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# LIGHT ASSISTED PHYTOEXTRACTION OF LANDFILL LEACHATE: AN EFFECTIVE TOOL TO ATTENUATE POLLUTANTS IN LANDFILL LEACHATE

**Abstract:** In the presented experiment, we tested the influence of irrigation with different proportions of leachate collected from the leachate sump of municipal solid waste landfill in combination with LED light on the capacity of model plant to remove some heavy metals (HMs) (As, Cr, Cd, Ni and Hg) from the soil and to store them in the biomass of model plant *Sinapis alba* L. There were six variants irrigated either with distilled water (control) or with a combination of distilled water and leachate (20 %, 50 %, 75 %, 90 %, 100 %). The variants were divided into two groups: A (LED) and B (no LED) and concentrations of HMs and their interdependence (concentration of HM in the soil and in the plant) were monitored in the soil/plant samples. The measured values of HMs concentration in the soil and plant samples did not show a positive influence of LED light on the phytoextraction of HMs. A significantly increased ($p \leq 0.05$) sorption of HM by the model plant was demonstrated only in Cd and Hg where the difference between Group A and Group B was approximately 10 mg $kg^{-1}$. Furthermore, enrichment coefficient, *EC*, was established for the respective HMs and experimental variants. *EC* values > 1 indicating "high accumulator plants" were measured for Cd and Hg for all variants of Group A. In Group B, EC values > 1 were recorded for Cd and Ni only in the control variants, and for Hg in all variants with the application of leachate (20 % → 100 %).

**Keywords:** landfill, heavy metals, leachates, phytoextraction, LED, irrigation

## Introduction

The ever-growing industrial production and trade over the last decade resulted in the rapidly increasing generation of municipal solid waste (MSW) in many countries [1-3]. Thanks to economic advantages, landfilling remains a generally accepted and used method for the final disposal of waste [1, 4-7]. The global trend of landfilling applies not only to big cities where daily amounts of landfilled waste represent thousands of megagram, but also to rural areas where similar waste amounts are landfilled every year [8, 9]. Disposal of MSW in landfills brings a number of environmental risks and causes concern about a possible harm to human health due to the pollution of air, soil and groundwater, risk of

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fires and explosions, bad odour or damage to vegetation [10-13]. Main problems connected with landfilling include the generation of potentially explosive gases (waste dump gas) and leachate (seepage water) [1, 8, 14]. Seepage from waste dumps arouses ever more fears, particularly for its toxicity if uncontrolled leaking occurs into the environment [1, 8, 15]. Long-term environmentally sustainable management of landfill leachates is a priority for landfill operators [16, 17].

MSW consists of a wide range of inorganic, natural and xenobiotic compounds, the mixture of which is affected by the composition and pollution potential of leachate [16, 18, 19]. Seepage water (leachate) from MSW landfills is formed when chemical and biological products are released from waste through the decomposition of putrid and organic fractions. These fractions come from the municipal waste which has not been sufficiently sorted and contains for example organic substances that should be disposed ideally by composting rather than by landfilling [1, 18, 19]. It is a result of numerous processes including microbial degradation [16, 20]. Leachates are considered one of the most important issues for landfill operators [17].

Many studies demonstrated leachates to be a significant source of pollutants due to leaching from the waste of dangerous substances [1, 21-23]. Moreover, leachate composition is specific for the given locality and variability within one landfill is often a case [1, 16, 24]. Landfill leachate may contain large amounts of organic matter (biodegradable, and refractory to biodegradation), as well as ammonia-nitrogen, heavy metals (HM), chlorinated organic and inorganic salts [22]. Lead, cadmium, mercury, and chromium can simultaneously prevail in the environment as a result of various human activities. Landfill leachate is certainly one of the main anthropogenic sources for these HMs. Besides, HMs are widely known to be non-essential elements for plants, and they can cause adverse effects on the plant's photosynthetic system, chlorophyll synthesis, and antioxidant enzyme production, resulting in various forms of damage to the plants [17].

Conventional methods used for leachate treatment can be classified into three major groups: (a) leachate transfer: recycling and combined treatment with domestic sewage, (b) biodegradation: aerobic and anaerobic processes and (c) chemical and physical methods: chemical oxidation, adsorption, chemical precipitation, coagulation/flocculation, sedimentation/flotation and air stripping [25-28].

In the most developed countries, legislative requirements for waste water treatment are becoming stricter. However, due to the ageing of waste dumps with ever more stabilised leachates, the conventional methods of treatment (biological or physical and chemical) are not enough to achieve the level of treatment that would be needed to reduce the general impact of landfill leachates on the environment. It means that new, alternative methods have to be designed and can be considered necessary to improve the environment. One of them is phytoextraction which represents one of phytoremediation techniques [28, 29]. These methods should be tested in the laboratory, in lysimetric and field experiments [25, 28, 30]. Fundamental reasons for adopting the process of phytoextraction and its further investigation are mentioned by Nevel et al. [29] and Wang et al. [31]. The authors claim that it represents a low-cost and environment-friendly technology for the remediation of heavy metals or soils contaminated with organic matters. Nevertheless, according to Lasat [32] and Soudek et al. [33], the technology features some essential disadvantages that have to be further explored: 1) limited efficiency of phytoextraction caused by the low production potential of plants realising the process of phytoextraction; 2) length of the phytoextraction process (usually several years or decades); 3) contaminated sites cannot be

used for other purposes during the phytoextraction; 4) pollutants have to be localised in the rhizosphere, i.e. within the reach of root zone; 5) phytotoxicity of pollutants and their concentration in the soil must not exceed the value that can be tolerated by the plant realising the phytoextraction.

Phytoextraction appears to be a promising method [34]. Phytoremediation is a process which utilises live plants to reduce risk on sites with contaminated soils, sludge, sediments and waters through the elimination, degradation or retention of contaminating substances [35]. Phytoremediation represents an extensive package containing phytotransformation, phytostabilisation, rhisofiltration, phytovolatilisation and phytoextraction [33]. Namely the phytoextraction (or phytoaccumulation/phytoabsorption) is an environment-friendly method the principle of which is removal of contaminants from the soil using absorption into plant biomass [33]. Application of phytoextraction is a cheaper and more efficient alternative for the removal and immobilisation of HMs from aqueous solutions [17, 36] and from the contaminated soil [37] (Fig. 1). According to He et al. [37], the main advantage of phytoremediation is low technical cost, easy operation and convenient promotion, all this leading to the low cost of the whole process of phytoremediation and its easy realisation. Phytoremediation covers a wide range of pollutants such as inorganic chemicals including HMs and metalloids, many organic substances including persistent organic pollutants and radioactive elements. Phytoremediation has gained much popularity over the last 20 years and has been considered as an acceptable technique in many countries due to its cost-effectiveness compared to traditional practices [38, 39].

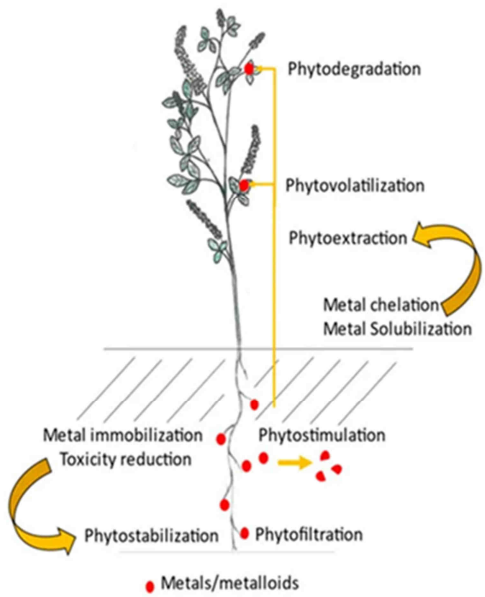


Fig. 1. Processes used in phytoremediation of heavy metal pollutants from the soil according to Ojuederie et al. [40], modified by Kintl et al. [36]

First of all, phytoremediation systems make use of the potential of natural or actively controlled soil-plant system for detoxication, degradation and inactivation of potentially

toxic elements in leachates. Selection of plant species is essential for the long-term success of phytoremediation (phytomanagement). Suitable is a mixture of plant species tolerant to HMs. In this respect, grasses are popular colonisers of soils contaminated with HMs, which can become established and develop an extensive root system thanks to their capability to overcome growth restriction under extreme ecological characteristics of such soils [36, 40-42]. According to OECD [43], plants commonly used for phytoextraction are *Lactuca sativa* (L.), *Lepidium sativum* (L.), *Raphanus sativus* (L.), *Brassica napus* (L.), *Brassica rapa* (L.), *Trifolium pratense* (L.), *Vicia sativa* (L.), *Avena sativa* (L.), *Oryza sativa* (L.), *Lolium perenne* (L.), *Sorghum bicolor* (L.) and *Triticum aestivum* (L.) and *Sinapis alba*.

