Methodological aspects of life cycle assessment of integrated solid waste management systems

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Abstract

Environmental life cycle assessment (LCA) developed rapidly during the 1990s and has reached a certain level of harmonisation and standardisation. LCA has mainly been developed for analysing material products, but can also be applied to services, e.g. treatment of a particular amount of solid waste. This paper discusses some methodological issues which come into focus when LCAs are applied to solid waste management systems. The following five issues are discussed. (1) Upstream and downstream system boundaries: where is the ‘cradle’ and where is the ‘grave’ in the analysed system? (2) Open-loop recycling allocation: besides taking care of a certain amount of solid waste, many treatment processes also provide additional functions, e.g. energy or materials which are recycled into other products. Two important questions which arise are if an allocation between the different functions should be made (and if so how), or if system boundaries should be expanded to include several functions. (3) Multi-input allocation: in waste treatment processes, different materials and products are usually mixed. In many applications there is a need to allocate environmental interventions from the treatment processes to the different input materials. The question is how this should be done. (4) Time: emissions from landfills will continue for a long time. An important issue to resolve is the length of time emissions from the landfill should be considered. (5) Life cycle impact assessment: are there any aspects of solid waste systems (e.g. the time horizon) that may require specific attention for the impact assessment element of an LCA? Although the discussion centres around LCA it is expected that many of these issues
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1. Introduction

Environmental life cycle assessment (LCA) is a systems analysis tool. It developed rapidly during the 1990s and has reached a certain level of harmonisation and standardisation. An ISO standard [1] has been developed as well as several guidelines [2–4].

An LCA studies the environmental aspects and potential impacts throughout a ‘product’s’ life (i.e. cradle-to-grave) from raw material acquisition through production, use and disposal [1]. This is done by compiling an inventory of relevant inputs and outputs of a system (the inventory analysis), evaluating the potential impacts of those inputs and outputs (the impact assessment), and interpreting the results (the interpretation) in relation to the objectives of the study (defined in the goal and scope definition in the beginning of a study) [1].

In the definition of LCA, the term ‘product’ includes not only product systems but can also include service systems [1], for example waste management systems. LCA is currently being used in several countries to evaluate different strategies for integrated solid waste management and to evaluate treatment options for specific waste fractions [5–13].

Although improvements have been made to LCA methodology, there are still a number of unresolved issues which need further attention [14]. This paper discusses aspects of LCA methodology which come into focus when applying LCA to integrated solid waste management systems (ISWMS). Topics which will be discussed are the upstream and downstream system boundaries, the open-loop allocation problem, the multi-input allocation problem, time as a system boundary and life cycle impact assessment. It is expected that these topics are also of relevance for other types of systems analyses of waste management systems, although terminology may be different. In another paper, some additional topics related to modelling of landfills in LCA are discussed [15].

2. Upstream and downstream system boundaries

A key aspect of LCA is that the system should be modelled in such a manner so that inputs and outputs to the system are followed from the ‘cradle’ to the ‘grave’, which means that inputs should be flows that are drawn from the environment without human transformation, and outputs should be flows that are discarded to the environment without subsequent human transformations [1]. In LCAs of waste management systems, this is typically not done. Instead, the inputs are often solid
waste as they appear, e.g. from households. This is, however, still compatible with the LCA definition, if the same inflow appears in all systems which are to be compared. This is because those parts of the systems which are identical in all systems which are compared, can be disregarded. The upstream system boundary may, however, have to be changed, if one of the systems to be compared produce more or less waste than the others. In this situation the system inputs are no longer identical, and in principle the system boundary should be moved and upstream activities should be included, at least those parts which differ between different systems. This may in practise prove to be very difficult and therefore not done. In that case it should, however, be carefully noted that the impacts of the system which produces less waste is overestimated compared to the others.

