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Permanent land use and microbial contributions shape soil organic carbon stabilization in lake floodplains

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ABSTRACT

Floodplain ecosystems play a key role in soil organic carbon (SOC) storage, as they integrate inputs from both vegetation and deposited sediments. Promoting land uses with low anthropogenic disturbances helps maintain the function of these ecosystems as soil carbon (C) sinks. However, the tipping points along a disturbance gradient where land use transitions generate the largest SOC losses or gains remain unclear, though they are key for effective land use management and climate change mitigation. Because flood events can mobilize and enhance the loss of labile C, determining the main origin of stable SOC, whether from plants or soil microorganisms, is also important to identify the optimal combination of land use, vegetation, and soil type for SOC stabilization in floodplains. We examined how SOC stabilization and origin vary in the floodplain of Lake Saint-Pierre, Québec, Canada along an anthropogenic disturbance gradient of six land uses: conventional and improved croplands, temporary and permanent meadows, marshes, and forested swamps. In all 6 land uses at the same elevation, we quantified SOC stocks, mineral-associated organic matter - C (MAOM-C), soil $\delta^{13}\text{C}$ and sugar biomarkers. Driven by land use change, variations in plant inputs were associated with changes in topsoil SOC storage. Forested swamps had the highest MAOM-C due to greater plant biomass inputs that increased both microbial- and plant-derived C. Across all land uses, microbial-derived sugars, particularly of fungal origin, were more abundant in MAOM than plant-derived sugars. Notably, MAOM remained below its saturation capacity. Soil C pools, including MAOM-C, and both microbial- and plant-derived C, increased nonlinearly with decreasing disturbance, with a tipping point occurring at the transition from temporary to permanent meadows. Our results highlight the importance of increasing both microbial- and plant-derived C inputs to promote SOC stabilization, while emphasizing the persistent key role of fungal metabolism in floodplain soils.

1. Introduction

Soils represent the second-largest carbon (C) sink after the oceans, storing approximately 1500 Pg of C in the top meter of soil (Schmidt et al., 2011; Scharlemann et al., 2014). Increasing soil organic carbon (SOC) stocks not only contributes to climate change mitigation but also improves soil fertility and structure (Lal et al., 2015). Floodplain ecosystems, in particular, are hotspots for SOC storage due to their unique position at the interface between terrestrial and aquatic environments

and their dynamic hydrological regimes (Mitra et al., 2005; Bridgman et al., 2006; Mitsch et al., 2013). Despite occupying only a small fraction (6.6 %) of global land surface, floodplain wetland soils play a disproportionately large role in global C cycling by integrating both autochthonous organic inputs from vegetation and allochthonous materials deposited by sediment-rich floodwaters (Omengo et al., 2016; Scott and Wohl, 2018; Zheng et al., 2024). These attributes render floodplains critical yet understudied components of the terrestrial C sink.

Although floodplains naturally emit greenhouse gases, with methane

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emissions ranging from 0.7 to 263 g C m⁻² year⁻¹ compared to 0 to 203 g C m⁻² year⁻¹ for peatlands (Mitsch et al., 2013; Sjögersten et al., 2014), their capacity to sequester C generally surpasses these emissions, with net annual sequestration rates between 3 and 340 g C m⁻² year⁻¹ (Mitsch and Mander, 2018; Hupp et al., 2019). From a management perspective, maintaining or enhancing this C sink as well as transforming it into SOC stocks depends on selecting appropriate land uses (Ji et al., 2020). While intensive agriculture, such as annual cropping systems, can be particularly productive in floodplains (Kwesiga et al., 2019), repeated soil disturbance and harvest biomass exportation lead to reduced SOC stocks (Bronick and Lal, 2005; Ciais et al., 2010). In contrast, the conversion of croplands to forested floodplains has enhanced SOC due to higher litter inputs (Nyssen et al., 2008; Omengo et al., 2016; D'Elia et al., 2017) and increased root production and turnover in deep soil layers (Ferré et al., 2014; Sariyildiz et al., 2022). However, the effect of forested floodplains on SOC stocks compared to other vegetation types such as floodplain grasslands remains unclear, with some studies reporting higher SOC stock under forests (Smith and Reid, 2013; Hernandez et al., 2015), while others found higher SOC in grasslands (Del Galdo et al., 2003; Sariyildiz et al., 2022) or reported no differences (Cierjacks et al., 2010; Scott and Wohl, 2018). Research should thus consider SOC as a heterogeneous pool composed of fractions with different properties for a more nuanced understanding of land use effects along a gradient of anthropogenic disturbance in floodplain ecosystems.

Importantly, SOC responses to changes in land use are unlikely to be linear along the anthropogenic disturbance gradient. Tipping points may occur, where shifts in vegetation or management type can result in disproportionately large gains or losses in SOC storage (Lininger, 2020; Yeasmin et al., 2020). For example, a transition from croplands to meadows may represent a stage where SOC gains are most pronounced, due to the cessation of tillage and crop residue removal (Ciais et al., 2010). Shifts in vegetation type and input quality, from herbaceous-dominated meadows to forests dominated by trees, could also lead to substantial changes in SOC storage (Filley et al., 2008; Córdova et al., 2018). Identifying the critical SOC gain along the disturbance gradient is essential to prioritize land use transitions that maximize SOC persistence in floodplains and support climate change mitigation.

Floodplains, by definition, are characterized by episodic flooding that can mobilize labile (i.e., easily decomposed) forms of organic C (Saint-Laurent et al., 2017). As a result, the long-term stabilization of SOC is likely to depend on mineral-associated organic matter (MAOM), which forms through the adsorption of microbial by-products or plant-derived compounds onto mineral surfaces (Balesdent, 1996; Six et al., 2002; Cotrufo et al., 2013; Angst et al., 2021). MAOM is typically considered to be more persistent than particulate organic matter (POM), which is derived from partially decomposed plant materials (Cotrufo et al., 2013). MAOM accumulation is promoted in fine-textured soils, abundant in floodplain environments (Ricker and Lockaby, 2015; Heger et al., 2024). Nevertheless, its accumulation is not infinite and recent work by Georgiou et al. (2025) underlines the importance of considering MAOM saturation, i.e. its maximum observed capacity to stabilize C based on the physicochemical properties of the soil mineral matrix. Determining the degree of MAOM saturation in floodplains could therefore help to better recognize the potential of these ecosystems for storing more stable SOC.

Because plant- and microbe-derived compounds differ in their binding strength to minerals and vulnerability to decomposition, determining their relative proportion in MAOM composition is key to understanding and managing SOC persistence in floodplains (Kleber et al., 2021; Zhao et al., 2025). Yet, the main origin and composition of MAOM remain debated (Blankinship et al., 2018). Stable $\delta^{13}\text{C}$ isotopes, combined with C/N ratios, can distinguish more recently plant-derived C from older, more microbially processed C (Balesdent and Mariotti, 1996; Tonucci et al., 2017; Lamarque et al., 2023), while sugar

biomarkers help identify plant-vs microbe-derived compounds (Angst et al., 2021). Some studies have suggested predominant microbial residues in the MAOM (Cotrufo et al., 2013; Liang et al., 2017), whereas others have observed substantial plant-derived inputs (Angst et al., 2017; Córdova et al., 2018). Grasslands may favor microbe-derived sugars in MAOM compared to forests (Turrión et al., 2002) and croplands (Ding et al., 2017) likely due to continuous inputs of easily decomposed organic materials (Cotrufo et al., 2013; Angst et al., 2021). However, plant-derived neutral sugars in MAOM can be higher in forests than in croplands, but lower than in grasslands (Jolivet et al., 2006; Glaser et al., 2000). Understanding these patterns could facilitate the choice of land uses and vegetation types that promote long-term SOC stabilization in floodplain soils (Angst et al., 2021).

In this study, we investigated how SOC storage and stabilization vary across a large temperate floodplain under different land uses, along a disturbance gradient. Specifically, we aimed to trace the origin of MAOM in floodplain soils and to identify which land use transitions most effectively increase SOC and MAOM. We hypothesized that:

- i) Greater contributions of recently plant-derived C, as indicated by $\delta^{13}\text{C}$ signatures, would be associated with greater SOC storage across land uses in both topsoil and subsoil.
- ii) In meadows, microbial-derived C would be more abundant in MAOM than plant-derived C due to higher easily decomposed inputs, promoting *in vivo* microbial assimilation and thus higher microbial by-products (Angst et al., 2021), whereas in forests, plant-derived C would be more abundant in MAOM due to higher inputs of coarse and recalcitrant plant material, possibly promoting *ex vivo* microbial transformation and transfer of plant-derived compounds onto soil minerals (Liang et al., 2017; Whalen et al., 2022).
- iii) Non-linear responses of all SOC pools (SOC, POM-C, MAOM-C, plant-derived C and microbial-derived C) would be expected along the disturbance gradient, with tipping points occurring at the transition from annual to perennial systems due to tillage cessation, and from herbaceous systems to forests due to changes in input quality.

