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Physico-chemical characteristics of fine fraction materials from an old crystal glass dumpsite in Sweden

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ABSTRACT

Physico-chemical characteristics of waste, particularly fine fraction (FF), from an old crystal glass waste dump in Sweden were studied to assess recycling or disposal alternatives. Hand-sorting of the waste indicated glass content of 44.1% while sieving established the FF as a more soil-like mix of glass and other materials constituting 33.3% of all excavated waste. The FF was around neutral pH with 24.4% moisture content, low values of Total Dissolved Solids, Dissolved Organic Carbon and fluorides, but hazardous concentrations of As, Cd, Pb and Zn according to the Swedish Environmental Protection Agency guidelines. While the FF leached metals in low concentrations at neutral pH, it leached considerably during digestion with nitric acid, implying leaching risks at low pH. Thus, the waste requires safe storage in hazardous waste class 'bank account' storage cells to avoid environmental contamination as metal recovery and other recycling strategies for the glass waste are being developed. The study could fill the information gap regarding preservation of potential resources in the on-going, fast-paced excavation and re-landfilling of heavy metal contaminated materials in the region.

1. Introduction

Industrialisation has resulted in improved quality of life, but also in an increase in polluted sites globally. In Europe and the United States, for instance, about 2 million potentially polluted sites have been identified, while in Sweden alone over 80,000 polluted sites have been linked to past industrial activities [1,2]. One such activity was crystal glass production in Småland region, south-eastern Sweden, famously known as '*Glasriket*' (Kingdom of Crystal). In glass production, metal oxides were vital and were used as glass component stabilisers/modifiers, refining agents and colouring/decolouring agents [3,4]. Through raw material residues and factory wastes, metals from the production process ended up in open dumps within factory premises and accumulated over time. Human and

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environmental exposure risks and impacts in the region have been documented due to high contamination of these sites by As, Cd, Pb, Zn, etc. [5,6].

Based on previous investigations recommending site remediations [7], and given the high toxicities and persistence of heavy metals in landfill environments [8,9], the remediation practice has been excavation, transfer and landfilling of the polluted masses followed by covering of the sites with cleaner soils. However, the practice may involve transferring the pollution problem to other sites. Since landfills and dumpsites are regarded as secondary resource stores for future resource recovery [10–15], research on extraction of metals from the dumped glass waste has been ongoing [16,17]. Thus, the fast-paced 'excavation-landfilling' remediation model against the moderately-paced research on metal extraction possibilities and assessment of circular economy applications calls for safe intermediate storage of the waste for later valorisation as opposed to landfilling. This implies designing an approach that prevents contamination from stored waste, while evaluating their most suitable fate for the benefit of both human health and environmental systems. Planning and implementation of such an approach requires a thorough understanding of characteristics of the contaminated materials [8,18]. As such, this study serves as one of the initial steps in the planning of safe intermediate storage of the excavated waste.

Of particular importance to this study is the fine fraction (<11.3 mm) of excavated waste since information about its physico-chemical properties and recycling alternatives in Sweden is lacking [19]. Furthermore, available studies on this fraction have mainly focussed on municipal solid waste (MSW) landfills. Although it can account for as high as 80% of total excavated waste [20], it is the most challenging fraction to sort due to its small size, since most sorting mechanisms and valorisation processes are size-dependent [21]. Re-landfilling of the fraction is thus preferred as it has also been associated with high heavy metal concentrations in waste, where concentration increases with decrease in particle size [22]. This study focuses particularly on the metals As, Cd, Pb and Zn in FF given their abundance and availability in dumps around the region [7], as well as the imminent scarcity against rise in global production for some of these metals. Zn production, for instance, rose by 5% (13.2 million tonnes out of 230 million tonnes world reserves) in 2017, though the year ended at a supply deficit of 3%, while Pb production was at 4.7 million tonnes out of 88 million tonnes world reserves [23,24]. These trends make excavated wastes, and FF in particular, vital candidates for study towards avoided landfilling.

