

RESEARCH ARTICLE

Varying effects of stream restoration on riparian soil carbon persistence and methane emissions in the southern Rocky Mountains, United States

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Abstract

Introduction: Channel incision can disconnect streams from their floodplains, potentially depleting carbon (C) in riparian soils by stimulating microbial decomposition. Stream restoration may offer an opportunity to replenish soil C pools by saturating riparian soils with water and slowing microbial activity.

Objectives: Assess how stream restoration affects soil C persistence and greenhouse gas emissions in riparian soils.

Methods: We measured soil C pools and methane (CH₄, a strong greenhouse gas) emissions adjacent to stream reaches restored with plug-and-ponds or beaver dam analogs (BDAs) in northern New Mexico. We also measured mineral-associated organic carbon (a persistent C pool) and particulate organic matter (particulate organic C [POC], a more labile C pool) as indicators of soil C persistence and their isotopic composition to inform formation pathways.

Results: Plug-and-ponds increased POC from 22.3 ± 2.34 g C/kg to 62.5 ± 9.71 g C/kg within 3 years, with increases in soil moisture and decreases in $\delta^{13}\text{C}$ together suggesting suppression of microbial decomposition. While plug-and-ponds also increased CH₄ emissions (from 0.14 ± 0.055 $\mu\text{g C m}^{-2} \text{s}^{-1}$ to 1.73 ± 0.93 $\mu\text{g C m}^{-2} \text{s}^{-1}$), preliminary calculations suggest that this did not offset the climate benefit of increased POC. BDAs did not consistently increase soil C or soil moisture; steeper slopes at the BDA sites may have prevented water saturation that is necessary to slow microbial decomposition.

Conclusions: Stream restoration efforts can increase soil C accumulation enough to offset increased CH₄ emissions, but success depends on initial site characteristics, like slope.

Implications for Practice: Stream restoration practices can provide a net climate benefit by saturating soils with water and allowing soil carbon (C) accumulation to outpace soil methane emissions. Because steeper slopes adjacent to streams may prevent water saturation and soil C accumulation in riparian soils, restoration practitioners should consider initial site characteristics—including the slope of riparian areas—if their aim is to increase soil C stocks and soil moisture. Furthermore, gains in soil C were tightly coupled with increased soil moisture, suggesting that monitoring soil moisture may be an effective way to assess how stream restoration efforts affect C cycling in riparian soils.

Key words: ¹³C, beaver dam analogs, MAOM, meadow rewetting, plug-and-ponds, POM, pond-and-plugs

Introduction

Channel incision from overgrazing, timber harvest, and wildfire can decrease lateral connectivity between streams and their floodplains (Galay 1983; Schilling et al. 2006; Simon & Rinaldi 2006), potentially depleting carbon (C) stocks in riparian soils (Sutfin et al. 2016; Wohl et al. 2017). Over 30% of global soil organic C is stored in wetlands—including riparian wetlands—despite accounting for just 6% of the land surface (Poulter et al. 2021; Stewart et al. 2024). Microorganisms can deplete C stored in riparian wetlands if channel incision allows more oxygen (O₂) to diffuse into soil pore space to stimulate decomposition (Wohl et al. 2017). To reestablish anoxic conditions in degraded riparian soils, practitioners often implement restoration methods—including plug-and-ponds and beaver dam analogs (BDAs)—that aim to slow stream flow, raise groundwater, and increase soil moisture (Bernhardt et al. 2005; Wohl et al. 2015; Reed et al. 2022). Yet, it is not clear how these practices affect soil C cycling in restored riparian wetlands.

Persistent water saturation of wetland soils slows O₂ diffusion and establishes anoxic conditions that hinder microbial decomposition and allow soil C to accumulate (Estop-Aragónés et al. 2013; Jackson et al. 2017; Stewart et al. 2024). Wetland C accumulation is further maintained by the persistence of

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plants that tolerate anoxic soils and pump C belowground despite low O₂ availability (Vartapetian & Jackson 1997). The combination of inhibited C outputs from decomposition and persistent C inputs from plants favors the accumulation of organic C in wetland soils. Riparian wetlands become degraded when disturbances alter the balance between stream sediment supply and transport capacity, lowering water tables and decreasing lateral connectivity between streams and adjacent riparian soils (Galay 1983; Schilling et al. 2006; Simon & Rinaldi 2006). As degraded riparian soils dry out, more oxygenated soils can limit the success of native riparian plants while also promoting the loss of soil C via microbial decomposition (Estop-Aragónés et al. 2013; Jackson et al. 2017; Stewart et al. 2024; Driscoll et al. 2025). To restore riparian wetlands, management efforts likely need to reestablish connections between streams and riparian areas to limit the diffusion of O₂ into surface soils.

Whether stream channel incision depletes riparian soil C stocks also depends on how resistant soil C is to microbial decomposition. Microorganisms can readily decompose particulate organic C (POC), while mineral-associated organic C (MAOC) is more protected from decomposers (Lavalley et al. 2020). Much of the C that accumulates in wetlands is POC because slowed microbial decomposition under anoxic conditions prevents the conversion of plant C inputs into the smaller biopolymers and monomers that comprise MAOC (Lehmann & Kleber 2015; Sokol et al. 2022), suggesting soil C may be susceptible to losses if stream degradation aerates riparian soils. In addition to depleting total C pools, the stimulation of microbial decomposition can enrich the isotopic composition of soil organic C by preferentially decomposing plant material that is isotopically lighter (i.e. less ¹³C) than microbial biomass (Ehleringer et al. 2000; Werth & Kuzyakov 2010; Klink et al. 2022). As such, monitoring soil POC, MAOC, and their isotopic composition can indicate which processes deplete C in degraded riparian wetlands, and if restoration efforts are restoring C cycling processes common in wetland soils.

In addition to affecting soil C pools, stream incision and restoration may alter the emission of methane (CH₄) and nitrous oxide (N₂O)—strong greenhouse gases (GHGs)—from riparian soils (Zou et al. 2022; U.S. Environmental Protection Agency 2024; Schuster et al. 2024). Wetlands can emit more CH₄ and N₂O than drained riparian soils because high soil organic C and low O₂ create conditions that favor the production of CH₄ via methanogenesis and N₂O via denitrification (Knowles 1982; Estop-Aragónés et al. 2013; Rosentreter et al. 2021; Sun et al. 2024). However, not all wetlands emit much CH₄ and N₂O; highly reducing conditions in some wetlands may limit net N₂O emissions by promoting the reduction of N₂O to N₂ (Morse & Bernhardt 2013; Zou et al. 2022), while some wetland plants can facilitate the oxidation of CH₄ and prevent its release to the atmosphere (Cui et al. 2024; Schuster et al. 2025). Because multiple factors determine how much CH₄ and N₂O are emitted from wetlands, it is not clear how restoration efforts impact the emission of these strong GHGs from riparian soils.

