

Effects of natural rewetting on soil respiration and temperature sensitivity (Q_{10}) under contrasting water table regimes: A case study in a Belgian floodplain

Nicolas Kovacs^{a,*}, Gilles Colinet^a, Bernard Longdoz^a, Jens Leifeld^b, Suzanna Lettens^c, Maud Raman^c, Karen Vancampenhout^d, Gert Verstraeten^e, Jeroen Meersmans^a

^a TERRA Teaching and Research Center, Gembloux Agro-Bio Tech, University of Liège, Gembloux, 5030, Belgium

^b Climate and Agriculture Group, Agroscope, Reckenholzstrasse 191, Zurich, 8046, Switzerland

^c Research Institute for Nature and Forest (INBO), Havenlaan 88, Brussels, 1000, Belgium

^d Department of Earth & Environmental Sciences, Division Forest, Nature and Landscape, KU Leuven campus Geel, Kleinhofstraat 4, Geel, 2240, Belgium

^e Department of Earth & Environmental Sciences, Division of Geography and Tourism, KU Leuven, Celestijnenlaan 200E, Heverlee, Leuven, 3001, Belgium

ARTICLE INFO

Keywords:

Soil respiration
Temperature sensitivity (Q_{10})
Soil moisture
Land use
Floodplain
Rewetting

ABSTRACT

Floodplains store large amounts of soil carbon, but hydrological changes strongly regulate its release into the atmosphere. Floodplains in Belgium are representative of temperate ecosystems extensively modified by drainage and land-use change, yet the combined effects of hydrology and land use on soil respiration remain unknown. In the Dijle valley, Belgium, we measured in situ soil carbon dioxide (CO_2) emissions across a freely drained plateau and two floodplain zones characterized by moderately and poorly drained soils, under representative land uses including forest, grassland, cropland, and marsh. Measurements conducted during the wettest year on record were used to quantify basal respiration (R_{10}) and assess the effects of rewetting and drainage on temperature sensitivity (Q_{10}). Additional measurements assessed potential methane (CH_4) emissions. Basal respiration followed the order: poorly drained < freely drained < moderately drained for forest and grassland (forests: 0.65, 1.16, and 1.51 $\mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$; grasslands: 1.13, 1.61, and 1.88 $\mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$, respectively). Overall, Q_{10} was not affected by soil moisture on the plateau. In moderately drained sites, Q_{10} peaked at intermediate moisture and strongly decreased at higher moisture, while in the poorly drained sites, Q_{10} increased with decreasing soil moisture. CH_4 contributed substantially to combined greenhouse gas emissions only in the poorly drained sites. These results suggest that maintaining water table below but close to the soil surface through rewetting could be an effective management strategy to reduce floodplain soil CO_2 emissions and their temperature sensitivity.

1. Introduction

Floodplains are relatively flat areas adjacent to rivers that experience periodic flooding, often driven by river overflow or groundwater rise. Although these ecosystems cover only 0.5 to 1% of the Earth's land area (D'Elia et al., 2017), they play a key role in valley systems by attenuating water flows, transporting dissolved or sediment materials and sequestering carbon (C) from the atmosphere (Sutfin et al., 2016; Wohl, 2021). Temperate floodplains can store a large amount of C in their soils, particularly in Belgium where floodplains and other poorly drained soils are considered hotspots for soil organic carbon (SOC) storage (Meersmans et al., 2008; Ottoy et al., 2022; Swinnen et al., 2020a).

At the same time, floodplains maintain a delicate balance between long-term C storage and greenhouse gas (GHG) emissions. Their capacity to act as stable C sinks is increasingly threatened by climatic variability, including changes in temperature and precipitation patterns, as well as by anthropogenic pressures, particularly artificial drainage for agricultural purposes (Karamouz et al., 2011; Tiemeyer et al., 2020). In Belgium, Meersmans et al. (2011) found that soils that were naturally poorly drained lost substantial SOC after being artificially drained during the 20th century, illustrating how hydrological alterations can directly accelerate soil C loss. This dependence of SOC stocks on hydrological conditions is consistent with earlier findings showing a strong correlation between groundwater levels and SOC stocks in northern

* Corresponding author.

E-mail address: nicolas.kovacs@uliege.be (N. Kovacs).

<https://doi.org/10.1016/j.agrformet.2026.111321>

Received 13 January 2026; Received in revised form 6 May 2026; Accepted 21 June 2026

Available online 6 July 2026

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Belgium, with the relationship being strongest for the mean lowest water table (MLW) (Meersmans et al., 2008). These observations underscore the need to better understand carbon dioxide (CO₂) fluxes in floodplains under changing hydrological conditions to ensure they can contribute positively to climate regulation.

Most studies investigating C in humid environments have focused on typical wetland systems (e.g. river deltas, lakes, fens), whereas floodplains without permanent standing water remain much less studied (Swinen et al., 2020b). However, interest in these ecosystems has increased in recent years (Wohl, 2021). Floodplains show unique flux dynamics because of their variable flooding conditions and frequent shifts between aerobic and anaerobic phases driven by groundwater table fluctuations. In Belgium, they also include different land uses, such as forests, grasslands, and croplands, which further increases their spatial heterogeneity (Swinen et al., 2020a). In this context, both land use and water regime can strongly influence soil respiration (Buysse et al., 2016; Ratcliffe et al., 2019), yet studies that combine multiple land uses along hydrological gradients in the field are lacking.

Soil respiration represents a key process linking soil C to atmospheric CO₂ emissions and is mainly driven by soil temperature, often with an exponential response. This temperature sensitivity is commonly expressed as Q_{10} and reflects how strongly soil respiration responds to changes in temperature, with higher values indicating a greater increase

in CO₂ flux for a given temperature rise (Davidson and Janssens, 2006; Reichstein et al., 2005b). Although Q_{10} has historically been considered constant, accounting for environmental variability in Q_{10} appears important to reduce uncertainties in C modelling (Davidson et al., 2006; Meyer et al., 2018). Soil respiration can also be influenced by variations in soil moisture and water table, which can modulate its temperature sensitivity. For example, Miao et al. (2013) showed that using a Q_{10} that accounts for water table variations improved the predictions of CO₂ fluxes in a forested wetland. These results highlight that accounting for hydrological variability is crucial for accurately predicting CO₂ fluxes in floodplain soils.

Recently, Heger et al. (2023) identified soil texture and pH as important factors controlling the annual total soil CO₂ fluxes in German floodplain forests. However, forests are still rarely included in field studies that explicitly compare multiple land uses along hydrological gradients. This limitation is particularly important given that hardwood floodplain forests can store substantial SOC (Heger et al., 2021) and cover significant areas in European floodplains. In the Dijle floodplain, which is the focus of the present study, forests cover approximately 30% of the land area (Fig. 1a).

Most studies investigating the effects of hydrological fluctuations, including soil drainage and rewetting, have been conducted under controlled laboratory conditions, and their extension to field conditions

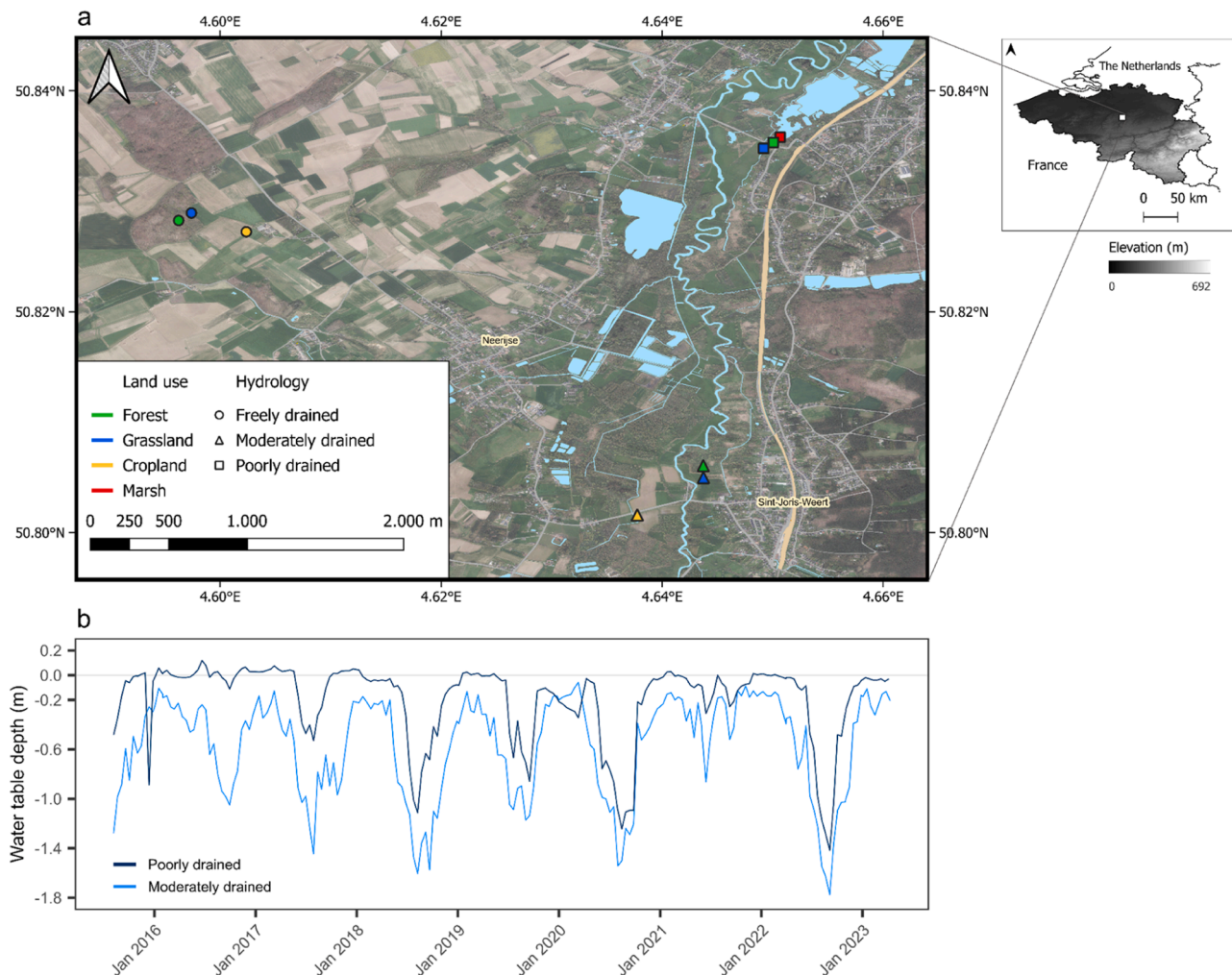


Fig. 1. (a) Location of the study area in the Dijle valley, Belgium. Each point represents one site defined by a unique combination of land use and hydrological condition, with four PVC collars used for soil flux measurements at each site. (b) Temporal evolution of water table depth (m) over 8 years prior to the experiment in the moderately drained and poorly drained floodplain sites. The soil surface (0 m) is shown as a horizontal grey line. The piezometer in the poorly drained zone is located approximately 10 m east of the forest measurement point, and the piezometer in the moderately drained zone is located approximately 90 m east of the grassland measurement point.