2. Materials and methods

2.1. Site description

The study was conducted in the floodplain of Lake Saint-Pierre (LSP), Québec, Canada (46.12°N, 72.49°W, Fig. 1). The total area of the flood recurrence zone is approximately 28,000 ha, indicating that the LSP floodplain is the largest wetland in the 1197-km-long Saint Lawrence River system, Québec, Canada (Hudon et al., 2018; Watson et al., 2024). The LSP is a widening of the Saint Lawrence River formed by an immense rocky cavity partially filled by Quaternary marine clay sediments from the Champlain Sea, which gradually retreated around 12,000 years ago. The LSP is recognized as a RAMSAR site (1998) and a UNESCO Biosphere Reserve (2000). This region is categorized by a humid continental climate with an average daily temperature of 5.6 °C and an average annual precipitation of 948 mm (716 mm rain and 213 cm snow) based on 39-year data (1981–2020, Nicolet station, Environment Canada, 2025). The soil type of the floodplain is essentially composed of Gleysols, typical of flood-affected areas, with a poorly developed profile on sandy or clayey sedimentary material, covered by an alluvial loam with a high organic matter (soil profile analysis in Fig. S1). This floodplain is driven by an annual spring flooding event following snowmelt, which lasts for 5–9 weeks and inundate croplands and surrounding natural ecosystems. Therefore, we refer to the floodplain as the relatively flat topographic area adjacent to Lake Saint-Pierre that experience flooding at biennial intervals (Watson et al., 2024).

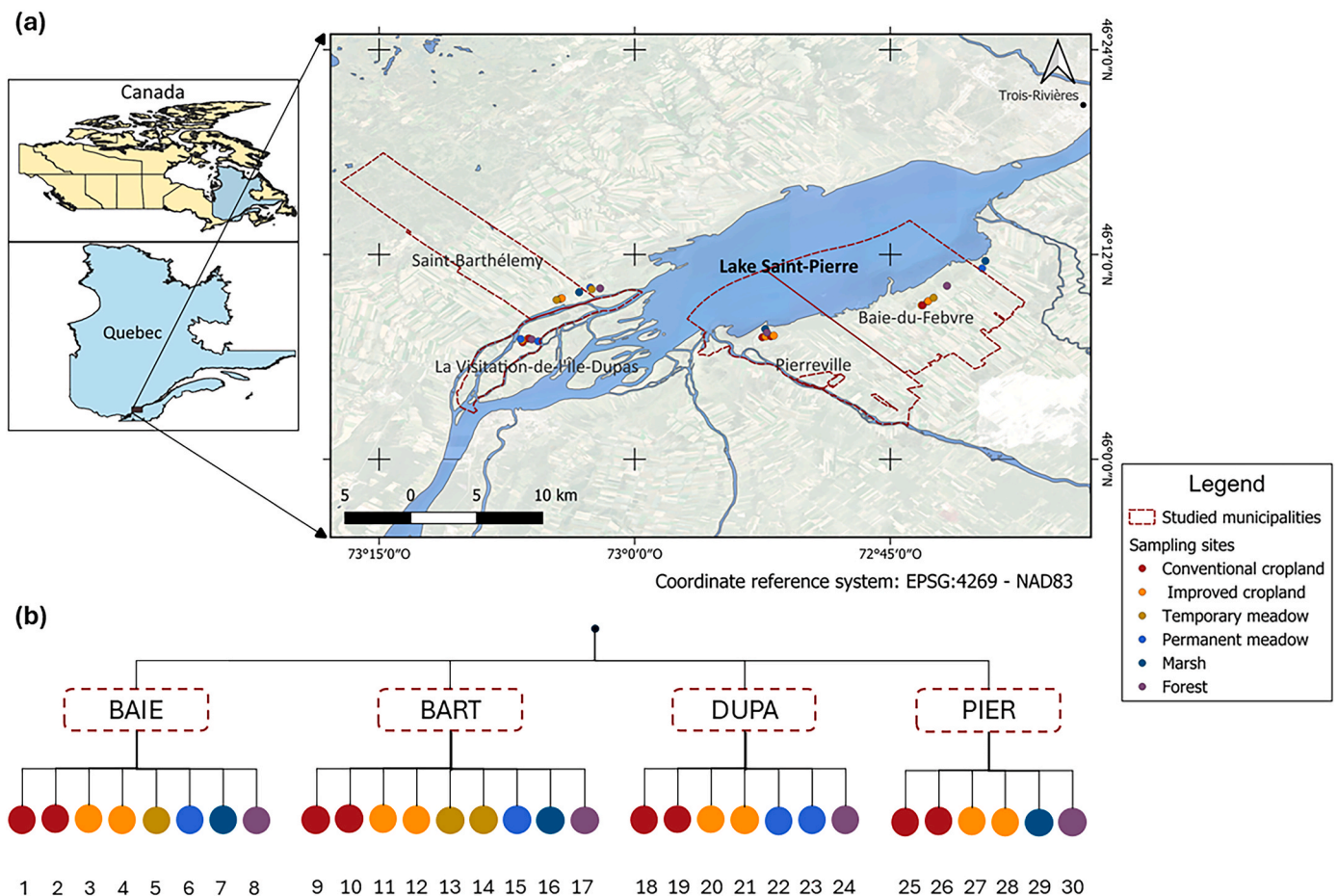


Fig. 1. Location of the study sites in the province of Quebec, eastern Canada (a), and schematic representation of the experimental design (b). The blocks (municipalities) were: BAIE: Baie-du-Febvre; BART: Saint-Barthélemy; DUPA: La-Visitation-de-l'Île Dupas and PIER: Pierreville. These numbers correspond to the site numbers, for a total of 30.

2.2. Experimental design

We established four study blocks corresponding to four areas of the littoral zone (floodplain) to represent regional differences (municipalities of Baie-du-Febvre [BAIE], Saint-Barthélemy [BART], La-Visitation-de-l'Île Dupas [DUPA] and Pierreville [PIER]) (Fig. 1). In each block, different land uses were present, ranging from the most to the least disturbed: 1) conventional croplands (maize [*Zea mays* L.]-soybean [*Glycine max* (L.) Merr.] alternation), 2) improved croplands (maize-soybean alternation with three-year-old intercropping and canary grass riparian strip), 3) temporary meadows (cut-grazed, established less than 5 years ago), 4) permanent meadows (cut-grazed, established more than 5 years ago), 5) marshes and 6) forested swamps (dominated by silver maple [*Acer saccharinum* L.] and black ash [*Fraxinus nigra* M.]) (Table 1, Fig. S2). Note that all land uses were not always present within the

landscape of each block, and thus we sampled a total of 30 sites, distributed as follows: eight conventional croplands (two in each block), eight improved croplands (two in each block), three temporary meadows (one in BAIE and two in BART), four permanent meadows (one in BAIE and BART and two in DUPA), three marshes (one in BAIE, BART and PIER), and four forested swamps (one in each block) (Fig. 1, Campeau et al., 2024). All 30 sites were sampled at the same elevation of 6.2 m a.s.l (above sea level) to ensure similar flooding frequency and sediment deposition. In addition, all sites were located within the same floodplain, ensuring comparable hydroperiods.

2.3. Measurement of vegetation biomass and Normalized Difference Vegetation Index (NDVI)

We measured aboveground and belowground vegetation biomass

Table 1
Means ($\pm$ SEM) of soil characteristics at the 0-10 cm depth across all sites for each land use.

Soil Variable	Conv_Crop	Imp_Crop	Tem_Meadow	Per_Meadow	Marsh	Forest
OM (mg g ⁻¹)	58.3 $\pm$ 9.2	51 $\pm$ 5.6	72.3 $\pm$ 10.3	95.2 $\pm$ 19.4	120.4 $\pm$ 3.5	149.3 $\pm$ 28.8
pH	6.2 $\pm$ 0.2	6.3 $\pm$ 0.3	6.4 $\pm$ 0.6	5.8 $\pm$ 0.3	5.8 $\pm$ 0.5	5.6 $\pm$ 0.3
Eh (mV)	258.2 $\pm$ 13.6	244.3 $\pm$ 13.4	260.1 $\pm$ 30	272.4 $\pm$ 24.4	277.4 $\pm$ 24.2	281.3 $\pm$ 19.1
Clay (%)	21.4 $\pm$ 3.9	19.9 $\pm$ 3.4	28.8 $\pm$ 3.1	27 $\pm$ 3.5	25.9 $\pm$ 7.6	24.7 $\pm$ 3.9
Silt (%)	55.4 $\pm$ 4.2	52.2 $\pm$ 4.4	61 $\pm$ 3.3	59.3 $\pm$ 2.4	68.1 $\pm$ 4.8	69.9 $\pm$ 1.7
Fe oxides (mg g ⁻¹)	1.1 $\pm$ 0.2	1.2 $\pm$ 0.2	1.5 $\pm$ 0.3	1.6 $\pm$ 0.2	1.1 $\pm$ 0.3	1.7 $\pm$ 0.2
Al oxides (mg g ⁻¹)	0.5 $\pm$ 0.1	0.4 $\pm$ 0.1	0.9 $\pm$ 0.4	0.8 $\pm$ 0.1	0.8 $\pm$ 0.2	0.8 $\pm$ 0.2

