Titre: Title:	Substitution modelling in life cycle assessment of municipal solid waste management
Auteurs: Authors:	Stéphanie Viau, Guillaume Majeau-Bettez, Laurent Spreutels, Robert Legros, Manuele Margni, & Réjean Samson
Date:	2020
Type:	Article de revue / Article
Référence: Citation:	Viau, S., Majeau-Bettez, G., Spreutels, L., Legros, R., Margni, M., & Samson, R. (2020). Substitution modelling in life cycle assessment of municipal solid waste management. Waste Management, 102, 795-803. https://doi.org/10.1016/j.wasman.2019.11.042

SUBSTITUTION MODELLING IN LIFE CYCLE ASSESSMENT OF MUNICIPAL SOLID WASTE MANAGEMENT

S. VIAU[1,2], G. MAJEAU-BETTEZ[1,2,3], L. SPREUTELS[1,2], R. LEGROS[1,2], M. MARGNI[4], R. SAMSON[1,2]

[1] Chaire de Recherche sur la Valorisation des Matières Résiduelles (CRVMR), Department of Chemical Engineering, Polytechnique Montréal, Montréal, Canada

[2] CIRAIG, Department of Chemical Engineering, Polytechnique Montréal, Montréal, Canada

[3] Industrial Ecology Programme, Norwegian University of Science and Technology (NTNU), Trondheim, Norway

[4] CIRAIG, Department of Mathematical and Industrial Engineering, Polytechnique Montréal, Montréal, Canada

Abstract

Life cycle assessment (LCA) is gaining importance worldwide in guiding waste management policies. The capacity of co-products such as recycled materials and recovered energy to avoid primary production of equivalent products largely determines the environmental performance of waste treatment technologies. Estimating the reductions in resource use, emissions, and impacts enabled by this substitution of primary production is often the most influential and controversial factor in quantifying the overall environmental performance of a waste management strategy. This study aims to critically evaluate the modelling of substitution in LCAs of recovered material from municipal solid waste management systems (MSWMS) by answering two questions. First, to what extent is substitution modelling transparently documented in the literature? Second, are the substitution ratios justified to represent physically realistic replacement of one product by another? To address these questions, we performed a systematic analysis of 51 LCA studies on MSWMS published in the peer-reviewed literature. We found that 22% of the substitution ratios are only implicitly expressed. A significant proportion of substitution ratios is not justified (65%), while for the remaining 35%, justifications do not represent physically realistic substitutions. We call for more rigor and transparency, and we propose guidance for the documentation of substitution ratios, with the aim of reaching more credible and robust analyses. For the justification of a substitution ratio to be considered physically realistic, information should notably be provided concerning loss of quality, the function performed by substitutable materials, and the sector of use.

Keywords

LCA; substitution; municipal solid waste; modelling; recycling; energy recovery

1 Introduction

Life cycle assessment (LCA) is gaining importance worldwide in guiding waste management policies (European Commission, 2005; Hellweg & Milà i Canals, 2014; LegisQuébec, 2018). It allows policy makers to go beyond mere avoidance of landfilling and directly consider multiple environmental impacts and benefits caused both directly and indirectly by their decisions. Three factors mainly determine the environmental performance of a waste treatment technology:

1. its direct emissions and related impacts;
2. its material and energy needs, as well as the impacts associated with their production chains;
3. the capacity of co-products such as recycled materials and recovered energy to avoid primary production of material or energy and related emissions and impacts.

Quantifying the influence of this last factor is typically performed in LCA through *substitution* modelling (Azapagica & Clift, 1999).

Substitution modelling in MSWMS LCA is best understood within a so-called "consequential perspective", where we strive to capture all the consequences of the implementation of a waste management strategy. The typical rationale for substitution modelling in MSWMS LCA is as follows. Waste treatment and recycling activities are usually understood as having little control over the amount and composition of waste to be treated (Nakamura & Kondo, 2002). The production of secondary materials is then assumed to roughly scale with this amount of waste treated, leading to an inelastic supply of secondary materials (Weidema, 2003). In other words, waste treatment processes are represented as having little control over the amount of secondary materials that they produce, "forcing" these coproducts on the market. Then, assuming a fixed overall demand for products, any increase in this inflexible supply of secondary products must be met by a proportionate reduction in primary production of the competing products. The environmental impacts typically associated with the value chains of these displaced primary productions is then considered to be "avoided" thanks to the waste treatment strategy. The validity and credibility of substitution modelling in LCA therefore hinges on its capacity to capture (1) how the amounts and physical characteristics of waste streams *determine coproduct* flows (Eriksen, Damgaard, Boldrin, & Astrup, 2019); (2) how secondary materials effectively compete as *physical and functional equivalents* to primary materials (Ekvall & Weidema, 2004; Vadenbo, Hellweg, & Astrup, 2017); and (3) how demand levels adjust and *markets respond* to changes in supply (Ekvall & Tillman, 1997).

It has been demonstrated that substitution modelling is often the most influential factor in determining the overall environmental performance of a waste management strategy (Andreasi Bassi, Christensen, & Damgaard, 2017; L Rigamonti, Grosso, & Giugliano, 2010). Given this importance, the complexity of the mechanisms described, and the numerous simplifying assumptions that are typically introduced, it is no surprise that modelling of substitution is the object of numerous debates and constitutes a long-standing challenge for the LCA community (Heijungs & Guinée, 2007; Majeau-Bettez et al., 2017; Weidema, 2000). Challenging the simple rationale for substitution outlined above, Geyer and colleagues (2016), among others (Ekvall & Tillman, 1997; Ekvall & Weidema, 2004; Zink, Geyer, & Startz, 2016), notably urge that price elasticities be taken into account, since demand for the primary commodity does not necessarily decline because of an increased offer for the secondary commodity. Consumers also have a say in whether they have the willingness and the ability to buy the secondary product (Zink et al., 2016). In short, it remains unclear to what extent LCAs follow best practice in modelling substitution.

Deploring a lack of clarity and transparency in the communication of substitution modelling in LCA, Vadenbo and colleagues (2017) proposed a framework that decompose the substitution potential of a MSWS into four multiplicative parameters, as shown in equation 1. This substitution potential (γ) is defined as the amount of primary material potentially avoided per functional unit of treatment (for example, the treatment of 1 ton of paper to be recycled).

$$\gamma = U^{\text{rec}} * \eta^{\text{rec}} * \alpha^{\text{rec:disp}} * \pi^{\text{disp}} \tag{1}$$

The term U^{rec} is the physical resource potential of the waste stream, and it represents the amount of potentially recoverable material available in the waste stream. For example, in the case of processing 1 ton of waste paper, a U^{rec} of 0.840 would indicate the presence of 840 kg of potentially recoverable paper, and 160 kg of contaminants and irrecuperable material. The term η^{rec} is the resource recovery/recycling efficiency that represents conversion efficiency. In the example, η^{rec} = 94% means that the recycling process has an efficiency of 94% and has a loss of 6% during the recycling process. The term $\alpha^{\text{rec:disp}}$ is the substitutability, a.k.a. the substitution ratio, as it represents the functional relationship that exists between two intersubstitutable materials. According to Vadenbo et al. (2017), it can be influenced by technological constraints, institutional constraints or user-perceived constraints. In the example $\alpha^{\text{rec:disp}}$ = 0.9 kg primary paper/kg secondary paper, which means that 1 kg of secondary paper is equivalent to 0.9 kg of primary paper in order to fulfill the same function. Finally, π^{dis} defines the market response. It indicates the expected changes in the level of market activity of the system of the substituted primary product. It is expressed as a percentage and it is set to π^{dis} = 100% for the example which means that the primary paper substituted is the marginal supply. In this way, a substitution potential γ of 711 kg primary paper/ton of paper to be recycled is calculated.

