

# Modular life cycle assessment of municipal solid waste management

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2                                   Waste Management

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11

## 12 **Abstract**

13 Life cycle assessment (LCA) is commonly applied to examine the environmental performance of waste  
14 management systems. The system boundaries are, however, often limited to either one tonne of  
15 material or to specific waste treatments and are, therefore, lacking a systems perspective. Here, a  
16 framework is proposed to assess complete waste management systems based on actual waste flows,  
17 assessed with a detailed material flow analysis (MFA) in a modular MFA/LCA approach. The  
18 transformation of the MFA into a product-process-matrix facilitates a direct link between MFA and  
19 LCA, therefore allowing for the assessment of variations in flows. To allow for an up-to-date and  
20 geographically specific assessment, 190 LCA modules were set up based on primary industrial data  
21 and the ecoinvent database. The LCA modules show where there have been improvements in  
22 different recycling processes over the past years (e.g. for paper recycling) and highlight that, from an  
23 environmental perspective, closed-loop recycling is not always preferable to open-loop recycling. In a  
24 case study, the Swiss municipal solid waste management system, of which there is already a detailed  
25 MFA, was modeled using the new LCA modules and applying the modular MFA/LCA approach. Five  
26 different mass flow distribution scenarios for the Swiss municipal solid waste management system  
27 were assessed to show the environmental impact of political measures and to test the sensitivity of the  
28 results to key parameters. The results of the case study highlight the importance of the dominant  
29 fractions in the overall environmental impacts assessment; while the metal fraction has the highest  
30 impact on a per kilogram basis, paper, cardboard, glass and mixed municipal solid waste were found  
31 to dominate the environmental impacts of the Swiss waste management systems due to their mass.  
32 The scenarios also highlight the importance of the energy efficiency of municipal solid waste  
33 incineration plants and the credits from material substitution as key variables. In countries with  
34 advanced waste management systems such as Switzerland, there is limited improvement potential  
35 with further increases in recycling rates. In these cases, the focus of political measures should be laid  
36 on (i) the utilization of secondary materials in applications where they replace high-impact primary  
37 production, and (ii) an increased recovery of energy in waste-to-energy plants.

## 38 **Key words (min. 6):**

39 Life cycle assessment (LCA), material flow analysis (MFA), recycling, energy recovery, indirect  
40 energy, policy scenario

## 41 **1. Introduction**

42 Waste recycling and disposal is an important part of the life cycle of a product and is associated with  
43 environmental burdens like any other life cycle stage. Legislation often refers to the so-called waste  
44 hierarchy (prevention, re-use, recycling, other recovery, disposal) but the hierarchy alone is in-  
45 sufficient in providing guidance when considering a combination of recycling and waste treatment  
46 technologies in integrated waste management systems (WMS). A comprehensive environmental  
47 assessment such as life cycle assessment (LCA) (Finnveden et al., 2005; Gheewala, 2009), as is  
48 recommended by the European legislation (EC, 2008), is necessary to provide decision support to  
49 minimize the environmental burdens associated with waste. Generally, the first priority of the waste  
50 hierarchy (waste reduction) is well accepted as the preferred solution. The remaining waste hierarchy  
51 priorities, however, have been subject to debate. Exceptions to this hierarchy, particularly with regard  
52 to material recycling and energy recovery (see e.g. Finnveden et al. 2005; Pires et al. 2011; Laurent et  
53 al. 2014a), have been shown to exist. The scope of LCA studies on WMS is, however, often limited;  
54 several authors have analyzed either limited geographies (e.g. Rigamonti et al. 2013) or single  
55 material streams (e.g. Humbert et al. 2009; Damgaard et al. 2009; Paraskevas et al. 2013). Most  
56 studies focus on the treatment of a defined amount of waste, e.g. 1 tonne, and do not assess the  
57 overall impact of regional WMS (Andreasi Bassi et al., 2017).

58 LCA allows for expanding the perspective beyond the core WMS, which is important since the  
59 consequences of waste management often depend more on the impacts to surrounding systems than  
60 on the WMS specific emissions (Ekvall et al., 2007). The energy systems model, for example, is of  
61 great importance when assessing thermal waste treatments, as it defines the credits for recovered  
62 energy, and for recycling processes, for which energy consumption is often responsible for the largest  
63 share of the environmental impacts (Laurent et al., 2014a, 2014b; Münster et al., 2013). In addition,  
64 the displacement of primary material production through secondary materials from recycling, as well  
65 as the allocation of burdens and benefits, has substantial influence on the result of waste  
66 management oriented LCAs and has been extensively discussed (Gala et al., 2015; Geyer et al.,  
67 2015; Vadenbo et al., 2016; Zink et al., 2015). In order to stimulate more critical reflection and scrutiny  
68 of the underlying assumptions used for assigning avoided burdens in LCA, Vadenbo et al. (2016)  
69 proposed a reporting framework for the systematic estimation of substitution potentials from resource  
70 recovery. The framework, however, has yet to be tested on a large scale system.

71 Besides safe disposal and emission abatement, waste management aims to conserve resources  
72 through material and energy recovery. Hereby, energetic and material utilization of waste are often  
73 competing. Detailed and comprehensive accounts of waste generation and treatment are increasingly  
74 seen as a quantitative basis for designing and assessing measures to move towards a circular  
75 economy (Tisserant et al., 2017), thus focusing on material resource efficiency while neglecting the  
76 waste's potential energetic use (Haupt and Zschokke, 2017). In parallel, the energetic recovery from  
77 waste is an important contribution to the energy system, and maximum recovery potentials have been  
78 quantified as a benchmark in Liechti (2009) and Münster and Meibom (2011). The indirect energy  
79 savings through substitution of primary material production (and associated energy demand) with  
80 recycled material, however, often exceeds the lower heating values (LHV) of the materials.  
81 Assessments of the WMS should, therefore, include an analysis of the direct energy recovered from  
82 thermal treatment coupled with an assessment of the indirect energy flows for material production.

83 MFAs help to understand the stocks and flows of a WMS and are a natural starting point for an  
84 environmental assessment. MFA and LCA have been applied in parallel for waste management  
85 decision support, providing information on the environmental performance of different mass flow  
86 scenarios (Boesch et al., 2014; Turner et al., 2016; Vadenbo et al., 2013). The combination of regional  
87 MFA and LCA allows for assessing environmental impacts while ensuring consistency between  
88 variables such as MFA and LCA process model transfer coefficients, capacity restrictions of waste-  
89 treatment infrastructure, and waste resource availability. Furthermore, it enables one to capture the  
90 input-dependency of process performance and the resulting environmental implications, both for  
91 dedicated waste treatments such as municipal solid waste incineration (MSWI; Boesch et al., (2014))  
92 and for alternative resources used as feedstock in industries such as cement or iron production  
93 (Boesch et al., 2009; Boesch and Hellweg, 2010; Vadenbo et al., 2013).

94 Laurent and colleagues (2014) analyzed 222 LCA studies on waste management systems (WMS) and  
95 revealed that waste composition and the surrounding energy system highly influence the outcome of  
96 the LCA. The strong dependence on these local conditions justifies the need for regional WMS  
97 assessments. The use of recent and geographically specific data in life cycle inventories (LCIs) is  
98 important for many of the outlined research gaps. First, process models of geographically varying  
99 industrial practices allow one to model the recycling output quality more appropriately and, therefore,  
100 to credit material substitution more accurately. Second, coupling MFA and LCA requires an inventory

101 for all processes identified within the case study region. Third, if site-specific process models are used,  
102 knowledge of the waste composition, which allows for input-dependent modelling of emissions from  
103 waste management facilities (e.g. incinerators), is more coherent (Boesch et al., 2014; Gheewala,  
104 2009; Laurent et al., 2014a). The need for regionally specific assessments has led to the development  
105 of several tools for WMS LCAs (e.g. Clavreul et al. (2014)).