remains very limited (Rousk and C. Brangari, 2022). In addition, floodplains in Belgium are representative of systems that have been modified by long-term drainage and land-use change. This is particularly the case for the Dijle floodplain (Ateeq-Ur-Rehman et al., 2022; Swinnen et al., 2020a), which therefore provides a relevant setting to assess soil CO₂ flux responses to hydrological change under field conditions. The present study directly addresses key priorities by studying soil respiration and its temperature sensitivity across forest, grassland, cropland, and marsh sites, under contrasting hydrological regimes. These flux data are essential for improving the management of temperate floodplains and for understanding how land use and water regime jointly control GHG emissions. Given that floodplain soils often experience high and variable moisture conditions, which can trigger methane (CH₄) emissions (Mitsch et al., 2013), we also considered the additional role of CH₄ in the soil C fluxes. We hypothesized that soil CO₂ fluxes would be lower at wetter sites with shallow water tables, while the temperature sensitivity of soil respiration would differ across hydrological regimes. We also expected that CH₄ emissions would contribute more to combined GHG emissions under consistently wet conditions. The objectives are therefore: (i) to compare soil CO₂ fluxes across contrasting hydrological conditions and land uses, (ii) to evaluate how changes in soil water content impact the temperature sensitivity of soil respiration (Q₁₀), and (iii) to quantify the contribution of CH₄ to combined GHG emissions.

2. Materials and methods

2.1. Study site description and selection of measurement points

The study area was located in the Dijle valley, central Belgium (Fig. 1). According to the Köppen climate classification, the region has a temperate oceanic climate, fully humid with warm summers (Cfb). The mean annual precipitation is 785 mm, and the mean annual temperature is 10.9 °C. The average growing season begins in mid-May and ends in October. Frost can typically occur from early November until April. The average incoming solar global radiation (R_g) reaches a maximum in June (220.8 W m⁻²) and a minimum in December (20.8 W m⁻²).

Sampling was based on three distinct hydrological conditions, hereafter referred to as freely drained (plateau), moderately drained, and poorly drained soils. Three sites were selected within each setting to represent the dominant land uses, resulting in a total of nine sampling sites (Fig. 1a).

The freely drained sites on the plateau were located northwest of the valley, at higher topographic positions, and included forest, grassland, and cropland. The other two site groups were situated within the floodplain and defined according to the methodology of Van Heesen (1970) and Van Der Sluijs & De Gruijter (1985). These floodplain sites are characterized by temporary flooding resulting from groundwater rise rather than direct river overflow. In this approach, hydrological regimes are characterized by the mean highest water table (MHW) and the mean lowest water table (MLW), based on fortnightly piezometer measurements. For each hydrological year (April–March), MHW and MLW were calculated as the averages of the three highest and three lowest water levels, respectively, using records spanning at least eight consecutive years.

Table 1 presents the two floodplain site groups and their contrasting characteristics. In the moderately drained sites, water table depth varied considerably with the seasons, with higher levels in winter (corresponding to MHW) and lower levels in summer (corresponding to MLW) (Fig. 1b). These soils were located in the southern floodplain, where forest, grassland, and cropland were present. In contrast, the poorly drained sites were characterized by a consistently shallow water table, occasionally reaching above the soil surface, with seasonal fluctuations of much lower amplitude. They were located in the northeastern part of the floodplain, with land uses restricted to forest, grassland, and marsh (cropland was absent due to persistently wet soils). The two floodplain

Table 1

Hydrological classification of the three site groups, indicating the symbols used in Fig. 1. Mean highest water table (MHW) and mean lowest water table (MLW), as well as the drainage class, are based on the observed hydrological regime recorded by piezometers and classified after Van Ranst & Sys (2000) and Meersmans et al. (2008). The complete drainage classification system for northern Belgium can be found in Supplementary Table 1.

Hydrological condition	Symbol	MHW (m)	MLW (m)	Drainage class
Freely drained	○	–	–	b
Moderately drained	△	–0.15	–1.22	i
Poorly drained	□	0.03	–0.70	f

hydrological conditions mainly differed in water table depth within the top 10 cm of the soil profile, where microbial activity relevant for soil respiration is generally highest (Fig. 1b). These two drainage classes are also associated with contrasting soil physicochemical properties in northern Belgium, further supporting their distinction as two hydrologically contrasting conditions (Meersmans et al., 2008).

During the experiment, groundwater table depth was monitored continuously in the floodplain sites using the same two piezometers, allowing for the characterization of temporal hydrological dynamics (Fig. 2c). Monthly air temperature and precipitation data were obtained from the nearby Zaventem meteorological station to provide regional climatic context for the study period (Figs. 2a and b). This location has historically shown very similar monthly precipitation patterns to the Dijle valley at the regional scale. Precipitation data were used to describe seasonal conditions and climatic anomalies rather than to explain soil respiration responses.

There has been no change in land use since at least 1979. Active river management through dredging, deepening, and widening stopped in 1990, after which the floodplain was left to natural hydrological processes (Ateeq-Ur-Rehman et al., 2022). Forests were composed of broadleaf deciduous species, primarily alder (*Alnus glutinosa*), ash (*Fraxinus excelsior*), and oak (*Quercus petraea*). Grasslands mainly consisted of perennial ryegrass (*Lolium* spp.). Floodplain grasslands also contained grass typical of humid conditions, such as *Alopecurus pratensis*. All grasslands were mowed during the study, except for the small areas surrounding the soil respiration collars (see Section 2.2, Supplementary Fig. 1), which were left untouched.

The freely drained grassland was also grazed, but the soil respiration area was protected from livestock by a fence throughout the study. The freely drained cropland initially grew white mustard (*Sinapis alba*). The entire plot was ploughed for summer beet cultivation, and the soil respiration collars were temporarily removed during the operation. However, beets did not grow within the collar area. The moderately drained cropland was left unmanaged during the study and consisted of clover species, such as *Trifolium* spp.

2.2. CO₂ flux measurements

Soil respiration (R_s, μmol CO₂ m⁻² s⁻¹) was measured by placing four PVC collars (10 cm diameter, 7 cm high) in a uniform pattern at each of the nine sites (Supplementary Fig. 1) to facilitate a reliable comparison of CO₂ flux measurements. The collars were inserted 6 cm into the mineral soil, leaving a 1 cm headspace above the ground. This setup enabled for the capture of both heterotrophic respiration from soil organic matter (OM) mineralization and autotrophic respiration from root activity. A single set of four collars per site was used following the approach of Deng et al. (2010). This number was considered sufficient because the plots were relatively homogeneous in vegetation and topography, therefore minimizing the within-site variability. Consequently, statistical analyses were performed at the level of the nine study sites, while PVC collars were treated as technical replicates rather than independent experimental units.