Plug-and-ponds are created by using excavated alluvial material from a degraded stream to build a series of “plugs” that slow stream flow and lead to ponding that traps sediment, raises water

tables, and restores riparian plant communities (Loheide & Gorkel 2007; Hammersmark et al. 2009; Pope et al. 2015). However, these benefits are not always observed due to local confounding factors (e.g. climate and topography) (Natali & Kondolf 2018), leading to studies showing both an increase (Reed et al. 2022) or no effect (Pope et al. 2015) on soil C. BDAs are a widely implemented restoration technique that also aim to reduce erosion and raise water tables, but do so by building permeable dams out of branches, rocks, and sediment to trap sediment, aggrade channels, and raise stream channels (Pollock et al. 2014; Pilliod et al. 2018; Lautz et al. 2019; Munir & Westbrook 2021; Nash et al. 2021). However, like plug-and-ponds, these outcomes are not always achieved (Scamardo & Wohl 2020; Munir & Westbrook 2021), and effects on soil C persistence remain unclear (Hinshaw & Wohl 2023). To better understand the relationship between riparian restoration and soil C accumulation with the goal of informing climate-smart restoration practices, we ask: how does stream restoration affect soil C cycling and GHG emissions in restored and degraded riparian soils?

We hypothesized that stream restoration would increase riparian soil moisture and suppress microbial decomposition, allowing soil C to accumulate as POC while also stimulating anaerobic processes that produce CH₄ and N₂O. To test this hypothesis, we measured soil POC and MAOC from restored and degraded stream reaches in northern New Mexico. We also measured the isotopic composition (δ¹³C) of these C pools to assess which processes contributed to C loss or accumulation. Finally, we measured soil CH₄ and N₂O emissions to assess the climate benefit of restoration treatments. Together, these data inform how widely implemented stream restoration techniques affect the plant and microbial processes that control soil C persistence in riparian wetlands.

Methods

Study Site

We measured soil C pools and GHG emissions from restored and control stream reaches in the Carson (2024) and Santa Fe (2023) National Forests in northern New Mexico. Two reaches along Vidal Creek in the Carson National Forest were restored with plug-and-ponds in 2021 to combat channel incision from livestock overgrazing and mining (The Quivira Coalition 2005). One control reach was chosen upstream of the plug-and-ponds and another control reach was established along nearby Comanche Creek (Fig. 1; Driscoll et al. 2025). Riparian soils near the restored and control reaches are classified as deep loamy Cumulic Cryaquolls (Soil Survey Staff, Natural Resources Conservation Service, United States Department of Agriculture 2019).

Two reaches of San Antonio Creek in the Santa Fe National Forest were restored with BDAs in 2020, where livestock overgrazing and timber harvest have contributed to channel incision (USDA Forest Service 2011). Two degraded reaches with similar characteristics (i.e. channel incision and soil series) were chosen as control sites along a one-mile section of nearby Rio Cebolla in the subwatershed neighboring San Antonio Creek

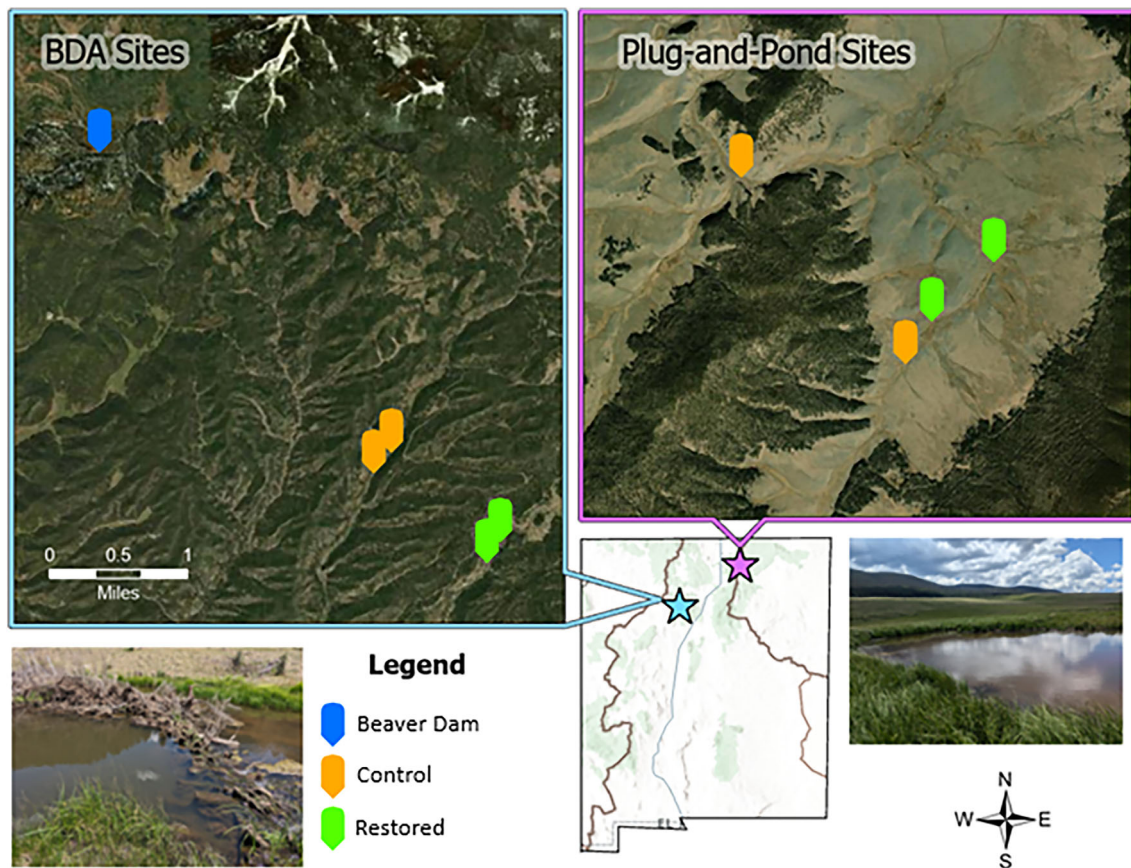


Figure 1. Map of restored, control, and beaver dam reaches in northern New Mexico. Beaver dam analog (BDA) control sites (orange, upper left panel, lat 35°58'23.2"N, long 106°39'30.3"W; lat 35°58'10.7"N, long 106°39'42.8"W) are in the Headwaters Rio Cebolla subwatershed. BDA-restored sites (green, upper left panel, lat 35°57'09.4"N, long 106°38'05.2"W; lat 35°57'24.3", long 106°37'60.0"W) are in the Outlet San Antonio Creek subwatershed. Plug-and-pond restored (green, upper right panel, lat 36°44'35.8"N, long 105°14'58.8"W; lat 36°44'53.7"N, long 105°14'35.7"W) and control sites (orange, upper right panel, lat 36°45'21.6"N, long 105°16'09.5"W; lat 36°44'26.1"N, long 105°15'07.3"W) are in the Little Costilla watershed. The Beaver Dam is in the Rio de Las Vacas subwatershed (lat 36°01'52.7"N, long 106°43'32.5"W). The bottom left photo shows one of the BDAs and the bottom right photo shows one of the plug-and-ponds.