OM: organic matter concentration; pH: potential of hydrogen in soil; Eh: soil redox potential; Fe oxides: crystalline iron oxides; Al oxides: crystalline aluminum oxides. Conv: conventional; Imp: improved; Per: permanent; Tem: temporary.

from August 16 to 19, 2021, during the peak of the vegetation period. Only roots with a diameter of 3 cm or less were sampled to ease extraction and minimize disturbance to vegetation growth in all land uses. For croplands, we measured the interrow spacing and sampled aboveground biomass along 1 m and 2 m rows for soybeans and maize, respectively, while we sampled belowground biomass in pits matching the interrow width, with dimensions of 30 cm in length and 30 cm in depth. For meadows and marshes, we set up protected zones (30 × 30 cm) at the start of the growing season and harvested aboveground and belowground biomass to a depth of 30 cm. For forested swamps, we harvested litter and belowground biomass of trees from two pits per site, each measuring 25 × 25 cm in surface area and 30 cm in depth. All fresh biomass collected from each land use was oven-dried at 80 °C for 3 days and then weighed to calculate the dry biomass by the sampled area. In addition, we measured aboveground biomass of forests as the mean aboveground biomass of trees multiplied by their mean density. We determined the mean tree density using the mean point-tree distance method at a series of 10 stations spread over a 500 m distance (Elzinga et al., 1998). We calculated the mean aboveground biomass (kg) by measuring 20 trees and using the allometric equation Eq. (1):

$$AB = a_{sp} \times DBH^{b_{sp}} \quad (1)$$

Where *AB* is aboveground biomass, *DBH* is diameter at breast height and a_{sp} and b_{sp} represent the parameters associated with the two dominant tree species, silver maple ($a = -2.047$; $b = 2.3852$) and black ash ($a = -2.0314$; $b = 2.3524$) (Chojnacki et al., 2014).

We also measured Normalized Difference Vegetation Index (NDVI), which is another metric of terrestrial vegetation density (Huang et al., 2021). NDVI values were obtained from the European Union's Copernicus Land Monitoring Service information. (2024), providing 10-day NDVI composites at 300 m resolution for our study area. For each site, NDVI was averaged over the period from May to September 2021.

2.4. Soil sampling

We conducted two events of soil sampling. The first, conducted in October 2019, targeted soil samples for the main analyses (SOC stocks, soil $\delta^{13}C$, MAOM-C, POM-C, and sugar biomarkers). At each site, we randomly collected three soil cores, each measuring 1 m in depth and 3.3 cm in internal diameter, using a thermal drill and plastic tubes (Fig. S2) (AMS Gas Powered REDI Boss Hammer, American Falls). The cores were transported in their original tubes and subsequently processed in the laboratory.

The second events of sampling focused on soils for microbial biomass and dissolved organic carbon (DOC) analyses. We sampled composite soils at the 0–10 cm depth at the 30 sites on five occasions: October 2019, June and October 2020, October 2021, and November 2022. We transported these samples to the laboratory in a cooler, sieved them to 2 mm and stored them at −80 °C within 3–4 h of collection to standardize the time interval between field sampling and laboratory biomass measurements.

2.5. Soil organic C analyses

We sliced the soil cores designated for the main analyses into 12 sections at the following depth intervals: 0–5, 5–10, 10–15, 15–20, 20–25, 25–30, 30–35, 35–40, 40–50, 50–60, 60–80, and 80–100 cm (Fig. S2). The corresponding slices from the three cores were pooled to obtain one composite sample for each depth interval at each site. We then sieved soil samples to 2 mm, air-dried them, and removed inorganic C by acid fumigation (HCl) (Harris et al., 2001). After, we measured organic C concentrations of all samples, along with their $\delta^{13}C$ values using an elemental analyzer coupled with an isotope ratio mass spectrometer (EA-IRMS, technology Agilent, Santa Clara, CA, USA). We took 10 g subsamples and oven-dried them at 105 °C for 36 h to estimate the dry

mass of samples.

We calculated soil *bulk density* ($g\ cm^{-3}$) using the equation Eq. (2) (Ferchaud et al., 2023):

$$Bulk\ density = (M_{soil} - M_{coarse}) / (V_{core}) \quad (2)$$

Where M_{soil} and M_{coarse} are the masses (g) of oven-dried soil (<2 mm) and coarse fragments (pebbles and plant debris), respectively, and V_{core} is the volume of the soil core (cm^3).

As large variations in bulk density were expected among land uses, we used the equivalent soil mass (ESM) approach to compare SOC stocks at a common reference soil mass, thereby reducing potential bias (Von Haden et al., 2020; Ferchaud et al., 2023). We calculated the soil mass per unit area ($Mg\ ha^{-1}$) at each soil depth as follows (Eq. (3)):

$$Soil\ mass = Bulk\ density \times d \times 100 \quad (3)$$

Where d is the depth (cm) and the factor 100 converts the units to $Mg\ ha^{-1}$. We selected the least compacted soil profile (i.e., the smallest soil mass within the 0–100 cm depth) to define the reference soil masses for each depth and avoid extrapolation (Table S1). Finally, we used *bulk density*, *SOC concentration*, and the reference soil mass to calculate SOC stocks ($Mg\ ha^{-1}$) using the SimpleESM R script provided by Ferchaud et al. (2023). To facilitate interpretation, we reported SOC stocks at ESM depths, which correspond to the depths in the reference profile at which the reference soil masses are located.

2.6. Microbial biomass and DOC analyses

We conducted a chloroform fumigation-extraction using 0.5 M K_2SO_4 on the soil samples stored for microbial biomass and DOC analyses (Vance et al., 1987). Non-fumigated samples were used to quantify DOC, while microbial biomass was calculated from the difference between fumigated and non-fumigated samples. We obtained the extracts by shaking soil samples with the K_2SO_4 solution, followed by centrifugation, and we filtered the supernatant prior to analysis. The DOC analyses were performed using a portable M9 TOC analyzer (Sievers, SUEZ) via $K_2S_2O_8$ oxidation.

2.7. Soil fractionation

Particle-size fractionation by wet sieving was carried out on 10 g of air-dried soil from each sample to study SOC stabilization. Each 10 g subsample was placed into a 250 ml plastic bottle and agitated in a solution of sodium hexametaphosphate (0.5 %) with 10 glass beads (6 mm diameter) overnight under continuous and regular movement to completely disperse soil particles (Balesdent et al., 1991). The soil fractions that passed through the 53 μm sieve and those remaining on the sieve were defined as the mineral-associated (<53 μm) and the particulate ($\geq 53\ \mu m$) fractions, respectively. We determined the concentration of mineral-associated organic matter - C ($MAOM-C_{concentration}$, $mg\ g^{-1}$ fraction) and particulate organic matter - C ($POM-C_{concentration}$), using the same method as for SOC concentration in bulk soil. To account for the distribution of each fraction in bulk soil, we determined the amount of *MAOM-C* or *POM-C* ($mg\ g^{-1}$ soil), which was calculated using the equation Eq. (4) (d'Annunzio et al., 2008):

$$MAOM-C\ or\ POM-C_{amount} = \frac{(MAOM-C\ or\ POM-C_{concentration} \times M_{fraction})}{M_{soil}} \quad (4)$$

where $MAOM-C$ or $POM-C_{concentration}$ is the concentration of mineral-associated organic C or particulate organic C ($mg\ g^{-1}$ fraction), $M_{fraction}$ is the dry mass of the soil mineral-associated or particulate fraction, and M_{soil} is the dry mass of the bulk soil used for fractionation.

2.8. Determination of plant-derived C

Considering the fact that $\delta^{13}\text{C}$ in the subsoil for all land uses converged towards similar values (see Results section), which can be considered as the reference or old $\delta^{13}\text{C}$ of past vegetation, and that $\delta^{13}\text{C}$ in the topsoil tended towards that of current vegetation (Table S2), we were able to estimate the C derived from current vegetation in the topsoil (Balesdent and Mariotti, 1996). The verification of C/N ratio also supports that OM in the topsoil (mean C/N of 9.7 at the 0–10 cm depth) may be more recently derived from vegetation compared to that in deeper soil layers (8.9 at the 10–30 cm depth). For each land use, we calculated the C derived from current vegetation in each soil fraction (Plant-derived $C_{\text{MAOM or POM}}$, mg g⁻¹ soil) using the equation Eq. (5) (Balesdent and Mariotti, 1996; Tonucci et al., 2017):

$$\text{Plant-derived } C_{\text{MAOM or POM}} = \frac{\delta_f - \delta_{\text{old}}}{\delta_{\text{plant}} - \delta_{\text{old}}} \times (\text{MAOM-C or POM-C}) \quad (5)$$

Where MAOM-C or POM-C was the amount of C in MAOM or POM, δ_f was the $\delta^{13}\text{C}$ of MAOM or POM in the topsoil, δ_{old} was the reference or old $\delta^{13}\text{C}$ values in the subsoil and δ_{plant} was the $\delta^{13}\text{C}$ of current vegetation.

