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INTRODUCING A NEW METHOD FOR CALCULATING THE ENVIRONMENTAL CREDITS OF END-OF-LIFE MATERIAL RECOVERY IN ATTRIBUTIONAL LCA

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ABSTRACT

Purpose This paper aims to provide an alternative method for calculating the environmental credits associated with material recycling in life cycle assessment (LCA) of waste management systems. The method proposed here is more consistent with the general attributional approach in LCA than the hitherto common practice of simply assuming a 1:1 substitution of primary material production.

Methods The formula proposed for estimating the environmental credit is applicable for the recovered materials that are reintroduced into the market (outputs of the recycling facilities), after all process losses in the various stages of the waste management system have been accounted for. It considers the displacement of materials by using the mix of virgin and recycled materials for each individual material that is used in the market for the production of goods. Moreover, it also considers the changes in the inherent properties of the materials undergoing a recycling process ('down-cycling'), by introducing a quality (Q) factor, affecting the proportion of virgin material that is accounted for.

Results and discussion Example applications of the proposed formula to a number of different materials (aluminium, steel, paper and cardboard and plastics) illustrate the range of possible results obtained.. The environmental credit calculated using the proposed formula can be interpreted as an indication of the remaining margin for improvement, since it depends on the existing mix of virgin and recycled materials already on the market, and on the potential of the recycled material to actually replace the primary one on a functional basis. We also discuss the possible use of a material's Q factor to estimate the maximum allowable % of recycled material in a product consistent with the quality demands of selected applications.

Conclusions and recommendations We have introduced here a consistent and unified formula for the evaluation of the credits associated with material recovery of all waste materials in waste management systems (paper, glass, plastics, metals, etc.). Such a formula requires the knowledge of the current average market consumption mixes of primary and secondary materials (or the application-specific average mixes when the final application of the recovered materials is known), and of suitable Q factors for the material(s) that are recycled. As the latter are often not readily available, more research is called for to arrive at a ready-to-use Q factors database.

KEY WORDS: attributional LCA, avoided impact, environmental credit, LCA, material recycling, system expansion, waste management.

1. INTRODUCTION

Integrated waste management systems can be seen as multi-functional systems, from a life cycle assessment (LCA) perspective, in which the treatment of waste is the main function of the system and the recovered energy and materials are additional functions. To be able to compare different waste management alternatives and maintain the same functional unit, it is necessary to take into account both the credits of material and energy recovery as well as the environmental impacts due to the collection and treatment of all waste fractions. In this context, system expansion (also referred to using synonymous terms such as 'substitution', 'crediting', and 'system enlargement') is the common method to avoid resorting to the allocation of environmental impacts in all sub-steps of the waste management system and to maintain the same functional unit for the comparison (see for instance, Bjarnadóttir et al. 2002; EC 2010; EC 2011).

When carrying out system expansion, uncertainty in identifying the alternative system to produce the same product is introduced, regardless of the application of a consequential or attributional LCA perspective (see section 2.1). As mentioned by several authors (i.e. Finnveden & Ekvall 1998; Shonfield 2008; Michaud et al. 2010), deciding which systems are displaced may have a strong influence on the results of an LCA. Different ways of modelling recycling have been extensively discussed over the past two decades (EC 2010). Whereas the effects of using different assumptions or approaches in relation to the energy that is substituted or displaced in waste management systems have been widely analysed (see for instance Finnveden et al., 2005; Smith et al. 2001; Eriksson et al. 2005; Bernstad & la Cour Jansen 2011; Laurent et al. 2014), the effects of material substitution have not been studied to the same extent. The vast majority of the LCA studies analysing the effects of recycling so far have assumed a 1:1 substitution ratio of recycled to virgin materials (Laurent et al., 2014). Such a substitution ratio applies at the point where the recycled materials are reintroduced into the market, after all losses due to impurities and process inefficiencies have been considered. A 1:1 substitution ratio implicitly means that recycled materials are supposed to replace the *same amount* of virgin materials with the *same quality*. Examples of this common practice can be found in: Björklund et al. 1999; Bovea et al. 2009; Dodbiba et al. 2008; Finnveden et al. 2005; Grant, et al. 2001; Merrild et al. 2008; Michaud et al. 2010; Muñoz et al. 2004;

Perugini et al. 2005; Shen et al. 2010; Shonfield 2008; Smith et al. 2001; and US EPA 2006. A few studies also account for a decrease in the quality of the recycled materials and use varied assumptions and reduced substitution ratios (see for instance Bernstad, A. et al 2011 for paper recycling), sometimes applying different criteria depending on the type of material (plastics, paper, metals or glass) recovered (e.g., Finnveden et al. 2000; Prognos et al. 2008; Smith et al. 2001; and US EPA 2006).

However, only a few authors have addressed the influence of the substitution ratio through a sensitivity analysis, especially for those materials for which a 'down-cycling' occurs when they are recycled (i.e. for which a direct substitution on a like-for-like basis is not possible), such as paper and plastics. Among them, Gentil et al. (2009) undertook a sensitivity analysis for a range of substitution ratios ranging from 1:2 (i.e. 50% replacement of virgin material) to 1:1 (100% replacement). Although no substantial effects on the results were identified, compared with the effects associated with changes in other technology parameters, it was concluded that a country relying strongly on material recovery with a poor substitution ratio would have a higher GWP, compared to systems with better substitution ratios. Rigamonti et al. (2009), analysed the effects of a substitution ratio <1 for paper and plastics and observed a worsening of around 15-20% in several impact category indicators and up to 45% for GWP. Using the same substitution factors, the sensitivity analysis performed by Bovea et al. (2010) concluded that this choice has a significant influence on the results, up to 20-42% in some impact categories. Thus, it seems that the employed substitution ratio is a significant factor to take into account.

When analysing all these facts, two potential methodological issues arise: (1) whether the recycled materials effectively displace virgin materials in all cases (or a mix of virgin and secondary materials, see section 2.2), *and* (2) whether the technical quality of the recycled materials remains the same as that of the original virgin materials.