Previous studies indicated that light of proper intensity, spectral quality, and photoperiod can stimulate the growth and increase the efficiency of the absorption of soil nutrients by plants [43-45]. Accumulation of metals in plant organs is contradictory during the phytoextraction process. Higher concentrations of metals in plants facilitate the removal of a greater mass of pollutants from the soil [46]. However, the accumulation of metals often leads to the overproduction of reactive oxygen species (ROS) in various plant tissues, decreases the biomass yield of the species, and subsequently reduces phytoremediation efficiency [47, 48]. Light irradiation has been applied to activate antioxidant systems in plants in order to scavenge ROS overproduction induced by external stresses in agricultural products. Ma et al. [48] suggested that the activities of superoxide dismutase (SOD) and ascorbate peroxidase (APX) in *Malus domestica* exposed to a greater amount of light were significantly greater than those in the control, which subsequently decreased the production induced by drought stress; Dong et al. [49] found that the increased activity of peroxidase (POD) and SOD in *Triticum aestivum* L. treated with red-blue light improved the straw height and thousand kernel weight of the species; and Levine and Pare [50] observed that light irradiation mitigated the negative effects on the total antioxidant capacity of *Allium fistulosum* L. caused by elevated $CO_2$.

However, the information about how the light irradiation of different quality and intensity influences metal distribution and antioxidation mechanisms of plants during the phytoextraction processes is largely lacking. Once the mechanisms by which the light increases phytoextraction efficiency are determined, the light associated phytoextraction can be widely used in real-scale fields. Ning et al. [51] confirmed that different light spectra can positively affect the capability of plants to mobilise some heavy metals. According to Wang et al. [31], the low efficiency of phytoremediation processes, or more precisely the rate of heavy metals being removed from the contaminated environment, is one of the main disadvantages of phytoremediation. Based on these findings, we can assume that if the use of light diodes (LED) would remove the drawback, phytoremediation processes (including phytoextraction) could be used in real-scale fields in the sense of special facilities, for example vegetation halls with aquaponics or a special form of root wastewater treatment plants that would be able to remove heavy metals from the contaminated environment efficiently by the form of phytoremediation.

LED lamps can be used to provide the narrow-band light spectrum. Compared to conventional fluorescent lamps, they have several advantages such as low energy consumption, longer service life, low heat generation or high conversion efficiency. Moreover, LEDs can influence the growth of plants and accumulation of contaminants [52]. Fu et al. [53] found a positive correlation between light intensity and plant biomass, irrespective of nitrogen concentration, after studying the interaction influences of light and nitrogen fertiliser on the quality of *Lactuca sativa* L. Improvement of light conditions

optimum for achieving maximum plant growth was investigated by several researchers [45, 54, 55]; however, no research of using LED for landfill leachate phytoremediation (phyto-treatment) has been conducted so far. Therefore, the aims of this study are to evaluate the feasibility of using different concentrations of landfill leachate as a fertiliser and, more importantly, the effect of light photoperiod on the process of phytoremediation. Phytoremediation or phytoextraction of contaminants (pollutants) using the plants represents a low-cost method [56] which is however characterised by low efficiency [31]. It is therefore necessary to deal with the feasibility of phytoremediation in the practical decontamination of sites contaminated with heavy metals.

According to Wang et al. [31], Santa-Cruz et al. [56] and Kwon et al. [57], essential influence on the process of phytoextraction of HM's pollutants (its feasibility) and resulting effectiveness is that of 1) plants used for phytoremediation thanks to their growth potential in the contaminated environment and fixation of individual heavy metals; 2) the share of biologically available heavy metals; 3) the influence of abiotic factors affecting the plant growth - exposure to light, temperature and precipitation. Based on this information, it is obvious that feasibility of phytoremediation can be affected especially in points 1) and 2), i.e. by selecting plants suitable for the process of phytoremediation and by stimulating their growth, e.g. by additional LED light, complementary irrigation and the like. Although the research of LED light-irradiated plants is limited, it is safe to hypothesise that light irradiation can change HM distribution and antioxidation mechanisms of plants as well as the pollutant accumulation by stimulating the growth of plants. It follows that the simulated growth of plants together with their capability to deposit pollutants in their biomass is essential for the increased feasibility of phytoremediation [58].

Kwon et al. [57] observed that red LED light improved the ability of *Chlorella vulgaris* to extract metals at significantly high levels by increasing its Cu and Zn concentrations. Therefore, there is a reasonable assumption that the LED photoperiod can also influence the phytoextraction and phytoremediation process efficiency of higher plants.

Therefore, main objectives of this study are to verify the effects of LED irradiation on (1) the potential of phytoextraction of selected species, and (2) HM concentration changes in the soil and plant samples irrigated with landfill leachate at different rates. Two hypotheses were tested: ($H_0$) and an alternative ($H_1$).

$H_0$: Increased light intensity does not have any influence on the sorption of heavy metals by the model plant and on the reduced concentration of heavy metals in the soil.

$H_1$: Increased light intensity applied in selected variants beyond the natural daylight leads to increased sorption of heavy metals by the model plant and hence to the reduced concentration of heavy metals in the soil.

## Materials and methods

### Collection and analysis of samples

In 2018, a mixed sample of leachates (12 litres) was collected from the Kuchynky controlled MSW landfill (category S-OO - Other waste; sub-category SOO-3 Low proportion of biodegradable waste). HM concentrations in the collected seepage water are presented in Table 1. The landfill of this classification receives more than 10 tonnes of waste per day or its total capacity is over 25,000 megagrams of waste. The landfill is situated near Zdounky, cadastral area of Netcice, in the Zlín Region, Czech Republic (CZE). The sample of leachates consisted of a mixture of more samples collected from

different places and different depths of leachate sump on the same date. The samples were collected into sterile receptacles, then mixed, and the resulting sample of leachates was placed into a cooling box and brought to accredited laboratory for analysis. The mixed leachate sample was analysed for the concentration of heavy metals (HMs) As, Cr, Cd, Ni and Hg. Concentrations of As, Cr, Cd and Ni were determined according to CSN EN ISO 11885 [59] and the concentration of Hg was determined according to CSN 75 7440 [60]. The presence of HM in the leachate was established in $\mu g\ L^{-1}$. The analysis was performed in the accredited laboratory LITOLAB Ltd. (Litovel, CZE), accreditation no. 1255, accredited by the Czech Accreditation Institute according to ISO/IEC 17025:2005.

Table 1
Average concentrations of HMs in seepage water collected from the Kuchynky MSW landfill for laboratory experiment irrigation

| Heavy metals | Measured value [$\mu g\ L^{-1}$] | *RSD* [%] |
|---|---|---|
| Cd | 0.183 | 3 |
| As | 29.73 | - |
| Cr | 812.4 | 50 |
| Ni | 153.7 | 20 |
| Hg | 1.32 | 1 |

Average values ($n$ = 3) of HM concentration in the leachate are presented; *RSD* is the value of Relative Standard Deviation that was calculated: 100 · (*SD*/mean)

## Research of light influence in laboratory conditions

The sampled leachates were used in the pot experiment for watering seeds of the tested plant which was white mustard (*Sinapis alba* L.). The plant is highly sensitive to changes in environmental conditions to which it reacts by leaf necrosis or by slowing or stopping its growth. This model plant was chosen thanks to its ability of responding even to relative changes in the concentration of HMs in the soil (environment) as described in OECD [43], Hernandez et al. [44], Šourková et al. [61] and Vasile et al. [62]. The pot experiment lasted 28 days. Its scheme and course are shown in Figure 2. Laboratory conditions: ambient temperature (23 ±5) °C, relative humidity (50 ±2) %.

The pots were filled with 300 g of test substrate which consisted of OECD soil mixture, garden substrate and silica sand, without added artificial fertilisers (hereinafter only substrate), allowing natural germination of seeds and growth of plants in accordance with the European national norm in a ratio of 1 : 1 : 1. OECD soil composition: 5 % of dried peat sifted and homogenised through a 2 mm sieve, 85 % of dried silica sand with a min. 50 % proportion of grains sized 0.05 mm - 0.2 mm, 10 % of dried kaolin clay with a min. 30 % kaolinite content, and 0.3 % - 1% of calcium carbonate for pH treatment to 6.0 ±0.5. One hundred of *Sinapis alba* L. seeds were evenly spread on the surface of each container. The seeds were then covered with a thin layer of silica sand to prevent water evaporation. The plant biomass was cultivated under stable laboratory conditions (temperature (23 ±5) °C, relative air humidity (50 ±2) %). Leachates were applied onto the substrate three times a week for 28 days in different proportions (20 %, 50 %, 75 %, 90 % and 100 %). Individual variants were prepared by diluting the collected (mixed) sample of leachates in the following ratios: 1 : 4 (var. 20 %), 1 : 1 (var. 50 %), 3 : 1 (var. 75 %), 9 : 1 (var. 90 %) and an undiluted sample, i.e. irrigation with concentrated leachates. The irrigation of experimental containers (either distilled water or a mixture of leachate and

distilled water in determined ratio) was performed according to model plant development as follows:

