



Decomposition dynamics of plant litter from *Salvia rosmarinus* grown in a contaminated soil and the potential release of toxic elements

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Abstract

Background and aims *Salvia rosmarinus* Spenn. (rosemary) was used in an off-site phytoremediation experiment conducted in contaminated soil from the Sierra Minera of La Unión-Cartagena mining area (Murcia, Spain). This study examined the potential reintegration in the soil of potentially toxic elements (PTEs) coming from plant litter degradation and their environmental impact, aiming to evaluate the efficiency of the remediation process.

Methods The sustainability of the phytoremediation approach used was assessed through the analysis of the decomposition dynamics of phytoremediation-derived biomass in the soil, and its role in the biogeochemical cycling of carbon (C) and PTEs. A 201-day incubation experiment was conducted, and both plant litter and soil were analyzed before and after the incubation to evaluate the changes in the DTPA-extractable fraction of the PTEs.

Results The results of C mineralization kinetics revealed a spectrum of microbial activity patterns, with CO₂-C emission dynamics ranging from zero–first order in the absence of litter, to first-order or two-step decay models in littered samples, depending on the biomass degradation stage. The decomposition process, which led to the mineralization of 21–50% of litter C, also influenced the cycling of PTEs, significantly decreasing the DTPA-extractable concentrations of copper (Cu, 15%), manganese (Mn, 50%) and lead (Pb, 11%).

Conclusions These findings revealed the role of organic matter age and decomposition stage in regulating C mineralization, and support the potential of phytoremediation as a sustainable ecosystem restoration method, enhancing microbial activity, promoting C sequestration, and reducing PTEs bioavailability in the soil.

Keywords C-mineralization · Mineralization kinetics · Phytoremediation · Rosemary · Potentially toxic elements

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Introduction

The carbon (C) cycle plays a crucial role in regulating Earth's climate, maintaining soil fertility (Sharma et al. 2021), supporting biodiversity (Sokol et al. 2022), and providing mechanisms for resilience against climate impacts (Wu et al. 2024). A

key aspect of this cycle is the decomposition of plant material (Hristovski et al. 2014; Lang et al. 2023), which releases C back into the atmosphere as carbon dioxide (CO₂) through soil microbial respiration (Yan et al. 2018). The interaction between organic matter from biomass degradation and micronutrients such as copper (Cu), iron (Fe), and manganese (Mn) is critical for the biogeochemical cycling of nutrients and the dynamics of C storage (Cao et al. 2018). These elements can significantly influence microbial communities involved in decomposition processes, often inhibiting microbial activity and altering nutrient release patterns (Gadd 2010; Huang et al. 2023).

Plant litter, a vital source of organic matter input into the soil, significantly influences the cycling of both nutrients and potentially toxic elements (PTEs). The decomposition of plant litter and the subsequent release of nutrients are crucial processes that affect soil fertility, microbial activity, and C fluxes (Hristovski et al. 2014; Lang et al. 2023; Sokol et al. 2022). The interaction between specific PTEs and nutrients with organic matter during decomposition can significantly influence the rate of biomass degradation. For instance, Martínez-Alcalá et al. (2016) demonstrated that mine soils can partially inhibit litter decomposition, as observed in the slower degradation of roots in soils with high PTEs concentrations compared to those in less contaminated environments. However, elements such as Al, Fe and Mn have been reported as modulators of biomass degradation processes. In fact, Fe and aluminum (Al) have been identified as rate-suppressing elements during the advanced stages of litter decomposition (Huang et al. 2023), while Mn bioavailability may enhance lignin degradation (Hristovski et al. 2014).

Human activities, such as mining can affect natural areas in vicinity. The Sierra Minera of La Unión-Cartagena is located in Murcia (SE Spain), very close to Calblanque, Monte de las Cenizas and Peña del Águila Regional Park. This mining area is currently the subject of a remediation program that seeks the generation of a permanent vegetation cover using species compatible with the nearby natural area, dominated by Mediterranean forest. Some notable species in this area include *Pinus halepensis* Mill., *Pistacia lentiscus* L., *Tetraclinis articulata* (Vahl) Mast., *Chamaerops humilis* L., *Olea europaea* subsp. *europaea* L., and *Quercus coccifera* L., along with various shrubs typical of the Mediterranean region, such

as *Salvia rosmarinus* Spenn. (Real Jardín Botánico, CSIC 2025). In ecosystems impacted by mining activities, phytoremediation plays a pivotal role in soil restoration. Certain plant species can stabilize or detoxify contaminated soils by accumulating PTEs and enhancing microbial activity, which, in turn, promotes organic matter decomposition and improves C flux dynamics (Lacalle et al. 2023; Yan et al. 2018). Previous studies have evaluated the restoration of mining-impacted soils through phytoremediation (phytostabilization), assessing recovery using various biological parameters (Alvarenga et al. 2018; Clemente et al. 2019; Pardo et al. 2014), including soil basal respiration (Lacalle et al. 2023). However, the sustainability of phytoremediation processes depends significantly on how post-phytoremediation biomass is managed (Mukherjee et al. 2025). Poor disposal of plant residues can alter soil microbial communities due to the risk of pollutants release in bioavailable forms during the degradation of fresh organic matter (Khan et al. 2023). The magnitude of these impact depends on the soil microbial biodiversity, the nature and composition of the litter accumulated on soil surface and the experimental conditions of the study.

Previous studies on leaf degradation have reported the reincorporation of PTEs (Cd and Zn) to the soil in an incubation experiment with *Arabidopsis halleri* (Boucher et al. 2005). Contaminated plant litter poses a potential risk of secondary contamination, as its mineralization process can lead to an increase in the soluble concentrations of PTEs in the soil (Cao et al. 2018). Despite these risks, decaying plant tissues can also enrich the soil with organic matter and nutrients, supporting microbial populations and contributing positively to soil remediation (Al Souki et al. 2020). In addition, Martínez-Alcalá et al. (2012) demonstrated that the degradation of plant roots does not significantly enhance the solubility of metal contaminants in neutral or calcareous soils, although it may slightly increase the soluble concentration of certain elements, such as Zn, in acidic soils. However, the rise in soil pH resulting from plant material degradation in acidic conditions can counteract this effect, effectively preventing increased metal solubility (Vázquez et al. 2008).