Leachate characterisation is an important tool in the planning and implementation of waste management strategies, and thus leaching tests are employed on the FF to assess its physico-chemical characteristics towards recovery and disposal. The tests are valuable sources of information about potential leaching risks during waste storage, recovery or disposal. They are also valuable for cost-saving in case the waste does not leach in hazardous levels that require design of costly storage or disposal sites. Thus, the study partly aimed to understand the composition of the waste from Orrefors glass dump to create a decision-making basis about waste valorisation with focus on waste sorting and metal extraction mechanisms. It also aimed to understand the amount and characteristics of the FF in order to identify safe intermediate material storage options for future recycling, or disposal options in case of undesired characteristics. Thus, waste composition was assessed through sieving and hand-sorting of the waste, while physico-chemical characterisation was assessed through leaching tests and metal analyses on the FF. Furthermore, the FF

was also acid-digested to assess potential leaching risks under acidic conditions which could also prevail during waste handling.

2. Materials and methods

2.1. Site description

The site investigated in this study was Orrefors glass factory dumpsite in Nybro Municipality (56°50′35.9″N, 15°44′47.3″E), south-eastern Sweden, which is co-owned by Nybro Municipality and a private entity. The factory was active from 1898 to 2012 and produced a mix of crystal and household glasses. The dumpsite covers about 4000 m^2 surface area and contains approximately 5600 m^3 of contaminated soil, glass, and other factory wastes [7]. With As and Pb as the major contaminants, the site is categorised as high risk and ranked 9th out of 38 other high-risk objects in Kalmar County [25].

2.2. Excavation and sampling

The excavation was achieved using a 5 tonne excavator based on a method from Kaczala et al. [8]. Excavated materials from every half-metre depth of each test pit (TP) were stockpiled separately for sampling. In each case, the top 0.15 m to 0.20 m soil layer was preserved for later use as cover material after sampling and closing of the TPs. The process also involved verification measurements of each depth interval to ensure material matching with associated stockpiles. This sampling method was adopted in order to understand the variations of waste materials over time. In total, 8×2 m deep TPs were excavated i.e. about 23 tonnes of waste and 32 stockpiles. Stockpiles were sampled according to the Nordtest Method NT ENVIR 004-1996/05. From each TP, materials from 0.5 and 1 m depth stockpiles made a composite sample while materials from 1.5 and 2 m depth stockpiles made another composite. Thus, 16×10 l composite samples were collected and stored at 4°C until they were analysed. Furthermore, results obtained from the individual composite samples were aggregated according to layers (L_X) to evaluate the vertical distribution of contaminants in the layers (L_X), as presented below:

$$L_Z = \mathrm{TP}_1 C_Z + \mathrm{TP}_2 C_Z + \ldots + \mathrm{TP}_7 C_Z + \mathrm{TP}_8 C_Z \tag{1}$$

where L_Z is layer number Z (1 or 2) aggregated from all composite samples Z (1 or 2) from all the 8 TPs, L_1 representing 0–1 m depth and L_2 representing 1–2 m depth. Thus, 8 composite samples each from L_1 and L_2 were analysed.

2.3. Sieving and hand-sorting of samples

The waste was sieved onsite and the resulting particle sizes were categorised into coarse (>31.5 mm), medium (31.5–11.3 mm) and fine (<11.3 mm) fractions, henceforth referred to as CF, MF and FF respectively. Afterwards, each sieved size fraction was hand-sorted and categorised into glass, inert (stones and demolition waste like ceramics, concrete and bricks), organics (wood, plant debris and paper) and other materials (metals and plastics), henceforth referred to as 'residual' in the study. Although glass is an inert material, it was categorised alone since it was the main subject of study. To obtain mass balance, the total

waste mass before sieving and sorting was recorded together with subsequent masses of each particle size category and associated waste fractions. To understand FF composition, materials 4–11.3 mm were also hand-sorted in size categories of 4–8 mm and 8–11.3 mm, using magnifying lenses. Materials <4 mm were only assessed visually without hand-sorting.

2.4. Leaching tests

Sample FF was leached with deionised water (18.2 MΩ/cm^2 Milli-Q^{TM} water) at neutral pH and room temperature according to the Swedish Standard method for the characterisation of waste (SS-EN_12457-4). The standard liquid to solid ratio was maintained while the particle size was modified from the standard <10 mm to <11.3 mm. Using an ELMI Rotamix (RM1), a weighed sample portion and water were agitated in 50 mL tubes at 10 rpm for 24 h (± 30 min). Leaching was done in triplicates per sample, and the obtained leachate was sieved through 0.45 µm sieves for further analyses.

