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Effect of logging residues management on the distribution of potentially toxic elements in soils of large-scale clearcuts resulting from bark beetle forest damage

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Abstract: This research focuses on the effects of large-scale clearcuts resulting from salvage logging after spruce (*Picea abies*) forest dieback caused by an extreme bark beetle infestation, and on the effect of logging residues management (chopping vs. clearing) on the distribution of potentially toxic elements (PTEs) in soil. Pseudo-total contents of Cd, Cr, Cu, Ni, Pb and Zn were determined in soil samples collected separately from the organic (F+H) and mineral (0–10, 10–20, and 20–30 cm depths) soil layers. The distribution of elements was influenced mainly by sampling locality and position in the soil profile. In general, the contents of Cd, Ni and Cr were higher in the mineral layers, whereas Pb was more concentrated in the FH layer. A significant effect of logging residues management on the distribution of PTEs was observed only for Pb and Zn. We expect that the relative decrease of Pb and increase of Zn contents in the "chopped" treatment was mostly due to the higher input of mineral soil and wood residues to the FH layer. Since the stand was harvested relatively recently, the effects of soil preparation have probably outweighed those of spreading or removing logging residues.

Keywords: forest soil; harvesting; risk elements; spruce

Monitoring and studying the behaviour and distribution of potentially toxic elements (PTEs) in agricultural and forest soils in the Czech Republic has a long history (Suchara & Sucharová 2002; Poláková et al. 2017). In terms of soil pollution, the Czech Republic has several anomalies, both natural – geological and anthropogenic – areas affected by mining and processing of ores or coal combustion. Many anthropo-

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genic anomalies are related to the deposition of dust particles. For example, the area surrounding the lead smelter in Příbram is marked as a multi-element pollution hotspot containing Ag, Bi, Cd, Hg, Pb, Sb, Sn and Zn (Suchara & Sucharová 2004; Ettler et al. 2005, 2006; Sucharová et al. 2011). The Silesian Beskids is an example of an area with prevailing combined effects of contamination and acidification of forest soils (Šrámek et al. 2008; Pavlů et al. 2015).

Forest soils and specifically their upper organic horizons serve as accumulators of PTEs (e.g. Juřička et al. 2023). PTEs enter the topsoil horizons either directly from atmospheric deposition (Sucharová et al. 2011) or from the deeper soil layers through root uptake and the subsequent litter fall of leaves, bark and needles (Juřička et al. 2022). When metals are released from decomposing litter, they are adsorbed on deeper located transformed organic matter with strong binding capacity (e.g. Reimann et al. 2007). Therefore, the bioavailability and mobility of PTEs is controlled mainly by the content and quality of soil organic matter (SOM) and soil pH (e.g. Pavlů et al. 2015; Gąsiorek et al. 2017). The importance of controlling factors is specific for each PTE (Hernandez et al. 2003). For example, Zn and Cd migrate together with soluble organic fractions, which allows them to penetrate more easily into the deeper soil layers (Hernandez et al. 2003). On the other hand, PTEs like Pb and Cu with high affinity to high molecular organic matter can be immobilized directly in organic horizons and do not represent an immediate environmental risk (Yelpatyevsky et al. 1995; Ruan et al. 2008). The organic matter accumulates in surface organic horizons of forest soils along with the forest development, and any disturbance of these organic-rich soil layers may mobilise the accumulated PTEs.

