Annual net CO_2 fluxes from drained organic soils used for agriculture in the hemiboreal region of Europe

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Abstract. Carbon dioxide (CO_2) emissions from drained organic soils used for agriculture contribute significantly to the overall anthropogenic greenhouse gas budget in the land use, land-use change and forestry (LULUCF) sector. To justify the implementation of climate change mitigation measures on these lands, it is important to estimate at least the regional variation in annual net CO_2 fluxes. This study presents the first estimates of annual net CO_2 fluxes from drained nutrient-rich organic soils in cropland (8 sites) and grassland (12 sites) in the hemiboreal region of Europe, represented by Estonia, Latvia and Lithuania. The study sites represented both deep, and shallow highly decomposed, organic soils, categorised based on the concentration of organic carbon in the top 20-cm soil layer. CO_2 flux measurements were conducted at least over two years in each site. To estimate annual net CO_2 fluxes, ecosystem respiration (R_{eco}) and soil heterotrophic respiration (R_{het}) were measured using a manual dark chamber technique, and carbon (C) input to soil through plant residues was estimated. R_{eco} was strongly dependent on temperature, particularly soil temperature at 10 cm depth, but rather independent of soil water-table level and soil moisture. The overall mean annual net soil CO_2 fluxes, calculated as the difference between annual output (R_{het}) and input (plant residues), were 4.8 ± 0.8 t CO_2-C ha^{-1} yr^{-1} for croplands and 3.8 ± 0.7 t CO_2-C ha^{-1} yr^{-1} for grasslands, while the means for “true” or deep organic soil were 4.1 ± 0.7 t CO_2-C ha^{-1} yr^{-1} in cropland and 3.2 ± 0.6 t CO_2-C ha^{-1} yr^{-1} in grassland (mean ± standard error). Both the annual R_{eco} and net CO_2 fluxes for shallow highly decomposed organic soils, currently not recognised as organic soil by the Intergovernmental Panel on Climate Change (IPCC), were of similar magnitude or even higher than those from deep organic soil, suggesting a need to separate them from mineral soils in emission estimation.

1 Introduction

Organic soils drained for agriculture contribute significantly to anthropogenic greenhouse gas (GHG) emissions and are carbon dioxide (CO_2) emission hotspots in the agricultural and land use, land-use change and forestry (LULUCF) sectors in many countries (Tubiello et al., 2015; Tiemeyer et al., 2016; Tubiello et al., 2016; Säurich et al., 2019a; European Environment Agency, 2023). When evaluating the overall impact of drained organic soils used for agricultural production on the greenhouse effect, CO_2 is considered the most important GHG (Houghton et al., 2001; Maljanen et al., 2007). Maljanen et al. (2007) reported that CO_2 emissions accounted for around 80 % of the total emissions of CO_2, methane (CH_4), and nitrous oxide (N_2O) in drained organic cropland soils in the boreal region. The soil CO_2 emissions result from two main processes: autotrophic respiration, which is the respiration of living plant roots, and heterotrophic respiration (R_{het}), which involves respiration of soil biota such as microorganisms and soil fauna responsible for decomposing litter and soil organic matter (SOM) (Kuzyakov, 2006; Berglund et al., 2011; Bader et al., 2017; Tang et al., 2020a; Tang et al., 2020b). SOM-derived CO_2 emissions, along with estimates of C input to soil by vegetation, are key components in the assessment of soil as a source or sink of atmospheric CO_2 (Kuzyakov, 2006; Tiemeyer et al., 2016).

According to the European Union (EU) GHG inventory for the year 2021, 4.1 Mha or 1 % of the total land area in the EU comprised managed organic soils under cropland and grassland, corresponding to emissions of 76 Mt of CO_2 (European Environment Agency, 2023). Thus, these soils were responsible for the largest share (~70 %) of GHG emissions from managed organic soils in the EU (European Environment Agency, 2024). The largest area of drained organic soils used for agriculture is in Eastern and Northern Europe. As of 2019, this region comprised 45 % of the worldwide agricultural land on organic soil (FAO, 2020). In order to achieve international climate change mitigation goals, like the Paris Agreement (UNFCCC, 2015) and the European Green Deal (Fetting, 2020), an increase in the sequestration of atmospheric CO_2 and a reduction in GHG emissions from organic soils, especially from soils drained for agricultural use, is urgently required. To take effective and practical measures to reduce emissions, it is necessary to know where and why the emissions are highest, and to understand how they respond to changes in the environmental variables regulating them.

It is well documented that improved soil aeration caused by lowering the soil water-table level (WTL) through ditch drainage along with mechanical practices (e.g., repeated ploughing), as well as the application of lime and fertilizers, enhance the conditions for SOM mineralisation and the associated CO_2 production (Nykänen et al., 1995; Lohila et al., 2004; Maljanen et al., 2007). However, CO_2 emissions from drained organic soils vary considerably. They depend on complex interactions of many physical and chemical variables, including local climate and physical soil conditions (mainly temperature, moisture, and WTL), soil properties (e.g., peat type, composition, degree of decomposition), as well as the type and intensity of management, including the type of vegetation (Oleszczuk et al., 2008; Norberg et al., 2016; Tiemeyer et al., 2016; Minasny et al., 2017, Bader et al., 2018; Fairbairn et al., 2023).

Relative to the number of the affecting variables and their interactions, as well as variation in management practices and intensity, there is still a rather limited number of studies that provide comprehensive information on the annual net CO_2 fluxes from drained organic soils used for agriculture. For instance, the IPCC (Hiraishi et al., 2014) default CO_2 emission factors for drained nutrient-rich organic soils in the temperate and boreal regions are based on data from 39 sites for croplands and 60 sites for grasslands. The categories temperate and boreal are broad and comprise a lot of variation in climatic, hydrological and geomorphological conditions that are likely to shape the emissions. Still, there is too little data to adjust the emission factors correspondingly. Further, many studies have focused on deep peat soils known as Histosols, which have high soil organic carbon (SOC) content. Yet, some studies have highlighted that also soils with comparatively low SOC concentration (<15.0 %, Tiemeyer et al., 2016), which do not fall under the definition of organic soils by the IPCC (Eggleston et al., 2006), may have high CO_2 emissions (Leiber-Sauheitl et al., 2014; Eickenscheidt et al., 2015; Liang et al., 2024). These soils include formerly drained peatlands that have transformed into organo-mineral soils due to prolonged agricultural activities. Thus, the total GHG emissions from soils used in agriculture may be underestimated if such soils are treated as mineral soils in the estimation, but their emissions are higher.

In the Baltic countries, which, according to the vegetation zone classification (Ahti et al., 1968), fall within the hemiboreal region of Europe, the share of croplands and grasslands with organic soils comprises 3–6 % and 5–19 % of the total land area and produce up to 156 % and 75 % of total net GHG emissions (including CO_2 removals) in cropland and grassland, respectively (Estonia's National GHG inventory, 2023; Latvia's National GHG inventory, 2023; Lithuania's National GHG inventory, 2023). To provide knowledge-based recommendations for land-use and climate policymakers regarding the management of organic soils, the magnitude of ecosystem CO_2 fluxes and the variables affecting them need to be quantified under climatic and management conditions that are pertinent at a national or regional level (Wüst-Galley et al., 2020). In the hemiboreal region of Europe that falls between the boreal and temperate regions, region-specific CO_2 emission factors for cropland and grassland with drained organic soils have not been elaborated so far, due to limited data.

The primary aims of this study were to produce the first estimates on annual net CO_2 fluxes from drained organic soils in cropland and grassland in the Baltic countries, and to elaborate corresponding CO_2 emission factors for this hemiboreal region of Europe. In addition, we evaluated the impacts of organic carbon (OC) concentration in topsoil and other potentially controlling environmental variables on the magnitude of the CO_2 fluxes. The study was conducted at 20 sites covering managed grasslands and croplands with both deep, and shallow highly decomposed, organic soils, grouped depending on the OC concentration in the topsoil layer.

2 Material and methods

2.1 Study sites

The study was conducted in Estonia, Latvia and Lithuania, which are part of the hemiboreal vegetation region of Europe. In total, 20 sites were included in the study (Fig. 1, Table 1, Table S1): 8 croplands (arable land) and 12 grasslands with low management intensity (grazing or fodder production with up to two grass cuttings per year). The sites, established on formerly drained peatlands, included both i) deep organic soils with an OC concentration above 12 % in the 0–20 cm soil layer, and ii) shallow highly decomposed organic soils with an OC concentration below 12 % in the 0–20 cm soil layer. The latter type of soil, in the current classification, does not meet the IPCC criterion for organic soils (Eggleston et al., 2006). The thickness of the soil organic layer ranged from 16 to 72 cm (mean 43 ± SD 19 cm) in cropland and from 17 to 95 cm (mean 46 ± SD 25 cm) in grassland (Table 1). All cropland sites were deep drained (mean WTL > 30 cm) according to the IPCC (Hiraishi et al., 2014), while the grassland sites included both deep (*n* = 10) and shallow drained (mean WTL < 30 cm, *n* = 2) sites (Table 1, Fig. S1). Description of the vegetation species composition in the grassland sites is summarised in Table S2. All study sites represented a steady-state level of land use, i.e., the land had been used for agricultural production for at least the past 20 years. The long-term mean (1991–2020) annual air temperature was 6.3 °C in Estonia, 6.9 °C in Latvia, and 7.4 °C in Lithuania, while the mean annual precipitation was 665 mm in Estonia, 681 mm in Latvia, and 679 mm in Lithuania (Climate Change Knowledge Portal, 2024).

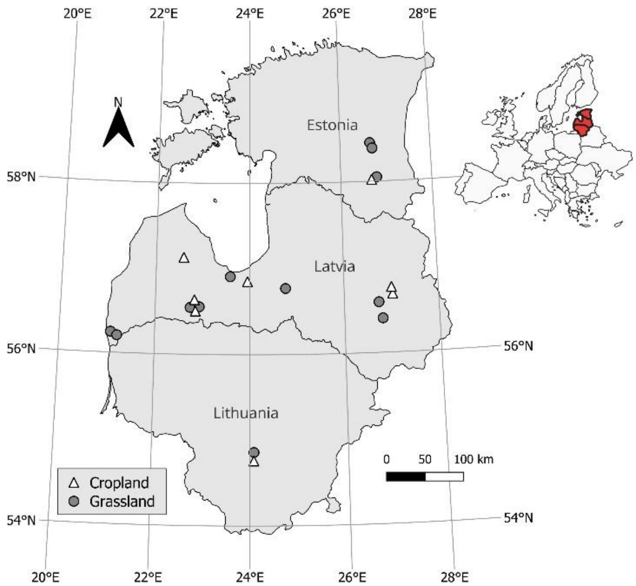

Figure 1: Location of the study sites in the Baltic countries (Estonia, Latvia and Lithuania) belonging to the hemiboreal vegetation region of Europe (maps prepared using QGIS 3.34.4).

