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# Distribution and leaching characteristics of trace elements in ashes as a function of different waste fuels and incineration technologies

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## ABSTRACT

Impact of waste fuels (virgin/waste wood, mixed biofuel (peat, bark, wood chips) industrial, household, mixed waste fuel) and incineration technologies on partitioning and leaching behavior of trace elements has been investigated. Study included 4 grate fired and 9 fluidized boilers. Results showed that mixed waste incineration mostly caused increased transfer of trace elements to fly ash; particularly Pb/Zn. Waste wood incineration showed higher transfer of Cr, As and Zn to fly ash as compared to virgin wood. The possible reasons could be high input of trace element in waste fuel/change in volatilization behavior due to addition of certain waste fractions. The concentration of Cd and Zn increased in fly ash with incineration temperature. Total concentration in ashes decreased in order of Zn > Cu > Pb > Cr > Sb > As > Mo. The concentration levels of trace elements were mostly higher in fluidized boilers fly ashes as compared to grate boilers (especially for biofuel incineration). It might be attributed to high combustion efficiency due to pre-treatment of waste in fluidized boilers. Leaching results indicated that water soluble forms of elements in ashes were low with few exceptions. Concentration levels in ash and ash matrix properties (association of elements on ash particles) are crucial parameters affecting leaching. Leached amounts of Pb, Zn and Cr in >50% of fly ashes exceeded regulatory limit for disposal. 87% of chlorine in fly ashes washed out with water at the liquid to solid ratio 10 indicating excessive presence of alkali metal chlorides/alkaline earths.

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## Introduction

Incineration is a rapidly growing technology for solid waste management in Sweden and all over the world (Swedish Waste Management, 2012). During 2006–2007, there were 2000 waste incineration plants in Asia, 460 in Europe (32 in Sweden) and 100 in North America (Huang et al., 2006; Sora, 2013; Swedish Waste Management, 2011). It is an efficient way to reduce volume and mass of solid waste. Resulting fly and bottom ashes can contain high concentrations of hazardous

trace elements such as Cd, Pb, Zn, As, Cr, Cu and other harmful substances (Chang et al., 2009; Chou et al., 2009). If not properly managed, these can pose a serious threat to human health and environment by releasing trace elements into soil and ground water (Ito et al., 2006). The presence of trace elements, alkali metals, Cl and S in the waste is a challenge in waste incineration with respect to operational problems and environmental concerns. Therefore, improved knowledge and understanding about the formation of ashes, distribution of trace elements during incineration, and how

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trace element partitioning responds to changes in waste composition, is important with respect to optimization of combustion process and plant efficiency.

It is crucial to understand the influential factors for partitioning of trace elements between bottom and fly ash. During incineration, high volatile elements are mostly found in flue gas or fly ash, partially volatile elements distribute equally or more to fly ash whereas low volatile elements mainly stay in bottom ash (Arena and Gregorio, 2013; Shi et al., 2004). Many studies have investigated the important factors affecting the trace element partitioning during incineration. Zhang et al. (2012) have studied the impact of temperature and moisture content on trace element partitioning to fly ash during municipal solid waste (MSW) incineration. They reported that Zn and Cu compounds transferred from chlorides to oxides with decrease in temperature and increase in moisture content while Pb and Cd distribution was not much affected by temperature. Ménard et al. (2006) studied thermodynamic behavior of trace elements during MSW incineration in grate boiler and reported that Cd volatilize as  $\text{CdCl}_2$  (g) and that Pb is more volatile than Zn. Astrup et al. (2011) investigated the trace element distribution by adding individual waste fractions such as shoes, salt (NaCl), chromate-copper-arsenate impregnated wood and batteries in MSW and described that added waste materials significantly changed the emissions in fly ash particularly for As, Cd, Cr and Sb. Generally, the focus has been on MSW with varying amounts of industrial/commercial waste in conventional grate boilers (Chang et al., 2009; Morf et al., 2000). Limited work is reported about partitioning of trace elements during biofuel incineration such as virgin wood, waste wood or mixed waste fuels including peat, bark and wood chips (Pedersen et al., 2009; Wiinikka et al., 2013; Saqib and Bäckström, 2014). Since these waste fractions are part of waste fuel being used for heat production especially in Sweden and other European countries, therefore, it is imperative to evaluate the distribution behavior of trace elements during incineration of these materials in both types of combustion systems; grate fired and fluidized bed boilers.

Leaching characteristics of ashes provide useful information for selecting the appropriate management strategy and/or their possible reuse. Fly ash is of particular concern regarding environmental safety, recovery of resources and landfilling. Since trace elements are preferentially deposited on surface of fly ash particles and are readily available for leaching (due to high alkalinity and small ash particle size), it is therefore important to assess their mobility in fly ash (Van der Sloot et al., 2001; Shi et al., 2004).

Leaching tests are common to assess risk and select proper management and disposal strategies for residues (Buchholz and Landsberger, 2012). Leaching characteristics of Korean and Japanese municipal solid waste incinerator (MSWI) fly and bottom ashes indicated that leaching of Pb exceeded regulatory limit in ashes for both countries and leached concentrations of Zn and Cd were pH dependent (Shim et al., 2005). Previous studies have showed that the liquid to solid ratio (L/S), leaching reagents and speciation were the most significant factors affecting the release of Pb and Zn from air pollution control residues (APC), while leaching from fresh bottom ash is also influenced by several factors such as ash matrix, type and concentration of extractant, leachate's

pH, L/S and chemical speciation (Kida et al., 1996; Wang et al., 2001; Zhang et al., 2006, 2008; Fedje et al., 2010). Usually MSWI ashes have been discussed with respect to trace element leachability and for reuse purposes (Yao et al., 2011; Quina et al., 2009). However; in many countries waste wood, bark, peat and virgin wood are part of waste fuels for incineration (Krook et al., 2004), therefore it is important to understand leaching behavior of trace elements from these types of ashes.

Today, the major part of fly ash from incinerators in Sweden is transported to Norway where it is used for neutralization of sulphuric acid products from paint production. The resulting slurry is disposed off in an old open pit near Langøya, Holmestrand, Norway (Swedish Waste Management, 2011). But this is not a permanent solution and domestic alternative strategies must be developed for ash management. Therefore, it is important to investigate the leaching behavior of residues as they might have potential to be considered for trace element recovery or any other secondary use.

Reuse of incineration ashes is often limited due to the presence of excessive chlorine content especially in fly ash. Average chloride content is more than 10 and 200 g/kg dry matter weight (dw) in MSWI bottom and fly ash, respectively. For instance, there is a limit of <1 g/kg dw on chloride content in fly ash for use in eco-cement production in Europe (Boghetich et al., 2005; Tokoro et al., 2013). Water washing of ashes prior to any other treatment is a common method for removal of major and trace elements. Wang et al. (2001) reported the removal of 65% Cl and 50% of major elements from MSWI fly ash through water extraction. Similarly, 95% Cl removal was reported with water from hazardous waste incineration fly ash in Finland (Kinnarinen et al., 2013).

Bottom ash, in particular, contains a significant amount of unburned organic matter. Total organic carbon (TOC) and dissolved organic carbon (DOC) contents must be kept low for possible reuse purpose. Belevi et al. (1992) mentioned DOC level of 200–800 mg/L in leachate from MSW bottom ash; while in another study it was reported to be in excess of 2000 mg/L (Brocca et al., 1997). According to European regulations, TOC and DOC must be lower than 5% and 0.5%, respectively, for waste to be landfilled (European Commission, 2003). DOC may also enhance mobility of pollutants from incineration residues when landfilled or used in constructions (Olsson et al., 2007). It has been observed that DOC facilitates the leaching of trace elements through complex formation especially for Cu, which is why it is important to evaluate DOC release from residues (Yao et al., 2011; Zhang et al., 2008).

The present study focuses on trace element partitioning during incineration between fly and bottom ash as a function of waste fuel (virgin wood, waste wood, mixed biofuel, industrial, household, mixed waste), incineration technology and combustion temperature as well as leaching characteristics of bottom and fly ashes.