The aim of this study was therefore to assess the role of decomposing biomass from *S. rosmarinus* during the phytostabilization of soils contaminated by mining activities. Specifically, the long-term

decomposition dynamics of plant litter biomass and the potential release of PTEs from decomposing leaf litter to the soil were assessed through C mineralization and soluble PTEs determination in incubation experiments. The starting hypothesis was that the plant litter from *Salvia rosmarinus* grown in mine-contaminated soil during soil remediation enhances soil quality and microbial biomass activity promoting soil phytoremediation. Simultaneously, it is also hypothesized that the solubility of PTEs in the soil would not increase after plant litter degradation but would rather undergo an immobilization process. A deeper understanding of these processes is essential to develop effective soil recovery strategies and to ensure long-term ecosystem stability.

Materials and methods

Experimental setup and sample collection

A phytoremediation experiment was set under semi-real field conditions using large-volume (600 L) plastic containers filled with ca. 400 kg of PTEs contaminated soil (Lacalle et al. 2023). Soil affected by mining activity from the Sierra Minera of La Unión-Cartagena (Murcia, Spain; 37° 37' 22.3'' N, 0° 50' 21.8'' W), nearby the natural area of Calblanque, Monte de las Cenizas and Peña del Águila Regional Park (Fig. S1, Supplementary Information), was selected for this experiment (pseudo-total PTEs concentration: Fe 68 10³ mg kg⁻¹; Mn 3845 mg kg⁻¹; Cu 271 mg kg⁻¹; Zn 18575 mg kg⁻¹; As 46 mg kg⁻¹; Cd 20 mg kg⁻¹; Pb 1519 mg kg⁻¹). The soil from the surrounding area was classified as sandy loam, with a particle size distribution of 79.5% sand, 15% silt, and 5.2% clay. The mineralogical composition was dominated by gypsum, muscovite, and plumbojarosite (Arco-Lázaro et al. 2016). The soil contained less than 3% CaCO₃, with total P of 2.0 ± 0.1 g kg⁻¹, total N of 4.0 ± 0.04 g kg⁻¹, and total organic C of 35 ± 0.2 g kg⁻¹. Based on 2024 climatological data from two nearby meteorological stations (7012C and 7012D), the region experiences an average annual temperature of approximately 18.8 °C and an average annual precipitation of around 115 mm, primarily concentrated in spring and late summer to autumn months (AEMET 2025). This area is currently the subject of a remediation program that seeks the generation

of a permanent vegetation cover using species compatible with the nearby natural area. The soil used in this experiment was amended (5% w/w) with a mature compost (pH 6.8 ± 0.1; electrical conductivity 8.3 ± 0.1 dS m⁻¹; organic matter 82.6 ± 0.1%) made from separated solid fraction of pig slurry, to ensure plant survival and adequate development (amended soil pH 6.9 ± 0.04; electrical conductivity 2.7 ± 0.04 dS m⁻¹; organic matter 6 ± 0.1%). The harsh physicochemical and toxicological conditions of this soil make it entirely inhospitable, and no plants are able to grow in it without the appropriate soil condition (Lacalle et al. 2023). Plants of *S. rosmarinus* were grown in the amended soil for three years. After that, samples of plant litter naturally deposited on the soil surface were collected with the soil just underneath (Fig. 1). Eight samples were taken using a cylindrical structure (inner area 11.2 cm²; depth 2 cm), six of which corresponded to biological replicates representing different degradation states, and two control points of soil without leaf litter on the surface.

Dynamics of organic carbon mineralization in the soil

The mineralization of organic matter from leaf litter in the soil was assessed through a 201-day laboratory incubation experiment conducted under controlled temperature conditions in hermetically sealed glass containers (26 ± 0.5 °C; 500 mL capacity), as described in De la Fuente et al. (2013). A glass vial with NaOH was placed inside each container to trap the CO₂ released from organic-C decomposition. The vial with NaOH solution was replaced at established intervals and the excess of NaOH solution was titrated with an adequate HCl solution. The incubated samples included two control soils, labeled S1 and S2, each subdivided into three independent incubation units (n = 3 replicates each). In addition, six individual samples of leaf litter with underlying soil (L1–L6, biological replicates) were incubated separately and kinetic models were fitted individually to capture the variety of degradation patterns. Three blank containers without soil were also included. The plant litter of each sampling point was manually separated from the soil and characterized before setting the mineralization experiment. The amount of leaf litter (soil/plant litter ratio) was calculated based on the average weight of both materials in the different samples

