Dealing with waste products and flows in Life Cycle Assessment and Emergy Accounting: methodological overview and synergies

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ABSTRACT

This paper considers the different approaches taken in dealing with waste products and flows in Life Cycle Assessment (LCA) and Emergy Accounting (EMA), from a methodological point of view, and aims to develop more standardized and synergistic procedures. LCA deals with the waste issue from the point of view of the impact of their disposal, as well as the potential benefit ('environmental credit') afforded by the avoided extraction and processing of additional primary resources when waste is recycled or its energy content recovered. The 'environmental burden' associated to the entire production and consumption chain leading to the waste item is generally not included in LCAs of waste management systems, due to the boundary being placed – consistently with the intended goal – around the actual disposal processes (including recycling alternatives and associated environmental credits). Instead, Emergy

Accounting, a donor-side approach with its implicit boundary set at the biosphere level, in principle keeps track of the entire supply-chain at all times, considering even waste flows as products (or co-products), and calculating their intensity factors and assessing their role within the ecosystem's web and hierarchy. However, when the focus is limited to evaluating processes under human control, within the narrower space and time boundary of human-dominated production and consumption processes, waste products can arguably be regarded as something to be recycled or disposed of to minimize the environmental burden. When this is the case, and particularly in comparative analyses, the emergy perspective thus becomes closer to the LCA perspective and interesting methodological synergies may emerge. A clearly defined set of emergy algebra rules for waste products and flows, and specifically for recycling, was found to be still lacking in the available emergy literature. We propose here that a better and more consistent methodological solution may be arrived at by leveraging the work done in LCA.

1. INTRODUCTION

In natural ecosystems, all material flows are circular and the very concept of waste does not apply: ‘waste’ products and flows from a process always become inputs to other processes. Instead, human-dominated systems are typically incapable of continuously re-using all waste flows, which puts increased pressure on the environment in terms of pollution as well as ever-increasing depletion of natural resources. Waste management strategies are aimed at minimizing such problems, but they entail additional resource use too, and so must be carefully assessed and optimized.

As already advocated and explained elsewhere (Ulgiati et al., 2006; 2011), there is much to be gained from the comparison, parallel application and, where appropriate, integration (Raugei et al., 2006; Ingwersen, 2011; Rugani and Benetto, 2012; Marvuglia et al., 2013; Arbault et al., 2014; Raugei et al., 2014) of life cycle assessment (LCA) and emergy accounting (EMA), when the intended object of analysis is human-dominated systems. Waste management systems are often especially complex, and therefore require extra care when making all the necessary methodological choices and assumptions, in order to ensure both strict internal adherence to the dictates of the underlying theories, and, no less importantly, external consistency and comparability to pre-existing and possible follow-up studies.

While a number of waste management case studies have already been investigated by emergy analysts (Brown and Buranakarn, 2003; Marchettini et al., 2007; Lei et al., 2008; Amponsah et al., 2011; Yuan et al, 2011; Zhang et al., 2011; Mu et al., 2011; Agostinho et al., 2013; Giannetti et al., 2013; Liu et al., 2013; Song et al., 2013), it seems reasonably safe to conclude that coherent and agreed-upon methodological guidelines as to how to approach this particular field of application are still lacking. On the other hand, a large body of scientific and technical literature exists in which LCA has been used as the method of choice when tackling waste management systems from the point of view of their energy and environmental performance from a user-side perspective (e.g. Finnveden & Ekvall, 1998; Eriksson, 2003; Coleman, 2006; Thorneloe et al, 2007; Gentil, 2011; Koci & Trecakova, 2011). Additionally, in recent years a considerable effort has been made to standardize LCA and provide clear methodological guidelines on how it should be implemented for waste management systems (Bjarnadóttir et al., 2002; JRC, 2010; 2011a; 2011b; 2011c), and on the trade-offs that are inherent in the adoption of alternative assumptions in those cases where no single clear-cut distinction can be made between absolutely ‘right’ or ‘wrong’ approaches.

We herein provide a brief overview of the main critical points that are specific to waste products and flows (with selected illustrative examples) and of how they have so far been addressed in LCA. We then discuss the extent to which the work done in the LCA community may be leveraged to improve the clarity and consistency of EMA when applied to waste management. At the same time, we also highlight and discuss those instances where underlying perspective of LCA conflicts with that of EMA, thereby rendering some of the assumptions and solutions proposed by the former essentially inapplicable within the framework of the latter.

2. METHODS

2.1 Life Cycle Assessment

Life Cycle Assessment (LCA) is a relatively recent methodology that has rapidly grown to become a standard tool to investigate the environmental performance of a wide range of human-dominated processes (ISO, 2006a,b; JRC, 2010). LCA is based on the basic principle that in order to accurately assess the environmental impact of the analysed system or product, all its life stages must be addressed, also including in the analysis, where appropriate, the end-of-life recovery and/or recycling of the system’s components (for subsequent re-use in other product systems). Methodologically, an LCA is structured in four consecutive stages, namely: (i) goal and scope definition (including a clear definition of the functional unit, system boundaries and associated assumptions); (ii) life cycle inventory (the compilation of all the inputs and outputs respectively from and to nature associated to all processes that form part of the system’s life cycle); (iii) life cycle impact assessment (in which the full inventory of inputs and outputs is translated into a number of aggregated metrics of environmental impact); and (iv) interpretation (in which results are discussed and compared to suitable benchmarks).