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## 1. Materials and methods

### 1.1. Sampling and storage of ashes

Fly and bottom ash from 13 Swedish waste incineration facilities, 4 equipped with grate fired, 6 circulating fluidized

**Table 1 – Summary of plants used in the study.**

Facility	Capacity (ton/day)	Boiler	Waste fuel (%)	Temperature (°C)	Bottom ash (ton/day)	Fly ash (ton/day)
Umeå	480	Grate	Industrial + household (40:60)	1087	60	12
Kiruna	240	Grate	Household (100)	1091	65	6
Sundsvall	144	CFB	Household + industrial + wood chips (40:40:20)	860	10	10
Lidköping	288	BFB	Household + industrial (70:30)	874	12	5
Händelö P13	360	CFB	Wood chips (100)	899	13	24
Händelö P14	600	CFB	Household + industrial (50:50)	870	54	50
Söderenergi	672	Grate	Industrial + fuel pellets (60:40)	1000	48	13
Nynäshamn	168	BFB	Cont. wood chips (100)	850	2	4
Mälarenergi	1000–1200	CFB	Peat + bark + wood chips + waste wood (24:27:38:11)	782	13	42
Munksund	N.A.	CFB	Bark + wood chips (70:30)	809	5.5	10
Braviken	N.A.	Grate	Bark + cont. wood chips + DIP sludge (33:16:51)	1023	71	35
Eskilstuna	N.A.	BFB	Wood chips (100)	870	21	10
Högdalen	750–800	CFB	Industrial waste (100)	850	45	45

BFB: bubbling fluidized bed; CFB: circulating fluidized bed; Cont.: contaminated; DIP: de-inking pulp; N.A.: not available.

bed (CFB) and 3 with bubbling fluidized bed (BFB) boilers were sampled (Tables 1 and 2). A short summary of the selected boilers including capacity, total ash production per day, fuel used and incineration conditions is presented in Table 1. From every boiler, 1 kg of each ash was sampled by staff on four occasions during two days to collect representative samples. Sample jars were sealed with tight lids to prevent oxidation and carbonation. The four sub-samples of each ash were thoroughly mixed to get a possibly homogenized sample. During the sampling period, operation was stable and same waste fuel (Table 1) was used during the entire period. Bottom ashes were collected from ash pits after water quenching, while fly ash sampling locations are described in Table 2. Dry samples were collected when possible, however; some wet samples were dried at 40°C and coarse ashes were crushed down to a size of <4 mm to get homogenous and manageable samples.

### 1.2. Total concentrations

About 100 g of possibly homogenized sample of each ash was sent to an external laboratory for chemical characterization.

For analysis of As, Cd, Zn, Pb, Sb, Co, Cu, Ni, and S, acid digestion of ash sample (0.2 g) was performed before analysis according to ASTM D3683 (HF/HNO<sub>3</sub>/HCl digestion), while other main elements were determined after alkaline fusion using LiBO<sub>2</sub> (ASTM D3682) (Sun et al., 2001). All analyses were performed according to EPA methods 200.7 using inductively coupled plasma-atomic emission spectroscopy (ICP-AES) and 200.8 using inductively coupled plasma, quadrupole mass spectrometry/sector field mass spectrometry (ICP-QMS/SFMS). Total chlorine concentration was determined by XRF (X-Ray Fluorescence). Results of chemical characterization are reported in Table 3.

### 1.3. Standard leaching procedure and analysis of leachates

Samples were subjected to leaching according to standard leaching procedure (EN 12457-3). This method is used to assess whether the criteria for disposal at landfill are met or not (SEPA, 2006). According to this method, ashes are initially leached with ultra-pure water at L/S 2 for 6 hr with shaking at 30 ± 2 r/min in an end-over-end fashion at room temperature. Phase separation is done by centrifugation at 20,000 g for

**Table 2 – Summary of additives and flue gas treatment systems used by plants.**

Facility	Residence time (sec)	Ammonia (kg/hr)	Addition to flue gases (kg/day)	Fly ash sampling location	O <sub>2</sub> concentration (%)	FGTS
Umeå	3	Yes	AC + lime	Bag filter	6.53	Bag filter
Kiruna	3.2	Urea	N.A.	ESP	6.45	ESP
Sundsvall	2–4	Yes	Dolomite + lime	ESP + bag filter	6.5	ESP + bag filter
Lidköping	2	40	2400 quick lime	Bag filter	5.58	Bag filter
HändelöP13	2–3	60	12,720 lime	ESP	4.5	ESP
HändelöP14	2	79	8136 lime + 103 AC	Bag filter	N.A.	Bag filter
Söderenergi	3–4	90	N.A.	ESP	N.A.	ESP + bag filter + semi dry scrubber
	1200–1800 fuel					
Nynäshamn	2	35	108 AC + 576 lime	Bag filter	3.18	Bag filter
Mälarenergi	N.A.	Yes	N.A.	Bag filter	2.45	Bag filter
Munksund	2	N.A.	N.A.	ESP	2.34	ESP
Braviken	N.A.	25	70 AC + 1400 hydrated lime (fabric filter)	ESP	4.38	ESP + fabric filter
Eskilstuna	N.A.	108	N.A.	ESP	2.20	ESP
Högdalen	2	Yes	Lime + AC	Bag filter	6	Bag filter

AC: activated carbon; ESP: electrostatic precipitator; FGTS: flue gas treatment system. N.A.: not available.

**Table 3 – Trace element concentrations in incineration residues (unit: mg/kg dw).**

Facility	Ash	Cd	As	Sb	Pb	Zn	Mo	Cu	Cr	Ni	Co	Cl (%)	S
Umeå	Fly ash	179	131	1050	3170	26,400	35	1270	1050	94	70	9	38,400
	Bottom ash	5.5	56	61	733	4240	17	17,300	593	135	36	1.1	4300
Kiruna	Fly ash	380	136	1870	6880	26,900	38	2400	707	150	30	13.3	35,600
	Bottom ash	6.3	26	115	2750	10,600	26	3680	649	120	59	0.5	7750
Sundsvall	Fly ash	27	84	239	2060	5040	31	4130	483	154	26	8.1	30,100
	Bottom ash	3	49	72	870	4640	9	4370	370	280	18.3	<0.1	1440
Lidköping	Fly ash	97	54	538	6920	8120	28	16,400	421	75	20	21.4	26,300
	Bottom ash	1.5	66	321	563	2490	24	3550	528	153	30	0.5	4610
Händelö P13	Fly ash	8	26	17	108	17,200	3	317	101	35	124	0.3	32,800
	Bottom ash	0.3	13	9	95	14,800	<6	144	64	18	52	<0.1	790
Händelö P14	Fly ash	40	30	351	2830	11,600	42	7190	643	311	40	7.6	28,100
	Bottom ash	8.4	52	228	1110	3290	14	10,900	361	147	21.5	0.2	6830
Söderenergi	Fly ash	129	147	1600	7460	21,800	35	2390	605	121	42	12	27,600
	Bottom ash	0.8	22	112	943	2400	22	7350	806	190	37	0.2	4900
Nynäshamn	Fly ash	21	392	64	1900	12,100	7	864	651	69	26	4	19,100
	Bottom ash	0.4	104	22	1730	5760	<6	1890	308	24	12	<0.1	620
Mälarenergi	Fly ash	6.3	104	17	1090	3510	12	309	369	58	19	0.5	15,200
	Bottom ash	0.2	55	4	193	1450	<6	365	329	16	5.8	<0.1	1350
Munksund	Fly ash	14	20	21	118	2010	<6	228	56.5	27	11	1.5	10,500
	Bottom ash	0.2	29	3	17	2380	<6	169	50	19	14	<0.1	903
Braviken	Fly ash	11	34	32	1810	2940	<6	618	237	88	12	1.3	7880
	Bottom ash	0.1	6	5	99	395	<6	470	226	86	9.5	<0.1	27,300
Eskilstuna	Fly ash	12	2.2	2	140	1290	<6	74	47	26	9	0.6	16,000
	Bottom ash	0.3	<10	0.5	12	821	<6	36.7	43	9.6	3.39	<0.1	532
Högdalen	Fly ash	44	69	390	4160	8840	23	7630	557	156	32	6.8	60,000
	Bottom ash	2.1	62	120	912	2690	9	14,200	249	86	21	0.2	2320

30 min (Avanti J-20 XPI, Beckman Coulter, Avanti J-20 XPI, Minnesota, USA) and filtration using 0.40  $\mu\text{m}$  polycarbonate filters (style 3120, Nalgene, Titan filtration system, Nylon filter membranes, Tennessee, USA). During the second stage, fresh ultra-pure water is added to the already leached ash sample and is shaken again at L/S 8 for 18 hr resulting in a cumulative L/S 10. Ash leachate samples were analyzed for electrical conductivity (CDM 210, Radiometer Analytical, Radiometer Analytical SAS, CDM 210, Villeurbanne Cedex, France) and pH according to Swedish standards.