It should be noted that we weight-averaged the $\delta^{13}\text{C}$ of maize and soybean to obtain the plant $\delta^{13}\text{C}$ in both croplands (Table S2). We also calculated the percentage of plant-derived C (%) in MAOM and POM relative to the total MAOM-C and POM-C, respectively.

2.9. Neutral and amino sugar analyses

We determined neutral and amino sugars from air-dried and ground (<0.15 mm) soil subsamples and fine fractions (MAOM) using the method detailed in Chantigny and Angers (2008). Amino sugar biomarkers are frequently used to trace microbial-derived C, as they are specific indicators of bacterial (muramic acid) and fungal (glucosamine) cell walls (Ni et al., 2020). To further investigate OM origin, neutral sugars were also used, as they indicate both plant- and microbial-derived C (Gunina and Kuzyakov, 2015; Song et al., 2025). Namely, arabinose and xylose are mainly derived from plant hemicellulose, whereas galactose and mannose are associated with microbial exopolymers and cell wall components (Gunina and Kuzyakov, 2015).

Briefly, we extracted neutral sugars using H₂SO₄ solution, neutralized the hydrolysates with NaOH solution, sequentially passed the extractant through anion- and cation-exchange resins, and analyzed the purified solutions for plant- (arabinose and xylose; AX) and microbe-derived (galactose and mannose; GM) sugar concentrations (Gunina and Kuzyakov, 2015) using high-performance anion-exchange chromatography (Model Dionex ICS 5000+, Thermo Scientific, Sunnyvale, CA).

To estimate fungal and bacterial amino sugars, we quantified glucosamine (GluN, corrected to take account only of the fungi-derived fraction) and muramic acid (MurA), respectively (Samson et al., 2020). Briefly, we extracted amino sugars using HCl solution and N₂ bubbling and analyzed their concentration by high-performance liquid chromatography (Model Dionex ICS 5000+, Thermo Scientific, Sunnyvale, CA). The amounts of each sugar in MAOM were calculated based on the C mass present in that sugar and by taking into account the distribution of the fine fraction in bulk soil (i.e., mg C g⁻¹ soil). Sugar amounts in POM were estimated by subtracting sugar amounts in MAOM from that in bulk soil.

2.10. Statistical analysis

All statistical tests were performed using R software version 4.4.0 and RStudio version 2024.04.0 (R Development Core Team, 2013). We produced all figures using the visreg and ggplot2 packages of R software (Breheny and Burchett, 2017).

2.10.1. Comparison of biomass and the different forms of SOC between land uses

We first compared plant aboveground and belowground biomass, NDVI, and soil microbial biomass among land uses using linear mixed-effect models (LMMs), where ‘block’ was a random effect (lme4 and lmerTest R packages; Bates et al., 2015; Kuznetsova et al., 2017). Model assumptions (independence of residuals, homoscedasticity and normality of residuals) were assessed using diagnostic graphs and Shapiro-Wilk tests. We applied log transformations to aboveground, belowground and soil microbial biomass to meet model assumptions. We used Tukey's Honest Significant Difference (HSD) test as a *post-hoc* method to make pairwise comparisons where appropriate using the “emmeans” function (emmeans R package; Lenth et al., 2018).

We assessed the relationship between soil depth (0–100 cm) and SOC concentration, SOC stock, BD or soil $\delta^{13}\text{C}$ across land uses to obtain an overview of the variability of these variables along the soil profiles. We used generalized linear mixed-effects models (GLMMs) to produce trend curves for SOC concentration, SOC stock and BD using gamma distributions (log, log, and identity link, respectively) in the “glmer” function (lme4 R package); land use and log-transformed depth were set as fixed effects since model assumptions were not met (Hengl and MacMillan, 2019). We fitted LMMs for $\delta^{13}\text{C}$ after applying a square-root transformation of $\delta^{13}\text{C} + 30$ and a log transformation of soil depth. For all these models, block and sample core were included as random effects to account, respectively, for block effects and the dependence of these variables across soil depths. We also conducted analyses at each soil interval separately to determine the depth of the effect of land use on these soil variables (Table S3); for these models, LMMs with ‘block’ as a random effect met assumptions without transformation.

Based on the results of the comparison above, we conducted a more detailed analysis (LMMs and pairwise comparisons) only at the 0–10 cm for SOC concentration, SOC stock, DOC, POM-C, and MAOM-C, following the same method as for plant biomass. We obtained the values for the 0–10 cm depth by averaging SOC concentrations and summing SOC stocks. We included clay percentage as a predictor in the MAOM-C model as fine particles are considered to be an important factor for C retention on soil minerals and within soil aggregates (Six et al., 2002; Prout et al., 2021). We applied log transformations for POM-C to meet the model assumption.

2.10.2. Assessment of the origins of stable SOC

We compared $\delta^{13}\text{C}$ and plant-derived C in MAOM and POM (mg g⁻¹ soil and %) at the 0–10 cm soil depth between land uses following the same method as for plant biomass. We tested the effects of land use and sugar origin (microbial versus plant) on neutral sugars in MAOM and POM at the 0–10 cm depth using LMMs followed by Tukey's HSD test. For amino sugars in MAOM and POM, we used the same approach to determine the effects of land use and sugar origin (fungal versus bacterial). We applied log transformations for plant-derived C, neutral and amino sugars in MAOM and POM to meet model assumptions.

2.10.3. Assessment of variation in all soil C pools along a disturbance gradient

We conducted a principal component analysis (PCA) to obtain an overview of all soil C pools at the 0–10 cm depth across land uses using the FactoMineR R package. Variables included SOC concentration, MAOM-C, POM-C, plant-derived C, plant- and microbe-derived sugars in bulk soil, and soil microbial biomass. We then correlated the first PCA axis, which explained 83% of the overall variation, with the disturbance gradient using a local polynomial regression (LOESS) via “geom_smooth” function of the ggplot2 R package to capture non-linear relationships. This gradient was created by assigning numeric values at a constant interval, using the “as.numeric” base R function, to each land use arranged from the most to the least disturbed: (1) conventional croplands, (2) improved croplands, (3) temporary meadows, (4) permanent meadows, (5) marshes, and (6) forested swamps. For clarity,

numeric values were subsequently replaced with the corresponding land use names.

3. Results

3.1. Vegetation and soil microbial biomass and NDVI

We found significant variations in aboveground ($P < 0.001$) and belowground ($P < 0.001$) biomass, NDVI ($P < 0.01$), and soil microbial biomass ($P < 0.001$) among land uses. Forested swamps had much higher aboveground biomass compared to the two croplands, the two meadows (temporary and permanent) and marshes (Fig. 2a). Forested swamps, marshes and permanent meadows had higher belowground and soil microbial biomass than both croplands and temporary meadows (Fig. 2cd). NDVI values were higher in forested swamps than in conventional croplands (Fig. 2b).

3.2. Soil organic carbon, bulk density and $\delta^{13}\text{C}$ along the soil profile

The interaction between land use and soil depth significantly affected SOC concentrations and stocks, BD, and $\delta^{13}\text{C}$ along the soil profile of 0–100 cm ($P < 0.001$ for all, Fig. 3). For each depth separately, SOC concentration was influenced by land use down to the 10 cm depth and consistently decreased with increasing soil depth (Fig. 3a). Land use also significantly affected SOC stock down to the 10 cm ESM depth ($605.49 \text{ Mg ha}^{-1}$ of soil), with an additional effect observed at the 35–40 cm ESM depth ($P = 0.04$, Fig. 3c). In addition, land use had a significant impact on BD down to the 20 cm depth (Fig. 3b). The effect of land use on soil $\delta^{13}\text{C}$ was highly significant down to the 25 cm depth, but its effects were also observed at the 50–80 cm depth (Fig. 3d). Across all land uses, the soil $\delta^{13}\text{C}$ values converged towards -25.5‰ with increasing depth, aligning with the mean soil $\delta^{13}\text{C}$ at the subsoil (40–100 cm) (Fig. 3d).

3.3. Comparison between land uses of different forms of organic C at the 0–10 cm soil depth

Land use significantly affected SOC concentrations, DOC, SOC stocks, POM-C ($P < 0.001$ for these variables) and MAOM-C ($P < 0.01$) at the 0–10 cm depth. Forested swamps had higher SOC concentrations and stocks than both croplands and temporary meadows at the 0–10 cm depth (Fig. 4a and S3). Forested swamps and marshes showed higher DOC than both croplands at this depth (Fig. 4b). In addition, MAOM-C was greater in forested swamps than in both croplands (Fig. 4d) and was positively correlated with clay percentage ($P = 0.01$, estimate = 0.50) at the 0–10 cm depth. The MAOM-C amounts for all floodplain land uses remained below the maximum observed capacity (saturation) of $87 \text{ mg C g}^{-1} \text{ clay} + \text{silt}$ (Fig. 4e; Georgiou et al., 2025). Overall, MAOM-C varied less among land uses than POM-C at the 0–10 cm depth (Fig. 4cd). Higher POM-C amounts were observed in forested swamps, marshes and permanent meadows than in temporary meadows and both croplands at this depth (Fig. 4c).