(Fig. 1; Driscoll et al. 2025). Soils in the restored and control riparian areas are both classified as fine-loamy, mixed, superactive, frigid Cumulic Endoaquolls (Chamita series) derived from shale, sandstone, and metamorphic rock (Soil Survey Staff, Natural Resources Conservation Service, United States Department of Agriculture 2019). To provide a reference for how much C natural beaver dams can store in the Santa Fe National Forest, we also sampled one abandoned beaver dam with soils classified as fine-loamy mixed, superactive, mesic Aridic Haplustolls derived from basalt and pyroclastics (Valle grande series; Soil Survey Staff, Natural Resources Conservation Service, United States Department of Agriculture 2019).

Experimental Design

We collected soil samples from two restored and two control reaches (100 m each) for each restoration method, as well as from one abandoned beaver complex in the Santa Fe National Forest (Fig. 1). We established three transects perpendicular to each reach (0, 50, and 100 m upstream of the restoration

structure) that extended 10 m from the stream on either side. We also used digital elevation models to estimate the slope of each transect in both the Santa Fe National Forest (U.S. Geological Survey 2024) and the Carson National Forest (U.S. Geological Survey 2025). Soil samples were collected 1, 5, and 10 m from the stream along each transect in each of the control, BDA-restored, and reference beaver complex reaches in the Santa Fe National Forest in the summer of 2023 (18 locations $\times$ 5 reaches = 90 samples) and from the control and plug-and-pond-restored reaches in the Carson National Forest in 2024 (18 locations $\times$ 4 reaches = 72 samples) using a 5.1-cm diameter, hollow, steel cylinder. In the Santa Fe National Forest, three depth increments were collected at each sampling location: 0–10, 10–30, and 30–50 cm. Due to difficulties sampling below 30 cm in the water-saturated soils in the Carson National Forest, we sampled soils from 0 to 10 cm, 10 to 20 cm, and 20 to 30 cm depth increments.

All soil samples were transported to the Rocky Mountain Research Station in Albuquerque, New Mexico within 72 hours of collection, where they were air-dried and sieved to 2 mm

prior to analysis. Prior to air-drying, a 10-g subsample from each 0–10 cm soil sample was analyzed for pH in a 2:1 water to soil slurry, and a separate 10-g subsample was oven-dried at 105°C to measure gravimetric soil moisture. Bulk density was calculated by subtracting the weight of coarse rock fragments from the mass of dried soil and dividing this by the volume of the soil corer. We also measured soil CH₄, N₂O, and CO₂ emissions from each sampling location in the Santa Fe National Forest in June 2024, and in the Carson National Forest in July 2025, using the methods described below.

Soil C Pools

Soil organic matter from surface soils (0–10 cm) was separated into three fractions: particulate organic matter (POM), mineral-associated organic matter (MAOM), and sand-associated organic matter (Püspök et al. 2022). To do this, 5-g of air-dried soil was oven-dried overnight at 60°C and was then dispersed in 25 mL of 1.65 g/cm³ sodium polytungstate (SPT) at 200 J/mL. The dispersed sample was centrifuged at 2500g for 60 minutes, and the floating POM fraction was aspirated and collected on a 0.45 µm glass fiber filter. The heavy fraction pellet was then washed with deionized (DI) water and passed through a 53-µm sieve to separate the sand (>53 µm) from the MAOM (<53 µm). The remaining sand fraction was rinsed with DI water and centrifuged at 2500g for 20 minutes before decanting the water. All fractions were oven-dried (60°C) for 24 hours before being weighed and finely ground.

Fractionated soil samples were analyzed for C amount and isotopic composition at the UC Riverside Facility for Isotope Ratio Mass Spectrometry (FIRMS; <https://ccb.ucr.edu/facilities/firms>) using an elemental analyzer (Costech Analytical Technologies) coupled to an isotope ratio mass spectrometer (Delta-V Advantage, Thermo Fisher Scientific). Peach leaf standards of known C content were used to calculate C amount in each sample, and United States Geological Survey (USGS) standards 40 and 41 were used to calculate the isotopic composition ($\delta^{13}\text{C}$) of each sample (Qi et al. 2003). We also analyzed $\delta^{13}\text{C}$ of different masses of peach leaves to account for C mass dependence in our isotope measurements. Using the total C data from the elemental analyzer, POC, MAOC, and sand C were calculated as the percent C multiplied by the mass of the fraction divided by the initial soil mass (5 g). Because sand-associated C accounted for less than 18% of total C, did not differ among treatments, and may not relate to persistence as much as POC and MAOC (Püspök et al. 2022), we do not present it here (Fig. S1).

To calculate total soil C stocks to 30 or 50 cm depth, each bulk soil sample was analyzed for C and N amount and isotopic composition at FIRMS, using the same protocols described above. To account for differences in soil bulk density that can hinder comparisons of soil C stocks, we used an equivalent soil mass approach to normalize elemental stocks to a consistent cumulative mineral soil mass per unit area (von Haden et al. 2020). One of the control sites from each forest was used as the reference bulk density for these calculations.

Soil Greenhouse Gas Emissions

Soil CH₄, N₂O, and CO₂ fluxes were measured along the same transects described above using a portable Fourier Transform Infrared Gas Analyzer (Gasmeter GT5000 Terra FTIR; Gasmeter Technologies Oy; Vantaa, Finland). Polyvinyl chloride (PVC) collars (20-cm diameter × 11-cm height) were hammered 3–5 cm into the ground at least 1 hour prior to the measurement. Because collars included grasses or forbs in their footprint, CO₂ fluxes represent plant and soil respiration, and CH₄ fluxes may be influenced by plants (Bhullar et al. 2013).

