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Dynamics of heavy metals during litter decomposition in fire-affected boreal forests

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Abstract

Purpose Russian boreal forests represent a globally significant carbon stock, been suffering from frequent surface fires that modify natural cycles of elements, including heavy metal (HM). The behaviour of HM, exerting various ecosystem effects, is not well understood, especially in northern larch forest ecosystems affected by fires.

Methods The dynamic of Fe, Pb, Mn, Zn, Cu, Co, Ni and Cr was studied in the 850-day field decomposition experiment in a natural unburned larch stand (*Larix gmelinii* (Rupr.)) and adjacent burned forest on the Russian Far East. We observed mass loss, HM release/accumulation and correlation of HMs with soil properties.

Results The litter decomposed slower in the burned site, with pronounced differences in the late decomposition stage. The concentrations of HMs except Mn had increased by the end of the experiment in both forest sites. Among all the HMs, Fe, Cr and Ni showed pronounced accumulation in burned stand compare to the unburned forest. Fire does not modify the patterns of HM release/accumulation but significantly alters the final values. In unburned forest, soil pH and water content strongly influenced only Fe dynamics, whereas, on burned site, soil properties correlate with the group of HMs.

Conclusion Our experiment showed that HM dynamics are coupled with the mass loss only in the late stages of litter decomposition. We found that fire's legacy effect in natural larch forests could last over 15 years, creating favourable conditions for significant accumulation of Fe, Pb, Cr and Ni.

Keywords Litter decomposition · Heavy metals · Forest fire · Gmelin larch · Boreal forest

1 Introduction

Forest litter decomposition is an essential part of the biogeochemical cycles and recognized as a two-phased process of mass loss, accumulation and release of elements. Decomposition is mainly mediated by plant traits, litter chemistry and environmental factors in forests vastly changed by human activity (Cornwell et al. 2008). Among other influences, frequent fires represent repeating forest disturbance acting as a powerful, long-lasting factor of ecosystem processes, including decomposition (Brennan et al. 2009). If the dynamics of major elements in burned forests are well understood, then heavy metal (HM) behaviour during decomposition in fire-affected forests has been largely overlooked.

HMs are a group of elements participating in metabolic and redox processes, biologically essential in small concentrations (Lenart-Boron and Boron 2014). However, high concentrations of the HMs have ambiguous ecosystem effects, including suppressing the soil microbes (Laskowski and Berg 1993), mycelia and soil respiration (Rühling et al. 1973). These elements attract scientific attention due to their ability to form highly resistant complexes with organic matter and the tendency to accumulate in forest litter and upper soil horizons (Berg and McClaugherty 2014a, b; Laskowski and Berg 1993; Tyler 2005). For example, Fe creates a stronger bond with humic acids in a high pH, but migration Cr is hindered in acidic conditions (Boguta et al. 2019; Catrouillet et al. 2014; Vodyanitskii 2008; Yue et al. 2019). Therefore, depending on environmental conditions, HMs might suppress the decomposition with implications to ecosystem functioning (Berg and Ekbohm 1991; Hattori

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1992; Laskowski et al. 1994; De Santo et al. 2002; Srivastava et al. 2017).

The data on the behaviour of HM in fire-affected forests are very scarce. Thus, modern investigations in the Mediterranean eucalypt and pine-dominated forests demonstrate that post-fire runoffs carrying HM significantly reduce stream microbial decomposer communities (Carvalho et al. 2019; Pradhan et al. 2020). The upper horizon of Cambisols in fire-affected larch forests in the Russian far East may accumulate a substantial stock of HMs (Bryanin and Sorokina 2019). The investigations from boreal larch forests in north China (Kong et al. 2018) and pine forests in East Siberia (Sosorova et al. 2013) revealed that HM concentrations returned to the pre-fire level in 8–11 years after a fire. However, long-lasting pyrogenic changes of soil properties (pH, temperature, moisture) and decreased litter input previously found in the study area (Bryanin et al. 2020) may create conditions for HM accumulation in the litter layer. To our best knowledge, investigations of HM dynamics coupled to litter decomposition in fire-affected forests have never been conducted in the boreal larch forests of Eurasia.

Decomposing litter may serve as temporary or long-term storage for some elements and be a source for others (Rustad and Cronan 1988). Incubation of litterbags is one of the most common approaches to studying decomposition in forest ecosystems, often including investigation of major elements dynamics. To date, we know that decomposition is a two-phased process comprised of early and late stages in which major elements dynamics are tightly coupled to mass loss (Prescott et al. 1993; Berg and McClaugherty 2014b). We know that decomposition is restrained in the fire-affected boreal forest even after decades since fire, and this effect is more pronounced in the later stage of decay (Hart et al. 2005; Bryanin et al. 2020). If the major element dynamic during decomposition stages is well understood, our knowledge of HMs is very limited even in background forests and missing in fire-affected forests.