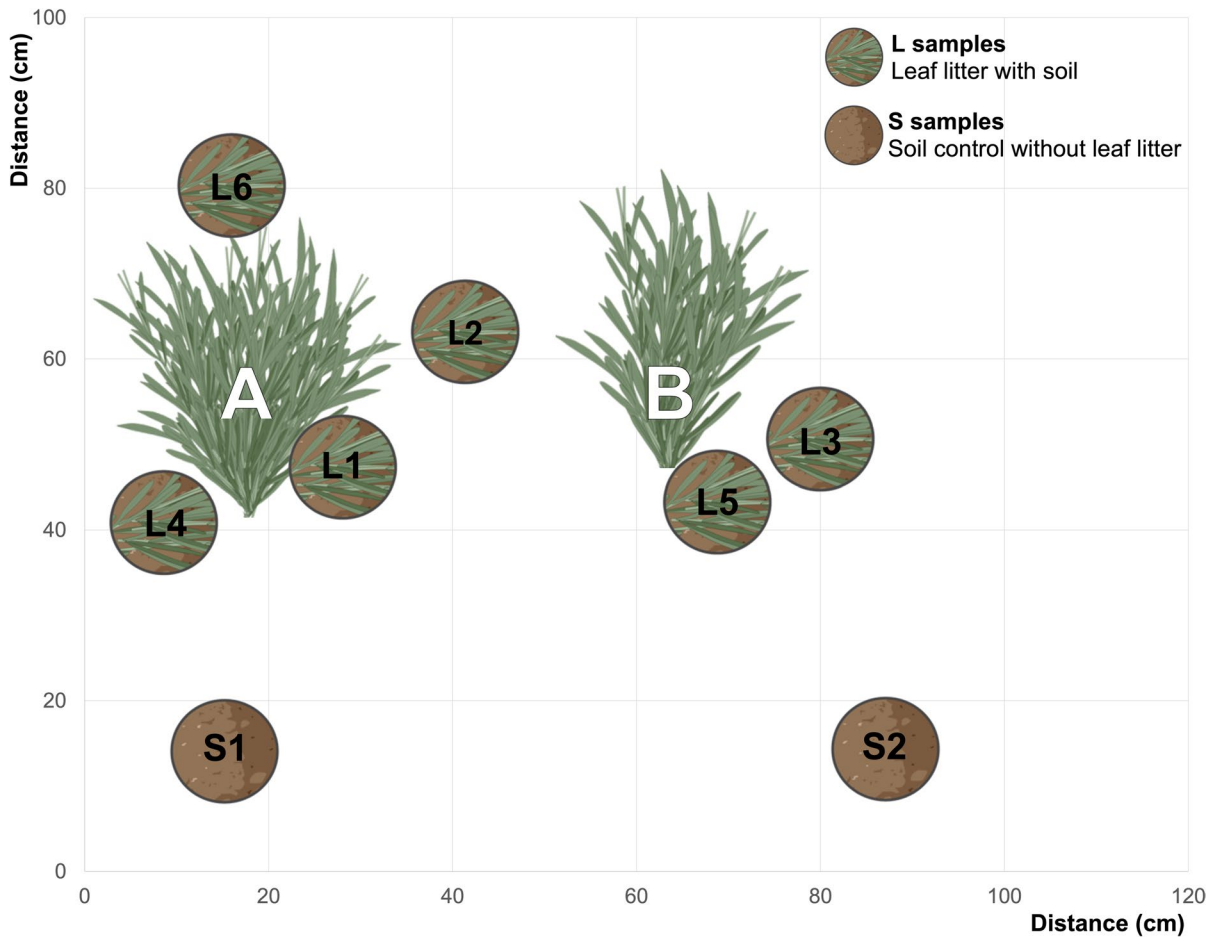


Fig. 1 Sampling points distributed around two different specimens of *S. rosmarinus*. Samples of soil with leaf litter (L1–L6) were taken at the points of greatest accumulation of litter, around two plants of different size (A and B). The samples vis-

ibly represented different states of degradation, according to their distribution around the plants. Soil control samples were taken outside the plant litter distribution area (S1 and S2)

collected (25/1.45 g). For the incubation experiment, the same amount (25 g) of each soil was placed in a glass container and 1.45 g of the corresponding plant litter was added on the soil surface to simulate real soil conditions (the same soil/plant ratio was used in each incubation sample). The samples were maintained at 60% of soil water holding capacity (WHC; 35%) throughout the experiment.

Cumulative $\text{CO}_2\text{-C}$ evolution (C_{CO_2}) was determined from the acid–base titration data by summing the measurements taken at consecutive sampling intervals, compared to the data obtained from the blank containers:

$$C_{\text{CO}_2} = \frac{[(V_b - V_s)MF]}{2} m_C$$

where V_b represents the volume of titrant used in the titration of the blank samples (average value), V_s represents the volume of titrant used in the titration of the S or L samples, M is the molarity of the titrant, F is the correction factor, and m_C is the molar mass of C (12 g mol^{-1}).

Carbon mineralization data from leaf litter (C_m) were calculated by subtracting the $\text{CO}_2\text{-C}$ generated in the S samples ($C_{\text{CO}_2\text{S}}$) from that in the L samples ($C_{\text{CO}_2\text{L}}$) and expressed as a percentage of the total

C content of the leaf litter ($C_{\text{Leaf litter}}$; previously analyzed).

$$\% C_m = \frac{[(C_{CO_2L} - C_{CO_2S})100]}{C_{\text{Leaf litter}}}$$

The dynamics of both C_{CO_2} and C_m were analyzed using nonlinear regression equations to derive various kinetic functions:

$$C_{CO_2} \text{ or } C_m = C_0(1 - e^{-kt}) \text{ First-order kinetic}$$

$$C_{CO_2} \text{ or } C_m = C_R + C_S(1 - e^{-k_S t}) \text{ Zero-first order kinetic}$$

$$C_{CO_2} \text{ or } C_m = C_R(1 - e^{-k_R t}) + C_S(1 - e^{-k_S t}) \text{ Two-step first-order kinetic}$$

where C_{CO_2} or C_m represents the CO_2 -C production or carbon mineralization at time t , C_0 indicates the potentially mineralizable C or CO_2 -C release, C_R and C_S indicate rapidly and slowly C_{CO_2} or C_m , respectively, and k , k_R and k_S are their respective rate constants. The half-life degradation time ($t_{1/2}$) of plant litter was calculated as:

$$t_{1/2} = \frac{-\ln(1/2)}{k_S \text{ or } k_R}$$

Analytical methods

Soil samples were collected at the beginning and at the end of the incubation experiment to determine the concentrations of extractable PTEs using a chelating solution containing 0.005 M DTPA, 0.01 M $CaCl_2$ and 0.1 M triethanolamine (pH 7.3; 1:5 w/v; 2 h). PTEs concentrations were analyzed by inductively coupled plasma-optical emission spectrometry (ICP-OES; ICAP 6500 DUO+ONE FAST, Thermo Scientific). Soil pH was also determined in the soil samples, in deionized (di) water suspensions (1:2.5, w/v). Samples were homogenized using a vortex mixer before being left to stand for 30 min and measured with a calibrated pH meter (Crison Basic 20 pH meter).

Plant litter was processed both at the start and the end of the incubation experiment, after careful separation from the soil surface. The samples were oven-dried (60 °C; 24 h), and ground to a fine powder using an analytical mixer mill (Retsch MM200). Then, the samples underwent acid digestion using a mixture of H_2O_2 (33% w/v) and concentrated HNO_3

(1:4, v/v) and a microwave digester (Milestone Ethos 1 MW digester) to determine pseudo-total concentrations of PTEs. Additionally, at the end of the experiment, plant litter samples were freshly sieved (1 mm mesh size) and homogenized using an analytical grinder (IKA A10). These samples were subjected to a di water extraction (1:20, w/v, 2 h) to determine their concentration of soluble PTEs. Element concentrations in both extracts (acid digestion and di-water) were determined by ICP-OES.

Total organic carbon (TOC) and total nitrogen (TN) in plant litter were determined at the beginning of the experiment using an automatic microanalyzer (EuroEA 3000, EuroVector). In addition, the source plants (SP; plants of *S. rosmarinus* growing in the sampled soil) were also processed following the described protocol for their elemental analysis. All the analyses were done at least in duplicate.