As simple as it may sound when taken at face value, most of the key methodological dilemmas in the application of LCA to waste management arise in that first all-important step of a clear and unambiguous definition of the intended goal and scope of the study. In fact, all that LCA requires is that whatever the stated goal and scope of the analysis is, the analysis be then carried through in strict adherence to those same goal and scope at all times. In other words, it is perfectly permissible to carry out two independent LCAs of the very same system starting with different 'questions' in mind and, consequently, arriving at quite different 'answers' in the end. Indeed, this is the principal reason why not all methodological assumptions and alternatives that have legitimately been adopted in LCA may be equally applicable to EMA (whether specifically dealing with end-of-life and waste management processes or otherwise).

In all cases, LCA only accounts for matter and energy flows occurring under human control, whereas flows outside of market dynamics (such as environmental services and renewable resources that do not flows through human controlled devices) as well as flows which are not associated to significant matter and energy carriers (such as labour, culture, information) are not generally included. Moreover, the supply-side 'quality' and degree of renewability of resources, in terms of biosphere activity leading to resource generation processes, are not explicitly taken into account in LCA evaluations (Ulgiati et al., 2006). Where renewable flows are included, such as e.g. in the calculation of the CED metric (VDI, 1997), their inclusion only refers to the renewable fraction captured under human control (e.g. the amount of sunlight actually captured by photovoltaic modules).

2.2 Emergy Accounting

Emergy is defined as the available energy (exergy) of one kind (usually solar) previously required, directly and indirectly, to make a service or product (Odum, 1996). The boundary of the analysis is always set at the biosphere level, thereby keeping track of the entire supply chain (from resource generation to processing and disposal), and accounting for the environmental support needed to generate all the storages and flows of (renewable and non-renewable) raw natural resources which flow through the web of natural processes supporting the analysed process either directly or indirectly (e.g. in the form of ecosystem services). The unit of emergy is the solar emergy Joule (seJ), and the emergy to generate one unit of available energy or mass along a particular pathway is named tranformity (units: seJ/J) or, more generally, Unit Emergy Value (UEV, units: seJ/unit). Incidentally, it is worth noting that in a natural ecosystem, which is not only subject to, but *the product of* natural selection, the transformity also indicates the position of each type of energy flow in the ecosystem's energy hierarchy (Brown et al., 2006), while this only applies loosely and at a very coarse level to human-dominated systems, many of which co-exist without having yet been vetted by long-term natural selection. The total emergy driving a system, calculated as the sum of all emergy inflows, is assigned to the product or service delivered (for further details see Odum, 1996; Brown and Ulgiati, 2004, 2010). After all the flows of interest have been quantified, a set of additional indicators: Environmental Loading Ratio (ELR), Emergy Yield Ratio (EYR), etc., can be developed for better understanding of a system's dynamics as well as for environmental policy making (sustainable resource use), by assessing the environmental performance of the process itself (Brown and Ulgiati, 2004).

One fundamental difference between LCA and EMA is arguably that in the latter, unlike in the former, the analyst is required to always abide by the same underlying 'donor side perspective' that is at the very core of emergy theory. Also, the concept of waste (something useless and devoid of any ability to drive further transformation processes) has little meaning from an emergy point of view, because every flow or residue from a process inevitably has a 'history' of its own (hence the concept of 'energy memory' introduced by Brown and Herendeen, 1996), becomes an input to and has an impact on some other (human-dominated or natural) process (Genoni et al., 2003).

As a consequence, EMA should always consider all waste flows as products or co-products, and calculate their intensity factors accordingly (but paying careful attention not to double-count the emergy inputs when dealing with multiple functional units). On the contrary, LCA distinguishes between 'waste flows', 'waste products' and co-products based on market value (Guinée et al., 2004), and applies different allocation rules accordingly. This is better detailed in section 3 below.

3. KEY METHODOLOGICAL ASPECTS

3.1 Treatment of elementary flows vs. products and waste products

LCA makes a fundamental distinction between what it calls 'elementary flows', i.e. flows which are directly sourced and/or emitted to the environment *as is* (including 'waste flows'), and 'products' (including 'waste products'), which on the other hand are the product of, and are output to, a range of human-dominated systems (the latter collectively referred to as the 'technosphere'). While it is the elementary flows which directly contribute to environmental impact (in terms of resource depletion, and of a number of emission-related impact categories such as global warming potential, acidification potential, etc.), a life-cycle impact potential is computed and assigned to products and waste materials, depending on the inputs and outputs of elementary flows that they have been 'responsible for' along their life cycle. The rules for the allocation of such 'responsibility' amongst (co)-products and waste materials in LCA are detailed in the following sub-section.