The aim of this paper is to propose a novel method for calculating the environmental credits due to material recycling in LCA of waste management systems, applicable to all waste materials, and more in line with the attributional approach in LCA than the extended practice of simply assuming the substitution of primary material production. The proposed formula takes into account the average mix of virgin and recycled materials used in the market for the production of new goods as the displaced material to be considered. Moreover, it also considers the changes in the inherent properties of the materials undergoing a recycling process ('down-cycling'), by introducing a quality (Q) factor, affecting the proportion of virgin material that is accounted for.

2. METHODOLOGICAL KEY POINTS

2.1 Attributional vs. Consequential approach

As mentioned above, two main modelling approaches to LCA are possible, namely attributional and consequential. The choice between using the former or the latter should be based on the fundamental context and purpose of the study. The ILCD Handbook (EC, 2010) defines four major types of contexts: Situation A (micro-level decision support), Situation B (Meso/macro-level decision support) and Situations C1 and C2 (accounting with no decision support). As depicted in Figure 1, all approaches can be used for assessing the environmental performance of a system or to compare different waste treatment alternatives. In addition to the micro-, meso-, or macro-level decision context, we agree with Brandão and colleagues (2014) in claiming that attributional LCA is not an appropriate basis for policy development, but may be applicable in the context of policy implementation. Thus, a clear analysis of the purpose of the study and whether it is intended for policy development or implementation must also be taken into account for deciding the most appropriate modelling approach.

A consequential approach assumes that the changes in the system under study have large-scale effects on the background system. Accordingly, the "avoided impacts" are estimated on the basis of the displaced marginal technologies –those that are directly affected by changes in demand (Weidema et al., 1999). This is arguably the most appropriate approach to be used for strategic decisions (situation B), including decisions on new investment policies (Finnveden et al. 2005), and to answer questions of the type: "*what would be the consequences of developing a policy that would achieve an overall increased recycling rate for a given waste product/material?*" In this case, the "what if" scenario would clearly entail a change in the background system, a change in the composition of the virgin/recycled mix for the particular material consumed, and the additional recycled material recovered would clearly displace its virgin counterpart.

At the same time, an attributional LCA approach is more appropriate to answer questions like: *"what would happen if an existing source-sorting waste collection policy were implemented in additional 'X' sites/villages/etc.?"* This is because the attributional LCA approach assumes that the analyzed system does not modify its environment or, in other words, does not affect *in a significant way* the environmental performance of the background systems that supply the materials and energy inputs required (situations

C1 and A). In this case, the system should be modelled as it is (or was, or is forecast to be) using historical data. In this context, one may claim that each additional unit of waste material collected and recycled would displace an equivalent quantity of the current mix of virgin+recycled material being used as raw material by the market, without significantly affecting the composition of the overall mix; accordingly, the "environmental credits" of the recycled material should be calculated on the basis of the same mix of virgin+recycled (and not as the "avoided" 100% virgin) material.

Admittedly, if, in the same example above, the number of additional collection sites were large enough, the composition of the mix might end up being affected anyway, so the boundary of application of the two approaches is not clear-cut, but rather blurred. Moreover, as stated by Zamagni et al. (2012), "One should be careful, however, to note that the attributional/consequential dichotomy is constructed for the sake of argument. In practice, many LCAs are prospective based on scenarios for identified variables or explore the effect of identified causal changes while modelling the remainder of the system in an attributional manner".

This paper is however strictly meant to be confined to attributional LCA, and applicable to situations C1, A and B (if no large scale consequences in the background processes are produced).. Also, it is recommended that the formula be applied in the context of waste policy implementation.

2.2 Virgin (marginal) *vs.* market mix (attributional) substitution

As discussed above, it has so far been common practice in LCA to assume a 1:1 substitution ratio of recycled to virgin materials (albeit sometimes accounting for a loss of technical properties in the recycled materials leading to a reduced ratio). Additionally, and as stated by Laurent et al. (2014), this practice has often been accompanied by a lack of transparency about whether average or case-specific primary production data were used to perform the system expansion.

From a strictly theoretical point of view, using primary production as the displaced process entails a linear vision of the economy, since it assumes that every single unit of secondary material that is introduced into the market always avoids the production of the primary material. This can be interpreted in some way as the potential or *marginal gain* that is sought by implementing a recycling system.

However, it is arguably more fitting for a strictly *attributional* analysis, and from the point of view of a more 'circular' economy (Stahel and Reday-Mulvey 1981; Ayres 1998), to assume that each time a material is reintroduced into the market (i.e. at each cycle), it does not displace the primary production of the virgin material, but the average mix of technologies that provide an average unit of the material itself. According to this latter view, the environmental credits of one unit of recycled material should be calculated as the weighted average of the impacts of producing the primary (i.e. virgin) and secondary (i.e. recycled) materials being used by the market as input materials for the production of new goods. This is methodologically similar to the calculation of the credits associated with energy recovery in attributional LCA, where, if the technology mix that is effectively being displaced is not known, the average mix of technologies (e.g. grid mix) should be employed (Ripa et al. 2014).

Let us illustrate the difference between using the 100% primary *vs*. the market mix substitution approach by comparing the environmental credits of recycling aluminium and steel in a simplified example, using a single impact metric (Cumulative Energy Demand). The cumulative energy demand of virgin aluminium production is 194 MJ/kg and that of recycled aluminium is 23.8 MJ/kg; for steel those values are respectively 30 MJ/kg and 8.9 MJ/kg (Classen et al. 2009). If we apply a 'marginal' gain approach (primary substitution), the net impact in each case is simply the energy used for recycling the material minus that for primary production (23.8-194 = -170.2 MJ/kg for aluminium and 8.9-30 = -21.1 MJ/kg for steel, where a resulting negative sign indicates an environmental gain); hence, collecting 1 kg of aluminium for recycling is *always* 8 times more beneficial than collecting 1 kg of steel (170.2/21.1 = 8). In contrast, if we apply the 'attributional' approach based on (variable) market mixes, the relative benefit of collecting 1 kg of aluminium or steel for recycling changes depending on how much of those metals are already being recycled in their respective market mixes (Figure 2). For instance, only 50% of the steel and as little as 25% of the aluminium used in the packaging sector in Europe is of primary (virgin) origin (Table 1). Using these percentages, Figure 2 shows that the net gain of recycling aluminium is only 42 MJ/kg, while that of recycling steel is 12 MJ/kg; the relative benefit ratio in such sector-specific real-life conditions is thus still in favour of aluminium, but only 42/12 = 3.5. It could then be argued that if, hypothetically, the two market mixes became sufficiently different from one another, collecting 1 kg of steel for recycling might become more beneficial than collecting 1 kg of aluminium. Specifically, this would happen if the amount of virgin aluminium in the aluminium mix were to fall below 10% and, at the same time, the amount of virgin steel in the steel mix remained higher than 80% (see Figure 2).