To measure trace gas fluxes, a chamber lid (approximately 6 L volume) with a 0.64-cm diameter vent was placed on top of each collar to create a gas-tight seal and headspace gas was recirculated through the analyzer for at least 5 minutes at a rate of 2 L/s. Before entering the instrument, gas in the sample loop passed through a desiccant to remove water vapor (Drierite Drying Tube, WA Hammond Drierite Co LTD, Xenia, OH, U.S.A.). During the incubation, the gas analyzer recorded concentrations of CH₄, CO₂, and N₂O every 20 seconds. After approximately 5 minutes, the chamber lid was removed, and ambient air was allowed to pass through the instrument for at least 2 minutes before measuring fluxes from the next collar. Fluxes were calculated as the linear change in trace gas concentrations over the course of the 5-minute incubation. Fluxes were considered as 0 if the coefficient of determination for the relationship between time and gas concentration was less than 0.7. Because only 7 out of the 144 measurements had detectable N₂O fluxes, and these fluxes were always less than 0.02 µg N m⁻² s⁻¹, we only report N₂O fluxes in the supplemental material (Fig. S2).

We also calculated the climate benefit of restoration by comparing the global warming potential of emitted CH₄ to the global warming potential of CO₂ sequestered as soil C. To do this, average CH₄ emissions were multiplied by the number of seconds in 1 year, by 16/12 to convert g C to g CH₄, and by the global warming potential of 30 CO₂ equivalents over a 100-year time frame (U.S. Environmental Protection Agency 2024). We then calculated the amount of POC or total C stored during the 3-year period post-restoration by subtracting the average C stock in control reaches from the average C stock in restored reaches. POC stocks were calculated using average bulk density from stream reaches in each forest multiplied by POC concentrations and soil depth (10 cm). The accumulated soil C stocks were then converted to CO₂ equivalents by multiplying by 44/12 because CO₂ equivalents are calculated based on the mass of CO₂ rather than the mass of C (U.S. Environmental Protection Agency 2024).

Statistical Analyses

We used linear mixed effects models (lme4 package in R; Bates et al. 2015) to assess differences in soil C pools, moisture, pH, CH₄ fluxes, and CO₂ fluxes between restored and control sites. Separate models were run for reaches restored with plug-and-ponds versus those restored with BDAs. For each model, either soil C, moisture, pH, CH₄ fluxes, or CO₂ fluxes were included as the dependent variable, and restoration status, distance from stream, and their interaction were included as fixed

independent variables. Reach was included as a random factor to account for experimental design. The “anova” function was used to calculate F values, Satterthwaite’s degrees of freedom, and statistical significance ($p < 0.05$; “lmerTest” package in R; Kuznetsova et al. 2017). Model residuals were visually inspected to assess for normality; CH_4 fluxes were log transformed to meet assumptions of normality.

Results

Soil C Pools and Isotopic Composition

POC was 2.8 times higher in surface soils restored with plug-and-ponds (62.5 ± 9.71 g C/kg) than in nearby control reaches (22.3 ± 2.34 g C/kg, Treatment: $F_{[1,66]} = 17.2$, $p < 0.001$; Fig. 2A). However, MAOC did not differ between plug-and-pond and control reaches ($F_{[1,2]} = 0.04$, $p = 0.85$; 21.8 ± 0.83 g C/kg; Fig. 2B). Unlike plug-and-ponds, BDAs did not consistently increase POC (Fig. 2C). However, there was a significant interaction between BDA restoration and distance from the stream ($F_{[1,2]} = 3.4$, $p = 0.04$), which likely reflected an increase in POC closer to the stream in restored reaches (10.2 ± 1.72 g C/kg at 1 m vs. 5.30 ± 0.76 g C/kg at 10 m) but not in control reaches (7.48 ± 1.24 g C/kg at 1 m vs. 6.81 ± 0.68 g C/kg at 10 m). Like plug-and-ponds, BDAs did not affect MAOC ($F_{[1,2]} = 5.8$, $p = 0.14$; 13.9 ± 0.79 g C/kg; Fig. 2D).

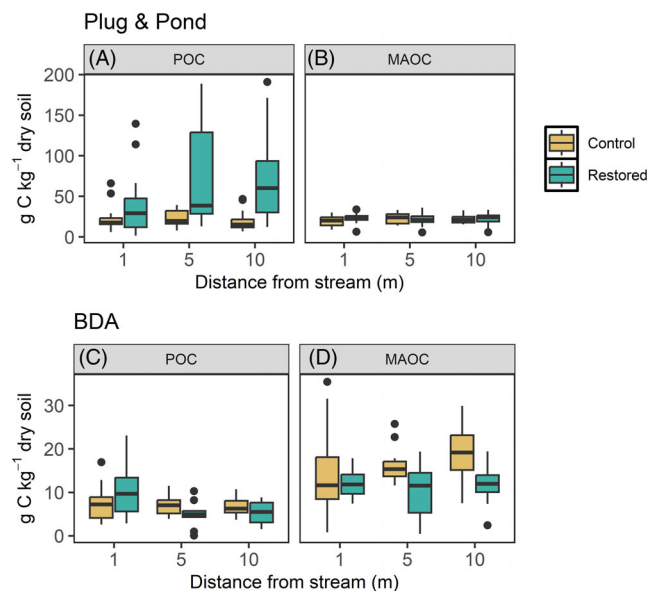


Figure 2. Soil carbon (C) in (A, C) particulate organic carbon (POC) and (B, D) mineral-associated organic carbon (MAOC) in sites restored with (A, B) plug-and-ponds or (C, D) beaver dam analogs (BDA) as a function of distance from the stream. Box color corresponds to restoration status (restored or control). Within each box, horizontal lines represent the median ($n = 12$), the upper and lower bounds of the box represent the interquartile range, vertical lines represent the largest and smallest values within 1.5 times the interquartile range, and solid black dots represent outliers.

POC was isotopically lighter (i.e. lower $\delta^{13}\text{C}$) in plug-and-ponds reaches ($-28.4 \pm 0.078\text{‰}$) than control reaches ($-27.9 \pm 0.11\text{‰}$; $F_{1,66} = 13.0$, $p < 0.001$; Fig. 3A). POC $\delta^{13}\text{C}$ also increased with distance from the stream in plug-and-pond and control reaches ($F_{[2,66]} = 4.44$, $p = 0.02$). While MAOC $\delta^{13}\text{C}$ did not differ between plug-and-pond and control reaches ($F_{[1,2]} = 0.53$, $p = 0.54$), it increased with distance from the stream ($F_{[2,64]} = 4.51$, $p = 0.015$; Fig. 3B). $\delta^{13}\text{C}$ in soils restored with BDAs had an opposite pattern than in soils restored with plug-and-ponds: it was higher in BDA reaches, especially further away from the stream (treatment by distance interaction: $F_{[2,63]} = 3.3$, $p = 0.04$; Fig. 3C). MAOC $\delta^{13}\text{C}$ also increased with distance from the stream ($F_{[2,64]} = 18$, $p < 0.001$) but did not differ between BDA and control reaches ($F_{[1,2]} = 2.23$, $p = 0.27$; Fig. 3D).