It is important to stress here that the substitution *potential* (γ) differs from the substitution *ratio* (α). The latter is a multiplicative factor in calculating the former. The substitution potential represents the overall amount of primary production potentially substituted per unit of waste treated; it is therefore expressed in relation with the functional unit under study and takes into account multiple factors, such as process efficiency, etc. In contrast, the substitution ratio aims, as Vadenbo et al. (2017) stated, "to designate the degree of functional equivalence between alternative resource/products for specific end use". For example, let us consider the electricity generation by the incineration of waste-to-energy (WtE) facility. This produced electricity is indistinguishable from the electricity that comes from the grid mix. Thus, 1 kilowatt-hour (kWh) of electricity generated by WtE avoids the production of 1 kWh of electricity from the grid mix. In this way, the substitution ratio is 1 kWh: 1 kWh. This ratio may not equal 1:1 when the two substitutable products differ in their nature (compost replacing mineral fertilizer) or in their levels of purity and quality (e.g. damaged recycled fibers substitution primary wood fibers). Thus, a substitution ratio will be less than 1:1 when a larger amount of the secondary materials is required to fulfill the same function as a given amount of primary material.

Establishing a standard approach to estimate substitution ratios has proved challenging, since most materials simultaneously involve multiple properties in the fulfillment of multiple functions. For example, the diverse use of plastics throughout the economy is not explained by a single property enabling a single function. Thus, Ekvall and Weidema (2004) stressed that two materials must share some obligatory properties to be substitutable, although Heijung and Guinnée (2007) pointed out that these criteria are difficult to determine quantitatively in practice. When comparing materials differing quality and composition, Werner and Richter (2000) suggested the use of a "value-corrected substitution ratio" based on the price difference between primary material and secondary material. Thus, establishing the level of functional equivalence of secondary and virgin materials, and translating this intersubstitutability in a simple substitution ratio, remains a particularly controversial step in the border estimation of the overall substitution potential of MSWMS.

2 Aim and Scope

LCA practitioners generally apply simplifying modelling assumptions based on their understanding of secondary material markets, availability of data, and research resources. to determine the substitution potential of recycled resources. The aim of this study is to critically evaluate the modelling of substitution in LCAs of municipal solid waste management systems (MSWMS) published in peer-reviewed literature. This study attempts to answer two questions. The first question is: To what extent are substitution potential transparently documented in the literature? Specifically, we will strive to determine to what extent the different contributing factors identified by Vadenbo et al. (2017) are explicitly disclosed and justified by LCA practitioners in a wide sample of waste management LCAs. The second question is: Are the substitution ratios set out in the literature justified to be physically realistic? In other words, are the different substitution ratios explicitly based on an assessment of the physical characteristics of secondary materials, and their effects on these materials' functional equivalence with primary materials within various sectors of use?

To answer the first question, we made a systematic literature review on LCAs of MSWMS. To better focus on municipal solid waste, i.e. post-consumer waste, we excluded the treatment of industrial waste, notably "new scrap", from this review. Moreover, specialized LCA on e-waste and wastewater management are beyond the scope of this study. To answer the second question, we made a thorough analysis of the substitution ratios and their accompanying justifications.

3 Methodology

To perform the literature review, we searched for the keywords "life cycle assessment", "substitution", "system expansion", "recycling", "municipal solid waste management" in the Compendex and Inspec databases (Elsevier, 2019). We selected only articles published in English in peer-reviewed journals between the years 2000 and 2017.

Research in the literature with the keywords detailed previously yielded 188 scientific articles between the years 2000 and 2017. Several rejection criteria were applied to streamline the study. First, many articles used allocation (a.k.a. partition) methods, splitting emissions across all functional flows proportionately to a common property (monetary value, mass, etc.), rather than modeling the substitution of primary materials. Since we specifically focus on substitution, these allocation methods are outside the scope of the study, and we rejected these studies. Second, as announced in the title, we focus exclusively on LCAs of municipal solid waste; and consequently, LCAs on wastewater treatment and e-waste were excluded. Finally, we only kept peer-reviewed journal articles; and conference articles and technical reports were not retained. Applying these rejection criteria, 137 papers were rejected leaving only 51 available for our analysis.

For each reviewed article, we determined whether the 4 parameters expressed by Vadenbo and colleagues (2017) were explicitly mentioned. The survey is performed using a table (see Table S1 in the supporting information) where we also note the context of use of each identified parameter. The value of the parameter, its units, and any restriction to its applicability are also noted. A distinction is made between the waste material input for treatment (from the functional unit), the substituent secondary material, and the substituted primary material. It is also noted whether the substitution ratios are stated implicitly or explicitly. An implicit ratio is expressed for example as follows: "The electricity production is then assumed to be the electricity grid in Finland" (Hupponen, Grönman, & Horttanainen, 2015). The value of the ratio is not explicitly stated as 1: 1 but is implicit.

In parallel, when a substitution ratio is listed, we have noted if a justification is provided to the ratio and what type of justification it is. We determined that a substitution ratio is unjustified when (1) it is implicitly stated, (2) it is mentioned with no explanation, or (3) it is justified solely with a citation without recontextualization.

4 Results

4.1 Survey of parameters υ, η, α, and π[1]

The detailed characteristics of coefficients used to model the substitution in the 51 analyzed papers are presented in Table S1 of the supporting information. Within this body of literature, 25% of articles mention only one of the four parameters υ, η, α, and π; 39% of the reviewed articles mention at least two parameters; 33% mention at least three parameters and none of them make explicit mention of all four coefficients.

The substitution ratio (α) is the most frequently mentioned coefficient, in 100% of selected articles (see Figure 1). The 2% portion in red on Figure 1 represents substitution ratios considered invalid in the rest of our analysis. A ratio is deemed invalid if: (1) the substitution was not made at the appropriate "point of substitution", or (2) it could not be determined to which measurement unit the multiplicative ratio should be applied. This first case occurs when, for example, recycled paper use is modelled as directly avoiding wood extraction used in the primary's manufacture paper. In this way, all downstream impacts of wood extraction, i.e., the processes of converting wood into paper, are not included in the calculation of avoided environmental impacts. Also, avoided wood extraction and secondary paper are not functionally equivalent and cannot be

---

[1] For simplification we assigned new variables to the parameters. The physical resource potential U^{rec} becomes υ, the resource recovery efficiency η^{rec} becomes η, the substitutability $\alpha^{rec:disp}$ becomes α, and the market response π^{disp} becomes π.

substitutable. Another example that demonstrates the application of substitution to the wrong "point of substitution" may be when a waste substitutes a primary material directly. A waste material will always be sorted, split, decomposed or transformed prior to competing with virgin materials. Thus, it is an error to consider waste as a substitute material. The absence of units in the second case prevents the reader from knowing the basis of comparison used: does a substitution ratio α mean that α kg will replace 1 kg of primary material, or is it rather that α m^3 will replace 1 m^3 of primary product, or else is the substitutability on a per energy content basis, etc.? The interpretation of the 51 selected articles made it possible to identify 148 recoverable resource potential values (υ), 97 conversion efficiency values (η), 267 substitutability or substitution ratio values (α), and no value for the market response (π).

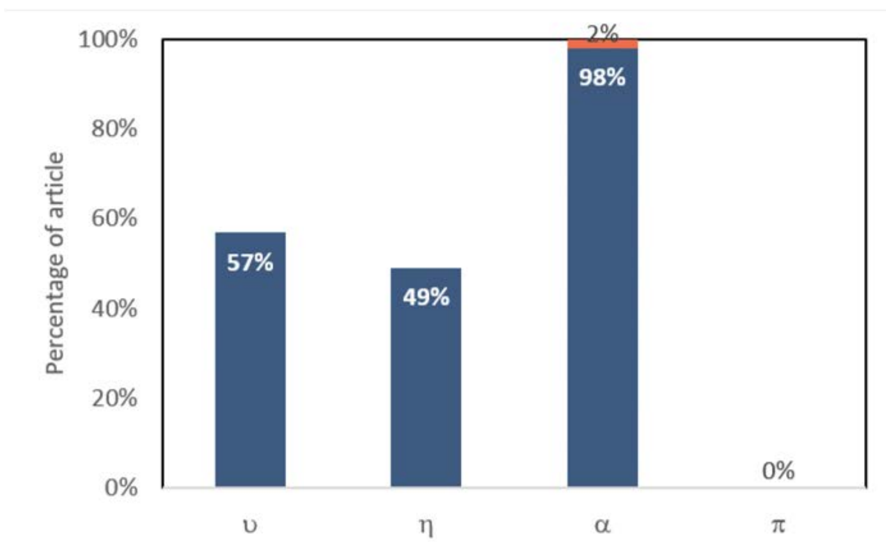

Figure 1: Proportions in which the parameters υ, η, α and π are represent in the literature (υ: physical resource potential; η: resource recovery/recycling efficiency; α: substitutability; π: market response)

Resource recovery efficiency (η) coefficients are present in 49% of articles. The physical resource potential (υ) is present in 57% of the studied articles. This resource potential is sometimes expressed in energy content: the biogas potential for anaerobic digestion and the low-heating value (LHV) of waste for incineration, in mineral content (nitrogen, potassium, and phosphorus) for composting and anaerobic digestion, and in waste composition for material recycling. None of the 51 analyzed articles expressed value for the market response coefficient (π). According to Vadenbo et al. (2017), this coefficient should be equal to 100% if the substituted technology represents the marginal technology. Although some articles mentioned marginal technology only with electricity substitution, none of them mentioned a value near to market response.

