UNIVERSIDADE PAULISTA

PROGRAMA DE PÓS-GRADUAÇÃO STRICTO SENSU EM ENGENHARIA DE PRODUÇÃO

CIRCULAR ECONOMY AND FOOD RECOVERY HIERARCHY OPTIONS APPLIED TO ORGANIC BY- PRODUCTS MANAGEMENT IN FOOD DISTRIBUTION CENTERS

Tese apresentada ao Programa de Pós-Graduação em Engenharia de Produção da Universidade Paulista – UNIP, para obtenção do título de Doutor em Engenharia de Produção.

FEDERICO SULIS

SÃO PAULO

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Post-graduation program of Production Engineering of Paulista University (UNIP)

Scientific Advisor: Prof. Dr. Feni Agostinho

Concentration Area: Sustainability in Production Systems

Research Line: Advances in Cleaner production and Industrial Ecology

Research Project: Searching for more sustainable agro-industrial production chains

Post-graduation program of Production Engineering, Paulista University (UNIP)

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Data de aprovação: ______/_____/_______

BANCA EXAMINADORA

____/____/_____
Prof. Orientador Dr. Feni Dalano Roosevelt Agostinho
Universidade Paulista – UNIP

____/____/_____
Prof. Dr. Biagio Fernando Giannetti
Universidade Paulista – UNIP

____/____/_____
Prof.ª Dra. Cecília Maria Villas Boas de Almeida
Universidade Paulista – UNIP

____/____/_____
Prof. Dr. Alexandre Monteiro Souza
Swedish University of Agricultural Sciences – SLU

____/____/_______
Prof. Dr. Luca Coscieme
Hot or Cool Institute

DEDICATION

To my parents, Angelina (in memoriam) and Angelo (in memoriam).

A special dedication to my stepmother Paola, who gave me back my dreams and guided me for so many years until I found my way.

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RESUMO

Cerca de um terço da produção global anual de alimentos para consumo humano é desperdiçado, com implicações ambientais e sociais. A redução do desperdício alimentar e a sua valorização, considerando-se os alimentos rejeitados como um subproduto orgânico da cadeia de abastecimento alimentar, é de fundamental importância para um mundo mais sustentável, como reconhecido pelos princípios da economia circular (EC) e pela hierarquia de recuperação de alimentos (HRA). Seguindo as recomendações da EC e HRA, este estudo avalia as opções de hierarquia de recuperação de alimentos, buscando as melhores soluções para valorizar os subprodutos orgânicos (SPO) gerados pelo centro de distribuição de alimentos brasileiro denominado CEAGESP, localizado na cidade de São Paulo. Dentre as opções de HRA, são avaliados oito cenários para os SPO da CEAGESP (37.652 toneladas/ano), incluindo o aterro (cenário atual) e cenários de doação e biorrefinaria, como alternativas. A avaliação do ciclo de vida (ACV), por meio da contabilização de nove categorias de impacto, e a síntese de Emergia, por meio do cálculo de indicadores de sustentabilidade tradicionais e novos, são aplicadas para verificar o desempenho ambiental dos cenários atuais e propostos, segundo uma perspectiva complementar. Os resultados de ACV mostram que os cenários de doação têm os menores impactos ambientais para todas as nove categorias de impacto da ACV avaliadas, os cenários de biorrefinaria têm um desempenho intermediário e os piores cenários correspondem ao aterro do SPO, considerando-se ou não a geração de eletricidade. Diferenças superiores a dez vezes nas bases comparativas entre os melhores e piores cenários foram obtidas por uso dos combustiveis fósseis, aquecimento global, toxicidade humana, consumo de água e consumo de metais nas categorias de impacto da LCA. As características brasileiras de geração de eletricidade a partir de usinas hidrelétricas têm influência considerável em algumas categorias de impacto da ACV, como Aquecimento Global e Consumo da Água, principalmente quando as emissões evitadas são contabilizadas. Os resultados da síntese de Emergia mostram emergia líquida muito maior obtida pelo cenário de doação em comparação com biorrefinaria e aterro. A doação é capaz de salvar 29 vezes mais emergia se comparada com emergia investida, enquanto a biorrefinaria, 1,5 vezes e as opções localizadas na parte inferior do HRA apresentam menor economia de emergia aliada a um maior investimento em emergia. Os resultados destacam as vantagens ambientais em doar-se a fração comestível de SPO da CEAGESP, seguidos de um cenário de biorrefinaria capaz de recuperar energia e materiais, ambas opções alinhadas com os conceitos da EC para um desenvolvimento sustentável. As opções de doação e biorrefinaria são capazes de atingir simultaneamente um maior número de metas de desenvolvimento sustentável da Nações Unidas e um melhor desempenho para os indicadores ACV e Emergia, portanto, esses cenários devem ser promovidos pelas políticas públicas.

Palavras-Chave: Economia Circular; Hierarquia de Recuperação de Alimentos; Avaliação do ciclo de vida, Emergia, Doação de Alimentos, Bioreffinaria.

ABSTRACT

About one-third of annual global food production for human consumption is wasted, with environmental and social implications. Therefore, food waste reduction and its valorization by considering it an organic by-product of the food supply chain is of fundamental importance towards a more sustainable world, as recognized by both the circular economy (CE) principles and the food recovery hierarchy (FRH). By following CE and FRH recommendations, this study assesses the food recovery hierarchy options searching for the best solutions to valorize the organic by-products (OBP) generated by the Brazilian food distribution center called CEAGESP, located in São Paulo city. Among the FRH options, eight scenarios for the CEAGESP's OBP (37,652 tons/yr) are evaluated, from landfilling (current scenario) to food donation and biorefinery alternatives. Life cycle assessment (LCA), by accounting for nine impact categories, and Emergy synthesis, by calculating traditional and new sustainability indicators, are applied to verify the environmental performance of the current and modelled scenarios to achieve a complementary perspective. LCA results show that donation scenarios have the least environmental burdens for all the nine LCA impact categories assessed, biorefinery scenarios have an intermediate performance and worst-case scenarios correspond to landfilling the OBP, considering or not the electricity generation. Differences over ten times in comparative bases among the best and worst-case scenarios were obtained by fossil depletion, global warming, human toxicity, water and metal depletion on LCA's impact categories. The Brazilian characteristic of electricity generation from hydropower plants have considerable influence in some LCA impact categories as Global Warming and Water depletion, especially when accounting for the avoided emissions. Results from Emergy synthesis shows, by far, higher net emergy obtained by the donation scenario, in comparison with biorefinery and landfilling. Donation can save 29 times more emergy than the emergy invested, while biorefinery 1.5 times, and those options located at the bottom of the FRH show lower emergy savings allied to a higher emergy investment. Results highlight the environmental advantages in donating the edible fraction of OBP of CEAGESP, followed by a biorefinery scenario recovering energy and materials, both options aligned with the concepts of CE towards a sustainable development. Donation and biorefinery options are able to achieve, simultaneously, a higher number of UN sustainable development goals, and the best performance for LCA and Emergy indicators, therefore, these scenarios should be promoted by public policies.

Keywords: Circular Economy; Food Recovery Hierarchy; Life Cycle Assessment; Emergy; Food Donation; Biorefinery.

Practical and theoretical contribution of this dissertation

This dissertation contributes to the advances of science by providing a scientific-based environmental performance assessment of food donation and industrial transformation options, as proposed by the concept of food recovery hierarchy management. Both options are not well explored in the literature, from life cycle assessment and emergy perspectives. In particular, the relation between invested and saved emergy for different management options is assessed to bring scientific robustness for the food recovery hierarchy concept under a donor side perspective value quantification.

Focusing on public policies, a new approach called “sustainable performance score” is proposed by combining the LCA and emergy indicators of the modelled scenarios for organic by-products management, and their achievement for the sustainable development goals proposed by the United Nations. This approach allows for a more holistic perception of advantages of the modelled scenarios evaluated.

LIST OF FIGURES

LIST OF TABLES

LIST OF ACRONYMS

AD: anaerobic digestion
BE: bioeconomy
BP: by-products
CBE: circular bioeconomy
CE: Circular Economy
CEAGESP: Companhia de Entrepostos e Armazéns Gerais de São Paulo
CHP: Combined Heat and Power plant
EMA: emergy analysis
EMI: invested emergy
EMS: saved emergy
ERI: emergy return index
ETSP: Entreposto Terminal São Paulo
FDC: Food Distribution Centers
FDP: Fossil Depletion Potential
FEP: Freshwater Eutrophication Potential
FD: Food Donation
FL: Food Loss
FLW: Food Loss and Waste
FRH: Food Recovery Hierarchy
FU: Functional Unit
FW: Food Waste
FSC: Food Supply Chain
GHG: Green-house gases
GWP: Global warming potential
HTP: Human Toxicity Potential
LCA: Life Cycle Assessment
LCI: Life Cycle Inventory
LCIA: Life Cycle Impact Assessment
MDP: Metal Depletion Potential
MSW: Municipal Solid Waste
MSW – OF: MSW organic fraction
NEB: net-emergy benefit

NMF: non-marketable food

OBP: organic by-products

OF: Organic Fraction

NMOC non-methane organic compounds

NMVOC: No-Methane Volatile Organic Compounds

PMFP: Particular matter formation potential

POFP: photochemical oxidant formation potential

RDF: Refuse derived fuel

RF: Residual fraction

SPS: sustainability performance score

TAP: terrestrial acidification potential

TS: Total Solids

UEV: Unit Emergy Value

VS: Volatile Solids

WDP: water depletion potential

CONTENTS

1. INTRODUCTION & JUSTIFICATION

Over the last few decades, the relationship between economic growth and environmental pollution has become the subject of passionate investigation. Societal development has resulted in a large and growing worldwide consumption of fossil fuels, and in an increased amount of CO_2 released into the atmosphere (Sharma, 2011). In modern societies, the environmental pollution mainly relies on two key issues: 1) the depletion of fossil fuels and limited availability of other non-renewable resources; and 2) waste generation that is pushing biosphere's carrying capacity to its limits. Both could be considered as by-products of the technological development of human society (Brown and Ulgiati, 2002).

In response to this situation, during the last few decades, the idea of Sustainable Development has gradually become a key factor, described as "development that meets the needs of present generation without compromising the ability of future generations" (IUCN, 1980). According to Daly (1990; 2017), two principles define the main characteristics of sustainable development: "First, that harvest rates should equal regeneration rates (sustained yield). Second, that waste emission rates should equal the natural assimilative capacities of the ecosystems into which the wastes are emitted". Therefore, a milestone of sustainable development is the establishment of affordable, effective and truly sustainable waste management able to generate multiple health, safety and environmental co-benefits (Cherubini et al., 2009).

Waste generation causes pressure on both the environment and the human health, thus calling for improved waste management strategies to replace the traditional and current methods. Landfilling is one of the most used waste disposal methods. By considering the municipal solid waste (MSW) fraction, landfill disposal accounts for approximately 23% in Europe (Eurostat, 2020), 50% in the United States (EPA, 2018) and 58.6% in Brazil (Coelho and Lange, 2018), potentially generating environmental consequences such as leachate contamination of underground water as well as methane release into the atmosphere. Incineration, most often considered as another mainstream technology, has faced a rapid development in recent years, although toxic substances such as heavy metals and dioxins released during combustion may cause negative effects on the environment and human health (Wang et al., 2019) entailing high costs for management (Martinez-Sanchez et al. 2015) and negatively impacting the standard of living of populations in urbanized centers.

In regions where landfilling (instead of, for example, incineration) is the most common disposal method, the recovery of the organics (e.g., kitchen waste, tissues, etc.) becomes a priority to minimize landfilling volume and comply with legislative targets (EC, 2008). The material recycling, and thus the minimization of waste to be disposed of, is a basic concept that must be implemented to meet the sustainable development goals in both industrialized and developing countries. It has been claimed that the carrying capacity of the planet has already been exceeded in several areas, for example, regarding temperature regulation processes that were altered by greenhouse gases emissions (Rockström et al., 2009). Energy efficiency and cleaner energy have been recognized as key factors to minimize the cost and negative effect of climate change on the environment and society (EU, 2006), and for this purpose, the circular economy concept plays an important role.

Circular economy (CE) is seen as a new business model expected to lead to a sustainable development and a harmonious society. According to Kirchherr et al. (2017, p.229) who analyzed 114 different definitions of CE, it is "*an (economy) that replaces the end-of life concept, with reducing, alternative 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 benefit the current and the future generations. It is enabled by novel business models and responsible consumers.*" Therefore, CE promotes waste prevention and reduction, efficiency increase, resource exchange, reusing and recycling across scales, to get out of the old paradigm "take, make and dispose" towards more sustainable production and consumption patterns. This concept, among others, can be applied to food production and consumption chains.

Globally, nearly one third of food produced for human consumption is lost or wasted, matching a total of 1.3 billion tons of food per year (FAO, 2011). Food loss and waste are responsible for many environmental impacts, such as soil erosion, deforestation, water, and air pollution, as well as greenhouse gas emissions that occur in all steps of the food supply chain (Shanes et al., 2018); this uneaten food represents an unnecessary exploitation of natural resources.

Food is wasted throughout the food supply chain (FSC), from initial agricultural production to final household consumption, and this wasted food could be considered as a by-product of the FSC. In medium and high-income countries, food is largely wasted, meaning that it is discarded even if it is still suitable for human consumption. Significant food loss and waste, however, also occur early in the food supply chain. In low-income countries food is mainly lost during the early and middle stages of the food supply chain; much less food is wasted at the consumer level (FAO, 2011; FAO 2019). Figure 1 shows that the per capita food loss in Europe and North America is about 280-300 kg/year; in sub-Saharan Africa and South/Southeast Asia it is 120-170 kg/year. The total per capita production of edible parts of food for human consumption is, in Europe and North America, about 900 kg/year and, in sub-Saharan Africa and South/Southeast Asia, 460 kg/year. Per capita food wasted by consumers in Europe and North America is 95-115 kg/year, while this figure in sub-Saharan Africa and South/Southeast Asia is only 6-11 kg/year (FAO, 2011).

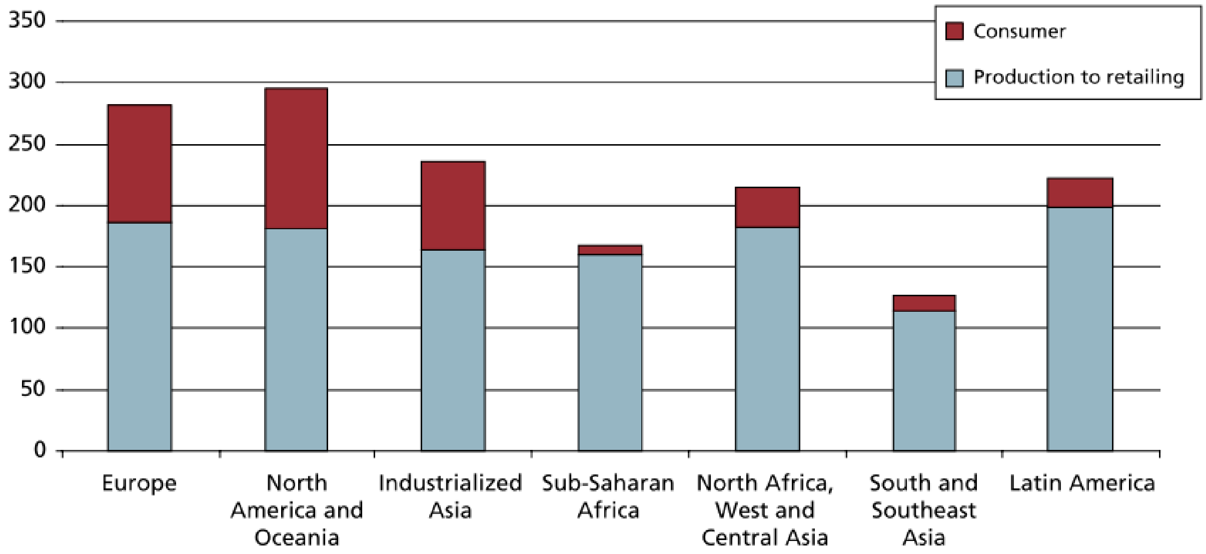

Figure 1: Per capita food losses and waste, at consumption and pre-consumption stages, in different regions (FAO 2011).

Food waste mainly occurs due to losses at consumer level (i.e., wasting of food still suitable for consumption), and losses through the food supply chain (i.e. losses and damage of food during harvest operation, spillage or industrial processing and distribution). According to Schneider (2013), more appropriate management could be designed for the edible fraction, since it has higher potential to be used as food than as waste. Moreover, due to its composition and energy content, the discarded food could be valorized as alternative feedstock, facing, at the same time, the problems related to the use of traditional feedstocks (Ebner et al., 2014). To reduce the quantity of waste to be landfilled and/or incinerated, and in order to promote a circular economy framework by recovering energy and material resources, technological solutions could be implemented at different levels of the supply chain.

Most waste generation (food waste, in particular) occurs in cities, where more than 50% of the world population live. It is necessary to find new ways to improve the efficiency of the existing food waste management, focusing on all steps of the food production chain, from waste prevention to innovative waste refining processes; the latter could provide potential solutions for energy and materials recovery. Nowadays, food waste generation is so abundant and so centralized that there is insufficient capacity for its natural degradation. Therefore, a better management model for food waste other than landfills is mandatory, including composting, anaerobic digestion, industrial uses, and, for the edible portion, reusing scenarios where wasted food is still managed as edible food. Donation scenarios, for example, are options that cause lower load on the natural environment and reduce social issues related to food insecurity. “Biorefinery” scenarios, which are capable of exploiting all reusable fractions of organic by-products generated by the food supply chain, could represent a way to avoid the downstream impacts related to food waste landfilling and provide the society with useful products, thus saving, at the same time, the natural resources necessary to produce the same product elsewhere.

Food distribution centers (FDC) are companies that provide an efficient circulation of products in highly populated cities, allowing for products transfer from agricultural areas to urban centers. Considering that cities, mainly the highly populated ones, import their own food from other regions, rather producing it, a market centralization through FDC proves important, so as to avoid logistic problems that can have a negative influence on the quality of the transported food, such as mechanical injuries that reduces its market value.

Due to the high concentration of by-products derived from food trading operations, and because they are usually located in big cities, FDC could represent an interesting scenario to apply the previously mentioned circular economy principles and verify their advantages. The CEAGESP food distribution center in São Paulo city, in Brazil, is particularly interesting. It is the biggest food supply center of all Latin America and the third one in the world, with more than three million tons of products (mainly horticultural) traded yearly. An average of ~52,300 tons of waste is generated per year, mostly food waste. Currently, around 90% of this waste is landfilled, without any attempts to valorize it, and causing high environmental, social, and economic costs (CEAGESP, 2018). This situation claims for new efforts towards food waste reduction and valorization; the CEAGESP FDC is, therefore, considered an interesting case study for the implementation of circular economy alternatives.

2. OBJECTIVES

2.1. General objective

This study uses the Life Cycle Assessment and Emergy Synthesis as methods to identify and quantify environmental costs and benefits when circular economy and food waste hierarchy concepts are applied to different options for food by-products management. The CEAGESP food supply center, located in São Paulo, Brazil, is considered as a case study, to exemplify procedures and numbers.

2.2. Specific Objectives

(a) Perform a literature review on the available scientific literature related to food waste generation, waste prevention, food redistribution, energy and resources recovery from organic by products generated by the food sector, and integrated environmental assessment methods to identify the main problems and limitations in scenarios for food by-product management options;

(b) Obtain information (qualitative and quantitative from fieldwork, technical reports, databases and published data) about the current food waste management applied by the “CEAGESP” Food Supply Center;

(c) Apply the Life Cycle Assessment and Emergy Synthesis on the current food waste management adopted by CEAGESP;

(d) Modeling scenarios for food donation and biorefinery as options to substitute the current management of organic by-products generated by CEAGESP;

(e) Applying the Life Cycle Assessment and Emergy Synthesis on the donation and biorefinery modelled scenarios;

(f) Propose a framework to support public policies based on the environmental performance of current scenario versus the proposed scenarios for the CEAGESP organic by-products management.

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3. LITERATURE REVIEW

This literature review is developed according to the following criteria: section 3.1 focuses on the food supply chain, investigating the main processes linked to food waste generation; section 3.2 focuses on the environmental impacts of landfilling, currently the most common food waste disposal; section 3.3 presents which scientific methods are the most appropriated to estimate the environmental impacts generated by landfilling waste; section 3.4 assess which tools and strategies are available to obtain a more sustainable food waste management and their effectiveness from a scientific point of view, seeking for the state of the art in research topic. This approach covers all the subjects dealt with in this thesis, allowing for a better understanding about the reasons (criteria) and methods considered herein, emphasizing its scientific contribution in the topic studied.

3.1. Food supply chain (FSC)

The food supply chain is defined as the movement of products and services along the value-added chain of food commodities that aims at realizing better value for the customer alongside cost minimization (Folkerts and Koehorst, 1998). It could be divided into five steps, which include (i) farm production, (ii) handling and storage, (iii) processing, (iv) distribution and (v) consumption (Porter et al., 2016). Food is wasted along all stages of the FSC. The generation of food waste implies several impacts in the three pillars of sustainability, including crop losses during harvest or storage, hunger in low-income countries, and the deprivation of natural resources without accomplishing its final purpose (Vandermeersh et al., 2014).

Different definitions of food waste have been proposed in literature (Alexander et al., 2017; FAO, 2011; Henz and Porpino, 2017; Lipinski et al., 2013; Monier et al., 2010; Porter et al., 2016; Stenmark et al., 2016; Tisserant et al., 2017), with a further distinction among food loss, food waste, bio waste, solid waste and other subcategories, depending on the author(s). By comparing the different authors, the common distinction between food loss (FL) and food waste (FW) (Figure 2) is recognizable, where the former usually accounts for losses at the upstream stages of FSC (Alexander et al., 2017; FAO, 2011; Henz and Porpino, 2017; Lipinski et al., 2013; Porter et al., 2016) while the latter indicates losses at the final stages, by including losses at retail level (FAO, 2011) or by exclusively considering losses at consumer level (Alexander et al., 2017; Henz and Porpino, 2017; Lipinski et al., 2013; Porter et al., 2016).