A similar situation may occur for the downstream system boundary when materials or energy are recycled into new products. In LCAs of waste management systems, products from recycling are normally not followed to the ‘grave’. Again this is compatible with the LCA definition, if the products are ‘identical’ in all systems which are compared. In these cases the products can be disregarded. ‘Identical’ does not mean that they have to be exactly identical in all aspects. It is enough if they are providing a comparable function to a user, and if they have the same environmental impacts. If the products are not providing comparable functions, they cannot replace each other. If the products do not have the same environmental impacts, at least the differences should be included in the LCA.

3. Open-loop recycling

3.1. Introduction

Open-loop recycling takes place when a product is recycled after its use into another product (Fig. 1). This can be a problem since the system boundary between products 1 and 2 is not clear-cut. Open-loop recycling has been much discussed in LCA literature [2–5,14,16–18]. The problem can be solved in two ways: by allocating environmental interventions between products 1 and 2 and studying only one of them, or by expanding system boundaries and including both products within the system.

3.2. Allocation in open-loop recycling systems

If an allocation is to be made, there are three parts of the system described in Fig. 1 which should be allocated between products 1 and 2 [4]: (1) the recycling system (box E); (2) production of primary material used in both products 1 and 2 (parts of box A); (3) disposal of material used in both products 1 and 2 (parts of box I). The other boxes in Fig. 1 are concerned with only one of the products and there is therefore no need to allocate the environmental interventions between the two products.
Many methods for the allocation have been suggested in the literature and used in case studies [4,16,18]. There does not seem to be any procedure that proves that any specific method is the ‘correct’ one. Instead arguments are usually based on what intuitively seems reasonable or fair. However, such arguments can lead to very different conclusions [18]. Examples of suggested and implemented methods include (based on reviews in Refs. [4,18]):

(1) allocation of primary material production used in both products, and disposal of material used in both products to product 1, and the recycling process to product 2;

(2) allocation of primary material production used in both products, and disposal of material used in both products to product 2, and the recycling process to product 1;

(3) allocation of 50% of the primary material production used in both products, the disposal of material used in both products and the recycling process to product 1, and 50% to product 2;

(4) allocation of primary material production used in both products to product 1, disposal of material used in both products to product 2, and the recycling process to either product 2 or as a refinement use of gross sales values for the allocation of the recycling process. This procedure can be described as a cut-off, since a cut is made between the two systems. This procedure is sometimes used without realising that an allocation is being performed.

From these examples it is clear that different allocation procedures can yield different results. There is currently no international consensus on which procedure to use, although different recommendations occur in different types of national guidelines.

Fig. 1. An open-loop recycling system.
3.3. System expansion

In order to avoid allocation problems, system boundaries can sometimes be expanded to include several products within the system. An example is the comparison between landfilling and incineration of solid waste [9]. The main function (or ‘product’) of landfilling is treatment of solid waste. Incineration with heat recovery also treats solid waste, but in addition also produces heat or electricity, thus providing a second function (Fig. 2). Since the two processes provide different functions, it is difficult to directly compare them.

In the expanded system (Fig. 3), an alternative method of producing an equivalent amount of heat has been added to the landfill system. It is thus possible to compare the incineration system to the combined landfill and heat producing system. The systems compared are multi-functional systems. Another way of presenting the expanded systems is to subtract the heat-producing system using an alternative heat source from the incineration system, as described in Fig. 4. Since the same components are included, it is essentially the same systems which are presented in Fig. 3 and Fig. 4. In the system shown in Fig. 4, so called ‘avoided emissions’ will occur and environmental interventions may become negative. The system in Fig. 4 is sometimes described by saying that the incineration is ‘credited’ with an equivalent amount of heat (or electricity) being produced in an alternative manner.
The expanded (but subtracted) systems shown in Fig. 4 are single functional but will not produce the same results as a single-functional system in which an allocation has been performed. This is because the system boundaries are different. Instead, the subtracted, expanded system will produce the same results in qualitative terms as the expanded multi-functional system (Fig. 3), although the exact numbers will be different. The choice between a subtracted, enlarged single-functional system and an allocated single-functional system can be decisive for the outcome of a comparison [4,19,20].

