The Use of the Direct Test on Substrate in the Assessment of Phytotoxicity of Foundry Waste

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Research Article

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Abstract

The article presents the results of phytotoxicity tests on foundry dust and landfilled waste. Currently, all of this waste is being reused. The results supplement the previous study on the phytotoxicity of waste leachate. The research has focused on phytotoxicity tests performed directly on the waste. Watercress (*Lepidium sativum* L.) was used as the test plant. The germination test (GI) and the accumulation test were used to assess phytotoxicity. The results show that the dust from electric furnaces, classified as hazardous waste, was the most phytotoxic. Most of the dust samples inhibited germination and root growth. A possible cause of this phytotoxicity was the high content of heavy metals and low pH. The phytotoxicity were different from previous studies on waste leachate. A lower phytotoxicity effect was found for those waste leachates. The differences could have been caused by the higher concentration of toxic substances available to plants in the direct test. Moreover, the direct contact of sprouts and roots of *L. sativum* could have contributed to the higher phytotoxicity of the wastes than the leachate. Therefore, it seems appropriate to use both tests simultaneously to assess the phytotoxicity of waste.

Introduction

Landfilled industrial waste may have a negative impact on the environment and biota. The toxicity depends on its composition, physical properties, and the leaching of the contaminants. Some industrial waste may be landfilled in heaps or piles, separated from the environment in order to minimize infiltration of pollutants into the groundwater and to reduce dusting. For this purpose, geomembranes or natural materials (loams and clays) are used, as is biological reclamation with plants to protect against dusting and water infiltration into the heap/pile. Plants used for biological reclamation should have high tolerance to toxic substances in waste. Until now, most of the Polish foundry waste has been landfilled in heaps. Recently, the landfilled waste are recovered and used in road and construction industries. The largest type of foundry waste by mass is spent foundry sands (SFS), approx. 90% by weight (Bożym 2019, 2020). Many foundries regenerate SFS and re-use it for the production of castings, which reduces the consumption of raw materials. The demand for SFS is currently considerable, so it is profitable to recover waste that is landfilled (Bożym 2019, 2020). SFS may be use as road aggregate, in the construction industry, and as a soil substitute in horticulture and agriculture (Fahnlne 1995; Lindsay and Logan 2005; Dungan et al. 2006, 2007, 2009; Dungan and Dees 2007; Dayton et al. 2010; Siddique et al. 2010; EPA Report 2014; Zhang et al. 2014; Bożym 2017, 2018). The last application is popular in several countries, such as the USA, Argentina, Brazil, and South Africa. This application is conditional on the SFS having a low content of contaminants, including heavy metals. The Best Available Techniques Reference Document for Foundry (BAT 2007) describes the use of some foundry waste in the production of artificial crop layers, as a filler in the production of fertilizer, or as a soil modifier. Foundry dust may be generated at various stages of casting production, mainly during metal smelting, sand preparation, cleaning and knocking out of castings, and in the processes of dry sand regeneration. The place of collecting dust samples affects its physical and chemical properties. For example, electric arc furnace dust (EAFD) is classified as hazardous waste because it may contain significant amounts of heavy metals (Strobos and

Friend 2004; Salihoglu et al. 2008; Mymrin et al. 2016; Bożym 2020). The preferred management of foundry dust is to reuse it. It may be recycled in the foundry process or used for other purposes. Dust with a high content of heavy metals, which cannot be managed, should be solidified and then landfilled in properly prepared landfills. Due to the high costs of waste disposal, foundries independently manage all types of dust. To assess the use of foundry waste, its quantity, physical/mechanical properties, binder content, composition, and toxicity – the content of contaminants and their leachability – are taken into account.

Biotoxicity assessment may complement the physicochemical analysis of foundry waste. Biotoxicity tests are useful for assessing waste used in agriculture or deposited in bio-remediated landfills, because of its direct impact on biota. Phytotoxicity tests may be useful to determine the impact of waste on test species of plants and to calculate bioaccumulation factors. So far, foundry waste has not been assessed for its phytotoxicity. The leachability of contaminants and the enzymatic activity of soil substitutes based on SFS were investigated (Bastian and Alleman 1998; Dungan et al. 2006, 2009; Dungan and Dees 2009; Zhang et al. 2014),..

Many tests are used to assess the biotoxicity of waste. Phytotoxic tests are a popular method of assessing the biotoxicity of the municipal and industrial waste, sewage sludge, compost, fly ash, or contaminated soils (Gyuricza et al. 2010; Mitelut and Popa 2011; García-Lorenzo et al. 2014; Visioli et al. 2014; Jadhav et al. 2015; Machado et al. 2016; Seneviratne et al. 2017; Bożym 2020; Bożym et al. 2020). The germination of seeds and elongation of the roots after 72 h on leachate are usually evaluated. Both parameters are used to calculate the germination index (GI). Some authors observed that the root elongation is a more useful test than the germination test to assess the phytotoxic effect (Fuentes et al. 2004; Mitelut and Popa 2011). The root is more sensitive on toxins because it is directly exposed to the toxic effects of the contaminated solution (Fuentes et al. 2004; Gyuricza et al. 2010; Mitelut and Popa 2011; Jayasinghe 2012; García-Lorenzo et al. 2014; Visioli et al. 2014; Vochozka and Marouskova 2017). Some tests, bioaccumulation and bioremediation, for example, may be carried out directly in contaminated soil, waste, or sediment (Keser 2013; Baran et al. 2014; García-Lorenzo et al. 2014; Hattab et al. 2014). Although these tests are more time-consuming than germination tests, they allow for a wide assessment of the phytotoxicity and accumulation of contaminants (Jayasinghe 2012). Gyuricza et al. (2010) suggest that direct tests on the substrate are more useful than leachate tests in assessing phytotoxicity, because plants depend directly on the substrate. For this reason, the contact of seeds with the substrate reflects the real conditions (Bożym et al. 2020). However, for GI tests directly on a substrate, root germination and root elongation may be more difficult to evaluate because roots may penetrate the substrate and visual assessment may be more complicated. Common plants are usually used in phytotoxicity tests: sorghum, cress, mustard, radish, wheat, cucumber, barley, and others. However, each of the species is characterized by a different tolerance to contamination. Selecting an appropriate species for the type of toxin and waste may be problematic (Manas and De las Heras 2018). In many international and European standards as well in commercial tests, a wide range of plant species for phytotoxicity assessment on water leachate or mixture of substrate is used (Baumgarten and Spiegel 2004; Baran and Tarnawski 2013). One of the species used to assess phytotoxicity is watercress