Chloride was analyzed using capillary zone electrophoresis. Prior to analysis, ash leachates were filtered (0.40  $\mu\text{m}$  polycarbonate filter). The buffer consisted of 5 mmol/L chromate at pH 8 and 0.5 mmol/L tetradecyl trimethyl ammonium bromide (TTAB) to reverse electro osmotic flow. Hydrostatic injection was done at 10 mbar for 30 sec.

DOC was determined using total organic carbon (TOC) analyzer (TOC-VWP, Shimadzu, Copenhagen, Shimadzu, TOC-V CPH, Japan), where  $\text{CO}_2$  formation was detected by IR. All leachates were analyzed for trace elements by inductively coupled plasma-mass spectrometry (ICP-MS, 4500, Agilent, USA) using  $^{103}\text{Rh}$  as internal standard.

## 2. Results and discussion

### 2.1. Trace element distribution in residues

The incineration residues were dominated by trace elements such as Zn, Cu and Pb (Table 3). An overall distribution behavior of trace elements will be discussed in this section with the help of literature. Distribution ratio for trace element transfer to fly ash (as a function of weighted average concentration (mg/kg dw) of

trace elements in residues (fly + bottom)) is shown in Fig. 1. Weighted average concentration ( $X$ -axis) and distribution ratio of trace element to fly ash ( $Y$ -axis) were calculated by following equations:

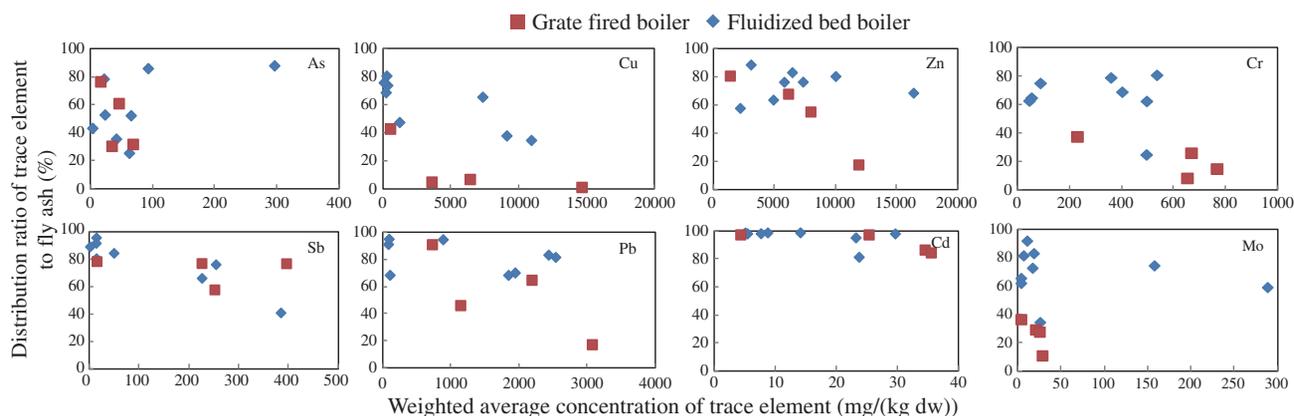
$$X_i = \frac{C_{\text{Bi}}M_{\text{B}} + C_{\text{Fi}}M_{\text{F}}}{M_{\text{B}} + M_{\text{F}}} \dots \dots \quad (1)$$

$$\text{Distribution ratio} = \frac{C_{\text{Fi}}M_{\text{F}}}{X_i \times (M_{\text{B}} + M_{\text{F}})} \dots \dots \quad (2)$$

where,  $X_i$  is the weighted average concentration of a metal in fly and bottom ash,  $C_{\text{Bi}}$  and  $C_{\text{Fi}}$  (mg/kg) are the mass concentration of heavy metal  $i$  in the bottom ash and fly ash respectively, and  $M_{\text{B}}$  and  $M_{\text{F}}$  (ton/day) are the mass of bottom and fly ash produced.

Cd is highly volatile trace element and chlorine in the waste also favors formation of chlorides. Therefore, it is believed that most of Cd and CdO entering the incinerator will be volatilized in elemental form and/or converted into  $\text{CdCl}_2$  (g) during incineration. This is also supported by thermodynamic calculations confirming that Cd(g),  $\text{CdCl}_2$ (g) and to some extent CdO can be vaporized (Shi et al., 2004). Cd might also be adsorbed onto small ash particles when the flue gas temperature is lowered (Shi et al., 2004). Therefore, a major part of Cd (an average value of 97%) was distributed to fly ash in this study (Fig. 1).

For Pb, formation and vaporization of  $\text{PbCl}_2$  (g) are the most preferential reaction during incineration (Chandler et al., 1997). In this study, 72% of Pb (based on weighted average value of all samples) was distributed to fly ash (Fig. 1). Sources of organic and inorganic chlorine (PVC, NaCl) in waste caused high transfer of Pb to fly ash through the formation of lead chloride (Astrup et al., 2011). A kinetic study by Asthana et al. (2010) also supported rapid and complete vaporization of Pb during MSW incineration. Ménard et al. (2006) have reported that the Pb and Zn chloride



**Fig. 1 – Distribution ratio of trace element to fly ash as function of total weighted average concentration in residues (fly + bottom) ash. dw: dry matter weight.**

species are stable in the entire incineration temperature range, and that evaporation and entrainment processes govern distribution, thus explaining excessive presence of Pb in some fly ashes in this study (Fig. 2).

Zn is a medium volatile element and volatilize only partially as  $ZnCl_2$  (g). The formation of stable oxides and compounds such as  $ZnAl_2O_4$  and  $ZnSiO_4$  prevents Zn from leaving the bottom ash (Zhang et al., 2012). However, if high chlorine contents are available or reducing conditions prevail, complete volatilization of Zn might be expected (Astrup et al., 2011). In this study around 67% of Zn distributed to fly ash (Fig. 1). Various input waste compositions, incineration temperatures and types of boiler affect distribution. These factors will be discussed in the next section.

In case of Sb, an average of 73% was distributed to fly ash which might be attributed to the formation of  $SbCl_3$ (g). Thermodynamic calculations also supported the formation of  $SbCl_3$ (g) (Verhulst et al., 1996; Watanabe et al., 1999). Paoletti et al. (2001) studied Sb partitioning by increasing chlorine content in waste, and found that volatilization significantly increased. Partial presence in bottom ash for some facilities might be due to the formation of thermally stable antimonates.

Arsenic showed variation by distributing around 56% into fly ash overall, while retaining in the bottom ash for few grate facilities (Figs. 1 and 2). Watanabe et al. (1999) reported that the formation and volatilization of arsenic as chloride are predominant process, however; it also reacts with ash matrix to form compounds like  $Ca_3(AsO_4)_2$ , which tends to retain As in bottom ash.

Mo partitioned around 56% (based on average values) into fly ash (Fig. 1). Similar results were reported by Astrup et al. (2011). However; Chandler et al. (1997) mentioned that it is usually enriched in bottom ash (typical concentration range from 2.5 to 276 mg/kg). In this study it ranged from 5 to 42 mg/kg. Powellite ( $CaMoO_4$ ) has been stated a possible species in bottom ash (Meima et al., 2002).

Lithophilic elements Cu and Cr are hardly volatilized and stay mainly in the bottom ash. Similar results were obtained in present study with an average distribution of 45% and 51% (Fig. 1), respectively. However; there was a surprisingly high transfer of Cu and Cr to some fly ashes at fluidized bed facilities (Fig. 2). Equilibrium analysis (Abanades et al.,

2002) indicated the presence of  $CuCl$  (g) as a dominant species during MSW incineration, so Cu might have vaporized to a small degree. Astrup et al. (2011) reported that firing CCA (chromate-copper-arsenate) wood with the base load (MSW) increased the transfer and concentration of Cr in fly ash. However, these findings were only true for CCA wood incineration.