Monier et al. (2010), Stenmark et al. (2016) and Tisserant et al. (2017) have adopted different nomenclature patterns. Monier et al. (2010) call “bio waste” the biodegradable garden and park waste, food and kitchen waste from households, restaurants, caterers and retail premises, and comparable waste from food processing plants. It does not include forestry or agricultural residues, manure, sewage sludge or other biodegradable waste. “Food waste” is part of biowaste, composed of raw or cooked food materials. It includes food materials discarded at any time between farm and fork. Stenmark et al. (2016) call Food Waste the "fractions of food and inedible parts of food removed from the food supply chain to be recovered or disposed", including all the available disposing or recovering options, such as anaerobic digestion, composted crops, bioenergy production, disposal to sewer, incineration and landfilling. Tisserant et al. (2017) consider a more general "solid waste" definition, referring to any solid output from human activity that remains inside the Technosphere, and that requires further treatment before it can be released to the environment or be used as a substitute for other industrial products.

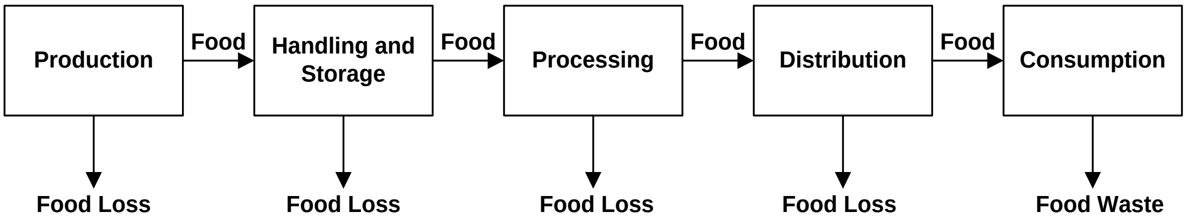

Figure 2: Food Loss and Waste along the FSC according to most common definitions in literature

With a general definition of FL and FW provided, the following paragraphs show further insights regarding edible/inedible parts, avoidable and unavoidable fractions as well as information regarding the main causes of food loss and waste generation along the food supply chain.

Almost all the previously mentioned studies have included both the edible and inedible parts of food lost or wasted, while FAO (2011) considers only the edible fraction. Two approaches are recognizable, depending on food destination: (I) food intended for human consumption, but eventually not eaten by humans (FAO, 2011; Porter et al., 2016); (II) food waste sent to waste management facilities (Alexander et al., 2017; Monier et al., 2010; Stenmark et al., 2016; Tisserant et al. 2017;). Only the first of these approaches recognizes the wealth of Food Waste stream as a potential resource.

Corrado and Sala (2018), when assessing some of the above-mentioned works, made an attempt to quantify both the average breakdown of food waste per food type, as well as the average waste coefficients (the percentages of inputs to a certain stage of the supply chain which end up being FW) along the FSC. Only two of those (FAO, 2011; Porter et al., 2016) have calculated these values at global scale, and results are shown in Table 1. Besides a great variability related to the food type and author's assumptions, Table 1 shows that most of the cases that have the steps of agricultural and marine production and final consumption are the most impacting, usually followed by distribution.

Differences in percentages of FW along the FSC also depend on geographical areas. As shown in Figure 3, that depicts the breakdown of the 100% of FLW at the different steps of FSC, in developing countries, the highest percentage of FW generation is concentrated at the

Table 1: Food Waste percentage coefficients considered in the studies per food product group and per food supply chain stage. The breakdown proposed by Porter et al. (2016) was considered both for the supply chain and for the food product groups. (p) = processed product, (f) = fresh product. Adapted from Corrado and Sala (2018).

Food Group	Production (Agr. and Marine)		Storage and Handling		Manufacturing		Distribution		Consumption	
	FAO, 2011	Porter et al., 2016	FAO, 2011	Porter et al., 2016	FAO, 2011	Porter et al., 2016	FAO, 2011	Porter et al., 2016	FAO, 2011	Porter et al., 2016
Cereals	2	4.33	4	3.85	10	10.5	2	3	25	27
Fruit and Vegetables	20	20	5	7.32	2	2	10 (f); 2(p)	4.87 (f); 2(p)	19 (f); 15 (p)	19 (f); 15 (p)
Marine	9.4	9.4	0.5	7.9	6	6	9 (f); 5(p)	9 (f); 5(p)	11 (f); 10 (p)	11 (f); 10 (p)
Meat	3.2		0.7		5	5	4	4.05	11	11
Bovine		2.3		0.63						
Mutton and Goat		10		0.59						
Pig		2.5		0.32						
Poultry		7		0.94						
Eggs	4	4	-	1.86	0.5	0.5	2	2	8	8
Milk	3.5	3.5	0.5	1.67	1.2	1.2	0.5	0.82	7	7
Oilseeds and Pulses	10	5.28	1	1.15	5	5	1	1	4	4
Roots and Tubers	20	20	9	7.61	15	13.82	7 (f); 3(p)	7 (f); 3(p)	17 (f); 12(p)	17 (f); 12(p)

production stage, achieving a maximum value of 39% in sub-saharian Africa. Conversely, in developed countries, the problem is at the consumption stage, with North America showing the maximum value of 61%. A decreasing trend of the percentage of FLW at consumption level that corresponds to an increase at production and handling and storage level from the most developed to the least developed countries is evident.

There is an important distinction between avoidable and unavoidable food waste. Avoidable food waste is food thrown away because it is no longer wanted or has lost the minimum quality level to be accepted by consumers. It is composed of formerly edible material, in many cases, at some point prior to disposal, even though not edible at the time of disposal, due to deterioration. Avoidable food waste comprehends food recognized as edible by the vast majority of people, while unavoidable food waste is waste generated by non-edible food, under normal circumstances, such as fruit skin, apple cores and meat bones (Papargyropoulou et al., 2014).

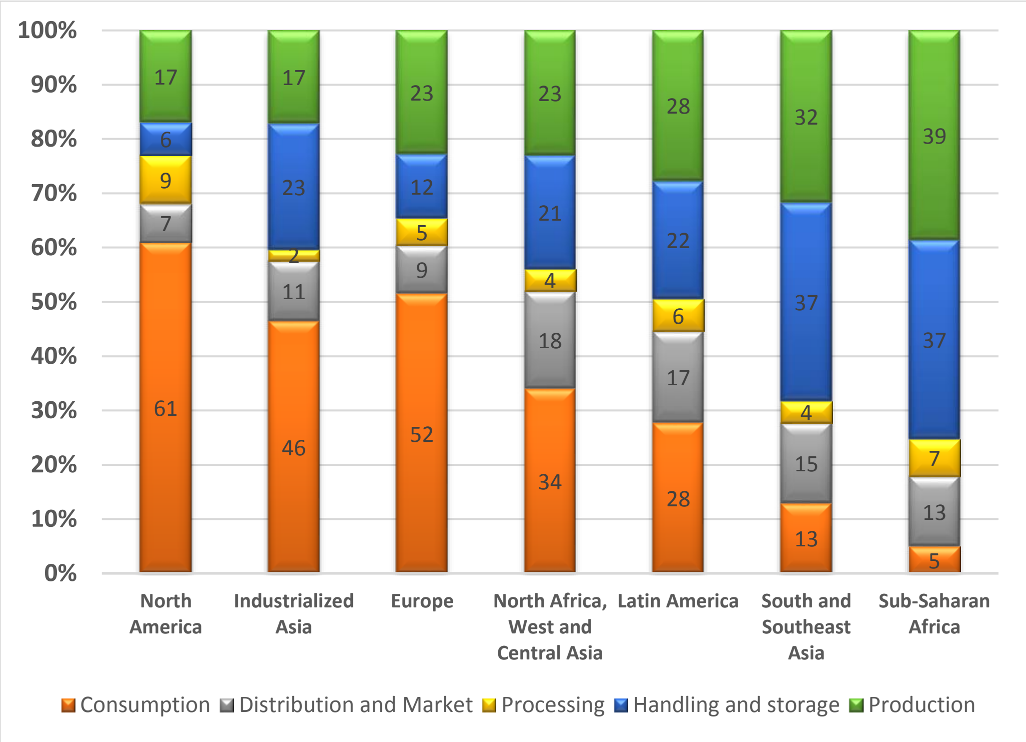

Figure 3: FLW for region and Stage along the Food Supply Chain (Percent of kcal lost and wasted; adapted from Lipinski et al., 2013).

According to FAO (2011), food could be wasted for different reasons, with some differences between developed and developing countries. In the former case, food is wasted mainly when production exceeds demand, when it does not have the minimum aesthetical standards to be attractive for consumers, and when disposing is cheaper than the 'using or re-using' attitude. In the latter case, it is mainly wasted due to poor storage facilities, lack of infrastructure and premature harvesting. The case when food is wasted only due to aesthetical reasons that do not affect its edibility is very relevant and has been explored by different authors in literature (Albizzati et al., 2019; Brancoli et al., 2020; Fagundes et al., 2014; Legaspe, 2006; Papargyropoulou et al., 2014).

This type of wasted food is named "surplus food, unsold food or non-marketable food", and all the authors recognize its avoidable nature and highlight the importance of recovering its potential nutritional value.

This overview of FLW generation and characteristics has depicted the global importance of the problem, highlighting the differences between developed and developing countries. In this context, the Brazilian situation is particularly interesting, because a high FLW generation is associated with a widespread food insecurity.

In fact, according to Henz and Porpino (2017), there were still 52 million Brazilians in 2017 (about ¼ of the population) threatened by food insecurity, if one considers its three levels (low, moderate and severe) while six out of 10 people were in a situation of food insecurity during the recent pandemic, reducing the consumption of food items important for their regular diets (Silva et al., 2021). People struggling to access food or constrained in their food choices jointly to the high wastage of food in developing countries are the result of a controversial situation (Schneider, 2013; Silva et al., 2021).

By assessing the Brazilian FLW reality from a quantitative perspective, 35% of all agricultural production is lost before consumption,10% of such loss is related to the harvesting process, 50% to handling and transport issues, 30% at wholesale markets and the remaining 10% is divided between supermarket and consumers (RIM, 2013).

Regarding wholesale markets, Fehr and Romano (2001) have assessed the food loss from distribution to consumption of a medium size town (Uberlandia, population 440,000). The authors considered the food arrival at the wholesale market as 100% and found the following percentages: (I) wholesale market loss, 6.28%; (II) Street trading, 11.67%; (III) Supermarkets, 8.76%; the average value at retail level was 11%.

FLW is an important component of Municipal Solid Waste (MSW). At a global level, most of the MSW is disposed into landfills and open dumps. While developed countries such as North America eliminated all its open dumps, this is still a widespread solution in developing countries; Brazil included (De Campos et al., 2021).

The literature review has shown the origin of food loss and waste along the food supply chain. It was identified that a fraction of this food is still edible (named surplus food, unsold food or non-marketable food), calling for new management strategies able to recognize its nutritional value. Moreover, if the above-mentioned option is not possible, it is important to find new pathways capable of rediscovering the potential of wasted food as alternative feedstock.

Currently, without any distinctions between edible and not edible fractions, FLW is treated as organic waste fraction of municipal solid waste (MSW – OF) and follow the municipal solid waste (MSW) most common destination: landfill disposal. This choice causes environmental impacts that will be explored in the next section.

3.2. Environmental impacts of food loss and waste landfilling

Landfilling is strictly connected with environmental impacts, among which methane emissions into the atmosphere and leachate generation are the most impacting. These impacts are mainly caused by the degradation of the organic fraction (OF), which is an important component of MSW that is raising concerns around the globe (Paritosh et al., 2018).

Shortly after MSW disposal, the organic components start to undergo a series of biochemical reactions. In the presence of atmospheric air, which is near the surface of the landfill, the natural organic compounds are oxidized aerobically, a reaction similar to combustion, since the products are carbon dioxide and water vapor. However, the principal bioreaction in landfills is anaerobic digestion, which occurs in three stages. In the first, fermentative bacteria hydrolyze the complex organic matter into soluble molecules. In the second, these molecules are converted by acid forming bacteria to simple organic acids, carbon dioxide and hydrogen; the main acids produced are acetic acid, propionic acid, butyric acid and ethanol. Finally, in the third stage, methanogen bacteria form methane, either by breaking down the acids to methane and carbon dioxide, or by reducing carbon dioxide with hydrogen. Two examples among the most important chemical reactions involved in this process (Eq. 1-3) are presented below (Themelis and Ulloa, 2007).

Anaerobic fermentation of glucose to form ethanol and carbon dioxide:

$$C_6H_{12}O_6 \rightarrow 2C_2H_5OH + 2CO_2 \quad (1)$$

Methanogenesis:

$$CH_3COOH \rightarrow CH_4 + CO \quad (2)$$

$$CO_2 + 4H_2 \rightarrow CH_4 + 2H_2O \quad (3)$$

The maximum amount of natural gas that may be generated during anaerobic decomposition can be determined by Eq.4:

$$C_6H_{10}O_4 + 1.5H_2O = 3.25CH_4 + 2.75CO_2 \quad (4)$$

This reaction releases a small amount of heat and the product gas contains about 54% methane and 46% carbon dioxide. Other important components in biogas are nitrogen (N_2), Hydrogen Sulphide (H_2S) and non-methane organic compounds (NMOC), with average concentration of 5%, 1% and 2700 ppmv respectively (Themelis and Ulloa, 2007).

Another source of environmental impacts caused by FLW waste landfilling is leachate generation, defined as liquid effluents generated by the percolation of rainwater through the solid waste disposed of in landfills, as well as the moisture present in the waste and the degradation products of the residues (Salem et al., 2008). The composition of landfill leachate is highly variable and heterogeneous. However, it generally contains the following components: a high concentration of dissolved organic material (volatile fatty acids and refractory organic compounds, such as humic and fulvic acids); macro inorganic components, including ammoniacal nitrogen ($N\text{-}NH_4^+$), sodium (Na^+), potassium (K^+), chloride(Cl^-), calcium (Ca^{2+}), magnesium (Mg^{2+}), iron (Fe^{2+}), manganese (Mn^{2+}), sulfate (SO_4^{2-}) and hydrogen carbonate (HCO_3^-); heavy metals such as cadmium Cd^{2+}) nickel (Ni^{2+}) chromium (Cr^{3+}), lead (Pb^{2+}), copper (Cu^{2+}) and zinc (Zn^{2+}); and xenobiotic organic compounds as aromatic hydrocarbons, phenols and pesticides (Slack et al., 2005; Schiopu and Gavrilescu, 2010). The volume and the chemical characteristics of leachate changes under different climate conditions (Zierguer – Rodriguez, 2019).

Another environmental impact of MSW-OF landfilling is related to the collection and transportation process to the disposal site, which causes vehicular emissions derived by fuel combustion as CO_2, SO_2, CO, No-Methane Volatile Organic Compounds (NMVOC), NO_X and Particular Matter (Buratti et al., 2015; Larsen et al., 2009).

Furthermore, the impact of food waste is also linked to the processes of the previous life cycle stages of food before it had become waste, such as agriculture (including land use change), processing, manufacturing, transportation, storage, refrigeration, distribution and retail (Papargyropoulou et al., 2014). This overview on MSW landfilling has shown the environmental impacts derived by this technological route. It is evident the connection with transport steps and organic waste degradation after landfill disposal, and that upstream impacts related to food production cannot be neglected.

3.3. Environmental Impacts Assessment

The environmental impacts of OF-MSW management claims for either an appropriate method or a combination of different methods to be quantified. The complexity of the aspects involved for an effectively integrated MSW management leads to the development of different approaches to improve decision-making (Liu et al., 2017a). The two most common approaches are from a “user side perspective” and from “a donor side perspective”. The first approach, typical of methods as Life Cycle Assessment or Ecological footprint, accounts for those flows of matter and energy under human control, while flows outside the economic system as well as flows of labor, culture, or information are usually not included. The second approach, typical of methods such as Emergy Synthesis, considers the anthropic processes as embedded in natural systems, including all the inputs demanded to support them, and enlarging the space-time scale that generates and sustains them.

The following three subsections explore characteristics, strengths and weaknesses of the most used approaches, by assessing their single or joint use. It is discussed whether a parallel application, a complementary approach, or an integration among different methods is able to provide more reliable results, as claimed by some authors (Gala et al., 2015; Patterson et al., 2017; Pizzigallo et al., 2008; Raugei et al., 2014).

3.3.1 Life cycle assessment (LCA): an user-side approach

Over the last years, studies were published in which LCA is applied to evaluate different waste management scenarios in several countries: Australia (Edwards et al., 2018), UK (Evangelisti et al., 2015; Tunesi, 2011), Spain (Bovea and Powell, 2006; Bueno et al., 2015; Fernandez-Nava et al., 2014), Switzerland (Rossi et al., 2015), Norway (Slagstad and Brattebø, 2012), Sweden (Carlsson Reich, 2005; Bernstad and la Cour Jansen, 2011), Denmark (Andersen et al., 2012; Boldrin et al., 2011; Jensen et al., 2016), Germany (Jensen at al., 2016), Italy (Buratti et al, 2015; Cherubini et al., 2009; Ripa et al., 2017), Perù (Ziegler-Rodriguez et al., 2019) and Brazil (Liikanen et al., 2018; Mendes et al. 2004; Oliveira et al., 2017). Some of the above show interesting aspects that are useful for this present study, due to the system analyzed, the geographical location, or innovative approaches, as discussed next.

Mendes et al. (2004) compare the environmental impacts of incineration and landfilling of municipal solid waste in São Paulo City, Brazil, by analyzing five different scenarios under the LCA perspective. Three of those involve incineration with energy recovery while two are landfilling scenarios with and without energy recovery. The authors assessed emissions into air and water, energy recovery, recovered resources and energy consumption, by evaluating global warming, acidification, and nutrient enrichment as impact categories. The results show that landfilling has higher impact than incineration in all LCA categories assessed, with a slight advantage for the scenario with energy recovery. As for the global warming potential, the small improvement in the scenario with energy recovery is caused by the peculiarity of Brazilian electricity, generated mainly by hydropower plants.

Cherubini et al. (2009), in their work on waste management in the biggest Italian city, Rome, face the challenge of finding an affordable, effective, and sustainable waste management model, investigating scenarios that could be applicable to other big European cities with the same waste composition. By using LCA as a method and the amount of waste produced in Rome in 2003 as the functional unit, four different scenarios were studied: (0) wastes are delivered to landfill without any further treatment; (1) part of the biogas released by the landfill is collected and used to produce electricity; (2) a sorting plant is present at the landfill site for separation of the inorganic and organic fractions, and of ferrous metal recovery; electricity, biogas, and compost are then produced on site; (3) waste is directly incinerated to produce electricity. For each scenario, liquid, solid and gas emissions are evaluated and classified into impact categories to estimate indicators such as Global Warming Potential, Acidification Potential, Eutrophication potential. The results show that scenario (2) appears as the best option, as it is the only one which takes into account both components of waste: the organic one to produce biogas, and the inorganic one to produce electricity via combustion. Scenario (1) only exploits the organic part (landfill gas) and scenario (3) the inorganic part (direct combustion). At local scale, landfill options have the lowest values for CO_2, NO_x, PM_{10}, and dioxin, and higher values for CH_4 and H_2S, while scenarios (2) and (3) have more relevant emissions of CO_2, NO_x, PM_{10} and dioxin (due to the percentage of presence of plastic) generated by the incineration steps. Thus, there is a conflict between global and local scales, since what is positive at a global scale is negative at a local scale. Eventually, the authors remark the importance of waste sorting as a preliminary step, especially the separation of the organic fraction from the inorganic one, to maximize the material and energy recovery and, consequently, reduce the environmental impacts.

Buratti et al. (2015) focused on the organic fraction and the two most common methods of treatment in Italy: landfilling and composting. From an LCA point of view, the authors considered one ton of organic fraction as a functional unit and evaluated fifteen different impact categories at mid-point and end-point levels for two scenarios: (1) undifferentiated collection, mechanical and biological treatment followed by disposal in landfills, (2) source separated collection and production of high-quality compost. The results show that landfilling of the undifferentiated organic waste has the lowest impacts in all categories analyzed, except global warming potential, mostly due to the methane released into the air by the landfill. As for the composting of the source segregated organic fraction, to reduce the impacts, it is necessary to focus on the reduction of emissions into the atmosphere (hydrogen sulfide, particulate, ammonia and no-methane volatile organic compounds NMVOC) from the bio stabilization process.

The study of Oliveira et al. (2017) aims to examine six alternatives to composting organic waste generated in the city of Bauru, in São Paulo state, where there is neither a composting plant nor planning for home composting development in the future. The LCA is implemented using the Recipe (2008) method for the impact categories of climate change, ozone depletion, particulate matter formation, human and freshwater ecotoxicity. The authors assessed 7 scenarios: (a) sending organic waste to BAURU landfill (base); (b) shipping the organic waste to another city that has a composting plant; (c) building a composting plant in Bauru; (d); (e); (f); (g); using home composting for respectively 10%, 25%, 60%, 90 % of the organic waste. GWP accounted for CO_2, CH_4, and N_2O emissions. Methane emissions have the most important role, 90% in all cases. Scenarios B and C showed lower emissions since they consider the dispatch to composting plants. Scenarios from (d) to (g) home composting have less emissions of methane but higher of N_2O. As for the ozone depletion, home composting has the best performance, followed by industrial composting. Particulate matter shows the worst performance in scenarios (b) and (c). The authors conclude that the composting and home composting of organic fraction usually have a lower environmental impact, but this was not always true for all impact categories.

Liikanen et al. (2018) implemented LCA to evaluate the environmental impacts of different management alternatives for MSW in Sao Paulo to determine a pathway towards more environmentally sustainable MSW management. The Impact categories assessed were global warming potential (GWP), acidification potential (AP) and eutrophication potential (EP). The authors assessed 5 scenarios, with progressively more waste separation and recycling. In some scenarios, electricity production is included. Functional unit is the total amount of waste managed in 2015. The results show that GWP is higher for the landfilling scenario and landfilling + composting + incineration; moreover, general greenhouse gases emissions from landfilling are by far higher than emissions from transportation. Electricity production does not affect the results due to the special characteristics of electricity production in Brazil (mainly from hydropower plants). As for AP, except for scenarios with incineration, in all cases the emissions are negative due to electricity substitution. In scenarios with incineration, the direct emissions overcome the emissions avoided by producing electricity. The results for EP are similar to those for AP, with more importance of the collection and transportation steps. In conclusion, the authors highlight the role of source separation and recycling to reduce the environmental impacts.

