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Home composting as an alternative treatment option for organic household waste in Denmark: an environmental assessment using life cycle assessment-modelling

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ABSTRACT

An environmental assessment of the management of organic household waste (OHW) was performed from a life cycle perspective by means of the waste-life cycle assessment (LCA) model EASEWASTE. The focus was on home composting of OHW in Denmark and six different home composting units (with different input and different mixing frequencies) were modelled. In addition, incineration and landfilling was modelled as alternatives to home composting. The most important processes contributing to the environmental impact of home composting were identified as greenhouse gas (GHG) emissions (load) and the avoided emissions in relation to the substitution of fertiliser and peat when compost was used in hobby gardening (saving). The replacement of fertiliser and peat was also identified as one of the most sensible parameters which could potentially have a significant environmental benefit. Many of the impact categories (especially human toxicity via water (HTw) and soil (HTs)) were affected by the heavy metal contents of the incoming OHW. The concentrations of heavy metals in the compost were below the threshold values for compost used on land and were thus not considered to constitute a problem. The GHG emissions were, on the other hand, dependent on the management of the composting units. The frequently mixed composting units had the highest GHG emissions. The environmental profiles of the home composting scenarios were in the order of -2 to 16 milli person equivalents (mPE) Mg^{-1} wet waste (ww) for the non-toxic categories and -0.9 to 28 mPE Mg^{-1} ww for the toxic categories. Home composting performed as good as or better than incineration and landfilling in several of the potential impact categories. One exception was the global warming (GW) category, in which incineration performed better due to the substitution of heat and electricity based on fossil fuels.

Keywords: Life cycle assessment, home composting, organic household waste.

Abbreviations: AC, acidification; CHP, combined heat and power; CO_2-eq., CO_2-equivalents; EF, emission factor; ETs, eco-toxicity in soil; ETw, eco-toxicity in water; GHG, greenhouse gas; GW, global warming; GWP, global warming potential; HTa, human-toxicity via air; HTs, human-toxicity via soil; HTw; human-toxicity via water, LCA, life cycle assessment; LCI, life cycle inventory; LCIA, life cycle impact assessment; LHV, lower heating value; MFA, mass flow analysis; NE, nutrient enrichment; OHW, organic household waste; POF, photochemical ozone formation; SNCR, selective non-catalytic reduction; SFA, substance flow analysis; UOD, upstream-operation-downstream; WtE, waste-to-energy; ww, wet waste.

1. INTRODUCTION

One alternative, or rather complimentary, technology for the treatment of Organic Household Waste (OHW) is home composting. In home composting, the OHW is taken out of the waste stream at the source, thereby lowering the amount of waste in the municipal waste stream. There are some potential disadvantages of home composting. The most important one is Greenhouse Gases (GHG) emissions during the microbial degradation of the waste, which was assessed and quantified in an experiment by Andersen et al. (2010a). Other environmental contributions (e.g. leachate generation) were assessed by Andersen et al. (2011) in a study that provided a full life cycle inventory (LCI) of home composting. There have, however, only been a few limited attempts at a full environmental assessment of home composting. Colón et al. (2010) performed a life cycle assessment (LCA) of home composting including all upstream processes (environmental loads associated with the production of the composting units and the tools used during composting). Some of the most important factors such as GHG emissions, leachate production and the downstream processes (the substitution of peat in growth media and fertiliser) were poorly investigated or even left out of the assessment. Martínez-Blanco et al. (2010) did a more complete LCA study, comparing home composting of OHW with central composting (tunnel composting) in Spain. In a study by Martínez-Blanco et al. (2010), their assessment of GHG emissions was more complete than the one by Colón et al. (2010) (in which the detection limits on the measuring instruments were too high), but the downstream processes were still not included. Martínez-Blanco et al. (2010) concluded that home composting is an interesting alternative (or rather accompaniment) to central composting, especially in areas with low density population. One of the main differences between other studies on home composting and this study was that the load of waste used as input into the composting unit was much higher in the other mentioned studies. This could change the rate of mineralisation of the organic material, which would result in a different composting process. The amount of input waste in Colón et al. (2010) was 14.3 kg OHW and 3.7 kg of garden waste on average per week and in Martínez-Blanco et al. (2010) the input waste was 8.3 kg OHW and 3.1 kg of garden waste on average per week. This is compared to 2.6-3.5 kg OHW on average per week in the present study. There is a general lack of reliable environmental assessments with consistent data, which is emphasised by the studies by Weidema et al. (2006), and Lundie and Peters (2005). In these studies, the authors modelled home composting (in a LCA context) as an intermediate between aerobic and anaerobic digestion (Weidema et al., 2006) and as two scenarios with aerobic (no methane) and anaerobic home composting (Lundie and Peters, 2005). These options were not considered to be realistic since they represent extremes.

Most of the OHW in Denmark is currently incinerated in waste-to-energy (WtE) plants. This technology has been chosen as the alternative to home composting, since this is where the waste ends up if it is going into the municipal waste stream. Landfills were considered as another alternative technology, since they are still a major treatment route in many countries in Europe (40% of biowaste still ends up in landfills in the European Union) (European Commission, 2010), despite the introduction of the landfill directive in 1999 (CEC, 1999).

The aim of this paper is to perform a full LCA on home composting of OHW in Denmark based on the LCI data for single-family home composting (of six differently managed composting units) that was provided by Andersen et al. (2011). Their environmental performance was compared with that of incineration of OHW in WtE plants, which is the main management route for OHW in Denmark. Finally, landfills were also included in the assessment.

2. METHODOLOGICAL APPROACH

The functional unit of this study was defined as management of 1 Mg of OHW and the time scale of the assessment was 100 years. According to Petersen and Kielland (2003), 1 Mg of OHW is equivalent to the amount of OHW that 20 households compost at home in one year in Denmark (50 kg per household per week). The boundaries of the studied system were expanded to include benefits and burdens from the upstream- and downstream processes linked to waste treatment activities. However, some aspects were not included in the assessment, e.g. the construction and demolition of waste treatment facilities (e.g. composting units, asphalt pads, excavators), solid outputs from the WtE plant and wastewater generation. The reasons for the exclusion of these parameters were non-availability of data or data uncertainty.