- Day 1 of the experiment 10 mL
- Day 2 - 7 of the experiment 40 mL
- Day 8 - 14 of the experiment 120 mL
- Day 15 - 21 of the experiment 150 mL
- Day 22 - 28 of the experiment 150 mL
- Total irrigation applied for one variant for the entire duration of the experiment was ∑ 470 mL

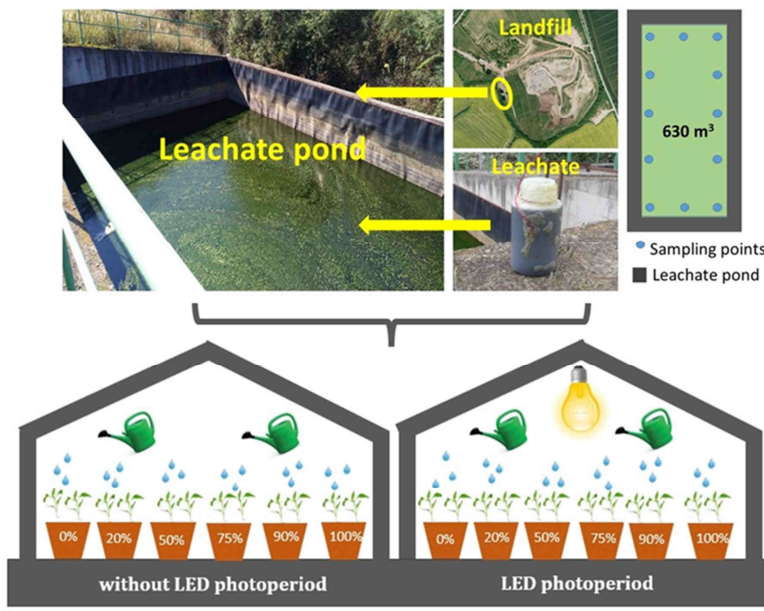


Fig. 2. Scheme of experiment

The substrate that was not irrigated with leachate but with distilled water represented a standard and served as reference. The test was at all times performed in three repetitions. Designation of variants based on the applied leachate proportions was as follows: 0 (control), 25, 50, 75, 90 and 100. Samples exposed to the LED photoperiod and samples without the LED photoperiod are marked with A and B, respectively. Thus, two groups of variants were created: A - LED; B - daylight only (no LED).

In order to find the effect of LED photoperiods on the efficiency of phytoextraction abilities of higher plants (*Sinapis alba* L.) using different concentrations of leachates, the tested containers (i.e. ½ of them) were exposed to the 16 hour cycle (from 6:00 am to 10:00 pm) of controlled LED illumination (Philips cultivation system with the function of timer and brightness of 850 lm; light radiation range 400 nm - 660 nm, angle of exposure to light 120°; Fig. S1). The remaining containers were exposed only to daylight. After the end of the experiment period, soil and biomass were sampled from the containers (soil sample =

soil mixture formed by mixing samples of soils I, II and II for the particular concentration; biomass sample = biomass mixture formed by mixing samples of biomass I, II and II for the particular concentration) and the samples were analysed.

**Analysis of soil and plant biomass samples**

The soil and plant biomass samples were analysed in the accredited laboratory LITOLAB Ltd. (Litovel, Czech Republic), accreditation no. 1255, accredited according to ISO/IEC 17025:2005 by the Czech Accreditation Institute. An overview of used analytical methods is presented in Table 2. Two main methods were used: CSN 75 7440 (Water quality - Determination of total mercury by thermal decomposition, amalgamation and atomic absorption spectrometry [60]) and CSN EN ISO 11885 (Water quality - Determination of selected elements by inductively coupled plasma optical emission spectrometry ICP-OES [59]). These methods can be used not only for water quality monitoring but also for sediments, sludge, plant and soil matrices [63].

Table 2

Methods used to determine the concentrations of HMs in the soil and plant biomass samples

| Parameter | Material | Method |
|---|---|---|
| As | Plant/Soil | CSN EN ISO 11885 [59] |
| Cr | Plant/Soil | CSN EN ISO 11885 [59] |
| Cd | Plant/Soil | CSN EN ISO 11885 [59] |
| Ni | Plant/Soil | CSN EN ISO 11885 [59] |
| Hg | Plant/Soil | CSN 75 7440 [60] |

CSN - Czech Technical Standards, EN - European Standard, ISO - International Organisation for Standardisation

**Enrichment coefficient for the plant/soil system**

Enrichment coefficient, *EC*, was calculated to assess the accumulation of HMs from the soils to plants, and it is described as the following formula [64, 65]:

$$EC = \frac{M_{plant}}{M_{soil}} \quad (1)$$

where *EC* is the enrichment coefficient for the plant/soil system, [-]; $M_{plant}$ is the metal concentration in the plant tissue, [mg $kg^{-1}$], in dry matter; $M_{soil}$ is the total metal concentration in the soil where the plant is grown, [mg $kg^{-1}$], in dry matter.

Malayeri et al. [66] grouped the plant species according to their HM uptake capacities and sensitivity to pollution by metals (Table 3).

Table 3

Classification of *EC* according to Malayeri et al. [66]

| The ability of plants to accumulate heavy metals | ***EC* [-]** |
|---|---|
| High accumulator plants | between 1-10 |
| Moderately accumulator plants | between 0.1-1.0 |
| Low accumulator plants | between 0.01-0.1 |
| Non accumulator plants | < 0.01 |

**Statistical data processing**

The measurements were processed in the statistical programme Statistica 12 (Dell software, Round Rock, USA) using the following analyses: (a) input data analysis -

determination of the homogeneity of measured values - the aim was to determine extreme values possibly occurring in the measured parameters. For the purpose, exploratory data analysis (EDA) was used, which included diagnostic diagrams and a comparison of the distribution of measured values; (b) one factor and two-factor analysis of variance (ANOVA) of the measured sets of values - these methods were used to determine the influence of individual factors on the measured values; (c) post-hoc Tukey's Honest Significant Difference (HSD) to determine significant differences in the measured values among the individual variants; (d) pair *t*-test to identify differences between the groups of data (photoperiod with LED and without LED). All analyses were performed at a significance level of $\alpha < 0.05$.

## Results

### Analysis of leachates

The collected leachates were analysed for the presence of HMs in the accredited laboratory (LITOLAB Ltd., Litovel, Czech Republic). Results for individual HMs (Cd, As, Cr, Ni and Hg) are presented in Table 1. The highest values were recorded in Cr (812.4 µg $L^{-1}$) and Ni (153.7 µg $L^{-1}$). Concentrations of the other HMs in leachates did not exceed 100 µg $L^{-1}$ and the concentration of Hg approached the limit of detectability.

### HM concentrations in the soil and plant biomass samples

The aim of the experiment was to monitor the concentration of some HMs (As, Cr, Cd, Ni, Hg) in the soil in which the model plant (*Sinapis alba* L.) was grown. The LED light variant (Table 4) did not exhibit any significant difference in the concentrations of As, Cd and Hg; more precisely, the concentration of leachate in irrigation did not affect the concentration of these HMs in the soil samples. Moreover, Cd and Hg exhibited the lowest concentrations (< 0.2 mg $kg^{-1}$) in this experimental group. On the other hand, the highest concentrations were recorded in Cr (> 2.9 mg $kg^{-1}$), which was a significant difference ($\alpha < 0.05$) as compared with the other variants.

In case of the second experiment group (Table 4) which was not exposed to another source of light but to daylight only, the structure of measured values was similar in As, Cd and Hg. The concentrations of As and Cd did not show any significant differences among the individual experimental variants. The concentration of As exceeded 1.9 mg $kg^{-1}$ in all variants and conversely, the concentration of Cd in the soil did not exceed 0.09 mg $kg^{-1}$ in any variant. The concentration of Hg exhibited a non-significant difference among the variants “0; 20 and 90“. On the other hand, in case of the other experimental variants (50; 75 and 100), a small but significant decrease in concentration occurred below 0.014 mg $kg^{-1}$ as compared with the control variant. The highest HM concentrations were recorded in Ni (variant 50 and variant 75), where the values exceeded 13 mg $kg^{-1}$. Other significant differences were found in the concentration of Cr which was demonstrably increasing in the soil samples from variant “0” up to variant “50” where the highest Cr concentration was recorded (5.05 mg $kg^{-1}$). The following variants (75, 90, 100) then showed a demonstrable decline.

We also monitored HM concentration in the above-ground biomass of the model plant (*Sinapis alba* L.) both in the experiment group with additional LED lighting (Table 5) and with no LED (Table 5). Average concentrations of heavy metals in Group A greatly varied. The highest concentrations of HMs in the plant biomass were those of Cr and Ni. In these

two HMs, the recorded values ranged around 1.00 mg kg$^{-1}$ in case of Ni and above this limit in case of Cr. The concentration of Cr in the plant biomass was demonstrably increasing from the control variant (1.11 mg kg$^{-1}$) to (1.73 mg kg$^{-1}$) in the variant in which irrigation was applied with 75% of leachates. The growing proportion of leachates resulted in the decreased concentration of Cr at first, and then a mild increase followed to 1.65 mg kg$^{-1}$. The absolutely highest concentrations of all HMs across the variants were those of Cr in variants "50" (1.68 mg kg$^{-1}$); 75 (1.73 mg kg$^{-1}$) and 100 (1.65 mg kg$^{-1}$). The concentration of Ni in the plant biomass was not so variable; however, it copied the trend with the concentration of 1.00 mg kg$^{-1}$ in the control variant and the demonstrably highest value of 1.09 mg kg$^{-1}$ in variant "75". Then a decline was recorded to 0.91 mg kg$^{-1}$ in the variant with the absolute concentration of leachate in irrigation.

