Application of the absolute environmental sustainability assessment framework to multifunctional systems – The case of municipal solid waste management

Jérémy Rodrigues, Natacha Gondran, Adrien Beziat, Valérie Laforest

Version of Record: https://www.sciencedirect.com/science/article/pii/S0959652621032248
Manuscript_b66d0f0b194081c198964099b1f71b29

Application of the Absolute Environmental Sustainability Assessment framework to multifunctional systems – The case of municipal solid waste management

Jérémy RODRIGUES[a], Natacha GONDRAN[a,*], Adrien BEZIAT[b], Valérie LAFOREST[a]

[a] Mines Saint-Etienne, Univ Lyon, Univ Jean Moulin, Univ Lumière, Univ Jean Monnet, ENTPE, INSA Lyon, ENS Lyon, CNRS, UMR 5600 EVS, Institut Henri Fayol, F - 42023, Saint-Etienne, France

[b] Laboratoire d'Economie des Transports, 14 avenue Berthelot, F - 69003 Lyon, France

* Corresponding author: natacha.gondran@emse.fr ; +334 77 42 01 75

Abstract

The Absolute Environmental Sustainability Assessment (AESA) framework is an emerging field in environmental research that aims at comparing the estimated environmental burden generated by a system on its life cycle to the carrying capacity that can be assigned to this system, *i.e.* the amount of impacts that it can cause without causing unacceptable impairment of ecosystem functional integrity. This paper aims to expand the AESA framework to multifunctional systems. We applied it to a simplified case study based on municipal solid waste (MSW) management, since such a system provides several functions such as reduction of the quantity and toxicity of waste, production of heat, electricity or secondary materials. Thus, we could identify the specific questions that arise from this multifunctionality in the context of AESA.

Based on the theory of AESA, we developed a four-steps methodology to identify the most significant impacts at both system and global scales. That methodology consisted of (step 1) quantifying the impacts of the studied system with conventional Life Cycle Assessment (LCA) followed by a normalisation step; (step 2) quantifying the number of beneficiaries from this system; (step 3) identifying impact categories for which the studied system causes significant impacts and global carrying capacities are exceeded; and (step 4) quantifying the Assigned Carrying Capacity (ACC) of the system, based on a utilitarian perspective and its contribution to the satisfaction of human needs.

We then applied this methodology to a simplified model of the MSW management implemented in the Lyon Metropolitan Area (France), as a proof of concept. Steps 1 to 3 helped elaborate a hierarchy of impact categories, highlighting which ones should be reduced in priority. Step 4 required significant knowledge about how the functions of the studied system were used, both directly and indirectly, to satisfy human needs. We used national Supply and Use Tables and several simplifying hypotheses to that end. We also tested different sharing principles based on macroeconomic indicators. This variety of sharing principles helped refine the hierarchy developed in Step 3 by identifying which carrying capacities were most certainly exceeded by the studied system.

Key words

Absolute Environmental Sustainability Assessment; Life Cycle Assessment; Waste management; Circular Economy

Abbreviations

ACC	Assigned Carrying Capacity
AESA	Absolute Environmental Sustainability Assessment
CC	Carrying Capacity
CE	Circular Economy
EoL	End of Life
FCE	Final Consumption Expenditures
GDP	Gross Domestic Product
GVA	Gross Value Added
IHR	Incineration with Heat Recovery
LCA	Life Cycle Assessment
LCI	Life Cycle Inventory
LCIA	Life Cycle Impact Assessment
MSW	Municipal Solid Waste
PB-LCA	Planetary Boundary based Life Cycle Assessment
RSW	Residual Solid Waste
SSW	Source-Separated Waste

1. Introduction

One of the key strategies towards sustainability is the implementation of a Circular Economy (CE), which can be defined as "an economic system that replaces the 'end-of-life' concept with reducing, alternatively reusing, recycling and recovering materials in production/distribution and consumption processes (...) with the aim to accomplish sustainable development, thus simultaneously creating environmental quality, economic prosperity and social equity, to the benefit of current and future generations" (Kirchherr et al., 2017). This definition encompasses several key concepts, such as the hierarchy between resource and waste management strategies and sustainable development, the former being the means and the latter being the end of circular economy.

One of the conditions of the so-called "environmental quality" is that human activities do not alter ecosystem functioning beyond their carrying capacities (CCs), which can be defined as the "maximum persistent impact" that ecosystems can sustain "without suffering perceived unacceptable impairment of their functional integrity" (Bjørn et al., 2020). This concept originates from the planetary boundaries framework (Rockström et al., 2009 ; Steffen et al., 2015), which defined a set of "control variables", for nine Earth Systems processes, and the associated thresholds that human impacts should not cross to maximize the probability of staying in a Holocene-like state and avoid potentially "unacceptable environmental changes" (Rockström et al., 2009). Then, as long as the environmental impacts of human activities do not exceed the associated carrying capacities (Bjørn et al., 2020), these planetary boundaries are not crossed, and Humanity would remain within its "safe operating space" (Rockström et al., 2009; Steffen et al., 2015).

In this context, when assessing the environmental impacts of their activities, stakeholders need to go beyond a mere "relative environmental sustainability assessment" (*i.e*. comparing two systems with similar functions to identify which one is best). They need to determine whether their efforts to reduce their impacts are “good enough”, from a strong sustainability perspective (Bjørn, 2015), *i.e.* if their activities are compatible with planetary boundaries. To answer this question, Bjørn and Hauschild (2015) proposed to base upon both LCA methodology and the planetary boundaries framework to develop the “Absolute Environmental Sustainability Assessment” (AESA) approach that aims to “address whether a production or consumption activity can be considered sustainable in an absolute sense” (Bjørn et al., 2019b).

This framework proposes to assign studied systems (territories, human activities or individuals) with a share of the global carrying capacities, later referred to as their assigned carrying capacity (ACC) (Bjørn et al., 2020), and to compare this ACC with the environmental pressures exerted by these systems. In the literature, several concepts similar to ACC can be found, such as “acceptable environmental burden” (Sala et al., 2020) or “share of safe operating space” (Ryberg et al. 2018). They all respond to the same constraint that human activities should not exceed them to be considered sustainable in the absolute sense. This paper focuses on the concept of ACC for more clarity.

The AESA framework offers many opportunities to highlight which impacts should be reduced in priority (Rodrigues, 2016). It could also contribute to setting reduction targets, depending on by how much human activities exceed their ACC. It is attractive from a decision-support point of view (Clift et al, 2017), but considerable research and political challenges rely within the definition of this so-called ACC, especially when it must be defined at the scale of a given product or service. These research questions are the subject of several dozen publications since 2015, see for example Ryberg et al. (2020). One of the most common approaches is to define an ACC for the system, based on the equal per capita sharing principle of the global carrying capacity among countries or individuals, possibly combined with a so-called utilitarian distribution principle between human activities (Ryberg et al., 2020). This utilitarian distribution principle states that the ACC of a specific activity should be linked with its “utility”, *i.e.* its contribution to satisfying human needs, and to “maximizing material well-being in society” (Bjørn et al. 2020). Several metrics have been proposed for this “utility”, such as economic (or gross) value-added, final consumption expenditure, calorific content, physical output (Bjørn et al. 2020), and the choice of one particular metric has a great influence on the results (Ryberg et al., 2018).

Assigning a carrying capacity to an activity can be even more complex if it has several functions, and therefore contributes in several forms to satisfying human needs. That is especially the case with activities related to Circular Economy, and in particular waste management, since it both helps to mitigate potential sources of pollutions and produces a variety of coproducts (secondary materials, energy, etc.). Furthermore, many of these functions do not *directly* satisfy human needs. For instance, metal production (from primary or secondary resources) does not contribute *per se* to human needs, but is essential for many applications – housing, transportation, etc. – that do.

This paper proposes to apply the AESA framework to complex and multifunctional systems such as municipal solid waste (MSW) management and suggests a methodology to highlight which impact categories should be monitored and reduced in priority. It is structured in several sections. Section 2 describes the theoretical background of the AESA framework used in this paper and expands this framework to multifunctional systems, based on the insights of MSW management. Section 3 presents some data and LCIA results of the case study of the MSW management implemented in the Lyon Metropolitan Area (France), later referred to as Grand Lyon. Section 4 presents the data and hypotheses used to apply our expanded AESA framework, and the results. Section 5 concludes the case study and, finally, Section 6 discusses the conclusions and methodological perspectives of this work.

2. AESA framework

This section describes the theoretical framework of AESA and its current applications in LCA. Then, it presents how we propose to use and expand this framework to the study of complex systems.

2.1. Planetary boundary-based Life Cycle Assessment (PB-LCA)

One of the AESA methods, also called PB-LCA method, aims to develop normalisation factors that express impact indicators, at mid-point level, as shares of the Earth's carrying capacities $\boldsymbol{CC_i}$ (impact unit per year) corresponding to the studied impact categories $\boldsymbol{i}$.

In particular, in the European Union, the Joint Research Centre has recommended a set of impact assessment methods and indicators, grouped within the Environmental Footprint (EF) methodology (Fazio et al., 2018), which is an update of previous works from the International reference Life Cycle Data (ILCD) (EC- JRC, 2011). Based on this methodology, AESA based normalisation factors were also proposed to highlight the most significant impact categories of a system, with regards to planetary boundaries (Sala et al., 2020). These factors estimate the ACC of an individual, *i.e*. the maximum amount of impact that an average person can generate each year without compromising proper ecosystem functioning (Bjørn et al., 2020), noted $\frac{\boldsymbol{CC_i}}{\boldsymbol{pop}}$ (impact unit per capita per year) in this paper, where $\boldsymbol{pop}$ is the world population (capita). These factors imply that all humans are treated equally and granted with the same ACC, which is described by Ryberg et al. (2020) as an egalitarian distribution of $\boldsymbol{CC_i}$ between individuals worldwide.

Table 1 shows the impact categories and normalisation factors in question in 2010 (Sala et al., 2020). We updated these factors in this paper to account for the increased population in 2019. That implies that the same CCs have to be shared between more individuals, thereby reducing the amount of impacts that each individual can safely cause. Table 1 also shows the ratios by which impact categories $\boldsymbol{i}$ exceed their global carrying capacities – which were drawn from Sala et al. (2020), in their Supplementary Material, and described as the ratios over planetary boundaries for global normalisation factors. These ratios, later referred to as $\boldsymbol{\gamma_i}$ factors, were considered necessary by Rodrigues (2016) to identify different levels of priority for impact reduction. They will be useful in section 2.2.

Note that land use (LU) related impacts were not normalized within the AESA framework used in this paper because of scope and unit differences. Indeed, the Land Use impact category considered in EF calculates a score aggregating the degradation of several soil properties (erosion control, freshwater filtration, groundwater recharge, water cycle regulation, fertility), based on the LANCA model (Bos et al., 2016; Horn and Maier, 2018; De Laurentiis et al., 2019). Sala et al. (2020) could assign a Carrying Capacity for soil erosion only, which is why no normalisation factor is available (yet) for the single score indicator. Furthermore, these indicators still need major improvements to be fully integrated and relevant (Bjørn et al., 2019a; Thoumazeau et al., 2019; Rodrigues, 2016). Finally, this pressure indicator does not fit with the control variable that was defined as a planetary boundary by Steffen et al. (2015) for land-system as they proposed the change area of forested land as % of original forest cover, at the global scale, or the area of forested land as percent of potential forest at the biome scale.