4.2 Analysis on the substitution ratio (α)

Figure 2 shows, by material category, the percentage of substitution ratios that are either equal to 1:1 or are different from 1:1. Figure 2 also shows the proportion of substitution ratios that are implicitly equal to 1:1 (hatched part). Our analysis revealed that 68% of the substitution ratios identified have a value of 1:1. Except for fuel (kg), fuel (MJ), paper (kg) and plastic (kg), all material categories have the majority of their ratios equal to 1:1.

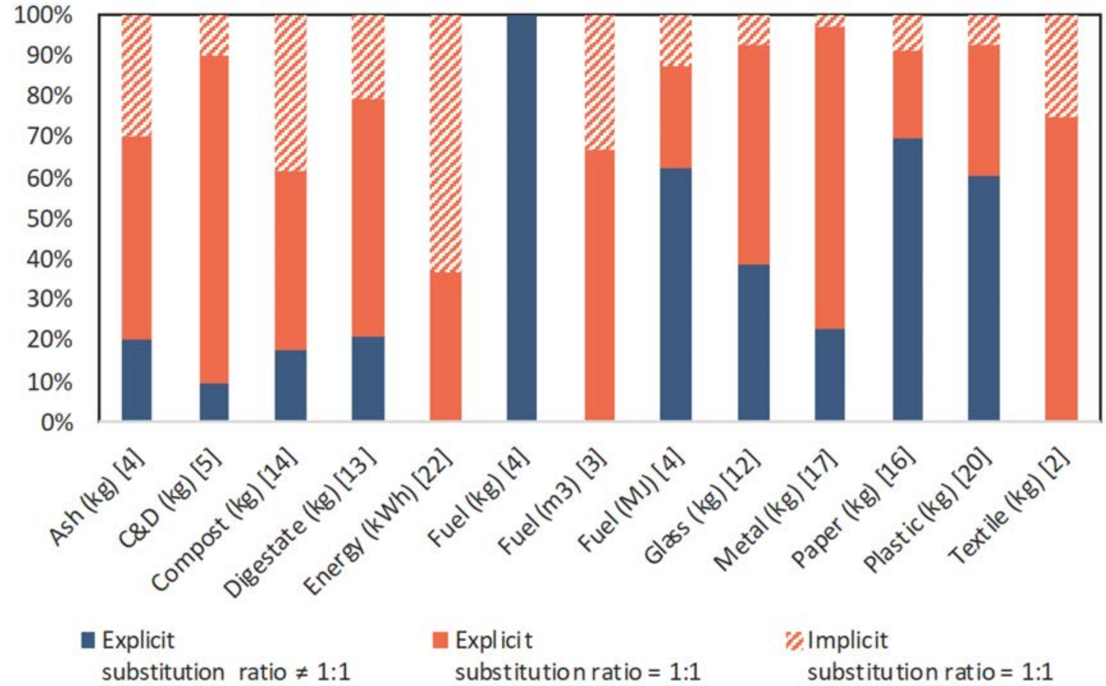

Figure 2: By category of waste, proportion of the substitution ratios expressed implicitly and explicitly, and proportion of substitution ratios equal and not to 1:1 which are explicitly. Number in hook represents the number of articles listed by categories.

The disaggregation of the material categories from Figure 2 allows for the representation of substituting-substituted pairs, resulting in 79 pairs. Substituting-substituted pairs indicates the match between two materials: a secondary material, which is the substituting one and a primary material, which is the substituted one. Of these pairs, 55 were encountered less than three times in the present literature review and do not allow for a dispersion analysis of their substitution ratios. Figure 3 shows the ranges of distribution of substitution ratio values for substituting-substituted pairs with over six identified values[2].

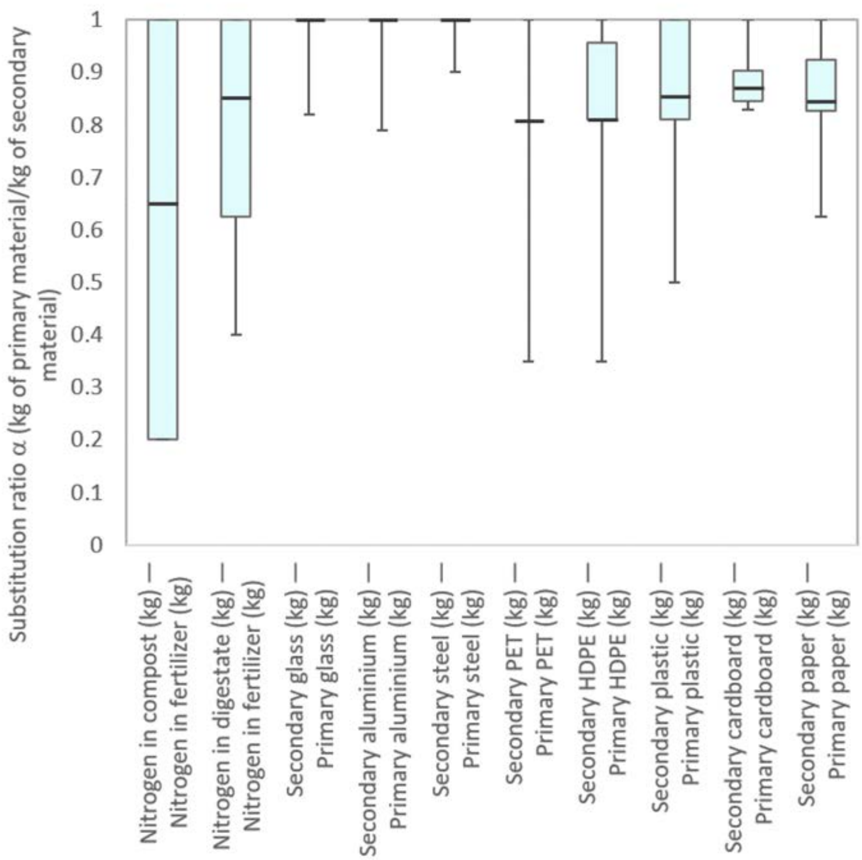

Figure 3: By substituting-substituted pairs, distribution of substitution ratio values for categories with more than six listed values. The lines of the boxes represent the first quartile, the median, and the third quartile; the error bars represent the minimum and maximum values for each distribution.

---

[2] To simplify the reading of figure 3, the substituting-substituted pairs all having substitution values equal to 1 have not been presented (See Table S2 in SI for a complete table of all 79 ranges of distribution).

The survey of substitution ratios illustrated that the substitution of nitrogen in conventional fertilizers by nitrogen in compost has the largest value range with a dispersion coefficient of 0.8 (i.e. a difference of 0.8 kg of nitrogen in fertilizer/kg of nitrogen in compost between the first quartile and the third quartile). The nitrogen in compost and digestate is predominantly present in organic form (Amlinger, Götz, Dreher, Geszti, & Weissteiner, 2003; Bernstad & la Cour Jansen, 2011). However, plants need nitrogen in mineral form, i.e., in the form of ammonia and nitrate to grow. In this way, it is essential to make a distinction while establishing the substitution ratio between organic and inorganic nitrogen present in the substituting compost or digestate and the substituted fertilizer.

From Figure 3, the distribution ranges for glass, aluminum and steel substitution ratios are concentrated towards 1. Almost all the authors have hypothesized that there was no change in inherent properties when recycling these materials (see Figure 5). They are thus assumed to be infinitely recyclable, with their properties unaltered by recycling. However, the presence of a contaminant in metals can significantly affect the ability of recycled metals to substitute primary metals (Reck & Graedel, 2012). Also, the glass containers must be well separated by type and color to avoid altering the properties of the secondary glass (Dyer, 2014).