Generally, the dynamics of HMs during decomposition depend on their chemical properties, qualities of forest litter (Gautam et al. 2019; He et al. 2016; Zhang et al. 2014), soil pH, moisture, nutrient concentrations, dissolved organic matter (De Santo et al. 2002; Lomander and Johansson 2001) and microbial activity (Tyler 2005). A comprehensive study from the temperate European forest has revealed that HMs either release slowly (e.g., Zn) or accumulate (e.g., Fe) during litter decomposition, except Mn, which often easily releases (Tyler 2005). Therefore, it was clearly shown that HMs have particular dynamics during the decomposition process. Considering possible HMs hazardous effect, we must better understand their dynamics during litter decomposition in fire-affected forests.

Despite broad recognition of the importance of vast boreal forests and litter decomposition as one of the primary sources of soil organic matter, the role of HMs in this process remains unclear. This study aimed to reveal HM dynamics during litter decomposition in fire-affected larch forests. The following questions have been addressed: (1) if decomposition is restrained in burned forests, does this affect HM dynamics? (2) Do HMs undergo stages of decomposition as mass loss, or do they have their independent dynamics? (3) Does HM release/accumulation differ in the fire-affected forest, and do soil properties affect these processes?

This study investigates the changes of HM (Fe, Mn, Zn, Cu, Co, Cr, Ni and Pb) concentrations and their release/accumulation in the larch needles during 3-year decomposition. We performed a field litterbag study in the unburned boreal larch forest, and the adjacent forest burned 15 years ago. We analyze HM dynamics, mass loss and soil properties, which allows us to reveal distinct patterns of post-fire HMs dynamic during litter decomposition in larch forests on the Russian Far East.

2 Material and methods

2.1 Site characteristics

The field experiment was conducted in Zeysky State Nature Reserve, which spans the Tukuringra mountain range in the Russian Far East. Natural larch/birch *(Betula platyphylla)*/(*Larix gmelinii (Rupr.)* Rupr.*)* stands represent typical vegetation cover of eastern Siberia and the Far East. Our research area (53° 50′ N, 127° 10′ E) located on a gentle south-facing slope of the mountain range in two stands: one was a background larch/birch forest with evenly distributed tree species and at least 100 years have not been influenced by the forest fire. We consider this stand as an unburned site (*Unburned*). Within the forest stand, there is a clear border of fire that occurred in the year 2003. It was a long-lasting surface fire that consumed the litter layer, charred the upper humus horizon of soil and killed most birches and part of larch trees, but mature trees of larch have survived. This stand was considered a burned site (*Burned*). Study territory belongs to the protected area, and according to surveys of Zeya Nature Reserve, this stand was uniform in terms of forest and soil characteristics before the fire event in 2003. Therefore, we recognize all observed differences in our experiment as fire legacy or effect of the post-fire environmental difference between sites. Fifteen years after the fire, some soil characteristics and HM concentration in the litter layer still differ significantly (Table 1).

Study sites were distributed in the discontinuous permafrost region with a mean annual temperature of −0.7 °C. The minimum month average temperature occurs in January (−19.3 °C), with the maximum in July (+19.1 °C). The sum of annual precipitation is 528 mm, most of which falls as rain

Table 1 Soil (0–5 cm) and litter characteristics

Characteristic	Unburned	Burned
Bulk density, g cm^{-3}	0.191 ± 0.01a	0.223 ± 0.04a
Water content, %	65 ± 1.5a	60 ± 2.0b
Clay, %	31.22 ± 3.09a	29.4 ± 4.07a
pH (H_2O)	4.8 ± 0.1a	5.4 ± 0.1b
CEC, meq 100 g^{-1}	19.1 ± 1.8a	25.3 ± 3.9a
P_2O_5, mg kg^{-1}	191 ± 27.1a	363 ± 54.6b
K_2O, mg kg^{-1}	484 ± 82.9a	556 ± 69.5a
Total organic C, %	20.1 ± 2.0a	21.1 ± 2.9a
Total N, %	0.77 ± 0.08a	0.86 ± 0.11a
Dissolved organic C, mg l^{-1}	43.9 ± 4.35a	24.1 ± 2.04b
Dissolved organic N, mg l^{-1}	1.11 ± 0.17a	0.77 ± 0.09b
Total litter stock, g m^{-2}	251 ± 20.4a	94.4 ± 11.2b
Fe, g kg^{-1}	10.3 ± 0.52a	10.4 ± 1.98a
Cu, mg kg^{-1}	11.0 ± 0.37a	13.4 ± 1.31a
Zn, mg kg^{-1}	70.2 ± 6.12a	108 ± 9.87b
Mn, mg kg^{-1}	942 ± 133a	518 ± 65.2b
Co, mg kg^{-1}	5.27 ± 1.16a	2.63 ± 1.08b
Cr, mg kg^{-1}	< 0.01	< 0.01
Pb, mg kg^{-1}	10.1 ± 1.81a	8.9 ± 1.29a
Ni, mg kg^{-1}	11.4 ± 0.55a	11.4 ± 1.62a

from July to September (Amur Center for Hydrometeorology and Environmental Monitoring 2015). The soil in the study area refers to Dystric Cambisols formed on granite rocks (WRB 2014). Detailed soil characteristics and profile morphological description is presented in previous studies (Procopchuk and Bryanin 2007; Bryanin and Sorokina 2019).

