# 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, ~~categorized~~ categorised based on the concentration of organic carbon in the top 20-cm soil layer. ~~In each site, $CO_2$ flux measurements were conducted at least over two years~~ $CO_2$ 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 ~~$CO_2$~~ output ($R_{het}$) and ~~annual C~~ input (plant residues), ~~was~~ were 4.8 ± 0.8 t $CO_2$-C ha$^{-1}$ yr$^{-1}$ ~~in~~ for croplands and 3.8 ± 0.7 t $CO_2$-C ha$^{-1}$ yr$^{-1}$ ~~in~~ 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 ~~recognized~~ 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~~a~~). 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~~s~~ 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~~a~~). Thus, these soils ~~are~~ 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~~ the~~ 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. ~~For~~ To take effective and practical ~~mitigation actions~~measures to reduce emissions, it ~~needs to be~~is necessary to know~~n~~ where and why the emissions are highest, and to understand how they respond to changes in the ~~factors~~ environmental ~~parameters~~variables regulating them.

It is well documented that improved soil aeration caused by lowering the soil water-table level (WTL) through ditch drainage~~,~~ ~~and~~ along with mechanical ~~disturbance~~ practices (e.g., repeated ploughing), as well as the application of ~~liming~~ lime and fertilizers~~fertilizationfertilisation~~, ~~improve~~ enhance the conditions for SOM ~~mineralization~~ 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 ~~factors~~variables, including local climate and physical soil conditions (mainly ~~soil~~ 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~~factors~~ ~~parameters~~ ~~in effect~~ and their ~~potential~~ interactions, as well as variation in management practices and intensity, there is still a rather limited number of ~~sites~~studies that provide comprehensive information on the annual net $CO_2$ fluxes from drained organic soils used for agriculture ~~from a rather limited number of sites~~. 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~~, but~~. Still ~~currently,~~ there is too little data to adjust the emission factors correspondingly. Further, many ~~of the GHG emission~~ 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).

~~Only a few studies have highlighted the important contribution of organic soils with comparatively low SOC concentration (<15.0 %, Tiemeyer et al., 2016),, and even soils not falling under the definition of organic soils provided by the IPCC (Eggleston et al., 2006), to total GHG emissions (Leiber-Sauheitl et al., 2014; Eickenscheidt et al., 2015; Liang et al., 2024).~~

These soils include formerly drained peatlands ~~undergoing transformation into~~ 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 ~~actually~~ higher.

In the Baltic ~~states~~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 ~~the relevant $CO_2$ emissions from organic soils account for~~ ~~correspond to emissions of~~ 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 ~~factors~~variables affecting them need to be quantified under climatic and management conditions that are pertinent at a national or regional level ~~relevant nationally or at least regionally~~ (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 ~~insufficient data availability~~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 ~~states,~~ 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 ~~study~~ 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 ~~study~~ sites were included in the study (Fig. 1, Table 1, Table S1): 8 ~~sites in~~ croplands (arable land) and 12 ~~sites in~~ grasslands with low management intensity (grazing or fodder production ~~that involves~~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 ± S~~.~~D~~E.~~ ~~7~~19 cm) in cropland and from 17 to 95 cm (mean 46 ± S~~.~~D~~.~~ 25~~7~~ 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 ~~drained~~ ($n$ = 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 ~~summarized~~ 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 ~~average~~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 ~~average~~mean annual precipitation was 665 mm in Estonia, 681 mm in Latvia, and 679 mm in Lithuania (Climate Change Knowledge Portal, ~~2023~~2024).

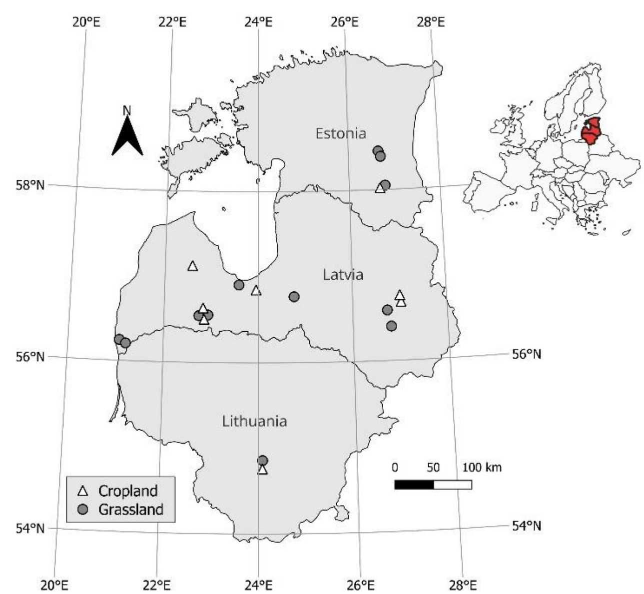



**Figure 1:~~.~~ Location of the study sites in the Baltic countries ~~States~~ (Estonia, Latvia and Lithuania) belonging to the hemiboreal vegetation region of Europe (maps prepared using QGIS 3.34.4).**

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

| Land use type | Country | Study site (name, identification code)* | Soil group (WRB, 2015~~4~~) | Management during the study period (type of cultivated arable crop/ perennial grass, tillage, N input with ~~fertilization~~fertilisation) | Mean thickness of organic soil layer (range), cm | Mean soil water-table level ± S~~.~~E~~.~~ (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) |
| | | 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 ~~characterized~~ characterised as ‘deep organic soils' are marked with upper index D, while sites ~~characterized~~ 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 ~~five~~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~~, Lithuania~~) or 60-minute (Estonia, Lithuania) period, four consecutive gas samples (~~100~~ 50 mL~~cm$^3$~~) were taken in 10- (Latvia~~, Lithuania~~) or 20-minute (Estonia, Lithuania) intervals, respectively, using ~~underpressurized~~ pre-evacuated (0.3 mbar) glass vials. All measurements were made during daytime, ~~randomizing~~ ~~randomising~~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 ~~analyzed~~ analysed using Shimadzu GC-2014 ~~equipped with ECD detector~~ (Shimadzu Corporation, Kyoto, Japan) at the Laboratory of the Geography Department, University of Tartu in ~~(~~Estonia,~~)~~ ~~and~~ or Shimadzu Nexis GC-230 ~~equipped with ECD detectors~~ (Shimadzu U.S.A Manufacturing, Inc., Canby, OR, USA) ~~and LabSolutions software 5.93~~ 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)~~20 ppm of $CO_2$~~.