The environmental assessment was performed by means of the waste-LCA-model EASEWASTE. The general concept of EASEWASTE, its functioning and its different modules was described by Boldrin et al. (2011a), Hansen et al. (2006), Kirkeby et al. (2006), Manfredi and Christensen (2009) and Riber et al. (2008). Mass flow analysis (MFA) was performed by means of the mass-balance model STAN (short for subSTance flow ANalysis) (Cencic and Rechberger, 2008).

The life cycle impact assessment (LCIA) was based on the EDIP 1997 methodology (Wenzel et al., 1997). The normalised results for the impact potentials were expressed in terms of person equivalents (PEs), and calculated according to normalisation factors defined for Denmark (Stranddorf et al., 2005; EASEWASTE, 2008). All normalisation factors are shown in Table 1. The potential impact categories considered were: global warming (GW), photochemical ozone formation (POF), nutrient enrichment (NE), acidification (AC), eco-toxicity in water (ETw), eco-toxicity in soil (ETs), human toxicity via water (HTw), human toxicity via soil and human toxicity via air (HTa). Emissions of biogenic CO_2 were accounted as neutral in terms of GW during the characterisation phase of the LCA (Christensen et al., 2009).

Table 1. Normalisation factors for the environmental impact categories in EDIP 1997 (Stranddorf et al., 2005).

	Impact category	**Geographical scale**	**Characteri-sation unit**	**Normalisation reference (characterisation unit person^{-1} year^{-1})**
Non-toxic impacts	Global Warming (GW)	Global	kg CO_2-eq.	$8.7\cdot10^3$
	Acidification (AC)	Regional	kg SO_2-eq.	$7.4\cdot10^1$
	Nutrient enrichment (NE)	Regional	kg NO_3-eq.	$1.19\cdot10^2$
	Photochemical ozone formation (POF)	Regional	kg C_2H_4-eq.	$2.5\cdot10^1$
Toxic impacts	Human toxicity via air (HTa)	Local	m^3 air	$6.09\cdot10^{10}$
	Human toxicity via water (HTw)	Regional	m^3 water	$5.22\cdot10^4$
	Human toxicity via soil (HTs)	Regional	m^3 soil	$1.27\cdot10^2$
	Ecotoxicity via water (ETw)	Regional	m^3 water	$3.52\cdot10^5$
	Ecotoxicity via soil (ETs)	Regional	m^3 soil	$9.64\cdot10^5$

In addition to the environmental impacts, the phosphorous (P) saving from the substitution of compost for fertilisers was also evaluated. The normalisation factor for P was 28 kg person^{-1} year^{-1} (EASEWASTE, 2008). The weighting of P is not included in the EDIP methodology and so the weighting (according to the supply horizon) was estimated at 0.00848 yr^{-1} based on data from Cordell et al. (2009). The normalised results were expressed in PE, where 1 PE represents the potential impact of an average person over one year. The weighted results for the P savings were expressed in person reserves (PR), where 1 PR expresses the amount of P that is available for one average person and that person's descendants (Wenzel et al., 1997).

3. INVENTORY AND MODELING OF RELEVANT DATA

3.1 Life cycle inventories

An overview of all aspects included in the environmental assessment is summarised in Table 2. The aspects are divided into direct (emissions occurring during the waste treatment) and indirect (associated with the waste treatment, but not taking place at the treatment facility) contributions. The indirect contributions are further divided into upstream (processes taking place prior to the treatment) and downstream (processes taking place after the treatment). This choice was made according to the UOD (upstream-operation-downstream) concept developed by Gentil et al. (2009). All relevant processes were accounted for. There are no upstream aspects involved in home composting since there is no fuel (there is no collection and transportation involved like central composting (Andersen et al., 2010b)), electricity or material used during the process. The LCI of home composting is presented in Table 3 (and described in section 3.4) and the LCIs of incineration and landfills were taken from Riber et al. (2008) and Manfredi and Christensen (2009) respectively (and described in Sections 3.5 and 3.6).

3.2 Waste quantity and composition

Six composting units were all fed with different input material from six different families. The families were supplied with a guideline that discouraged composting of meat and dairy products, for example, but apart from this, they were not given any guidance. The home composting units were all identical and they were described in details by Andersen et al. (2010a). They consisted of recycled PE and PP and they were located in a shady area on paved surface. They weighted 22 kg and had a total volume of 320 litres each. The input waste was OHW and it mainly consisted of food waste (fresh and cooked) and smaller amounts of flower clippings and herbs (including soil), for example, from the household. In addition, small amounts of garden waste were added as structural material. The amounts of waste varied between 2.6 and 3.5 kg OHW per week across the six units. In order to be able to compare the technologies, 1 Mg of OHW was chosen as the functional unit. Details regarding the amounts and the chemical composition of the input waste can be found in Andersen et al. (2011). The input waste fed into the incinerator and into the landfill was modelled as the average input waste from the six composting units.

3.3 Collection and transportation distances

For home composting, no collection and transportation was accounted for, since the waste was processed at the source. For the transportation of waste from the recycling centres to the WtE plant and the landfill, a distance of 25.4 km was chosen based on a study in Aarhus, Denmark (Boldrin et al., 2011b).

Table 2. Overview of different aspects considered in the assessment according to the upstream-operation-downstream (UOD) concept described by Gentil et al. (2009).

