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Interactions between metals and soil organic matter in various particle size fractions of soil contaminated with waste water

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A B S T R A C T

Only scarce field studies concern the consequences of natural soil organic matter (SOM) and metal interactions on SOM dynamics in soils. We investigated the interactions of four metals (Pb, Zn, Cu and Cd) with the SOM associated to five different size fractions (between 2000 μm and $<2 \mu\text{m}$) of a sandy top soil contaminated by waste water. Metal, organic carbon and nitrogen concentrations were measured and chemical extractions (with $\text{Na}_4\text{P}_2\text{O}_7$ and EDTA) were also performed to assess the variations of SOM–metal interactions irrespective of the size fraction. In addition, as in that selected contaminated site, maize (C_4 plant), replaced C_3 crops 15 years ago, natural isotopic ^{13}C labelling gave new insights into SOM turnover. First, the results suggest that metals influence the SOM dynamics in that sandy soil: a C_3 “old carbon” enrichment was observed in the small or clay size fractions, while the “new” C_4 carbon associated with sandy soil particles presents a rapid turnover. Metal accumulation in the clay fraction is attributed to particulate organic matter (poorly associated) and SOM decay which overtime accumulated metals and eventually these metal–SOM associations prevent the biological decomposition of such carbon pools. Moreover, the $\delta^{13}\text{C}$ signals, C/N ratios and results from chemical extractions clearly showed differences in the origin, nature and reactivity of the SOM as a function of the size fraction with consequences on the metal behaviour. Differences were observed between metals studied: Zn seems to be mainly bound to SOM associated with clay particles, while Pb seems to prefer to interact directly with the mineral surfaces versus the SOM.

Keywords:

Soil organic matter turnover

Stable isotopic carbon ratio

Trace metal behaviour

Size fractionation

Waste water

1. Introduction

Due to various anthropogenic activities, potentially toxic metals are accumulated in soils, with a risk of water and biota contamination (Alloway, 1995). The design of pertinent soil criteria for environment protection and remediation relies on an understanding of the mechanisms controlling metal behaviour (Cecchi et al., 2008; Arshad et al., 2008). Among the various reactive soil constituents, soil organic matter (SOM) has a large sorption capacity towards metals (Yin et al., 2002). The metal–SOM interaction has various and complex consequences both on the solubility, mobility and bioavailability of metals (Impellitteri et al., 2002) and on SOM turnover (Boucher et al., 2005).

Generally, solid phase SOM is associated with retention, decreased mobility and reduced bioavailability of trace metals (Sauvé et al., 2000). But cationic metals that would ordinarily precipitate at the pH values of most soils are sometimes maintained in solution through complexation with soluble organics (Ferrand et al., 2006). Furthermore, metals participate and/or affect biogeochemical cycles in soils and influence the soil biota (Barajas-Aceves et al., 1999).

Reactive soil components are obviously in interaction (Dumat et al., 1997) and SOM consists of various components with a wide range of turn-over times, weight, reactivity and chemical nature: therefore the way such interactions are studied is complex (Woomer et al., 1994). The studies concerning one extracted soil component are open to criticism, conversely, methods of soil particle-size fractionation can be used to assess the distribution of metals in soils (Ducaroir and Lamy, 1995) and provide information on the SOM behaviour.

The metal distribution among the different soil components can be estimated using specific chemical reagents in order to discriminate the total metal content over various operationally defined fractions in the solid phase (Tessier et al., 1979; Dumat et al., 2001). For instance, sodium pyrophosphate ($\text{Na}_4\text{P}_2\text{O}_7$) is used to dissolve SOM:

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pyrophosphate complexes the metals which stabilize the SOM and the hydrogen bonds between OM and minerals are broken (Dumat et al., 1997). In contrast, EDTA is a powerful but unspecific complexing agent: the metals adsorbed both by SOM and oxides are extracted (Miller et al., 1986). The comparison of Na₄P₂O₇ and EDTA metal extractions should improve the estimation of metals specifically linked to SOM.

