The Mass Flow and Proposed Management of Bisphenol A in Selected Norwegian Waste Streams

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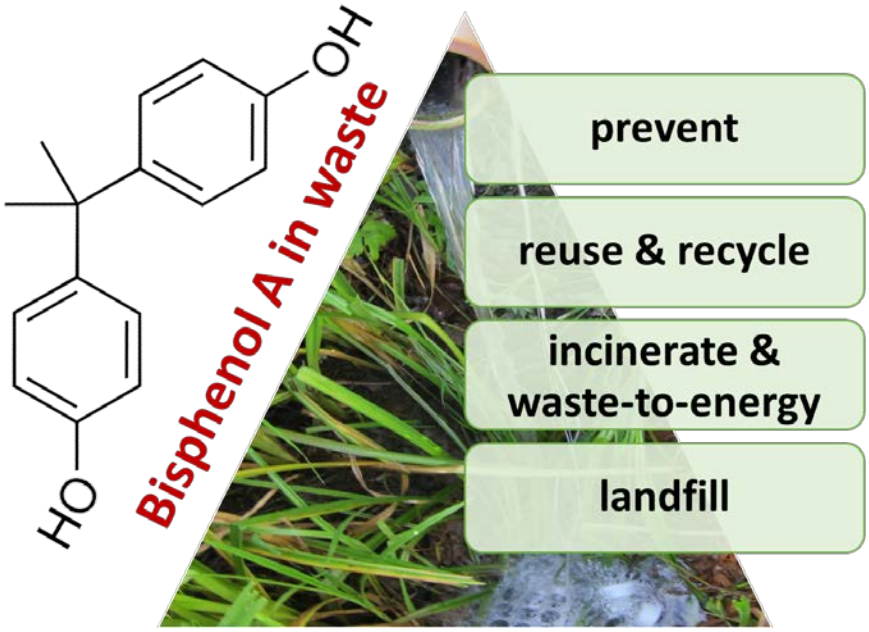

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Abstract

Current initiatives for waste-handling in a circular economy favor prevention and recycling over incineration or landfilling. However, the impact of such a transition on environmental emissions of contaminants like bisphenol A (BPA) during waste-handling is not fully understood. To address this, a material flow analysis (MFA) was constructed for selected waste categories in Norway, for which the amount recycled is expected to increase in the future; glass, vehicle, electronic, plastic and combustible waste. Combined, 92 tons/y of BPA are disposed of via these waste categories in Norway, with 98.5% associated with plastic and electronic waste. During the model year 2011, the MFA showed that BPA in these waste categories was destroyed through incineration (60%), exported for recycling into new products (35%), stored in landfills (4%) or released into the environment (1%). Landfilling led to the greatest environmental emissions (up to 13% of landfilled BPA), and incinerating the smallest (0.001% of incinerated BPA). From modelling different waste management scenarios, the most effective way to reduce BPA emissions are to incinerate BPA-containing waste and avoid landfilling it. A comparison of environmental and human BPA concentrations with CoZMoMAN exposure model estimations suggested that waste emissions are an insignificant regional source. Nevertheless, from monitoring studies, landfill emissions can be a substantial local source of BPA. Regarding the transition to a circular economy, it is clear that disposing of less BPA-containing waste and less landfilling would lead to lower environmental emissions, but several uncertainties remain regarding emissions of BPA during recycling, particularly for paper and plastics. Future research should focus on the fate of BPA, as well as BPA alternatives, in emerging reuse and recycling processes, as part of the transition to a circular economy.

Keywords: Bisphenol A, waste hierarchy, mass flow, WEEE, plastic

1 Introduction

Bisphenol A (BPA) is a hazardous, endocrine disrupting compound (Vandenberg et al., 2007) that is used in many plastics, epoxy resins, paper and paper products (EC, 2008; Liao and Kannan, 2011). BPA is subject to an increasing number of restrictions in certain products that present a human exposure risk, such as infant feeding bottles (e.g. EU directive 2011/8/EU). Despite these restrictions, the worldwide production of BPA is expected to grow by 5.1% from 2014 to 2019 (TechNavio, 2015), with 2012 production levels estimated at 4.6 million tons . An emerging issue in Europe is what effect policy initiatives that set targets to change waste management practices as part of the transition to a circular economy (EC, 2015b) will have on hazardous compounds like BPA. The two main implications under discussion are, firstly, how this will influence environmental emissions, and secondly, how this will effect exposure through occurrence in recycled products. (Pivnenko et al., 2016a). In this study, we provide an examination of the first issue, how waste-handling alternatives will influence the environmental emissions of BPA.

BPA is water soluble (water solubility 300 mg/L, log Kow = 3.4) (Cousins et al., 2002), and therefore BPA-containing waste can readily release BPA into the aqueous environment (Cousins et al., 2002; Morin et al., 2015). In Norway and elsewhere, municipal and industrial waste landfill leachate often has BPA concentrations that are substantially higher than its chronic predicted no-effect concentration of 1.6 µg/L (e.g. a Norwegian survey reported an interquartile range of 1 – 62 µg/L, maximum 692 µg/L) (Arp, 2013; Morin et al., 2015). Therefore, waters receiving landfill leachate are particularly vulnerable. Wastewater treatment plants (WTP), be they municipal or for a particular waste handling facility, are poorly equipped to eliminate BPA from leachate, as is evident by the frequent detection of BPA in WTP effluent (Guerra et al., 2015; Lee

et al., 2015; Mohapatra et al., 2010; Yu et al., 2015). BPA is not prone to long-range environmental transport, having a half-life in water of 0.5 to 6 days (Klečka et al., 2001). However, in waters impacted by waste-handling facilities, a sustained elevated presence can be expected. A sound understanding of BPA emissions from waste is therefore important in order to establish effective waste-management strategies that reduce BPA-related risks to the environment.

Previous studies to investigate emissions of BPA into the environment have focused on BPA-rich products such as polycarbonate plastic and bottles (Cooper et al., 2011; Sajiki and Yonekubo, 2003, 2004), waste electrical and electronic equipment (WEEE) (Zhang et al., 2016) and paper (Geens et al., 2012; Liao and Kannan, 2011; Pivnenko et al., 2016b; Pivnenko et al., 2015b), or specific waste products such as WTP sludge (Mohapatra et al., 2010). Modelling and monitoring studies have also been developed for an area near a paper-processing plant (Fürhacker et al., 2000), as well as the EU and US on the regional scale (Cousins et al., 2002).

In this current study, we aim to better understand how changes in waste-management can influence environmental emissions of BPA from waste-handling practices, and in particular those within Norway. The context for this work is that European policy makers have recently set ambitious targets to landfill less waste and recycle more as part of the shift to a circular economy (EC, 2015a, b, c). In order to address this, the present study presents: (i) a Material Flow Analysis (MFA) of BPA in selected types of wastes in Norway as they undergo sorting and defragmenting, landfilling, incineration and recycling, accounting for environmental emissions to air, water and water treatment plant (WTP) sludge; (ii) an assessment of whether waste-handling emissions contribute substantially to environmental and human exposures on a regional scale; and (iii) a method that allows the manipulation of the MFA to estimate differences in BPA emissions caused

by different waste-management scenarios that may be adopted in the near future as part of the transition towards a circular economy.

2 Materials and methods

2.1 System description

Table 1 presents a conceptual sketch of the waste-management system boundaries used to construct the MFA for the Norwegian waste-stream. Herein, five waste categories were considered in this study: glass, vehicles, waste electrical and electronic equipment (WEEE), plastics, and combustibles. These waste categories undergo sorting and defragmenting, where subfractions are sent to be either landfilled, incinerated or recycled. During incineration waste is transformed to gases, fly ash and bottom ash. In Norway, fly ash is landfilled as hazardous waste, and bottom ash undergoes metal recycling followed by landfilling as normal waste. All waste treatments produce atmospheric emissions or water leachate. Water leachate can be emitted directly to the environment, or be treated by a water treatment plant (WTP), which would emit treated water and sludge.

There are three ways that waste and BPA mass can exit the system boundaries presented in Figure 1; i) thermal-destruction through incineration, ii) the export of recycled/recyclable materials, and iii) environmental emissions in the form of air, water and WTP sludge. New products from recycled materials are considered to be outside the system boundaries. Further, products of WTP sludge, which in Norway includes biogas and soil fertilizer, are also considered outside the system boundaries. The only way for BPA to be stored as a "stock" in the MFA is by landfilling.