2.5. Acid digestion

A portion of each sample FF was digested with 7M nitric acid (HNO$_3$) according to the Swedish Standard method (SS-028150 – modified) for determination of metals in soil and other wastes including demolition waste, sediments, etc. As per standard, 10 mL of acid was titrated into 0.5 g of each sample in 15 mL tubes, which were then digested in an autoclave at 120°C for 30 min. The samples were also digested in triplicates, resulting in 48 digested samples.

2.6. Other analytical procedures

Each FF sample was analysed for moisture content (MC), fluorides (F$^-$), total dissolved solids (TDS), dissolved organic carbon (DOC), pH and metal contents. The MC was determined according to the Swedish Standard method SS-EN 14346:2007 while metal contents in solid phase were analysed with a portable X-Ray Fluorescence (XRF) analyser (Olympus DS-4000 Innov-X). A portable pH metre (Radiometer PHM 210) was used for pH measurements while DR Lange cuvettes were used for F$^-$ and DOC. They were digested in a DR Lange digester (HT200S) and analysed using a DR Lange Spectrophotometer (DR 5000). The gravimetric method 8163 by Hach Lange was used to determine TDS while metals in liquid phase were analysed using an ICP-MS.

2.7. Statistical analysis

Statistical analysis on all parameters was achieved using GraphPad Prism version 7.0c for Mac (GraphPad Software Inc.). Descriptive statistics (minimum, maximum, mean and standard deviation) was calculated at $p < 0.05$. In addition, one-way ANOVA and Tukey's multiple comparison tests were done to compare such datasets as metal concentrations among TPs and layers (Ls). In some cases, given the large data spreads due to heterogeneity, Grubbs test for outliers ($a = 0.05$) was done to validate the data, while taking caution not to knock out valid data.

3. Results and discussions

3.1. Characteristics of all excavated materials

3.1.1. Waste composition

The waste composition was determined as shown in Figure 1(a). Glass was dominating (44.1 ± 11.4%) followed by inert (43.6 ± 15.9%), organic (9 ± 13.4%) and 'residual' (1.7 ± 1.1%) fractions. A one-way ANOVA ($p < 0.05$) and Tukey's multiple comparisons test showed that the difference between inert and glass fractions was not significant statistically while both fractions differed from organic and 'residual' fractions significantly. The difference between organic and 'residual' was not significant statistically either. Similarities in spreads of glass (27.4%–56.1%) and inert (27.9%–64.6%) fractions explain their lack of significant differences. The high standard deviation for organic on the other hand was due to an unusually wide data spread (0–30.4%) owing to one sample with more organics.

These results, however, could not be compared with other published studies on Swedish glass waste dumps as those studies have mainly focused on toxicity assessments and remediation aspects [5,6,26,27].

3.1.2. Particle size distribution

As shown in Figure 1(b), the PSD was dominated by CF (44.8 ± 6.6%) followed by FF (33.3 ± 7.7%) and MF (20.5 ± 5.8%). Tukey's multiple comparisons test showed that the differences among the three size categories were significant statistically. The results were in agreement with other studies that quantified CF as 46 ± 11% [28], MF as ranging between 21.8% and 31.4% [29] and FF (<24 mm) averaging 55% [30]. However, the results may differ from a number of other studies owing to different factors such as sampling procedure, selected particle size categorisation, waste type and waste age [19].

From a circular economy perspective, old crystal glass waste could be potentially used in both open and closed-loop recycling, where constituent metals are extracted for use in
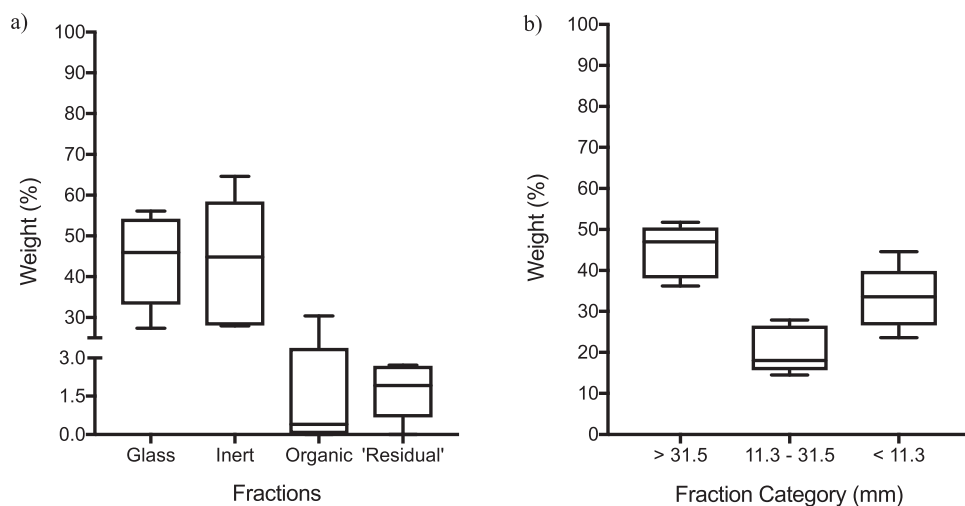