By considering all these reported studies, the main outcome is that landfilling always showed the highest environmental impacts on global warming potential, due to methane emissions. As for the other categories evaluated, especially for acidification potential and eutrophication potential, in some cases composting has shown the worst results (Buratti et al., 2015). Moreover, it is important to consider the differences between the impacts assessed at both local and global scales.

3.3.2 Emergy: a donor-side approach

Several works have used emergy analysis (EMA) to evaluate MSW environmental impacts. Among them, the following ones were selected for being more closely related to this present study.

Marchettini et al. (2007) applied EMA to assess different waste management options in Italy,to verify which one had better performance for sustainability. The work was implemented by using Emergy Yield Ratio (EYR; emergy released per unit invested) and Net emergy indicators. Three scenarios were assessed: landfilling, incineration, and composting plant. The results show that composting has the lowest demand for resources per gram of waste compared to the incineration and landfilling options; on the other hand, incineration and composting are more efficient in recovering eMergy from refuse. Both options succeed in recovering at least part of the potential of the waste into becoming a valuable resource. For incineration, this potential is represented by the energy content of the refuse, while for composting, this potential is represented by the organic component. EYR and NET eMergy are able to measure, respectively, the efficiency and the amount of eMergy recovery; the former has higher efficiency than incineration, while the latter is able to recover a greater amount of eMergy. Both indicators highlight landfilling as the worst option for eMergy recovery. In this work, the authors did not consider the emissions.

Liu et al. (2013) used emergy synthesis for a MSW disposal practice in Liaoning province, China, considering the emissions impacts on ecosystem, economy and human health integrity. The study has used emergy synthesis to directly calculate the resources demanded, while the impacts were calculated by considering the damage caused by emissions on human health and ecosystems as an additional indirect demand for resource. Human resources (considering all its complexity: life quality, education, know-how, culture, social values and structures, hierarchical roles, etc.) are considered as a local slowly renewable storage that is irreversibly lost due to the pollution and high rate of demand for it in the processes. The same approach was used regarding ecosystems, by considering them as the results (stock) of slow renewable processes. The study estimated the damage using the Disability Adjusted Life Years (DALY) and Potentially Disappeared Fraction of Species (PDF). Such effects can be quantified as the emergy loss of local ecological resources. The work compared four scenarios: sanitary landfills systems, fluidized bed incineration system, grate type incineration system and the current landfills system (without leachate capture and treatment). Results of the total emergy use, including the impacts of emissions, are: sanitary landfills (3.87×10^{16} seJ/t-waste) > current landfills (3.71×10^{16} seJ/t-waste) > grate type incineration (2.39×10^{16} seJ/t-waste) > fluidized bed incineration (2.38×10^{16} seJ/t-waste).The authors concluded that the emergy based urban solid waste model can be considered a useful tool for decision makers to compare different MSW options. The limit of this approach is the impacts estimations executed only indirectly by calculating their environmental costs in emergy, without the integration of emissions impacts into specific performance indicators.

These reported studies that have used emergy synthesis as the only method to evaluate environmental impacts have highlighted landfilling as the worst option. They also depicted the limits of the emergy method in quantifying emissions, that were simply ignored (Marchettini et al., 2007) or estimated indirectly (Liu et al., 2013).

3.3.3 LCA and Emergy accounting: two complementary perspectives

This literature review on the application of a single one-dimensional approach, LCA or Emergy synthesis, besides emphasizing landfilling as the worst scenario in most cases, has shown that applying one single method will hardly provide a complete vision of the functioning and interactions that characterize a systemic perspective. LCA has shown its potential regarding emissions assessment and short time-space scale, but it ignores the upstream nature’s work necessary to generate resources. On the other hand, Emergy Synthesis focuses on the upstream aspects, but it ignores emissions at downstream. To be really understood, environmental aspects should be evaluated from a 360-degree approach, which means a holistic perspective. Life Cycle Assessment and Emergy Synthesis have different approaches, the former from the user-side, and the latter, from the donor-side, each one focusing on different aspects of system performance.

According to Raugei et al. (2014), two closely connected and fundamental aspects in which LCA diverges from EMA are those of system boundaries and implied point of view. LCA designs the time and space boundaries of the system under analysis in function of its own "life cycle", defined as the list of anthropic processes exchanging commodities (through market relations) that are directly or indirectly influenced by modifications of a functional unit. Conversely, EMA always considers the assessed system as part of a larger natural system that underpins it and includes all direct and indirect inputs that converged to support it over a much larger time and space scale (Figure 4). The choice of boundaries and impact assessment methods in LCA are perfectly consistent with its user-side perspective, where the interest lies in how many resources were directly consumed during the process under investigation, converted into emissions. On the other hand, EMA is fully consistent with its donor-side perspective, since it encompasses all the resources directly and indirectly supplied by nature in order to support the product or system under study, independently of the actual user-side usefulness.

The joint use of these two methods, both through complementary and parallel perspectives, or by integrating them to overcome individual deficiencies, was and still is discussed in the literature (Gala et al., 2015; Liu et al., 2017a,b; Lye et al., 2021; Pizzigallo et al., 2008; Raugei et al., 2014; and Santagata et al., 2020). In particular, Pizzigallo et al. (2008) have evaluated the joint use of LCA and emergy from a complementary perspective, by considering two Italian wine farms as a case study. Authors argue that emergy evaluation offers a wider overview than LCA as it includes the productive cycle into the environmental context in which it is found, then it quantifies, in terms of energy flows, its relations with the natural environment. While emergy aggregates the ''gate'' phase and can miss details needed for actions to be taken, the LCA quite neglects the ''cradle'' phase. The joint use of these two methods has proved to be very useful, providing a much wider range of direct usable information, when compared to using both separately.

Rugani and Benedetto (2012), in an attempt to integrate both methods, assessed the fundamental requirements to improve the Emergy synthesis by using LCA. They focus on the weaknesses of emergy, which, despite its capability to compare the amount of resources embodied in production systems, has various problems such as vague accounting procedures and lack of accuracy, reproducibility, and completeness. According to the authors, an improvement of Emergy synthesis can be achieved by (1) technical implementation of Emergy algebra in the Life Cycle Inventory (LCI), (2) selection of consistent Unit Emergy Values (UEVs) as characterization factors for Life Cycle Impact Assessment (LCIA), and (3) expansion of the LCI system boundaries to include supporting systems usually considered by Emergy but excluded in LCA (e.g., ecosystem services and human labor).

Whereas Emergy rules must be adapted to life-cycle algebra structures, LCA should enlarge its inventory to give emergy a broader computational framework.

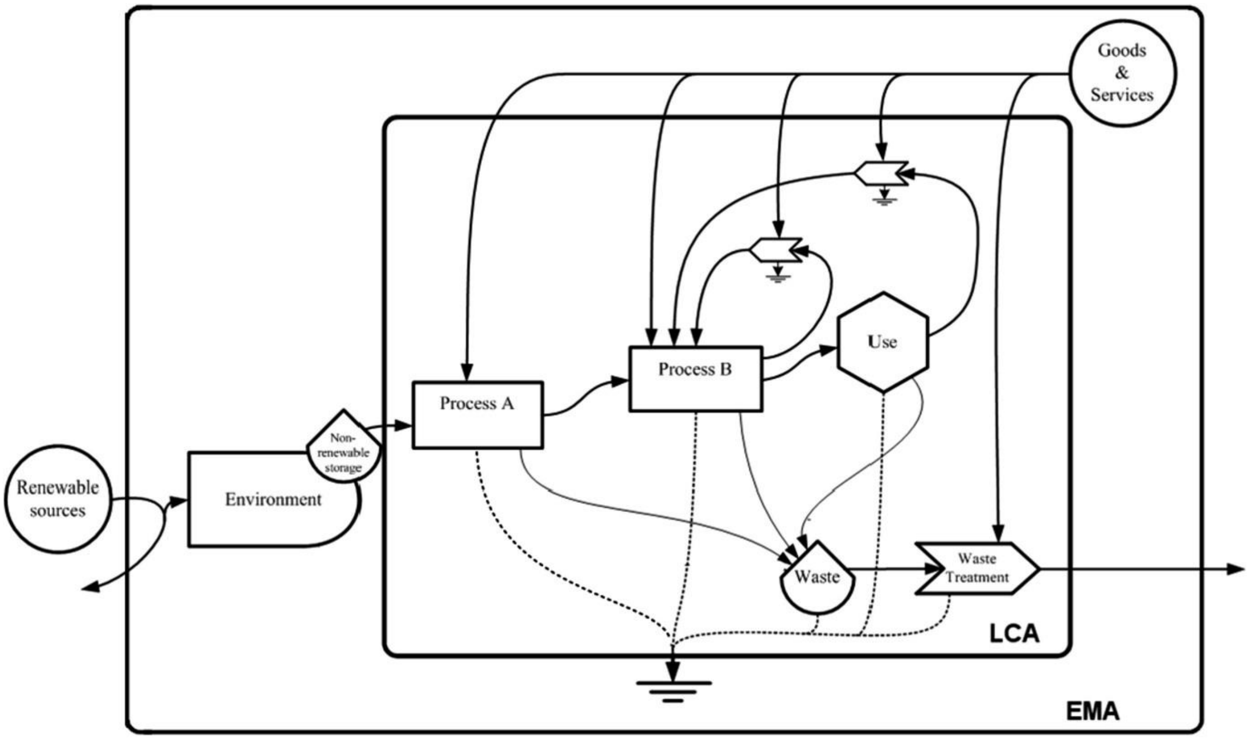

Figure 4: Different approaches and scales in LCA and Emergy Accounting (adapted from Santagata et al., 2019).

The work of Gala et al. (2015) provides a brief overview of the main critical points when dealing with waste management systems (with selected illustrative examples) and how they have so far been addressed in LCA. Authors discuss the extent to which the work done in the LCA community may be leveraged to improve the clarity and consistency of EMA when applied to waste management. At the same time, they also highlight and discuss those instances where underlying LCA conflicts with EMA perspective, thereby rendering some of the assumptions and solutions proposed by the former essentially inapplicable within the framework of the latter. They conclude that, despite the many steps already made towards the fruitful comparison and integration of LCA and EMA regarding waste management assessment, there is still a number of unsolved issues that call for further research. The need for further standardization to achieve a fully consistent and comparison-friendly boundary and accounting procedures in LCA and EMA was recognized. The necessity for better and more widespread comprehension and awareness of the different inherent perspectives offered by the two methods was also highlighted. Therefore, a forced integration in those cases when the intended goal of the study does not require it is not necessary.

Patterson et al. (2017) highlighted that, along the last few decades, several methods of environmental accounting have been developed to conceptualize and quantify the direct and indirect effects of human activity on the environment, with the purpose of helping decision-makers towards the best decision to achieve sustainable targets. These methods range from ecological footprint, carbon footprint, energy analysis, eMergy analysis, ecological pricing and life cycle assessment to environmental input-output analysis. The development of these tools was implemented in isolation from each other, even though they often seek to serve common analytical and evaluative purposes. The authors try to find the common features of these methods that are often the same issues on logical and mathematical quantifications as, for example, the co-products problem, weighting, commensuration, double counting, boundary setting, and analyze how the various environmental accounting tools can 'learn' from each other. A better understanding of any given environmental issue could be easily achieved using a mix of these environmental accounting tools, rather than relying on just one tool, one perspective, or one criterion.

From a general view, scientific literature shows that a multidimensional approach offers better opportunities to achieve a more complete environmental assessment (Patterson et al., 2017). In particular, the joint use of LCA and EMA methods provides a much wider range of directly usable information when compared to their use separately (Pizzigallo et al., 2008). Moreover, it was depicted that several problems are still present in the attempts to integrate the methods (Gala et al., 2015; Rugani and Benedetto, 2012) but, nevertheless, it has also been proven that a forced integration, in those cases when the intended goal of the study does not require it, is not necessary (Gala et al., 2015). For these reasons, due to the better reliability and completeness of a multidimensional approach and the problems related to the integration of the two methods, in this present study the LCA and EMA are jointly applied according to a complementary and parallel perspective.

3.4 Food Recovery Hierarchy

The environmental impacts derived by FLW landfilling call for new tools and strategies towards a more sustainable FLW management, able to reduce the amount of landfilled food and recognizing its potential hidden wealth. Among them, the waste hierarchy pyramid has received increasing attention in recent years by scientists and policy makers.

The waste hierarchy concept was developed during the late 1970's, and it is commonly described as a priority order to be chosen for at least three waste management options based on their environmental impacts (Hultman and Corvellec, 2012; Van Ewijk and Stegemann, 2016). In 2008, the waste hierarchy principle was included in the Waste Framework Directive 2008/98/EC (WFD) established by the European Commission (EC, 2008) that has updated and refined the conceptual model for waste hierarchical management. In this model, the priority order is waste prevention and management policies, reducing the demand for new products and/or reducing the amount of generated waste. The alternative options proposed, from highest to lowest priority, are waste prevention, preparing for reuse, recycling, recovering (i.e.energy recovering), and disposal.

In 2015, the Circular Economy Strategy from EU COM/2015/0614 (EC, 2015) defended the role of waste hierarchy as the main reference regarding waste management, considering it as the path to lead to the best overall environmental outcome and to get valuable materials back into the economy (Pires and Martinho, 2019). Therefore, waste management hierarchy and circular economy are strictly connected.

Precisely, while the waste management hierarchy categorizes waste management approaches into more and less desirable ones (including waste prevention, material re-use and recycling preferable over energy recovery and landfilling), circular economy pursues the idea that materials open-end flows are closed through re-use and recycling, by including the waste hierarchy concept as an integral part of itself (Traven, 2019).

Countries worldwide have different laws to suggest the most appropriate waste management options, incorporating the main principles derived by the food waste hierarchy. In Brazil, this occurred in 2010 with the promulgation of the Law 12305 – Brazilian National Policy on Solid Waste (NPSW, 2010) -, which in its article #9 recommends: "When managing solid waste, the following priority shall be observed: non-generation, reduction, reutilization, recycling, solid waste treatment and finally environmental-adequate waste disposal". Nevertheless, after about 10 years of its promulgation, the new national policy has not yet accomplished the desired changes in municipal waste management, particularly no significant upgrading can be observed in the indicators studied: municipal waste generation, frequency of waste collection, rate of recyclable waste recovered, and proportion of sanitary landfill (Cetrulo et al., 2018; Silva et al., 2021). This situation shows the urgent necessity to apply new strategies of MSW management in the Brazilian context, by following the recommendations from the waste management pyramid.

Figure 5 shows the specific Waste Hierarchy related to food recovery management, named Food Recovery Hierarchy (FRH). Along the hierarchy, from the most to the least preferred scenarios, source reduction is the best option, followed by donation, feeding animals, industrial use (as Biorefinery scenarios), composting, with incineration and landfill as the worst options. Despite the priority order for waste management, few studies (e.g., Beretta and Hellweg, 2019) have assessed the higher priority levels, such as waste prevention, considered a key aspect towards sustainability. As discussed by Van Ewijk and Stagemann (2016), a possible explanation is that waste prevention is the most difficult option to execute among all others in the waste management hierarchy, and that waste collection is the only process that managers can easily control.

As previously shown in section 3.1, food supply chains are constantly generating FLW. Part of this FLW is considered waste due to market logics, which makes the risk of underestimating the real potential of these by-products beyond their economic value. In particular, the fraction recognizable as non-Marketable food (NMF) plays a key role. Rejected by consumers or retailers only due to little

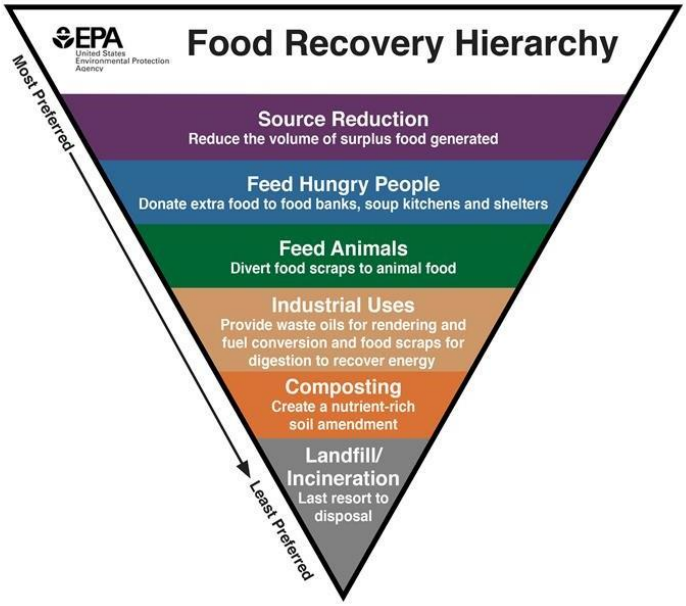

Figure 5: Food Recovery Hierarchy management (source: www.epa.com).

physical imperfections or excessive level of ripeness in relation to retailers selling times, the NMF could show its potential if managed according to the preferable options of the Food Recovery Hierarchy. Food Donation is one of these options, and it could be considered as a waste prevention approach. In fact, according to Cakar (2022), NMF is not food waste, but it has the potential to become waste if not managed suitably. Therefore, recognizing NMF as a nutritional resource to people in need rather than discarding it into landfills would represent a more sustainable option regarding waste prevention (Salhofer et al., 2008).

The relationship between the food that has lost market value and the options proposed by the food recovery hierarchy implies a process of recognition of a potential value (Figure 6). In this present study, the potential value recognition of NMF and its link with the food recovery hierarchy options are explicitly presented and discussed. Focusing on the food importance to man, the process consists in a hierarchical classification of a potential wealth – (i) Nutritional; (ii) Material; (iii) Energetic; (iiii) Not – recognized, which is connected to the different options of the pyramid. The next sections explore the above-mentioned potential values along the different scenarios proposed by the Food Recovery Hierarchy.

3.4.1 Food Donation

Recognizing the potential nutritional value of FLW is the first and most recommended option. This operation could be executed by valorizing the edible fraction of NMF, through the implementation of donation policies. As shown in section 3.1, donation could be particularly suitable in the Brazilian context, where a considerable amount of NMF discarded is associated with a high level of food insecurity. According to Silva et al. (2021), a recent updating of the Brazilian law has defined a better background to promote food donation scenarios. In fact, the new legislation addressing the fight against hunger (LF - 14.016/2020)

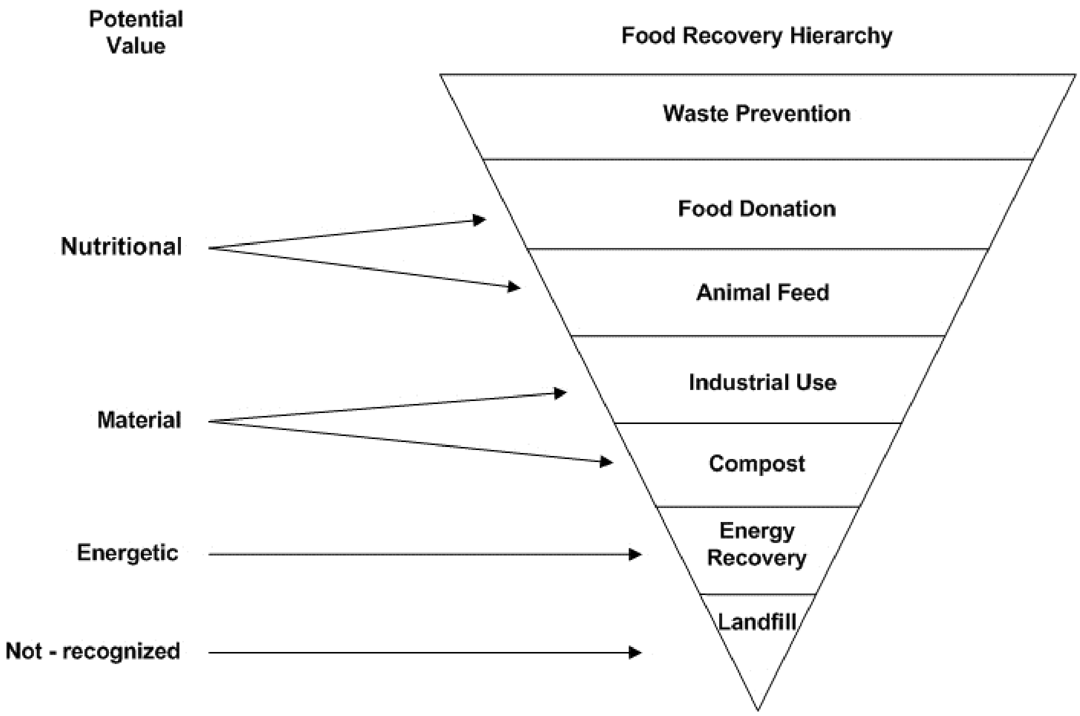

Figure 6: Pyramid of recognized values associated with the food recovery hierarchy.

exempts the donor and the involved intermediaries from any responsibility in case of damages derived from food donation, except in case of an explicit intention to harm. This new legislation removes barriers to donation while ensuring the prevention of food loss and waste, as recommended by the Law 12305 – Brazilian National Policy on Solid Waste (NPSW, 2010).

The most common way to implement these food donation policies is the creation of Food Banks. As discussed by Schneider (2013), since their first appearance in 1960 in the United States, food banks have demonstrated to be a valid option to help people in need. Food banks are defined as "organizations that solicit food and grocery products from a variety of sources, receive and store the products in warehouses and distribute them to impoverished families and individuals through charitable human service agencies".

Scientific literature shows that few works have explored the environmental impacts of FD under a LCA perspective, while the Emergy evaluation of food donation scenarios is still in its infancy, as shown by the following presented works.