(*Lepidium sativum* L.). This species is used in the toxicity assessment of many substances such as ammonia, fatty acids (Hoekstra et al. 2002), heavy metals (Hoekstra et al. 2002; Gill et al. 2013; Visioli et al. 2014; Masarovičová and Král'ová 2017), low-weight carboxylic acids (Himanen et al. 2012), benzalkonium chlorides (Khan et al. 2018), and others. Cress is highly tolerant to salinity, drought, and high concentrations of heavy metals and metalloids (Janecka and Fijalkowski 2008; Visioli et al. 2014; Masarovičová and Král'ová 2017; Praveen et al. 2017); for this reason it is often used in phytoremediation (Dursun and Ayturan 2018; Novo et al. 2017; Das and Osborne 2018). The advantage of using cress in phytotoxicity tests is its rapid growth, common occurrence, the availability of its seeds, and the ease of analysis (Masarovičová and Král'ová 2017). *L. sativum* L. is used to assess the phytotoxicity of soils (Bianchi et al. 2008; Mekki and Sayadi 2017; Manas and De las Heras 2018), sewage sludge (Fuentes et al. 2006), composts (Aslam et al. 2008), compost, or industrial waste (Courtney and Mullen 2009; Božym 2020; Božym et al. 2020). The ability to absorb and accumulate heavy metals, as well as the mechanisms of metal concentration, exclusion, and breakdown, vary from species to species and between plant parts (Amin et al. 2018). In order to assess the accumulation of metals in the test plant, the Bioconcentration factor (BCF) (Ashraf et al. 2011) or the transfer factor (TF) (Saha et al. 2017) are used. BCF can be calculated for all or part of a plant, e.g. for the roots (BCFr) or shoots (BCFs) (Marchiol et al. 2004; Kandziora-Ciupa et al. 2017). Some authors state that BCF is only concerned with the accumulation of metals in the root, while the bioaccumulation coefficient (BAC) is concerned with the metal content in shoots (Amin et al. 2018). These factors are calculated as the ratio of the metal content in the part of plant to the content in the substrate or leachate (extract).

Previous studies have assessed the phytotoxicity of leachate from foundry waste. The current research presents the results of phytotoxicity and accumulation of contaminants using direct tests on the substrate. The aim of the study was I) to assess the phytotoxicity of foundry waste in a contact test using cress seeds (*L. sativum* L.), II) to assess the condition of the plants and the accumulation of heavy metals in a contact test, and III) to compare the results of a contact test and leachate test from previous research, in order to evaluate the usefulness of both tests in assessing the toxicity of foundry wastes.

Sampling And Methodology

Samples of waste were collected from one of the Polish foundries, which produces cast iron and steel castings. For the production of foundry molds, organic binders based on phenol-formaldehyde resins, and less frequently bentonite, are used. Samples of the landfilled foundry waste (LFW) were taken from the landfill located next to the foundry. The samples were taken from six piles, after prescreening of the waste, in order to select suitable grains. This recovered waste is used as a commercial material in the production of road aggregate. The waste includes SFS (approx. 80% wt.), slag (approx. 10% wt.), spent refractory materials (approx. 5% wt.), and other waste such as dust and metalliferous inclusions. Increment samples were taken from several places in each pile. Subsequently, increment samples weighing approx. 25 kg were mixed on site and reduced by quartering to a mass of approx. 5 kg. The second group of waste, collected for phytotoxicity tests, were dusts from dust collectors located in various units of the foundry, i.e. the forming unit (regeneration, transport, and shock grating sections),

steelworks unit (electric furnace section), and shot blasting unit (pneumatic cleaning section). The dust samples were divided into five groups: shock grating dust (SGD), regeneration (of SFS) dust (RD), transport (of molding sands) dust (TD), EAFD, and pneumatic blast cabinet dust (PBCD). All dust produced in the foundry is used for various purposes. EAFD and SGD are substrates for the production of briquettes for foundries, while the other dust is used as inert material for filling inactive mines or producing building materials. Foundry waste samples were collected twice in 2017–18. The location of the foundry and landfill, as well as the sampling methodology, were presented in previous papers (Bożym 2020, Bożym and Kłojzy-Karczmarczyk 2021).

Phytotoxicity assessment and accumulation test

In this study, phytotoxicity tests directly on the waste (contact test) were carried out. A germination index (GI) and an accumulation test were performed to assess the extent to waste impact on plants. To the tests untreated *L. sativum* L. seeds of commercial certified material (No. PL–EKO–01) were used.

Germination index (GI)

The GI test was based on the analysis of the number of germinated seeds and the degree of root elongation over 72 days. The seeds were sown directly on the waste substrate, 20 per Petri dish. Previously, about 10.0 ± 0.5 g of waste was introduced into the dishes and soaked in deionized water (approx. 10 ml) for 24 h, to dissolve the contaminants (similarly to the water extract). The Petri dishes with seeds were covered and incubated in the dark at room temperature for 72 h. The GI value was calculated from the relative seed germination (RSG, %) and relative root growth (RRG, %), in accordance with Zucconi's methodology (Zucconi et al. 1985). Petri dishes filled with waste were used for the accumulation test, as with the GI method. The waste was previously soaked in deionized water (approx. 10 ml) for 24 h. In the next stage, 1.0 ± 0.1 g of *L. sativum* L. seeds was sown. The dishes with seeds were covered and incubated for 7 days at room temperature for 16 h in the light and 8 h in the dark (ST2BD Smart incubator, Pol-Eko-Aparatura SP). After 7 days, the color of the cotyledons and the length of the shoots was assessed. Only the aerial part (shoots) was used for the study of heavy metal accumulation, because it was difficult to clean the roots from the dusty substrate. Root contamination may have affected the heavy metal analysis. The aerial parts of *L. sativum* L. were weighed to assess their biomass. For both tests, pretreated sand (sieved and treated with HNO_3) was used as a control. The bioconcentration factor (BCF), the ratio of heavy metal concentration in the shoots to the total content in the substrate, was calculated. $\text{BCF} > 1$ indicates bioaccumulation of the metal by the plant (Marchiol et al. 2004; Kandziora-Ciupa et al. 2017).

Sample preparation and analysis

LFW samples were dried at 105°C , ground in a mortar and sieved through a 1mm sieve. Dust samples were not ground and sieve, but dried same as LFW samples and analyzed. The evaluation of germination and root elongation was carried out with an accuracy of 1 mm. Germination was found when the sprout was at least 1 mm. The GI and accumulation test was performed in triplicate. Waste and plant samples

were mineralized for the determination of heavy metals. The mineralization was carried out in a microwave oven (Start D, Milestone) with the use of inorganic acids (EN 13656, EN 16173). The loss of ignition (LOI) at 550⁰C (EN 15169) was analysed. The CHNS (carbon, hydrogen, nitrogen, sulfur) analysis in the Elementar Vario Macro Cube elemental analyzer was measured (PN-G-04571 and PN-G-0458). Heavy metals were determined by the method flame atomic absorption spectrophotometry (FAAS) using a spectrophotometer Solaar 6M (Thermo) (US EPA Method 7000B, ISO 8288).