### ~~2.3 Flux calculations and data quality check~~

Quality control of the data ~~(GC results)~~ 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 ~~data~~ 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):

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

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~~4~~ 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 (~~i.e.,~~a total of nine measurement points per study site, Table S1) ~~with an area of 0.36 $m^2$~~ during~~were established in~~ the previous growing season ~~prior~~before the commencement ~~measurements~~ of ~~soil heterotrophic respiration (~~$R_{het}$~~) were started~~ measurements. Vegetation was removed, soil trenching to a depth of at least 40 cm was done to exclude autotrophic respiration from ~~existing~~ 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 ~~(for the measurement periods, see Table S1)~~ using a portable $CO_2$ gas ~~analyzer~~ analyser (EGM-5~~,~~; P.P. Systems, Amesbury, MA, USA) and opaque fan-equipped chambers with a~~-~~ volume of ~~0.017 or~~ 0.021~~3~~ $m^3$ that covered an area of 0.07 $m^2$. ~~$R_{het}$ measurements were conducted by~~The chamber was positioned~~ing a chamber (area 0.07 $m^2$) open lower edge~~ 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 ~~possible~~ the impacts of mechanical disturbances ~~(that couldimpact on affect $CO_2$ concentration changes over the time) – stemming from due to from (~~chamber placement~~,~~ and removal, ~~or positioning, removing of and~~ movement near the chamber~~)~~,~~ –~~ the concentration values ~~obtained~~ recorded during the first 15 and the last 30 seconds of the 180-second measurement period ~~(180 seconds in total)~~ were excluded from the regression, ~~(~~based on results of~~btained during~~ the method validation~~)~~. Similarly to $R_{eco}$, instantaneous $R_{het}$ was calculated based on the equation of Ideal G~~g~~as L~~l~~aw using the slope of the constructed linear regression (Eq. (1)).

### 2.4~~5~~ 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 ~~once to thrice~~ per study period. The biomass sampling dates for each study site are ~~summarized~~ summarised in Table S1. The sampling areas ~~(1 m distance between replicates) were representative for~~represented each ~~the~~ $R_{eco}$ measurement plot, avoiding atypical microrelief and ~~disturbance of vegetation~~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 ~~part~~ biomass was sampled ~~by excavating roots~~ down to 20–30 cm depth. ~~Vegetation~~ The samples were ~~transported~~ brought to the laboratory, ~~and~~ where their dry mass was determined after drying at 65–70°C temperature for 48 h or untill a constant mass ~~was reachedachieved~~. Before drying, the ~~samples of~~ 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 ~~analyzer~~ analyser Elementar El Cube) according to the LVS ISO 10694:2006 and LVS ISO 13878:1998, respectively.

### 2.5~~6~~ 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 ~~analyzer~~ 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~~7~~ 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 ~~of 5,~~ 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~~1.6~~ m. In addition, continuous soil temperature measurements at depth of 10 cm were carried out~~on~~ 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~~8~~ 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 ~~utilize~~ 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 $R_{eco}$. ~~Use of the directly measured $R_{het}$ values would thus have resulted~~Using the directly measured $R_{het}$ values would thus overestimate ~~in overestimation of~~ the $CO_2$ output. The discrepancy between $R_{het}$ and $R_{eco}$ indicates that the ~~parameters~~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 ~~separately~~ 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 ~~respective~~ each month, spanning ~~covering~~ all months ~~in~~ 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 ~~measured~~ total aboveground biomass and the C stock in harvested products, ~~(Eq. 2). C stock in harvested products~~which was calculated using a harvest index (HI, Table 2), ~~which is~~i.e., the ratio of harvested product to total aboveground biomass (Palosuo et al., 2015)~~, resulting in Eq. (2)~~:

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

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 ~~be equal to~~equal the C stock in the ~~measured~~ belowground biomass. ~~The assumptions of annual C input into the soil with belowground biomass litter are summarized summarised in Table 2.~~

For study sites where ~~data on~~ above- and/or below-ground biomass (including belowground biomass litter) was not ~~available~~measured, ~~theoretical~~ values derived from earlier research ~~summarized summarised in Table 2~~ 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) ~~in~~ for cases where ~~for which no~~ 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, ~~it was either assumed that~~ the C input into soil with aboveground ~~parts of~~ vegetation was assumed to equal~~led~~ the C stock in aboveground biomass measured ~~in~~ 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, ~~, or~~ the C input was calculated using the harvest index (HI, Table 2)~~, depending on the study site and management practicees.~~. 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 ~~summarized~~ 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~~9~~ Statistical analysis

Statistical analyses and ~~visualization~~ 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 ~~results of the~~ Shapiro-Wilk normality test, ~~both~~ neither when all study sites were pooled ~~and~~ 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, $R_{eco}$ ~~and~~, $R_{het}$ 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 ~~and pairwise comparisons using Wilcoxon rank sum test with continuity correction was used;~~on ~~plot mean values~~ wereas used ~~for analysis~~. Plot~~Site~~-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. ~~between different types~~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 ~~(cropland, grassland)~~, soil types ~~(deep organic soil, shallow highly decomposed organic soil)~~ or drainage ~~(deep drained, shallow drained)~~.

Spearman's correlation coefficient (ρ) was used to assess the degree of dependence between pairs of variables. ~~To explain the variation in mean instantaneous $R_{eco}$ (Y) calculated as the average of monthly means (Y) among study sites (plot-level mean values were used),~~ 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. P~~p~~artial 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 ~~as~~ insignificant and were not ~~used~~ retained in the PLS regression, while X variables with VIP values exceeding 1.0 were considered ~~as~~ important.

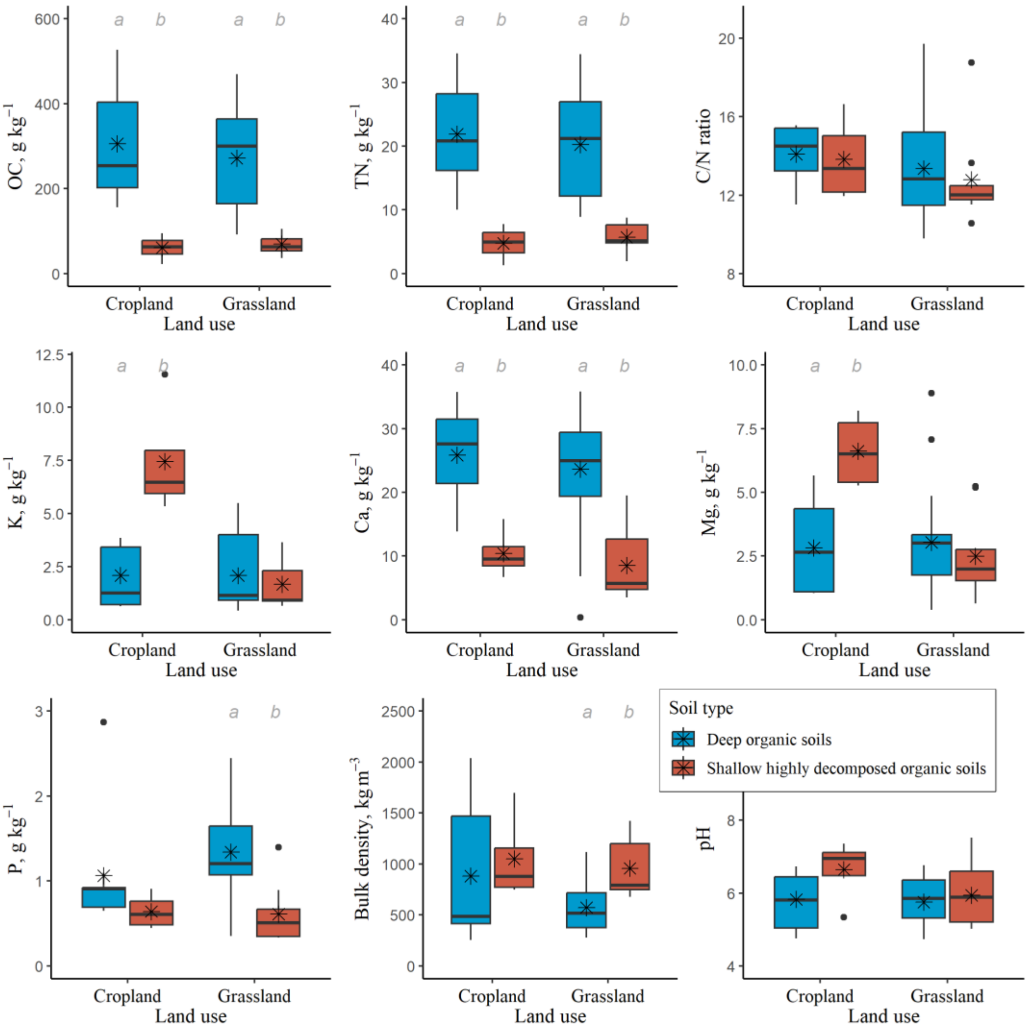
All statistical analyses were carried out with a significance level of 95 % ($\alpha = 0.05$). Results are expressed as arithmetic mean values ± standard error (S~~.~~E~~.~~) unless stated otherwise.