From a policy point of view, it can easily be argued that a marginal approach encourages material recycling, which was the original aim of starting up an integrated waste management system, whereas using a market mix approach can lead to the seeming paradox that the more we recycle the less credit we get. However, we argue here that moving from the 'marginal' to the 'market mix / attributional' approach can lead to a better evaluation of what happens in reality due to waste policy implementation, especially if we are at a stage where a more circular economy is in place (almost fully closed recycling loops).

2.3 Accounting for quality

In line with the recommendations of the EC (2010), in order to correct the possible overestimation of the environmental credits associated with material recycling (which often produces lower-quality secondary materials), some authors calculate the amount of primary production displaced by applying correction factors based on technical properties of the secondary material, or on its price (for further details see Rigamonti et al. 2009), leading to the use of substitution ratios < 1. However, using market prices for calculating the substitution ratios is based on the assumption that price elasticity or, in other words, the way a change in price affects the demand, is equal for recycled and virgin materials, which has been demonstrated by some authors (Ekvall 1999; Weidema 2001; Frees 2008) to be wrong. Bearing this in mind, using a physical basis seems to be more appropriate for accounting for the substitution ratios and credits of material recovery.

3. METHODS

The formula proposed here (*Eq.1*) estimates the credits associated with the recovery of materials by means of using the actual mix of virgin and recycled materials that is used as a source of raw materials in the market (*cf.* section 2.2). Moreover, it also considers the deterioration of the inherent properties of the materials undergoing the recycling process ('down-cycling'), by introducing a quality factor (*cf.* section 2.3). This factor is used as 'proxy' to indirectly take into account that, because of its lower technical quality, the recycled material cannot replace an equal quantity of virgin material being part of the mix, but only a smaller quantity thereof (quality factor ≤ 1).

The formula to calculate the environmental credit associated to 1 tonne of recycled material is:

Eq.1) $$\text{Environmental credit} = x * REC + (1-x) * Q * VIR$$

Where:

x = proportion of recycled material in the average market mix

(1-x) = proportion of virgin material in the average market mix

Q = quality factor of recycled material *vs.* virgin material (Q ≤ 1)

REC = environmental load of the recycling process (1 tonne of recycled material in output)

VIR = environmental load of the production process of the virgin material (1 tonne in output)

This same approach is to be applied consistently to all recovered materials, and for all life-cycle impact categories/metrics.

4. PUTTING THE FORMULA INTO PRACTICE

4.1. Representative mixes

The first step to apply the proposed formula is to identify the average mix of virgin and recycled materials that is displaced. If the appropriate mix for a particular application or sector is known, and this is where the recovered material effectively ends up, then such a mix should be used. If not, average market-mix data such as those in Table 1 may be used instead. Import and export effects are considered in the model by adopting suitable material consumption (as opposed to production) mixes.

4.2. Quality factors

The second step for applying the formula is to determine the quality factors for those materials for which a down-cycling occurs. These should reflect the loss of quality of recycled *vs.* virgin materials. Obviously, this is not an easy task. In fact, we have identified a lack of studies in which the properties of recycled *vs*. primary materials are compared, especially in the case of plastics.

These quality factors can be likened to the technical correction factors used by some authors in the 'marginal' approach. In the case of paper products, for instance, the European Topic Centre on Waste Materials Flows (2004) suggests using a ratio not higher than 1:1.25 (i.e. Q = 0.8) for paper and cardboard, very close to the 1:1.23 (i.e. Q = 0.81) ratio calculated by Rigammonti et al. (2009). Instead, other authors such as Gentil et al. (2008) suggest using a ratio of 1:1.11 (i.e. Q = 0.9) for paper and also for plastics. However, we propose that the Q factors should always be strictly calculated on the basis of the actual physical properties of the materials and their contamination levels (to be determined by appropriate laboratory tests).

4.2.1 An example for calculating a quality factor based on mechanical properties of the materials

The process whereby recycled wood fibres behave differently from virgin ones is in itself complex and, contrary to common belief, cannot be reduced to a simple matter of 'fibre shortening'. Other properties related to the quality of the product such as water retention, tensile strength or tear index can also be significant, depending on the final application of the recycled pulp (Wistara & Young 1999). What is undeniable is that recycling paper products always results in down-cycling, and additional cycles (beyond the first one) result in progressively worse properties. Since it is impossible to distinguish between fibres that have undergone one, two or three recycling processes, it is common practice to counteract this loss of quality by adding a certain amount of virgin paper to the recycled products. Villanueva & Wenzel (2007) for instance, quantified this amount as about 20%, which means a $Q = 0.8$. Based on the study by Wistara and Young (1999), and taking into account the tensile strength indicator, we have arrived at a similar number ($Q \approx 0.83$), which implies a loss of quality of about 17% compared to the virgin paper (see Figure 3).

4.3. Examples of application

In this section, our proposed formula is applied to a set of materials, which serve as typical examples of different situations that may occur in the market: aluminium, paper and high density polyethylene (HDPE), considering an average market consumption mix substitution.

Figure 4 illustrates the varying trends of, respectively: a) the impact of the average market mix (red dotted line) calculated as per Eq. 2 , and b) the credit corresponding to one unit of recycled material (green dashed line), calculated according to Eq.1.

Eq.2) $$\text{Production impact of mix} = x * REC + (1-x) * VIR$$

The horizontal axis shows the percentage of secondary material present in the market mix, whereas the vertical axis shows the % of Global Warming Potential (GWP), normalized to the GWP of the virgin production (expressed as 100%).