Statistics and quality control

The statistical analysis was carried out using the program Statistica ver. 13.3. All samples, were carried out and analysed in triplicate. For quality control a certified reference materials (CRM): '*Metals in soil*' (SQC001, Merck), '*Urban particulate matter*' (SRM 1648a, Sigma Aldrich), '*Fine dust PM10–LIKE*' (ERM®–CZ120, IRMM), '*Lichen (trace elements)*' (BCR–482, IRMM) were also analysed. Heavy metals recovery in CRM samples ranged from 90-110%.

Results

Table 1 presents the results of the elemental analysis, pH, and loss of ignition (LOI) of the tested waste. All waste samples were characterized by different pH levels. A low pH was found for the dust samples (SGD, RD, and PBCD). Excessively high or low pH may influence the biotoxicity of the waste (Phoungthong et al. 2016); for example, the toxicity of heavy metals may increase at low pH (Emamverdian et al. 2015; Phoungthong et al. 2016). Seneviratne et al. (2017) found that phytotoxicity is the result of a combination of several factors that inhibit plant growth. For example, fluoride phytotoxicity increases at low pH (Stevens i in. 2000), while heavy metal cations in solution may reduce the phytotoxicity of fluorides (Palmieri et al. 2014).

Table 1
The characteristic of tested foundry waste (Bożym and Kłojzy-Karczmarczyk 2020, 2021)

sample	Elementar analysis [% wt.]				pH	LOI [% wt.]
	C	H	N	S		
LFW	2.7 ± 1.2	0.05 ± 0.05	0.04 ± 0.02	0.09 ± 0.02	7.9 ± 0.4	4.0 ± 2.0
SGD	5.3 ± 0.0	0.11 ± 0.00	0.03 ± 0.02	0.40 ± 0.00	5.1 ± 0.1	7.1 ± 0.6
RD	11.4 ± 4.1	0.12 ± 0.03	0.08 ± 0.01	0.65 ± 0.07	5.4 ± 0.5	13.8 ± 3.2
TD	6.1 ± 3.3	0.12 ± 0.04	0.07 ± 0.02	0.40 ± 0.14	6.0 ± 0.2	7.9 ± 5.2
EAFD	7.8 ± 1.0	0.13 ± 0.04	0.04 ± 0.00	1.05 ± 0.21	7.6 ± 1.5	10.0 ± 0.7
PBCD	2.1 ± 0.3	0.11 ± 0.03	0.04 ± 0.02	0.37 ± 0.13	5.4 ± 0.5	2.5 ± 0.4

The LOI value is an indicator of the presence of organic matter in the waste. Among the tested wastes, EAFD and RD samples demonstrated the highest LOI values. The dust from the regeneration section (RD) contained organic binder residues, hence its high LOI value (Bożym 2018). While EAFDs, similar to dust from thermal regeneration, may contain organic compounds from organic pollutants – such as adhesives, paints, or lubricants – contained in scrap, which may evaporate at high temperature during the metal melting process (Zanetti and Godio 2006; Salihoglu and Pinarli 2008). The elemental composition of the tested waste was similar to the foundry dust analyzed by Chirila and Ionescu Luca (2011).

Foundry waste usually contains metal residue from castings. Waste from steel and iron foundries is characterized by the highest iron content (Skvara et al. 2002; Sofilic et al. 2004; Li et al. 2010; Gengel et al. 2010). This finding was confirmed in the current research. The percentage of Fe varied within a wide range: 35.9–41.9% for EAFD, 16.0–19.4% for SGD, 12.7–13.0% for TD, 10.4–84.0% for PBCD, 4.1–9.8% for RD (Bożym and Kłojzy-Karczmarczyk 2020), and 14.1% for LFW (Bożym 2020). Iron is not a toxic metal, so its content in waste is not limited. Foundry waste may also contain heavy metals. The content of metals, metalloids, and nutrients in foundry waste was investigated in previous studies (Bożym 2020). Figure 1 shows the sum of heavy metals (Co, Mo, Ni, Cr, Zn, Cu, Pb, and Cd) in the tested foundry waste.

Among the heavy metals tested in this study, zinc characterized the highest percentage (23–81%), then copper (6–31%) and chromium (4–27%) for most wastes, and lead (33%) for EAFD. The highest content of all heavy metals was found in EAFD dust. It is known that EAFDs are the most problematic waste in foundries due to the toxic content of heavy metals and organic pollutants (Salihoglu and Pinarli 2008; Bożym and Zalejska 2014; Chirila and Ionescu Luca 2011; Mymrin et al. 2016; Bożym 2020). As Salihoglu and Pinarli (2008) found, metals such as Zn and Pb are very volatile at the temperature of molten steel and therefore accumulate in the furnace dust (Salihoglu and Pinarli 2008). Disposal of EAFD is problematic for foundries, as this dust may be generated in large quantities. It is estimated that in the typical operation of an electric arc furnace, about 2% of the input is converted into dust and 10–20 kg of EAFD are produced per 1 ton of castings. Worldwide, foundries produce about 8 million tons of EAFD annually; in the USA it is about 0.7 million tons, in Europe up to 1 million tons, and in Poland 4,000 tons of EAFD (Chirila and Ionescu Luca 2011; Bożym and Kłojzy-Karczmarczyk 2020). One method of utilizing EAFD with a high percentage of iron is to recycle it back into the foundry furnace. In addition, some heavy metals are recovered from the EAFD, but the cost-effectiveness of the process depends on the amount of metal in the dust. In the current study, a large amount of heavy metals, especially Zn, was also found in the dust from the shot blasting section (PBCD) (Strobos, and Friend 2004; Miyoshi et al. 2008; Chirila and Ionescu 2011; Bożym and Zalejska 2014; Bożym and Kłojzy-Karczmarczyk 2020). Moreover, PBCDs were characterized by the highest variability of heavy metal content, which was caused by the variability in the composition of the castings. The lowest metal content of all tested dust samples was found in RD and TD, i.e. dusts from the regeneration and transport units. On the other hand, the lowest heavy metal content of all waste samples was found in the LFW. The tested LFWs also demonstrated low leachability of heavy metals (Bożym 2017, 2019, 2020). LFW consisted mainly of SFS; therefore, this waste may be used for the production of road aggregates or other applications.

Phytotoxicity Tests

The assessment of phytotoxicity is based on the GI test, plant growth, leaf size and color, chlorophyll content, biomass, and the accumulation of pollutants (Gyuricza et al. 2010; Awasthi et al. 2017). The phytotoxicity assessment of the tested wastes based on the condition of the *L. sativum* L. (cotyledons, biomass, heavy metal accumulation) and the GI test is presented below. The GI results obtained in the direct test were compared with the leachate test performed in the previous study (Bożym 2020).