## 3 Results

### 3.1 Soil physical and ~~o-~~chemical variables

The soils of the study sites were ~~characterized~~ 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~~5~~, Fig. S8~~6~~). In the topsoil (0–

20 cm layer), OC concentrations ranged from <120 g $kg^{-1}$, ~~in~~ at sites ~~where the mean thickness of the soil organic layer was <30 cm and soil organic matter was highly mineralized mineralised and mixed with the underlying mineral soil as a result of soil ploughing (, i.e., the~~with shallow highly decomposed organic soils~~)~~, to 527~~6.8~~ g $kg^{-1}$ ~~in~~ at sites with deep organic soil. In the topsoil of deep organic soils, the mean OC concentration in cropland was 365~~.0~~ ± 59~~.2~~ g $kg^{-1}$, and in grassland 276~~.0~~ ± ~~36.8~~37 g $kg^{-1}$. In the topsoil of shallow highly decomposed organic soils, the mean OC concentration was significantly lower in both cropland (~~up to 73.7~~48 ± 26~~11.8~~ g $kg^{-1}$) and grassland (~~up to 81.9~~69 ± ~~22.7~~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~~s~~ ~~tended to be~~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~~; no significant differences between the two soil types were found~~. 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. S~~5~~7 and Fig. S~~6~~8).


OC, g kg$^{-1}$
TN, g kg$^{-1}$
C/N ratio
K, g kg$^{-1}$
Ca, g kg$^{-1}$
Mg, g kg$^{-1}$
P, g kg$^{-1}$
Bulk density, kg m$^{-3}$
pH
Cropland
Grassland
Land use
a
b
Soil type
Deep organic soils
Shallow highly decomposed organic soils

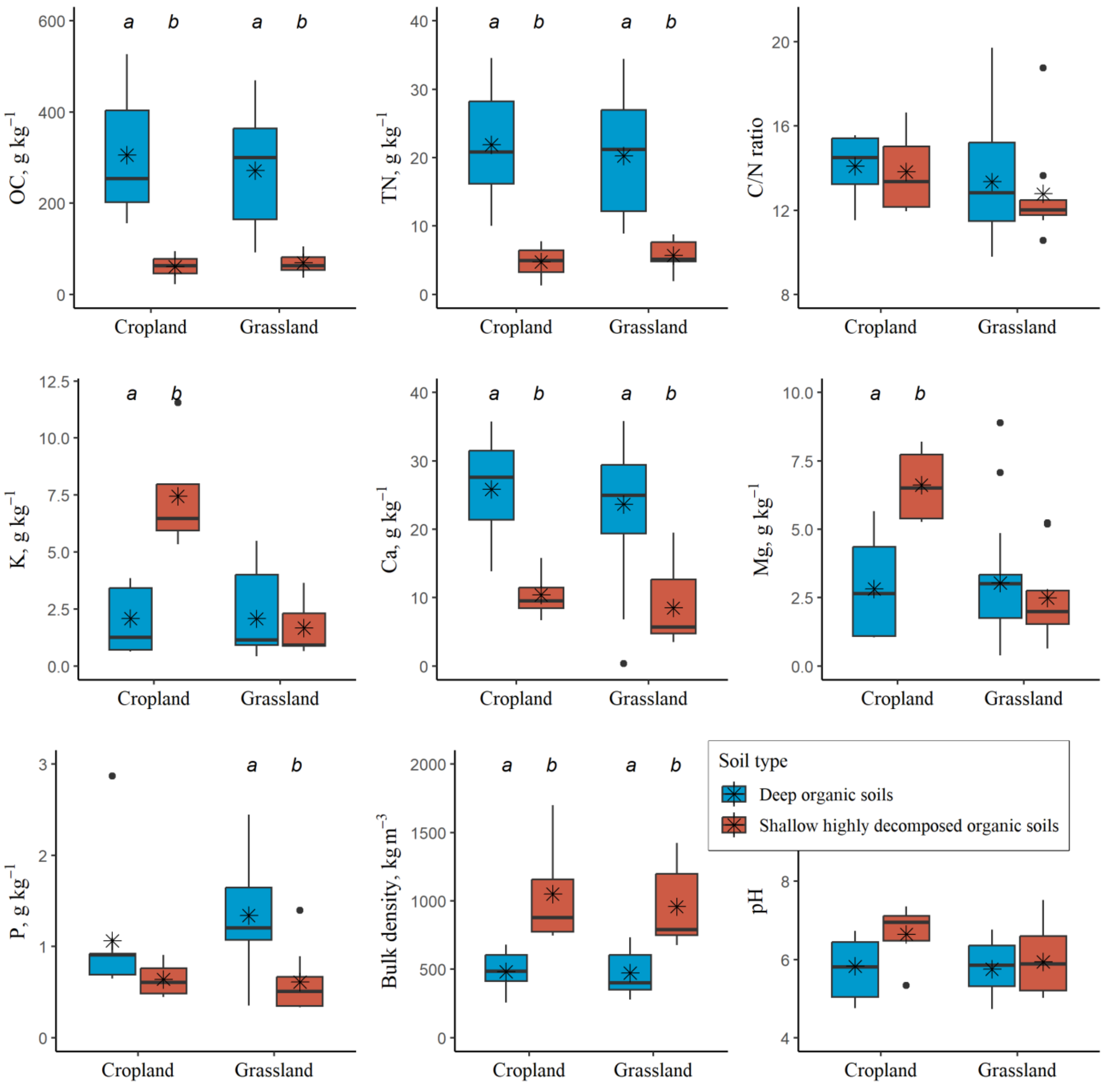



**Figure 2~~.~~: Variation ~~in~~ ~~of~~in topsoil (0–20 cm soil layer) characteristics (organic carbon (OC), total nitrogen (TN), organic carbon/total nitrogen (C/N) ratio, $HNO_3$-extractable potassium (K), calcium (Ca), magnesium (Mg) and phosphorus (P) concentration, soil bulk density, soil pH) in 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 ~~(bold horizontal lines and asterisks, respectively)~~ are presented as bold horizontal lines and asterisks, respectively; ~~plot~~ the plot-level mean values were used ~~for data analysis~~. The boxes indicate the interquartile range (from $25^{th}$ to $75^{th}$ 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)