Eriksson et al. (2015) compare the outcome, regarding greenhouse gas emissions, of different food waste management scenarios available to supermarkets in Uppsala, a city located in Sweden. Life cycle assessment (LCA) was used to calculate the environmental impact of the impact category global warming potential (GWP). Six waste management scenarios were considered according to the food waste hierarchy (FWH): landfilling, incineration, composting, anaerobic digestion, animal feeding, donations. Five kinds of products were selected for the analysis: bananas, iceberg lettuce, grilled chicken, stewing beef and wheat bread. In the six scenarios investigated, results have shown a decreasing GWP trend from higher to lower priority FRH levels. For all investigated products, landfill was the option with the highest greenhouse gas emissions rate. Donation and anaerobic digestion were the alternatives with the lowest greenhouse gas emissions rate. Donation was the alternative with the lowest emissions for grilled chicken and bread, however, for bananas, lettuce and beef, anaerobic digestion generated the lowest emissions. The other scenarios did not fully agree with the waste hierarchy. Incineration was a good option for bread and grilled chicken, but a poor option for lettuce and bananas, for which composting provided a better alternative. Similarly, anaerobic digestion was a better alternative than animal feeding and, surprisingly, for some products, it was better than donation. The study demonstrated that the properties of individual food products have a great influence in determining which waste management option is most favorable. However, the waste management scenarios studied in supermarkets in Uppsala corresponded, to some extent, with the priority levels in the waste hierarchy.

Eriksson and Spangberg (2017) implemented a comparison regarding greenhouse gas emissions and primary energy use of different food waste management scenarios available in supermarkets in Växjö, a city located in Sweden. Life cycle assessment (LCA) was used to calculate the environmental impact of four different food waste management scenarios (donation, conversion, anaerobic digestion, and incineration) concerning the impact categories GWP and the primary energy use (PEU). Results show the existence of high potential for reducing greenhouse gas emissions and primary energy use by changing the waste management of surplus fruit and vegetables to more favorable options in the waste hierarchy. When food waste was assumed to be managed by a method with higher priority in the waste hierarchy, it was found that it normally generated lower greenhouse gas emissions, compared with less prioritized waste management options. Being the modelling applied in a local context with specific data, general conclusions should be made with caution. However, there were clear similarities between the incineration and anaerobic digestion waste scenarios regarding GWP and PEU, as well in the results for the conversion and donation scenarios, with the last two scenarios showing a considerably better environmental performance.

Moult et al. (2018) focused on greenhouse gas emissions of food waste disposal scenarios for UK retailers. Authors assessed the net GHG emissions of eight different waste disposal options for five core food types using life cycle assessment, accounting for both emissions incurred in transport and processing, and those mitigated by the creation of useful products. Results followed the waste hierarchy priorities: donation of edible food to food banks is the best option, followed by anaerobic digestion, conversion to animal feed, incineration with energy recovery, aerobic composting, landfill with gas collection and utilization, landfill with gas collection and flaring, landfill without gas collection. If waste food from retailers is unfit for human consumption, to minimize greenhouse gas emissions, conversion to animal feed or anaerobic digestion are the best options. For all food types, landfill has demonstrated to be the worst management option.

Albizzati et al. (2019), by using data from twenty French retail outlets that have implemented surplus food redistribution and diversion to animal feed, (I) evaluated the environmental benefits associated with surplus food management as implemented in selected retailers in France, and (II) quantified the associated economic implications for retailers. The study is a cradle-to-grave LCA, encompassing the entire life cycle of the surplus food generated at the retail outlets. This included transport, redistribution of surplus food, reuse of the surplus food as animal feed, and other treatment pathways for the waste. Four scenarios were considered:Scenario I, representing the current management of surplus food (constituted by almost 100% food donation pathways with a negligible percentage recovered as animal feed), Scenario II, where surplus food is sent to anaerobic digestion, Scenario III, where surplus food is sent to incineration, and Scenario IV, representing prevention of surplus food, and used as benchmark for the ideal management. Ten impact categories were accounted for: Global Warming, Terrestrial Acidification, Photochemical Ozone Formation, Particulate Matter, Aquatic Eutrophication Nitrogen, Aquatic Eutrophication Phosphorous, Human Toxicity, Cancer Ecotoxicity, Fossil Resource Depletion and Water Depletion. The results show that all impact categories supported a clear hierarchy: surplus food prevention was the best scenario, followed by current management, which included both redistribution and use-as-feed; the waste management scenarios (aerobic digestion and incineration) were, evidently, the worst.

Brancoli et al. (2020), by using LCA's Recipe 2016 method, assessed the environmental impacts associated with different options for managing bread surplus in Sweden. The goal of the LCA was to compare the following options: source reduction, donation, animal feed production, ethanol production, beer production, anaerobic digestion and incineration. Although the exact amounts sent to each treatment are unknown, the alternatives included in this study are the ones which are already implemented in Sweden or the ones that can come to be implemented. The environmental savings offered by these waste management schemes are also compared to reducing the production of bread by the amount of surplus bread. The relative environmental savings offered by the different waste management options and their comparison with waste prevention are then compared to the waste hierarchy concept. The trend seen by the results in the eighteen impact categories assessed has supported the waste hierarchy: source reduction of bread waste is the preferred option, followed by feed production, donation, beer production and ethanol production. There is no clear preference among these four latter valorization pathways. The worst waste management options, with the exception of four impact categories, are anaerobic digestion and incineration, which are the most common waste management schemes in Sweden. Source reduction has the highest environmental savings in the sixteen impact categories.

Damiani et al. (2021) apply LCA to study environmental burdens and benefits of food redistribution following attributional and consequential LCA approaches. Data on surplus food recovered is collected from local charities in Veneto (Italy) and the impact of their activities is compared with a mixed treatment of food waste through incineration, anaerobic digestion and composting. All midpoint impact categories of ReCiPe (hierarchist) method are considered in life cycle impact assessment of 1 kg of food wasted or donated. The results highlight the great variability of food locally recovered, with respect to quantity and type that influence the outcomes. Food donation reduces the average impact of the studied systems (e.g. 1.9 kg CO_2 eq/kg net environmental benefit for GWP). However, efficient mechanisms of recovery and redistribution are required, in terms of sizing, consumptions and logistics, to ensure a significant environmental improvement over food waste treatment.

Cakar (2022) assessed the redistribution of fresh fruit and vegetable surpluses from Istanbul's supermarkets, in Turkey, compared with three traditional waste management options: landfilling, composting and anaerobic digestion. Climate change, water consumption and energy use were the impact categories assessed, while 1 kg of surplus food was the chosen functional unit. The donated food was assumed able to substitute for the same product from a 1 to 1 product substitution, while the energy and the heat generated by landfill and anaerobic digestion plants were assumed to replace the electricity and heat of the Turkish matrix. Results showed a better environmental performance of food redistribution for all the three impact categories assessed, with one exception related to the better performance of biogas production over food distribution in climate change. This is due to the characteristics of the Turkish energetic matrix constituted mainly by fossil fuels.

Sundin et al. (2022) compared food donation with anaerobic digestion in the city of Uppsala, in Sweden, by including the rebound effect. An attributional LCA was implemented by considering only the GWP impact category. Two scenarios were assessed: (1) food redistribution compared to (2) anaerobic digestion, the latter being the typical organic food waste management treatment in Sweden. Regarding system expansion, donated food was considered to replace the same kind of food while biogas and fertilizers generated by anaerobic digestion were considered to replace Natural Gas as vehicular fuel and mineral fertilizers. Regarding the rebound effect, the potential changes in the purchases of people in need receiving the donations were assessed. In particular, it was accounted for the re-spending of accrued savings due to receiving donated food substituting food that would have otherwise been purchased. This is because the re-spending leads to environmental emissions that are quantified and added to the net carbon footprint results in contrast to emissions corresponding to the substitution that are credited to the net results. The functional unit (FU) chosen was 1 kg of surplus food. The results show that food redistribution is able to save 0.40 kg CO_2 eq /FU while anaerobic digestion does only 0.22 kg CO_2 eq / FU. The rebound effect was equal to 0.50 kg CO_2 eq / FU in the case of food redistribution and negligible (2%) in the case of anaerobic digestion. The authors conclude that results confirm the FRH recommendations but highlight that the rebound effect is able to reduce a considerable fraction of the net carbon savings.

The common conclusions of all these referenced works are that valorization pathways related to the highest levels of the waste hierarchy management (including food waste prevention and food donation) usually generate more environmental benefits than other options, and therefore have to be prioritized. Additionally, all these authors have shown that local characteristics can influence the results, emphasizing that each case study must be assessed in detail, and considered with care.

Almost all the referenced studies have explored the food donation potential for European characteristics and in developed countries, with the exception of Cakar (2022). Besides, all the studies are limited at the retail sector. Furthermore, exclusively three of them (Albizzati et al., 2019; Brancoli et al., 2020; Damiani et al., 2021) have implemented a wide-ranging LCA by considering many impact categories. Notwithstanding, none of those studies has proposed, beyond the diagnostic of different options currently in use, new plausible scenarios modelled to improve the environmental benefits of food donation pathways. These characteristics make evident the lack of knowledge on the potential environmental benefits of FD alternatives in developing countries outside Europe, such as in Brazil, where food insecurity is widespread and food waste is mainly managed in compliance with the lowest levels of waste hierarchy management.

Regarding Emergy synthesis, food donation research is still in its infancy, therefore hardly found in the scientific literature, which claims for additional efforts towards assessing the environmental performance of food donation, compared to the food waste management options with less priority, by considering a donor side point of view. This present study aims to overcome the above-mentioned lacks by modelling and assessing, from LCA and EMA perspectives, the environmental impacts of plausible FD scenarios for the non-marketable food of the wholesale market ‘CEAGESP’, São Paulo City, Brazil.

Conversion to animal feed is the second most recommended option proposed by the FRH to recover the nutritional value. Such option is suggested when the food is no more suitable for human consumption, while still maintaining nutritional and sanitary properties that allow its conversion into animal feed. Few authors (e.g., Salemdeeb et al., 2017; San Martin et al., 2016) have evaluated animal feed scenarios compared with other FRH options under an LCA perspective. Despite the good environmental performance of animal feed scenarios when compared with the lowest levels of the FRH, the presence of high microbiological contamination risks is not negligible, as highlighted by the above mentioned authors, and confirmed by Socas-Rodriguez et al. (2021). For this reason, the European Union has banned the recycling of food waste to animal feed (EC, 2002). The high risk related to healthcare issues increases doubts regarding the safety and the reliability of this recovering pathway, which is the reason why animal feed scenarios are not covered in this present study.

3.4.2 Industrial Use: Waste-based Biorefinery

Industrial use, after donation and conversion to animal feed, is the third option along the FRH, located at the intermediate level. When the nutritional value is lost, no more exploitable or has never been present, this is the most recommended option. It is the first level where the food is no longer considered as NMF or surplus food but could be properly identified as organic waste (Albizzati et al., 2019).

Over the last few decades, the use of biomass as an alternative source of material/energy in industrial systems has received increasing attention. With the growing demand for energy, the rapid increase of greenhouse gas (GHG) emissions, and the depleting of fossil fuels, the role of biomass as a pivotal renewable energy source has emerged to overcome the current and future needs of humankind (Ubando et al., 2020). In this context, the development of the biorefinery concept has received special attention.

Several definitions of biorefinery are available in literature, varying according to the context, the period, and the different perspectives related to their definition. Among them, one of the most exhaustive was performed by the IEA Bioenergy Task 42 (IEA, 2009) that defines biorefining as "'the sustainable processing of biomass into a spectrum of marketable products and energy" (Figure 7).

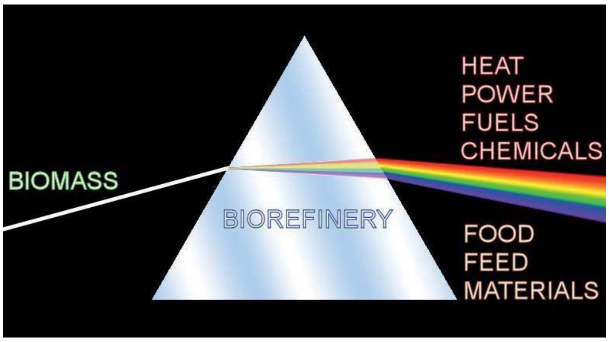

Figure 7: Graphical representation of the concept of Biorefining Processing (adapted from IEA, 2009)

Another widely accepted definition is provided by Cherubini (2010), in which "the biorefinery concept embraces a wide range of technologies able to separate biomass resources (wood, grasses, corn) into their building blocks (carbohydrates, proteins, triglycerides) which can be converted to value added products, biofuels and chemicals. Biorefinery is a facility (or network of facilities) that integrates biomass conversion processes and equipment to produce transportation biofuels, power, and chemicals from biomass. This concept is analogous to today's petroleum refinery, which produces multiple fuels and products from petroleum".

The recent development of the bioeconomy (BE) framework and, later, its union with the circular economy concept to obtain the definition of "circular bioeconomy" (CBE) have created the appropriate background where biorefineries finds their place (Figure 8). BE has been defined by the European Commission (EC, 2018) to indicate the generation of different renewable biological resources and their conversion into various high-value bio-based products such as food, feed, biochemical, and bioenergy. Its main purpose is to mitigate the effects of global warming while supplying a renewable carbon source (biomass) as well as creating business and employment opportunities, especially in the rural areas. The biorefinery concept plays a key role in fulfilling these expectations, as the main actor capable of optimizing the conversion of biomass and to achieve the goals set for the BE concept (Ubando et al., 2020). The CBE adopts the CE framework, utilizing biomass as an integral component to generate various bio-products, biochemicals, and bioenergy in a biorefinery (EC, 2017). According to this BE perspective, Conteratto et al. (2021) have recently updated the traditional biorefinery concept, by assessing more than 30 biorefinery definitions available in literature. After the epistemological analysis of the words "bio" and "refine", and the classification of the several types of biorefineries according to input-based, process-based and product-based concepts (Figure 9), the authors have identified the necessity to update the concept by adding contemporary elements to the terminology. Therefore, taking into account the epistemological

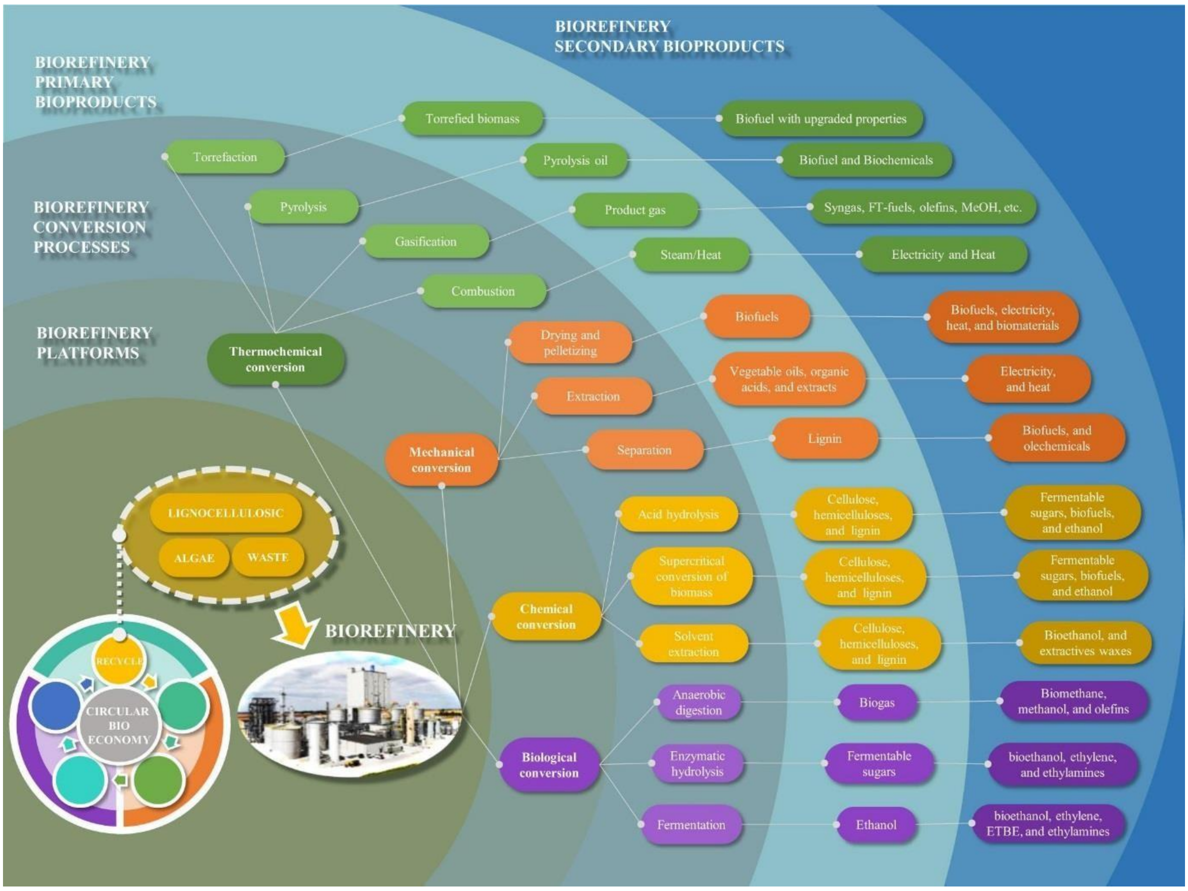

Figure 8: The biorefinery framework on circular bioeconomy (Source: Ubando et al., 2020)

elements, the conceptual basis previously recorded in the literature, and the context of the bioeconomy, the authors define biorefining as: “a physical, chemical, or biological process which purifies, separates, refines, or transforms elements constituting biological assets from the kingdoms Monera, Protista, Plantae, Animalia, or Fungi, originating from the terrestrial or oceanic environment, in bioproducts for final use or that serve as raw material for other bioproducts.” This updated definition is considered as reference in this present study.

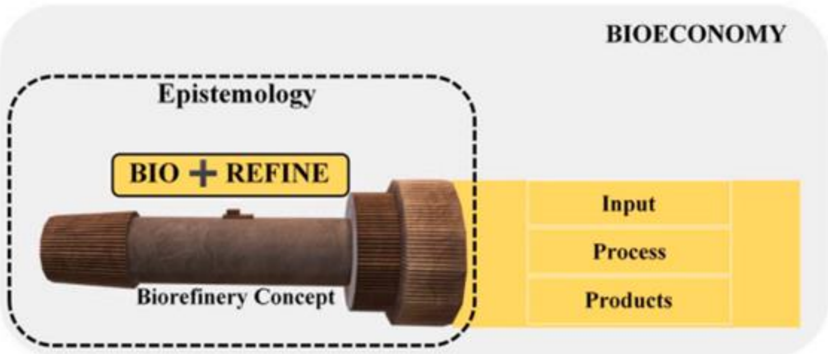

Figure 9: the concept of biorefinery from its morphological decomposition and epistemological analysis (Source: Conteratto et al., 2021)

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The biorefinery is composed of 4 main conversion platforms such as the thermochemical, biological, chemical, and mechanical conversions These allow for the appropriate conversion of various biomass feedstocks into different bioproducts identified as either primary or secondary. The former refers to the raw bioproducts while the latter represents the refined bioproducts (Ubando et al., 2020).

This study is focused on a particular kind of biorefinery, the "waste-based biorefinery" that uses organic waste as feedstock. Several authors have explored waste-based biorefinery scenarios and options, as Alibardi et al. (2020), Caldeira et al. (2020), Dahiya et al. (2018), Sawatdeenarunat et al. (2016), Teigiserova et al. (2019), Tsegaye et al. (2021), Ubando et al. (2020), Yang et al. (2015), among others.

According to Alibardi et al. (2020), "waste biorefineries offer platforms for integrated utilization of a wide range of resources in organic waste". The development and implementation of the waste biorefinery concept offer a range of economic, environmental, social and political benefits: (a) stimulate the engagement of local communities to promote and apply sustainable waste management strategies; (b) provide a profitable alternative solution for waste management in areas with growing urbanization; (c) support the implementation of circular economy principles; (d) reduce the pressure on non-renewable resources; (e) help diversify sources of strategic supply and decrease dependence on imported resources; (f) promote distributed production systems and sustain regional and rural development; (g) contribute to mitigate climate change impacts by providing useful products and off-setting the use of fossil carbon.

As highlighted in (b), biorefineries could become pivotal as a more sustainable solution regarding waste concentration in urban areas, as for example in the case of organic waste generated by food distribution centers as studied in this present work.

The feedstocks used by waste-based biorefineries can be different, including: (1) Organic waste from agricultural residues, basically constituted by lignocellulosic raw materials; (2) Organic waste from industrial residues, basically constituted by not edible residues generated by, for example, orange juice, instant noodles or potato chips production and (3) Organic waste from urban residue, as home scraps and catering waste (Yang et al., 2015). The organic waste generated by food distribution centers, despite showing some similarities with (1) and (2), have peculiar characteristics, such as the fact that it is mainly constituted by edible parts and showing a high grade of diversification.

In the passage from traditional to waste-based biorefinery systems, all the theoretical, technological, economic assumptions and perspectives are not directly applicable. Waste materials range in composition and can contain impurities, such as small plastics that are not easily removable, highlighting the importance of an appropriate pretreatment (Alibardi et al., 2020).

Regarding the strategies, the simplest layouts of a waste-based biorefinery are those aimed at recovering low-added-value products, i.e., biofuels or energy carriers, soil improvers and fertilizers, through a pathway that has sugar fermentation as a key process (Alibardi et al., 2020). Nevertheless, over the last ten years the interest on Biorefineries with anaerobic digestion as core process has been increasing (Sawatdeenarunat et al., 2016).

By considering the most common low added – value products generated by MSW–OF - biorefining routes, several authors have explored bioethanol and/or biomethane production (Ardolino et al., 2018; Chester and Martin, 2009; Ebner et al., 2014; Guo et al., 2021; Kalogo et al., 2007; Moreno et al., 2021; Papadaskalopoulou et al., 2019; Sofokleus et al., 2022; Stichnothe and Azapagic, 2009), by considering single production according to an alternative perspective or joint production according to sequential processes. Higher complexity is required to generate pure streams of chemical platform to produce biomaterials, where more specific technical standards must be met (Alibardi et al., 2020).

When compared with traditional oil refineries, both conventional (ethanol, biodiesel) and advanced biofuels (lignocellulosic methanol, ethanol, butanol) generally cannot be produced in a profitable way at current oil prices, therefore, a significant reduction in production costs is necessary to make biofuels competitive. A new approach towards this target is the implementation of biofuels-driven biorefineries for the coproduction of both value-added products (chemical, materials, feed) and biofuels through an efficient and integrated approach (IEA, 2012).

