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Greenhouse Gas Fluxes from Cranberry and Highbush Blueberry Plantations on Former Peat Extraction Fields Compared to Active Peat Extraction Fields and Pristine Peatlands in Latvia

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Abstract: Emissions of carbon dioxide (CO_2), nitrous oxide (N_2O), and methane (CH_4), particularly those from organic soils, need to be reduced in the context of climate change mitigation (CCM). Here, we estimated the greenhouse gas (GHG) fluxes from nutrient-poor organic soils in cranberry (*Vaccinium macrocarpon*) and highbush blueberry (*Vaccinium corymbosum*) plantations established on former peat extraction fields compared to active peat extraction fields and pristine raised bogs in Latvia. A two-year study (2016–2018) was conducted using the manual closed chamber method. In berry plantations and active peat extraction fields, annual net CO_2 fluxes contributed the most to total GHG emissions, accounting for over 67%, and temperature had the most significant impact on CO_2 fluxes. Conversely, annual CH_4 fluxes were the primary contributor to total net GHG emissions in the pristine raised bog, which simultaneously acted as a slight CO_2 sink. N_2O fluxes were relatively low among all studied land use types. This study provided quantitative insights into the variation in GHG fluxes and the environmental variables influencing them, and the obtained data are valuable to estimate the impact of the establishment of berry plantations on former peat extraction fields on CCM in the hemiboreal region of Europe.

Keywords: drained organic soil; highbush blueberry; cranberry; commercial berry plantations; peat extraction fields; raised bog; greenhouse gas emissions

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1. Introduction

Peatlands are distributed worldwide [1] and cover 4.23 million km^2 or nearly 3% of the global land area [2]. They contain significant carbon (C) stocks—almost twice the C stock of the total biomass of the world's forests [3]. Wetlands provide various ecosystem services that help humanity adapt to climate change and sequester and store C, thereby reducing the rate of climate changes [4]. Therefore, wetlands have been the focus of numerous restoration and reclamation programmes aimed at improving ecosystem services, including C sequestration, land hydrology, flood mitigation, and biodiversity [5]. Importantly, while wetlands offer valuable nature-based solutions to the goals of the 2015 Paris Climate Agreement [6], restoration and reclamation measures must be combined with a dramatic reduction in GHG emissions [4]. Today, peatlands that have been drained for economical use or abandoned after peat extraction contribute to almost 5% of global anthropogenic GHG emissions [7]. The proper management of these drained peatlands is therefore a key element for reducing GHG emissions. At the same time, climate change, including increased evapotranspiration and the irregularity of rainfalls as well as artificial drainage networks, can hinder the ability of peatland ecosystems to continue acting as C sinks [3,8].

Thus, scientific discussions and efforts for the most appropriate management strategies for peatlands in the context of CCM are still ongoing.

In the Baltic states (Estonia, Latvia, and Lithuania), peat extraction has directly affected roughly 90,000 ha of peatlands. Of these, about 30% have already been extracted and are now abandoned, while the rest of the areas have been afforested, converted into agricultural lands, including berry plantations [9], or turned into water bodies. Most of the still-extant abandoned peat extraction sites in the Baltic states were abandoned during and shortly after the Soviet period (1940–1991) without any restoration efforts [10].

Since berries, such as cranberries and blueberries (*Vaccinium* spp.), can tolerate wetter soil conditions [11], establishing berry plantations on former peat extraction sites is one of the options that has gained the interest of researchers in both socioeconomic and GHG emission mitigation contexts (e.g., [12]). Collecting berries is a long-standing tradition in the Nordic and Baltic countries mostly due to their medical and dietary properties [13]. Nowadays, wild berries, such as cranberries, blueberries, and lingonberries, are harvested in economically significant quantities for both self-consumption and for sale [14,15]. In Latvia, cranberry cultivation for commercial purposes began in 1985 [13]. The cultivation of large cranberries (*Vaccinium macrocarpon*) and lowbush and highbush blueberries (*Vaccinium angustifolium* and *Vaccinium corymbosum*, respectively) in Latvia is commercially important, has a growing market demand, and is a rapidly expanding agricultural sector [16]. Both cranberry and blueberry plantations can be established on extracted peatlands with bare peat and on mineral soils with a high organic matter content [17,18]. Thus, abandoned peat extraction areas can be transformed into profitable land resources. For blueberries, the optimum pH range of the upper peat layer is 4.5–5.0 and the residual thickness of the peat layer should be greater than 0.5 m. If the pH of the peat layer is lower, between 3.5 and 4.5, cranberries can be considered for cultivation. This crop is also less sensitive to the residual peat layer thickness if the optimal pH is maintained [19].

The establishment of berry plantations on former peat extraction areas requires land reclamation measures, which involve preparing the abandoned sites for further use. Reclamation and construction designs of drainage and irrigation systems must be approved according to the procedures set by legislation. If the former peat extraction area was abandoned a long time ago, the remaining vegetation, tree trunks, and stumps must be removed. The land surface must be leveled, irrigation/drainage systems must be constructed, and then the soil must be cultivated. The soil should be tested and fertilizers, including gypsum, should be applied if necessary [19,20].

Thus, following reclamation activities and berry plantation establishment, abandoned peat extraction areas can be used for economic activities, which boost employment and generate income. Another significant advantage of establishing berry plantations on former peat extraction areas in the context of CCM is that the topsoil is completely covered by berry plants, which helps to reduce GHG emissions. In Latvia, the total area of cultivated blueberry and cranberry plantations in 2023 was 499 and 189 ha, respectively [21].

Among the main factors affecting GHG fluxes from peatlands, as mentioned in the literature, are air temperature, groundwater level, composition of soil organic matter, living biomass, dead organic matter, nutrient supply in the soil, and management activities [22]. The GHG balance consists of several components that can act as sources or sinks of the gases. CO_2 is released into the atmosphere through autotrophic and heterotrophic respiration [23]. Autotrophic respiration, caused by metabolic processes in plants, results from the living above- and belowground plant biomass, whereas heterotrophic respiration results from the decomposition of soil organic matter by bacteria or fungi [23]. Besides emissions, the ecosystem CO_2 balance also includes photosynthetic CO_2 uptake. CO_2 captured through photosynthesis is released in the soil with litter and dead parts of plants, thus contributing to C input to the soil [24]. Bacterial activity in the soil leads to the release of CH_4 and N_2O [25].

Earlier studies conducted in Latvia showed that the total GHG emissions from organic soils in Latvia are equal to the emissions of the whole energy sector [26]. These studies analyzed surface-to-atmosphere GHG fluxes from rewetted and permanently flooded former peat extraction areas in comparison to pristine peatlands. Results showed that among the studied types of land use, the highest annual CO_2 fluxes from soil heterotrophic and autotrophic respiration were recorded in rewetted former peat extraction areas with restored vegetation and in pristine peatland, while the lowest fluxes were recorded in flooded former peat extraction areas. Air temperature and groundwater level were identified as the most significant influencing factors. The highest annual CH_4 fluxes were found in pristine peatland, followed by significantly smaller CH_4 fluxes in flooded and rewetted areas, respectively. Also, they found that N_2O fluxes were negligible in all the studied land use types, with the highest N_2O fluxes again observed in pristine peatland [26].

The aim of this study was to estimate surface-to-atmosphere GHG fluxes from cranberry and highbush blueberry plantations established on former peat extraction fields (Histosols) in Latvia and to identify the main factors influencing these fluxes. In addition, we compared GHG fluxes from cranberry and highbush blueberry plantations on former peat extraction fields to GHG fluxes from active peat extraction areas and pristine peatlands.

2. Materials and Methods

2.1. Study Sites

The study was conducted over a two-year period between December 2016 and November 2018 in Latvia, an area belonging to the hemiboreal vegetation region of Europe. Four different types of land use and vegetation (16 study sites in total, soil group Histosols) were examined across 11 different raised bogs (Figure 1 and Table 1): (i) cranberry (*Vaccinium macrocarpon*) plantations on former peat extraction fields, where the groundwater level is slightly lowered or close to the surface (the dominant peat type in the upper layer—raised bog peat); (ii) highbush blueberry (*Vaccinium corymbosum*) plantations on former peat extraction fields, where the groundwater level is slightly lowered or close to the surface (the dominant peat type in the upper layer—raised bog or mixed peat); (iii) active peat extraction fields with an effective drainage system, where peat was extracted using the milling method; and (iv) pristine raised bogs, where trees do not exceed a height of 5 m, the projective cover in mature stands does not exceed 20%, and the area continuously covered with trees does not exceed 0.1 ha.

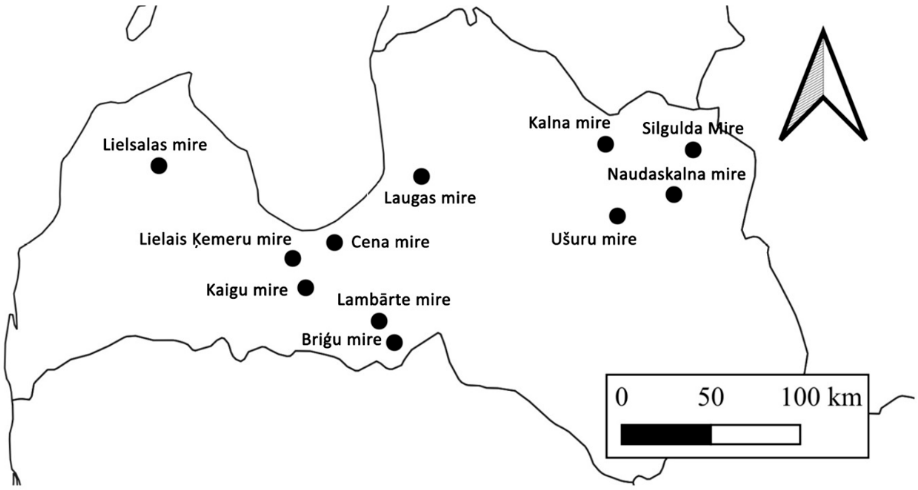

Figure 1. Distribution of raised bogs (black dots in the map) in Latvia where the study sites were established.

The long-term (1991–2020) average annual air temperature in Latvia was 6.8 °C, while the average annual precipitation was 685.6 mm. In the study period (2016–2018), the average annual air temperature in Latvia ranged from 6.9 to 7.6 °C, while the average annual precipitation ranged from 472.7 to 809.8 mm [27].

Table 1. General description of the study sites in Latvia.

Type of Land Use and Vegetation	Study Sites	Soil Layer, cm	Mean Soil Bulk Density, Mean Value ± S.E. (Range), kg m^{-3}
Cranberry plantations on former peat extraction fields	Kalna_12 (Kalna mire), Naud_12 (Naudaskalna mire), Usuri_12 (Ušuru mire), Lauga_12 (Laugas mire), Brigi_13 (Briģu mire)	0–10	166.6 ± 24.2 (110.8–321.3)
		10–20	113.0 ± 24.6 (74.1–333.5)
		20–30	122.1 ± 23.6 (80.6–333.1)
		30–40	117.1 ± 24.8 (72.2–339.2)
		40–50	106.0 ± 19.1 (76.5–276.6)
Highbush blueberry plantations on former peat extraction fields	Kaigu_11 (Kaigu mire), Kalna_11 (Kalna mire), Naud_11 (Naudaskalna mire)	0–10	152.6 ± 15.0 (107.3–226.3)
		10–20	120.9 ± 18.5 (46.3–205.8)
		20–30	106.4 ± 18.8 (20.4–178.7)
		30–40	111.7± 69.3 (70.3–196.6)
		40–50	109.9 ± 13.2 (71.4–170.0)
Active peat extraction fields	Kaigu_1 (Kaigu mire), Usuri_1 (Ušuru mire), Lamb_1 (Lambārte mire), Cena_1 (Cena mire), Silg_1 (Silgulda mire)	0–10	123.9 ± 12.5 (73.6–183.9)
		10–20	106.3 ± 4.6 (87.2–136.1)
		20–30	100.3 ± 5.2 (73.9–128.9)
		30–40	94.2 ± 6.8 (71.8–134.0)
		40–50	105.2 ± 8.6 (67.5–163.8)
Pristine raised bog	Lauga_9 (Laugas mire), Kem_9 (Lielais Ķemeru mire), Liels_9 (Lielsalas mire)	0–10	150.9 ± 11.6 (134.9–173.5)
		10–20	128.9 ± 1.3 (126.3–130.4)
		20–30	97.5 ± 1.2 (95.7–99.9)
		30–40	101.1 ± 21.0 (79.7–143.1)
		40–50	75.3 ± 11.2 (56.6–95.2)

2.2. GHG Flux Measurements and Calculations

Between December 2016 and November 2018 (over a period of 24 months), gas sampling was conducted using a manual closed chamber method [28]. Before gas sampling was initiated, each study site was prepared by installing five permanent circular collars extending to a depth of 5 cm. Collars were evenly distributed with a 2–3 m distance between individual collars. During the installation of the collars, the disturbance of vegetation was avoided or minimized. Gas sampling was conducted during the daytime once a month by positioning chambers (non-transparent, volume 0.0655 m^3, diameter 50 cm) on the collars and taking four consecutive gas samples (100 cm^3) at 20 min intervals (immediately after positioning the chamber on the collar as well as after 20, 40, and 60 min) using underpressurized (0.3 mbar) glass vials. The gas samples were transported to the Climate Change Laboratory of the Department of Geography at the University of Tartu (Estonia), where CO_2, CH_4, and N_2O concentrations in gas samples were determined using the Shimadzu GC-2014 gas chromatograph (Shimadzu Corporation, Kyoto, Japan) equipped with an electron capture detector, flame ionization detector, and a Loftfield autosampler [29].

GHG fluxes (mg GHG m^{-2} h^{-1}) were calculated using the ideal gas law equation and the slope coefficient of linear regression describing the change in the gas concentration over time (during a 60 min period) in the chamber (based on the results of the gas chromatography analysis of four consecutive gas samples). A detailed description of GHG flux calculations is provided by Bardule et al. [26].

In study sites where the soil is covered with vegetation (cranberry plantations, highbush blueberry plantations, pristine raised bog), the estimated CO_2 fluxes reflect ecosystem respiration (R_{eco}), which includes both soil heterotrophic respiration (R_{het}) due to the decomposition of dead organic matter and autotrophic respiration by the aboveground and belowground parts of plants. For the sites where the soil is covered with vegetation, it was assumed that the proportion of R_{het} to R_{eco} is 0.5 [30] if the air temperature is above 5 °C, while R_{eco} equals R_{het} if the air temperature is below 5 °C. In study sites with bare soil (active peat extraction fields), the estimated CO_2 fluxes reflect R_{het}, as there is no vegetation cover.