~~Among~~ Across the different seasons, the widest variation ~~as well as the highest ($p < 0.001$), as well as~~and the highest ($p < 0.001$) mean intensity of instantaneous $R_{eco}$ was observed in summer (Fig. 3, Fig. S9~~7~~). Both in cropland and grassland, mean instantaneous $R_{eco}$ decreased in the following order: summer > spring ≈ autumn > winter. The mean instantaneous $R_{eco}$ (mean~~average~~ 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~~319~~).

In cropland, ~~a significant difference ($p < 0.001$) in mean instantaneous $R_{eco}$ was observed between study sites with deep organic soil and shallow highly decomposed organic soil. H~~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 ~~Although, A similar tendency of a higher mean instantaneous $R_{eco}$ was observed in~~ sites with shallow highly decomposed organic soil ~~was observed also in grassland~~ (~~respectively,~~ 129.4 ± 30.8) and sites with deep organic soil ~~vs.~~ (102.5 ± 7.7 mg $CO_2$-C m$^{-2}$ h$^{-1}$)~~, but the difference~~ was not statistically significant ($p$ = 0.689). Additionally, in grassland, ~~a slight tendency of a higher~~the difference in mean instantaneous $R_{eco}$ in deep drained sites (mean WTL > 30 cm) compared to shallow drained sites (mean WTL < 30 cm) ~~was observed~~ (~~respectively,~~ 109.6 ± 11.4 vs~~.~~ 107.4 ± 2.1 mg $CO_2$-C m$^{-2}$ h$^{-1}$)~~, but the difference~~ was not statistically significant ($p$ = 0.924).

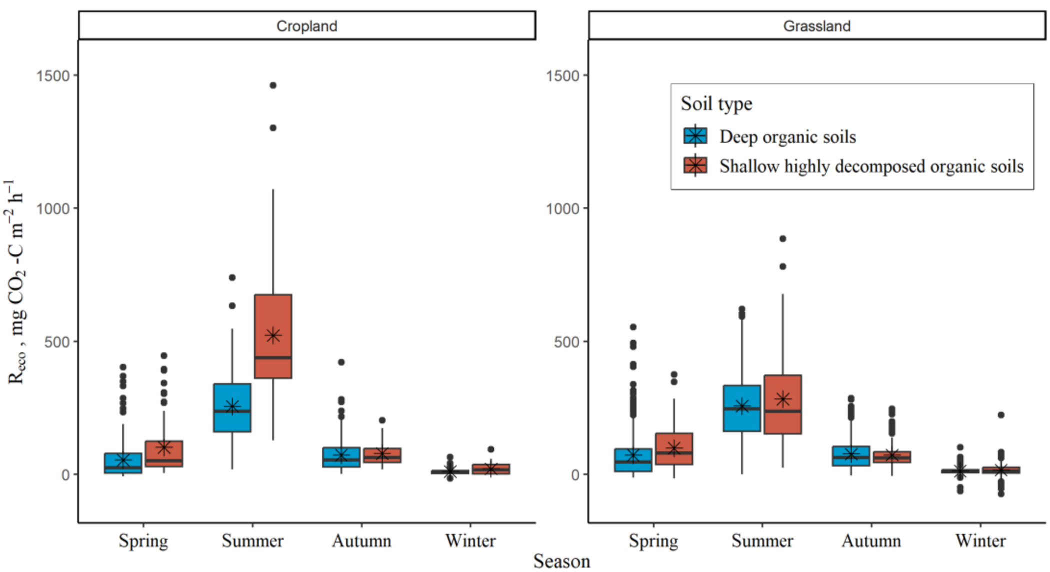


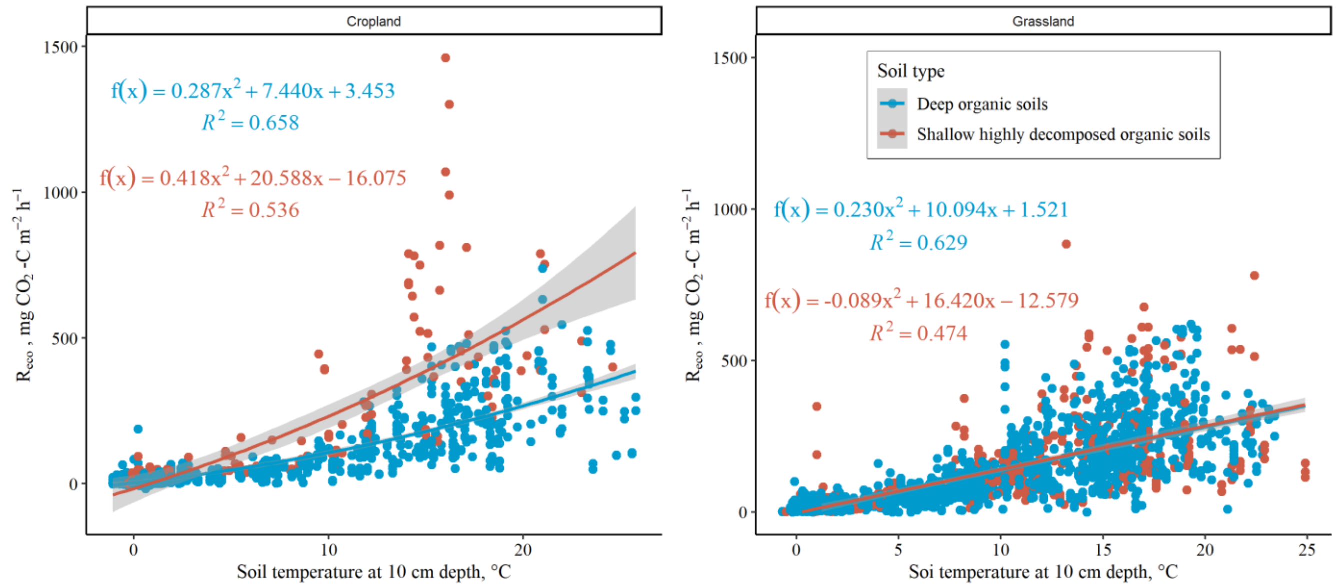

**Figure 3:~~.~~ Variation in instantaneous ecosystem respiration ($R_{eco}$) among different seasons in the cropland (left graph) and grassland (right graph) sites, separately for the two soil types (deep organic soil and shallow highly decomposed organic soil).** In the boxplots,

median and mean values (bold horizontal lines and asterisks, respectively) calculated from all ~~performed~~ $R_{eco}$ measurements from all study sites ~~and plots~~ are presented. The boxes indicate the interquartile range (from 25th to 75th 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).