We carried out fortnightly CO₂ measurements starting on 31 October

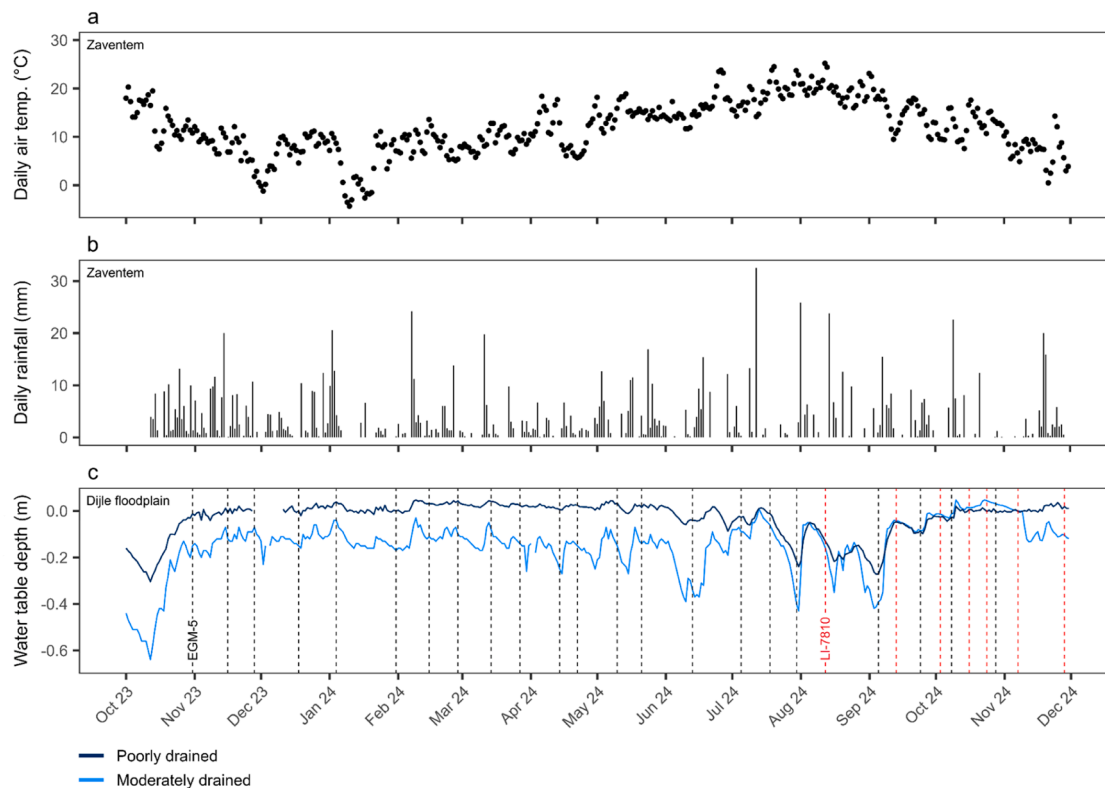


Fig. 2. (a) Daily air temperature (°C) and (b) daily rainfall (mm) recorded at the METAR station in Zaventem, located 9 km northwest of the Dijle valley, used to describe the regional climatic context during the study period. (c) Temporal evolution of water table depth (m) during the experiment, measured with two piezometers in the Dijle floodplain at the poorly and moderately drained sites. Vertical dashed black lines mark CO₂ flux measurement days with the EGM-5 ($n = 22$), and vertical dashed red lines indicate simultaneous CH₄/CO₂ flux measurements with the LI-7810 ($n = 7$).

2023, two weeks after collar placement to allow fluxes to stabilize following the disturbance caused by the installation (Mills et al., 2011), and continued until 28 October 2024. Measurements were not taken immediately after heavy rainfall to avoid temporary respiration pulses (Xu et al., 2004), ensuring that R_s was assessed under steady soil moisture conditions representative of each hydrological regime. Vegetation within the PVC collars was manually clipped and removed, as well as dead leaves, to prevent interference with CO₂ released by decaying fresh OM and with aboveground photosynthesis. Throughout the experiment, the collars remained installed, except during agricultural activities in the freely drained cropland. They were removed and then reinstalled once the work was completed. When the site was flooded and the collars were submerged in the floodplain, CO₂ measurements were not taken. Flooding periods ranged from one month in the moderately drained grassland to around seven months in the poorly drained grassland, and up to eight months in the poorly drained marsh (Fig. 3). In addition, measurements across the three hydrological conditions of the valley (freely drained, moderately drained, and poorly drained) were taken in random order to avoid bias from the diurnal hysteresis between R_s and temperature (Bahn et al., 2008).

R_s was measured over five minutes in a closed-loop configuration by placing a respiration chamber on the collar and connecting it to an infrared gas analyzer (IRGA) EGM-5 (PP Systems, USA). CO₂ flux rate ($\mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$) was calculated as follows:

$$R_s = \frac{dC}{dt} \frac{V_t}{V_{\text{mol}} A} \quad (1)$$

Where $\frac{dC}{dt}$ is the slope of the linear increase of CO₂ concentration over time (ppm s^{-1}), V_t is the sum of the chamber's internal volume and the headspace volume enclosed between the soil surface and the top of the collar (m^3), A is the soil surface area covered by the chamber (m^2), and

V_{mol} is the molar volume of CO₂ ($22.41 \cdot 10^{-3} \text{ m}^3 \text{ mol}^{-1}$ at standard temperature and pressure). The CO₂ readings of the IRGA were automatically corrected for ambient temperature and pressure measured inside the chamber during each flux measurement. The portable IRGA has a measurement range of 0–1000 ppm and an accuracy of less than 1% of span concentration. The soil respiration chamber (SRC-2, 10 cm diameter, 15 cm height) was also equipped with a stainless-steel ring to maintain a seal to the collar, and an internal fan ensured proper air mixing.

During each CO₂ measurement, soil temperature (T_s , °C) and volumetric water content (SWC, $\text{m}^3 \text{ m}^{-3}$) were measured adjacent to each collar ($n = 4$) using a probe inserted into the top 0–10 cm of soil. The sensor (5TM, Decagon Devices, USA) measured the dielectric constant at 70 MHz (with an accuracy of $\pm 2\%$ SWC) and had been previously calibrated using a third-degree polynomial equation.

2.3. Temperature dependence

To quantify the dependence of R_s on soil temperature (T_s), we employed a commonly used exponential Q_{10} function (Buyse et al., 2017; Jonard et al., 2007), hereafter referred to as Model I (Eq. (2)):

$$R_s = R_{10} Q_{10}^{(T_s - 10)/10} \quad (2)$$

Where R_s is the measured soil respiration rate ($\mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$), R_{10} is the basal respiration rate at a reference temperature of 10 °C, which is very close to the annual average temperature of the Dijle valley, and Q_{10} is the temperature sensitivity of CO₂ flux, representing the factor by which R_s increases for every 10 °C rise in T_s . Model I was used to derive a single site-specific estimate of R_{10} and Q_{10} from the annual dataset, allowing comparison among ecosystems.

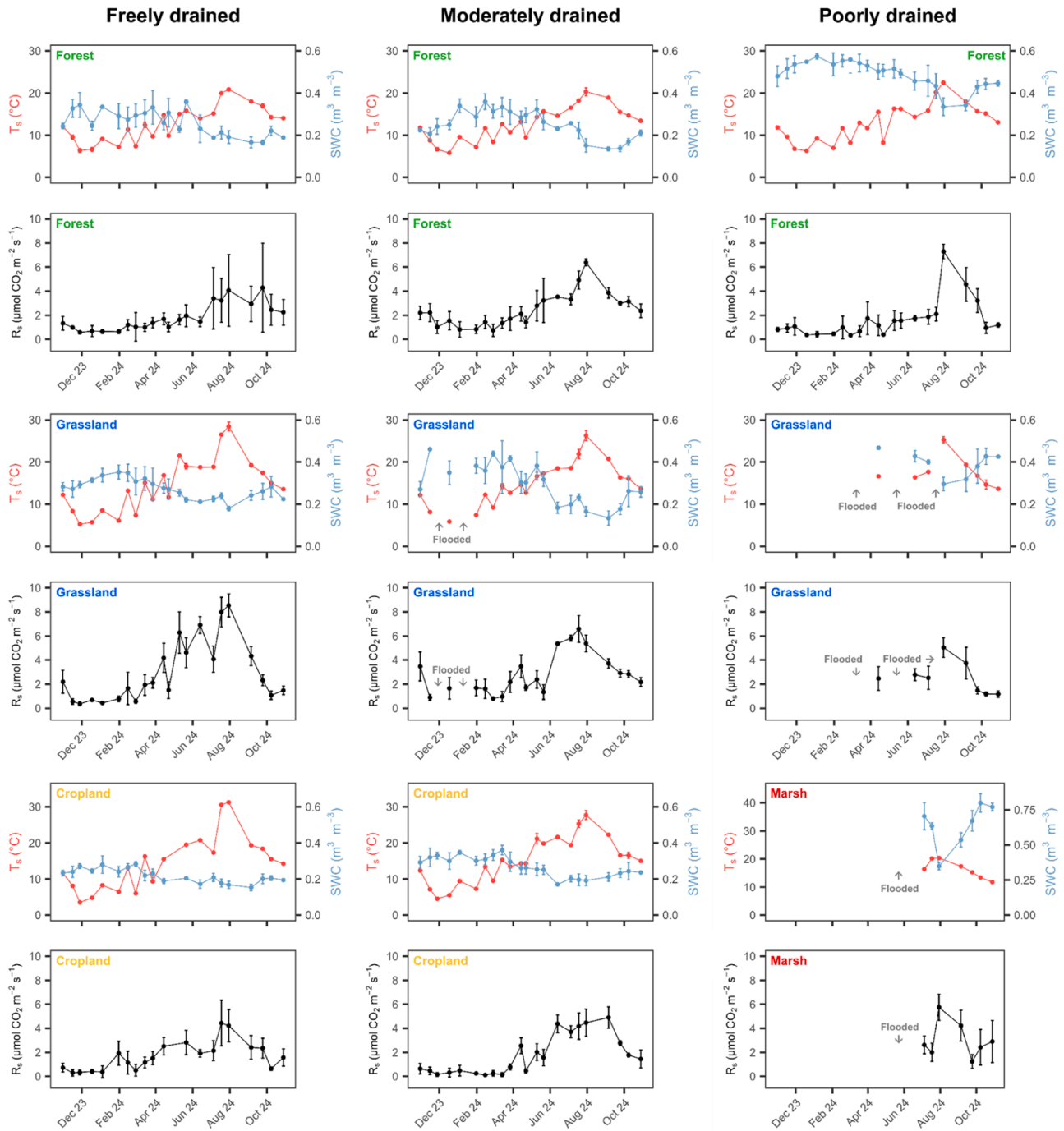


Fig. 3. Mean soil temperature T_s ($^{\circ}\text{C}$), volumetric water content SWC ($\text{m}^3 \text{m}^{-3}$) and soil respiration rate R_s ($\mu\text{mol CO}_2 \text{m}^{-2} \text{s}^{-1}$) measured in forest, grassland, cropland and marsh ecosystems under three hydrological conditions from October 2023 to October 2024. Error bars indicate $\pm$ standard deviation (SD). Flooded periods without measurements are indicated by arrows.