For polyethylene terephthalate (PET), the distribution range of substitution ratios is centered around 1:0.8. The majority of the analyzed articles based their estimates on the price variation between the primary PET and the secondary PET to evaluate the substitution ratio between the two materials. This substitution method is named "value-corrected substitution ratio" (Werner & Richter, 2000). It is also used to determine the substitution ratio between primary HDPE and secondary HDPE.

The distribution ranges for paper and cardboard are relatively small, with dispersion coefficients less than 0.1, which means that all of the substitution ratio surveyed had similar values. This low value can be explained by the fact that paper and cardboard are not infinitely recyclable; they can be recycled approximately five times before they are no longer of good quality and are eliminated (USEPA, 2016). In this way, the possibilities of substitution ratio are limited since there is a loss of quality at each recycling loop and the number of loops is limited to five. This explains why the third quartile of the range of distribution of these two material categories is less than 1:1.

4.3 Evaluating the physical realism of the justification of the substitution ratios

By definition, the substitutability between two materials is related to the primary function that each of them performs. Thus, a realistic justification for a substitution ratio should make it possible to compare the difference in functionality between the two objects. Surprisingly, 65% of the substitution ratios sampled in the present literature review were not supported by any justification, let alone by a physically realistic one. Figure 4 breaks down this proportion in terms of the different categories of residual materials. This part of the analysis focusses on the functional equivalence of secondary *materials*, with the exclusion of recovered *energy* flows, such as heat and electricity, whose functionality depends on comparatively few intrinsic physical characteristics.

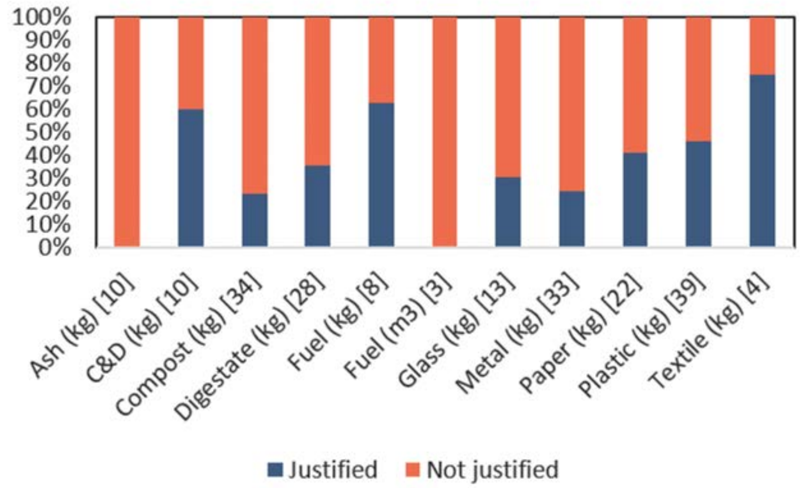

Figure 4: By category of residual materials, proportions of substitution ratios justified and not justified by the authors of LCAs studies. Number between brackets represents the number of articles listed by categories.

For the remaining 35% of substitution ratios that are justified by their authors, Figure 5 shows the different proportions of each type of the justification by category of residual materials.

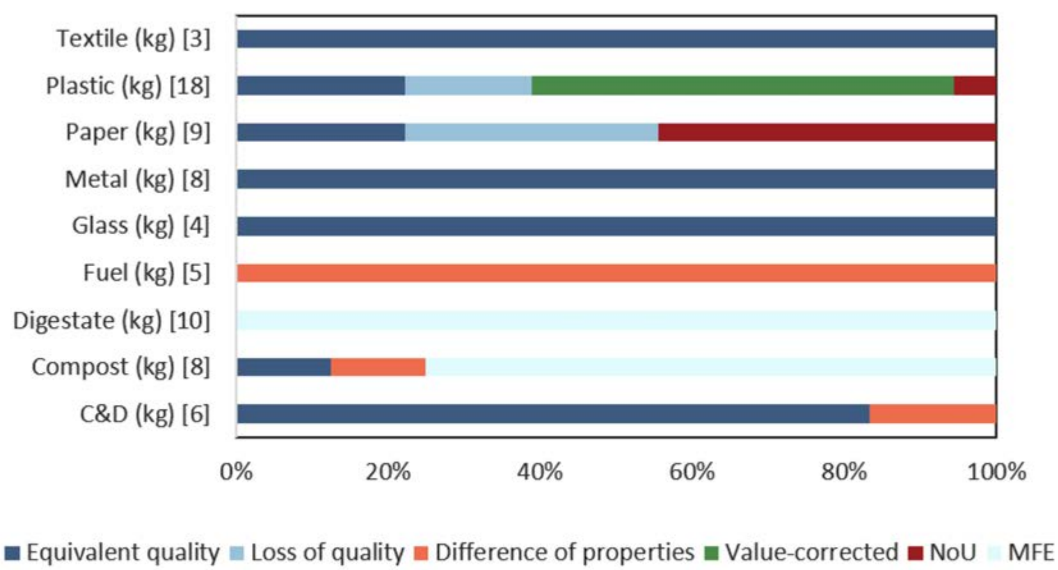

Figure 5: By category of residual materials, proportions of the different justifications used for justified substitution ratios. Number between brackets represents the number of articles listed by categories. (NoU: Number of uses, MFE: Mineral fertilizer equivalent)

Six types of justification were identified: (1) Equivalent quality: there is no loss of quality so that materials can substitute in a 1:1 ratio; (2) Loss of quality: there is a qualitative difference in quality between the secondary and the primary material; (3) Difference of properties: there is a quantitative difference in inherent properties between the secondary and the primary material; (4) Value-corrected: There is a difference in the price between the secondary and primary material; (5) Number of uses (NoU): The materials are recycled in a cascade and cannot be recycled an unlimited amount of cycles; (6) Mineral fertilizer equivalent (MFE): The MFE value is used for compost and digestate substitution with conventional fertilizer.

From Figure 5, 100% of justified ratios for glass and metals used quality equivalence as justification. In all cases, the authors used the assumption that there is no change in the inherent properties of the materials during the recycling process. For C&D and textile substitution ratios, the authors only mentioned that primary and secondary materials have the same properties or that both materials are of the same quality.

Loss-of-quality justifications have a qualitative aspect rather than a quantitative aspect. The authors use this justification to specify that the value of the substitution ratio is different from 1 because of a loss of quality, but they do not use a quantitative method to determine this value. By using the loss of quality as an argument, some authors of the studies seek to penalize from an environmental point of view the secondary material since it is not of the same quality as the primary matter. This justification is used in 33% of cases for paper substitution and almost 20% of cases for substitution of plastics. We must stress that such an approach confers a different meaning to substitution ratio, in a very significant way. For the authors of such studies, a coefficient of 0.5 clearly does not indicate that 200 grams of secondary plastic are necessary in order to achieve the same properties (structural, etc.) as 100 g of virgin material, for example. This puts forth a different understanding of "functional equivalence" whose interpretation in terms of effectively avoided primary productions and impacts is less transparent.

The difference-in-properties-between-materials justification is almost only used to explain the substitutability between different fuels. This substitutability is calculated using the difference between heat values since it directly relates them to the fuel mass. To get the same amount of energy from both fuels, a higher or lower amount of alternative fuel is needed. In Figure 5, about 10% of C&D and compost substitutions used this type of justification. In these two cases, the difference in density with the substituted product explains the non-equality in the substitution.

The value-corrected substitution ratio method is based on the price difference between the primary and the secondary material to determine the substitutability between the two materials (Werner & Richter, 2000). The assumption behind this method is that the price difference reflects the difference in quality. This method of justification is used to justify more than half of the substitution ratios of plastics. The value-corrected substitution ratio serves as a proxy to determine the difference in quality between two materials. It is not founded on a required amount to physically achieve the same properties, but it rather tries to establish a rough correlation between quality (or functionality) and price. As such, we cannot consider this method as a way to physically justify a substitution ratio.