Neither plug-and-ponds ($F_{[1,2]} = 3.9$, $p = 0.19$; Fig. 4A & 4B) nor BDAs ($F_{[1,2]} = 9.11$, $p = 0.09$; Fig. 4C & 4D) increased total bulk soil C stocks. Reaches restored with plug-and-ponds averaged 159 ± 17.3 Mg-C/ha compared to 88.5 ± 4.7 Mg-C/ha in control reaches. Reaches restored with BDAs averaged 42.9 ± 3.54 Mg-C/ha compared to 70.2 ± 3.86 Mg-C/ha in control reaches. Total C stocks also did not vary with distance from stream bed with either restoration method ($p > 0.1$).

POC increased as total bulk C increased in surface soils from all reaches (Fig. 5A & 5C). In the plug-and-pond reaches, there was no consistent positive relationship between MAOC and

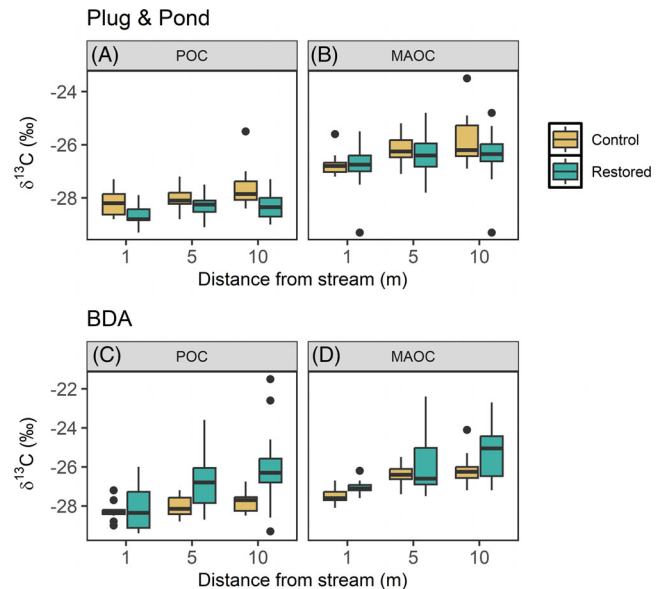


Figure 3. Stable isotopic composition soil Carbon ($\delta^{13}\text{C}$) in (A, C) particulate organic carbon (POC) and (B, D) mineral-associated organic carbon (MAOC) in sites restored with (A, B) plug-and-ponds or (C, D) beaver dam analogs (BDA) as a function of distance from the stream. Box color corresponds to restoration status (restored or control). Within each box, horizontal lines represent the median ($n = 12$), the upper and lower bounds of the box represent the interquartile range, vertical lines represent the largest and smallest values within 1.5 times the interquartile range, and solid black dots represent outliers.

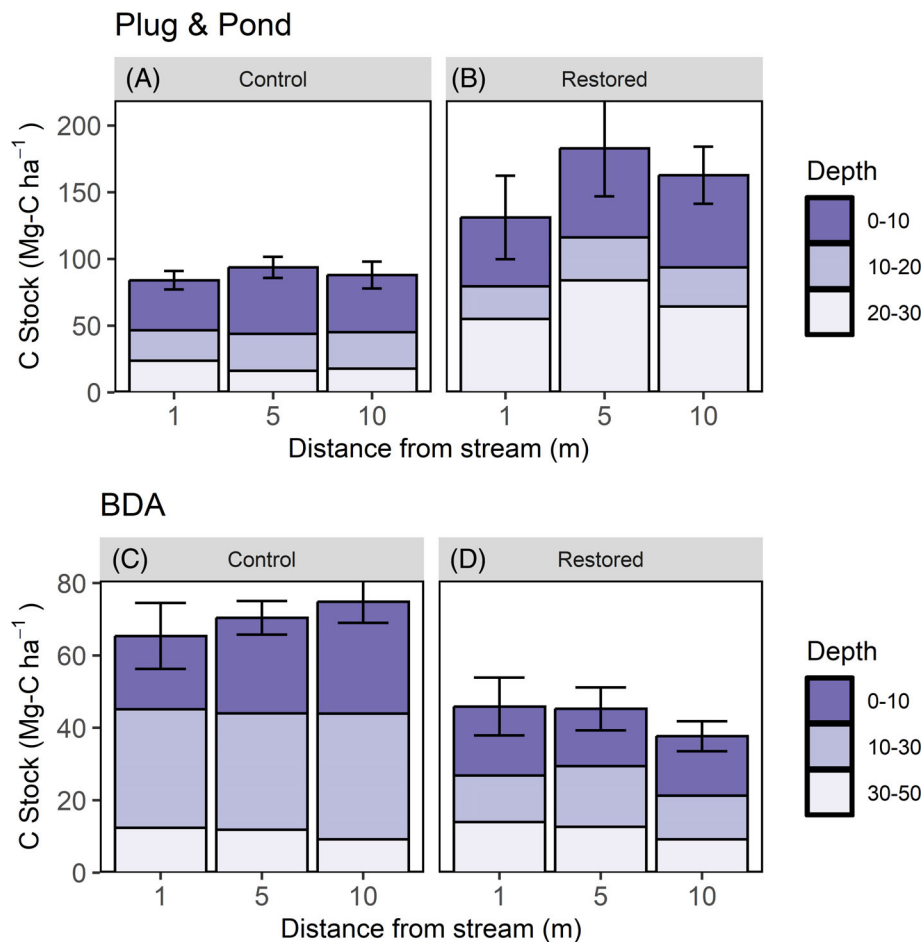


Figure 4. Total carbon (C) stocks in (A, C) control and (B, D) restored stream reaches treated with (A, B) plug-and-ponds or (C, D) beaver dam analogs (BDA). Carbon stocks were calculated using an equivalent soil mass method and depth increments are therefore estimated (von Haden et al. 2020). Bar height corresponds to mean carbon stocks ($n = 12$) and error bars correspond to ± 1 SE (calculated from total cumulative soil stocks, not individual depth increments).

total C (Fig. 5B). In the BDA reaches, there was a general positive relationship between bulk percent C and MAOC that reached a horizontal asymptote around 40 g C/kg when soils from the historic beaver dam complex were included (Fig. 5D).