Table 1. Description of the study sites with drained nutrient-rich organic soil in agricultural land in the Baltic countries.

Land use type	Country	Study site (name, identification code)*	Soil group (WRB, 2015)	Management during the study period (type of cultivated arable crop/ perennial grass, tillage, N input with fertilisation)	Mean thickness of organic soil layer (range), cm	Mean soil water-table level ± SE (range) during the study period, cm below the surface
Cropland	Latvia	Diervanīne I, CL_LV_1[D]	Histosols	Winter wheat; annual tillage; N input 120 kg N ha^{-1} yr^{-1}	55	87.3 ± 3.9 (12–155)

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		Diervanīne II, CL_LV_2^{D}	Histosols	Maize; annual tillage; N input 120 kg N ha^{-1} yr^{-1}	57	96.2 ± 2.8 (53–160)
		Gaveņpurvs, CL_LV_3^{D}	Histosols	Winter wheat; annual tillage; N input 120 kg N ha^{-1} yr^{-1}	45	41.7 ± 3.3 (−3–93)
		Mārupe, CL_LV_4^{D}	Histosols	Maize; annual tillage; N input 120 kg N ha^{-1} yr^{-1}	72	86.3 ± 2.3 (33–140)
		Lazdiņi I, CL_LV_5^{S}	Gleysols	Winter wheat, winter rape; annual tillage; N input 189 kg N ha^{-1} yr^{-1}	18 (15–20)	59.1 ± 1.3 (30–100)
		Lazdiņi II, CL_LV_6^{S}	Gleysols	Beans; annual tillage; no information on N input	16 (10–21)	54.7 ± 3.4 (1–91)
	Estonia	Saverna I, CL_EE_1^{D}	Histosols	Maize; annual tillage; no information on N input	33 (30–40)	46.7 ± 0.9 (29–78)
	Lithuania	Dobilija, CL_LT_1^{D}	Histosols	Winter wheat, spring wheat, winter rape; no-tillage > 5 years; N input 188 kg N ha^{-1} yr^{-1}	45 (45–45)	> 150 (110–>150)
Grassland	Latvia	Kašķu, GL_LV_1^{D}	Histosols	Perennial grass (managed); no-tillage; no N input	42	91.1 ± 3.3 (1–150)
		Krista, GL_LV_2^{D}	Histosols	Perennial grass (managed); no-tillage; no N input	50	25.5 ± 2.9 (−2–98)
		Stabulnieku, GL_LV_3^{D}	Histosols	Perennial grass (managed); no-tillage; no N input	50	42.2 ± 3.1 (−4–110)
		Rucava, GL_LV_4^{D}	Gleysols	Perennial grass (managed); no-tillage; no N input	31 (30–32)	30.3 ± 2.7 (−3–91)
		Lazdiņi III, GL_LV_5^{S}	Gleysols	Perennial grass (managed); no-tillage; no N input	28 (20–35)	47.7 ± 2.1 (1–85)
		Andrupēni, GL_LV_6^{S}	Phaeozems	Perennial grass (managed); no-tillage; no N input	22 (15–30)	94.2 ± 1.5 (47–127)
		Lazdiņi IV, GL_LV_7^{D}	Phaeozems	Perennial grass (managed); no-tillage; no N input	43 (20–70)	46.3 ± 2.1 (0–125)
		Ķegums, GL_LV_8^{S}	Umbrisols	Perennial grass (managed); no-tillage; no N input	17 (10–25)	83.0 ± 2.3 (0–146)
	Estonia	Maramaa I, GL_EE_1^{D}	Histosols	Perennial grass (managed); no-tillage; no N input	37 (30–40)	22.6 ± 0.9 (−3–51)
		Saverna II, GL_EE_2^{D}	Histosols	Perennial grass (managed); no-tillage; no N input	47 (40–50)	58.4 ± 1.0 (32–84)
		Maramaa II, GL_EE_3^{D}	Histosols	Perennial grass (managed); no-tillage; no N input	92 (75–100)	30.6 ± 1.4 (−1–96)
	Lithuania	Dubrava, GL_LT_1^{D}	Histosols	Perennial grass (managed); no-tillage; no N input	95 (78–120)	43.3 ± 3.7 (−3–150)

* Sites characterised as ‘deep organic soils' are marked with upper index D, while sites characterised as ‘shallow highly decomposed organic soils ' are marked with upper index S.

2.2 Measurements of ecosystem respiration

To estimate ecosystem respiration (R_{eco}), which includes both soil heterotrophic (R_{het}) respiration from organic matter decomposition (including decomposition of the plant residues specific to the study site) and autotrophic respiration of above- and belowground plant biomass, gas sampling was conducted once or twice a month (Table S1). The measurement periods varied between sites, as shown in Table S1, falling between December 2016 and June 2023. One to five plots per site (Table S1) were prepared for gas sampling by installing permanent circular collars (area 0.1995 m^2) in the soil, extending down to 5 cm depth, at least one month before the first gas sampling to avoid the disturbance to the vegetation affecting the results. Gas sampling was conducted using manually-operated closed static opaque chambers (volume 0.0655 m^3). The chambers were positioned air-tightly on the collars, and during the next 30- (Latvia) or 60-minute (Estonia, Lithuania) period, four consecutive gas samples (50 mL) were taken in 10- (Latvia) or 20-minute (Estonia, Lithuania) intervals, respectively, using pre-evacuated (0.3 mbar) glass vials. All measurements were made during daytime, and the time of measurement events was randomised among sites and plots.

The CO_2 concentration in the R_{eco} gas samples was determined using a gas chromatography (GC) method. The gas samples were analysed using Shimadzu GC-2014 (Shimadzu Corporation, Kyoto, Japan) at the Laboratory of the Geography Department, University of Tartu in Estonia, or Shimadzu Nexis GC-230 (Shimadzu U.S.A Manufacturing, Inc., Canby, OR, USA) at the Latvian State Forest Research Institute Silava (LVS EN ISO 17025:2018-accredited laboratory) in Latvia. Both instruments were equipped with an ECD detector. The expanded uncertainty (equal to two times the combined uncertainty) of the method was estimated to be 4.8 % (Magnusson et al., 2017).

Quality control of the data involved the assessment of the fit of the CO_2 concentrations in the gas samples to a linear regression representing the gas concentration change in time within the closed chamber. Data were excluded from further processing if the coefficient of determination (R^2) of the regression was lower than 0.9, except when the difference between the maximum and minimum CO_2 concentration in the four consecutive gas samples of a measurement event was lower than that method uncertainty (20 ppm of CO_2). In total, 6.5% of all instantaneous R_{eco} results were excluded from further data processing based on data quality check.

Instantaneous R_{eco} was calculated based on the equation of Ideal Gas Law using the slope of the linear regression describing the change in the CO_2 concentration over time following Eq. (1):

$$R_{eco} = \frac{M \times P \times V \times Slope}{R \times T \times A \times 1000}, \quad (1)$$

where R_{eco} is instantaneous ecosystem respiration (mg CO_2-C m^{-2} h^{-1}), M is the molar mass of CO_2-C (12.01 g mol^{-1}), P is air pressure in the chamber during sampling (assumption) (101 300 Pa), V is chamber volume (0.0655 m^3); Slope is the slope of the constructed linear regression describing the change in CO_2 concentration over time (ppm h^{-1}), R is the universal gas constant (8.314 m^3 Pa K^{-1} mol^{-1}), T is air temperature (K), and A is collar area (0.1995 m^2).

2.3 Measurements of soil heterotrophic respiration

On 13 study sites (four croplands and nine grasslands), soil heterotrophic respiration (R_{het}) was measured to allow comparison between the use of direct R_{het} measurements versus R_{het} estimates derived from R_{eco} measurements in estimation of annual net CO_2 fluxes. Three measurement points with an area of 0.36 m^2 were established per plot (a total of nine measurement points per study site, Table S1) during the previous growing season before the commencement of R_{het} measurements. Vegetation was removed, soil trenching to a depth of at least 40 cm was done to exclude autotrophic respiration from roots, and a geotextile was installed to prevent new root ingrowth into the measurement points. Soil R_{het} was measured once or twice a month during the vegetation periods (April–November); the measurement periods varied between sites, as shown in Table S1, falling between April 2021 and October 2023. Soil R_{het} was measured using a portable CO_2 gas analyser (EGM-5; P.P. Systems, Amesbury, MA, USA) and opaque fan-equipped chambers with a volume of 0.021 m^3 that covered an area of 0.07 m^2. The chamber was positioned air-tightly on bare soil without a collar.

The duration of each R_{het} measurement was 180 seconds, during which the CO_2 concentration in the closed chamber was recorded every second. Measurement results (CO_2 concentration, ppm) were used to construct linear regressions reflecting changes in CO_2 concentration over time. To avoid the impacts of mechanical disturbances (chamber placement and removal, movement near the chamber) the concentration values recorded during the first 15 and the last 30 seconds of the 180-second measurement period were excluded from the regression, based on results of the method validation. Similarly to R_{eco}, instantaneous R_{het} was calculated based on the equation of Ideal Gas Law using the slope of the constructed linear regression (Eq. (1)).

