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Impact of pig slurry and green waste compost application on heavy metal exchangeable fractions in tropical soils

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ABSTRACT

Six soil/waste samples were incubated at 28 °C under standard conditions to assess the impact of organic waste applications on heavy metal (HM) exchangeable fractions. The three studied soils are representative of the pedological diversity that prevails in Réunion and characterized by high natural total Cr, Cu, Ni and Zn concentrations. For organic waste, we tested two products that differed in their chemical composition, physical state and different HM contents, i.e. pig slurry (PS) and green waste compost (CT), which were incorporated at dosages consistent with the actual agricultural setting: 350 and 1000 kgN ha^{-1} (subsequently called 350 and 1000). Variations in pH and electrical conductivity (EC) were monitored during incubation, while also measuring carbon mineralization and HM exchangeable fractions (Cr, Cu, Ni and Zn). The physicochemical parameter variation patterns were not identical in the three soils. The greatest variations were noted in the cambisol: pH reduction and EC increase. Conversely, in the andosol, there were only slight variations in pH and EC during incubation. The C–CO_2 quantities detected were derived from PS mineralization, and were always higher than those from the mineralization of CT, irrespective of the soil considered, and C mineralization was proportional to the initial quantity of incorporated waste (350 vs 1000). We explained the difference in the andosol mineralization coefficient between the 350 and 1000 dosages as being the result of an interaction between the organic constituents and the pseudo-cristallized aluminosilicates from the soil (allophane and/or imogolite), which limited degradation of the incorporated organic matter (OM). Despite organic waste mineralization, the pH variations recorded during the incubations and the increased salinity, we did not detect any increase in the HM exchangeable fractions, but instead noted a decrease in the Cu exchangeable fraction (soil + PS incubation), which could be explained by complexation of this element by the OM.

1. Introduction

Resources must be recycled to ensure sustainable development, and this especially applies to organic wastes of agricultural, municipal and agroindustrial origins (Clapp et al., 2007). Organic wastes may be used as fertilizer and soil amendments, renewable energy (liquid fuels, electricity) or the production of chemicals (volatile organic acids, alcohols) (Westerman and Bicudo, 2005). From a waste management standpoint, agricultural recycling is a way of making effective use of organic waste of different origins, e.g. agricultural (slurry, manure, guano (Moreno-Caselles et al., 2002)), urban (compost, sewage sludge (Doelsch et al., 2006a)) and agroindustrial (vinasse from fermentation industries (Tejada and Gonzalez, 2005)). Agricultural recycling can give rise to products that have a fertilizer value paralleling that of standard fertilizer (Madejon et al., 2001), while improving the soil–organic matter (OM) quantity and quality (Rivero et al., 2004), enhancing the physical properties of cultivated soils, e.g. the texture, porosity and infiltrability (Zebarth et al., 1999; Pascual et al., 1999; Albiach et al., 2001), and increasing soilborne microflora and fauna activity (Emmerling et al., 2000; Saison et al., 2006). Although the use of organic waste in agriculture has many benefits, there are several factors that significantly limit the applicability of this strategy. These factors were listed and discussed by Westerman and Bicudo (2005). They include: acceptable integration into agriculture, quality control of organic wastes being applied, economic viability, logistics and organization, public acceptance (nuisance or environmental concern) and environmental regulations. The agronomically efficient and environmentally safe use of any organic amendment requires adequate control of the chemical quality of the added OM (Doelsch et al., 2009) but also of the quantities and

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dynamics of nutrients (Feder and Findeling, 2007) and potential pollutants (Doelsch et al., 2006a). Among these pollutants, heavy metals (HM) have been closely examined due to their toxicity to humans, animals and plants. Several authors have documented Cu and Zn contamination of cultivated soils following application of pig slurry (Coppenet et al., 1993; Nicholson et al., 2003; Novak et al., 2004) or compost (Ramos, 2006; Hargreaves et al., 2008). The application of sewage sludge and compost in crop fields can boost soil levels of Cd, Pb, etc. (Pinamonti et al., 1997; Genevini and Adani, 1997; McBride, 2003). Apart from problems of accumulation and induced risks of crop phytotoxicity, another main challenge in organic waste recycling is to ensure that there is no transient or disseminated pollution to the natural environment (McBride, 1999; Xue et al., 2003; McBride, 2003; Xue et al., 2005). After spreading organic wastes, the chemical and biological conditions of the medium can be substantially altered during mineralization of this waste through changes in the OM stock and dynamics, water flows, or in the physicochemical properties (pH, Eh, etc.) of the soil solution (Stevenson, 1994; Sanchez-Martin et al., 2007; Doelsch et al., 2009). All of these modifications affect HM speciation. Organic waste amendments can, for instance, increase the quantity of soil soluble OM that is able to complex HM (Stevenson, 1994) and the mobility of these complexes in the aqueous phase, thus facilitating their transport to the deep unsaturated zone and water table (Xue et al., 2003). Moreover, massive organic waste inputs can create reducing conditions (de Cockborne et al., 1999; de Cockborne et al., 2001; Doelsch et al., 2009), which could markedly alter HM speciation, especially the exchangeable fractions (Quantin et al., 2002). Several studies have thus focused on the impact of the mineralization of various organic wastes (compost, slurry, dung, manure) on HM speciation patterns during incubation experiments. For example, Guerra-Rodriguez et al. (2006) showed that biosolid application on soil (mollisol) increased the total Cu and Zn content. Moreover, the more labile Zn fractions increased after biosolid incorporation into the soil, while increasing the Zn bioavailability for test plants, but no significant differences were found for Cu, Cr, Ni, and Pb. Usman et al. (2004) showed that the addition of sewage sludge to a calcareous soil decreased the soil pH and increased the salinity and extractability of Cd, Cu, Ni and Zn, which could limit the use of this organic waste as an amendment in this kind of soil. Only a few studies have focused on heavy metal speciation in tropical volcanic soils (Doelsch et al., 2008), but even fewer have attempted to determine heavy metal speciation patterns after application of organic waste on tropical volcanic soils.