In soils where the vegetation changes from C₃ to C₄ (or vice versa), the variations in the natural abundance of ¹³C (¹³C/¹²C ratio) in SOM over time can be used to identify organic carbon sources in the soil (Schwartz et al., 1996) or to determine the SOM turnover rate (Balesdent and Mariotti, 1996). Using that tool, Dumat et al. (2006) highlighted the influence of metals on the SOM turnover on pseudogley brown leached soils strongly contaminated by inorganic atmospheric fallout from a smelter. However, only scarce field studies concern the consequences of natural SOM–metal interactions on SOM dynamics: supplementary studies in various pedologic and pollution contexts are therefore needed.

In the present work a contaminated sandy top soil under maize that had received waste water applications was used to investigate the interactions between four metals (Pb, Cd, Zn and Cu) and SOM. The tested hypothesis throughout these experiments was a significant influence of organic matter–metal interaction both on SOM dynamics and metals fate.

2. Materials and methods

2.1. Soil sample preparation and soil properties

The studied sandy luvisols, in the north of the Paris basin, has been precisely described by Lamy et al. (2006). Moreover the site management history is widely exposed in several publications (Bourennane et al., 2006; Lamy et al., 2006; Dère et al., 2006, 2007): wastewater irrigation of garden markets occurred from 1899 to 2002 (average 40,000 m³ ha⁻¹ year⁻¹).

In comparison with experiments performed in controlled laboratory conditions, only few study dealing with organic matter–metals interactions concern field experimentations based on concrete case. Our study offers an original experimental design from this point of view. The studied site has been considered only after metal contamination has been noticed, which means: several years after that the spreading has been forbidden, thus the waste water was not available for analysis. However Dère et al. (2006, 2007) observed that metals concentrations in the sandy topsoil have increased by tenfold compared to the pedogeochemical background and concluded to a soil metal pollution due to waste water spreading.

Field sampling was oriented to collect representative soil samples weighing 40 kg in closed bags. After mixing, all the samples were air dried and sieved to exclude particles larger than 2 mm to give the bulk soil. Thereafter trace metal, carbon and nitrogen concentrations were measured. The cation exchange capacity (CEC), determined by the ammonium acetate method at pH 7 (Metson's method, AFNOR NF X 31130) was 4.9 Cmol(+) kg⁻¹ and the pH_{water} = 7.1.

2.2. Physical fractionation of the various soil size fractions

Before particle size fractionation, an initial physical dispersion of the soil samples was performed by end-to-end shaking overnight in pure de-ionised water in order to minimise the potential variations of metal speciation that could occur with chemical dispersion. The solid/solution ratio was 1/5 (m/V) with the use of glass balls favouring the soil dispersion. The suspensions were then sieved through a series of grids (AFNOR French reference) with decreasing mesh size (2000, 200 and 50 µm). Then the fractions (20 µm < size < 50 µm) and (2 µm < size < 20 µm) were obtained by sedimentation at normal gravity (g), during times determined by Stokes' law. The <2 µm

fractions were finally obtained after concentration of the suspensions by centrifuging at 39,200 g for 1 h at 15 °C on a Beckman Avanti J301 apparatus. All the particles recovered were air dried at 60 °C, weighed, ground (200 µm) and homogenised.

2.3. Measure of metal concentrations and chemical extractions

The total metal concentrations in the various soil size fractions were analysed by inductively coupled plasma atomic emission spectrometry (ICP-AES) on a Jobin Yvon JYESS 4.03 apparatus, after triple acid digestion (HF, HCl and HNO₃) of the solid samples. The digestion protocol and measurement accuracy were checked by reference to a known standard soil (GRX2). It indicated the efficiency of the digestion protocol and of the ICP-AES assay (within ±1%). All the results correspond to the mean of two replicates.

Two extractions were performed on separate sub-samples. The Na₄P₂O₇ extractions were performed with the procedure of Vieiriae et al. (1993). The contact time was 16 h, with 0.1 M Na₄P₂O₇ at pH = 10: maximum efficiency is for pH > 9 (Asher and Bar-Yosef, 1982). The EDTA extraction conditions were chosen such that EDTA was in excess at equilibrium (Gesthem and Bermond, 1998). The contact time was 24 h, with 0.05 M EDTA at pH = 6.5. The various solid phase fractionations were performed at room temperature (20 °C), with a solid/solution ratio of 1/10, with two replicates per sample. In order to obtain a solution free of solid material before filtration, a strong centrifugation (50,000 g) was performed on the Na₄P₂O₇ extracts (20,000 g for EDTA extracts). After each extraction, the soil was rinsed with deionised water to limit precipitation of metals and to improve the extraction efficiency. The extracts and rinses were analysed after filtration (0.2 µm), for metals using ICP-AES. Analyses of soil extractions were calibrated using standard solutions made with the extracting agents.