2.3. Measurements of Environmental Parameters

In each study site, unmixed soil samples were collected in three replicates in 2016 using a soil sample probe from the following soil layers: 0–10 cm, 10–20 cm, 20–30 cm, 30–40 cm, 40–50 cm, and 50–100 cm. Pretreatment and physico-chemical analyses of soil samples were performed according to the ICP Forests guidelines [31] and ISO methodology. Soil moisture content was determined according to the LVS ISO 11465 (LVS ISO 11465:2006 +TC1 A/L; Soil quality—Determination of dry matter and water content on a mass basis—Gravimetric method. Latvian Standards: Riga, Latvia, 2006) (gravimetric method); soil bulk density was determined according to the LVS EN ISO 11272 (LVS EN ISO 11272:2014; Soil quality—Determination of dry bulk density (ISO 11272:1998). Latvian Standards: Riga, Latvia, 2014); pH was measured according to the LVS ISO 10390 (LVS ISO 10390:2002 A/L; Soil quality—Determination of pH. Latvian Standards: Riga, Latvia, 2002) (suspension of soil in 0.01 mol L^{-1} calcium chloride ($CaCl_2$) solution); total carbon ($C_{tot.}$) content equaled the organic carbon ($C_{org.}$) content and was measured with an elemental analyzer (dry combustion) according to the LVS ISO 10694 (LVS ISO 10694:2006; Soil Qualitsy—Determination of Organic and Total Carbon after Dry Combustion (Elementary Analysis). Latvian Standards: Riga, Latvia, 2006); total nitrogen ($N_{tot.}$) content was determined with a modified Kjeldahl method according to the LVS ISO 11261 (LVS ISO 11261:2002 L; Soil quality—Determination of total nitrogen—Modified Kjeldahl method. Latvian Standards: Riga, Latvia, 2002); concentrated nitric acid (HNO_3) extractable phosphorus (P) concentration in soil samples was determined according to ISO 11466 (ISO 11466:1995; Soil quality—Extraction of trace elements soluble in aqua regia. ISO International standard: Geneva, Switzerland, 1995) and LVS EN 14672 and potassium (K) concentration was determined using atomic absorption spectroscopy method.

In each study site, two groundwater wells (PVC pipes) were installed vertically at a depth of 1.5 m. Concurrent with GHG flux measurements, the groundwater level was measured manually, as well as the dissolved oxygen (DO) content; electrical conductivity (Cond.), water pH, and the oxidation-reduction potential (ORP) were determined using YSI ProDSS Multiparameter Digital Water Quality Meter (YSI, a Xylem brand, Yellow Springs, OH, USA).

Concurrent with GHG flux measurements, air temperature and soil temperature at 5, 10, 15, and 30 cm depths were measured using a Comet data logger (Comet system, s.r.o., Roznov pod Radhostem, Czech Republic) equipped with temperature probes. Soil moisture (volumetric water content) at a 5 cm depth was determined using a ProCheck meter (Decagon Devices, Pulman, WA, USA) equipped with a moisture sensor.

2.4. Estimation of Carbon Input with Plant Litter

In the pristine raised bogs, C input with plant organic matter (1.43 $\pm$ 0.65 t C ha^{-1} y^{-1}) was assumed according to the earlier estimates reported by [32–35]; the mean value was calculated from these studies.

In the cranberry plantations, C input with the plant organic matter (1.82 $\pm$ 0.21 t C ha^{-1} y^{-1}) was assumed based on the estimates of above- and belowground biomass of vegetation in cranberry plantations in Latvia [36], and assuming that the root turnover rate is 0.41 [37], the leaf retention time is two years [33] and the C content in biomass is 48% [32].

In highbush blueberry plantations, aboveground litter from blueberry bushes was collected using ten litter traps (0.049 m^2) in each study site. Litter was collected once a month for a whole year (12 consecutive months). After collecting the litter samples, they were combined and the dry mass was determined. The C concentration in litter samples was determined using an elemental analyzer (dry combustion) according to the LVS ISO 10694. Additionally, we used a coefficient of 1.2 to estimate C input with belowground litter to C input with aboveground litter based on the approximate proportion reported by Moore et al. [33]. We assumed that the cover of blueberry bushes in the highbush blueberry plantations is 40%.

2.5. Estimation of Annual Greenhouse Gas Fluxes

Annual GHG fluxes were calculated as a cumulative value of the mean monthly fluxes (expressed as t CO_2-C ha^{-1} $month^{-1}$, kg CH_4-C ha^{-1} $month^{-1}$, kg N_2O-N ha^{-1} $month^{-1}$) covering all calendar months (from January until December) and expressed as t CO_2-C ha^{-1} y^{-1}, kg CH_4-C ha^{-1} y^{-1}, and kg N_2O-N ha^{-1} y^{-1}. Annual net CO_2 fluxes or CO_2 emission factors for each land use type were calculated as the difference between annual soil R_{het} and C input into soil with plant litter.

2.6. Statistical Analysis

All statistical analyses were performed using licensed Statistica software (StatSoft Statistica 12) with additional integrated R modules [38] and the software environment R (version 4.3.3) and RStudio (2023.12.1) [39]. To test the hypothesis of the normal distribution and the homogeneity of the variance in the obtained study data, the Shapiro–Wilk W test and a histogram of the density of the normal distribution were used and a comparison graph was built on a normal probability plot. In the case of non-compliance of the study data to the theory of the normal distribution, further statistical processing was performed using nonparametric statistics.

Statistically significant differences in variables of soil general chemistry between different soil layers and types of land use and vegetation were estimated using the Wilcoxon rank-sum exact test, with pairwise comparisons adjusted for multiple testing using the Bonferroni correction. To link mean GHG fluxes to various environmental variables, a simple regression analysis and Spearman correlation (r) analysis were performed. A significance level of $p < 0.05$ was used.

3. Results

3.1. Soil and Groundwater Physico-Chemical Variables

In general, within the same type of land use and vegetation, variation in soil chemical variables (Table 2) between different soil layers was relatively small (additionally confirmed by the analysis of confidence intervals). The mean $C_{org.}$ concentration tended to increase in deeper soil layers compared to upper soil layers across all studied types of land use. However, statistically significant differences in $C_{org.}$ concentrations were found only in active peat extraction fields where a significantly higher $C_{org.}$ concentration was found in the 50–100 cm soil layer compared to the 0–10 cm and 10–20 cm soil layers ($p = 0.028$ and $p = 0.007$, respectively). In none of the studied types of land use, statistically significant differences in $N_{tot.}$ concentrations and soil pH between different soil layers were found. Some individual cases of significantly lower concentrations of P and K in the deeper soil layers compared to the upper soil layers were observed (Table 2).

Among the studied types of land use and vegetation, the lowest mean $C_{org.}$ concentrations were found in cranberry plantations on former peat extraction fields and pristine raised bogs. Furthermore, several significant differences in $C_{org.}$ concentrations between the soil layers at depths of 10 and 100 cm were found (Table 2). The highest mean $C_{org.}$ concentration was observed in the deepest analyzed soil layer (50–100 cm) in active peat extraction fields (598.6 ± 10.15 g kg^{-1}). Statistically significant differences in this soil layer compared to all other studied types of land use were found ($p < 0.035$). A statistically significant difference in $N_{tot.}$ concentration was found only for the 0–10 cm soil layer between the pristine raised bogs where the highest $N_{tot.}$ concentration among all studied soil layers and types of land use was found (15.0 ± 1.24 g kg^{-1}) and cranberry plantations ($p = 0.044$). No statistically significant differences in soil pH between different types of land use were found, while several significant differences in P and K concentrations between different types of land use were found for 20–30 cm, 30–40 cm, and 40–50 cm soil layers (Table 2).

Table 2. Soil chemical variables in different soil layers by types of land use and vegetation. Different lowercase letters denote significant differences ($p < 0.05$) between different soil layers within the same type of land use and vegetation. Different capital letters denote significant differences ($p < 0.05$) between types of land use and vegetation within the same soil layer.

Type of Land Use and Vegetation	Soil Layer, cm	$C_{org.}$, Mean ± S.E. (Range), g kg^{-1}	$N_{tot.}$, Mean ± S.E. (Range), g kg^{-1}	P, Mean ± S.E. (Range), g kg^{-1}	K, Mean ± S.E. (Range), g kg^{-1}	pH $CaCl_2$, Mean ± S.E. (Range)
Cranberry plantations on former peat extraction fields	0–10	512.2 ± 23.22 ^{a,A} (304.3–577.0)	10.3 ± 0.97 ^{a,A} (5.3–17.5)	0.25 ± 0.02 ^{a,A} (0.16–0.35)	0.46 ± 0.05 ^{a,A} (0.26–0.84)	3.1 ± 0.22 ^{a,A} (2.4–4.4)
	10–20	547.6 ± 6.97 ^{a,AB} (514.5–590.2)	8.9 ± 0.88 ^{a,A} (5.65–15.43)	0.17 ± 0.02 ^{b,A} (0.09–0.30)	0.82 ± 0.50 ^{a,A} (0.12–6.31)	3.0 ± 0.20 ^{a,A} (2.5–4.3)
	20–30	561.6 ± 14.8 ^{a,A} (529.1–711.1)	9.6 ± 0.97 ^{a,A} (4.7–15.9)	0.19 ± 0.03 ^{ab,A} (0.08–0.47)	0.35 ± 0.04 ^{a,A} (0.12–0.54)	3.0 ± 0.21 ^{a,A} (2.5–4.4)
	30–40	548.6 ± 6.62 ^{a,A} (509.2–577.7)	8.0 ± 0.67 ^{a,A} (5.1–11.9	0.13 ± 0.01 ^{b,A} (0.09–0.19)	0.32 ± 0.05 ^{a,A} (0.14–0.79)	3.1 ± 0.23 ^{a,A} (2.5–4.7)
	40–50	546.2 ± 6.07 ^{a,AB} (511.3–583.5)	8.7 ± 0.76 ^{a,A} (5.7–15.2)	0.15 ± 0.01 ^{b,AB} (0.09–0.21)	0.32 ± 0.04 ^{a,A} (0.14–0.72)	3.2 ± 0.26 ^{a,A} (2.5–5.1)
	50–100	554.2 ± 5.98 ^{a,A} (524.1–592.5)	10.1 ± 1.17 ^{a,A} (6.6–20.9)	0.19 ± 0.02 ^{ab,A} (0.10–0.35)	0.33 ± 0.08 ^{a,A} (0.13–1.03)	3.1 ± 0.30 ^{a,A} (2.0–4.8)
Highbush blueberry plantations on former peat extraction fields	0–10	543.2 ± 8.83 ^{a,A} (510.8–588.2)	11.3 ± 1.24 ^{a,AB} (7.3–18.9)	0.28 ± 0.08 ^{a,A} (0.09–0.82)	0.82 ± 0.30 ^{a,A} (7.3–18.9)	3.1 ± 0.22 ^{a,A} (2.6–4.2)
	10–20	554.6 ± 12.79 ^{a,AB} (510.3–616.2)	10.1 ± 0.83 ^{a,A} (7.0–13.3)	0.26 ± 0.09 ^{a,A} (0.09–0.93)	0.41 ± 0.12 ^{ab,A} (0.14–1.04)	3.1 ± 0.23 ^{a,A} (2.5–4.1)
	20–30	549.3 ± 11.85 ^{a,AB} (514.9–630.6)	9.3 ± 0.94 ^{a,A} (6.0–14.1)	0.34 ± 0.12 ^{a,A} (0.04–0.89)	0.41 ± 0.10 ^{ab,AB} (0.13–1.08)	3.2 ± 0.25 ^{a,A} (2.5–4.3)
	30–40	557.3± 15.79 ^{a,AB} (510.5–652.4)	9.2± 0.97 ^{a,A} (6.6–15.4)	0.14± 0.04 ^{a,A} (0.02–0.34)	0.21± 0.03 ^{b,A} (0.07–0.34)	3.3± 0.28 ^{a,A} (2.6–4.4)
	40–50	555.8 ± 11.56 ^{a,AB} (514.5–616.6)	10.4 ± 1.62 ^{a,A} (5.5–20.0)	0.09 ± 0.01 ^{a,A} (0.03–0.17)	0.19 ± 0.03 ^{b,AB} (0.06–0.29)	3.3 ± 0.28 ^{a,A} (2.6–4.5)
	50–100	552.9 ± 8.61 ^{a,A} (526.1–601.5)	12.0 ± 1.52 ^{a,A} (6.9–19.9)	0.18 ± 0.08 ^{a,A} (0.01–0.79)	0.32 ± 0.09 ^{ab,A} (0.09–0.94)	3.4 ± 0.25 ^{a,A} (2.7–4.5)
Active peat extraction fields	0–10	546.2 ± 3.87 ^{a,A} (529.9–561.3)	10.5 ± 1.35 ^{a,AB} (6.4–20.3)	0.21 ± 0.05 ^{a,A} (0.03–0.53)	0.28 ± 0.08 ^{ab,A} (0.07–0.65)	2.9 ± 0.11 ^{a,A} (2.6–3.5)
	10–20	546.8 ± 2.59 ^{a,A} (538.1–559.6)	7.5 ± 0.71 ^{a,A} (5.4–11.1)	0.28 ± 0.08 ^{a,A} (0.12–0.84)	0.28 ± 0.06 ^{a,A} (0.09–0.59)	3.0 ± 0.15 ^{a,A} (2.6–3.8)
	20–30	556.7 ± 4.23 ^{ab,A} (537.3–575.2)	9.4 ± 1.39 ^{a,A} (5.3–16.0)	0.25 ± 0.09 ^{a,A} (0.10–0.95)	0.16 ± 0.04 ^{ab,B} (0.06–0.42)	3.1 ± 0.19 ^{a,A} (2.7–4.0)
	30–40	563.0 ± 6.28 ^{ab,A} (532.8–590.0)	9.1 ± 1.37 ^{a,A} (4.8–17.6)	0.12 ± 0.01 ^{a,A} (0.08–0.17)	0.08 ± 0.02 ^{b,B} (0.02–0.15)	3.2 ± 0.22 ^{a,A} (2.7–4.1)
	40–50	571.2 ± 9.85 ^{ab,A} (523.1–609.1)	11.6 ± 2.44 ^{a,A} (4.4–25.6)	0.14 ± 0.01 ^{a,AB} (40.11–0.24)	0.37 ± 0.26 ^{ab,B} (0.03–2.43)	3.3 ± 0.23 ^{a,A} (2.7–4.3)
	50–100	598.6 ± 10.15 ^{b,B} (550.2–644.0)	11.9 ± 1.22 ^{a,A} (7.6–18.2)	0.15 ± 0.02 ^{a,A} (0.03–0.21)	0.20 ± 0.09 ^{ab,A} (0.01–0.90)	3.6 ± 0.29 ^{a,A} (2.8–4.8)
Pristine raised bog	0–10	517.7 ± 11.33 ^{a,A} (479.3–575.6)	15.0 ± 1.24 ^{a,B} (9.7–21.3)	0.32 ± 0.04 ^{a,A} (0.12–0.45)	0.52 ± 0.11 ^{ab,A} (0.06–1.20)	2.8 ± 0.04 ^{a,A} (2.6–2.9)
	10–20	513.9 ± 8.51 ^{a,B} (475.6–549.6)	10.9 ± 5.5 ^{a,A} (5.1–23.6)	0.23 ± 0.09 ^{a,A} (0.08–0.32)	0.36 ± 0.12 ^{a,A} (0.22–0.56)	2.8 ± 0.04 ^{a,A} (2.7–3.0)
	20–30	525.8 ± 4.64 ^{a,B} (507.8–550.6)	12.8 ± 1.21 ^{a,A} (8.3–20.0)	0.22 ± 0.03 ^{a,A} (0.07–0.30)	0.52 ± 0.19 ^{ab,AB} (0.12–1.51)	2.8 ± 0.05 ^{a,A} (2.6–3.1)
	30–40	523.0 ± 6.00 ^{a,B} (500.1–555.9)	10.5 ± 1.38 ^{a,A} (6.7–19.2)	0.16 ± 0.04 ^{a,A} (0.06–0.35)	0.32 ± 0.16 ^{ab,AB} (0.17–0.26)	2.8 ± 0.05 ^{a,A} (2.6–3.2)
	40–50	514.0 ± 8.37 ^{a,B} (481.5–547.9)	11.7 ± 1.95 ^{a,A} (6.1–21.1)	0.17 ± 0.02 ^{a,B} (0.12–0.26)	0.14 ± 0.03 ^{b,B} (0.05–0.32)	2.8 ± 0.05 ^{a,A} (2.7–3.1)
	50–100	548.2 ± 12.27 ^{a,A} (511.1–601.5)	10.5 ± 1.01 ^{a,A} (7.1–14.9)	0.14 ± 0.01 ^{a,A} (0.10–0.19)	0.15 ± 0.03 ^{ab,A} (0.08–0.36)	3.0 ± 0.08 ^{a,A} (2.8–3.5)