2.2 Litterbag experiment

For this investigation, we selected larch needles based on the domination of larch species in the boreal forests of Eastern Eurasia (Sato et al. 2016). We used the field litterbag approach to investigate the decomposition. Litterbags were made of white nylon with a mesh size of 50 µm; bag size was 100×100 mm with sealed edges. We chose a 50-µm mesh size to prevent washing out of litter fragments and root ingrowth. Such an approach allowed us to focus on microbial and fungi decomposition, excluding soil fauna, showing a contradictory effect on the decaying process (Makita and Fujii 2015). We used the same litter material on both sites. Each bag contained about 5 g of air-dried larch litter in six replications per sampling date. The bags were randomly set out on an area of 400 m^2 within unburned and burned sites. Samples were placed on the forest floor and slightly covered by fresh litter to create more tight contact with the surface. We did not bury litterbags into the litter layer, resulting in mass loss overestimation (Xie 2020). The experiment started in 2016 and lasted for 3 years, with an intermediate sampling of litterbags after 75, 140, 500 and 850 days. After sampling, litterbags were transported directly to the laboratory. Samples were cleaned from all adhering materials, and dried at 40 °C to the constant weight. For chemical analysis, samples were ground to pass through a 0.2-mm sieve.

2.3 Chemical analysis

All laboratory procedures were performed at the Analytical Centre of Mineral-Geochemistry Investigation in the Institute of Geology and Nature Management. Dissolved organic C and N were analyzed in the soil water extract by combustion with a TOC-L analyzer; total forms of organic C and N were analyzed on the same machine on the module for solid sample measurement, SSM-5000A (Shimadzu, Japan). Soil pH was measured electrometrically from a 1:2.5 soil H_2O using Hanna HI 2215 (HANNA Instruments Deutschland GmbH, Germany). During the measurement for the suspension, the pH was stable. The concentration of HMs in soil and litter samples was analyzed by digesting with a mixture of hydrofluoric acid and aqua regia. Element total concentrations were measured by atomic absorption spectrometry (180-50S, Hitachi, Tokyo, Japan).

2.4 Calculations and statistical analysis

The remaining mass was calculated in each sampling day and for the whole experiment period using Eq. (1):

$$Mass\ remaining(\%) = \frac{M_t}{M_0} \times 100\% \quad (1)$$

where M_0 – the initial mass of litter, g; M_t – the mass of litter in t day.

Release/accumulation (%) was calculated as the dynamic of mass-normalized element concentration following Eq. (2):

$$\text{Release / accumulation } (\%) = \frac{M_t \times C_t - M_0 \times C_0}{M_0 \times C_0} \times 100\% \quad (2)$$

where M_0 – the initial mass of litter, g; M_t – the mass of litter in t day; C_0 – initial concentration of HM, mg kg^{-1}; C_t – concentration of HM on the t day, mg kg^{-1}.

Firstly, all data were checked for normality and homogeneity of variances. Differences in mass loss between sampling days checked after data normalization by ANOVA followed the Tukey HSD test. Relation element concentration and mass loss were estimated by Pearson correlation after data normalization. Principal component analysis (PCA) was employed to reveal tendencies and patterns in all datasets. All statistical analysis was done in R-studio (R Development Core Team 2020).

3 Results

3.1 Litter decomposition and HM dynamic

Needle decomposition in unburned and in burned sites did not differ until the 140th day ($p = 0.507$); mass loss during the period accounted for 25% (Fig. 1). From the 500th day, the burned site decomposition rate started to reduce, and at the end of the experiment, mass loss in the unburned site was much higher (67.5%) than in the burned (44.2%) ($p < 0.01$).

The concentration of HM at the early stages of decay (0–140 days) differed little from the initial; change of concentrations mainly occurred in the late stages (500–850 days, Fig. 2). The final concentration at day 850 of the experiment of Cu, Zn, Fe, Co, Ni, Cr and Pb increased and Mn decreased related to initial concentration both in the unburned and burned sites (Table 2). Despite equality of initial concentrations at the end of the experiment, Fe on the burned site was twofold higher and Cr 1.7 lower than those on the unburned site ($p < 0.05$, Table 2). At the end of the experiment, there were no significant differences in Cu, Zn, Mn, and Co concentrations between burned and unburned sites; the only apparent trend of reducing the element concentrations in the burned site.

Initially, mass loss and HM concentrations did not correlate on both sites (Fig. 3). But in the late decomposition stage, we found a strong correlation for the majority of HMs and mass loss. For the majority of studied HMs, this correlation was similar on both sites. Thus Cu, Fe, Zn and Pb were positively correlated to mass loss. The group of HMs (Ni, Cr and Mn) negatively correlated to mass loss on the burned site, while positive or no relation (Mn) was on unburned.