Indirect: Upstream	Direct: Operation	Indirect: Downstream
	Accounted	
Home composting • None WtE plant • Diesel provision • Electricity provision Landfill • Diesel provision • Electricity provision	Home composting • Gas emissions (CH_4, N_2O, CO, NH_3) • Leachate (NO_3^-, NH_4^+, TOC, heavy metals) WtE plant • Use of materials and energy needed for the combustion process • Gas emission from the stack • Combustion of diesel for collection and transportation of OHW Landfill • CH_4 emissions • Use of materials and energy (electricity) • Leachate losses • Combustion of diesel for collection and transportation of OHW • C storage	Home composting *Peat substitution* • Peat substitution • CO_2 biogenic from compost degradation • N_2O emissions *Fertiliser substitution* • Mineral fertiliser substitution • CO_2 biogenic from compost degradation • C binding in soil • N_2O emissions WtE plant • Substitution of electricity • Substitution of heat Landfill • Substitution of electricity • Substitution of heat
	Non-accounted	
Home composting • Construction of home composting units and associated tools WtE plant • Construction of waste facilities and machinery • Provision of other materials Landfill • Construction of waste facilities and machinery • Provision of other materials	Home composting • Any trace gas release (e.g. VOCs) WtE plant • None Landfill • Any trace gas release (e.g. VOCs)	Home composting • Improved soil properties and associated plant growth • Reduced water use • Reduced soil erosion • Other effects of compost use-on-land WtE plant • Treatment of residues (etc. fly ash, bottom ash) • Wastewater Landfill • None

Table 3. LCI data for home composting of organic household waste. Data adapted from Andersen et al. (2011) and Andersen et al. (2010e).

	LCI data	Amount	Unit
Input waste	Organic household waste	113-273	kg ww yr^{-1}
	Garden waste	6-22	kg ww yr^{-1}
Energy and materials consumption	Electricity	0	kWh Mg^{-1} ww
	Water	0	L Mg^{-1} ww
Gaseous emissions (to atmosphere)	CO_2-C (biogenic)	177-252	kg Mg^{-1} ww
		51-95	(% of total C emitted)
	CH_4-C	0.4-4.2	kg Mg^{-1} ww
		0.3-3.9	(% of total C emitted)
	CO-C	0.07-0.13	kg Mg^{-1} ww
		0.04-0.08	(% of total C emitted)
	N_2O-N	0.30-0.55	kg Mg^{-1} ww
		2.8-6.3	(% of total N emitted)
	NH_3	~0	kg Mg^{-1} ww
Liquid emissions (to groundwater)	Leachate generation	130	L Mg^{-1} ww
	Total N losses	0.05	kg Mg^{-1} ww
		0.3-0.6	(% of total N emitted)
	Total C losses	0.33	kg Mg^{-1} ww
		1.3-3.0	(% of total C emitted)
	BOD	3.5	kg Mg^{-1} ww
	COD	9.9	kg Mg^{-1} ww
	K	6.4	kg Mg^{-1} ww
	P	0.08	kg Mg^{-1} ww
	As	$2.4 \cdot 10^{-5}$	kg Mg^{-1} ww
	Cd	$2.5 \cdot 10^{-6}$	kg Mg^{-1} ww
	Cr	$3.2 \cdot 10^{-5}$	kg Mg^{-1} ww
	Cu	$2.9 \cdot 10^{-4}$	kg Mg^{-1} ww
	Hg	$2.8 \cdot 10^{-7}$	kg Mg^{-1} ww
	Ni	$8.7 \cdot 10^{-5}$	kg Mg^{-1} ww
	Pb	$9.9 \cdot 10^{-5}$	kg Mg^{-1} ww
Finished product	Compost	0.27-0.45	kg Mg^{-1} ww
	Substitution of peat	21	% substitution
	Substitution of fertiliser	18	% substitution
	Substitution of manure	0	% substitution

3.4 Modelling of home composting

The home composting setup was described in detail by Andersen at al. (2010a). It consisted of six composting units with different types of management. The main difference was the mixing frequency and the fact that all composting units received OHW from individual and therefore different households. The composting units were divided into three groups, depending on the frequency of mixing, was believed to be the most important parameter that could significantly change the environmental performance of the different composting units, as explained in details in Andersen at al. (2010a). Different mixing frequencies were modelled, including some extremes situations: Units 1 and 2 were mixed every week, Units 3 and 4 were mixed every sixth week and Units 5 and 6 were not mixed at all. All of the data presented were based on experiments performed during one year of composting (May 2008-May 2009) and an additional three months of maturation. The gaseous emissions were quantified as described by Andersen et al. (2010a) over the entire composting phase and the leachate was collected and sampled during two periods of two months each (November-December 2008 and March-April 2009). The inventory (Andersen et al., 2011) developed for home composting comprised all of the processes that contribute to any of the potential impact categories from the actual composting process. In addition, the downstream processes were included in the assessment as described in Sections 3.4.1 and 3.4.2.

3.4.1 Modelling of peat substitution

Peat substitution was modelled on a 1:1 volume basis, assuming that the densities of peat and compost are 200 kg m^{-3} and 700 kg m^{-3} (wet weight), respectively, meaning that 1 Mg of compost can replace 285 kg peat (Boldrin et al., 2010). The actual substitution of peat was estimated based on the results from user surveys (Andersen et al., 2010e). The responses to the user surveys indicated that 21% of the compost users actually replace peat when applying compost in their gardens. In a LCA context, this means that the benefits from peat replacement reach only around one fifth of their potential. This is modelled in EASEWASTE by assuming that 1 Mg of compost can replace 58 kg of peat (instead of 285 kg). All benefits and burdens of substituting peat by compost have been accounted for in the EASEWASTE model (Boldrin et al., 2011a). Carbon emitted during the degradation of peat was considered fossil (during the 100 year timeframe of this study) (Boldrin et al., 2010).

3.4.2 Modelling of fertiliser substitution

The substitution of fertiliser was modelled based on the nutrient (N, P and K) content of compost and their utilisation rate (the fractions of the nutrients that can replace inorganic fertilisers, which is dependent on the availability of the nutrients) (Hansen et al., 2006). The chemical composition of the compost produced in the home composting units was presented in Andersen et al. (2011). Utilisation rates were assumed to be 20% for N and 100% each for P and K. The substitution of fertilisers was estimated from the same user survey as was used for peat substitution (Andersen et al., 2010e) and the answers indicated that 18% of the compost users actually replace fertilisers when applying compost in their gardens. From the nutrient content, the utilisation rates and the estimated substitution from the user surveys, the amount of nutrients that can be substituted per Mg of compost was 0.15-0.20 kg N, 0.23-0.33 kg P and 0.79-1.44 kg K. The carbon that will still be bound in the soil at the end of the 100 year time-frame has been accounted for by crediting the system with avoided CO_2 emissions. The carbon binding was assumed to be 14% of the carbon content of the compost (Bruun et al., 2006).