EMA, on the other hand, by virtue of its intrinsic 'historical' perspective on the exergy cumulatively spent to provide any given flow at any given moment, has no use for such distinction, and treats flows from/to the environment and those from/to the technosphere in the same way, from a methodological point of view.

3.2 Different approaches to multi-functionality

Based on their market value, LCA then also clearly differentiates between: (i) useful (co)-products, which jointly carry the environmental burden of a production system, and (ii) waste products, which (like waste elementary flows) are considered devoid of any useful value, and whose environmental impact is therefore re-distributed amongst the (useful) (co)-products.

The general recommended way to tackle co-products in LCA (both those of the same physico-chemical nature – which are usually named 'splits' in EMA - and those of different physico-chemical nature) is by system expansion (ISO 2006b; JRC, 2010). When adopting the system expansion approach in LCA, the analyst is free to select those output products which are considered to be of primary interest, and the impact associated to the remaining co-products is removed by (i) expanding the analysis to also assess *alternative* product systems which generate those same (and only those) outputs whose impact needs to be removed, and then (ii) subtracting the impact associated to the latter systems from that of the original system under study (on a per-functional unit basis).

If such system expansion is impossible or impractical, then allocation may alternatively be employed (similarly to what is done by default in EMA in the case of product splits – see below); however, in LCA the analyst has a choice to opt for either energy-, mass-, or economic-based allocation. In fact, depending on the specific system under study and on the goal of the analysis, any of these options may be preferable in order to better reflect the user-side perspective (i.e. "to which degree is each co-product responsible for the operation of the entire system?").

Contrary to what happens in LCA, in EMA *all* system outputs (including waste products) are, at least in principle, always considered to be either co-products or ‘splits’. Additionally, according to the basic emergy definition, computation procedures in EMA follow a special ’algebra’ that keeps track of all steps from resource generation up to the product at stake, and differentiates between 'co-products' (two or more products or flows characterized by different physico-chemical nature and generated simultaneously: one cannot be generated without also generating the other one) and 'product splits' (two or more products or flows sharing the same physico-chemical nature: in principle it is possible to generate only one of them without also generating the others). When only one product is obtained from a process, all source-emergy is assigned to it. Instead, when two split products are generated, the source-emergy is assigned (allocated) to them according to their available energy (or mass). Finally, when two or more co-products are generated, the total source-emergy is assigned to all of them (no allocation). Consequently, when two co-product pathways re-unite in a downstream process, the emergy carried by those converging flows must not be added together, lest their common original driving source be double-counted. In such cases, the traditional approach has been to only account for the largest flow when computing the total emergy of the final product.

This peculiarity of the ‘emergy algebra’ represents a potential stumbling block for the seamless integration of EMA into an existing LCA workflow. Marvuglia et al. (2013) proposed an interesting way to address and solve this issue with their SCALE software. However, the fly in the ointment of their solution as it may be implemented using the currently available LCA databases is that all those flows which appear to be co-products in the database are treated as if they were *actual* co-products of the same real process. In reality, however, the same database process is often used as a proxy for independent processes taking place at different locations and at different moments in time, which removes the requirement for any special emergy algebra rule in the first place. So, while worthy of praise from a theoretical point of view, in its current practical implementation the solution proposed in SCALE may often end up ‘over-compensating’; the resulting uncertainty and loss of accuracy should be the subject of a proper analysis, e.g. by running SCALE with and without considering the ‘co-product rule’. It is important to note, though, that this state of matters is an intrinsic shortcoming not of SCALE itself but of the LCI networks as they are modelled in the currently available databases, which are lacking spatial and temporal differentiation (Tiruta-Barna and Benetto, 2013).

3.3 End-of-life processes, avoided impact and environmental credit

When specifically dealing with those end-of-life processes that result in the production of secondary materials (recycling) or recovered energy (incineration and sometimes landfilling), the recommended way to address them in *attributional* LCAs (i.e. those LCAs whose goal is not to investigate the potential long-term consequences of large-scale policy choices, but to actually assess the real impact associated to the life cycle of a system as it is now) is again by system expansion. The analysis is thus extended to also include the average mix of technologies that at the time of the analysis provide an average unit of, respectively, the material and/or energy that is recovered, and the impact associated to the latter is then subtracted from that of the original system under study. Figure 1 illustrates this logic in the case of aluminium. From to this viewpoint, the ‘environmental credit’ associated to one unit of recycled material is calculated as the weighted average of the impacts of producing the primary (i.e. virgin) and secondary (i.e. recycled) material used in the market. Likewise, for energy, the appropriate average mix of technologies (e.g. the grid mix) should be employed.