Table 2. *Cont.*

Type of Land Use and Vegetation	Soil Layer, cm	$C_{org.}$, Mean ± S.E. (Range), g kg^{-1}	$N_{tot.}$, Mean ± S.E. (Range), g kg^{-1}	P, Mean ± S.E. (Range), g kg^{-1}	K, Mean ± S.E. (Range), g kg^{-1}	pH $CaCl_2$, Mean ± S.E. (Range)
95% confidence interval (CI), all types of land use and vegetation pooled	0–10	490.0–562.5	7.47–15.48	0.20–0.34	0.15–0.88	2.60–3.15
	10–20	512.1–565.6	6.02–12.08	0.16–0.31	0.24–0.43	2.52–3.23
	20–30	523.4–565.4	6.83–13.07	0.15–0.35	0.11–0.59	2.49–3.36
	30–40	517.9–574.8	6.74–11.21	0.11–0.16	0.05–0.40	2.45–3.50
	40–50	507.2–584.5	7.50–13.20	0.08–0.19	0.05–0.31	2.52–3.53
	50–100	528.5–601.5	7.83–13.47	0.13–0.20	0.11–0.34	2.35–3.90

The mean values of $C_{org.}$, $N_{tot.}$, P, and K concentrations and the soil pH in the soil layer of 0–50 cm are shown in Figure 2. The lowest mean $C_{org.}$ concentration and simultaneously, the highest $N_{tot.}$, P, and K concentrations, were observed in the pristine raised bogs. However, the mean values of analyzed chemical variables between different types of land use were in a relatively narrow range (Figure 2).

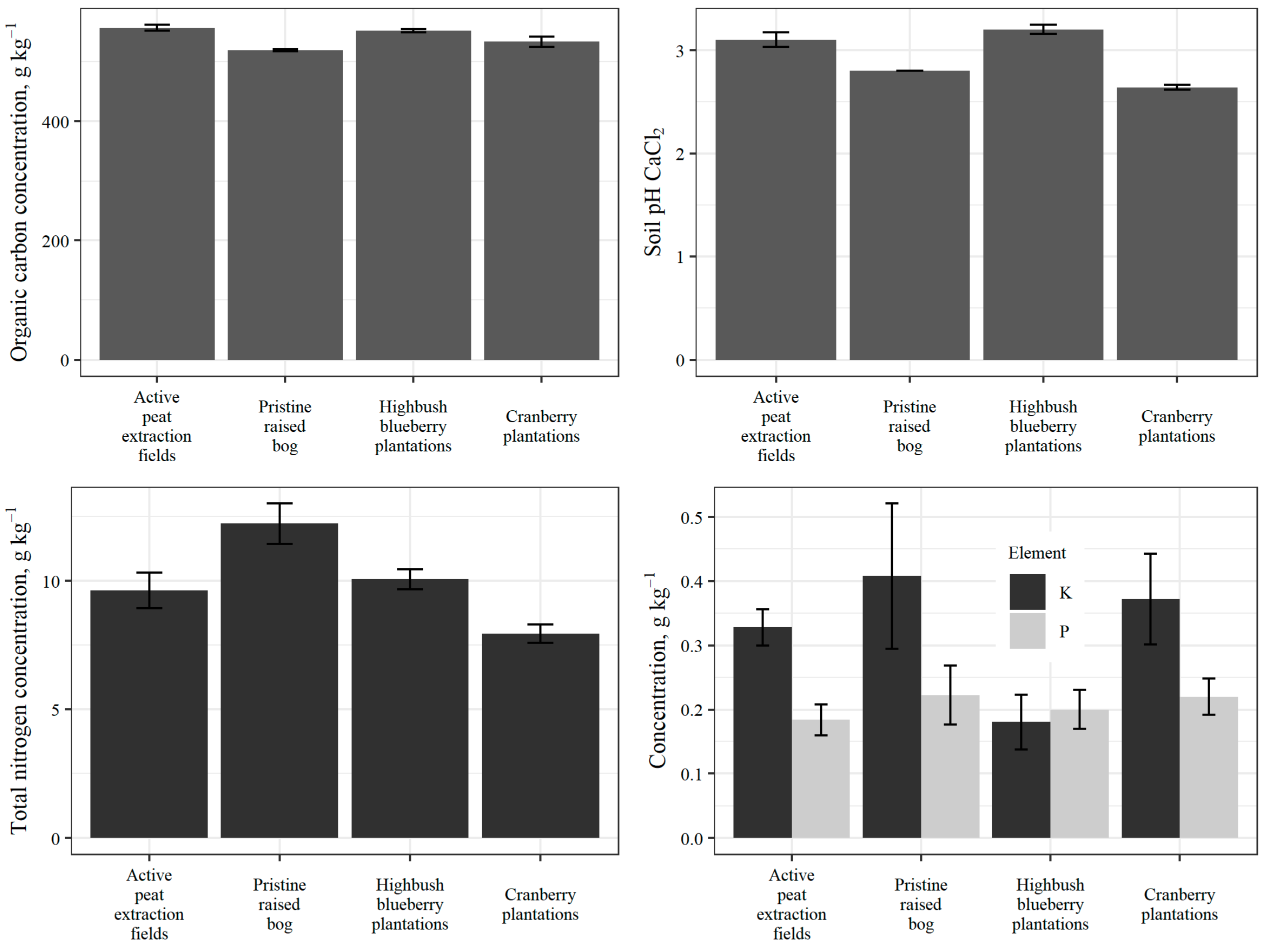

Figure 2. Mean values of soil chemical variables in the soil layer of 0–50 cm. Error bars reflect standard error.

In the soil layer of 50–100 cm (Appendix A, Figure A1), the highest levels of $C_{org.}$, $N_{tot.}$, and P were found in active peat extraction fields. In contrast, the lowest average $C_{org.}$ content was observed in the pristine raised bogs, while the concentrations of $N_{tot.}$, P, and K, as well as the soil pH, were lowest in cranberry plantations on former peat extraction fields.

Groundwater pH is influenced by the complex interactions between groundwater and soil. Among the studied types of land use and vegetation, the highest mean groundwater pH (the most alkaline environment) was observed in active peat extraction fields (5.15 ± 0.04), while the lowest mean pH (the most acidic environment) was observed in the pristine raised bogs (4.27 ± 0.03, Appendix A, Figure A2). In all studied types of land use and vegetation, groundwater pH (Appendix A, Figure A2) was higher (more alkaline) than the soil pH of $CaCl_2$ (Figure 2). In active peat extraction fields, the mean soil pH of $CaCl_2$ in 0–50 cm layer was lower than the groundwater pH by 2.0 units, while in highbush blueberry and cranberry plantations, the difference was 1.62 and 1.25 pH units, respectively. However, in order to accurately compare the obtained results, it is necessary to develop models that relate the measurements of soil pH in water and $CaCl_2$, as presented in the publications of Minasny et al. [40].

3.2. Environmental Variables (Temperature and Groundwater Level)

In each of the studied land use types, a specific microclimate (air and soil temperature, groundwater level) was observed (Table 3). The highest mean groundwater level was observed in the pristine raised bogs (7.0 ± 0.56 cm) and during the study period (field surveys), the groundwater level did not drop below 35 cm. In active peat extraction fields and berry plantations, the mean groundwater level ranged from 41.6 to 52.3 cm.

Table 3. Mean air temperature and soil temperature in different soil layers and the groundwater level in studied types of land use and vegetation.

Type of Land Use and Vegetation	Temperature		Groundwater Level *, Mean Value ± S.E. (Range), cm
	Measurement Point	Mean Value ± S.E. (Range), °C	
Highbush blueberry plantations on former peat extraction fields	Air	11.3 ± 0.57 (−6.8–31.8)	45.3 ± 1.99 (0.0–160.0)
	Soil, 5 cm	9.0 ± 0.44 (−1.9–23.4)	
	Soil, 10 cm	8.3 ± 0.39 (−1.5–20.9)	
	Soil, 15 cm	7.6 ± 0.33 (−0.3–19.3)	
	Soil, 30 cm	7.7 ± 0.32 (−0.1–19.5)	
Cranberry plantations on former peat extraction fields	Air	13.0 ± 0.59 (−9.4–32.8)	41.6 ± 1.27 (−12.0–118.5)
	Soil, 5 cm	10.8 ± 0.46 (−4.0–27.0)	
	Soil, 10 cm	9.5 ± 0.40 (−4.4–24.2)	
	Soil, 15 cm	8.5 ± 0.34 (−0.6–21.3)	
	Soil, 30 cm	8.6 ± 0.33 (−0.5–21.3)	
Active peat extraction fields	Air	11.9 ± 0.59 (−6.6–33.0)	52.3 ± 1.98 (−0.5–150.0)
	Soil, 5 cm	9.5 ± 0.46 (−2.7–25.9)	
	Soil, 10 cm	8.3 ± 0.42 (−4.0–22.0)	
	Soil, 15 cm	7.6 ± 0.36 (−1.2–18.9)	
	Soil, 30 cm	7.6 ± 0.35 (−1.0–18.7)	
Pristine raised bog	Air	9.4 ± 0.49 (−10.1–26.7)	7.0 ± 0.56 (−23.0–35.0)
	Soil, 5 cm	7.9 ± 0.42 (−1.5–24.4)	
	Soil, 10 cm	7.4 ± 0.38 (−1.0–20.7)	
	Soil, 15 cm	7.2 ± 0.34 (−0.5–19.0)	
	Soil, 30 cm	7.3 ± 0.33 (−0.3–19.1)	

* Negative values reflect the groundwater level above the ground surface.

Slight differences in mean air and soil temperatures between the different types of land use and vegetation were observed (Table 3). However, it was not possible to reliably determine whether these differences in air and soil temperature were caused by interactions with vegetation or by specific relief conditions.

The mean groundwater temperature (Appendix B, Figure A3) in the active peat extraction fields was 8.27 ± 0.20 °C. In the pristine raised bogs, the lowest mean groundwater temperature was observed (7.45 ± 0.18 °C). The mean groundwater temperature in the highbush blueberry plantations was 0.12 °C lower than in the active peat extraction fields (8.15 ± 0.19 °C), while the highest value of the mean groundwater temperature was observed in cranberry plantations (9.04 ± 0.17 °C).

3.3. Variation in Greenhouse Gas Fluxes and Its Affecting Factors

The highest mean CO_2–C fluxes reflecting soil heterotrophic respiration were observed in highbush blueberry and cranberry plantations on former peat extraction fields (higher by 8.8 and 8.6 mg m^{-2} h^{-1} compared to active peat extraction fields). The lowest mean CO_2–C fluxes (16.0 mg CO_2–C m^{-2} h^{-1}) were recorded in the pristine raised bogs, which was even lower (by 0.4 mg CO_2–C m^{-2} h^{-1}) than in the active peat extraction fields (Figure 3). If we evaluate the mean CO_2–C fluxes using the coefficient of variation (CV), the smallest deviations were observed in the highbush blueberry and cranberry plantations on former peat extraction fields (110.0 and 112.0%, respectively). In active peat extraction fields, the CV was 135.0%, while in the pristine raised bogs—119.6%.

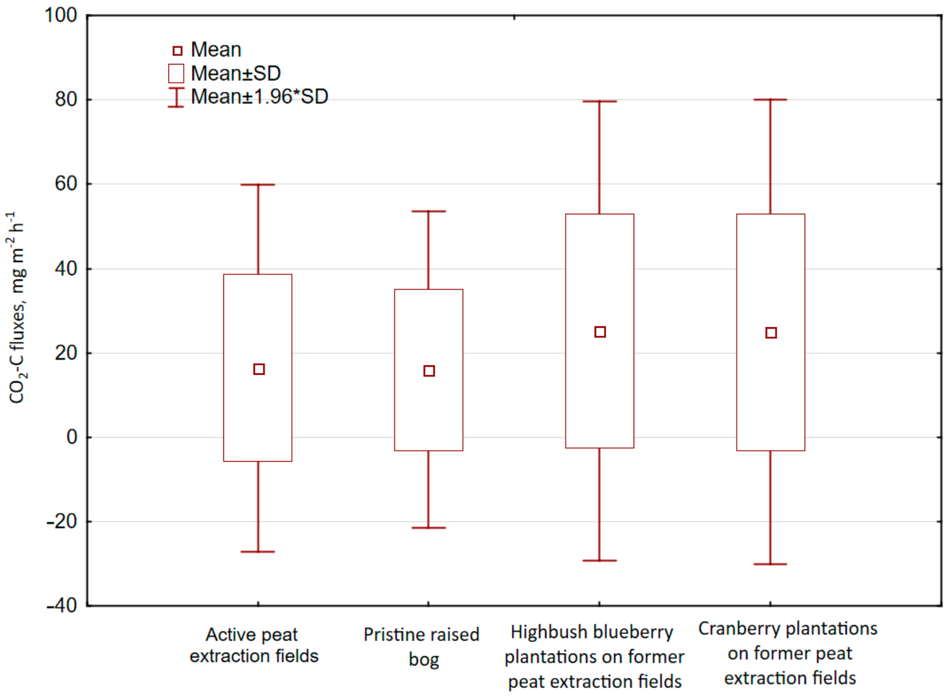

Figure 3. Variation in CO_2–C fluxes reflecting soil heterotrophic respiration depending on the type of land use and vegetation.

The highest mean CH_4-C fluxes were observed in the pristine raised bogs (1.43 mg CH_4-C m^{-2} h^{-1}, Figure 4), while the CV in the pristine raised bogs was the lowest (395.4%). In other types of land use, significantly lower mean CH_4-C fluxes were observed, with 0.12 mg CH_4-C m^{-2} h^{-1} in active peat extraction fields (CV 580.8%), 0.24 mg CH_4-C m^{-2} h^{-1} in highbush blueberry plantations (CV 495.8%), and 0.07 mg CH_4-C m^{-2} h^{-1} in cranberry plantations (CV 471.8%).