Three classes of situations may occur when applying the proposed formula.

- **Situation a)** (illustrated by the case of aluminium), in which the calculation of the credit mainly depends on the market mix. In this case, the impact of virgin production (VIR) is about 10 times higher than that of the recycling process ($REC/VIR \approx 0.1$). At the same time, the quality factor is virtually equivalent to 1 (the same also applies to many other metals and glass). Thus, from a pragmatic point of view, in these cases, using the market mix alone is considered a reasonably good proxy, and the credit closely matches that of the simple weighted average of the mix itself.

- **Situation b)** (illustrated by the case of high density polyethylene). In this case, the impact of the recycling process is still lower than that of virgin production, but the difference is not so large (REC/VIR ≈ 0.2). Additionally, the credit is strongly influenced by the application of the quality factor (Q ≈ 0.75, as obtained through laboratory tests, to be published shortly). Thus, the lower the quality of the recovered material, the less credit one has. The result is that the credit line lies lower than that indicating the production impact of one unit of material according to the market mix. This is typically the case for most other plastics too.
- **Situation c)** (illustrated by the case of paper). This situation merits special attention due to the fact that the line indicating the credit ends up having a positive slope, instead of the normal negative one seen in all other cases. This counterintuitive result is due to the fact that the Q factor is actually *lower* than the ratio of the impact of recycling to that of virgin production (Q ≈ 0.83, and REC/VIR ≈ 0.9). As a consequence, the credit actually *increases* as the recycling replaces more and more secondary material (since the quality reduction only affects the replacement of the virgin material). This indicates that, because of the inevitable quality loss inherent in the recycling process, recycling waste paper is actually more beneficial (in terms of credits) if the output can be used to contribute to a well-established mix of already mainly secondary paper products (e.g. in the packaging sector[2]) than if it were employed to provide its inevitably low-quality fibre to a production mix still dominated by virgin paper (e.g. in the publishing sector).

4.4. Minimum acceptable quality for selected applications

A further issue that may be analysed by properly taking into account the relative difference in quality between the recycled and virgin forms of a material is the minimum acceptable *technical* quality of the mix of the two for specific applications (this discussion does not take into account other possibly important but unrelated ‘quality’ considerations, including aesthetics, colour homogeneity, etc., which may lead to a lower amount of recycled material being acceptable in a specific final product). Knowing this minimum acceptable *relative* quality (i.e. assuming that the quality of the primary material is 1) for a

[2] In fact, when dealing with the waste management of packaging paper and cardboard, the sector-specific mix is so close to the right end of the graph already (% of secondary paper > 90%), that the effect of the Q factor on the calculation of the credit becomes negligible, as shown in Figure 3c by the proximity of the dashed and dotted lines (respectively resulting from Eq. 1 and Eq.2).

specific application, and expressing the average quality of the mix of virgin + secondary material ($\bar{Q}$) as dependent on the fraction (x) of recycled material in the mix itself:

Eq.3) $\bar{Q}$ = Q * x * REC + 1 * (1-x) * VIR

using a figure similar to Figure 4, the cross-over point between the line indicating such average quality of the mix ($\bar{Q}$) and the horizontal line indicating the minimum acceptable quality for the particular application at hand will point to the percentage of secondary material (x) that may be accepted in input (along the horizontal axis). Such estimates can be used for specific analyses where the final application and the minimum acceptable quality of the material mix in input are known. Figure 5 shows three simple examples for aluminium, paper and HDPE, assuming for instance minimum acceptable relative qualities Q = 0.9, 0.83 and 0.8, respectively.

5. DISCUSSION

We stated that there is a common practice of using a substitution factor of 1:1 in LCAs of waste management, considered at the point where the recycled materials are ready to be reintroduced into the market, after having considered all process losses because of impurities in the input waste materials or technology efficiencies. We argue that this practice originates from a time when the market for recycled goods and materials was very limited, and the economy was perceived and described as a linear chain of processes. However, the waste management systems for recovering and recycling goods and the effective reintroduction of secondary materials in the market have improved and become more widespread in many countries, thereby moving towards the goal of a ‘circular economy’. As a result, continuing with the use of this simple substitution factor can lead to a misrepresentation of reality, and in particular to an overestimation of the environmental credits associated with recycling practices. Let us illustrate this fact by focussing for instance on the case of platinum. This valuable metal is used by the automotive industry in the production of catalytic converters, and is recovered and reused by the industry in an almost perfectly closed loop. Thus, when analysing a car recycling facility, it no longer makes sense to assume that by recovering platinum we are displacing the extraction and production of virgin platinum every time we recover it - because this is not what is happening in reality. Considering primary production as the displaced impact would thus lead to an inaccurate estimation of the immediate environmental consequences of the recovering facility, when we are under the framework of an attributional analysis.

Applying the formula proposed here to all LCAs of waste management systems to calculate the credits for all recycled materials may lead to the seeming paradox that the more one substitutes, the less credit one gets. This, according to some authors (IFEU & Öko-Institut 2012), may be problematic when comparing LCAs performed in different countries, because in those countries where the percentages of recycled materials in the market mixes are still small, the credit will end up being larger than in those countries where the recycling practices are more established and the amounts of recycled materials in the mixes are already larger. However, in our opinion this should not be considered a 'problem', but instead a necessary consequence of methodological consistency in strictly adhering to the attributional approach in LCA. The credit calculated by using the formula proposed here (Eq. 1) can essentially be interpreted as an indication of the remaining margin for improvement, since it depends on the existing mixes of virgin and recycled materials on the market, and on the potential of the recycled material to actually replace the primary ones on a functional basis.