During last decades, extreme bark beetle infestations caused by preceding period of drought have been a major challenge for forests throughout the Northern Hemisphere, including the Czech Republic (Neumann et al. 2017; Hlásny et al. 2021; Brázdil et al. 2022). Extensive dieback of spruce-dominated forest stands led to fundamental changes in the light, temperature and humidity regimes. Disruption of the canopy leads to soil surface heating up, which increases evaporation and, on the other hand, the absence of trees transpiration often leads to temporary waterlogging of the soil (Kohout et al. 2018). However, aside from extreme drying or over-wetting of the soil, an increase in temperature and fresh litter inputs accelerate microbial activity (Štursová et al. 2014; Kohout et al. 2018) and the decomposition of organic matter (Clarke et al. 2015; Keenan 2016). Tree dieback also alters the cycles, pools and fluxes of nutrients in forest ecosystems (e.g. Kaňa et al. 2019). Additionally, disturbing the soil surface during logging accelerates the decomposition of organic matter and releases carbon from the upper soil horizons (Yanai et al. 2003; Mayer et al. 2020). Changes in light conditions and resource availability trigger the growth of herbaceous plants and the rejuvenation of forests. This relatively quickly leads to the soil surface becoming shaded again. The relatively quick response of soil chemistry to environmental changes has been observed, particularly in the upper organic soil layers (e.g., Porębska et al. 2008; Kaňa et al. 2013).

In natural forests, dead trees stay in place, becoming a source of organic matter and nutrients. In contrast, in managed forests, they are harvested, resulting in a significant loss of biomass that would otherwise replenish soil reserves of organic matter and nutrients. The harvesting methods themselves, such as tree or trunk harvesting, the size and shape of the clearcut and management of logging residues, all begin to play an important role. Prescott and Grayston (2023) demonstrate that the properties of the rhizosphere and soil ecosystem are preserved even at a distance of 10 m from the base of the trees. The narrow shape of the cut, or the leaving of individual trees, can to some extent limit the impact of clearcutting on both soil biodiversity and soil organic matter (Šrámek et al. 2024). And while the maximum permitted size of clearcuts after planned logging takes this effect into account, the size of areas affected by bark beetles ranges from tens to hundreds of hectares.

The behaviour of PTEs in soils of clearcuts could be significantly influenced by the intensity of surface disturbance, as well as by the management of logging residues (i.e. removal of biomass or leaving it in place). We hypothesize that extensive deforestation and soil disturbance accompanying logging will lead to changes in the content and distribution of PTEs in the soil profile. Therefore, there will be differences in the content and distribution of PTEs between undisturbed forests and clearcuts, and between clearcuts depending on whether logging residues are removed (clearing) or chopped and dispersed in the form of chips across the clearcut area and partially incorporated into the soil. Larger differences can be expected for elements with a higher affinity for SOM (Cu and Pb) that has accumulated in the

https://doi.org/10.17221/119/2025-SWR

surface organic horizons. The accelerated decomposition in the clearcuts may cause these elements to be mobilised and transported to deeper layers. Compared to the original forest, their concentrations in the clearcuts in the FH horizon would be lower, whereas in the deeper layers they would be higher. Removing all logging residues from the clearcuts can further intensify this process.

MATERIAL AND METHODS

Research was focused on clearcuts > 1 ha resulting from salvage logging after spruce (*Picea abies*) forest dieback caused by extreme bark beetle infestation in the highlands (altitude range 500–800 m) of the Czech Republic between 2015 and 2020. Eleven sites from the long-term monitoring in the ICP Forest program affected by forest dieback and salvage logging were selected (Figure 1). Soils of all sites belong to Cambisol reference soil group (IUSS Working Group WRB 2022). On each site, three or four plots with different treatments were defined. Two of them were maximum 4 years (predominantly 2–3 years) old clearcuts with different management: (*i*) logging residues chopping (including stumps) and scattering of wood chips over the cleared area – WCH, and (*ii*) removal of logging residues (branches and tree tops, no stumps), i.e. their concentration into mounds or piles – RLR, and one or two surviving forest stands (FOR) as the control.

Soil samples were collected from surface layer (FH = mixture of fermented – F and humified – H soil surface organic horizons (Green et al. 1993) and with wood chips in WCH variant), and from the mineral soil in 0–10, 10–20, and 20–30 cm depths. At each plot, subsamples were collected from the central point and four points at 10 m in the cardinal directions to form one composite sample for each soil layer.