The magnitude of N_2O-N fluxes (Figure 5) was significantly lower compared to other GHGs. However, the highest mean N_2O-N fluxes were observed in highbush blueberry plantations (0.008 mg N_2O-N m^{-2} h^{-1}, CV 697.5%), while in cranberry plantations, the mean N_2O-N fluxes were lower (0.003 mg N_2O-N m^{-2} h^{-1}, CV 432.5%). The lowest mean N_2O-N fluxes were observed in the pristine raised bogs (0.002 mg N_2O-N m^{-2} h^{-1}); however, the CV was the highest (1097.0%).

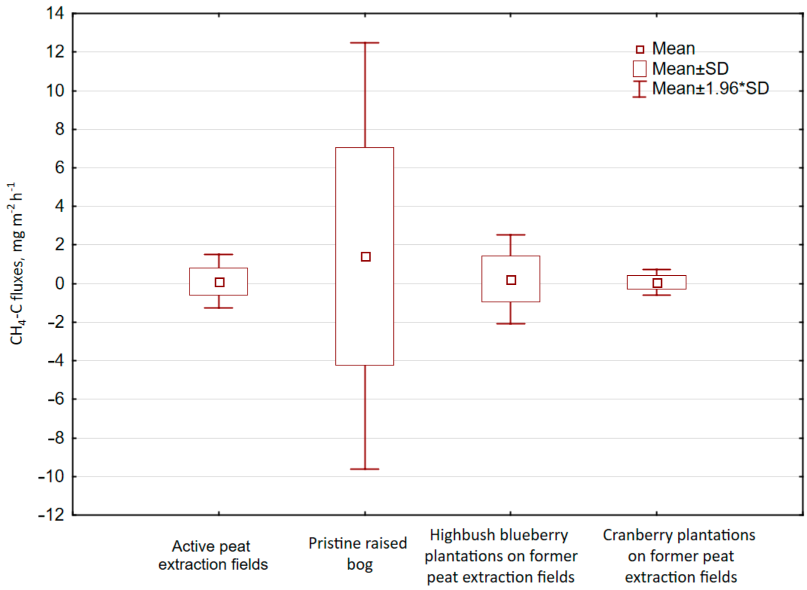

Figure 4. Variation in CH_4–C fluxes depending on the type of land use and vegetation.

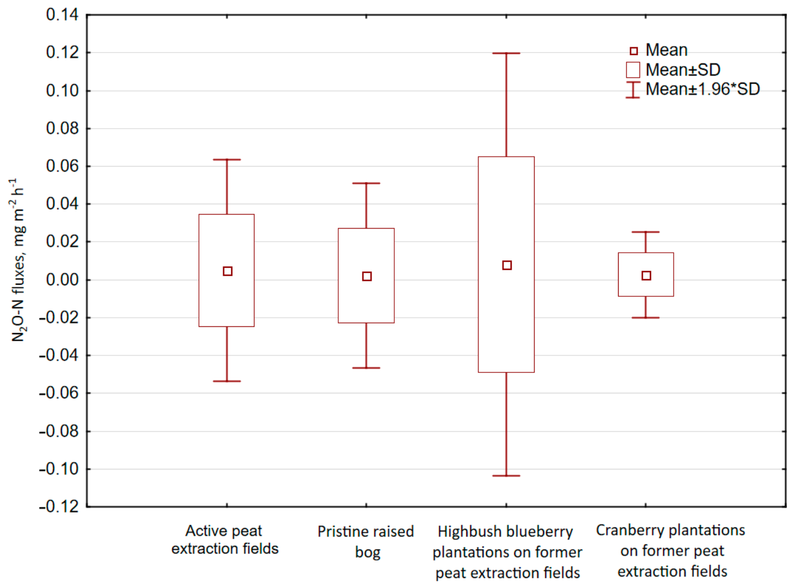

Figure 5. Variation in N_2O-N fluxes depending on the type of land use and vegetation.

To determine the most significant correlations between GHG fluxes and different environmental variables, Spearman rank correlation coefficients were calculated (Table 4). Air temperature, soil temperature at different depths, and groundwater temperature showed the most significant impact on CO_2–C fluxes (heterotrophic respiration). CH_4-C emissions were more dependent on soil moisture, groundwater pH, and oxidation-reduction potential (ORP). The impact of different environmental variables on the N_2O-N fluxes was quite insignificant; only soil moisture, groundwater pH, and conductivity had weak correlations (Table 4).

Table 4. Spearman rank order correlations (significant at $p < 0.05$) between GHG fluxes and environmental variables. CO_2–C fluxes reflecting soil heterotrophic respiration. Marked (bold) values represent significant correlations at $p < 0.05$.

Variable	Air Temp., °C	Soil Temp., 5 cm, °C	Soil Temp., 10 cm, °C	Soil Temp., 15 cm, °C	Soil Temp., 30 cm, °C	Soil Moisture, m^3 m^{-3}	Ground-Water Level, cm	Water Temp., °C	Water pH	Water ORP, mV	Water Cond., µS cm^{-1}	Water ODO, mg L^{-1}	CO_2–C, mg m^{-2} h^{-1}	CH_4–C, mg m^{-2} h^{-1}	N_2O–N, mg m^{-2} h^{-1}
Air temp., °C	1.00														
Soil temp., 5 cm, °C	**0.95**	1.00													
Soil temp., 10 cm, °C	**0.93**	**0.98**	1.00												
Soil temp., 15 cm, °C	**0.86**	**0.94**	**0.96**	1.00											
Soil temp., 30 cm, °C	**0.86**	**0.93**	**0.96**	**1.00**	1.00										
Soil moisture, m^3 m^{-3}	**0.10**	**0.17**	**0.19**	**0.22**	**0.21**	1.00									
Groundwater level, cm	**−0.13**	**−0.14**	**−0.14**	**−0.15**	**−0.15**	−0.02	1.00								
Water temp., °C	**0.71**	**0.80**	**0.83**	**0.90**	**0.90**	**0.51**	**−0.12**	1.00							
Water pH	**0.11**	**0.13**	**0.15**	**0.16**	**0.16**	**−0.39**	**−0.08**	**0.37**	1.00						
Water ORP, mV	**−0.20**	**−0.19**	**−0.18**	**−0.16**	**−0.16**	**0.38**	**0.09**	**−0.06**	**−0.45**	1.00					
Water cond., µS cm^{-1}	**0.19**	**0.21**	**0.21**	**0.23**	**0.24**	**−0.12**	**−0.11**	**0.26**	−0.03	**−0.06**	1.00				
Water ODO, mg L^{-1}	**−0.25**	**−0.24**	**−0.24**	**−0.22**	**−0.23**	**−0.11**	0.00	**−0.25**	0.01	**0.38**	**−0.20**	1.00			
CO_2–C, mg m^{-2} h^{-1}	**0.76**	**0.79**	**0.79**	**0.80**	**0.80**	**0.15**	**−0.13**	**0.73**	**0.15**	**−0.12**	**0.23**	**−0.24**	1.00		
CH_4-C, mg m^{-2} h^{-1}	0.01	0.03	0.03	**0.06**	**0.06**	**0.54**	**−0.11**	**0.08**	**−0.29**	**0.28**	**−0.13**	**−0.10**	**0.12**	1.00	
N_2O-N, mg m^{-2} h^{-1}	0.03	0.02	0.01	0.02	0.03	**−0.17**	−0.03	0.04	**0.16**	**−0.09**	**0.13**	0.04	**0.12**	**−0.12**	1.00

The variation in CO_2–C fluxes (soil heterotrophic respiration) depending on air temperature (Figure 6) and soil temperature at a depth of 5 cm (Figure 7) was best described by nonlinear (polynomial type) regressions. We obtained similar regression equations of the polynomial type for all the studied types of land use showing that the magnitude of CO_2–C fluxes increased with increasing air temperature. The main part of the experimental data (recorded fluxes) was within the predicted deviations (red dotted line, Figures 6 and 7) according to the obtained regression equations. The polynomial curve of the equation most accurately describes the experimental data, since the air temperature cannot increase exponentially or linearly. Despite some data points fitting linear or exponential curves, their application for the analysis is not justified due to actual temperature changes with alternating warmer and cooler periods.

The regression equations of CO_2–C fluxes (soil heterotrophic respiration) depending on soil temperature (Figure 7) are based on the results of temperature measurements at a soil depth of 5 cm. Strong correlations between CO_2–C fluxes and soil temperature in the deeper soil layers were also determined (Table 4). However, a decrease or increase in soil temperature at depth is only a matter of time and the resulting dependencies on measurements at depths of 10, 20, and 30 cm had similar trends. Similarly, a significant impact of changes in the groundwater temperature on the intensity of CO_2–C fluxes was found (Figure 8). Consequently, the temperature of the environment (air, soil, and groundwater) constitutes a mechanism that influences the intensity of CO_2–C fluxes.

Although correlations between CH_4 fluxes and soil moisture, groundwater pH, and the ORP were determined (Table 4), no reliable regression equations describing the variation in CH_4 fluxes were found (Appendix C, Figures A4–A6). However, given the presence of reliable Spearman correlation coefficients obtained from the cumulative analysis of experimental data (without stratification in types of land use), an increase in the number of measurements could have a clarifying impact on the strength of correlations and regressions describing the relationships between CH_4 fluxes and different potentially affecting factors.

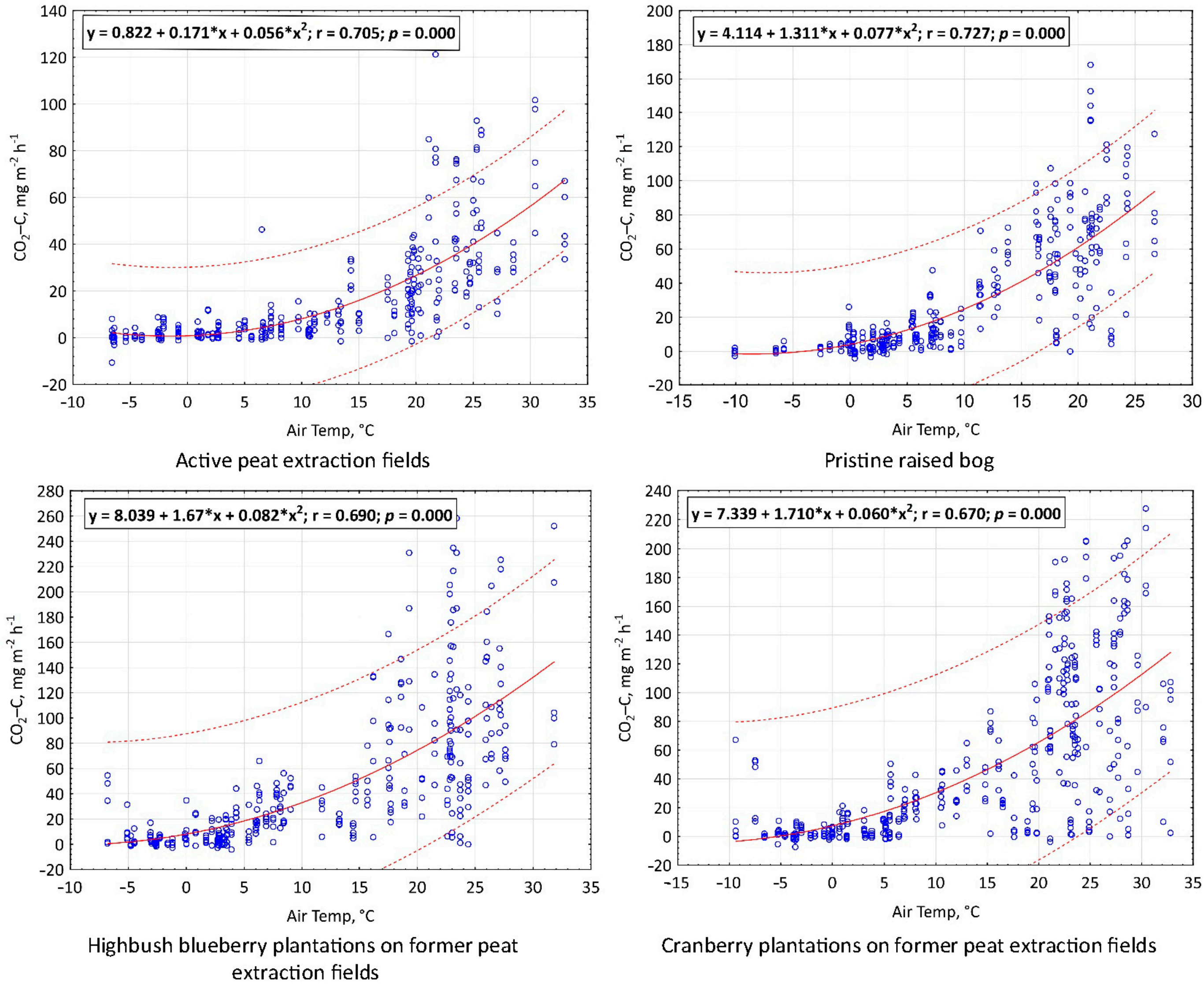

Figure 6. Polynomial regressions (red lines) describing dependence of CO_2–C fluxes (soil heterotrophic respiration, blue points) on air temperature in studied types of land use and vegetation. Area between red dashed lines corresponds to 95% confidence interval.

Based on the results of Spearman rank order correlations (Table 4), regressions describing the dependence of N_2O fluxes on changes in groundwater pH were analyzed (Appendix D, Figure A7). We obtained a reliable equation at a significance level of $p < 0.05$ only for cranberry plantations. However, the correlation coefficient indicated only a moderate strength in the relationship between the magnitude of N_2O fluxes and groundwater pH.

3.4. Annual Greenhouse Gas Fluxes

Estimated annual GHG fluxes, including net CO_2 fluxes from soil in the studied types of land use and vegetation, are shown in Table 5; default GHG emission factors for drained organic soils and rewetted areas provided by the Intergovernmental Panel on Climate Change (IPCC) [41] are added for comparison. Among the studied types of land use and vegetation, the highest mean annual R_{het} (2.23 ± 0.46 t CO_2-C ha^{-1} y^{-1}) was estimated in highbush blueberry plantations followed by cranberry plantations (2.14 ± 0.18 t CO_2-C ha^{-1} y^{-1}), while the lowest mean annual R_{het} (1.36 ± 0.19 t CO_2-C ha^{-1} y^{-1}) was observed in the pristine raised bog. Mean annual net CO_2 fluxes, calculated as the difference between the annual R_{het} and the annual C input with plant litter, ranged from slight removals

(-0.07 ± 0.68 t CO_2-C ha^{-1} y^{-1}) in the pristine raised bogs to 1.56 ± 0.19 t CO_2-C ha^{-1} y^{-1} in active peat extraction fields.