Conversely, in *consequential* LCAs a different line of reasoning is adopted, which is often referred to as ‘marginal replacement’. This leads to the identification of the production of virgin material(s), and of energy carriers produced by those technologies whose use it is the industry’s or government’s intention to curb, as the best candidates for the calculation of an ‘avoided impact’. The latter corresponds to arguing that, after all, it is essentially *in order to* reduce the demand for primary materials (and *in order to* replace polluting energy technologies) that, respectively, recycling and energy recovery are implemented.

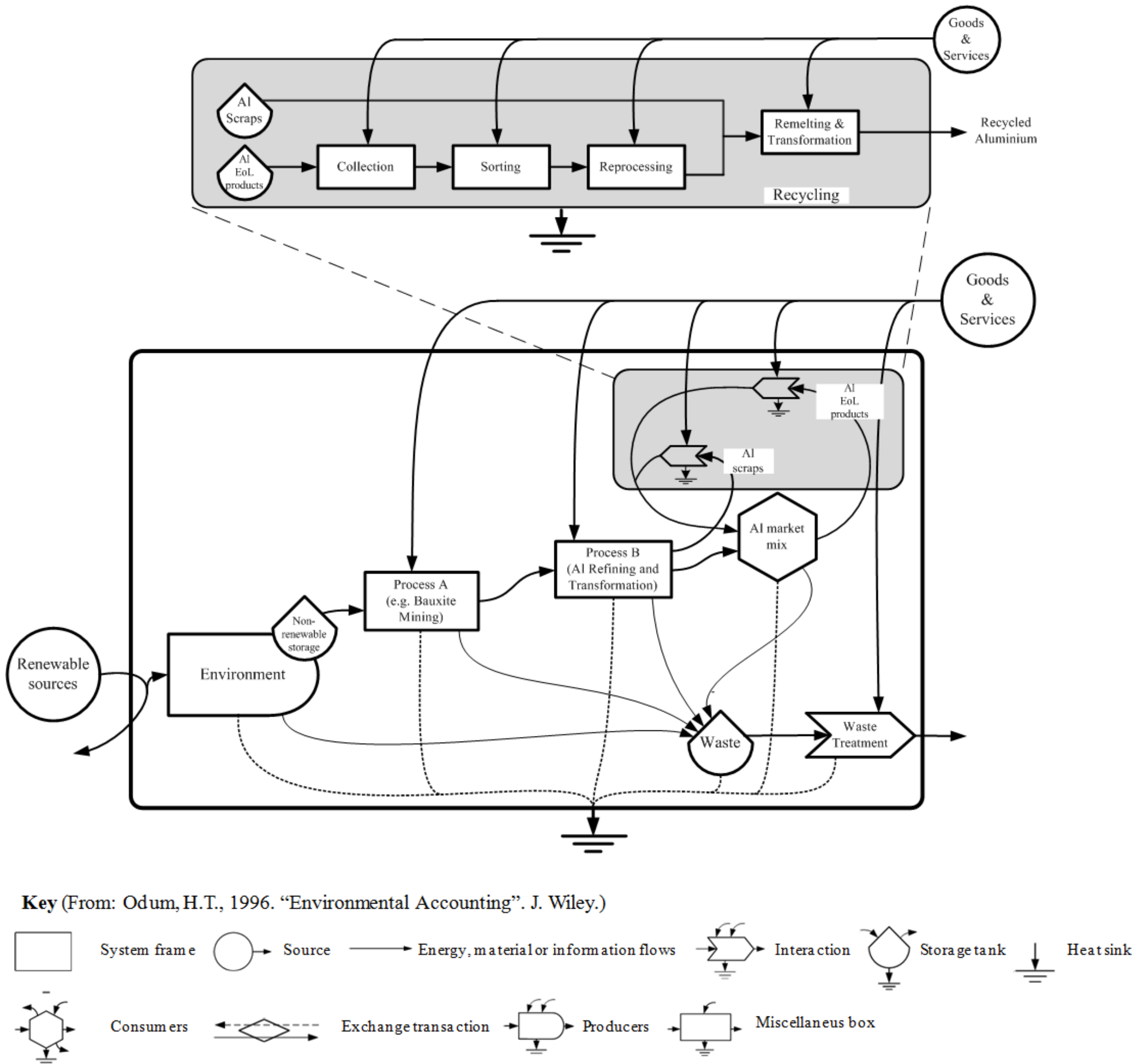


Figure 1. Energy system diagram for primary and secondary aluminium production, both contributing to an average mix of Al on the market (hexagon-shaped symbol on the right hand side of the main diagram).

At least in principle, an 'environmental credit' logic similar to that of attributional LCA discussed above and illustrated in Figure 1 may generally be considered applicable to EMA too. For instance, when waste materials are produced which could be recycled or put to new use elsewhere (via open-loop recycling), be they categorized as *co-products* (e.g. corn straw which could be used as soil fertilizers in another system) or *split flows* (e.g. saw dust of wood processing, which could be used as a source of energy), a virtual decrease of input emergy to the analysed system could be considered. In the two examples above, such ‘credited emergy’ would be respectively that for the production of chemical fertilizers, and that for the production of conventional thermal energy.