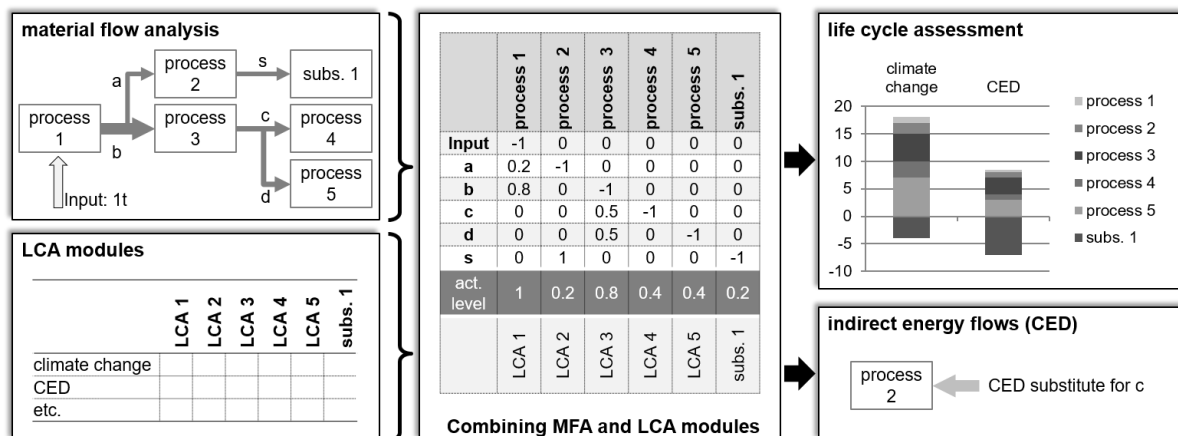
106 This study aims at building a modular LCA model coupled with a national MFA study, while  
107 incorporating up-to-date waste flows and LCIs for thermal treatments and recycling processes. The  
108 new inventories account for the quality of both the recyclables (e.g. amount of residues) and of the  
109 resulting secondary materials. Furthermore, closed-loop and open-loop recycling is differentiated and  
110 is included to evaluate the environmental impact of moving towards a circular economy. The modular  
111 MFA/LCA design allows for a detailed assessment of recycling and treatment pathways as well as  
112 national waste management strategies. The application is illustrated by quantifying the environmental  
113 impacts of the current Swiss WMS as well as five alternative waste management strategies. The  
114 results of this case study are intended to be used by local, regional, and national decision-makers,  
115 related associations and industries as a basis for waste management policies, communication  
116 strategies and future waste management planning and investing.

## 117 **2. Methodology**

118 Life cycle assessment (LCA) is a comprehensive framework for systematically evaluating the potential  
119 environmental impacts of product systems throughout their entire life cycles (ISO 14040:2006). This is  
120 achieved by modeling the cause-effect relationships in the environment induced by resource extraction  
121 or emissions (Hellweg and Milà i Canals, 2014). In comparison, material flow analysis (MFA) maps the  
122 existing waste flows and the treatment processes in place (Brunner and Rechberger, 2004). MFA  
123 provides, for example, the transfer coefficients of all treatment processes as well as data on residues  
124 in separate collection systems that can be used in the LCA calculations. MFA is also the basis for an  
125 input dependent modeling of waste management LCAs, e.g. for the thermal treatment of waste in  
126 MSWI and the recycling processes. The knowledge of the transfer coefficients for all treatments allows  
127 for modeling the sorting and recycling process inventories in accordance with the MFA and assures  
128 mass balance conservation in the system. In addition, the modular study set-up enables the simulation  
129 of material flow variations. As the modules are defined based on system decision nodes that result

130 from the direct coupling of regional MFA and LCA, environmental consequences of political decisions,  
 131 e.g. a change in a process model or on the mass-flow distribution, can be efficiently assessed and  
 132 visualized.

133 Figure 1 exemplifies the combination of MFA and LCA. First, the MFA is completed, including the  
 134 systems input (e.g. 1 tonne of waste material or all wastes occurring in a region), the distribution to the  
 135 processes (flows a to d) and the secondary products (flow s to substitution process 1 including only  
 136 the credits for the substitution). Second, LCA modules for all processes and substitutions are modeled  
 137 and the impacts calculated for as many indicators as preferred (climate change and cumulative energy  
 138 demand (CED) in Figure 1). The MFA is then translated into a technology matrix with a product-  
 139 process-structure showing all transfer coefficients of the processes (positive numbers) and the  
 140 processes absorbing the flows ("-1" in the matrix). Multiplying the input of each process with the  
 141 respective transfer coefficients then yields the activity levels of all processes. By further multiplying the  
 142 activity levels with the LCA modules, the environmental impacts for both individual flows and the  
 143 overall system can be calculated. The substitution achieved through recycling is also integrated in the  
 144 product-process-structure and modeled using the avoided burden approach.



145  
 146 **Figure 1:** Schematic representation of combination of material flow analysis with life cycle assessment  
 147 using a modular LCA structure. LCA modules describe process LCAs (collection, sorting, recycling,  
 148 thermal treatment, etc.) as well as the substitution of materials and energy (as individual modules), as  
 149 indicated for process 1 to 5 and the substitution "subs. 1". CED = cumulative energy demand.

150 The present model aims at an analysis of the status quo of the system and therefore follows an  
 151 attributional approach. The system boundary includes the collection, sorting, preparation for recycling,

152 recycling, thermal treatment, thermal treatments of residues and the final disposal of residual fractions.  
153 Background processes, such as energy inputs and capital equipment are also considered. In general,  
154 the framework allows for assessing a wide range of indicators. The focus was placed on climate  
155 change due to its importance for policy makers (using the Global Warming Potentials 100 years of  
156 IPCC (2013) as characterization factors) and the CED (Frischknecht et al., 1998) for assessing the  
157 indirect energy implications of the WMS. Biogenic CO<sub>2</sub> emissions are assumed to be climate neutral.  
158 Both climate change impacts and CED have been shown to correlate with many other impact  
159 indicators (Huijbregts et al., 2006). An important exception is toxicity, which may show diverging  
160 results for waste treatment systems. Therefore, the USEtox methodology was added to assess any  
161 toxic impact on humans and the ecosystem (Rosenbaum et al., 2008). Besides these specific  
162 indicators, the ILCD indicators (EU-JRC 24708 EN, 2010) were assessed on a midpoint level and are  
163 presented in the Supporting Information (Section 2.2). Long-term emissions have been modelled in the  
164 life cycle inventories for all processes.

165 The framework of Vadenbo et al. (2016) was followed to report the assumptions for material  
166 substitution. The resulting substitution factors were integrated when coupling the activity level with the  
167 respective LCA module (Figure 1). The main components of the framework are the waste-specific  
168 resource potential, the recovery efficiency, and the displacement rate. The latter is determined by  
169 technical limitations, user-perceived equivalence and institutionally prescribed functionalities. The  
170 physical resource potential, i.e. the amount of secondary resources in the waste stream, and the  
171 recovery efficiency are modeled in the MFA. The point of substitution of all fractions is defined to be  
172 after all recycling processes at the level of secondary raw materials, i.e. materials substituting other,  
173 often primary, streams, without undergoing further treatments.

174 The LCA, based on the product-process-matrix from the MFA, was calculated using Brightway2  
175 (Mutel, 2017), an open source framework for LCA in Python (version 3.6). This allows for coupling a  
176 large number of inventories from different sources and allows for the inclusion of additional modelling  
177 steps such as the substitution modelling.

## 178 **2.1. Analysis of direct and indirect energy flows**

179 In parallel to the environmental impact assessment, an analysis of the direct and indirect energy flows  
180 was performed. Direct energy flows denote the waste materials' energetic content, i.e. the lower



181 heating value, and represent the maximum energy potential that can be exploited by thermal  
182 utilization. Indirect energy savings stem from avoided energy consumption due to the substitution of a  
183 material's primary production route with recycling. Indirect energy gains were taken from the full LCA  
184 as described above. The indirect energy gains are given as net CED total (Frischknecht et al., 1998),  
185 i.e. as the difference between the CED of primary and secondary production. As the study focuses on  
186 the net savings from the production of secondary materials and the respective substituted production  
187 routes, alternative waste treatment of substituted materials, e.g. XPS insulation versus foam glass,  
188 were not taken into account.

## 189 **2.2. LCA modules**

190 Because the study aims for a modular design, the waste treatments were split into several small  
191 processes describing individual treatment steps. The treatment of recyclables is divided into collection  
192 (curbside, centralized, etc.), sorting, pre-treatment and recycling phases. Between these phases,  
193 material can be shifted from one process chain to another. Transport, therefore, often marks the  
194 beginning and end of these modules giving the opportunity to choose the further treatment options.  
195 Thermal treatments are modeled as six modules, namely the collection, the incineration (process-  
196 specific and input-specific emissions), the bottom and fly ash treatment processes, and the electricity  
197 and heat recovery. The individual modules for fly ash and bottom ash treatments allow for modeling  
198 the ash processing independently of the incineration process, e.g. at different locations and with  
199 different technological specifications.