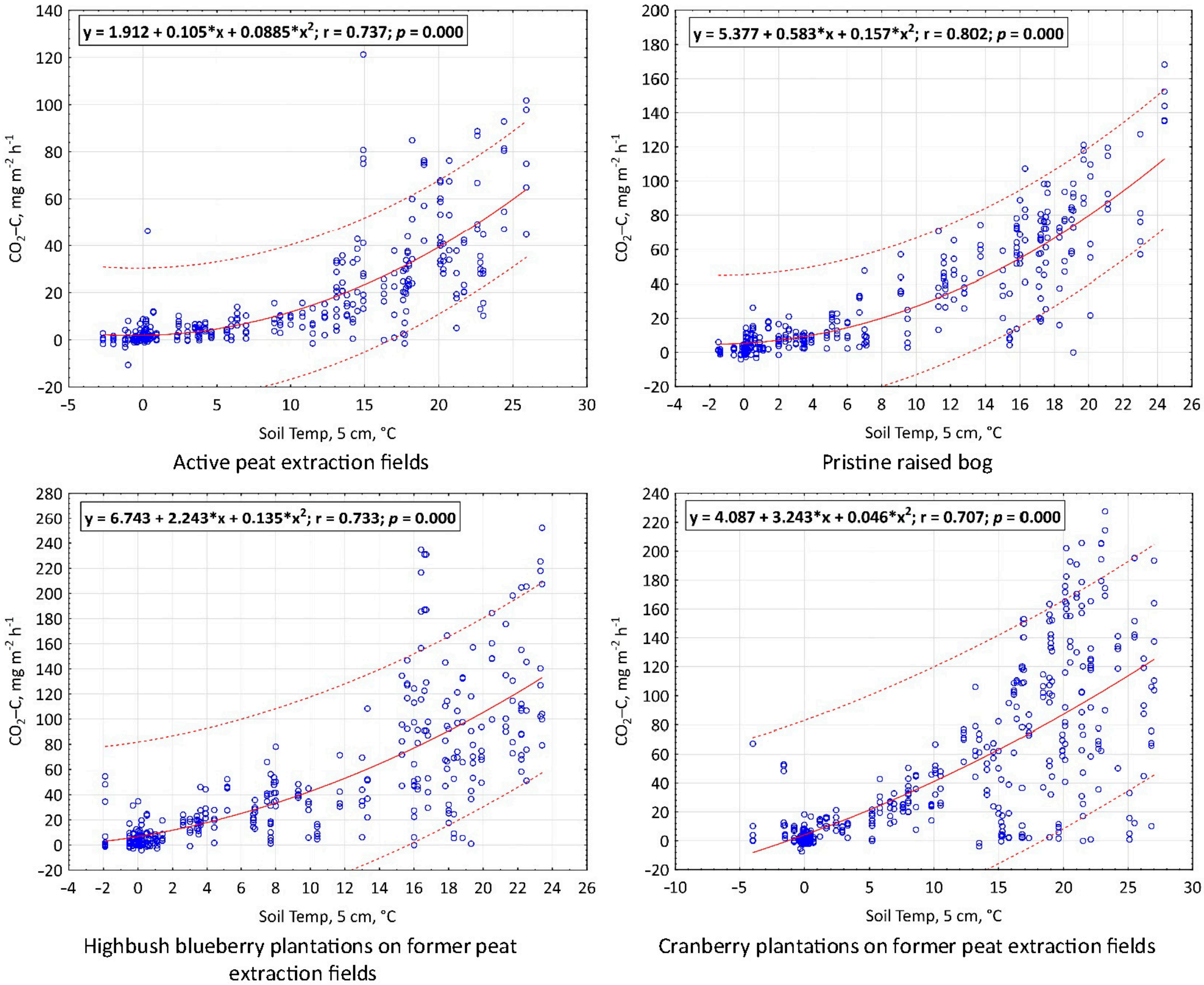

Figure 7. Polynomial regressions (red lines) describing dependence of CO_2–C fluxes (soil heterotrophic respiration, blue points) on soil temperature at a depth of 5 cm in studied types of land use and vegetation. Area between red dashed lines corresponds to 95% confidence interval.

Estimated annual CH_4 fluxes varied widely from 6.65 ± 1.77 kg CH_4-C ha^{-1} y^{-1} in cranberry plantations to 128.0 ± 27.5 kg CH_4-C ha^{-1} y^{-1} in pristine raised bogs. In contrast, annual N_2O fluxes varied in a narrow range from 0.18 ± 0.15 kg N_2O-N ha^{-1} y^{-1} in cranberry plantations to 0.65 ± 0.33 kg N_2O-N ha^{-1} y^{-1} in highbush blueberry plantations.

A summary of estimated cumulative annual GHG fluxes expressed in CO_2 equivalents using global warming potential (GWP) values for a 100-year time horizon [42] is provided in Figure 9. The highest total GHG fluxes (sum of annual net CO_2 fluxes, CH_4 fluxes, and N_2O fluxes) were observed in active peat extraction fields (6.23 t CO_2 eq. ha^{-1} y^{-1}), while the lowest were in cranberry plantations (1.50 t CO_2 eq. ha^{-1} y^{-1}). In active peat extraction fields, highbush blueberry, and cranberry plantations, the net CO_2 fluxes formed the largest contribution to total GHG emissions—from 67.6% in highbush blueberry plantations to 91.8% in active peat extraction fields. On the contrary, in the pristine raised bogs, CH_4 fluxes had the largest contribution to total GHG emissions. Furthermore, the combined emissions of CH_4 and N_2O (4.92 t CO_2 eq. ha^{-1} y^{-1}) in the pristine raised bogs significantly exceeded CO_2 removals (-0.26 t CO_2 eq. ha^{-1} y^{-1}), indicating that pristine raised bogs are a source of GHG emissions.

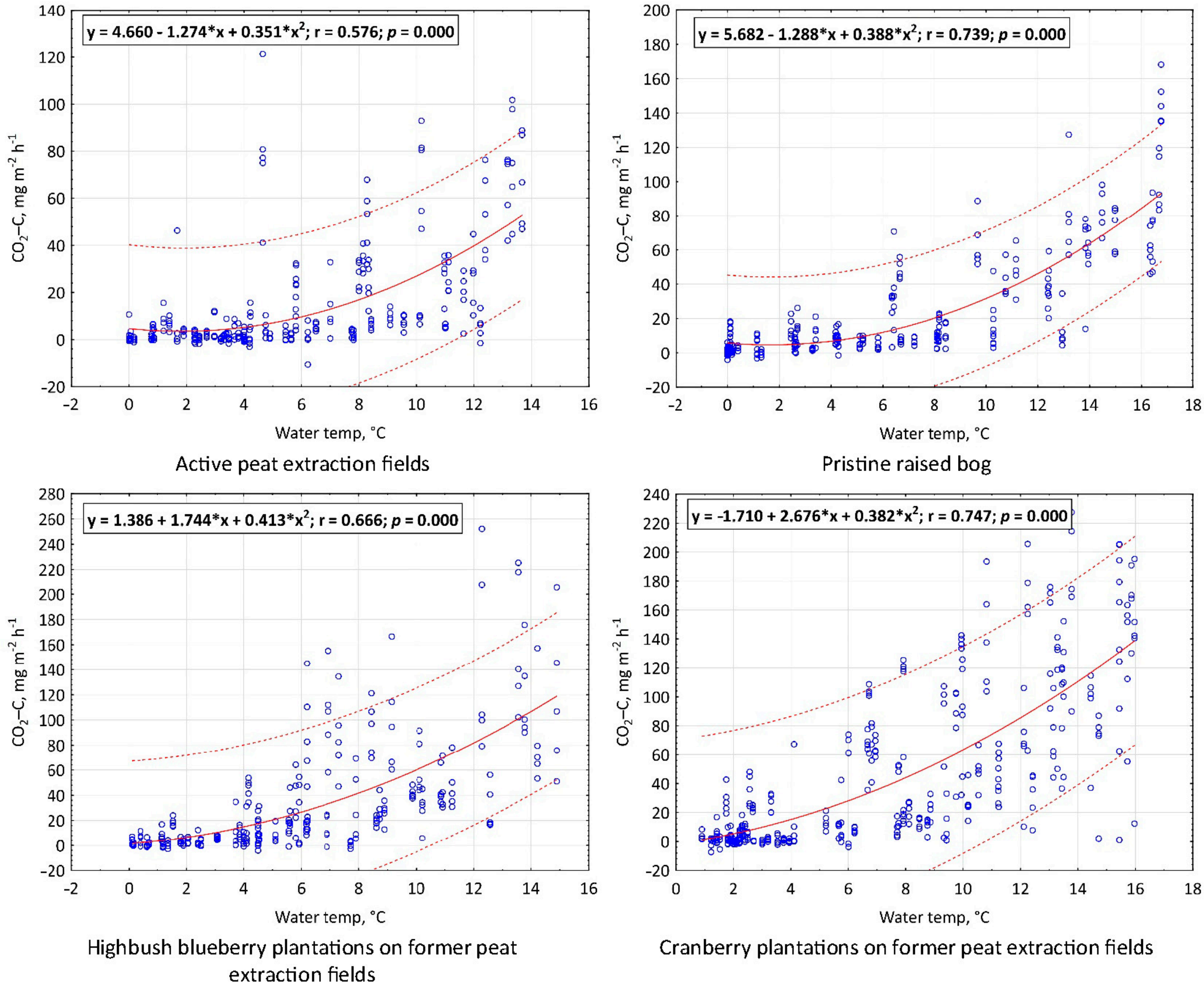

Figure 8. Polynomial regressions (red lines) describing dependence of CO_2–C fluxes (soil heterotrophic respiration, blue points) on groundwater temperature in studied types of land use and vegetation. Area between red dashed lines corresponds to 95% confidence interval.

Table 5. Estimated annual GHG fluxes (mean value ± S.E.) in the studied land use types. Negative values of annual GHG fluxes indicate the removal of GHG from the atmosphere. Default GHG emission factors for drained organic soils and rewetted areas (temperate climate zone, nutrient status—poor) provided by the IPCC [41] are added for comparison.

Annual GHG Fluxes	Unit	Studied Type of Land Use and Vegetation			
		Active Peat Extraction Fields	Pristine Raised Bog	Highbush Blueberry Plantations on Former Peat Extraction Fields	Cranberry Plantations on Former Peat Extraction Fields
Estimated annual soil heterotrophic respiration (R_{het})	t CO_2–C ha^{-1} y^{-1}	1.56 ± 0.19	1.36 ± 0.19	2.23 ± 0.46	2.14 ± 0.18
Annual carbon input with plant litter	t C ha^{-1} y^{-1}	-	1.43 ± 0.65	1.63 ± 1.12	1.82 ± 0.21
Annual net CO_2 fluxes	t CO_2–C ha^{-1} y^{-1}	1.56 ± 0.19	−0.07 ± 0.68	0.60 ± 1.21	0.32 ± 0.28
IPCC (2014) default CO_2 emission factor [41]	t CO_2–C ha^{-1} y^{-1}	2.8 (95% CI 1.1…4.2)	Rewetted organic soils, poor: −0.23 (95% CI −0.64…0.18)	Cropland: 7.9 (95% CI 6.5…9.4)	
Estimated annual CH_4 fluxes	kg CH_4–C ha^{-1} y^{-1}	10.6 ± 6.0	128.0 ± 27.5	21.0 ± 18.3	6.65 ± 1.77
IPCC (2014) default CH_4 emission factor [41]	kg CH_4–C ha^{-1} y^{-1}	4.6 (95% CI 1.2…8.3)	Rewetted organic soils, poor: 92 (95% CI 3…445)	Cropland: 0 (95% CI −2.1…2.1)	

Table 5. *Cont.*

Annual GHG Fluxes	Unit	Studied Type of Land Use and Vegetation			
		Active Peat Extraction Fields	**Pristine Raised Bog**	**Highbush Blueberry Plantations on Former Peat Extraction Fields**	**Cranberry Plantations on Former Peat Extraction Fields**
Estimated annual N_2O fluxes	kg N_2O–N ha^{-1} y^{-1}	0.28 ± 0.18	0.33 ± 0.30	0.65 ± 0.33	0.18 ± 0.15
IPCC (2014) default N_2O emission factor [41]	kg N_2O–N ha^{-1} y^{-1}	0.3 (95% CI −0.03. . .0.64)	Rewetted organic soils: negligible	Cropland: 13 (95% CI 8.2. . .18)	

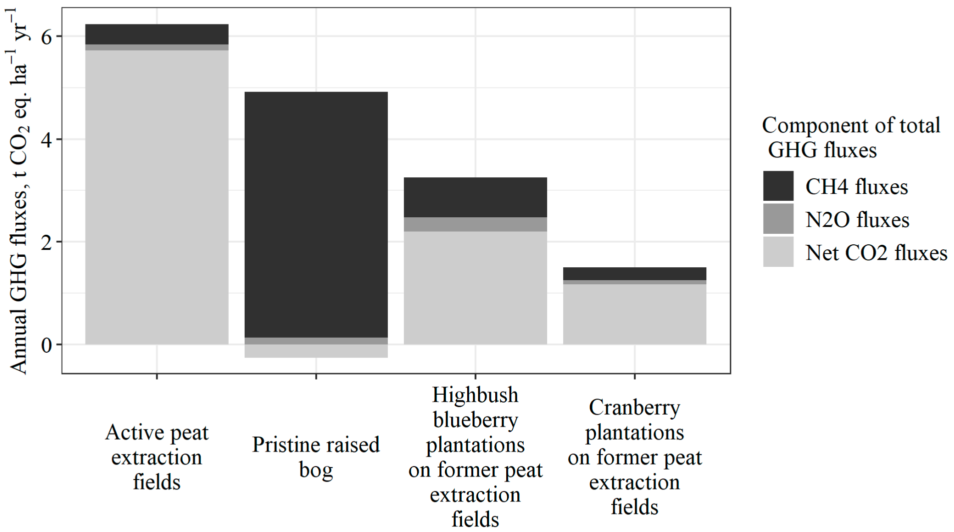

Figure 9. Cumulative annual GHG (CO_2, CH_4, and N_2O) fluxes among different studied types of land use and vegetation, recalculated to CO_2 equivalents using global warming potential values for a 100-year time horizon [42].

4. Discussion

Here, we estimated the annual net GHG (CO_2, CH_4, and N_2O) fluxes from soil in cranberry and highbush blueberry plantations established on former peat extraction fields compared to active peat extraction areas and pristine raised bogs. This study did not cover the contribution of GHG fluxes from drainage ditches, dissolved organic carbon exported from organic soils, and off-site CO_2 emissions associated with the use of peat to the total (ecosystem level) GHG budget.

All studied sites covering both pristine and managed areas were with raised bogs, also called ombrotrophic bogs, and corresponded to nutrient-poor organic soils. In topsoil layers (0–10 cm and 10–20 cm), the soil C/N ratios, which indirectly characterize the intensity of mineralization (decomposability of organic matter) and thus soil fertility [43], were above 20 (among individual soil samples), while the mean C/N ratios at different sites grouped depending on type of land use and vegetation were above 35, indicating potential microbial immobilization [43] and a low rate of N release per unit of organic matter [44]. The establishment and management of commercial berry plantations is one of the potential scenarios of the after-use of peat extraction areas with nutrient-poor organic soils. In such a case, respective areas can be accounted for under the agricultural land category within national GHG inventories following the IPCC guidelines. However, an important nuance that distinguishes drained organic soils used for highbush blueberry and cranberry plantations on former peat extraction fields from typical drained organic soils used for agriculture is a relatively low soil fertility (high soil C/N ratio) and pH (acidic environment) suitable for these berry plantations, contrary to conventional agriculture.