The MFA was conducted with annual emissions as the functional unit by using Microsoft Excel 2013 and STAN (subSTance flow ANalysis) Version 2.5.1072 (Vienna University of Technology, http://www.stan2web.net/).

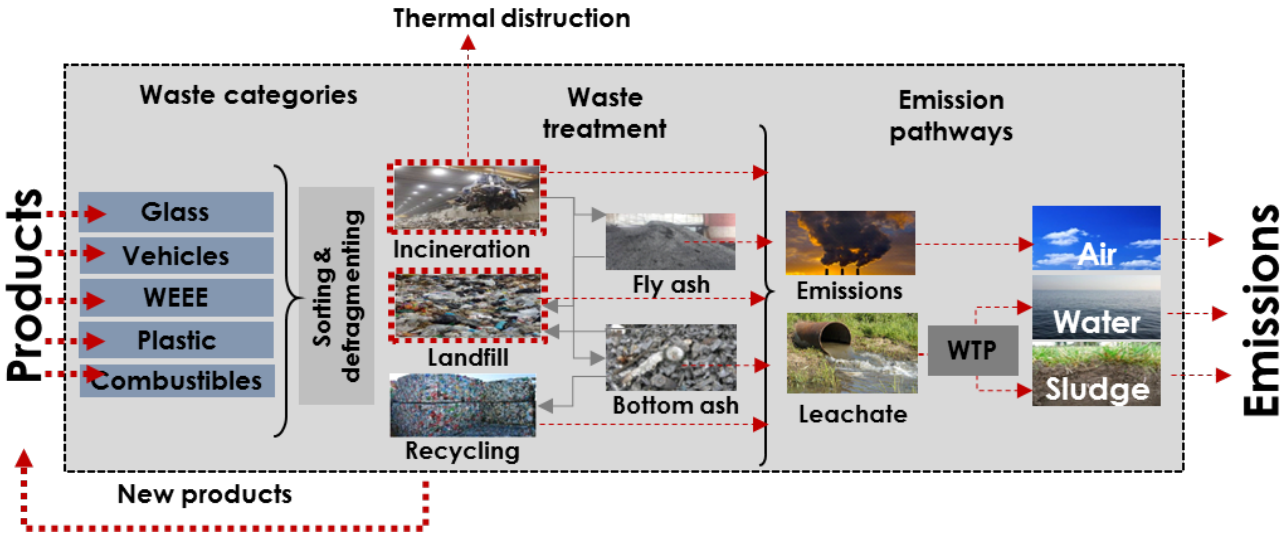


Figure 1. System boundaries for the material flow analysis (MFA). The mass of waste and its bisphenol A are distributed through the various processes from left-to-right. Dotted lines represent removal pathways from the system

2.2 Waste categories

The five waste categories considered in this study and their subfractions are defined in Table 1.. The quantities presented in this table are the total Norwegian mass flow of all waste sources (household, manufacturing industries, construction, etc.) as defined by Statistics Norway (www.ssb.no). These waste categories were selected based on data availability of both waste statistics and BPA concentrations. In addition, targets to recycle more of these types of waste have been set where for instance by 2030 the EU has a target of recycling 65% of municipal waste and 75% of packaging waste (EC, 2015a, c). One important BPA-containing waste stream that was not included here was paper (and cardboard) waste sorted for recycling as we did not have access to paper recycling facilities. The concentrations and mass flows of BPA in paper recycling in

Denmark was recently the focus of series of papers by Pivnenko et al. (Pivnenko et al., 2015a; Pivnenko et al., 2016a; Pivnenko et al., 2016b; Pivnenko et al., 2015b). Therefore, the results of Pivnenko et al. will be discussed alongside ours, in order to include this important waste stream..

The waste categories we selected also contain subfractions that, for the purposes of the MFA, are categorized as being either unique or composite. A unique waste fraction is one in which both mass flow statistics and a BPA concentrations (e.g. recycled glass, composite glass, cable plastic, etc.) could be obtained. A composite waste fraction is one that is of general interest, enough to warrant inclusion in the MFA, but a fraction that consists of different unique fractions. For instance, in order to carry out emissions estimates for the composite fraction "vehicle waste", the sum of its unique waste fractions: "coarse fluff", "fine fluff", "lead batteries" and "other (metal)" was taken. In this manner, scenarios can be implemented in the MFA on both unique and composite waste fractions, such as "recycle all coarse vehicle fluff" or "recycle all vehicle waste", respectively, in order to estimate the impact on environmental emissions.

Table 1. Approximate mass of diverse waste fractions generated in Norway for the year 2011 ($\dot{m}_{waste}$) along with the relative mass percentages recycled $f_{recycled}$, landfilled $f_{landfilled}$, incinerated, $f_{incinerated}$, and incombustible after incineration, f_{incom}.

Group	Fraction	Fraction ID and Definition	Type[a)]	$\dot{m}_{waste}$ (kilotons/year)	$f_{recycled}$ (%)	$f_{landfilled}$ (%)	$f_{incinerated}$ (%)	f_{incom} [b)] (%)	Notes
Glass	Recycled glass	(1) Glass that is sorted for recycling	unique	100 ± 11	100.0	0.0	0.0	100	c
	Composite glass	(2) All other glass	unique	114 ± 13	0.0	7.2	92.8	100	c
	All glass	(3) = (1) + (2)	comp.	214 ± 17	46.7	3.9	49.4	100	
Vehicle	Coarse Fluff	(4) Non-metallic automobile shredder residue (ASR) > 8 mm	unique	23 ± 3	0.0	0.0	100.0	20	d,e
	Fine Fluff	(5) ASR between 2 – 8 mm	unique	9 ± 1	10.0	90.0	0.0		d,e
	Lead batteries	(6)	unique	13 ± 1	100.0	0.0	0.0		f
	Other (metal)	(7)Other materials (metal, glass, etc.)	unique	94 ± 11	100.0	0.0	0.0		d
	All vehicle	(8)= (4) +(5) + (6) + (7)	comp.	140 ± 11	77.5	5.7	16.8	3.2	
WEEE	Remaining plastic	(9) WEEE plastic that is not cable plastic or BFR plastic	unique	48 ± 5	85.5	6.9	7.6	0.5	g, h
	BFR plastic	(10) WEEE plastic separated for containing BFRs	unique	2 ± 0	0.0	0.0	100.0	0.5	g, i
	Cable Plastic	(11) Plastic stripped from cables	unique	15 ± 2	91.5	1.6	6.8	0.5	j
	Remains / Metal	(12) Other materials (metal, glass, etc.)	unique	80 ± 9	84.2	5.5	10.3	20	k
	All WEEE	(13) = (9) + (10) + (11) + (12)	comp.	145 ± 11	84.2	5.5	10.3	11	k
Plastic	Packaging plastic	(14) Plastic separated for potential recycling	unique	194 ± 22	60.1	1.3	38.6	0.5	g,
	Composite, other	(15) Plastic in non-sorted wastes, excluding WEEE&vehicles	unique.	207 ± 23	0.0	8.1	91.9	0.5	g, l,
	Composite, WEEE&vehicle	(16) = (9) + (10) + (11) + 0.6(4) + 0.3(5)	comp.	82 ± 9	67.2	7.3	25.5	0.5	g, l
	All plastic	(17) = (14) + (15) +(16)	comp.	482 ± 33	35.6	5.2	59.2	0.5	c
Comb.	All combustibles	(18) All waste combusted in Norway	comp.	1326 ± 150	0.0	0.0	100.0	16	c,m
	Considered combustibles	(19) = 0.494(3) + 0.168(8) + 0.103(13) + 0.386(14)+0.919(15)	comp.	409 ± 46	0.0	0.0	100.0	28	c,m
	Remaining combustibles	(20) = (18) – (17)	unique	917 ± 156	0.0	0.0	100.0	11	c,m

WEEE = Waste of electrical or electric equipment; BFR = brominated flame retardants; a) unique indicates a unique waste fraction, composite indicates if it is a combination of diverse waste fractions from this table; b) fraction of incinerated waste that becomes ashes (i.e. % ash = $f_{incinerated}$ x f_{incom}), see the SI Table S1, c) Data from Statistics Norway (Statistisk sentralbyrå) for 2011 accessed December 2014 from http://www.ssb.no/; d) Mass estimates from Autoretur AS (http://www.autoretur.no/gjenvinning-av-metaller-gir-stor-energi-og-miljogevinst/, accessed January 2015); e) Fraction estimates based on personal communication with the company Norsk Gjenvinning June 2014, note that this does not include metals in the fluff that are recycled post incineration; f) (Agency, 2012a) data for 2009; g) (Agency, 2012b)); h) Based on a 2010 estimate of plastic in WEEE minus the reported mass flow for cable plastic and BFR plastic; i) BFR plastics are not allowed to be recycled under RoHS; j) 60% of cable plastic is handled in Norway for sorting, 40% sent abroad, this study assumes 100% of cable plastic is handled in Norway; k) Data from (EE-registeret, 2012); l) percentages are based on mass balance calculations, assuming 60% of coarse fluff is plastic and 30% of fine fluff is plastic, based on fluff typically containing 40% plastic and fine fluff containing more wood and stone grains; m) many of the considered combustible fractions (E.g. WEEE, vehicle combustibles) are incinerated abroad, as Norway incinerates primarily municipal waste, though in this study we are assuming they are incinerated in Norway (in reality the "considered combustibles" is lower and "remaining combustibles" is higher, relative to "All combustibles").