200 The foreground system of the study includes the set of processes directly affected by the decision of  
201 the study (Clift et al., 2000). While some inventories exist from database providers such as ecoinvent  
202 (Wernet et al., 2016) and literature (e.g. Arena et al. (2003), Merrild et al. (2009), Chilton et al. (2010),  
203 European aluminium association (2013)), many are outdated or geographically not applicable to  
204 Switzerland. Therefore, most inventories have been revised in collaboration with industrial partners  
205 and experts in a large primary data collection effort (Haupt et al., submitted). The data for the LCIs has  
206 been collected in 2015 and 2016. The LCA modules described are modeled primarily for Switzerland  
207 but can also be used for, or adapted to, similar WMS. All processes, methodological and technical  
208 assumptions, and inventories are described and provided in Haupt et al. (submitted). Data for the  
209 background system, i.e. data on processes that interact with the foreground system supplying or  
210 receiving material or energy, was taken from ecoinvent (system model cut-off, version 3.3 (Wernet et

211 al., 2016)).

### 212 **2.3. Case study**

213 The case study serves to both illustrate the application of the modular LCA and to draw policy  
214 conclusions from it. It encompasses the region of Switzerland for the reference year 2012, and is  
215 based on a detailed assessment of both municipal solid waste (MSW) flows from Swiss households  
216 and industrial waste with a similar composition (Haupt et al., 2017a). This MFA served as the basis for  
217 the process-product matrix for Switzerland. The functional unit of the case study is the treatment of all  
218 MSW fractions occurring in Switzerland in 2012. A zero burden assumption was chosen (Finnveden,  
219 1999), therefore the environmental burden of the production of the materials which reach the WMS  
220 was neglected. A total of 67% of the MSW was specified on the level of material fractions, i.e. paper,  
221 cardboard, glass, PET bottles and other plastics, ferrous metals incl. tinsplate, aluminum incl. used  
222 beverage cans, and biogenic waste. As data regarding the exact composition of the total MSW was  
223 lacking, 9% were assumed to be mixed MSW (consisting of composites, minerals, textiles, and  
224 natural products such as leather). Of the national MSW, 21% are direct deliveries to MSWI plants  
225 (from small business or private persons). These direct deliveries were assumed to have the same  
226 composition as regular mixed MSW. The remaining 3% of the MSW, i.e. electronic waste, batteries  
227 and other hazardous wastes, were not taken into account due to missing data regarding both their  
228 mass flows and treatment process LCAs of their treatment processes. The material substitution was  
229 modeled by taking the material quality into account (e.g. open-loop and closed-loop systems as  
230 defined in Haupt et al. (2017a)). The substitution modeling for Switzerland is described based on  
231 Vadenbo et al. (2016) and can be found in the Supporting Information S2.

232 The direct energy flows in the waste fractions are calculated by quantifying the LHV to each individual  
233 flow of the Swiss MFA (Table S1.1 in the Supporting Information). The same LHV are used in the LCA  
234 of thermal waste treatments. It was assumed that residual fractions, e.g. from the sorting process of  
235 separate collected fractions, have the same composition as mixed MSW. The energy recovery in  
236 MSWI was calculated by categorizing the MSWI plants in Switzerland into 5 archetypes as described  
237 in Meylan et al. (submitted) and as presented in the Supporting Information (Section 1.4).

## 238 **2.4. Scenario Analysis for Case study**

239 Five scenarios were developed and assessed to highlight the improvement potential of political  
240 measures in the Swiss MSW management system. In all scenarios, the functional unit was kept  
241 constant (treating all MSW occurring in Switzerland in 2012). If mass flows into a specific treatment  
242 process are changed, this is balanced with a reduction in the amount of the same material in the  
243 previous treatment pathways. These scenarios, therefore, present what-if scenarios of waste  
244 management actions. All assumptions are described in the following sections. It is important to keep in  
245 mind that the scenarios vary in the level of changes necessary for their implementation. Some are  
246 rather extreme scenarios that assume drastic changes in WMS, while others reflect changes that have  
247 already taken place.

### 248 **2.4.1. Scenario 1: Increase recycling rates**

249 The collection rates of paper, cardboard, glass, PET, aluminum, tinplate and organic waste were  
250 increased by 5%, as this value corresponds to a realistic improvement potential as identified in Steiger  
251 (2014). Plastics were kept constant, as an increase in collection rate was tested in scenario 4. The  
252 distribution in closed-loop and open-loop recycling pathways was kept consistent with 2012 data. It  
253 was assumed that the additional paper and cardboard would be collected together, as this facilitates  
254 higher recycling rates. In contrast to paper, 30% of the glass was assumed to be collected as mixed  
255 color cullet and 70% as color separated fractions, as found in 2012.

### 256 **2.4.2. Scenario 2: Increase closed-loop recycling**

257 The recycling rates of 2012 were kept at the same level as identified in Haupt et al. (2017a), however,  
258 all materials (exceptions being PET bottles and mixed aluminum) entered the closed-loop recycling  
259 pathways. For PET, only blue and transparent bottles are currently used in bottle production and  
260 therefore qualify for closed-loop recycling. For aluminum, only source separated used beverage cans  
261 are allowed in the closed-loop recycling. In addition, the amount of glass reused was kept constant  
262 while substantially increasing the use of cullet in packaging glass production and decreasing the  
263 production of foam glass. The recycling of plastic and the utilization of organic waste remain  
264 unchanged.

### 265 **2.4.3. Scenario 3: Increase open-loop recycling**

266 Scenario 3 shows the environmental impact of moving towards more open-loop recycling under the  
267 assumption that all material is treated in open-loop processes as defined in (Haupt et al., 2017a). The  
268 recycling rates were kept constant. Similar to scenario 2, the recycling of plastic and utilization of  
269 organic waste was kept constant.

### 270 **2.4.4. Scenario 4: Increase plastic recycling**

271 In 2012, only PET bottles and a minor share of the polyethylene bottles were collected separately.  
272 Since 2012, however, the amount of separately collected plastic has increased and has reached  
273 11,000 tonnes in 2016. Of this plastic, 48% was collected at retailers (mostly bottles), 10% at  
274 communities and public collection points and 42% in a mixed plastic collections bag (Dinkel et al.,  
275 2017). This scenario therefore evaluates the effects of the increase in plastic recycling in Switzerland  
276 between 2012 and 2016. To enable for comparison with other scenarios, it assumes that overall  
277 plastic consumption has not changed. It also assumes that the plastic would be recycled and sorted in  
278 Switzerland and residues would be treated in Swiss cement kilns.

### 279 **2.4.5. Scenario 5: Increase efficiencies in the MSWI**

280 The environmental impact of waste incineration is largely dependent on the energy recovery from the  
281 waste (Boesch et al., 2014; Cherubini et al., 2009; Christensen et al., 2009; Harrison et al., 2000;  
282 Jeswani et al., 2012). An increase in the energy recovery efficiency is therefore modeled in scenario 5.  
283 The efficiencies for the MSWI archetypes are taken from Meylan et al. (submitted) (scenario for 2035,  
284 “low MSW amount”) and provided in the Supporting Information (Section 1.4).

## 285 **3. Results**

286 In total, 190 LCA modules for MSW management were either newly developed or adapted from  
287ecoinvent processes (Table 1). The data for 51 recycling process LCI has been collected to either  
288 update existing processes or create new process models. Two modules for biogenic processes were  
289 taken from literature and 10 processes were based on the existing ecoinvent processes with minor  
290 adjustments. The substitution of 36 materials from recycling processes was modeled. In addition, the  
291 thermal treatment of 12 waste fractions was modeled within 84 modules comprising the thermal waste  
292 treatment and the recovery and recycling of recovered fractions from fly and bottom ash. The

293 inventories and a description of all modules are provided in (Haupt et al., submitted). These modules  
294 can be combined flexibly to model a WMS.

295 In the Swiss MSW management system case study, the LCA modules are used to model the more  
296 than 500 processes of the Swiss MSW management system. The number of processes exceeds the  
297 number of modules, as some modules were used to approximate several processes. For example, the  
298 same LCA module was used for sorting used beverage cans, sorting aluminum from tinplate and  
299 sorting tinplate. The new LCA modules, their contribution to the overall system and the overall  
300 environmental impact of the Swiss MSW management system are described in the following sections.