Soil CH₄ and CO₂ Emissions

CH₄ emissions were higher in plug-and-pond reaches ($1.73 \pm 0.93 \mu\text{g C m}^{-2} \text{s}^{-1}$) than control reaches ($0.14 \pm 0.055 \mu\text{g C m}^{-2} \text{s}^{-1}$) and increased closer to the stream (treatment by distance interaction: $F_{[2,65]} = 8.52$, $p < 0.001$; Fig. 6A). CH₄ emissions reached $4.59 \pm 2.67 \mu\text{g C m}^{-2} \text{s}^{-1}$ in plug-and-pond reaches 1 m from the stream compared to $0.40 \pm 0.15 \mu\text{g C m}^{-2} \text{s}^{-1}$ in control reaches 1 m from the stream. In the BDA soils, CH₄ emissions also increased closer to the stream ($F_{[2,64]} = 12.6$, $p < 0.001$), reaching $0.89 \pm 0.25 \mu\text{g C m}^{-2} \text{s}^{-1}$ at 1 m. However, soil CH₄ emissions did not differ between BDA and control reaches ($F_{[1,2]} = 0.87$, $p = 0.45$; Fig. 6B).

Assuming summer CH₄ emissions remain consistent all year—an assumption that will likely lead to the overestimation of CH₄ emissions because cold winter temperatures should constrain CH₄ production (Li et al. 2024)—we calculated that plug-and-pond reaches emitted 0.67 Mg CH₄/ha more CH₄ than control reaches over a 1-year period. Because CH₄ has a global warming potential 30 times that of CO₂ over a 1-year period, we calculated that these CH₄ emissions offset the climate benefit of accumulating 20.1 Mg CO₂ ha⁻¹ yr⁻¹ in riparian soils. Over 3 years, plug-and-ponds increased POC in surface soils by 28 Mg C/ha, equivalent to 102 Mg CO₂/ha, meaning it would take approximately 5 years for CH₄ emissions to offset 3 years of soil POC gains in 0–10 cm soils. Over this same time, plug-and-ponds may have increased total C stocks to 50 cm depth by 121 Mg C/ha, equivalent to 444 Mg CO₂/ha, which would take approximately 22 years of CH₄ emissions to offset.

CO₂ emissions did not significantly differ between plug-and-pond reaches ($82.2 \pm 5.48 \mu\text{g C m}^{-2} \text{s}^{-1}$) and control reaches ($56.2 \pm 2.70 \mu\text{g C m}^{-2} \text{s}^{-1}$; $F_{[1,2]} = 14.5$, $p = 0.06$) and did not change with distance from the stream

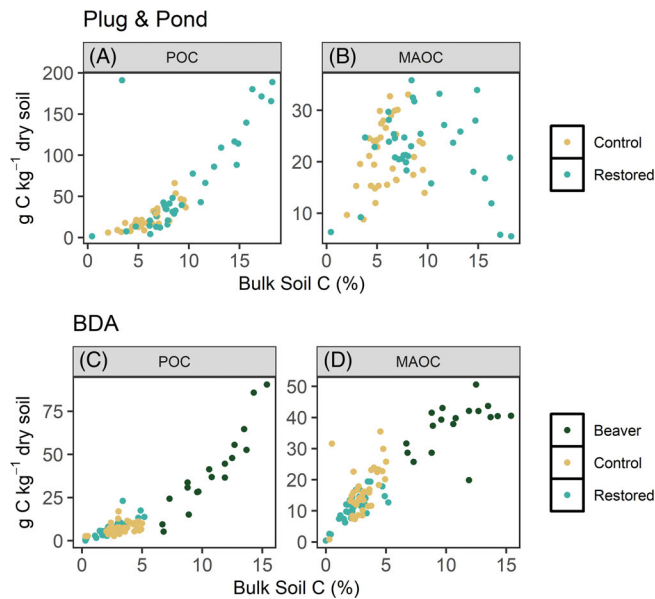


Figure 5. Relationship between total bulk soil carbon (C) percent and (A, C) particulate organic carbon (POC) or (B, D) mineral-associated organic carbon (MAOC) in sites restored with (A, B) plug-and-ponds or (C, D) beaver dam analogs (BDA). Dots represent individual measurements from surface soils (0–10 cm) and dot color corresponds to whether sites were restored, control, or from an abandoned beaver complex in the Santa Fe National Forest.

($F_{[2,64]} = 0.065$, $p = 0.53$; Fig. 6C). Like plug-and-ponds, BDAs also did not affect CO_2 emissions ($78.7 \pm 4.88 \mu\text{g C m}^{-2} \text{ s}^{-1}$ in restored versus $74.7 \pm 5.78 \mu\text{g C m}^{-2} \text{ s}^{-1}$ in control reaches; $F_{[1,2]} = 0.10$, $p = 0.78$) and emissions did not change with distance from the stream ($F_{[2,64]} = 0.52$, $p = 0.60$; Fig. 6D).

Soil Moisture, pH, and Slope

Soil moisture was higher in plug-and-pond reaches ($0.42 \pm 0.029 \text{ g}^{-1} \text{ H}_2\text{O g}^{-1} \text{ dry soil}$) than control reaches ($0.22 \pm 0.018 \text{ g}^{-1} \text{ H}_2\text{O g}^{-1} \text{ dry soil}$; $F_{[1,66]} = 36.8$, $p < 0.001$) and did not change with distance from the stream ($F_{[1,66]} = 1.09$, $p = 0.34$; Fig. 7A). BDAs did not have a significant effect on soil moisture ($F_{[1,2]} = 0.82$, $p = 0.46$) and moisture increased closer to the stream (Fig. 7B; $F_{[2,64]} = 36.2$, $p < 0.001$).

Soil pH was higher in plug-and-pond reaches (5.86 ± 0.088) than control reaches (5.60 ± 0.057 ; $F_{[1,66]} = 6.04$, $p = 0.017$) and did not change with distance from the stream ($F_{[2,66]} = 1.16$, $p = 0.32$; Fig. 7C). Unlike plug-and-ponds, BDAs did not have a significant effect on soil pH ($F_{[1,2]} = 9.20$, $p = 0.09$) and pH did not change with distance from the stream ($F_{[2,64]} = 0.25$, $p = 0.78$; Fig. 7D).

Using digital elevation models, we calculated that the ground was $1.00 \pm 0.13 \text{ m}$ higher than the stream 10 m from the stream in the Santa Fe National Forest, where the BDAs were installed. In the Carson National Forest, where the plug-and-ponds were

installed, the ground was $0.47 \pm 0.10 \text{ m}$ higher than the stream at 10 m from the stream.

Discussion

We monitored soil C pools and greenhouse gas emissions to assess the impact of stream restoration on riparian ecosystems in northern New Mexico. In support of our hypothesis—that restoration would increase soil moisture and suppress microbial decomposition, allowing soil C to accumulate as POC while also stimulating anaerobic processes that produce CH_4 —we found that POC and soil CH_4 emissions increased in stream reaches restored with plug-and-ponds compared to nearby control reaches. However, BDAs only increased POC within 1 m of the stream and did not affect CH_4 emissions, potentially due to steeper slopes preventing soil water saturation. Together, these results suggest that stream restoration can restore wetland C cycling, but that these changes depend on initial site characteristics.