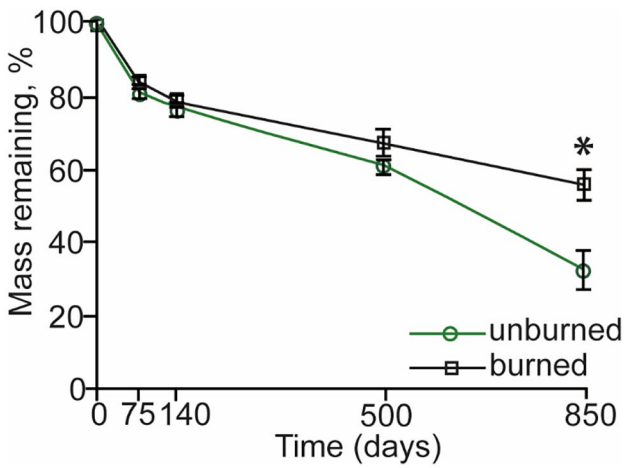

Fig. 1 Percent mass remaining (% of initial litter mass). Data are means; whiskers indicate standard errors for $n = 6$. The asterisk denotes a significant ($p < 0.05$) difference between sites based on ANOVA followed by Tukey HSD

The dynamic of Co was independent of mass loss on both studied sites.

3.2 Effect of soil properties on the HM release/accumulation

Release/accumulation of all studied HMs on both study sites was unidirectional; however, the final values of release/accumulation were site-dependent. Thus, Fe, Cr, Pb and Ni show accumulation, while Mn, Zn, Co and Cu showed release at the end of the experiment (Fig. 4). Overall accumulation was more pronounced in the burned site, whereas release – on the control. The apparent difference observed for Fe and Pb showed 4- and twofold higher final accumulation in burned site compared to the unburned ($p < 0.05$, Fig. 4).

The effect of soil properties was element- and site-dependent. Thus, on the unburned site, the PCA depicts several groups of HMs with interact release (Fig. 5a). However, soil properties are included in other PC unrelated to groups of HM. The only release of Fe and soil pH pointed in the same direction. On the burned site, Fe Cr, Ni and Cu pointed in the same direction with soil pH and dissolved nitrogen, whereas soil water content pointed in the opposite direction (Fig. 5b).

4 Discussion

4.1 Dynamics of HMs during decomposition

Our results show that the concentrations of HMs, except Mn, increase to the end of the experiment from twofold for Cu, Zn and Co up to sixfold for Fe and 15-fold for Pb (Table 2, Fig. 2). We observed significantly restrained late-stage litter decomposition on the burned plot (Fig. 1) along with higher final concentrations of Fe and Cr (Table 2). In general, our result supports previous studies in natural forests that found an increase of HM concentrations with decomposition time (Laskowski and Berg 1993; Tyler 2005; Brun et al. 2008; Gautam et al. 2019). Therefore, answering our first question, we conclude that a long-lasting fire legacy, in terms of modified soil parameters, had restrained litter decomposition but did not alter the overall dynamics of HMs. However, we revealed a stage-dependent correlation between HMs and mass loss. Thus, in the early stage, when the mass loss was more rapid, there was no significant correlation between mass loss and HM concentration, and this pattern was equal in unburned and burned forests (Fig. 3). Overall, in the late stage of our experiment in both studied forests, most HMs significantly correlating with mass loss showed similar trends except Ni and Cr. These elements are shown a decreasing trend in the burned forest and increasing in the unburned (Fig. 3). Therefore, answering our second