3.5 Modelling of the WtE plant

The thermal treatment of OHW was performed in the Aarhus WtE plant (this scenario is henceforth referred to as Aarhus 2006). The facility was equipped with a grate furnace with a combined heat and power (CHP) energy recovery system and the efficiency was 20.7% for electricity production and 74% for heat production (calculated using the lower heating value (LHV)). The electricity and the heat provided by the WtE plant were assumed to be based on coal. The flue gas cleaning was performed

with semidry and wet systems. Dioxins and Hg was removed by activated carbon and NO_x was removed by a selective non-catalytic reduction (SNCR) system. The inventory of the plant included the input of materials and energy into the process, but not the treatment of the solid outputs, as the treatment options for solid residues adopted in Aarhus result in emissions which are significantly lower than the direct emissions from the combustion process (Fruergaard & Astrup, 2011). Bottom ashes generated at Aarhus WtE plant are aged and used as base layer in road construction, while fly ashes and APC residues are exported either to Germany or Norway, where they are used for backfilling of salt mines or for neutralization of waste acid respectively.
In addition, another WtE facility was modelled in order to show a range of performances by Danish incinerators. This facility had a less developed flue gas cleaning system and lower energy recovery, which was equivalent to how the Aarhus WtE plant performed in 2003 (the scenario is henceforth referred to as Aarhus 2003). The efficiencies were 11% for electricity and 69% for heat at that time. The two WtE technologies represent "efficient" and "less efficient" technologies.

3.6 Modelling a landfill

The landfill was modelled as a conventional landfill with energy recovery. The landfill included a bottom liner, a leachate collection system (and leachate treatment prior to discharge to surface water bodies), top soil cover, gas collection system, flares and gas utilization for energy recovery (Manfredi and Christensen, 2009). It was assumed that the gas was collected from year 2 to 45 with an efficiency of 50% and it was utilised in a CHP plant for the production of electricity and heat. The inventory did not include direct emissions occurring during the treatment of leachate in the wastewater treatment plant, because of lack of data.

4 RESULTS

4.1 Environmental profile of home composting of OHW with an example of Unit 1

The environmental performance of home composting in Unit 1, divided into sub-processes, is presented in Figure 1. All of the composting units showed the same tendency as Unit 1 and they have thus not been presented here. The main contributions to GW in the home composting scenarios were CH_4 and N_2O emissions from the degradation of the organic matter. The only benefit in terms of GW was from the use of compost, and this was mainly from peat substitution (CO_2 emissions were avoided when compost was used instead of peat). The environmental impact potentials were very small in terms of POF and AC, while benefits were gained in terms of NE from the substitution of fertilisers (mainly from saved chromium emissions when avoiding production of P fertilisers). The most significant toxic impact category was HTs, with large environmental loads resulting from the leaching of (mainly) arsenic, chromium and mercury. On the other hand, the substitution of fertilisers due to lower emissions of Cr and, to a lesser extent, Hg and Cd facilitated environmental savings. HTw was affected in the same way as HTs, but with lower contributions from all sub-processes. The other toxic impact categories were relatively insignificant.

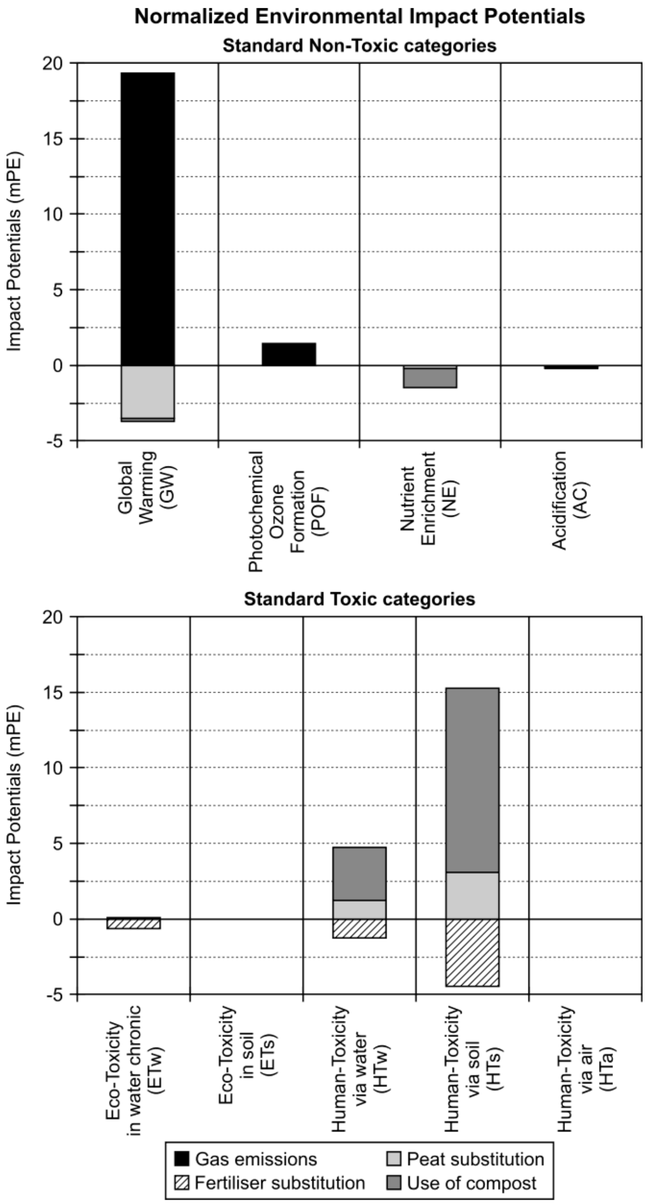

Figure 1 - Normalised potential non-toxic (top) and toxic (bottom) environmental impacts of home composting of 1 Mg of OHW in Unit 1.