2.4. Amounts of organic carbon and stable carbon isotopic signals (δ¹³C)

The ploughed contaminated topsoil (0–30 cm), which has been under maize (C₄ metabolism with δ¹³C from –10 to –12‰) for 15 years, was previously used for garden markets cultures, presenting a C₃ metabolism (δ¹³C from –27.6 to –30.5‰). That C₃–C₄ sequence allows natural labelling of organic matter in relation with the dynamics of the carbon sources (Bender, 1971; O'Leary, 1981; Collister et al., 1994). Organic carbon and nitrogen contents of the soil samples were determined by dry combustion in a Carlo Erba NA 1500 CHN elemental analyser. Stable carbon isotope ratios of SOM and plants were measured on a Fisons SIRA 10 Isotope Ratio Mass Spectrometer. ¹³C natural abundance was expressed in δ units, by reference to the international standard PDB (marine carbonate: belemnite from the PEE Dee formation) (Craig, 1957), according to the following equation:

$$\delta^{13}\text{C}\text{‰} = \frac{[\text{^{13}C/^{12}C}]_{\text{sample}} - [\text{^{13}C/^{12}C}]_{\text{PDB}}}{[\text{^{13}C/^{12}C}]_{\text{PDB}}} \times 1000.$$

All samples were sieved to 1 mm, ground to 100 µm and homogenised. Organic carbon, nitrogen and δ¹³C were measured in duplicate, with previous elimination of carbonates by acidification. According to Balesdent and Mariotti (1996) and Boutton (1996), mean δ¹³C (‰) reference values for carbon isotopic composition are around –27 for C₃ plants (mainly trees, rice, wheat, certain grass species), –14 for C₄ plants (mainly maize, tropical grasses and certain other grasses) and –8 for current atmospheric CO₂.

2.5. Statistical data treatment

Data obtained were subjected to analysis of variance (ANOVA) with one factor, using the software Statistica, Edition '98 (StatSoft Inc., Tulsa,

Table 1Metal concentrations (mg kg⁻¹), organic C (%), N (%) and C/N of the various size fractions of the contaminated sandy top soil

Soil size fraction	Concentration (mg kg ⁻¹)				% C/N		
	Pb	Zn	Cu	Cd	C	N	C/N
2000–200 μm	43.2±5	258±19	32±11	0.06±0.02	1.26±0.075	0.04±0.005	31.5
200–50 μm	94.2±7	429±16	93.6±15	0.46±0.05	1.24±0.05	0.07±0.005	17.71
50–20 μm	340±22	1893±98	296.7±21	4±0.2	3.4±0.1	0.21±0.015	16.33
20–2 μm	749±54	4112±165	429±33	13.15±0.3	5.5±0.3	0.42±0.03	13.12
<2 μm	1560±19	7688±73	1267±94	22.5±2	7.2±0.3	0.67±0.05	10.75

OK, USA). For each bioassay, mean values with different letters represent a significant difference ($p < 0.05$) as measured by the LSD Fisher test. Letters are reported on the figures.

3. Results

3.1. Amounts of trace metals, carbon and nitrogen in the various size fractions

For the bulk topsoil sample: (i) trace metal concentrations (mg kg⁻¹) were [Pb]=142, [Zn]=1233, [Cu]=68 and [Cd]=1.8; (ii) organic carbon (%) and nitrogen (%) concentrations are respectively 1.75 and 0.15, with a C/N value of 11.7. These results are given as the means of the two replicates and the relative standard deviation never exceeded 7%. Compared with the natural pedogeochemical background values ([Pb]=5.9 mg kg⁻¹, [Zn]=13.4 mg kg⁻¹, [Cu]=3.9 mg kg⁻¹ and [Cd]=0.1 mg kg⁻¹), our results indicate a significant contamination of the top soil (Lamy et al., 2006).