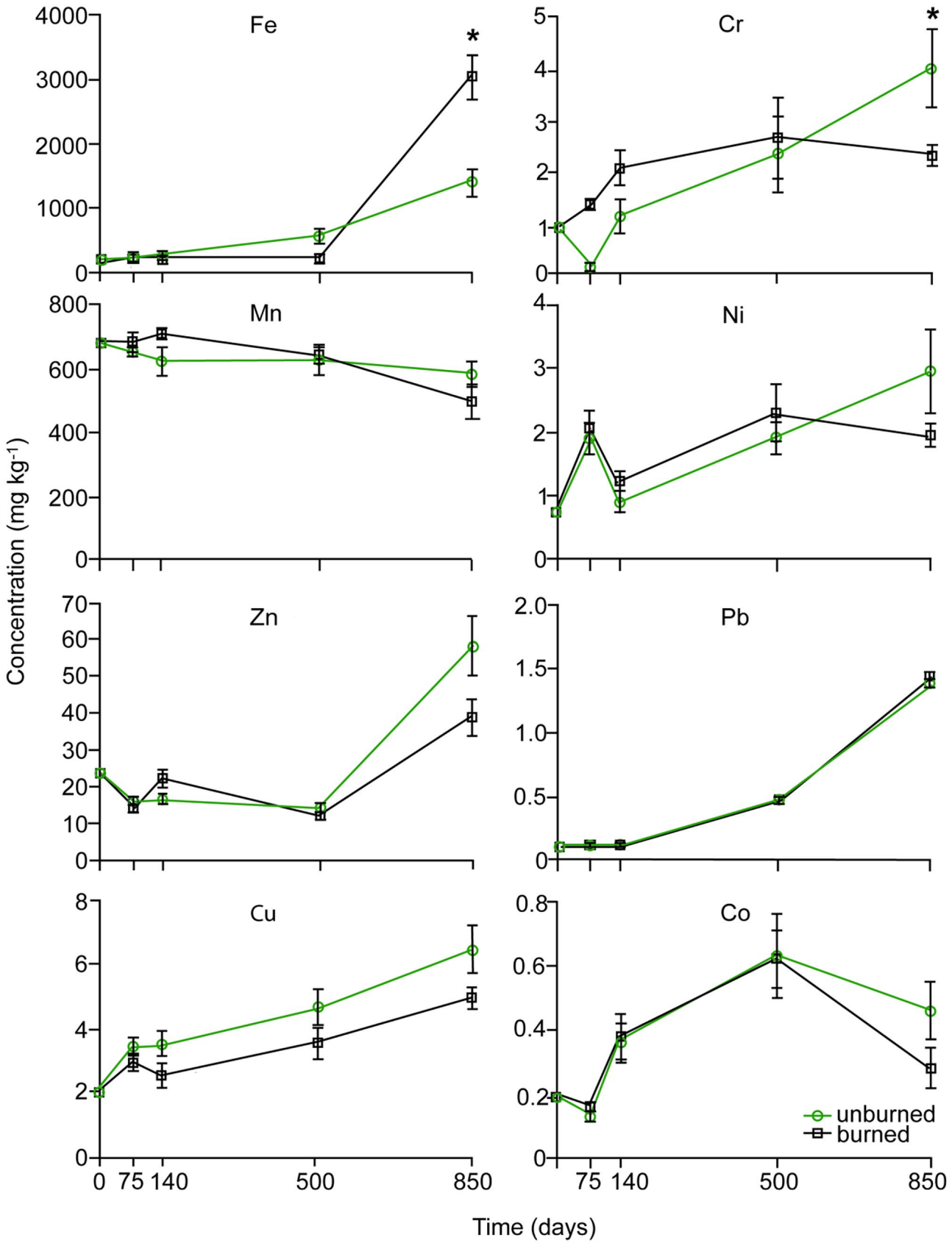

Fig. 2 Dynamics of HM concentrations during decomposition time. Data are means; whiskers indicate standard errors for $n = 6$

Table 2 Initial and the final concentrations of heavy metals on unburned and burned sites

Metals	Initial concentration	Final concentration			
		Unburned		Burned	
	mg·kg^{-1}	mg·kg^{-1}	*p values*	mg·kg^{-1}	*p values*
Cu	3.79	6.47 ± 0.74a	*0.0012*	4.96 ± 0.34a	*0.0001*
Zn	20.8	58.3 ± 8.09a	*0.0053*	38.8 ± 4.86a	*0.0216*
Mn	697	596 ± 40.5a	*0.0465*	507 ± 54.6a	*0.0131*
Cr	0.91	4.06 ± 0.77a	*0.0178*	2.35 ± 0.22b	*0.0115*
Ni	0.71	2.95 ± 0.66a	*0.0162*	1.94 ± 0.18a	*0.0007*
Co	0.19	0.46 ± 0.09a	*0.0192*	0.25 ± 0.06a	*0.1285*
Pb	< 0.1	1.42 ± 0.54a	*0.0519*	1.45 ± 0.41a	*0.0256*
Fe	211	1412 ± 224a	*0.0017*	3035 ± 354b	*0.0025*

Data in the table are means ± standard errors, *p* values represent the difference of final HM concentration compared to initial based on *t*-test. Different letters denote the significant difference of HM concentrations between sites based on ANOVA followed by Tukey HSD

question, we conclude that HM dynamics coupled the mass loss only on the late stage of decomposition, but this relation was site-dependent.