2.4 Estimation of C stock in above- and belowground parts of vegetation

To estimate the vegetation C stocks, above- and belowground plant biomass was sampled in each plot with at least three replicates (1 m distance between replicates), one to three times per study period. The biomass sampling dates for each study site are summarised in Table S1. The sampling areas represented each R_{eco} measurement plot, avoiding atypical microrelief and vegetation disturbance in the CO_2 flux measurement points (permanent circular collars). The sampling area of aboveground biomass was 625 cm^2 in Latvia, 1600 cm^2 in Lithuania, and 10000 cm^2 in Estonia, and the sampling area of belowground biomass was 625 cm^2 in Latvia, 1600 cm^2 in Lithuania, and 15 soil cores (diameter 48 mm) were randomly sampled per each site in Estonia. The belowground biomass was sampled down to 20–30 cm depth. The samples were brought to the laboratory, where their dry mass was determined after drying at 65–70°C temperature for 48 h or untill a constant mass. Before drying, the belowground biomass samples were cleaned of soil particles by washing with cold tap water and using wet sieving. Total C and nitrogen (N) concentrations in all biomass samples were determined with the elementary analysis method (elemental analyser Elementar El Cube) according to the LVS ISO 10694:2006 and LVS ISO 13878:1998, respectively.

2.5 Soil sampling and analyses

At each plot, the soil was sampled in one to three replicates using a soil sample probe (diameter 5 cm) from the following depths: 0–10 cm, 10–20 cm, 20–30 cm, 30–40 cm, 40–50 cm, 50–75 cm, and 75–100 cm. The soil samples were first pretreated for physico-chemical analyses, including drying at a temperature not exceeding 40 °C and sieving (aperture size of 2 mm) according to the LVS ISO 11464:2005. The following soil variables were then determined: soil pH according to LVS EN ISO 10390:2021 (suspension of soil in 1 mol L^{-1} potassium chloride (KCl) solution, pH KCl; pH-meter Adrona AM 1605); total C (TC) and total N (TN) concentrations by dry combustion according to the LVS ISO 10694:2006 and LVS ISO 13878:1998 (elemental analyser Elementar El Cube); carbonate concentration using a digital soil calcimeter UGT/BD Inventions FOG II Calcimeter Field Kit; ash content according to the LVS EN ISO 18122:2022; and concentrations of HNO_3-extractable potassium (K), calcium (Ca), magnesium (Mg) and phosphorus (P) according to the LVS EN ISO 11885:2009 with the inductively coupled plasma-optical emission spectrometry (ICP-OES) method (Thermo Fisher Scientific iCAP 7200 Duo). Organic C (OC) concentration was calculated as the difference between TC and inorganic C (carbonate) concentration or by multiplying the SOM content derived using results of ash content by a factor of 0.5, thus assuming that SOM is 50 % carbon (Pribyl, 2010). In addition, the soil OC/TN ratio (C/N ratio) was calculated.

2.6 Other environmental variables

Concurrently with the soil respiration measurements, the following environmental variables were measured in each plot: air temperature; soil temperature at depths of 10, 20, 30 and 40 cm using the Comet data logger (COMET SYSTEM, s.r.o., 756 61 Roznov pod Radhostem, Czech Republic) equipped with Pt1000 temperature probes; soil moisture (volumetric water content) and soil temperature at 5 cm depth using the ProCheck meter (Decagon Devices, Pulman, WA, USA) equipped with a moisture sensor (in Estonia and Latvia) and HH2 Hand Held Moisture Meter (Delta-T Devices, Burwell, UK) with SM150T moisture sensor (in Lithuania); soil water-table level (WTL) using groundwater wells (piezometer tubes, 5-7.5 cm in diameter, perforated and coated with nylon mesh) installed vertically down to a depth of 1.5–2.0 m. In addition, continuous soil temperature measurements at depth of 10 cm were carried out at 10 study sites (four croplands and six grasslands), at 30-minute intervals using Maxim Integrated DS1922L2F loggers (iButtonLink Technology, Whitewater, WI 53190 USA).

2.7 Estimation of annual soil net CO_2 fluxes and CO_2 emission factors

Annual net CO_2 fluxes from soil were calculated as the difference between annual CO_2 output (annual soil R_{het}) and annual C input into the soil with above- and belowground parts of vegetation (plant residues). We initially intended to utilise the directly measured R_{het} values for these calculations; however, preliminary analyses showed that the directly measured R_{het} values, which, unlike R_{eco}, do not include autotrophic respiration, were higher than R_{eco} in several study sites (Fig. S2, Fig. S3). Under similar conditions, R_{het} should not be higher than R_{eco}. Using the directly measured R_{het} values would thus overestimate the CO_2 output. The discrepancy between R_{het} and R_{eco} indicates that the variables regulating respiration, such as soil temperature and moisture, critically differed between the respective measurement locations. Also, unlike R_{het}, R_{eco} data was available over the whole years, including the winter season.

Consequently, mean annual soil R_{het} was calculated assuming that i) our R_{eco} is equal to soil surface respiration (R_s), which includes R_{het} and the dark respiration of the belowground plant biomass, and that ii) the proportion of annual soil R_{het} from R_s is 64 %, based on results of previous studies (n=61, Fig. S4) conducted in temperate and boreal regions (Jian et al., 2021). These assumptions were consistent with the most conservative approach and should clearly avoid underestimation of R_{het} since our R_{eco} values additionally included the dark respiration of the aboveground plant biomass, not included in the R_s. The measured R_{eco} values include the CO_2 output due to decomposition of the plant residues specific to the study site. Thus, we did not need to consider the decomposition rates of the plant residues separately, and did not overestimate the C input into the soil by the vegetation.

Annual R_{eco} was calculated for each study site individually as a cumulative value consisting of mean hourly values of R_{eco} multiplied by the number of hours in a day and days in the each month, spanning all months of the calendar year, and the final result is expressed as t CO_2-C ha^{-1} yr^{-1}. The annual CO_2 output from soil (annual soil R_{het}) was then estimated as the 64 % value of the annual R_{eco}.

To assess the potential overestimation of annual R_{eco} due to measurements conducted only during daytime, when the temperature usually is higher than the daily mean temperature, a study-site-specific comparison of the applied method and modelling approach based on continuous soil temperature measurements at depths of 10 cm at 10 study sites was made. The modelling approach included constructing study-site-specific models to describe the relationships between logarithmically or Box-Cox transformed (data normalisation, Box and Cox, 1964) instantaneous R_{eco} and soil temperature at 10 cm depth (Fig. S5, Fig. S6). Hourly R_{eco} estimates were then calculated using the models (Fig. S5, Fig. S6) with the continuous soil temperature data

For cropland, the annual C input into the soil by the vegetation was divided into three components: aboveground harvest residues, belowground harvest residues, and belowground biomass litter. C input from aboveground harvest residues was calculated as the difference between the C stock in the total aboveground biomass and the C stock in harvested products, which was calculated using a harvest index (HI, Table 2), i.e., the ratio of harvested product to total aboveground biomass (Palosuo et al., 2015):

$$Annual\ C\ input_{AGBHR} = C\ stock_{AGB} - (C\ stock_{AGB} \times HI), \quad (2)$$

where Annual C $input_{AGBHR}$ is annual C input from aboveground harvest residue (t C ha^{-1} yr^{-1}), C $stock_{AGB}$ is C stock in total aboveground biomass (t C ha^{-1}), HI is harvest index (Table 2).

For cropland, C input from belowground harvest residues was assumed to equal the C stock in the belowground biomass. For study sites where above- and/or belowground biomass (including belowground biomass litter) was not measured, values derived from earlier research were used (Table 2).

Table 2. The estimated annual C inputs into the soil with above- and belowground parts of vegetation (arable crops, perennial grass) for cases where data was not collected in this study.

Arable crop, perennial grass	Harvest index (HI)	Annual C inputs into soil, t C ha^{-1} yr^{-1}		
		Aboveground harvest residues	Belowground biomass	Belowground biomass litter
Winter wheat	0.39 [a]	3.00 [a]	0.50 [a]	0.21 [a]
Spring wheat	0.44 [a]	2.21 [a]	0.43 [a]	0.18 [a]
Maize	0.84 [b]	0.95 [a,b]	0.72 [a]	0.30 [a]
Beans	0.28 [a]	3.11 [a]	0.23 [a]	0.09 [a]
Rape	0.35 [b]	1.95 [b]	0.58 [b]	0.40 [b]
Fallow	0.00 [a]	1.50 [a]	0.25 [a]	0.10 [a]
Perennial grass	0.84 [b]	0.81 [a,b]	1.14 [a]	0.77 [a]

[a] Source: Latvian State Forest Research Institute “Silava”, 2024
[b] Source: Palosuo et al., 2015

For grassland sites where measured aboveground biomass was left on the field, the C input into soil with aboveground vegetation was assumed to equal the C stock in aboveground biomass measured at the end of vegetation season. For grassland sites where measured aboveground biomass was harvested and removed from the field after biomass measurements were taken, the C input was calculated using the harvest index (HI, Table 2). The C input into the soil with belowground parts of vegetation was calculated assuming that the root turnover rate is 0.41, according to Palosuo et al. (2015). For study sites where data on above- and/or belowground biomass was not available, values summarised in Table 2 were used.

Mean annual net CO_2 fluxes from the soil, corresponding to emission factors as outlined by IPCC, were calculated from the site-level annual net fluxes.

2.8 Statistical analysis

Statistical analyses and visualisation were conducted using the software environment R (version 4.3.3) and RStudio 2023.12.1 (R Core Team, 2024). The datasets of CO_2 fluxes (both R_{eco} and R_{het}) were not normally distributed according to the Shapiro-Wilk normality test, neither when all study sites were pooled nor when each study site was tested separately ($p < 0.001$).

To evaluate the differences between independent variables, for instance, differences in soil physico-chemical variables, R_{eco}, R_{het} and annual net CO_2 fluxes between different types of land use (cropland, grassland), soil types (deep organic soil, shallow highly decomposed organic soil) or drainage (deep drained, shallow drained), Wilcoxon rank sum exact test was used. Plot-level mean values were used when differences in soil physico-chemical variables, R_{eco} and R_{het} between different soil types (deep organic soil, shallow highly decomposed organic soil) or drainage (deep drained, shallow drained) within the same type of land use were estimated. Site-level mean values were used when differences in independent variables between different types of land use (cropland, grassland) were evaluated, as well as when differences in annual net CO_2 fluxes were estimated between different types of land use, soil types or drainage.

Spearman’s correlation coefficient (ρ) was used to assess the degree of dependence between pairs of variables.

For assessing the variation in R_{eco} among sites, the plot-level mean instantaneous R_{eco} (Y) was first calculated from means of instantaneous R_{eco} for each month spanning all months of the calendar year. Partial least squares (PLS) regression, a multivariate method suitable for dealing with variables that are linearly correlated to each other, such as soil physico-chemical variables, was then used. PLS regression analysis includes the evaluation of X variables depending on their importance in explaining Y, expressed as variables important for the projection (VIP values). X variables with VIP values below the threshold of 0.5 were considered insignificant and were not retained in the PLS regression, while X variables with VIP values exceeding 1.0 were considered important.