2.4. Soil water content impact

To thoroughly evaluate the impact of soil moisture, we extended Model I by allowing Q_{10} to vary as a function of soil conditions, following Wen et al. (2006) and Fu et al. (2009):

$$Q_{10} = a_1 - a_2 T_s + a_3 \text{SWC} + a_4 \text{SWC}^2 \quad (3)$$

Where a_i ($i = 1, 2, 3, 4$) are site-specific parameters. Substituting Eq. (3) into Eq. (2) yields the multivariate model of R_s , including both T_s and SWC :

$$R_s = R_{10} (a_1 - a_2 T_s + a_3 \text{SWC} + a_4 \text{SWC}^2)^{(T_s - 10)/10} \quad (4)$$

Eq. (4) is hereafter referred to as Model II, whereas Eq. (2) corresponds to Model I. Model II is based on biological processes in which R_s decreases under both dry and wet soil conditions. It is relevant in the floodplain contexts, where fluctuations in groundwater levels may lead to distinct hydrological periods, and is therefore suited to capture long-term (annual-scale) variations in soil conditions. In addition, Model II accounts for the decrease in intrinsic temperature sensitivity of biological processes with increasing temperature (Davidson and Janssens, 2006), therefore allowing Q_{10} to vary continuously as a function of

both T_s and SWC. The model was also fitted to the annual dataset for each site.

To complement the modelling results of Model II, the annual dataset of each site was divided into three equal-width SWC ranges spanning the observed moisture gradient (Wagle and Kakani, 2014; Wen et al., 2006). Model I (Eq. (2)) was fitted independently to each class, resulting in three Q_{10} estimates per site. The number of SWC classes was chosen as a trade-off between maintaining sufficient data points within each class and achieving a meaningful interpretation of the soil moisture effect.

2.5. CH_4 flux measurements

Given that most soils in this study were subject to variable, sometimes high, moisture conditions, CH_4 fluxes were expected. Because CH_4 emissions can substantially affect GHG balances in rewetted soils, we assessed whether including CH_4 would alter the interpretation of rewetting effects on gaseous C emissions. Therefore, we complemented the first CO_2 data collected using the EGM-5 by conducting additional measurements with a CH_4 and CO_2 gas analyzer LI-7810 (LI-COR Environmental, Lincoln, NE, USA) connected to a soil respiration chamber (8200–01S Smart Chamber, 4244.1 cm^3). CH_4 and CO_2 fluxes were simultaneously measured over a 2-minute period at four PVC collars (20 cm diameter, 11 cm height) installed in a uniform pattern (Supplementary Fig. 1), following the methodology described in Section 2.2. Seven days of simultaneous CH_4/CO_2 flux measurements were conducted between August and November 2024, covering a period of increasing SWC (simulating rewetting) and decreasing T_s .

To compare the relative radiative effects of CH_4 and CO_2 emissions, CH_4 fluxes ($g\ CH_4\ m^{-2}\ s^{-1}$) were converted to CO_2 -equivalents (hereafter called CO_2 -equivalent CH_4 flux) using a 100-year global warming potential (GWP) factor of 27, following the Sixth Assessment Report of the Intergovernmental Panel on Climate Change (IPCC) (Forster et al., 2023). For each combination of land use and hydrological condition (9 sites), the contribution of CH_4 to combined GHG emissions (%) was calculated as follows:

$$CH_4\ share = \left(\frac{CH_4}{CH_4 + CO_2} \right) * 100 \quad (5)$$

Where all fluxes are expressed as CO_2 -equivalents.

2.6. Statistical analysis

The following statistical analyses concern the CO_2 fluxes measured with the EGM-5. All analyses were performed for the non-frozen period using nonlinear least squares regression based on the Levenberg-Marquardt algorithm, implemented via the *nlsLM* function in R. Regression parameters (i.e. R_{10} and Q_{10}) were considered statistically different among sites after calculating a t statistic and performing pairwise comparison tests, following the approach of Ngao et al. (2012). However, instead of applying a Bonferroni correction to adjust the significance level (α), we used a Benjamini–Hochberg correction to control the false discovery rate (FDR). Given the 36 pairwise comparisons performed, the Bonferroni approach was considered too conservative.

We computed the corrected Akaike information criterion (AIC_c) to evaluate the performance of the two R_s models (Burnham and Anderson, 2010):

$$AIC_c = -2\log(L(\hat{\theta})) + 2k + \frac{2k(k+1)}{n-k-1} \quad (6)$$

Where $L(\hat{\theta})$ is the log-likelihood of the model, n and k are the number of observations and parameters, respectively. AIC_c is recommended in model selection when $n/k < 40$. We used this estimator to assess whether adding SWC improves the trade-off between model fit and complexity and thus supports the existence of a distinct relationship between R_s and SWC (Hsieh et al., 2016). We also computed the

coefficient of determination (R^2) for Model I only, and the adjusted coefficient of determination (R_{adj}^2) for both Model I and II, to provide descriptive statistics and facilitate comparisons with other studies (Burnham and Anderson, 2010).

3. Results

3.1. Environmental conditions

The weather conditions are presented in Figs. 2a and 2b, while the evolution of the water table depth (m) is shown in Fig. 2c. Air temperature and precipitation data are used here to describe the regional climatic context during the study period and are not intended to represent local floodplain microclimatic conditions. Overall, the study period was wetter than the 1991–2020 climatic baseline, with several months exceeding long-term precipitation by 15–72 mm. Mean air temperatures in October–December 2023 and February–March 2024 were 2–4 °C above the long-term average.

In October 2023, the two water tables in the Dijle floodplain rose closer to the soil surface, just before the first soil respiration measurement (Fig. 2c). From November to May, water depth remained relatively stable at both sites. However, the moderately drained site had more pronounced week-to-week fluctuations than the poorly drained site, which consistently maintained a shallower water table. In contrast, greater fluctuations occurred between May and October at both sites. The groundwater level appeared to be visually correlated with rainfall, showing short-term rises following precipitation events. Although temporary decreases occurred during summer, water table depths never reached the calculated MLW (−0.70 m for poorly drained and −1.22 m for moderately drained, Table 1).

3.2. Temporal dynamics of soil CO_2 flux

Soil CO_2 flux (R_s , $\mu mol\ CO_2\ m^{-2}\ s^{-1}$) is presented for each land use and across the three hydrological classes in Fig. 3. Soil temperature (T_s) and volumetric water content (SWC) followed clear seasonal dynamics. T_s was similar across forest sites, whereas SWC differed. The freely drained and moderately drained forest sites showed comparable soil moisture over time, while the poorly drained forest consistently had higher SWC than the other two forest sites. In terms of CO_2 flux, R_s remained relatively low and stable during the first half of the year, increasing gradually from spring onward. In the moderately and poorly drained forest sites, R_s exhibited pronounced seasonal variability, with a distinct summer peak coinciding with maximum T_s , minimum SWC, and a deeper water table. Overall, these patterns indicate that R_s was primarily driven by soil temperature.

In all nine ecosystems, R_s was positively correlated with T_s (Pearson's $r = 0.43$ to 0.94) and negatively correlated with SWC ($r = -0.66$ to -0.90) (Supplementary Table 2). However, T_s and SWC were strongly correlated with each other ($r = -0.50$ to -0.82). It is therefore essential to disentangle their respective effects on R_s . The following results first address the general influence of T_s on R_s using Model I, and then examine the impact of SWC using Model II.

3.3. Temperature-driven soil respiration (Model I)

CO_2 fluxes ($\mu mol\ CO_2\ m^{-2}\ s^{-1}$) are shown in Supplementary Fig. 2 as a function of soil temperature (T_s). Overall, the CO_2 flux ranged between 0 and 10 $\mu mol\ CO_2\ m^{-2}\ s^{-1}$ (Fig. 3). A clear exponential relationship was observed between soil respiration and soil temperature, with R^2 values ranging from 0.52 (moderately drained grassland) to 0.77 (freely drained grassland). Only the freely drained forest and, particularly, the poorly drained marsh showed weaker fits for Model I, as indicated by lower R^2 values (0.40 and 0.18, respectively).

Basal respiration at 10 °C (R_{10}) in forest and grassland ecosystems

followed the trend poorly drained < freely drained < moderately drained (Table 2). However, differences between hydrological conditions were statistically significant ($p < 0.05$) only for comparisons involving poorly drained soils. In forests, R_{10} increased from $0.65 \pm 0.10 \mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$ (poorly drained) to $1.16 \pm 0.16 \mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$ (freely drained) and $1.51 \pm 0.10 \mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$ (moderately drained). Grasslands showed the same pattern, with R_{10} increasing from 1.13 ± 0.17 , to 1.61 ± 0.14 and $1.88 \pm 0.18 \mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$ across the same hydrological classes. Furthermore, for a given hydrological condition, R_{10} was consistently higher in grasslands than in forests, and the lowest R_{10} values occurred in cropland soils.

The temperature sensitivity of R_s ($Q_{10} \pm \text{SE}$) estimated with Model I was higher in forest ecosystems than in other land uses across all hydrological classes (3.25 ± 0.58 , 3.71 ± 0.33 and 6.34 ± 0.94 for the freely drained, moderately drained and poorly drained conditions, respectively) (Table 2). On the plateau, grassland exhibited a significantly higher Q_{10} (2.63 ± 0.16) than cropland (1.97 ± 0.13), whereas in the moderately drained floodplain the opposite pattern was observed, with cropland showing a slightly higher (but not significant) Q_{10} (2.79 ± 0.28) than grassland (2.14 ± 0.19). Overall, for a given hydrological regime, Q_{10} was consistently higher in forest soils, whereas R_{10} was higher in grassland soils, except under poorly drained conditions where marsh R_{10} was higher but not significantly different.

3.4. Model selection and performance (Model I vs Model II)

To evaluate the distinct effect of soil water content (SWC) on R_s , the residuals of Model I (i.e. the difference between measured and predicted R_s) were first plotted against SWC (Supplementary Fig. 3). No clear relationship was observed. We then used a robust approach by testing the impact of SWC on the temperature sensitivity (Q_{10}) for each site (leading to Model II). Table 2 summarizes the performance of the two R_s models for each land use and hydrological class. In the freely drained sites, only the grassland showed a negative ΔAIC_c , indicating better support for the dynamic Q_{10} model ($\Delta\text{AIC}_c = -26.92$). For five floodplain ecosystems, varying Q_{10} according to soil conditions reduced AIC_c and improved model performance. Only the grassland under poorly drained conditions, which had a smaller SWC range (c. 0.3 to 0.45 $\text{m}^3 \text{ m}^{-3}$) (Fig. 3), behaved differently ($\Delta\text{AIC}_c = 4.42$). For grassland and marsh in this hydrological class ($n = 31$ and 25, respectively), the application of Model II led to substantially larger standard errors.