Interestingly, the most pronounced changes in HM concentrations occurred in the late stage when the decomposition rate decreased compared to the initial (Figs. 1 and 2). One of the possible explanations of observed phenomena is that, in the beginning, the rate of decomposition is higher than the release of HMs later; when decomposition rate decreases, these rates became equal and showed a relation. Previous papers studying forest litter decomposition also proposed this mechanism (Gautam et al. 2019; Lomander and Johansson 2001). Another possible way of HM accumulation in late stages might be accumulation from the outer soil environment. In the late stage, due to microbial and fungal activity, larch needles became porous and might be easily penetrated by percolating, containing HMs from the outer soil and litter. Previous studies mainly explained HM accumulation from external sources (Brun et al. 2008), including air pollution (Rühling et al. 1973; Laskowski et al. 1995). Our study was performed in a nature reserve far from industrial activities and busy roads; therefore, we consider observed accumulation as immobilization from litter and underlying soil layer. The mechanism of this transport of HMs might be explained by percolating soil solution and simultaneous fungal activity. Soil fungi are known to concentrate and retain Fe and other elements (Stark 1972; Levinskaite et al. 2009; Kabata-Pendias and Szteke 2015). We suggest that the above accumulation mechanism is most likely in our system due to the high Fe concentration in the organic layer (Table 1). Calculations of Fe mass in abscised litter covering litterbags during two seasons of the experiment indicated that this source might have contributed only 14% of observed accumulated Fe (Table 1, Fig. 2). A similar calculation showed that the proposed mechanism of soil solution and fungal HM transport might occur for most studied elements except Mn and Cr. As the concentration of Cr in the surrounding litter was < 0.1 mg/kg, its input from the litter layer to litterbags was unlikely. Simultaneously to decay, litter decomposition is the process of synthesis of various organic substances with high cation exchange capacity. In such conditions, HMs are easily bounded to newly formed organic compounds and accumulated in litterbags. Usually, these bounds are tight that protect soil biota from the hazardous effect of HMs; however, soil properties considerably mediate the dynamic of these compounds.

4.2 Effect of soil properties on HM dynamics

Element concentration is an unsteady characteristic reflecting only the given sampling event in the whole decaying process. In contrast, the accumulation or release of the element during decomposition could provide an overall understanding of the element behaviour and its possible implications for the ecosystem. Although in our experiment, most HMs increased in decaying needles; only Fe, Pb, Ni and Cr accumulated in absolute value, and the most pronounced accumulation was observed for Fe and Pb (Fig. 4). Despite release/accumulation processes being unidirectional in both study sites, we noticed more release for all HMs in the unburned forest and more accumulation in the burned. The most considerable difference observed for Ni, Pb and Fe is that these HMs accumulate 3 and 9 times more on the burned site than on the unburned after 3 years of the experiment. Given that we used the same litter for this investigation, observed phenomena could be attributed to the effect of fire-altered environment and soil properties. Previous studies highlighted the importance of ecological factors as mediators of HM release/accumulation during decomposition (De Santo et al. 2002; Goya et al. 2008; He et al. 2020; Lomander and Johansson 2001).

We found out that 15 years after the fire, concentrations of the most studied HMs except Zn, Mn and Co had returned to pre-fire