All statistical analyses were carried out with a significance level of 95 % ($\alpha = 0.05$). Results are expressed as arithmetic mean values ± standard error (SE) unless stated otherwise.

3 Results

3.1 Soil physical and chemical variables

The soils of the study sites were characterised by high variation in both the thickness of the soil organic layer (Table 1) and soil OC concentration, as well as other physico-chemical variables (Fig. 2, Fig. S7, Fig. S8). In the topsoil (0–20 cm layer), OC concentrations ranged from <120 g kg^{-1}, at sites with shallow highly decomposed organic soils, to 527 g kg^{-1} at sites with deep organic soil. In the topsoil of deep organic soils, the mean OC concentration in cropland was 365 ± 59 g kg^{-1}, and in grassland 276 ± 37 g kg^{-1}. In the topsoil of shallow highly decomposed organic soils, the mean OC concentration was significantly lower in both cropland (48 ± 26 g kg^{-1}) and grassland (69 ± 8 g kg^{-1}). Similarly, significantly higher TN concentrations were found in the topsoil of deep organic soils compared to shallow highly decomposed organic soils (25.7 ± 3.9 vs. 3.6 ± 2.3 g kg^{-1} in cropland and 20.0 ± 2.4 vs. 5.7 ± 0.8 g kg^{-1} in grassland, respectively). The mean P concentration was higher in deep organic soils, with a significant difference observed only for grassland. No significant differences in soil C/N ratio were found between deep organic and shallow highly decomposed organic soils; the overall mean soil C/N ratio in the topsoil was 14.4 ± 0.6 in cropland and 13.6 ± 0.8 in grassland. In both cropland and grassland, significantly higher Ca concentrations were observed in the topsoil of deep organic soils compared to shallow highly decomposed organic soils. In contrast, higher K and Mg concentrations were found in the topsoil of shallow highly decomposed organic soils, although a significant difference was observed only for cropland. The mean topsoil bulk density also tended to be higher in shallow highly decomposed organic soils, but a significant difference was observed only for grassland. The mean pH of the topsoil was 6.1 ± 0.3 in cropland and 5.9 ± 0.1 in grassland sites, with no statistically significant differences observed between the two soil types. Similar tendencies in differences in soil physico-chemical variables between deep organic soil and shallow highly decomposed organic soil were also observed for the 20–40 cm and 40–80 cm soil layers (Fig. S7 and Fig. S8).

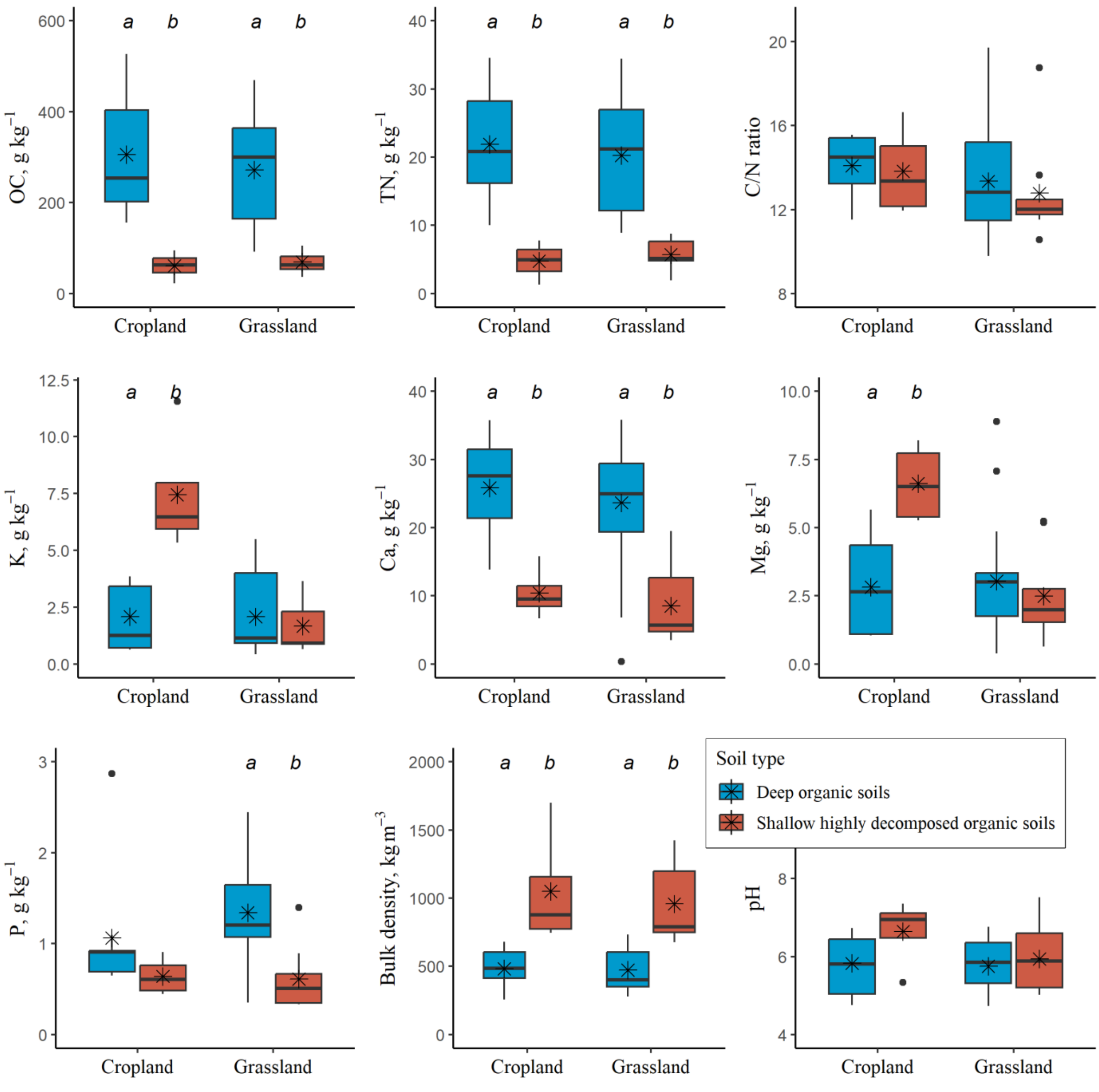

Figure 2: Variation in topsoil (0–20 cm soil layer) characteristics (organic carbon (OC), total nitrogen (TN), organic carbon/total nitrogen (C/N) ratio, HNO_3-extractable potassium (K), calcium (Ca), magnesium (Mg) and phosphorus (P) concentration, soil bulk density, soil pH) in the cropland and grassland sites, separately for the two soil types (deep organic soil and shallow highly decomposed organic soil). In the boxplots, median and mean values are presented as bold horizontal lines and asterisks, respectively; the plot-level mean values were used. The boxes indicate the interquartile range (from 25th to 75th percentiles), the whiskers denote the minimum and maximum values, and the black dots show outliers. Statistically significant differences ($p < 0.05$, Wilcoxon rank sum exact test) between deep organic soil and shallow highly decomposed organic soil within the type of land use are denoted by lowercase letters a and b.

3.2 Ecosystem respiration (instantaneous)

Across the different seasons, the widest variationand the highest ($p < 0.001$) mean intensity of instantaneous R_{eco} was observed in summer (Fig. 3, Fig. S9). Both in cropland and grassland, mean instantaneous R_{eco} decreased in the following order: summer > spring ≈ autumn > winter. The mean instantaneous R_{eco} (mean of monthly means) reflected emissions (CO_2 release into the atmosphere) that ranged among different study sites from 71.9 ± 21.0 to 194.8 ± 79.0 mg CO_2-C m^{-2} h^{-1} in cropland and from 59.9 ± 15.4 to 190.9 ± 60.4 mg CO_2-C m^{-2} h^{-1} in grassland. No statistically significant difference in mean instantaneous R_{eco} between study sites in cropland and grassland was observed ($p = 0.181$).

In cropland, higher ($p < 0.001$) mean instantaneous R_{eco} was observed in sites with shallow highly decomposed organic soils (mean 177.7 ± 17.0 mg CO_2-C m^{-2} h^{-1}) compared to sites with deep organic soils (mean 117.1 ± 13.5 mg CO_2-C m^{-2} h^{-1}). In grassland, the difference between sites with shallow highly decomposed organic soil (129.4 ± 30.8) and sites with deep organic soil (102.5 ± 7.7 mg CO_2-C m^{-2} h^{-1}) was not statistically significant ($p = 0.689$). Additionally, in grassland, the difference in mean instantaneous R_{eco} in deep drained sites (mean WTL > 30 cm) compared to shallow drained sites (mean WTL < 30 cm) (109.6 ± 11.4 vs 107.4 ± 2.1 mg CO_2-C m^{-2} h^{-1}) was not statistically significant ($p = 0.924$).