Particle size fractionation of soil samples resulted in the recovery of 98±0.5% of the initial sample mass. The size distribution observed was: 42.3% (2000 μm > fraction > 200 μm), 40.4% (200 μm > fraction > 50 μm), 5.4% (50 μm > fraction > 20 μm), 4.5% (20 μm > fraction > 2 μm) and 7.4% fraction < 2 μm i.e. the sand fractions (fine and coarse) accounted for 80% of the total weight.

The results of total metal and organic carbon concentrations in the different soil particle size classes are given in Table 1. Whatever the metal considered, the concentration strongly increased as the size decreased according to Harter and Naidu (1995): 30–40 times for Zn, Pb and Cu and 375 times for Cd. The clay fraction is also that with the highest levels of organic carbon and nitrogen. The concentrations measured increased from 1.3% C and 0.04% N in the 2000–200 μm fraction to 7.2% C and 0.7% N in the clay fraction. The proportion of the different soil particle size classes was multiplied by the concentration of the metals in each class to give the actual quantity of each metal in

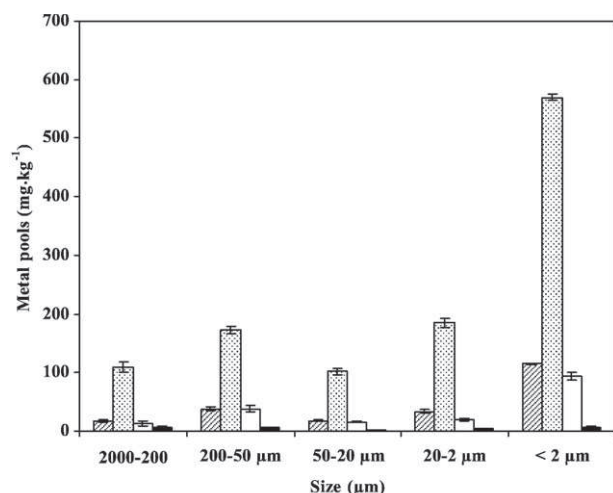


Fig. 1. Metal pools (Pb: ▨, Zn: ▩, Cu: □ and Cd: ■) in the different size fractions of the topsoil.

each size class (Fig. 1). The determination of these different metal pools demonstrated the relevance of focussing both on the clay fraction and on the sum of the different coarse fractions in order to study the SOM–metal interaction.

3.2. Chemical extractions

The results of the chemical extractions performed on the sandy top soil size fractions for Zn (A), Cu (B), Pb (C) and Cd (D) are reported both in Fig. 2: quantities of metals extracted (mg of the metal per kg of soil fraction) by EDTA or Na₄P₂O₇ and in Fig. 3: ratios of the quantities of metals extracted by EDTA or Na₄P₂O₇ divided by the total metal content of that size fraction. The data of Fig. 2 indicate that the quantities extracted per unit weight of the fraction increased with decreasing particle size. However, for the ratios (Fig. 3), the reverse trend was generally observed, in particular with Na₄P₂O₇ extraction: a large proportion of metals extractable from the coarse fraction and in the 20–2 μm fraction.

In the clay fraction, more Zn and Cu were extracted with Na₄P₂O₇ than with EDTA (Fig. 2). Moreover, higher proportions of Zn and Cu were extracted by Na₄P₂O₇ than Cd and Pb (Fig. 3) suggesting two different patterns depending on the metal.

3.3. Use of the δ¹³C signal to study the organic matter dynamics in relation with metals

Fig. 4 gives: (a) δ¹³C and (b) C₄/C₃: ratio between carbon of C₄ origin and total organic carbon for the different size fractions (Balesdent and Mariotti, 1996; Dumat et al., 2006). The value of δ¹³C for the topsoil was between –21.8‰ for the coarse fraction and –23.2‰ for the fine fraction. No δ¹³C values for organic matter from waste water are available from the literature, however a δ¹³C signature of –25.4‰ measured for anaerobically digested sludge from South West France (Parat et al., 2007) could give a general idea. In fact, the quantity of organic carbon in waste waters is generally very low compared to the carbon inputs from plants, therefore the influence of the organic matter from waste waters on the δ¹³C signature of the SOM can be ignored considering the much greater inputs of organic matter from C₃ and C₄ plants.