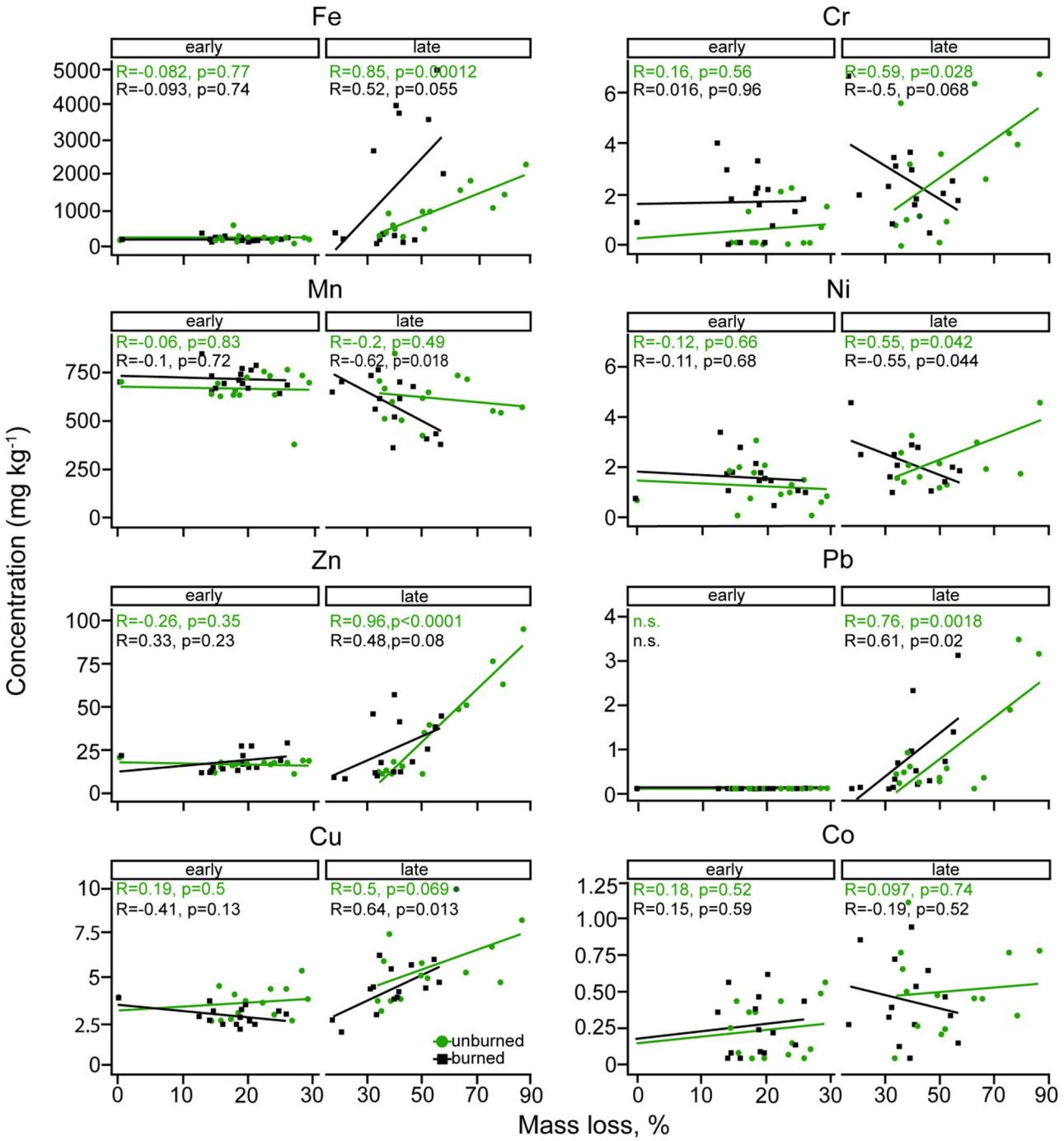

Fig. 3 Correlation of HM concentrations with mass loss in early and late decomposition stages

level (Table 1), proving the earlier findings in boreal forest ecosystems (Kong et al. 2018; Sosorova et al. 2013). However, soil properties essential to HM dynamics remain altered in the burned site. Thus, we noticed the rise of pH_{H2O}, decreasing soil water content, dissolved organic carbon, and microbial activity in the burned site (Table 1; Bryanin et al. 2020). These soil properties could be responsible for the larger immobilization of HMs with high accumulation factors such as Fe, Pb and Cr.

The formation of stable complexes of Fe with organic matter increases with the rise of pH, while its solubility decreases (Boguta et al. 2019; Catrouillet et al. 2014). In our system, higher pH and lower water content on burned sites create favourable conditions for the HM accumulation (Fig. 4). The reduced accumulation of Fe in litterbags on the unburned forest could be attributed to its complexation with water-soluble organic matter and migration to underlying soil layers,

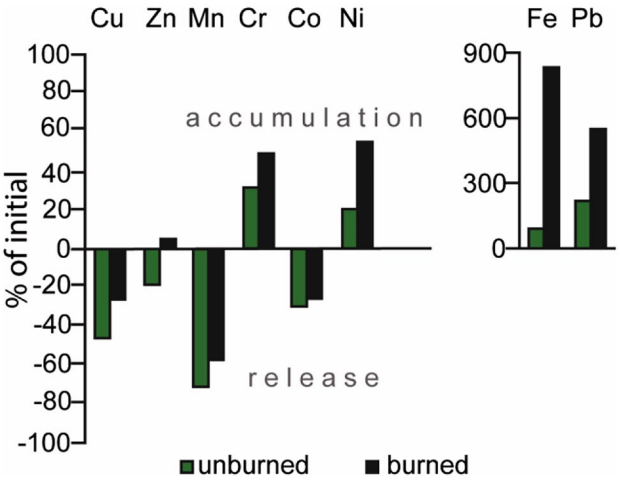

Fig. 4 Final release/accumulation of HMs calculated by Eq. (2). Zero is the initial mass-normalized concentration at the beginning of the experiment

as previously described by Staaf (1980). The concentration of dissolved organic carbon (DOC) was higher ($p < 0.05$) in the unburned site even 15 years after the fire (Table 1). A decadal post-fire decrease of DOC was noticed in larch stands in Central Siberia (Startsev et al. 2017). The higher soil water content derives more DOC to be released in the soil solution, making complexation with Fe likely (Gmach et al. 2020; Rustad and Cronan 1988). In our system, DOC concentration in unburned forest correlates to water content, soil pH and concentrations of Fe, Mn and Cu, supporting the above mechanism (Table 1, Fig. S1a). On the burned site, water content, pH and DOC located in different directions on PCA ordinates, proving largely altered soil properties even 15 years after the fire (Fig. S1b). However, Fe, Cu, Cr and Ni grouped with DOC in the same direction as on unburned site (Fig. S1B).

All mechanisms considered above for Fe could apply to other elements from the group of HMs. For example, Cr and Ni show similar behaviour, closely related to pH and water content (Vodyanitskii 2008; Yue et al. 2019). However, Pb has a different dynamic: we observed a threefold larger accumulation in the burned site compare to those on the unburned (Fig. 4) In our opinion, this is explained not by environmental factors but restrained mass loss (Fig. 1). Pb concentration's dynamic was almost identical on both sites showing a steady increase during the experiment (Fig. 2) and a strong positive correlation to mass loss (Fig. 3). Supporting this assumption, Pb did not correlate to soil properties and lied to another direction from other elements and soil properties on PCA (Fig. 5). Same as accumulation, the concentration of Pb was also positively correlated only to mass loss on both sites (Fig. S1).

