolation to cover all months of the calendar year. Annual CH_4 or N_2O (kg ha^{-1} y^{-1}) fluxes for each measurement year were calculated separately as a cumulative flux value per site. The proportions of growing season (May-Oct) and non-growing season (Nov-Apr) fluxes at each site were estimated from cumulative annual fluxes. Gaseous emissions from the soil to the atmosphere are positive, and removals or uptakes from the atmosphere are negative.

Soil environmental analysis

Several environmental drivers of GHG fluxes, including soil volumetric water content (SWC at 0–7.5 cm depth, m^3 m^{-3}), soil temperature at 10, 20, 30 and 40 cm depths (T_{soil_10}, T_{soil_20}, T_{soil_30} and T_{soil_40}, °C), and water table level (WTL, cm), were monitored during the gas sampling events. These parameters were measured near each plot i.e., 3 measurements per site (Fig. 1C). Topsoil SWC was measured with a TEROS 11 sensor connected to a Procheck meter (Meter Group, Inc., Pullman, WA, USA), and the T_{soil} profile was measured with Pt100 probes with the data logger U0141 (COMET SYSTEM, Rožnov pod Radhoštěm, Czech Republic). To measure distance to the WTL, three observation wells (perforated PVC tube, internal diameter of 7.5 cm, extending down to 150 cm) were installed close to gas monitoring points at each site. The observation well was covered with nylon mesh and inserted to the maximum depth of the organic layer. Dry wells could occur during dry periods, especially in sites with deeper WTL conditions that are ecologically meaningful. Therefore, plausible substitute values were examined from nearby wells to avoid bias in calculating daily or annual average WTL (see Supplementary). In the end, they were not used in statistical models, and a value of −145 cm was identified as the true value for maximum WTL depth, based on observations. Concurrently with the gas sampling, soil water properties were measured from the observation wells (Fig. 1C) using a multiparameter analyser YSI ProDSS (YSI/Xylem Inc., Yellow Springs, OH, USA). The measured parameters were water temperature (T_{water}, °C), pH, dissolved oxygen content (O_2, %), and temperature-compensated specific conductivity (SPC, μS cm^{-1}) when water was present in the wells. Soil solution-specific conductance (equal to electrical conductivity normalised at 25 °C) measures soil water's dissolved solids (organic and inorganic) in the observation wells (Bansal et al. 2023). The YSI sensors were calibrated regularly after every 2 to 3 sampling events. Further, the measurement-day mean values for soil and soil–water parameters were calculated from all plots within the site, expressed as daily means. Site annual means were obtained by averaging all sampling events.

Soil sampling and chemical analysis

Soil samples (0–20 cm depth) were taken once (Aug-Sep 2022) with a peat auger near the gas monitoring points for the determination of soil nutrient and ash concentrations (Fig. 1C). The samples were pooled into one composite sample per plot. A separate set of soil samples for determining bulk density (BD, kg m^{-3}) was collected at the same depths (0–20 cm) using a soil sampling ring kit (model A53; Royal Eijkelkamp, Giesbeek, the Netherlands). Soil samples from all sites were transported in temperature-controlled boxes to the Latvian State Forest Research Institute (Silava) laboratory and stored at 4 °C before physico-chemical analysis according to LVS ISO 11464:2005 standards. The BD soil samples were weighed after oven drying at 105 °C (LVS EN ISO 17828:2016). Dry-mass-based total C and N (g kg^{-1}) concentrations were analysed using the Elementar El Cube elemental analyser (Dry combustion; LVS ISO 10694:2006, LVS ISO 13878:1998). The concentrations of nitric acid (HNO_3) extractable potassium (K), calcium (Ca), magnesium (Mg) and phosphorus (P) (mg kg^{-1}) were determined with an inductively coupled plasma-optical emission spectrometer Thermo Fisher Scientific iCAP 7200 Duo (LVS EN ISO 11885:2009) after mineralisation of the sample in concentrated HNO_3 by digestion in a microwave oven CEM Mars 6 iWave. Ash concentration was determined according to LVS EN ISO 18122:2022, and pH was measured in a 1:5 soil solution of 1 M potassium chloride (KCl) according to the LVS ISO 10390:2021 method. Total soil organic carbon (SOC) was calculated as the ash-free organic matter (100% − Ash (%)) multiplied by 0.5, assuming that 50% organic matter is SOC (Pribyl 2010). Furthermore, the C/N ratio was calculated by dividing soil organic C by the total N concentration.

Statistical analysis

Statistical analysis was conducted using R software v.4.3.1 (R Core Team 2023), and figures were drawn using packages ggplot2 and tidyplot (Wickham 2016). Shapiro–Wilk normality tests were applied to check the normality of the fluxes and other parameters. Correlations between soil physical (Tsoil, SWC and WTL) and soil water (SPC, pH, and O_2) characteristics, as well as simultaneously measured soil hourly mean CH_4 and N_2O fluxes per site, were first assessed with Spearman's rank correlation analysis. Linear mixed-effect (LME) models were then developed using *lme4* (Bates et al. 2015) to evaluate the effects of Tsoil, WTL and drainage regime (DF versus UF) on hourly mean CH_4 and N_2O fluxes. All interactive effects of the prior fixed effects were used initially, and only significant ones were kept in the best-fitted model. Random effects include site, year and measurement day to account for repeated measurements, using Restricted Maximum Likelihood (REML). Both CH_4 and N_2O flux data were normalised by log10 function, and integer numbers were added for conversion of negative values; $CH_4 = \log10(CH_4 + 200)$ and $N_2O = \log10(N_2O + 5)$, before transformation. WTL values higher than − 145 cm, and T_{soil_30} were used as it had the strongest correlation of the T_{soil} options with hourly mean CH_4 and N_2O fluxes. Soil water parameters were not used in these models; details are explained in Appendix 2. Predictor variables were selected based on AIC (Akaike Information Criterion) and tested for multicollinearity by computing VIFs (variance inflation factors) for the final models, with $VIF < 5$ set as the threshold. The ANOVA type III test examined significant variation in fixed effects, and the results of the models were printed using libraries *sjPlot* (Lüdecke 2024) and *ggpredict* (Lüdecke 2018).

Nonparametric Wilcoxon rank-sum exact tests with continuity correction were employed to evaluate differences between total annual fluxes of DF vs UF in general, site-types within DF, and tree species. Post-hoc tests with Bonferroni correction were then used for multiple comparisons, such as CH_4 and N_2O fluxes, soil, and soil water physicochemical characteristics. These prior analyses were done using hourly mean fluxes, mean of soil physical and water chemical parameters per sampling event, and plot-level mean values of soil chemical parameters. The significance of results is reported as $p < 0.05$ and presented as mean ± standard error (SE).

Principal Component Analysis (PCA) on plot-level mean (3 values per site) was applied using *FactoMineR* (Le et al. 2008) to identify key continuous drivers of annual emissions and determine the categorical classification of the associated factors. A graph of PC1 vs PC2 was visualised using *factoextra* (Kassambara and Mundt 2020), with samples grouped into categorical factors (e.g., drainage regime, tree species, and site type). Correlation coefficients between the continuous variables (e.g., annual fluxes, soil and water variables) and the first four principal components were extracted from the PCA, which explained 75.4% of the total variance. In addition, linear mixed-effect models were constructed to assess the effects of both categorical and continuous variables on CH_4 and N_2O fluxes, after log transformation. Plot-level mean values were used in the analysis, including sites as random effects, and results were reported as significant if $p < 0.05$.