#### 2.1.1. Impact of input waste fuel

Various types of waste fuel (pure biofuel, pure household/ industrial and mixed waste materials) were incinerated in both sorts of combustion system (grate and fluidized bed incinerators). Categorically, four types of waste fuel were treated at the investigated facilities: household, biofuel, industrial and mixed waste fuel (Table 1). Facilities with the same combustion system treating different waste fuels, are combined here to explain how input waste fuels affects the partitioning of trace elements between fly and bottom ash. For example:

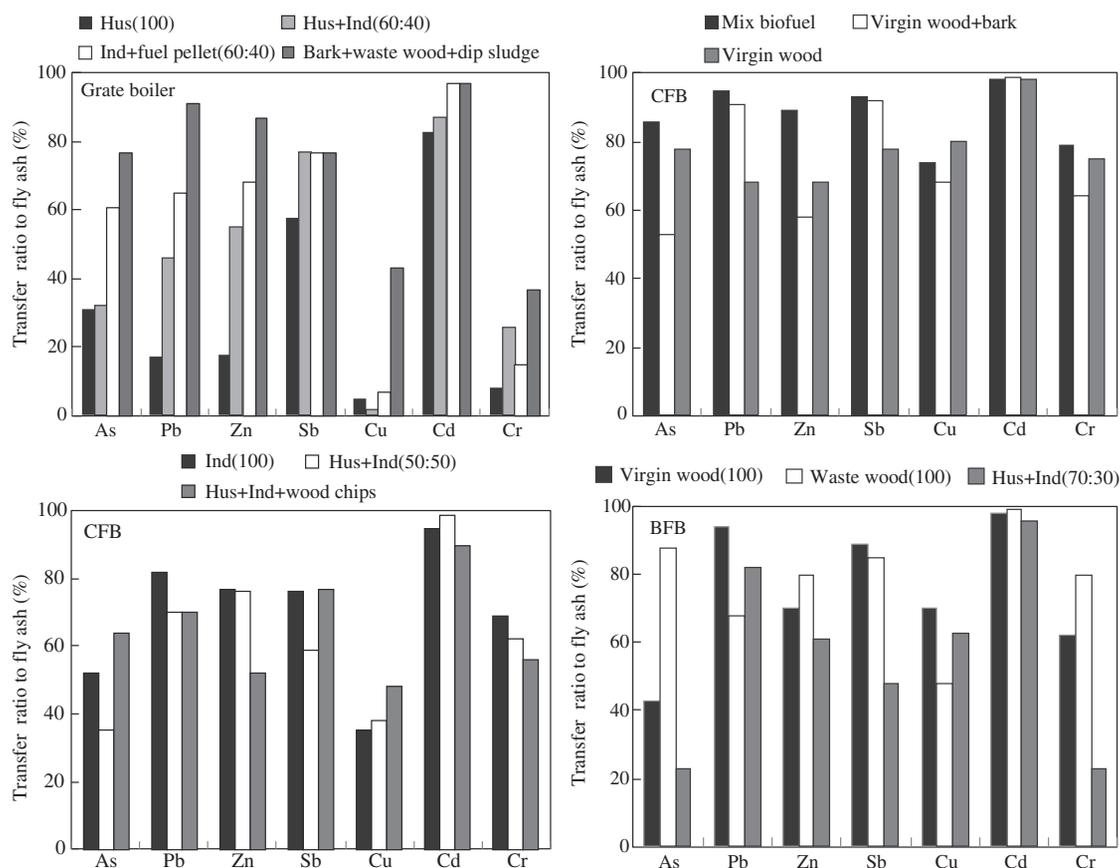
- The incineration facilities at Umeå, Kiruna, Söderenergi and Braviken treated pure waste fuel/mixtures of fuel (Table 1) in grate incinerators.
- (b, c) Six other facilities treated different waste fuel in CFB incinerators.
- (d) Similarly, Nynäshamn, Eskilstuna and Lidköping used BFB incinerators for waste incineration (Table 1).

Partitioning of trace elements in fly ash for different waste materials in various boilers is presented in Fig. 2. Transfer ratio ( $R_{Fi}$ ) of trace elements to fly ash was calculated by following equation:

$$R_{Fi} = \frac{FC_{Fi}}{BC_{Bi} + FC_{Fi}} \dots \dots \quad (3)$$

where,  $F$  and  $B$  (%) are the dry mass percentage of fly ash and bottom ash in the incineration residues, respectively, and  $C_{Fi}$  and  $C_{Bi}$  (mg/kg) are the mass concentration of heavy metal  $i$  in the fly ash and bottom ash respectively. It was assumed that input waste fuel matches the mass balance of fly and bottom ash, since no flue gas measurement was made.

Distribution behavior in Fig. 2 shows that, for pure household waste incineration in grate boiler (facilities (a)),



**Fig. 2 – Impact of input waste fuel on distribution of trace elements to fly ash. CFB: Circulating fluidized bed; BFB: Bubbling fluidized bed; Hus: household waste; Ind: industrial waste.**

most part of the studied trace elements such as Pb, Zn, Cu and Cr stayed in the bottom ash. Produced bottom ash (65 ton/day) at this facility (Kiruna) was almost 10 times higher than fly ash (6 ton/day), indicating the presence of higher total content in bottom ash. Cd was mainly found in fly ash with 60% of Sb as well. For incineration of mixed waste including household and industrial waste at another grate boiler (Umeå), partitioning to fly ash comparatively increased for Pb, Zn, Sb and Cr, while for Cu it decreased. Swedish industrial waste includes fractions like mixed metals, mixed plastics and even wood products that contain high concentrations of these elements and chlorine as well (Olofsson, 1998), which could be the reason for high transfer to fly ash. Arm et al. (2006) have reported an average of 1.5% chlorine in the incoming waste fuel (at the same facility), which could be responsible for increased volatilization and metal chloride formation during incineration.

Further increase in transfer of As, Pb, Zn and Sb to fly ash was observed for the incineration of mixed waste containing fuel pellets (Söderenergi). Blomqvist and Jones (2012) reported high chlorine content (2.1%) for the waste fuel pellets being used at this facility. Therefore enrichment of these elements in fly ash is likely during incineration of mixed waste. Additions of certain waste fractions such as wood products and plastics can also affect the partitioning behavior of trace elements. Astrup et al. (2011) reported that the addition of impregnated wood, PVC and batteries to the

normal household waste significantly increased the emissions of As, Cr, Cd and Sb.

Highest transfer to fly ash in grate boilers was observed for the incineration of waste containing biofuel (Braviken). Increased transfer for As, Cu, Cr and Zn was noted. Waste fuel characterization at the same facility showed high concentrations of Zn (950 mg/kg dw) and Cu (230 mg/kg dw) in the sludge which is making up a major part (51%) of the fuel mixture (Blomqvist and Jones, 2012). Contaminated waste wood that makes up about 16% of this waste fuel (Table 1) also often contains high concentration of these elements. In Sweden, the most important sources contributing to contamination of recovered waste wood are impregnated wood, copper-chromium-arsenic (CCA), industrial preserved wood, plastics and galvanized fastening system (Krook et al., 2004). The surface-treated wood is considered the major source of Pb, Zn, Cr and As. Therefore, an increased loading (input concentrations) might have caused high transfer of these elements to fly ash.

For pure industrial waste incineration in CFB (Högdalen), as described above in facilities (b), >50% of all trace elements distributed to the fly ash except Cu. Swedish industrial waste is a complex mixture containing waste from several sectors (pulp and paper, wood, steel, metal and textile industry), so it is hard to say specifically how much of a certain trace element is contributed by a particular sector. According to Olofsson (1998), major individual fractions are paper and cardboards

(33%), wood products (27%), mixed metals (11%) and mixed plastics (9.5%). For mixed waste (household + industrial) incineration in another CFB (P14), >75% of Pb, Zn and Sb partitioned to fly ash. Addition of household waste can also cause an increased input of organic and inorganic chlorine. According to SEPA (2005), food waste (source of NaCl) accounts for about 43% of typical Swedish household waste with 22% packaging (source of vinyl chloride monomer (PVC)), 8% newspaper and 2% chlorinated plastics (source of PVC), therefore, a considerable amount of inorganic and organic chlorine will enhance the formation and vaporization of  $PbCl_2$ ,  $ZnCl_2$  and  $SbCl_3$ . Incineration of mixed waste containing wood chips increased the transfer to fly ash for Cu and As, while the transfer of Zn was lowered. Cr in all the cases was found >50% in fly ash.