The number of use (NoU) method has been developed to quantify the allocation of the environmental impact of materials that have a limited number of recycling loop, like paper. It splits an initial production impact on all future use phases of a material. However, according to ISO/TR 14049, this method is an allocation rule and not a substitution method (ISO, 2000). When using it to determine substitution ratios between secondary paper and primary paper (almost half of the ratios identified), LCAs authors apply it as a proxy to assess the loss of quality. Note also that this approximation is used to justify a little less than 10% of substitution ratios between plastics.

The sixth justification identified is the mineral fertilizer equivalent value (MFE) in substitution of compost and digestate with conventional fertilizers. This justification is a specific case of the difference-in-properties-between-materials and can only be applied to the substitution of digestate or compost. The MFE indicates the amount of nutrient available in the digestate and compost that is in assimilable form for plants and thus allows to quantify the substitutability compared to mineral fertilizers.

5 Discussion

5.1 Limitations of the analysis

Our analysis suffers from two central limitations. First, it is oriented toward the management of post-consumption, municipal solid waste, not the recycling within industrial sectors, such as the treatment of new scrap. These material cycles are very different with respect to material purity and market dynamics and are better analyzed separately. Our restricted scope lead to a reduced sample size for substituting-substituted pairs (Figure 3) which limit the conclusions we can draw. Nevertheless, we were able to evaluate the transparency in the application of the substitution ratio across material categories.

Second, the search for coefficients was done only in the published peer-reviewed literature on LCAs for waste management. LCAs for complete life cycles of specific materials were not considered. For example, we did not look into cradle-to-grave LCAs of steel, even though these LCAs could consider waste management of steel. It would therefore have been possible to get additional substitution coefficients by studying LCA on the entire life cycle of steel. We also didn't review the whole body of grey literature consisting in governmental reports, consultant studies, etc. Nevertheless, despite the limited scope of the review to the context of MSWM, our study already provided a clear overview of the situation.

5.2 Lack of transparency and rigor

The results presented in this analysis prove a general lack of transparency in the LCA modelling of waste management activities and the associated substitutions. First, since the physical resource potential coefficient of the waste stream (υ) and the conversion efficiency of the treatment technology (η) are disclosed in less than 60% and 50% of studies, respectively, it becomes very difficult to recontextualize the findings and to evaluate whether similar substitution benefits may be expected in other situations. Indeed, the purity of the waste stream and the efficiency of treatment processes are known to vary significantly and directly impact the environmental performance.

Second, we found significant proportions of substitution ratios that are not explicit (22%) or not justified (65%). It is beyond question that this factor can be decisive in determining the overall environmental performance of a waste management system. Consequently, such a lack of transparency on this coefficient significantly affects the credibility and legitimacy of the LCA results. If a substitution ratio is assumed, and no justification is provided, it becomes imperative to do a sensitivity analysis to test whether these assumptions significantly affect the results. It should be reminded that these studies frequently serve as a decision support tool to guide environmental policies, and decision-makers should be aware of the assumptions and limitations of these.

Third, in 4% of studies overall, which represent 13% of studies on paper recycling and 13% of studies on metal recycling, we found a recurrent conceptual error concerning substitution. As mentioned previously, substitution takes place between two functionally equivalent products or at least those competing on the same market. There is, therefore, a conceptual error when the secondary material that is produced *directly* avoids not a competing product but the extraction of raw material such as wood or ores. These errors may have a significant impact on LCA results, as they leave out potentially impactful transformation and refinement processes. Also, we have found in 14% of all substitution ratios a low level of detail in the definition of materials. For example, there is a distinction to be made between the different grades of paper. Newspaper is not equivalent to graphic paper, and they do not have the same recycling process. It is therefore essential to make this distinction concerning substitution to avoid having secondary paper that substitutes primary paper without further detail on the grade of paper.

Although substitution modelling is inevitably based on simplifying assumptions, we must emphasize the importance of communicating these assumptions with rigor and transparency, to support credible and useful results. The credibility and the robustness of results are already seen to increase simply with the disclosure of the ratios of substitution, the nature of the treatment technology under study with its coefficient of efficiency, and the level of detail in the material's definition. Of course, everything can and should be validated with a sensitivity analysis.

5.3 More than functional equivalence

In response to our second question, we found that none of the studies analyzed defended its substitution ratio in terms of a physical ratio of the quantities needed to perform a specific task or function. In particular, a good physical justification should include the function performed by substitutable materials, the information in relation to loss of quality by associating quality with intrinsic properties, and the sector of use.

First, the ability to substitute is not only determined by equivalence in the functional unit. Ekvall and Weidema (2004) mention obligatory properties that take into account several aspects, including the aesthetic aspect, the image, the technical aspect, etc. In their framework, Vadenbo et al. (2017) propose three types of substitutability, one influenced by the technical aspect, another by the institutional aspect, and the last by user-perceived constraints.

Second, when taking into account all aspects mentioned by Ekvall and Weidema (2004) and by Vadenbo et al. (2017) the sector of use of the secondary material must be taken into account in the assessment of its capacity to substitute primary materials. The sector of use is determined by the choice of the processing technology. A simple example is the case of plastic recycling. The mechanical recycling of plastic, a widely used technology, does not make it possible to get a grade of sufficient quality to be reintegrated into the food industry. This technology does not remove additives that give color to plastic, for example. A mixture of mechanically recycled polyethylene terephthalate (PET) containers will lead to PET granules whose color is about gray-black. These granules cannot be used in the manufacture of transparent water bottles. The gray-black color thus becomes a physical property to be taken into account to decide the sector of use of the recovered secondary material. This physical aspect is not negligible in the determination of the substitution’s potential. This physical aspect should be taken into account on a sound understanding of transformation processes. The LCA practitioner must question whether, and for what purpose, mechanically recycled PET is really equivalent to virgin PET. In the example stated above, considering the sector of use, the food industry, the color of recycled PET implies that the substitution ratio (α) will be 0 compared to virgin PET. In another case, recycled PET can perfectly substitute virgin PET in the fleeces industry which result in substitution ratio equal to 1.

In regard to all these aspects, we propose to the LCA community that a good physical justification for substitution ratio, in the context of material recycling be stated as:

> Because of the loss of quality q caused by the transformation of the secondary material, it is necessary to incorporate an additional x kg of this secondary material when it is used for purpose p in order to achieve the same functionality as 1 kg of virgin material, leading to a substitution ratio of $\alpha^p = \frac{1}{1+x}$.

Here again, we know of the efforts needed to obtain this justification, but we must stress the importance of bringing more rigor and transparency, with the aim of having credible and robust results. Moreover, it also constitutes a first step toward an end of the use of generic substitution ratios that can be found the many LCAs we reviewed. Taking the time to justify the substitution ratios in terms of the qualities that matter for a given purpose will help consolidate our community's understanding of the transformation processes, recycling value chain, and their role in facilitating to a circular economy where the value of materials is preserved.

By better capturing the consequences of loss of quality in secondary products, our proposal will lead to a better documentation of the effects of “downcycling” on the environmental performance of MSWMS. The consequences of downcycling-induced quality loss cannot be captured by substitution ratios, since the modelling of substitution as presented in this article is based on relatively short-term considerations. Downcycling may have long-term system-wide consequence, such as a lock-in situation with important stocks of low-grade materials, contaminant accumulation over multiples recycling loops, etc. (Eriksen et al., 2019). We believe that these complex dynamics go beyond mere substitution modelling, and that these issues can and should be tackling through a close integration of LCA and material flow analysis (MFA). However, further research is needed on the operationalization of the combination MFA-LCA.

6 Conclusion

This article demonstrates that there is a lack of transparency and rigor in LCAs of municipal solid waste management with respect to the evaluation of substitution potentials. It is particularly problematic that a majority of studies do not communicate the observed or assumed efficiencies (h) of waste treatment processes, hindering inter-study comparisons and the critical evaluation of assumption. Our analysis further reveals that 22% of the studies consulted do not mention an explicit substitution ratio between secondary and primary products, and that 65% of the explicit ratios are not justified by the authors of the LCA studies. We noted that six types of justification are used and only three of them are based in one way or another on the physical properties of the materials. We also found that there appears to be a misunderstanding of the substitution concept.

Some authors put the substitution between the secondary product formed and the raw material necessary for producing the substituted primary product.