Site-level annual CH_4 and N_2O fluxes were used to calculate mean annual fluxes across drainage regimes, tree species, and site-types, and the results were compared using Wilcoxon rank sum exact tests for significant differences. As a supplement to inform about factors driving site-level annual fluxes, non-linear regression models were fitted to mean WTL with CH_4 and CN ratios with N_2O fluxes. The mean annual fluxes at the DF and UF sites are presented as emission factors. To estimate the uncertainties in the EFs of CH_4 and N_2O from DF and UF sites, the annual fluxes were bootstrapped using 1000 resamples with the "boot" package. The 95% confidence interval (CI) was calculated using the "basic" method.

Results

Soil physical and chemical differences between drained and undrained conditions

The organic layer thickness varied across sites, with DF sites exhibiting a significantly shallower profile (98 ± 6 cm) than UF sites (166 ± 11 cm). The mean BD of the organic layer (0–20 cm) and mean SOC were both higher in DF sites (243 ± 16 kg m^{-3}; 460 ± 16 g kg^{-1}) than in UF sites (174 ± 8 kg m^{-3}; 443 ± 8) (Fig. 2). Further, UF sites showed significantly higher Ntot concentrations (27 ± 0.9) than DF sites (23 ± 0.9). The C/N ratio also differed significantly, with DF sites averaging 18 ± 0.8 and UF sites 15 ± 0.7, and the highest ratios observed in pine stands and the lowest in alder stands (Table S4; Table S5). Among the DF stands, the spruce- and birch-dominated stands had the highest SOC concentrations, while the alder stands exhibited the lowest. A similar trend was observed for total N, with pine stands showing the lowest concentrations. Soil pH values ranged from 2.7 to 5.8, with pine stands having the lowest mean pH and alder-dominated stands the highest. The mean top-soil pH was lower in DF than UF sites (Fig. 2). UF sites also had significantly higher concentrations of Ca, Mg, K, and P than

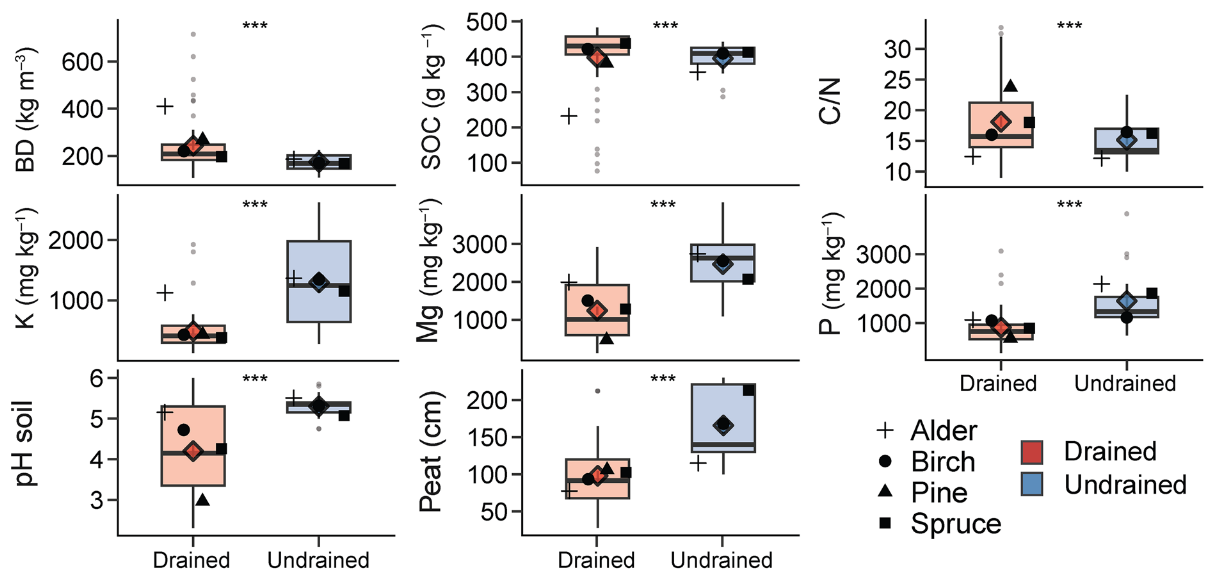

Fig. 2 Soil characteristics of drained and undrained forests at 0–20 cm depths and peat layer depth, showing mean (diamond in center) and median values (center line) along with 25th and 75th percentiles (bottom and top of the box). The different symbols show the means for stands dominated by different tree species

DF sites. Within DF sites, alder-dominated stands showed the highest concentrations of soil P and Mg (Table S5).

During the two-year study period, the mean T_{soil_10} varied across sites, ranging from 5.8 to 10.8 °C annually and from 10.1 to 15.4 °C during the growing season (May-Oct), while it remained between 0 and 2 °C during the non-growing season (Nov-Apr) (Fig. 3A). The mean Tsoil at 10 and 30 cm depths was 7.5 ± 0.2 and 7.7 ± 0.2 °C respectively, based on overall measurements across sites (Table S3). A higher mean Tsoil_10 (9.4 ± 0.4 °C) was recorded at Lithuanian sites than at sites in Estonia and Latvia (7.4 ± 0.2 and 7.2 ± 0.1), corresponding to higher T_{air} (Table S2). No significant difference was observed between drained (DF) and undrained (UF) sites, whereas alder-dominated stands had the highest soil temperatures (Table S5). Temporal variability in WTL was observed at all sites, with the lowest levels during Jul-Sep and the highest in Jan-Mar (Fig. 3B). The DF sites had significantly lower mean WTL (− 66 ± 5.6 cm) than UF sites (− 8.8 ± 1.6 cm), with ranges of − 27.1 to − 144.9 cm and − 1.5 to − 15.4 cm, respectively (Table S3). Among tree species, birch- and pine-dominated DF stands had the deepest WTL, while alder stands had the shallowest in both DF and UF sites (Table S5). Mean SWC was 0.37 ± 0.08 m^3 m^{-3} in DF sites and 0.74 ± 0.05 m^3 m^{-3} in UF sites (excluding flooded conditions). Soil water SPC was significantly higher in DF sites (283.2 ± 9.5 µS cm $^{-1}$), especially in alder stands, while pine stands had the lowest SPC and pH (5.31 ± 0.08).