Fig. 4 shows predicted versus observed CO_2 fluxes for the best model

selected at each site. Freely drained forest and cropland, as well as poorly drained grassland, are represented by Model I, whereas all other ecosystems are represented by Model II. Overall, predicted fluxes were in reasonable agreement with those measured using the EGM-5, with generally higher R^2 values for sites represented by Model II. This was especially the case for freely drained grassland ($R^2 = 0.84$), moderately drained cropland and forest ($R^2 = 0.79$ and 0.76 , respectively), and poorly drained forest ($R^2 = 0.75$). Intermediate values were obtained for moderately and poorly drained grasslands ($R^2 = 0.67$ and 0.64 , respectively). The lowest R^2 values were found for freely drained cropland and forest ($R^2 = 0.59$ and 0.40 , respectively), as well as for the poorly drained marsh ($R^2 = 0.55$). A slight underestimation of high CO_2 fluxes was observed, more pronounced in sites where Model I was selected than in those using Model II (Fig. 4).

3.5. Effect of soil water content on temperature sensitivity

Fig. 5 shows Q_{10} values as a function of SWC. The values predicted by fitting Model II also reflect the effect of T_s through the color gradient. Overall, Q_{10} was primarily controlled by SWC across floodplain sites. A stronger T_s effect was observed for freely drained forest and grassland sites, as well as in the moderately drained cropland site.

Under freely drained conditions, two ecosystems (forest and cropland) showed positive ΔAIC_c values (Table 2), indicating that Model II (dynamic Q_{10}) performed worse than Model I (fixed Q_{10}). This statistical outcome was consistent with the three measured Q_{10} values, which did not show a clear effect of SWC on Q_{10} (Fig. 5). In contrast, the grassland showed stronger support for Model II (negative ΔAIC_c), although the measured Q_{10} values did not perfectly align with the model predictions, suggesting that T_s exerted a stronger control on Q_{10} than SWC in this case.

In the moderately drained floodplain, a similar trend was observed across all land uses. Q_{10} exhibited an optimum within a moderate range of SWC and declined, with variable intensity, at both low and high SWC values (Fig. 5). All three ΔAIC_c values were negative (Table 2), indicating strong support for Model II. The measured Q_{10} values closely matched Model II, again reaching a maximum at intermediate SWC and declining towards both extremes, with a stronger decrease at high SWC.

In the poorly drained floodplain, the forest and marsh showed higher SWC values than the other ecosystems in the valley. In both ecosystems, Q_{10} increased as SWC decreased (Fig. 5). The measured Q_{10} values closely matched those predicted by Model II, which in both cases had a

Table 2

Nonlinear regression results of soil respiration (R_s), using Model I ($R_s = R_{10} Q_{10}^{(T_s - 10)/10}$) and Model II ($R_s = R_{10} (a - bT_s + c\text{SWC} + d\text{SWC}^2)^{(T_s - 10)/10}$). The parameters R_{10} ($\mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$) and Q_{10} are presented with their standard error (SE). ΔAIC_c values correspond to the difference between Model II and Model I. Positive values indicate poorer support for Model II. The best model is highlighted in bold.

Hydrology	Land use	<i>n</i>	Model	R_{10}	Q_{10}	R^2_{adj}	AIC_c	ΔAIC_c
Freely drained	Forest	77	I	1.16 (0.16)	3.25 (0.58)	0.39	247.23	–
			II	1.08 (0.18)	–	0.41	248.09	0.86
	Grassland	88	I	1.61 (0.14)	2.63 (0.16)	0.77	294.96	–
			II	1.03 (0.15)	–	0.83	268.04	- 26.92
	Cropland	79	I	1.07 (0.11)	1.97 (0.13)	0.58	212.56	–
			II	0.95 (0.14)	–	0.59	214.35	1.79
Moderately drained	Forest	84	I	1.51 (0.10)	3.71 (0.33)	0.73	202.51	–
			II	1.53 (0.10)	–	0.75	200.31	- 2.20
	Grassland	72	I	1.88 (0.18)	2.14 (0.19)	0.51	241.50	–
			II	1.48 (0.18)	–	0.65	221.52	- 19.99
	Cropland	85	I	0.87 (0.12)	2.79 (0.28)	0.62	253.66	–
			II	0.60 (0.11)	–	0.78	211.85	- 41.82
Poorly drained	Forest	83	I	0.65 (0.10)	6.34 (0.94)	0.66	241.25	–
			II	0.79 (0.13)	–	0.74	221.86	- 19.38
	Grassland	31	I	1.13 (0.17)	2.74 (0.36)	0.63	84.48	–
			II	0.57 (0.26)	–	0.63	88.90	4.42
	Marsh	25	I	1.60 (0.56)	2.46 (1.01)	0.14	100.42	–
			II	2.65 (1.82)	–	0.46	93.90	- 6.52

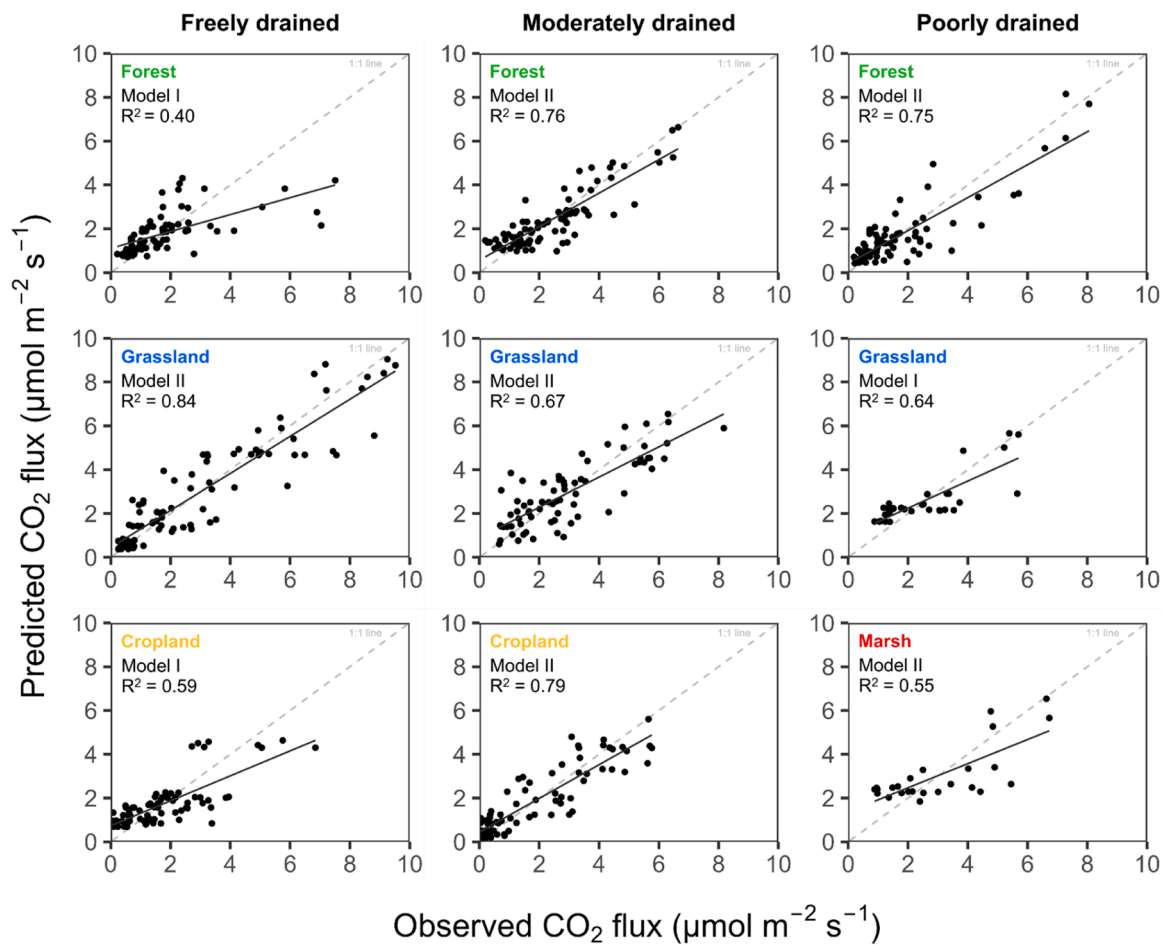


Fig. 4. Predicted versus observed CO₂ flux ($\mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$) across land uses and hydrological conditions. Each panel shows the best-performing model selected based on ΔAIC_c . The solid black line represents the fitted linear regression, while the dashed grey line indicates the 1:1 relationship.

lower AIC_c than Model I (Table 2). In contrast, for the poorly drained grassland, the SWC range was lower (c. $0.15 \text{ m}^3 \text{ m}^{-3}$). No clear trend in the data could be identified. Additionally, the AIC_c of Model II was higher than that of Model I, indicating weaker support for Model II in this case.

3.6. Contribution of CH₄ to combined GHG emissions

Supplementary Fig. 4 shows the CO₂ and CH₄ fluxes measured with the LI-7810, and Fig. 6 illustrates their conversion into CO₂-equivalent fluxes. Seasonal declines in CO₂ fluxes from August to December were consistent across most ecosystems, corresponding to decreasing T_s and increasing SWC. In the freely drained and moderately drained sites, CO₂ dominated the combined GHG flux (expressed as CO₂ equivalents) across all land uses, while CH₄ remained very low and often negative (CH₄ uptake). Conversely, substantial positive CH₄ fluxes were observed only under the poorly drained condition. In grasslands and forests, combined GHG emissions were higher under moderately drained conditions than in freely drained sites, primarily driven by higher CO₂ fluxes. Under poorly drained conditions, combined emissions occasionally increased due to substantial CH₄ contribution.