4.1. CO_2 Fluxes

Peatlands, in general, and northern peatlands, in particular, are one of the key C-sequestering ecosystems, storing one-third of global C stocks in soil [45]. Usually, pristine peatlands act as CO_2, one of the major GHGs, sinks; however, the balance can vary significantly depending on environmental conditions, including drought periods and subsequent rewetting and land use type and management activities [33]. Also, the results of our study (annual net CO_2 fluxes) showed that pristine raised bogs were slight CO_2 sinks (-0.07 ± 0.68 t CO_2-C ha^{-1} y^{-1}), while all other studied types of land use and vegetation were CO_2 sources. The highest annual net CO_2 fluxes were observed in active peat extraction fields (1.56 ± 0.19 t CO_2-C ha^{-1} y^{-1}), where the contribution of annual net CO_2 fluxes to total GHG emissions reached 91.8%. It was reported earlier [46] that in the first few years of peat extraction, residual labile C contained in the top peat layer promotes C mineralization and high levels of CO_2 emissions. Within 3−4 years, the respiration rate reaches a plateau and then declines over time. We also observed a lower annual heterotrophic respiration in active peat extraction fields compared to highbush blueberry and cranberry plantations, which may also be the result of the accumulation of inhibitory compounds such as lignins, phenolic, or humic substances that hamper the activity of extracellular enzymes [47]. At the same time, the use of former peat extraction areas for plant (including highbush blueberry and cranberry) cultivation probably leads to a reduction in the role of these inhibitory compounds due to the influence of agricultural activities.

We found that temperature (air, soil, and groundwater) was the most important influencing factor of CO_2 fluxes in all studied types of land use and vegetation. In general, temperature is one of the most frequently reported CO_2 flux (both ecosystem respiration and soil heterotrophic respiration)-affecting factors [19]. On the contrary, other researchers (e.g., [46]) reported that temperature and soil moisture, which are also typical influencing factors of CO_2 emission, had a relatively low impact on the soil (peat) respiration rate. According to Oestmann et al. [48], the groundwater level affected the net ecosystem exchange of CO_2 in *Sphagnum* sites on former peat extraction areas, also due to the drying of mosses. In addition to the hydrological characteristics of sites, the development of vegetation cover affects the net ecosystem exchange of CO_2, as vegetation cover contributes to increasing CO_2 fluxes for ecosystem respiration. Evidence suggests that abandoned peat extraction areas can become significant sources of CO_2 emissions during dry periods [3,49]. Therefore, leaving them without reclamation or cultivation is not advisable for CCM.

Our estimates of annual net CO_2 fluxes in active peat extraction fields were slightly lower (while within the 95% confidence interval) than the IPCC default emission factor provided for peatlands managed for extraction in boreal and temperate climates/vegetation zones (Table 5). While our estimates of annual net CO_2 fluxes in highbush blueberry and cranberry plantations on former peat extraction fields were significantly lower than the IPCC default emission factor provided for croplands with drained organic soils, a land use category under which commercial berry plantations can be accounted within national GHG inventories.

4.2. CH_4 Fluxes

CH_4 has a global warming potential (GWP) of 28 times that of CO_2 over a 100-year horizon [42]. Among the studied types of land use and vegetation, annual CH_4 fluxes ranged from 6.65 ± 1.77 kg CH_4-C ha^{-1} y^{-1} in cranberry plantations on former peat extraction fields to 128.0 ± 27.5 kg CH_4-C ha^{-1} y^{-1} in pristine raised bogs. In active peat extraction fields, the contribution of annual CH_4 fluxes to total annual GHG fluxes was comparatively low (6.3%). In cranberry and highbush blueberry plantations, the contribution of annual CH_4 fluxes was higher—16.6% and 24.1% of total annual GHG fluxes, respectively, while in the pristine raised bogs, annual CH_4 fluxes were the main contributor to total annual net GHG fluxes (102.6%). A significant difference in the amount of annual CH_4 fluxes between pristine raised bogs and the other studied types of land use and vegetation was related to soil moisture conditions. In the pristine raised bogs, the mean groundwater level was 7.0 ± 0.56 cm, while in the active peat extraction fields and berry plantations, the mean groundwater level was in a range from 41.6 to 52.3 cm (Table 3). It is well known that CH_4 fluxes released into the atmosphere result from two contrary microbial metabolic processes (CH_4 production and CH_4 oxidation) controlled by the groundwater level and thus, oxygen availability. A shallow groundwater table or a permanently waterlogged condition causes anaerobic conditions that favor methanogenesis, while drainage and the consequent downward movement of the groundwater level causes aerobic conditions that enhance methanotrophy [50].

According to Oestmann et al. [48], CH_4 fluxes increased with an increase in the average daily soil temperature. However, in our studies, no reliable correlation between these environmental variables and CH_4 fluxes was found (Table 4), whereas at the sites of distribution of *Eriophorum angustifolium* and *Eriophorum vaginatum*, Oestmann et al. [48] obtained higher CH_4 emissions, similar to our results. The researchers explained such patterns by the fact that wetland plants, in particular, moss, have aerenchyma tissues that contribute to better gas transport. Similar patterns were also described by other researchers [51–54].

Among all studied types of land use and vegetation, our estimates of annual CH_4 fluxes were higher than the latest default IPCC CH_4 emission factors (Table 5). In the active peat extraction fields and berry plantations, estimated annual CH_4 fluxes even exceeded 95% confidence intervals of CH_4 emission factors provided by the IPCC for peat extraction fields and drained croplands, respectively, while our estimates of annual CH_4 fluxes in the pristine raised bogs were within the 95% confidence intervals of CH_4 emission factors provided by the IPCC for rewetted organic soils with poor nutrient status (Table 5).

4.3. N_2O Fluxes

Among the studied GHGs, N_2O is the most potent GHG, with a GWP 265 times that of CO_2 over a 100-year horizon [42]. Although N_2O has the highest GWP, the contribution of annual N_2O fluxes to total annual GHG fluxes was comparatively low—ranging from 1.9% in active peat extraction fields to 8.3% in highbush blueberry plantations in former peat extraction fields. However, all studied types of land use and vegetation acted as slight N_2O sources with mean annual N_2O emissions in a range from 0.18 ± 0.15 kg N_2O-N ha^{-1} y^{-1} in cranberry plantations to 0.65 ± 0.33 kg N_2O-N ha^{-1} y^{-1} in highbush blueberry plantations (Table 5).

Contrary to Yao et al. [55], we did not find a close relationship between the N concentration in the soil and annual N_2O emissions, although our estimates of the intensity of N_2O fluxes depending on the type of land use and soil general chemistry (Figure 2) agreed with their findings. Other researchers [56–58] also could not find a strong relationship between N concentration in soil and N_2O fluxes. It is considered that N fertilizers boost N_2O production by providing a substrate for microbial N conversion through nitrification and denitrification [59]. However, other researchers, e.g., [60] reported that high concentrations of nitrates (NO_3^-) would not necessarily lead to high N_2O fluxes. When growing plants without the application of fertilizers, the effect of anthropogenic factors on the processes of nitrogen denitrification in the soil becomes more significant [57,58]. At the same time, anaerobic conditions in the soil occurring simultaneously with a sufficient amount of NO_3^- probably lead to a more intensive denitrification process [61,62]. This explains insignificant but reliable correlations between N_2O emissions, soil moisture, and groundwater characteristics in our studies (Table 4).

Our estimates of annual N_2O fluxes in active peat extraction fields were in line with the latest [41] N_2O emission factors for peatland managed for extraction in boreal and temperate climates/vegetation zones provided by the IPCC (Table 5). Under saturated conditions and sequentially low oxygen availability, it is considered that N_2O fluxes are negligible due to limited nitrification and denitrification processes, two complementary processes that produce N_2O. This assumption is also provided by the IPCC as Tier 1 level for rewetted organic soils [41]. However, our estimates showed that annual N_2O fluxes (emissions) in pristine raised bogs were as high as in the active peat extraction fields, while estimated annual N_2O fluxes from drained organic soils in highbush blueberry and cranberry plantations were significantly lower than the IPCC default N_2O emission factor provided for drained organic soils in croplands in the whole boreal and temperate climates/vegetation zones. Furthermore, our estimates were even significantly lower than the 95% confidence interval of the IPCC default N_2O emission factor for croplands. Most likely, the significantly lower estimated annual N_2O fluxes in highbush blueberry and cranberry plantations can be explained by the low decomposability of organic matter (high soil C/N ratio associated with microbial N immobilization [44]). For instance, Yao et al. [63] found that annual N_2O fluxes' dependence on the soil C/N ratio follows an optimum Gaussian curve, with a threshold at a C/N ratio of about 18–19. Similarly, Klemedtsson et al. [64] reported a strong negative correlation between N_2O fluxes and the soil C/N ratio across *Histosols*.

5. Conclusions

Establishing commercial berry plantations is one of the potential scenarios for the temporal further use of peat extraction areas (on raised bogs). However, these soils have a relatively low fertility (high C/N ratio) and an acidic environment (low pH). This distinguishes these soils from typical drained organic soils used for conventional agriculture.

The estimated cumulative annual net GHG (sum of net CO_2, CH_4, and N_2O) fluxes in cranberry and highbush blueberry plantations (1.50 and 3.25 t CO_2 eq. ha^{-1} y^{-1}, respectively) were notably lower than those in active peat extraction fields (6.23 t CO_2 eq. ha^{-1} y^{-1}) and pristine raised bogs (4.66 t CO_2 eq. ha^{-1} y^{-1}). Thus, the establishment of cranberry and highbush blueberry plantations on former peat extraction fields (on raised bogs) can be considered as a CCM measure to reduce GHG emissions from soil in peat extraction fields.

Among studied type of land use, the highest annual net CO_2 fluxes were observed in active peat extraction fields (1.56 ± 0.19 t CO_2-C ha^{-1} y^{-1}), while pristine raised bogs were slight CO_2 sinks (−0.07 ± 0.68 t CO_2-C ha^{-1} y^{-1}). Temperature was the most important environmental variable influencing CO_2 fluxes across all studied land use types. The contribution of annual CH_4 fluxes to total cumulative annual net GHG fluxes in all studied land use types was relatively low (6.3–24.1%), excluding pristine raised bogs, where annual CH_4 fluxes (128.0 ± 27.5 kg CH_4-C ha^{-1} y^{-1}) were the main contributor. The significantly higher annual CH_4 fluxes in pristine raised bogs in comparison to the other studied land use types were related to different soil moisture conditions that depend on the groundwater level. All studied land use types were minor sources of N_2O fluxes, with average annual N_2O emissions ranging from 0.18 ± 0.15 kg N_2O-N ha^{-1} y^{-1} in cranberry plantations to 0.65 ± 0.33 kg N_2O-N ha^{-1} y^{-1} in highbush blueberry plantations. However, no clear relationships were found between annual N_2O emissions and different environmental variables, including soil N concentration.

Future research should include more frequent (including diurnal measurements) and long-term GHG flux monitoring to further improve the estimation of the contribution of organic soils to overall GHG emissions from land use, land use change, and forestry (LULUCF) sector. Additionally, future research should focus on the potential impacts of climate change and weather extremes, including periods of drought, on the magnitude of GHG fluxes, not only in managed land with organic soils, but also in pristine ecosystems (peatlands).

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Appendix A

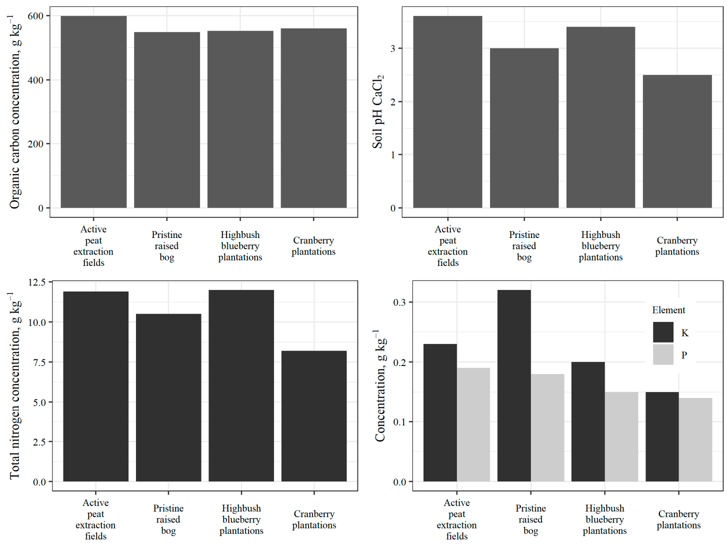

Figure A1. Mean values of soil chemical variables in the soil layer of 50–100 cm.

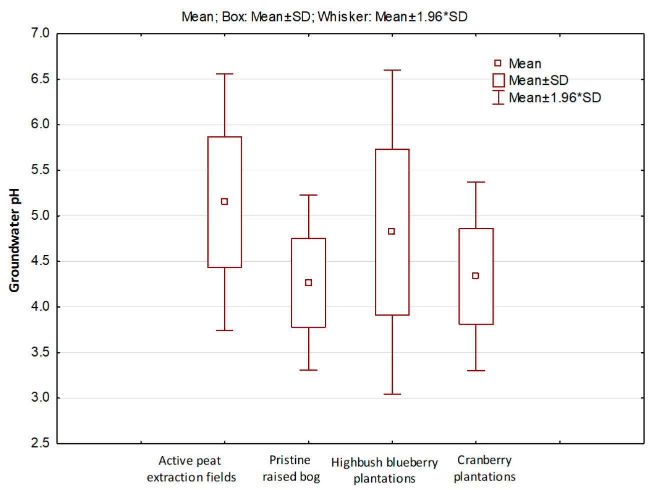

Figure A2. Variation in groundwater pH depending on the type of land use and vegetation.

Appendix B

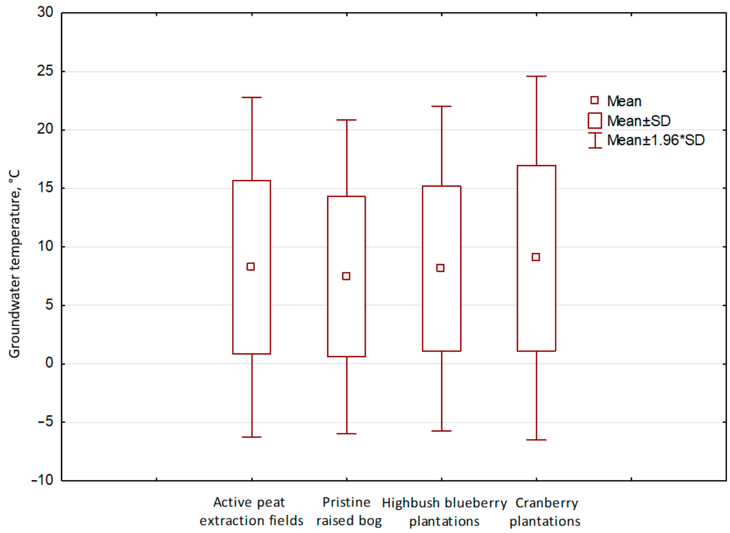

Figure A3. Variation in groundwater temperature depending on the type of land use and vegetation.