Dynamic controls of hourly CH_4 and N_2O fluxes

Generally, the DF sites acted as soil CH_4 sinks with a mean (± SE) hourly CH_4 flux of –58 ± 0.9 (µg m^{-2} h^{-1}) throughout the measurement period, while the UF sites showed CH_4 emissions (386 ± 73 µg m^{-2} h^{-1}). The hourly mean CH_4 fluxes showed clear seasonal patterns, with relatively low values during the non-growing season (Nov-Apr) and increased fluxes during the growing season (May-Oct) (Fig. 3C; Fig. S1). On average, CH_4 fluxes during the growing season (mean ± SE) were –75 ± 1.3 in DF sites and 762.4 ± 148 µg m^{-2} h^{-1} in UF sites (Table S6). The hourly mean CH_4 fluxes (uptake in DF, emission in UF) were positively correlated with Tsoil at both 10 and 30 cm depths, with a stronger correlation at the deeper Tsoil_30 ($r = –0.42$, $p < 0.05$) (Fig. S2). The soil wetness indicators WTL and SWC also influenced the CH_4 fluxes ($r = 0.67$ and 0.46, respectively), with WTL showing a more pronounced effect across both DF and UF sites. In contrast, soil water SPC showed a weak, non-significant correlation ($r = –0.14$), and neither pH nor O_2 concentration significantly affected CH_4 fluxes in pooled observations of all sites.

Overall, in DF sites, lower WTL increased CH_4 uptake, while the UF sites showed higher CH_4 emissions, particularly during the growing season when WTL was close to or above the soil surface (Fig. 3). A linear mixed-effects (LME) model including Tsoil, WTL, drainage regime, and their interaction explained 55% of the variation in hourly CH_4 fluxes (Table 1; Table S7). The CH_4 fluxes differed significantly between DF and UF sites, with effects of both Tsoil and WTL. The optimal WTL depth for CH_4 uptake was determined to be − 22 cm (Fig. S3). No significant differences in CH_4 fluxes were found between dominant tree species ($p > 0.05$) of either DF or UF sites (Fig. 4). Generally, however, birch stands showed the highest uptake rates in DF sites and alder stands the lowest. In UF sites, both birch- and alder stands exhibited higher CH_4 emissions than spruce stands.

The mean hourly N_2O flux was significantly higher in DF sites (64.6 ± 3.9 µg m^{-2} h^{-1}) than UF sites (16.3 ± 2.2 µg m^{-2} h^{-1}). The hourly N_2O fluxes in DF sites showed a distinct seasonal pattern, with peaking emissions in early spring (Mar and Apr), typically during snowmelt events (Fig. 3D; Fig. S1) and showed higher emissions during the growing season (93.6 ± 7.3 µg m^{-2} h^{-1}) than in non-growing season (34.7 ± 2.1 101 µg m^{-2} h^{-1}) (Table S7). The UF sites exhibited higher N_2O fluxes during growing season (28.9 ± 4.3 µg m^{-2} h^{-1}), primarily during dry–wet transition periods from Jun to Sep when WTL was at lowest, compared to non-growing season (3.7 ± 0.7 µg m^{-2} h^{-1}). Overall, N_2O fluxes correlated positively with Tsoil at both depths 10 and 30 cm ($r = 0.24$, $r = 0.23$, respectively) and SPC ($r = 0.25$), and negatively with WTL ($r = –0.18$) and O_2 concentration of soil water ($r = –0.22$) (Fig. S2).

A linear mixed-effects model including WTL, drainage regime, and their interactions with T_{soil_30} explained 19% of the variation in hourly N_2O fluxes (Table 1). The significant Tsoil × WTL

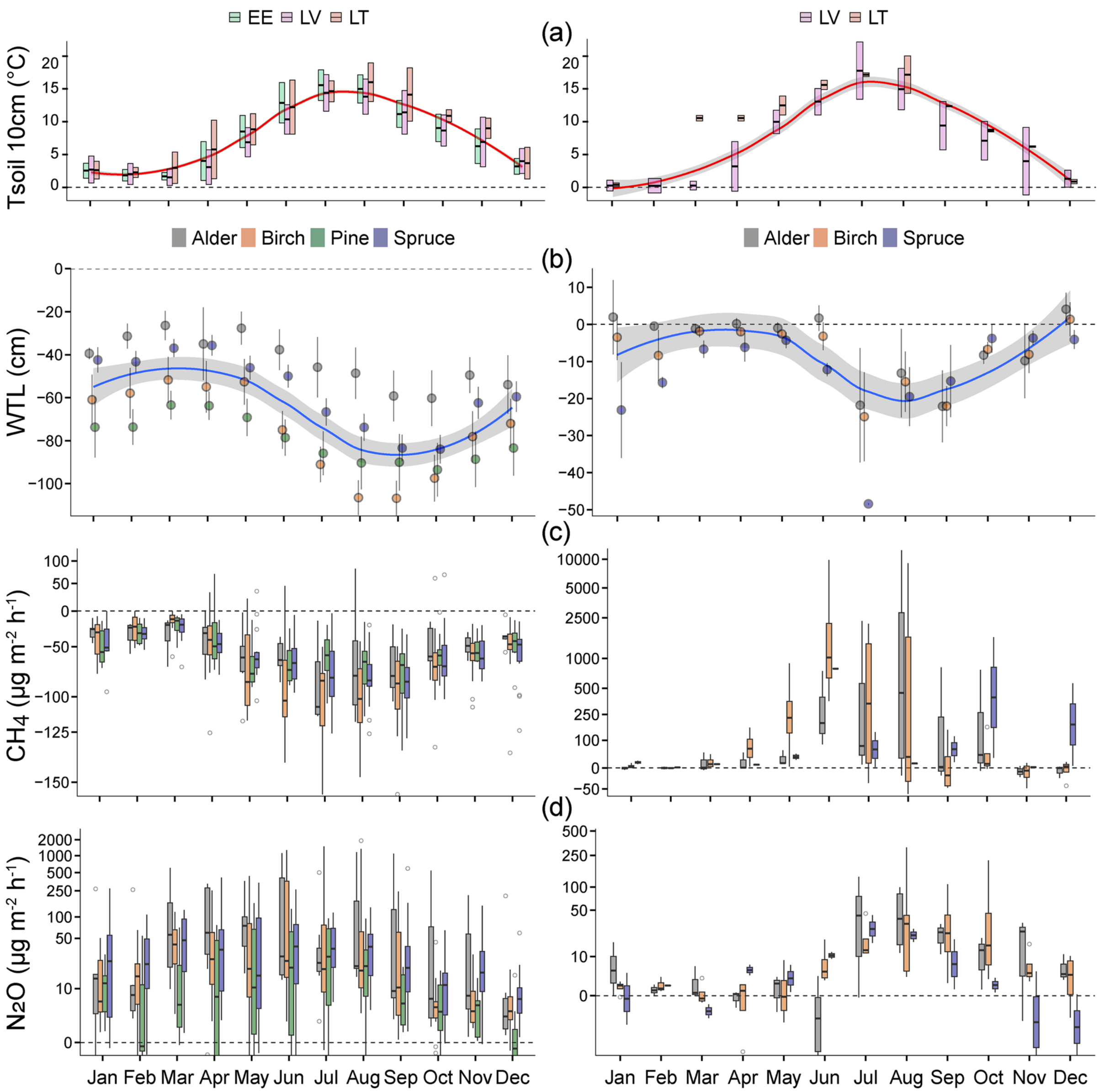

Fig. 3 Monthly variation at the drained (left panel) and undrained (right panel) study sites of **a**) soil temperature at 10 cm (T_{soil_10}; °C), **b**) soil water-table level (WTL; cm from the soil surface), and the log-transformed hourly **c**) CH_4 and **d**) N_2O fluxes (μg m^{-1} h^{-1}) during the measurement period (2021–2022 or 2022–2023, depending on site). Different colours of the boxplots indicate sites with different dominant tree species: alder (Alnus incana), birch (Betula pubescens), pine (Pinus sylvestris), and spruce (Picea abies). The red line in (**a**) shows the mean Tsoil, and the box shows the means (± SD) of Tsoil observed in different countries with a gradient red colour for EE-Estonia, LV-Latvia, and LT-Lithuania. The blue line in (**b**) shows the mean WTL, and the circle represents tree species means (± SE). Variations in Tsoil (red) and (WTL blue) were smoothed with LOESS based on mean hourly values. The boxplots (**b**,**c**) show median in the centre with the 25th and 75th percentiles, and whiskers at the top and bottom are the minimum and maximum values