To further explore the role of soil moisture in regulating the partitioning between CO₂ and CH₄ fluxes, we examined how the relative contribution of CH₄ to combined GHG emissions varied with SWC across sites. Given the non-normal distribution of CH₄ share, no inferential statistical model was applied.

The impact of SWC on the relative contribution of CH₄ to combined GHG emissions (% CO₂-eq.) is illustrated in Fig. 7 for each site. In the

freely drained sites, the CH₄ share remained stable across the observed range of SWC. In forest and grassland sites under moderately drained conditions, CH₄ contribution showed a slight increase with increasing SWC, although values remained close to zero (ranging from -0.1 to 0.1%). In the poorly drained grassland, CH₄ contribution appeared to decrease with increasing SWC, but not in the marsh or in the forest.

4. Discussion

4.1. Meteorological conditions

The main objective of this study was to investigate R_s in two hydrologically contrasting areas of a temperate floodplain, using a plateau with freely drained soils from the same valley as a reference. Meteorological conditions played an important role during the experiment, as the study period was exceptionally wet. In Belgium, 2024 was the wettest year on record (Supplementary Fig. 5), and precipitation amounts in both 2023 and 2024 exceeded the long-term climatic average by more than 100 mm. These high amounts of rainfall were part of a broader climatic anomaly in Western Europe (Copernicus Climate Change Service (C3S) and World Meteorological Organization (WMO), 2025; Schoofs et al., 2025). These extreme hydrometeorological conditions allowed us to study soil respiration under very high soil moisture and natural rewetting rarely seen in temperate Europe, providing insight into how water table changes affect soil flux dynamics. Such wet conditions are relevant in the context of climate change, which is expected to increase precipitation extremes (IPCC, 2023; Schoofs et al., 2025).

However, drought effects were not assessed in this study and may

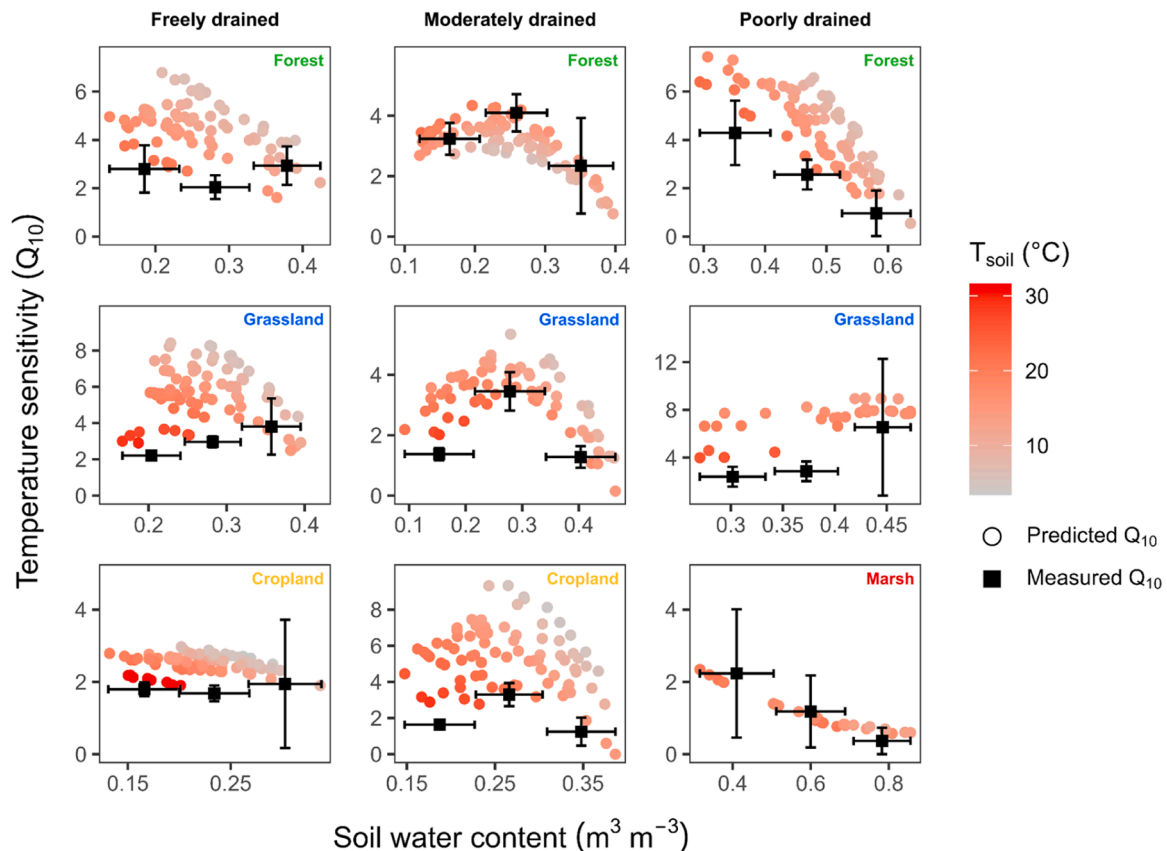


Fig. 5. Effect of soil volumetric water content (SWC, $\text{m}^3 \text{m}^{-3}$) on the temperature sensitivity of soil CO_2 flux (Q_{10}) across land uses and hydrological conditions. Colored circles show Q_{10} values predicted by the Model II, in which Q_{10} varies with T_s and SWC. Black squares represent observed Q_{10} values derived from Model I by dividing each dataset into three SWC classes. Horizontal bars indicate the SWC range around the midrange, and vertical bars denote the standard error (SE) of the fitted coefficient from nonlinear regression.

lead to contrasting responses. In dry extreme years, water limitation would become a dominant control over C fluxes (Patel et al., 2021), while rewetting of dry soils would induce short-lived CO_2 pulses (Védère et al., 2022). Compared to more typical hydrological years (Fig. 1b), the present study reflects conditions with prolonged periods of shallow water tables and reduced seasonal drawdown (Fig. 2c). Under such conditions, R_s dynamics are expected to be strongly influenced by hydrological constraints, particularly the alternation between oxygen-limited and more aerated states. While wetter than average, these conditions remain representative of floodplain systems with fluctuating water tables.

4.2. Basal respiration rate

Soil CO_2 flux was positively correlated with T_s and negatively correlated with SWC (Fig. 3), in line with the seasonal co-variation of temperature and moisture in temperate climates (Jonard et al., 2007; Ngao et al., 2012). Given that R_s was strongly regulated by T_s , we used a modelling approach using the basal respiration rate (R_{10}) and the temperature sensitivity (Q_{10}) (Eq. (2)). Few studies provide a rigorous comparison of multiple land-use types at the same time (Kochiieru et al., 2023). In addition, comparing soil respiration across studies is often difficult because sites can differ in their intrinsic characteristics (Ngao et al., 2012).

In the Dijle valley, R_{10} was consistently higher in grassland soils than in other land uses under the same hydrological condition, in line with previous studies (Buyse et al., 2016; Kochiieru et al., 2023; Raich and Tufekcioglu, 2000). This likely reflects greater belowground C allocation, higher availability of C substrates and nutrients, and a more

abundant fine-root network in grasslands.

Consistent with our hypotheses, the wettest ecosystems had the lowest R_{10} for a given land use (Table 2), in line with the lower R_{10} reported for wetland soils by Giasson et al. (2013). This can be attributed to the prevalence of anoxic conditions that limit OM decomposition and rhizospheric respiration. The marsh nevertheless exhibited noticeable CO_2 fluxes, which may partly reflect seasonal aeration of surface soil layers following inundation. In addition, CO_2 production can persist at high soil moisture through anaerobic metabolisms, but also through aerobic mineralization supported by oxygen diffusion at the soil–air interface or by oxygen entrapped in surface soil layers (Fairbairn et al., 2023). Conversely, grassland and forest under moderately drained conditions had the highest R_{10} , despite similar average SWC to the freely drained sites. It has been shown that microtopography can influence R_s (Miao et al., 2013). In our study, the grassland under moderately drained conditions was located in a slight depression, which may accumulate C substrates and maintain soil moisture at levels favorable to respiration, as also observed in a frequently flooded floodplain site (Schindlbacher et al., 2022). The succession of rewetting and drying phases is also known to break soil aggregates and make SOC more available to microorganisms (Védère et al., 2022), although the role of physical protection by aggregates was not investigated in this study.