Accumulation Test

According to Phoungthong et al. (2016), an excess of heavy metals in the substrate causes stress in plants and leads to reduced germination, delayed growth, and leaf chlorosis. For this reason, the condition of *L. sativum* L. (cotyledon color, shoot length, and biomass) was additionally assessed during the accumulation test. A blackening of the seed coat of *L. sativum* L. from the EAFD, SGD, and RD substrates was found. The same effect was observed in previous studies on leachate from this waste (Bożym 2020). Presumably, the cause of this effect was the high iron concentration and the low pH. Some authors have reported that heavy metals may cause seeds, roots, and leaves to blacken. Mossor-Pietraszewska (2001) noted that Al inhibited root growth and darkened them, while Emamverdian et al. (2015) found blackened plants with a high Mn content in the substrate. However, in other studies with *L. sativum* L. growing on slag from copper and zinc smelters with high concentrations of Zn, Cu, and Pb in the substrate, this effect was not found. In the current research, *L. sativum* L. sprouts died after 2–3 days of the accumulation test on EAFD, SGD, and RD substrates (Bożym et al. 2020). In previous studies on leachate from the same foundry waste (Bożym 2020), however, plant material was collected from all variants in an accumulation test. Another parameter of *L. sativum*'s condition was shoot length, which was measured from the end of the root to the cotyledons. The longest shoots for *L. sativum* L. were from the control (mean: 13.5 cm) and LFW (mean: 14.5 cm) groups. The length of the shoots from the dust substrate ranged from 12 to 13 cm (TD) and from 5 to 12 cm (PBCD). In the case of PBCD, the large variance of the results depended on the composition of the samples (n = 10). The best condition (cotyledon color, stem length, and biomass) of *L. sativum* L. was found in the LFW and control groups. On the other hand, the cotyledons of *L. sativum* L. of the control, LFW, TD, and PCBD groups were colored well, indicating no negative effect of those substrates on chlorophyll. It is known that increased accumulation of heavy metals in the aerial parts may cause leaf chlorosis, a reduction in yield, leaf area, relative growth rate, and assimilation rate (Wahid et al. 2007; Keser 2013; Gill et al. 2013; Emamverdian et al. 2015; Masarovičová and Král'ová 2017). On the other hand *L. sativum* L. is resistant to negative environmental factors, including heavy metals (Dursun et al. 2018) and it is used as a bioindicator of soil contamination, as it shows changes in protein expression even at low concentrations of heavy metals (Janecka and Fijalkowski 2008; Seneviratne et al. 2017).

In addition to the assessment of the condition of the plants, a yield analysis was performed. The aerial parts of *L. sativum* L. from the LFW, TD, and PCBD substrates and the control group were collected. The

roots were not analyzed due to the potential for contamination by the substrate, especially with dust, which could have affected the results. Figure 2 presents the dry mass of *L. sativum* L. shoots obtained during the 7-day accumulation test. The results were compared with the control (sand). No plants were taken from the SGD, RD, and EAFD substrates. On the basis of the results, it may be stated that the biomass of *L. sativum* L. was significantly higher from the LFW substrate than in the control ($p \leq 0.05$), which may indicate that LFW stimulated the growth of *L. sativum* L.. This is confirmed by the results of the GI test. The reason for this effect could be the low content and leachability of heavy metals and the neutral pH of LFW compared to foundry dust samples (Bożym 2020). Moreover, the higher biomass of *L. sativum* growing on the TD substrate compared to the control ($p \leq 0.05$) were also analyzed. Keser (2013) found a reduction in the biomass of the edible parts of *Eruca sativa* irrigated with wastewater in comparison with the control, influenced by high heavy metal content. The author did not note these effects for *L. sativum* L., which explains the higher resistance of cress to heavy metals. Many studies have confirmed the negative effect of heavy metals on the biomass of plants (Wahid et al. 2007; Kachout et al. 2009; Farouk et al. 2011; Vijayarengan and Mahalakshmi 2013).

The phytoaccumulation of heavy metals from waste and soil depends on the total content, leachability, pH, and metal interactions in substrate. For example, Keser (2018) found slight differences in the content of Cd, Pb, and Ni in the edible parts and roots of *L. sativum* L. after 20 days of the test. In the case of Cr and Cu, an accumulation of these metals in the root and a low content in the edible parts was reported. The distribution of heavy metals in plant organs is determined by the species and age of the plant, the transport mechanism, the degree of environmental pollution by heavy metals, and the interaction between metals (Seregin and Ivanov 2001; García-Lorenzo et al. 2014; Seneviratne et al. 2017). In the current study, a wide range of heavy metal content was found in *L. sativum* L. after 7 days of the experiment. The results are presented in Fig. 3. No plants were collected for analysis from the SGD, RD, or EAFD substrates due to the atrophy of the sprouts. This effect was not found in the leachate test (Bożym 2020). This may suggest that an accumulation test carried out on leachate may not be sufficient for phytotoxicity assessment of waste. Similar results were obtained for hazardous waste from zinc and copper smelters in previous studies (Bożym et al. 2020).

Zinc displayed the highest accumulation out of all heavy metals in *L. sativum* from all substrates (Fig. 3). This effect may be influenced by the high percentage of this metal in the waste under analysis. In previous studies on leachate from foundry waste, the content of Zn, Cu, and other metals in *L. sativum* L. was lower than in the current study (directly on the substrate). For example, Zn and Cu accumulation in both experiments differed by 5–37% and 48–66%, respectively. Based on the heavy metal content in *L. sativum* L., the BCF value was calculated (Table 2). No metal accumulation by *L. sativum* L. in any substrate was found in the current study ($BCF < 1.0$). Slightly higher BCF values were calculated for Cu and Zn from LFW, Zn from TD, and Cd and Pb from PBCD ($BCF = 0.27–0.67$).

Table 2
Bioconcentration factors (BCF) in shoots of *L. sativum* L.

metal	LFW (n = 18)	SGD (n = 6)	RD (n = 6)	TD (n = 6)	EAFD (n = 6)	PBCD (n = 30)
Cd	<LOQ	nd	nd	0.19	nd	0.44
Pb	0.07	nd	nd	0.18	nd	0.63
Cu	0.27	nd	nd	0.05	nd	0.08
Zn	0.67	nd	nd	0.28	nd	0.03
Cr	0.06	nd	nd	0.01	nd	0.02
Ni	0.02	nd	nd	0.01	nd	0.02
Mo	0.02	nd	nd	0.03	nd	0.08
Co	0.05	nd	nd	0.05	nd	0.03
nd – no data (no plants to analyze)						
<LOQ - the metal content of the substrate or plants was below the limit of quantification						

Germination Index Directly On The Substrate

The GI was calculated from the values for *L. sativum* L. germination and root elongation in the direct test on the substrate compared to the control (sand). Due to the fact that the root length has a direct impact on the GI value, Fig. 4 shows the GI value while Fig. 5 shows the root length of *L. sativum* L. in the direct test (GI 1) and on the leachate (GI 2) (Bożym 2020). Additionally, Fig. 4 shows the ranges of the inhibitory/stimulating action of waste on plants according to Zucconi et al. (1985). A GI value of < 50% indicates high phytotoxicity, 50–80% indicates medium phytotoxicity, and 80–100% no phytotoxicity. On the other hand, a GI of > 100% suggests a stimulating effect of the waste or leachate on the plant.