Table 1 Parameter estimates of linear mixed-effect models of log-transformed CH_4 and N_2O hourly fluxes as response variables. Fixed effects were soil temperature (T_{soil_30}, °C), soil water-table level (WTL, cm), and drainage regime (drained, undrained). Detailed output tables describing random effects and predictions of interaction effects are given in the supplementary (Table S7; Fig. S3)

Fixed Variable	CH_4 fluxes			N_2O fluxes		
	Wald statistics	df	p-value	Wald statistics	df	p-value
Intercept	7630	1	**< 0.001**	179.9	1	**< 0.001**
WTL	11.6	1	**< 0.001**	80.1	1	**< 0.001**
T_{soil_30}	33.4	1	**< 0.001**	-	-	-
Drainage	-	-	-	20.4	1	**< 0.01**
Drainage × T_{soil_30}	286.6	1	**< 0.001**	-	-	-
Drainage × WTL	95.3	1	**< 0.001**	84.6	1	**< 0.001**
T_{soil_30} × WTL	-	-	-	68.9	1	**< 0.001**
Marginal/Conditional R^2	0.55/0.76			0.19/0.75		

Abbreviation: Wald Statistics – shows the Wald chi-square value of individual fixed effects (calculated as the squared ratio of estimated coefficients to their standard error), *p*-value – indicates the significance ($p < 0.05$) of the Wald chi-square test, and df – degree of freedom

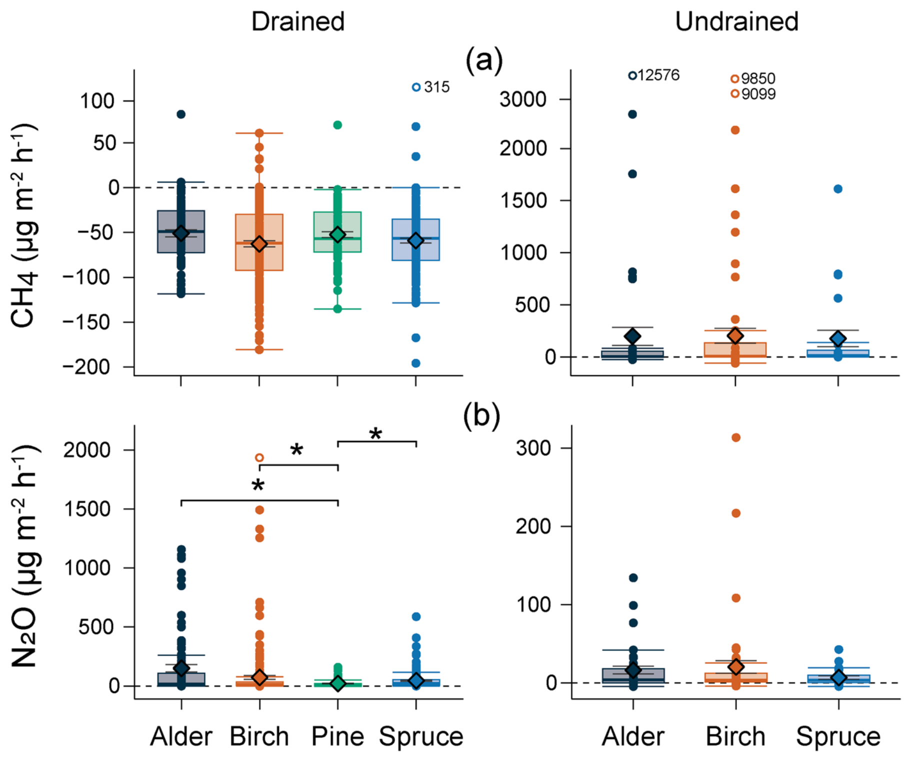

Fig. 4 Variation in mean hourly **a**) CH_4 and **b**) N_2O fluxes across different tree species and drainage regimes. Boxplots show median with 25th and 75th percentiles and whiskers, and diamond in center display mean ± SE. The dots indicate individual observations or outliers

interaction suggests that the influence of temperature on N_2O emissions varies with water table level depth (Fig. S3). On average, the alder- and birch-dominated DF stands showed the highest N_2O emissions, with alder stands emitting more N_2O (150 ± 32.1 µg m^{-2} h^{-1}) although not differing significantly from other species (Fig. 4). However, pine stands had significantly lower emissions (20.8 ± 3.4 µg m^{-2} h^{-1}) and occasionally acted as soil N_2O sinks (Fig. 3).

Water table thresholds regulated annual CH_4 flux

Drained forest sites consistently acted as annual CH_4 sinks, with annual fluxes ranging from –8.3 to –0.8 kg ha^{-1} y^{-1} (overall mean, –4.6 ± 0.4 kg ha^{-1} y^{-1}), while UF sites functioned as CH_4 sources, emitting between 1.3 to 120 kg ha−1 y−1, and a mean of 38.2 ± 15.4 kg ha^{-1} y^{-1} (Fig. 5A, Table 2). The growing season fluxes accounted for 65% and 98% of the total annual CH_4 fluxes at the DF and UF sites, respectively (Table S6).

The peatland forest site-types showed distinct patterns, with Oxalis-type DF sites (spruce- and birch-dominated) showing the highest annual CH_4 uptake and Myrtillus-type DF (pine stands) the lowest, while the Dryopteris-type (all UF sites) showed annual net emissions (Fig. 5B). The LME analysis identified mean WTL as the strongest predictor, explaining 48% of the variation in mean annual CH_4 fluxes, followed by forest site type, drainage regime, drainage effect in different tree stands and soil nutrients (Table S9). A non-linear

Table 2 Emission factors of CH_4 and N_2O-N from drained organic forest soils for the hemiboreal Baltic region as compared to Ipcc 2014) EFs for temperate and boreal regions. For the Baltic region, values for undrained organic forest soils are also provided

Region	Drainage regime	No. of sites (n)	CH_4 kg ha^{-1} y^{-1} (95% CI)	No. of sites (n)	N_2O-N kg ha^{-1} y^{-1} (95% CI)
IPCC, Temperate	Drained	13	2.5 (−0.6, 5.7)	13	2.8 (−0.57, 6.1)
IPCC, N−rich Boreal	Drained	83	2.0 (−1.6, 5.5)	75	3.2 (1.9, 4.5)
Baltic region*	Drained	18	−4.6 (−5.3, −3.9)	18	3.7 (2.0, 6.0)
	Undrained	7	38.2 (10.7, 68.0)	7	0.9 (0.4, 1.6)