Figure 1. (a) Waste fractions abundance and (b) particle size distribution for the three fraction size categories ($n = 5$).

other industrial applications and the detoxified glass residue is used in production of new glass artefacts [16,17]. The relatively high glass content at Orrefors dump could support such recycling potential of the fraction. However, the complex mix of the waste requires well-planned sorting mechanisms to isolate the desired glass fraction. The FF on the other hand, being the most challenging fraction to sort, only accounts for about a third of the materials, which is good for recycling alternatives and sorting mechanisms.

3.2. Characteristics of the FF

3.2.1. Waste composition

Composition of FF was based on waste fractions 4–8 mm and 8–11.3 mm and was determined as shown in Figure 2. It was dominated by inert fraction (55.1 ± 24.6%) followed by glass (29 ± 8.6%), organic (11.2 ± 15.3%) and 'residual' (0.8 ± 1.1%) fractions. The percentage contents were based on five random samples from the 16 composite samples gathered around the site. The high standard deviations indicate the level of heterogeneity among samples, especially in terms of the organic and 'residual' fractions. The fraction <4 mm was only visually categorised as soil-like material due to the complexity of fraction mixtures. Since it was dominated by inert fractions other than glass, glass-centred recycling potential is low. It is almost impossible to sort out the individual fractions for recycling. Their contamination status further jeopardises re-use potential, and thus this fraction could be better landfilled. Moreover, the fraction <4 mm accounted for only a fifth of the total excavated waste, unlike landfilling of all excavated materials.

3.2.2. Metal concentrations in solid phase

The total metal concentrations varied among TPs and between layers, with huge differences in ranges in some cases as shown in Table 1. The results were evaluated against Swedish guidelines for hazardous waste [31]. L_1 had higher maximum concentrations than L_2 for Cd, Pb and Zn whereas for As L_2 recorded a higher maximum concentration. Maximum concentrations of all metals exceeded the Swedish guidelines in all layers, except for Cd in L_2. As shown in Figure 2, heterogeneity of FF materials, dominated by inert fractions other than glass, contributed to the high concentration spreads observed in Table 1.
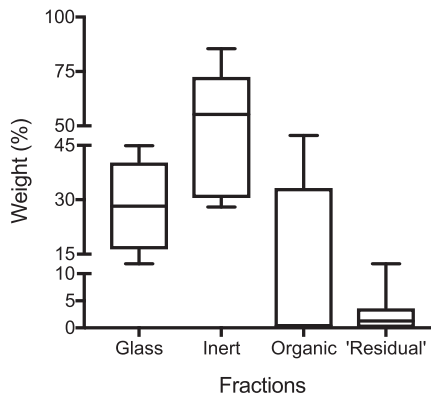

Figure 2. Waste composition of the FF based on weight percent contribution of each waste fraction.

Table 1. Metal concentrations of FF materials per sampled layer compared with Swedish guidelines for hazardous solid waste according to Avfall Sverige [31], $n = 8$, (mg kg^{-1}).