Statistical analysis

Graphical representation and Nonlinear Regression analysis of C mineralization dynamics were performed using SigmaPlot 15.0 (Systat Software, Inc., CA, USA, 2021). The statistical software IBM SPSS Statistics Version 29.0.2.0 (IBM Corp., NY, USA, 2022) was used for statistical analysis. The PTEs concentration of the leaf litter and their available soil concentrations (DTPA extractions) at the beginning and at the end of the incubation experiment were compared using paired-sample t-tests. Pearson's correlation coefficient between C mineralization parameters and C/N ratio of plant litter was also calculated.

Results

Cumulative CO_2 -C release in soil

The results of the soil incubation experiment demonstrated clear differences in microbial activity between control samples (soil-only, S1 and S2) and those containing leaf litter on the soil surface (L1-L6). Samples with litter exhibited significantly higher levels of released CO_2 -C throughout the incubation period compared to the control samples (Fig. 2). Cumulative CO_2 -C release reached values within the range 7.2–11.9 g C kg^{-1} soil in L samples, compared to values of 3.1–4.0 g C kg^{-1} soil in S samples (Table 1).

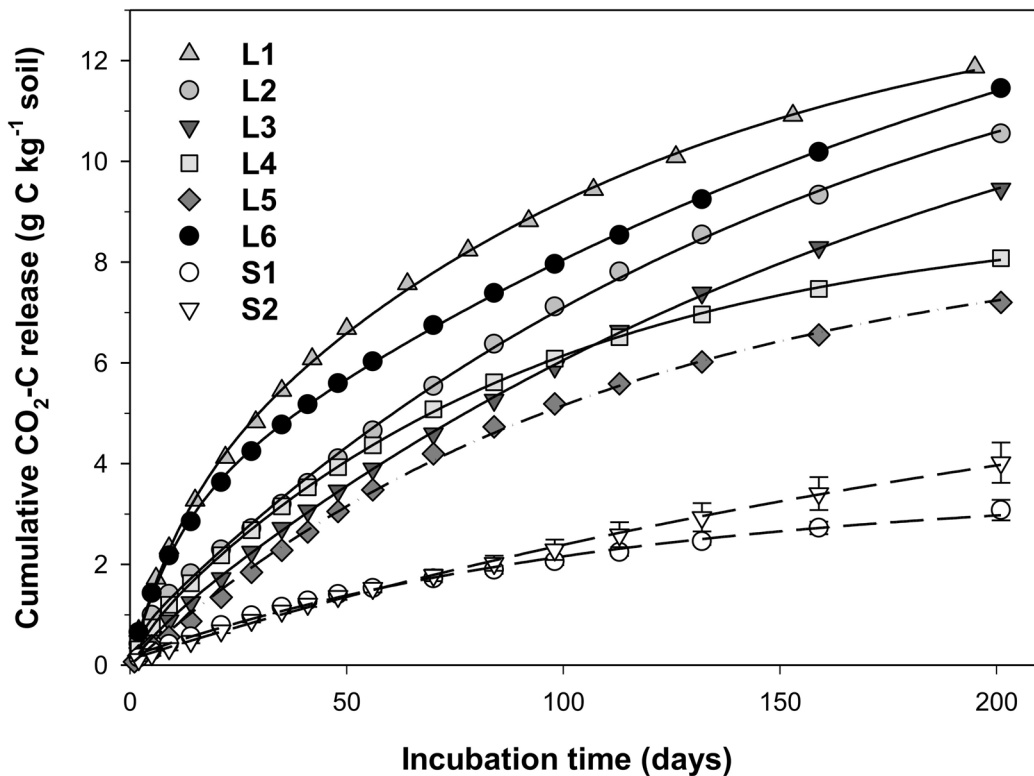


Fig. 2 Cumulative $\text{CO}_2\text{-C}$ release (g C kg^{-1} soil) in soil with leaf litter (L1-L6) and control soil (S1 and S2; mean $\pm$ SE, $n=3$), during the incubation experiment. The symbols are

experimental data and lines represent predicted data by the first-order kinetic (dash-dotted line), zero-first-order kinetic (dashed line), and two-steps first-order kinetic (solid line)

Table 1 Cumulative $\text{CO}_2\text{-C}$ (C_{CO_2} , g C kg^{-1} soil) release in soil during the incubation of leaf litter decomposition (L1-L6) and control soil (S1 and S2; mean $\pm$ SE, $n=3$), parameters of

the different kinetic models used to describe microbial activity, and the statistical significance of non-linear curve fitting

Sample	C_{CO_2}	Model	C_R	K_R	C_0 or C_S	k or k_S	RMS	F
L1	11.9	2FO	2.52	0.091	11.2	0.009	0.006	9912***
L2	10.5	2FO	0.54	0.461	14.0	0.006	0.010	4962***
L3	9.45	2FO	0.32	0.157	14.1	0.005	0.002	28,199***
L4	8.07	2FO	0.41	0.231	8.55	0.011	0.001	54,520***
L5	7.20	FO			8.65	0.009	0.008	11,769***
L6	11.5	2FO	2.73	0.094	14.3	0.005	0.002	24,820***
S1	3.07 ± 0.20	ZFO	0.17		3.36	0.009	0.004	1473***
S2	4.01 ± 0.40	ZFO	0.12		7.41	0.004	0.002	4588***

FO, First-order kinetic; ZFO, Zero-first-order kinetic; 2FO, Two-step first-order kinetic; RMS, Residual mean square; F, analysis of variance corrected value; C_R and C_S indicate rapid and slow released $\text{CO}_2\text{-C}$ concentration (g C kg^{-1} soil), respectively; k_R and k_S are the rate constants of the two different stages (day^{-1}) in the two-step model. *** Significant at probability level $P < 0.001$

The kinetic models used to describe microbial activity further underscored the differences between sample groups. In sample L5, where leaf litter was initially

more decomposed than in the rest of the samples (lowest TOC concentration; Table 2), C release was best described by a first-order kinetic model (Table 1).