Our results demonstrate that forest fire, at least for 15 years, could modify later stages of mass loss and HM behaviour during litter decomposition. The lack of differences in mass loss and HM dynamics between burned and unburned sites in the early stages of destruction could suggest that abiotic and biotic conditions changed by the fire do not play a critical role both in decomposition and HM dynamics when leaching of low-molecular substances predominate.

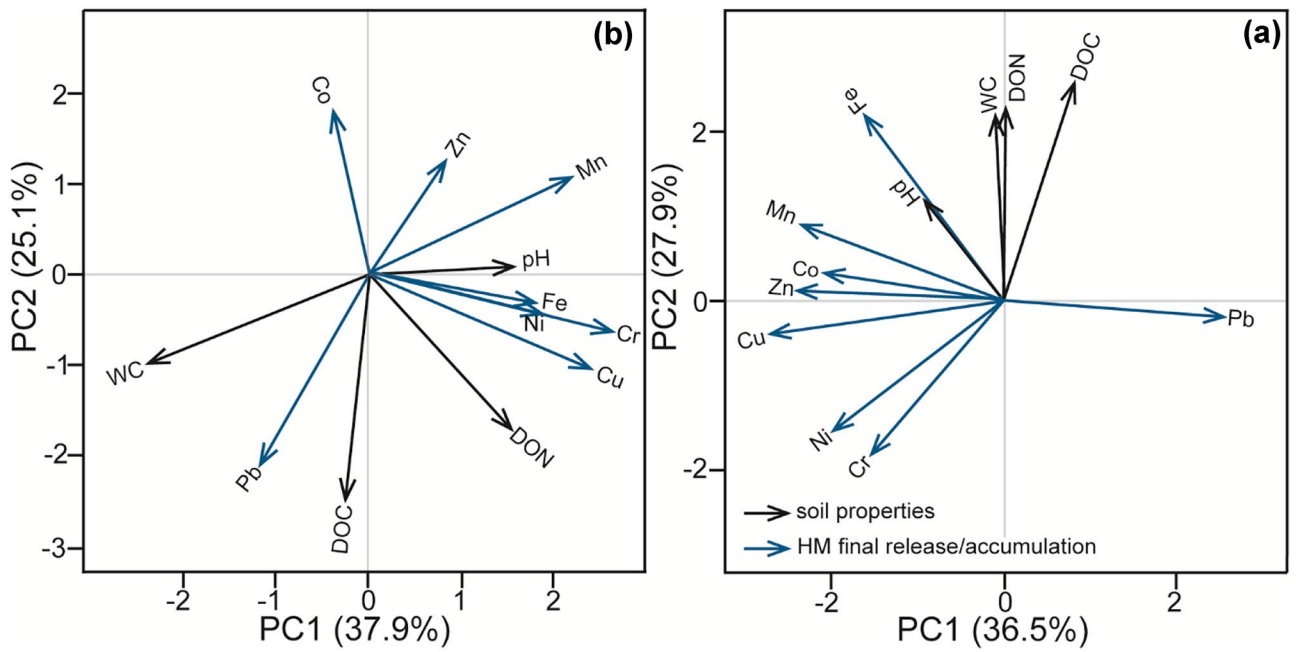

Fig. 5 Biplot obtained from the analysis of the main components of soil properties and HM release/accumulation after 850 days of the experiment on unburned (**a**) and burned (**b**) sites

5 Conclusions

Field investigation of the temporal dynamics of heavy metals during litter decomposition in fire-affected forests is of great significance for improving our mechanistic understanding of the legacy effect of fire. As a case of such an approach, our 3-year field experiment showed that the legacy effect of fire derives some fluctuations, but generally, most heavy metals concentrations increased during litter decomposition in both studied forests. However, due to the discrepancy of mass loss rates and element release, we observed accumulation in absolute value only for Fe, Pb, Cr and Ni. Moreover, restrained mass loss and altered soil properties have led to a more considerable accumulation of these elements in the burned forest than the unburned. This effect may slow down the turnover of some metals in post-fire boreal forests. Besides, our results revealed the stage-dependent dynamics of heavy metals during litter decomposition. This finding, along with the larger accumulation of heavy metals in post-fire forest cooperatively, highlights the importance of temporal variability in accessing the legacy effect of fire during litter decomposition.

Even though our investigation was held in the nature reserve, excluding possible pollution, we observed the legacy effect of fire, creating favourable conditions for heavy metals accumulation. These conditions have led to up to 9-time larger element accumulation from natural sources such as litter and soil. Therefore, in the case of a pollution source near the burned forest, a long-lasting favourable condition for heavy metals accumulation may lead to a much considerable increase of elements in the litter layer with possible hazardous implications to the environment.

Supplementary Information The online version contains supplementary material available at https://doi.org/10.1007/s11368-021-03027-5.

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Author contribution A.K. Conceptualization, investigation, writing original draft, validation. S.B. Conceptualization, investigation, review and editing, supervision, funding acquisition

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Data availability Data attached as supplementary material.

Declarations

Conflict of interest The authors declare no competing interests.

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