For mixed biofuel (peat, bark, waste wood) incineration in CFB (Mälarenergi), as described in facilities (c), >75% of all trace elements partitioned to the fly ash. Wiinikka et al. (2013) also reported high concentration of Pb and Zn in flue gas during incineration of peat and bark. They mentioned that a clear and strong correlation exists between concentrations of these elements in original fuel and in fly ash particles. However, other factors such as fly ash sampling location and flue gas treatment system might also affect the distribution.

Mixing virgin wood with bark showed a slight decrease in transfer to fly ash for As, Zn, Cu and Cr. Overall, for biofuel incineration in CFB (Munksund), most of the trace elements were distributed to the fly ash. Similar results about fate of trace elements during combustion of Swedish mixed biofuel have also been reported (Andersson and Tullin, 1999). They found that 88% Pb and 77% Zn were distributed to fly ash during incineration and increased loading in waste fuel and entrainment in fluidized boilers, affecting the distribution of these elements.

For virgin wood incineration in BFB (Eskilstuna, facilities (d)), most of the trace elements found their way to the fly ash especially Pb, Sb, Zn and Cd. Lind et al. (2003) investigated the vaporization of trace elements such as Zn, Cd, Pb and Cu during the incineration of forest waste and observed that the major fractions of all these elements were enriched on fly ash particles. For pure waste wood incineration (Nynäshamn), distribution to fly ash for Cr, Zn and As clearly increased. The reason might be an increased input concentration of these elements originating from CCA impregnated wood as shown by Blomqvist and Jones (2012) at the same facility (Nynäshamn). Pedersen et al. (2009) also reported high transfer of Zn to fly ash for CCA wood incineration and concluded that it might be either due to increased input concentration or because of a change in vaporization behavior of elements due to change in feedstock chemistry. For mixed waste incineration in BFB (Lidköping), the major part of Pb, Zn, Sb and Cd distributed into fly ash.

Overall, for fluidized bed facilities, especially those using waste wood/virgin woods, most of the trace elements partitioned to fly ash were also observed by Jermer et al. (2001). For pure household waste incineration, only Cd and Sb partitioned into fly ash, while for pure industrial or mixture of household and industrial waste incineration, 2/3 or more of Pb, Zn, Sb and Cd transferred into fly ash. For virgin wood,  $\geq 60\%$  of all studied elements transferred to fly ash. Zn, Cr and As were mainly present in fly ash for waste wood incineration. For fluidized boilers, low volatiles also showed an increased transfer to fly ash.

Although the composition of the residues has been shown to vary substantially with time as a function of process conditions and input waste composition. However, from our results and previous research about partitioning characteristics, one or more of following reasons might be written for an increased transfer of elements to fly ash:

- (1) An elevated input concentration of certain trace elements triggered an increased concentration in resulting ash, e.g., Cr, As, Zn and Cu from CCA wood or Zn, Pb, Cu and Cl from household/industrial waste.
- (2) Mixing certain fractions in the incoming waste might change the partitioning or volatilization behavior of elements (Pedersen et al., 2009).
- (3) High combustion efficiency due to pre-treatment of waste fuel in fluidized bed boilers.

#### 2.1.2. Incinerator temperature

The volatility of trace elements generally increases with furnace temperature due to vapor pressure increase. Shi et al. (2004) showed decreasing order of volatility and vaporization rate as  $Cd > Pb > Zn > Cu$  during MSW incineration. In this study, only Zn and Cd concentrations were increased in fly ash with incineration temperature (Fig. 3). The enrichment of these trace elements in fly ash at high temperatures was possibly due to vaporization and entrainment processes.

#### 2.1.3. Incinerator type and flue gas treatment system

Trace element concentrations of the ashes from facilities using various incineration technologies are shown in Table 3. The average total concentration of trace elements including Cr, Cu, Ni and Cl was higher in fly ash as compared to bottom ash for fluidized boilers. In grate boilers, average concentration of Cu and Ni was high in bottom ash. In this study, mostly low volatiles (Cr, Cu) distributed more into fly ash for fluidized boilers, mainly those using biofuel for incineration. Lundholm et al. (2008) observed a small increase in volatilization of Cr during CCA wood incineration, but Cr is not considered thermally mobile during incineration. The raw particulate carried over from the furnace and even clean flue gas dust has been reported to contain considerable concentrations (100–1200  $\mu\text{g/g}$ ) of Cr (Chandler et al., 1997).

A simple comparison between boiler types can also be made here. Mixed waste fuel containing household and industrial waste was incinerated in both grate (Umeå) and CFB (Händelö P13) boilers. Moving from grate to CFB boiler for this mixed waste incineration, transfer of all studied trace elements increased in fly ash. Similarly, wood chips were incinerated in both CFB (Händelö P13) and BFB (Eskilstuna) boilers. Moving from CFB to BFB for virgin wood incineration, volatilization of As, Cr, Pb and Cu decreased which might be due to the comparatively lower incineration temperature (782°C) in BFB (Table 1).

The amount of produced bottom ash was 2–5 times higher than that of fly ash in grate facilities, while fly ash production in most of the fluidized boilers was equal to or higher than that of bottom ash. This indicates the presence of higher total element content in fly ash for fluidized boilers. The difference might be due to high combustion efficiency in fluidized boilers. Since input fuel is usually pretreated (sizing,

shredding) before it goes into incinerator for fluidized facilities, slightly more uniform and smaller composition of waste fuel would enhance combustion, while in case of grate facilities, uneven distribution of fuel on grate can cause lower combustion and high unburned content in spite of high incineration temperature.

Fly ash was collected mostly from electrostatic precipitators (ESP) or bag filters while for one facility (Sundsvall) it was sampled as a mix between ash from the bag filter and the ESP (Table 1). Mechanisms governing the distribution between bottom ash and fly ash take place in upstream of the flue gas treatment system, however, just for comparison between sampling locations in the current study, for facilities using ESP, most of the Pb partitioned to fly ash, and for (ESP + bag filter), low volatiles distributed more to fly ash.

## 2.2. Leaching characteristics of ashes

### 2.2.1. Fly ash leaching at the liquid to solid ratio (L/S) of 10

Leaching characteristics of trace elements as per standard leaching procedure with water at L/S 10 are presented in Fig. 4. Facilities (at X-axis) are arranged in plots such that it represents a variation of input waste from pure household to pure industrial, waste/virgin wood and finally mixed biofuel. It starts with pure household waste (Kiruna), proceeds with decreasing proportions of household waste in mixed waste (household + industrial), to pure industrial waste (Högdalen), waste wood, virgin wood (Nynäshamn, P13 and Eskilstuna) and at the end mixed biofuel. Results showed that as we move from pure household to pure biofuel waste or mixed biofuel, the leaching of As, Cu, and to some extent Pb and Zn, decreased. Correlation between total and leached concentrations was calculated for Cu ( $r^2 = 0.8$ ) and Pb ( $r^2 = 0.5$ ). For other elements no significant correlations were found. Average amounts of trace elements leached with water from the fly ashes followed the order of Pb > Cu > Cr > As > Zn > Mo > Sb. pH of fly ash leachate was around 11–13.5, indicating high alkalinity and buffering capacity. This is not only partly attributed to the use of Ca(OH)<sub>2</sub> for removal of acidic gases and flue gas treatment, but also due to the natural content of oxides (CaO and MgO) in ashes.