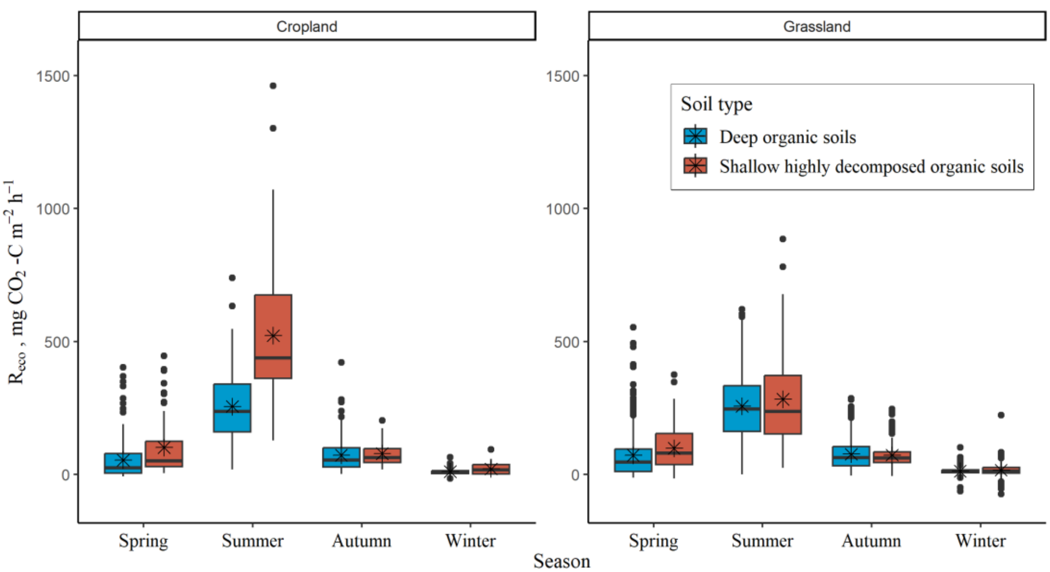

Figure 3: Variation in instantaneous ecosystem respiration (R_{eco}) among different seasons in the cropland (left graph) and grassland (right graph) sites, separately for the two soil types (deep organic soil and shallow highly decomposed organic soil). In the boxplots, median and mean values (bold horizontal lines and asterisks, respectively) calculated from all R_{eco} measurements from all study sites are presented. The boxes indicate the interquartile range (from 25^{th} to 75^{th} percentiles), the whiskers denote the minimum and maximum values, black dots show outliers. Spring – March, April, May; summer – June, July, August; Autumn – September, October, November; Winter – December, January, February (relevant environmental variable data in Fig. S10-S12).

In general, among the environmental variables measured during each gas sampling event (WTL, soil moisture, air and soil temperatures), variation in instantaneous R_{eco} was best described by a polynomial regression where the independent variable was soil temperature at 10-cm depth both in cropland and grassland (47–66 % of the variation explained depending on the type of land use and soil, Fig. 4). Although WTL varied widely during the study period from slightly (3–4 cm) above soil surface to >150 cm below soil surface (Fig. S10), no clear relationship between WTL and instantaneous R_{eco} was observed, neither when data from all study sites were pooled (Fig. S13) nor at single study site level. The response of instantaneous R_{eco} to WTL was highly site-specific and R^2 of site-level polynomial regressions was mostly below 0.25, with some exceptions of higher R^2 showing an increase in instantaneous R_{eco} with higher WTL. Similarly, no clear relationship between soil moisture and instantaneous R_{eco} was observed. However, there were some indications of comparatively lower instantaneous R_{eco} both at very dry and water-saturated conditions (R_{eco} as a function of WTL reflected as a downward-opening parabola, Fig. S13).

In cropland, mean instantaneous R_{eco} was negatively correlated with soil Ca concentration and positively with soil K and Mg concentrations (Table S3). Although a moderate negative correlation between mean instantaneous R_{eco} and soil TC, OC and TN concentrations, and a moderate positive correlation between mean instantaneous R_{eco} and soil bulk density, was also found, these correlations were not statistically significant (Table S3). The PLS analyses that attempted to explain the variation in mean instantaneous R_{eco} among the study sites with the soil physico-chemical variables resulted in a strong model for cropland (number of selected components is 4) with goodness of fit (R^2) of 0.95 and goodness of prediction of 0.66 (Q^2, full cross-validation). The soil physico-chemical variables that best explained the variation (VIP > 1) were the concentrations of K and Ca in the 0–20 and 20–40 cm soil layers. The PLS model also included variables with a VIP > 0.5 (TC, OC, TN, Mg, P concentration, C/N ratio, pH, soil bulk density and thickness of organic soil layer, Table S4). The soil physico-chemical variables that were positively related to the mean instantaneous R_{eco} were K, Mg, and P concentration, as well as pH and soil bulk density, while the other soil physico-chemical variables were related negatively.

In contrast, for grassland, no significant correlations between mean instantaneous R_{eco} and soil physico-chemical variables were found (Table S3). Also, the PLS analyses using soil variables resulted in weak models (R^2 < 0.25).

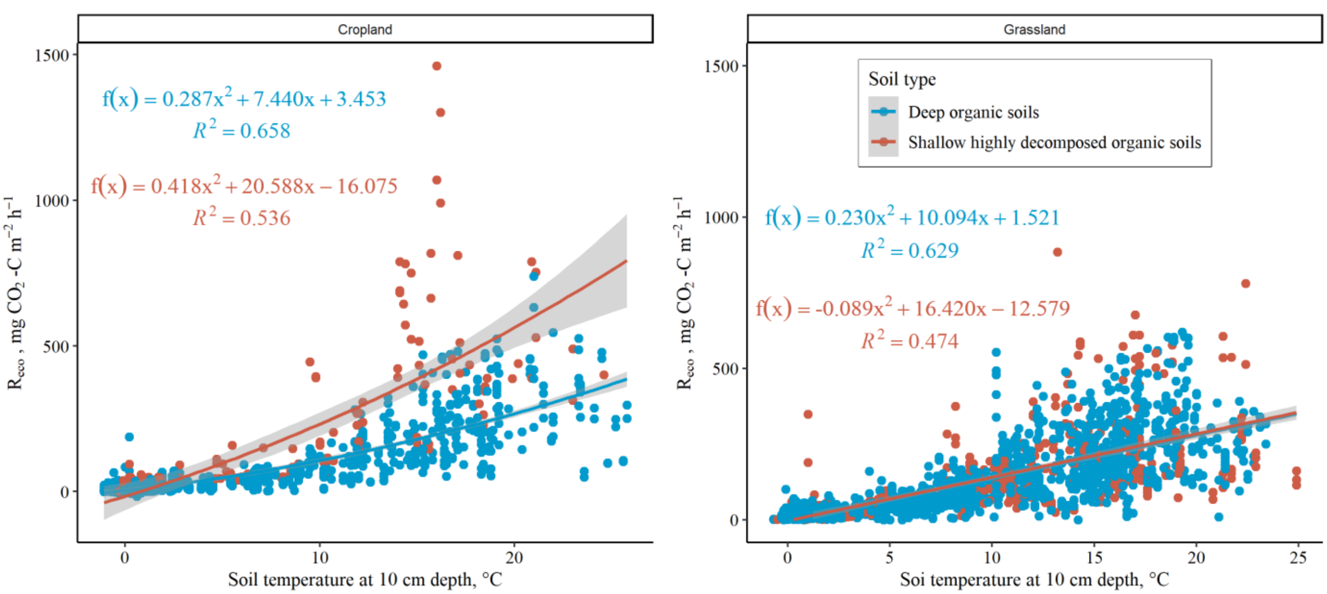

Figure 4: Instantaneous ecosystem respiration (R_{eco}) in cropland (left graph) and grassland (right graph) as a function (polynomial regression) of soil temperature at 10 cm depth measured during each gas sampling event. Data of instantaneous ecosystem respiration is grouped by soil type (deep organic soil denoted by blue colour and shallow highly decomposed organic soil denoted by red colour). The grey area around the regression line reflects the 95% confidence interval of regression.

3.3 Soil heterotrophic respiration (instantaneous)

During the period covered by the measurements (April–November), the widest variation and highest mean intensity of instantaneous R_{het} was observed in the summer months (June–August) both in cropland and grassland (Fig. 5, Fig. S14). Monthly mean instantaneous R_{het} among different study sites ranged from 41.1 ± 5.2 in November to 662.6 ± 69.6 mg CO_2-C m^{-2} h^{-1} in July in cropland and from 18.6 ± 3.2 in November to 652.3 ± 58.1 mg CO_2-C m^{-2} h^{-1} in July in grassland. The mean instantaneous R_{het} (mean of monthly means) ranged among different study sites from 158.4 ± 30.7 to 295.8 ± 72.9 mg CO_2-C m^{-2} h^{-1} in cropland and from 90.0 ± 19.8 to 291.8 ± 56.6 mg CO_2-C m^{-2} h^{-1} in grassland. No statistically significant difference in mean instantaneous R_{het} between cropland and grassland was observed ($p = 0.825$).

In cropland, the overall mean instantaneous R_{het} (mean of monthly means) was significantly higher ($p = 0.009$) in the study sites with shallow highly decomposed organic soils (mean 237.3 ± 58.5 mg CO_2-C m^{-2} h^{-1}) compared to the study sites with deep organic soils (mean 158.8 ± 0.4 mg CO_2-C m^{-2} h^{-1}). No statistically significant differences in mean instantaneous R_{het} were observed for grasslands, neither between soil types nor between deep or shallow drained sites ($p = 0.495$ and $p = 0.743$, respectively). The mean instantaneous R_{het} in grassland was 192.3 ± 25.5 mg CO_2-C m^{-2} h^{-1}.

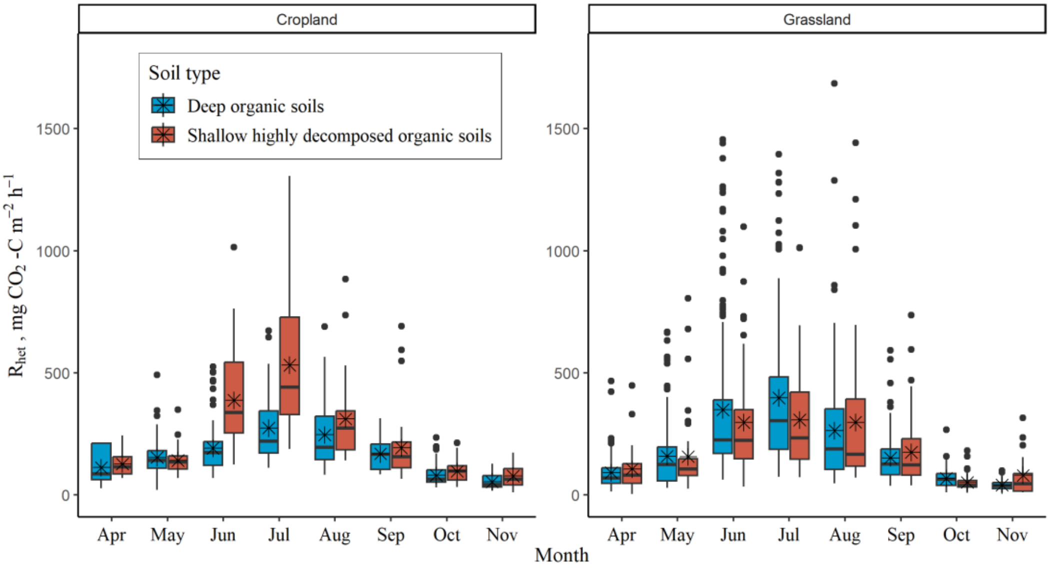

Figure 5: Variation of instantaneous soil heterotrophic respiration (R_{het}) in cropland (left graph) and grassland (right graph) from April to November grouped depending on soil type (deep organic soil and shallow highly decomposed organic soil). In the boxplots, median and mean values (bold horizontal lines and asterisks, respectively) calculated from all performed R_{het} measurements in four study sites in cropland and nine study sites in grassland are presented. The boxes indicate the interquartile range (from 25^{th} to 75^{th} percentiles), the whiskers denote the minimum and maximums values, and the black dots show outliers.