4. Discussion and conclusion

4.1. SOM and metal interaction

The comparison between EDTA (a powerful but unspecific metal chelating agent) and Na₄P₂O₇ (which mobilises metals bound to SOM) (Fig. 2) suggests that a significant part of all the metals is bound to the SOM. That hypothesis is moreover supported by total concentrations results: carbon from the clay fraction represented more than 25% of the total soil carbon and the highest proportion of metals was also Na₄P₂O₇ extracted from this fraction. For lead and zinc, two metals with high total concentrations in the sandy top soil, the following empirical equations between metal concentrations (expressed in mg kg⁻¹) and organic carbon concentrations (expressed in percent) were determined: [Pb]=228.59×[C]–294.93 with a correlation coefficient (r) of 0.96 and [Zn]=1150.4×[C]–1412.6 with $r=0.95$.

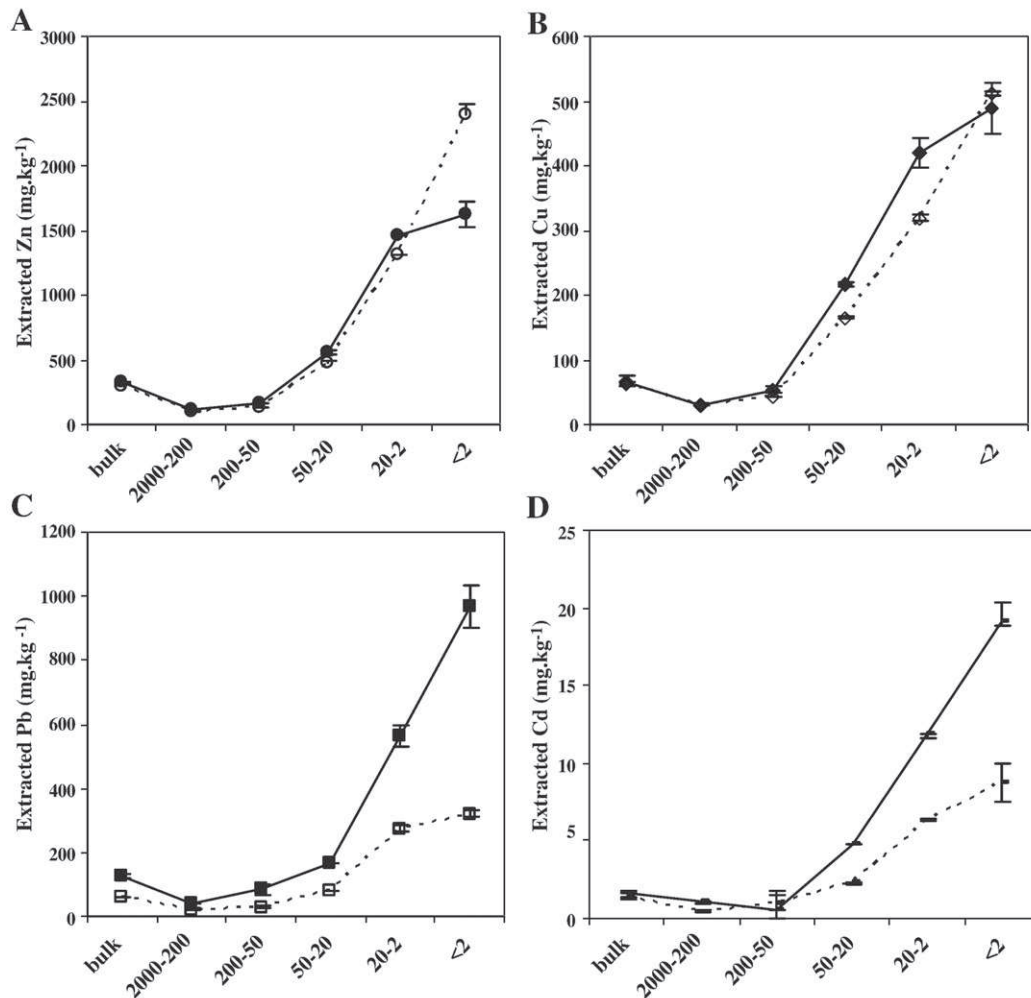


Fig. 2. Chemical extractions performed on various size fractions of the sandy topsoil. Quantities of metals extracted (mg of the metal per kg of soil fraction) by EDTA or Na₄P₂O₇. A) Zn-EDTA: ● and Zn-Na₄P₂O₇: ○; B) Cu-EDTA: ◆ and Cu-Na₄P₂O₇: ◇; C) Pb-EDTA: ■ and Pb-Na₄P₂O₇: □; D) Cd-EDTA: ▲ and Cd-Na₄P₂O₇: △.