4.2 Comparison of environmental profiles for the treatment of OHW

Table 4 shows the potential impacts of the nine treatments (six home composting units, two incineration and one landfill scenario) of OHW, which are then graphically compared in Figure 2. For each of the impact categories, the potential impacts originating from the different processes have been aggregated into single indicators. The total impacts of home composting were quite low and the six scenarios (the first six bars in the figure) showed values of -2 to 16 mPE Mg^{-1} ww for the non-toxic categories and -0.9 to 28 mPE Mg^{-1} ww for the toxic categories. For the potential impact category GW, home composting was an environmental burden in all units due to CH_4 and N_2O emissions. The difference in emissions of CH_4 was the main reason for the difference in the GW impact potential for the six composting units. Units 1 and 2 (frequent mixing) had by far the highest emissions of CH_4, whereas the emissions from Units 5 and 6 (no mixing) were low (Andersen et al., 2010a). Incineration made an overall saving in terms of GW due to the utilization of electricity and heat resulting in a better performance than home composting in this category. However, the landfill was a burden in the same range as Units 3 and 6, due to CH_4 emissions.

In several categories other than GW, home composting performs (in environmental terms) as good as or better than incineration or landfilling. Especially in the impact categories AC, NE and ETw, home composting made overall savings. Incineration did not perform as well, in the same categories, mainly due to NO_X (NE), NO_X and SO_2 (AC) and PAH (ETw) from the stack emissions and emissions relating to diesel combustion. The landfill performed worse than home composting in most impact categories, except for AC and HTw and, in some cases for GW (Units 1, 2 and 4). The greatest impact resulting from the landfill was found in the categories ETw (due mainly to PAH emissions) and HTs (due to benzene, vinylchloride and arsenic among other parameters).

The results in terms of P savings are presented in Table 5 and they show that there were savings in the order -0.97 to -2.01 kg P Mg^{-1} ww in the six home composting scenarios. The weighted results were -0.29 to -0.61 mPR. The variation is due to the different concentrations of P in the input material.

Table 4. Potential environmental impacts associated to the treatment of 1 Mg of OHW in the analysed scenarios and the sensitivity scenarios (unit: mPE).

Scenario (mPE)		Global Warming (GW)	Photochemical Ozone Formation (POF)	Nutrient Enrichment (NE)	Acidification (AC)	Eco-Toxicity in water chronic (ETw)	Eco-Toxicity in Soil (ETs)	Human-Toxicity via Water (HTw)	Human-Toxicity via Soil (HTs)	Human-Toxicity via Air (HTa)
Unit 1	Original	15	1.4	-4.8	-0.40	-2.3	-0.002	-2.6	-11	0.002
Unit 2		14	1.2	-4.0	-0.33	-1.9	-0.002	1.3	-5.5	0.003
Unit 3		4.6	0.34	-4.6	-0.38	-2.2	-0.002	4.0	8.7	0.001
Unit 4		9.9	0.47	-2.6	-0.21	-1.2	-0.001	2.8	1.8	0.007
Unit 5		1.9	0.21	-5.4	-0.45	3.4	-0.002	11	2.7	-0.0005
Unit 6		4.2	0.27	-3.7	-0.30	-1.7	-0.001	6.3	1.4	0.002
Incineration Aarhus 2003		-18	1.1	17	24.2	24	-0.0003	3.8	1.2	2.7
Incineration Aarhus 2006		-28	0.81	12	8.6	23	$4.3e^{-05}$	-2.5	-1.0	2.4
Landfill conventional		5.5	3.8	3.5	-0.53	43	0.032	0.80	20	4.1
Unit 3 - NH_3	Sensitivity	4.6	0.34	8.1	5.6	-2.2	8.7	-0.002	4.0	0.001
Unit 3 - Fertiliser		4.2	0.34	-0.59	-9.0	-4.4	-20	-0.004	-3.9	-0.002
Unit 3 - Peat		1.1	0.32	-0.57	-4.9	-2.2	6.0	-0.002	4.0	-0.002
Incineration - Heat substitution		-13	-1.1	8.3	12.0	25	-0.28	$9.4e^{-05}$	-0.45	2.7
Incineration - Waste composition		-38	-0.028	9.1	5.7	11	-0.0003	-3.0	-1.3	1.0
Landfill - Gas collection		45	7.4	4.1	6.9	39	32	0.064	4.1	3.1
Landfill - Waste composition		4.8	3.4	-2.3	1.8	32	0.036	0.52	22	2.9