Samples were air dried at room temperature and sieved through a 2-mm sieve for analysis of selected soil properties. Basic soil characteristics were determined in accordance with ICP Forests methodology (Cools & De Vos 2016). Total contents of C and N (Ctot, Ntot) were measured using LECO CNS elemental analyser (LECO, St. Joseph, MO, USA). The active pH_{H_2O} (in soil-water suspension; 1 : 5/v : v) and exchangeable pH_{CaCl_2} (in soil – 0.01M $CaCl_2$ suspension) were determined potentiometrically using pH meter 798 MPT Titrino (Metrohm, Herisau, Switzerland) with a glass electrode. Pseudo-total contents of potentially toxic elements (PTEs: Cd, Cr, Cu, Ni, Pb, and Zn) were determined in aqua regia extract (mixture of nitric acid and hydrochloric acid in a molar ratio of 1 : 3) using a Thermo iCAP 7000 inductively coupled plasma optical emission spectrometer (ICP–OES; Thermo Scientific, USA).

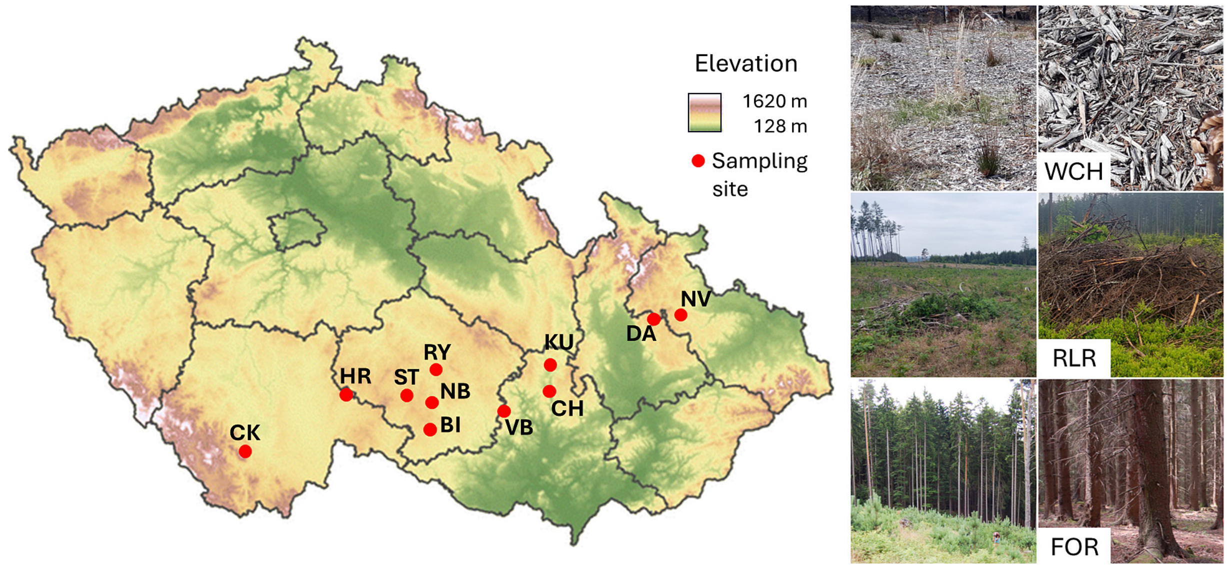

Figure 1. Map of the Czech Republic with sampling site designation and experiment variants presentation
BI – Bítovánky; CH – Černá Hora; CK – Český Krumlov; DA – Dalov; HR – Horní Radouň; KU – Kuničky; NB – Nová Brtnice; NV – Nové Valteřice; RY – Rytířsko; ST – Stonařov; VB – Velká Bíteš; WCH – logging residue chopping and scattering over the cleared area; RLR – removal of logging residues, i.e. their concentration into mounds or piles; FOR – a control forest

https://doi.org/10.17221/119/2025-SWR

All the analytical procedures were secured from point of view of Quality Assurance (QA) and Quality Control (QC) according to the standard laboratory practise including repeated measurements, processed blanks, fortified standards, spiked samples and the use of certified referential materials (IAEA 2003; NATA 2012).