However in relation to SOM influence, two different behaviours were observed for the four studied metals. For Zn and Cd, the quantities extracted by EDTA and Na₄P₂O₇ were fairly similar (except for Zn in the case of clay fraction: Na₄P₂O₇>EDTA) suggesting that these two metals were preferentially linked to the SOM (Fig. 2). In contrast, for lead and cadmium, the quantities extracted by EDTA were significantly higher than the quantities extracted by Na₄P₂O₇; other soil components (like clay minerals or oxides) significantly interact with these two metals (Dumat et al., 2001; Cecchi et al., 2008).

The general trend observed was that of a decrease of metal proportions extracted by Na₄P₂O₇ with the decrease of the soil particle size: the metal extraction yield was greater in sand and in the 20–2 μ m fractions compared with the clay fraction (Fig. 3). The efficiency of Na₄P₂O₇ could be influenced by the nature of the SOM and by the interactions between SOM and other soil components. Indeed, with decreasing particle size, Table 1 shows a decrease in C/N ratio, in agreement with a change in the chemical nature of the SOM (Dormaer, 1984; Besnard et al., 2001). Two different pools of metals with contrasting behaviours (in relation with the SOM characteristics) could be present in the soil: metals in the coarse fractions and metals in the clay fraction.

The coarsest fraction was mainly composed of sand and poorly degraded particulate organic matter (POM) (Cambaradella and Elliott, 1992). Minerals in the 2000–20 μ m fraction were relatively inert, so the interaction between these minerals and metals should mainly be weak such as electrostatic bonding; POM (with high specific surface area) may therefore play a significant role (Besnard et al., 2001).

Sequential extraction performed by Dère et al. (2007) on the bulk fraction of the same sandy topsoil ranked lability (potential to be remobilized from solid phase to soil solution) as: Zn \gg Cu > Pb > Cd. Doelsch et al. (2006) also observed the high lability of Zn in sewage sludge. In the coarse fractions of the soil, the metals associated with POM will therefore mainly be under relatively labile forms.

During biodegradation, the level of SOM oxidation increases (Baldock et al., 1997), leading to an increase of its reactivity towards metals. Since the level of humification increases as particle size decreases, SOM from the fine fraction constitutes a highly reactive pool for binding metals. Moreover, the metals arrived in the sandy top soil simultaneously with organic matter from the waste water, the metals could therefore have already been associated with dissolved organic matter (Vaca-Paulín et al., 2006; Doelsch et al., 2006). In addition, Wong et al. (2007) concluded that dissolved organic matter interacts with the clay minerals becoming less accessible to microorganisms. Due to its higher stability against microorganism degradation, the SOM of the clay fraction could have a stabilization effect on the metals and reduce their bioavailability (Dabkowska-Naskret, 2003).

4.2. Use of the $\delta^{13}\text{C}$ signal to study organic matter dynamics in relation with metals

The measurement for the topsoil (under maize for 15 years) of $\delta^{13}\text{C}$ (‰) between –21.8 (coarse fraction) and –23.2 (clay fraction), whereas the value of –12.5‰ has been measured on a maize sample