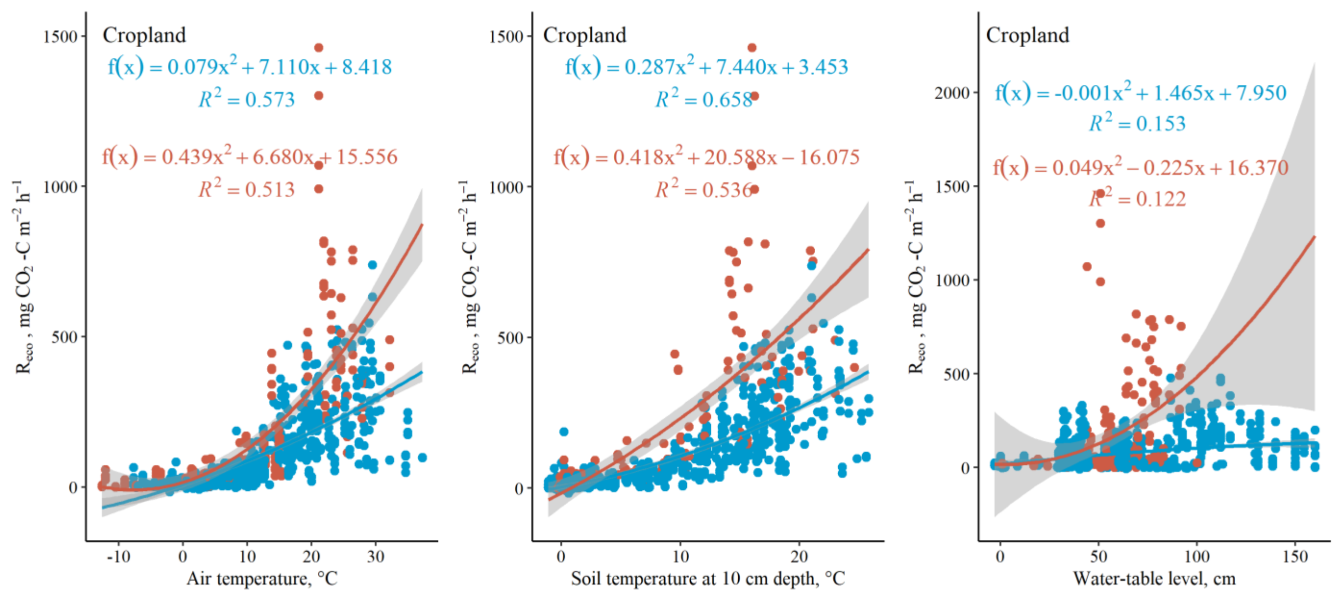
In general, among the environmental variables measured during each gas sampling event (WTL, soil moisture, air and soil temperatures), variation in instantaneous $R_{eco}$ was best described by a polynomial regression where the independent variable was soil temperature at 10-cm depth both in cropland 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~~4~~) nor at single study site level. The response of instantaneous $R_{eco}$ to WTL was highly ~~site~~ 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 ~~water table level~~WTL reflected as a downward~~s~~-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 0.9~~9~~5 and goodness of prediction of 0.~~75~~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).



Cropland
Grassland
Soil type
Deep organic soils
Shallow highly decomposed organic soils
$f(x) = 0.287x^2 + 7.440x + 3.453$
$R^2 = 0.658$
$f(x) = 0.418x^2 + 20.588x - 16.075$
$R^2 = 0.536$
$f(x) = 0.230x^2 + 10.094x + 1.521$
$R^2 = 0.629$
$f(x) = -0.089x^2 + 16.420x - 12.579$
$R^2 = 0.474$
$R_{eco}$, mg $CO_2$-C $m^{-2}$ $h^{-1}$
Soil temperature at 10 cm depth, °C
Soi temperature at 10 cm depth, °C

Cropland
$f(x) = 0.079x^2 + 7.110x + 8.418$
$R^2 = 0.573$
$f(x) = 0.439x^2 + 6.680x + 15.556$
$R^2 = 0.513$
Cropland
$f(x) = 0.287x^2 + 7.440x + 3.453$
$R^2 = 0.658$
$f(x) = 0.418x^2 + 20.588x - 16.075$
$R^2 = 0.536$
Cropland
$f(x) = -0.001x^2 + 1.465x + 7.950$
$R^2 = 0.153$
$f(x) = 0.049x^2 - 0.225x + 16.370$
$R^2 = 0.122$
$R_{eco}$, mg $CO_2$-C $m^{-2}$ $h^{-1}$
Air temperature, °C
Soil temperature at 10 cm depth, °C
Water-table level, cm

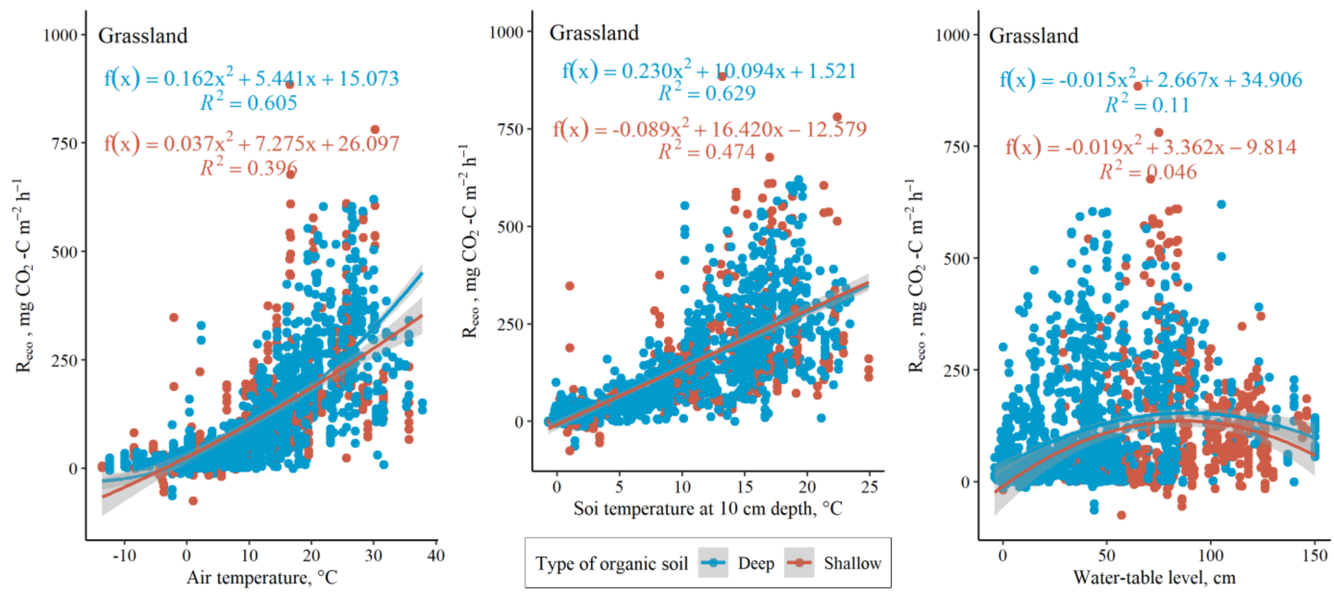



**Figure 4:~~.~~ Instantaneous ecosystem respiration ($R_{eco}$) in cropland (left graph) and grassland (right graph) as a function (polynomial regression) of ~~air temperature,~~ soil temperature at 10 cm depth ~~and water-table level~~measured during each gas sampling event.** Data of instantaneous 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 g~~G~~rey area around the regression line reflects the 95% confidence interval of regression.