As briefly mentioned in section 2.2, another reason for adopting this approach is the fact that it is strictly consistent with common practice in attributional LCAs when dealing with electricity production from waste management, where the national grid mix is used to calculate the environmental credits when the real substitution is not known (EU 2011). Let us imagine a case in which one wishes to compare the gains of recycling to the gains of incineration with energy recovery. Applying a 'marginal' approach to material recycling (1:1 substitution ratio) while adopting the common attributional praxis of assuming grid mix replacement for electricity production, would result in a methodological bias against energy recovery. While favouring material recycling may in fact be a good decision in many cases, especially when the recycling market is still in its infancy, applying the same, strictly attributional, approach to both waste management alternatives is unquestionably more even-handed and allows the analysis of the situation from a more neutral starting point.

6. CONCLUSIONS AND RECOMMENDATIONS

We have introduced here a unified formula for the evaluation of the environmental credits associated with material recovery in waste management, which represents a viable methodological alternative to the common marginal replacement approach (1:1 substitution factor) for many practical case studies. This formula is in line with the fundamental aim of the attributional approach in LCA, and may be applied to all waste materials, thereby ensuring methodological consistency among them. Such formula relies on the knowledge of the application-specific or market-average mix of primary and secondary material currently in use, which is assumed to be displaced by the recycled material. It also requires the evaluation of a quality factor (Q) to account for the reduced relative technical quality of the recycled material (vs. that of the virgin one). While information on the composition of the average market-consumption material mixes for many common materials is easily obtained, there is a dearth of specific studies addressing the quality of secondary *vs*. primary materials, and more research is called for to arrive at a ready-to-use database of suitable Q factors for many materials and applications.

Finally, the same approach recommended here for waste management systems is, in principle, equally valid for LCAs of product systems. However, while the system boundary for the former is almost invariably the same (namely, a cut-off rule is invoked whereby all input waste materials carry no environmental burdens), many alternatives exist when dealing with product systems. In fact, products may be parts of complex chains or even webs of other upstream and downstream processes and systems, and may already have secondary, as well as primary, material inputs. Utmost care is therefore needed in order to avoid any implicit or even explicit double counting, where the same product system is credited twice for the same amount of recovered material used as raw material and being recycled at the end of the product's life. A more in-depth discussion of all the possible intricacies arising from the application of system expansion in the LCA of products is however beyond the scope of the present paper, which is confined to LCA of waste management systems.

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References

Ayres RU (1998) Viewpoint: Toward a Zero-Emissions Economy. Environ. Sci. Technol. 32(15), 366-367.

Bernstad A, la Court Jansen J (2011) A life cycle approach to the management of household food waste – A Swedish full-scale case study. Waste Manage. 31: 1879-1896.

Bernstad A, la Cour Jansen J, Aspegren H (2011) Life cycle assessment of a household solid waste source separation programme: a Swedish case study. Waste Manage. Res. 29(10), 1027-1042.

Bjarnadóttir H J, Fridriksoon G B, Johnsen T, Sletsen H (2002) Guidelines for the use of LCA in the waste management sector. Nordtest Report. TR 517.

Björklund A, Dalemo M, Sonesson U (1999) Evaluating a municipal waste management plan using ORWARE. J. Clean. Prod. 7, 271-280.

Bovea MD, Ibáñez-Forés V, Gallardo A, Colomer-Mendoza FJ (2010) Environmental assessment of alternative municial solid waste management strategies. A Spanish case study. Waste Manage. 30: 2383-2395.

Brandão M, Clift R, Cowie A, Greenhalgh S (2014). Letter to the editor. "The Use of Life Cycle Assessment in the Support of Robust (Climate) Policy Making: Comment on "Using Attributional Life Cycle Assessment to Estimate Climate-Change Mitigation . . . ". J. Ind. Ecol. 18(3): 461-463.

CEPI- European Pulp and Paper Industry (2010) Annual Statitics 2010 (http://www.cepi.org). Accessed, July 2012.

Classen M, Althaus HJ, Blaser S, Tuchschmid M, Jungbluth N, Doka G, Faist Emmenegger M, Scharnhorst W (2009) Life Cycle Inventories of Metals. Final Report Ecoinvent data v2.1, Nº 10. EMPA Dübendorf, Swiss Centre for Life Cycle Inventories, Dübendorf, CH, Online-Version under: www.ecoinvent.ch

Dodbiba G, Takahashi K, Sadaki J, Fujita T (2008) The recycling of plastic wastes from discarded TV sets: comparing energy recovery with mechanical recycling in the context of life cycle assessment. J. Clean. Prod. 16, 458-470.

EAA- European Aluminium Association (2011) Activity Report, 2011. Brussels.

Ekvall T (1999) System expansion and allocation in life cycle assessment. Göteborg: Chalmers University of Technology, ARF Report 245.

Eriksson O, Carlsson Reich M, Frostell B, Björklund A, Assefa G, Sundqvist J-O, Granath J, Baky A, Thyselius L (2005) Municipal solid waste management from a system perspective. J. Clean Prod. 13:241–252

European Commission - Joint Research Centre - Institute for Environment and Sustainability (2010) International Reference Life Cycle Data System (ILCD) Handbook - General guide for Life Cycle Assessment - Detailed guidance. First edition March 2010. EUR 24708 EN. Luxembourg. Publications Office of the European Union.

European Commission (2011) Supporting environmentally sound decisions for waste management – A technical guide to Life Cycle Thinking (LCT) and Life Cycle Assessment (LCA) for waste experts and LCA practitioners. Luxembourg: Publications Office of the European Union .

EUROFER- European Steel Association (2014), http://www.eurofer.org/Sustainable%20Steel/Steel%20Recycling.fhtml. Accessed 10 January 2014.

European Topic Centre on Waste and Material Flows (2004) Review of existing LCA studies on the recycling and disposal of paper and cardboard. http://waste.eionet.europa.eu/. Accessed 10 January 2014

Finnveden G, Ekvall T (1998) "Life Cycle Assessment as a decision-support tool – The case of recycling versus incineration of paper". Resour. Conserv. Recycl. vol 24, no 3-4, pp 235-256.

Finnveden G, Johansson J, Lind P, Moberg A (2005) Life cycle assessment of energy from solid waste – part 1: general methodology and results. J. Clean Prod. 13: 213-229.

Frees N (2008) Crediting aluminium recycling in LCA by demand or by disposal. Int. J. Life Cycle Assess. 13(3): 212-218.