This study was aimed at determining whether organic waste mineralization modifies the speciation of HM naturally in the soil or derived from waste amendments. We thus studied the mineralization of two different organic wastes (pig slurry and compost) in contact with three soils representative of the diverse range of soil–climate conditions in Réunion (nitisol, andosol and cambisol):

(i) by assessing CO_2 production over a time course and monitoring variations in the physicochemical properties of the mixtures (pH, salinity, cation and anion exchange capacity);
(ii) by determining variations in the exchangeable fractions of four HMs (Cr, Cu, Ni and Zn) during incubation.

To ensure that the results would be as representative as possible of the actual agricultural conditions, dosages of organic wastes incorporated during mineralization were selected on the basis of the nitrogen input (350 and 1000 kgN ha^{-1}).

2. Materials and methods

2.1. Soil and organic waste characteristics

The three soils selected for the present study are representative of the soil–climate conditions on the tropical volcanic island of Réunion (Fig. 1). The first soil was a nitisol (Food and Agriculture Organization of the United Nations, 1998) that was sampled at La Mare in a lowland area along the northern windward coast. The second soil was an andic cambisol collected at Les Colimaçons in a medium elevation area on the western windward side of the island. The third soil was a silandic andosol collected at Sainte-Rose in a lowland area on the eastern side of the island. All soils were sampled in the 0–25 cm horizon (Table 1 shows the characteristics of the soils used for the incubations). A more complete description of the three soils can be found elsewhere (Doelsch et al., 2008). Only steel or plastic tools were used for sampling in order to avoid heavy metal contamination of soils. After air-drying and declumping the aggregates, the soil was sieved through a 2 mm mesh sieve. Two organic wastes from Réunion were used: pig slurry and compost. The slurry was sampled fresh from pig fattening slurry pits. The commercial compost was obtained from composting green wastes (municipal and garden wastes) after a 6-month composting period.

2.2. Soil incubations

An incubation experiment was carried out using three soils, two organic wastes and two input rates. The amount of fresh pig slurry and fresh compost added was equivalent to 350 kgN ha^{-1} which corresponded to the legally recommended dose and 1000 kgN ha^{-1}. Indeed, until the recent order of 02/07/05 (NOR: DEVP0540077A), regulations stipulated that total applications had to be less than 350 kgN/ha/year for forage and pasture crops, or 200 kgN/ha/year for other crops. The soil samples, equivalent to 50 g of dry soil and corresponding amounts of organic waste, were mixed thoroughly and then incubated in the dark in a temperature controlled incubator at 28 °C for 28 days. Deionized water was added to the soils in order to bring their moisture content to 70% of the water-holding capacity. The soil moisture was controlled by weighting, and readjusted if necessary, during the experiments. To ensure that organic waste decomposition was not limited by the experimental conditions, inorganic N was added to the soil in the form of KNO_3 to obtain an initial concentration of 35 mg N kg^{-1} dry soil. Each experiment included a treatment without organic waste (control soil). Eight replicates were performed for each treatment and each sampling date. The C-mineralization dynamics were determined with four replicates per treatment. The control and amended soils were placed in closed vessels. The CO_2 formed during the incubation period was trapped in 10 ml of 1 M NaOH in small tubes, which were placed on the top of the soil, inside the incubation vessels. Empty vessels were used as blanks. After 1, 3, 7, 14, and 28 days, CO_2 was measured by titration of the NaOH solution with 1 M HCl and excess $BaCl_2$.

2.3. Extraction procedure

Variations in the exchangeable HM fractions during incubation were determined with four replicates per treatment. The control and amended soils were placed in open vessels to avoid anaerobic conditions. After 1, 3, 7, 14 and 28 days, the samples were freeze-dried to stop OM mineralization and kept in a dark room before extraction and analysis.

The first step of the sequential extraction procedure proposed by the Geological Survey of Canada (Hall et al., 1996) was applied to the soil and waste samples in order to characterize the salt-extractable HM fractions (subsequently called exchangeable fractions) and their patterns during incubation.

One gram of dry soil or organic amendment sample was introduced in a 50 ml screw-cap centrifuge tube (PPCO). The solid phase was shaken with 30 ml of 0.1 mol l^{-1} $NaNO_3$ for 90 min using a rotary shaker at 16 rpm. The solution was separated by centrifugation (30 min at 12 000 rpm) and the supernatant was stored in a 250 ml bottle (HDPE). The whole procedure was repeated on the