This lack of rigor and understanding undermines the credibility of LCAs results, which are often used as a decision support tool by government authorities. We understand the complexity of determining substitution ratios and that knowledge of a lot of information is needed such as market knowledge, intrinsic properties, use sectors, and processing technology to name a few. Nevertheless, we believe that it is possible with a limited effort to add a minimum of rigor in LCA studies applying substitution by justifying the assumption of the substitution ratios in respect to the main function of the two intersubtituting materials, the use sector, the losses quality caused by the process of transformation and the amount of additional secondary material to achieve the same functionality as the primary material.

7 Acknowledgments

The authors wish to thank the CRVMR partners: City of Gatineau, City of Laval, City of Montreal and Recyc-Quebec, and the International Life Cycle Chair (ILCC) partners for their financial support to this study.

8 Appendix A

Table 1 : List of reviewed articles

Reference	Waste types
(Al-Salem, Evangelisti, & Lettieri, 2014)	MSW[3]

[3] MSW = Municipal Solid Waste

(Andreasi Bassi et al., 2017)	MSW
(Arena, Mastellone, Perugini, & Clift, 2004)	Paper and board packaging waste
(Arena, Mastellone, & Perugini, 2003)	Plastic packaging waste
(Astrup, Fruergaard, & Christensen, 2009)	Plastic waste
(Bernstad & la Cour Jansen, 2011)	Organic waste
(Bernstad Saraiva, Souza, & Valle, 2017)	Organic fraction from MSW
(Birgisdóttir, Bhander, Hauschild, & Christensen, 2007)	MSW incineration residues
(Bueno, Latasa, & Lozano, 2015)	MSW
(Butera, Christensen, & Astrup, 2015)	C&DW[4]
(Carlsson, Naroznova, Moller, Scheutz, & Lagerkvist, 2015)	Food waste
(Cossu, Garbo, Girotto, Simion, & Pivato, 2017)	Plastic waste
(Cremiato, Mastellone, Tagliaferri, Zaccariello, & Lettieri, 2018)	MSW
(Damgaard, Riber, Fruergaard, Hulgaard, & Christensen, 2010)	MSW
(Di Maria & Micale, 2014)	MSW
(Di Maria, Micale, Contini, & Morettini, 2016)	Organic fraction from MSW
(O. Eriksson et al., 2005)	MSW
(Ola Eriksson, Finnveden, Ekvall, & Björklund, 2007)	MSW + C&DW
(Feraldi, Cashman, Huff, & Raahauge, 2013)	Scrap tires
(Finnveden, Johansson, Lind, & Moberg, 2005)	MSW
(Fruergaard & Astrup, 2011)	Organic waste + SRF[5]
(Fyffe, Breckel, Townsend, & Webber, 2016)	SRF
(Galgani, van der Voet, & Korevaar, 2014)	Organic fraction from MSW
(Gentil, Clavreul, & Christensen, 2009)	MSW
(Ghose, Pizzol, & McLaren, 2017)	C&DW
(Hossain, Wu, & Poon, 2017)	C&DW
(Hupponen et al., 2015)	MSW
(Jensen, Møller, & Scheutz, 2016)	Organic waste
(Laner, Rechberger, De Soete, De Meester, & Astrup, 2015)	MSW

[4] C&DW = Construction and Demolition Waste

[5] SRF = Solid Derived Fuel

(Larsen, Merrild, & Christensen, 2009)	Glass waste
(Lombardi & Carnevale, 2018)	MSW
(Mah, Fujiwara, & Ho, 2017)	C&DW
(Margallo, Aldaco, & Irabien, 2014)	Bottom ash
(Morris, 2017)	C&DW
(Parkes, Lettieri, & Bogle, 2015)	MSW
(Pires, Chang, & Martinho, 2011)	MSW
(Poeschl, Ward, & Owende, 2012)	Organic waste
(Reza, Soltani, Ruparathna, Sadiq, & Hewage, 2013)	RDF[6]
(L. Rigamonti et al., 2014)	Plastic fraction in MSW
(L Rigamonti et al., 2010)	MSW
(Lucia Rigamonti, Grosso, & Sunseri, 2009)	MSW
(Ripa, Fiorentino, Giani, Clausen, & Ulgiati, 2017)	MSW
(Sevigné-Itoiz, Gasol, Rieradevall, & Gabarrell, 2015a)	Plastic waste
(Sevigné-Itoiz, Gasol, Rieradevall, & Gabarrell, 2015b)	Paper waste
(Tagliaferri et al., 2016)	MSW
(Tunesi, Baroni, & Boarini, 2016)	MSW
(Van Ewijk, Stegemann, & Ekins, 2017)	Paper waste
(Yoshida, Gable, & Park, 2012)	Organic waste
(Zamani, Svanström, Peters, & Rydberg, 2015)	Textile waste
(Zhao, Wang, Lu, Damgaard, & Christensen, 2009)	MSW
(Zhao, Xing, Lu, Zhang, & Christensen, 2012)	MSW

9 References

Al-Salem, S. M., Evangelisti, S., & Lettieri, P. (2014). Life cycle assessment of alternative technologies for municipal solid waste and plastic solid waste management in the Greater London area. *Chemical Engineering Journal*, *244*, 391–

[6] RDF = Refuse Derived Fuel

402. https://doi.org/10.1016/j.cej.2014.01.066

Amlinger, F., Götz, B., Dreher, P., Geszti, J., & Weissteiner, C. (2003). Nitrogen in biowaste and yard waste compost: Dynamics of mobilisation and availability - A review. *European Journal of Soil Biology*, *39*(3), 107–116. https://doi.org/10.1016/S1164-5563(03)00026-8

Andreasi Bassi, S., Christensen, T. H., & Damgaard, A. (2017). Environmental performance of household waste management in Europe - An example of 7 countries. *Waste Management*, *69*, 545–557. https://doi.org/10.1016/j.wasman.2017.07.042

Arena, U., Mastellone, M. L., Perugini, F., & Clift, R. (2004). Environmental assessment of paper waste management options by means of LCA methodology. *Industrial and Engineering Chemistry Research*, *43*(18), 5702–5714. https://doi.org/10.1021/ie049967s

Arena, Umberto, Mastellone, M. L., & Perugini, F. (2003). Life Cycle assessment of a plastic packaging recycling system. *The International Journal of Life Cycle Assessment*, *8*(2), 92–98. https://doi.org/10.1007/BF02978432

Astrup, T., Fruergaard, T., & Christensen, T. H. (2009). Recycling of plastic: accounting of greenhouse gases and global warming contributions. *Waste Management & Research*, *27*(8), 763–772. https://doi.org/10.1177/0734242X09345868

Azapagica, A., & Clift, R. (1999). Allocation of environmental burdens in multiple-function systems. *Journal of Cleaner Production*, *7*(2), 101–119. https://doi.org/10.1016/S0959-6526(98)00046-8

Bernstad, A., & la Cour Jansen, J. (2011). A life cycle approach to the management of household food waste - A Swedish full-scale case study. *Waste Management*, *31*(8), 1879–1896. https://doi.org/10.1016/j.wasman.2011.02.026

Bernstad Saraiva, A., Souza, R. G., & Valle, R. A. B. (2017). Comparative lifecycle assessment of alternatives for waste management in Rio de Janeiro – Investigating the influence of an attributional or consequential approach. *Waste Management*, *68*, 701–710. https://doi.org/10.1016/J.WASMAN.2017.07.002

Birgisdóttir, H., Bhander, G., Hauschild, M. Z., & Christensen, T. H. (2007). Life cycle assessment of disposal of residues from municipal solid waste incineration: Recycling of bottom ash in road construction or landfilling in Denmark evaluated in the ROAD-RES model. *Waste Management*, *27*(8), 75–84. https://doi.org/10.1016/j.wasman.2007.02.016

Bueno, G., Latasa, I., & Lozano, P. J. (2015). Comparative LCA of two approaches with different emphasis on energy or material recovery for a municipal solid waste management system in Gipuzkoa. *Renewable and Sustainable Energy Reviews*, *51*, 449–459. https://doi.org/10.1016/j.rser.2015.06.021

Butera, S., Christensen, T. H., & Astrup, T. F. (2015). Life cycle assessment of construction and demolition waste management. *Waste Management*, *44*, 196–205. https://doi.org/10.1016/j.wasman.2015.07.011