Gentil E, Potter A, Boldrin A (2008) Carbon footprinting of export of second-life materials using life-cycle thinking. Proceedings of Waste 2008 Conference, Stratford-upon-Avon (England), 16–17, September 2008.

Gentil E, Clavreul J, Christensen TH (2009) Global warming factor of municipal solid waste management in Europe. Waste Manage. Res. 27:850-860.

Gentil E, Gallo D, Christensen TH (2011) Environmental evaluation of municipal waste prevention. Waste Manage., 31: 2371-2379.

Grant T, James KL, Lundie S, Sonneveld K (2001) Stage 2 Report for Life Cycle Assessment for Paper and Packaging Waste Management Scenarios in Victoria. Report to EcoRecycle Victoria, Melbourne. Centre for Design at RMIT University, Centre for Packaging Transportation and Storage at Victoria University and the Centre for Water and Waste Technology at University of New South Wales, Melbourne and Sydney.

IFEU & Öko-Institut e.V (2012) Minutes of the Workshop "Methods to calculate GHG mitigation potentials in Solid Waste Management" on behalf of a project commissioned by Unweltbundesamt (Federal Environment Agency) Germany. 18 June 2012, Berlin, Germany.

Laurent A, Clavreul J, Bernstad A, Bakas I, Niero M, Gentil E, Christensen TH, Hauschild MZ (2014) Review of LCA studies of solid waste management systems – Part II: Methodological guidance for a better practice. Waste Manage. 34:589-606.

Merrild H, Damgaard A, Christensen TH (2008) Life cycle assessment of waste paper management: The importance of technology data and system boundaries in assessing recycling and incineration. Resour. Conserv. Recycl. 52 : 1391-1398.

Michaud JC, Farrant L, Jan O, Kjaer B, Bakas I (2010) Waste and Resources Action Program – WRAP (2006): Environmental benefits of recycling- 2010 update.

Muñoz I, Rieradevall J, Domenech X, Milà L (2004) LCA application to integrated waste management planning in Gipuzkoa (Spain). Int. J. Life Cycle Assess. 9, 272-280.

Perugini F, Mastellone ML, Arena U (2005) A life cycle assessment of mechanical and feedstock recycling options for management of plastic packaging wastes. Environ. Prog. 24(2): 137-154.

Prognos AG, Ifeu, INFU (2008). Resource savings and CO2 reduction potential in waste management in Europe and the possible contribution to the CO2 reduction target in 2020. Brussels.

Rigamonti L, Grosso M, Sunseri MC (2009) Influence of assumptions about selection and recycling efficiencies on the LCA of integrated waste management systems. Int. J. Life Cycle Assess. 14: 411-419.

Ripa M, Raugei M, Bala Gala A, Ulgiati S (2014) Dealing with waste management in Emergy Accounting and LCA: methodological overview and synergies. Presented at: 8th Biennial Emergy Research Conference. January 16-18, 2014, University of Florida, USA.

Roldán E, Pino MR (2012) FENIX-Giving Packaging a New Life Project. Life Cycle Inventory of pretreatment and glass recycling. Universidad San Jorge, unpublished.

Shen L, Worrell E, Patel MK (2010) Open-loop recycling: a LCA case study of PET bottle-to-fibre recycling. Resour. Conserv. Recycl. 55: 34-52.

Shonfield P (2008) LCA of Management Options for Mixed Waste Plastics-Final Report. WRAP, 2008. ISBN: 1-84405-397-0.

Smith A, Brown K, Ogilvie S, Rushton K, Bates J (2001) Waste Management Options and Climate Change. Final report to the European Commission, DG Environment. http://europa.eu.int

Stahel W, Reday-Mulvey G (1981) Jobs for tomorrow: The Potential for Substituting Manpower for Energy.Vantage Press. New York. Publication in book form of: The potential for substituting manpower for energy (1977), Report to DG V for Social Affairs, Commission of the EC, Brussels.

United States Environmental Protection Agency (2006) Solid Waste Management and Greenhouse Gases: A life Cycle Assessment of Emissions and Sinks. 3rd Edicion. September, 2006.

Villanueva A, Wenzel H (2007) Paper waste-Recycling, incineration or landfill ? A review of existing life cycle assessments. Waste Manage. 27: S29-S46.

Weidema B, Frees N, Nielsen AM (1999) Marginal Production Technologies for Life Cycle Inventories. Int. J. Life Cycle Assess. 4(1): 48-56.

Weidema B (2001) Avoiding co-product allocation in Life-Cycle Assessment. J. Ind. Ecol. 4(3): 11-33.

Wistara N, Young RA (1999) Properties and treatments of pulps from recycled paper. Part I. Physical and chemical properties of pulps. Cellulose, 6: 291-324, 1999.

Zamagni A, Guinée J, Heijungs R, Masoni P, Raggi A (2012) Lights and shadows in consequential LCA. Int. J. LCA, 17(7): 904–918.

Table 1- *Average European market mixes for different materials*

Material	% virgin	% recycled	Source
Aluminium*	63	37	Calculated from EAA, 2011
Steel	50	50	EUROFER, 2014
Glass	55	45	Roldán & Pino, 2012
Cardboard	16	84	Calculated from CEPI, 2010
Paper	71	29	Calculated from CEPI, 2010
Beverage cardboard	57	43	Calculated from CEPI, 2010
Plastics**	**	**	-

* For the packaging sector these percentages move to 25% of virgin and 75% of recycled.
** The percentage of recycled plastic is difficult to quantify.