Table 1: Impact categories recommended by the ILCD (EC- JRC, 2011; Fazio et al., 2018) and associated ACCs, adapted from Bjørn and Hauschild (2015) and Sala et al. (2020)

Code name	Impact category (i)	Unit	Assigned Carrying Capacities per capita and per year $\left(\frac{CC_i}{pop}\right)$		Ratio over Planetary Boundaries* (γ_i)
			Value 2010**	Value 2019**	
AC	Acidification	mol H+-Eq	1.45E+02	1.30E+02	0.38
GW	Global Warming***	kg CO2-Eq	9.85E+02	8.83E+02	8.17
ECOTOX	Ecotoxicity	CTU	1.90E+04	1.70E+04	0.62
FEU	Freshwater Eutrophication	kg P-Eq	8.40E-01	7.53E-01	1.92
FRD	Fossil Resources Depletion	MJ	3.24E+04	2.91E+04	2.01
HTOX_c	Human Toxicity (cancer)	CTUh	1.39E-04	1.25E-04	0.28
HTOX_nc	Human Toxicity (non-cancer)	CTUh	5.93E-04	5.32E-04	0.80
IR	Ionising Radiations	kg U235-Eq	7.62E+04	6.83E+04	0.00
LU	Land Use	kg soil loss****	1.84E+03	-	0.02
MEU	Marine water eutrophication	kg N-Eq	2.90E+01	2.60E+01	0.67
MRD	Mineral Resources Depletion	kg Sb-Eq	3.18E-02	2.85E-02	2.01
ODP	Ozone layer Depletion	kg CFC-11-Eq	7.80E-02	6.99E-02	0.62
PM	Particulate Matter	disease incidence	7.47E-05	6.70E-05	7.95
POF	Photochemical Ozone Formation	kg NMVOC-Eq	5.88E+01	5.27E+01	0.69
TEU	Terrestrial Eutrophication	mol N-Eq	8.87E+02	7.95E+02	0.20
WU	Water Use	m3 water-Eq	2.63E+04	2.36E+04	0.44

* Factors by which global impacts exceed their assigned carrying capacities, mentioned by Sala et al. (2020) in their supplementary material (global normalisation factors)

** Values for years 2010 and 2019, considering a world population of respectively 6,916,183,482 inhabitants (Sala et al., 2020) and 7,713,468,000 (UNO, 2019)

*** In their work, Sala et al. (2020) refer to Climate Change instead of Global Warming. However, we preferred the latter in this paper to avoid confusion within acronyms between Climate Change and Carrying Capacity.

**** No normalisation factor for the single score indicator used in the EF (unit: point). Only one component (soil erosion) has an associated normalisation factor.

2.2. Methodology for AESA at system and global scales

This section describes some of the indicators that can be produced by the AESA framework to facilitate decision making, how they are calculated, as well as the procedure that we implemented to assign a carrying capacity to waste treatment processes (see results in section 4). Its theoretical roots are based on the works of Bjørn and Hauschild (2015) and Sala et al. (2020) on normalisation (see section 2.1) and on those on sustainability coefficients, initiated by McElroy et al. (2008) and further developed by more recent studies (Rodrigues, 2016; Wolff, 2017; Wolff et al., 2017).

Thus, the latter propose that the carrying capacity $\boldsymbol{ACC_{\Sigma,i}}$ (impact unit per year) that can be assigned to a monofunctional system $\boldsymbol{\Sigma}$ for an impact category $\boldsymbol{i}$ be a fraction $\boldsymbol{a_{\Sigma,i}}$ (dimensionless) of the CC assigned to the $\boldsymbol{b_\Sigma}$ individuals (capita) who benefit from that system $\boldsymbol{\Sigma}$, as expressed by Equation 1:

Equation 1

$$\boldsymbol{ACC_{\Sigma,i} = a_{\Sigma,i} * b_\Sigma * \frac{CC_i}{pop}}$$

The choice of the allocation key $\boldsymbol{a_{\Sigma,i}}$ raises ethical and political issues, since its value, between 0 and 100%, indicates the share that the system $\boldsymbol{\Sigma}$ can use out of the carrying capacity of overall human activities. According to Ryberg et al. (2020), this distribution principle should be "utilitarian", meaning that this factor $\boldsymbol{a_{\Sigma,i}}$ should ideally reflect the importance of the function fulfilled by system $\boldsymbol{\Sigma}$ to satisfy human needs. This value being intrinsic to the delivered function in question, it should not depend on the studied impact category. Therefore, $\boldsymbol{a_{\Sigma,i}}$ can be simplified as $\boldsymbol{a_\Sigma}$[1]. In practice, a relevant metric remains to be found (Diener, 2000; Ryberg et al., 2020). For instance, functions related to healthcare services and luxury products would be assigned with very different carrying capacities if the sharing principle was based on the vital nature of these items or if it was based on their contribution to Gross Domestic Product (GDP).

In order to compare the life cycle impacts $\boldsymbol{I_{\Sigma,i}}$ of $\boldsymbol{\Sigma}$ for impact category $\boldsymbol{i}$ for its yearly production (impact unit per year) and its $\boldsymbol{ACC_{\Sigma,i}}$, we can calculate the ratio between these two indicators using Equation 2 (Rodrigues, 2016; Wolff et al., 2017). This environmental sustainability ratio $\boldsymbol{\beta_{\Sigma,i}}$ (dimensionless) can highlight different levels of severity, as illustrated by Table 2.

Equation 2

$$\boldsymbol{\beta_{\Sigma,i} = \frac{I_{\Sigma,i}}{ACC_{\Sigma,i}}}$$

Now, consider the dimensionless ratio $\boldsymbol{p_{\Sigma,i}}$ between the impacts $\boldsymbol{I_{\Sigma,i}}$ generated by this monofunctional system $\boldsymbol{\Sigma}$ and the carrying capacity attributed to the $\boldsymbol{b_\Sigma}$ inhabitants benefiting from $\boldsymbol{\Sigma}$ (Equation 3). This ratio is the fraction of the CC of the beneficiaries that is consumed by $\boldsymbol{\Sigma}$ and reflects the magnitude of this impact.

Equation 3

$$\boldsymbol{p_{\Sigma,i} = \frac{I_{\Sigma,i}}{b_\Sigma * \frac{CC_i}{pop}}}$$

Factoring Equation 1 and then Equation 3 into Equation 2, we can derive the following Equation 4, which can be used to compare more easily the contribution of $\boldsymbol{\Sigma}$ to the consumption of the CC of its beneficiaries ($\boldsymbol{p_{\Sigma,i}}$) to its contribution to their needs ($\boldsymbol{a_\Sigma}$), regardless of the chosen metrics. Table 2 shows how this comparison highlights different levels of severity for a given impact category.

Equation 4

[1] Note that if the allocation key was based on the "grandfathering approach" (Wolff et al., 2017), *i.e.* if the $\boldsymbol{ACC_i}$ allocated to an industry were proportional to its past contributions to global impacts $\boldsymbol{i}$, (1) coefficients $\boldsymbol{a_{\Sigma,i}}$ would remain dependant on impact categories $\boldsymbol{i}$ and (2) the distribution principle between industries would be based on "acquired rights" instead of "utilitarian".

$$\boldsymbol{\beta_{\Sigma,i}} = \frac{\boldsymbol{p_{\Sigma,i}}}{\boldsymbol{a_{\Sigma}}}$$

Table 2: Different severity levels of environmental impacts - adapted from Wolff et al. (2017) and Rodrigues (2016)

Value of $\boldsymbol{\beta_{\Sigma,i}}$	Equivalent equation	Corresponding severity level
$\boldsymbol{\beta_{\Sigma,i} \leq 1}$	$\boldsymbol{p_{\Sigma,i} \leq a_{\Sigma}}$ Or: $\boldsymbol{I_{\Sigma,i} \leq ACC_{\Sigma,i}}$	Studied system uses less than its ACC and can be considered as sustainable for this impact category.
$\boldsymbol{\beta_{\Sigma,i} > 1}$	$\boldsymbol{p_{\Sigma,i} > a_{\Sigma}}$	Studied system uses more than its ACC and is not sustainable.
$\boldsymbol{\beta_{\Sigma,i} > \frac{1}{a_{\Sigma}}}$	$\boldsymbol{p_{\Sigma,i} > 1}$ Or: $\boldsymbol{\frac{I_{\Sigma,i}}{\left(\frac{CC_i}{pop}\right)} > b_{\Sigma}}$	Studied system uses more than the ACC of the individuals it supports and is strongly unsustainable.
With $\boldsymbol{\beta_{\Sigma,i}} = \frac{\boldsymbol{I_{\Sigma,i}}}{\boldsymbol{ACC_{\Sigma,i}}}$, $\boldsymbol{ACC_{\Sigma,i} = a_{\Sigma} * b_{\Sigma} * \frac{CC_i}{pop}}$ and $\boldsymbol{p_{\Sigma,i}} = \frac{\boldsymbol{I_{\Sigma,i}}}{\boldsymbol{b_{\Sigma} * \frac{CC_i}{pop}}}$		

Based on these elements, the following four-steps procedure is proposed to conduct a rigorous application of the AESA framework:

Step 1. Calculation of the normalised life cycle impacts of the system $\boldsymbol{\Sigma}$ over one year. Indeed, the values $\boldsymbol{I_{\Sigma,i}/\frac{CC_i}{pop}}$ highlight the most significant impacts with regards to Earth's carrying capacity;

Step 2. Comparison of these impacts, expressed in sustainable capita, with the number of beneficiaries $\boldsymbol{b_{\Sigma}}$ of the function provided by $\boldsymbol{\Sigma}$ by calculating the fraction $\boldsymbol{p_{\Sigma,i}}$ of their CC that is consumed by $\boldsymbol{\Sigma}$ using Equation 3;

Step 3. Comparison of this fraction $\boldsymbol{p_{\Sigma,i}}$ (or the normalized impacts, if $\boldsymbol{b_{\Sigma}}$ could not be estimated) with the factors $\boldsymbol{\gamma_i}$ by which global carrying capacities are exceeded by human activities at global scale (see Table 1). This allows identifying impact categories that raise issues at both global and system scales. Indeed, for a given impact category $\boldsymbol{i}$, if $\boldsymbol{\Sigma}$ showed a high value of $\boldsymbol{p_{\Sigma,i}}$ or exceeded its assigned carrying capacity ($\boldsymbol{\beta_{\Sigma,i} > 1}$), that would be much more acceptable if this carrying capacity was not exceeded globally ($\boldsymbol{\gamma_i}$**<1**) than if we had $\boldsymbol{\gamma_i}$**>1** (Rodrigues, 2016);

Step 4. Calculation of one – or preferably several values of – $\boldsymbol{a_{\Sigma}}$ and comparison with $\boldsymbol{p_{\Sigma,i}}$ to assess the sustainability of $\boldsymbol{\Sigma}$ and calculation of the corresponding $\boldsymbol{\beta_{\Sigma,i}}$ coefficients using Equation 4. The motivation behind the use of different sharing principles is to comply with the recommendations of Ryberg et al. (2020) to "obtain a range of AESA results rather than a single result". It also increases the number of severity levels introduced in Table 2.

2.3. Expansion to multifunctional systems

The previous procedure assumes a system providing only one function directly to final users. In the case of a system providing several functions simultaneously, it may be more difficult to estimate the number of beneficiaries of the various functions and the fraction of the carrying capacity that can be assigned to the system. This section proposes how the AESA framework could be expanded to assess complex systems while remaining coherent with the framework presented above. Considering the latest definition and Equation 1, the system $\boldsymbol{\Sigma}$ is considered as providing only one function that was assessed with factors $\boldsymbol{b_{\Sigma}}$ (number of people whose needs are satisfied) and $\boldsymbol{a_{\Sigma}}$ (share of total CC allocated to $\boldsymbol{\Sigma}$, or rather: to the function $\boldsymbol{f}$ it provides).

However, some systems such as municipal solid waste (MSW) management, for example, may be more complex for at least two reasons:

(1) They may provide several functions, such as elimination of wastes that can be an environmental/health hazard and provision of secondary products (heat and electricity recovery, recycled materials);
(2) Many of these functions $\boldsymbol{f'}$ do not satisfy human needs *directly* but are used as inputs by activities that satisfy these needs. For instance, final consumers do not use recycled iron *per se*, but products made from this material (and other inputs).