Carlsson, M., Naroznova, I., Moller, J., Scheutz, C., & Lagerkvist, A. (2015). Importance of food waste pre-treatment efficiency for global warming potential in life cycle assessment of anaerobic digestion systems. *Resources, Conservation and Recycling*, *102*, 58–66. https://doi.org/10.1016/j.resconrec.2015.06.012

Cossu, R., Garbo, F., Girotto, F., Simion, F., & Pivato, A. (2017). PLASMIX management: LCA of six possible scenarios. *Waste Management*, *69*, 567–576. https://doi.org/10.1016/j.wasman.2017.08.007

Cremiato, R., Mastellone, M. L., Tagliaferri, C., Zaccariello, L., & Lettieri, P. (2018). Environmental impact of municipal solid waste management using Life Cycle Assessment: The effect of anaerobic digestion, materials recovery and secondary fuels production. *Renewable Energy*, *124*, 180–188. https://doi.org/10.1016/j.renene.2017.06.033

Damgaard, A., Riber, C., Fruergaard, T., Hulgaard, T., & Christensen, T. H. (2010). Life-cycle-assessment of the historical development of air pollution control and energy recovery in waste incineration. *Waste Management*, *30*(7), 1244–1250. https://doi.org/10.1016/j.wasman.2010.03.025

Di Maria, F., & Micale, C. (2014). A holistic life cycle analysis of waste management scenarios at increasing source segregation intensity: The case of an Italian urban area. *Waste Management*, *34*(11), 2382–2392. https://doi.org/10.1016/j.wasman.2014.06.007

Di Maria, F., Micale, C., Contini, S., & Morettini, E. (2016). Impact of biological treatments of bio-waste for nutrients, energy and bio-methane recovery in a life cycle perspective. *Waste Management*, *52*, 86–95. https://doi.org/10.1016/j.wasman.2016.04.009

Dyer, T. D. (2014). Glass Recycling. In *Handbook of Recycling: State-of-the-art for Practitioners, Analyst, and Scientist* (pp. 191–210). London: Elsevier Science.

Ekvall, T., & Tillman, A. M. (1997). Open-loop recycling: Criteria for allocation procedures. *International Journal of Life Cycle Assessment*, *2*(3), 155–162.

https://doi.org/10.1007/BF02978810

Ekvall, T., & Weidema, B. P. (2004). LCA Methodology System Boundaries and Input Data in Consequential Life Cycle Inventory Analysis. *The International Journal of Life Cycle Assessment*, *9*(3), 161–171. https://doi.org//10.1065/lca2004.03.148

Elsevier. (2019). Engineering Village. Retrieved May 31, 2019, from https://www.engineeringvillage.com/search/quick.url?CID=quickSearch&database=1

Eriksen, M. K., Damgaard, A., Boldrin, A., & Astrup, T. F. (2019). Quality Assessment and Circularity Potential of Recovery Systems for Household Plastic Waste. *Journal of Industrial Ecology*, *23*(1), 156–168. https://doi.org/10.1111/jiec.12822

Eriksson, O., Reich, M. C., Frostell, B., Björklund, A., Assefa, G., Sundqvist, J. O., … Thyselius, L. (2005). Municipal solid waste management from a systems perspective. *Journal of Cleaner Production*, *13*(3), 241–252. https://doi.org/10.1016/j.jclepro.2004.02.018

Eriksson, Ola, Finnveden, G., Ekvall, T., & Björklund, A. (2007). Life cycle assessment of fuels for district heating: A comparison of waste incineration, biomass- and natural gas combustion. *Energy Policy*, *35*(2), 1346–1362. https://doi.org/10.1016/j.enpol.2006.04.005

European Commission. (2005). *DIRECTIVE (EU) 2018/851 OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL of 30 May 2018 amending Directive 2008/98/EC on waste*. Retrieved from https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:32018L0851&from=EN

Feraldi, R., Cashman, S., Huff, M., & Raahauge, L. (2013). Comparative LCA of treatment options for US scrap tires: Material recycling and tire-derived fuel combustion. *International Journal of Life Cycle Assessment*, *18*(3), 613–625. https://doi.org/10.1007/s11367-012-0514-8

Finnveden, G., Johansson, J., Lind, P., & Moberg, Å. (2005). Life cycle assessment of energy from solid waste - Part 1: General methodology and results. *Journal of Cleaner Production*, *13*(3), 213–229. https://doi.org/10.1016/j.jclepro.2004.02.023

Fruergaard, T., & Astrup, T. (2011). Optimal utilization of waste-to-energy in an LCA perspective. *Waste Management*, *31*(3), 572–582. https://doi.org/10.1016/j.wasman.2010.09.009

Fyffe, J. R., Breckel, A. C., Townsend, A. K., & Webber, M. E. (2016). Use of MRF residue as alternative fuel in cement production. *Waste Management*, *47*, 276–284. https://doi.org/10.1016/j.wasman.2015.05.038

Galgani, P., van der Voet, E., & Korevaar, G. (2014). Composting, anaerobic digestion and biochar production in Ghana. Environmental-economic assessment in the context of voluntary carbon markets. *Waste Management*, *34*(12), 2454–2465. https://doi.org/10.1016/j.wasman.2014.07.027

Gentil, E., Clavreul, J., & Christensen, T. H. (2009). Global warming factor of municipal solid waste management in Europe. *Waste Management and Research*, *27*(9), 850–860. https://doi.org/10.1177/0734242X09350659

Geyer, R., Kuczenski, B., Zink, T., & Henderson, A. (2016). Common Misconceptions about Recycling. *Journal of Industrial Ecology*, *20*(5), 1010–1017. https://doi.org/10.1111/jiec.12355

Ghose, A., Pizzol, M., & McLaren, S. J. (2017). Consequential LCA modelling of building refurbishment in New Zealand- an evaluation of resource and waste management scenarios. *Journal of Cleaner Production*, *165*, 119–133. https://doi.org/10.1016/j.jclepro.2017.07.099

Heijungs, R., & Guinée, J. B. (2007). Allocation and “what-if” scenarios in life cycle assessment of waste management systems. *Waste Management*, *27*(8), 997–1005. https://doi.org/10.1016/j.wasman.2007.02.013

Hellweg, S., & Milà i Canals, L. (2014). Emerging approaches, challenges and opportunities in life cycle assessment. *Science (New York, N.Y.)*, *344*(6188), 1109–1113. https://doi.org/10.1126/science.1248361

Hossain, M. U., Wu, Z., & Poon, C. S. (2017). Comparative environmental evaluation of construction waste management through different waste sorting systems in Hong Kong. *Waste Management*, *69*, 325–335. https://doi.org/10.1016/j.wasman.2017.07.043

Hupponen, M., Grönman, K., & Horttanainen, M. (2015). How should greenhouse gas emissions be taken into account in the decision making of municipal solid waste management procurements? A case study of the South Karelia region, Finland. *Waste Management*, *42*(2015), 196–207. https://doi.org/10.1016/j.wasman.2015.03.040

ISO. (2000). ISO/TR 14049:2000 - Environmental management -- Life cycle assessment -- Examples of application of ISO 14041 to goal and scope definition and inventory analysis. Retrieved February 14, 2019, from https://www.iso.org/standard/29834.html

Jensen, M. B., Møller, J., & Scheutz, C. (2016). Comparison of the organic waste management systems in the Danish-German border region using life cycle assessment (LCA). *Waste Management*, *49*, 491–504. https://doi.org/10.1016/j.wasman.2016.01.035

Laner, D., Rechberger, H., De Soete, W., De Meester, S., & Astrup, T. F. (2015). Resource recovery from residual household waste: An application of exergy flow analysis and exergetic life cycle assessment. *Waste Management*, *46*, 653–667. https://doi.org/10.1016/j.wasman.2015.09.006

Larsen, A. W., Merrild, H., & Christensen, T. H. (2009). Recycling of glass: Accounting of greenhouse gases and global warming contributions. *Waste Management and Research*, *27*(8), 754–762. https://doi.org/10.1177/0734242X09342148

LegisQuébec. Environment Quality Act, c. Q-2, art.53-4-1 (2018).