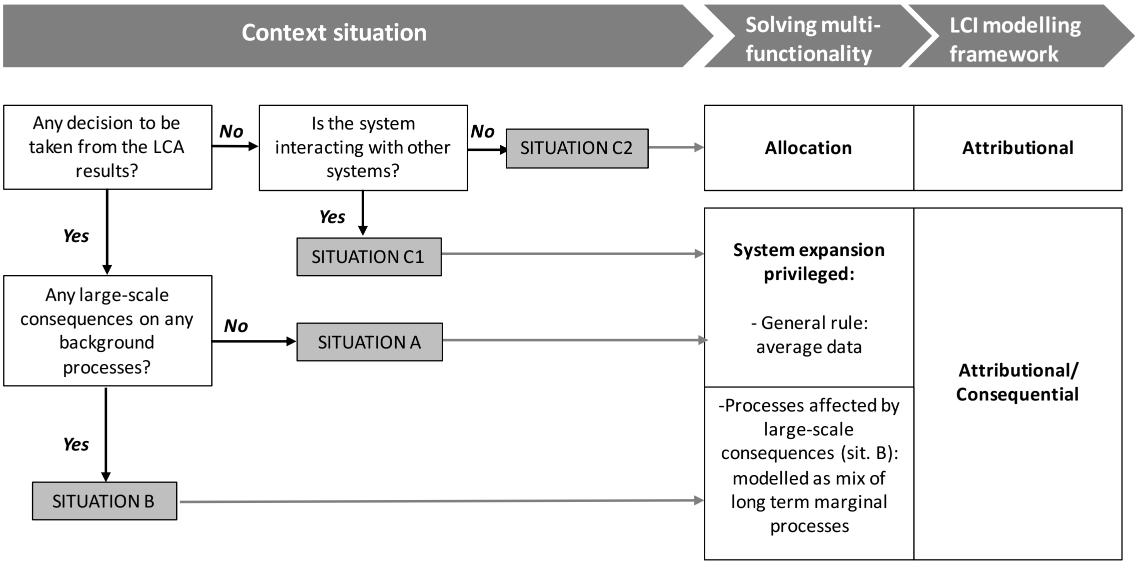

Figure 1–*Identification of context situations from the ILCD Handbook*

(Source: Laurent et al., 2014)

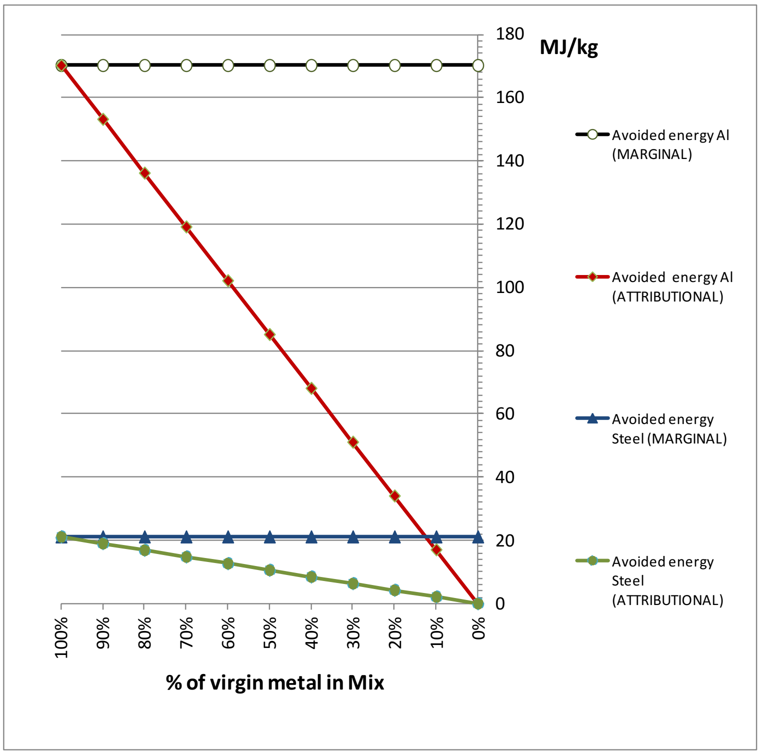

Figure 2– *Net benefits of recycling aluminium and steel under a marginal approach (1:1 replacement of virgin material) vs. an 'attributional' approach (replacement of virgin and recycled market mix).*

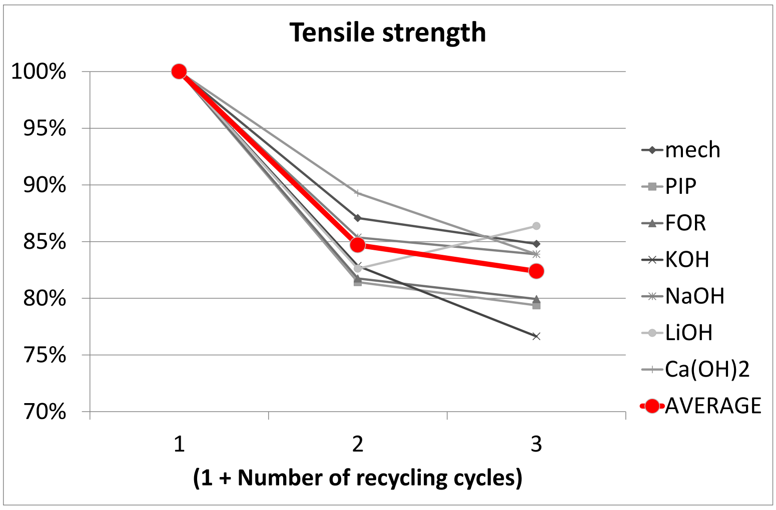

Figure 3 *– Change in Tensile Strength as a proxy of quality factor for paper and cardboard produced from (1) virgin pulp, (2) first-cycle secondary pulp and (3) second-cycle secondary pulp [after Wistara and Young, 1999]. mech = purely mechanical recycling; PIP = recycling with piperidine treatment; FOR = recycling with formide treatment; KOH = recycling with potassium hydroxide treatment; NaOH = recycling with sodium hydroxide treatment; LiOH = recycling with lithium hydroxide treatment; Ca(OH)2 = recycling with calcium hydroxide treatment; AVERAGE = average of all of the above.*