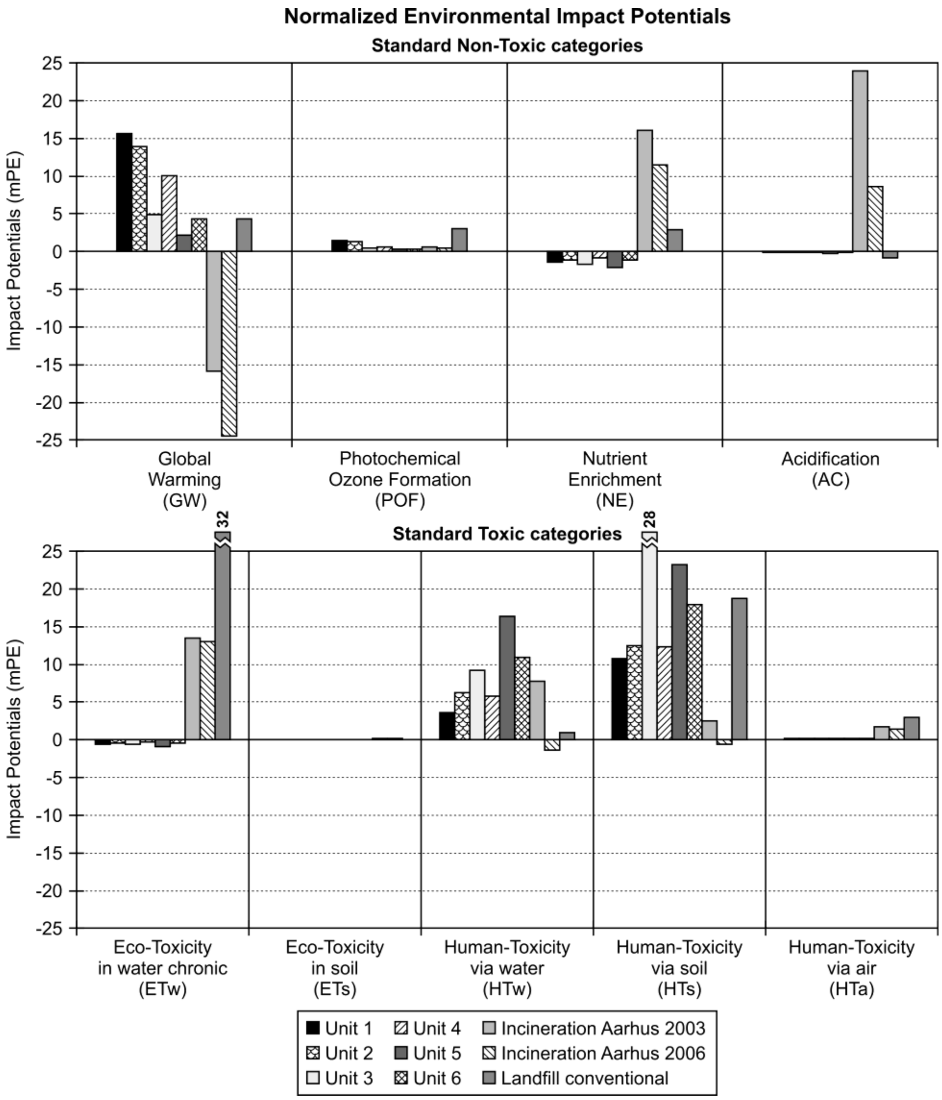

Figure 2 - Comparison of the potential non-toxic (top) and toxic (bottom) environmental impacts of home composting of 1 Mg of OHW in Units 1-6 and the alternative treatments (incineration and landfilling).

Table 5. Phosphorous savings in the six home composting scenarios (all values are per Mg ww treated).

	P savings (kg)	**Normalised (PE)**	**Weighted (mPR)**
Unit 1	-1.80	-0.064	-0.54
Unit 2	-1.48	-0.053	-0.45
Unit 3	-1.72	-0.062	-0.52
Unit 4	-0.97	-0.034	-0.29
Unit 5	-2.01	-0.072	-0.61
Unit 5	-1.37	-0.049	-0.41

4.3 Sensitivity analysis

The most important contributions in terms of the environmental benefits and burdens of home composting have been identified as being the GHG emissions occurring during home composting and the actual substitution of peat and fertiliser when the compost is used on soil. The GHG emissions were comprehensively assessed in the study by Andersen et al. (2010a) and the data have been considered to be quite robust. The difference in GHG emissions among the six composting units can be considered to be realistic, and thus the range acts as a form of sensitivity analysis. As discussed in the study by Andersen et al. (2010a), the most likely management of the home composting units was the form practised in Unit 3 (infrequent mixing and relatively low additions of waste). The contributions to GW from this unit were relatively low (lower than the contributions from the landfill), and thus Unit 3 was used for the sensitivity analysis. The sensible parameters that were assessed are described here:

The substitution of peat and fertiliser was considered quite uncertain since it was based on an estimate based on user surveys (Andersen et al., 2010e). The parameters were doubled for Unit 3, from 18% to 36% for fertiliser substitution and from 21% to 42% for peat substitution.

The NH_3 emissions were found to be insignificant in the study (Andersen et al., 2011); however, other studies have shown that NH_3 is emitted during composting. Amlinger et al. (2008) reported that NH_3 concentrations are highest at temperatures of 40-50°C, which could explain why the NH_3 concentrations (and emissions) are so low in the experiments described here. Temperatures of 40-50°C will only be reached by having very high additions of waste (and/or a warmer outside temperature). An emission of 6.3% of the total N, as reported by Amlinger and Peyr (2002), was considered in the sensitivity test of NH_3 emissions in Unit 3.

The energy efficiency and the flue gas treatment had already been assessed by modelling two different incineration scenarios (Aarhus 2003 and Aarhus 2006). The heat substitution was changed from coal-based to biomass-based, since the coal-based power plant in Aarhus plans to incinerate biomass in the future. The power plant produces heat for the central heating system and the change in energy source will thus affect the entire Aarhus area. The sensitivity analysis was also chosen in order to represent possible future trends of heat substitution.

The levels of gas collection efficiency may vary a great deal between landfills. In the analysis, the gas collection efficiency decreased from 50% to 0%, in order to show the sensitivity of this parameter.

An average from the six waste inputs (Units 1-6) was used as the input waste for the incinerator and the landfill. The composition of this average waste was tested by increasing all compositional values by one standard deviation (STDEV).

The changes chosen for the sensitivity analysis are shown in Table 6. Results are reported in Table 4 and shown graphically in Figure 3. The increased fertiliser substitution resulted in general savings, especially in the impact categories NE, ETw, HTs and HTw. Increased peat substitution resulted in savings in terms of GW but only minor savings in the other impact categories. The NH_3 emissions contributed significantly to NE and AC, making home composting worse than incineration in terms of these two categories when NH_3 is included in the assessment. The change from coal-based to biomass-based energy in the incineration scenario generally decreased its environmental performance, especially in regard to GW (due to less fossil CO_2 being saved). When the gas collection system was removed from the landfill (as in the sensitivity test) the contributions to environmental loads increased, especially in the GW category (where they increased from 5 mPE to 45 mPE). The change in waste composition resulted in relatively low changes in the impacts.

Table 6. Sensitivity analysis for the most important parameters in the modelling.

