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Research article

Mobility of heavy metals in sandy soil after application of composts produced from maize straw, sewage sludge and biochar

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ABSTRACT

Studies on the availability of heavy metals in composted organic materials and in soil amended with these materials are of practical significance. They are used in the assessment of the purity of the soil environment and of the biological value of plants intended for human and animal consumption. Composting of organic materials has a significant effect on changes in mobile forms of heavy metals. Therefore, the aim of this study was to determine the effect of the addition of biochar and sewage sludge on (i) the contents of water soluble forms of Cu, Cd, Pb, and Zn in composts; and (ii) the contents of mobile forms of these elements in sandy soil after the addition of composts. Addition of sewage sludge and biochar to maize straw did not increase the heavy metal forms extracted with water in total content of heavy metals. The content of Cd and Cu extracted with water in composts produced from maize straw and sewage sludge, and produced from maize straw, sewage sludge and biochar was higher than the one determined in compost produced from maize straw. The content of Pb and Zn extracted with water in compost produced from maize straw, sewage sludge and biochar was lower than in compost produced from maize straw. The addition of sewage sludge and biochar to maize straw had an immobilizing effect on mobile forms of the studied elements compared to compost produced from maize straw and sewage sludge. The addition of composts to soil decreased the contents of mobile forms of Cu, Cd, and Pb extracted with 1 M NH_4NO_3 compared to the contents in the control soil. However, the content of Zn extracted with NH_4NO_3 increased in treatments with 0.5% dose of compost produced from maize straw and sewage sludge and 0.5% dose of compost produced from maize straw, sewage sludge and biochar. In none of the analyzed cases, the application of the composts produced did not exceed the acceptable content of studied elements in the soil.

1. Introduction

Composting of organic waste is an aerobic process resulting in mineralization and humification of organic matter. The process should give a stable product with no phytotoxic properties, safe due to its hygienic quality. Furthermore, the product should have the characteristics of humic compounds (Bernal et al., 2009). In general, it can be assumed that simple carbon compounds are readily mineralized and metabolized by microorganisms at an early stage of the composting process, which contributes to the formation of CO_2, NH_3, H_2O, organic acids and considerable amounts of heat (Sánchez-García et al., 2015). Composting is often regarded as an environmentally friendly process aimed at disposing of waste containing organic matter (Larney and Hao, 2007; Tandy et al., 2009; Gondek et al., 2014). However, reports from the literature show that, under certain conditions, composting may pose a significant environmental burden. The results obtained by Amon et al. (2006) show clearly that such risks actually exist. The authors indicated that composting of manure is a source of greenhouse gases, ammonia, hydrogen sulfide, and volatile organic compounds. The selection of components affects the dynamics of changes, including gas emissions during the composting process (Kopeć et al., 2015; Czekała et al., 2016).

A criterion limiting the natural use of composts is the content of undesirable substances, such as heavy metals. This problem applies especially to waste composts which, due to their origin (e.g. municipal sewage sludge), are contaminated with heavy metals. Many authors point to the fact that the total content of heavy metals is not a good indicator of their availability to plants (Cai

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et al., 2007). It should be noted that the organic matter content is significantly reduced during the composting process as a result of microbiological degradation. Consequently, the structure of organic compounds is also modified, which often changes the properties of composted biomass (Bernal et al., 2009). The intensity of changes in composted biomass is varied and determined by the nature of organic compounds and the availability of nutrients to microorganisms. According to Neklyudov et al. (2008), the rate of degradation of organic matter decreases gradually over time of composting. This is due to the reduction of readily available sources of carbon and increase of polymerized organic compounds.

Degradation of organic compounds and synthesis of new organic connections can significantly alter relations between not only C and N, the content of humic and fulvic acids, but also between total and mobile forms of heavy metals in composted materials. He et al. (2009) and Martinho et al. (2015) concluded that the content and speciation of heavy metals in composted sewage sludge are a major cause of the negative impact of these materials on the environment and health of living organisms. Degradation of organic compounds, resulting from mineralization may activate the initially inaccessible forms of heavy metals. This will increase their availability in soil and accumulation in plant biomass, which is one of the initial elements of the food chain (Gondek et al., 2014). Ingelmo et al. (2012) suggested that composting of organic materials increases the content of mobile forms of heavy metals. However, as previously mentioned, composting is not only a degradation of organic connections, but also a synthesis of humic compounds in the humification process (Kopeć et al., 2016). This process significantly increases the number of functional groups capable of binding heavy metals in composted biomass (Czekała, 2006; Zeng et al., 2014). Therefore, there is a need to search for materials that, once introduced into composted biomass, will increase the effectiveness of immobilization of heavy metals and make them more stable in soil after applying the compost. Biochar may be such a material (Czekała et al., 2016).