Table 4

Average HM concentration in the soil samples of individual experiment variants A - LED; B - no LED

| **Variant A** | | | | | |
|---|---|---|---|---|---|
| **Treatment** | **As ±*SD* [mg kg$^{-1}$]** | **Cr ±*SD* [mg kg$^{-1}$]** | **Cd ±*SD* [mg kg$^{-1}$]** | **Ni ±*SD* [mg kg$^{-1}$]** | **Hg ±*SD* [mg kg$^{-1}$]** |
| 0 | 1.97 ±0.08 a | 4.14 ±0.17 ab$^{*}$ | 0.099 ±0.004 a | 0.99 ±0.04 d | 0.019 ±0.001 a |
| 20 | 2.05 ±0.07 a | 4.87 ±0.20 a$^{*}$ | 0.103 ±0.004 a | 6.88 ±0.29 a | 0.015 ±0.001 a |
| 50 | 1.97 ±0.15 a | 2.94 ±0.12 c$^{*}$ | 0.106 ±0.007 a | 1.06 ±0.04 d | 0.016 ±0.001 a |
| 75 | 2.01 ±0.13 a | 3.29 ±0.14 c$^{*}$ | 0.103 ±0.009 a | 1.43 ±0.06 d | 0.014 ±0.001 a |
| 90 | 1.91 ±0.15 a | 3.33 ±0.14 c$^{*}$ | 0.103 ±0.009 a | 2.09 ±0.09 bc | 0.016 ±0.001 a |
| 100 | 2.09 ±0.12 a | 3.52 ±0.15 b$^{*}$ | 0.103 ±0.009 a | 1.48 ±0.06 c | 0.017 ±0.001 a |
| **Variant B** | | | | | |
| **Treatment** | **As ±*SD* [mg kg$^{-1}$]** | **Cr ±*SD* [mg kg$^{-1}$]** | **Cd ±*SD* [mg kg$^{-1}$]** | **Ni ±*SD* [mg kg$^{-1}$]** | **Hg ±*SD* [mg kg$^{-1}$]** |
| 0 | 1.97 ±0.08 a | 3.00 ±0.12 b$^{*}$ | 0.099 ±0.004 a | 1.07 ±0.04 c | 0.016 ±0.001 a |
| 20 | 1.95 ±0.05 a | 2.84 ±0.12 b | 0.099 ±0.004 a | 4.37 ±0.18 b | 0.014 ±0.001 a |
| 50 | 2.03 ±0.08 a | 5.05 ±0.21 a | 0.099 ±0.004 a | 13.02 ±0.54 a$^{*}$ | 0.013 ±0.001 b |
| 75 | 1.99 ±0.04 a | 2.75 ±0.11 b | 0.099 ±0.004 a | 13.71 ±0.57 a$^{*}$ | 0.012 ±0.001 b |
| 90 | 2.05 ±0.04 a | 2.82 ±0.12 b$^{*}$ | 0.099 ±0.004 a | 1.96 ±0.08 c | 0.014 ±0.001 a |
| 100 | 1.98 ±0.11 a | 2.87 ±0.15 b | 0.099 ±0.004 a | 6.98 ±0.29 b$^{*}$ | 0.012 ±0.001 b |

Different lower-case letters indicate significant differences in the concentration of specific HMs between the individual variants either in Group A or in Group B. Symbol $^{*}$ indicates significant differences ($\alpha < 0.05$) in the concentration of particular HM among all HMs in the given variant

The other HMs exhibited a similar trend, for example As or Hg, but differences among the individual variants were small, which was the case of Cd whose concentrations in the plant biomass ranged from 0.20 mg kg$^{-1}$ to 0.34 mg kg$^{-1}$. On the contrary, the As concentrations ranged from 0.30 mg kg$^{-1}$ (control variant) to 1.02 mg kg$^{-1}$ (variant with 20 % of leachates in irrigation). The Hg concentration showed relatively small differences; the highest value (0.15 mg kg$^{-1}$) was recorded in variant "75" and the lowest value (0.07 mg kg$^{-1}$) was found in the variant with the highest concentration of leachates.

In absence of additional LED lighting (Table 5), the highest concentrations were found in Cr and Ni again and ranged above 1.3 mg kg$^{-1}$ (with the exception of control variant for Cr) while only partial differences were found within the individual variants with different leachate concentrations in irrigation. The concentration of Cr in the plant biomass was gradually increasing ($P < 0.05$) with the share of leachates in irrigation water up to 50 % where a decline occurred. A similar trend ($P < 0.05$) was recorded in Ni, too. However, a declined concentration of this HM in the plant biomass to ca. 1.6 mg kg$^{-1}$ occurred only

from the concentration of 75 % of leachates in irrigation water. The remaining HMs (As, Cd and Hg) showed a similar trend ($P < 0.05$); their slightly increasing concentrations in the plant biomass were recorded from variant "0" but then the trend stopped and their decline started with the increasing proportion of leachates (from the concentration of 75 %) in irrigation water.

Table 5

Average concentrations of HMs in the plant biomass samples from individual experiment variants A - LED; B - no LED

| **Variant A** | | | | | |
|---|---|---|---|---|---|
| **Treatment** | **As ±*SD* [mg kg$^{-1}$]** | **Cr ±*SD* [mg kg$^{-1}$]** | **Cd ±*SD* [mg kg$^{-1}$]** | **Ni ±*SD* [mg kg$^{-1}$]** | **Hg ±*SD* [mg kg$^{-1}$]** |
| 0 | 0.30 ±0.01 c | 1.11 ±0.05 cd | 0.27 ±0.011 b | 1.00 ±0.04 ab | 0.12 ±0.005 b |
| 20 | 1.02 ±0.04 a | 1.40 ±0.06 c | 0.31 ±0.013 ab | 1.00 ±0.05 ab | 0.11 ±0.004 bc |
| 50 | 0.41 ±0.02 dc | 1.68 ±0.07 b* | 0.25 ±0.011 b | 0.88 ±0.04 b | 0.09 ±0.004 cd |
| 75 | 0.75 ±0.03 b | 1.73 ±0.07 a* | 0.21 ±0.009 c | 1.09 ±0.06 a | 0.15 ±0.006 a |
| 90 | 0.34 ±0.01 c | 1.25 ±0.05 c | 0.20 ±0.008 c | 0.94 ±0.04 ab | 0.12 ±0.005 b |
| 100 | 0.46 ±0.02 c | 1.65 ±0.04 b* | 0.34 ±0.014 a | 0.91 ±0.05 ab | 0.07 ±0.003 d |
| **Variant B** | | | | | |
| **Treatment** | **As ±*SD* [mg kg$^{-1}$]** | **Cr ±*SD* [mg kg$^{-1}$]** | **Cd ±*SD* [mg kg$^{-1}$]** | **Ni ±*SD* [mg kg$^{-1}$]** | **Hg ±*SD* [mg kg$^{-1}$]** |
| 0 | 1.13 ±0.05 b | 0.93 ±0.04 c | 0.15 ±0.006 a | 1.39 ±0.06 b | 0.01 ±0.001 e |
| 20 | 1.60 ±0.07 a | 1.69 ±0.07 b | 0.08 ±0.004 b | 1.45 ±0.06 b | 0.09 ±0.003 a |
| 50 | 1.08 ±0.04 cb | 2.41 ±0.10 a | 0.09 ±0.004 b | 1.97 ±0.08 a | 0.07 ±0.003 b |
| 75 | 1.00 ±0.04 cb | 2.30 ±0.10 a* | 0.03 ±0.001 c | 1.44 ±0.07 b | 0.04 ±0.002 c |
| 90 | 0.30 ±0.01 d | 2.49 ±0.09 a* | 0.04 ±0.002 c | 1.56 ±0.06 b | 0.02 ±0.001 d |
| 100 | 0.92 ±0.04 c | 2.12 ±0.06 a | 0.09 ±0.004 b | 1.66 ±0.04 b | 0.05 ±0.002 c |

See Table 4

Table 6

Linear correlation (Pearson's correlation coefficient *R*) between HMs concentration in soil and plant biomass - A: with LED

| **Variables** | **As PLANT** | **Cr PLANT** | **Cd PLANT** | **Ni PLANT** | **Hg PLANT** |
|---|---|---|---|---|---|
| As SOIL | 0.28 | 0.46 | **0.51** | **0.59** | 0.08 |
| Cr SOIL | **0.56** | –0.34 | **0.56** | 0.44 | 0.12 |
| Cd SOIL | 0.13 | 0.45 | 0.22 | 0.48 | 0.16 |
| Ni SOIL | **0.81** | –0.12 | 0.33 | 0.18 | 0.00 |
| Hg SOIL | **–0.51** | **–0.54** | 0.26 | 0.14 | 0.08 |

***Bold R*** indicates a significant correlation ($\alpha < 0.05$)

In addition to the HM concentrations in the soil and plant biomass samples, we also analysed their mutual relationships. Table 6 and Table 7 bring information about relationships between the content of particular HMs in the soil and in the plants. The above tables present the values of "Pearson's correlation coefficient *R*" among the measured parameters. Bold *R* indicates a significant correlation $\alpha < 0.05$ The group of experiment that was exposed to LED light (Table 6) did not show any positive correlation between the same HMs concentration in the soil and in the plant. For example, the *R* value for As in the soil and As in the plant biomass was only 0.28. In terms of significance, only partial dependencies were recorded. The soil concentration of As only weakly correlated with Cd and Ni in the plant biomass. Cr in the soil positively correlated with As and Cd concentrations in the plant samples. The only HM detected in the soil that had no

correlation with the concentration in the plant biomass was Cd. Conversely, the concentration of Ni exhibited the highest *R* value (*R* = 0.81) towards the content of As in the plant biomass.