The relationship between instantaneous R_{het} and soil temperature at 10-cm depth differed somewhat between the two soil types (Fig. 6). Further, it was in several cases different from that of R_{eco} (Fig. S2 and Fig. S3). Comparison of instantaneous soil R_{eco} and R_{het} as a function of soil temperature at 10-cm depth showed that instantaneous R_{het} tended to exceed R_{eco} in several study sites.

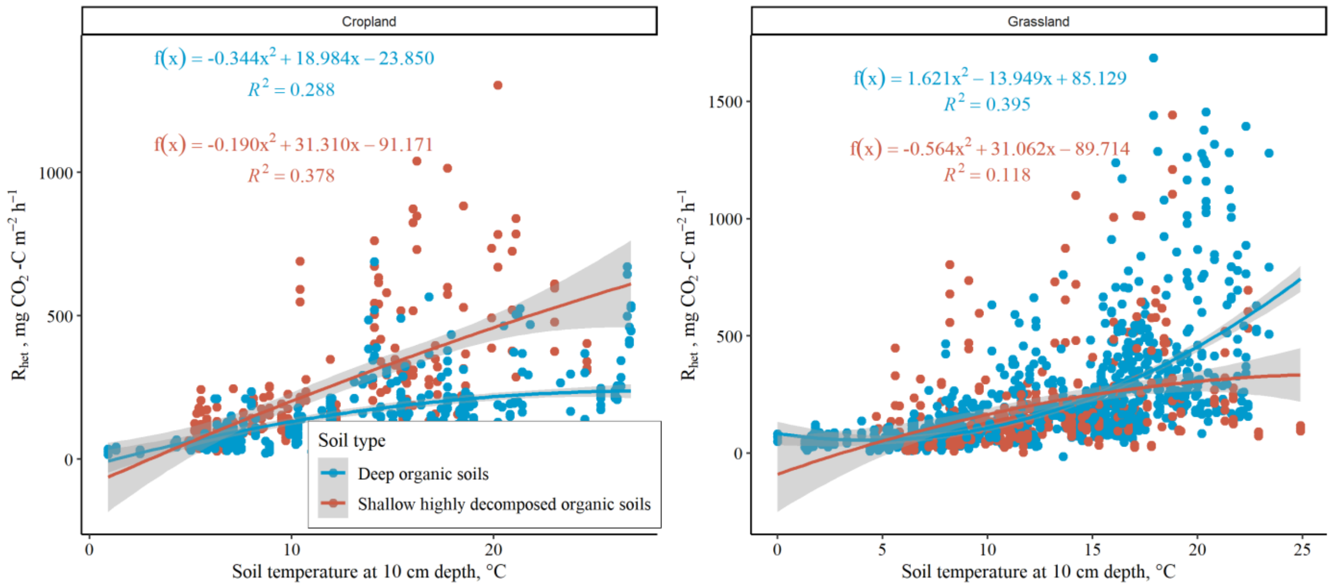

Figure 6: Instantaneous soil heterotrophic respiration (R_{het}) in cropland (left graph) and grassland (right graph) as a function (polynomial regression) of the soil temperature at 10 cm depth. Data of instantaneous soil heterotrophic respiration is grouped depending on soil type (deep organic soil denoted by blue and shallow highly decomposed organic soil denoted by red colour). The grey area around the regression line reflects the 95% confidence interval.

3.4 Carbon stocks in, and inputs into soil through, vegetation

At the end of the growing season, the mean C stock in plant biomass (total including above- and belowground parts) was 4.94 ± 0.55 t C ha^{-1} in cropland and 3.65 ± 0.97 t C ha^{-1} in grassland (Fig. 7). In cropland, the largest part of the C stock was in the aboveground biomass (91 % of total C stock; 4.50 ± 0.49 t C ha^{-1}), while in grassland a larger C stock was found in the belowground biomass (63 % of total C stock; 2.31 ± 0.78 t C ha^{-1}) than in aboveground part (1.26 ± 0.26 t C ha^{-1}). Among the studied arable crops, the largest C stock in plant biomass (total) was estimated for maize (6.7 t C ha^{-1}), and the lowest for spring wheat (2.3 t C ha^{-1}, Fig. S15).

Estimated mean annual C input (total including above- and belowground parts) was 2.65 ± 0.31 t C ha^{-1} yr^{-1} in cropland and 2.35 ± 0.36 t C ha^{-1} yr^{-1} in grassland (Fig. 7). In cropland, the largest annual C input was aboveground harvest residues (74 % of total annual C input; 1.97 ± 0.34 t C ha^{-1} yr^{-1}), while in grassland the amount of belowground inputs (1.19 ± 0.27 t C ha^{-1} yr^{-1}) was similar to the aboveground inputs (1.16 ± 0.26 t C ha^{-1} yr^{-1}). The largest annual C input (total) was estimated for rape (4.1 t C ha^{-1} yr^{-1}), and the lowest for spring wheat (1.5 t C ha^{-1} yr^{-1}, Fig. S15).

The mean concentrations of both C and N were higher in aboveground biomass compared to belowground biomass, while the C/N ratio was higher in belowground biomass both for arable crops and perennial grass (Table S5).

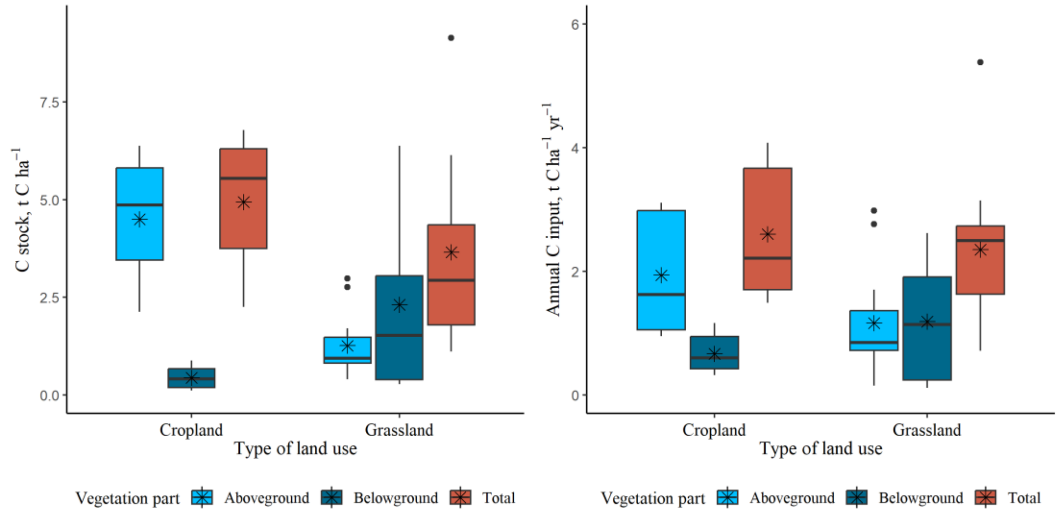

Figure 7: Carbon (C) stock in above- and belowground plant biomass at the end of the growing season (left graph) and annual C inputs into the soil (right graph) in cropland and grassland ecosystems. In the boxplots, the median and mean values are presented by bold lines and asterisks, respectively. The boxes indicate the interquartile range (from 25th to 75th percentiles), the whiskers denote the minimum and maximums values, and the black dots show outliers.

3.5 Annual net CO_2 fluxes

The annual net CO_2 fluxes depended equally on both C losses and C inputs into soil, which both, but in particular the C losses, varied widely. The studied drained organic agricultural soils were all net sources of CO_2, i.e. C losses due to the estimated soil heterotrophic respiration exceeded C inputs into the soil as plant residues (Table 3). The mean annual net CO_2 emissions in cropland and grassland were overall quite similar ($p = 0.270$). In cropland, the mean annual net CO_2 emissions were 4.8 ± 0.8 t CO_2-C ha^{-1} yr^{-1} (all sites pooled); higher net emissions were observed for sites with shallow highly decomposed organic soil (7.0 ± 1.5 t CO_2-C ha^{-1} yr^{-1}) than for deep organic soils (4.1 ± 0.7 t CO_2-C ha^{-1} yr^{-1}). In grassland, the mean annual net CO_2 emissions were 3.8 ± 0.7 t CO_2-C ha^{-1} yr^{-1} (all sites pooled), while, similarly to cropland, higher net emissions were observed for sites with shallow highly decomposed organic soil (mean 5.6 ± 2.1 t CO_2-C ha^{-1} yr^{-1}) than deep organic soils (mean 3.2 ± 0.6 t CO_2-C ha^{-1} yr^{-1}). However, the differences in mean annual net CO_2 emissions between deep organic soils and shallow highly decomposed organic soils were not statistically significant ($p = 0.143$ for cropland sites and $p = 0.209$ for grassland sites). It should be noted, however, that sites with shallow highly decomposed organic soil were relatively less represented in the study. The mean annual net CO_2 emissions from deep-drained and shallow-drained study sites in grassland were similar as well ($p = 0.889$).

The contribution of winter (December–February) CO_2 emissions to total annual R_{eco} was on average 2.4 % in cropland and 3.2 % in grassland, while the contribution of summer (June–August) CO_2 emissions was on average 63.9 % in cropland and 60.3 % in grassland.

Based on the comparative analysis done for the ten study sites for which continuous temperature data were available, our annual R_{eco} estimates were overestimates by a mean of 9 % because of the flux measurements being all done in daytime (Fig. S16).