301

302  
303  
304

**Table 1:** List of process life cycle inventories developed and applied in this paper, ordered according to the material waste fractions. A detailed description of the models as well as the resulting inventories can be found in (Haupt et al., submitted).

material	process	process description	LCI
paper	collection	collection of waste paper	new LCI
	sorting	sorting of waste paper CH	new LCI (same as for cardboard)
	recycling	paper recycling to secondary fibers for newsprint production CH	new LCI
	recycling	reuse of waste paper in the production of insulation material CH	new LCI
	recycling	paper recycling to secondary fibers for newsprint production RER	LCI adapted from ecoinvent
	substitution	substitution of newsprint from virgin fibers	S, LCI from ecoinvent
	substitution	substitution of insulation made from cellulose	LCI adapted from ecoinvent
cardboard	collection	collection of waste cardboard CH	new LCI
	sorting	sorting of waste cardboard CH	new LCI
	recycling	cardboard recycling to secondary fibers for linerboard production CH	new LCI
	recycling	cardboard recycling to secondary fibers for fluting medium production CH	new LCI
	recycling	reuse of waste cardboard in the production of insulation material CH	new LCI (same as for paper)
	recycling	cardboard recycling to secondary fibers for cardboard production RER	LCI adapted from ecoinvent
	substitution	substitution of corrugated board from virgin fibers	S, LCI from ecoinvent
biogenic waste	substitution	substitution of insulation made from cellulose	LCI adapted from ecoinvent
	collection	collection of biogenic waste CH	new LCI
	composting	composting of biogenic waste from household	Zschokke and Schleiss (2016)
	anaerobic digestion	anaerobic digestion of biogenic waste from household	Zschokke and Schleiss (2016)
	substitution	N fertilizer replaced by compost	S, LCI from ecoinvent
	substitution	P fertilizer replaced by compost	S, LCI from ecoinvent
	substitution	K fertilizer replaced by compost	S, LCI from ecoinvent
	substitution	N fertilizer replaced by liquid digestate	S, LCI from ecoinvent
	substitution	P fertilizer replaced by liquid digestate	S, LCI from ecoinvent
	substitution	K fertilizer replaced by liquid digestate	S, LCI from ecoinvent
	substitution	N fertilizer replaced by solid digestate	S, LCI from ecoinvent
	substitution	P fertilizer replaced by solid digestate	S, LCI from ecoinvent
	substitution	K fertilizer replaced by solid digestate	S, LCI from ecoinvent
	substitution	substitution of natural gas through biogas	S, LCI from ecoinvent
	substitution	substitution of peat from compost and digestate	S, LCI from ecoinvent
substitution	substitution of straw from compost and digestate	S, LCI from ecoinvent	
plastics	collection	collection of post-consumer plastic bottles or mixed plastics at curbside CH	new LCI
	collection	collection of post-consumer plastic bottles or mixed plastics at collection points CH	new LCI
	collection	collection of post-consumer plastic bottles or mixed plastics at retailers CH	new LCI
	sorting	sorting of mixed plastics or plastic bottles CH	new LCI
	sorting	sorting of mixed plastics or plastic bottles DE	new LCI
	sorting	sorting of mixed plastics or plastic bottles AT	new LCI
	recycling	recycling of polyethylene to polyethylene granulate, non-food-grade CH	new LCI
	recycling	recycling of polyethylene to polyethylene granulate, non-food-grade RER	new LCI
	substitution	substitution of primary, high impact polystyrene granulates	S, LCI from ecoinvent
	substitution	substitution of primary high-density polyethylene granulates	S, LCI from ecoinvent
	substitution	substitution of primary low-density polyethylene granulates	S, LCI from ecoinvent
	substitution	substitution of primary polypropylene	S, LCI from ecoinvent
	substitution	substitution of wood	S, LCI from ecoinvent
	substitution	substitution of concrete	S, LCI from ecoinvent

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<b>material</b>	<b>process</b>	<b>process description</b>	<b>LCI</b>
PET bottles	collection	collection of waste PET bottles	new LCI
	sorting	sorting of waste PET bottles (from residues and according to color) CH	new LCI
	sorting	sorting of waste PET bottles (from residues) and transport for export CH	new LCI
	recycling	recycling of PET bottles to PET, bottle-grade CH	new LCI
	recycling	recycling of PET bottles to PET, amorphous CH	new LCI
	recycling	recycling of PET bottles to PET, bottle-grade RER	new LCI
	recycling	recycling of PET bottles to PET, amorphous RER	LCI adapted from ecoinvent
	substitution substitution	substitution of primary bottle-grade PET granulates substitution of primary amorphous PET granulates	S, LCI from ecoinvent S, LCI from ecoinvent
beverage cartons	collection	collection of post-consumer beverage cartons at curbside CH	new LCI
	collection	collection of post-consumer beverage cartons at collection points CH	new LCI
	collection	collection of post-consumer beverage cartons at retailers CH	new LCI
	sorting	sorting of beverage cartons from mixed plastics or plastic bottles CH	new LCI
	sorting	sorting of beverage cartons from mixed plastics or plastic bottles DE	new LCI
	sorting	sorting of beverage cartons from mixed plastics or plastic bottles AT	new LCI
	recycling	recycling of fibers in beverage cartons to linerboard CH	new LCI
	recycling	recycling of fibers in beverage cartons to linerboard DE	new LCI
	recycling substitution	recycling of fibers in beverage cartons to linerboard AT substitution of linerboard from primary production	new LCI S, LCI from ecoinvent
ferrous metals	collection	collection of waste tinplate and aluminum CH	new LCI (same as for tinplate)
	sorting	sorting of waste tinplate and aluminum CH	new LCI (same as for tinplate)
	transport	transport of ferrous metals from CH to RER	new LCI
	recycling	recycling of ferrous scrap to low-alloyed steel in EAF CH	new LCI
	recycling	recycling of ferrous scrap to low-alloyed steel in EAF RER	LCI from ecoinvent
	substitution	substitution of primary steel from converter	S, LCI from ecoinvent
tinplate	collection	collection of waste tinplate and aluminum CH	new LCI
	collection	collection of waste tinplate and aluminum and export CH	new LCI
	sorting	sorting of waste tinplate and aluminum CH	new LCI
	sorting	sorting of waste tinplate and aluminum RER	new LCI
	recycling	detinning of waste tinplate CH	new LCI
	recycling	tin production from concentrate RER	new LCI
	recycling	recycling of black plate to low-alloyed steel in EAF CH	new LCI (same as steel scrap)
	substitution substitution	substitution of primary tin substitution of primary steel from converter	S, LCI from ecoinvent S, LCI from ecoinvent
aluminum, UBC	collection	collection of waste tinplate and aluminum (mixed) CH	new LCI (same as for tinplate)
	sorting	sorting of waste tinplate and aluminum CH	new LCI (same as for tinplate)
	sorting	sorting of waste tinplate and aluminum RER	new LCI (same as for tinplate)
	recycling	remelting of sorted aluminum to cast aluminum RER	LCI from ecoinvent
	recycling	remelting of sorted used beverage cans to wrought aluminum RER	LCI from ecoinvent
	substitution	substitution of primary wrought aluminum from UBC	S, LCI from ecoinvent
	substitution	substitution of cast aluminum from mixed post-consumer aluminum	S, LCI from ecoinvent

308

309

<b>material</b>	<b>process</b>	<b>process description</b>	<b>LCI</b>
glass	collection	collection of waste packaging glass	new LCI
	sorting	sorting of waste packaging glass CH	new LCI
	sorting	sorting of waste packaging glass RER	new LCI
	recycling	washing of packaging glass for reuse CH	new LCI
	recycling	recycling of glass to green packaging glass CH	new LCI
	recycling	recycling of glass to green packaging glass DE	LCI from ecoinvent
	recycling	production of green packaging glass from primary raw materials CH	new LCI
	recycling	production of green packaging glass from primary and secondary raw materials CH	new LCI
	recycling	recycling of glass to foam glass gravel CH	new LCI
	recycling	recycling of glass to foam glass plates RER	LCI from ecoinvent
	recycling	recycling of glass to foam glass gravel RER	LCI from ecoinvent
	recycling	recycling of glass to glass sand CH	new LCI
	substitution	substitution of average Swiss insulation mix	S, LCI from ecoinvent
	substitution	substitution of XPS	S, LCI from ecoinvent
	substitution	substitution of green packaging glass made from primary materials	S, LCI from ecoinvent
	substitution	substitution of primary foam glass plates RER	S, LCI from ecoinvent
	substitution	substitution of foam glass plates from primary and secondary materials	S, LCI from ecoinvent
substitution	substitution of sand by glass sand from cullet	S, LCI from ecoinvent	
mixed MSW	collection	collection of mixed municipal solid waste from curbside CH	LCI from ecoinvent
	collection	transport of sorting residues to municipal solid waste incineration CH	LCI from ecoinvent
	incineration	incineration of various fractions CH	12 LCIs from Boesch et al. (2014)
	incineration	incineration of MSW (incl. ash treatments and energy recovery) DE	LCI from ecoinvent
	bottom ash treatment	advanced bottom ash treatment for various fractions CH	12 LCIs from Boesch et al. (2014)
	bottom ash treatment	basic bottom ash treatment for various fractions CH	12 LCIs from Boesch et al. (2014)
	fly ash treatment	fly ash in underground deposit	12 LCIs from Boesch et al. (2014)
	fly ash treatment	fly ash in residual material landfill (after stabilization with cement)	12 LCIs from Boesch et al. (2014)
	fly ash treatment	acid fly ash washing and treatment of hydroxide sludge in Waelz kiln (FLUWA process)	12 LCIs from Boesch et al. (2014)
	fly ash treatment	acid fly ash washing and direct zinc electrolysis (FLUREC process)	12 LCIs from Boesch et al. (2014)
	heat recovery	flows of recovered heat were integrated in product-process matrix (60 flows)	-
	electricity recovery	flows of electricity produced were integrated in product-process matrix (60 flows)	-
	substitution	material substitution according to Boesch et al. (2014)	S, LCI from ecoinvent
	substitution	substitution of alternative energy carriers in district heating networks	new LCI
substitution	substitution of alternative heating systems	new LCI	
substitution	substitution of Swiss electricity mix	S, LCI from ecoinvent	
cement kiln	treatment	use of PE as alternative fuel in cement kiln (clinker production) CH	LCI adapted from ecoinvent
	substitution	substitution of clinker produced from coal as energy carrier CH	LCI adapted from ecoinvent
all fractions	building	waste preparation facility	new LCI