Seeds from some substrates had blackened in the GI test, similar to the effect in the accumulation test and previous leachate tests (Bożym 2020). The *L. sativum* L. growing on these substrates was also characterized by shortened and blackened root tips. It is known that heavy metals in the substrate may shorten the roots and shoots of plants (Shanker et al. 2005; Zou et al.; Emamverdian et al. 2015; Sharma et al. 2010; Vijayarengan and Mahalakshmi 2013; Janecka and Fijalkowski 2008; Courtney and Mullen (20019; Phoungthong i in. (2016). An additional phytotoxic factor may be a low pH, which increases the toxicity of heavy metals (Emamverdian et al. 2015; Phoungthong et al.2016; Courtney and Mullen 20019). At the same time, some metals may stimulate seed germination and root elongation at lower concentrations, while at higher concentrations they are phytotoxic (Phoungthong et al.2016). Many scientific reports indicate the effect of some toxic substances contained in the substrate – e.g. phenols,

cyanides, salinity, fluorides, ammonia, or low-weight carboxylic acids – on the germination and elongation of *L. sativum* L. roots (Hoekstra et al. 2002; Himanen et al. 2012; Palmieri et al. 2014; Manas and De las Heras 2018).

The leachate test showed a stimulating effect of LFW, TD, and PBCD leachate on *L. sativum* in a previous study (Bożym 2020). This effect was not observed in the current study, directly on the substrate. The contact test showed that LFW were characterized by moderate or no phytotoxicity; the dust samples, SGD, TD, RD, and PCBD, demonstrated high phytotoxicity (Fig. 4). EAFD was characterized by the highest phytotoxicity, in both the leachate and direct tests. In a previous study, SGD and RD leachate were less phytotoxic than the substrate in a direct test. This effect could have been influenced by the dilution of pollutants in the leachate (liquid/solid, L/S, 1/10); at the same time, in the direct test L/S was 1/1. The presence of organic pollutants, for example, phenol and formaldehyde, could also have contributed to the phytotoxicity of the tested wastes. An additional phytotoxic factor could have been the low pH. In a study of hazardous wastes from a zinc and copper smelter, higher phytotoxicity of the waste was also found in the direct test than in the leachate test (Bożym et al. 2020).

The root length of *L. sativum* L. from the control, LFW, and PBCD substrates was higher in the direct test than in the leachate test (Fig. 5). These results may suggest that *L. sativum* L. root elongation is faster on a solid substrate than on leachate. On the other hand, for SGD, RD, TD, and EAFD substrates, a much shorter root length of *L. sativum* L. was observed in the contact test than in the leachate test. Nevertheless, the final GI value was lower due to the considerable root length of *L. sativum* L. from the control (sand) in the direct test.

Summarizing the current research, it may be concluded that *L. sativum* L. is useful for assessing the phytotoxicity of foundry wastes. In addition, different phytotoxicities were found for waste in the direct and leachate tests. Phytotoxicity tests have an impact on the assessment of the use of foundry waste, especially SFS in road engineering and for agrotechnical purposes, e.g. for the production of Technosols (Bożym and Kłojzy-Karczmarczyk 2021). Studies have shown that LFW with SFS was characterized by low phytotoxicity, because of the low content of heavy metals and neutral pH; its leachate may stimulate the growth of *L. sativum* L. Tests on foundry dusts showed different phytotoxicities, depending on the composition of the waste and the type of the test. Higher phytotoxicity was observed in the direct test than in the leachate test. For this reason, it is recommended to use both the leachate and the direct tests simultaneously to assess the phytotoxicity of waste.

Conclusions

Foundry dusts were characterized by high phytotoxicity in the direct test. On the other hand, the dust leachate showed a slightly lower phytotoxicity. In previous studies, it was found that the phytotoxicity of the leachate is influenced by the content of heavy metals, metalloids, inorganic anions, high EC value, and the concentration of organic pollutants (phenol and formaldehyde) (Bożym 2020). The wastes which were tested displayed higher phytotoxicity in the direct test than the leachate test. The reason for this

effect may be the higher concentration of pollutants in the substrate than in the leachate. The degree of dilution also contributed to the higher phytotoxicity of the waste in the contact test than leachate test. Direct contact of the roots of *L. sativum* L. with the substrate could also have increased the phytotoxicity of the foundry waste. However, it was found that *L. sativum* L. roots from the control and LFW groups were longer in the direct test than in the leachate test, which may indicate the effect of the solid substrate on root elongation. Due to the conflicting results of the phytotoxicity of waste in the direct test and the leachate test, it is suggested that both tests should be performed at the same time to assess the phytotoxicity of waste. The leachate test may not indicate the actual phytotoxicity of the waste. This is important when assessing the suitability of waste for agrotechnical purposes (e.g., SFS for the production of Technosols). In such applications, the direct contact of the plants with the waste-based substrate will have an impact on their growth and condition.

Declarations

Conflict of interest

The author declares no competing interests.

Ethical approval

This article does not contain any studies with human participants or animals performed by the author.

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References

1. Amin H, Arain AB, Jahangir TM, Abbasi MS, Amin F (2018) Accumulation and distribution of lead (Pb) in plant tissues of guar (*Cyamopsis tetragonoloba* L.) and sesame (*Sesamum indicum* L.): profitable phytoremediation with biofuel crops. *Geology Ecology Landscapes* 2(1):51–60. <https://doi.org/10.1080/24749508.2018.1452464>
2. Ashraf MA, Maah J, Yusoff I (2011) Heavy metals accumulation in plants growing in ex tin mining catchment. *Int J Environ Sci Techn* 8(2):401-416.
3. Aslam DN, Horwath W, VanderGheynst JS (2008) Comparison of several maturity indicators for estimating phytotoxicity in compost-amended soil. *Waste Manage* 28:2070–2076. <https://doiorg/10.1016/j.wasman.2017.08.026>