Table 3. Annual ecosystem respiration (R_{eco}), heterotrophic soil respiration (R_{het}) estimated from R_{eco} (64 % of annual R_{eco} as described in Sect. 2.7), C input into soil as plant residues, and the estimated net soil CO_2 emissions in cropland and grassland in the Baltic countries, hemiboreal region of Europe. The deep organic soils of the sites were either Histosols, Gleysols or Phaeozems, while the shallow highly decomposed organic soils were either Gleysols, Phaeozems or Umbrisols (WRB, 2015).

Study site	Value	R_{eco}, t CO_2-C ha^{-1} yr^{-1}	R_{het}, t CO_2-C ha^{-1} yr^{-1}	C_{input}, t C ha^{-1} yr^{-1}	Net CO_2 emissions, t CO_2-C ha^{-1} yr^{-1}
Type of land use: Cropland					
All sites (*n* = 8)	Mean ± SE	11.7 ± 1.3	7.5 ± 0.8	2.7 ± 0.3	4.8 ± 0.8
	Median	12.3	7.8	2.4	5.0
	Range (min–max)	6.3–17.2	4.1–11.0	1.5–3.7	2.4–8.5
Sites with deep organic soil (*n* = 6)	Mean ± SE	10.3 ± 1.2	6.6 ± 0.8	2.5 ± 0.4	4.1 ± 0.7
	Median	10.6	6.8	2.2	3.6
	Range (min–max)	6.3–13.7	4.1–8.8	1.5–3.7	2.4–6.8
Sites with shallow highly decomposed organic soil (*n* = 2)	Mean ± SE	15.7 ± 1.5	10.0 ± 1.0	3.0 ± 0.5	7.0 ± 1.5
	Median	-	-	-	-
	Range (min–max)	14.2–17.2	9.1–11.0	2.4–3.5	5.5–8.5
Type of land use: Grassland					
All sites (*n* = 12)	Mean ± SE	9.6 ± 0.8	6.2 ± 0.5	2.3 ± 0.4	3.8 ± 0.7
	Median	9.1	5.9	2.5	3.4
	Range (min–max)	5.3–16.8	3.4–10.8	0.7–5.4	0.7–9.7
Sites with deep organic soil (*n* = 9)	Mean ± SE	9.0 ± 0.7	5.8 ± 0.4	2.6 ± 0.5	3.2 ± 0.6
	Median	9.3	6.0	2.7	3.2
	Range (min–max)	5.3–11.3	3.4–7.2	0.7–5.4	0.7–6.5
Deep-drained sites with deep organic soil (*n* = 7)	Mean ± SE	8.9 ± 0.9	5.7 ± 0.6	2.5 ± 0.6	3.2 ± 0.8
	Median	8.3	5.3	2.7	2.2
	Range (min–max)	5.3–11.3	3.4–7.2	0.7–5.4	0.7–6.5
Shallow-drained sites with deep organic soil (*n* = 2)	Mean ± SE	9.5 ± 0.2	6.1 ± 0.1	2.7 ± 0.02	3.3 ± 0.1
	Median	-	-	-	-
	Range (min–max)	9.3–9.6	6.0–6.2	2.7–2.8	3.2–3.4
Sites with shallow highly decomposed organic soil (*n* = 3)	Mean ± SE	11.4 ± 2.7	7.3 ± 1.7	1.7 ± 0.4	5.6 ± 2.1
	Median	9.0	5.8	1.8	3.5
	Range (min–max)	8.3–16.8	5.3–10.8	1.1–2.3	3.5–9.7

4 Discussion

This is the first region-level study to estimate annual net soil CO_2 fluxes from cropland and grassland on drained organic soils in the hemiboreal region. Our study sites covered organic soils with a wide range in both the thickness of the organic soil layer and the OC concentration in the topsoil (0–20 cm). Thus, we could examine separately deep organic soils and soils that we defined as shallow highly decomposed organic soils. Based on existing soil information, we believe that the soils of all our sites were originally (before drainage) deep peat soils. OC concentration of 12 % in 0–20 cm soil layer was set as the threshold value for the definition of organic soils by the IPCC (Eggleston et al., 2006), and our soils with shallow organic layer would not be classified as organic soils according to this definition. Yet, we recorded as high CO_2 emissions from them as from the deep organic soils meeting the threshold. It supported the recent highlight by Liang et al. (2024) of global underestimation of area-scaled CO_2 emissions from drained organic soils undergoing the transition from organic to organo-mineral soils due to agricultural management.

All the studied drained organic soils in cropland and grassland were sources of CO_2 into the atmosphere. In general, our results are in line with the tendency stated by the IPCC (Eggleston et al., 2006; Hiraishi et al., 2014) and several previous studies (Kasimir-Klemedtsson et al., 1997; Alm et al., 2007; Elsgaard et al., 2012; Fell et al., 2016) that mean annual net CO_2 emissions in cropland exceed the net emissions in grassland. Within our study, this tendency was mainly related to the higher mean annual R_{eco} and subsequently the estimated annual R_{het} in cropland than grassland. Further, the belowground C inputs tended to be higher, and the growing period longer under grass compared with arable crops. The slightly higher total C input into the soil as plant residues in cropland did not compensate for the higher R_{het}. However, there was overlap in the site-level values in cropland and grassland, as has been noted in previous studies as well (Couwenberg, 2011).

Although our estimates may contain slight overestimations rather than underestimations, the mean annual net CO_2 emissions of both cropland and grassland with deep-drained organic soil were generally lower than the emission factors provided by the IPCC (Hiraishi et al., 2014) for nutrient-rich soils in the temperate and boreal zones (Table 4). The net CO_2 emissions from shallow highly decomposed organic soils, instead, were rather similar to the IPCC emission factors. Our estimates of annual net CO_2 emissions for grasslands with shallow-drained organic soils were in line with the IPCC emission factor, while for deep-drained sites our estimate was lower but within the confidence interval (Table 4). Interestingly, we found no difference in annual net CO_2 emissions between deep- and shallow-drained organic soils in grassland, although WTL was < 30 cm in 69 % and < 20 cm in 50 % of all measurement events in shallow-drained sites. Distinguishing these is suggested by IPCC (Hiraishi et al., 2014). Logically, WTL should regulate the thickness of the soil layer where efficient aerobic decomposition may take place, and thus the CO_2 flux. However, we only had a two grassland sites with shallow-drained organic soils, which increases uncertainty in the respective emission estimate. Yet, based on current data, the same emission factor could be used for both deep- and shallow-drained grassland in the hemiboreal region.

Table 4. Comparison of estimated and IPCC (Hiraishi et al., 2014) default net CO_2 emissions expressed as emission factors for drained nutrient-rich organic soils in cropland and grassland.

Land use, drainage status	Net CO_2 emissions, t CO_2-C ha^{-1} yr^{-1}				
	Estimates from this study (mean ± SE)			IPCC (Hiraishi et al., 2014) default CO_2 emission factors (95 % confidence interval)	
	Hemiboreal			Boreal	Temperate
	All study sites	Deep organic soil	Shallow highly decomposed organic soil	Deep organic soil	
Cropland	4.8 ± 0.8	4.1 ± 0.7	7.0 ± 1.5	7.9 (6.5–9.4)	
Grassland, deep-drained	3.8 ± 0.7	3.2 ± 0.8	5.6 ± 2.1	5.7 (2.9–8.6)	6.1 (5.0–7.3)
Grassland, shallow-drained		3.3 ± 0.1	-		3.6 (1.8–5.4)

In the past decade, following the development of the latest IPCC default emission factors, several studies have been carried out in temperate and boreal regions, specifically in central and northern Europe. These studies have reported CO_2 emission values that fall within or just beyond the 95 % confidence interval of the IPCC default CO_2 emission factors (Hiraishi et al., 2014). For instance, mean C losses from drained organic soils of 6.45 t C ha^{-1} yr^{-1} were reported for arable land in Germany (Fell et al., 2016), while values between 3.1 and 7.55 t C ha^{-1} yr^{-1} were reported for drained organic soils in grasslands in Switzerland and Germany (Fell et al., 2016; Tiemeyer et al. 2016; Wang et al. 2021). In general, recent studies consistently underscore a notable variability, which can be attributed to the variations in climate and weather conditions, peat chemistry and degree of decomposition, the time since establishment and the maintenance measures of the drainage systems, WTL, and land-management practices and their intensity, including cultivation methods and types of crops grown (Maljanen et al., 2010; Leifeld et al., 2011, Tiemeyer et al., 2016). However, it should also be noted that various methods for measuring or estimating soil C losses are employed across studies, including estimation of peatland subsidence, measuring CO_2 fluxes using chamber methods or eddy covariance as well as by modelling (Kasimir-Klemedtsson et al., 1997; Fell et al., 2016), and all approaches contain some assumptions, advantages and disadvantages, which have to be considered (Kasimir-Klemedtsson et al., 1997; Maljanen et al., 2010; Phillips et al., 2017).

Although the CO_2 emission factors elaborated here for cropland and grassland with drained organic soils are regional in nature, covering the Baltic countries, they provide a general opportunity to improve national GHG inventories and fill knowledge gaps regarding the hemiboreal region of Europe. Further research is needed to elaborate dynamic temperature-dependent CO_2 emission factors, considering that differences in responses of CO_2 fluxes to temperature in different climatic subregions even within the same region or country are possible (Alm et al., 2007). In addition, elaboration of CO_2 emission factors in terms of accuracy would benefit from quantitative separation of R_{het} from R_{eco} in cropland and grassland, where the annual production, rotation and management of plant biomass are highly dynamic and differences in the proportion of R_{het} may be expected. Another aspect to pay attention to when assessing CO_2 emissions in the long-term, specifically in grasslands, is the impact of periodical ploughing of grasslands for renovation by reseeding – a widely used grassland management practice in the hemiboreal region of Europe. Such practices result in additional C inputs to the soil with belowground biomass (plant residues) and consequently can lead to increased R_{het} (Reinsch et al., 2018). In our study, we did not assess the effects of grassland renovation on CO_2 fluxes.