3.4 System boundary and closed-loop vs. open-loop recycling

In LCA, when materials are used in more than one product cycle, it is crucial to always set inter-system boundaries in such a way as to clearly separate the life cycles of the different product systems that make successive use of the same materials (Figure 2). A number of options are available as to where to locate such ‘cut-off’ points (Ekvall and Tillman, 1997).

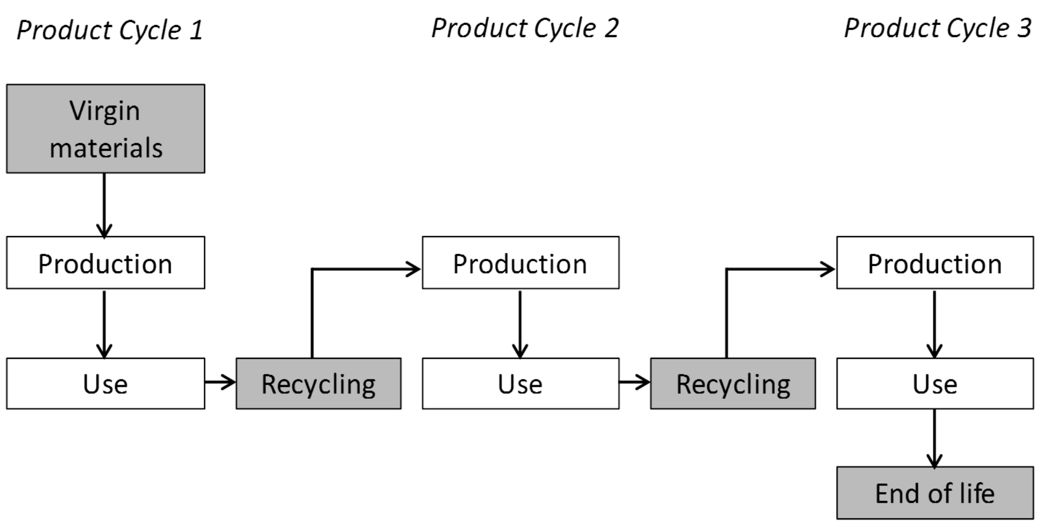


Figure 2. Simplified example of successive product cycles. Processes in grey are those susceptible to be assigned to different product cycles of shared among them.

One approach that is sometimes adopted when analysing one particular product system which happens to be located along any such chains of multiple material uses is to assign the impact associated to the first stages of its waste management (i.e. its collection, disassembly and transport to landfill, incinerator and/or sorting facilities) to the first product system, and then the additional impact due to the pre-treatment and recycling of those materials that are re-used in subsequent product systems to the latter systems. This corresponds to adopting the ‘rule’ that <u>secondary scrap used as input material carries ‘zero embodied impact’</u>. In so doing, though, the analyst foregoes the possibility to claim back any ‘environmental credit’ for the first product system (cf. previous sub-section) due to the recovery of materials at its end-of-life.

Alternatively, in many cases the system boundaries are often set so as to include all of the waste management in the life cycle of the first product system (including the recycling processes), and then an 'environmental credit' is claimed back for the same product system, based on what the recycled materials are assumed to replace. It is interesting to note, however, that whenever this second approach is adopted, a potential external inconsistency issue arises when results from independent analyses are combined. This, of course, is because the impacts of the recycling process and the associated 'credits' can only be assigned at any given time to *either* product system 1 *or* product system 2, along the chain.

In EMA, the following two basic scenarios are distinguished:

a) Recycling within the same process (i.e. 'closed-loop recycling'), *analysed assuming a steady state*. When a recycled flow (waste or co-product) is fed back to a process' earlier step, its emergy should not be double counted and only the additional emergy investment for collection, feedback and pre-treatment should be added. This essentially coincides with the LCA logic.

b) Waste flows from other processes (i.e. 'open loop recycling'). The rule to prevent double-counting does not automatically apply to this situation, and at first it might seem that if the recycled/reused material were allowed to carry its entire 'emergy memory', each reuse cycle would increase the emergy of the recycled fraction, in principle increasing its UEV without a limit - and in fact, a similar argument has sometimes been made in the literature (Amponsah et al., 2011). However, more careful scrutiny reveals that such interpretation stems from a fundamental misconception of the fundamentals of emergy theory (Ulgiati et al., 2004). In general terms, the emergy of a 'virgin' resource in input to a production process may be decomposed into: (Ef + Ep), where Ef is the emergy of natural resource 'formation', and Ep is the emergy of the subsequent processes taking place in the technosphere (i.e. extraction, refining/pre-treatment and delivery). It should be noted that Ef is in fact the contribution of nature's own work to slowly 'recycle' the resource <u>once</u> on the geological scale (e.g. through sedimentary deposition, or through remelting in the mantle, etc.), and does not take into account more than <u>one</u> successive 'loop' of such natural recycling process. According to the same logic, the emergy of a 'secondary' (i.e. recycled) resource in input to a process at any given moment should only be Er = the emergy of (technological) recycling. A secondary input should not be assigned any additional emergy besides Er, because:

(1) The material is already in the technosphere, and therefore its use does not entail any additional resource depletion; in other words, it does not require nature to perform another 'loop' of its slow 'recycling' work on the geological scale. Hence, in this case $Ef = 0$; to include this contribution again would be double counting.