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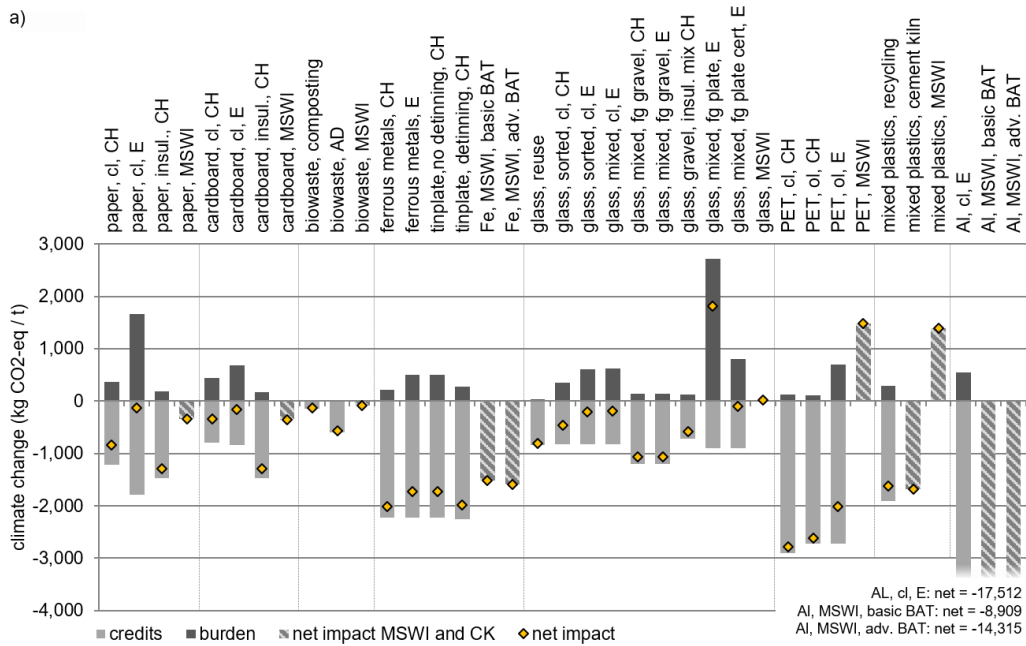
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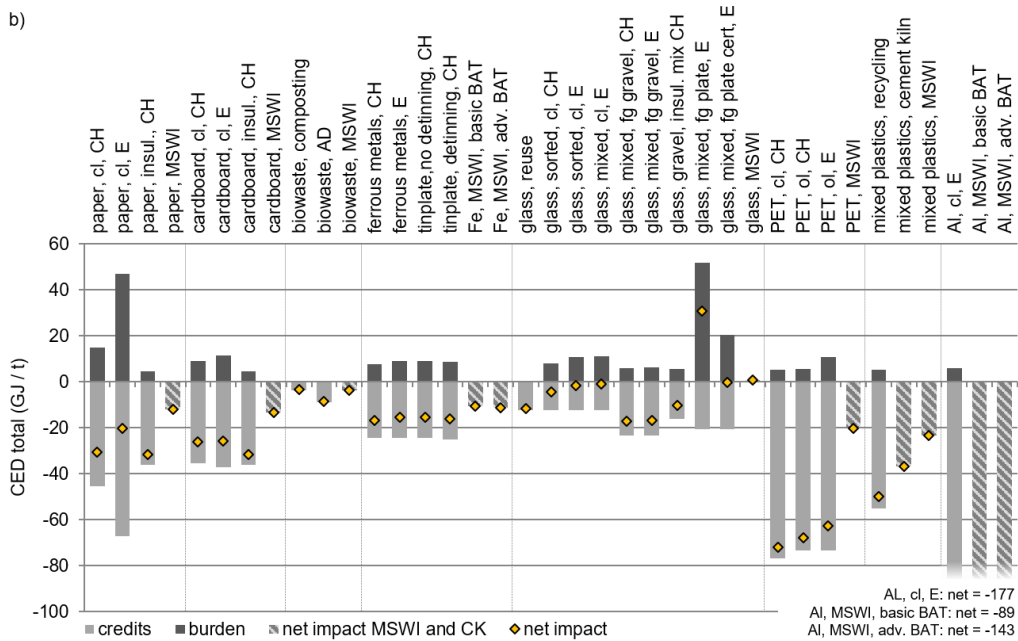


314 **3.1. LCA of individual modules**

315 Figure 2 shows the environmental impacts for the treatment of one tonne of various materials in the  
 316 given valorization pathway, including its collection, preparation for recycling, recycling and the  
 317 resulting material substitution. The results describe the collection of 1 tonne of the recyclable material  
 318 as well as the respective residues in their collection (e.g. the collection of 1 t of PET also includes 0.3 t  
 319 of mixed MSW, which is sent for treatment in a MSWI during the sorting or recycling process).



320



321

322 **Figure 2:** Climate change impacts (a) and CED (b) per tonne of material recycled including the  
323 treatment of the residues arising in the separate collection system. *cl* = closed-loop, *ol* = open-loop,  
324 *insul* = to cellulose insulation production, for glass: *mixed* = mixed color collection and *sorted* = color  
325 sorted collection, *fg* = foam glass, *cert* = produced with certified electricity, *MSWI* = municipal solid  
326 waste incineration, *CK* = cement kiln, *BAT* = bottom ash treatment, *adv* = advanced, *CH* = recycling in  
327 Switzerland, *E* = exported material which is recycled in Europe.

### 328 3.1.1. Paper and cardboard recycling

329 For paper and cardboard, both the recycling and the treatment in MSWI with energy recovery is  
330 environmentally beneficial. Because of industry wide environmental improvement initiatives, the new  
331 LCI for paper recycling in Switzerland has reduced impacts by half due to 25% lower electricity use  
332 and reduced water consumption. These changes have led to a substantial increase in the  
333 environmental benefits associated with paper recycling when recycling compared to previous studies  
334 in which paper recycling was not necessarily beneficial (depending on the indicator, for example in  
335 Merrild et al. (2009) and Dinkel and Kägi (2014)). Overall life cycle impacts from recycling amount to -  
336 350 kg CO<sub>2</sub>-eq/t of cardboard and -840 kg CO<sub>2</sub>-eq/t of paper. If paper and cardboard wastes were  
337 used to produce insulation material, the highest credits (-1290 kg CO<sub>2</sub>-eq/t of paper, see Figure 2)  
338 were yielded if XPS is substituted. For impacts measured as CED, the insulation production from  
339 waste paper and the closed-loop recycling reveal similar credits (-32 GJ-eq/t and -31 GJ-eq/t,  
340 respectively). If waste paper is recycled abroad or treated in MSWI, credits were -20 GJ-eq/t and -12  
341 GJ-eq/t, respectively.

### 342 3.1.2. Metal recycling

343 On a per tonne basis, metals are, from an environmental perspective, the most significant fraction  
344 being studied. Environmental benefits of aluminum recycling are as high as -17.5 t CO<sub>2</sub>-eq/t or -177  
345 GJ-eq/t of recycled aluminum assuming a global aluminum market. There is a large range of possible  
346 credits, as Damgaard et al. (2009) identified credits ranging between -3.8 to -19 t CO<sub>2</sub>-eq/t for recycled  
347 aluminum. For tinplate and ferrous scrap, the recycling resulted in -2 t CO<sub>2</sub>-eq/t of ferrous metals  
348 recycled (-17 GJ-eq/t) within Switzerland and -1.7 t CO<sub>2</sub>-eq/t (-15 GJ-eq/t) for material recycled  
349 elsewhere in Europe. While not evident from the chosen main impact indicators, the toxicity of slag  
350 disposal resulting from steel recycling can dominate other, toxicity based, impact categories. While the  
351 burdens of recycling differ geographically based on the electricity sources, the credits given (converter

352 steel from Europe) were assumed to be the same everywhere. This assumes a European market for  
353 primary steel, and hides the differences regarding the environmental burdens (primary production  
354 creates tenfold the environmental impact compared to recycling).