Our flux measurements covered both ecosystem respiration (R_{eco}) and heterotrophic respiration (R_{het}). R_{eco} represented the gross respiration rate: CO_2 produced by the plant–soil system, including soil heterotrophic respiration (aerobic and anaerobic decomposition processes, respiration of soil microorganisms and animals) and autotrophic respiration (CO_2 produced by living plant roots and associated rhizosphere as well as by dark respiration of the plants' aboveground parts) (e.g., Maljanen et al., 2002; Maljanen et al., 2007). Both R_{eco} and R_{het} were primarily regulated by temperature, especially soil temperature at 10 cm depth. Thus, our results align with previous research indicating that soil temperature is the main driver of both ecosystem respiration (Nieveen et al., 2005; Elsgaard et al., 2012) and heterotrophic respiration deriving from peat decomposition (Mäkiranta et al., 2009). For R_{eco} we were able to continue the measurements over the winter seasons. Due to cold temperatures, the contribution of winter (December–February) fluxes was minor, on average 2–3 % in both grassland and cropland. Nevertheless, CO_2 released during winter cannot be disregarded when annual CO_2 emissions are estimated, especially when the soil C balance is close to neutral.

Soil WTL has earlier been found to be the overriding variable explaining GHG emissions when examining a wide range of unmanaged and managed peatlands (Evans et al. 2021). Somewhat surprisingly, we found no clear evidence that the variations in WTL and soil moisture would have an impact on the magnitude of CO_2 fluxes. Yet, there were some indications of comparatively lower instantaneous R_{eco} and R_{het} under both very dry and water-saturated conditions, as the relationship between R_{eco} and WTL exhibited a downward-opening parabola. Similarly, no clear quantitative CO_2 response to WTL among study sites with drained organic soils under agricultural management was found in some previous studies (Nieveen et al., 2005; Elsgaard et al., 2012; Tiemeyer et al., 2016), being explained by the rather deep WTL in the sites, indicating that lack of moisture in the topmost soil layers may restrict R_{het}. There is also some earlier evidence that soil moisture may have a parabolic influence on CO_2 fluxes (e.g. Säurich et al., 2019b). Inconsistent results regarding WTL as a controlling variable of soil respiration can also be obtained due to different hydraulic conductivity of the studied soils (Parmentier et al., 2009). Soil moisture could then, in principle, be a more suitable variable; however, our results do not support that either. Put together, our findings, along with previous studies, suggest that among drained and managed sites - where the mean WTL is generally deeper than 30 cm - WTL may not be the sole linear factor regulating the soil CO_2 fluxes (Fig. S17 and Fig. S18). However, several studies that include study sites with a wider range of mean WTL, including sites where WTL fluctuates close to the soil surface, present WTL-driven response functions (asymptotic relations) for CO_2 emissions (Tiemeyer et al., 2020; Koch et al. 2023). These studies showed that CO_2 emissions increase almost linearly with deeper WTL under shallow drainage (down to around 40 cm threshold) before reaching an asymptotic level (Tiemeyer et al., 2020; Koch et al., 2023).

Higher R_{eco} and R_{het} were observed in shallow highly decomposed organic soil with OC concentration <12 % at 0–20 cm soil layer than in deep organic soils meeting the threshold. This was the case especially in cropland, where the difference was statistically significant. At the same time, no clear (strong and significant) correlation between mean R_{eco} and R_{het} and OC content in soil was identified. In general, our finding of higher R_{eco} and R_{het} in soils with highly decomposed soil organic matter layer is not surprising because some previous studies have highlighted similar tendencies. For instance, Säurich et al. (2019a) concluded that the magnitude of soil-specific basal respiration (CO_2 flux per unit SOC) increased with increasing soil disturbance caused by drainage-induced mineralisation and organic soil layer mixing with mineral soil (i.e. with lower soil OC concentration). Also, other previous studies (Leiber-Sauheitl et al., 2014; Eickenscheidt et al., 2015; Liang et al., 2024) highlighted that the magnitude of CO_2 emissions from drained organic soils used for agriculture was not affected by OC concentration in the soil histic horizon. In contrast, Norberg et al (2016) found significantly lower CO_2 emissions from peaty marls with low total C concentration (9.5–12.2 %) than from peats with much higher total C concentration (27.2–42.8 %). However, our study improves knowledge on soils that may have fulfilled the criteria of organic soil in the past but not any more after long-term land use.

Based on the comparative analysis done in ten sites our annual R_{eco} estimates were overestimated by a mean of 9 % as the flux measurements were all done in daytime (Fig. S16). Previous studies have concluded that the mean CO_2 flux during the daytime is 14–23 % higher than the mean daily fluxes (Maljanen et al., 2002). This is largely caused by diurnal variation in air temperature and consequently soil temperature, which are intercorrelated variables. Thus, a regression describing variation in R_{eco} depending on soil temperature could be used for further evaluations to avoid overestimation of R_{eco} due to lack of measurements during the nighttime. We did not revise our estimates as the comparison could only be done in ten sites.

Contrary to expectations, the magnitude of instantaneous R_{het} tended to exceed the R_{eco} in several study sites. It is inconsistent with the theoretical basis that R_{het} is a part of R_{eco}, and thus, recorded values should be lower than those of R_{eco} simultaneously. The observed inconsistency is most likely explained by methodological challenges. Measurement points established for R_{het} involved trenching, vegetation removal, and keeping the soil surface bare. This may elevate the magnitude of R_{het} firstly by higher temperature in bare soil than under vegetation. Further, soil moisture conditions may differ from vegetated soil. In permanent grassland, emissions from decomposition of the roots killed by the trenching are likely to further contribute to R_{het}. These aspects have also been discussed as challenges of root exclusion also before (Hanson et al., 2000; Kuzyakov, 2006; Norberg et al., 2016; Savage et al., 2018). In general, previous studies on cultivated peat soils in central and southern Sweden suggest that the contribution of R_{het} to cumulative total CO_2 emission (ecosystem respiration) is in the range from 37 to 73 % depending on soil type, crop type and season (Berglund et al., 2011; Norberg et al., 2016; Berglund et al., 2021), while the mean proportion of R_{het} from soil surface respiration is 64 % based on previous studies (n = 61) conducted in agricultural land in temperate and boreal regions (Jian et al., 2021). Considering the previously mentioned, we used R_{het} values derived from the results of R_{eco} for the estimation of annual R_{het} and, subsequently, annual net CO_2 emission from soil. Such an approach was applied to avoid overestimation of R_{het} or C losses from soil. Yet, even the method that we used may result in overestimates, as our R_{eco} values exceed the soil surface respiration because they also account for the dark respiration of the aboveground plant biomass. This “additional” CO_2 flux should logically be at its highest during late summer when the plants are fully developed. However, the share of aboveground autotrophic respiration in ecosystem respiration in cropland or grassland has rarely been reported, and the published results vary widely and have relatively large uncertainties (Phillips et al., 2017). Consequently, we could not estimate how much it contributed to our ecosystem flux. The estimated mean annual C input into the soil with vegetation (residues of above- and belowground parts returned to the soil) was comparatively similar in cropland and grassland, while the proportion of C input with above- and belowground litter differed – in cropland, significantly higher C input was from residues of aboveground part of plants, while in grassland even slightly higher C input was from belowground plant residues. Concerning plant residue inputs into soil, our results follow the previous finding that plant aboveground biomass tends to have a lower C/N ratio and, therefore, be more labile and decompose faster than their belowground counterparts (Almagro et al., 2021). At the same time, both the above- and belowground biomass of arable crops had a higher C/N ratio than grass biomass, potentially indicating slower decomposition of residues in arable crops. Thus, differences in C/N ratio of plant residues as well as the proportion of residues of the plant above- and belowground parts may introduce differences in the response of soil microbial community through altering decomposition and consequent OC incorporation in stable soil aggregates (Almagro et al., 2021). Nevertheless, previous findings have indicated that the total variation of CO_2 emissions from drained organic soil exceeds the variation between different cropping systems and, thus, the selection of certain arable crops has not become a viable option to reduce CO_2 emissions from cultivated organic soils thus far (Norberg et al., 2016).

Conclusions

This study examined the CO_2 fluxes and estimated annual net CO_2 emissions from drained nutrient-rich organic soils (both deep organic soils and shallow highly decomposed organic soils) in cropland and grassland in the Baltic countries (hemiboreal region of Europe). The intensity of both R_{eco} and R_{het} was strongly dependent on temperature (particularly soil temperature at 10 cm depth), while it was rather independent of water-table fluctuations and soil moisture. Although the results obtained within this study may contain slight overestimation rather than underestimation, our estimates of annual net CO_2 emissions both in cropland and grassland were lower than the IPCC default emission factors for the temperate climate/vegetation zone (Hiraishi et al., 2014). This highlights the need to specify emission factors for smaller, climatically and perhaps geomorphologically more uniform regions than the very wide regions for which the current emission factors are available. Mean annual net CO_2 emissions from pooled data recorded in our study were 4.8 ± 0.8 t CO_2-C ha^{-1} yr^{-1} in cropland and 3.8 ± 0.7 t CO_2-C ha^{-1} yr^{-1} in grassland, while the mean annual net CO_2 emissions specifically for deep organic soil were 4.1 ± 0.7 t CO_2-C ha^{-1} yr^{-1} in cropland and 3.2 ± 0.6 t CO_2-C ha^{-1} yr^{-1} in grassland. Both annual R_{eco} and R_{het} as well as the net CO_2 emissions from shallow highly decomposed organic soils were of similar magnitude or higher than from deep organic soils. This result highlights the need to estimate the emissions from these highly transformed soils as organic soils rather than as mineral soils, even though they do not fulfil the current IPCC definition of organic soils (Eggleston et al., 2006). A clear advantage of our study was that we were able to include several sites where comparable measurements were carried out, which allows rigorous inter-site comparison and search for explanatory variables. However, we recommend both continuation of data acquisition, including higher measurement intensity, and consequent further refinement of the first hemiboreal region-specific CO_2 emission factors that we here defined for national GHG inventories.

Data availability

Data used for estimation of annual net CO_2 fluxes is available at DOI: https://zenodo.org/records/14988737. Additional data can be provided by the corresponding authors upon request.

Author contribution

Conceptualisation: AL, RL, JJ, KS; Methodology: KS, JJ, AL, IO and KA; Formal analysis: ABā, ABu; Investigation: ABu, DČ, AK, MM, IO, GRO, MKS, TS, HV, EV; Resources: AL, KS, KA; Writing - Original Draft: ABā; Writing - Review & Editing: all authors; Visualisation: ABā; Supervision: RL, JJ, KS, AL, KA, IL; Project administration: IL; Funding acquisition: AL, KS, RL, JJ, KA, IL.

Competing interests

The authors declare that they have no conflict of interest.

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