*Using bootstrapping methods, the 95% CIs are based on annual fluxes from drained (n = 30) and undrained (n = 9) sites in this study

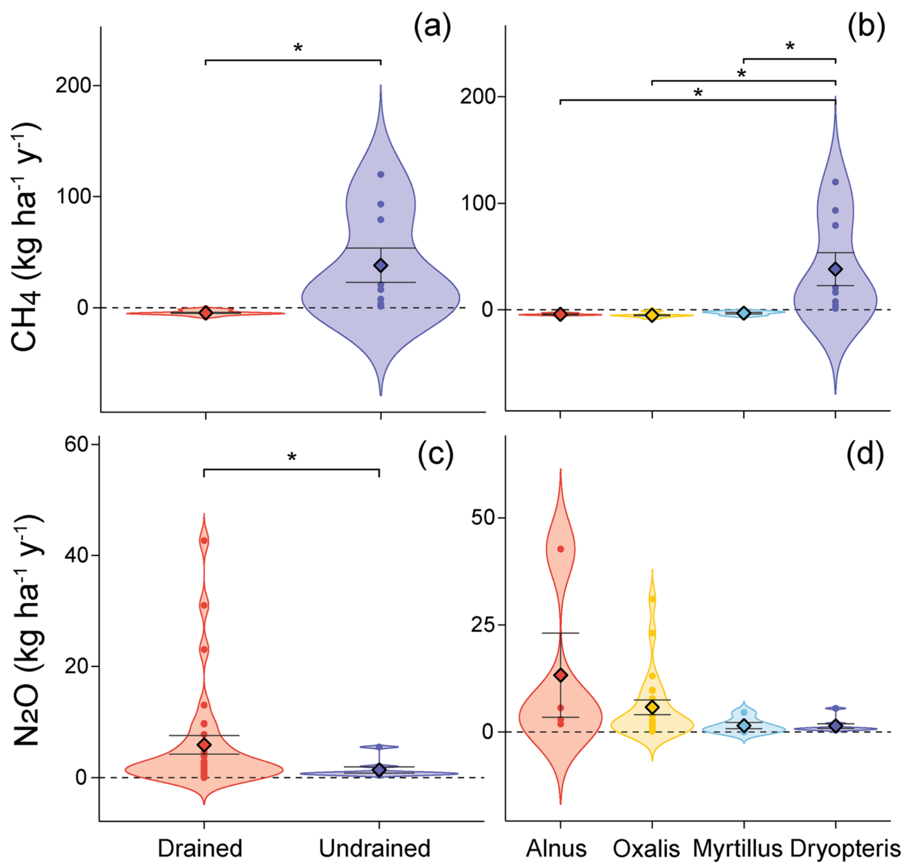

Fig. 5 Cumulative annual CH_4 and N_2O fluxes from drained and undrained sites (**a**, **b**), and across different forest site-types (**c**, **d**). The violin plot represents a density curve indicating higher and lower frequency of data points with wider and narrower sections, respectively, and a diamond in the centre of the violin shows mean ± SE. Significant p-values are derived from non-parametric pairwise comparisons of means

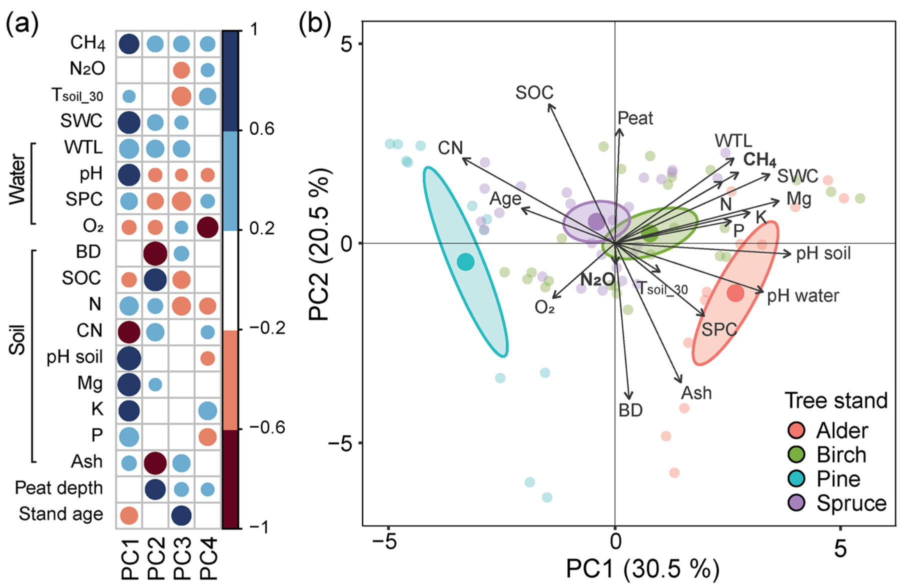

Fig. 6 Principal component analysis: **A**) PCA loadings on measured variable showing correlation matrix with first four components of PCA (PC1, PC2, PC3, and PC4). Significant correlation coefficients are colour-coded, with dark colours on the bar representing high values, blue indicating positive and brown indicating negative values. **B**) PCA biplot of first two components with arrows representing variable loadings. The coloured ellipses with large centroids show 95% confidence intervals for the dominant tree species, and small dots are the individual plot observations

relationship with WTL showed a threshold at –25 cm, below which CH_4 emissions shifted to net annual uptake in both DF and UF sites (Fig. 7A). However, different countries, sampling years and tree species within DF and UF sites showed no significant differences in annual CH_4 fluxes (Table S8).

In line with the specific results presented above, the PCA analysis highlighted major ecological gradients shaping the CH_4 dynamics (Fig. 6A, B). The first axis (PC1) represented a drainage and nutrient gradient, separating wetter, nutrient-rich deciduous-alder stands with high CH_4 emissions from drier, more acidic pine-conifer stands with lower fluxes. Higher CH_4 emissions were associated with deep, organic-rich soils, whereas CH_4 uptake was associated with shallow, mineral-dense soils (PC2). Methane fluxes aligned positively with PC1 and PC2, indicating that hydrology and nutrient availability are key controls on soil CH_4 exchange in these forested peatlands.

Soil edaphic factors control annual N_2O emissions

Both drained and undrained forest sites generally acted as annual sources of N_2O, with fluxes varying significantly between drainage regimes ($p<0.05$) (Fig. 5C). Annual N_2O emissions from DF sites ranged from –0.03 to 42.7 kg ha^{-1} y^{-1} (mean, 5.9 ± 1.7 kg ha^{-1} y^{-1}), while UF sites ranged from 0.4 to 5.5 kg ha^{-1} y^{-1} (mean, 1.4 ± 0.5 kg ha^{-1} y^{-1}). Growing season N_2O fluxes accounted for 65% and 85% of the annual totals at the DF and UF sites, respectively. The differences among dominant tree species were not statistically significant; however, a trend of increasing N_2O emissions was observed from pine < birch < spruce < alder stands (Table S8). The LME analysis identified forest site type, drainage regime, stand age and peat depth across different tree species as key factors influencing N_2O fluxes (Table S9). Furthermore, a non-linear relationship with the C/N ratio showed a threshold between 14 and 20, between which N_2O emissions occurred in both DF and UF sites (Fig. 7B).