Samples	As L_1	As L_2	Cd L_1	Cd L_2	Pb L_1	Pb L_2	Zn L_1	Zn L_2
Min	72	13	<LD	<LD	832	54	135	43
Max	4096[a]	5834[a]	252[a]	93	77,521[a]	75,645[a]	2749[a]	2548[a]
Mean	1433[a]	817	107[a]	n.a.	29,663[a]	10,666[a]	1036	599
Swedish Guidelines	1000		100		2500		2500	

Notes: [a]Values higher than the guidelines; <LD – 'Lower than the Limit of Detection' (1 mg kg^{-1}); n.a. – 'Not Applicable'

The pattern of abundance of metal concentrations for all TPs and Ls was thus in the order Pb > As > Zn > Cd. The findings are consistent with the purposes for which the metals were used in the crystal glass industry. Pb and Zn, from PbO (>24%) and ZnO (1.5%) respectively, were used as glass network stabilisers or modifiers [3]. Their abundance in dumpsites depends on the type and period of industrial production. In a crystal glass dump, more Pb than Zn is expected due to the EU directive of 1969 which imposed a minimum quality requirement of Pb > 24% weight for crystal glass [32]. Thus, Pb was more preferred than Zn. On the other hand, As was used as a refining agent through As$_2$O$_3$ (0.4%), and Cd as a colouring agent through CdS (amount dependent on the need for red-coloured glass) [3]. The higher abundance of As in the dump than Zn may also be traced back to the preference of Pb over Zn in crystal glass.

3.2.3. Leachate characteristics

Leachate parameters: The physico-chemical parameters pH, MC, DOC, TDS and F$^-$ were determined as shown in Table 2, and the results were compared with the Swedish Environmental Protection Agency (SEPA) guidelines for material landfilling in inert, non-hazardous and hazardous waste landfills [33]. The leachate was around neutral pH on average, ranging between 5.9 and 8.2, though the guideline for pH is not specified in the standard. Both DOC and TDS were below the guidelines for the three categories. F$^-$, on the other hand, was higher than the guideline for inert waste landfill, whereas it was below the guidelines for both non-hazardous and hazardous waste landfills. The MC (24.4 ± 11.4%) was quite high considering that waste is relatively loosely packed in a glass dump, implying a lower water-holding capacity. However, the dump is located besides a stream and on a swampy area where water starts collecting around 1.5–2 m depth of excavation. Though higher, the MC is comparable to some MSW landfill amounts of 23.5%–43.3% reported in other studies [19,34,35].

Table 2. Leachate parameters (mean ± standard deviation) against the Swedish EPA guidelines [33].

Parameter	Value	Swedish EPA Inert	Non-hazardous	Hazardous
pH	7.3 ± 0.8	n.s.	n.s.	n.s.
MC (%)	24.4 ± 11.4	n.s.	n.s.	n.s.
F$^-$ (mg kg^{-1} DS)	10.8 ± 2[a]	10	150	500
TDS (mg kg^{-1} DS)	2856 ± 467	4000	60,000	100,000
DOC (mg kg^{-1} DS)	56.4 ± 21.8	500	800	1000

Notes: [a]Values higher than the guidelines; DS – dry substance; n.s. – 'not specified'.

Leachate and its concentration of contaminants and other parameters are a consequence of the waste types, age and the conditions prevalent within a landfill or dump determining degradation and transformation processes [36]. Therefore, the observed low levels of the physico-chemical parameters could be explained in terms of the type of waste disposed, which was mainly glass and inert waste. The low DOC (and partly TDS) concentrations are due to the lack of putrescible wastes and their degradation products in the dump, since bio-decomposition of such wastes is related to increased DOC amounts [8,36]. As observed by Kaczala et al. [8], the leachate generated and the concentration of carbon contents in particulate and dissolved forms is dependent on environmental and physical conditions such as oxygen diffusion, MC, void spaces and waste compactness. This could be a further explanation for the observed low parameter values, since in a glass dump a number of void spaces are expected, which in turn primarily increases oxygen diffusion. Low values of some parameters, such as DOC, could also be a result of the absence of (or marginal presence of) anaerobic degradation processes in the dump [37] since over 90% of the waste is inert.