3.4. Origin of stable SOC at the 0–10 cm depth

Land use significantly impacted $\delta^{13}\text{C}$ in both MAOM and POM ($P < 0.001$ for both). Compared to the $\delta^{13}\text{C}$ values of POM, those of MAOM deviated less from the reference or old $\delta^{13}\text{C}$ (-25.5‰), which represents the mean $\delta^{13}\text{C}$ in the deep soil layer (Fig. 5ab). Only the $\delta^{13}\text{C}$ values of MAOM in forested swamps (-28.23‰), marshes (-28.16‰), and permanent meadows (-27.51‰), deviated from this reference, but not those in croplands and temporary meadows (Fig. 5a). The $\delta^{13}\text{C}$ values of POM deviated from this reference with depleted values in forested swamps (-29.33‰), marshes (-28.79‰), and permanent meadows (-28.35‰), whereas those from the conventional (-22.66‰) and improved (-21.64‰) croplands shifted toward enrichment (Fig. 5b). Plant-derived C in MAOM and POM was significantly different

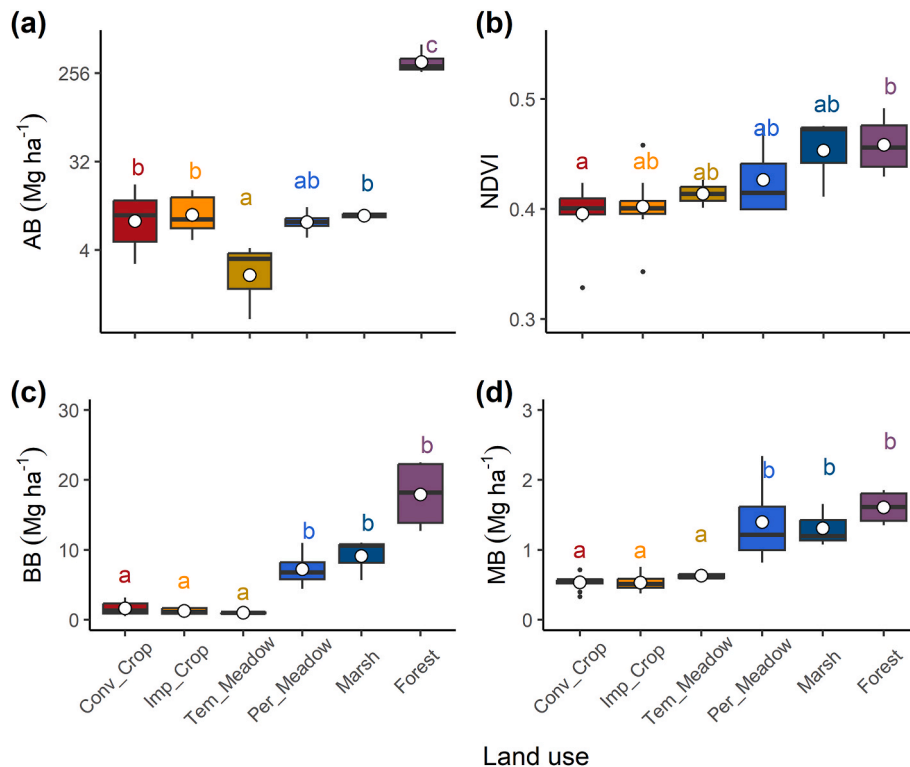


Fig. 2. Aboveground biomass (AB) (a), Normalized Difference Vegetation Index (NDVI) (b), belowground biomass (BB) (c), and soil microbial biomass (MB) (d) among studied land uses. Belowground biomass was collected at the 0–30 cm depth, while soil microbial biomass was estimated at the 0–10 cm depth. NDVI was averaged over May–September period. Boxplots include the mean (white circles) and the median values (black horizontal lines). Different lowercase letters indicate significant differences between land uses ($P < 0.05$). Conv: conventional; Imp: improved; Per: permanent; Tem: temporary.

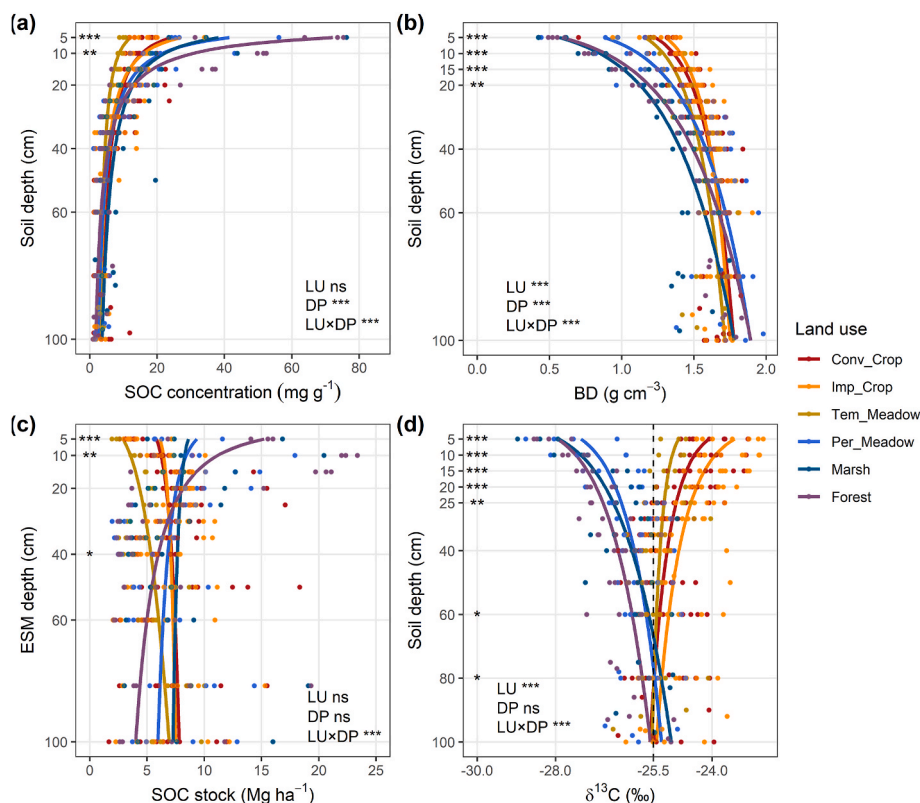


Fig. 3. Soil organic carbon (SOC) concentration (a), bulk density (BD) (b), SOC stock (c), and soil $\delta^{13}\text{C}$ (d) as a function of depth for the different land uses. Trend curves were obtained from mixed models with land use (LU) and soil depth (DP) as predictors. Significance code: ***: $P < 0.001$; **: $P < 0.01$; *: $P < 0.05$; ns: nonsignificant. Stars to the right of a depth indicate significant differences between land uses at that depth, considered separately (Table S3). Equivalent soil mass (ESM) depths in panel “c” were used to account for notable variations in bulk density among land uses and correspond to the soil depths in the reference profile at which the reference soil masses are located. The dotted line in panel “d” represents the mean $\delta^{13}\text{C}$ in the deep soil layer of 40–100 cm (-25.5‰), a value considered to be the reference $\delta^{13}\text{C}$. Conv: conventional; Imp: improved; Per: permanent; Tem: temporary.

between land uses at the 0–10 cm depth ($P < 0.01$ for both). More plant-derived C was present in MAOM in the forested swamps and marshes than in both croplands, and there was more plant-derived C in POM in the forested swamps than in both croplands and temporary meadows (Fig. 5cd). However, the percentage of plant-derived C in MAOM (19 to 71 %), relative to total MAOM-C, did not differ between land uses (Fig. 5e), whereas the percentage of plant-derived C in POM (10 to 89 %), relative to total POM-C, was higher in permanent meadows and marshes than in temporary meadows ($P = 0.01$, Fig. 5f).

Sugar origin (microbial versus plant) and land use ($P < 0.001$ for all), but not their interaction, had significant effects on neutral sugar amounts in both MAOM and POM at the 0–10 cm depth (Fig. 6ab). Microbe-derived neutral sugars were more abundant in MAOM than plant-derived neutral sugars, whereas the opposite trend was observed in POM (Fig. 6ab). Forested swamps and marshes had higher amounts of neutral sugars in MAOM and POM compared to both croplands and temporary meadows (Fig. 6ab).

Amino sugars in MAOM and POM at the 0–10 cm depth were affected by sugar origin (fungal versus bacterial) and land use ($P < 0.001$ for all), but not by their interaction (Fig. 6cd). Fungi-derived amino sugars were more abundant than bacteria-derived amino sugars in both MAOM and POM (Fig. 6cd). Forested swamps and marshes had higher amino sugars in MAOM than both croplands, while forested swamps and permanent meadows showed higher amino sugars in POM than both croplands and temporary meadows (Fig. 6cd).