Several authors have explored the production of high-added value products, standing alone and with the joint generation of bioethanol, energy and other co-products. For example, Scaglia et al. (2020) have assessed a tomato pomace biorefinery for the production of lycopene, bioenergy and digestate. Joglekar et al. (2019) have explored a citrus waste biorefinery with the joint production of Limonene, Bioethanol and Methane. Pathak et al., (2018) developed a biorefinery approach for the valorization of potato peel through the production of animal feed, dietary fibers, antioxidants, phenolic compounds, ethanol and fertilizers. Nevertheless, according to Caldeira et al. (2020), although numerous valorization options exist regarding the recovering of value-added products from food waste, most of them are still at lab-scale and studies analyzing their feasibility at industrial scale are missing. Moreover, as shown by Albizzati et al. (2021), by considering the current level of technologies, the production of high added-value products from food waste is not always convenient, when compared with their market alternative, both from an environmental and economic perspective. This is true especially in the case of biochemicals. Figure 10 shows a layout of a multi-form biorefining producing biofuels and biomolecules.

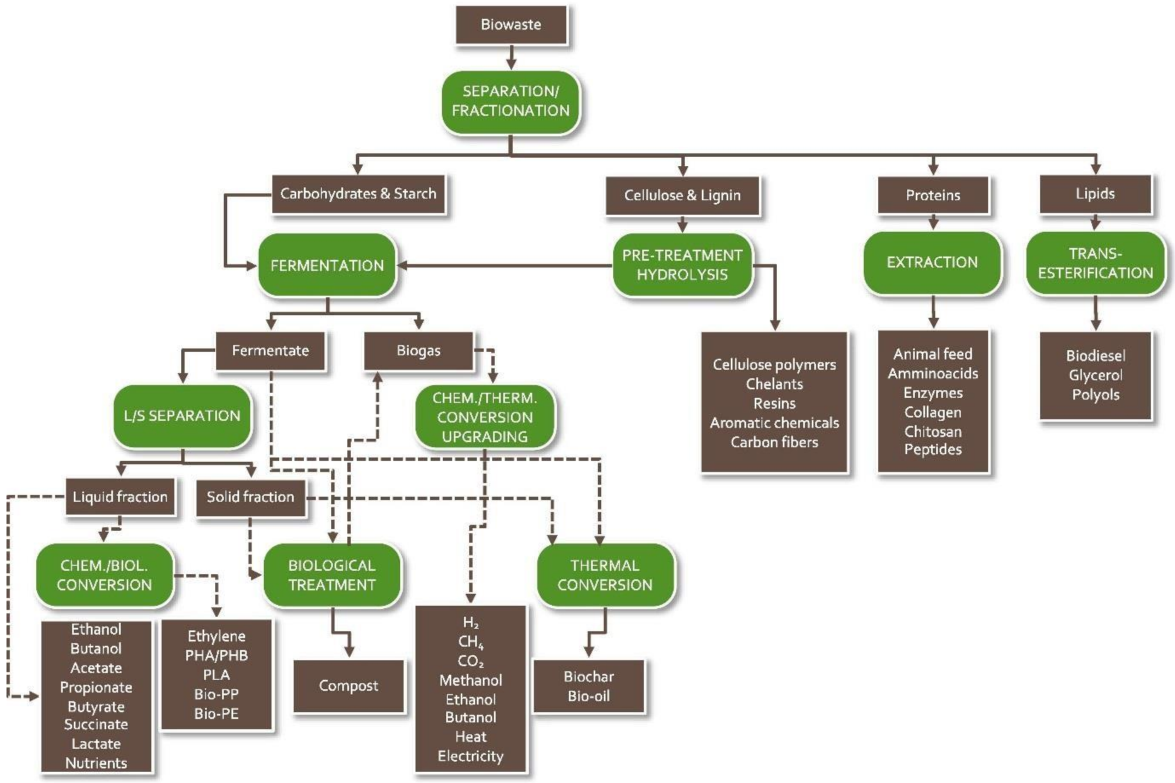

Figure 10: Layout for a multi-platform anaerobic biorefinery producing biofuels and biomolecules. Dashed lines represent alternative options. Green blocks represent processes and brown blocks represent materials (from Alibardi et al., 2020)

In regard to the assessment of biorefineries sustainability, their performance has been previously measured in terms of economic valuation through the net present value and other temporal adjusted methods, and in terms of environmental evaluation through life cycle assessment (Ubando et al., 2020). Two different LCA perspectives may be considered when evaluating the environmental performance of organic waste biorefineries: (I) waste management perspective and (II) output perspective. The former focuses on the comparison of the waste-based biorefinery with other (traditional) waste management options such as composting or landfilling, while the latter evaluates one or more waste biorefinery products with alternative (traditional) production options (Alibardi et al., 2020). This study will focus on the waste management perspective, by comparing the biorefinery scenarios with the other alternatives proposed by the food recovery hierarchy.

As previously shown in section 3.3, the available literature on the evaluation of environmental burdens generated by MSW-OF management is abundant, however, most studies have focused on the least recommended options proposed by the waste management hierarchy. Conversely, few studies (Ardolino et al., 2018; Chester and Martin, 2009; Ebner et al., 2014; Guo et al., 2021; Kalogo et al., 2007; Papadaskalopoulou et al., 2019; Stichnothe and Azapagic, 2009) have considered biowaste treatment for ethanol and or biomethane production from an LCA waste management perspective (Table 2).

Ardolino et al. (2018) study aims to demonstrate the overall environmental sustainability of biomethane production from anaerobic digestion of MSW-OF compared to biogas directly burned in a combined heat and power plant (CHP). Four scenarios are assessed following the LCA procedure: (0) biowaste-to-biomethane base case scenario, where a flow rate of 400 Nm^3/h of raw biogas is sent to the membrane upgrading unit and the CHP unit only partially covers the electricity needs. (1) all the biogas is used to electricity production at CHP plant; (2) all the biogas is converted to biomethane, and all the electricity needs are covered by the Italian matrix; (3) “energetic autonomy” where all the electricity needs to produce biomethane are covered by the internal CHP plant”. The study assessed the environmental impacts by considering 15 different LCA impact categories.

The substituted products are diesel for automotive use replaced by biomethane and Italian matrix electricity replaced by electricity generated at CHP. The results show that GWP, NREP (non-renewable energy potential), RINP (respiratory inorganics potential) and TECP (terrestrial ecotoxicity potential) are the impact categories that play a key role. The total values for each impact category are negative (for GWP and NREP) or about zero, highlighting that the examined biowaste-to-biomethane plant implies a substantial reduction of the overall environmental impacts. Avoided burdens related to the biomethane production and utilization are larger than the direct and indirect burdens. A large part of the avoided impacts derives from the missed production of diesel (“from crude oil to diesel”) and avoided “tank-to-wheels” emissions for its utilization in passenger cars and small rigid trucks. The comparison among the different scenarios showed that plant configurations aimed to biomethane production (scenarios 0, 2, 3) have the best environmental performances. The authors conclude that the use of biogas to Biomethane Production has less environmental impacts than traditional use in CHP plants to produce electricity, and therefore is preferable.

Chester and Martin (2009) have assessed cellulosic ethanol production from MSW in California, United States. The authors have examined the main processes required to support a lignocellulosic MSW - to ethanol biorefinery, through enzymatic hydrolysis, by accounting for cost, energy and from an LCA – greenhouse gases perspective compared to the current scenario where California state imports bioethanol from the Midwest. The analysis exclusively considers MSW destined for landfill, and it was assumed a 75% ethanol yield per ton of dry matter. To estimate avoided emissions from California landfills, average state-specific material emission factors were computed based on the mix of landfill types (31% no recovery, 21% flared, and 48% used to produce electricity). The CO_2 emissions from ethanol combustion is interpreted as follows: because the total system considers emissions that do not occur as the result of avoided landfill decomposition, it is appropriate to consider the additional emissions that result from the combustion of ethanol. The results show that the impact of waste diversion from landfills is significant if a large fraction of the organic material is diverted from landfills that do not control methane emissions. The energy saved from not landfilling the materials is roughly equivalent to the operational energy spent classifying the material for ethanol production. The authors found that the avoided impact of diverting organic waste from the landfill presents the greatest system uncertainty. This uncertainty is linked to the existence and the efficiency of methane capture systems at landfill, varying from GHG positive emissions where methane capture systems are not present to negative GHG emissions where methane capture systems are highly efficient. In conclusion, the authors affirm that ethanol production from MSW cannot be unequivocally justified as an alternative to traditional landfilling from the perspective of net-GHG avoidance.

Ebner et al. (2014) have implemented a life cycle greenhouse gases assessment of a novel process for converting food waste into ethanol and co-products. This new process uses the combination of an input of a “sugary diluent” and food scraps. In fact, fruit juice and cannery industrial waste have been reported as potential biofuel feedstocks. Food scraps, which are generally more complex lignocellulosic materials, also have the potential for conversion into ethanol. The study analyzes a pilot fermentation plant where lignocellulosic food scraps are combined with a sugar rich diluent. The food scraps are ground without any other pretreatment and simultaneously co-fermented with diluent, at ambient temperature. The process produces ethanol as well as compost and animal feed co-products. The functional unit is 1 L of ethanol that is converted to a unit of transport energy (1 MJ) for comparison to conventional gasoline. The results are compared to those of corn ethanol and traditional gasoline production as well as to traditional landfilling process and compost production. The bio-refinery consists of two phases: fermentation and dehydration that occur in two different facilities, with intermediate outputs transported from phase 1 plant to phase 2 plant by diesel trucks. The greenhouse gases considered are carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O), and the emissions of the replaced co-products (compost and animal feed) production processes were considered as avoided emissions. The results of GHG impacts comparison among the waste-based biorefinery, traditional bioethanol production from corn and gasoline production have shown a net carbon-negative production process with 553% improvement to corn ethanol and 460% relative to conventional gasoline. This reduction is almost entirely due to the avoided methane emissions that would be incurred by food waste disposal in a landfill. Without the inclusion of avoided landfill impacts, the net bio-refinery emissions (phase 1 and phase 2) show a 9% improvement over commercial corn ethanol production (including agricultural phase impacts). Authors conclude that the use of readily convertible, source-separated commercial or industrial food waste as a feedstock for ethanol offers significant potential for GHG reduction when compared to traditional ethanol production, especially when avoided emissions at landfill are included.

Guo et al. (2021) aim to quantify the GWP of alternative biorefinery technologies including consumptions and emissions during the biorefining process as well as the substitution of fossil fuels by the produced biofuels. The study considers typical Chinese food waste as input, that show around 80% of moisture and a dry fraction composed by about 51%, 22%,16% and 11% of Carbohydrates, Lipids, Proteins and Ashes respectively, while 1000 kg of FW is the functional unit.

Five alternative biorefineries and a reference scenario are assessed:

- S0 - Anaerobic digestion: This is the reference scenario. No upgrading is considered, and the process produces biogas used for electricity generation and fertilizers.
- S1 - Biomethane: This scenario is an extension of S0 with upgrading of the biogas to pure methane (greater than 97% methane). Water scrubbing was chosen as upgrading technology. The authors assume that removed carbon dioxide is not utilized and has no GWP due to its biological origin.
- S2 - Bioethanol: The carbohydrates in the FW are by means of enzymes saccharified and afterwards fermented to bioethanol which is removed by distillation. The residue is anaerobically digested as in S0.
- S3 - Biodiesel: the FW is pre-treated to separate the lipids from the mixed residues. The lipids are separated by centrifugation of the FW and converted by transesterification to biodiesel and some side products. The non-lipid residue is anaerobically digested as in S0.
- S4 - Biodiesel and bioethanol: The lipids are converted into biodiesel like in S3 and the carbohydrates are used for bioethanol production as in S2. The residual flow is anaerobically digested as in S0.
- S5 - Biodiesel and biomethane: The FW is pre-treated to separate the lipids from the mixed residues. The lipids are converted into biodiesel like in S3, and the non-lipid residue is anaerobically digested, and the biogas upgraded to biomethane as in S1.

All scenarios include anaerobic digestion of the liquid residue stream followed by composting of the solid residue stream, the latter after addition of wood chip. GWP was calculated on emissions and savings in terms of fossil carbon dioxide (CO_2), methane (CH_4) and dinitrogen monoxide (N_2O). CO_2 of biogenic origin is considered neutral with respect to GWP.

The results show a saving of 75 kg CO_2eq per t FW for scenario S0, mainly derived from electricity replacement. Compared to the reference scenario with anaerobic digestion only, upgrading the biogas into biomethane increases the CO_2 savings by 37% (S1: 103 kg CO_2/1000 kg FW), and introducing biodiesel prior to the anaerobic digestion can improve the savings by around 60% (S3: 120 kg CO_2/1000 kg FW). Combining biodiesel and biomethane can obtain even better improvements by around 84% compared to the reference scenario (S5: 138 kg CO_2/1000 kg FW). Introducing bioethanol has no GWP benefits with the current technological performance, since extracting the sugars for bioethanol production consumes energy and also reduces the subsequent biogas or biomethane production.

The authors conclude that modelling biorefining scenarios by introducing biogas upgrading to biomethane and biodiesel production before the anaerobic digestion significantly increases the CO_2 saving, while bioethanol production, alone or joint with biodiesel production does not show GWP benefits.

Kalogo et al. (2007) have modelled a MSW – to ethanol facility and implemented the following comparisons from a Life Cycle Energy use and air emissions perspective: (I) Evaluation of the environmental burdens associated with using MSW-ethanol as a Light Duty Vehicles (LDV) fuel; (II) MSW-ethanol-fueled LDV comparison with LDV fueled with gasoline or with ethanol produced from corn or cellulosic biomass from energy crops; (III) MSW-ethanol production with landfilling. The energy use, the GHG and the air pollutants (AP) emissions were quantified. The GHG considered are CH_4, N_2O, and CO_2 and these are weighted by their 100-year global warming potentials in calculating CO_2 equivalents. The AP considered are volatile organic compounds (VOC), carbon monoxide (CO), nitrogen oxides (NO_X), particulate matter less than 10 micrometers in size (PM_{10}), and sulfur oxides (SO_X). The results show that MSW to ethanol pathway show better environmental performance both when compared to corn-ethanol production and gasoline and when compared with landfilling. In the last case, MSW to ethanol depicts a better performance when compared with landfilling without biogas recovering, while more uncertainties were found when biogas capture was considered.

The authors conclude that the net life cycle energy used in producing MSW-ethanol is less than the energy used for producing corn-ethanol or cellulosic biomass-ethanol. In terms of global warming effects, MSW-ethanol performs better than corn-ethanol and gasoline. Similarly, converting MSW into ethanol instead of landfilling will result in significant fossil energy savings, and less GHG emissions when LFG are not captured and recovered.

Papadaskalopoulou et al. (2019) assessed, from an LCA perspective, a waste-to-ethanol biorefinery system versus conventional waste management methods in the Attica Region, Greece. The examined conventional methods include: (I) landfilling with energy recovery (current method applied for mixed municipal waste in the study area); (II) windrow composting (current method applied for biowaste in the study area); (III) anaerobic digestion; (IV) incineration. For each scenario avoided emissions were calculated through a system expansion that includes the conventional production of secondary products; in this way, the reference system is credited with the avoided emissions due to the replacement of the respective conventional products. The boundaries of the biorefinery system also include the substitution processes of conventional products by the system bioproducts. The functional unit is 1 ton of municipal wet biowaste. The examined systems were assessed against fourteen impact categories included in the ILCD 2013 LCA method, and a final sensitivity analysis was implemented. The results show that biorefinery scenario presents very good environmental performance as the net emissions to the environment are quite low for all the impact categories examined, while in many cases the emissions are negative, meaning that the examined system creates a net benefit for the environment. The highest net benefits are recorded for the impact categories "Freshwater eutrophication" and "Human toxicity-carcinogenic" while the highest net burdens are recorded for the categories "Ecotoxicity" and "Marine eutrophication". These higher burdens in the last two impact categories derive from the substitution of mineral fertilizer by the produced digestate. The total net emissions for the "Climate change" category are estimated to be -15 kg CO_2eq per ton of biowaste. Regarding the comparison between biorefinery and the conventional waste management methods, biorefinery presents a better performance in most of the impact categories assessed, while composting shows the worst performance, followed by anaerobic digestion and landfilling.

The authors conclude that the biorefinery system presents, when compared with other traditional management systems, a particularly good environmental performance, because net emissions to the environment are quite low for all the impact categories examined, while in many of the cases the emissions are negative.

Stichnothe and Azapagic, (2009) assessed two alternative feedstocks for bioethanol production, both derived from household waste: (I) Refuse Derived Fuel (RDF) and (II) Biodegradable Municipal Waste (BMW), in the United Kingdom. LCA was performed to estimate the GHG emissions from bioethanol using these two feedstocks, and data were compared with the current municipal solid waste management in UK. An integrated waste management system was considered, taking into account recycling of materials and production of bioethanol in a combined gasification/bio-catalytic process. The functional unit defined was the 'total amount of waste treated in the integrated waste management system'. The results showed that, among the assessed scenarios, the best option is to produce bioethanol from RDF— which can save up to 196 kg CO_2 equiv. per ton of MSW, compared to the current waste management practice in the UK. The authors conclude that, despite the good results obtained by bioethanol production in GWP, the overall environmental sustainability of bioethanol from waste cannot be assessed without investigating other environmental and socio-economic impacts. Furthermore, the production of ethanol from waste might compete with other recycling or material recovery options that should also be analyzed by using a full life cycle approach.

Table 2: overview of some biorefinery scenarios assessed from LCA perspective.

Biorefinery Type	Impact Categories / Pollutants/ Sustainability Index	Method	References
Biomethane from MSW – OF	Carcinogenic, Non – carcinogenic; Respiratory Inorganics, Ionizing Radiation, Ozone layer depletion, Respiratory organics, Aquatic ecotoxicity, Terrestrial ecotoxicity, Terrestrial aci / nutri, Land occupation, Aquatic acidification, Aquatic eutrophication, Global warming, Non – renewable energy, Mineral extraction	LCA	Ardolino et al. (2018)
Ethanol from MSW	Climate change, economic and energy impact	GHG LCA, economic and energy assessment	Chester and Martin, (2009)
Ethanol from Food Scraps and Sugary diluent	Climate Change	GHG LCA	Ebner et al. (2014)
Five Biorefineries options (Biomethane, Bioethanol, Biodiesel and combined) from FW	Climate Change	GHG LCA	Guo et al. (2021)
Ethanol from MSW - OF	Climate Change; Air Pollutants (CO, NO_X, PM_{10}, SO_X) Energy use	GHG LCA, pollutants and energy assessment	Kalogo et al (2007)
Ethanol from MSW	Climate Change, Terrestrial Acidification, Terrestrial eutrophication, Photochemical Oxidant Formation, Non-carcinogenic Human Toxicity, Ionizing Radiation, Freshwater Eutrophication, Marine Eutrophication, Ecotoxicity total, Abiotic resources depletion fossil, Particular matter	Life Cycle assessment	Papadaskalopoulou et al., (2019)
Ethanol from MSW-OF	Climate Change	GHG LCA	Stichnothe and Azapagic, (2009)

Regarding emergy synthesis of biorefinery scenarios, the most relevant identified studies to the purposes of the present dissertation are the works of Baral et al. (2016), Patrizi et al. (2016) and Santagata et al. (2019).

Baral et al. (2016) assess and compare the sustainability and environmental impacts of fast pyrolysis and direct combustion systems of lignin utilization. The identified problem is related to biofuels production waste, in particular lignin and other cellulosic waste that cannot be converted in biofuels through normal biorefinery processes. Both solid and liquid wastes of cellulosic biorefineries are collectively known as stillage. Thus, a low cost and low energy stillage recovery method is essential for economic and sustainable biofuel production in the future. Currently, the economic use for lignin is direct combustion to produce process steam and electricity, however, bio-oil and bio-char production are largely used as well. The authors evaluate fast pyrolysis and bio-oil and bio-char production scenarios by using the most common emergy indicators. Results show that fast pyrolysis demands more emergy than direct combustion, and that recovering electricity and products through this kind of biorefinery is not convenient, as the required emergy is too high when compared with standard production systems. This is because, as confirmed by the yield, external energy is required to transform the low-grade energy available in the stillage into high grade or more concentrated energy.

Patrizi et al. (2016) evaluated the sustainability of bioethanol production in a biorefinery fed by straw from agriculture and residual heat from geothermal electricity production; the output is calibrated to replace 10% of gasoline production within the province of Siena (Italy). The system is fed by local residual inputs (geothermal heat and residual straw from crop production). An annual input of 38,000 tons of straw was used to produce 8,200 tons of bioethanol to replace 5,000 tons of traditional gasoline. Two scenarios were considered: a biorefinery located in Siena province able to take advantage of residual geothermal heat, and a biorefinery located elsewhere in Italy fed by the Italian electricity grid, which is based on Natural Gas. As for the system boundaries, the emergy investment represents the emergy required to collect, transport the residual straw, and convert it into bioethanol, by means of a biorefinery fed by residual geothermal energy. Embodied straw emergy was not included. The results related to the biorefinery fed by geothermal heat have shown a total emergy investment U of 7.52E+18 seJ/yr, with industrial phase and straw collection showing the highest emergy cost: 45% and 53% respectively. In the case of Siena province, the emergy benefit derived from replacing 5,000 t of gasoline (1.48 E+19 seJ/yr) doubles the total emergy investment. In the case of the same biorefinery elsewhere in Italy, the recovered emergy is balanced with the emergy saved.