Metal concentrations in leachate: The metals exhibited different leaching trends among TPs as shown in Figure 3(a). Among the four metals overall, the highest leached metal was Pb (4.7 mg kg^{-1}) followed by Zn (1.1 mg kg^{-1}), As (0.6 mg kg^{-1}) and Cd (0.4 mg kg^{-1}). A one-way ANOVA and Tukey's multiple comparisons test showed no significant statistical differences among TPs for As, except between TP1 and TP6. For Cd, significant differences were observed between TP2 and all other TPs except TP1. Apart from TP2, there were no significant differences among all other TPs for Cd. Although for Pb differences among TPs are seen in the graph, they were not statistically significant among all TPs. This was the case for Zn also. As per Swedish EPA guidelines [33], all metals from across the dump leached in concentrations lower than the guidelines for non-hazardous and hazardous waste, but higher than the guideline for inert waste in most instances. Thus, while the materials qualify to be classified as hazardous waste according to solid-phase metal concentrations, and based on the other Swedish guidelines for hazardous solid waste [31], in terms of leachate concentrations of metals the materials would qualify to be stored under the conditions for both hazardous and non-hazardous waste landfills. However, caution needs to be taken in storage design and ultimate storage since biological or chemical transformations in solid phases, of low initial leachate concentrations, could lead to formation of toxic substances from relatively innocuous compounds [36]. Taking As, for instance, although its concentration in leachate was low, chemical transformations during storage affecting or reducing Fe content in the waste would render As available for leaching in possibly toxic levels.

Metal leachability: Leachability of metals, defined as the percentage of leaching concentration of a metal in relation to its total concentration in the solid phase, was computed as presented below [38]:

$$\text{Leaching ratio (\%)} = \frac{\text{Leaching concentration}}{\text{Total concentration}} \times 100\% \qquad (2)$$

Leachability was dominated by Cd (0.15%), As and Zn (0.05%), and Pb (0.02%). It was not influenced by the total concentrations of metals in solid phase (Pb > As > Zn > Cd), since total metal concentrations do not correspond to the amounts leached into water and the potential mobility of each metal [8]. However, leachability is dependent on the
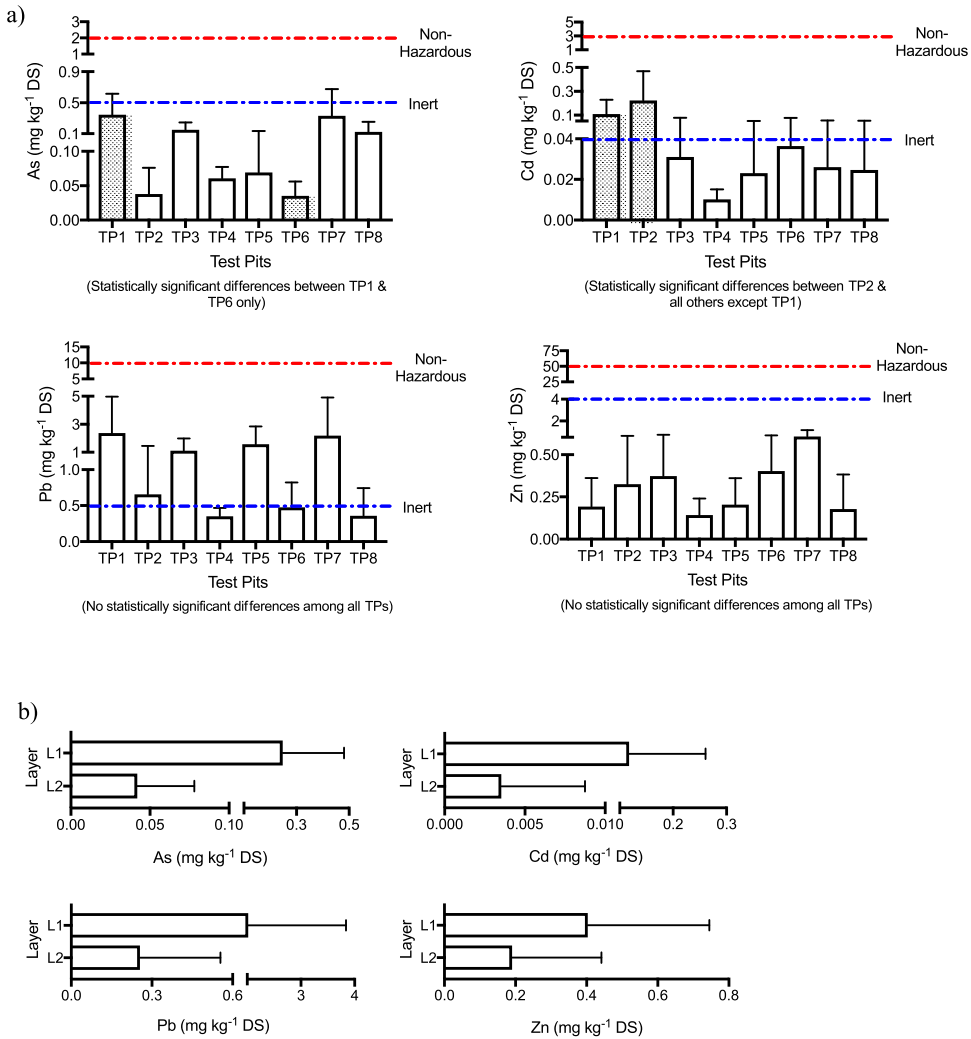
Figure 3. Leaching of As, Cd, Pb and Zn according to; (a) TPs ($n = 6$), and (b) Ls ($n = 24$). The blue and red lines in (a) indicate Swedish guidelines for inert waste and non-hazardous waste landfills respectively. Threshold values for hazardous waste landfill are 25 mg kg^{-1} for As, 5 mg kg^{-1} for Cd, 50 mg kg^{-1} for Pb and 200 mg kg^{-1} for Zn [33].