Plug-and-ponds increased both POC and soil moisture by over 90% at all distances from the stream, suggesting that they saturated soils and suppressed microbial decomposition to allow POC to accumulate. However, CO_2 emissions, an index of decomposition, did not differ between restored and unrestored sites. Because our chambers included plant material, autotrophic respiration likely masked any decrease in CO_2 emissions in restored sites (Baggs 2006). Rather, the decrease in $\delta^{13}\text{C}$ in restored sites is consistent with plants making up a larger proportion of POC than microbial biomass, which is expected with decreased microbial decomposition (Ehleringer et al. 2000; Werth & Kuzyakov 2010; Klink et al. 2022). While changes in plant communities can also affect soil $\delta^{13}\text{C}$ if there is a shift in the relative abundance of plants that use C3 versus C4 photosynthesis (Krüger et al. 2023), there was no change in the relative abundance of C3 versus C4 plants at our sites (Driscoll, Rocky Mountain Research Station, 2025, personal communication). However, plug-and-ponds did increase the abundance of obligate wetland plants (Driscoll et al. 2025), which likely sustained plant-derived soil C inputs in water-saturated soils. Plant-derived soil C inputs could have also increased the export of dissolved organic carbon (DOC) in restored streams (Hallberg et al. 2026), although we did not measure DOC in this study. Despite the increase in POC, we did not detect a difference in total C stocks to 30 cm with restoration, likely due to our limited sample size coupled with high variation in total soil C stocks. Because POC comprised the majority of soil C at these sites, and POC was significantly higher in restored plug-and-pond sites, the increase in mean total C stocks was likely ecologically significant. As soil POC continues to accumulate in water-saturated soils, the accumulation of total C stocks in the plug-and-pond reaches may become more pronounced. Regardless, these results show that plug-and-ponds successfully slowed microbial decomposition and allowed POC to accumulate.

Unlike plug-and-ponds, BDAs only increased POC within 1 m of the stream. Because soils were generally wetter closer to the stream, a small increase in moisture here would likely be

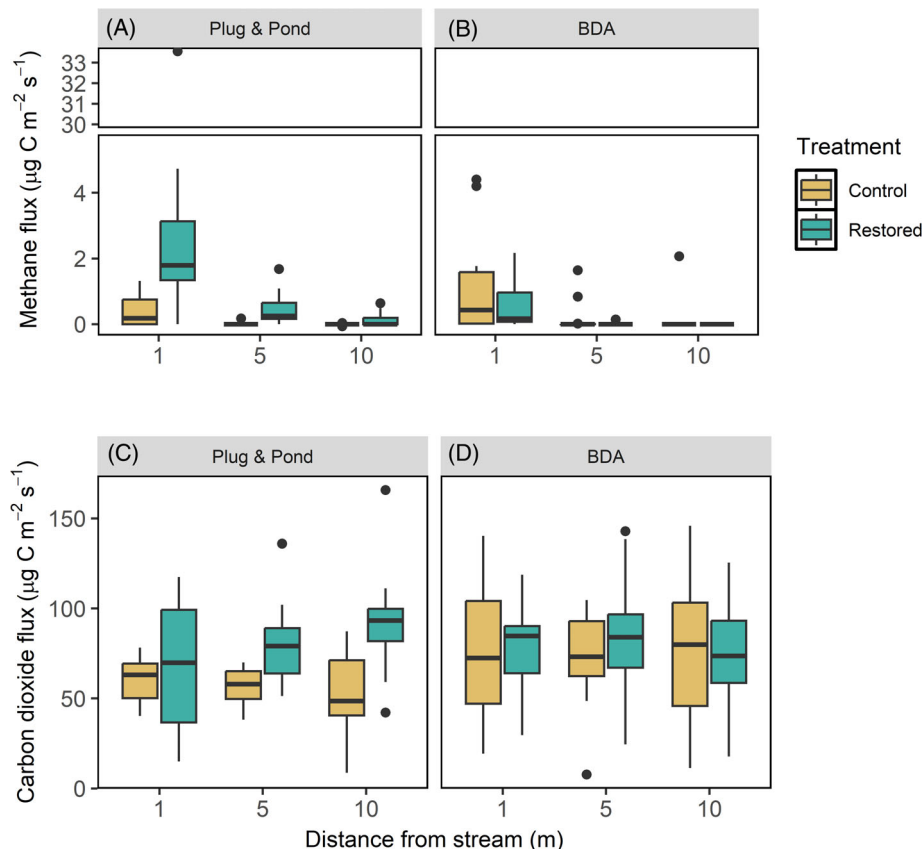


Figure 6. Net flux of (A, B) methane and (C, D) carbon dioxide from sites treated with (A, C) plug-and-ponds (BDA) or (B, D) beaver dam analogs as a function of distance from the stream. Box color corresponds to restoration status (restored or control). Within each box, horizontal lines represent the median ($n = 12$), the upper and lower bounds of the box represent the interquartile range, vertical lines represent the largest and smallest values within 1.5 times the interquartile range, and solid black dots represent outliers.

enough to generate suboxic conditions that allow C to accumulate (Estop-Aragónés et al. 2013; Jackson et al. 2017). While we did not measure an increase in moisture with BDAs, our measurements reflect one point in time and do not reflect daily or seasonal changes in soil moisture. Indeed, the greater presence of anaerobic-tolerant wetland plants in BDA reaches suggests that BDAs are increasing soil moisture (Driscoll et al. 2025). The increase in wetland plant species can also explain the increase in POC- $\delta^{13}\text{C}$ in BDA sites; C4 wetland plants *Blepharoneuron Tricholepsis*, *Bouteloua gracilis*, *Schizachyrium scoparium*, and *Muhlenbergia montana* increased in abundance (Driscoll, Rocky Mountain research Station, 2025, personal communication), which should produce isotopically heavier organic matter (Von Fischer et al. 2008). Given the change in plant community composition, it is surprising we did not observe corresponding changes in POC or bulk soil C. Perhaps, soil moisture increased just enough to favor wetland plants without suppressing microbial decomposition. Indeed, BDAs were built out of woody material that may be more permeable than the sediment used in the construction of plug-and-ponds, potentially limiting increases in riparian soil moisture. Drier soils at the BDA sites could also arise from slightly steeper riparian areas than in the plug-and-pond sites that can slow water

infiltration into surface soils (Pacific et al. 2011). Additionally, it may take longer than 3 years for BDAs to stimulate soil C accumulation, consistent with prior studies suggesting BDAs may not change soil C stocks within 10 years of implementation (Hinshaw & Wohl 2023). However, high POC and MAOC in a nearby beaver dam suggest that these riparian soils can support more C, especially if treatments promote the reestablishment of native beaver populations that may maintain more effective dams. These results should not necessarily be taken as evidence that plug-and-ponds are better at restoring riparian C stocks than BDAs, but rather that accounting for initial site characteristics and time since implementation is important in evaluating the success of riparian restoration efforts.