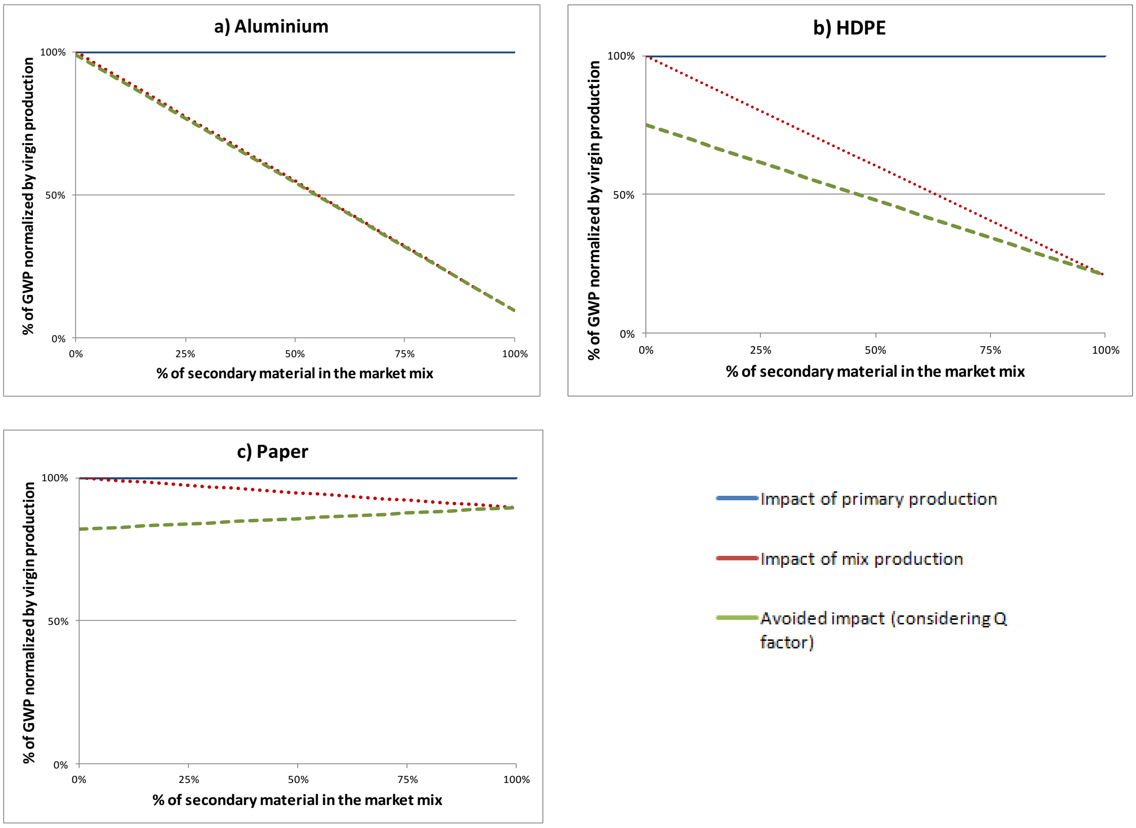

Figure 4 *– Comparison of (1) GWP impact of primary (virgin material) production; (2) GWP impact of the representative market mix of primary (virgin material) and secondary (recycled material) production, calculated according to Eq. 2; and (3) avoided GWP impact relative to the representative mix, calculated according to Eq. 1 (Q factor aluminium=0.99; Q factor HDPE=0.75; Q factor paper=0.83).*

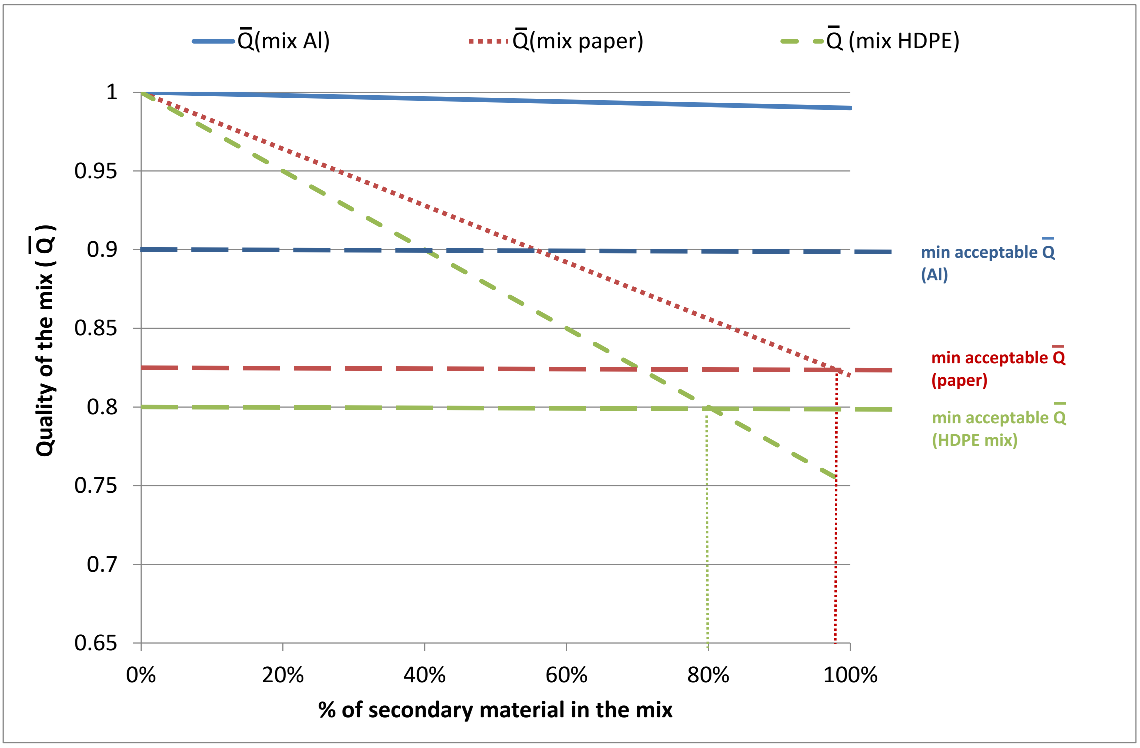

Figure 5 *– Example of estimation of the maximum acceptable % of secondary material in the mix in order to comply with a pre-set minimum average quality demand. Examples for Aluminium (blue solid line); Paper (red dotted line); and HDPE (green dashed line). The minimum values employed in the figure are only for illustrative purpose and do not correspond to any real case.*