Results indicated that there were significant differences between samples with respect to leaching. The amount of leachable trace elements in fly ashes was generally low with leaching below 1% of the total content except for some

samples such as Lidköping (household + industrial) fly ash. This ash leached 43% Pb and 10% Cu, which means that significant amounts of these elements were present as water soluble phases. Water soluble compounds of these elements are likely to be chlorides, bromides or sulfates such as CuCl<sub>2</sub>, PbBr<sub>2</sub>, ZnSO<sub>4</sub>, PbClOH, Pb<sub>4</sub>(OH)<sub>6</sub>SO<sub>4</sub> and CaZn<sub>2</sub>(OH)<sub>6</sub>·2H<sub>2</sub>O (Karlfeldt and Steenari, 2007; Quina et al., 2009). Leaching of these trace elements in other samples was low, e.g., Pb in Kiruna (1.4%) and Sundsvall (0.7%). Under the highly alkaline environment, metal hydroxide formation is favored that might limit the release of some elements. Fedje et al. (2010) mentioned that dissolution and precipitation control leaching of Pb at alkaline pH favoring the formation of Pb(OH)<sub>3</sub><sup>-</sup> and Pb(OH)<sub>4</sub><sup>2-</sup> complexes at high pH over solubility controlling phase Pb(OH)<sub>2</sub>. However, speciation of trace elements is not the only factor affecting leachability, but ash matrix and position of trace element on ash particles are also decisive factors in determining leached amounts.

Trace elements behave in three different ways to water leaching. Some elements are leached easily and in high concentrations, e.g., Pb and Cu from Lidköping (household industrial mix) and Mo from Söderenergi ash (industrial and fuel pellets mix). Most probably such elements were deposited on the surface of ash particles and being water leachable species easily washed away on contact with water. A similar study (Fujimori et al., 2002) on industrial waste fly ash showed that Zn, Cu and Mo were enriched on the surface while Pb, Ni and Co were present inside as well as on the surface of ash particles. However, leaching of trace elements is strongly dependent on pH of leachate, geochemical nature of ash matrix and L/S, but in present study, trace element leaching was not related to pH ( $r^2$  value ranges from -0.01 to 0.1). Secondly, some elements might have existed in the center of ash particles or enclosed inside the particles, they were leached moderately and slowly only after surface deposited elements were washed away. Possible example from current study can be Sb and Pb release from Sundsvall fly ash, because higher amount of Sb and Pb was leached at L/S 8 as compared to L/S 2. In another study (Pinzani, 2003) using XRF spectroscopy, it was concluded that Cd and Pb were mainly present in the core of ash particles while Zn was present on both positions inside and on surface of ash particles. Thus, encapsulation of trace element species inside ash particle can also cause low leach abilities. However, further investigation is needed to quantify distribution of elements on and inside the ash particles, to clarify the mechanism.

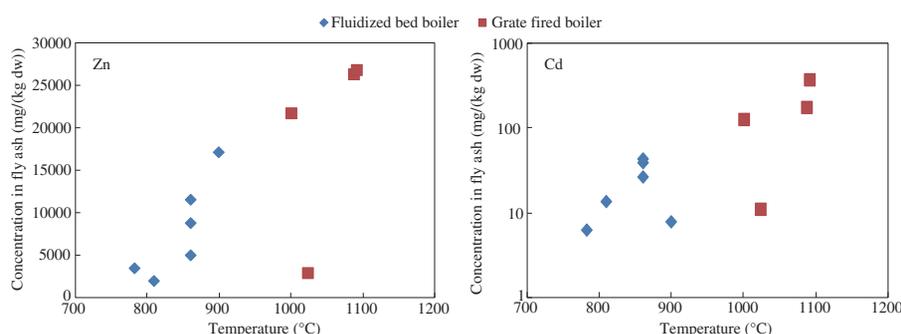
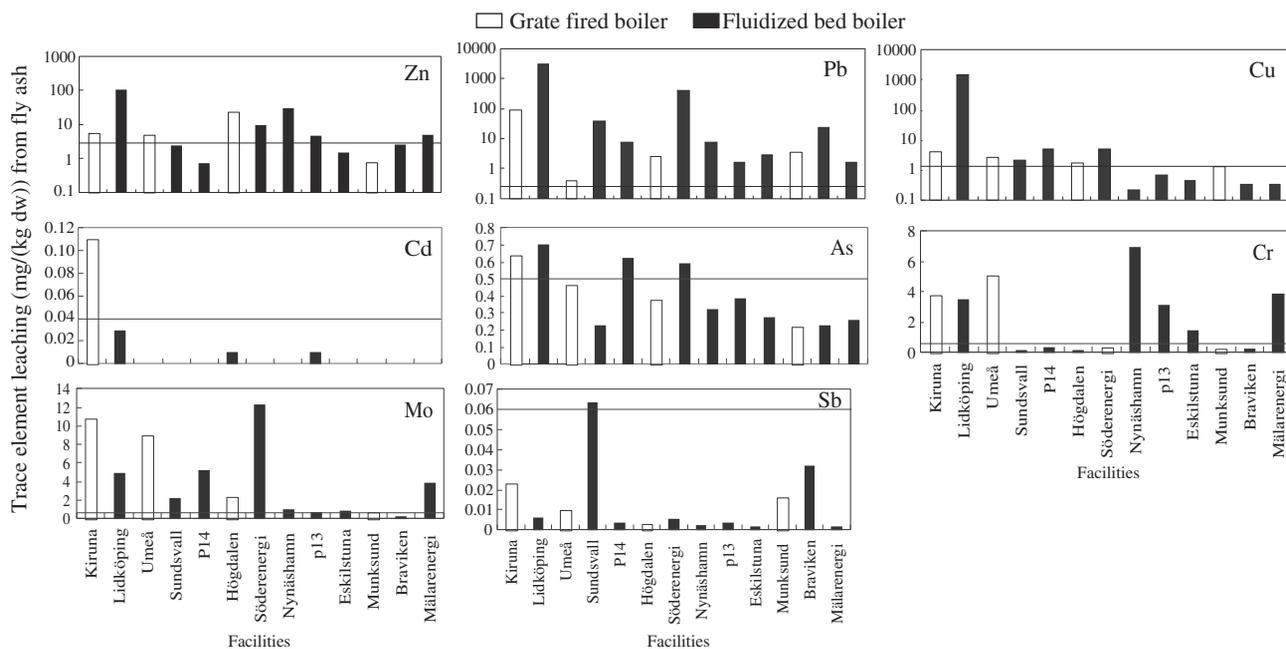


Fig. 3 – Correlation between furnace temperature and trace element concentration in fly ash.



**Fig. 4 – Trace element leaching from fly ash with water at the liquid to solid ratio (L/S) of 10. The horizontal line in plots indicates the limit value of an element in ash for disposal criteria.**

Another category is of those elements which are insoluble and are below detection limit. For example Cd in many ashes was below the detection limit (Fig. 4). It was also reported by Ramesh and Kozinski (2001) that Cd and Pb concentration increases towards core and decreases on the surface of fly ash particles. Further, these elements form stable compounds with alumino-silicate matrix, and are therefore not easily leached.

Leaching of elements varied in all the ashes, possibly also due to different concentration levels in ashes and the existence of different chemical speciation.

With respect to the impact of boiler technology on leaching characteristics of trace elements; average leached concentration of Cu (181 mg/kg dw), Pb (343 mg/g dw) and Zn (19 mg/kg dw) in fluidized boiler fly ash was higher than in grate fly ash (Cu 3 mg/kg dw, Pb 132 mg/kg dw and Zn 5 mg/kg dw), while for As, Cr and Mo it was *vice versa*. Correlation coefficient ( $r^2$ ) between total and leached concentrations in fluidized boiler fly ash for Cu (0.83) and Pb (0.8) is high. Similarly, the correlation coefficient for As, Cr and Mo in grate fly ash is around 0.9, which is an indication of availability controlled leaching.

The horizontal line in leaching plots (Fig. 4) represents the limit value of an element in ash for disposal criteria (SEPA, 2006). Following general and specific points can be written for leaching of fly ash from current study:

1. For facilities using virgin wood, only Cr and Pb exceeded the regulatory limit, while for waste wood incineration fly ash Zn, Cr and Pb exceeded the regulatory limit.
2. Facilities using mixed waste (household and industrial) had Cu and Mo in common exceeding the regulatory level in the respective fly ashes.

3. Braviken fly ash (mixed biofuel) only had Pb leaching more than the regulatory limit.
4. Kiruna fly ash (household) was the most contaminated with Pb, Cd, Cu, Zn, Cr, and Mo leaching, exceeding the regulatory limit.