The Statistica software (Ver. 14.1.0.8) (TIBCO Software Inc., USA) was used to perform statistical analyses. The basic statistical parameters, such as the mean and the standard errors were computed. The normality of all data sets was tested using the Kolmogorov-Smirnov test. Outliers (Cu content > 100 mg/kg) limiting normal distribution were excluded from the dataset before further statistical processing. A main effect analysis of variance (mANOVA) was used to assess the differences between sites (site; N = 11), soil layers (layer; N = 4), and the two variants of logging residues management plus control variant (management; N = 3). A Tukey HSD test was computed for the three categorical variables (site, layer, and management); significant difference was considered at level α = 0.05. Paired t-test was used for comparing control (FOR) with both clearcut variants (WCH and RLR), and these two variants with each other (without site and layer effect). Relationships between the soil properties were assessed by factor analysis with Varimax normalisation, and two factors that explained more than 60% of the data variability were selected. Data sets from the organic (FH) and mineral (0–30 cm) soil layers were evaluated separately. Variables N and pH_{H2O} were excluded from the analysis due to data collinearity (with C and pH_{CaCl_2}) causing instability of the model.

RESULTS AND DISCUSSION

A main effect analysis of variance showed significantly different loads of the studied PTEs in individual localities (Table 1). The highest concentrations of Cd, Cr, Ni and Zn were found in Bítovánky and Nová Brtnice, and the highest concentration of Cu was found in Černá Hora. Highest concentrations of Pb were found in Dalov and Nové Valteřice. In contrast, the experimental area near Český Krumlov was the least affected by PTEs. A more detailed assessment, taking into account individual soil layers and comparing the data (https://zenodo.org/records/17787545) with the values reported in the literature (Kabata-Pendias & Pendias 1992; Borůvka et al. 2015), shows that the Cd concentration in the

Table 1. Results of the main effect analysis of variance

PTE variable	Cd		Cr		Cu		Ni		Pb		Zn	
	F	*P*	*F*	*P*	*F*	*P*	*F*	*P*	*F*	*P*	*F*	*P*
Site	**21.32**	***0.000***	**23.61**	***0.000***	**19.27**	***0.000***	**25.95**	***0.000***	**10.63**	***0.000***	**19.17**	***0.000***
Layer	**11.08**	***0.000***	**19.20**	***0.000***	0.107	*0.956*	**13.72**	***0.000***	**74.04**	***0.000***	0.665	*0.575*
Management	1.191	*0.307*	0.626	*0.536*	0.338	*0.714*	0.676	*0.511*	**8.804**	***0.000***	2.289	*0.106*
PTE	Cd		Cr		Cu		Ni		Pb		Zn	
Bítovánky	ab		a		b		a		b		a	
Černá Hora	abc		cde		a		de		b		bc	
Český Krumlov	g		f		d		e		b		f	
Dalov	bcd		bcd		bc		bcd		a		cd	
Horní Radouň	bcde		bcd		cd		bcd		b		cd	
Kuničky	ef		ef		cd		e		b		de	
Nová Brtnice	a		b		cd		a		b		ab	
Nové Valteřice	def		def		bc		de		a		cd	
Rytířsko	cde		bcd		cd		bc		b		de	
Stonařov	def		bc		cd		b		b		cd	
Velká Bíteš	fg		def		d		cde		b		ef	

PTE – potentially toxic element; significant effects are in bold; Tukey HSD test was computed for the categorical variables (site, N = 11); not sharing of any letter (a–f) in the columns of relevant elements indicates difference between sites at the significance level α = 0.05 (letter "a" represents the highest element concentration)

https://doi.org/10.17221/119/2025-SWR

mineral horizons at all sites exceeds 0.5 mg/kg which is the upper limit of the common values interval for Cambisols (Kabata-Pendias & Pendias 1992). In Nová Brtnice, the Cd concentrations range between 2.5–2.8 mg/kg. When compared with the reference values specified in Borůvka et al. (2015), the Cd concentration is only in the FH layer at the Český Krumlov site below value of 90th percentile of values common in forests in the Czech Republic. Further exceeding of the upper Zn limit (75 mg/kg) of values common for Cambisols was found in Bítovánky (80–100 mg/kg) and Nová Brtnice (80 to 90 mg/kg). According to Borůvka et al. (2015), the 90th percentile values for forest soils in the Czech Republic were exceeded for Zn in the FH layer in Bítovánky and Černá Hora and for Pb in the 10–20 cm layer in Nové Valteřice. All other values were below reference values and the locations can be considered as non-polluted.