### 3.3 Soil heterotrophic respiration (instantaneous)

During the ~~study~~ 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. S~~8~~14). 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 $R_{het}$ (~~average~~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 $R_{\cancel{eco}het}$ between ~~study sites in~~ cropland and grassland was observed (p = 0.~~776~~825). In cropland, the overall mean instantaneous $R_{het}$ (~~average~~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 $m^{-2}$ $h^{-1}$). N~~For grassland, n~~o statistically significant differences in mean instantaneous $R_{het}$ were observed for grasslands, neither between ~~study sites grouped depending on~~ soil types nor between ~~study sites grouped depending on drainage (~~deep or shallow drained sites~~) status~~ (p = 0.495 and p = 0.743, respectively). ~~In grassland,~~The mean instantaneous $R_{het}$ in grassland was 192.3 ± 25.5 mg $CO_2$-C $m^{-2}$ $h^{-1}$.

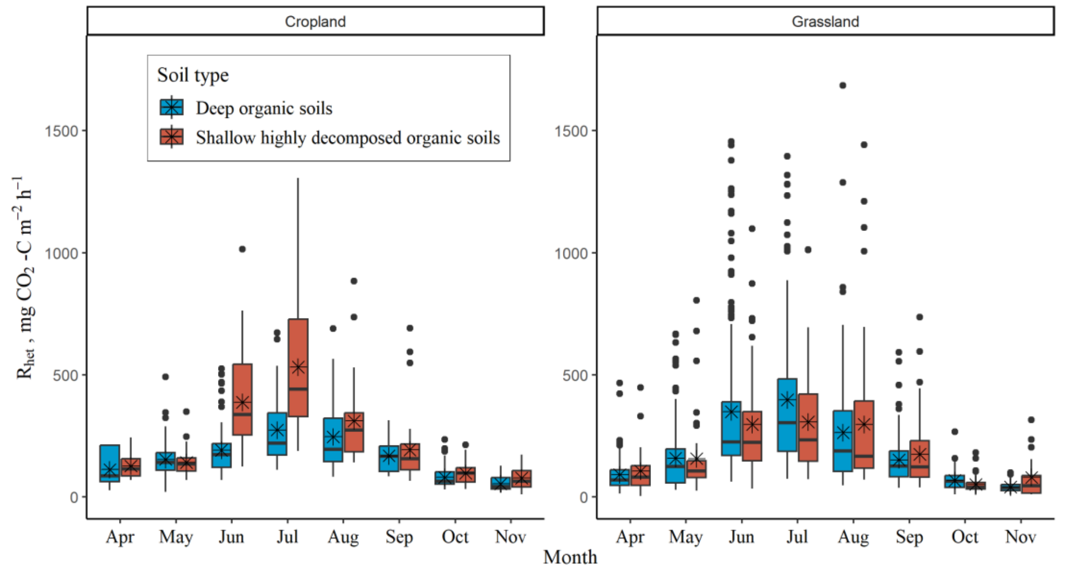
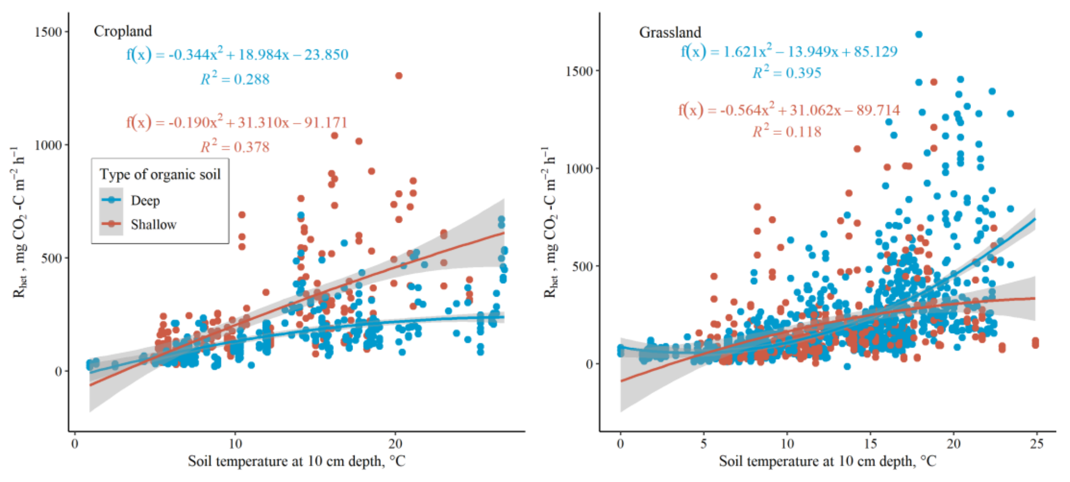



**Figure 5: Variation of instantaneous soil heterotrophic respiration ($R_{het}$) in cropland (left graph) and grassland (right graph) from April to November grouped depending on soil type (deep organic soil and shallow highly decomposed organic soil).** In the boxplots, median and mean values (bold 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 25th to 75th 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.