Results reported in the literature indicate that properties of biochar, such as porous structure and large specific surface area, may immobilize harmful organic and inorganic compounds (Park et al., 2011; Al-Wabel et al., 2015). However, the potential of biochar in reducing the content of heavy metal ions depends on the presence of surface functional groups (e.g., carboxyl, hydroxyl, or phenol). Assuming that the surface structure of biochar is diversified, the material can, as a potential sorbent, significantly affect the binding of heavy metals with functional groups, which results from exchange or complexation reaction and sorption on its surface (Paz-Ferreiro et al., 2014; Rees et al., 2014; Vithanage et al., 2014). However, as stated by Rees et al. (2014), apart from the application of biochar, pH also strongly influences immobilization of heavy metals. Numerous studies showed that both biological and chemical properties of soil change depending on the amount and type of organic matter applied to the soil, including biochar (Leifeld et al., 2002; Valarini et al., 2010; Houben et al., 2013; Meier et al., 2015; Mierzwa-Hersztek et al., 2016, 2017). These factors also determine the nature of organic and mineral connections due to the degradation of the complex organic component (Gondek and Mierzwa-Hersztek, 2016). A direct consequence of changes in these connections is altered bioavailability of heavy metals to plants (Houben et al., 2013; Paz-Ferreiro et al., 2014; Al-Wabel et al., 2015).

Apart from the commonly known benefits of using composts, there is still lack of information on the long-term effect of these materials on the soil environment. The great diversity of technological factors in the production of composts combined with a significant variability of environmental conditions can directly affect the conversion of composts in soil and thus the mobility of heavy metals. Consequently, the initial equilibrium between the presence and availability of heavy metals in soil will be impaired.

Composting of organic materials has a significant effect on changes in mobile forms of heavy metals. Therefore, the aim of this study was to determine the effect of the addition of biochar and sewage sludge on (i) the contents of water soluble forms of Cu, Cd, Pb, and Zn in composts; and (ii) the contents of mobile forms of these elements in sandy soil after the application of composts.

2. Material and methods

2.1. Materials and conditions of the composting process

Studies of organic material composting and its effect were conducted for 140 days (from mid-May to the end of September 2015). The process was carried out in $1.2 \times 1.0 \times 0.8$ m bioreactors with perforated bottoms to allow an active aeration of composted material. Laboratory bioreactors were sheltered against precipitation, but exposed to the outside temperature and sunlight. This ensured heat exchange between the composted material and the surrounding environment.

The scheme of the experiment included the following treatments (composts): control – shredded maize straw (M), shredded maize straw with municipal sewage sludge (M+SS), and shredded maize straw with sewage sludge and willow biochar (M+SS+BC). In terms of chemical composition and structure, the applied maize straw constituted and equivalent of grass and wood chips which are commonly used in industrial composting. Sewage sludge used in the study came from municipal wastewater treatment plant (mechanical and biological system) located in the Malopolska Province (southern Poland). Before sampling, sewage sludge was subjected to oxygen stabilization in separate open chambers in which continuous aeration was carried out at ambient temperature. The aeration process lasted for 5 days. After that period, sewage sludge was dewatered using a settling centrifuge. Biochar used in the study was produced from willow. Thermal conversion of willow was conducted by Fluid S.A. (Sędziszów, Poland). At an early stage of the process, shredded biomass (max 50 mm) with a humidity of approx. 45% was transported to a dryer in which its humidity was reduced to approx. 25%. Then, the biomass was transported to a reactor. Organic material was converted into biochar at 350 °C under a limited supply of air (1–2%) (IBI, 2012). According to the production technology of Fluid S.A., air supply was controlled by forcing the process gases to return to the reactor chamber.

Chemical composition of materials used in the composting process are shown in Table 1 and the proportions of feedstocks used in individual treatments are shown in Table 2.

The basic feedstock used in the composting process was shredded maize straw. After mixing the materials, the mixture humidity was equilibrated to 60%. Aeration of composted materials was performed in cycles, 6 times a day; air was flowing through the bioreactor at the rate of 15 dm^3 per minute for 60 min; composted materials were manually shifted every 10 days. The outside temperature and temperature of the composed materials (at half height of the composted matter) were recorded every 30 min using DT-171 data loggers (EMOS Si, d.o.o. Slovenija).

2.2. Chemical composition of materials and composts

The pH of fresh composts (material: water = 1: 5) was determined electrochemically using a pH meter (pH – meter CP - 505, Elmetron, Poland), electrical conductivity (material: distilled water = 1: 5) using a conductivity meter (Conductivity/Oxygen meter CCO - 501, Elmetron Poland) (Meier et al., 2015). Feedstocks and composts were dried at 105 °C for 12 h (Jindo et al., 2012), 1 mm