As evidenced from the increasing number of studies that examine expanded systems, conclusions drawn at workshops [5], recommendations in standards [17] and guidelines [4], system expansion is now generally recognised as a way of avoiding the allocation problem. There are, however, some critical questions to consider when using system expansion [18,21]. For example:

(1) What material will the recycled material replace? In Fig. 1, the material that is recycled from product 1 into product 2, replaces some other material. It is often assumed that the recycled material will replace virgin material of the same kind. For example, recycled paper is often assumed to replace virgin paper. However, in some cases, recycled paper may replace another type of recycled paper or another material, e.g. plastic.

(2) If more or less waste is incinerated with heat recovery, what is the alternative energy source? This question is of interest when comparing recycling and incineration as waste management strategies for different materials. It is often assumed that it is the average energy source that will replace the waste. It was, however, noted in a site-specific study that it was often other types of solid waste which would have been landfilled that replaced the recycled waste [22]. This was because incineration capacity was the factor limiting the amount of solid waste incinerated. Therefore, if parts of the solid waste are recycled they will be replaced by other types of solid waste which in other cases would have been landfilled. In the case of comparing recycling to incineration of paper packaging materials, it has been shown that assumptions concerning the alternative energy source can be decisive to the outcome of the study [9].

(3) Are the demands independent of how the products are produced? It is generally assumed in LCAs that the demands for the functions fulfilled by the
systems are independent of how they are fulfilled [23]. The system modelled is thus a simplification of the real system. The question of how severe this simplification is in different applications has, however, received very limited attention.

(4) Are the functional qualities of the products and/or materials similar and independent of how they are produced? It is often assumed that the recycled material can replace another material having similar functional qualities. If the materials are similar in the two systems that are to be compared, the processes downstream are identical and can thus be disregarded in a comparative assessment, greatly facilitating the study, as discussed above.

The answers to some of these and other questions are likely to depend on the goal of the study. Different types of goals can be analysed in different dimensions. A first fundamental dimension is concerned with whether the study is analysing the effect of a choice or not [24–26]. Studies interested in the effects of different choices are likely to be the most common, but exceptions, which may be called environmental reports, may occur. If the study concerns the effects of a choice, another important dimension concerns the time-frame of interest. Is the time-frame short (years), long (decades) or very long (centuries, millennia, etc.)? Another important dimension concerns the scale of the change. Are the changes discussed small or large compared to the scales of the systems being studied?

3.4. Discussion

A drawback of using system expansion is that the models get larger and more complicated. As noted above, the models used in system expansion are often based on several critical assumptions concerning, for example, materials and energy sources being replaced. When using system expansion there is sometimes a loss of transparency. This has been the case in some studies when subtracted systems have been used and the avoided function has not been adequately described and justified. Since the avoided function is in practice a second function, it should be as clearly described and justified as the functional unit of the study [5].

When using system expansion, several functions are studied at the same time [27]. This is not always apparent since the subtracted systems are single-functional. But, as discussed above, subtracted single-functional systems are in effect multi-functional, since they produce the same results. When using system expansion it is thus no longer possible to study one product in isolation and this can be seen as a drawback. However, this can also be seen as an advantage since it reflects the real situation. In a situation where different products or functions cannot be chosen independently, it can be seen as an advantage if the LCA model reflects this fact. For example, if a choice is made to incinerate solid waste with heat recovery, it is no longer possible to choose another energy source and this is illustrated in the expanded system. It can therefore be claimed that the model is more complete and correct when expanded system boundaries are used. Also, if expanded system boundaries are used, it can be noticed if the question posed by a decision-maker (e.g. recycle or incinerate a certain waste fraction) is too narrow and depend on other policy decisions [9].
A major advantage of expanding system boundaries is that a difficult allocation problem is avoided. The approach is recommended in the ISO standard [17]. It should, however, be noted that it is not always possible to expand the system boundaries. This is, for example, the case if no alternative production exists.