2.3 Waste treatment

Table 1 shows estimates of the total yearly production of waste, $\dot{m}_{waste}$ (kilotons/y), for the different waste fractions alongside relative mass percentages for waste that is recycled ($f_{recycled}$), landfilled ($f_{landfilled}$) and incinerated ($f_{incinerated}$). These data are based on national statistics, industry reports as well as personal communication with industry representatives (see the footnotes to Table 1). The majority of data is for the year 2011, or as close to 2011 as possible, as this was the most recent year that complete national statistics were available.

Regarding ashes produced by waste incineration, one study estimated the total wet mass of bottom and fly ash to be 20 – 40 % and 3 – 8 % of the original waste mass, respectively (Sabbas et al., 2003). According to Norwegian Statistics (www.ssb.no/en/avfhand/) in 2011 the mass of "incineration residues" (bottom and fly ash) was approximately 16% of the total mass incinerated in Norway, which is on the low end of the literature estimate. This is possibly due to the the exportation of less-calorific combustible fractions to Sweden, where the demand is higher (Naturvårdsverket, 2012). Based on these considerations, the fraction of waste sent to incineration that is incombustible, f_{incom}, was derived (Table 1) and was further subdivided to landfilled fly ash, landfilled bottom ash and recycled bottom ash (i.e the metallic fraction of bottom ash); see Section S1 of the Supplementary Material for further details.

2.4 Emission pathways

2.4.1 Dust and suspended solid concentrations

To obtain information related to release pathways of BPA as it is being handled throughout the waste stream, a combination of field measurements and existing data were utilized. In a previous study, BPA was analyzed in air, leachate water and waste from 12 waste-handling facilities located in southeastern Norway from mid-2013 to mid-2014 (Morin et al., 2015).

Concentration data used from this study are presented in Table 2. Three of the sampled facilities were landfills, each containing various quantities of municipal and industrial waste, digested WTP sludge for composting (i.e. digestate, which is landfilled temporarily before being deployed in agriculture), bottom ash and fly ash. Seven of the sampled locations were sorting and defragmenting facilities for WEEE and/or vehicle waste. The remaining two sampled facilities were for sorting and incinerating combustible waste (one municipal and one industrial). With the exception of one combustible waste facility and one WEEE/vehicle facility, all the facilities were in rural environments and represent the likely key source of BPA in the area. For the other sites, sampling was done very close to the sources (e.g. air sampling near the shredder or vents), to minimize the influence from other sources of BPA. This previous study also discussed partitioning mechanisms of BPA from waste to air and water, showing that BPA primarily is in the particle phase in air and the dissolved phase in water (Morin et al., 2015). Additionally, concentrations of BPA in 46 landfill leachates, 34 WTP effluents, and 34 WTP sludge samples were taken from a report compiling data from 2002 – 2012 in Norway (Section S2 in the Supplementary Material) (Arp, 2013).

2.4.2 Emission Factors

Emission factors (EF) were used to estimate particulate and BPA emissions from waste-handling processes to the atmosphere and leachate. The EF for dust to the atmosphere, EF_{dust}, was defined as:

$$EF_{\text{dust}}\ (\text{tons}_{\text{dust}}/\text{tons}_{\text{waste}}) = \dot{m}_{\text{dust}}\ /\dot{m}_{\text{waste}} \quad (1)$$

where $\dot{m}_{dust}$ is the mass flow of atmospheric dust emissions ($tons_{dust}$ /y) originating from waste and no other sources (e.g. diesel dust, brake dust, ambient dust) that is transported away from a facility. Atmospheric emissions of BPA occur mainly via BPA-bound to dust (Morin et al., 2015), and thus emission factors of BPA emitted into air per mass of waste at a specific type of facility, $EF_{air,BPA}$ (kg_{BPA}/kg_{waste}), were derived as follows:

$$EF_{air,BPA} = \dot{m}_{dust}\ C_{dust,BPA}\ /\ (1000\ kg_{waste}/tons_{waste}\ x\ \dot{m}_{waste}\ ) \quad (2)$$

where $C_{dust,BPA}$ is the BPA concentration measured in the dust particles (kg_{BPA}/kg_{dust}) produced by a waste treatment process.

Similarly, an EF for suspended solid in leachate, EF_{ss}, can be defined:

$$EF_{ss}(tons_{ss}/tons_{waste}) = \dot{m}_{ss}\ /\dot{m}_{waste} \quad (3)$$

where $\dot{m}_{ss}$ is the mass flow of suspended solids in leachate ($tons_{ss}$ /y). For this analysis, EF_{ss} is predominantly used to assess the amount of suspended solids that end up in WTP from waste-handling. Suspended solids in leachate only contain a negligible amount of BPA compared to the amount dissolved in water, except at extreme particulate concentrations (Morin et al., 2015). The facility specific *EF* of BPA in leachate per mass of waste, $EF_{leachate,BPA}$ (kg_{BPA}/kg_{waste}), were derived based on the flow rates of leachate $Q_{leachate}$ (L/y)

$$EF_{leachate,BPA} = Q_{leachate}\ C_{BPA\_Leachate}\ /\ (1000\ \dot{m}_{waste}\ ) \quad (4)$$

where $C_{BPA\_Leachate}$ (kg_{BPA}/L) is the total leachate concentration.

Data to derive $\dot{m}_{dust}$ and $\dot{m}_{ss}$ were taken from a compilation of dust and leachate measurements at sampled facilities (Morin et al., 2015), complemented by data from other landfills and annual reports from Norwegian waste-to-energy plants, as presented in Section S3 (Supplementary Material). $EF_{air,BPA}$ and $EF_{leachate,BPA}$ for the whole of Norway were derived by using the average of available $C_{dust,BPA,}$ and $C_{BPA\_Leachate}$ for a specific type of waste or waste-handling facility for which data were available in combination with estimated national values for $Q_{leachate}$ and $\dot{m}_{waste}$ (see Section S3).

2.4.3 Mass flow modelling

The mass flow of BPA in waste entering the waste stream for sorting and defragmenting (SD), $\dot{m}_{SD\_BPA}$ (kg_{BPA}/y) was determined by multiplying the relevant $\dot{m}_{waste}$ and the corresponding waste BPA concentration $C_{waste,BPA}$ (kg_{BPA}/kg_{waste}) (Table 2):

$$\dot{m}_{SD\_BPA} = \dot{m}_{waste}\ C_{waste,BPA} \quad (5)$$

A similar equation was used to account for bottom ash, $\dot{m}_{BA}$, and fly ash, $\dot{m}_{FA}$, being transferred from the incinerator to landfill (e.g. $\dot{m}_{landfilled\ ash,,BPA} = \dot{m}_{BA}C_{BA,BPA}$, where $C_{BA,BPA}$ is the concentration of BPA in bottom ash). To calculate BPA in waste being incinerated, $\dot{m}_{I\_BPA}$, landfilled $\dot{m}_{L\_BPA}$ and recycled $\dot{m}_{RE\_BPA}$, which occurs after sorting and defragmenting (Figure 2), an equation that accounts for losses to the environment during sorting and defragmenting was used as follows, using incineration as an example:

$$\dot{m}_{I\_BPA}\ (kg_{BPA}/y) = (\dot{m}_{SD\_BPA} - EF_{SD\_\ BPA,dust} - EF_{SD\_BPA,leachate})\ f_{incinerated} \quad (6)$$

where $EF_{SD\_BPA,dust}$ and $EF_{SD\_BPA,leachate}$ (kg_{BPA}/kg_{waste}) are emissions factors for air and leachate, respectively, from the initial sorting and defragmenting. In this manner, emission factors from waste that is incinerated ($EF_{I\_BPA,dust}$ and $EF_{I\_,BPA,leachate}$), landfilled ($EF_{L\_BPA,dust}$ and $EF_{L,BPA,leachate}$) and recycled ($EF_{RE\_BPA,dust}$ and $E_{RE,BPA,leachate}$) were estimated for the unique waste categories.