Test name	Parameter changed	From	To
Unit 3 – Fertiliser	Fertiliser substitution	18 %	36 %
Unit 3 – Peat	Peat substitution	21 %	42 %
Unit 3 – NH_3	NH_3 emissions	0 % of total input N	6.3 % of total input N
Incineration – Heat substitution	Heat substitution	Coal	Biomass
Incineration – Waste composition	Waste composition	Average of Unit 1-6	+ 1 STDEV
Landfill – Gas collection	Gas collection efficiency	50 %	0 %
Landfill – Waste composition	Waste composition	Average of Unit 1-6	+ 1 STDEV

*STDEV, standard deviation

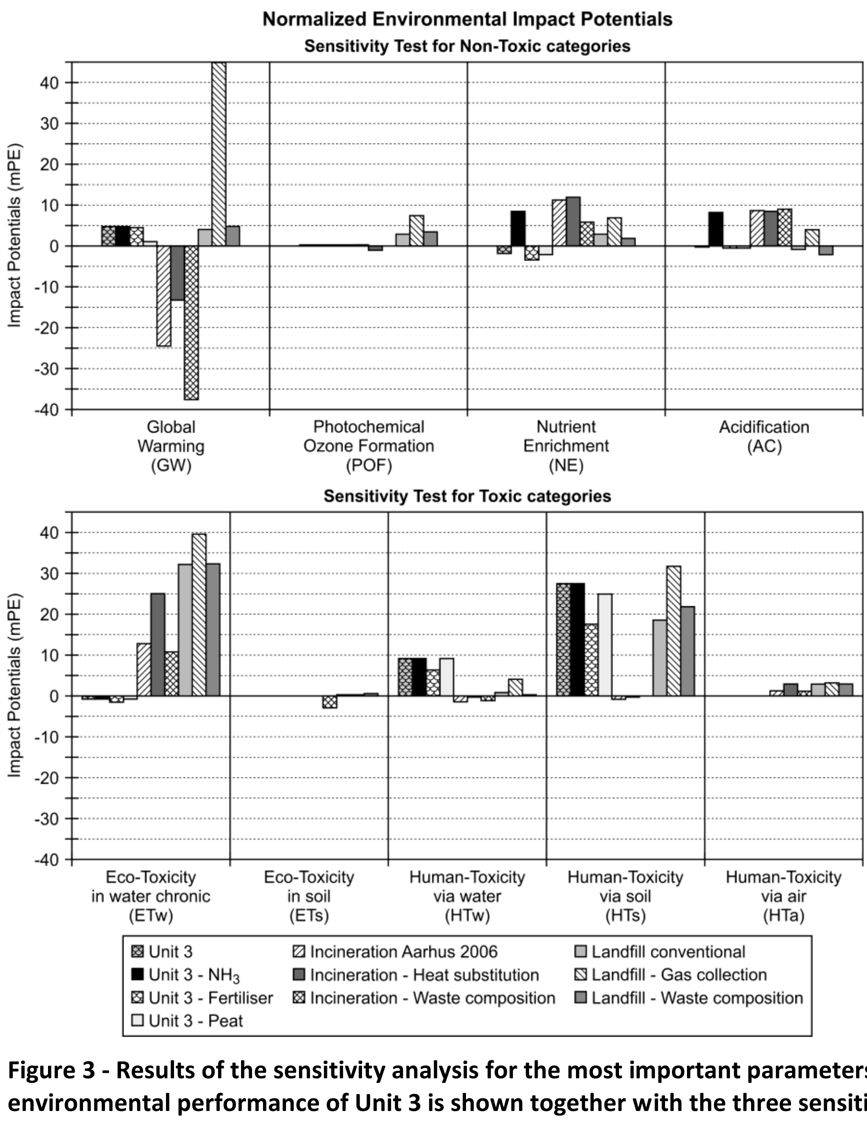

Figure 3 - Results of the sensitivity analysis for the most important parameters/processes in the modelling. The environmental performance of Unit 3 is shown together with the three sensitivity tests for Unit 3 in the first four columns. The next three columns represent the results for the incinerator and the two related sensitivity tests and the last three columns represent the results for the landfill and the two related sensitivity tests.

5. DISCUSSION AND RECOMMENDATIONS

Environmental burdens and savings from the treatment of OHW. The differences in the environmental performance between the six home composting units were relatively large. The largest differences were found in the impact categories of GW, HTs and HTw. The reason for the differences in GW contributions was mainly due to the difference in CH_4 emissions, caused by the varied management of the composting units. The variation in the mixing frequency was ascribed the greatest importance in the impact category GW according to Andersen et al. (2010a). Differences in the impact categories of HTs and HTw were related to the chemical composition of the compost which is governed by the composition of the input material. The most important substances contributing to HTs and HTw were chromium, (HTs), arsenic (HTs) and mercury (HTs and HTw), and these compounds were the ones with the lowest concentrations in the compost in Units 1, 2 and 4 (Andersen et al., 2011), thereby causing the lowest total impact in these impact categories (see Figure 2). Even though the heavy metal content significantly affected some impact categories (especially the toxic categories), the concentrations were all below the threshold values for use of compost on land (Andersen et al., 2011; Hogg et al., 2002) and were therefore not considered to constitute a problem when applied on soil. The EDIP method accounts for the total amount of heavy metals in the compost, instead of taking the effective concentrations into considerations. This means that the heavy metals are considered as environmental burdens (in the toxic categories), even in low concentrations. Since the home composting units were operated individually by the home owners, there was no overall control of the input waste and thus nothing could be done in order to improve the environmental performance in the categories which were governed by the composition of the input waste. A guideline which is delivered together with the home composting units is already in use, in which the composting of meat and dairy products, for example, is discouraged. A range of the potential additional benefits related to the use of the compost cannot easily be quantified and have therefore been excluded from this environmental assessment. These include: improved soil properties and associated plant growth, reduced water use and reduced soil erosion.

In general, the home composting units performed well compared to incineration and landfills, except in terms of GW (and HTs and HTw) where incineration gave the best environmental performance. This suggests that home composting is a good alternative for, at least in part, disposal of OHW. The only other comprehensive LCA of home composting was performed by Martínez-Blanco et al. (2010). In this study, the contributions to GW, AC and NE (only mutual impact categories) were similar to the ones reported in this paper. Martínez-Blanco et al. (2010) included upstream processes such as the production of the composting units and the tools that were used during composting. The contributions of these upstream processes to the total impacts were, however, limited in the aforementioned impact categories (4.6%, 3.1% and 1.5% of the total impacts on GW, AC and NE, respectively). Other impact categories (other than those included in the EDIP method) were affected in more significant ways by the inclusion of the upstream processes. No savings were reported since no downstream processes were included in the assessment by Martínez-Blanco et al. (2010).