4. Awasthi AK, Pandey AK, Khan J (2017) Municipal solid waste leachate impact on metabolic activity of wheat (*Triticum aestivum* L) seedlings. Environ Sci Pollut Res 24:17250–17254. <https://doi.org/10.1007/s11356-017-9412-8>
5. Baran A, Czech T, Wieczorek J (2014) Chemical properties and toxicity of soils contaminated by mining activity. Ecotoxicology 23:1234–1244. <https://doi.org/10.1007/s10646-014-1266-y>
6. Baran A, Tarnawski M (2013) Phytotoxkit/Phytotestkit and Microtox tools for toxicity assessment of sediments. Ecotoxicol Environ Saf 98:19–27. <https://doi.org/10.1016/j.ecoenv.2013.10.010>
7. Bastian KC, Alleman JE (1998) Microtox characterization of foundry sand residuals, Waste Manage 18:227-234.
8. BAT 2007. BAT 2007. Reference Document for Best Available Techniques for Forging and Foundry Engineering. Polish Ministry of the Environment, BAT in forging and foundry - IPPC 2007. (in Polish)
9. Baumgarten A, Spiegel H (2004) Phytotoxicity (Plant tolerance), HORIZONTAL – 8, Agency for Health and Food Safety, Vienna. Available online <https://horizontal.ecn.nl/>
10. Bianchi V, Masciandaro G, Giraldi D, Ceccanti B, Iannelli R (2008) Enhanced heavy metal phytoextraction from marine dredged sediments comparing conventional chelating agents (citric acid and EDTA) with humic substances. Water Air Soil Pollut 193:323–333. <https://doi.org/10.1007/s11270-008-9693-0>
11. Bożym M (2017) The study of heavy metals leaching from waste foundry sands using a one-step extraction. E3S Web of Conferences 19:1–6. <https://doi.org/10.1051/e3sconf/20171902018>
12. Bożym M (2018) Alternative directions for the use of foundry waste, especially for energy management, Sci Notebooks of the Institute of Mineral Raw Materials and Energy Management of the Polish Academy of Sciences 105:197–212. <https://doi.org/10.24425/124358> (in Polish)
13. Bożym M (2019) Assessment of leaching of heavy metals from the landfilled foundry waste during exploitation of the heaps. Pol J Environ Stud 28(6):1–10. <https://doi.org/10.15244/pjoes/99240>
14. Bożym M (2020) Assessment of phytotoxicity of leachates from landfilled waste and dust from foundry. Ecotoxicology 29:429–443. <https://doi.org/10.1007/s10646-020-02197-1>
15. Bożym M, Kłojzy-Karczmarczyk B (2020) The content of heavy metals in foundry dusts as one of the criteria for assessing their economic reuse, Mineral Resources Management 36(3):111–126. <https://doi.org/10.24425/gsm.2020.133937>
16. Bożym M, Kłojzy-Karczmarczyk B (2021) Assessment of the mercury contamination of landfilled and recovered foundry waste – a case study. Open Chemistry 19:462–470. <https://doi.org/10.1515/chem-2021-0043>
17. Bożym M, Król A, Mizerna K (2021) Leachate and contact test with *Lepidium sativum* L. to assess the phytotoxicity of waste, Int J Environ Sci Technol 18:1975–1990. <https://doi.org/10.1007/s13762-020-02980-x>
18. **Bożym M, Zalejska K (2014)** The study of heavy metals leaching from foundry dusts in terms of their impact on the environment, Arch Waste Manage Environ Protection 16(3):1-6. (in Polish)

19. Chirila E, Ionescu Luca C (2011) Characterization of the electric arc furnace dust, Annals of the Oradea University. Fascicle of Management and Technological Engineering, Volume X (XX), 2, 4.1-4.6.
20. Courtney R, Mullen G (2009) Use of germination and seedling performance bioassays for assessing revegetation strategies on bauxite residue. *Water Air Soil Pollut* 197:15–22.
21. Das A, Osborne JW (2018) Bioremediation of Heavy Metals. Chapter 9, KMGothandam et al. (eds.), *Nanotechnology, Food Security and Water Treatment, Environmental Chemistry for a Sustainable World* 11, https://doi.org/10.1007/978-3-319-70166-0_9
22. Dayton EA, Whitacre SD, Dungan RS, Basta NT (2010) Characterization of physical and chemical properties of spent foundry sands pertinent to beneficial use in manufactured soils. *Plant Soil* 329:27–33. <https://doi.org/10.1007/s11104-009-0120-0>
23. Dungan RS, Dees NH (2009) The characterization of total and leachable metals in foundry molding sands. *J Environ Manage* 90:539-548. <https://doi.org/10.1016/j.jenvman.2007.12.004>
24. Dungan RS, Kim JS, Weon HY, Leytem AB (2009) The characterization and composition of bacterial communities in soils blended with spent foundry sand. *Ann Microbiol* 59(2):239-246. <https://doi.org/10.1007/BF03178323>
25. Dungan RS, Kukier U, Lee B (2006) Blending foundry sands with soil: Effect on dehydrogenase activity. *Sci Total Environ* 357: 221– 230. <https://doi.org/10.1016/j.scitotenv.2005.04.032>
26. Dungan RS, Lee BD, Shouse P, De Koff JP (2007) Saturated hydraulic conductivity of soils blended with waste foundry sands. *Soil Sci* 10:751–758.
27. Dursun S, Ayturan ZC (2018) Phytoremediation of metal industry wastewaters: A Review. Chapter of A. Kallel et al. (eds.), *Recent Advances in Environmental Science from the Euro-Mediterranean and Surrounding Regions, Advances in Science, Technology & Innovation*, https://doi.org/10.1007/978-3-319-70548-4_134
28. Emamverdian A, Ding Y, Mokhberdoran F, Xie Y (2015) Heavy Metal Stress and Some Mechanisms of Plant Defense Response. Hindawi Publishing Corporation *The Scientific World Journal* Volume 2015, Article ID 756120, 18 pages. <http://dx.doi.org/10.1155/2015/756120>
29. EN 13656 Characterization of waste - Microwave assisted digestion with hydrofluoric (HF), nitric (HNO₃) and hydrochloric (HCl) acid mixture for subsequent determination of elements. CEN.
30. EN 15169 Characterization of waste. Determination of loss on ignition in waste, sludge and sediments. CEN.
31. EN 16173. Sludge, treated biowaste and soil - Digestion of nitric acid soluble fractions of elements. CEN.
32. EPA Report 2014: Risk Assessment of Spent Foundry Sands In Soil-Related Applications. Evaluating Silica-based Spent Foundry Sand From Iron, Steel, and Aluminum Foundries. EPA-530-R-14-003. October 2014. Available online <https://www.epa.gov/>