From reason (1), we claim that $\boldsymbol{ACC_{\Sigma,i}}$, the CC allocated to $\boldsymbol{\Sigma}$, should ideally reflect the full magnitude of the functions it provides. That implies changing the way the coefficients mentioned in section 2.2 are considered:

- $\boldsymbol{a_{\Sigma}}$ becomes $\boldsymbol{a_f}$ and reflects the value of a function $\boldsymbol{f}$ (product or service) provided to humans;
- $\boldsymbol{b_{\Sigma}}$ becomes $\boldsymbol{b_{\Sigma,f}}$ and reflects, for said function $\boldsymbol{f}$, the number of people whose needs are satisfied by $\boldsymbol{\Sigma}$. It can be calculated using Equation 5, as the ratio between the quantity $\boldsymbol{q_{\Sigma,f}}$ of function $\boldsymbol{f}$ provided by $\boldsymbol{\Sigma}$ (number of functional units per year, *e.g.* number of tons of municipal solid waste treated by $\boldsymbol{\Sigma}$ per year) and the yearly final demand per capita $\boldsymbol{d_f}$ for that function (e.g. number of tons of municipal solid waste treated per capita per year)

For more clarity, Table 3 compiles the meaning of the different terms, with examples in the case of Municipal Solid Waste management.

Equation 5

$$\boldsymbol{b_{\Sigma,f}} = \frac{\boldsymbol{q_{\Sigma,f}}}{\boldsymbol{d_f}}$$

Then the carrying capacity $\boldsymbol{ACC_{\Sigma,i}}$ allocated to the system $\boldsymbol{\Sigma}$ becomes the sum of all the shares related to the functions $\boldsymbol{f}$ it provides, as shown in Equation 6:

Equation 6

$$\boldsymbol{ACC_{\Sigma,i}} = \left(\sum_{f}\left(\boldsymbol{a_f} * \boldsymbol{b_{\Sigma,f}}\right)\right) * \frac{\boldsymbol{CC_i}}{\boldsymbol{pop}} = \left(\sum_{f}\left(\boldsymbol{a_f} * \frac{\boldsymbol{q_{\Sigma,f}}}{\boldsymbol{d_f}}\right)\right) * \frac{\boldsymbol{CC_i}}{\boldsymbol{pop}}$$

Furthermore, reason (2) implies that all functions $\boldsymbol{f'}$ that are not used directly by humans to satisfy their needs should be redistributed to other systems that provide such functions $\boldsymbol{f}$. For instance, recycled iron produced by waste treatment ($\boldsymbol{f'}$) should be redistributed to systems that require iron throughout their lifecycle to produce equipment for final users (cars, buildings, electric grid, etc.). Therefore, the amounts $\boldsymbol{q_{\Sigma,f}}$ of functions $\boldsymbol{f}$ provided indirectly to humans by $\boldsymbol{\Sigma}$ (quantity of function for final consumption per year) can be estimated using Equation 7, knowing (a) the quantities $\boldsymbol{q_{\Sigma,f'\to f}}$ of intermediate functions $\boldsymbol{f'}$ produced by $\boldsymbol{\Sigma}$ (quantity of intermediate function per year) and used to produce functions $\boldsymbol{f}$ and (b) the intermediate demand $\boldsymbol{d_{f'\to f}}$ for function $\boldsymbol{f'}$ required to produce one unit of $\boldsymbol{f}$:

Equation 7

$$\boldsymbol{q_{\Sigma,f} = \frac{q_{\Sigma,f'\to f}}{d_{f'\to f}}}$$

However, these quantities need to be weighted by a factor $\boldsymbol{c_{f'\to f}}$ (between 0 and 100%) to account for the fact that the production of functions $\boldsymbol{f}$ requires many other inputs except for just functions $\boldsymbol{f'}$. For instance, a car factory would require steel – which it may get from a steel recycling facility – but also motor components, magnets, rubber, etc. which may not be produced by MSW management. This factor $\boldsymbol{c_{f'\to f}}$ could be calculated based on the contribution of inputs $\boldsymbol{f'}$ to the expenses required by (or the price formation of) $\boldsymbol{f}$. These elements can then be computed into Equation 8:

Equation 8

$$\boldsymbol{ACC_{\Sigma,i} = \frac{CC_i}{pop} * \left[\left(\sum_f \left(a_f * \frac{q_{\Sigma,f}}{d_f}\right)\right) + \left(\sum_f \left(a_f * \frac{\sum_{f'}\left(c_{f'\to f} * \frac{q_{\Sigma,f'\to f}}{d_{f'\to f}}\right)}{d_f}\right)\right)\right]}$$

$$\boldsymbol{= \frac{CC_i}{pop} * \left[\sum_f \left(a_f * \frac{\left(q_{\Sigma,f} + \sum_{f'}\left(\frac{c_{f'\to f}}{d_{f'\to f}} * q_{\Sigma,f'\to f}\right)\right)}{d_f}\right)\right]}$$

In other words, it is much easier to quantify the ACC of systems only providing functions directly to final users (*e.g.* food, clothing), using Equation 1 (for monofunctional systems) or Equation 6 (for multifunctional systems). The AESA of systems with intermediate functions (*i.e.* that do not directly satisfy human needs) requires extensive knowledge about all the products and services used by final users ($\boldsymbol{d_f}$), their respective demands for such functions ($\boldsymbol{d_{f'\to f}}$), as well as an appreciation of the importance of these functions to satisfy these needs or to produce these products and services ($\boldsymbol{a_f}$, $\boldsymbol{c_{f'\to f}}$). Such data may be available, at least partially, within economic or physical Input / Output Tables.

One way to simplify Equation 8, when sufficient data is missing, is to consider a specific function $\boldsymbol{All}$ that is the satisfaction the overall needs of a person. Then, by definition:

- $\boldsymbol{a_{All}}$ = 100%,
- $\boldsymbol{q_{\Sigma,f'\to All}}$ is the quantity of function $\boldsymbol{f'}$ produced by $\boldsymbol{\Sigma}$ (number of functional units per year) and used by other activities to satisfy overall needs – assuming that $\boldsymbol{f'}$ is made available to an average market of consumers,
- $\boldsymbol{d_{f'\to All} * d_{All}}$ is the intermediate demand for function $\boldsymbol{f'}$ required to satisfy the overall needs of one person (number of functional units per capita per year),

- and $\boldsymbol{c_{f' \to All}}$ is an average estimation of the importance of function $\boldsymbol{f'}$ to satisfy overall needs. It is written in a different format from contributions $\boldsymbol{a_f}$ to highlight that this contribution to overall needs is indirect and can be estimated using a different metrics.

Then we can define $\boldsymbol{b_{\Sigma, f' \to All}}$ as the number of persons whose overall needs are satisfied by the function $\boldsymbol{f'}$ provided by $\boldsymbol{\Sigma}$. It would be calculated using Equation 9:

Equation 9

$$\boldsymbol{b_{\Sigma,f'\to All}} = \frac{\boldsymbol{q_{\Sigma,f'\to All}}}{\boldsymbol{d_{f'\to All}} * \boldsymbol{d_{All}}}$$

Factoring Equation 9 into Equation 8, we can derive the following Equation 10:

Equation 10

$$\boldsymbol{ACC_{\Sigma,i}} = \frac{\boldsymbol{CC_i}}{\boldsymbol{pop}} * \left[\sum_{f}\left(\boldsymbol{a_f} * \frac{\boldsymbol{q_{\Sigma,f}}}{\boldsymbol{d_f}}\right) + \sum_{f'}\left(\boldsymbol{c_{f'\to All}} * \boldsymbol{b_{\Sigma,f'\to All}}\right)\right]$$

It is important to remember, though, that despite their apparent complexity, Equation 8 and Equation 10 boil down to assessing how many people benefit from $\boldsymbol{\Sigma}$ to satisfy their needs, both directly and indirectly, and the relative importance of each one of these needs ($\boldsymbol{a_f}$). Thus, when applying the methodology mentioned in section 2.2 to a multifunctional system, one may retain one function as the main function of this system (*e.g.* waste treatment). The choice of the main function in question could be guided by the facility to estimate the number of beneficiaries $\boldsymbol{b_\Sigma}$ of this main function (see **Step 2**). Then the calculation of an average contribution to human needs $\boldsymbol{a_\Sigma}$ (**Step 4**) can be done using Equation 11.

Equation 11

$$\boldsymbol{a_\Sigma} = \frac{\boldsymbol{1}}{\boldsymbol{b_\Sigma}} * \left[\sum_{f}\left(\boldsymbol{a_f} * \frac{\left(\boldsymbol{q_{\Sigma,f}} + \sum_{f'}\left(\frac{\boldsymbol{c_{f'\to f}}}{\boldsymbol{d_{f'\to f}}} * \boldsymbol{q_{\Sigma,f'\to f}}\right)\right)}{\boldsymbol{d_f}}\right)\right]$$
$$= \frac{\boldsymbol{1}}{\boldsymbol{b_\Sigma}} * \left[\sum_{f}\left(\boldsymbol{a_f} * \frac{\boldsymbol{q_{\Sigma,f}}}{\boldsymbol{d_f}}\right) + \sum_{f'}\left(\boldsymbol{c_{f'\to All}} * \boldsymbol{b_{\Sigma,f'\to All}}\right)\right]$$

Table 3: Summary of the parameters required to calculate the Assigned Carrying Capacity of a system

Notation, parameter	Meaning	Example (and associated unit)
$\boldsymbol{\Sigma}$	Studied system	Municipal Solid Waste management chain
$\boldsymbol{i}$	Studied impact category	Global Warming
$\boldsymbol{CC_i}$	Global carrying capacity for $\boldsymbol{i}$	Yearly total greenhouse gas emissions allowed (kg CO2-Eq per year)
$\boldsymbol{pop}$	Global human population	(capita)
$\boldsymbol{ACC_{\Sigma,i}}$	Carrying Capacity assigned to $\boldsymbol{\Sigma}$ for $\boldsymbol{i}$	Maximum amount of CO2 emissions allowed (kg CO2-Eq per year)
$\boldsymbol{f}$	Function provided directly to final users	MSW disposal (avoidance of health hazard), heat (household use), cars, buildings, etc.
$\boldsymbol{f'}$	Function provided by $\boldsymbol{\Sigma}$ to other industries that satisfy human needs	Production of recycled metals, plastics, paper, heat (industrial use)
$\boldsymbol{b_{\Sigma,f}}$	Number of final users whose needs are satisfied by $\boldsymbol{\Sigma}$ for a given function $\boldsymbol{f}$	Number of people whose needs for MSW treatment or household heating are fully covered (capita)
$\boldsymbol{q_{\Sigma,f}}$	Quantity of function $\boldsymbol{f}$ provided yearly by $\boldsymbol{\Sigma}$ to final users	Quantity of MSW treated, quantity of heat made available to final users (t, MWh, etc. per year)
$\boldsymbol{q_{\Sigma,f'\rightarrow f}}$	Quantity of intermediate function $\boldsymbol{f'}$ produced yearly by $\boldsymbol{\Sigma}$ and used to produce some function $\boldsymbol{f}$	Quantity of metals, plastics, paper, heat used by industries to satisfy final users (cars, buildings, books, electricity, food, etc.) (t, MWh, etc. per year)
$\boldsymbol{d_f}$	Yearly final demand per capita for function $\boldsymbol{f}$	Yearly MSW production, household heating demand, manufacture of cars, buildings, electricity production, etc. per capita (t, MWh or item per capita per year)
$\boldsymbol{d_{f'\rightarrow f}}$	Intermediate demand for function $\boldsymbol{f'}$ required to produce one unit of some function $\boldsymbol{f}$	Quantity of recycled iron used to manufacture one car (t, MWh, etc. per item)
$\boldsymbol{a_f}$	Contribution of function $\boldsymbol{f}$ to human needs	Contribution of MSW treatment, household heating, cars, to final demand expenditure (%)
$\boldsymbol{c_{f'\rightarrow f}}$	Contribution of input $\boldsymbol{f'}$ to the expenses required by (or the price formation of) $\boldsymbol{f}$	Contribution of recycled iron to the price of a car (%)

3. Case study of Municipal Solid Waste management

In order to demonstrate the feasibility of the proposed AESA framework for a multifunctional system, we used the case study of the MSW management implemented by Grand Lyon. Before applying the AESA framework to this case study, an evaluation of the impacts of this system is needed. This section shows the first 3 steps of the LCA that we conducted (goal and scope definition, life cycle inventory, life cycle impact assessment), according to ISO14040 standards, using simplifying data and the OpenLCA 1.9 software.