355 Impacts from tinplate recycling differ depending on the pre-treatment of the recyclables. Separately  
356 collected tinplate that undergoes a detinning procedure (electro-chemical process to separate tin from  
357 steel) reveals an environmental benefit of -2 t CO<sub>2</sub>-eq/t (-16 GJ-eq/t). Tinplate that is recycled without  
358 prior detinning (where tin is lost in the steel product) has lower (12%) environmental benefits than  
359 detinned tinplate. As the tin content of collected tinplate is lower than Swiss and European scrap  
360 specification tin limits (Eurofer, 2008; Haupt et al., 2017b; Stahl Gerlafingen AG and Swiss Steel AG,  
361 2010), it was assumed that tin would not restrict the future utilization of the steel in its first recycling  
362 cycle. The accumulation of tin, however, was not modeled, as this would require a global model for the  
363 accumulation of impurities (similar to Daehn et al. (2017) for copper) for the alloying element tin  
364 (similar to Nakamura et al. (2017)). The net benefits for both metals are within the range outlined in  
365 Brogaard et al. (2014).

366 For comparison, metals recovered from the bottom ash of thermally treated aluminum or ferrous scrap  
367 treatment and later recovery of metals was modeled. The differences between the basic and advanced  
368 bottom ash treatment are described using different recovery efficiencies for metals (Haupt et al.,  
369 submitted). Relative to the credits from separately collected metals (see Figure 2), the credits that can  
370 be realized through metal recovery in bottom ash treatment plants are between 75% and 80% for  
371 ferrous metals and between 50% and 81% for aluminum.

### 372 **3.1.3. Glass recycling**

373 The impacts of glass recycling vary depending on the secondary product and the substitution of  
374 primary material. The new inventories lead to benefits of -805 kg CO<sub>2</sub>-eq/t of waste for glass reuse, -  
375 466 kg CO<sub>2</sub>-eq/t of waste cullet used in domestic packaging glass production and -211 kg CO<sub>2</sub>-eq/t if  
376 the glass is exported for recycling to packaging glass (Figure 2). The production of foam glass gravel  
377 is highly beneficial if XPS insulation material and foam glass plates are substituted, and results in -  
378 1066 kg CO<sub>2</sub>-eq/t. The results in terms of CED also highlight the benefit of foam glass gravel and  
379 bottle reuse (Figure 2).

380 Foam glass gravel was assumed to replace a mixture of XPS and primary foam glass plates based on

381 Stettler et al. (2016), but the sensitivity of the results to this assumption was tested by substituting a  
382 common Swiss insulation mix (EPS, XPS, PUR, mineral wool, and glass wool (Kasser et al., 2016)),  
383 which lead to a reduced foam glass gravel credit by 45% (Figure 2). The production of foam glass  
384 plates is highly energy intensive and the related substitution of XPS only results in credits if certified  
385 electricity is used for the recycling (as currently done in Belgium (IBU, 2015)). Also, the energy inputs  
386 required for the production of packaging glass are responsible for a large share of the burdens and,  
387 therefore, the overall impacts are largely dependent on the location (-1320 kg CO<sub>2</sub>-eq/t in Switzerland  
388 and -800 kg CO<sub>2</sub>-eq/t in Europe).

#### 389 **3.1.4. Plastic recycling**

390 The Swiss collection rates of PET are high (85%) compared to international rates. The PET recycling  
391 system is regularly assessed (Dinkel and Hauser, 2008; FOEN, 2013), and was updated for this study.  
392 The potential net savings arising from substitution of virgin plastic were estimated to be between -  
393 2,567 and -1,766 kg CO<sub>2</sub>-eq/t (at collection rates of 1% and 99%, respectively, assuming the 2012  
394 shares between open- and closed-loop recycling) of plastic waste, depending on the collection rate,  
395 the related purity and the transport distance (Haupt et al., submitted).

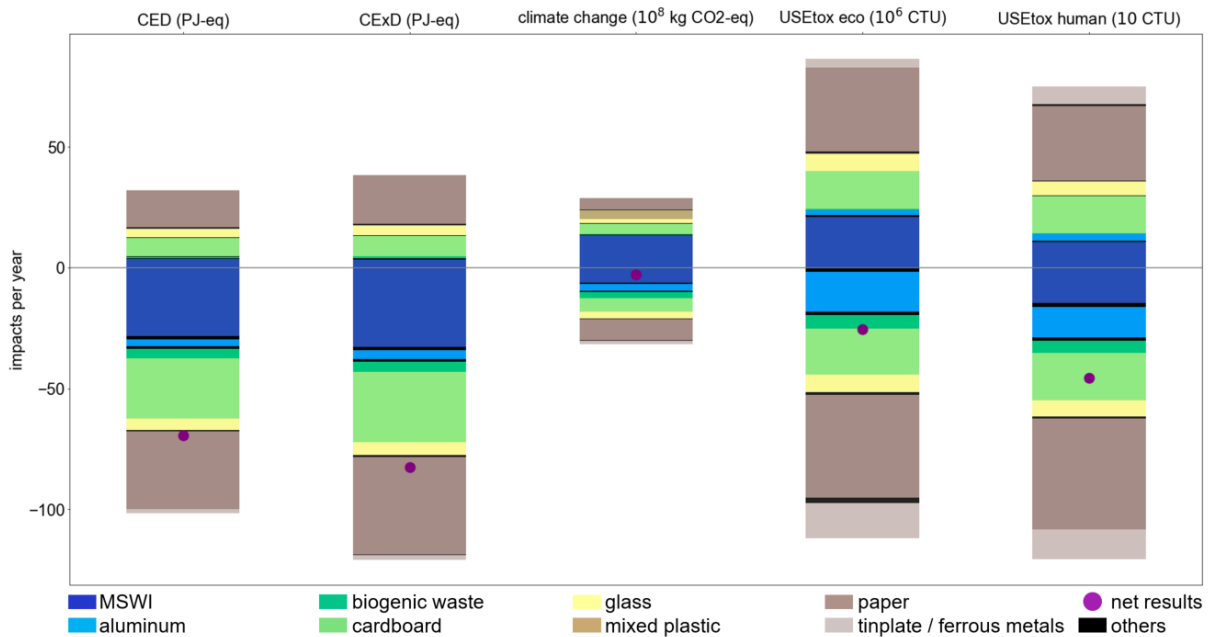
396 The recycling of polyethylene and other plastics is less environmentally beneficial than the recycling of  
397 PET (-1,600 kg CO<sub>2</sub>-eq and -50 GJ-eq/t compared to -2,800 kg CO<sub>2</sub>-eq and -72 GJ-eq/t, respectively)  
398 because their primary production is less energy intensive. The use of plastic as auxiliary fuel in cement  
399 kilns reveals credits in the same order of magnitude as the recycling of polypropylene, polyethylene  
400 and polystyrene for climate change impacts. By contrast, co-processing of PET in clinker kilns results  
401 in lower credits than material recycling (-50% for climate change impacts). In this study, only  
402 packaging material was considered, which means pollutant content was relatively low compared to  
403 waste electric and electronic equipment. The inclusion of polluted fractions could substantially  
404 increase toxicity impacts of plastic utilization, e.g. in cement kilns. Direct landfilling of plastic was not  
405 considered based on Swiss regulations (VVEA, 2016).

#### 406 **3.2. LCA of the Swiss MSW management system**

407 Taking a zero burden assumption (burden free reception of waste by the WMS), the environmental  
408 assessment results in credits of -0.6 million tonnes of CO<sub>2</sub>-eq/a and -60 PJ-eq/a, therefore  
409 compensating for 1% of the total climate change impacts and 4% of the CED of total Swiss

410 consumption reported by Frischknecht et al. (2014) and shown in Figure 3. Paper, cardboard and  
411 mixed MSW represent 22%, 9% and 33%, respectively, of the 700 kg MSW per person per year in  
412 2012. Because of the relatively high collection rates for cardboard and paper (89% and 74%,  
413 respectively; Haupt et al. (2017a)), the result in terms of CED is dominated by paper (23%), cardboard  
414 (23%) and the MSWI contribution (38%). Compared to CED, where air emissions are burden-free, air  
415 emissions of the MSWI lead to its dominating role in terms of climate change impacts; MSWI is  
416 responsible for the vast majority of burdens, with paper and cardboard contributing 36% to the credits,  
417 aluminum 27% and biogenic waste 18%.