Lombardi, L., & Carnevale, E. A. (2018). Evaluation of the environmental sustainability of different waste-to-energy plant configurations. *Waste Management (New York, N.Y.)*, *73*, 232–246. https://doi.org/10.1016/j.wasman.2017.07.006

Mah, C. M., Fujiwara, T., & Ho, C. S. (2017). Concrete waste management decision analysis based on life cycle assessment. *Chemical Engineering Transactions*, *56*(January), 25–30. https://doi.org/10.3303/CET1756005

Majeau-Bettez, G., Dandres, T., Pauliuk, S., Wood, R., Hertwich, E., Samson, R., & Strømman, A. H. (2017). Choice of allocations and constructs for attributional or consequential life cycle assessment and input-output analysis. *Journal of Industrial Ecology*, *22*(4), 656–670. https://doi.org/10.1111/jiec.12604

Margallo, M., Aldaco, R., & Irabien, Á. (2014). Environmental management of bottom ash from municipal solid waste incineration based on a life cycle assessment approach. *Clean Technologies and Environmental Policy*, *16*(7), 1319–1328. https://doi.org/10.1007/s10098-014-0761-4

Morris, J. (2017). Recycle, Bury, or Burn Wood Waste Biomass?: LCA Answer Depends on Carbon Accounting, Emissions Controls, Displaced Fuels, and Impact Costs. *Journal of Industrial Ecology*, *21*(4), 844–856. https://doi.org/10.1111/jiec.12469

Nakamura, S., & Kondo, Y. (2002). Input-Output Analysis of Waste Management. *Journal of Industrial Ecology*, *6*(1), 39–63. https://doi.org/10.1162/108819802320971632

Parkes, O., Lettieri, P., & Bogle, I. D. L. (2015). Life cycle assessment of integrated waste management systems for alternative legacy scenarios of the London Olympic Park. *Waste Management*, *40*, 157–166. https://doi.org/10.1016/j.wasman.2015.03.017

Pires, A., Chang, N. Bin, & Martinho, G. (2011). Reliability-based life cycle assessment for future solid waste management alternatives in Portugal. *International Journal of Life Cycle Assessment*, *16*(4), 316–337. https://doi.org/10.1007/s11367-011-0269-7

Poeschl, M., Ward, S., & Owende, P. (2012). Environmental impacts of biogas deployment - Part I: Life Cycle Inventory for evaluation of production process emissions to air. *Journal of Cleaner Production*, *24*, 168–183. https://doi.org/10.1016/j.jclepro.2011.10.039

Reck, B. K., & Graedel, T. E. (2012). Challenges in metal recycling. *Science (New York, N.Y.)*, *337*(6095), 690–695. https://doi.org/10.1126/science.1217501

Reza, B., Soltani, A., Ruparathna, R., Sadiq, R., & Hewage, K. (2013). Environmental and economic aspects of production and utilization of RDF as alternative fuel in cement plants: A case study of Metro Vancouver Waste Management. *Resources, Conservation and Recycling*, *81*, 105–114. https://doi.org/10.1016/j.resconrec.2013.10.009

Rigamonti, L., Grosso, M., Møller, J., Martinez Sanchez, V., Magnani, S., & Christensen, T. H. (2014). Environmental evaluation of plastic waste management scenarios. *Resources, Conservation and Recycling*, *85*, 42–53. https://doi.org/10.1016/j.resconrec.2013.12.012

Rigamonti, L, Grosso, M., & Giugliano, M. (2010). Life cycle assessment of sub-units composing a MSW management system. *Journal of Cleaner Production*, *18*(16–17), 1652–1662. https://doi.org/10.1016/j.jclepro.2010.06.029

Rigamonti, Lucia, Grosso, M., & Sunseri, M. C. (2009). Influence of assumptions about selection and recycling efficiencies on the LCA of integrated waste management systems. *The International Journal of Life Cycle Assessment*, *14*(5), 411–419. https://doi.org/10.1007/s11367-009-0095-3

Ripa, M., Fiorentino, G., Giani, H., Clausen, A., & Ulgiati, S. (2017). Refuse recovered biomass fuel from municipal solid waste. A life cycle assessment. *Applied Energy*, *186*, 211–225. https://doi.org/10.1016/j.apenergy.2016.05.058

Sevigné-Itoiz, E., Gasol, C. M., Rieradevall, J., & Gabarrell, X. (2015a). Contribution of plastic waste recovery to greenhouse gas (GHG) savings in Spain. *Waste Management*, *46*, 557–567. https://doi.org/10.1016/j.wasman.2015.08.007

Sevigné-Itoiz, E., Gasol, C. M., Rieradevall, J., & Gabarrell, X. (2015b). Methodology of supporting decision-making of waste management with material flow analysis (MFA) and consequential life cycle assessment (CLCA): Case study of waste paper recycling. *Journal of Cleaner Production*, *105*, 253–262. https://doi.org/10.1016/j.jclepro.2014.07.026

Tagliaferri, C., Evangelisti, S., Clift, R., Lettieri, P., Chapman, C., & Taylor, R. (2016). Life cycle assessment of conventional and advanced two-stage energy-from-waste technologies for methane production. *Journal of Cleaner Production*, *129*(2016), 144–158. https://doi.org/10.1016/j.jclepro.2016.04.092

Tunesi, S., Baroni, S., & Boarini, S. (2016). Waste flow analysis and life cycle assessment of integrated waste management systems as planning tools: Application to optimise the system of the City of Bologna. *Waste Management and Research*, *34*(9), 933–946. https://doi.org/10.1177/0734242X16644520

USEPA. (2016). Paper Making and Recycling. Retrieved May 28, 2019, from https://archive.epa.gov/wastes/conserve/materials/paper/web/html/papermaking.html

Vadenbo, C., Hellweg, S., & Astrup, T. F. (2017). Let's Be Clear(er) about Substitution: A Reporting Framework to Account for Product Displacement in Life Cycle Assessment. *Journal of Industrial Ecology*, *21*(5), 1078–1089. https://doi.org/10.1111/jiec.12519

Van Ewijk, S., Stegemann, J. A., & Ekins, P. (2017). Global life cycle paper flows, recycling metrics, and material efficiency. *Journal of Industrial Ecology (Accepted)*, *00*(0), 1–8. https://doi.org/10.1111/jiec.12613

Weidema, B. (2000). Avoiding Co-Product Allocation in Life-Cycle Assessment. *Journal of Industrial Ecology*, *4*(3), 11–33. https://doi.org/10.1162/108819800300106366

Weidema, B. (2003). *Market information in life cycle assessment*. Retrieved from http://citeseerx.ist.psu.edu/viewdoc/download?doi=10.1.1.197.5739&rep=rep1&typ

e=pdf

Werner, F., & Richter, K. (2000). Economic Allocation in LCA : A Case Study About Aluminium Window Frames. *The International Journal of Life Cycle Assessment*, *5*(2), 79–83. https://doi.org/https://doi.org/10.1007/BF02979727

Yoshida, H., Gable, J. J., & Park, J. K. (2012). Evaluation of organic waste diversion alternatives for greenhouse gas reduction. *Resources, Conservation and Recycling*, *60*, 1–9. https://doi.org/10.1016/j.resconrec.2011.11.011

Zamani, B., Svanström, M., Peters, G., & Rydberg, T. (2015). A Carbon Footprint of Textile Recycling: A Case Study in Sweden. *Journal of Industrial Ecology*, *19*(4), 676–687. https://doi.org/10.1111/jiec.12208

Zhao, Y., Wang, H. T., Lu, W. J., Damgaard, A., & Christensen, T. H. (2009). Life-cycle assessment of the municipal solid waste management system in Hangzhou, China (EASEWASTE). *Waste Management and Research*, *27*(4), 399–406. https://doi.org/10.1177/0734242X09103823

Zhao, Y., Xing, W., Lu, W., Zhang, X., & Christensen, T. H. (2012). Environmental impact assessment of the incineration of municipal solid waste with auxiliary coal in China. *Waste Management*, *32*(10), 1989–1998. https://doi.org/10.1016/j.wasman.2012.05.012

Zink, T., Geyer, R., & Startz, R. (2016). A Market-Based Framework for Quantifying Displaced Production from Recycling or Reuse. *Journal of Industrial Ecology*, *20*(4), 719–729. https://doi.org/10.1111/jiec.12317