3.1. Goal and scope of study

The goal of this LCA is to assess how much of the Assigned Carrying Capacity of the Grand Lyon is consumed/exhausted by the municipal solid waste management and whether it can be considered sustainable or not.

3.1.1. Functional unit, allocation procedure and background modelling

The studied system is the supply-chain that handles the MSW generated by Grand Lyon's inhabitants, from the moment they leave households to the moment they are either eliminated or valorised. The year 2017 is chosen as reference because it is the most recent year for which we have sufficient data. Then, 1,385,927 people inhabited the 59 municipalities included within Grand Lyon (INSEE, 2020a) and they generated 466,486 tons of waste presented by Table 4, plus 48,075 tons of other wastes (wood products, furniture, electronic waste, gypsum, mineral oil, textiles), for a total of 535,358 tons of waste.

When valorised, either through recycling or incineration with heat recovery (IHR), the associated coproducts (recycled materials, energy) are considered to avoid the production of primary products, following a scope expansion procedure. The detailed MSW mix, and associated valorisation coproducts are described in Table 4. The other generated waste types mentioned above represent less than 10% of the total mass of generated wastes and 5% of corresponding transportation needs, as illustrated in Supplementary Material. They are not part of the scope of this study.

In this context, the main functional unit associated with the studied system is the collection, transportation, sorting, storage, treatment and/or valorisation of the quantity of MSW that are generated by Grand Lyon's inhabitants in one year (466,486 tons mentioned in Table 4). Incineration and recycling processes have other functions that are included in this study: the provision of the secondary products (materials and energy) mentioned in Table 4.

Background processes, *i.e* processes upstream or downstream the studied system that cannot be influenced directly by the decision-makers (*e.g.* production of the equipment and inputs necessary for MSW treatment), are modelled following the zero-burden assumption, which states that any background waste treatment process takes the whole burden of waste treatment, and that their potential coproducts are burden-free. Therefore, the CutOff model of the ecoinvent 3.6 database is used (Moreno Ruiz E. et al., 2019).

Table 4: Characteristics of the Municipal Solid Waste generated by Grand Lyon in 2017

Waste category	Quantity generated in 2017 (ton per year)	Treatment chain	Coproducts	Avoided products	Quantities of secondary products**
Residual waste	310,505	Incineration / Landfill	Heat, electricity, metals, bottom ash	Heat	182,343 MWh
				Electricity	95,953 MWh
Sorting waste*	20,797			Steel	4,283 t
				Aluminum	664 t
				Gravel	52,148 t
Paper and cardboard	44,675	Recycling	Recycled paper and cardboard	Paper	21,580 t
				Cardboard	26,332 t
Glass	30,681	Recycling	Recycled glass	Glass	49,090 t
Gravel	30,637	Recycling	Recycled gravel	Gravel	30,637 t
Green waste	28,621	Composting	Compost	Mineral fertilizers	108 t nitrogen
					62 t phosphate
					93 t potassium
Plastics	3,093	Recycling	Recycled plastics	Plastics	2,418 t
Bulk material	18,274	Landfill	//	//	//
Total	**466,486**				

* Sorting waste is waste that has been collected separately from residual waste, sent to a sorting facility, sorted and discarded because of its insufficient quality or of its inadequacy with downstream recycling requirements.

** Quantities are measured for secondary products generated by incineration, using data from Grand Lyon, or estimated with ecoinvent for the other wastes. Some quantities of secondary products exceed those of recycled wastes (paper and cardboard, glass). That is because, according to ecoinvent datasets, waste represent only a fraction of the inputs used to produce secondary materials. Other products and raw materials are also used.

3.1.2. Scope definition

Waste elimination encompasses several steps:

- Waste collection and transportation to a collection, sorting, treatment, disposal, or valorisation plant,
- Sorting, treatment, and disposal processes,
- Production of coproducts (energy, secondary materials) and the associated avoided production of primary products,
- Transportation of said coproducts and avoided primary products.

Several phases are excluded from the scope of this study:

- The use of the product before it became waste: we assume that it would happen anyway;
- Waste transportation by inhabitants to containers (glass), which is assumed to be done by walking, which is the overwhelmingly predominant transport mode for glass containers in French urban context (ADEME, 2012). Modelling their transportation with motorised vehicles would require (1) hypotheses on the involved vehicle fleet, average transported quantities,

transportation distances and (2) assuming that waste disposal is the only motivation of such trips, which can safely be assumed *not* to be the case (ADEME, 2012);

- The construction and end of life of waste collection containers, centres and equipment, the impacts of which are assumed to be negligeable, when compared to those of sorting plants and other treatment processes;
- The transportation of coproducts and avoided transportation of primary products: no data were available to model them and they can be allocated to the final users of said products. Sensitivity analyses could however highlight when changes in transportation distances may favour recycling over primary production or not (Merrild et al., 2012).

3.1.3. Chosen Impact Assessment method

At the time when this study was realised, the EF methodology had not yet been implemented within the OpenLCA software. Therefore, a previous version of EF, referred to as ILCD2018, was used for this LCA. Some updates were implemented manually to ensure full characterisation of nitrogen and sulphur compounds and particulate matter. Indeed, that methodology did not account for the elementary flows for which the country of emission was mentioned. For instance, in OpenLCA, the flow "Ammonia, FR" describes an emission of ammonia in France. It is not characterised by ILCD2018. We added characterisation factors related to this flow with the same values as those related to the flow "Ammonia", and repeated that for all relevant substances.

3.2. Life Cycle Inventory

This section presents some of the main life cycle phases of the studied system, as well as the corresponding inputs and outputs. Complete modelling data and hypotheses for each phase are provided in Supplementary Material. Figure 1 gives the distribution of the different types of MSW produced between the different collection strategies implemented by Grand Lyon (including a very small fraction of waste from other territories). These strategies include (the figures in brackets being the quantities collected in 2017) (Grand Lyon, 2017):

- Residual Solid Waste (RSW) collection from kerbside bins (310,505t), whereby all wastes are mixed, do not undergo any sorting process but instead are either incinerated (301,895t) or landfilled in a residual landfill facility (8,610t). The composition of RSW is known thanks to measurement campaigns;
- Source-separated waste (SSW) collection from a mix of dedicated kerbside bins and drop-off containers (64,066t), whereby all wastes are sent to a sorting facility. High quality materials (paper, paperboard, plastics, metals) are recycled. However, a significant proportion of materials (20,797t *i.e.* 34% of SSW) is in fact misplaced RSW that cannot be recycled and is either incinerated (16,888t) or landfilled (3,909t);
- Glass containers collect glass (30,681t), before it is transported to a sorting site and recycled;
- Waste collection centres collect different types of wastes (132,761t). Waste producers transport themselves their waste to these facilities. They are then sorted and recycled (glass, metals, construction waste), composted (gardening waste) or landfilled (bulky waste), with all the subsequent transportations.

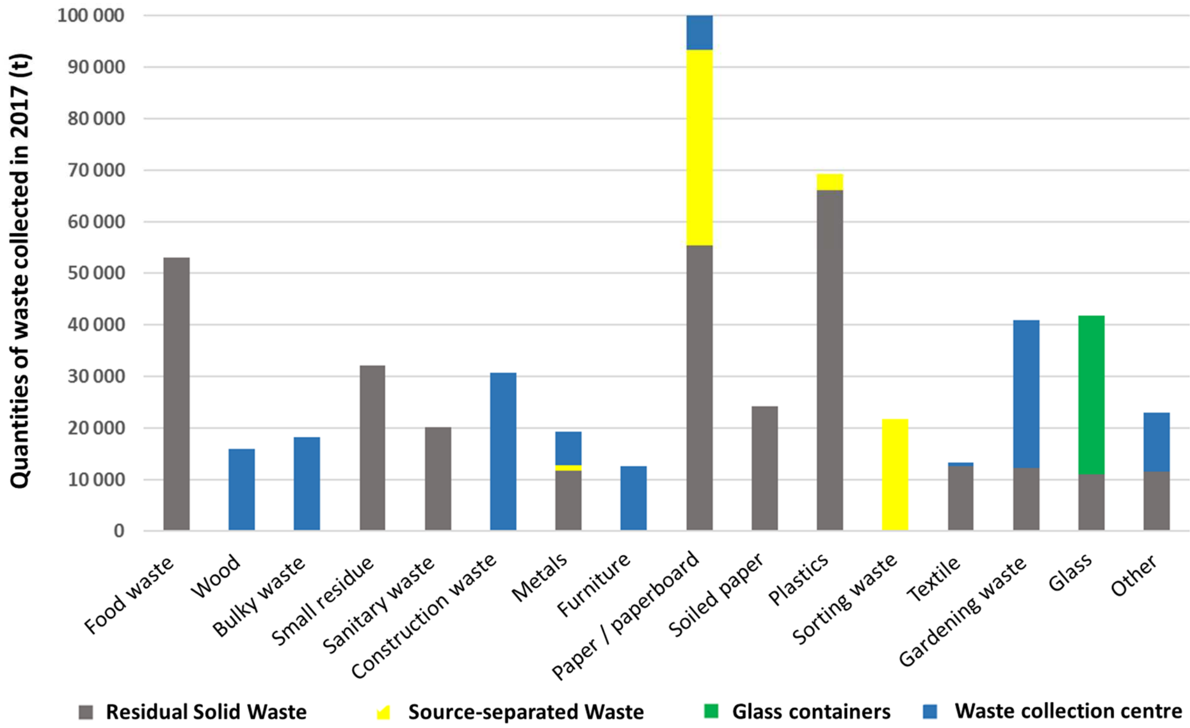

Figure 1: Waste repartition according to their type and collection mode (Total: 538,012 t)

3.3. Life Cycle Impact Assessment

The main Life Cycle Impact Assessment (LCIA) results of the studied system are presented in Supplementary Material, since they are not the main subject of this paper, which we chose to focus on the AESA perspective.

4. Contribution analysis and interpretation within AESA framework

This section shows the data that were used for the AESA of MSW management and highlights the challenges faced when implementing the four-steps methodology introduced in sections 2.2 and 2.3. It also presents the results obtained for each step, under some simplifications.

For **Step 1**, the life cycle impacts of the studied system $\boldsymbol{I_{\Sigma,i}}$ are those quantified in Section 3.3 and detailed in the Supplementary Material, and the normalisation factors $\frac{\boldsymbol{CC_i}}{\boldsymbol{pop}}$ are those shown in Table 1 for 2019. For **Step 2**, the number $\boldsymbol{b_\Sigma}$ of people benefiting from the main function (*i.e.* MSW management service) is the population inhabiting Grand Lyon. For **Step 3**, the $\boldsymbol{\gamma_i}$ ratios are those shown in Table 1. The main difficulty lies in the estimation of one or several appropriate factors $\boldsymbol{a_\Sigma}$, that are necessary to quantify the CC assigned to the studied system (**Step 4**). Sub-section 4.1 describes the used hypotheses in more detail. Sub-sections 4.2 and 4.3 present the results, merged by pairs of steps (Steps 1 and 2; Steps 3 and 4) to avoid redundancies.

4.1. Hypotheses used for Step 4

For the sake of simplicity, we consider that the studied system has 4 functions: MSW safe collection, transportation and treatment (the main function, $\boldsymbol{W}$), provision of heat ($\boldsymbol{H}$), provision of electricity ($\boldsymbol{E}$), provision of secondary raw materials (all types included, $\boldsymbol{RM}$). We also assume that the first 3 functions serve exclusively the final users – no electricity or heat produced by the system is used by industries, but only by households. In contrast, provision of raw materials is supposed to serve only industries, that transform them into products for final users (indirect function $\boldsymbol{All}$). Then, the average factor $\boldsymbol{a_\Sigma}$ can be calculated using Equation 12. This equation is a direct application of Equation 11, where each sum has been developed and each function has been highlighted.