3.5. Variation in soil C pools along a disturbance gradient

A clear distinction of all soil C pools was observed between temporary (croplands and temporary meadows) and permanent land uses

(permanent meadows, marshes, and forested swamps), as shown by PCA (Fig. 7a). The overall soil C pools, represented by PCA1, increased with decreasing anthropogenic disturbance intensity, ranging from conventional croplands to forested swamps (Fig. 7b). However, this increase did not follow a linear pattern along the disturbance gradient, with a tipping point occurring at the transition between temporary and permanent meadows (Fig. 7b).

4. Discussion

4.1. Key role of land uses in SOC storage in floodplain topsoil

The effects of land use on SOC concentrations and stocks in the LSP floodplain were mostly observed in the top 10 cm. This is likely driven by greater contributions of recently plant-derived C at that depth, as indicated by soil $\delta^{13}\text{C}$ values shifting toward vegetation values, consistent with the topsoil but not the subsoil component of our first hypothesis. Forested swamps had higher SOC and DOC than croplands in our floodplain topsoil, consistent with previous studies (Omengo et al., 2016; D’Elia et al., 2017), likely due to high inputs from vegetation, as indicated by above- and belowground biomass and NDVI. Notably, root C may be retained more efficiently in SOC than aboveground inputs, as the disproportionate difference in aboveground biomass between forested swamps and marshes was not reflected in SOC stocks. This is likely because roots are in direct contact with the soil and release continuous inputs through turnover and exudates, which soil microbes can efficiently process into SOC (Ghafoor et al., 2017; Sokol et al., 2019; Rabearison et al., 2025). For both croplands, low biomass inputs combined with repeated soil tillage likely explain the low SOC storage in the topsoil (Bronick and Lal, 2005; Ciais et al., 2010).

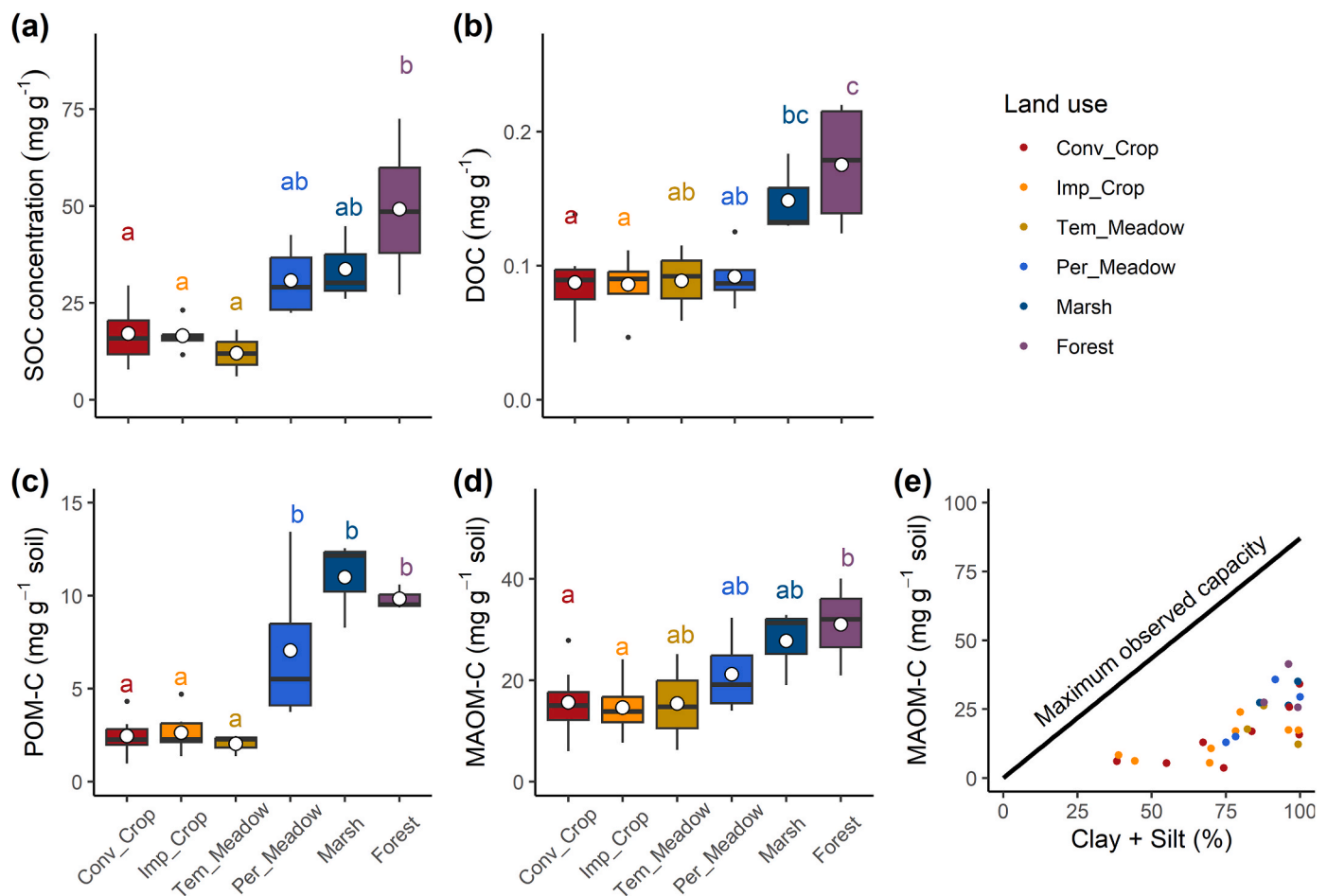


Fig. 4. The effects of land use on the different forms of organic carbon (C) at the 0–10 cm soil depth and the maximum observed capacity for mineral-associated organic matter - C (MAOM-C) storage. (a) Soil organic carbon (SOC) concentration; (b) dissolved organic carbon (DOC), (c) particulate organic matter - C (POM-C), (d) MAOM-C, (e) MAOM-C as a function of clay + silt percentage at the 0–10 cm depth, with the maximum observed capacity (87 mg C g⁻¹ clay + silt) for MAOM-C storage according to Georgiou et al. (2025). Boxplots include the mean (white circles) and the median values (black horizontal lines). Different lowercase letters indicate significant differences between land uses ($P < 0.05$). Conv: conventional; Imp: improved; Per: permanent; Tem: temporary.

Given that our experimental design (same elevation and comparable flooding frequency across sites) minimized variability in allochthonous inputs, the observed land use effects on topsoil SOC indicate that autochthonous vegetation inputs are a primary driver of SOC accumulation in floodplain topsoil (Omengo et al., 2016; Scott and Wohl, 2018). This is supported by the predominance of plant-derived C in POM (61% on average across land uses), highlighting the strong contribution of local vegetation inputs to this fraction. However, although variability in allochthonous inputs was constrained by design, their absolute contribution likely remains substantial in this floodplain due to its low elevation and recurrent sediment deposition during flooding events (Mayer, 2018; Heger et al., 2021). Overall, the relative contribution of autochthonous and allochthonous inputs to SOC reflected not only land use, but also the combined influence of geomorphic position, flooding regime (frequency and hydroperiod) and the quantity of suspended sediment (Heger et al., 2021; Shen et al., 2022).

Land use effects on SOC stocks were no longer evident below 10 cm depth, since vegetation inputs were found to have reduced contributions to SOC in floodplain subsoils (Omengo et al., 2016). Frequent flood events could cause partial loss of these vegetation inputs, limiting OM accumulation in deeper soil layers (Saint-Laurent et al., 2017). Contrary to SOC concentrations, SOC stocks of all land uses decreased less with soil depth due to increasing bulk density, highlighting the important contribution of subsoil to total SOC stocks in floodplains (D'Elia et al., 2017). In addition, soil $\delta^{13}\text{C}$ signatures indicate that recently

plant-derived C contributed to SOC down to 25 cm, comparable to the results reported by Kanari et al. (2025). The variation in soil $\delta^{13}\text{C}$, but not in SOC stocks, suggests that between 10 and 25 cm depth, land use effects were more related to SOC turnover than accumulation (You et al., 2021). The convergence of soil $\delta^{13}\text{C}$ values across all land uses toward approximately -25.5‰ below 40 cm indicates that recent vegetation inputs become negligible at this depth, suggesting a predominance of older or allochthonous C. Comparable subsoil $\delta^{13}\text{C}$ signatures have been reported in other floodplain systems (D'Elia et al., 2017; Steger et al., 2019), likely reflecting the isotopic composition of sedimentary C deposited during past flooding events.