33. Fahnl DE (1995) Leaching of metals from beneficially used foundry residuals into soils. 50th Purdue Industrial Waste Conference Proceedings 1995, Chelsea, Michigan, 339-347.
34. Farouk S, Mosa AA, Taha AA, Ibrahim HM, EL-Gahmery AM (2011) Protective Effect of humic acid and chitosan on radish (*Raphanus sativus*, L. var. sativus) plants subjected to cadmium stress. J Stress Physiol Biochem 7:99–116.
35. Fuentes A, Lloréns M, Sáez J, Aguilar MI, Ortuño JF, Meseguer VF (2004) Phytotoxicity and heavy metals speciation of stabilised sewage sludges. J Hazard Mater A108:161–169.
<https://doi.org/10.1016/j.jhazmat.2004.02.014>
36. García-Lorenzo ML, Martínez-Sánchez MJ, Pérez-Sirvent C (2014) Application of a plant bioassay for the evaluation of ecotoxicological risks of heavy metals in sediments affected by mining activities. J Soils Sediments 14:1753–1765. <https://doi.org/10.1007/s11368-014-0942-0>
37. Gengel P, Pribulová A, Futáš P (2010) Possibilities of foundry dust utilization in foundry process. Acta Metallurgica Slovaca 16(1):20–25.
38. Gill S, Anjum N, Gill R, Hasanuzzaman M, Sharma P, Tuteja N (2013) Mechanism of Cadmium Toxicity and Tolerance in Crop Plants. In: Tuteja N., Gill S. (eds) Crop Improvement Under Adverse Conditions. Springer, New York, NY. https://doi.org/10.1007/978-1-4614-4633-0_17
39. Gyuricza V, Fodor F, Szigeti Z (2010) Phytotoxic effects of heavy metal contaminated soil reveal limitations of extract-based ecotoxicological tests. Water Air Soil Pollut 210:113–122.
<https://doi.org/10.1007/s11270-009-0228-0>
40. Hattab N, Soubrand M, Guégan R, Motelica-Heino M, Bourrat X, Faure O, Bouchardon JL (2014) Effect of organic amendments on the mobility of trace elements in phytoremediated techno-soils: role of the humic substances. Environ Sci Pollut Res 21:10470–10480.
41. Himanen M, Prochazka P, Hänninen K, Oikari A (2012) Phytotoxicity of low-weight carboxylic acids. Chemosphere 88:426–431. <https://doi.org/10.1016/j.chemosphere.2012.02.058>
42. Hoekstra NJ, Bosker T, Lantinga EA (2002) Effects of cattle dung from farms with different feeding strategies on germination and initial root growth of cress (*Lepidium sativum* L.). Agric Ecosyst Environ 93:189–196. [https://doi.org/10.1016/S0167-8809\(01\)00348-6](https://doi.org/10.1016/S0167-8809(01)00348-6)
43. ISO 8288. Water quality - Determination of cobalt, nickel, copper, zinc, cadmium and lead – Flame atomic absorption spectrometric methods
44. Jadhav SB, Chougule AS, Shah DP, Pereira CS, Jadhav JP (2015) Application of response surface methodology for the optimization of textile effluent biodecolorization and its toxicity perspectives using plant toxicity, plasmid nicking assays. Clean Techn Environ Policy 17:709–720.
<https://doi.org/10.1007/s10098-014-0827-3>
45. Janecka B, Fijalkowski K (2008) Using *Lepidium* as a test of phytotoxicity from lead/zinc spoils and soil conditioners. In: Simeonov L., Sargsyan V. (eds) Soil Chemical Pollution, Risk Assessment, Remediation and Security. NATO Science for Peace and Security Series. Springer, Dordrecht.
https://doi.org/10.1007/978-1-4020-8257-3_14

46. Jayasinghe GY (2012) Sugarcane bagasses sewage sludge compost as a plant growth substrate and an option for waste management. *Clean Techn Environ Policy* 14:625–632, <https://doi.org/10.1007/s10098-011-0423-8>
47. Kachout SS, Mansoura AB, Leclerc JC, Mechergui R, Rejeb MN, Ouerghi Z (2009) Effects of heavy metals on antioxidant activities of *Atriplex hortensis* and *A. rosea*. *J Food Agric Environ* 7:938–945.
48. Kandziora-Ciupa M, Nadgórska-Socha A, Barczyk G, Ciepal R (2017) Bioaccumulation of heavy metals and ecophysiological responses to heavy metal stress in selected populations of *Vaccinium myrtillus* L. and *Vaccinium vitis-idaea* L. *Ecotoxicology* 26:966–980. <https://doi.org/10.1007/s10646-017-1825-0>
49. Keser G (2013) Effects of irrigation with wastewater on the physiological properties and heavy metal content in *Lepidium sativum* L. and *Eruca sativa* (Mill.). *Environ Monit Assess* 185:6209–6217. <https://doi.org/10.1007/s10661-012-3018-x>
50. Khan AH, Libby M, Winnick D, Palmer J, Sumarah M, Ray MB, Macfie SW (2018) Uptake and phytotoxic effect of benzalkonium chlorides in *Lepidium sativum* and *Lactuca sativa*. *J Environ Manage* 206: 490-497. <https://doi.org/10.1016/j.jenvman.2017.10.077>
51. Li HX, Wang Y, Cang DQ (2010) Zinc leaching from electric arc furnace dust in alkaline medium. *J Cent South Univ Technol* 17:967–971. <https://doi.org/10.1007/s11771-010-0585-2>
52. Lindsay BJ, Logan TJ (2005) Agricultural reuse of foundry sand. Review. *J Resid Sci Technol* 2(1):3-12.
53. Machado RM, Monteggia LO, Arenzon A, Curia AC (2016) Assessment of the toxicity of wastewater from the metalworking industry treated using a conventional physico-chemical process, *Environ Monit Assess* 188: 373.
54. Manas P, De las Heras J (2018) Phytotoxicity test applied to sewage sludge using *Lactuca sativa* L. and *Lepidium sativum* L. seeds, *Int J Environ Sci Technol* 15:273–280. <https://doi.org/10.1007/s13762-017-1386-z>
55. Marchiol L, Assolari S, Sacco P, Zerbi G (2004) Phytoextraction of heavy metals by canola (*Brassica napus*) and radish (*Raphanus sativus*) grown on multicontaminated soil. *Environ Pollut* 132:21–27. <https://doi.org/10.1016/j.envpol.2004.04.001>
56. Masarovičová E, Kráľová K (2017) Essential Elements and Toxic Metals in Some Crops, Medicinal Plants, and Trees. In: Ansari A., Gill S, Gill R, R. Lanza G, Newman L. (eds) *Phytoremediation*. Springer, Cham. https://doi.org/10.1007/978-3-319-52381-1_7
57. Mekki A, Sayadi S (2017) Study of heavy metal accumulation and residual toxicity in soil saturated with phosphate processing wastewater. *Water Air Soil Pollut* 228:215. <https://doi.org/10.1007/s11270-017-3399-0>
58. Mitelut AC, Popa ME (2011) Seed germination bioassay for toxicity evaluation of different composting biodegradable materials. *Roman Biotechnol Lett Supplement* 16(1):121–129.
59. Miyoshi Y, Mogi Y, Saima H, Takagi K (2008) Method for reforming exhaust gas generated from metallurgical furnace, method for cooling exhaust gas and apparatus therefore. US patent