Equation 12

$$\boldsymbol{a_\Sigma = \frac{1}{b_\Sigma} * \left[a_W * b_W + a_H * \frac{q_{\Sigma,H}}{d_H} + a_E * \frac{q_{\Sigma,E}}{d_E} + c_{RM\to All} * b_{RM\to All}\right]}$$

All the terms used in this equation are presented in Table 5. Yearly energy demand is estimated using data from the French Ministry of Environment (Ministère de la Transition Ecologique, 2020). Note that using these hypotheses, the energy production from MSW incineration covers the needs of $\boldsymbol{\frac{q_{\Sigma,H}}{d_H}}$ = 39,900 capita for heat and $\boldsymbol{\frac{q_{\Sigma,E}}{d_E}}$ = 45,475 capita for electricity, *i.e.* about 30% of the population of Grand Lyon. In the absence of more detailed data, the value of $\boldsymbol{b_{RM\to All}}$ is set to a value as low as 5% of the population of Grand Lyon to account for the fact that:

- the raw materials produced by recycling are only a small fraction of the total MSW produced by households;
- a significant part of these MSW comes from packaging, which weighs only a small fraction of the total mass of the goods purchased by households;
- with increasing stocks of materials within the economy, we can assume that household consumption for goods (and therefore raw materials) outweighs MSW production;
- a significant part of the produced raw materials is used by industries along their value chain.

The contribution of raw materials to human overall needs $\boldsymbol{c_{RM\to All}}$ is assessed with the Supply and Use Tables of the whole economy (INSEE, 2018). It is measured as the ratio between (a) the monetary value of the intermediate consumption, by the French economy, of products from industries producing these raw materials, and (b) the total output for final use. Detailed figures are presented in the Supplementary Material. Note that the resulting value of $\boldsymbol{c_{RM\to All}}$ is overestimated, since the industries considered for this estimation produce a wide variety of products besides those studied in this paper.

Table 5: Summary of the terms used in Equation 12

Term	Meaning	Value
$\boldsymbol{a_W, a_H, a_E}$	Factors $\boldsymbol{a_f}$ quantifying the contribution of functions $\boldsymbol{W}$, $\boldsymbol{H}$, and $\boldsymbol{E}$ to human needs	Depends on the sharing principle chosen to quantify the satisfaction of human needs (see Table 6)
$\boldsymbol{b_\Sigma = b_W}$	Number of people benefiting from the main function delivered by $\boldsymbol{\Sigma}$, *i.e.* MSW management ($\boldsymbol{W}$), see section 3.1	1,385,927 capita
$\boldsymbol{q_{\Sigma,H}}$	Quantity of heat delivered by $\boldsymbol{\Sigma}$, see section 3.1	182,343 MWh/year
$\boldsymbol{d_H}$	Yearly heating demand per capita, see Supplementary Material	4.57 MWh/capita/year
$\boldsymbol{q_{\Sigma,E}}$	Quantity of electricity delivered by $\boldsymbol{\Sigma}$, see section 3.1	95,953 MWh
$\boldsymbol{d_E}$	Yearly electricity demand per capita, see Supplementary Material	2.11 MWh/capita/year
$\boldsymbol{c_{RM\to All}}$	Contribution of the studied raw materials to the satisfaction of overall needs, see Supplementary Material	8.7%
$\boldsymbol{b_{RM\to All}}$	Average number of people whose overall needs are satisfied by the raw materials produced by $\boldsymbol{\Sigma}$, set to 5% of the population of Grand Lyon.	69,296 capita

The last step, for the estimation of $\boldsymbol{a_\Sigma}$, is to propose candidate values for the different contributions $\boldsymbol{a_f}$. We propose several sharing principles:

HE_AV: Contribution of f to the annual expenditures of average households, to measure how much of their resources these households dedicate to the satisfaction of this function;

HE_D1: Contribution of f to the annual expenditures of the 10% of the households with the lowest income (decile 1). The underlying hypothesis is that with constrained revenue, households would tend to give priority to the most "essential" goods and services (Colombi, 2020);

GVA_D: Direct contribution of f to total national Gross Value Added (GVA), *i.e.* the main component of Gross Domestic Product (GDP) before adding taxes and subtracting subsidies on products (like VAT, etc.). This is a reference indicator that is commonly used to measure wealth, and as a sharing principle (Ryberg et al. 2020);

GVA_T: Cumulative direct and indirect contribution of f to total national GVA, measured by dividing the total production of f (and not just its value added) by national Gross Value Added, assuming that each euro spent on f will eventually contribute to that indicator. This line of reasoning cannot, however, be applied to other macroeconomic indicators without proper Input Output modelling;

COE: Direct contribution of f to total Compensation of employees, *i.e.* the sum of all wages and salaries. We propose to use this indicator, assuming that one utility of an industry is to provide income to its employees, thereby giving them the financial resources to satisfy their needs;

FCE: Contribution of f to Final Consumption Expenditures (FCE), to measure how much of their resources households and administrations spend to satisfy their needs. We choose to use it as an estimate of the importance of that function to their overall needs.

The data used for these indicators, shown in Table 6, are based on the Family Budget Survey (INSEE, 2020b) and on the Supply and Use Tables (INSEE, 2018), expressed with different levels of detail and nomenclatures – respectively the COICOP (European Commission, 2002) and the NACE (EUROSTAT, 2008). In the Family Budget Survey, expenses related to waste treatment management and energy supply (electricity, gas and other fuels) are respectively categorised in COICOP subclass 444 and class 45. In the Supply and Use Tables, they are categorised in NACE sections E and D.

However, section E aggregates data on MSW management – what is wanted for this study – with other activities such as: water supply, sewerage, remediation and management of wastes that are not MSW (*e.g.* industrial waste). Therefore, using these data just for MSW management would overestimate the value of its assigned carrying capacity. We therefore chose to assign MSW management with a fraction of section E's economic flows, based on the ratio between average household expenses for MSW management (COICOP subclass 444) and their expenses for water supply and miscellaneous services relating to the dwelling (COICOP class 44).

Table 6: Contribution of waste treatment and energy supply activities to several macroeconomic indicators in France in 2017 (INSEE, 2018; INSEE, 020b)

	COICOP code		**TOTAL**	**44**	***444***	***45***
Family Budget Survey	Average	€	27,408	834	*578*	*1,353*
(€ per household per year)	**(HE_AV)**	%	100%	3.04%	*2.11%*	*4.94%*
(INSEE,2020b)	Decile 1	€	16,123	728	*537*	*967*
	(HE_D1)	%	100%	4.52%	*3.33%*	*6.00%*
	NACE Code		**TOTAL**	**E**	***E*****	***D***
	Gross value added	B€	2,043,997	15,053	***10,433***	*33,756*
	(GVA_D)	%	100%	0.74%	***0.51%***	*1.65%*
Supply and Use Table	Output by industry*	B€	4,039,953	37,370	***25,899***	*107,147*
(billion € per year)	**(GVA_T)**	%	197.65%	1.83%	***1.27%***	*5.24%*
(INSEE 2018)	Compensation of employees	B€	1,198,018	7,592	***5,600***	*12,414*
	(COE)	%	100%	0.63%	***0.47%***	*1.04%*
	Final consumption expenditure	B€	1,783,093	14,130	***9,793***	*40,772*
	(FCE)	%	100%	0.79%	***0.55%***	*2.29%*

* The ratio between "total output by industry" and "total gross value added" exceeds 100% because a significant part of the output is consumed by industries (intermediate consumption).

** Bold figures are based on estimates for the Supply and Use Table, since no data were available at that level of detail.

Column 444 is used to estimate $\boldsymbol{a_W}$ and column 45 is used to estimate $(\boldsymbol{a_H} + \boldsymbol{a_E})$
COICOP codes: 44 - Water supply and miscellaneous services relating to the dwelling; 444 - Miscellaneous services relating to the dwelling; 45 - Electricity, gas and other fuels
NACE codes: E – Water supply, sewerage, waste management and remediation activities; D – Electricity, gas, steam and air conditioning supply

Based on these data, it is possible to calculate the different values of $\boldsymbol{a_\Sigma}$, depending on the scenario. Table 7 shows the resulting values, as well as the respective contributions of each function. It follows that accounting for the provision of energy and materials increases the ACC of the studied MSW management chain by between 67 and 204%, as opposed to if only the MSW management function had been accounted for. That increase is mainly due to the provision of energy, since we assumed that it benefited directly to 30% of Grand Lyon's inhabitants, and that the provision of materials only benefited indirectly to 5% of Grand Lyon's inhabitants. Furthermore, choosing to account for the direct contribution to household expenditures (**HE_AV**), and especially of those with the lowest income (**HE_D1**) grants a much more significant ACC to MSW management than other macroeconomic indicators. That may be because COICOP subclass 444 includes other services except for just MSW management, but the data from INSEE (2020b) could not tell them apart.

Finally, the ACC of function $\boldsymbol{RM}$ is not affected by the change of sharing principle. That is normal, since it is based on the average contribution of intermediate consumption of the studied products by the French economy to overall French final demand. That raises two issues. Firstly, this overall final demand is the sum of the final expenditures of households, governments, non-profit institutions, gross fixed capital formation and exports. The structure of this overall demand may differ from that of just household expenditure – which would be more relevant to our case study, but it is not possible to estimate the intermediate consumption assigned specifically to household expenditure without proper Input Output modelling. Secondly, changing the sharing principle used for $\boldsymbol{RM}$ would require disaggregating household final demand expenditure into its different components $\boldsymbol{f}$, estimating the contribution $\boldsymbol{c_{RM \to f}}$ of function $\boldsymbol{RM}$ to each of these components, and to assign alternative factors $\boldsymbol{a_f}$ to each function $\boldsymbol{f}$. Then again, proper Input Output modelling seems necessary.

Table 7: Average contribution of the studied MSW management chain to the overall needs of Grand Lyon's inhabitants, depending on sharing principle

	MSW management* W	Heat and electricity supply** H, E	Raw material supply*** RM	TOTAL (a_{Σ})	Increase of contribution by multifunctionality ****
HE_AV	**2.11%**	1.48%	0.44%	**4.03%**	91%
HE_D1	**3.33%**	1.80%	0.44%	**5.57%**	67%
GVA_D	**0.51%**	0.50%	0.44%	**1.44%**	182%
GVA_T	**1.27%**	1.57%	0.44%	**3.28%**	158%
COE	**0.47%**	0.31%	0.44%	**1.22%**	159%
FCE	**0.55%**	0.69%	0.44%	**1.67%**	204%

* Calculated using $\boldsymbol{a_W} * \frac{\boldsymbol{b_W}}{\boldsymbol{b_\Sigma}}$

** Calculated using $(\boldsymbol{a_H} + \boldsymbol{a_E}) * \frac{\boldsymbol{b_H}}{\boldsymbol{b_\Sigma}}$**.** With $\boldsymbol{b_H}$ the estimated number of people benefiting from heat and electricity (30% of Grand Lyon's inhabitants). Electricity and heat supply were aggregated because some of the Family Budget Survey data related to these expenses were not detailed enough.

*** Calculated using $\boldsymbol{c_{RM \to All}} * \frac{\boldsymbol{b_{RM \to All}}}{\boldsymbol{b_\Sigma}}$

**** Difference between $\boldsymbol{a_\Sigma}$ and contribution of MSW management divided by the latter.

4.2. Steps 1 and 2: normalised impacts

This sub-section presents the LCIA results within the AESA framework. First, the impacts and benefits of MSW management are normalised and expressed as a fraction of the ACC of the inhabitants of Grand Lyon - see steps 1 and 2 of the procedure presented in sub-section 2.2. These steps are presented together, since estimating the number of inhabitants benefiting from the main function of MSW management did not raise any issue and only adds information to the most important results provided by step 1. The normalised impacts shown in Figure 2 are calculated using Equation 3 (detailed figures and contribution analysis in Supplementary Material). The bars above 0 represent detrimental environmental impacts, while the bars below 0 represent the environmental benefits associated with resource savings from waste valorisation. Several observations can be drawn from this Figure 2:

Step 1: Judging only the hight of the bars, regardless of the scale, it appears that the most significant impacts of MSW management, vis-à-vis AESA, are particulate matter emissions (PM), global warming (GW), fossil resource depletion (FRD), emissions of substances that are toxic for human health (HTOX_c, HTOX_nc) and freshwater eutrophication (FEU). For all these impact categories (except those regarding human toxicity), the resource savings ("Substitution" bar) allowed by valorisation outweigh the environmental impacts of transportation and treatment processes, which means that MSW valorisation is a cleaner way to produce these resources than primary production. For the other impact categories (POF: photochemical ozone formation; ECOTOX: ecotoxicity; MEU & TEU: marine and terrestrial eutrophication; MRD: mineral resource depletion; IR: ionizing radiation; AC: acidification; WU: water use), the data and LCIA methods used seem to highlight that MSW management is not a major issue with regards to planetary boundaries – or at least much less than the former impact categories.