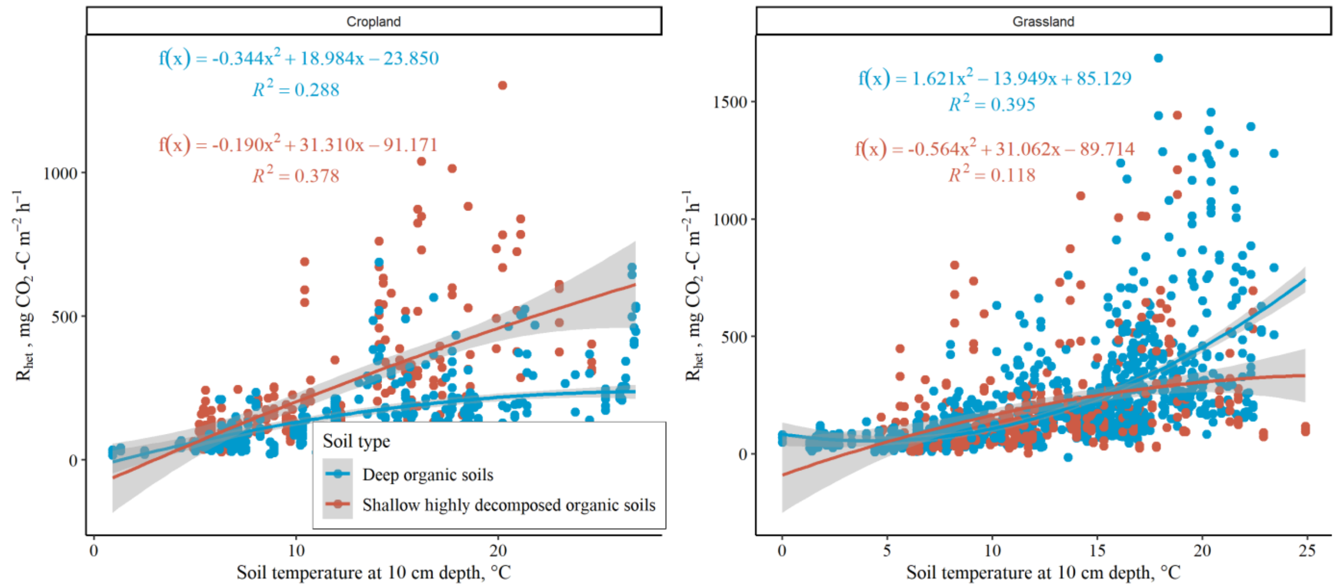



**Figure 6~~.~~: Instantaneous soil heterotrophic respiration ($R_{het}$) in cropland (left graph) and grassland (right graph) as a function (polynomial regression) of the soil temperature at 10~~-~~ cm depth.** Data of instantaneous soil heterotrophic respiration is grouped depending on soil type (deep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

### 3.4 Carbon stocks in, and ~~annual~~ inputs into soil through, vegetation

~~In~~ At the end of the growing season, the ~~average~~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 ~~agricultural~~ crops, the largest C stock in plant biomass (total) was estimated for maize (6.7 t C ha$^{-1}$), ~~but~~ and the lowest for spring wheat (2.3 t C ha$^{-1}$, Fig. S~~9~~15).

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 ~~than~~ to the aboveground inputs (1.16 ± 0.26 t C ha$^{-1}$ yr$^{-1}$). ~~Among the studied agricultural crops, t~~The largest annual C input (total) was estimated for rape (4.1 t C ha$^{-1}$ yr$^{-1}$), ~~but~~ and the lowest for spring wheat (1.5 t C ha$^{-1}$ yr$^{-1}$, Fig. S~~9~~15).

The mean concentrations of both C and N were higher in aboveground biomass compared to ~~below ground~~belowground biomass, while the C/N ratio was higher in belowground biomass both for arable crops and perennial grass (Table S5~~3~~).

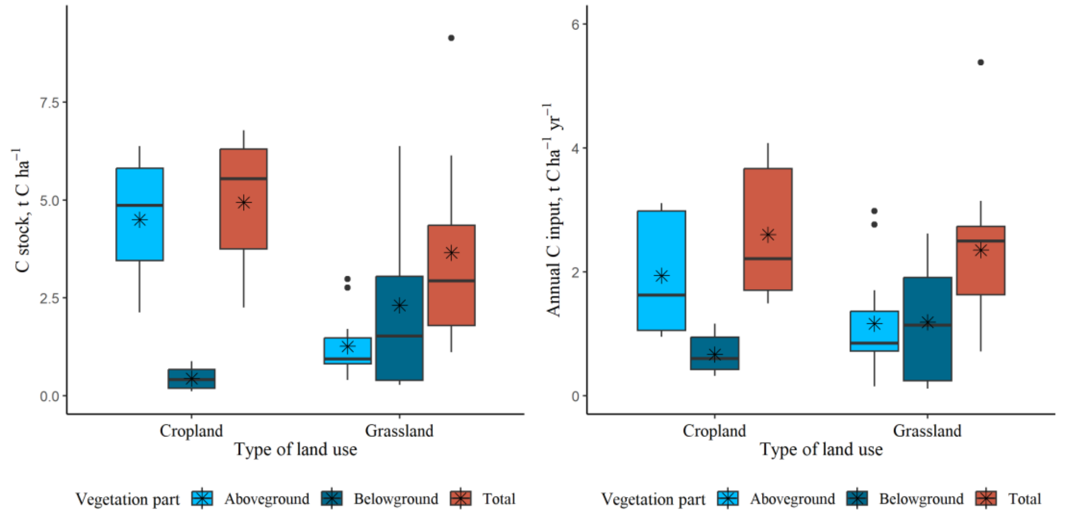



**Figure 7:~~.~~ ~~Variation in c~~Carbon (C) stock in above- and belowground plant biomass at the end of the growing season (left graph) and annual C inputs into the soil (right graph) in cropland and grassland ecosystems.** In the boxplots, the median and mean values ~~(~~ are presented by bold lines and asterisks, respectively~~) 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.

### 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 ~~4~~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 ~~in~~ for sites with shallow highly decomposed organic soil (7.0 ± 1.5 t $CO_2$-C ha$^{-1}$ yr$^{-1}$) than ~~in~~ 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 ~~in~~ 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 ~~was~~ 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 ~~under~~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 43. Annual ecosystem respiration ($R_{eco}$), heterotrophic soil respiration ($R_{het}$) estimated from $R_{eco}$ (64 % of annual $R_{eco}$ as described in Sect. 2.78), C input into soil as plant residues, and the estimated net soil $CO_2$ emissions in cropland and grassland in the Baltic ~~states~~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, 20154).

| 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 ± S.E. | 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 ± S.E. | 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 ± S.E. | 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 ± S.E. | 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 ± S.E. | 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 ± S.E. | 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 ± S.E. | 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 ± S.E. | 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 $CO_2$ ~~emissions~~ 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 ~~call~~ defined as shallow highly decomposed organic soils. ~~We consider based on existing soil information~~Based on existing soil information, we ~~consider~~believe that the soils of all our sites were originally (before drainage) deep peat soils. ~~Organic carbon (~~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 ~~shallow~~ 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. ~~To estimate the annual net $CO_2$ fluxes from soil or emission factors, we combined the ecosystem inputs and outputs as $CO_2$ fluxes into net fluxes by combining estimates from study sites on current or former peat soils in the Baltic statescountries with similar characteristics regarding geoclimatic conditions and intensity of land 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 ~~$CO_2$~~ emissions in grassland. Within our study, this tendency was mainly related to the higher mean annual $R_{eco}$ and subsequently ~~also~~ the estimated annual $R_{het}$ ~~(calculated as a proportion of $R_{eco}$)~~ in cropland than grassland.~~,~~ Further, the belowground C inputs tended to be higher, and the growing period longer under grass compared with arable crops. ~~despite~~ The slightly higher total C input into the soil as plant residues in cropland~~, as well as larger belowground (root) biomass and longer vegetation period under grass compared with arable crops~~ 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 ~~both our study and~~ previous studies as well (Couwenberg, 2011).