Above discussion indicates that water soluble forms of elements are generally low in fly ash. Leaching behavior of trace elements is different in various waste fuel ashes. Further ash matrix and total concentration in the ash are critical factors affecting the leaching of trace elements.

**2.2.2. Bottom ash leaching at the liquid to solid ratio (L/S) of 10**  
Results for bottom ash leaching are shown in Fig. 5. Leached concentrations of As decreased as household waste proportion in the mixture decreased (from Kiruna to Sundsvall). Waste wood ash at Nynäshamn showed highest release of As, probably due to high concentration in respective bottom ash due to use of CCA wood (Table 3). Cu and Sb also showed an increased leaching with increasing concentration in ashes. Leached amounts of Pb decreased from household to biofuel ashes. Correlation coefficients ( $r^2$ ) found between total and leachable contents for Cu (0.8), Sb (0.7) and As (0.6) were also high. The average amounts of trace elements leached with water from bottom ashes followed the order of Pb > As > Cr > Sb > Cu > Zn > Mo. Mälarenergi and Munksund (mixed biofuel) bottom ashes showed low leaching for all trace elements.

Average leached concentrations of Cu, Pb and Zn (2.4, 2.5 and 3.0 mg/kg dw) were higher in grate bottom ashes than in fluidized bottom ashes (1.4, 1.6 and 1.0 mg/kg dw). The same trend was observed for total concentrations of these elements in bottom ashes. Following comments can be written with respect to disposal criteria (SEPA, 2006) for bottom ash leaching:

1. Most of the facilities using household waste had Cu and those using industrial waste had Sb and Cu in common, exceeding the regulatory limit. For facilities using contaminated wood chips, Cr exceeded the limit.
2. Nynäshamn and Högdalen bottom ashes were the most contaminated with As, Cr, Sb, Pb, Cu, and Cr leaching, surpassing regulatory level, respectively.

Leachate's pH ranged from 10.7 to 13.1 and again trace elements released from bottom ash showed no pH dependency in this pH interval. At high pH, amphoteric trace elements like Cu, Zn and Pb usually form soluble hydroxides. Zhang et al. (2008) reported that dissolution and precipitation were the major mechanisms controlling leaching under alkaline conditions and that  $\text{Pb}(\text{OH})_2$ ,  $\text{CuO}$ ,  $\text{ZnO}$ ,  $\text{Cr}_2\text{O}_3$  and  $\text{Cd}(\text{OH})_2$  were major leaching controlling phases of respective elements in MSW bottom ash. Pb precipitates as  $\text{Pb}(\text{OH})_2$  (s), and under very strong alkaline conditions it tends to combine with  $\text{OH}^-$  to form  $\text{Pb}(\text{OH})_4^{2-}$  and  $\text{Pb}(\text{OH})_3^-$  that are more leachable fractions (Chiang et al., 2009). Various solubility controlling minerals have been reported in fresh bottom ash, e.g.,  $\text{Pb}(\text{OH})_2$  (s),  $\text{PbSO}_4$  (s),  $\text{ZnO}$  (s),  $\text{ZnSiO}_3$  (s),  $\text{Cu}(\text{OH})_2$  (s),  $\text{CuO}$  (s) and  $\text{CdCO}_3$  (s) (Chou et al., 2009; Chiang et al., 2009). Calcite ( $\text{CaCO}_3$  (s)), which forms due to weathering process and operates as buffer for ashes keeping pH high for a long time, may also affect the trace element release. However, many factors such as ash particle size, trace element speciation, ash mineralogy, redox potential, presence of complexing agents and L/S govern the element mobility.

Dissolved organic carbon (DOC) has been reported to affect trace element leaching from bottom ash especially for Cu (Yao et al., 2011). Studies have confirmed Cu-DOC complex formation using geochemical speciation model and a 95% association of Cu with DOC is found in Swedish bottom ash (Olsson et al., 2007). DOC release from bottom ash from this study is shown in Fig. 7. There might be some Cu leaching

associated with Cu-DOC complex formation. However further investigation is needed to confirm and determine the possible fractions of DOC in bottom ash leachates. Processes that are reported mostly controlling the leaching of elements are mostly dissolution, precipitation and adsorption mechanism (Zhang et al., 2008; Quina et al., 2009).

### 2.2.3. Chlorine leaching

Average chlorine concentrations in fly ash from grate facilities were found to be higher than that of fluidized bed facilities (Table 3). Results of leaching are presented in Fig. 6.

Chlorine leaching in 61% of fly ash samples exceeded set limit which is 0.8% for landfill disposal (SEPA, 2006). Exposing fly ash to water removed >87% of total chlorine at L/S 10, indicating the importance of water washing process for incineration residues before any other treatment. Astrup et al. (2011) reported that affinity between chlorine and cations follows decreasing order of  $\text{H} > \text{Na/K} > \text{Pb} > \text{other trace elements}$ . Thus, formations of highly water extractable alkali chlorides ( $\text{NaCl}$ ,  $\text{KCl}$  and  $\text{CaCl}_2$ ) are much likely during incineration, resulting in the removal of large proportion of available chlorine. Zhu et al. (2010) observed that in fly ashes, where  $\text{Ca}(\text{OH})_2$  was used for acid gas removal, contained 35% chlorine as  $\text{NaCl}$ , 11% as  $\text{KCl}$ , 37% as  $\text{CaCl}_2$  that are easily soluble in water and 13% was found as Friedel's salt ( $3\text{CaO}\cdot\text{Al}_2\text{O}_3\cdot\text{CaCl}_2\cdot 10\text{H}_2\text{O}$ ), an insoluble chloride. In current study, most of the facilities had used lime for flue gas treatment (Table 2), so probably a similar chlorine distribution is possible. A water washing step must be included for fly ashes before landfilling or for certain applications to avoid any adverse environmental effect.

For bottom ashes only Umeå ash showed a removal of 45% at L/S 10. For most facilities using biofuel for incineration, leached chlorine concentration was well below the limit for inert waste. The leaching of chlorine is availability controlled and is independent of pH because  $\text{Cl}^-$  offers rapid and complete dissolution at all pH values (Wang et al., 2001).

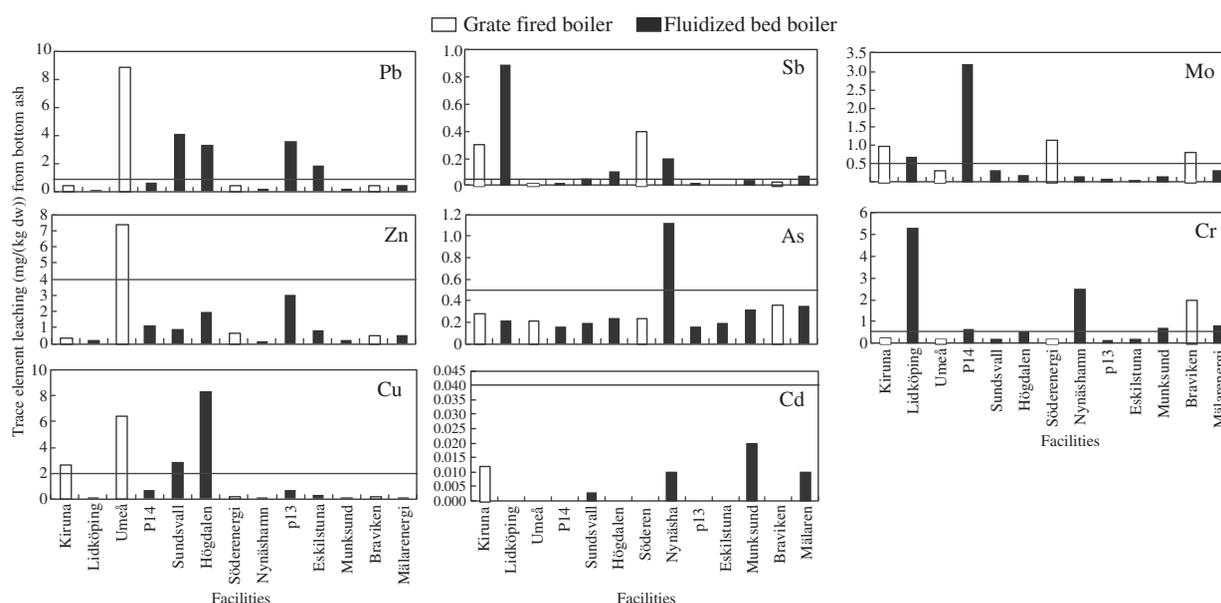


Fig. 5 – Trace element leaching from bottom ash with water at the liquid to solid ratio (L/S) of 10. The horizontal line in plots indicates the limit value of an element in ash for disposal criteria.