4.3. Temperature sensitivity

In the freely drained sites, Q_{10} derived from Model I (Eq. (2)) followed the order cropland < grassland < forest (Table 2), which is consistent with the findings of Zheng et al. (2009) and Meyer et al. (2018). In cropland, tillage redistributes SOM and modifies porosity,

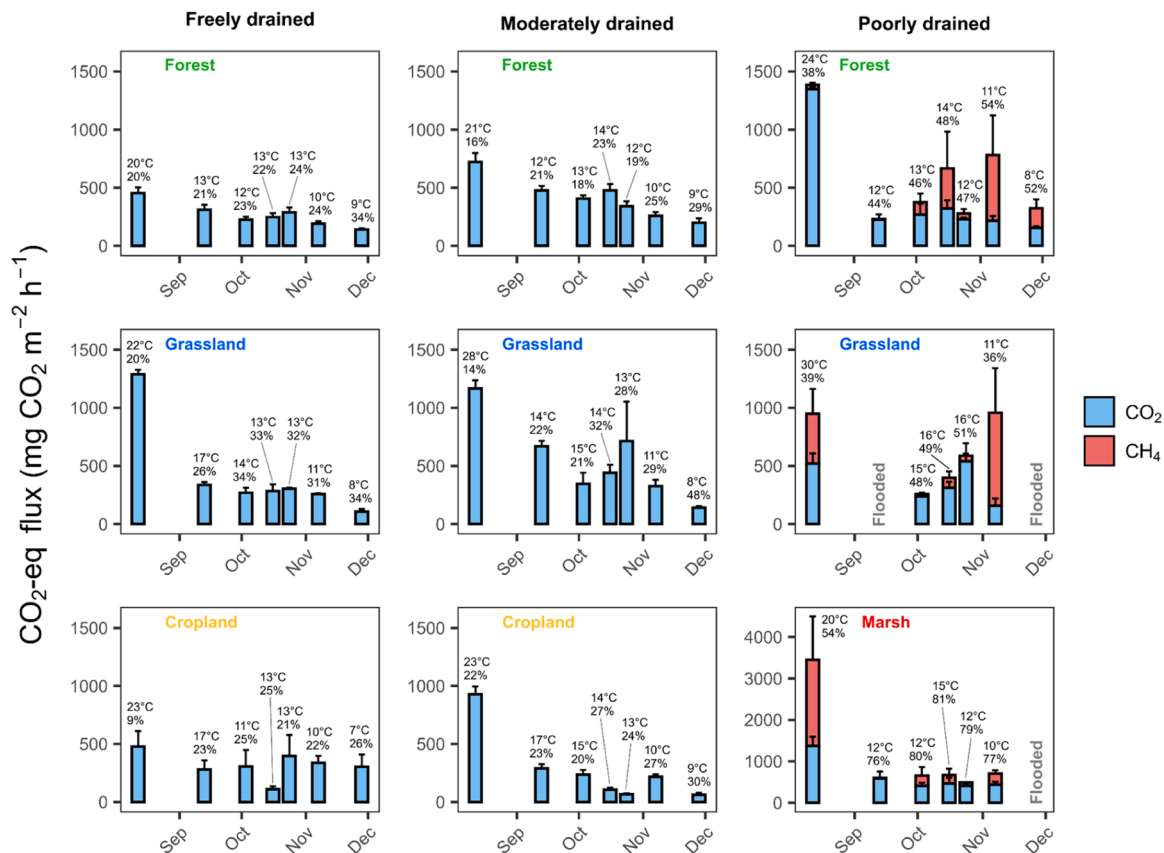


Fig. 6. CO₂-equivalent GHG fluxes (CO₂, CH₄ × 27) measured across different land uses and hydrological conditions using the LI-7810. Data are presented as mean + SE. Average soil temperature (°C) and volumetric water content (%) are indicated above the bars. The mean and SE of the CO₂-equivalent CH₄ flux in the freely drained and moderately drained sites are too small to be visible. Note that the y-axis for the marsh is shown on a different scale.

weakening the apparent temperature dependence of R_s (Zheng et al., 2009). However, apparent Q_{10} values obtained with long-term field measurements in the floodplain should be interpreted with caution, as they integrate not only temperature effects but also concurrent changes in substrate and oxygen availability. Rhizospheric respiration is generally more temperature-sensitive than heterotrophic respiration at the seasonal time scale (Gaumont-Guay et al., 2008; Li et al., 2020). In temperate deciduous forests, the strong increase in root and overall biological activity during the growing season increases annual Q_{10} estimates (Curiel Yuste et al., 2004; Reichstein et al., 2005a; Zheng et al., 2009). In addition, Keiluweit et al. (2017) showed that a transition from anaerobic to aerobic soil conditions can markedly enhance OM mineralization, in some cases by up to 10 times. When soil warming coincides with increased substrate availability or improved soil aeration, as observed during the first water table drawdown below 10 cm (Figs. 2c and 3), the R_s - T_s relationship can be further amplified, leading to very high apparent Q_{10} values (Davidson et al., 2006). This likely explains the $Q_{10} > 6$ observed in the forest under poorly drained conditions (Table 2), which cannot be attributed to temperature sensitivity of biological processes alone.

To disentangle the effects of soil temperature and moisture, a modified Q_{10} model was applied (Wen et al., 2006), considering Q_{10} as a seasonal variable that changes with T_s and SWC (Xu and Qi, 2001). When soil moisture is low, CO₂ emissions are constrained by limited diffusion of organic C substrate and enzymes, as well as by physiological stress on microbial cells (Fairbairn et al., 2023; Jassal et al., 2008; Skopp et al., 1990). Conversely, when soil moisture is high, aerobic respiration is restricted by the limited diffusion of oxygen (O₂). In this context, Model II does not reflect the intrinsic temperature sensitivity but rather integrates the confounding effects of soil moisture on substrate and

oxygen availability.

At sites under moderately drained conditions, we found an optimal Q_{10} between 0.25 and 0.3 m³ m⁻³, consistent with previous studies reporting maximum R_s and Q_{10} near 30% soil moisture (Tan et al., 2021; Wagle and Kakani, 2014). Given the predominantly silt loam soils in the study area, the observed optimal SWC was slightly below typical field capacity values. Q_{10} then decreased under both drier and wetter conditions, suggesting that rewetting previously drained floodplain soils with fluctuating water table could reduce the temperature sensitivity of R_s and potentially limit CO₂ emissions. Conversely, in forest and marsh ecosystems under poorly drained conditions, Q_{10} increased with decreasing SWC (Fig. 5), which is consistent with a potential increase in the temperature sensitivity of R_s under drainage scenarios, and thus higher CO₂ emissions. This pattern was not observed in grassland under poorly drained conditions, likely due to a narrow range of SWC and summer soil compaction by agricultural machinery, which maintained the area flooded for part of the summer. Nonetheless, applying Model I (Eq. (2)) to three ranges of SWC revealed higher R_{10} values under lower soil moisture, corresponding to drainage conditions (Supplementary Table 3).

The soil moisture- Q_{10} relationships differed between the hydrological conditions (Fig. 5). This underscores the influence of water regime, as hypothesized. In this study, soil moisture exerted a strong control on Q_{10} by regulating oxygen availability and thus the balance between aerobic and anaerobic processes. This could explain the contrasting soil moisture- Q_{10} relationships observed in the two floodplain zones. In addition, microbial communities are adapted to the historical moisture conditions of their site and may therefore exhibit site-specific responses to drying or rewetting events, further contributing to these contrasting patterns (Patel et al., 2021; Zhou et al., 2014). In the literature,

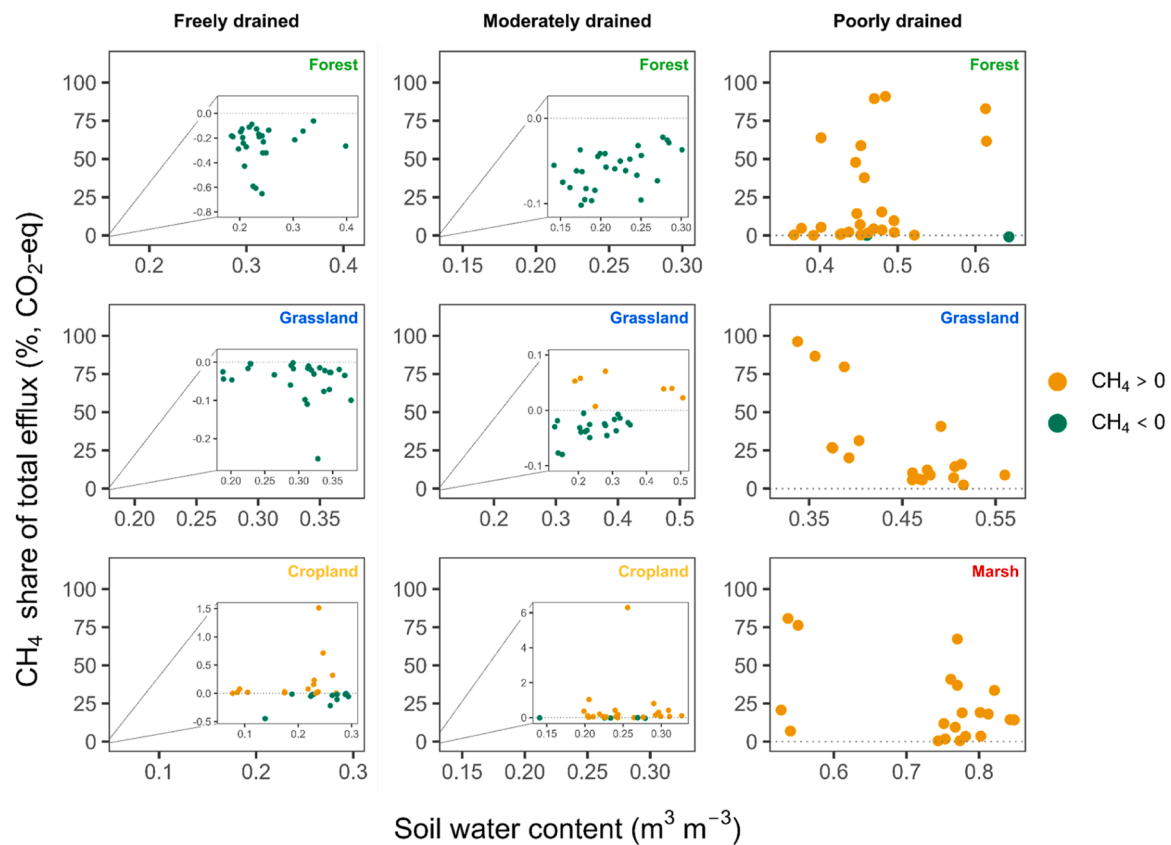


Fig. 7. Effect of soil water content (SWC, $\text{m}^3 \text{m}^{-3}$) on methane (CH_4) contribution to combined GHG flux (% $\text{CO}_2\text{-eq}$) across land uses and hydrological conditions. The CH_4 share is expressed as the ratio of CH_4 to the combined CO_2 and CH_4 fluxes (CO_2 -equivalent basis). Orange points indicate positive ratios (CH_4 flux > 0), and green points indicate negative ratios (CH_4 flux < 0).

conflicting relationships have been observed between SWC and Q_{10} , probably because of confounding factors in field conditions, or because of varying experimental methodologies (e.g. soil type, range of soil moisture or absence of roots) (Reichstein et al., 2005b; Zhou et al., 2014). Our results are consistent with the findings of Craine and Gelderman (2011), who showed that topographic position can influence the soil moisture– Q_{10} relationship during soil incubation.