The recycling of phosphorous is a debated issue at the moment due to the scarcity of the resource. The supply horizon has been estimated at only 50-100 years (Cordell et al., 2009). When the waste is incinerated, the phosphorous (and other nutrients) is lost. It is possible to recover the phosphorous from dedicated biomass power plants, but the MSW is not clean enough to recover the phosphorous from WtE plants. When OHW is composted, the phosphorous is recycled instead. One limitation of the EDIP resource consumption methodology is that peat is not included. This should be considered as a resource together with P when comparing composting technologies with alternative treatment options.

Limitations and sensitivity. The study is based on a direct comparison of the treatment of 1 Mg of OHW. This is not completely realistic, since OHW would not be incinerated on its own, but together with other fractions of MSW and industrial waste. The most realistic procedure would be to model the entire waste stream, but in this system most waste would be treated in the same way as in the different scenarios and the relative difference between the environmental performances of the two would be very small. This is the reason behind only modelling the OHW.

The sensitivity analysis showed that increased peat and fertiliser substitution has the potential to significantly increase the environmental performance of home composting. This indicates that the substitution should be optimised; however, this is very difficult to do in practice, since it is difficult to change the habits of compost users. In addition, the NH_3 emissions have a potentially large effect on some impact categories (NE and AC). The NH_3 emissions were believed to be a sensible parameter, since only concentration measurements were performed (and thus no direct emission measurements). It was then considered to include emissions measured by other authors (Amlinger and Peyr, 2002). When including a loss of NH_3 of 6.3 % of the total N in the input, the contributions to the impact categories of NE and AC were quite significant, but still lower than for incineration. The benefits of utilising energy from the OHW in WtE plants decreased significantly when the heat substituted was based on biomass instead of coal (mostly in the GW impact category). The benefits of substituting fossil energy with energy (electricity and heat) produced at the WtE-plant is expected to decrease in the future due to more effective energy utilisation and due to the fact that the general electricity grid is becoming less dependent on fossil fuels.

Alternative options for treatment of OHW. In Denmark, OHW is in most cases not source-separated at home but instead incinerated in the nearest WtE plant together with the rest of the household waste. The OHW is only source-separated and sent to a central composting facility (together with garden waste) or for anaerobic digestion (AD) in a few municipalities. This is not very common, however, and does not take place with OHW as the only substrate, and this is the reason why home composting has not been compared with some of these technologies. Central composting plants and AD plants treating OHW are, however, gaining popularity in many countries in Europe. The trend is that the OHW is source-separated and then undergoes treatment such as tunnel composting or AD. In most cases, it is desirable to get the energy out of the waste (AD) first, and then to compost the residue (in open windrows) in order to get a nutrient-rich end product that can be used on land. In this case, there will be less material (due to the energy production) and the end product will most likely not be of the same good quality as compost since some of the organic matter has been removed in the digestion phase.

Comparison of home composting with central composting. The home composting systems described in this study were not directly compared to any central composting plants due to a lack of data and to a lack of composting facilities that treat OHW. There is a tradition in Denmark to incinerate OHW in order to get energy utilization. No central composting plants in Denmark process OHW without addition of large quantities of “drier” waste such as garden waste. A few central composting plants in

Denmark (nine out of 142 central composting plants in 2001) (Petersen and Hansen, 2003) are treating OHW together with considerable amounts of garden waste (at least 50%, based on mass). The rest is solely treating garden waste. The best data from central composting plants are from plants that only treat garden waste, and it is believed that the environmental inventory would be different than from a plant that processed OHW.

In order to compare central and home composting qualitatively, it can be stated that in general the most important parameters are the input material, GHG emissions and fertiliser and peat substitution when compost is used on land (Andersen et al., 2010a; Boldrin et al., 2011b; this study). The values for peat and fertiliser substitution have been assumed to be the same for compost used in hobby gardening, whether it is produced centrally or at home. This is of course, not necessarily the case, but since no data is available, this was thought to be a reasonable assumption. This means that GHG emissions are one of the most important parameters (in terms of environmental impacts), and one of the parameters that can be optimised by the operation of the composting units. The GHG emissions from central composting plants are within the same range as for home composting. Amlinger et al. (2008) reported emission factors (EFs) of 14-41 kg CO_2-equivalents (eq.) Mg^{-1} wet waste (ww) for windrow composting of biowaste and 9-68 kg CO_2-eq. Mg^{-1} ww for windrow composting of garden waste, while Andersen et al. (2010d) reported an EF of 111±30 kg CO_2-eq. Mg^{-1} ww for windrow composting of garden waste. The EFs for home composting as reported by Andersen et al. (2010a) was 100-239 kg CO_2-eq. Mg^{-1} ww depending on the mixing frequency. Martínez-Blanco et al. (2010) reported an EF of 207 kg CO_2-eq. Mg^{-1} ww from home composting of OHW. The most representative operation of home composting was in Unit 3 and Unit 6 with 127 kg CO_2-eq. Mg^{-1} ww and 111 kg CO_2-eq. Mg^{-1} ww, respectively.

6. CONCLUSION

The environmental impacts of home composting was generally quite low, and the six scenarios showed values of -2 to 16 mPE Mg^{-1} ww for the non-toxic categories and -0.9 to 28 mPE Mg^{-1} ww for the toxic categories. The most important processes were identified as the emissions of GHGs (CH_4 and N_2O) from the composting process (contributing to GW) and environmental savings from substituting fertiliser and peat with compost in hobby gardening (causing savings in GW, NE, ETw, HTw and HTs). The latter was also identified as the most sensible parameter in the study. Home composting performed as good as or even better than incineration and landfilling in terms of several impact categories (especially in terms of NE, AC and ETw). Incineration had the lowest impact in terms of GW due to the substitution of energy based on fossil fuels. Based on this LCA of the management of OHW in Denmark, it was concluded that home composting is a suitable waste management option for some OHW.

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