Santagata et al. (2019) explored the environmental performances of the production of animal meal and fat from slaughterhouse waste, and of the subsequent production of electric energy from processed animal fat. The process, consisting of a rendering phase and an electricity generation phase, was analyzed under different emergy algebra perspectives, as the allocation according to splits and co-products features, in order to understand how assumptions on output flows affect the results. The work evaluates different possible approaches of EMA on the electric energy cogeneration plant: (1): Split with economic allocation: the driving emergy is allocated according to the economic value of the output flows. In the case of a slaughterhouse process, only the main products (i.e. meat and leather) have market value, while by-products are generally considered having zero economic value and are disposed of as waste. (2): co-products. Animal by-products and meat flows are considered as co-products of the slaughtering process (meat cannot be obtained without producing by-products), therefore, the total emergy of the process is assigned to both of them. (3): split with exergy-based allocation to the byproduct. The results show that case (1) depicts the best performance followed by case (3) and case (2). In particular, case (1) assumption is equivalent to considering the investigated process simply as a waste disposal process, with a “zero burden” approach: the electricity generated is comparable with the Italian electricity mix generated for the greater part using natural gas as well as with the electricity from the reference oil fired power plant. Generally speaking, considering electricity production jointly with the possible market opportunities of the other products (animal fat and animal meal), this biorefinery scenario shows an interesting environmental performance.

From all these referenced studies regarding waste-based biorefinery vs traditional waste management techniques, a general conclusion is that waste-based biorefineries have better LCA and Emergy performance than traditional waste management options, although uncertainties are still present, and such general conclusions should be considered with care. In fact, under the LCA lens, many studies have evaluated only GWP, highlighting the necessity to assess the environmental performance in other impact categories, while from an emergy perspective, biorefining processes related to FLW are a scarcely explored area. These gaps in literature were also recently confirmed by Jones at al. (2022) in their review study, arguing that “the relative environmental impacts of most biorefining processes are unknown, compared to more established FLW management activities. Therefore, it is uncertain how biorefining processes should be prioritized within FLW management frameworks.”

According to the literature review carried on in this work, it is possible to recognize a general asymmetry in the scientific exploration and validation of the different options proposed by the food recovery hierarchy. The lowest levels have been explored in a very detailed way and over the five continents, while the highest levels (prevention and donation) are hardly explored, especially outside Europe, at wholesale level and from an emergy perspective. The intermediate level, called industrial use, is well explored only from an output perspective, while from a waste management perspective, it is still unclear how biorefining processes should be prioritized within FLW management frameworks. Furthermore, considering that only the work of Brancoli et al. (2020) includes donation and biorefinery options, limitedly to LCA and assessing bread – a very specific food with low moisture -, a lack of studies where different options proposed by the FRH (by including the highest to the lowest ones) are jointly assessed from a holistic and multidimensional point of view is recognizable.

This present thesis contributes to overcome these scientific gaps by assessing the environmental sustainability of donation, biorefinery, energy recovery, and landfilling scenarios of FLW generated by a wholesale market in Brazil, from the holistic and complementary approach provided by the joint use of LCA and emergy synthesis.

4. METHODS

4.1. Nomenclature and Definitions

Literature review shows that terms such as food loss, food waste, surplus food, unsold food, and non-marketable food are used according to different authors' interpretations, and sometimes the term food loss and/or waste is used as a reference to the still edible fraction. In order to avoid misinterpretations and recognizing that food donation is a waste prevention approach, this section proposes a new framework of definitions to clarify the relationships among food, food waste, and non-marketable food along the food supply chain.

Food supply chain is defined as the movement of products and services along the value-added chain of food commodities that aims to achieve higher value for the customers alongside cost minimization. It can be divided into five steps, which include (i) farm production, (ii) handling and storage, (iii) processing, (iv) distribution and (v) consumption (Porter et al., 2016). Each step generates losses that could be defined as "by-product" of the production and distribution system. In fact, according to Brown (2015), a by-product is "an incidental or secondary product produced in a process in addition to the principal product. Generally, it is not valuated as high as the product". A by-product could be considered useful and, sometimes, also marketable or it could be considered as waste. For example, the plastic used in plastic shopping bags started as a by-product of oil refining (Muthu and Li, 2016).

By-products generation regards all steps of the food supply chain, with differences in kind and quantity, according to the existing specific processes. It consists of organic and inorganic materials derived from the processing of issues and trading agreements. For example, bran is a by-product of the milling of wheat into refined flour, orange skins are the by-products of orange juice production while in a wholesale market, the food rejected by retailers, and therefore unsold, could be considered a by-product of the trading process. Figure 11 shows the last case in detail, considering, for example, a food distribution center of fresh and perishable products. The wholesalers arrive to the market with a certain amount of food products. In the normal trading operations, when correctly concluded, the food is transferred from wholesalers to retailers, with the generation of by-products derived from wholesalers' packaging made of a non-edible organic fraction (for example, the straw used to protect watermelons during transport or broken wooden crates) and an inorganic fraction (the plastics used to pack products or broken plastic crates). Both are considered organic and inorganic waste, respectively.

As previously discussed, due to rigorous quality standards concerning weight, size, and shape, a fraction of the food is unattractive for consumers, or it could be too ripe for the sales timing of

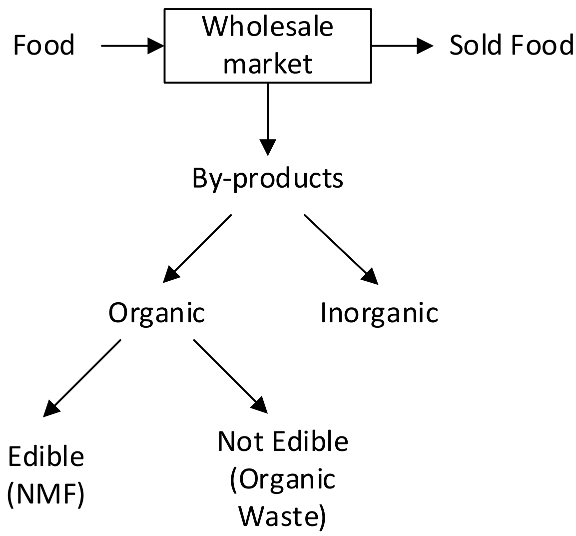

Figure 11: Flowchart of by-product generation at a wholesale market. NMF = non-marketable food.

buyers at retail level, therefore, it remains unsold, although still edible, and with high nutritional value. This unsold food could be considered as an organic by-product of the trading operations. In this present study (Figure 11), that edible fraction of organic by-products (OBP) is referred to as “non-marketable food” (NMF), while the fraction of food that is not suitable for human consumption is classified as food waste and, jointly with the other organic fractions, classifiable as organic waste. Figures 12 and 13 show some examples of marketable and non-marketable food, while figure 14 shows the current NMF management at CEAGESP.

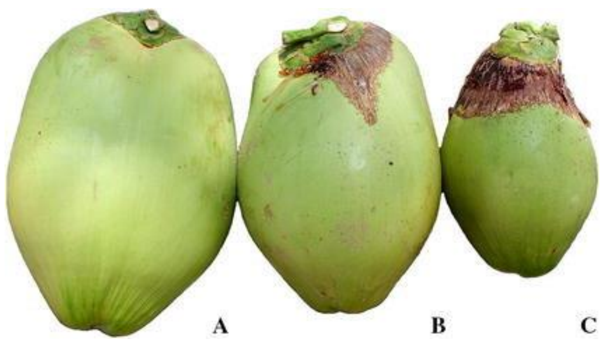

Figure 12: Products classification: A - marketable product; B - intermediate, non – marketable for the most exigent consumers; C - non-marketable for all consumers. Adapted from Redenze et al. (2016).

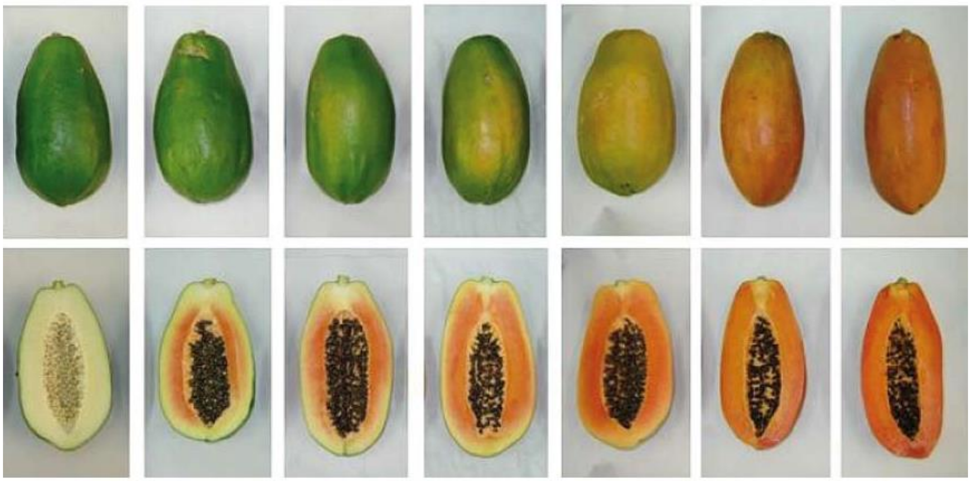

Figure 13: NMF – level of ripeness - the papayas that are too ripe (the two last photos on the right) are normally rejected by retailers due to insufficient shelf life. Adapted from Basulto et al. (2009).

Figure 14: non-marketable food discarded at CEAGESP. Source Uratani et al. (2014)

4.2. Case Study Description

This study focuses on the OBP generated in food distribution centers (FDC), companies that provide an efficient circulation of products in highly populated cities, allowing for the products transfer between croplands and urban centers. They are concentrated wholesale markets for products, usually horticultural, where sellers and buyers directly perform market agreements.

The 'Companhia de Entrepostos e Armazéns Gerais de São Paulo' (CEAGESP) is a federal public company, in the form of a corporation, linked to the Ministry of Economy, and is an important link in the supply chain of vegetables. It allows agricultural production from several Brazilian states and other countries to reach the tables of people, with regularity and quality.

The Company guarantees the necessary infrastructure for wholesalers, retailers, rural producers, cooperatives, importers, exporters, and agro-industries to develop their activities.

CEAGESP maintains the largest public network of warehouses, silos, and bulk carriers in the State of São Paulo, with 18 active units distributed throughout the state. It also has a network of warehouses, with 13 active units, also distributed throughout the State of São Paulo, including the largest central supply of fruits, vegetables, flowers, fish and miscellaneous items (garlic, potatoes, onion, dry coconut and eggs) in Latin America - Entreposto Terminal São Paulo (ETSP). Located on the west side of São Paulo capital, around 50 thousand people and 12 thousand vehicles circulate within its premises daily. Due to the high concentration of organic by-products generated daily by CEAGESP – ETSP, it was selected as a case study (Figure 15).

The food distribution center (FDC) of São Paulo is the largest one in Latin America and the third one in the world, after New York and Paris. The main steps of CEAGESP internal organization include a preliminary weighing and checking of new products upon arrival, the trading phase, the output of sold products, and the management of by-products (organic and inorganic), a small fraction of which is checked and recovered through a Food Bank and a recycling system, while most of them is not checked, treated as waste, and sent to landfill.

From 2007 to 2018, CEAGESP traded more than 3 million tons of products yearly; among them, horticultural products have played a key role. The most traded products were oranges (11.5%), tomatoes (9%), potatoes (7%), papayas (4.5%) and apples (4%). The quantity of products commercialized has varied (Figure 16) between about 3,033,000 tons in 2007 to 3,412,000 tons in 2014 with an average value of 3,200,000 tons/yr, with a waste generation rate ranging from 39,500 tons in 2007 to 60,200 tons in 2014, with an average yearly value of 52,300 tons. Comparing the waste production with the total volume commercialized, an annual waste generation in percentage between 1.30% in 2007 to 1.79% in 2015, with an average value of 1.61% (CEAGESP REPORTS from 2008 to 2019, for further insights see Table A1 in Appendix A) was found. According to information provided during technical staff interviews, the organic fraction (OF) of all by-products sent to landfill is about 80%, and the increase in percentage of waste generation along the last few years was caused by more effective controls on products quality (Figure 17).

Analyzing the waste management, from 2007 to 2018, about 77% of the waste generated were sent to landfill, while 23% were recycled. The recycled fraction was composed by straw, wood,paperboard, sent to appropriate recycling facilities, while part of the organic fraction derived from food waste was sent to a composting plant. Despite the good intentions, due to technical and organizational problems, the recycling system did not work properly along the years, conversely, the amount of recycled waste declined over the last few years reaching its minimum value in 2018, with

Figure 15: top view representation of CEAGESP. Black rectangles represent the locations for the different traded food products evaluated in this study. Source: adapted from http://www.ceagesp.gov.br/entrepostos/etsp/localize-se/.

Figure 16: comparison of Volume Traded and Waste Generated (in ton/yr) at CEAGESP between 2007 and 2018

9% (Figure 18). Focusing on the organic fraction derived from food, the performance was even worse. After a maximum value of about 16,000 tons of organic fraction sent to a compost plant in 2009, the amount of organic waste recycled in 2018 was only 19 tons. Therefore, in 2018, around 100% of the organic by-products were sent to landfill.

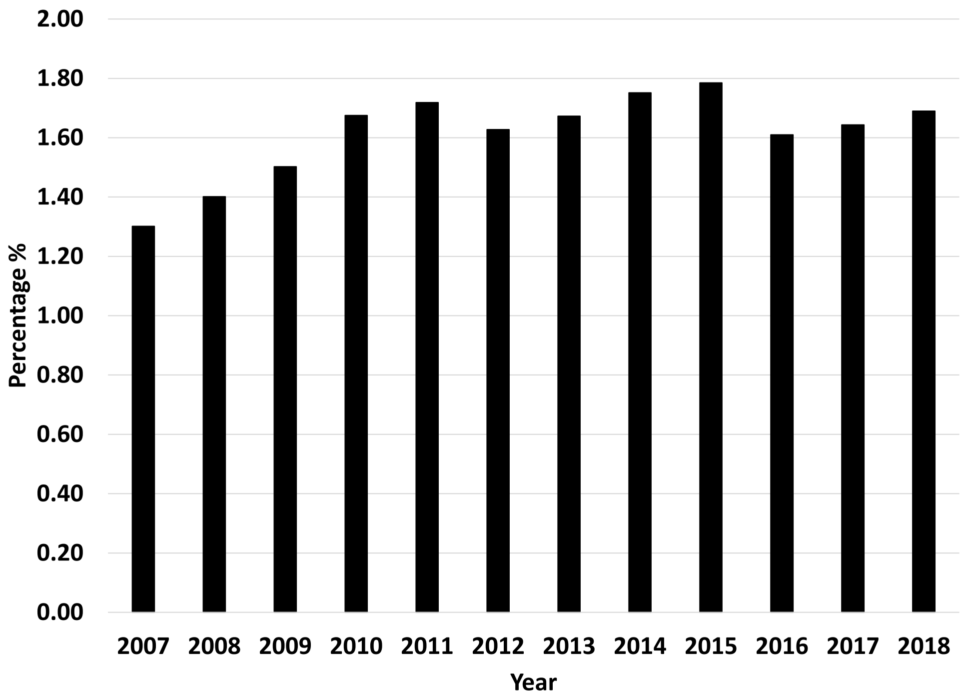

Figure 17: Percentage (%) of waste generated on volume traded at CEAGESP between 2007 and 2018

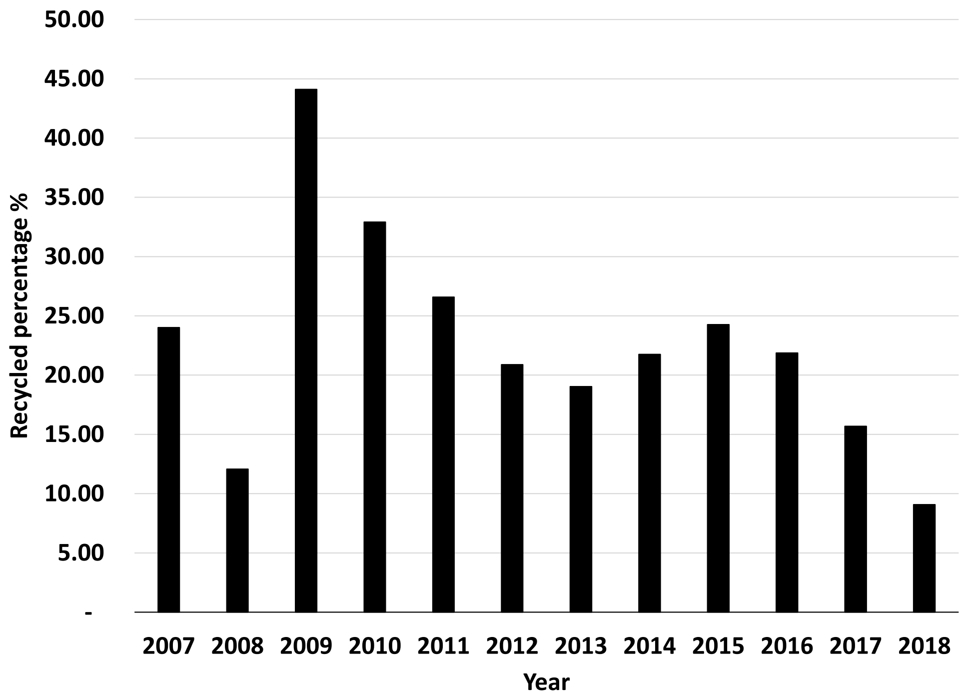

Figure 18: Percentage of Recycled Waste in CEAGESP from 2007 to 2018

Figure 19 shows the detailed flowchart related to food and by-products management at CEAGESP in 2018. The products input was equal to 3,063,098 tons/yr, while the output (sold food) was equal to 3,011,332 tons/yr. This process generated 51,766 tons of by-products, of which 4,701 tons/yr were recycled. The recycled fraction was composed by the NMF recovered in the Food Bank (905 tons/yr), non-edible by products such as straw, paper, plastic and wood (3777 ton/yr) and food waste sent to a composting plant (19 tons/yr). The unrecycled fraction corresponded to 47,065 tons/ yr (including the 135 tons discarded by the Food Bank) and it was directly sent to landfill, without any checking regarding a potential recovery. By considering 51,766 ton/yr as 100%, in 2018 ~9% of by-products were recovered, while ~91% were sent to landfill.

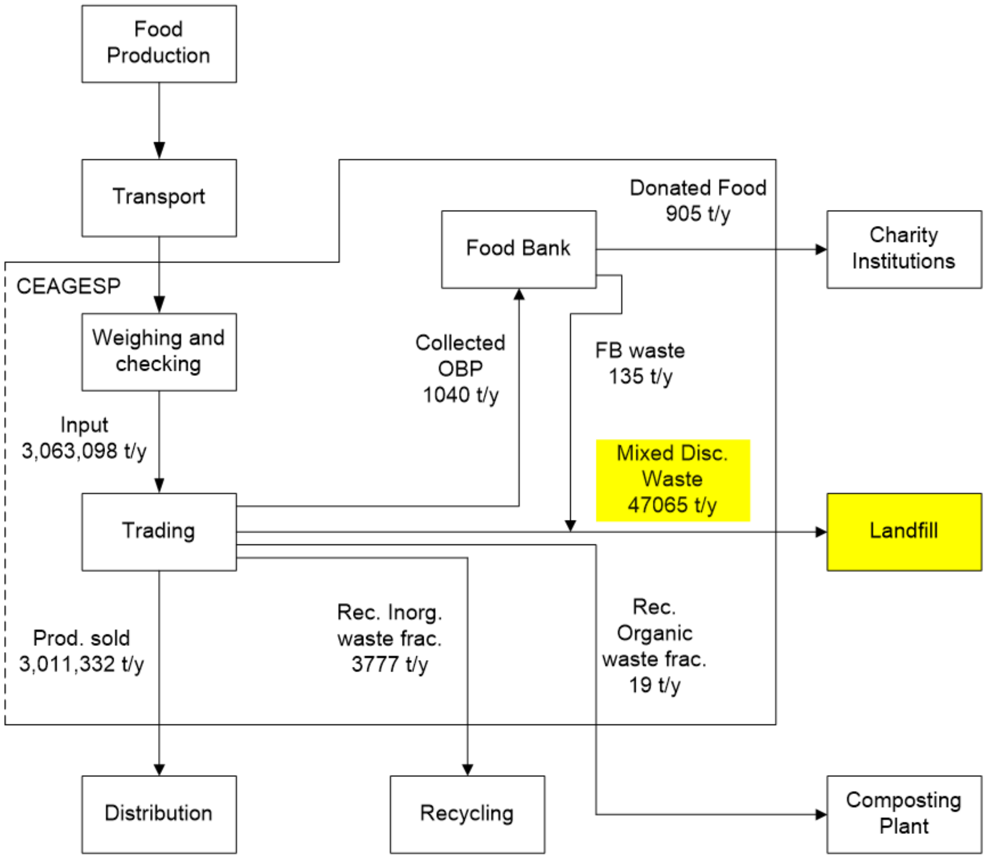

Figure 19: Flowchart depicting the food and the food waste flows at CEAGESP in 2018

4.3. Establishing scenarios for evaluation

The most recently updated data available are used to perform the inventory step, by considering 2018 (CEAGESP report, 2019) as the reference year. Fieldwork was also performed in CEAGESP and in the landfill where the organic by-products are discharged. In 2018, CEAGESP generated about 51,766 tons of by-products, of which 47,065 tons (~91%) were treated as waste and landfilled. According to information provided by staff interviews, the organic by-products fraction constituted by potentially edible unsold food correspond to 80% of the discarded material, therefore an amount of about 37,652 tons of potential valuable organic by-products sent to landfill in 2018 was assumed as a reference in this study.

Besides the diagnostic of the environmental impacts derived from the current OBP management at CEAGESP, several scenarios based on two of the most recommended options proposed by the food recovery hierarchy (donation and waste-based biorefinery) are modelled.

The first option suggested by the food recovery hierarchy is to assess if these OBP still have a potential nutritional value, and therefore can be considered NMF and donated to people in need.

According to information obtained during fieldwork, and consistent with Fagundes et al. (2014), the current CEAGESP food donation management system is ineffective, mainly due to (i) low participation (~15%) of the wholesalers, (ii) high inefficiency of potential OBP's collection system, which is executed on a voluntary basis with manual trolleys and without specific collecting points, (iii) the large distance between the food bank location to the wholesale areas, which demands extra costs for OBP transportation, (iv) and the claimed lack of time by the wholesalers that must return to their agricultural farms as soon as possible. In an attempt to solve these issues, food donation (FD) scenarios are modelled by considering the present scenario (landfilling 100% of OBP) as a baseline. The limits or system boundaries related to the current food donation system, as well as the main environmental burdens of the present OBP management, are considered in modelling the potentially more effective and sustainable FD scenarios. To maximize OBP collection, the most recently updated technologies available in logistic are considered.