Next criterion that significantly influences the distribution of PTEs is their position within the soil profile. Individual PTEs behave differently, but their distribution in the soil is mainly determined by their source (atmospheric deposition or bedrock), organic matter content and soil pH (e.g. Gąsiorek et al. 2017). Elements with a high affinity to organic matter, such as Pb or Cu, accumulate more in the organic horizons, whereas elements of geogenic origin or with higher mobility may accumulate in the deeper horizons. Figure 2 shows the distribution of individual PTEs in the studied soil layers. Copper and zinc are evenly distributed in the top 30 cm of soil. The distribution of copper corresponds to a mixed source (deposition and bedrock) similarly to the work of Borůvka et al.

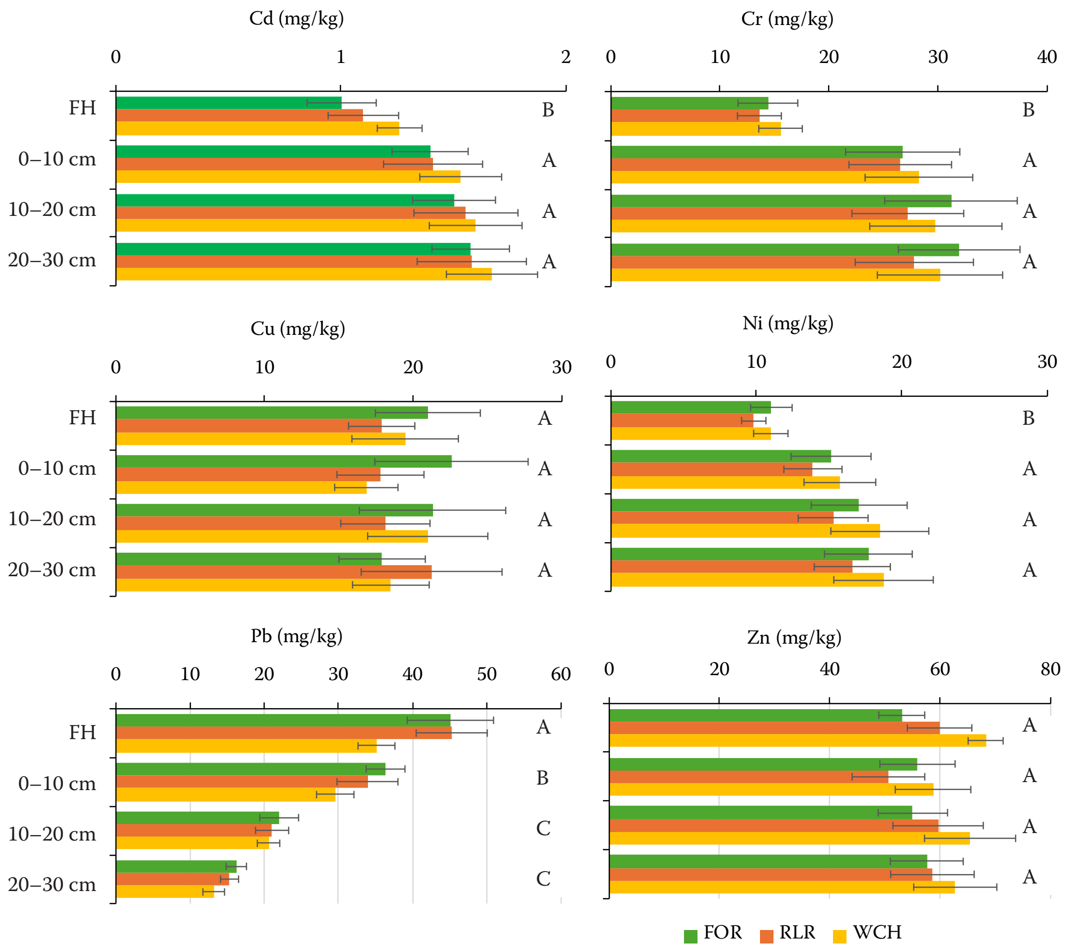

Figure 2. Distribution of studied potentially toxic elements (PTEs) in soil profile