Table 2 Elemental composition (g kg^{-1}) of the different leaf litter samples (L1-L6) and the source plant (SP) before the incubation experiment (mean $\pm$ SE; $n = 2$)

Sample	TOC	TN	C/N	P	K	Mg	Ca	Na
SP	488 ± 2	7.73 ± 0.27	63.4	2.21 ± 0.13	9.56 ± 0.29	2.72 ± 0.35	12.9 ± 2.00	0.84 ± 0.03
L1	458 ± 4	10.7 ± 0.19	42.8	1.97 ± 0.02	3.19 ± 0.06	3.45 ± 0.02	39.1 ± 2.18	0.83 ± 0.01
L2	266 ± 7	10.9 ± 0.12	24.4	2.28 ± 0.04	4.84 ± 0.31	4.66 ± 0.03	35.3 ± 0.29	1.32 ± 0.001
L3	203 ± 24	8.39 ± 0.89	24.2	1.36 ± 0.001	3.44 ± 0.08	3.31 ± 0.16	37.0 ± 3.59	1.23 ± 0.005
L4	423 ± 4	10.6 ± 0.68	39.9	1.78 ± 0.03	4.06 ± 0.02	5.80 ± 0.99	27.5 ± 0.18	2.41 ± 0.08
L5	167 ± 8	6.21 ± 0.07	26.9	1.63 ± 0.05	3.34 ± 0.03	3.83 ± 0.10	58.6 ± 1.98	1.19 ± 0.02
L6	441 ± 3	11.5 ± 0.28	38.3	2.02 ± 0.008	3.92 ± 0.04	4.41 ± 0.01	30.7 ± 0.27	1.40 ± 0.005

For the soil-litter mixtures L1-L4 and L6, a two-step first-order kinetic model provided the best fit for CO_2 -C release (Table 1). For the control samples, which contained only soil, a zero-first order kinetic model provided the best description of CO_2 -C release (Table 1), with very low C_R . The highest proportion of rapidly mineralizable-C against total mineralizable-C ($C_R + C_S$) was found in soil with L1 (18%) followed by L6 (16%); these samples also had the lowest litter degradation at sampling time, with final values of TOC concentration close to that in the source plant (Table 2). Sample L4 equally showed similar TOC concentrations to that in the source plant; however, the rapidly mineralizable-C in this sample accounted

unexpectedly only for 4.6% of total mineralizable-C. The rest of the samples showed similar proportions of C_R (with respect to $C_R + C_S$) to those in the soil without litter (<5%).

Mineralization of litter organic C

The differences found between the different litter (L) samples were further evidenced by the calculation of the mineralization of the TOC from leaf litter (Fig. 3) and the associated kinetic models (Table 3). Three ranges of TOC values were found (Table 2): L1, L6, L4 showed TOC values near that in the source plant ($\text{TOC} > 400 \text{ g kg}^{-1}$) and were collected close

Fig. 3 Percentage of C mineralization of the different samples of leaf litter during the incubation experiment. The symbols are experimental data and lines represent predicted data by the first-order kinetic (dash-dotted line), zero-first-order kinetic (dashed line), and two-steps first-order kinetic (solid line)

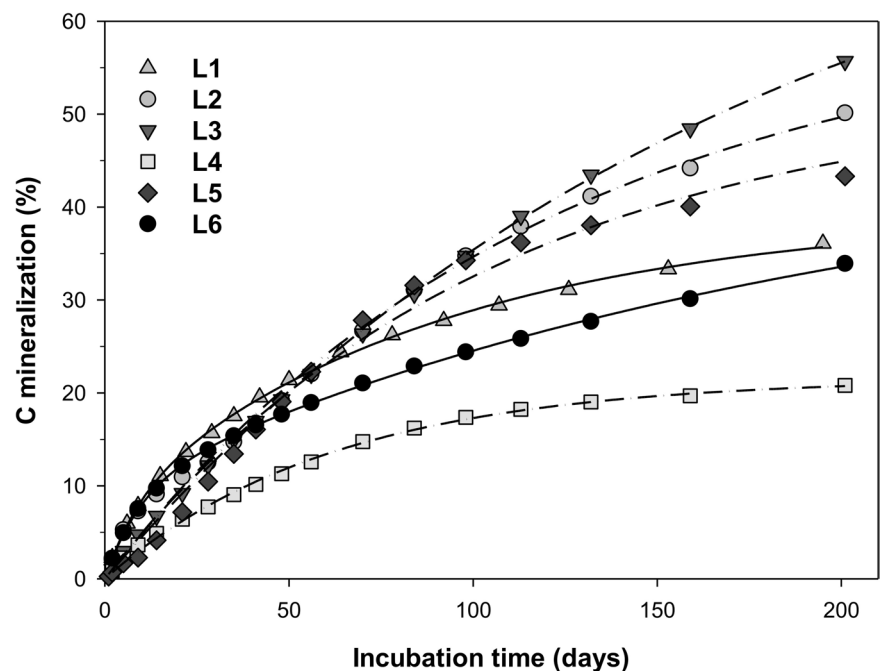


Table 3 Mineralized C (C_m) of initial TOC (%) leaf litter at the end of the experiment, parameters of the different kinetic models used to describe degradation dynamic, and the statistical significance of the non-linear curve fitting

Sample	C_m	Model	C_R	k_R	C_0 or C_S	k or k_S	RMS	F
L1	36.1	2FO	7.65	0.110	31.6	0.011	0.10	5025***
L2	50.1	FO			61.2	0.008	1.91	1826***
L3	55.7	FO			82.1	0.006	0.13	33774***
L4	20.8	FO			21.6	0.016	0.11	5916***
L5	43.3	FO			52.5	0.010	2.88	1345***
L6	33.9	2FO	9.66	0.101	37.9	0.005	0.04	11601***

FO, First-order kinetic; 2FO, Two-steps first-order kinetic; RMS, Residual Mean Square; F, analysis of variance corrected value; C_R and C_S indicate rapid and slow potentially mineralizable C concentration (mg C kg⁻¹ soil); k_R and k_S are the rate constants of the two different stages (day⁻¹) in the two-step model. *** Significant at probability level $P < 0.001$

to the largest plant (Specimen A); L2 (TOC=266 g kg⁻¹) was taken from an intermediate point; and L3, L5 (TOC=203, 167 g kg⁻¹) near the smallest plant (Specimen B). In contrast to the microbial activity results, TOC mineralization reached relatively low values of C degradation ($35 \pm 1\%$) in samples whose leaf cuticular structure was not significantly altered at the time of sampling (L1 and L6; Fig. 3). Although L4 had a comparable C content, it showed an unexpected low C mineralization, and its behavior was better described by a first-order kinetic model (Table 3). The remaining samples (L2, L3 and L5), also fitted to a first-order kinetic model and exhibited the highest TOC mineralization ($50 \pm 4\%$) (Fig. 3; Table 3). The initial TOC concentration in the different litter samples reflected diverse stages of leaf litter degradation before the incubation, related to two distinct ranges of C loss when compared to the source plant: $< 15\%$, represented by the fresh litter samples L1, L4, and L6; and $> 45\%$, observed in the samples L2, L3, and L5 (Table 2), which reflected a more advanced state of degradation.