Apart from the relationships of HMs in the soil and in the plant biomass, we also analysed mutual dependencies of all parameters. A complete correlation matrix is presented in Supplementary materials to Table S1 for the group with the additional light and in Table S2 for the group without the additional light. The concentrations of individual HMs in the soil exhibited only strong positive correlations (*R* > 0.75): As vs Cd, and Cr vs Ni. There was no demonstrable negative correlation. As to the content of HMs in the plant biomass, only a moderately strong negative correlation (*R* = –0.6) was found between Cd and Hg, and a very strong positive correlation (*R* = 0.78) between Hg and Ni.

Table 7

Linear correlation (Pearson´s correlation coefficient *R*) between HMs content in soil and plant biomass - B: no LED

| Variables | As PLANT | Cr PLANT | Cd PLANT | Ni PLANT | Hg PLANT |
|---|---|---|---|---|---|
| As SOIL | –0.02 | 0.35 | 0.03 | **0.54** | 0.08 |
| Cr SOIL | 0.13 | 0.31 | 0.22 | **0.87** | 0.41 |
| Cd SOIL | 0.15 | 0.21 | 0.12 | 0.42 | 0.10 |
| Ni SOIL | 0.15 | **0.57** | –0.41 | 0.49 | 0.34 |
| Hg SOIL | 0.17 | **–0.54** | **0.63** | –0.10 | –0.20 |

See Table 6

Correlation between HM contents in the soil and in the plant biomass was analysed also in the group that was not exposed to LED light (Table 7). We found again no correlation between the concentration of particular HMs in the soil and in the plant biomass. There were only four demonstrably positive and one negative correlations. One of positive correlations was that between the Ni concentration in the biomass of plants and the contents of As and Cr in the soil. The correlation was very strong (*R* = 0.87) in case of Cr. Furthermore, a positive correlation and a negative correlation were found between Cr in the plant biomass and Ni (*R* = 0.57) and Hg (*R* = –0.54) in the soil.

Similarly, as in the group with LED light, mutual correlations of all parameters were analysed also in the group with no LED light (see the correlation matrix in Table S2). Provability of correlations among HMs in the soil was similar as in the previous group. A positive correlation was found in As vs Cd (*R* = 0.88) and in Cr vs Ni (*R* = 0.52). The trend was similar as in the group with LED light, only with a lower *R* value in case of Cr vs Ni. Furthermore, a moderately strong (*R* = 0.62) negative correlation was found between the concentration of Hg in the soil and Ni. Other dependencies were not detected.

As to the correlations of individual HMs in the plant biomass, Group B exhibited diametrically different *R* values than the group with LED light. A strong negative correlation (*R* = –0.79) was found between Cr and Cd. Conversely, moderately strong positive correlations were recorded in Hg vs As (*R* = 0.64) and Ni vs Cr (*R* = 0.59).

## Effect of light intensity on the sorption of HMs by plants and HMs concentrations in the soil

The effect of different light amount on the sorption of HMs was studied from the point of HM concentration in the plant biomass (Fig. 3) and in the soil samples (Fig. 4). Average concentration of HMs in the soil without distinguishing individual experiment variants in

terms of irrigation are shown in Figure 2. In Group A (LED), the highest value was recorded in Cr (3.68 mg $kg^{-1}$), demonstrably towards the other HMs. The lowest value was detected in Cd (0.10 mg $kg^{-1}$) and Hg (0.017 mg $kg^{-1}$). In Group B (no LED), the highest value was recorded in Ni (6.8 mg $kg^{-1}$) and the lowest values were found in Cd (0.09 mg $kg^{-1}$) and Hg (0.013 mg $kg^{-1}$), i.e. in the same HMs as in Group A. Comparing the groups, we found differences only for Ni and Hg. The concentration of Ni in Group B was demonstrably higher than that in Group A. Conversely, the concentration of Hg was demonstrably lower. The average values are compared in Table 8 with significant differences being highlighted.

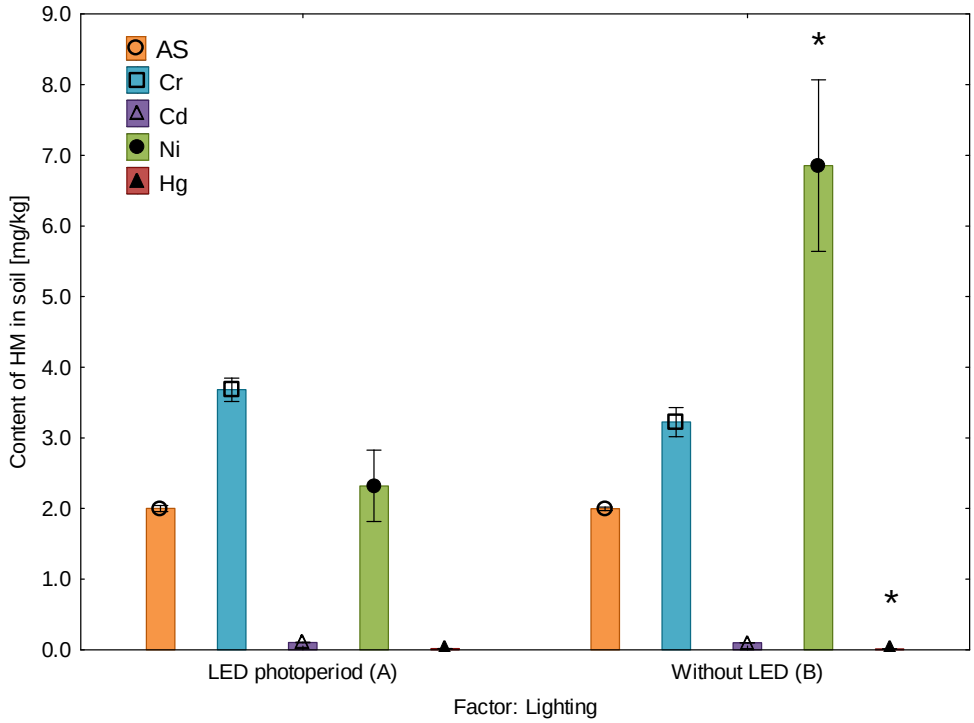


Fig. 3. Average concentrations of HMs in the soil in groups with and without LED photoperiod. Average concentrations of HMs in the soil are shown for experimental groups A with LED ($n$ = 18) and B with no LED photoperiod ($n$ = 18) ±SD after the end of the experiment. Symbol * indicates a significant difference (pair $t$-test; $\alpha < 0.05$) in the concentration of individual HMs between Group A and Group B

The measured values (Fig. 3 and Table 8) show that additional LED lighting had no clear influence on the concentration of HMs in the soil. This is evident from the values listed in Table 8, where the results of the pair $t$-test. Above all this assumption was to be confirmed by two-way ANOVA combined with the multivariate significance test whose results are presented in Table S3. One of the important goals of the laboratory experiment was to analyse the influence of leachate concentrations in irrigation water and additional light. A graphical representation of the performed analysis from which it is obvious that neither factor dominated is shown in Figure 4. The Shapiro-Wilk test revealed that the

supplement of leachates affected the concentration of HMs in the soil. $\alpha$-value was smaller than 0.000000002 and in case of additional LED light it was $\alpha < 0.000000020$. Therefore, it follows that the effect of the two factors on the concentration of HMs in the soil was similar.