FOR – undisturbed forest, RLR – removing of logging residues, WCH – spreading wood chips on a clearcut; FH – mixture of fermented (F) and humified (H) soil surface organic horizons ; average values for management variants with bars representing standard error and with letters denoting significant difference between soil layers at the level α = 0.05

https://doi.org/10.17221/119/2025-SWR

(2005) and not to dominant deposition source and binding to organic matter as e.g. in Pavlů et al. (2015). The contents of Cd, Cr and Ni are significantly lower in the surface organic horizons as compared to the mineral horizons, whereas Pb is (according to the assumptions and the affinity to SOM described above) most concentrated in the FH layer and least in the subsoil (10–20 and 20–30 cm depths).

For lead, the effect of management was evident in this first evaluation, i.e. its content in the soil of a clearcut with chopping significantly differed from the control forest. The RLR variant did not differ from the other treatments. To eliminate the influence of site and layer, a paired t-test was used (Table S1 in Electronic Supplementary Material (ESM)). In the case of Cu and Zn, the entire dataset was tested because there were no significant differences between layers. In the case of Cd, Cr, and Ni, the FH and mineral layer data were tested separately. For Pb, FH, 0–10 cm and 10–30 cm data were tested separately. A significant difference between the treatments was found for Pb and Zn in all the tested datasets. In all cases, the WCH variant differed from the FOR variant. Higher Pb (FH: 50.48 mg per kg > 35.11 mg/kg) and lower Zn (58.04 mg/kg < 63.74 mg/kg) contents were found in the FOR than in the WCH variant.

Factor analysis was used to test the effect of basic soil properties on the PTEs distribution in organic and mineral soil layers. Two factors per dataset were extracted. In the organic soil layer, they explain 62.30% of the total variability of the data and in the mineral layer, 72.36% (Table 2). The first factor (F1) is represented by high loadings of Cd, Cr, Ni and Zn in both datasets. The second factor (F2) includes loadings of Pb with C/N in organic layer and with C and pH in mineral layers. The first factor could be explained as an effect of various sites burden with Cd, Cr, Ni and Zn in whole profile. The second factor primarily indicates a different distribution of Pb compared to other PTEs. This is also documented by Figure 3, which shows distribution of factor scores in mineral layers and differentiates soil samples according to site (visibly separated variously PTEs loaded sites along the x-axis – F1) and layer (visibly separated along the y-axis – F2). The explanation of the causal relationships between basic soil properties (pH, C/N) and PTEs content is limited, especially in the FH layer, by the varying PTE loads in different locations. However, it can be said that at higher pH levels, the mobility and consequently the leaching of Cd, Cr, Ni, and Zn are smaller, which may lead to higher element contents in FH layer and that Pb is better bound by organic matter with a lower C/N ratio. In the mineral layers, a relationship is also visible between Pb and C/N, however, even stronger relationship is shown with Pb and C content, which indicates the limitation of lead binding by sorption sites on SOM in these layers.

CONCLUSION

We hypothesized that extensive deforestation and soil disturbance by salvage logging would lead to changes in the contents and distribution of PTEs

Table 2. Factor loading matrix after varimax rotation separately for the FH layer and mineral layers with values of dataset variability explained by selected factor

Variable	FH layer		Mineral layers	
	factor 1	factor 2	factor 1	factor 2
pH_{CaCl2}	0.773	–0.285	0.200	0.802
C/N	–0.170	–0.843	–0.242	–0.665
C	–0.385	0.006	0.025	–0.917
Cd	0.923	–0.004	0.911	0.136
Cr	0.849	0.250	0.893	0.120
Cu	0.564	–0.204	0.595	–0.194
Ni	0.712	0.435	0.867	0.247
Pb	–0.381	0.722	0.161	–0.868
Zn	0.822	–0.090	0.916	0.074
Expl. variability (%)	44.36	17.94	41.07	31.29