Although the differences were not significant, the Myrtillus-type pine stands had the lowest N_2O emissions and were comparable with the Dryopteris UF sites, while Alnus-type DF sites, dominated by alder, showed the highest emissions (13.3 ± 9.8 kg ha^{-1} y^{-1}) (Fig. 5D). Based on the PCA results, N_2O fluxes exhibited significant associations only with PC3 (negative) and PC4 (positive), which reflected stand development and mineral nutrients contrasts (Fig. 6A, B). Overall, N_2O fluxes were strongly associated with

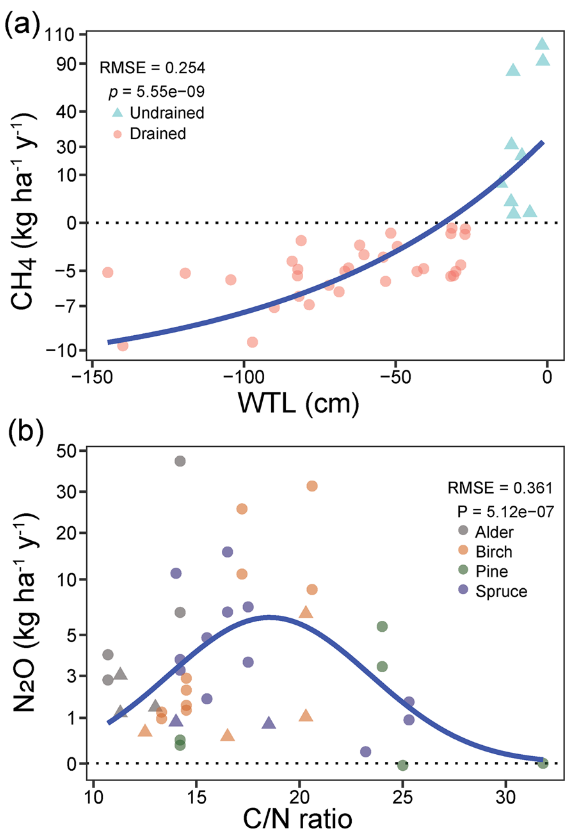

Fig. 7 Relationship between **a**) mean soil water-table level and annual CH_4 fluxes, **b**) soil C/N ratio and annual CH_4 fluxes in all drained (circle) and undrained (triangles) forest sites. Both models were fitted on normalised CH_4 and N_2O fluxes to improve residual distribution and then back-transformed to plot them on the original scale. Detailed estimates of the fitted models are given in Table S10

Tsoil, WTL, BD, stand age, and soil nutrients such as SOC, N, C/N, K, and P. Elevated N_2O emissions were observed in drained, mineral-rich, and younger stands under warmer conditions, highlighting the role of mineralisation, nutrient cycling and forest succession. Annual N_2O fluxes did not differ significantly between countries, supporting the use of mean values as Baltic region-specific emission factors for peatland forest soils (Table 2).

Discussion

We investigated soil fluxes of CH_4 and N_2O from a wide range of drained and undrained nutrient-rich peatland forests representing different site-types under hemiboreal conditions. The observed annual CH_4 emissions were clearly lower, whereas the N_2O emissions were somewhat higher but fell in the range of, the default Ipcc 2014) EFs for drained, nutrient-rich boreal and temperate organic forest soils (Table 2). The differences could be due to our study sites' regional climate, soil drainage regime, and nutrient status (Alm et al. 2007a; Jauhiainen et al. 2023; Laine et al. 2006). The IPCC EFs for the temperate region were further based on spatially very limited data, including only eight sites, and lacked division into nutrient-rich and nutrient-poor sites. The findings from this study can be used to update Baltic regional or country-specific EFs and serve as a valuable resource for the national GHG inventory.

Seasonal and environmental drivers influencing hourly CH_4 and N_2O fluxes

Soil physical variables, WTL and Tsoil, mainly drove the seasonal dynamics of CH_4 fluxes across all sites. In the drained peatland forests, CH_4 uptake increased with decreasing WTL depths, likely due to enhanced methanotrophic activity in the wider oxic zone (Lai 2009). Our predictions from the mixed-effect model indicated that the soils shifted from CH_4 sources to CH_4 sinks when the daily mean WTL dropped below 22 cm. Temperature also affects CH_4 consumption as shown by a positive correlation between Tsoil and CH_4, and consistent with findings from previous studies on drained peatlands (Korkiakoski et al. 2017; Olson et al. 2013). Episodic weather events also contributed to the seasonal variation of CH_4. For instance, in drained sites, short-term CH_4 emissions occurred mainly after rainfall events that temporarily saturated the soil. While at undrained sites, CH_4 emissions were dominant and increased exponentially as the WTL approached the soil surface, corresponding to enhanced methanogenesis under anoxic conditions (Bridgham et al. 2013; Lai 2009). Soil temperature strongly influenced CH_4 emissions also under these conditions, as higher temperatures stimulate methanogen activity, it is also related to the vegetative cycles of wetland plants (Le Mer and Roger 2001; Luan et al. 2019; Ojanen et al. 2010). In addition to physical drivers, chemical properties of soil water, such as SPC, pH, and O_2 concentrations, varied temporally and spatially, affecting dissolved gas solubility (Bansal et al. 2023). These variations likely indicate shifts in water sources, and evaporation and dilution effects, which may explain higher specific conductivity at drained sites than in undrained sites. Nevertheless, the availability of labile organic substrates affects methanogen activity (Bridgham et al. 2013), and higher groundwater inflow raises electrical conductivity, which is a key predictor of CH_4 emissions in wetlands (Pennock et al. 2010).

Regarding N_2O fluxes, the drained sites showed temporal variation linked with soil physical and water variables, which regulate N cycling processes affecting N_2O emissions from the soil (Oertel et al. 2016). The hourly N_2O emission was generally higher in DF sites, with persistently lower WTL throughout the year, than in UF sites, and consistent with findings from drained nutrient-rich forest organic soils (Butlers et al. 2023; Minkkinen et al. 2020). A lower WTL enhances soil aeration, promoting the mineralisation of organic N to NH_4^+, as well as subsequent nitrification (NH_4^+ to NO_3^-) and denitrification processes (Butterbach-Bahl et al. 2013; Klemedtsson et al. 2005). Additionally, higher concentrations of electron donors and acceptors in soil water stimulate N_2O production through intensified N transformations (Quick et al. 2019; Zhang et al. 2020). This elevated substrate availability is reflected in the higher conductivity (SPC) in soil water of drained sites compared to undrained sites, which is attributed to an excess of nutrients (e.g., NO_3^-), that further drive N_2O emissions (Hill et al. 2016). In contrast to drained sites, N_2O fluxes at undrained sites showed minimal temporal variation, peaking modestly only during the summer. Similarly, as reported in an undrained peatland forest (Minkkinen et al. 2020; Rautakoski et al. 2024), the hot moments of N_2O occurred following heavy rainfall after a drought in the growing season, i.e., dry-wetting cycle (Congreves et al. 2018; Ranniku et al. 2024).. Rückauf et al. (2004) showed that N_2 is the final product of denitrification in water-saturated soils, and accordingly, lower N_2O emissions were observed from UF sites. In this study, a few negative hourly N_2O fluxes were observed in pine stands of DF sites and also in the UF sites, indicating the consumption of N_2O by heterotrophic denitrification in anaerobic and wet soil conditions (Chapuis-Lardy et al. 2007). Moreover, these negative fluxes and positive fluxes during the non-growing seasons were included, because their absolute values were above the flux detection limit for the N_2O GC analyser (Jordan et al. 2016; Salm et al. 2011).