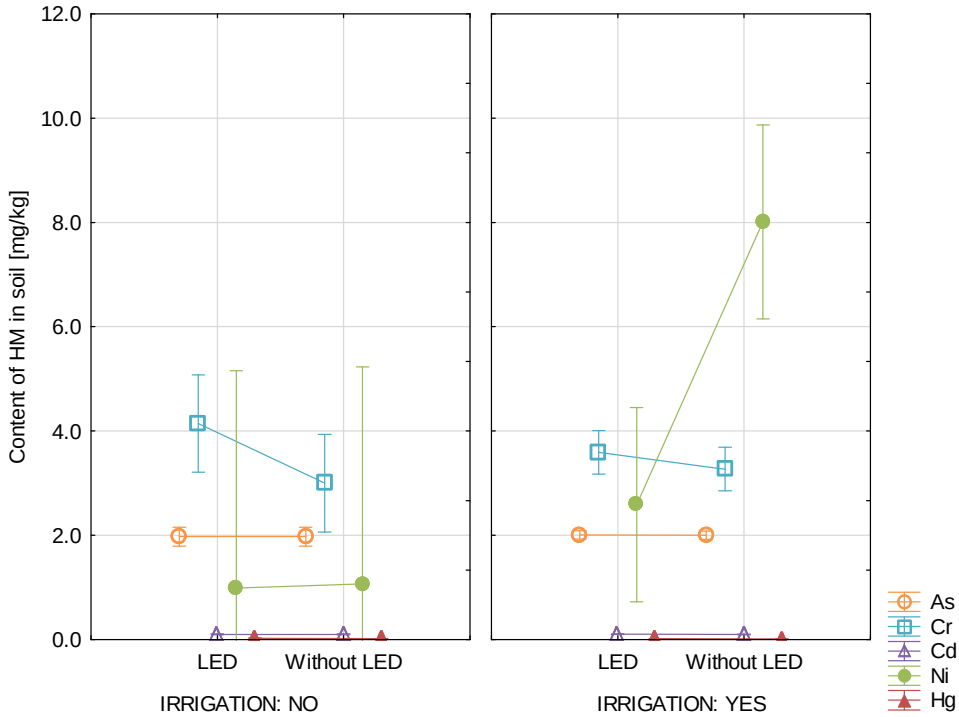


Fig. 4. Comparison of the effect of leachates and lighting on the concentration of HMs in the soil. Results of two-way ANOVA are shown as decomposition of effective hypothesis. Vertical lines represent confidence intervals at a level of $\alpha = 0.95$

Table 8

Comparison of average HM concentrations in the soil samples in dependence on light intensity using pair *t*-test

| Variable | Average (LED) | Average (w/o LED) | *t* | *α* | *n* (LED) | *n* (w/o LED) | *SD* (LED) | *SD* (w/o LED) | *F* |
|---|---|---|---|---|---|---|---|---|---|
| As SOIL [mg kg$^{-1}$] | 2.001 | 1.9956 | 0.1095 | 0.9134 | 18 | 18 | 0.1848 | 0.1103 | 2.81 |
| Cr SOIL [mg kg$^{-1}$] | 3.6819 | 3.2231 | 1.7441 | 0.0902 | 18 | 18 | 0.7009 | 0.8685 | 1.54 |
| Cd SOIL [mg kg$^{-1}$] | 0.1029 | 0.0987 | 1.4575 | 0.1541 | 18 | 18 | 0.0107 | 0.0060 | 3.23 |
| Ni SOIL [mg kg$^{-1}$] | 2.3203 | 6.8524 | **–3.4490** | **0.0015** | 18 | 18 | 2.1375 | 5.1488 | 5.80 |
| Hg SOIL [mg kg$^{-1}$] | 0.0171 | 0.0133 | **3.5764** | **0.0011** | 18 | 18 | 0.0042 | 0.0016 | 6.65 |

Significant differences between Group A - LED and Group B - w/o LED (no LED photoperiod) are in bold ($\alpha < 0.05$). $\alpha$ = level of significance at which the significant difference was found, $t$ = test criterion value, $F$ = ratio of the variances of data sets from Group A and Group B, $n$ = range of the analysed set of data, *SD* = standard deviation of these sets

The concentration of HMs in the plant biomass was analyzed from the point of total average concentration in the group without the additional light (A) and in the group with the additional light (B). The measured values show (Fig. 5) that the concentrations of selected HMs (As, Cr, Cd, Ni, Hg) differed between the two groups. Group B exhibited demonstrably higher concentrations of As (1.01 mg $kg^{-1}$), Cr (1.99 mg $kg^{-1}$) and Ni (1.57 mg $kg^{-1}$). Concentrations of Hg (0.11 mg $kg^{-1}$) and Cd (0.26 mg $kg^{-1}$) were demonstrably higher in Group A. From the statistical point of view, the effect of additional light was different for the individual HMs. The analysis of average HM concentrations in the plant biomass is summarized in Table 9 where pair *t*-test results are shown.

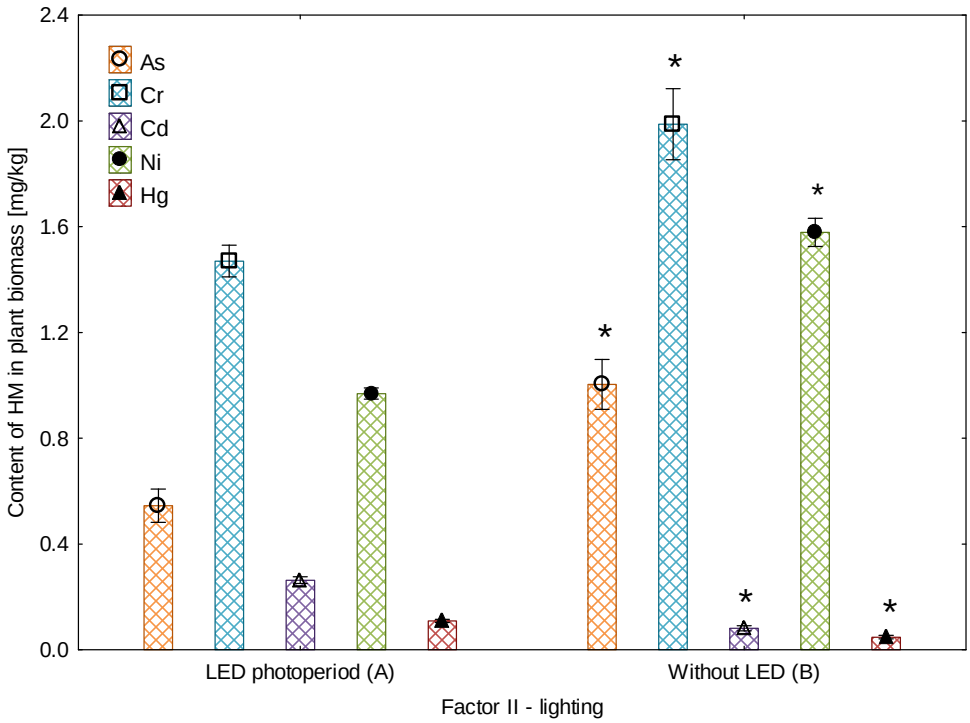


Fig. 5. Average HM concentrations in the plant biomass in groups with and without LED photoperiod. Average values of HM concentrations in the plant biomass are presented for individual experimental groups A with LED (n = 18) and B without LED (n = 18) ±*SD* after the end of the experiment. Symbol * indicates a significant difference (pair t-test; $\alpha < 0.05$) in the concentration of particular HMs between Group A and Group B

The influence of individual factors on the concentration of HMs in the plant biomass was analysed similarly as in case of their influence in the soil. The recorded values show (Fig. 5 and Table 9) that the concentrations of HMs differed between the groups with the additional LED light and without it. In order to confirm the assumption, two-ways ANOVA was performed in combination with the multivariate significance test whose results are presented in Table S4 and graphically shown in Figure 6 illustrating the influence of individual factors on the concentration of HMs in the biomass of plants. The analysis shows that the factor of using leachates affected ($\alpha < 0.001$) the total concentration of HMs in the

plant biomass. Its effect on the sorption of individual HMs was however lower than the effect of additional lighting ($\alpha < 0.001$).