If the OBP have no potential nutritional value for humans, and excluding their use as animal feed, the most recommended option by the FRH is industrial use. This option is explored by assessing the environmental performance of a plausible waste-based biorefinery scenario. The biorefinery scenario is modelled by taking into account the following criteria: literature recommendations (type), scale, biomass characteristics, feasibility, circularity and energy self-sufficiency.

Eight different scenarios (Figure 20) are modelled according to the concept of food recovery hierarchy. In scenarios #I (landfilling) and #II (electricity production), 100% of OBP are carried to landfill, however, while the former is a traditional landfill that does not feature energy recovery, the latter captures 40% biogas generated for electricity production, which replaces marginal Brazilian electricity. Scenarios #III (donation 80 + landfilling 20) and #IV (donation 80 + electricity 20) comprehend donation scenarios in which 80% of OBP are diverted to donation and the residual 20% still go to landfill, respectively without and with electricity production and replacement. Scenarios #V (avoided production 80 + landfilling 20) and #VI (avoided production 80 + electricity 20) are similar to scenarios #III and #IV respectively, however, the avoided emissions and the resources savings related to the consumption of donated food are included here; in other words, all the related emissions and natural resources use of conventional food production are avoided once they are being replaced by the donated food. Finally, Scenario #VII (biorefinery) and #VIII (biorefinery + avoided production) represent a biorefinery scenario where 100% of OBP are used as feedstock for a waste-based biorefinery. The former considers the environmental impacts related to the facility while the latter takes into account the potential environmental benefits derived from conventional production replacement.

In the scenarios where a system expansion is considered (#II, #IV, #V, #VI and #VIII), the avoided production derived from products substitution is included. As previously shown, LCA and emergy synthesis have a different and complementary perspective in assessing the environmental impacts, while avoided production generates avoided emissions and natural resources savings. Therefore, in all scenarios where avoided production is considered, avoided emissions are calculated through an LCA point of view, while emergy synthesis is considered when accounting for natural resources savings, according to a complementary and parallel perspective.

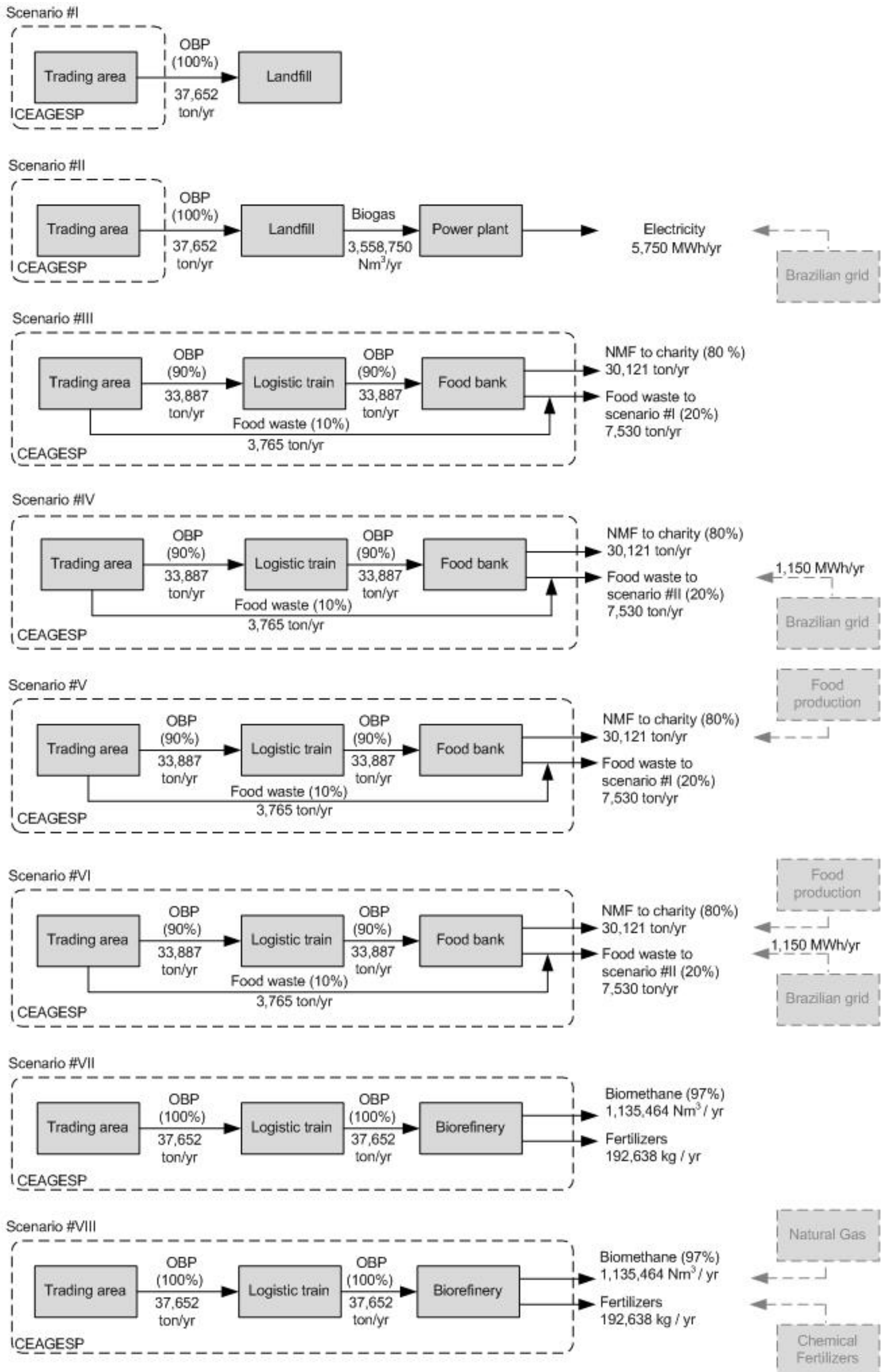

Figure 20: the evaluated scenarios. Legend: OBP, organic by-products; NMF, non-marketable food; Continuous- black lines indicate processes involved by the management of OBP. Dashed grey boxes on the far right represent the avoided productions related to production replacement.

4.3.1. Scenario #I: landfilling

This scenario represents the baseline for CEAGESP OBP management, with 37,652 tons generated in 2018, which comprehends five steps: internal OBP collection, transfer, transport to landfill, disposal and degradation (see flowchart in Figure 21). Internal OBP collection is performed by 8 diesel-fueled compactor trucks of 15 m^3 each and transported to a specific area located inside CEAGESP for temporary storage. The second step, named transfer, is also executed inside CEAGESP, in which one excavator transfers the OBP to a truck with 30 tons capacity. The third step is the OBP transportation from CEAGESP to 'Caieiras' Landfill, located 24.2 km away. The fourth step is the OBP disposal in the landfill, which is executed by five vehicles (1 excavator, 1 bulldozer, 1 compactor, 1 front loader and 1 truck), and finally, the last step is the OBP's natural degradation that generates biogas and leachate. The biogas is partially captured (80%) and burned in flares. The leachate is captured, temporarily stocked in an accumulation pond, and transported to the 'SABESP' wastewater treatment plant located in Barueri city, 39.4 km away. In the wastewater treatment plant, the leachate receives the same treatment as regular sanitary sewage, which comprehends a two-phase activated sludge system demanding energy, chemicals, and the infrastructure as the main needed inputs. Emissions to water (Tietê river) and to the atmosphere were also considered. Exclusively, the leachate components derived from the organic fraction degradation are considered, disregarding the products and effects derived from inorganic compounds. After the treatment, the purified water is released into the Tietê river, while the residual sludge is transported back to the 'Caieiras' landfill in nine annual trips executed by one 30-ton capacity truck.

4.3.2. Scenario #II: electricity production

This scenario represents the current management for CEAGESP's OBP that includes the electricity production at the 'Caieiras' landfill (Figure 21). It contemplates all the steps previously detailed for scenario #I, added to the processes related to electricity production in a power plant located inside the landfill. The amount of biogas generated by the landfill in 2018 was 142,350,000 Nm^3, 58% of which was methane. The biogas generated by the 'Caieiras' landfill follows three different pathways: 20% is directly released into the atmosphere, 40% is burned in flares (converting CH_4 into CO_2) without energy recovery, and the remaining 40% is burned in the power plant to produce electricity. According to IPCC (2006), the biogenic carbon dioxide has emissions factor equal to zero, while methane reaches a value twenty times higher (22.25; Goedkoop et al., 2009) this explains the preferred option in converting CH_4 into CO_2 through combustion. Annual electricity production reaches about 230,000 MWh/yr, from which 5,750 MWh/yr are allocated to CEAGESP's OBP.

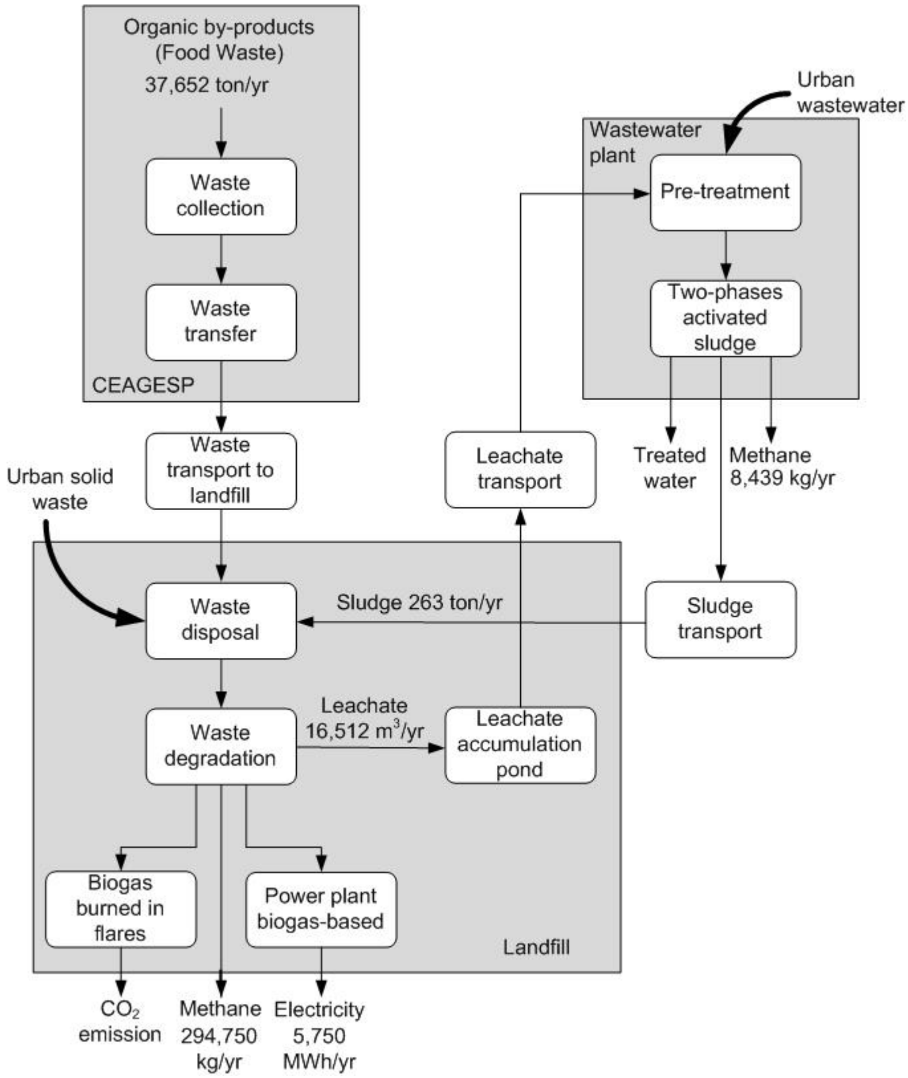

Figure 21: Processes involved in the current management of the non-marketable food (NMF) generated by the CEAGESP food distribution center.

4.3.3. Scenario #III: donation 80% + landfilling 20%

This scenario is modelled as an attempt to improve the existing food donation system and overcome its main deficiencies. According to Fagundes et al. (2014), CEAGESP reports and information provided by staffs during the fieldwork, the operation of receiving and donating food (including NMFs transportation, reception, storage, cleaning, and distribution, added to structure and materials cleaning) is carried out by the food bank and its team of professionals coordinated by a nutritionist. After OBP sorting to determine its edibility and perishability, the team of professionals establish the number of registered institutions that can receive the selected food, contacting them to schedule the collection within 24 hours. The crates of food are temporarily stocked into refrigerated rooms. Institutions collect the food, and the cold rooms are cleaned up afterwards, for the next cycle. The arrival of OBP occurs on daily basis. From all the CEAGESP's OBP, about 10-15% are not suitable food for human consumption and discarded as organic waste. Registered institutions collect the products using their own vehicles. About 70% of the registered institutions prepare meals with donated food, serving their beneficiaries, while 30% distribute food directly to families in needy communities. Beneficiaries are guests of non-profit institutions such as hospitals, recovery homes, nursing homes, other food banks, shelters, cooperatives, among others. From 2005 to 2018, 650 to 2,500 tons of food were donated yearly to charity institutions. Specifically for 2018, more than 227 registered charity institutions and 20 food banks were served with a total of 905 tons of donated food. About 15% of wholesalers have taken part in food donation programs, a considerable low percentage considering all its potential. The OBP collection is implemented on a voluntary basis mainly by CEAGESP's wholesalers, using manual trolleys and without any strategic organization and communication plan to involve all the wholesalers. This results in a lack of people involvement, a slow and ineffective food collection system that sometimes does not match with wholesalers' daily schedule, and, due to the distance between trading areas to the food bank, OBP collection does not cover 100% of trading areas. These aspects make the collection of potential NMF a service by far under its potentialities, failing to achieve the necessities of the current registered institutions, and as a result, increasing the amount of food landfilled, with all its economic, social, and environmental negative consequences.

The new scenario for food donation was modelled by considering the amount of by-products landfilled in 2018 as a baseline: 130 tons daily discarded by CEAGESP, composed by 104 tons of potential edible organic fraction (potential NMF), and 26 tons of non edible organic and inorganic fraction derived from baskets and packaging. Materials and energy sources with the lowest environmental impacts were chosen according to the available literature. The proposed collection scenario for NMF considers a recovering rate of 80% (best-case scenario), reaching 83.2 tonNMF/day. The residual 20% includes 10% loss due to mechanical injuries as a result of the transportation phase (from producers to CEAGESP), while the remaining 10% comes from quality checking at the food bank, according to Fagundes et al. (2014). This residual 20% was assumed to be landfilled through the current waste management practices.

The food donation scenario considers three steps: (1) OBP collection, (2) quality checking, and (3) storage and collection. Regarding the first step, a web of 180 food collection points was modelled considering 50 meters as the maximum distance between each wholesaler and the nearest food collection point. Collection points are the places where the wholesalers can put the OBP after their daily trading operations (mainly in the corner of black rectangles as shown in Figure 15). Each food collection point is constituted by a 1-ton capacity wooden euro pallet (1,200 x 800 mm) over a steel trolley. Wooden pallets are used because they cause lower global warming potential (Deviatkin et al., 2015). An electric logistic train derived from the 'Mizusumashi' concept was modelled, which, according to Coimbra (2009), Oliveira et al. (2018) and Vujanac et al. (2017), allows for a considerable reduction in the number of trips, the distance travelled, and the time spent, compared to both traditional forklifts and manual systems.

For the second and the third steps, an infrastructure made of roof steel of 900 m^2 surface (30l x 30w x 6h) was implemented to develop operations regarding potential NMFs quality checking and storage in refrigerated rooms. Upon the arrival of the logistic train at the quality checking area, the staff unload the pallets and transfer the crates above 108 (300kg-capacity) stainless steel tables (1.6x0.7m). After quality checking, about 10% of is discarded for landfilling, while selected and edible food is diverted to refrigerated cold rooms. This NMF is temporarily stocked on 72 plastic pallets inside 6 cold rooms with 20-tons capacity, made of steel panels with polystyrene insulation system, totaling 120 tons. Finally, the beneficiaries can collect the NMF within 24 hours.

4.3.4. Scenario #IV: donation 80% + electricity 20%

This scenario is modelled under the same assumptions as for scenario #III, in which 80% of NMF is donated and 20% is landfilled. The difference is in the electricity generated under the same conditions as scenario #II.

4.3.5. Scenario #V: avoided production 80% + landfilling 20%

This scenario is modelled under the same assumptions as for scenario #III, but the avoided emissions and the resources savings related to NMF donation are considered in this case. Since donation will avoid food production elsewhere, the emissions from food agricultural production are assumed to be negative, or avoided, while the natural resources that are not consumed are assumed to be saved. Table 3 shows the main food types donated by CEAGESP in 2014, in which case, according to fieldwork information, the values were maintained for years. The emission factors from ReCiPe 2008 midpoint (hierarchist) method v.1.13 available in Ecoinvent database (Ecoinvent, 2019) are considered to estimate the emissions of each product during its agricultural phase, which contributes to gas emission reduction.

Table 3: top-20 food types donated by CEAGESP in 2018. Products correspond to ~88% in mass units of values presented by Fagundes et al. (2014).

Product	% (in mass)	Product	% (in mass)
Tomato	35.58	Onion	2.39
Oranges	13.72	Banana	2.13
Potato	8.12	Eggplant	1.45
Apple	7.50	Peach	1.45
Papaya	6.12	Cucumber	1.32
Garlic	5.50	Manioc	1.21
Zucchini	4.37	Carrot	1.16
Chayote	3.49	Pear	0.88
Lettuce	2.74	Mango	0.87

4.3.6. Scenario #VI: avoided production 80% + electricity 20%

The same assumptions as for scenario #V are considered here, but the electricity generated by the residual landfilled fraction is included as in scenario #II.

4.3.7. Scenario #VII: biorefinery 100%

In this scenario, 100% of OBP are used as feedstock of a waste-based biorefinery. The waste composition is assumed to be the same as for donation scenario as shown in Table 3. The OBP collection system is the same as in scenario #III, the only difference regards the OBP allocation. In fact, in scenario #III, a residual fraction of 20% of OBP not suitable for human nutrition was considered, composed by 10% of highly damaged products not collected by the logistic train and directly discarded, added to another 10% discarded after the food bank quality-checking. In scenario #VII, 100% of OBP (37,652 ton/yr) are collected by the logistic train and used as a feedstock for the waste-based biorefinery. The biorefinery scenario was modelled according to the following criteria: (I) Biorefinery types suggested by literature; (II) Scale; (III) Biomass type; IV) Feasibility; (V) Circularity (VI) energy self-sufficiency

I. Biorefinery types suggested by literature: over the last few years, scientific literature converges towards the idea that anaerobic digestion, being a well-established biological process adopted for numerous and heterogeneous waste types at different scales, should play a key role in biorefinery schemes (Alibardi et al., 2020; Baral et al., 2016; Fuess et al., 2021; Sawatdeenarunat et al., 2016; among others). The work of Moreno et al. (2021), that assessed the sequential bioethanol and methane production from MSW at laboratory-level, demonstrated that anaerobic digestion of fermented residues results in similar and even higher methane yields than their raw counterparts. Guo et al. (2021) showed that introducing bioethanol production before AD has no GWP benefit with the current technological performance since extracting the sugars for bioethanol production and the distillation step consumes a great amount of energy. Ardolino et al. (2018) and Guo et al (2021) showed that the introduction of biogas upgrading to biomethane increases the environmental performance when compared to the traditional electricity and heat production at CHP plant. Therefore, AD followed by biogas upgrading to biomethane is able to recover the energy incorporated in the food waste with limited environmental impacts when compared with other options (bioethanol, CHP plant).

II. Scale: the input of ~ 38,000 tons/yr, associated to a low process complexity and a low process capacity (~100 tons/day) suggests a small scale biorefinery facility as the most appropriated for this case study (Ait Sair at al., 2021; Patrizi et al., 2015).

III. Biomass type: the average composition of the OBP was assumed to be the same as the donated food (Table 3). CEAGESP being a wholesale market, the OBP generated present peculiarities that distinguish it from OBP generated by household and industries. In fact, OBP derived by food trading operations is mainly made up of whole fruits and vegetables rather than peels and skins, which are the typical components of household organic waste, fruit juice and cannery industrial waste. Furthermore, there are no residues derived from meat or dairy products. This means that an important amount of pulp and liquid fraction is still available as biorefinery feedstock, which shows an average moisture of 89%, which corresponds to a total solid (TS) amount of 11%. According to the literature (Francini et al., 2020; Karthikeyan and Visvanathan, 2013, among others), a TS amount of about 10% is more suitable for a wet anaerobic digestion process. For this reason, in the biorefinery modelled in this work, the wet anaerobic digestion was considered as the “core” process.

IV. Feasibility: CEAGESP´s OBP presents high heterogeneity, where a high amount of OBP generation corresponds to a relatively small amount of a specific type of OBP. According to Lohrasbi et al. (2010), for example, an amount of 400,000 ton/yr of citrus waste is necessary to obtain an economically feasible production of limonene, ethanol and biogas through a biorefining process, but the citrus waste amount generated by CEAGESP, considering only the peel, is equal to ~ 2,000 ton/yr. For this reason, a biorefinery capable of producing value-added products jointly with low-added products is not suitable for CEAGESP OBP. Regarding the production of more traditional low-added products, a biological process capable of using all the components is preferable, and the anaerobic digestion to produce biogas and fertilizers is suitable for this purpose. As shown in (I), biogas upgrading to biomethane is recommended, therefore a purification process capable of removing the biogas impurities, mainly H_2S and CO_2, is necessary. According to Jeníček et al. (2017), microareation is a simple and effective way to remove H_2S, without the use of any kind of chemicals. Among the different processes to remove CO_2 from biogas, Bauer et al. (2013) and Sun et al. (2015) suggest water scrubbing as the most commonly used and easy to implement.

V. Circularity: during the normal trading operations, the wholesalers arrive at CEAGESP with trucks full of horticultural products and they return to the croplands with empty trucks. This fact was taken into account in the biorefinery scenario modelling. The digestate, produced during the anaerobic digestion process, after a solid-liquid separation, will be loaded by the wholesalers and sent back to the croplands during the return trip to allow for the close cycling of nutrients. Furthermore, the biomethane produced will be available to be sold by the main Brazilian distributor, since Brazil has very specific regulatory laws on that.