Most of the annual CH_4 (~64%) and N_2O fluxes (~67%) occurred during the growing season in both DF and UF sites, consistent with patterns observed in other drained peatlands (Berglund et al. 2021; Eickenscheidt et al. 2014; Minkkinen et al. 2020, 2007). Higher growing season fluxes are attributed to active plant and microbial activity and the greater availability of labile organic and inorganic substrates (Duan et al. 2022; Truu et al. 2020). While the share of non-growing season CH_4 and N_2O fluxes was around 36% and 32%, respectively, and in some DF sites, their proportions reached up to 78% and 60% of the total annual emissions. The observed non-growing season peak N_2O emissions in these exceptional DF sites were related to snow melting and soil thawing events, which probably released the soil N_2O trapped below the ice layer and increased soil wetness to optimal levels for CH_4 exchange with ice melting at increasing soil temperatures (Gerin et al. 2023; Kazmi et al. 2023). However, non-growing season N_2O peak emissions were not observed in undrained soils, likely due to soil inundation during freezing-thaw cycles, whereas incomplete denitrification under less inundated conditions dominates the N_2O emission during the thawing periods (Kazmi et al. 2023). The non-growing season or winter-time fluxes are mostly missing in previous studies conducted in both undrained and drained peatlands of temperate and boreal regions (Berglund et al. 2021; Minkkinen et al. 2020; Ojanen et al. 2013). In our studied sites, non-growing season fluxes contributed substantially to annual budgets, particularly in hemiboreal climates, where non-growing season emissions can exceed those in colder boreal climates (Jauhiainen et al. 2023). Capturing short-lived emission peaks triggered by rainfall or freeze–thaw events is therefore essential for accurate annual estimates (Gerin et al. 2023; Korkiakoski et al. 2017), which we may have missed due to our two-week and monthly gas sampling frequencies. Nevertheless, our study highlights the importance of measuring the non-growing season fluxes and the interaction effect of soil and soil water properties on the annual variation of CH_4 and N_2O fluxes.

Water table depth was a primary control on annual CH_4 fluxes

The annual CH_4 fluxes were strongly correlated with the mean WTL at the sites. The WTLs explained both the hourly mean and the annual CH_4 fluxes better than the drainage status did. In our study, the WTLs varied widely between and among drained and undrained sites. Generally, DF sites, where annual WTL typically remained below 25 cm, showed CH_4 consumption at the annual scale. Instead, the UF sites, where the WTL consistently exceeded this threshold, were CH_4 emitters. This behaviour aligns with findings from previous research (Evans et al. 2021; Korkiakoski et al. 2020; Mander et al. 2015), emphasising the importance of WTL over other controls on CH_4 fluxes in peatlands. The decrease in CH_4 emissions after drainage has been explained by WTL drawdown, which increases the volume of oxic surface soil, and a change in ground vegetation, with wetland plants that act as direct pipelines for CH_4 to move from anoxic soil to the atmosphere (Minkkinen et al. 2007). Soil characteristics and different site-types showed influence on annual CH_4 fluxes. Drained sites had about 60 cm of less peat layer depth and higher bulk density, compared to undrained sites, reflecting surface peat subsidence and compaction (Minkkinen and Laine 1998; Silins and Rothwell 1998). However, we did not observe any significant effect on soil CH_4 exchange. Peat depth and nutrient richness play a positive role on CH_4 consumption, as they determine substrate supply for methanotrophs (Lee et al. 2023). The Myrtillus type pine forest, with lower soil pH, showed a trend of reduced CH_4 uptake than the Oxalis site type, which is consistent with evidence that low pH suppresses CH_4 oxidation by altering methanotrophic communities and limiting copper availability essential for enzyme activity to oxidise CH_4 (D'Imperio et al. 2023). Notably, although drainage ditches were present at drained sites, our estimates exclude ditch fluxes. Ditches are known to emit CH_4 when water-filled (Köhn et al. 2021), while remaining near zero from dry ditches (Peacock et al. 2021; Tong et al. 2022). Accounting for CH_4 fluxes from ditches could change our estimates, but limited data are available on CH_4 fluxes from different types of ditches under different site and climatic conditions. (Korkiakoski et al. 2017; von Arnold et al. 2005b).

Significant drivers of variation in annual N_2O fluxes

Higher N_2O emissions have been reported from drained peatland forests (Berglund et al. 2021; Osterloh et al. 2018; Pearson et al. 2015) compared to undrained forests (Eickenscheidt et al. 2014; Minkkinen et al. 2020; Tiemeyer et al. 2016). In our study, the drainage regime had a stronger effect than mean WTL explaining differences in annual N_2O fluxes, contrasting the pattern in annual CH_4 fluxes. Hydrological conditions shape N_2O emissions, and our findings align with global trends, showing higher N_2O emissions at deeper WTLs (> 100 cm) (Yao et al. 2022). N_2O emissions are typically higher in drained nutrient-rich than nutrient-poor sites (Jauhiainen et al. 2023). Similarly, our pine stands on comparatively nutrient-poorer soil with the highest C/N ratios, emitting less N_2O than birch, spruce, and alder stands. More generally, our deciduous stands emitted more than coniferous, consistent with earlier research (von Arnold et al. 2005a, 2005b). This pattern is likely linked to species-specific effects on C and N cycling through differences in litter quality and decomposition rates, and N_2O shows a strong relation with soil chemistry (C/N ratio, pH) and rhizosphere microbiomes (Cornwell et al. 2008; Truu et al. 2020). Overall, variation in annual N_2O fluxes is best explained by the interacting controls of WTL, Tsoil, and soil properties (C, N, pH), highlighting the importance of spatial and environmental heterogeneity (McDaniel et al. 2019; Osterloh et al. 2018; Pärn et al. 2023). Previous studies suggest that emissions rise when the soil C/N ratio falls below a threshold of 15–19, and exhibit a curve-shaped relationship (Klemedtsson et al. 2005; Yao et al. 2022). Hence, the high N_2O fluxes in alder stands may result from N-enriched soils with N_2-fixing *Frankia* and subsequent mineralisation of organic N under semi-wet conditions (Eickenscheidt et al. 2014; Mander et al. 2015; Soosaar et al. 2011). Some drained sites with deeper WTLs and higher organic C also showed higher N_2O emissions, likely driven by denitrifiers utilising C substrates as an energy source (Sgouridis and Ullah 2014), and by enhanced N mineralisation from greater litter inputs (Kasimir-Klemedtsson et al. 1997; McDaniel et al. 2019). On the other hand, exceptionally high N_2O

emissions in two birch-dominated stands on former agricultural peatlands support evidence from N-rich cultivated peatlands (Ernfors et al. 2020; Ojanen et al. 2010), and justify classification of afforested sites as a distinct site category for EFs (Jauhiainen et al. 2023). Soil pH varied among dominant tree species but showed no clear relationship with N_2O emissions, contrary to studies linking low pH to greater emissions due to inhibited denitrification (Firestone and Davidson 1989; Weslien et al. 2009). These findings highlight that using a single characteristic may not reliably describe the soil processes underlying N_2O emission, but several characteristics should be evaluated in tandem to determine N_2O emissions in drained peatlands (Duan et al. 2022; Eickenscheidt et al. 2014; Gerin et al. 2023).