In addition to increasing soil POC, plug-and-ponds stimulated soil CH_4 emissions. To increase CH_4 emissions, plug-and-ponds must have saturated soils with water for enough time to establish highly reducing conditions that promote methanogenesis and also hinder microbial decomposition (Estop-Aragónés et al. 2013). Likewise, lower CH_4 emissions from BDA reaches, especially further from the stream, confirm that BDAs did not inundate riparian soils to the same degree as plug-and-ponds. We did not observe consistent N_2O emissions from restored or control reaches from either restoration method, likely

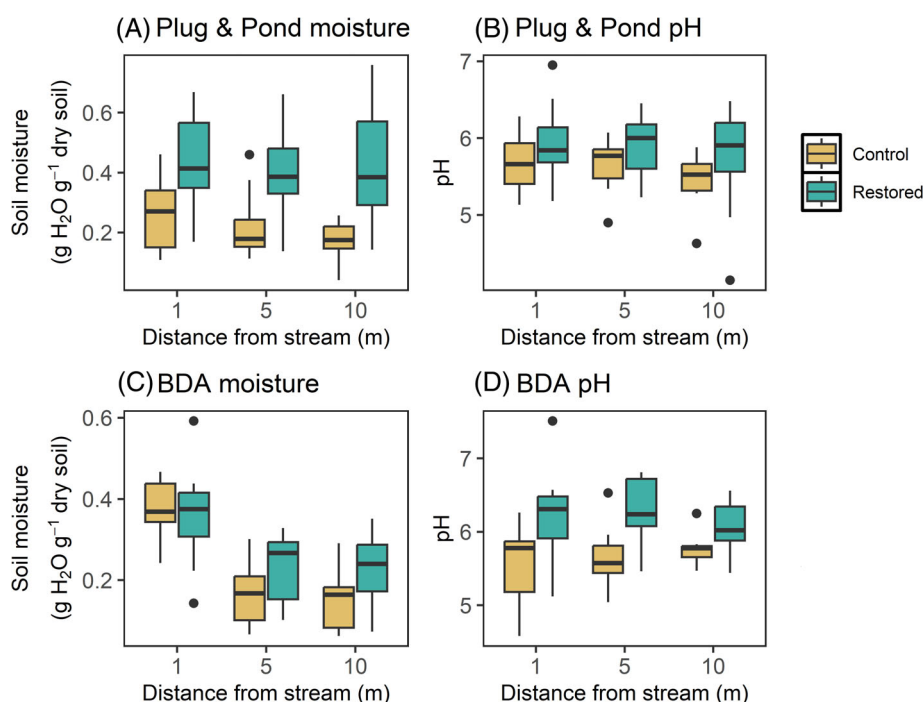


Figure 7. Soil (A, C) moisture and (B, D) pH from (A, B) plug-and-pond and (C, D) beaver dam analog (BDA) reaches. Box color corresponds to restoration status (restored or control). Within each box, horizontal lines represent the median ($n = 12$), the upper and lower bounds of the box represent the interquartile range, vertical lines represent the largest and smallest values within 1.5 times the interquartile range, and solid black dots represent outliers.

due to low soil N availability or highly reducing conditions that favored the reduction of N_2O to N_2 (Knowles 1982; Estop-Aragonés & Blodau 2012). While CH_4 emissions can offset climate benefits of increased C storage, they are also characteristic of wetland soils (Rosentreter et al. 2021), further supporting that the plug-and-ponds were successful in restoring wetland habitat. Additionally, we estimate that it would take at least 5 years for CH_4 emissions to offset the climate benefit of storing additional CO_2 as POC to 10 cm depth over a 3-year period in the plug-and-pond reaches, and at least 22 years to offset the total C stored to 50 cm depth. While using a single measurement of soil CH_4 flux to estimate annual CH_4 emissions is prone to errors, these estimates likely overestimate CH_4 emissions by applying summer rates—when CH_4 emissions should be highest (Li et al. 2024)—to annual budgets that include cold winters. Gains in soil C from plug-and-ponds are, therefore, unlikely to be offset by increases in soil CH_4 emissions, especially if soil C stocks continue to accrue over time.

All gains to soil C were observed as POC, and MAOC remained consistent regardless of restoration method. MAOC cycles more slowly than POC and may take more than 3 years to accumulate (Lavalée et al. 2020). Furthermore, MAOC is often comprised of microbial residues, which are slow to build up in wet soils that hinder microbial activity (Wieder et al. 2014; Lavalée et al. 2020). While soil mineral surfaces can become saturated with C to prevent increases in MAOC despite increases in total C, MAOC is unlikely to be saturated at our sites based on relationships between total C and MAOC. Specifically, MAOC increased linearly with bulk C at the BDA reaches, and when a nearby abandoned beaver dam complex was included, MAOC pools did not saturate until

approximately 40 g C/kg, greater than the highest MAOC levels we measured in BDA-restored sites. While the plug-and-pond sites had different soil types, MAOC also did not reach a clear horizontal asymptote when plotted against total percent C, suggesting that MAOC concentrations have not saturated (Georgiou et al. 2022). Rather, MAOC peaked at intermediate C concentrations, likely stemming from higher POC in restored sites that increased bulk C without affecting MAOC. Because MAOC did not saturate in BDA or plug-and-pond sites, it will take more time before accumulated POC can be converted to MAOC, limiting the stability of riparian soil C pools if streams are degraded in the future.

In conclusion, we found that restoring degraded streams with BDAs or plug-and-ponds can increase surface soil moisture, but initial site characteristics—such as slope—likely determine if restoration can establish anoxic conditions necessary for soil POC to rapidly accumulate post-treatment. Soil C gains were tightly coupled to increases in soil moisture, suggesting that monitoring surface soil moisture is an effective way to assess the success of riparian restoration efforts. While gains in soil C corresponded to increased soil CH_4 emissions, preliminary calculations suggest that these emissions do not offset the climate benefit of accumulating more C in soils. Restoring soil C sinks in wetland soil offers an opportunity for practitioners to promote natural climate solutions that curb the accumulation of carbon dioxide in the atmosphere.

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Supporting Information

The following information may be found in the online version of this article:

Figure S1. Soil C in (A, D) particulate organic carbon, (B, E) mineral-associated organic carbon, or sand organic carbon (C, F).

Figure S2. Net flux of nitrous oxide from sites treated with (A) BDAs or (B) plug-and-ponds as a function of distance from the stream.

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