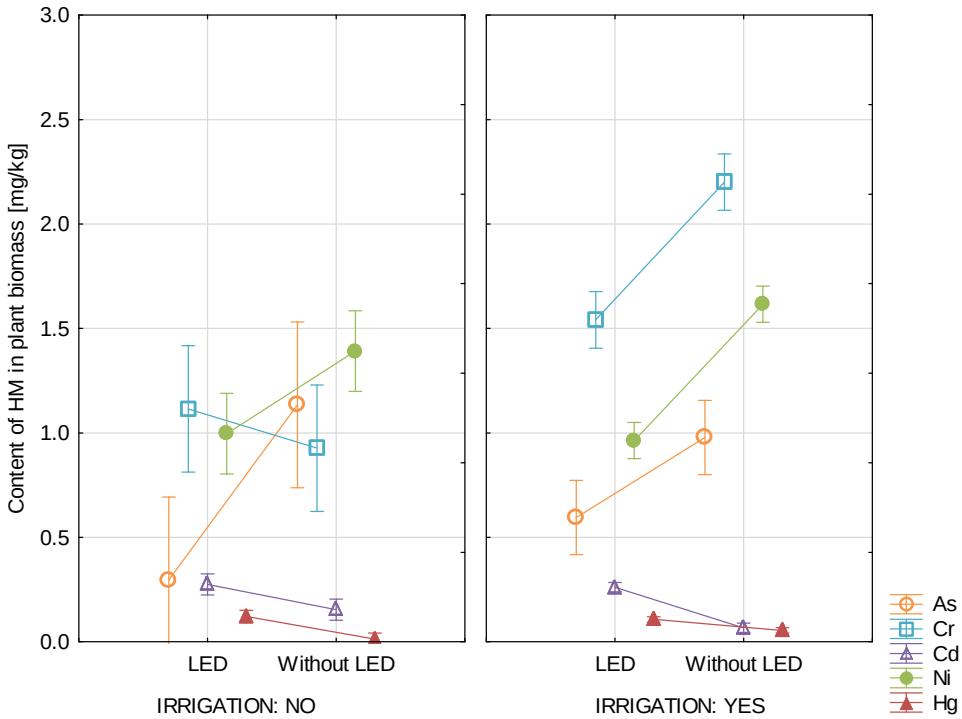


Fig. 6. Comparison of the effect of leachate and additional light on the concentration of HMs in the plant biomass. Results of two-way ANOVA are shown as decomposition of effective hypothesis. Vertical lines represent confidence intervals at a level of $\alpha = 0.95$

Table 9

Comparison of average HM concentrations in the plant biomass in dependence on light intensity using pair *t*-test

| Variable | Average (LED) | Average (w/o LED) | *t* | *α* | *n* (LED) | *n* (w/o LED) | *SD* (LED) | *SD* (w/o LED) | *F* |
|---|---|---|---|---|---|---|---|---|---|
| As PLANT [mg kg$^{-1}$] | 0.5451 | 1.0039 | **–4.0494** | **0.000281** | 18 | 18 | 0.2664 | 0.4001 | 2.26 |
| Cr PLANT [mg kg$^{-1}$] | 1.4701 | 1.9880 | **–3.5196** | **0.001252** | 18 | 18 | 0.2535 | 0.5704 | 5.06 |
| Cd PLANT [mg kg$^{-1}$] | 0.2636 | 0.0812 | **11.4352** | **0.000001** | 18 | 18 | 0.0529 | 0.0420 | 1.59 |
| Ni PLANT [mg kg$^{-1}$] | 0.9689 | 1.5787 | **–10.6929** | **0.000001** | 18 | 18 | 0.0904 | 0.2244 | 6.17 |

| Variable | Average (LED) | Average (w/o LED) | *t* | *α* | *n* (LED) | *n* (w/o LED) | *SD* (LED) | *SD* (w/o LED) | *F* |
|---|---|---|---|---|---|---|---|---|---|
| Hg PLANT [mg kg$^{-1}$] | 0.1097 | 0.0480 | **6.7447** | **0.000001** | 18 | 18 | 0.0252 | 0.0295 | 1.37 |

Significant differences between Group A - LED and Group B - w/o LED (no LED photoperiod) are in **bold** ($\alpha < 0.05$). $\alpha$ = level of significance at which the significant difference was found, $t$ = test criterion value, $F$ = ratio of the variances of data sets from Group A and Group B, $n$ = range of the analysed set of data, *SD* = standard deviation of these sets

## Enrichment coefficient for the plant/soil system

EC values (Eq. 1) were calculated based on the resulting HM concentrations in the samples of soil and plant biomass. Results of EC were then evaluated according to the classification presented in Table 2 and divided into Group A (LED) and Group B (no LED).

Table 10

Enrichment coefficient, *EC* [-], for the plant/soil system - Group A with LED

| Tested concentration | $EC_{As}$ | $EC_{Cr}$ | $EC_{Cd}$ | $EC_{Ni}$ | $EC_{Hg}$ |
|---|---|---|---|---|---|
| 0 | 0.15 | 0.27 | 2.70 | 1.01 | 4.00 |
| 20 | 0.50 | 0.29 | 3.10 | 0.15 | 11.00 |
| 50 | 0.21 | 0.57 | 2.27 | 0.83 | 4.50 |
| 75 | 0.37 | 0.53 | 2.10 | 0.76 | 15.00 |
| 90 | 0.18 | 0.38 | 2.00 | 0.45 | 6.00 |
| 100 | 0.22 | 0.47 | 3.40 | 0.61 | 3.50 |

| | |
|---|---|
| High accumulator plants | between 1-10 |
| Moderately accumulator plants | between 0.1-1.0 |
| Low accumulator plants | between 0.01-0.1 |
| Non accumulator plants | < 0.01 |

All tested leachate proportions (0 % - 100 %) in Group A with LED exhibited *EC* values for As and Cr ranging from 0.1 to 1.0 moderately accumulator plants (Table 10), indicating a mild degree of HM accumulation by the model plant (*Sinapis alba* L.). Ni reached these *EC* values at concentrations 20-100, and the standard ("0") *EC* exhibited a concentration of 1.01, i.e. high accumulation by plant. The high rate of accumulation (*EC* 1-10) was recorded in all tested concentrations of Cd and Hg as well as Ni in the control sample (standard). The tested samples in all experimental variants (at all leachate concentrations) exhibited accumulation capacity for HMs (*EC* 0.15- 15), i.e. for As, Cr, Cd, Ni and Hg.

Table 11

Enrichment coefficient, *EC* [-], for the plant/soil system - Group B (no LED)

| Tested concentration | $EC_{As}$ | $EC_{Cr}$ | $EC_{Cd}$ | $EC_{Ni}$ | $EC_{Hg}$ |
|---|---|---|---|---|---|
| 0 | 0.57 | 0.31 | 1.50 | 1.30 | 0.50 |
| 20 | 0.81 | 0.60 | 0.80 | 0.33 | 9.00 |
| 50 | 0.53 | 0.48 | 0.90 | 0.15 | 7.00 |
| 75 | 0.50 | 0.84 | 0.30 | 0.11 | 4.00 |
| 90 | 0.15 | 0.88 | 0.40 | 0.80 | 2.00 |
| 100 | 0.46 | 0.74 | 0.90 | 0.24 | 5.00 |

*EC* results for Group B with no LED indicated a lower rate of accumulation in all tested concentrations as compared with Group A. A high rate of accumulation (*EC* 1-10) was exhibited by the heavy metals only in the control variant "0": Cd (1.5), Ni (1.3) and also Hg in the tested concentrations 20 - 100 (2.00 - 9.00). All tested variants (leachate concentrations in irrigation water 0 % - 100 %) without LED exhibited EC values for As and Cr ranging from 0.1-1.0, i.e. a mild degree of accumulation by the plant. The mild degree of accumulation ($EC < 1$) was also exhibited by Cd and Ni in concentrations 20-100 and Hg in the control variant ("0").

## Discussion

### Analysis of leachates

HM concentrations in the tested leachates were very high, in all cases exceeding several times limits set by the World Health Organisation (WHO) [67, 68] for waste water (effluents) used in agriculture (Table 12). It is therefore obvious that they represent considerably contaminated waste water which has to be adequately managed to prevent environment pollution. Thus, leachate water captured from the landfill cannot be disposed for example for irrigating arable land during the cultivation of conventional crops (cereals, vegetables, legumes, etc.). This is also why the leachates were used in the presented laboratory experiment. Together with Pb and Zn, Cd, Cr and Ni are the main HMs in leachates from landfills [69]. The measured HMs concentrations are high but it should be borne in mind that leachates originate from the whole landfill body (in our case from the landfill volume of 569 000 $m^3$) coming to existence due to rain water infiltration [8]. Thus, created leachates have to be caught on the landfills by insulating layer and stored in impervious sumps before disposal [8, 69]. An option for the disposal of these hazardous leachates is offered therefore in using the process of phytoextraction or phytoremediation [31, 35] by plants that would have a sufficient capability to absorb HMs contained in landfill leachates.

Table 12
Comparison of measured HM values and permissible limits of selected heavy metal ions in waste water according to WHO standards [67, 68]

| Heavy metals | Measured value [µg $L^{-1}$] | WHO limit [µg $L^{-1}$] | Differences [µg $L^{-1}$] |
|---|---|---|---|
| Cd | 0.183 | 3 | –2.817 |
| As | 29.73 | - | - |
| Cr | 812.4 | 50 | +762.4 |
| Ni | 153.7 | 20 | +133.7 |
| Hg | 1.32 | 1 | +0.32 |

The monitoring of HM concentrations in seepage waters is necessary to determine a potential risk posed by disturbance of landfill insulation layers and by their leakage outside sumps [70, 71]. When rain water, groundwater or other precipitations get into the landfill, they can dissolve and transport substances occurring in the waste. Result of the process are seepage waters with contaminants such as heavy metals (e.g. lead, mercury, cadmium), organic chemicals (e.g. pesticides, solvents) or microorganisms which may penetrate to groundwater, depending on landfill security and age [72]. Contaminants that can migrate into groundwaters are heavy metals which may be present in batteries,

electronic equipment, chemical products, and may contaminate water for a very long time. Organic contaminants (e.g. pesticides, herbicides, PCB) pose another risk as they can be toxic to aquatic organisms and represent a health risk to human population, particularly if they get into drinking water. Another risk is ammonia and nitrate compounds from organic waste products, which may lead to eutrophication of water sources and impair quality of water [70-73]. Research conducted in the past in the locality of MSW Kuchynky landfill was focused on monitoring the movement of hazardous substances (e.g. HMs) within the landfill body. The research was described in Adamcova et al. [72] and Vaverkova et al. [73]. The two studies confirmed potentially hazardous substances occurring in the MSW landfill body that could be contained in leachates. On the other hand, data from these scientific works [72, 73] demonstrated no leakage of any hazardous substances from the MSW Kuchynky landfill body into the surrounding environment (farmland), the main reason of such a result being tightness of landfill insulation and functionality of drainage system including retention tanks. This system retains all leachates, thus preventing a potential leakage of hazardous substances outside the body of the MSW Kuchynky landfill.