To derive total BPA mass flows for "composite waste" and "all waste" categories (Table 1), $\dot{m}_{BPA}$ values (including $\dot{m}_{SD\_BPA}$, $\dot{m}_{I\_BPA}$, etc.) and EF_{BPA} values ($EF_{I\_BPA,dust}$, $E_{RE,BPA,leachate}$ etc.) were added based on the individual waste fractions they contained. For instance, for the waste category "considered combustibles" the $\dot{m}_{BPA}$ values for glass, vehicle fluff, WEEE and plastic sorted for incineration were added together.

To derive the amount of BPA that was emitted as water or sludge from WTP, $EF_{WTP\_BPA,water}$, and $EF_{WTP\_BPA,sludge}$, the ratio of the average concentration of BPA in Norwegian WTP effluent 0.923 µg/L and sludge 0.536 µg/g (Arp, 2013) (see also Section S3, supplementary material), was used to derive a log (C_{sludge}/C_{water}) ratio of 2.5.

2.5 Assumptions in the mass flow analysis

Certain simplifying assumption for the MFA were made. The first assumption is that all sorting, landfilling, incineration and initial recycling is handled in Norway, despite substantial quantities of Norwegian waste being shipped abroad for handling. As examples, for Norwegian WEEE 29% is handled in the EU and 16% outside the EU (EE-registeret, 2012), and for combustibles large amounts are exported to Sweden for incineration (Naturvårdsverket, 2012).

Regarding recycling, only "initial recyclable processing" is considered in the MFA, as in Norway we could find very few facilities that utilize recycled materials from the selected waste fractions for manufacture into new products. "Initial recyclable processing" herein refers to the grinding and reclamation of recyclable waste, such that it becomes a raw material (e.g. plastic chips or flakes, crushed glass, metal scrap) ready for smelting and remanufacture into new products, abroad. As a conservative assumption, emissions from "initial recyclable processing" were assumed to be twice that measured in Norwegian WEEE/Vehicle sorting and defragmenting facilities, as these facilities generally performed initial sorting and grinding. It is assumed that no processes that specifically add or remove BPA occurs during "initial recyclable processing".

The second group of assumptions relate to water leachate. Firstly, it is assumed leachate from "Defragmentation and Sorting", "Incineration" and "Initial Recyclable Processing" facilities are transported by a sewage system to a municipal WTP, and not emitted directly into the environment. This assumption was not applied to "Landfills", as it is documented that the leachate from 45% of Norwegian landfills directly enters the environment after on site treatment, and the remaining 55% is transported to a municipal WTP (Okkenhaug and Arp, 2012). To be conservative, it was assumed that no losses of BPA occurred during transport to the WTP or within the WTP, despite BPA having a half-life in sludge of 2.5 to 4 days (Staples et al., 1998) and that certain WTP processes can remove BPA (Fuerhacker, 2003); thereby such degradation was assumed to occur after sludge was removed from the WTP (outside the MFA boundaries).

The third set of assumptions is related to stocks and losses during the waste-handling processes. Waste or BPA mass that is sent to incineration and is converted by thermal destruction to gases (e.g. CO_2) is mass that is lost from the system; whereas, mass isolated in landfills is accumulated as a stock without further decomposition (BPA is considered stable in landfills under

anaerobic conditions (Limam et al., 2013)).The stock of BPA during defragmentation and sorting, as well as initial recyclable processing, is set to zero (thus no degradation or stocking is considered to occur). In the case of paper recycling, it is noted that BPA could be potentially removed or even added during the recycling process, depending on the process and the desired recycled product (Pivnenko et al., 2016a); however similar information for the waste fractions considered here could not be found.

The fourth assumption is related to using input from or close to the year 2011 as being representative, which in the case of landfills does not account for emissions originating from stocked, landfilled waste from previous years, when landfilling was more prevalent than in 2011. This will bias the presented EF_{landfill} leachate values for 2011 waste to be higher than the actual value.

2.6 Uncertainty Assessment in the mass flow analysis

MFA are inherently uncertain. In addition to requiring several assumptions, they are based on measured, estimated or extrapolated input parameters, such as mass estimates or concentrations, each with their own uncertainties. This study adopted the uncertainty approach described in Laner et al. (Laner et al., 2015). In this approach, data quality is evaluated in terms of its reliability, completeness, temporal correlation, geographical correlation and other correlations; and based on this evaluation, a coefficient of variation (CV) is assigned. CV values for input parameters are then propagated through the mass-flow model. The CV values for the input parameters of this MFA, and more information on the uncertainty analysis, are described in Section S6 (supplementary material).

2.7 Regional exposure outside the mass flow analysis

The MFA primarily predicts local (immediate) emissions, and does not account for regional impacts once BPA enters the environment. Therefore, measured BPA concentrations in the general Norwegian population and environment that would arise from the estimated waste emissions were predicted using CoZMoMAN (Breivik et al., 2010), a dynamic and non-spatially resolved linked model which simultaneously predicts the behaviour of organic contaminants in both the physical environment (CoZMo-POP2 (Wania et al., 2006)) and in the human food chain (ACC-Human (Czub and McLachlan, 2004)). In brief, CoZMoMAN requires three different types of model input parameters: (*i*) data describing environmental and food-chain characteristics, along with data representing the simulated chemical, (*ii*) chemical property data, and (*iii*) emission data. The number of input data required to describe environmental and food-chain characteristics (i) far exceeds the chemical property and emission input data specific for this study (ii and iii). This model was selected as it has previously been parameterized for the Nordic region (mainly Sweden, Norway and Denmark), reflecting relevant food-chains, environmental and climatic conditions; it has also been evaluated with respect to its capability to reproduce observations of both polychlorinated biphenyls (PCBs) (e.g. Breivik et al., 2010) and short-chain chlorinated paraffins (SCCPs) (Krogseth et al., 2013). These evaluations have shown agreement between modelled data and measurements within a factor of 2 to 4 in the case of individual PCBs and within a factor 6 for SCCPs. As the CoZMoMAN model is parameterized to cover Sweden and Denmark in addition to Norway, a similar per capita emission rate in the other countries as for Norway was assumed. This was done by accounting for the population of each country living within the domain, using an identical mode of emissions released to air, fresh water and agricultural soil (via the deposition of WTP sludge as fertilizer, while assuming no BPA loss as a conservative assumption).

There is therefore the following implicit assumptions in the regional exposure model. The first is that the current parameterisation for the Nordic region represents a reasonable approximation for Norway alone. The second is that there was no net exchange of BPA across the CoZMoMAN model boundaries (i.e. the model domain was surrounded by an equally contaminated region). The third is that the MFA output represents Norwegian emissions from waste-handling, including the WTP sludge value is representative for the amount of BPA applied to soils.

The property data compiled and used to simulate BPA concentrations using CoZMoMAN are summarized in Section S7 (supplementary material).

2.8 Management scenarios

After establishing the MFA for the current situation of Norwegian waste handling, the MFA was manipulated to assess the potential impact of alternative waste management scenarios on emissions. Following the waste hierarchy as outlined by European Commission directive 2008/98/EC (waste framework directive), the preferable management strategies for waste management in decreasing order are: 1) prevention, 2) preparing for re-use, recycling, 3) other recovery (including energy recovery by incineration), and 4) disposal (i.e. landfilling). With this in mind, four different alternative scenarios for BPA mitigation were considered.

Scenario 1 Prevention. This scenario models the outcome of Norwegians producing less BPA-containing waste; here by an ambitious target of 50%. This was modelled in the MFA indirectly by multiplying all measured BPA concentrations by 0.5, such that the mass flow entering the waste stream would be reduced by 50%. In addition to waste prevention, this scenario also reflects the case that the concentration of BPA in all waste materials was halved homogenously in

concentration across all considered waste fractions, yet the mass flow of waste was not changed, and thus reflects a potential (and extreme) decrease in demand for BPA (such as by a phase-out, or by replacing to another substance).