Appendix C

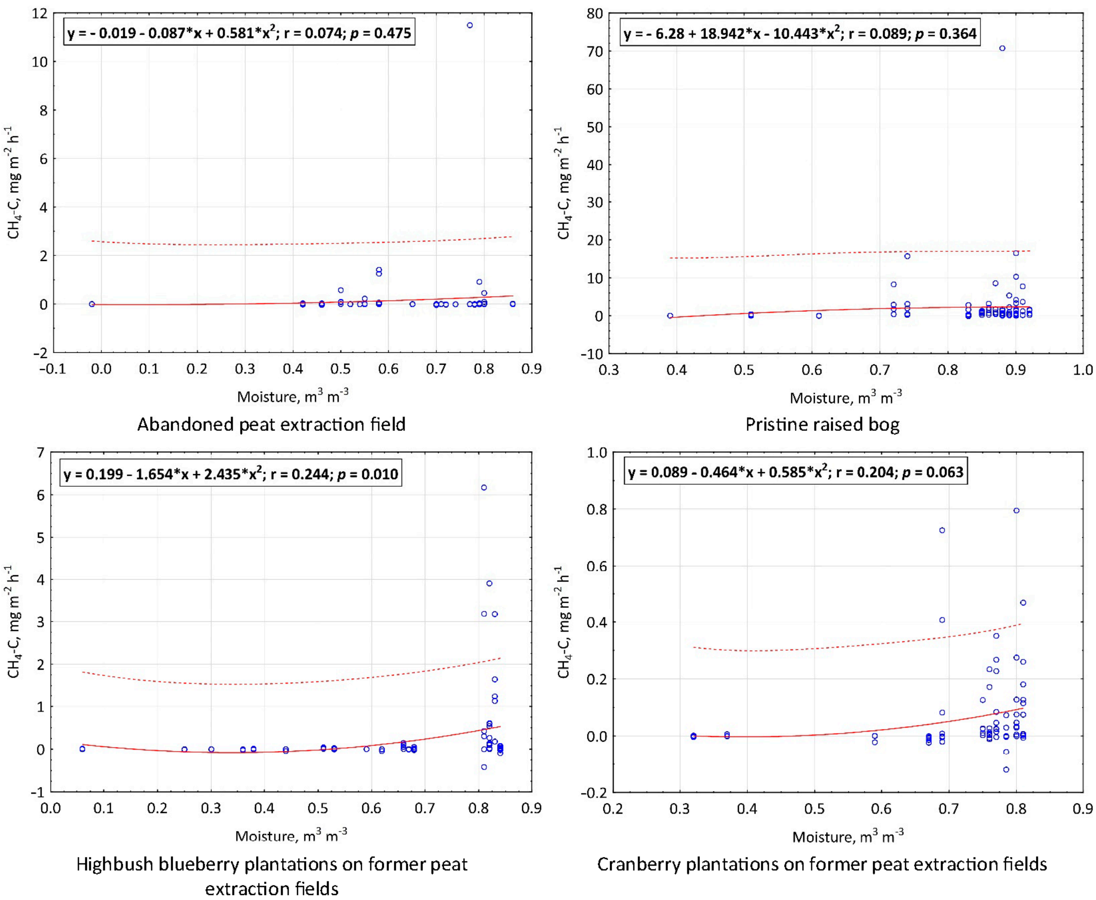

Figure A4. Polynomial regressions (red lines) describing dependence of CH_4 fluxes (blue points) on soil moisture in studied types of land use and vegetation. Area between red dashed lines corresponds to 95% confidence interval.

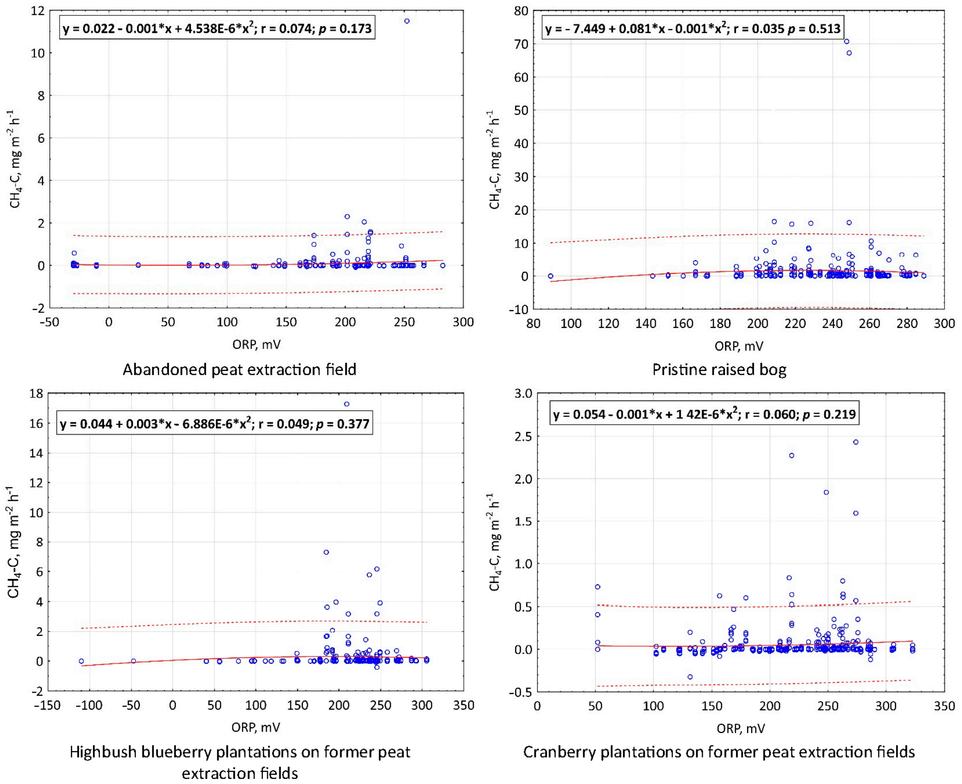

Figure A5. Polynomial regressions (red lines) describing dependence of CH_4 fluxes (blue points) on groundwater oxidation-reduction potential (ORP) in studied types of land use and vegetation. Area between red dashed lines corresponds to 95% confidence interval.

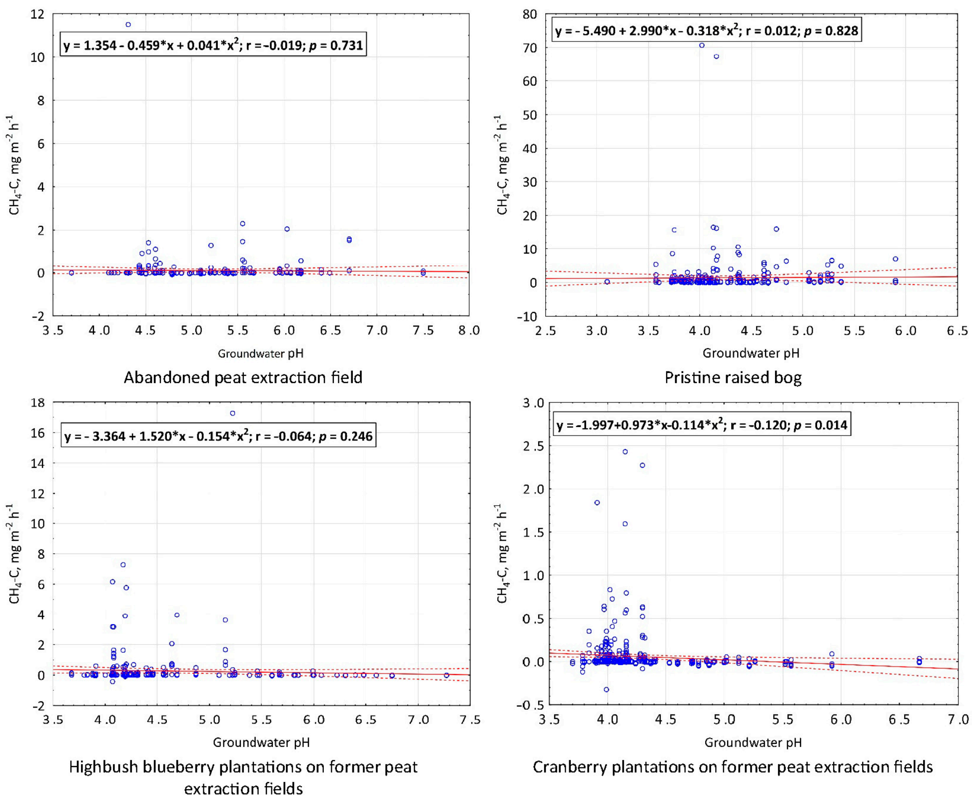

Figure A6. Polynomial regressions (red lines) describing dependence of CH_4 fluxes (blue points) on groundwater pH in studied types of land use and vegetation. Area between red dashed lines corresponds to 95% confidence interval.

Appendix D

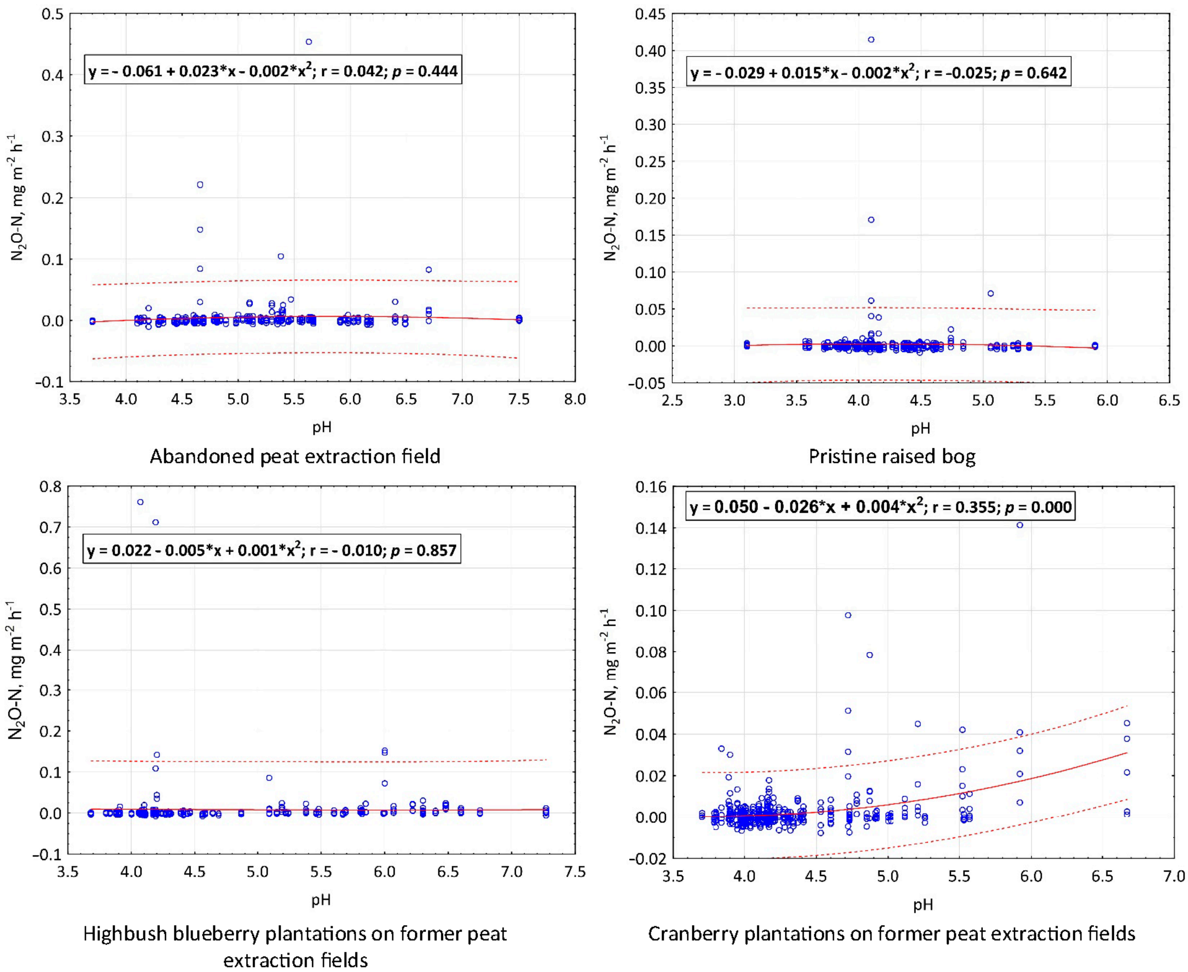

Figure A7. Polynomial regressions (red lines) describing dependence of N_2O fluxes (blue points) on groundwater pH in studied types of land use and vegetation. Area between red dashed lines corresponds to 95% confidence interval.

References

1. Müller, J.; Joos, F. Global peatland area and carbon dynamics from the Last Glacial Maximum to the present—A process-based model investigation. *Biogeosciences* **2020**, *17*, 5285–5308. [CrossRef]
2. Xu, J.; Morris, P.J.; Liu, J.; Holden, J. PEATMAP: Refining estimates of global peatland distribution based on a meta-analysis. *Catena* **2018**, *160*, 134–140. [CrossRef]
3. Strack, M.; Davidson, S.J.; Hirano, T.; Dunn, C. The potential of peatlands as nature-based climate solutions. *Curr. Clim. Change Rep.* **2022**, *8*, 71–82. [CrossRef]
4. Courouble, M.; Davidson, N.; Dinesen, L.; Fennessy, S.; Galewski, T.; Guelmami, A.; Kumar, R.; McInnes, R.; Perennou, C.; Rebelo, L.-M.; et al. *Convention on Wetlands. Global Wetland Outlook: Special Edition 2021*; Secretariat of the Convention on Wetlands: Gland, Switzerland, 2021.
5. Davies, M.; Gardner, T.; Gearey, B.; de Haas, T.; Schepers, M.; Stratigos, M.J. Between reclamation and restoration: The archaeology, historical ecology and future development of drained wetland landscapes. *J. Wetl. Archaeol.* **2022**, *22*, 1–6. [CrossRef]
6. UNFCCC. Adoption of the Paris Agreement. In Proceedings of the 21st Conference of the Parties, United Nations, Paris, France, 30 November–11 December 2015.
7. FAO. Peatlands and Climate Planning—Part 1: Peatlands and Climate Commitments. Available online: https://openknowledge.fao.org/handle/20.500.14283/cc2895en (accessed on 15 August 2024).
8. Ghezelayagh, P.; Oleszczuk, R.; Stachowicz, M.; Eini, M.R.; Kamocki, A.; Banaszuk, P.; Grygoruk, M. Developing a remote-sensing-based indicator for peat soil vertical displacement. A case study in the Biebrza Valley, Poland. *Ecol. Indic.* **2024**, *166*, 112305. [CrossRef]