**HM concentrations in the samples of soil and plant biomass and effect of light intensity on HM sorption**

Evaluating the concentrations of HMs in the soil and plant biomass samples we worked with an assumption that the increasing concentration of leachates in irrigation water would have a direct influence on the increased concentration of HMs in the soil as well as in the model plant (*Sinapis alba* L.) biomass. This was confirmed only partly as with the leachate concentration growing above 75 %, a non-significant decline was recorded in a majority of cases in the concentration of individual HMs in the soil. Based on Salehi et al. [74], it can be assumed that the increasing concentration of leachates in irrigation water was increasing the availability and mineralisability of HM ions, which could have led to their greater availability to plants.

Therefore, we assume that these would be partly corresponded to also by the *EC* value and by the concentrations of selected HMs in the plant biomass. However, this does not explain why a significant decline in the concentration of HMs in plants occurred in variants of 90 % and 100 %. Sayago et al. [75] maintain that the efficiency of the elimination of heavy metals depends on and is at the same time limited by the exchange of cations between hydroxyl groups and carboxyl groups present in biomass. More precisely, that each plant has a certain capacity and potential to absorb a specific HM [39, 66, 75]. Therefore, we can state that the presented results indicate exceeded limit values of HM concentration in irrigation water, which the plants of *Sinapis alba* L. were able to incorporate into their biomass. The situation is similar to that of soil with the excessive input of nutrients of which the soil and microorganisms in the soil are able to catch only a part which is then made available to plants. The remaining part is lost through flushing from the soil [73, 76]. Presumably, an excessive amount of HMs could have been partly flushed out also in our experiment. A similar conclusion was reported also by Kacalkova et al. [77] who inform that differences in the accumulation of HMs (Cd, Zn and Hg) into the biomass of different plant species are primarily affected by the total concentration of these HMs in the soil. On the other hand, Turan and Esringu [78] consider crucial the bioavailability of specific HMs in the soil, which primarily affects their absorption from the soil environment (rhisosphere) into roots and leaves of plants realising phytoremediation.

We also analysed the effect of additional light (LED) on the concentration of HMs in the soil and plant biomass samples. Designing the experiment, we assumed that light intensity greater than the natural daylight would result in the increased sorption of HMs by the model plant and hence in the decreased concentration of HMs in the soil (alternative hypothesis $H_1$). Comparing only the values of HM concentration in the soil or in the plant biomass, we would have to reject the $H_1$ hypothesis in most variants in spite of the fact that a demonstrably positive influence of the LED photoperiod on the concentration of HMs in the soil samples was not demonstrated and a positive influence in case of plant biomass samples was recorded only in Cd and Hg. It is therefore possible to state that the LED photoperiod contributed to the increased concentration of HMs in the plant biomass only in some HMs. Thus, the statistical hypothesis $H_0$ could not be rejected and the alternative $H_1$ hypothesis could not be fully confirmed. The measured values are partly in contradiction to the findings of Kwon et al. [57] who found a positive effect of LED leading to increased extractability of HMs from the soil. If we focus also on the results of two-way ANOVA (Fig. 3 and Fig. 5), it is clear that in the soil samples, the concentration of leachates in irrigation water had a higher influence (factor) on the concentration of HMs in the soil than on the concentration of HMs in the plant biomass. This brings us to assumption that *Sinapis alba* L. exhibited an increased susceptibility to the sorption of selected HMs, for example Cd and Hg as compared with the other HMs (As, Cr and Ni). According to Vasile et al. [62], Sayago et al. [75] and, Evangelou et al. [79] Fargasova [80], *Sinapis alba* L. has a high capacity of binding Cd and Hg as compared for example with As and Cr, while retaining tolerance - endurance to the other HMs although it is not able to bind them into plant tissues so effectively as it is able to bind Cd.

Another possibility to compare the effect of LED photoperiod is to use the Enrichment coefficient for the plant/soil system (Table 10 and Table 11). During the action of LED, higher *EC* values were recorded, which is relatively interesting because if we compare the average values of HMs in the plant biomass (Table 9) and in the soil samples (Table 8), we can see they are balanced. The concentrations of HMs in the plant biomass were higher in Group A (LED) for Cd and Hg. Conversely, in case of As, Cr and Ni, the concentrations were higher in the other experimental group. Therefore, it is not possible to determine clearly whether the LED technology had a positive influence on the deposition of HMs into the plant biomass or whether it only supported the sorption of HMs to which the model plant has greater susceptibility.

Nevertheless, *EC* values (Table 10 and 11) indicate that if the residual concentration of HMs in the soil after the end of the experiment and the amount of HMs absorbed in the plant biomass are taken into account, the LED light increased the capacity of plants to bind HMs. According to Malayeri et al. [66], values between 0.1 - 1.0 are characteristic of moderately accumulator plants and values > 1 are characteristic of high accumulator plants. When the LED photoperiod was used, all HMs exhibited *EC* values > 0.1 and for Cd, Hg and Ni (only in the control variant) even values > 1. On the other hand, the group without the LED photoperiod exhibited *EC* < 1 for a majority of HMs; only Cd and Ni in the control variant showed values > 1. The *EC* value of 1 was exceeded in Hg in variants with the concentrations of leachates in irrigation water ranging from 20 % - 100 %. The measured *EC* values similar in the two groups (with/without LED) confirm the higher susceptibility of *Sinapis alba* L. for binding Cd and Hg in its biomass, which was also corroborated by Vasile et al. [62], Evangelou et al. [79] and Fargasova [80]. The *EC* values show that *Sinapis alba* L. can bind also As, Cr and Ni; however, at a lower effectiveness

with this effectiveness not being increased by the LED technology. Last but not least, *EC* values were the highest in the group with the LED photoperiod, enhancing the sorption of Cd and Hg whose effectiveness was the highest in the model plant.

## Conclusion

The presented study is focused on the issue of phytoremediation and its higher effectiveness using a plant stimulated to phytoremediation by LED light. There were two main goals of the research: (1) to determine the effect of additional LED light on the phytoremediation potential of the model plant, and (2) to determine the effect of leachate concentration in irrigation water on the concentration of heavy metals in the soil and plant biomass samples. These two goals should result in new findings about phytoremediation, which could be further used for technologies of root wastewater treatment plants.

Designing the experiment, we assumed that the light intensity of individual experimental variants increased beyond natural daylight would lead to the increased sorption of heavy metals by the model plant and hence to the reduced concentration of heavy metals in the soil (= alternative hypothesis $H_1$). The measured values did not demonstrate a full impact of LED light on increased sorption of all heavy metals by the model plant *Sinapis alba* L. Such an effect was demonstrated only in Cd and Hg. The measured values are not significant enough to reject the zero hypothesis and to confirm the alternative hypothesis on their basis.

We found out that our assumption about the concentration of leachates in irrigation water having a direct effect on increased concentration of heavy metals in the soil and at the same time in the biomass of model plant was confirmed only partly. Such a direct effect was recorded only up to 75 % of leachate concentration in irrigation water; above this limit, the concentration of HMs was declining both in the soil (non-significantly) and in the plant samples (significantly). The phenomenon might have been caused by exceeded limit values of HMs concentrations in irrigation water, which the plants of *Sinapis alba* L. were able to incorporate into their biomass.

The measured values indicate that following research studies will have to involve different types of LED lights and model plants. As to the efficiency of phytoremediation, we have to find out what is the response of different plant species to the stimulation by LED light. The following research will have to be focused on a potential influence of LED light constructions in terms of light spectrum they radiate on the metabolic processes in the given plants. Further tests will, however, need experimental design modification so that a variant without the model plant is included. Values from this variant (only the soil irrigated with leachates) could provide a better explanation for the effect of growing *Sinapsis alba* L. on the sorption of HMs from the soil. Advanced LED technologies can simulate sunshine and also blue light which is necessary for photosynthesis, red light for the growth of plants, UV and violet radiation which affects the capacity of plants to create pigments. There are, however, considerable differences among the individual LED technologies (grow panels; grow reflectors, grow bulbs etc.) in spectral ratios, i.e. that these technologies are able to generate electromagnetic radiation of different wave lengths. We believe that a further research of the interconnection of LED technology and phytoremediation can contribute to the development of environment-friendly methods used in the treatment of waste waters of diverse origin.

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