The differences observed between samples characteristics align with potentially mineralizable C (C_0 or $C_R + C_S$ in one or two step models, respectively) values (Tables 2 and 3). Samples L2, L3, and L5 showed the highest C_0 . In those samples, when biomass lost prior to sampling is considered, the whole TOC mineralized (before sampling plus that after 201 days of incubation) reached 79–93% of TOC from SP (385, 452, and 409 g kg⁻¹ biomass, respectively), which is indicative of high labile organic matter content. In contrast, L1 and L6 had lower whole TOC mineralized values (210 and 257 g kg⁻¹ litter, representing 43% and 53% of TOC

from SP), consistent with their relatively low degradability (low $C_R + C_S$). The lowest value of TOC degradation was observed in L4 (whole TOC mineralized = 156 g kg⁻¹, 32% of TOC from SP), which indicates low degradability of this litter sample.

The substantial variability in the C/N ratio was also indicative of the degradation state of the leaf litter, showing a significant negative correlation with C mineralization (C_m) values ($r = -0.846$; $P < 0.05$). The source plant material exhibited elevated C and low nitrogen (N) concentrations and high C/N ratio (Table 2), much higher than those in the litter samples collected directly from the soil. Litter samples with the highest C/N ratio (38–43; L1, L4, and L6) were formed by slightly degraded material, compared to the source plant. In contrast, samples L2, L3 and L5 showed moderate-to-advanced degradation, with lower C/N ratios (24–27; Table 2).

The half-life degradation time ($t_{1/2}$) of the C pools further highlighted organic matter decomposition dynamics: the fast C pool (C_R) for L1 and L6 had a $t_{1/2}$ of 6–7 days, reflecting a rapid turnover of their labile fractions. Samples L3 and L6 had the highest $t_{1/2}$ values for C_S (124 and 139 days, respectively), which shows certain recalcitrant character of these materials. Intermediate $t_{1/2}$ for C_0 were found in L2 (84 days) and L5 (71 days), in line with their moderate to advanced decomposition states. L4 exhibited the shortest $t_{1/2}$ for C_0 (43 days), indicative of a relatively fast organic C mineralization.

PTEs release from plant litter decomposition

The PTEs concentration of both plant litter (pseudo-total concentrations) and soil samples

(DTPA-extractable) exhibited significant changes after the incubation period. Pseudo-total PTEs concentrations in the leaf litter declined significantly after incubation (Table 4), with concentrations of arsenic (As) falling below the detection limit, and maximum losses reaching up to 70% for Cu, 76% for Mn, 80% for Fe, 88% for lead (Pb), and 77% for zinc (Zn), which may indicate that those elements were released of into the soil during plant litter decomposition. Water solubility of the different leaf litter samples was also assessed after the incubation time (Table 5) as a measure of the potential release of PTEs into the soil. Manganese exhibited high variability among the different litter samples, with values reaching 9.1% and 5.6% of total Mn concentration in samples L2 and L4, respectively. However, most of the elements in the litter showed low solubility in water, indicating scarce potential to soil release.

Although the data of PTE concentration in the litter revealed that a significant proportion must be released into the soil, the soil samples showed significant decreases in extractable (DTPA) Cu, Mn, and Pb concentrations were observed at the end of the incubation period (Table 6). This pattern was observed for all the PTEs except for Cd, which showed no significant differences in either the plant litter or the soil analyzed in this experiment. This may suggest a certain degree of immobilization of the released elements within the soil components, rendering them in non-available forms to plants. In addition, the degradation of the plant litter did not affect soil pH, as the results at the beginning and end of the incubation period did not show any statistically significant differences (Table 6).

Discussion

Effects of litter decomposition on soil microbial activity

The presence of leaf litter enhanced soil microbial activity, likely due to the increased availability of easily degradable organic C substrates that promote the proliferation of the microbial community and, therefore, increase decomposition rate (De la Fuente et al. 2013; Hellequin et al. 2021). Kinetic fitting of the CO₂-C emission data further underscored the differences between soil with or without plant litter,

Table 4 Pseudo-total PTEs concentrations before (T_0) and after the incubation time (T_F) in *S. rosmarinus* leaf litter samples (L1-L6) and results of the t-test for paired samples performed

	Fe (mg kg ⁻¹)		Mn (mg kg ⁻¹)		Cu (mg kg ⁻¹)		Zn (mg kg ⁻¹)		As (mg kg ⁻¹)		Cd (mg kg ⁻¹)		Pb (mg kg ⁻¹)	
	T_0	T_F	T_0	T_F	T_0	T_F	T_0	T_F	T_0	T_F	T_0	T_F	T_0	T_F
L1	6724 ± 113	3557 ± 5	947 ± 31	696 ± 6	96 ± 3	37 ± 1	5518 ± 114	2554 ± 16	3.33 ± 0.07	BDL	5.30 ± 0.49	5.36 ± 0.64	304 ± 14	74 ± 2
L2	27,962 ± 41	5584 ± 51	1904 ± 39	483 ± 7	226 ± 4	67 ± 1	14,847 ± 243	3493 ± 17	15.3 ± 0.47	BDL	9.07 ± 0.04	5.84 ± 0.07	1213 ± 26	147 ± 3
L3	22,967 ± 4440	6556 ± 2	2690 ± 566	648 ± 4	164 ± 1	62.7 ± 1	9964 ± 152	4503 ± 7	9.2 ± 0.86	1.17 ± 0.37	7.21 ± 0.27	10.0 ± 0.05	600 ± 70	151 ± 1
L4	30,031 ± 3253	10,088 ± 201	2723 ± 1262	855 ± 15	164 ± 8	76.5 ± 1	8952 ± 49	5079 ± 93	15.7 ± 3.16	5.30 ± 0.69	5.97 ± 0.16	17.7 ± 0.27	1017 ± 388	226 ± 2
L5	30,776 ± 1646	8060 ± ND	1543 ± 218	593 ± ND	192 ± 1	79.7 ± ND	10,905 ± 145	5524 ± ND	11.7 ± 0.54	BDL	6.13 ± 0.47	9.90 ± ND	643 ± 47	175 ± ND
L6	10,158 ± 267	6043 ± 88	547 ± 1	522 ± 3	128 ± 3	65.4 ± 1	6652 ± 175	3552 ± 36	9.60 ± 0.63	1.40 ± 1.40	4.16 ± 0.02	7.18 ± 0.03	239 ± 1	132 ± 4
t-test Δ	14,788		1093		97.3		5355		9.50		-3.03		51,953	
p-value	**		*		***		***		***		NS		**	