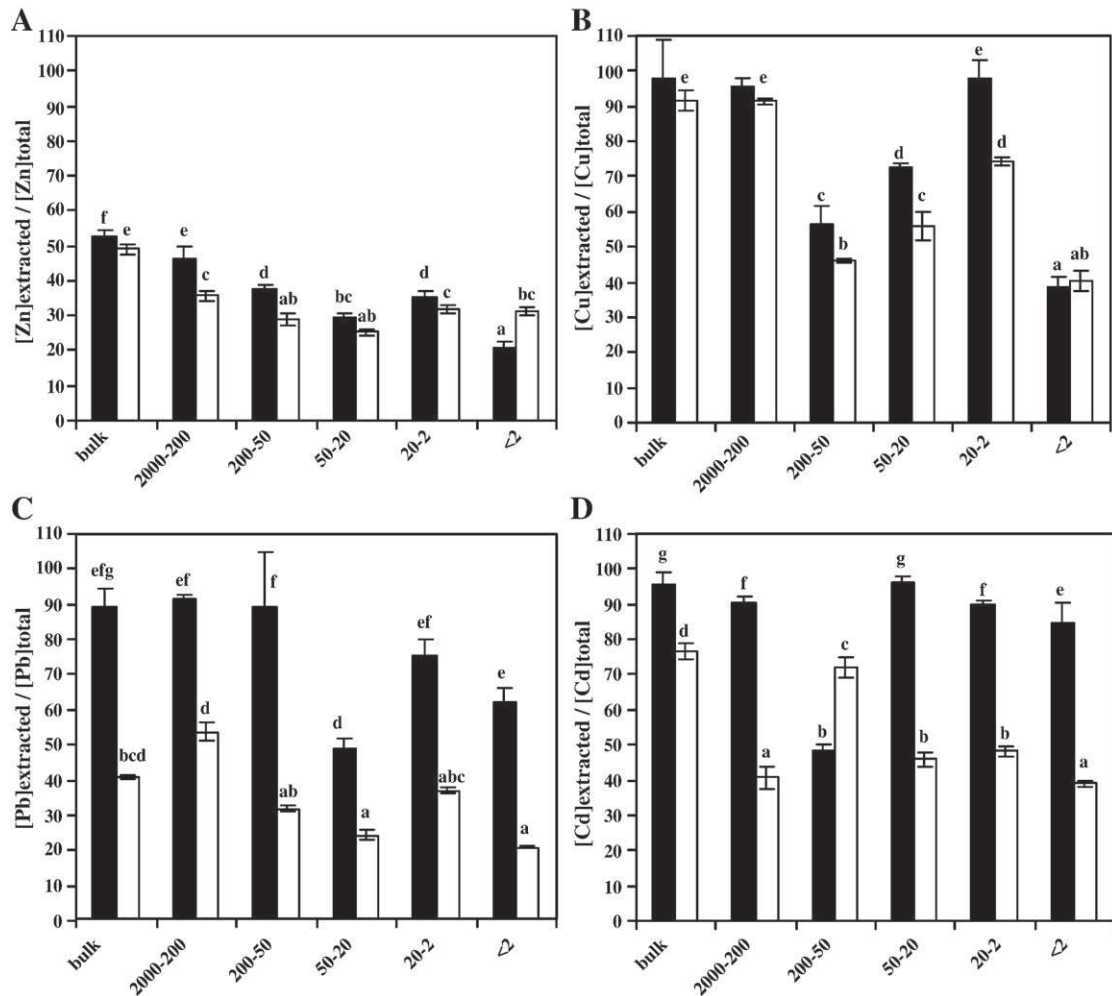


Fig. 3. Chemical extractions performed on various size fractions of the sandy topsoil. Ratios (%) of the quantities of metals extracted by EDTA (■) or Na₂P₂O₇ (□) divided by the total metal content of that size fraction. A) Zn, B) Cu, C) Pb and D) Cd.

composed of crushed leaves and stalks by Jolivet et al. (2003), indicates that the SOM is relatively unaffected by the maize $\delta^{13}C$. From the C₄/C ratios calculated, the amount of SOM of C₄ origin was seen to be under 40% and decreased with the soil size fraction: 0.4 for the 2000–200 μm fraction and 0.3 for the clay fraction. Balesdent and Mariotti (1996) or Balesdent et al. (1987) estimated the residence time of carbon in size fractions of different uncontaminated soils under maize for 13 years: the expected proportion of new C₄ carbon from maize in the coarse fraction (2000–200 μm) of these uncontaminated soils was 80%. Balesdent and Mariotti (1996) concluded that turnover times were 4 ± 1.5 years for the 2000–200 μm fraction and 13 ± 2 years for the 200–50 μm fraction. In contrast all fractions finer than 10 μm exhibited a slow turnover. Our results are therefore consistent with a preferential accumulation of old C₃ organic matter in the polluted soil, particularly in the clay fraction. Even in heavily metal contaminated soils, biological activity occurs (Valsecchi et al., 1995; Aoyama and Nagumo, 1997), but alterations of organic matter decomposition are reported (Berg et al., 1991; Cotrufo et al., 1995).