(2) The material does not need to be extracted, refined and delivered again from its natural source in the geobiosphere (e.g. from the ore in the ground). Hence, in this case $Ep = 0$; to include this contribution again would be double counting.

It should be noted that the same fundamental logic applies throughout emergy theory, and specifically to all natural ecosystem processes, where multiple recycling loops are ubiquitous. For instance, the emergy of the inorganic nutrients uptaken by a plant at any given moment do not carry the emergy that went into growing the previous generations of plants that grew and then decayed in the past, thereby releasing (i.e. recycling) the nutrients back into the soil. Nor does a blade of grass being fertilized by the decaying carcass of a lion see its emergy propelled to any higher level by virtue of the emergy accrued during the former ‘life cycle’ of its ‘donor system’ (i.e. the lion).

Additionally, it should also be considered that with each consecutive cycle, a new use is made (i.e. a new 'functional unit' is created) for the same amount of (recycled) material (assuming for the moment for the sake of simplicity that the recycling itself is 100% efficient). Thus, on average, the emergy of a unit of material after N cycles (Er_N) would amount to its original emergy of the ‘virgin’ material (Ef+Ep), plus N times the additional emergy required to recycle it once (Er), divided by (N+1) total functional units (Eqn. 1):

Eqn. 1) $$Er_N = \frac{(Ef + Ep) + N \cdot Er}{(N+1)}$$

For N >> 1, the expression above reduces to $Er_N \approx Er$. In other words, for those materials that may routinely be recycled multiple times (like e.g. glass and virtually all metals), the average emergy of one unit of recycled material is demonstrated to be approximated by the sole additional emergy required for the recycling process itself.

Operationally, this essentially coincides with adopting a simple 'cut-off' rule like is done in LCA, but, importantly, without calling for any special '*ad hoc* rules' or exceptions to the general emergy theory. For those materials for which the recycling process entails some degree of structural degradation, thereby limiting the maximum number of cycles (N) before terminal disposal becomes inevitable, Eqn. 1 also provides a theoretically sound way to compute the average emergy of a unit of recycled material. Since, typically, $Er \ll (Ef+Ep)$ (otherwise recycling would not make sense in the first place), we will have in these more general cases: $Er < Er_N \ll (Ef+Ep)$.

4. A SIMPLE APPLICATION EXAMPLE

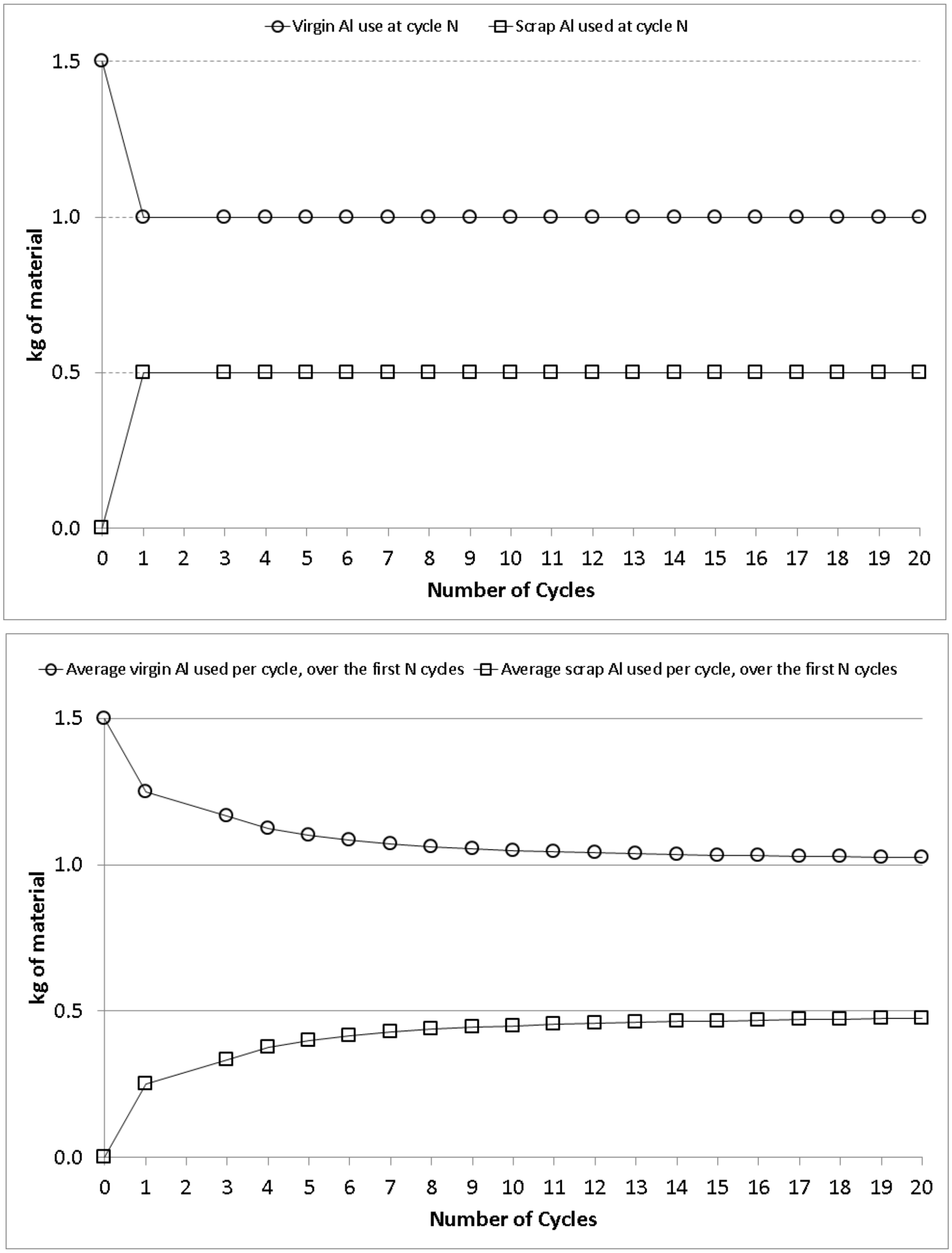
The streamlined example below is provided as a simple illustration of some of the theoretical points discussed in the previous section. For the sake of simplicity, we shall restrict ourselves to considering only the Cumulative Energy Demand (CED) indicator