Although our estimates ~~results obtained in this study~~ may contain slight overestimations rather than underestimations, ~~our estimates of~~the mean annual net $CO_2$ emissions of both ~~in~~ cropland and grassland with deep-drained organic soil were generally lower than the ~~mean values~~emission factors provided by the IPCC (Hiraishi et al., 2014) ~~, mean values of net $CO_2$ emissions expressed which are presented as emission factors~~ for nutrient-rich soils in the temperate and boreal ~~climate/vegetation~~ zones ~~(nutrient-rich soils)~~ (Table 4). The net $CO_2$ emissions from shallow highly decomposed organic soils, instead, ~~coincided with the 95 % confidence intervals of~~were rather similar to the IPCC emission factors ~~(Table 54)~~. 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 ~~5~~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)~~. and l~~Logically, WTL ~~that is at least in principle regulated by drainage depth, which~~should regulate~~s, in turn,~~ the thickness of the soil layer where efficient aerobic decomposition may take place, and thus the $CO_2$ flux. However, ~~i It could be relatedexplained to by the~~we only had a ~~limited number of~~two grassland ~~study~~ sites with shallow-drained organic soils ~~in grassland (*n* = 2)~~, which increases uncertainty in the respective emission estimate~~, however~~. Yet, based on current data, the same emission factor could be used for both deep- and shallow-drained grassland in the hemiboreal region.

**Table ~~5~~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 ± S~~.~~E~~.~~) | | | 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.~~During the last decade (i.e. in the period after the latest IPCC default emission factors were elaborated), several new studies were conducted in temperate and boreal regions (central and northern Europe), reporting~~ These studies have reported $CO_2$ emission values that fall within or just beyond ~~ranging within or slightly outside~~ 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 underscores a notable ~~continue to highlight a relatively large~~ variability, which can be attributed~~explained by~~ to the variations ~~differences~~ in climate and ~~(~~weather conditions~~)~~, peat chemistry and ~~extent~~degree of decomposition~~–~~, the time since establishment~~,~~ and ~~factors related to aspects of~~ 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 ~~also~~ noted that various methods~~different approaches to~~ for ~~measure~~ measuring or ~~estimate~~ estimating soil C losses are employed across~~used among~~ 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 ~~states~~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 ~~arable~~ cropland and grassland, where the annual production, rotation and management of plant biomass ~~is~~ are highly dynamic and differences in the proportion of $R_{het}$ ~~between cropland and grassland are~~ 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.~~WithiI~~Within ~~our study, the impact of ploughing of grasslands for renovation by reseeding on $CO_2$ fluxes was not estimated.~~

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~~study is in line with previous findings~~ that soil temperature is the main driver of ~~overall~~ both ecosystem respiration (Nieveen et al., 2005; Elsgaard et al., 2012) ~~as well as specifically affecting~~and heterotrophic respiration deriving from peat decomposition ~~specifically~~ (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 ~~of~~ WTL and soil moisture would have an impact on the magnitude of ~~the~~ $CO_2$ fluxes. Yet, there were some indications of comparatively lower instantaneous $R_{eco}$ and $R_{het}$ ~~at~~ under both very dry and water-saturated conditions, ~~the~~ as the relationship between $R_{eco}$ and WTL exhibited a downward-opening parabola~~$R_{eco}$ as a function of WTL reflected a downwards parabola~~. Similarly, no clear quantitative $CO_2$ response to WTL among study sites with drained organic soils under agricultural management ~~has been reported also previously~~was found in some previous studies (Nieveen et al., 2005; Elsgaard et al., 2012; Tiemeyer et al., 2016), being ~~and then~~ explained by the rather deep WTL ~~among~~in the ~~studied~~ 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)~~, our results now slightly supporting that~~. Inconsistent results regarding

WTL as a controlling ~~factor~~variable of soil respiration can also be obtained ~~also~~ due to~~, for instance,~~ 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~~ours and previous results indicate~~ that among drained and managed sites~~,~~ - where the mean WTL is generally deeper than 30~~25~~ cm~~ and varies less than when also undrained and unmanaged sites are included in the data,~~ - WTL may not be the sole linear factor ~~single overriding factor linearly~~ regulating the soil $CO_2$ fluxes (Fig. S17~~6~~ and Fig. S18~~7~~). 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~~, J.~~ et al. 2023). These studies showed that $CO_2$ emissions increase almost linearly with deeper WTL under shallow drainage (~~up~~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~~as~~ 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 ~~tendency~~tendencies. For instance, Säurich et al. (2019a) concluded that the magnitude of ~~soil~~ soil-specific basal respiration (~~i.e.~~ $CO_2$ flux~~es~~ per unit SOC) increased with increasing soil disturbance ~~(i.e. with lower soil OC concentration)~~ caused by drainage-induced ~~mineralization~~ mineralisation and organic soil layer mixing with mineral soil~~s~~ (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 ~~is~~ 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.

~~In general, the estimation of mean $CO_2$ flux within our study could be slightly overestimated as all $CO_2$ flux measurements were conducted during the daytime, when both air and soil temperatures isare typically higher than the average diurnal temperature. To estimate the , potential overestimation of annual $R_{eco}$ due to measurements conducted only during the daytime, a study site-specific comparison of the applied method and modelling approach based on continuous soil temperature measurements at depths of 10 cm was made (for the study sites where continuous soil temperature measurements were conducted).~~ 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). ~~and~~Also, ~~p~~Previous studies have concluded that the mean $CO_2$ ~~production~~ flux ~~occurring~~ 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 ~~parameters~~variables. Thus~~, for further evaluations~~, 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 simultaneously recorded values should be lower than $R_{eco}$~~, and thus, recorded values should be lower than those of $R_{eco}$ simultaneously. The observed inconsistency is most likely explained by methodological challenges ~~nuances~~. 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 ~~killed~~ ~~dead~~ roots ~~due to~~killed by the trenching ~~are~~ ~~is~~ are likely to further ~~add~~ contribute to $R_{het}$. These aspects have also been discussed as challenges of ~~the~~ root exclusion ~~method~~ 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~~are higher than~~ soil surface respiration ~~since~~ because they ~~additionally~~ also account for the~~include~~ 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. ~~This effect should logically be at its highest during the summer months, when the plants are fully developed. However, quantitative proportion of aboveground autotrophic respiration was rarely reported, and the published studies exhibited wide variability and had relatively large uncertainties – some studies have indicated negligible, while some even improbably high rates of aboveground autotrophic respiration (Phillips et al., 2017).~~

The e~~E~~stimated 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, ~~which can~~ potentially ~~indicate~~ indicating slower decomposition of residues ~~of~~ 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 ~~states~~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 ~~(Hiraishi et al., 2014)~~ 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 ~~from~~ 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. ~~In addition, b~~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 ~~even~~ higher than from deep organic soils. This result highlights the need to estimate the~~ir~~ emissions from these highly transformed soils ~~rather~~ as organic soils rather than as mineral soils, even though they do not fulfil~~l~~ 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, ~~both continuation of data acquisition including higher measurement intensity and further refinement of the first hemiboreal region-specific $CO_2$ emission factors that we here defined for national GHG inventories is recommended~~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~~10.5281/zenodo.13234237~~. Additional data can be provided by the corresponding authors upon request.

**Author contribution**

~~Conceptualization~~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; ~~Visualization~~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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