*, **, ***, significant at $P < 0.05$, 0.01 and 0.001, respectively. NS: not significant. ND: no data; analytical replicates were unavailable due to insufficient sample material. BDL: below detection limit (< 0.01 mg kg⁻¹). Δ: initial – final value. Values are mean ± standard error (SE) (n = 2 analytical replicates)

Table 5 Water solubility of pseudo-total PTEs concentrations (% of total concentration) in the different samples of *S. rosmarinus* leaf litter (L1–L6) after the incubation time

	Water solubility (% of total concentration)					
	Fe	Mn	Cu	Zn	Cd	Pb
L1	0.19	0.25	1.77	0.50	1.42	0.19
L2	0.14	9.07	3.27	3.03	4.17	0.08
L3	0.12	1.84	0.75	0.46	1.10	0.25
L4	0.10	5.58	1.32	0.81	1.98	0.13
L5	0.05	1.63	0.46	0.36	1.13	0.16
L6	0.32	1.83	0.83	0.41	1.38	0.18

which followed either a first-order or a two-step first-order kinetic model in soil with litter, unlike the soil samples without litter, that fitted to a zero-first order kinetic model. This suggests the absence or scarcity of readily degradable C sources present in the soil. When the soil had aged litter samples, a greater and uniform rate of $\text{CO}_2\text{-C}$ was released over time (first-order kinetic process), likely reflecting the predominance of homogeneous organic compounds, which are decomposed at a steady rate. Contrastingly, the dynamics of fresher material indicated the presence of two distinct C pools: one rapidly metabolized by soil microorganisms (labile C) described by C_R and k_R , and another one with higher resistance to degradation (recalcitrant C) described by C_S and k_S (Saviozzi et al. 2014; Wang et al. 2022), which corresponds to the gradual mineralization of stable organic matter (Berg 2014).

The TOC concentration of the litter revealed varying degrees of leaf litter degradation at the beginning of the study, which corresponded to the location of the sampling points (Fig. 1). Litter with high TOC concentration of the litter exhibited low C-mineralization, following two different degradation dynamic models (first-order or two-steps first-order), consistent with the decomposition state of the biomass. These results can be attributed to the presence of a cuticular wax in fresher samples, which—owing to its distinctive physicochemical properties and complex interactions with microbial communities (Ziv et al. 2018)—has been shown to inhibit the decomposition rate by 24–27% in sorghum biomass (Xu et al. 2024). Field experimental degradation of rosemary was studied by Almagro and Martínez-Mena (2012), whose results showed high C degradation (20–30%) after 6 months of plant decomposition in two different soils following a single exponential decay model, in good agreement with our results. However, these

values are lower than these found in the present experiment. Under field conditions, a lower mineralization rate could occur due to suboptimal conditions for microbial degradation compared to those observed under controlled experimental conditions. A similar kinetic pattern was observed during a 56-day incubation experiment that studied the mineralization of *S. rosmarinus* roots, where up to 13% of root C was mineralized by the end of the experiment (Moreno-Jiménez et al. 2009).

Bai et al. (2024) observed that litter with elevated C/N ratios slowed down microbial growth and resulted in a temporary delay in the degradation of all litter components in *Populus davidiana*, as evidenced in the material exhibiting lower levels of degradation in our experiment. In contrast, lower C/N ratios are indicative of greater C loss and N incorporation from plant litter to the soil (Martínez-García et al. 2021), which is consistent with its advanced degradation state (Zhang et al. 2021). Nitrogen is an essential nutrient and, therefore, when its concentration is low, both microbial activity (Craig et al. 2022) and plant litter decomposition are limited (Duan et al. 2018). During the early stages of decomposition, organic-N can boost microbial activity, leading to a fast breakdown of organic matter. However, as decomposition advances to later stages, lower C/N values affect microbial metabolism, which slow down the degradation process (Huang et al. 2023). In this context, the role of Ca is also noteworthy, as its concentration in biomass increases markedly due to direct inputs from soil to litter. In fact, Ca concentration rose three- to fourfold from the source plant to the litter (Table 2). Lovett et al. (2016) conducted a six-year litter decomposition experiment that showed how Ca strongly influences the extent of litter decomposition, and it is the concentration at the incubation site rather than the initial litter Ca that plays the most critical role.

Table 6 DTPA-extractable metal concentrations and pH before (T_0) and after the experiment (T_F) in soil samples incubated with *S. rosmarinus* leaf litter (L1–L6) and control samples (S1 and S2), and results of the t-test for paired samples performed