60. Mossor-Pietraszewska T (2001) Effect of aluminium on plant growth and metabolism, *Acta Biochimica Polonica* 48(3):673–686.
61. Mymrin V, Nagalli A, Catai RE, Izzo RLS., Rose J, Romano CA (2016) Structure formation processes of composites on the base of hazardous electric arc furnace dust for production of environmentally clean ceramics. *J Clean Prod* 137:888–894. <https://doi.org/10.1016/j.jclepro.2016.07.105>
62. Novo LAB, Onishi VO, Bernardino CAR, Ferreira da Silva E (2017) Metal Bioaccumulation by Plants in Roadside Soils: Perspectives for Bioindication and Phytoremediation. Chapter of N.A. Anjum et al. (eds.), *Enhancing Cleanup of Environmental Pollutants*, 2017. https://doi.org/10.1007/978-3-319-55426-6_10
63. Palmieri MJ, Lubber J, Andrade–Vieira LF, Davide LC (2014) Cytotoxic and phytotoxic effects of the main chemical components of spent pot–liner: A comparative approach. *Mutation Res* 763:30–35. <https://doi.org/10.1016/j.mrgentox.2013.12.008>
64. Phoungthong K, Zhang H, Shao LM, He PJ (2016) Variation of the phytotoxicity of municipal solid waste incinerator bottom ash on wheat (*Triticum aestivum* L.) seed germination with leaching conditions. *Chemosphere* 146:547–554. <https://doi.org/10.1016/j.chemosphere.2015.12.063>
65. PN-G-04571 Solid fuels - Determination of carbon, hydrogen and nitrogen content with automatic analyzers - Macro method (in Polish)
66. PN-G-04584 Polish Standard. Solid fuels - Determination of total sulfur and ash content with automatic analyzers. (in Polish)
67. Praveen A, Pandey C, Khan E, Gupta M (2017) Selenium enriched Garden Gress (*Lepidium sativum* L.): Role of antioxidants and stress markers. *Russ Agricult Sci* 43(2):134–137. <https://doi.org/10.3103/S1068367417020033>
68. Saha JK, Selladurai R, Coumar MV, Dotaniya ML, Kundu S, Patra AK (2017) Assessment of heavy metals contamination in soil. In: *Soil Pollution - An Emerging Threat to Agriculture*. Environmental Chemistry for a Sustainable World, vol 10. Springer, Singapore. https://doi.org/10.1007/978-981-10-4274-4_7
69. Salihoglu G, Pinarli V (2008) Steel foundry electric arc furnace dust management: Stabilization by using lime and Portland cement. *J Hazard Mater* 153:1110–1116. <https://doi.org/10.1016/j.jhazmat.2007.09.066>
70. Seneviratne M, Rajakaruna N, Rizwan M, Madawala HMSP, Ok YS, Vithanage M (2017) Heavy metal–induced oxidative stress on seed germination and seedling development: A critical review. *Environ Geochem Health*. <https://doi.org/10.1007/s10653-017-0005>
71. Seregin IV, Ivanov VB (2001) Physiological aspects of cadmium and lead toxic effects on higher plants. *Russian J Plant Physiology* 48(4):523–544.
72. Shanker K, Cervantes C, Loza-Tavera H, Avudainayagam S (2005) Chromium toxicity in plants, *Environ International* 31(5):739–753.

73. Sharma RK, Devi S, dan Dhyani PP (2010) Comparative assessment of the toxic effects of copper and cypermethrin using seeds of *Spinacia Oleracea* L. plants. *Tropical Ecology* 51(2) supplement: 375–387.
74. Siddique R, Kaur G, Rajor A (2010) Waste foundry sand and its leachate characteristics, *Res Conserv Recycling* 54:1027–1036. <https://doi.org/10.1016/j.resconrec.2010.04.006>
75. Skvara F, Kastanek F, Pavelkova I, Solcova O, Maleterova Y, Schneider P (2002) Solidification of waste steel foundry dust with Portland cement. *J Hazard Mater* B89:67–81.
76. Sofilic T, Rastovcan-Mioc A, Cerjan-Stefanovic S, Novosel-Radovic V, Jenko M (2004) Characterization of steel mill electric-arc furnace dust, *Journal of Hazardous Materials* B109:59–70.
77. Stevens DP, McLaughlin MJ, Randall PJ, Keerthisinghe G (2000) Effect of fluoride supply on fluoride concentrations in five pasture species: Levels required to reach phytotoxic or potentially zootoxic concentrations in plant tissue. *Plant Soil* 227:223–233. <https://doi.org/10.1023/A:1026523031815>
78. Strobos JG, Friend JFC (2004) Zinc recovery from baghouse dust generated at ferrochrome foundries. *Hydrometallurgy* 74:165–171. <https://doi.org/10.1016/j.hydromet.2004.03.002>
79. US EPA Method 7000B. SW-846 Test Method 7000B: Flame Atomic Absorption Spectrophotometry
- Vijayarengan P, Mahalakshmi G (2013) Zinc toxicity in tomato plants. *World Applied Sci J* 24(5):649–653.
80. Visioli G, Conti FD, Gardi C, Menta C (2014) Germination and root elongation bioassays in six different plant species for testing Ni contamination in soil. *Bull Environ Contam Toxicol* 92:490–496. <https://doi.org/10.1007/s00128-013-1166-5>
81. Vochozka M, Marouskova A (2017) Implications of the EU green energy policy on financial performance of crop production and water management of topsoil, *Clean Techn Environ Policy* 19:603–609. <https://doi.org/10.1007/s10098-016-1256-2>
82. Wahid A, Ghani A, Ali I, Ashraf MY (2007) Effects of cadmium on carbon and nitrogen assimilation in shoots of mungbean [*Vigna radiata* (L.) Wilczek] seedlings. *J Agron Crop Sci* 193:357–365.
83. Zanetti M, Godio A (2006) Recovery of foundry sands and iron fractions from an industrial waste landfill. *Resources Conserv Recycling* 48: 396–411.
84. Zhang H, Su L, Li X, Zuo J, Liu G, Wang Y (2014) Evaluation of soil microbial toxicity of waste foundry sand for soil-related reuse. *Front Environ Sci Eng* 8(1): 89–98. <https://doi.org/10.1007/s11783-013-0591-3>
85. Zou J, Wang M, Jiang W, Liu D (2006) Chromium accumulation and its effects on other mineral elements in *Amaranthus viridis* L. *Acta Biologica Cracoviensia Series Botanica* 48(1):7–12.
86. Zucconi F, Monaco A, Forte M, De Bertoldi M (1985) Phytotoxins during the stabilization of organic matter. In: Gasser JKR (ed) *Composting of agricultural and other wastes*. London: Elsevier pp.73–85.
87. Zucconi F, Pera A, Forte M, De Bertoldi M (1981) Evaluating toxicity of immature compost. *BioCycle* 22(2):54–57.

Figures

Figure 1

The content of heavy metals [mg/kg DM] in the tested foundry wastes (Bożym 2020).

Figure 2

Dry mass [g DM] of shoots of *L. sativum* L.. No plants have grown on the SGD, RD and EAFD substrate.

Figure 3

Concentration of heavy metals [mg/kg DM] in shoots of *L. sativum* L. Heavy metals content of control samples was below the LOQ (Cd 0.2; Pb 1; Cu, Zn, Ni, Cr, Co 0.5; Mo 2 mg/kg DM).

Figure 4

Germination index (GI) [%] calculated to direct test (GI 1) form current study and the leachate test (GI 2) from previous study (Bożym 2020).

Figure 5

The root length of *L. sativum* L. after 72h of GI test, directly on substrate (direct test) from current study and on leachate (leachate test) from previous study (Bożym 2020).