Scenario 2 Recycling. In this scenario vehicle fluff, WEEE and plastic that was incinerated in the original MFA is sent for initial recyclable processing instead, and no change is made to the amount directly landfilled (though the amount of incineration ashes landfilled is changed correspondingly). This is a relatively extreme shift in waste-management practices, considering that current targets for the transition towards a circular economy only target municipal and packaging waste (EC, 2015b, c). This was modelled by setting all $f_{\text{incinerated}}$ to 0 in Table 1 for WEEE, vehicles and plastics, and adjusting f_{recycled} accordingly.

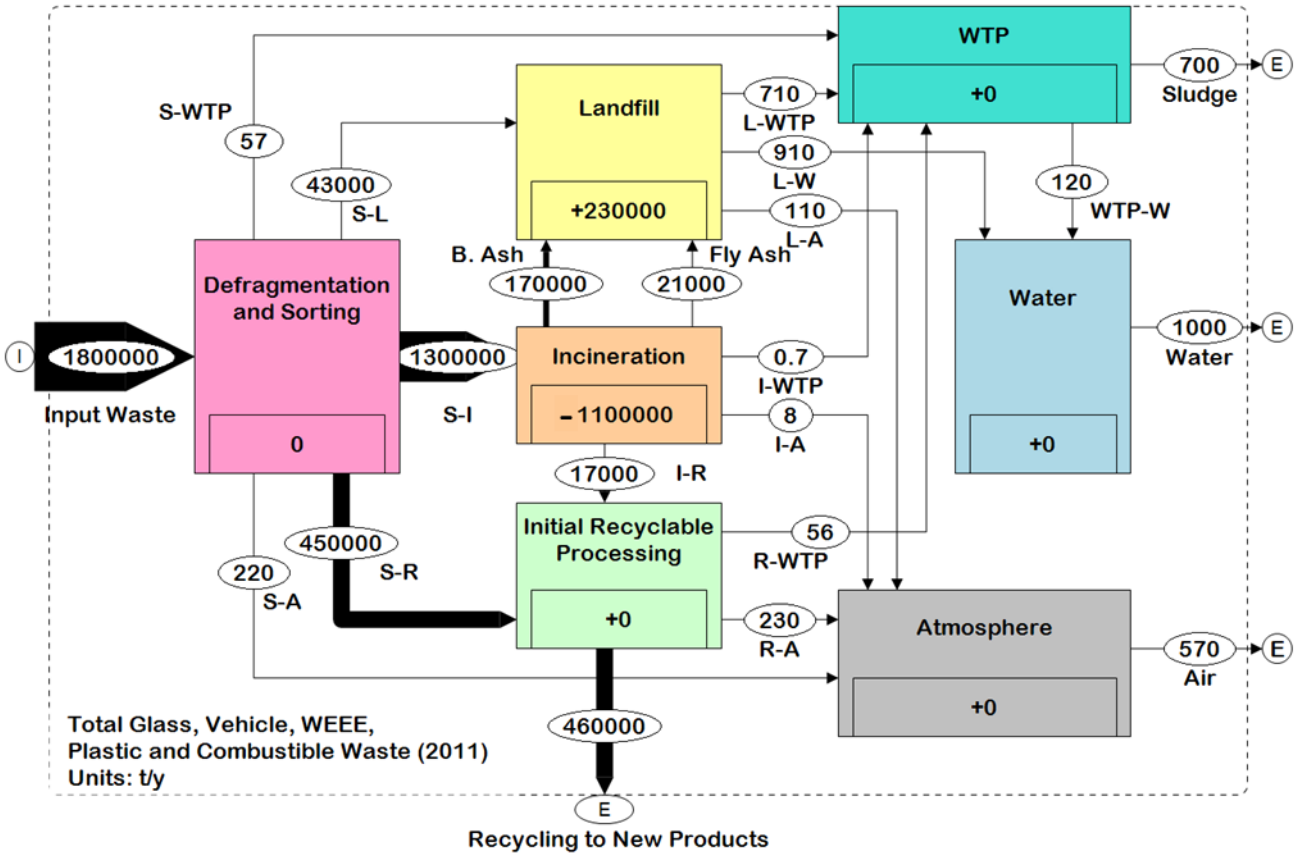
Scenario 3 Incineration. In this scenario, all non-ash waste that is currently landfilled directly is incinerated before landfilling as ashes; regardless of calorific value. This scenario was modelled by setting $f_{\text{landfiilled}}$ values in Table 1 to 0, and correspondingly increasing $f_{\text{incinerated}}$ values. No change was made to f_{recycled}.

Scenario 4 Landfilling of vehicle fluff. Finally, to consider a scenario that goes contra to the waste-hierarchy, in this scenario all vehicle fluff is landfilled instead of incinerated. This scenario was chosen as in Scandinavia coarse fluff is incinerated and fine fluff is landfilled (Table 1), but in many non-Scandinavian countries, vehicle fluff is commonly landfilled (Gerrard and Kandlikar, 2007; Santini et al., 2012). This was modeled setting the $f_{\text{landfiilled}}$ values in Table 1 to 100% for coarse and fine fluff, and f_{recycled} and $f_{\text{incinerated}}$ to 0.

3 Results and discussion

3.1 Mass flow of waste

Figure 2 presents the mass flow of the sum of glass waste, vehicle waste, WEEE, plastic wastes and combustibles going through the Norwegian waste stream, as well as emissions into air, water and sewage sludge. This MFA estimates that of the nearly 2 million tons/y of these wastes that are generated in Norway, 567 tons/y are emitted as dust, 1038 tons/y as suspended solids in water bodies, and 701 tons/y as WTP sludge sorted for biogas and fertilizer production. Approximately 235 ktons/y of the selected waste are stored in landfills (as stock) and these consist primarily of ashes, whereas 1116 kt/y are lost/destroyed via incineration. A further description of Figure 2 is presented in Section S4 (Supplementary Material), along with an alternative version of Figure 2 presenting uncertainties (which range from 11.3 to 60.1%).



412

413 **Figure 2. Principle presentation of connections between the processes in the MFA of the sum of glass**
414 **waste, combustibles, vehicle waste, WEEE and plastic wastes going through the Norwegian waste**
415 **stream (collectively referred to as "Input Waste") as well as emissions into air, water and WTP sludge**
416 **for the year 2011. All quantities have the units tons/y. Processes are given in boxes and flows as arrows**
417 **connecting the processes. Symbols used in the chart include S (sorting and defragmentation), L**
418 **(landfill), I (Incineration), R (Initial Recyclable Processing), WTP (water treatment plant), W**
419 **(environmental water recipients), A (atmosphere), E (Recycling to New Products or Emissions).**
420 **Numbers inside the processes Landfilling and Incineration indicate the mass that was lost or**
421 **accumulated from waste stream due to thermal destruction (negative values) or long term isolation**
422 **(positive values).**

423

424 ### 3.2 BPA in the waste categories

425 Table 2 presents the mass flow of BPA for the sum of the five selected waste categories.

426 The largest mass flow is associated with the plastic waste, being responsible for 90631 kg/year, or

427 98.5% of the total BPA in waste (91984 kg/y). Plastics can vary widely in their BPA content,

428 where BPA is found abundantly in polycarbonate and epoxy (e.g. glue and coatings). One of the

most common types of plastic, PVC (polyvinylchloride), is also a source of BPA, including when it is used for food packaging (Lopez-Cervantes and Paseiro-Losada, 2003). Representatives from the Norwegian waste industry informed us that alternatives to incinerate PVC are often sought after, as the produced chloride gas during PVC incineration is corrosive in waste incinerators (Sadat-Shojai and Bakhshandeh, 2011). Of the BPA associated with plastic, 60% was estimated to be incinerated, 5% landfilled (e.g. fine vehicle fluff, some WEEE fractions) and 35% recycled.