4. Multi-input processes

Waste treatment processes, e.g. landfilling and incineration, are often multi-input processes (Fig. 5). The allocation problem occurs when different waste materials are mixed and subsequently treated and the aim is to analyse the emissions caused by only one of the products (or fractions of the solid waste), e.g. product A in Fig. 5. The question is then: how should emissions be allocated to products A and B?

In this case, there now seems to be a general agreement that natural science-based causalities should be the basis for the allocation [5]. However, it should be realised that this only solves part of the question, since the causalities involved still need to be found. Understanding the causal relationships, and identifying approximations of them, are scientific and technical issues which have to be dealt with in co-operation with relevant specialists.

There is not one single factor which can be used for allocation in all cases. However, in the absence of understanding, allocation based on mass or energy or similar parameters can be used as a default, although it should be realised that the choice is more or less arbitrary. The model developed by White et al. [12] is an example where a mass-based allocation is used for calculation of most of the emission factors for incineration and landfilling of municipal solid waste. However, for some emissions which clearly derive from other waste products, other allocation principles are used, which demonstrates an appreciation for the limitations of a mass-based allocation.

Sometimes the causal relationships which should be the basis for the allocation may depend on the goal of the study, as briefly discussed at the end of Section 3.3. For example, if there is a non-linear relationship between the input of an element and the emissions of a pollutant, it may be of importance whether the change is small or large [5]. For example, when analysing which emissions should be allocated to incinerated polyvinylchloride (PVC), it may be of importance whether the change being discussed is small (e.g. increased incineration of PVC by 1 kg per year) or large (e.g. complete removal of PVC from incinerators).

The argument that natural science-based causation should be the basis for the allocation may sometimes lead to different conclusions in specific cases. This can be illustrated by the discussion on how chlorinated dioxins and other chloro–organic
compounds emitted from waste incinerators should be allocated to the inputs [5]. It is sometimes suggested that chlorinated dioxins should be allocated to different waste components in relation to their chlorine content. Others would however advocate that the emissions of chlorinated dioxins are more related to the operating conditions of the incinerator, and the pollutants should be allocated to waste components in relation to the energy content of the wastes, or some other similar parameter. This will of course have an influence on the emissions of chloro–organic compounds allocated to PVC. It will have an even larger impact on the emissions allocated to other waste components, e.g. polyethylene (PE). If chloro–organic compounds are allocated in relation to chlorine content, nothing will be allocated to PK. But, if chloro–organic compounds are allocated in relation to energy content, 1 kg of PE will cause more emissions of chlorinated dioxins and other chloro–organic compounds than 1 kg of PVC, which may come as a surprise to some.

5. Landfills and time aspects

One important difference between landfilling and most other processes which may occur in an LCA is the time-frame [28]. Emissions from landfills may prevail for a very long time, often thousands of years or more. In order to make the potential emissions from landfilling comparable to other emissions during the life cycle, the potential emissions have to be integrated over a certain time-period. This leads to two important questions: (a) what time period is of interest? (b) What happens with the remaining waste after the chosen time period?

Different principles for defining the time period can be used. One is in terms of years. Eggels and van der Ven [29] use 15 years, White et al. use 30 years [12], Rousseau et al. [30] suggest three different periods (20 100 and 500 years), Bez et al. [31] and Nielsen and Hauschild [32] use 100 years. These time periods are all rather short in relation to the time periods over which emissions of, for example, heavy metals are expected to occur. The fraction of heavy metals in municipal solid waste which is expected to be emitted during ‘the surveyable time period’ (which corresponds to approximately one century) is typically between $10^{-5}$ and $10^{-3}$ [33]. If the leachate concentrations remain constant, it will take thousands of years before half the amount of heavy metals have been leached. The suggestion by Rousseau et al. [30] that 500 years corresponds to an “indefinite reference at which point emission will have reached their theoretical yield” is therefore surprising. Why should the emissions stop after 500 years? Other time periods which have been discussed are longer, e.g. 10 000 or 1 million years [5].