Step 2: The scale of the environmental impacts of MSW management of Grand Lyon is well below that of the total CC of its inhabitants (all bars are below 10% of that CC). This is consistent with a former study carried on industrial waste landfill (Wolff, 2017).

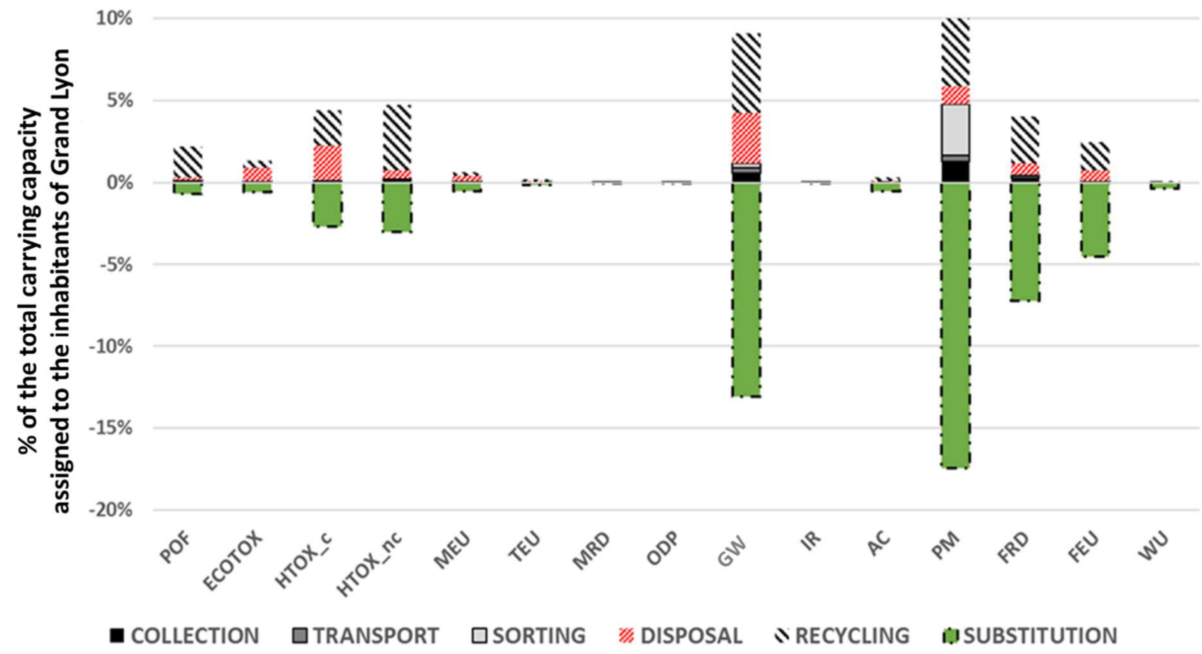

Figure 2: Contribution of life cycle stages to the overall environmental issues of MSW management – normalisation by carrying capacities assigned to beneficiaries ($p_{\Sigma,i}$ factors, see Equation 3)

4.3. Steps 3 and 4: Global scale and assignment of carrying capacity

This sub-section presents the results of the steps 3 and 4 of the procedure detailed in sub-section 2.2, focusing on collection, transportation, and treatment impacts. Resource savings are not considered since they are based on another scenario. However, for step 4 (comparison between the impacts of MSW management and its ACC), these resources significantly increased the studied ACC, as demonstrated in sub-section 4.1. Two layers of information can be found in Figure 3, corresponding to steps 3 and 4.

Step 3: Firstly, the LCIA results are put in perspective considering which impacts are the most significant at a global scale. Each point on Figure 3 describes an impact category, and their coordinates represent respectively the fraction $\boldsymbol{p_{\Sigma,i}}$ of the ACC of Grand Lyon's inhabitants used by MSW management (on the abscissa axis) and the $\boldsymbol{\gamma_i}$ factor by which planetary boundaries are exceeded at global scale (on the ordinate axis, see values in Table 1). The green-shaded area corresponds to impact categories for which carrying capacities are not (yet) exceeded globally: their corresponding $\boldsymbol{\gamma_i}$ values (ordinate axis) are below 100%.

Step 4: Secondly, for the purpose of assigning a CC to MSW management, the vertical straight lines represent the $\boldsymbol{a_\Sigma}$ factors that were calculated in Table 7, depending on different sharing principles. These factors all range between 1.2% and 5.6%. In other words, if we consider the used macroeconomic indicators as suitable interim estimates of utility to humans – and then again, this is a fragile assumption (Ryberg et al., 2020) – any impact category for which this activity uses more than 1.2% to 5.6% of the ACC of its beneficiaries – *i.e.* for which the coordinate on the abscissa axis in Figure 3 is more than 1.2% to 5.6% – would be considered as unsustainable.

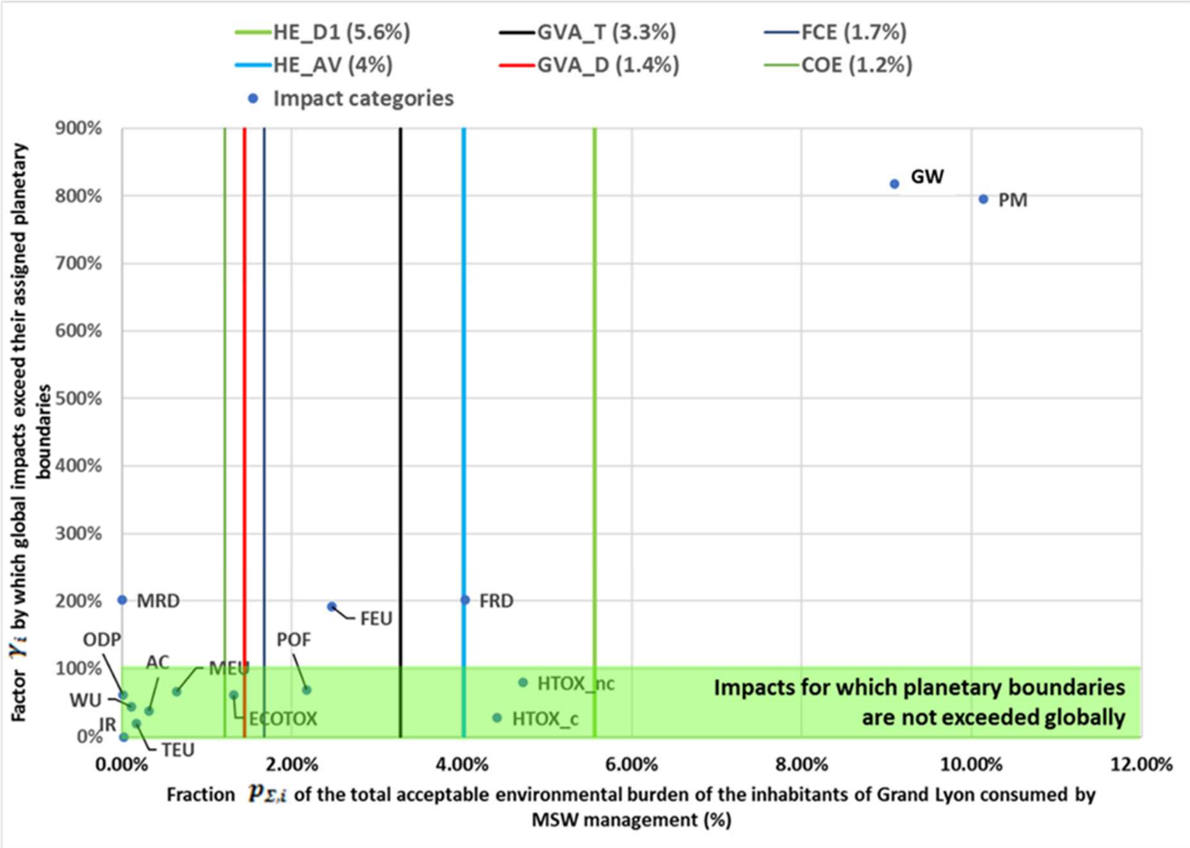

Figure 3: Comparison between fraction of per capita ACC used by MSW management in Grand Lyon ($p_{\Sigma,i}$ factors, see Equation 3) and exceedance factors of planetary boundaries at global scale (γ_i factors, see Table 1)

Different configurations can be observed and summarized by Table 8. These configurations are ranked from highest to lowest priority, in terms of needs for impact reduction by the studied system. This ranking first considers whether global carrying capacities are currently exceeded ($\gamma_i > 1$), in which case all human activities, including the one under study, should engage in significant impact reduction efforts, or at least avoid increasing their impacts. If $\gamma_i < 1$, the pressure for impact reduction may be lower. Such impacts may even be allowed to increase to some extent, especially if it is due to measures implemented to reduce higher priority impacts, as long as it does not result in crossing global carrying capacities.

Within these two cases, impact categories are then ranked from highest to lowest values of $p_{\Sigma,i}$. Note that this ranking is not influenced by the choice of sharing principle. Therefore, this ranking can be done during **Step 3** of the procedure. The main use of **Step 4** is to refine that ranking by adding different priority sub-levels. It does so by estimating if, for which impact categories, and by how much the studied system exceeds its ACC. In this case study, the ACC granted to MSW management can vary by a factor close to 5 between the most restrictive (COE) and the most "generous" sharing principle (HE_D1).

In this context, impact categories related to particulate matter, climate change, fossil resource depletion and freshwater eutrophication (PM, GW, FRD and FEU) are significant issues at both MSW management and global scales (high values of both $p_{\Sigma,i}$ and γ_i). They should be the ones to consider reducing in priority: depending on the chosen sharing principle, they exceed their ACC by a factor $\beta_{\Sigma,i}$ between 2 and 8.3 (for the direct contribution to wages and salaries, COE), or between 1.6 and 1.82

(for PM and GW, for the total contribution to the expenses of the 10% of households with the lowest income, HE_D1).

On the other hand, even though impact categories related to human and environmental toxicities (HTOX_nc, HTOX_c, ECOTOX) and photochemical ozone formation (POF) are significant for MSW management (high values of $\boldsymbol{p_{\Sigma,i}}$), and may exceed their ACC ($\boldsymbol{\beta_{\Sigma,i}}$ values above 1) for some or all sharing principles, at a global scale, they do not seem to be the most pressing issues yet ($\boldsymbol{\gamma_i}$ values below 100%). They should therefore be monitored, and possibly reduced, but not at the expense of higher priority impact categories. However, if their global carrying capacities were to be exceeded in the future, these impacts should also be reduced in priority.

It can also be observed that the only impact categories that would be considered as sustainable at both system and global scales would be those related to marine eutrophication, acidification, terrestrial eutrophication, water use, ionising radiation, and ozone depletion (MEU, AC, TEU, WU, IR, and ODP), whatever the macroeconomic indicator retained as sharing principle. These impacts may be allowed to increase due to changes of MSW management, especially if these changes allow significant reductions of higher priority impacts.

Finally, mineral resource depletion (MRD) cannot be really considered as sustainable, since its assigned global carrying capacity is exceeded by a factor 2. However, MSW management seems to be an example to follow by other industries, regarding this impact category, considering that it consumes an insignificant part of its ACC ($\boldsymbol{\beta_{\Sigma,i}}$ close to 0). Thus, MSW should avoid increasing its MRD impacts to avoid worsening the global situation, but most of impact reduction efforts should be done by other industries that must obviously exceed by far their own ACC.