In comparison with more contaminated “old” C₃ plant residues in the clay fraction, the “new” C₄ maize residues left in the soil (after the harvest) were mineralised more quickly. The POM is poorly associated with the soil mineral matrix and has a relatively rapid turnover in comparison with the finest SOM in the clay fraction (Cambaradella and Elliott, 1992; Balesdent, 1996). Parat et al. (2007) suggest an accumulation of plant residues in the coarse fraction of a sludge-contaminated soil by physical protection as the sludge could enhance

aggregate formation. However, the topsoil studied here had been intensively used for crops for several decades and its sandy texture limits the physical protection of SOM (Guggenberger et al., 1995). Moreover, the soil we studied received mainly waste water amendment, so the effect on aggregation was limited.

Some metals are progressively released during the decay of C₃ POM. Then, part of the metals was adsorbed by the reactive colloidal organic matter in the clay fraction (Besnard et al., 2001; Boucher et al., 2005). According to Brookes (1995) or Mhatre and Pankhurst (1997), decreased organic matter decomposition in polluted soils could result from the inhibition of microbial activity by bio-available toxic metals. However, Hattori (1996) concluded that the recalcitrance of plant residues towards microorganism degradation should also be considered. The formation, in the clay fraction, of organo-metallic complexes relatively resistant to biodegradation (Orlov et al., 1988) could therefore result. This hypothesis is consistent with the lack of accumulation of highly contaminated coarse plant residues in the top soil i.e. the microorganisms degrade the fresh organic matter (relatively free of metals and not protected by clay minerals), but tend to leave the old C₃ organic matter residues stabilized by chemical and physical phenomena.

Finally from the results obtained in the present study, it can be concluded that the presence of metals carried by waste water modified the turnover rate of the SOM in the contaminated sandy top soil. A decrease of SOM turnover due to the influence of metals was previously observed for pseudogley brown leached soils strongly contaminated by inorganic atmospheric fallout from a smelter (Dumat

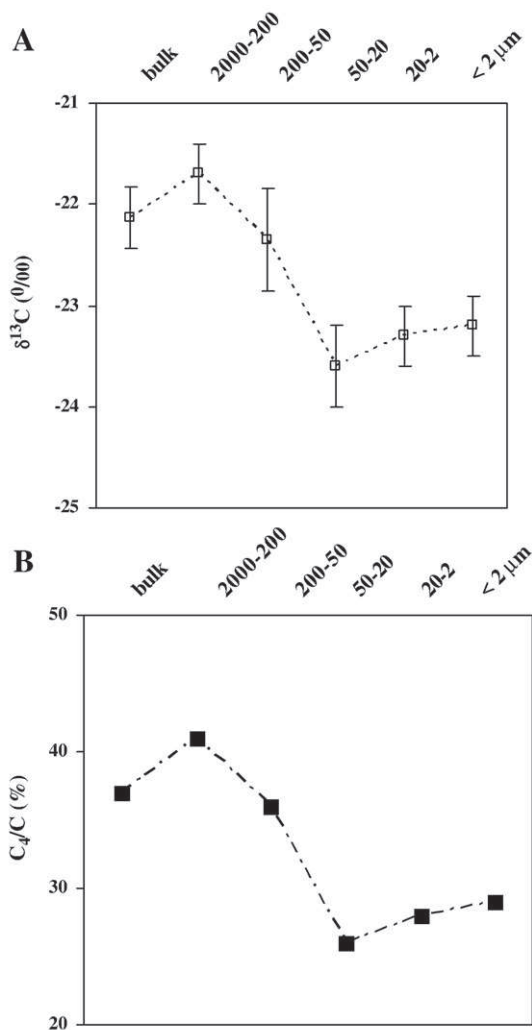


Fig. 4. (a) $\delta^{13}\text{C}$ and (b) C_4/C ratios (%) of the topsoil for the different size fractions.

et al., 2006), i.e. in a very different context. Decreased SOM turnover could therefore be considered as a general phenomenon to take into account when studying carbon sinks and organic matter turnover in polluted soils.

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