Another principle which has been discussed for defining the time period is to include the time period until concentrations are lower than ‘acceptable’ or lower than ‘background’ [5]. There are, however, several problems with these approaches. One concerns how to define ‘acceptable’ or ‘background’ concentrations. It can also be questioned whether such approaches are compatible with the LCA framework since all emissions should be considered in LCA whether they are below or above
thresholds [34]. Another open question concerns the handling of situations where concentrations can be expected to rise after some time, perhaps after being below an ‘acceptable’ level. Such situations can occur if a buffer capacity is consumed by leaching, or chemical or biochemical reactions, leading to a change in pH or redoxpotential and subsequent changes in concentrations. This is expected to happen, for example, in the weathering of sulphidic mining wastes [35]. A similar situation can also be envisaged for combustion residues [36]. Increased concentrations of heavy metals in leachate from municipal solid waste has also been hypothesised [37].

Despite problems with this approach it would still be interesting to see if it could be operationalised and what results it would lead to.

Zimmermann [38] and Hellweg et al. [39] use so called ‘availability tests’ to estimate ‘the total leaching potential’ without temporal differentiation. This approach has attractive features but the ability of the tests to actually measure the total leaching potential can be questioned. It has, for example, been noted that the ‘available fraction’ could be increased by nearly a factor of 10 by some minor adjustments of the standard for the leaching test [40]. How can it then be judged which test conditions measure the total leaching potential? From a theoretical point of view, it is difficult to understand what would define ‘the total leaching potential’. For example, if the endpoint of the availability test is the equilibrium concentration, as has been suggested [40], then solubility-controlling minerals in the waste will continue to dissolve into percolating water, and the ‘availability test’ would not measure the total leachable amount but only a part of it. It is difficult to understand why minerals in waste materials would stop dissolving into percolating water after a fraction had been dissolved.

When discussing the processes that may occur in a landfill, the exact kinetics are often difficult to predict [28]. The kinetics will often depend on site-specific characteristics such as climate and waste management practices that may differ and that will often be unknown in a specific LCA. On the other hand, the events and sequences of events and processes may be better known. Our knowledge of the processes can often be described in terms of phases that landfilled materials will go through. It may therefore be easier to define the time perspective in terms of the processes and the events rather than by years [41]. We have in our work used two time periods [28,33,41,42]:

(1) ‘The surveyable time period’ which is defined as the time period it takes to reach a pseudo steady state in the landfill. The surveyable time period should correspond to approximately one century. In this case the time period is defined by the processes in the landfill. For example, for municipal solid waste landfills, the surveyable time period is defined as the time it takes to reach the later part of the methane phase when gas production is diminishing. For some types of solid wastes, it may be difficult to define a pseudo steady state within this time frame. In such cases 100 years is used as a default. The surveyable time period is regarded as a short time period.

(2) ‘The hypothetical, infinite time period’ is defined by total emission of landfilled materials. This time period is introduced to get the maximum, potential impacts.
Since the emissions during shorter time periods (decades and centuries) in some cases will be only a small part of the total landfilled amount, it is clear that the choice between looking at a shorter time period and a longer time period can be decisive for the results. It is therefore suggested that long time periods should at least be considered in a complete life-cycle assessment in order not to miss any important impacts. It is noted by Sundqvist [43] that the difference between different shorter time frames may be less important for the outcome of the studies.