4.4. Soil CH_4 fluxes

The marsh showed the largest CH_4 emissions, with a mean flux of $340 \text{ nmol CH}_4 \text{ m}^{-2} \text{ s}^{-1}$, which falls within the range reported by Cui et al. (2024) for similar water table conditions. To assess the relative contribution of CH_4 to GHG emissions, we calculated the GWP-weighted CH_4 : ($\text{CH}_4 + \text{CO}_2$) ratio (Zheng et al., 2018). Changes in this ratio reflect shifts in C decomposition pathways across ecosystems (Jiang et al., 2025).

Under poorly drained conditions, all three ecosystems emitted CH_4 (Fig. 6). Shallow water tables create anaerobic conditions and promote methanogenesis (Zhao et al., 2020). In line with our hypotheses, the contribution of CH_4 to combined GHG emissions was much higher under poorly drained conditions than under the other hydrological regimes, with CH_4 : ($\text{CH}_4 + \text{CO}_2$) ratios occasionally reaching up to 80% (Fig. 7). Although moderately drained sites also experienced shallow groundwater levels (Fig. 2c), SWC was consistently higher under poorly drained conditions, likely limiting oxygen diffusion and favoring CH_4 production. Due to microtopography, the water table did not reach the soil surface within the measurement collars at the moderately drained sites. In addition, high precipitation intensity during the study period may have further stimulated CH_4 emissions (Radu and Duval, 2018; Xu et al., 2024).

Schindlbacher et al. (2022) reported much lower CH_4 contributions

in temperate forest floodplains, with CO_2 accounting for more than 97% of emissions. However, their wettest sites were more comparable to our moderately drained conditions than to our poorly drained sites. Based on the annual CO_2 -equivalent values from Cao et al. (2024), we calculated that their CH_4 : ($\text{CH}_4 + \text{CO}_2$) ratio of a marsh was 79%, which falls within the range of our values. Consistent with this, Li et al. (2024) showed that CH_4 can dominate C gas emissions in persistently flooded wetland systems, highlighting the importance of the degree and duration of soil saturation. Under poorly drained conditions, CH_4 contribution showed no consistent relationship with SWC (Fig. 7), in agreement with Cui et al. (2024). Taken together, our results suggest that high soil moisture conditions are associated with lower CO_2 emissions and reduced temperature sensitivity, without evidence of a proportional increase in CH_4 emissions.

Under moderately drained conditions, the forest acted as a CH_4 sink, whereas grassland and cropland alternated between CH_4 sources and sinks. In these ecosystems, the CH_4 : ($\text{CH}_4 + \text{CO}_2$) ratio increased with SWC (Fig. 7), reflecting a shift from aerobic toward more anaerobic conditions and reduced CH_4 oxidation as groundwater level rose (Fig. 2c) (Cao et al., 2024). Nevertheless, CH_4 contributions remained very low ($< 1\%$), indicating that CO_2 dominated the overall C gas balance. In the freely drained sites, soil CH_4 fluxes from both grassland and forest were consistently negative (Supplementary Fig. 4), in line with strong methane oxidation under well-aerated conditions (Rafalska et al., 2023; Roslev and King, 1996). Occasional CH_4 emissions from cropland were likely linked to manure application and soil compaction by agricultural machinery, which may have created partially anaerobic microsites following heavy rainfall (Fig. 2b) (Laubach et al., 2024; Tiemeyer et al., 2020).

4.5. Methodological considerations

This study focuses on gaseous C losses and does not quantify biomass production or long-term SOC accumulation. Conclusions regarding net ecosystem C balance cannot be drawn from soil gas fluxes alone as it also requires the assessment of C inputs. Although wetlands are often considered long-term C sinks (Mitsch et al., 2013), this involves sustained conditions where C inputs exceed gaseous losses over decadal timescales, which is particularly true in peatlands.

Each combination of land use and hydrological condition was represented by a single site (Fig. 1), meaning that treatments were not replicated across the landscape. Four PVC collars per site captured local spatial variability in GHG fluxes but did not constitute independent replicates. Consequently, the observed differences among land uses and hydrological regimes reflect site-level responses within the Dijle valley, rather than fully replicated treatment effects. Previous work in a northern Belgian floodplain has shown that within-site variability in soil CO₂ fluxes can be substantial and may contribute significantly to overall uncertainty (Tersago et al., 2025). In addition, collar installation can introduce within-site variability by disrupting the physical continuity between roots and soil and by modifying local soil moisture conditions, particularly in poorly drained floodplain soils (Mills et al., 2011). Together, these factors indicate that part of the observed variability likely reflects both microsite heterogeneity and measurement-related effects.

Despite these limitations, the dataset allows for the interpretation of functional relationships at the hydrological class level. For a given hydrological condition, broadly consistent SWC–Q₁₀ relationships were observed across land uses (Fig. 5), further suggesting a coherent functional response.

4.6. Perspectives

In our study, model predictions were in good agreement with measured CO₂ fluxes (Fig. 4). High fluxes were underestimated, particularly when Model I was selected in freely drained sites. Such deviations can also be observed in previous studies (Canteral et al., 2023; Qiu et al., 2018), for example when fluxes are measured with soil chambers that reflect local variability (Delogu et al., 2017). Our findings have direct implications for land surface models such as the Community Land Model version 5 (CLM5) and ORCHIDEE (Organising Carbon and Hydrology In Dynamic Ecosystems), which rely on fixed Q₁₀ values of 1.5 and 2, respectively. In CLM5, soil C decomposition processes remain unchanged from previous model versions, while the model still requires adjustments in soil moisture limitations on C turnover (Lawrence et al., 2019). In floodplain environments, where rapid shifts between aerobic and anaerobic conditions occur due to fluctuating water tables, incorporating a dynamic Q₁₀ approach could improve model performance by better representing moisture-driven transitions in decomposition processes.

From a management perspective, our results suggest that seasonally drained floodplains with periodically dry topsoil (i.e. our moderately drained sites) may emit more GHG than their freely drained counterparts. These floodplains sites were also associated with lower CH₄ contributions, whereas floodplains with persistently wet topsoil (i.e. our poorly drained sites) showed substantially higher CH₄ contributions to combined GHG. Maintaining water tables near, but below, the soil surface may reduce CO₂ emissions and Q₁₀ while avoiding large CH₄ releases. This pattern is consistent with Zhao et al. (2026) who identified a water table depth of 9 cm as an effective trade-off between CO₂ and CH₄ emissions. Increasing the frequency of CH₄ measurements and integrating biomass and SOC accumulation data would be necessary to fully assess the climate mitigation potential of floodplain restoration strategies.

5. Conclusion

In a temperate Belgian valley, we investigated soil C fluxes across dominant land uses (forest, grassland, cropland and marsh) and three contrasted hydrological conditions (reference plateau with freely drained soils, moderately drained and poorly drained floodplain ecosystems) during the wettest year on record. On an annual basis, basal soil respiration at 10 °C varied with both land use and hydrological regime, with the lowest value mainly observed under poorly drained conditions. Soil CO₂ respiration in forests was more sensitive to temperature than other land uses across all hydrological regimes. Using a dynamic temperature sensitivity (Q₁₀) improved the performance of soil respiration prediction models in floodplain sites. The soil moisture–Q₁₀ relationship differed between hydrological regimes, suggesting that rewetting historically drained floodplains would reduce their temperature sensitivity, likely decreasing CO₂ emissions, whereas draining wet floodplains would enhance it. CH₄ emissions were only substantial in very wet floodplain sites with near-surface water tables. Maintaining water tables close, but below, the soil surface through rewetting may represent an effective management strategy to mitigate CO₂ emissions while avoiding large CH₄ emissions. This study provides modelling insights to improve predictions of soil CO₂ fluxes in floodplain ecosystems with fluctuating water tables.

Declaration of generative AI and AI-assisted technologies in the writing process

During the preparation of this work the author used ChatGPT 5 only for English grammar correction. After using this tool, the author reviewed and edited the content as needed and takes full responsibility for the content of the published article.

Data availability statement

Data will be made available on request.

CRediT authorship contribution statement

Nicolas Kovacs: Conceptualization, Data curation, Investigation, Writing – original draft, Writing – review & editing. **Gilles Colinet:** Writing – review & editing. **Bernard Longdoz:** Investigation. **Jens Leifeld:** Writing – review & editing. **Suzanna Lettens:** Resources, Writing – review & editing. **Maud Raman:** Resources, Writing – review & editing. **Karen Vancampenhout:** Resources, Writing – review & editing. **Gert Verstraeten:** Funding acquisition, Writing – review & editing. **Jeroen Meersmans:** Conceptualization, Funding acquisition, Investigation, Resources, Writing – review & editing.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Acknowledgements

This research was funded by the Belgian Science Policy Office (BELSPO) in the framework of the Belgian Research Action through Interdisciplinary Networks (BRAIN-BE 2.0) programme (RECARBON project, contract number: B2/223/P1/RECARBON). We sincerely thank the Institute of Nature and Forest Research (INBO), the Agency for Nature and Forests (ANB), and the farmers involved for their kindness and for granting access to their land. The lead author would also like to thank Raquel Zapatero Cuadrado for her advice on figure design and color selection. Finally, we thank the anonymous reviewers for their valuable comments that helped improve the manuscript.

Supplementary materials

Supplementary material associated with this article can be found, in the online version, at [doi:10.1016/j.agrformet.2026.111321](https://doi.org/10.1016/j.agrformet.2026.111321).

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