Annual balances and GWP of CH_4 and N_2O

The overall mean annual CH_4 and N_2O fluxes in drained sites were − 4.6 kg ha^{-1} y^{-1} and 5.9 kg ha^{-1} y^{-1}; undrained sites were 38.2 kg ha^{-1} y^{-1} and 1.4 kg ha^{-1} y^{-1}, respectively. To sum up, the combined radiative forcing impact equals 1.48 t CO_2-eq ha^{-1} y^{-1} and 1.41 t CO_2-eq ha^{-1} y^{-1} for drained from undrained forest soils, respectively, while considering the global warming potentials for CH_4 (27) and N_2O (273) over 100 years, including non-fossil emissions (Ipcc 2021). These results show a lower emission potential from undrained soils, 0.07 t CO_2-eq less than drained soils, although they are not equal to zero from nutrient-rich soil conditions. Nevertheless, such high release of N_2O from nutrient-rich soil is not compensated by reduced CH_4 emission under drained conditions. However, CO_2 plays a significant role in soil GHG balance, more than CH_4 and N_2O in nutrient-rich peatland sites, as reported earlier (Ojanen et al. 2013). Mean annual CO_2 emission of 6.21 ± 0.43 t CO_2-C ha^{-1} year^{-1} was reported from drained and 4.38 ± 1.20 t CO_2-C ha^{-1} year^{-1} from undrained forest soils, similar to, and partly the same as, ours by (Butlers et al. 2025). Ultimately, the inclusion of both CH_4 and N_2O emissions thus increases total GHG impacts by 24% in drained and 26% in undrained soils, thereby increasing the significant difference between them. Therefore, all three major GHGs must be considered when implementing climate mitigation and restoration measures in the hemiboreal region.

Emission factors and their uncertainty

Since the cumulative annual emissions (kg ha^{-1} y^{-1}) of neither CH_4 nor N_2O varied significantly among countries, region-specific emission factors (EFs) are recommended for the whole Baltic region rather than country-specific ones. Such emission factors as single mean values from multi-year site measurements, and their associated uncertainties, as in (Ipcc 2014) are shown in Table 2. The CH_4 EFs for drained organic forest soils in the Baltic region showed removals from the atmosphere, contrasting with the default Tier 1 CH_4 EF, reported as net emissions for both boreal and temperate zones. In turn, the N_2O EFs were slightly higher, falling within the upper confidence limits of the Tier 1 EF range. As expected, localised conditions lead to differences between our regionally derived emission factors and broader-scale estimates. Temperature, WTL, drainage intensity, and ditch design varied little within our study area, contributing to the relatively narrow confidence intervals for EFs despite diverse tree species and site-types.

Some variability naturally persists, underscoring the need for dynamic, model-based EFs that account for changing climatic and biogeochemical conditions. As our site-level parameters offered limited predictive power, the calculated regional EFs carry an inherent degree of uncertainty. Moreover, diurnal variability can bias our estimates when extrapolating daytime measurements to annual emissions. In drained peatlands, where aerobic layers are extensive, diurnal variation in methane oxidation may play a key role in regulating net CH_4 fluxes. Also, plant-mediated CH_4 transport commonly peak during the day, making daytime sampling prone to overestimation (Knox et al. 2021; Long et al. 2010; Thomas et al. 1996). N_2O fluxes show widespread but inconsistent diurnal patterns; high emissions follow temporal variation in environmental conditions and often lack a clear diurnal structure, making sampling frequency more important than sampling time (Butterbach-Bahl et al. 2013; Francis Clar and Anex 2020; Rautakoski et al. 2024). However, our annual estimates are based on a variable number of observations across seasons, years, species, and site-types, allowing robust comparisons between drained and undrained soils. Therefore, averaging the data effectively served our purpose and ensured the comparability of annual values across divergent groups. Future research should employ high-frequency automated chamber measurements and continuous WTL monitoring to refine annual CH_4 and N_2O estimates and enable upscaling over extended periods. In this study, we prioritised regional representativeness over intensive measurements at fewer sites, a trade-off essential for broader applicability.

Conclusions

The study highlighted the significant evidence gap in understanding CH_4 and N_2O fluxes from forests on drained and undrained nutrient-rich organic soils in the hemiboreal region. Drained forest sites exhibited CH_4 consumption than CH_4 emissions in undrained sites. N_2O emissions increased with N-richness and were higher from drained peatland forests. However, the mere presence of a ditch was not always a good predictor of the WTL, and WTL, as such, was a better predictor of the CH_4 fluxes. We found that both annual and seasonal CH_4 and N_2O fluxes in peatland forests were primarily influenced by water table level and soil edaphic characteristics. Better water management can reduce emissions, but tradeoffs between CH_4 and N_2O emissions require careful consideration. The CH_4 emission factors (EFs) estimated for the Baltic region indicate net atmospheric removals, which differ from Tier 1 IPCC default EFs for boreal and temperate zones. N_2O EFs, in turn, were slightly in the higher range of the upper confidence interval than Tier 1 IPCC EFs and varied by dominant tree species and forest site-types. These findings provide region-specific EFs that can improve national and regional GHG inventory reporting, ensuring more accurate emission estimates for the LULUCF sector. It should be noted that any management-related direct emissions, such as those associated with different cutting regimes, site preparation, or fertilisation, are not included in our estimates and would require specific studies for quantification.

Authors contributions M.K-S: Writing – original draft, Investigation, Formal analysis, Visualisation; R.L: Writing – review & editing, Conceptualisation, Methodology, Supervision, Funding acquisition; J.J: Writing – Review & editing, Investigation, Conceptualisation, Methodology; A.Bu: Writing – Review & Editing, Investigation; T.S: Investigation, Writing – review & editing; H.V: Investigation, Writing – review & editing; D.C: Investigation, Writing – review & editing; E.V: Investigation, Writing – review & editing; A.T: Writing – review & editing; A.K: Writing – review & editing, Methodology; A.L: Writing – review & editing, Conceptualization, Funding acquisition; A.H: Writing – review & editing; A.Ba: Writing – review & editing; I.L: Writing – review & editing; V.S: Writing – review & editing; Ü.M: Review and editing, Conceptualisation; K.A: Funding acquisition; K.S: Writing – review & editing, Conceptualisation, Methodology, Supervision, Funding acquisition.

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Declarations

Declaration of generative AI and AI-assisted technologies in the writing process The author used Grammarly, an AI-powered editing tool, to enhance spelling, grammar, and readability.

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