The second question noted in the beginning of this section concerns the waste remaining after the chosen time period. One of the conclusions from a workshop on LCA and solid waste in 1995 [5] was that “if the time period considered is shorter than the period during which the total emissions will occur (or at least the major part of it) it may never be assumed that the remaining waste is fully inert. If for example a fixed time period is used, the remaining waste will generally not be fully inert. In that case a description of it in terms of ‘inert waste’ will not suffice. The composition of the waste in its not inert aspects will have to be specified.” It is sometimes suggested that the non-inert waste remaining after a certain time period should be dealt with in the impact assessment rather than in the inventory analysis. This shifting of the problem into another domain may, however, not make it easier to handle [5].

6. Life cycle impact assessment of ISWM systems

Life cycle impact assessment is the phase of an LCA aimed at understanding and evaluating the magnitude and significance of the potential environmental impacts of a product system [1]. In an LCA it will normally not be known where and when all the emissions take place. This is one of the reasons why LCA cannot predict actual impacts but is restricted to analysing potential impacts [34]. When analysing emissions from landfills, this situation is enforced since the emission will occur in a future situation. The emissions cannot be measured but only predicted. A consequence of the predictions is that it is potential emissions rather than actual emissions that can be included in the LCA for the landfilling processes. This can make the impact assessment more difficult because there are increased problems modelling background concentrations and other aspects which may be of importance for the impact assessment. There are, however, other situations in an LCA where emissions occur on different time scales [44]. The standard solution to this problem is to treat all emissions as if they occur at the same moment. If this is also assumed for the landfilling processes, methods that are being used for LCIA (see, e.g. the review in Ref. [34]) can also be used for landfilling processes in LCA.

There is, however, one additional aspect which relates to the discussion in the previous section: are future impacts more or less important than current impacts? If, for example, the valuation would be limited to effects within a certain time period, then the specification of emissions after this time period would be unnecessary [5]. Also, if the valuation placed a different importance on effects occurring at different times, the inventory analysis must indicate the time scale of emissions [5].
The principle that future generations should not be burdened with the environmental costs of current activities might involve placing more emphasis on future effects [5]. On the other hand, it will for many people seem quite senseless to take into account effects occurring after more than 100,000 or one million years, time scales which may be relevant for some types of waste [5].

The definition of LCA states that the assessment should include the complete life cycle, and there is no restriction in time. This suggests that all emissions should be included, regardless of when they occur. However, if one would like to put a lesser emphasis on future impacts there are two possible solutions:

6.1. A cut-off after a certain time period

If this approach is used, emissions after a certain time, and impacts associated with them are completely disregarded. The implicit assumption is that impacts after the chosen time are of no importance. This is consistent with a view that future generations are of no importance [45].

6.2. A discounting is made

The purpose of discounting is to discriminate against the future [46], for which there are several reasons. The choice of the discount rate is an ethical and ideological issue both in relation to the question of how future generations are valued and in relation to expected economic growth [45].

Discounting is currently not used in LCA. A cut-off is used in the approaches discussed above where emissions from landfills are only considered for a certain time period, and the remaining waste is described as inert if considered at all. When using that type of approach an ethical valuation is implicitly made to place no importance on impacts affecting future generations. It is important to realise that when deciding on which time period(s) to consider in an inventory analysis, an ethical valuation is in practice being made.

7. Concluding remarks

Although LCA is still a young scientific area, it has developed rapidly towards achieving consensus on a large number of key framework issues [14]. The adoption of LCA has grown along with these developments [14] and it is, for example, being used to evaluate different strategies for waste management. There are, however, methodological choices required and a number of aspects that still need to be worked out [14]. Some of these aspects are of special importance when performing LCAs of integrated solid waste management systems and have been discussed in this paper. It is, however, suggested that the aspects discussed here are not specific to LCA but generic to system models for waste management. For example, any analysis which includes landfilling processes has to make a choice concerning which time period to consider, although the choice is not always made consciously. Also,
in any analysis of a system which recovers energy and/or materials, a decision is made as to how the energy or material should be credited to the waste management system. It is therefore suggested that different areas of system analysis of waste management systems can learn and benefit from each other.

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