total concentrations since it is just a ratio. For instance, for a metal in low total concentration like Cd, the amount released may appear higher when expressed as a percentage although it is low in the leachate. Other factors affecting leachability include sample specific surface area, metal speciation in the solid phase as well as bulk solution chemistry [39,40]. The low leaching ratios could also be attributed to the metals being strongly bound to the glass matrix. Furthermore, especially for metals in the soil, it could be attributed to the low metal solubilities in neutral to alkaline pH, as well as to the metal immobilisation by organic/inorganic sorption and precipitation [8]. Since DOC is also known to affect the mobility of metals [36], the scenario of higher total metal concentrations in L_1 than L_2 as well as low concentrations of leached metals (Figure 3(a,b)) could also be

explained in terms of low DOC levels in the dump resulting in immobility. However, leaching tests only predict potential mobility of metals and their availability to environmental systems under natural conditions [41], they do not fully reflect actual leaching in real environments [42]. Actual leachate contains xenobiotic organic compounds (XOCs) and heavy metals that can bioaccumulate and persist in the environment, and is known for toxicity, corrosivity, reactivity, flammability, carcinogenicity and other hazards [36]. Thus, leaching tests do not correspond to higher safety, and so caution is required in the design of material storage facilities.

Since the metals were extracted according to the two layers (L_1 and L_2), the means and standard deviations of the metal concentrations for the TPs were also computed per layer resulting in Figure 3(b). The dump is highly heterogeneous in terms of metal concentrations both spatially and vertically, resulting in the observed high standard deviations. As further shown in Figure 3(b), the leached metal concentrations were more prominent in L_1 than L_2, which was confirmed by Tukey's multiple comparisons test that L_1 and L_2 concentrations were significantly different statistically for all the four metals. This corresponds to results of solid-phase metal concentrations in which metal concentrations on average were higher in L_1 than L_2.

3.2.4. Metal concentrations through acid digestion

Acid digestion yielded metal concentrations as shown in Table 3. The concentrations indicated huge variations spatially and vertically as indicated by the high standard deviations. Considered together with the results of waste composition and PSD, the huge variations in metal concentrations on either layer (L_1 or L_2) indicate large heterogeneity due to random waste disposal on one section of the dump and no disposal at all on the other section. This implies that even on the section where the waste was disposed, the disposal was not systematic. Results further indicate the order of dominance as Pb > Zn > As > Cd in both L_1 and L_2 as was the case with leaching. This only implies concentration differences between the two layers and further implies that while spatial heterogeneity in each layer was observed, in vertical terms it was only a concentration gradient.

The obtained metal concentrations were evaluated against the Swedish EPA guidelines for sensitive land use, less sensitive land use, non-hazardous and hazardous waste storage [31,43] as shown in Table 3. All metals were below the guidelines for hazardous and non-

Table 3. Metal concentrations (mean ± standard deviation), Swedish guidelines for land use, waste storage and metals recovery ($n = 8$) [31,43].