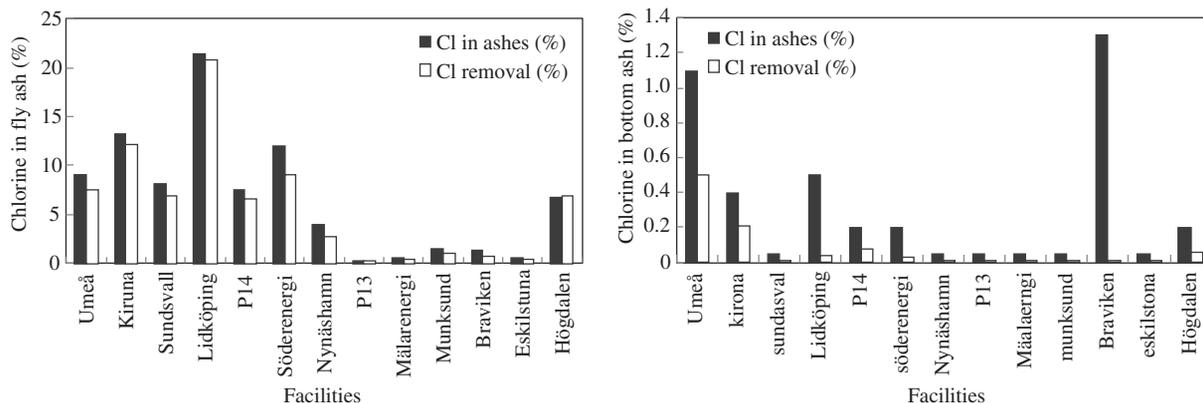


Fig. 6 – Chlorine (Cl) leaching from bottom and fly ash with water at the liquid to solid ratio (L/S) of 10.

2.2.4. Dissolved organic carbon (DOC) leaching

Organic carbon may enhance transport of contaminants from incineration residues due to its complex forming ability with trace elements (Fedje et al., 2010). Generally, bottom ash contains more water extractible organic compounds than fly ash. According to Ecke and Svensson (2008), about 60% of TOC (organic and elemental carbon) in typical Swedish MSWI bottom ash is water extractable, while for fly ash it is only 7%. DOC leaching results for residues in current study are presented in Fig. 7. It shows that comparatively high DOC amount was leached from bottom ash as compared to fly ash. DOC limit in waste leachate at L/S 10 is 500 mg/kg dw for inert and 800 mg/kg dw for non-hazardous waste (European Commission, 2003). Results showed that DOC content in all fly and bottom ash leachate was well under this limit. It was reported (Ecke and Svensson, 2008) that for Swedish waste incineration; bottom ash from grate boilers contains more water extractible organic carbon than fluidized bed boilers. Results of our study are also similar, since higher amounts of DOC leached for bottom ash from grate facilities Umeå, Kiruna and Braviken. One reason for low release from fluidized bed bottom ash might be the presence of excessive elemental carbon since it forms about 40%–90% of TOC and is reported as non-reactive and low water soluble (Ecke and Svensson, 2008).

DOC enhances metal mobility. According to Pavasars (1999), 23% of DOC in Swedish MSWI bottom ash leachates consists of organic molecules with characteristics similar to humic acids. Further, organic matter in Swedish MSWI bottom ash consists of 74% cellulose, 20% lignin, 4% water soluble organic compound with low molecular weight and 0.2% organic compounds extractable with organic solvents, indicating that a major part of organic matter is insoluble in water. At high pH, silicates and aluminum minerals are dissolved, and adsorbed organic compounds on these minerals are also mobilized. DOC leaching from fly ash was found lower as compared to bottom ash, possibly due to the enrichment of elemental carbon. Excessive presence of elemental carbon might be attributed to incomplete combustion, while low leaching of DOC can also be due to the formation of low soluble and high adsorption capacity minerals.

3. Conclusions

Impact of input waste fuel and incineration technology was investigated on partitioning and leaching characteristics of trace elements in bottom and fly ashes.

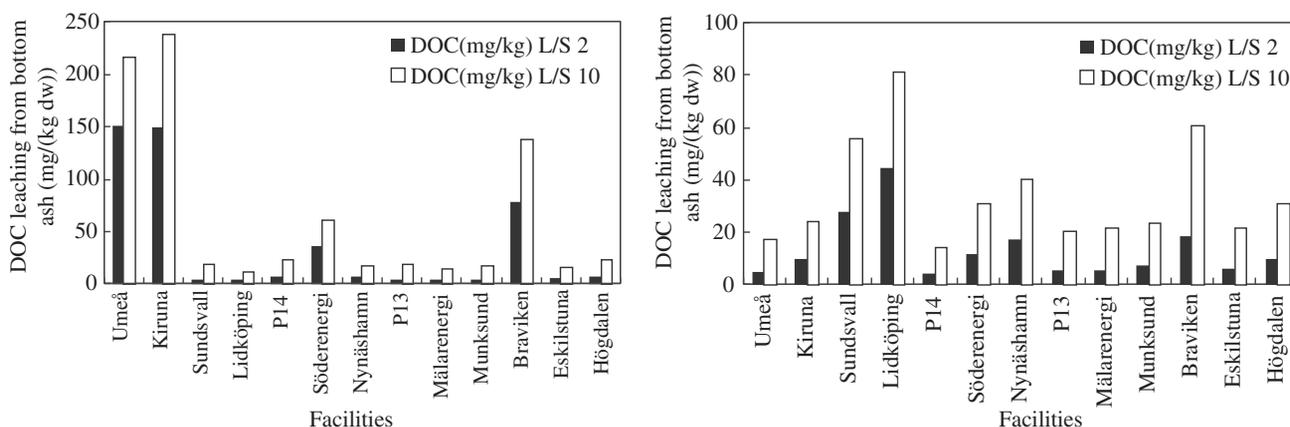


Fig. 7 – Dissolved organic carbon (DOC) leaching from bottom and fly ash with water at different liquid to solid ratios (L/S).

Firing mixed waste, especially biofuel mix, in grate or fluidized (CFB) boilers caused increased transfer into fly ash for almost all trace elements particularly Pb and Zn. Possible reasons might be either an increased input concentration of respective element in the waste fuel, or a change in volatilization behavior due to the addition of certain waste fractions. Similarly, firing of waste wood caused high partitioning to fly ash for Cr, As and Zn compared to firing of virgin wood, probably due to increased loading of these elements in waste wood.

Fluidized boilers mostly accommodated significant proportion of trace elements in fly ash during incineration even for non-volatile Cr and Cu, possible reason being the high combustion efficiency due to pre-treatment (sizing, shredding) of waste fuel. Concentration of trace element mostly follows the sequence of Zn > Cu > Pb > Cr > Sb > As > Mo for both residues.

During incineration, Cd was partitioned mostly in fly ash in all cases, most probably because of vaporization, condensation on fly ash particles and adsorption mechanisms. In most of cases, 2/3 of Pb was also transferred to fly ash, but As, Zn and Sb showed high variations, with most of Sb and Zn also distributing to the fly ashes. Low volatiles stayed in bottom ash for grate facilities, but distributed more into fly ash in fluidized boilers.

Cd and Zn concentration increased in fly ash with incineration temperature, possibly due to their chloride formation and volatilization processes.

Water soluble forms of trace elements in ashes were low with few exceptions. Ash matrix and total concentration in ash seem to play a vital role in the leaching of trace elements with water. For disposal criteria, leached amounts of Pb, Zn and Cr exceeded regulatory limit in 84%, 61% and 53% of the fly ash samples respectively, so all these fly ashes must be treated before landfill or application.

Most of the total chlorine in fly ash ( $\geq 87\%$ ) was leached out at L/S 10, implying that chlorine is availability controlled and water washing of residues is an efficient pre-treatment step. Further, none of the ash exceeded limit for disposal with respect to DOC leaching.

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