Table 2. Concentrations and mass flows of BPA in different waste categories and fractions, undergoing different waste-handling processes

Fraction	$C_{waste,BPA}$ [a]	Type	f_{BPA_waste} [b]	$\dot{m}_{SD\_BPA}$	$\dot{m}_{I\_BPA}$	$\dot{m}_{LA\_BPA}$	$\dot{m}_{RE\_BPA}$
				Sorting&. Defragmentation	Incineration	Landfilling	Recycling
	$\mu g_{BPA}/kg_{waste}$		(%)	(kg_{BPA}/y)	(kg_{BPA}/y)	(kg_{BPA}/y)	(kg_{BPA}/y)
Glass							
Recycled glass	11.0 ± 7.1	unique	0.0%	1.1 ± 0.7			
Composite glass	9.0 ± 2.8	unique	0.0%	1.0 ± 0.3	0.9 ± 0.3	0.1 ± 0.0	0.0 ± 0.0
All glass	*9.9 ± 5.5*	comp.	0.0%		0.9 ± 0.3	0.1 ± 0.0	1.1 ± 0.3
Vehicle [c]							
Coarse Fluff	4818.8 ± 1311.2	unique	0.1%	113.0 ± 33.3	111.6 ± 33.2	0.0 ± 0.0	0.0 ± 0.0
Fine Fluff	10914.4 ± 6983.7	unique	0.1%	96.9 ± 62.9	0.0 ± 0.0	86.7 ± 25.8	9.6 ± 2.9
Vehicle Fluff	*6492.1 ± 3350.1*	comp.	0.2%	209.9 ± 109.6	111.6 ± 33.2	86.7 ± 25.8	9.6 ± 2.9
WEEE							
Remaining plastic	200499 ± 122465	unique	10.5%	9630 ± 5981	735 ± 219	663 ± 197	8227 ± 2446
BFR plastic	84430 ± 27958	unique	0.2%	169 ± 59	169 ± 50	0 ± 0	0 ± 0
Cable Plastic	29087 ± 19357	unique	0.5%	435 ± 294	30 ± 9	7 ± 2	397 ± 118
Remains / Metal	1169 ± 301	unique	0.1%	94 ± 26	9 ± 3	5 ± 1	73 ± 22
All WEEE	*71188 ± 46656*	comp.	11.2%	10328 ± 6811	942 ± 280	675 ± 201	8697 ± 2586
Plastic							
Packaging & comp [d]	200499 ± 122465	unique	42.3%	80300 ± 49877	53052 ± 15774	3856 ± 1146	23392 ± 6955
Comp. WEEE&vehicle	*126884 ± 79309*	comp.	56.2%	10331 ± 6562	1000 ± 297	696 ± 207	8627 ± 2565
All plastic	*188052 ± 124720*	comp.	98.5%	90631 ± 60433	54052 ± 16071	4552 ± 1353	32020 ± 9520
Combustibles							
Considered combustibles [e]	*132766 ± 155648*	comp.	60.2%	54269 ± 63916	54269 ± 16136	0 ± 0	0 ± 0
Remaining combustibles [e]	1248 ± 373	unique	1.2%	1145 ± 394	1144 ± 340	0 ± 0	0 ± 0
All combustibles	*41790 ± 30738*	comp.	59.0%	55413 ± 41236	55413 ± 16476	0 ± 0	0 ± 0
Ashes							
Fly Ash	< LOQ	unique				neg	
Bottom Ash	235 ± 155	unique				40 ± 12	4.0 ± 1.2
Total							
All Waste				91984 ± 127812	55251 ± 16428	4617 ± 1373	32100 ± 9544

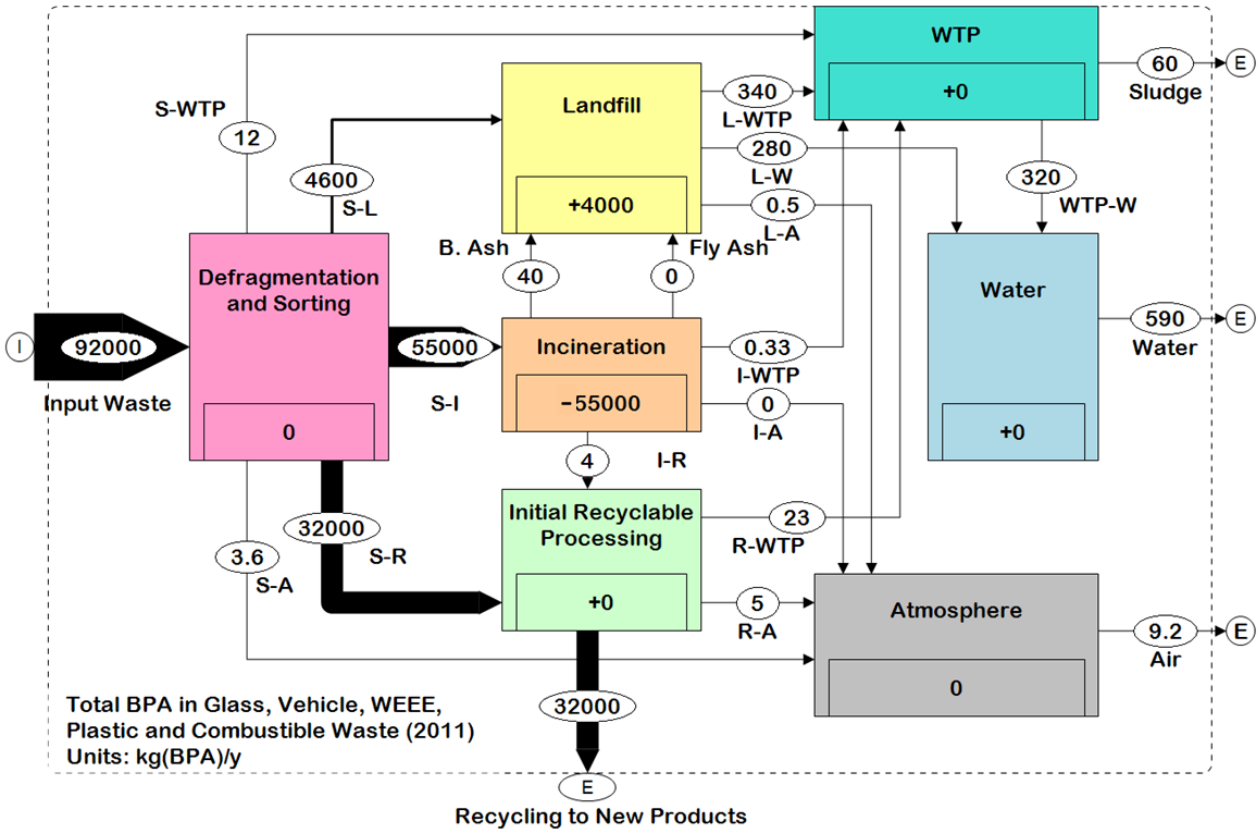
neg = negligible; LOQ = limit of quantification; italics = weighted averages for composite materials based on Table 1; unique =waste that remains as an unmixed fraction after sorting and defragmenting, such as glass-materials or plastic-materials sorted for recycling; comp. = waste mixtures of fractions that are either not separated during waste treatment, or were combined for this study due to not enough data being available to subdivide the fraction further. , a) concentrations presented in a previous study (Morin et al., 2015); b) the relative fraction of BPA in the specific waste fraction vs the total yearly produced BPA in the selected waste fractions entering Norwegian waste streams, c) vehicle metals and lead batteries are not included as they are assumed to have little BPA; d) due to lack of data on Packaging and Composite plastics, we assumed these were both the same as the "WEEE remaining plastic" concentration, which was for pooled "plastic fluff" samples from WEEE/Vehicle facilities, not including isolated BFR and cable plastics; e) considered combustibles referred to those already presented in the table (glass, vehicle fluff and plastic)

The mass flow of BPA in glass, vehicle fluff and "remaining combustibles" is very small compared to plastics. This was expected for glass based on the low concentrations found in glass (11.0 ± 7.1 µg/kg). Vehicle fluff has relatively low BPA concentrations (from 4819 ± 1311 to 10914 ± 6984 µg/kg) as well as $\dot{m}_{waste}$ values (ca. 32 ktons/year) compared to packaging and composite plastics (200499 ± 122465 µg/kg and ca. 482 ktons/year). However, the low BPA mass and concentration (1248 ± 349 µg/kg) and resulting mass flow (ca 1145 kg/y) in the "remaining combustibles" fraction was not initially expected, due to the potential sources of BPA (like plastics and paper waste (Fan et al., 2015)) that were visible within the analyzed samples. One potential explanation for combustibles having a low BPA content is that BPA containing-paper and plastics are removed by pre-sorting procedures prior to recycling. Pivnenko et al. (Pivnenko et al., 2016b) found that a selection of individual waste paper and cardboard materials had a median of 20000 µg/kg BPA (min 700 and max 4 800 000 µg/kg) in residual (non-sorted) waste paper, and 10000 µg/kg BPA (min 530 and max 3 600 000 µg/kg) in source-segregated waste paper, with the maximum concentrations being associated with receipts. This implies that a) the concentration for combustibles in this study is on the low end compared to Danish waste-paper, b) source-segregation would not on its own account for these low concentrations. Therefore, it could be that these measured concentrations and mass flow estimates for BPA in remaining combustibles are biased low, by up to an order of magnitude; though they would have to be biased low by two orders of magnitude in order for the BPA mass flow to be similar to that of plastic..