(MJ of primary energy per FU) in LCA, and the Unit Emergy Value (UEV) (seJ per FU) in EMA. The former indicator allows a comparison of alternative systems and scenarios on the basis of their different demand for existing commercial energy sources. The latter, instead, provides an overall assessment of the energy ‘cost’ of the analysed systems over the full evolutionary time scale of the biosphere (i.e. including resource generation in addition to resource processing), and may be used as a different measure of sustainability.

It is however important to note that the overall assessment of a system’s environmental performance typically calls for more indicators in both LCA (e.g. Global Warming Potential (GWP), Acidification Potential (AP), etc.) and EMA (e.g. Emergy Loading Ratio (ELR), Emergy Yield Ratio (EYR), etc.). In this simple, idealized example, we shall consider a factory that manufactures products made entirely of aluminium, and define our functional unit (FU) as 1 kg of product (for instance, we may refer to a 1 kg section of aluminium pipe). Virgin aluminium ingots are melted, cast, extruded and cut into the final products, which are then anodised. An amount of 0.5 kg of scraps and trimmings from the above processes per FU are reintroduced into the furnace, leading to what may be referred to as closed-loop recycling. The first time the aluminium product is produced (cycle N=0), an input of 1.5 kg of virgin aluminium is needed. Already in the first cycle (N=1), though, 0.5 kg of scraps from the first production run are reused, and the demand for virgin Al is down to 1 kg (Figure 3a). From then on, the average steady-state amount of virgin Al that is required will tend to be reduced as the number of cycles increases (as 1+[1/(N+1)]*0.5, where N is the number of cycles), up to a point in which a stable situation is reached (e.g. N > 10) where the average amount of virgin Al needed is ~ 1 kg (Figure 3b). In order to further simplify the example, we shall then analyse a case in which such a stable situation has already been reached (Figure 4).

Taking into account that no changes in the inherent properties of aluminium occur in the recycling process, we can assume that each unit of recycled aluminium replaces one unit of virgin aluminium.

Table 1 illustrates the calculations that would apply to a theoretical scenario where no recycling took place, and the Al scraps and trimmings were simply discarded. Table 2, instead, refers to the actual system including recycling, for N >> 1 (Figure 4).



a)

b)

Figure 3. Input of virgin and secondary materials in a closed-loop industrial recycling waste process. **a)** in each cycle; **b)** average over the first N cycles.

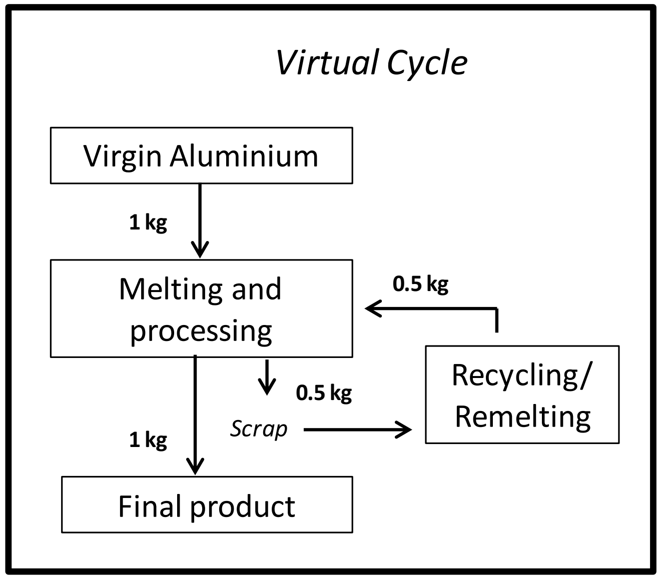


Figure 4. Closed-loop recycling of industrial waste (aluminium) when a steady state is reached (N>>1).