418 Figure 3 shows the distribution of burdens and benefits for CED, cumulative exergy demand (CExD),  
419 climate change, human toxicity and ecotoxicity, and highlights the shares of the different materials  
420 contributing to the different impact indicators. From these results and the assessment of the ILCD  
421 categories (see details in Supporting Information, section 2.2), four groups of indicators can be  
422 identified. For the first group, consisting of climate change, eutrophication, and photochemical ozone  
423 creation, the burdens related to the MSWI (mainly CO<sub>2</sub> and NO<sub>x</sub> emissions) are dominating the LCA  
424 outcome. Large credits from aluminum recycling occur, as primary production of aluminum releases  
425 substantial amounts of process-dependent CO<sub>2</sub> in the de-oxidization of alumina to aluminum metal.  
426 The eutrophication is substantially influenced by the utilization of biogenic waste from households and  
427 the related substitutions. The second group consists of the CED, CExD and the ozone layer depletion  
428 and is characterized through large credits from MSWI occurring based on the substitution of primary  
429 energy carriers. CED and CExD only account for energy inputs, which leads to lower burdens from the  
430 thermal process. Due to the lower energy demand in paper recycling compared to primary production,  
431 paper and cardboard together make up two thirds of the credits in terms of CED and CExD. The third  
432 group includes impacts related to the ionizing radiation, which are dominated by large credits from  
433 electricity substitution from MSWI plants (due to the high share of nuclear power in Switzerland). The  
434 toxicity related impact categories, ecotoxicity, human toxicity, and carcinogenic and non-carcinogenic  
435 effects, form the fourth group. The results of these impact categories are dominated by recycling  
436 processes, while the thermal treatment of waste plays an insignificant role. From the materially  
437 recycled fractions, metals and fibrous materials are important for both ecotoxicity and human toxicity.  
438 While paper and cardboard production mainly lead to non-carcinogenic emissions, primary metal  
439 production causes heavy metal emissions from the disposal of tailings and ore processing wastewater.



440

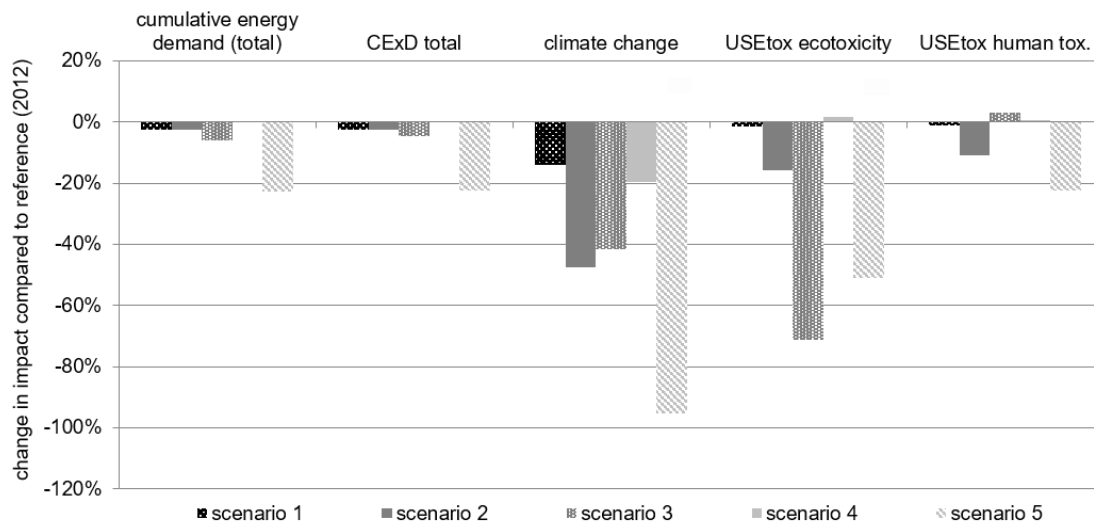
441 **Figure 3:** Positive, negative and net environmental impacts for all recycling fractions in Switzerland in  
 442 2012. The results for the MSW include the treatment of residues from recycling processes. The units  
 443 for the different impact categories are given on the x-axis. The numeric results of the impact  
 444 categories are not comparable with each other but should be interpreted individually.

445 For all categories, the results per tonne of material stream highlighted the large difference in  
 446 environmental impacts between primary and secondary metal resource use. Nevertheless, metal's  
 447 contribution to the overall impacts of the WMS is limited because of the small fraction of metal waste in  
 448 MSW. By contrast, paper, cardboard and glass cause a larger share of overall impact due to their  
 449 sheer mass.

### 450 3.3. Scenario analysis

451 Using the linked MFA and modular LCA model, various scenarios evaluating different waste related  
 452 political measures were calculated for Switzerland. Scenario 5, increasing energy recovery in MSWI,  
 453 shows a large overall environmental benefit (climate change benefit +44%) compared to the current  
 454 Swiss WMS. This is due to the rather low average electricity and heat recovery efficiencies from Swiss  
 455 MSWI (25% for heat and 15% for electricity (Rytec, 2013)) compared to other countries (65% for heat  
 456 and 29% for electricity in Denmark (Fruegaard and Astrup, 2011)). These low efficiencies are due to  
 457 the decentralized structure of the Swiss WMS and the small plant size. Furthermore, the plants are  
 458 often not optimized for energy recovery because of their size and age. Scenario 2 (promotion of

459 closed-loop recycling) also shows an improvement to the current system with a climate change benefit  
 460 of +22%. The increased savings are mostly related to additional paper recycling, which yields larger  
 461 credits than the use of paper fibers in cardboard production. Surprisingly, an increase of open-loop  
 462 recycling for all fractions (scenario 3) also results in an increase of the overall environmental benefit  
 463 (climate change impact +18%) compared to the reference scenario. The main reason is the large  
 464 credit for glass recycling related to the substitution of the insulation materials XPS and primary foam  
 465 glass plates. It is, therefore, not possible to deduce general recommendations for or against open- or  
 466 closed-loop recycling system. Instead, a mix of both, optimized for minimizing environmental impacts,  
 467 would provide the largest benefit. While for paper closed-loop recycling to paper fibers is favorable  
 468 compared to open-loop recycling to cardboard, the production of insulation material from glass cullet  
 469 results in higher credits than recycling to packaging glass. A 5% increase in the recycling rates leads  
 470 to a 7% net increase of climate change credits, highlighting that mere quantitative recycling rate  
 471 increases have a limited improvement potential in countries with already high recycling rates such as  
 472 Switzerland.



473  
 474 **Figure 4:** Results of the scenario assessment for the case study on the Swiss MSW management  
 475 system. The reference system (ref) is used to normalize the impacts for each impact category. The  
 476 following scenarios are shown: (1) increased collection rates by 5% for all source separated fractions  
 477 except plastic, (2) increased closed-loop recycling, (3) increased open-loop recycling, (4) increased  
 478 plastic recycling, and (5) higher energy recovery efficiency in the MSWI.

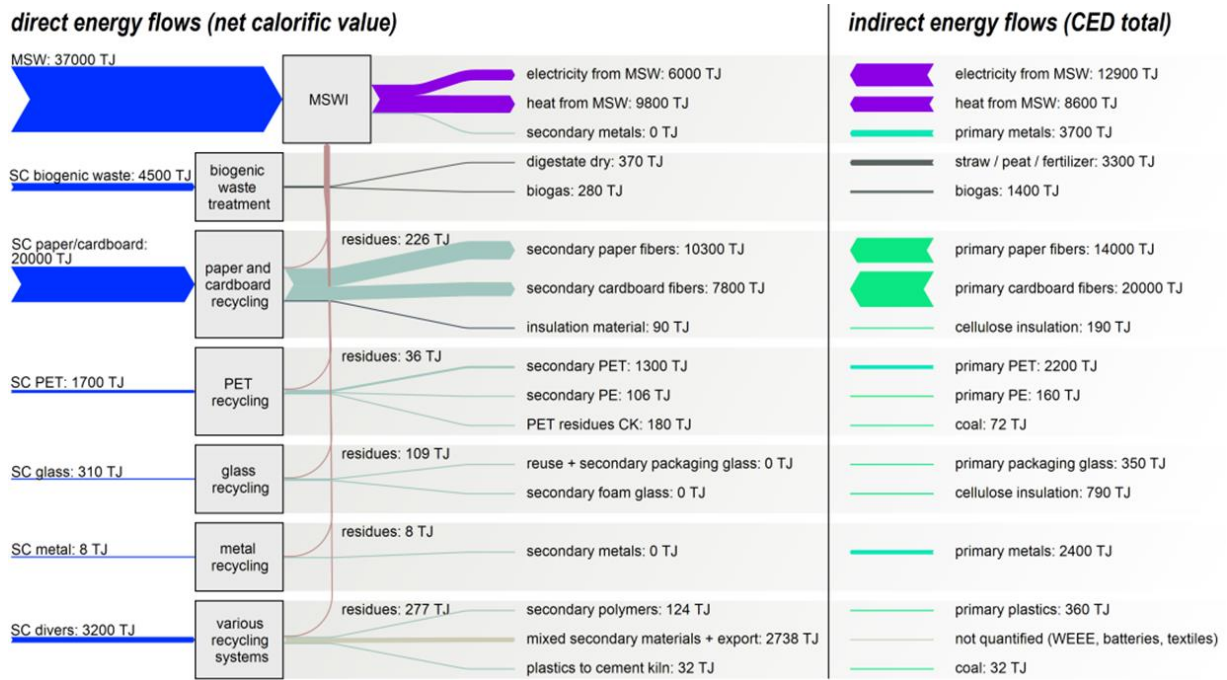
479 Scenario 4 represents a moderate increase of plastic recycling compared to the reference year 2012.