Metal	Value (mg kg^{-1} DS)	Sensitive land use	Less sensitive land use	Non-Hazardous waste storage	Hazardous waste storage	Recovery (%)
As$_1$	107.8 ± 61.2	<10[a]	<25[a]	<1000	≥1000	15.8 ± 11.3
Cd$_1$	77.8 ± 123.5	<0.8[a]	<12[a]	<1000	≥1000	47.3 ± 22.1
Pb$_1$	2811 ± 1517	<50[a]	<400[a]	<2500[a]	≥2500	24.2 ± 26
Zn$_1$	392.8 ± 343	<250[a]	<500[a]	<2500	≥2500	42.9 ± 18.8
As$_2$	37.7 ± 51.5	<10[a]	<25[a]	<1000	≥1000	17.9 ± 10.6
Cd$_2$	3.2 ± 4	<0.8[a]	<12	<1000	≥1000	7.6 ± 0
Pb$_2$	977.8 ± 1225	<50[a]	<400[a]	<2500[a]	≥2500	44.2 ± 25.8
Zn$_2$	358.9 ± 448	<250[a]	<500[a]	<2500	≥2500	58.2 ± 19.6

Notes: [a]Guidelines exceeded by maximum concentration in a group; subscripts on each metal indicates layers implied (L_1 or L_2); DS – dry substance.

hazardous waste storage, except for Pb, which was higher than the guideline for non-hazardous waste storage in both L_1 and L_2. The metalloid As exceeded the guidelines for sensitive and less sensitive land use in both L_1 and L_2. Whereas Cd exceeded both guidelines in L_1, it only exceeded the sensitive land use guideline in L_2. Both Pb and Zn exceeded both guidelines for sensitive and less sensitive land use in both L_1 and L_2. Except for Pb, these results agree with the leachate results on guidelines for non-hazardous and hazardous waste storage. Thus, they present the need for complementing of leaching tests with other more robust tests.

Metal recovery (%) was also computed for all TPs, expressing the metal amount released as a percentage of the total concentration in the solid phase (XRF), as presented below [38]:

$$\text{Recovery (\%)} = \frac{\text{Released amount}}{\text{Total concentration}} \times 100\% \tag{3}$$

On average, the metal recovery across the dump was 58.2%, 44.2%, 17.9% and 7.6% for Zn, Pb, As and Cd respectively. The Cd results in L_2 is the reflection of its low detection levels in the solid phase in which it was mainly below the limit of detection (see Table 1). The recovery also indicates the decontamination potential for the waste. However, recovery or decontamination potential could be optimised by more robust digestion methods, as well as deeper understanding of metal mobility through fractionation studies, which were not the focus of the present investigation. Although metals in glass could potentially be recovered, some are in low concentrations for viable recovery. In addition, upscaled recovery operations would require availability and steady supply of the glass fraction.

On the other hand, the heterogeneity of the FF, the complexity of its constituents and the relatively lower composition of the glass fraction would render recycling (metal extraction) of this fraction challenging since metal extraction processes (reduction-melting in this case) have minimum requirements for non-glass materials in their process feeds. Furthermore, this quality of FF may not be used as cover material (methane degradation layer) in existing landfills as a recycling option due to its contamination levels and porous nature (presence of glass particles), since landfill cover material should be non-porous and compact. Although decontamination potential of the FF is presented, remaining metals are still above the Swedish guidelines for safe disposal, rendering recycling potential of the FF challenging.

4. Conclusions

Physico-chemical characterisation of the waste at Orrefors glass dump in south-eastern Sweden showed that nearly half of the sortable fractions (CF and MF) contained glass, indicating the potential for recycling of the fraction. A potential barrier in obtaining recyclable glass, however, is the need for complex sorting mechanisms due to the complicated mixing of waste fractions. The materials contain As, Cd, Pb and Zn in hazardous but extractable amounts, although more robust extraction methods are required to enhance recovery. Furthermore, the metals are leachable depending on the prevailing environmental conditions but independent of their concentrations in the waste (a metal in low concentration could leach enough to pollute). Therefore, the recommendation is to sort the waste

according to fractions and store them temporarily in 'bank account' storage cells specially designed to prevent contamination from the materials. This is to preserve the resource while awaiting scale-up of metal extraction and other recycling options. On the other hand, the difficult to sort FF (about 33% of the materials) is too contaminated for sensitive and less sensitive land uses. Therefore, unless metals could be extracted from the FF in an economically sustainable way, safe disposal in a hazardous waste landfill is recommended. This study could fill the information gap regarding the preservation of potential resources in the on-going, fast-paced excavation and re-landfilling of materials.

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