4.2. Is stable SOC dominated by microbial- or plant-derived C?

Soils in forested swamps contained more C as MAOM than cropland soils, highlighting the role of trees in promoting accumulation of stable SOC in the LSP floodplain (Del Galdo et al., 2003; Shoumik et al., 2025). Regardless of land use, microbial-derived neutral sugars were more abundant in MAOM than plant-derived neutral sugars in our floodplain, refuting our hypothesis of different patterns between forests and meadows. The high plant biomass inputs from forested swamps would thus have been efficiently utilized by soil microbes, enhancing microbial by-products and MAOM-C accumulation (Cotrufo et al., 2013; Liang et al., 2017). As in van Leeuwen et al. (2017), higher microbial biomass in forest topsoil, relative to croplands, was found in our study,

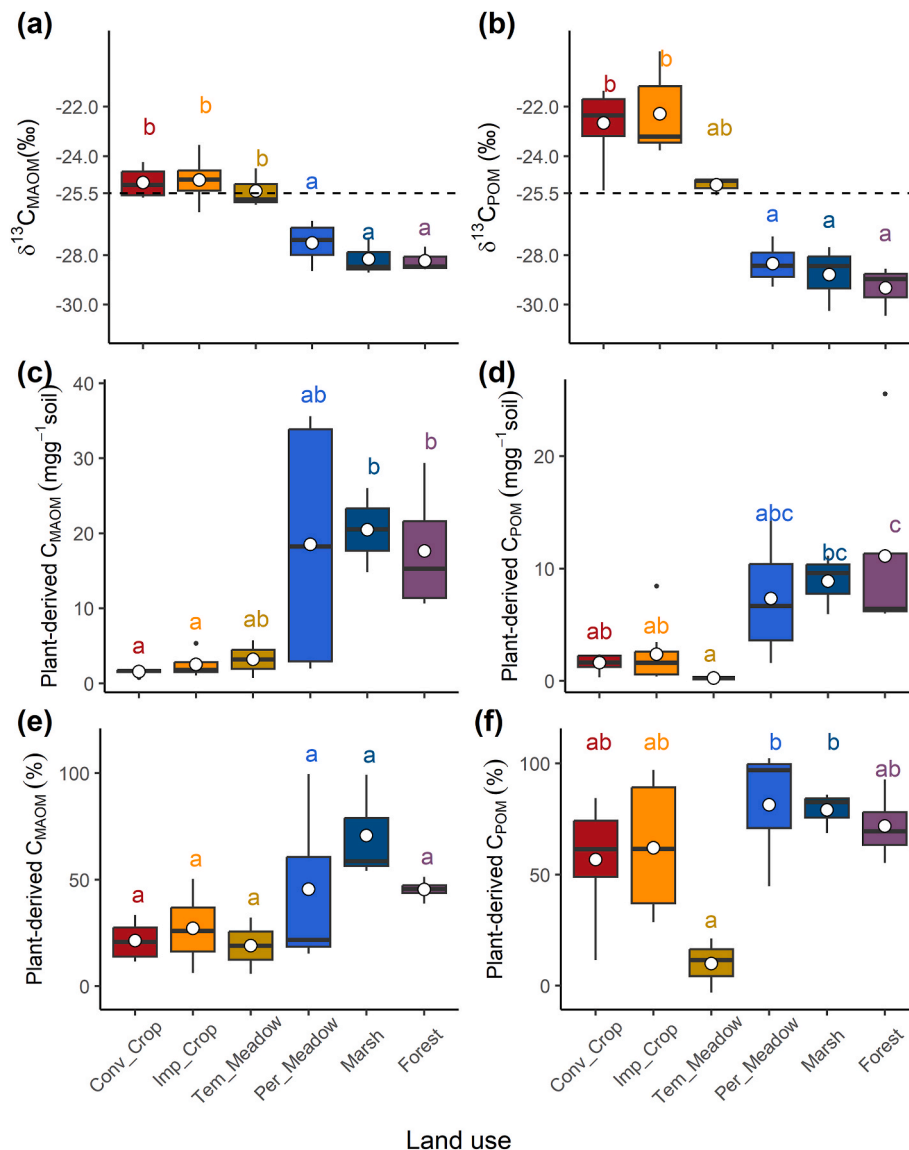


Fig. 5. $\delta^{13}\text{C}$ values and plant-derived C in mineral-associated organic matter (MAOM) (a,c,e) and in particulate organic matter (POM) (b,d,f) across land uses at the 0–10 cm depth. Plant-derived C was estimated using the $\delta^{13}\text{C}$ tracing technique (see section 2.6 for details). Boxplots include the mean (white circles) and the median values (black horizontal lines) and the dotted line represents the reference $\delta^{13}\text{C}$ (-25.5‰). The percentage of plant-derived C (%) in MAOM and POM was calculated relative to the total MAOM-C and POM-C, respectively. Different lowercase letters indicate significant differences between land uses ($P < 0.05$). Conv: conventional; Imp: improved; Per: permanent; Tem: temporary.

supporting the involvement of microbial metabolism in the SOC stabilization mechanism. Microbial by-products, particularly necromass, are often compositionally simpler and enriched in polar functional groups that have a stronger binding affinity for soil mineral phases, reducing their accessibility to microbial decomposers compared to plant-derived compounds (Kleber et al., 2021; Zhao et al., 2025). Within microbial communities, the higher fungi-derived amino sugars found in MAOM under all land uses, compared to bacteria-derived amino sugars, highlights the important contribution of fungi in the formation of stable SOC (Fu et al., 2025). In floodplains characterized by fine-textured soils, fungal compounds can be preferentially accumulated in MAOM due to their cell wall and necromass composition (including glucosamine), which resist decomposition and contain polar functional groups that strongly bind minerals (Kleber et al., 2021; Fu et al., 2025).

The $\delta^{13}\text{C}$ and sugar analyses also confirm that some plant-derived inputs accumulated directly in MAOM without microbial assimilation. In forested swamps, marshes and permanent meadows, the MAOM $\delta^{13}\text{C}$ values shifted towards those typical of trees and herbaceous plants,

indicating inputs of plant-derived C in MAOM. These inputs are likely mediated by roots, as supported by the high belowground biomass under these land uses. Previous studies reported that soluble compounds derived from roots (exudates and rhizodeposits) can be directly adsorbed onto soil mineral surfaces, contributing significantly to MAOM (Hagedorn et al., 2015; Angst et al., 2017; Córdova et al., 2018). Plant inputs may also first undergo *ex vivo* transformation by microbial extracellular enzymes, producing labile plant-derived compounds that readily interact with soil minerals (Liang et al., 2017; Sokol et al., 2019; Whalen et al., 2022). Thus, enhancing plant-derived C in MAOM could be achieved by promoting land uses that favor labile compounds, such as herbaceous-rich land uses.

High clay contents facilitated MAOM-C accumulation in our floodplain topsoil, as reported by several floodplain studies (Bullinger-Weber et al., 2014; Heger et al., 2024). These fine-textured particles promote the adsorption of organic C onto mineral and metal oxide surfaces, leading to the formation of stable organo-mineral complexes or MAOM (Cotrufo et al., 2013; Lavalée et al., 2020). Furthermore, fine texture

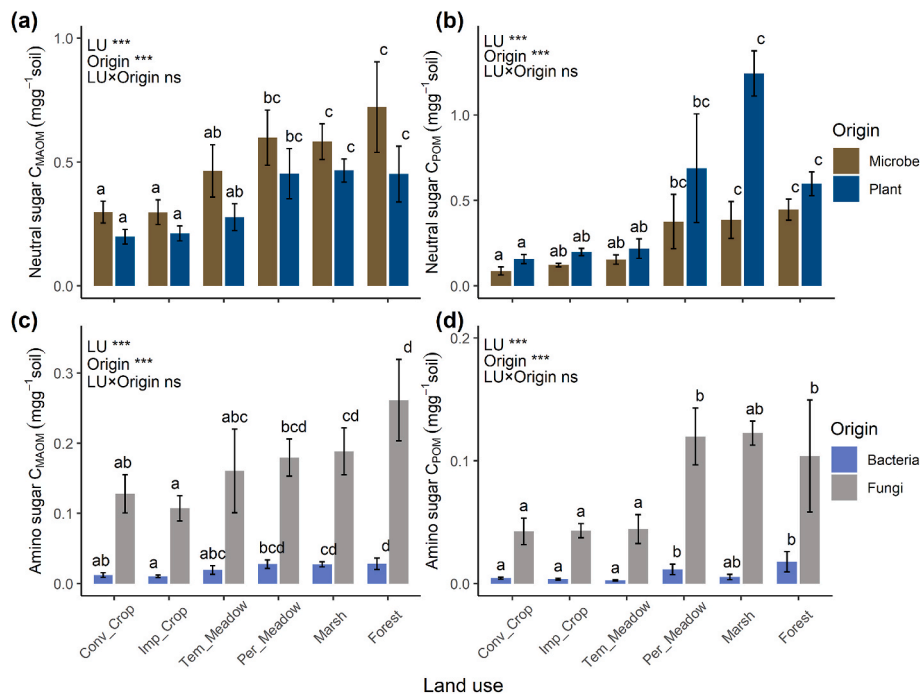


Fig. 6. Origins of neutral (a,b) and amino (c,d) sugars in mineral-associated (MAOM; a,c) and particulate (POM; b,d) organic matter across land uses (LU) at the 0–10 cm depth. Sugar amounts were calculated based on the C mass present in each sugar and by taking into account the distribution of each soil fraction in the bulk soil (i.e., mg C g⁻¹ soil). Neutral and amino sugars were determined using distinct analytical procedures; thus, their values are not directly comparable. Significance code: ***: $P < 0.001$; ns: nonsignificant. Different lowercase letters indicate significant differences between land uses ($P < 0.05$). Conv: conventional; Imp: improved; Per: permanent; Tem: temporary.