	pH		Fe (mg kg ⁻¹)		Mn (mg kg ⁻¹)		Cu (mg kg ⁻¹)		Zn (mg kg ⁻¹)		Cd (mg kg ⁻¹)		Pb (mg kg ⁻¹)	
	T_0	T_F	T_0	T_F	T_0	T_F	T_0	T_F	T_0	T_F	T_0	T_F	T_0	T_F
L1	6.67 ± ND	6.73 ± 0.02	5.70 ± 0.02	6.44 ± 0.07	7.33 ± 0.38	3.82 ± 0.61	6.62 ± 0.002	6.20 ± 0.07	1304 ± 15	1325 ± 14	4.88 ± 0.05	4.98 ± 0.17	9.24 ± 0.02	7.71 ± 0.41
L2	7.38 ± 0.07	7.20 ± 0.02	5.21 ± 0.01	5.26 ± 0.04	6.41 ± 0.40	4.64 ± 0.14	11.49 ± 0.01	10.51 ± 0.06	1179 ± 2	1216 ± 15	4.14 ± 0.08	4.43 ± 0.13	9.32 ± 0.41	8.18 ± 0.19
L3	6.57 ± 0.09	6.40 ± 0.05	5.98 ± 0.03	7.02 ± 0.08	33.2 ± 0.38	9.11 ± 0.06	6.77 ± 0.02	6.20 ± 0.13	1315 ± 10	1368 ± 6	4.77 ± 0.04	5.11 ± 0.002	9.02 ± 0.08	8.25 ± 0.30
L4	6.36 ± 0.01	6.43 ± 0.02	6.32 ± 0.28	5.54 ± 0.09	40.9 ± 0.78	28.3 ± 2.11	7.68 ± 0.09	5.67 ± 0.02	1355 ± 96	1223 ± 47	4.82 ± 0.17	4.05 ± 0.24	7.94 ± 0.07	7.75 ± 0.63
L5	6.61 ± 0.07	6.59 ± 0.01	5.57 ± 0.31	5.32 ± 0.21	5.19 ± 0.01	2.63 ± 0.46	6.66 ± 0.20	5.89 ± 0.04	1315 ± 14	1253 ± 1	4.71 ± 0.06	4.46 ± 0.18	8.88 ± 0.02	8.62 ± 0.48
L6	6.75 ± ND	6.81 ± 0.01	5.05 ± 0.21	4.02 ± 0.04	12.82 ± 1.17	4.63 ± 0.97	9.29 ± 0.30	6.92 ± 0.35	1241 ± 41	1120 ± 7	4.59 ± 0.27	4.21 ± 0.05	8.65 ± 0.16	6.70 ± 0.21
t-test Δ	0.07		0.04		8.79		1.19		33.8		0.11		0.978	
p-value	NS		NS		*		**		NS		NS		**	
S1	7.51 ± 0.01	7.44 ± 0.01	5.83 ± 0.11	4.91 ± 0.19	8.66 ± 0.14	8.12 ± 1.28	13.9 ± 0.19	13.2 ± 0.45	1330 ± 30	1386 ± 42	4.38 ± 0.01	4.90 ± 0.30	9.84 ± 0.11	8.09 ± 0.26
S2	7.42 ± ND	7.07 ± 0.11	7.32 ± 0.16	7.67 ± 0.41	7.30 ± 0.01	3.23 ± 0.30	8.99 ± 0.16	8.01 ± 0.10	1365 ± 23	1294 ± 21	4.88 ± 0.18	4.67 ± 0.09	8.71 ± 0.29	7.19 ± 0.08
t-test Δ	0.20		0.29		2.30		0.83		7.14		-0.15		1.63	
p-value	NS		NS		NS		NS		NS		NS		*	

*, **, significant at $P < 0.05$ and 0.01, respectively. NS: not significant. Δ: initial – final value. Values are mean ± standard error (SE) for both leaf litter samples ($n = 2$ analytical replicates) and control soils ($n = 3$ biological replicates per control)

Notably, high Mn concentrations were observed in L4 in both plant and soil samples, before and after the incubation period (Tables 4, 6), what could be related to the outlier $t_{1/2}$ observed in L4. This outlier may be attributed to its proximity to the soil edges, where microenvironmental variations, such as increased evaporation or temperature fluctuations, might have also interfered in soil microbial community. Furthermore, Mn concentration in plant litter has been previously associated to litter mass loss, suggesting that this element may play a role in accelerating decomposition rates (Hristovski et al. 2014). This relationship is likely attributable to the role of Mn in the production of manganese peroxidase (MnP), an enzyme commonly implicated in the breakdown of recalcitrant C compounds (Keiluweit et al. 2015; Wu et al. 2022).

Changes in PTEs extractability after litter decomposition

Over the course of the six-month study, up to 50% of the plant litter was decomposed, with a concomitant reduction in PTEs in the plant litter, which indicated a significant release into the environment, especially for As, Cu, Mn, Fe, Pb, and Zn. However, the concentration of plant bioavailable fraction of PTEs in the soil exhibited a significant reduction during the experiment, without significant changes in soil pH. This may be a consequence of the interaction between the elements and soil components, primarily through adsorption onto soil minerals—particularly iron oxides (such as hercynite)—which are known to play a key role in the fixation of heavy metals and As in soil (Arco-Lázaro et al. 2016; Li et al. 2022). Clay minerals, including silicate minerals, crystalline oxides, and amorphous oxides, are among the most active soil constituents. Arco-Lázaro et al. (2016) observed that the presence of various silicates and Fe oxides can endow the soil with a high capacity for the fixation of PTEs. In any case, the observed decrease in most of the PTEs concentrations in leaf litter after the incubation suggests that all those elements, including those with low water solubility (Cu, Fe, Pb, Zn; Table 5), had been gradually released from the biomass during decomposition. Once incorporated into the soil, they were likely retained in hardly soluble forms within the soil matrix, such as adsorbed onto soil particles or bound to organic matter

(Kwiatkowska-Malina 2018). In fact, DTPA-extractable concentrations of all PTEs in the soil either decreased or did not significantly increase (Table 6), which supports the idea of the immobilization of the released elements in the soil, either through microbial (Pande et al. 2022) or binding to soil particles (Li et al. 2022) processes, independently of the potential increase in their pseudo-total concentration in the soil. This pattern was observed for all the PTEs except for Cd, which showed no significant differences in either the plant litter or the soil analyzed in this experiment. Similar results regarding Cd concentration in plant litter decomposition have been previously reported for *Fagus sylvatica* L. (Hristovski et al. 2014) and mangrove plant species (Lang et al. 2023). Leaf litter from different species has been found to act as a source of nutrients and some PTEs during decomposition in long term (140–180 days) field degradation experiments (Cao et al. 2018; Lang et al. 2023). In an ex situ experiment studying Poaceae plants leaf decomposition, no significant changes in Cd CaCl_2 -extractable concentrations during the incubation period were found (Al Souki et al. 2020), in line with our results. In that experiment, a significant increase in CaCl_2 -extractable Zn and Pb concentrations that tended to decrease towards the end of the experiment was also reported. Further research is needed to deepen our understanding of litter decomposition process, considering the origin and condition of the biomass, the soil characteristics, and the nature of the extractant, all of which may critically influence the results.

Conclusions

The findings of this study highlight the dynamic interactions between PTEs and microbial processes during the decomposition of phytoremediation biomass. Carbon substrate availability in the plant litter regulated microbial activity, as shown by distinct mineralization patterns: partially degraded litter fit a first-order kinetic model, while fresher litter followed a two-step first-order model, characteristic of the initial stages of microbial colonization.

This process also affects PTEs cycling, as the PTEs released from plant litter did not increase their bioavailable fraction in the soil. On the contrary, a decrease in the DTPA-extractability of Mn, Cu, and

Pb was observed, likely due to interactions between the released elements and the soil matrix, which promoted their stabilization in non-bioavailable forms. These results support the potential of phytoremediation technologies as a sustainable approach to enhance soil microbial activity, promote soil organic-C sequestration, and reduce the bioavailability of PTEs across all phases of the plant biological cycle.

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Data availability The datasets generated during and/or analyzed during the current study are available from the corresponding author on reasonable request.

Declarations

Competing interests The authors declare no competing interests.

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