FH – mixture of fermented (F) and humified (H) soil surface organic horizons; loadings higher than 0.7 are written in red

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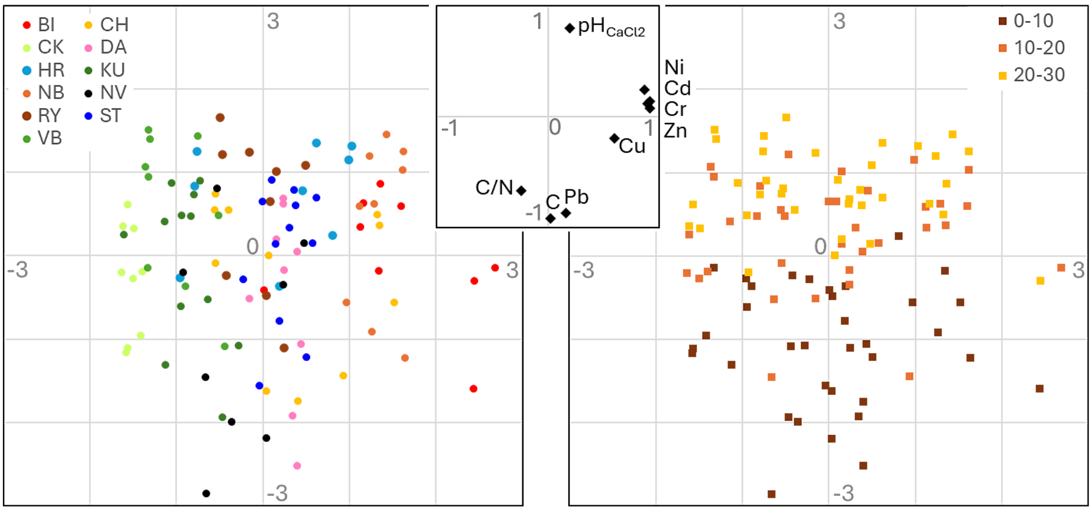

Figure 3. Results of factor analysis for mineral layers: factor loadings of selected soil properties after varimax rotation (middle); factor scores with the sorting of points into groups according to site (right) and layer (left)

BI – Bítovánky; CH – Černá Hora; CK – Český Krumlov; DA – Dalov; HR – Horní Radouň; KU – Kuničky; NB – Nová Brtnice; NV – Nové Valteřice; RY – Rytířsko; ST – Stonařov; VB – Velká Bíteš

in the soil profile, which would be observed as the differences in the PTEs content between undisturbed forests, clearcuts where logging residues were removed or chopped and spread out. This hypothesis was not confirmed for most of the PTEs studied except Zn and Pb, for which changes between treatments were detected. However, the expected changes in distribution were not confirmed. Due to the main accumulation of Pb in the surface organic layer, its distribution is most affected by clearcut management. The relationship of Pb and C content has been confirmed many times and was also confirmed by factor analysis in this observational experiment. Mixing of the surface organic layer with wood chips and the mineral soil are probably the main factors of relative Pb decrease in the upper layers at the WCH variant. The 0–10 cm layer mixed by chopping with the FH layer is the most acidic at studied sites (Table S2 in ESM). Given the key influence of pH on Pb mobility, documented in the literature, this fact can be understood as a potential health risk for newly forming ecosystems. However, in the studied localities, there is probably no direct threat to newly formed forest stands due to the low Pb concentrations. Nevertheless, in locations more heavily burdened by lead deposition, it would be appropriate to minimize mechanical interventions into organic horizons. In this observational experiment, with a relatively short time since the stand had been harvested, the effect of soil disturbance probably prevailed over the removing/spreading effect of logging residues. The longer-term impacts of logging and different clearing management may yet become apparent. Research at selected locations will therefore continue and aim at a more detailed assessment of qualitative changes in soil organic matter, which also determine the behaviour of PTEs in forest soils.

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