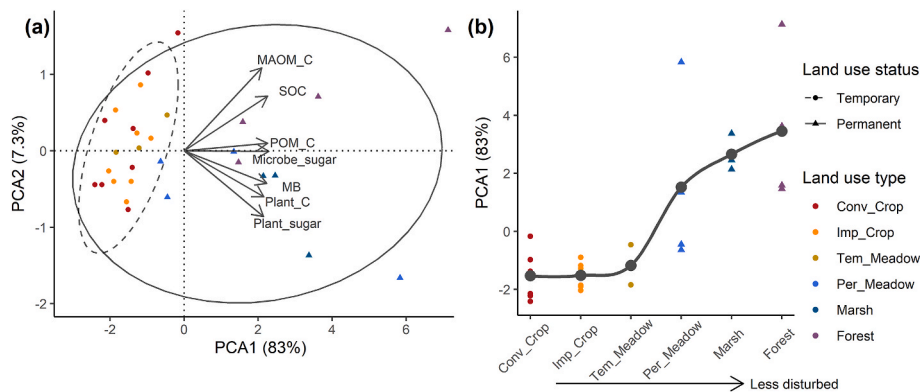


Fig. 7. Principal component analysis (PCA) of all soil carbon (C) pools (a) and variation of overall soil C pools along the anthropogenic disturbance gradient at the 0–10 cm depth (b). Black circles represent the means, and the curves are generated using a local polynomial regression (see Statistical analysis). Conv: conventional; Imp: improved; MAOM-C: mineral-associated organic C; MB: soil microbial biomass; Per: permanent; POM-C: particulate organic C; SOC: Soil organic C; Tem: temporary.

can also be a strong driver of MAOM accumulation in floodplain sub-soils, through repeated sediment deposition and reduced C mineralization rates (Heger et al., 2024). Regardless of land use, MAOM was not saturated in the LSP floodplain, compared to the maximum storage capacity of 87 mg C g⁻¹ clay + silt proposed by Georgiou et al. (2025). Therefore, there could be considerable potential to stabilize further SOC in this area. However, this potential also depends on the availability of reactive mineral surfaces, which control adsorption capacity, and continuous inputs of reactive minerals can be determinant in floodplain soils (Kleber et al., 2021).

4.3. Why does POM remain important?

Our sugar analyses indicated that plant-derived compounds were the

dominant source of C in POM. Accordingly, many studies have shown that POM is mainly comprised of partially decomposed OM from vegetation (Balesdent, 1996; Six et al., 2002). Significantly more POM-C was found in forested swamps, marshes and permanent meadows than in croplands and temporary meadows, highlighting the importance of this SOC reservoir in soils under forest and permanent herbaceous-dominated vegetation (Yeasmin et al., 2020). In addition, POM is a vital SOC reservoir under sub-oxic or anoxic conditions, such as those in floodplains and peatlands (Bridgham et al., 2006; Tolunay et al., 2024). Recalcitrant POM (e.g., lignin-rich) can persist longer in floodplains as its decomposition is likely to be slow due to its complex molecular structure and anoxic conditions (Poirier et al., 2018; Tolunay et al., 2024). However, its inherent recalcitrance may play a limited role, as flooding can also facilitate its loss (Saint-Laurent et al., 2017). Recent

findings suggest that POM constitutes a key compartment for additional C storage as it may not be subjected to saturation mechanisms (Dignac et al., 2025; Georgiou et al., 2025). The greater contrast in POM-C than MAOM-C found between croplands and permanently covered soils indicates that POM remains more vulnerable to anthropogenic disturbance than MAOM (Cotrufo et al., 2019; Angst et al., 2021).

We also detected microbial-derived compounds in POM, suggesting that some microbial by-products remain unprotected by mineral surfaces. During decomposition, POM is not entirely lost as CO₂, but a portion can subsequently be transferred into MAOM through microbial metabolism (Liang et al., 2017; Dignac et al., 2025). Thus, increased POM could serve as a potential reservoir for the eventual MAOM formation upon binding to mineral surfaces (Dignac et al., 2025; Georgiou et al., 2025). In this sense, it is important to promote land uses that enhance the transfer of C from POM to MAOM, such as forested swamps (Kallenbach et al., 2019).

4.4. Implications for land use management in floodplain ecosystems

The overall soil C pools were predominantly higher under land uses with permanent vegetation (permanent meadows, marshes, and forested swamps), emphasizing the critical role of conserving these stable ecosystems within floodplains. The variation of SOC pools (SOC, POM-C, MAOM-C, plant-derived C and microbial-derived C) was not linear along the disturbance gradient, with a tipping point occurring only in the transition from temporary to permanent meadows, but not during the shift from permanent herbaceous to woody vegetation (Fig. 7b and S4). In this context, prioritizing the conversion of temporary meadows or low-productivity croplands to permanent meadows, once prevalent in the early 1980s, is crucial (Yeasmin et al., 2020). Significant increases in SOC represent a long-term and gradual process, with permanent meadows having 150 % higher SOC stocks and 37% higher MAOM-C compared to temporary meadows. For instance, the permanent meadow in Baie-du-Febvre has been maintained by the same family for at least three generations, according to the owner, and showed SOC comparable to those in forested swamps. Nevertheless, such conversion efforts must be accompanied by targeted outreach to address negative perceptions among floodplain farmers, particularly regarding the forage quality and the competitive nature of bluejoint reedgrass (*Calamagrostis canadensis* [Michaux] Beauvois) (Sparrow and Panciera, 2005; Campeau et al., 2024).

As forested swamps were the most effective systems for the storage of stable SOC, we also recommend expanding their area through afforestation in low-productivity croplands while minimizing soil disturbance. Specifically, reforestation with silver maple in the LSP floodplain could not only increase SOC stabilization but also support maple syrup agricultural production, as this species is already dominant in the LSP landscape (Watson et al., 2024) and two silver maple sugarbushes equipped with tubing were observed in Pierreville. Fast-growing trees such as poplar (*Populus* spp.) may also be planted to promote rapid timber production and enhance SOC stabilization, as they tend to transfer more labile compounds that favor microbial by-product formation and hence MAOM accumulation (Cotrufo et al., 2013; Poirier et al., 2018; Rabearison et al., 2025). In addition, it should be noted that neither conventional croplands nor agri-environmental practices (improved croplands) were beneficial for SOC stabilization in the LSP floodplain, in line with the findings of Kanari et al. (2025).

5. Conclusion

Land use mainly influenced SOC storage and stabilization in the topsoil of the LSP floodplain. Soil $\delta^{13}\text{C}$ signatures indicated that recently plant-derived C contributed to SOC down to 25 cm depth, whereas the subsoil was increasingly dominated by older C. Forested swamps had more stable SOC (MAOM-C) than other land uses, due to greater plant biomass inputs that increased both microbial- and plant-derived C. Our

biomarker analysis showed that microbial-derived sugars, particularly of fungal origin, were more abundant in MAOM than plant-derived sugars across all land uses, highlighting the persistent key role of fungal compounds in floodplain SOC stabilization through interactions with fine-textured sediments. Across all land uses, MAOM remained below its saturation capacity, indicating substantial potential for further C stabilization. Permanent meadows and marshes also contributed significant C to POM, which appears to be an important SOC reservoir in herbaceous-dominated systems. Decreasing anthropogenic disturbance enhanced soil C pools, with a tipping point occurring during the transition from temporary to permanent meadows. Therefore, conserving and expanding land uses with permanent vegetation covers in floodplains would likely ensure their role as global C sinks, supporting climate change mitigation. To fully understand floodplain ecosystem functioning, further studies are needed to investigate how flooding intensity influences microbial community composition and SOC partitioning among different soil fractions.

CRediT authorship contribution statement

Toky Jeriniaina Rabearison: Conceptualization, Formal analysis, Methodology, Software, Validation, Visualization, Writing – original draft, Writing – review & editing. **Jim Félix-Faure:** Conceptualization, Investigation, Methodology, Writing – review & editing. **François Guillemette:** Resources, Validation, Writing – review & editing. **Michelle Garneau:** Resources, Validation, Writing – review & editing. **Christine Martineau:** Resources, Supervision, Validation, Writing – review & editing. **Louis Astorg:** Resources, Writing – review & editing. **Martin Chantigny:** Resources, Writing – review & editing. **Virginie Favreau:** Investigation, Writing – review & editing. **Jessika Malko:** Investigation, Writing – review & editing. **Alexandre Roy:** Investigation, Writing – review & editing. **Vincent Maire:** Conceptualization, Formal analysis, Funding acquisition, Investigation, Methodology, Project administration, Supervision, Validation, 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.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.soilbio.2026.110199>.

Data availability

The datasets generated during and/or analyzed during the current study are available in Borealis repository (Canadian Dataverse) [<https://doi.org/10.5683/SP3/ZIWLFP>].

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