9. Orru, M. Re-vegetation processes in cutaway peat production fields in Estonia in relation to peat quality and water regime. *Environ. Monit. Assess* **2016**, *188*, 655. [CrossRef]
10. Karofeld, E.; Jarašius, L.; Priede, A.; Sendžikaitė, J. On the after-use and restoration of abandoned extracted peatlands in the Baltic countries. *Restor. Ecol.* **2017**, *25*, 293–300. [CrossRef]
11. Freeman, B.W.J.; Evans, C.D.; Musarika, S.; Morrison, R.; Newman, T.R.; Page, S.E.; Wiggs, G.F.S.; Bell, N.G.A.; Styles, D.; Wen, Y.; et al. Responsible agriculture must adapt to the wetland character of mid-latitude peatlands. *Glob. Change Biol.* **2022**, *28*, 3795–3811. [CrossRef]
12. Mander, Ü.; Espenberg, M.; Melling, L.; Kull, A. Peatland restoration pathways to mitigate greenhouse gas emissions and retain peat carbon. *Biogeochemistry* **2024**, *167*, 523–543. [CrossRef]
13. Abolins, M.; Sausserde, R.; Liepniece, M.; Sterne, D. Cranberry and blueberry production in Latvia. *Latv. J. Agron.* **2009**, *12*, 7–13.
14. Abolina, L.; Osvalde, A.; Karlsons, A. Habitat characteristics and mineral nutrition status of *Rubus chamaemorus* L. in Latvia. *Plants* **2023**, *12*, 528. [CrossRef]
15. Bardule, A.; Jurmalis, E.; Libiete, Z.; Paulina, I.; Donis, J.; Treimane, A. Use of retail market data to assess prices and flows of non-wood forest products in Latvia. *Silva Fennica* **2020**, *54*, 10341. [CrossRef]
16. Osvalde, A.; Karlsons, A.; Cekstere, G. Diagnostics of nutrient status of soil for highbush blueberries in Latvia, 2014-2018. *IV Balk. Symp. Fruit Grow.* **2020**, *1289*, 295–301. [CrossRef]
17. Vahejoe, K.; Albert, T.; Noormets, M.; Karp, K.; Paal, T.; Starast, M.; Värnik, R. Berry cultivation in cutover peat lands in Estonia: Agricultural and economical aspects. *Balt. For.* **2010**, *16*, 264–272.
18. Osvalde, A.; Pormale, J.; Karlsons, A.; Cekstere, G. Differences in agrochemical characteristics of highbush blueberry peat and mineral soils in Latvia, 2011–2014. In Proceedings of the 15th International Multidisciplinary Scientific Geoconference (SGEM), Albena, Bulgaria, 18–24 June 2015.
19. Lazdiņš, A.; Lupiķis, A. LIFE REstore project contribution to the greenhouse gas emission accounts in Latvia. In *Sustainable and Responsible after-Use of Peat Extraction Areas*; Priede, A., Gancone, A., Eds.; Baltijas Krasti: Riga, Latvia, 2019; pp. 21–54.
20. Osvalde, A. *Amerikas Lielogu Dzērveņu un Krūmmelleņu Minerālās Barošanās Apstākļi Latvijā 2007–2009. g. ELFLA Projekts "Ilgtspējīgas Augļkopības Attīstība, Izmantojot vidi un Ūdeņus Saudzējošas, kā arī Lauku Ainavu Saglabājošas Integrētās Audzēšanas Tehnoloģijas Klimata Pārmaiņu Mazināšanai un Bioloģiskās Daudzveidības Nodrošināšanai" Nr. 080410/c-32*; Latvijas Valsts Augļkopības Institūts: Dobeles novads, Latvija, 2011.
21. The Rural Support Service of Latvia. Available online: https://www.lad.gov.lv/lv/platibu-maksajumu-statistika (accessed on 15 August 2024).
22. Hatano, R. Impacts of changes in peat soils due to agricultural activities on greenhouse gases (especially N_2O) emissions and their mitigations. *Pedosphere* **2023**, *in press*. [CrossRef]
23. Epron, D. Separating autotrophic and heterotrophic components of soil respiration: Lessons learned from trenching and related root-exclusion experiments. In *Soil Carbon Flux Measurements: An Integrated Methodology*; Kutsch, W.L., Bahn, M., Heinemeyer, A., Eds.; Cambridge University Press: Cambridge, UK, 2009; pp. 157–168.
24. Wohlfahrt, G.; Anderson-Dunn, M.; Bahn, M.; Balzarolo, M.; Berninger, F.; Campbell, C.; Carrara, A.; Cescatti, A.; Christensen, T.; Dore, S.; et al. Biotic, abiotic, and management controls on the net ecosystem CO_2 exchange of European mountain grassland ecosystems. *Ecosystems* **2008**, *11*, 1338–1351. [CrossRef]
25. Lévesque, V.; Rochette, P.; Hogue, R.; Jeanne, T.; Ziadi, N.; Chantigny, M.H.; Dorais, M.; Antoun, H. Greenhouse gas emissions and soil bacterial community as affected by biochar amendments after periodic mineral fertilizer applications. *Biol. Fertil. Soils* **2020**, *56*, 907–925. [CrossRef]
26. Bārdule, A.; Butlers, A.; Spalva, G.; Ivanovs, J.; Meļņiks, R.N.; Līcīte, I.; Lazdiņš, A. The surface-to-atmosphere GHG fluxes in rewetted and permanently flooded former peat extraction areas compared to pristine peatland in hemiboreal Latvia. *Water* **2023**, *15*, 1954. [CrossRef]
27. LEGMC. Klimata Portals. Available online: https://klimats.meteo.lv/operativais_klimats/laikapstaklu_apskati/arhivs/2016/gads/ (accessed on 15 August 2024).
28. Hutchinson, G.L.; Livingston, G.P. Use of chamber systems to measure trace gas fluxes. In *Agricultural Ecosystem Effects on Trace Gases and Global Climate Change*; Harper, L.A., Mosier, A.R., Duxbury, J.M., Rolston, D.E., Eds.; American Society of Agronomy: Madison, WI, USA, 1993; pp. 63–78.
29. Loftfield, N.; Flessa, H.; Augustin, J.; Beese, F.O. Automated gas chromatographic system for rapid analysis of the atmospheric trace gases methane, carbon dioxide, and nitrous oxide. *J. Environ. Qual.* **1997**, *26*, 560–564. [CrossRef]
30. Rankin, T.; Roulet, N.; Moore, T. Controls on autotrophic and heterotrophic respiration in an ombrotrophic bog. *Biogeosciences* **2022**, *19*, 3285–3303. [CrossRef]
31. Cools, N.; De Vos, B. Part X: Sampling and analysis of soil. In *Manual on Methods and Criteria for Harmonized Sampling, Assessment, Monitoring and Analysis of the Effects of Air Pollution on Forests*; UNECE ICP Forests Programme Co-ordinating Centre, Ed.; Thünen Institute of Forest Ecosystems: Eberswalde, Germany, 2010.
32. Inisheva, L.I.; Golovatskaya, E.A. Elements of carbon balance in oligotrophic bogs. *Russ. J. Ecol.* **2002**, *33*, 242–248. [CrossRef]
33. Moore, T.R.; Bubier, J.L.; Frolking, S.E.; Lafleur, P.M.; Roulet, N.T. Plant biomass and production and CO_2 exchange in an ombrotrophic bog. *J. Ecol.* **2002**, *90*, 25–36. [CrossRef]

34. Malmer, N.; Wallen, B. Input rates, decay losses and accumulation rates of carbon in bogs during the last millennium: Internal processes and environmental changes. *Holocene* **2004**, *14*, 111–117. [CrossRef]
35. Malmer, N.; Svensson, G.; Wallen, B. Carbon and mass balance in a south Swedish ombrotrophic bog: Processes and variation during recent centuries. *Mires Peat* **2011**, *8*, 1–16.
36. Lazdiņš, A. Above and Below Ground Biomass in Three Producing Cranberry Plantations in Latvia. Available online: https://zenodo.org/records/13321481 (accessed on 15 August 2024).
37. Palosuo, T.; Heikkinen, J.; Regina, K. Method for estimating soil carbon stock changes in Finnish mineral cropland and grassland soils. *Carbon Manag.* **2015**, *6*, 207–220. [CrossRef]
38. Cassell, A. Using R for Statistics: A Beginner's Manual. In *Resident's Handbook of Medical Quality and Safety*; Atanelov, L., Ed.; Springer International Publishing: Cham, Switzerland, 2016; pp. 361–387.
39. R Core Team. The R Project for Statistical Computing. Available online: https://www.R-project.org (accessed on 14 January 2024).
40. Minasny, B.; McBratney, A.B.; Bellon-Maurel, V.; Roger, J.M.; Gobrecht, A.; Ferrand, L.; Joalland, S. Removing the effect of soil moisture from NIR diffuse reflectance spectra for the prediction of soil organic carbon. *Geoderma* **2011**, *167*, 118–124. [CrossRef]
41. IPCC. *2013 Supplement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories: Wetlands*; Hiraishi, T., Krug, T., Tanabe, K., Srivastava, N., Baasansuren, J., Fukuda, M., Troxler, T.G., Eds.; IPCC: Geneva, Switzerland, 2014; Available online: https://www.ipcc-nggip.iges.or.jp/public/wetlands/index.html (accessed on 14 August 2024).
42. Myhre, G.; Shindell, D.; Bréon, F.-M.; Collins, W.; Fuglestvedt, J.; Huang, J.; Koch, D.; Lamarque, J.-F.; Lee, D.; Mendoza, B.; et al. Anthropogenic and natural radiative forcing. In *Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*; Stocker, T.F., Qin, D., Plattner, G.-K., Tignor, M., Allen, S.K., Boschung, J., Nauels, A., Xia, Y., Bex, V., Midgley, P.M., Eds.; Cambridge University Press: Cambridge, UK; New York, NY, USA, 2013.
43. Brust, G.E. Chapter 9—Management strategies for organic vegetable fertility. In *Safety and Practice for Organic Food*; Biswas, D., Micallef, S.A., Eds.; Academic Press: Cambridge, MA, USA, 2019; pp. 193–212.
44. Springob, G.; Kirchmann, H. Bulk soil C to N ratio as a simple measure of net N mineralization from stabilized soil organic matter in sandy arable soils. *Soil Biol. Biochem.* **2003**, *35*, 629–632. [CrossRef]
45. Hugelius, G.; Loisel, J.; Chadburn, S.; Jackson, R.B.; Jones, M.; MacDonald, G.; Marushchak, M.; Olefeldt, D.; Packalen, M.; Siewert, M.B.; et al. Large stocks of peatland carbon and nitrogen are vulnerable to permafrost thaw. *Proc. Natl. Acad. Sci. USA* **2020**, *117*, 20438–20446. [CrossRef]
46. Clark, L.; Strachan, I.; Strack, M.; Roulet, N.; Knorr, K.-H.; Teickner, H. Duration of extraction determines CO_2 and CH_4 emissions from an actively extracted peatland in eastern Quebec, Canada. *Biogeosciences* **2023**, *20*, 737–751. [CrossRef]
47. Hogg, E.H.; Lieffers, V.J.; Wein, R.W. Potential carbon losses from peat profiles: Effects of temperature, drought cycles, and fire. *Ecol. Appl. Publ. Ecol. Soc. Am.* **1992**, *2*, 298–306. [CrossRef]
48. Oestmann, J.; Dettmann, U.; Düvel, D.; Tiemeyer, B. Experimental warming increased greenhouse gas emissions of a near-natural peatland and *Sphagnum* farming sites. *Plant Soil* **2022**, *480*, 85–104. [CrossRef]
49. Wilson, S. Grasses and grassland ecology. *Ann. Bot.* **2009**, *104*, ix. [CrossRef]
50. Huang, Y.; Ciais, P.; Luo, Y.; Zhu, D.; Wang, Y.; Qiu, C.; Goll, D.S.; Guenet, B.; Makowski, D.; De Graaf, I.; et al. Tradeoff of CO_2 and CH_4 emissions from global peatlands under water-table drawdown. *Nat. Clim. Change* **2021**, *11*, 618–622. [CrossRef]
51. Waddington, J.; Day, S. Methane emissions from a peatland following restoration. *J. Geophys. Res.* **2007**, *112*, 1–11. [CrossRef]
52. Leroy, F.; Gogo, S.; Guimbaud, C.; Bernard-Jannin, L.; Yin, X.; Belot, G.; Shuguang, W.; Laggoun-Défarge, F. CO_2 and CH_4 budgets and global warming potential modifications in *Sphagnum*-dominated peat mesocosms invaded by *Molinia caerulea*. *Biogeosci. Discuss.* **2019**, *16*, 4085–4095. [CrossRef]
53. Rigney, C.; Wilson, D.; Renou-Wilson, F.; Müller, C.; Moser, G.; Byrne, K.A. Greenhouse gas emissions from two rewetted peatlands previously managed for forestry. *Mires Peat* **2018**, *21*, 1–23.
54. Henneberg, A.; Elsgaard, L.; Sorrell, B.; Brix, H.; Petersen, S. Does *Juncus effusus* enhance methane emissions from grazed pastures on peat? *Biogeosci. Discuss.* **2015**, *12*, 8467–8495. [CrossRef]
55. Yao, Z.; Pelster, D.E.; Liu, C.; Zheng, X.; Butterbach-Bahl, K. Soil N intensity as a measure to estimate annual N_2O and NO fluxes from natural and managed ecosystems. *Curr. Opin. Environ. Sustain.* **2020**, *47*, 1–6. [CrossRef]
56. Davidson, E.A.; Keller, M.; Erickson, H.E.; Verchot, L.V.; Veldkamp, E. Testing a conceptual model of soil emissions of nitrous and nitric oxides. *BioScience* **2000**, *50*, 667–680. [CrossRef]
57. Thomson, A.J.; Giannopoulos, G.; Pretty, J.; Baggs, E.M.; Richardson, D.J. Biological sources and sinks of nitrous oxide and strategies to mitigate emissions. *Philos. Trans. R. Soc. B Biol. Sci.* **2012**, *367*, 1157–1168. [CrossRef]
58. Wang, J.; Yan, X. Denitrification in upland of China: Magnitude and influencing factors. *J. Geophys. Res. Biogeosci.* **2016**, *121*, 3060–3071. [CrossRef]
59. Chataut, G.; Bhatta, B.; Joshi, D.; Subedi, K.; Kafle, K. Greenhouse gases emission from agricultural soil: A review. *J. Agric. Food Res.* **2023**, *11*, 100533. [CrossRef]
60. Zebarth, B.J.; Rochette, P.; Burton, D.L. N_2O emissions from spring barley production as influenced by fertilizer nitrogen rate. *Can. J. Soil Sci.* **2008**, *88*, 197–205. [CrossRef]
61. Fowler, D.; Coyle, M.; Skiba, U.; Sutton, M.A.; Cape, J.N.; Reis, S.; Sheppard, L.J.; Jenkins, A.; Grizzetti, B.; Galloway, J.N.; et al. The global nitrogen cycle in the twenty-first century. *Philos. Trans. R. Soc. B Biol. Sci.* **2013**, *368*, 20130164. [CrossRef]

62. Bouwman, A.F.; Boumans, L.J.M.; Batjes, N.H. Emissions of N_2O and NO from fertilized fields: Summary of available measurement data. *Glob. Biogeochem. Cycles* **2002**, *16*, 6-1–6-13. [CrossRef]
63. Yao, Z.; Yan, G.; Ma, L.; Wang, Y.; Zhang, H.; Zheng, X.; Wang, R.; Liu, C.; Wang, Y.; Zhu, B.; et al. Soil C/N ratio is the dominant control of annual N_2O fluxes from organic soils of natural and semi-natural ecosystems. *Agric. For. Meteorol.* **2022**, *327*, 109198. [CrossRef]
64. Klemedtsson, L.; Arnold, K.; Weslien, P.; Gundersen, P. Soil CN ratio as scalar parameter to predict nitrous oxide emissions. *Glob. Change Biol.* **2005**, *11*, 1142–1147. [CrossRef]

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