Table 8: Classification of impact categories according to decreasing levels of priority

Impact category	γ_i * (%)	$p_{\Sigma,i}$ ** (%)	$\beta_{\Sigma,i}$ factors depending on sharing principle *** COE 1.22%	GVA_D 1.44%	FCE 1.67%	GVA_T 3.28%	HE_AV 4.03%	HE_D1 5.57%	Comments on impact categories depending on the values of $\beta_{\Sigma,i}$ and γ_i Sharing principle and associated value of a_{Σ} (see Table 7)
PM	795	10.14	**8.31**	**7.04**	**6.07**	**3.09**	**2.52**	**1.82**	Exceed both their carrying capacity at global scale and ACC at system scale, no matter what the chosen sharing principle. **Should be reduced with maximum priority**
GW	817	9.09	**7.45**	**6.31**	**5.44**	**2.77**	**2.26**	**1.63**	
FRD	201	4.03	**3.30**	**2.80**	**2.41**	**1.23**	**1.00**	0.72	Exceed both their carrying capacity at global scale and ACC at system scale, for almost all sharing principles. **Should be reduced in priority**
FEU	192	2.47	**2.02**	**1.72**	**1.48**	0.75	0.61	0.44	
MRD	201	0.0004	0.0003	0.0003	0.0002	0.0001	0.0001	0.0001	Exceed their carrying capacity at global scale but not their ACC at system scale.
HTOX_nc	80	4.72	**3.87**	**3.28**	**2.83**	**1.44**	**1.17**	0.85	Exceed their ACC at system scale, for almost all sharing principles. Carrying capacity at global scale not (yet) exceeded. **Should be monitored and reduced, but not at the expense of higher priority indicators.**
HTOX_c	28	4.41	**3.61**	**3.06**	**2.64**	**1.34**	**1.09**	0.79	
POF	69	2.18	**1.79**	**1.51**	**1.31**	0.66	0.54	0.39	May exceed their ACC at system scale, depending on chosen sharing principle. Carrying capacity at global scale not (yet) exceeded. **Should be monitored and reduced (low constraint)**
ECOTOX	62	1.32	**1.08**	0.92	0.79	0.40	0.33	0.24	
MEU	67	0.65	0.53	0.45	0.39	0.20	0.16	0.12	Are within both their carrying capacity at global scale and ACC at system scale, no matter what the chosen sharing principle (ideal case). **Do not seem to be the most pressing issue.**
AC	38	0.32	0.26	0.22	0.19	0.10	0.08	0.06	
TEU	20	0.17	0.14	0.12	0.10	0.05	0.04	0.03	
WU	44	0.12	0.10	0.08	0.07	0.04	0.03	0.02	
IR	0	0.03	0.02	0.02	0.02	0.01	0.01	0.01	
ODP	62	0.01	0.01	0.01	0.01	0.003	0.002	0.002	

* Exceedance factors γ_i of planetary boundaries at global scale (Table 1)
** Fraction $p_{\Sigma,i}$ of per capita ACC used by MSW management in Grand Lyon (Equation 3)
*** Factors $\beta_{\Sigma,i}$ by which MSW exceeds its ACC, depending on chosen sharing principle $a_{\Sigma,i}$, calculated using Equation 4

Bold figures indicate when the studied system exceeds its assigned carrying capacity.
Shaded rows indicate different levels of priority for impact reduction, from high (dark red) to low (dark green). Grey rows indicate impacts for which no impact increase is desirable, but impact reduction efforts should be done by other actors than studied system, which can individually be considered as sustainable.

5. Discussion

This section puts the results in perspective from the case study point of view and from a methodological point of view.

5.1. Case study results

From the case study point of view, several conclusions can be drawn from the application of the AESA framework to the MSW management chain of Grand Lyon in 2017

For many impact categories, the resource savings allowed by the secondary products resulting from MSW valorisation (recycled materials, recovered heat and electricity) outweigh the negative impacts generated by MSW transportation and treatment. This means that waste valorisation is an improvement on waste disposal and primary resource production. That is coherent with previous works (Merrild et al., 2012; Andreasi Bassi et al., 2017; Beylot et al., 2018), and especially true for the most significant impact categories from an AESA perspective at both system and global scales, which should be reduced in priority (particulate matter emissions, climate change, fossil resource depletion, and freshwater eutrophication).

However, that improvement remains insufficient, since the MSW management chain of Grand Lyon, for the year 2017, exceeds its ACC by a factor of up to 8 for climate change and particulate matter – if the ACC is calculated with macroeconomic indicators. After all, the impacts associated with recycling processes remain significant and of comparable magnitude with those of primary production.

These conclusions need to be nuanced for at least two reasons.

Firstly, the primary function of waste management is to deal with materials that can raise health or environmental issues. Waste recycling (or valorisation) only makes sense if there is actually a market for the associated products. If the market for these products is constrained or saturated, several studies on MSW, industrial or construction and demolition waste recycling have highlighted that these products would probably (1) require higher transportation distances to be sellable that may reduce their environmental or economic relevance (Merrild et al., 2012; Rodrigues et al., 2019), or (2) not allow any savings at all, the other products being produced anyway (Mousavi, 2018). In other words, the so-called benefits associated with secondary products only exist under the condition that there *is* a market for such secondary products, and that they do allow savings of primary products that are polluting to produce. Without such a market, not only would these benefits be compromised, but also the ACC of MSW management would be significantly lower (see section 4.1). Waste source prevention may yield the same environmental benefits without causing nearly as much impacts as waste transportation and treatment, and therefore may be more compatible with planetary boundaries.

Secondly, it is important to remember the simplifications made for this case study (see Supplementary Material), which serves more as a proof of concept of the application of the AESA framework to complex systems such as MSW management, in order to highlight its challenges and opportunities.

5.2. Methodological perspectives

From the methodological point of view, now, we have highlighted some challenges associated with the AESA of multifunctional systems that contribute, at least partially, indirectly to human needs.

First and foremost, such systems require extensive knowledge about how their functions are used: what products or services destined to final users will they contribute making, and how much is required for each of these products and services? We lacked physical or even economic data with sufficient detail, for each recycled material produced by MSW management, to address this issue in our case study, so we chose several simplifying hypotheses. Thus, we assumed that the energy produced by MSW management was only destined to final users. That may be a realistic assumption for heat, which is essentially valorised through district heating, but it is more debatable for electricity, which is sold on the French or European market. We also assumed that the secondary materials produced were sold on an average national market that produced all the goods and services required by final users. In practice, that may not be the case, and this simplification makes it difficult to change the sharing principle for the provision of secondary materials, as pointed out in sub-section 4.1. Proper Input Output modelling seems an appropriate way to refine the used data and change the sharing principles applied to intermediate functions.

Another challenge is the choice of an appropriate sharing principle to assess a function's entitlement to carrying capacity, and even the relative importance of a given intermediate product (*e.g.* steel) to the manufacture of a good purchased by final users (*e.g.* a car). In both cases, the use of Supply and Use Tables and similar economic data proved useful, but such economic indicators based on monetary valuation are hardly relevant to capture the vital nature of certain needs (*e.g.* access to drinkable water, health services, food) or to deal with trade-offs (*e.g.* vital needs vs. psychological needs, culture, luxury products, etc.). This is why we proposed to focus on the final demand expenditures of the households with the lowest income (Colombi, 2020), in an attempt to better reflect what is most important under constrained resources. That approach would probably be more relevant at a global scale, considering that developed countries have sufficient infrastructure to ensure abundant access to (and low prices for) vital goods such as drinkable water, energy, etc. In any case, monetary valuation, however practical it may be, keeps raising ethical issues (Milanesi, 2010). Alternative indicators, based on the satisfaction of physiological and psychological needs and the preferences of individuals between them, would therefore offer promising sharing principles.

6. Conclusion

This paper has introduced several methodological developments for the PB-LCA of complex and multi-functionnal systems, and applied them to waste management, using Grand Lyon as a simplified case study. It has demonstrated that even without needing to define *one* single relevant sharing principle, we could highlight which were the most significant impact categories at both global and system scales, and define different levels of priority for impact reduction. These levels may be used to support decision making. Defining a proper sharing principle is a prerequisite for the next steps, *i.e.* assessing the "sustainable" or "unsustainable" nature of a studied system, and setting exact reduction targets. Yet, considering the challenges ahead and the need for quickly operational decision tools, it is a good start to keep CC allocation at a territorial level and to consider various simplified sharing principles, as suggested by previous works (Ryberg et al., 2020, 2018).

Improvements of the AESA framework, for its application at the product level with a utilitarian perspective, inevitably raise the issue of the purpose of the functions provided by a system: a product only has value because it contributes to satisfying human needs, according to a metric that needs to be developed (Ryberg et al., 2020). While conventional LCA usually considers *how* a product is *produced* to quantify its impacts, PB-LCA also requires insights on how it is *used* (or, alternatively, *why* it is produced) to quantify its benefits and its entitlement to CC. This raises several technical and political/ethical issues.

Firstly, as mentioned before, a relevant sharing principle of the total CC is required between the various needs of human beings. Macroeconomic indicators are relatively easily measurable, even though they are seldom available with an appropriate level of detail. They present the crucial advantages of being fact-based, and of giving an order of magnitude to the ACC. However, they are hardly relevant to capture the vital nature of certain needs (*e.g.* access to drinkable water, health services, food) or to deal with trade-offs (*e.g.* vital needs vs. psychological needs, culture, luxury products, etc.). Insights from social science to better quantify these preferences and priorities could help develop more relevant sharing principles.

Secondly, as illustrated in this paper, the case of multifunctional systems highlights that for each studied product, its contribution – *both* direct *and indirect* – to human final demand needs to be assessed. This is a non-trivial issue that may require methods such as Economic or Physical Input Output modelling with a sufficient level of detail. In this paper, many simplifications and hypotheses were required to overcome the lack of data. Furthermore, here again, the use of monetary value of goods and services, as a sharing principle between them, is highly debatable, since strategic goods and services may have low prices and/or be required in small quantities.

Therefore, using economic data as CC sharing principle should be done with extreme care. These data should be associated with transparent sensitivity analyses to better reflect the utility of the considered systems. Using a variety of sharing principles may also help provide fact-based levels of confidence in the sustainable or unsustainable nature of the studied system.

At last, it should be kept in mind that AESA methods assess *unsustainability* rather than environmental sustainability (Wolff, 2017). Indeed, it allows identifying unsustainable impacts for the impact categories for which the estimated environmental pressures are higher than the assigned carrying capacity. However, even if all the measured environmental pressures appear to be below the assigned carrying capacity, it cannot guarantee that there is no other ecological issue that is currently not measured but a potential source of unsustainability (biodiversity of pressure on biological or mineral resources, for example).

Acknowledgements

This research was conducted within the SIMODEM project (SImuler la MObilité des DÉchets Ménagers, or Simulating the mobility of household waste), which was funded by the LABEX "Intelligences des Mondes Urbains" (IMU, ANR-10-LABX-0088) of Université de Lyon, within the program « Investissements d'Avenir » (ANR-11-IDEX-0007) operated by the French National Research Agency (ANR). The authors wish to thank Florence Toilier who was responsible of the SIMODEM project as well as Elen Devauchelle, from Grand Lyon, for helping them accessing the necessary data. The authors also thank the reviewers for their insightful comments on our manuscript.

References

ADEME, 2014. Transport et logistique des déchets - Enjeux et évolutions du transport et de la logistique des déchets, Clés pour agir, https://librairie.ademe.fr/dechets-economie-circulaire/3030-transport-logistique-des-dechets-enjeux-et-evolutions-du-transport-et-de-la-logistique-des-dechets-9782358383553.html .

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

Beylot, A., Muller, S., Descat, M., Ménard, Y., Michel, P., Villeneuve, J., 2017. WILCI: a LCA tool dedicated to MSW incineration in France. Presented at the 16th Waste Management and Landfill Symposium, p. 12.