480 The recycling rates were set to Switzerland's 2016 rates (Dinkel et al., 2017). Although the amount of  
481 plastic recycled was doubled, only 6% of the plastic consumed was recycled in 2016. The increase in  
482 recycling rates leads to a 20% improvement in climate change impacts compared to the overall  
483 environmental performance of the Swiss MSW management system in 2012. This is due to the  
484 reduced CO<sub>2</sub> emissions, as less plastic is found in the thermally treated MSW. There was a small  
485 reduction of the CED credits, as the increased plastic recycling rate only reveals marginal credits and  
486 less energy is produced in the MSWI (-1%). With regard to toxicity, no significant improvement could  
487 be found when increasing plastic recycling. Note that if more ambitious increase in recycling rates had  
488 been assumed, the benefits would have been much larger.

### 489 **3.4. Direct and indirect contribution to energy system**

490 The lower heating value (LHV) of all MSW streams of the Swiss WMS summed to 71,000 TJ (Figure 5,  
491 based on Haupt et al. (2017a)). Approximately 40% of MSW, by calorific value, is currently collected in  
492 a source separated manner, and the remaining 60% was treated in waste-to-energy plants. Of the  
493 37,000 TJ contained in thermally treated waste, 10,000 TJ of heat and 6,300 TJ of electricity were  
494 recovered. Taking the substitution of primary heat and electricity into account, the resulting net  
495 cumulative energy benefits, i.e. indirect energy benefits, amount to 30,000 TJ-eq. The indirect savings  
496 from materially recycled fractions from households or utilized biogenic waste from gardens and  
497 kitchens were found to substantially surpass the LHV fractions (LHV: 30,000 TJ, net CED savings  
498 through substitution: 45,000 TJ (neglecting credits from WEEE, battery and textile recycling)). Relative  
499 to its primary production, aluminum shows the highest savings per tonne. Waste paper and cardboard,  
500 however, are the largest waste streams by mass and therefore offer the largest total indirect energy  
501 gain (34,000 TJ, 72% of all indirect net CED gains).





502

503 **Figure 5:** Direct energy flows in the Swiss MSW management system (left) compared to indirect  
 504 energy flows measured as total net cumulative energy demand savings (difference in CED of the  
 505 recycling processes and the related credits for material substitutions (right)). SC = material from  
 506 separate collection.

507 The results are particularly sensitive to the electricity and heat substituted by the recovered energy. In  
 508 the base case (shown in Figure 5) it was assumed that today's mix of energy carriers feeding into  
 509 district heating network was substituted. The assumption that heat production would take place at a  
 510 household level, if no district heating network would exist (resulting in a higher share of oil and gas  
 511 substituted), increased the CED gain of the heat recovery by 50%.

## 512 4. Discussion and Conclusions

513 The use of MFA as a basis of the LCA has enabled for evaluation of the input-dependent assessment  
 514 of emissions and impacts of all recycling processes, the thermal treatment of MSW, and the overall  
 515 impacts of the Swiss WMS. The LCA results are dominated by the impacts from paper and cardboard  
 516 recycling and the thermal treatment of residual waste. Although the environmental benefit per tonne of  
 517 paper and cardboard recycling is relatively low, this fraction dominates the total impact score due to  
 518 their high share of the overall waste produced (30%) and high collection rates (74% and 89%,  
 519 respectively (Haupt et al., 2017a)). The results of the Swiss WMS highlight the need to investigate

520 WMS with reference to the actual material flows in order to prioritize impact-reduction efforts on the  
521 waste flows that will provide the most impactful results.

522 Taking a zero burden assumption, the overall impact results are negative and, therefore, quantify the  
523 environmental benefit of the MSW management system. The results should not lead to the erroneous  
524 conclusion that waste production should be increased.

525 Overall, the WMS was found to compensate for 1% (CO<sub>2</sub>-emissions) and 4% (CED) of the impact of  
526 Swiss consumption. This is a significant contribution that highlights the need to optimally manage  
527 waste as a material and energetic resource.

528 Substitution benefits are much more uncertain than the direct impacts of the recycling and disposal  
529 processes. Therefore, the uncertainty on the net environmental impact is rather high. As argued by  
530 Zink and Geyer (2017), substitution benefits require a market where the recycled material can  
531 substitute primary material . A sensitivity analysis showed that a 20% demand reduction for all  
532 fractions results in a 66% environmental benefit reduction. The uncertainty with regard to substitution  
533 benefits is difficult to quantify due to the large dependence on the global market, consumer patterns  
534 and product development, and this uncertainty should be kept in mind when drawing conclusions from  
535 the LCA. Furthermore, the results are particularly sensitive to the Swiss electricity mix, which  
536 contributed between 13% (USEtox) to 61% (climate change) to the overall environmental impacts,  
537 taking all direct and upstream electricity consumptions into account.

538 While the concept of Circular Economy assumes that closed material cycles are preferred, several  
539 open-loop recycling pathways have been found to be environmentally superior. The substitution of  
540 insulation material with foam glass and recycled paper and cardboard, for example, leads to higher  
541 environmental credits than the production of packaging glass, linerboard or newsprint from recycled  
542 material. For glass, open-loop systems yield higher credits if the polystyrene material XPS is replaced.  
543 If paper and cardboard are used in insulation, fewer fibers are lost in the production of insulation  
544 material compared to the pulping required for closed-loop recycling, leading to higher environmental  
545 benefits. The share of paper and cardboard used as cellulose insulation material has increased in  
546 Switzerland over the last years and might also increase further as paper consumption and, therewith,  
547 the need for waste paper shrinks. The results highlight the need to integrate LCA in the Circular  
548 Economy concept in order to avoid environmentally adverse circles.

549 While the direct energy production from the Swiss WMS can be increased, the findings highlight that  
550 the indirect energy benefits are substantial. Although the indirect energy savings are much higher,  
551 most of the benefits are not local due to the high import share of goods. Therefore, they do not  
552 influence the national energy system as much as the direct energy recovered from incineration plants  
553 or fuel substitutions. Policy makers are therefore advised to take these indirect effects into account, as  
554 reaching the overall optimum requires thinking beyond national boundaries and taking a consumption  
555 footprint perspective (Frischknecht et al., 2014).

556 Several recommendations can be drawn from the scenario analysis. On the one hand, increasing the  
557 energy recovery efficiency of the MSWI is essential to improving system performance. This can be  
558 realized through a relocation of MSWI plants to industrial or urban areas with a high heat demand. On  
559 the other hand, recycling pathways should be optimized with a systems perspective which would allow  
560 for an optimal distribution of recyclables into open- and closed-loop recycling systems.

561 The modular approach chosen in this study allows for flexible evaluation of different mass flow  
562 distributions in the Swiss WMS and, therefore, assessing future waste management strategies. It also  
563 facilitates the integration of new modules (e.g. new technologies, treatments, collection routes, or  
564 WEEE recycling systems), the assessment of the influence of the new modules and their  
565 environmental contribution to the overall system. Also, changes in the waste composition, the  
566 respective treatment options and capacities as described in Meylan et al. (submitted) can be  
567 environmentally assessed. The presented model can, furthermore, be used in a mathematical  
568 optimization by providing the inputs for a linear programming extended matrix based LCA (Heijungs  
569 and Suh, 2002; Saner et al., 2014; Vadenbo et al., 2014a, 2014b). While the chosen attributional  
570 system model allows for modeling the current waste management, the application in optimization and,  
571 ultimately, in decision making requires a consequential perspective taking imposed system changes  
572 into account. The chosen modular structure enables for the formulation of a consequential LCA,  
573 therefore providing a framework for future decision making based on up-to-date LCIs.

## 574 **5. Acknowledgement**

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