3.3 BPA in the waste treatment process

Figure 3 presents the MFA of BPA in the selected waste categories through the entire Norwegian waste stream (totaling 91984 kg/y) for the model year 2011. As presented in Figure 3, *incineration* is estimated to remove the majority of BPA entering the waste stream (-55206 kg/y,

or 60%). BPA is known to be thermally unstable during waste incineration (Šala et al., 2010). Considering the difference between the total amount of BPA in waste that was sorted for incineration (-55251 kg/y), and the amount of BPA remaining in the bottom ash (40 kg/y) and fly ash (0 kg/y), we estimate that 99.9% of BPA sorted for incineration was removed by this process. Isolation due to *landfilling* accounted for accumulation of 4040 kg/y as stock in landfills, or 4% of all BPA entering the waste stream. *Initial recyclable processing* is responsible for having a potential of transferring up to 32076 kg/y, or 34% of the BPA into recyclables to be used for the manufacture of new products. Depending on further processing of the recycled material, the amount of BPA could be increased or decreased, which was not accounted for here. Some process such as deinking paper waste are known to be able to decrease BPA concentrations (Pivnenko et al., 2016a; Pivnenko et al., 2016b).



484

485 **Figure 3. Mass flow of BPA associated with glass waste, combustibles, vehicle waste, WEEE, and plastic wastes**
486 **(collectively referred to as input waste) going through landfilling, incineration and initial recyclable processing**
487 **in Norway. Symbols used in the chart are presented in Figure 2. Values along the arrows indicate the mass flow**
488 **of waste and waste particles from one process to the next (see Table 2). All values are presented in the units**
489 **kg/y. Numbers inside the processes Landfilling and Incineration indicate the mass that was lost or accumulated**
490 **from waste stream due to thermal destruction (negative values) or long term isolation (positive values).**
491

3.4 BPA emission pathways

As presented in Figure 3 and Section S5, the MFA estimates that 9 kg_{BPA}/y are emitted from Norwegian waste handling as dust to the air phase, 593 kg_{BPA}/y are released into downstream water bodies, and 60 kg_{BPA}/y end up in WTP sludge. The majority of water emissions are from landfills. (340 kg/y to WTP and 278 kg/y to receiving waters). Therefore, considering most BPA sent to landfills was associated with plastic (4552 kg of the total 4617 kg_{BPA}/y sent to landfills, Table 2), the primary source of BPA emissions from the selected waste categories are plastic-residues sent to landfills; though BPA-rich paper products (e.g. thermopaper) are another potential source. Recall that the mass flow model inherently does not account for any historical reservoir in landfills. Thus, the estimation herein of 134 g emission per kg landfilled BPA in 2011 (as derived from values in Table 5) represents an upper limit, as the BPA emissions were from the aggregate of several previous years of landfilling waste. Nevertheless it is clear from Figure 3 that landfilling, by far, represents the waste handling process resulting in the largest emissions of BPA.

Figure 3 also estimates that a total of 364 kg_{BPA}/y enters WTPs (339 kg/y from landfilling and 23 kg/y from initial recyclable processing). This value is slightly below the amount estimated to enter all Norwegian WTPs of 408 kg_{BPA}/y (with 342 kg_{BPA}/y leaving in water and 64 kg_{BPA}/y in sludge, Section S2). Taken together, this implies that the majority of BPA entering WTPs in Norway may originate from landfill deposited plastic and paper waste. However, it should be kept in mind that BPA can decompose in waste water, and this was not taken into account, therefore the contribution of plastic in landfills presented here would represent an upper value.

As a basis of comparison for WTP emissions, a recent study (Yu et al., 2015) estimated that country-wide WTP emissions in the USA for the year 2006/7 were 31800 kg_{BPA}/y in the water phase and 2900 kg_{BPA}/y in the sludge phase. On a per capita bases (298.4 million in the USA, 2006 and 5.0 million in Norway, 2011), this equates to 1065 and 680 mg_{BPA}/capita/y for the water phase

in the USA and Norway, respectively; for the sludge phase this would be 10 and 13 mg_{BPA}/capita/y, respectively. Thus, emission rates on a per capita basis are similar for these two countries (within a factor of 1.6). A study of BPA in emissions from Korean WTPs found that total per capita emissions were substantially lower for domestic WTP (16.1 mg_{BPA}/capita/y) than industrial WTP (885 800 mg_{BPA}/capita/y) (Lee et al., 2015). Taken together, these studies imply that industry and landfills are the two largest sources of BPA entering WTP and the aquatic environment.

There are several uncertainties regarding the estimations for initial recyclable processing in Figure 3 and Table 2, based on the assumptions presented above. Firstly, because recycling of plastic materials does not substantially occur in Norway, actual BPA emissions from initial recyclable processing within Norway would be much less than presented in Figure 3. Further, because recycling emissions can vary depending on the recycling process itself, and they were here based on the assumption that they were twice the measured value that WEEE/Vehicle sorting and defragmenting facilities as a conservative assumption (see Section 2.5 and Section S5 in the supplementary material), recycling emissions are more uncertain than what this MFA presents. In addition, outside of the model domain, BPA may be added or removed from recycled products (Pivnenko et al., 2016a). Resolving this uncertainty is important to study further in follow-up investigations, considering targets to recycle more types of plastic are increasing as part of the shift to the circular economy (EC, 2015b).

3.5 Regional exposure outside the mass flow analysis

The CoZMoMAN model was used to predict the resulting regional exposure of BPA based on Figure 3 emission estimates. The values are compared to observed regional exposure measurements in Norway in Table 3. The sampling sites for the observed concentrations are typically biased towards densely populated areas (e.g. Oslofjord, Lake Mjøsa, which are indeed

influenced by local waste-handling facilities and WTPs) while the model is not spatially resolved and therefore cannot reproduce any gradients in concentrations. Under these circumstances, one would therefore *a priori* expect CoZMoMAN to predict concentrations lower than measurements.

Table 3. Comparison CoZMoMAN model predictions for BPA and observations.

Environmental Media	Model (geomean)	Observed
Marine sediment (ng/g_{dw}), Oslofjord[a)]	2E-05	<0.8-44
Marine sediments (ng/g_{dw}), median, Norway[a)]		6.7
Freshwater sediment (ng/g_{dw}), Lake Mjøsa [a)]	0.01	2-3
Freshwater sediment (ng/g_{dw}), median, Norway [a)]		8.4
Cod liver (ng/g_{ww}) [a)]	4E-05	<0.3-1105
Milk, Norway, 2012 (μg/L fresh weight)[b)]	0.005	<0.02
Hamburgers, Norway, 2012 (μg/kg fresh weight)[b)]	0.01[d)]	0.17
Fish and fish products, Norway, 2012 (μg/kg fresh weight) [b)]	4E-06[e)]	2.0
Urine, pregnant women (mean age 30, mean conc. in μg/L)[c)]	2E-08[f)]	4.5

a) (Thomas et al., 2014), b) (Sakhi et al., 2014), c) (Ye et al., 2009), d) Beef, assuming 15% (w/w) lipid content, e) Cod filet (all age classes), f) 30 year old pregnant women

Indeed, predicted BPA concentrations in humans and food from waste emissions are typically orders of magnitude lower than recent environmental measurements (Table 3). This suggests that the far-field human exposure of the general population to BPA arising from emissions within the waste sector is likely to be insignificant. This is in accordance with past findings suggesting that pathways other than far-field environmental exposures, such as the intake of canned food are more likely to control human exposure of Norwegians to BPA (Sakhi et al., 2014; Ye et al., 2009). However, as monitoring studies have confirmed, such as downstream from landfills (Morin et al., 2015) and urban-influenced environments (Table 3), BPA can be present in local environments at concentrations that are cause for concern regarding ecosystem health. As