Table 1. Calculations for no recycling scenario.

	Amount	CED (MJ_{PE}/FU)[(a)]	EMERGY (seJ/FU)
INPUTS			
Virgin Al (kg/FU)	1.5	235.5	$2.43 \cdot 10^{13}$ [(b)]
Product manufacturing (electricity, kWh/FU)	1.2	14	$5.33 \cdot 10^{11}$ [(c)]
Total Impact		**249.5**	$\mathbf{2.49 \cdot 10^{13}}$

(a) Cumulative Energy Demand from CED impact assessment procedure in GaBi 6, based on PE International Database included in the GaBi 6 LCA software package (update: 1/12/2013)
(b) Unit Emergy Values of resource extraction, transport and processing to ingot, including biosphere work for ore concentration (Bargigli, 2003)
(c) Based on current ENTSO-E European mix; Unit Emergy Values of electricity production after Brown and Ulgiati (2002)

Table 2. Calculations for closed-loop recycling of industrial waste (N>>1).

	Amount	CED (MJ_{PE}/FU)[(a)]	EMERGY (seJ/FU)
INPUTS			
Virgin Al (kg/FU)	1	157	$1.62 \cdot 10^{13}$ [(b)]
Product manufacturing (electricity, kWh/FU)	1.2	14	$5.33 \cdot 10^{11}$ [(c)]
Al scrap recycling process	0.5	3	$1.23 \cdot 10^{11}$ [(d)]

(kg/FU)			
Total Impact		**174**	**$1.69 \cdot 10^{13}$**

(a) Cumulative Energy Demand from CED impact assessment procedure in GaBi 6, based on PE International Database included in the GaBi 6 LCA software package (update: 1/12/2013)
(b) Unit Emergy Values of resource extraction, transport and processing to ingot, including biosphere work for ore concentration (Bargigli, 2003)
(c) Based on current ENTSO-E European mix; Unit Emergy Values of electricity production after Brown and Ulgiati (2002)
(d) Calculated assuming 0.27 kWh/FU electricity use (Ecoinvent, 2010); Unit Emergy Values of electricity production (European mix) after Brown and Ulgiati (2002)

5. CONCLUSIONS

As previously discussed a number of times elsewhere, life cycle assessment and emergy accounting are independently developed methods that have a lot in common, but which also differ in some fundamental ways, making neither expendable and instead both potentially complementary to one another in many applications.

When dealing with end-of-life and waste management processes and systems, we have found that a comparative methodological review of LCA and EMA, as presented here, points to a significant convergence of the two methods, which represents a valuable opportunity for their integration. Specifically, LCA's clear and non-contradictory treatment of system and inter-system boundaries (as applies to chains of processes that are linked in ways that make the output and waste products of one the direct or indirect inputs of the next) may lead to a better understanding and to a less potentially ambiguous statement of emergy algebra rules as they apply to waste and recycled products. Additionally, the availability of a large body of LCA literature specifically focused on waste products and systems provides a valuable opportunity for EMA researchers and practitioners to reflect on a number of complex and sometimes subtle issues, thereby potentially improving the methodology further and facilitating its applicability to policy.

However, in spite of the many steps already made towards the fruitful comparison and integration of LCA and EMA, well-framed and carried out waste management case studies are still few and far between in the existing EMA literature, and there are still a number of unresolved issues that call for further research. On one hand, there is the need for further standardization, in order to arrive at fully consistent and comparison-friendly boundary and accounting procedures in LCA and EMA. On the other hand, though, there is also a need for a better and more widespread understanding and awareness of the different inherent perspectives offered by the two methods. In fact, in our opinion there is no need for a forced integration in those cases when the intended goal of the study does not require it. Also, it makes little sense to always adopt the largest possible system boundaries in those cases when the goal and scope of the analysis is intentionally restricted (e.g. when dealing with two alternative options for steel recycling).

Our systematic discussion of the main key methodological aspects of the analysis of waste products and systems in both LCA and EMA has helped identify a number of clear and non-contradictory practical guidelines that apply to both methods. We suggest that in the future such guidelines be vetted and, if confirmed to be sound, followed in all analyses of human-dominated systems that either focus on waste products and flows, or in which, in any case, the latter play a prominent role.

ACKNOWLEDGEMENTS

The authors Ripa and Ulgiati gratefully acknowledge the financial support received from the EU Project LIFE11 ENV/DE/343, MARSS “Material Advanced Recovery Sustainable Systems”, LIFE+ Environment Policy and Governance.

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