Beylot, A., Muller, S., Descat, M., Ménard, Y., Villeneuve, J., 2018. Life cycle assessment of the French municipal solid waste incineration sector. Waste Manag. 80, 144–153. https://doi.org/10.1016/j.wasman.2018.08.037

Bjørn, A., Chandrakumar, C., Boulay, A.-M., Doka, G., Fang, K., Gondran, N., Hauschild, M.Z., Kerkhof, A., King, H., Margni, M., McLaren, S., Mueller, C., Owsianiak, M., Peters, G., Roos, S., Sala, S., Sandin, G., Sim, S., Vargas-Gonzalez, M., Ryberg, M., 2020. Review of life-cycle based methods for absolute environmental sustainability assessment and their applications. Environ. Res. Lett. https://doi.org/10.1088/1748-9326/ab89d7

Bjørn, A., Sim, S., King, H., Keys, P., Wang-Erlandsson, L., Cornell, S. E., Margni, M., & Bulle, C., 2019a. Challenges and opportunities towards improved application of the planetary boundary for land-system change in life cycle assessment of products. Science of The Total Environment, 696, 133964. https://doi.org/10.1016/J.SCITOTENV.2019.133964

Bjørn A, Richardson K and Hauschild M Z, 2019b. A framework for development and communication of absolute environmental sustainability assessment methods *J. Ind. Ecol.* **23** 838–54, https://doi.org/10.1111/jiec.12820

Bjørn, A., Hauschild, M.Z., 2015. Introducing carrying capacity-based normalisation in LCA: framework and development of references at midpoint level. Int. J. Life Cycle Assess. 20, 1005–1018. https://doi.org/10.1007/s11367-015-0899-2

Bjørn, A. (2015). Better, but good enough? Indicators for absolute environmental sustainability in a life cycle perspective. DTU Management Engineering. DTU Management Engineering. PhD thesis No. 8.2015

Bos, U., Horn, R., Beck, T., Lindner, J.P., Fischer, M., 2016. LANCA® Characterization Factors for Life Cycle Impact Assessment - Version 2.0, FRAUNHOFER VERLAG. ed.

Clift, R.; Sim, S.; King, H.; Chenoweth, J.L.; Christie, I.; Clavreul, J.; Mueller, C.; Posthuma, L.; Boulay, A.-M.; Chaplin-Kramer, R.; Chatterton, J.; DeClerck, F.; Druckman, A.; France, C.; Franco, A.; Gerten, D.; Goedkoop, M.; Hauschild, M.Z.; Huijbregts, M.A.J.; Koellner, T.; Lambin, E.F.; Lee, J.; Mair, S.; Marshall, S.; McLachlan, M.S.; Milà i Canals, L.; Mitchell, C.; Price, E.; Rockström, J.; Suckling, J.; Murphy, R. The Challenges of Applying Planetary Boundaries as a Basis for Strategic Decision-Making in Companies with Global Supply Chains. Sustainability 2017, 9, 279. https://doi.org/10.3390/su9020279Colombi, D., 2020. Où va l'argent des pauvres. Fantasmes politiques, réalités sociologiques. Payot.

De Laurentiis, V., Secchi, M., Bos, U., Horn, R., Laurent, A., Sala, S., 2019. Soil quality index: Exploring options for a comprehensive assessment of land use impacts in LCA. J. Clean. Prod. 215, 63–74. https://doi.org/10.1016/j.jclepro.2018.12.238

Diener, E., 2000. Subjective well-being: The science of happiness and a proposal for a national index. - PsycNET. Am. Psychol. 55, 34–43.

EC- JRC, 2011. ILCD Handbook. Recommendations for Life Cycle Impact assessment in the European context – based on existing environmental impact assessment models and factors. Publications Office, Luxembourg.

European Commission, 2002. COMMISSION REGULATION (EC) No 113/2002 of 23 January 2002 amending Council Regulation (EC) No 2223/96 with regard to revised classifications of expenditure according to purpose [WWW Document]. URL https://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=OJ:L:2002:021:0003:0009:EN:PDF (accessed 3.24.21).

EUROSTAT, 2008. NACE rev. 2 - Statistical classification of economic activities in the European Community. Office for Official Publications of the European Communities, Luxembourg.

Fazio, S., Biganzioli, F., Laurentiis, V.D., Zampori, L., Sala, S., Diaconu, E., 2018. Supporting information to the characterisation factors of recommended EF Life Cycle Impact Assessment methods - Version 2 from ILCD to EF 3.0 (No. JRC114822). Joint Research Center.

François, C., Gondran, N., Nicolas, J.-P., 2021. Spatial and territorial developments for life cycle assessment applied to urban mobility—case study on Lyon area in France. Int. J. Life Cycle Assess. 26, 543–560. https://doi.org/10.1007/s11367-020-01861-2

Grand Lyon, 2017. Rapport annuel 2017 sur le prix et la qualité du service public de prévention et de gestion des déchets ménagers et assimilés. Métropole de Lyon, Direction Eau et Déchets.

Haupt, M., Kägi, T., Hellweg, S., 2018. Life cycle inventories of waste management processes. Data Brief 19, 1441–1457. https://doi.org/10.1016/j.dib.2018.05.067

Horn, R., Maier, S., 2018. LANCA® Characterization Factors for Life Cycle Impact Assessment - Version 2.0.

INSEE, 2020a. Comparateur de territoire − Intercommunalité-Métropole de Métropole de Lyon (200046977) | Insee [WWW Document]. URL https://www.insee.fr/fr/statistiques/1405599?geo=EPCI-200046977 (accessed 7.24.20).

INSEE, 2020b. Les dépenses des ménages en France en 2017 [WWW Document]. URL https://www.insee.fr/fr/statistiques/4648335?sommaire=4648339 (accessed 8.3.21).

INSEE, 2018. Supply and use table - Level A38 - Year 2017 [WWW Document]. URL https://www.insee.fr/en/statistiques/fichier/4132168/tes_38_2017.xls (accessed 8.11.20).

Kirchherr, J., Reike, D., Hekkert, M., 2017. Conceptualizing the circular economy: An analysis of 114 definitions. Resour. Conserv. Recycl. 127, 221–232. https://doi.org/10.1016/j.resconrec.2017.09.005

Larsen, A.W., Vrgoc, M., Christensen, T.H., Lieberknecht, P., 2009. Diesel consumption in waste collection and transport and its environmental significance. Waste Manag. Res. 27, 652–659. https://doi.org/10.1177/0734242X08097636

McElroy, M.W., Jorna, R.J., Engelen, J. van, 2008. Sustainability quotients and the social footprint. Corp. Soc. Responsib. Environ. Manag. 15, 223–234. https://doi.org/10.1002/csr.164

Merrild, H., Larsen, A.W., Christensen, T.H., 2012. Assessing recycling versus incineration of key materials in municipal waste: The importance of efficient energy recovery and transport distances. Waste Manag. 32, 1009–1018. https://doi.org/10.1016/j.wasman.2011.12.025

Milanesi, J., 2010. Éthique et évaluation monétaire de l’environnement : la nature est-elle soluble dans l’utilité ? VertigO - Rev. Électronique En Sci. Environ. https://doi.org/10.4000/vertigo.10050

Ministère de la Transition écologique, 2020. Consommation d’énergie par usage du résidentiel | Données et études statistiques [WWW Document]. URL https://www.statistiques.developpement-durable.gouv.fr/consommation-denergie-par-usage-du-residentiel (accessed 8.2.21).

Moreno Ruiz E., Valsasina L., FitzGerald D., Brunner F., Symeonidis A., Bourgault G., Wernet, G., 2019. Documentation of changes implemented in the ecoinvent database v3.6 [WWW Document]. URL https://www.ecoinvent.org/files/change_report_v3_6_20190912.pdf (accessed 4.8.20).

Mousavi, M., 2018. Territorial Environmental Modeling of Cement Concrete Demolition Waste (CCDW) Management with a Life Cycle Approach. PhD thesis. Université de Nantes. http://www.theses.fr/2018NANT4049

Rockström, J., Steffen, W., Noone, K., Persson, Å., Chapin, F.S.I., Lambin, E., Lenton, T., Scheffer, M., Folke, C., Schellnhuber, H.J., Nykvist, B., de Wit, C., Hughes, T., van der Leeuw, S., Rodhe, H., Sörlin, S., Snyder, P., Costanza, R., Svedin, U., Falkenmark, M., Karlberg, L., Corell, R., Fabry, V., Hansen, J., Walker, B., Liverman, D., Richardson, K., Crutzen, P., Foley, J., 2009. Planetary Boundaries: Exploring the Safe Operating Space for Humanity. Ecol. Soc. 14. https://doi.org/10.5751/ES-03180-140232

Rodrigues, J., 2016. Analyse de cycle de vie intégrative de filières de production de biomasse à usage industriel par la valorisation de délaissés. PhD thesis. Université de Lorraine. http://www.theses.fr/2016LORR0321

Rodrigues, J., Gérard, A., Séré, G., Morel, J.-L., Guimont, S., Simonnot, M.-O., Pons, M.-N., 2019. Life cycle impacts of soil construction, an innovative approach to reclaim brownfields and produce nonedible biomass. J. Clean. Prod. 211, 36–43. https://doi.org/10.1016/j.jclepro.2018.11.152

Ryberg, M.W., Andersen, M.M., Owsianiak, M., Hauschild, M.Z., 2020. Downscaling the planetary boundaries in absolute environmental sustainability assessments – A review. J. Clean. Prod. 276, 123287. https://doi.org/10.1016/j.jclepro.2020.123287

Ryberg, M.W., Owsianiak, M., Clavreul, J., Mueller, C., Sim, S., King, H., Hauschild, M.Z., 2018. How to bring absolute sustainability into decision-making: An industry case study using a Planetary Boundary-based methodology. Sci. Total Environ. 634, 1406–1416. https://doi.org/10.1016/j.scitotenv.2018.04.075

Sala, S., Crenna, E., Secchi, M., Sanyé-Mengual, E., 2020. Environmental sustainability of European production and consumption assessed against planetary boundaries. J. Environ. Manage. 269, 110686. https://doi.org/10.1016/j.jenvman.2020.110686

Steffen, W., Richardson, K., Rockstrom, J., Cornell, S.E., Fetzer, I., Bennett, E.M., Biggs, R., Carpenter, S.R., de Vries, W., de Wit, C.A., Folke, C., Gerten, D., Heinke, J., Mace, G.M., Persson, L.M., Ramanathan, V., Reyers, B., Sorlin, S., 2015. Planetary boundaries: Guiding human development on a changing planet. Science 347, 1259855–1259855. https://doi.org/10.1126/science.1259855

Steffen, W., Sanderson, R.A., Tyson, P.D., Jäger, J., Matson, P.A., III, B.M., Oldfield, F., Richardson, K., Schellnhuber, H.J., Turner, B.L., Wasson, R.J., 2005. Global Change and the Earth System: A Planet Under Pressure, Global Change - The IGBP Series. Springer-Verlag, Berlin Heidelberg.

Thoumazeau, A., Bustany, C., Rodrigues, J., Bessou, C., 2019. Using the LANCA® Model to Account for Soil Quality Within LCA: First Application and Approach Comparison in Two Contrasted Tropical Case Studies. Indones. J. Life Cycle Assess. Sustain. 3.

UNO, 2019. World Population 2019 - Population size and regional distribution [WWW Document]. URL https://population.un.org/wpp/Publications/Files/WPP2019-Wallchart.pdf (accessed 8.3.20).

VALORLY, 2016. Rapport annuel d'activité Traitement des Déchets Urbains.

Wolff, A., 2017. Responsabilité sociétale: quelles contributions des entreprises à la conservation de la biodiversité? PhD Thesis. Saint-Etienne. https://tel.archives-ouvertes.fr/tel-01695744/document

Wolff, A., Gondran, N., Brodhag, C., 2017. Detecting unsustainable pressures exerted on biodiversity by a company. Application to the food portfolio of a retailer. J. Clean. Prod. 166, 784–797. https://doi.org/10.1016/j.jclepro.2017.08.057