558 further indication of this in other counties, residents near e-waste facilities in China had higher
559 levels of BPA in their urine than those in rural areas (Zhang et al., 2016).
560
561
562 **3.7 Management of bisphenol A in the waste stream**
563
564 The way in which the alternative waste-management scenarios (Scenario 1 – Prevention of waste
565 or BPA use, Scenario 2 – recycling all WEEE, plastic and vehicle waste currently incinerated,
566 Scenario 3 – incinerate waste currently landfilled and Scenario 4 – landfill incinerated vehicle
567 fluff) impact changes in sludge, water and air emissions (kg/y) is presented in Figure 4. Percentage
568 changes and corresponding raw data is presented in Section S8 (supplementary material).
569

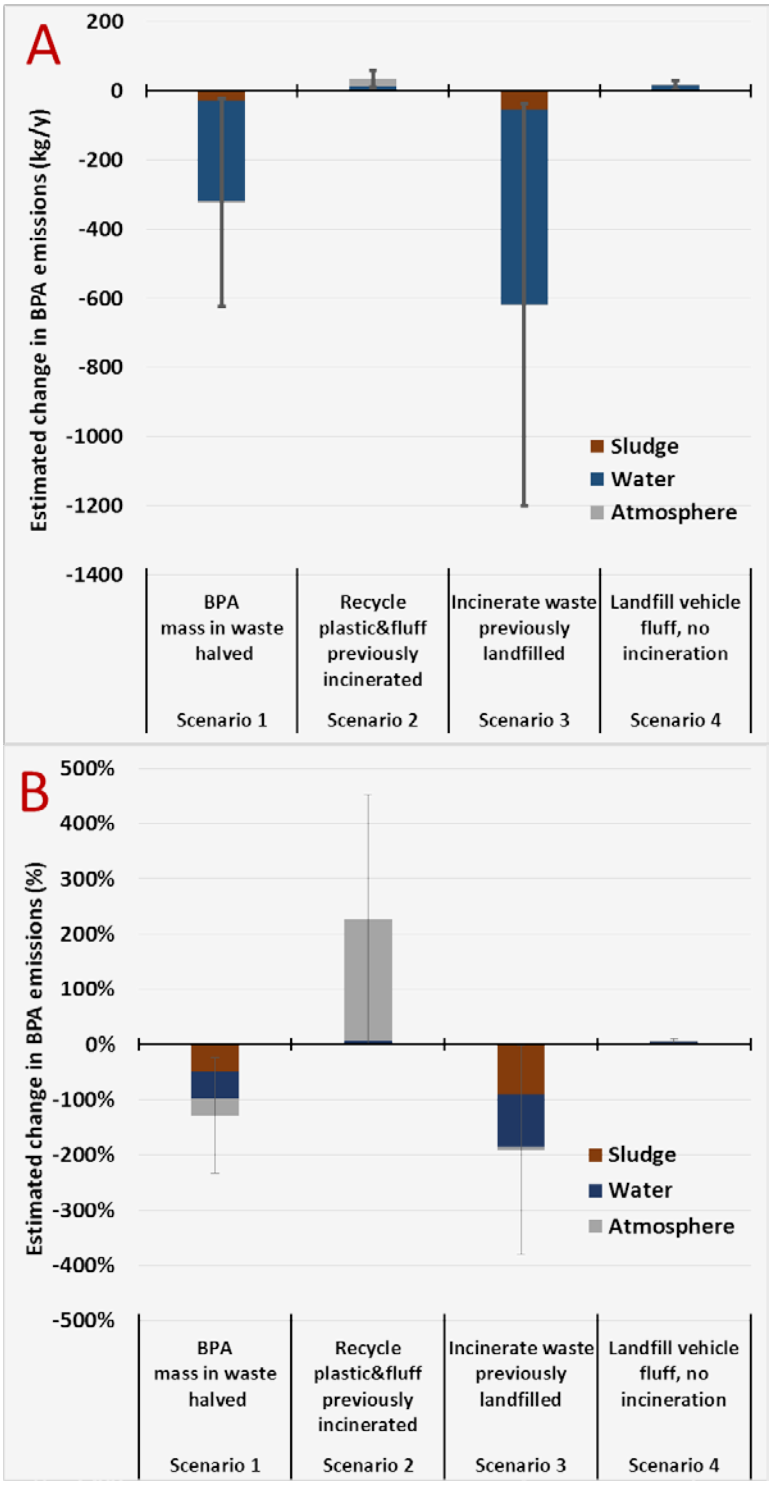


Figure 4. Comparison of BPA emissions resulting from different national waste management scenarios, showing changes in actual emissions (kg/y) in panel A or as percent difference (%) in panel B. Note that a positive change in the emissions would imply more BPA emitted to the environment than the current status quo model predicts, and vice versa.

The most substantial reduction in emissions of BPA presented in Figure 4 comes from scenario 3, representing incineration of all waste that is currently landfilled. This scenario reduced the total emissions of BPA by 618 kg/y (i.e. from 663 to just 45 kg/y). This is mainly due to preventing release of BPA from landfilled plastic waste into leachate. The next most effective scenario to reduce BPA emissions is Scenario 1, whereby reducing the concentrations in all materials by half, or alternatively the produced amount of waste by half, leads to a direct reduction in emissions by half, reducing the total amount of BPA emitted to 339 kg/y (and 321 kg/y just from plastic). Scenarios 2 and Scenario 4 increased the BPA emitted, but only slightly, giving totals of 678 and 695 kg/y compared to the current status quo of 663 kg/y. This is mainly because these scenarios reduced the amount of BPA-containing waste being incinerated, and incineration lowers BPA the most effectively. The emissions for initial recyclable processing, are mainly in the air phase in the form of BPA-containing dust (from any shredding or crushing activities in poorly ventilated areas).

4. Conclusions and Outlook

The results have importance from a regulatory perspective. Following Scenario 1, reducing the amount of BPA entering the waste stream by reducing the amount of BPA in waste, is in theory an efficient method to reduce emissions to air, water, and sludge. In practice, this does not seem to be a realistic outcome in the near future, considering the increased worldwide production of BPA, which is already estimated at over 4.6 million tons (Merchant, 2015), and the huge reservoir of materials already containing BPA. An emerging issue in relation to BPA phase-out is the use of potential BPA substitutes, which are most commonly other bisphenols such as bisphenol S and

599 bisphenol F. Many of these compounds are increasingly found in waste paper (Pivnenko et al.,
600 2015b), e-waste (Zhang et al., 2016) and WTP sludge (Lee et al., 2015; Yu et al., 2015); although
601 their risk to the environment is less well established.

602 The most immediate change in waste management that would result in lower BPA
603 emissions would be incinerating all BPA-containing waste that is currently sent directly to
604 landfills, following Scenario 3. This is further supported by Scenario 4, showing that landfilling
605 coarse car fluff in Norway would lead to higher emissions than the current practice of incineration.
606 Emissions could be further reduced by incinerating, rather, than recycling BPA-rich wastes
607 (Pivnenko et al., 2016a). However, this would go against current initiatives in Europe and Norway
608 to favor recycling over incineration (e.g. for vehicle fluff, Directive 2000/53/EC).

609 Management strategies at the facility level, or at the waste water infrastructure level, are
610 also important. For instance, the installation of air or water treatment systems at waste handling
611 facilities themselves, or ensuring that BPA emitting landfills are connected to a municipal WTPs
612 capable of removing BPA and its substitutes, would directly decrease environmental emissions,
613 particularly in regards to those stemming from landfill leachate.

614 As there is a drive to rapidly develop reuse and recycling schemes as part of the circular
615 economy, future research should focus on a better understanding of how contaminants like BPA
616 will be recycled into new products and emitted during recycling. Pivnenko et al. (Pivnenko et al.,
617 2016a) concluded that the most effective way to eliminate BPA from occurring in recycled paper
618 products is to eliminate BPA, but even if this is done it may take several decades before the
619 presence of BPA can be considered insignificant. Therefore, it may be needed to consider a
620 framework for the inclusion of compounds like BPA in recycled products, such as by specifying
621 acceptable levels of BPA in different types of recycled products, as part of the transition to the

circular economy. On the short-term, the best way to reduce environmental emissions based on the results of this study are to preferably incinerate BPA-rich wastes, such as those containing polycarbonate, epoxies and thermopaper, particularly if they are being landfilled, increase the infrastructure of water treatment systems for landfill leachate, or to produce less BPA containing waste.

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Supplementary Material

Additional data, tables and figures are found in the Supplementary Material.

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