VI. Energy self-sufficiency: part of the biogas produced by the biorefinery is burned inside the system, specifically in the CHP plant to cover the internal needs of electricity and heat.

Considering all the above-mentioned criteria, a biorefinery scenario was modelled, featuring anaerobic digestion as the core process to produce biomethane and fertilizers, using microareation to remove H_2S and water scrubbing to remove CO_2, while being self-sufficient from an energetic point of view.

The flowchart of the modelled biorefinery is shown in Figure 22. The biorefinery scenario was modelled by considering seven steps: (I) internal collection and transport; (II) manual separation; (III) mechanical grinding; (IV) anaerobic digestion; (V) biogas upgrading through water scrubbing; (VI) digestate solid – liquid separation; (VII) heat and power generation. A daily input of ~ 130,000 kg/day of by-products that corresponds to an amount of 104,000 kg/day of OBP after manual separation was considered. These values were obtained by dividing the total annual input by the number of annual working days at CEAGESP equal to 363 days/year. The biorefinery structure was dimensioned by accounting for a security buffer of ~10%, while the demand for materials and energy were modelled by considering the estimated daily input of 130,000 and 104,000 kg/day of by-products and OBP respectively.

The first step, internal collection and transport, is very similar to scenario #III. The difference regards the percentage of collected OBP equal to 100%, as it was assumed that all OBP generated by CEAGESP are suitable for the Biorefinery facility, without any residual fraction sent to landfill.

In the second step the separation between organic and inorganic fractions of CEAGESP's by-products is implemented, which is executed manually and facilitated by the use of a conveyor belt. In fact, according to Uratani et al. (2014), the amount of by-products generated by CEAGESP and the high organic fraction percentage do not justify the implementation of a mechanical separation facility. Therefore, the step was modelled by considering a conveyor belt of 20 meters in length and 7.29 kW power (Uratani et al., 2014), and a number of hours equal to 9.5 to complete the process, in two shifts, was estimated. This manual separation process removes the inorganic fraction (20%), therefore, the OBP input to the biorefinery is ~104,000 kg/day. The treatment of the inorganic fraction after the removal is outside the scope of this work.

The third step, mechanical grinding and shredding, includes the materials and electricity consumption related to the grinder motor. By assuming a maximum capacity of 7 ton/hour

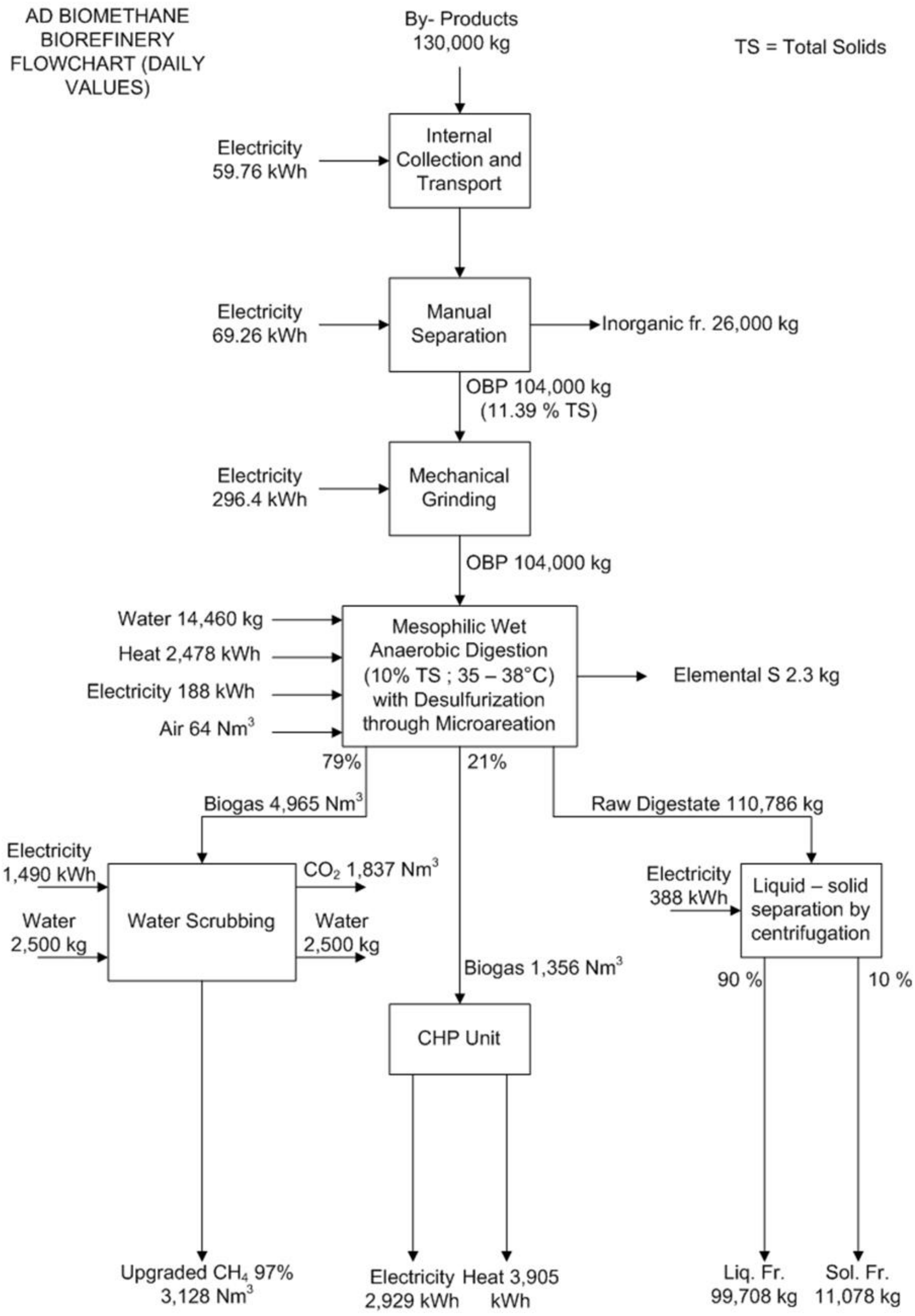

Figure 22: Biorefinery flowchart. Inputs and outputs calculated on daily basis, for an input of 130-ton by-products / day. Liq. Fr: liquid fraction; Sol. Fr: Solid fr

and a power of 30 kW (40.8CV) for each machine (Uratani et al., 2014), two grinding machines are included for normal use, added to another machine for emergency use in case of maintenance, therefore, three machines in total.

The fourth step regards the anaerobic digestion, which considers a wet mesophilic (35°C-38°C) AD process with 10% total solids inside to a one stage vertical biodigester with approximately cylindric shape of 3,000 m^3 that generates biogas (60% CH_4 – 40% CO_2, according to Francini et al., 2020; SGC, 2012) and digestate. Biogas desulfurization was modelled through air microinjections. Around 21% of the biogas is sent to an internal CHP plant while the remaining 79% is sent to the upgrading process. This step demands electricity, heat, air and water consumption as well the structural materials related to the biodigester.

The fifth step, biogas upgrading, considers water scrubbing technology with an input of raw biogas, water, and electricity. The upgraded biomethane (97%), CO_2 and water are the outputs. All materials and energy demanded by machines were considered as well.

The sixth step is the solid-liquid separation, which accounts for the electricity consumed by the centrifuge. The solid-liquid partition coefficients were assumed to be the same as for Tampio et al. (2014).

The seventh step is electricity and heat production at an internal CHP plant. Two CHP units of 100 kW each were accounted for to provide internal heat and energy needs. Direct emissions derived by biogas combustion within CHP and the materials used for the equipment were considered.

4.3.8 Scenario #VIII: biorefinery + avoided production 100%

Besides accounting for all those environmental impacts related to the biorefinery facility (scenario #VII), this #VIII scenario includes the potential environmental benefits resulting from conventional production replacement of natural gas and chemical fertilizers production (Nitrogen, Potassium and Phosphorous).

4.4 Life cycle Assessment

Considering a user-side approach, a common way to assess the environmental impacts is through a Life Cycle Assessment (LCA) Perspective. LCA is a structured, comprehensive, and internationally standardized method. It quantifies all relevant emissions and resources consumed, their related impacts on environment and human health, and resource depletion issues that are associated with any good or service delivered by a process (“products”). LCA considers a product’s full life cycle from “cradle to grave”: the extraction of resources, production, use, recycling, up to the final disposal of process waste and product after its useful life (JRC 2010). LCA measures the environmental impacts of every step in the life cycle of a product, starting with the extraction of the raw materials, the energy needed to manufacture the product, transportation, distribution to the consumer, the use of the product by the consumer, and ending with the ultimate disposal of the product at the end of its lifespan (Mcintosh et al, 2017). In other words, LCA looks at the process relation with the environment as a source and as a sink, and provides indicators related to many different environmental impact categories, such as climate change, stratospheric ozone depletion, depletion of resources, toxicological effects, among others (Pennington et al. 2004).

LCA methodology is standardized by ISO documents 14040/2006 and 14044/2006, as well as in the ILCD Handbook (ISO 14040 2006; ISO 14044 2006; JRC 2010), and includes the following four stages: definition of goal and scope, inventory analysis, impact assessment, and interpretation (Figure 23). Carrying out an LCA study is usually an iterative process: once the goal of the work is defined, the initial scope settings that define the requirements on the subsequent work are derived. However, as more information becomes available during the life cycle inventory phase for data collection and during the subsequent impact assessment and interpretation phases, the initial scope settings would be refined and sometimes also revised (JRC, 2010).

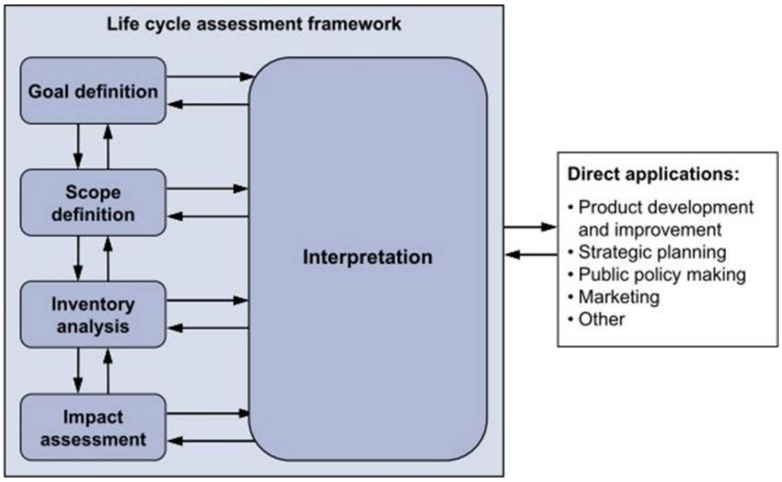

Figure 23: Framework for Life Cycle Assessment (ISO 14040:2006, modified).

In this study, the ReCipe 2008 (H) method was used to calculate the LCA impacts. The ReCiPe method, as well as many others Life Cycle Assessment tools, provides the possibility to calculate the impacts considering many different impact assessment methods. Although these methods vary in several aspects, the distinction between midpoint and endpoints methods is important. An endpoint method measures the environmental impact at the end of this cause- effect chain. A midpoint method measures the impacts earlier along the cause-effect chain before the endpoint is reached. The latter has a lower level of uncertainty compared with the former, for this reason it was the method chosen in this work. Figure 24 shows an example of the difference between midpoint and endpoint methods for climate change.

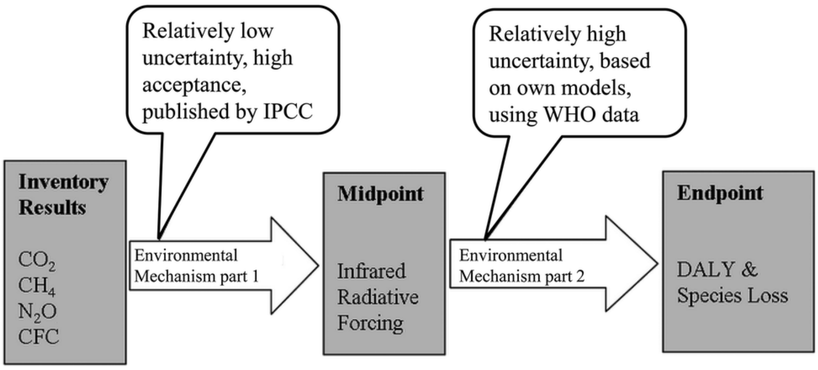

Figure 24: example of harmonized midpoint-endpoint model for climate change, linking to human health and ecosystem damage. At midpoint levelis measured the infrared radiative forcing (expressed in CO2 eq), at endpoint the DALY indicator and Species Loss (adapted from Goedkoop et al., 2009).

The ReCiPe method presents conversion factors based on three different perspectives: individualistic, hierarchist, and egalitarian. The first one is based on the short-term interest, impact types that are undisputed, technological optimism as regards human adaptation; the second one is based on the most common policy principles with regards to time-frame and other issues; the last one is the most precautionary perspective, taking into account the longest time-frame, impact types that are not yet fully established (Goedkoop et al., 2009). In this study, the hierarchist perspective was adopted, as it is an intermediate perspective that assumes the most common positions considering all aspects. The ReCiPe midpoint (H) method (Goedkoop et al., 2009) includes upstream categories (related to depletion of natural resources, such as water depletion, fossil depletion etc.) and downstream categories (related to impacts on natural ecosystems or human health, such as global warming, human toxicity, terrestrial acidification etc.).

The goal of this study is to compare the life cycle environmental performance of the current OBP management at CEAGESP with plausible donation and biorefinery scenarios. The work is performed in compliance with the guidelines of the international standard organization (ISO, 14040; 14044), by using the ReCiPe 2008 hierarchist (H) method (Goedkoop et al., 2009).

The functional unit of this study is the management of 1 ton of organic by products. Differently from other processes that produce a good or service for a specific function, waste management focuses downstream on production processes to find a more sustainable management for the generated by-product. The LCA was developed using Microsoft Visio® for figures, and Microsoft Excel® for quantitative analysis. The indirect impacts, such as fuel and vehicles and machines production, landfill and wastewater plant materials, chemical products, donation shed materials, biorefinery plant materials, electricity used and avoided emissions for the established scenarios were modelled by using the characterization factors provided by the ReCiPe (2008) midpoint hierachist method v.1.13, as available in the Ecoinvent database (Ecoinvent, 2019). Brazilian values for characterization factors were considered when available, and global values for all other cases. Direct impacts were also calculated by using characterization factors provided by the ReCiPe 2008 midpoint (Hierarchist) method (Goedkoop et al., 2009). The impact categories used in this work (Table 4) were chosen due to their representativeness for the evaluated system, as also considered by other authors (Albizzati et al., 2019; Brancoli et al., 2020; Buratti et al, 2015; Oliveira et al., 2017).

Table 4: LCA impact categories used in this work

Impact Category	Abbr.	Description
Fossil Depletion	FDP	The use of non-renewable energy sources, e.g., coal, crude oil, in kg oil equivalence
Freshwater Eutrophication	FEP	The causing of dense growth of algae or other plant life due to the excessive accumulation of nutrients in a body of freshwater (river, lakes), in kg P equivalent to freshwater
Global Warming	GWP	The causing of global atmospheric temperature increase due to specific air emissions, measured in kg of carbon dioxide equivalent to air
Human Toxicity	HTP	The endangering of human health due to toxic chemical emissions measured in kg of 1,4 dichlorobenzene equivalence (kg of 1-4DB eq)
Mineral (Metal) Depletion	MDP	The use of raw finite materials in (copper, lead…) in kg Fe eq extracted
Particular Matter Formation	PMFP	The damage to human health caused by fine particular matter with less than 10 µm (in kg PM_{10} eq to air) in diameter
Photochemical Oxidant Formation	POFP	The increased likelihood of harmful smog and haze caused by various emissions (in kg NMVOC eq to air)
Terrestrial Acidification	TAP	The causing of acid rain due to interactions in the atmosphere of specific emissions, measure in kg of sulfur dioxide equivalence (kg SO_2 eq to air)
Water depletion	WDP	Consumed water in m^3

4.5 Emergy Accounting

The development of the emergy concept and its theoretical base cannot be separated from the development of the concept of energy quality. This concept has been evolving since the 1950s with H.T. Odum's work on tracing energy flows in ecosystems. All forms of energy have different abilities to do work, because they have different "energy quality". Odum began using the term embodied energy to refer to energy quality differences in terms of their generation costs, and a ratio called quality factor for the calories (or Joules) of one kind of energy required to make those of another (Odum and Odum 1980). Later, the term embodied energy was abandoned and substituted by "emergy", and the quality factor ratio named "transformity".

Emergy is defined as "the availability of energy (exergy) of one kind that is (previously) used up in transformations directly and indirectly to make a product or service" (Odum, 1996). The unit of emergy is the emjoule, a unit referring to the available energy of one kind consumed in transformations. For example, sunlight, fuel, electricity and human service can be accounted for together by using the amount of emjoules of solar energy required to produce each one of these inputs a as a common basis. In this case, the value is a unit of solar emergy expressed in solar emjoules (abbreviated seJ).

Most of definitions of ‘value’ are based on a utility approach, or what is received from an energy transformation process. Thus, fossil fuels are evaluated based on the heat generated when they are burnt, while economic evaluations are based on the willingness to pay for perceived utilities. An opposite view of value in the biosphere is based on what is put into something rather than what is received from it, and this idea of “donor side” perspective constitutes the basis of the Emergy Accounting (EMA) approach (Odum, 1996). Emergy can be used to value flows of energy and materials within the biosphere, from a donor-side point of view. When expressed in units of the same form of energy, systems of varying scales and organization can be compared, and indices of performance and integrity can be calculated. Emergy flows, inputs and outputs are usually represented through specific diagrams (Figure 25) using specific symbols proposed by Odum (1996).

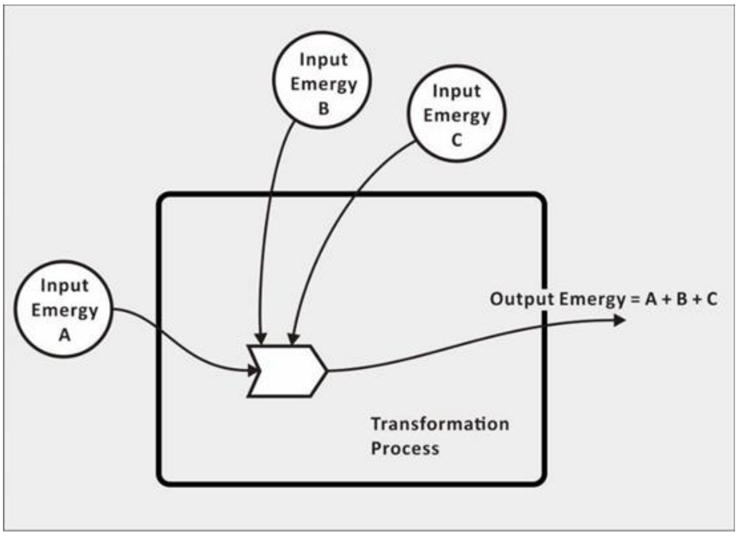

Figure 25: Example of Emergy Diagram (source Emergy Society http://www.emergysociety.com).

Unit Emergy Values (UEV’s or emergy intensities) are calculated on the emergy required to generate one unit of output. The emergy associated to a flow is easily calculated if the unit emergy value (UEV) is known. The flow expressed in its units is multiplied by the emergy per unit of that flow. When comparing and testing alternative parallel processes, the transformity measures their efficiency in delivering the same product. The total emergy use, U, measures the emergy that converges to produce the yield Y (output). Since it is a measure of the emergy cost of the yield, U is the emergy assigned to the yield Y or the environmental work supporting the yield itself. In addition to the total emergy input (U) and the UEV’s, the main emergy-based indicators are the Environmental Loading Ratio (ELR), the Emergy Yield Ratio (EYR), and the Emergy Sustainability Index (ESI) (Brown and Ulgiati 2004). Several other ratios can be calculated, depending on the objectives, type, and scale of the systems being evaluated.

Regarding the emergy accounting method, the indicators explored in this present work are **the total emergy U** (U = R + N + F) as the sum of renewable input flows (R), non-renewable input flows (N) and purchased input flows (F). F is splitted into purchased materials (M) and purchase services (S). Suffixes “n” and “r” mean non-renewable and renewable respectively, referring to the renewable and non-renewable component of material and energy flows (Figure 26). In this study the most recently published emergy baseline 12.00E+24 seJ/yr (Brown et al., 2016) was chosen as the reference to update and standardize all UEVs used. Other explored emergy indicators were:

Unit Emergy Value (UEV): the general label for all emergy intensities. It is defined as the solar emergy required to make one unit of system's product output. It is calculated by the ratio of total emergy (U) that was used in a process to the product amount (UEV = U / Product). When using Joules as the unit for the product, the UEV is referred to as Transformity.

Net emergy benefit (NEB = saved emergy) − (emergy investment): Any waste management system demands a certain number of resources used up to reduce the environmental impacts generated by the waste. For example, all the materials and processes related to the construction and operational steps of a sanitary landfill, or the resources used to build an incineration plant, or a recycling plant are quantified through emergy synthesis and referred to as “invested emergy” (EMI). Simultaneously, some waste management systems can provide useful outputs for society, such as electricity generated in landfills or incinerators, or the materials recovered in recycling plants. This useful emergy is referred to as “saved emergy” (EMS) or recovered emergy. The electricity generated from landfill biogas saves that electricity generated through the conventional process available in the national grid, which potentially would save a certain amount of emergy, as well. Here, the net emergy appears as an important indicator expressing the difference between the saved (recovered) and the invested emergy, where higher values mean better performance in saving emergy (Odum, 1996).

Keeping these concepts of EMS and EMI in mind, the studied scenarios in this work, as previously described, could be classified according to the following characteristics:

- Scenarios #I, #III and #VII account for the invested emergy EMI only.
- Scenarios #II, #IV, #V, #VI and #VIII, besides EMI, also have EMS according to the related avoided production.

Several authors have considered net emergy in their waste management studies under different levels in the HWM (Marchettini et al., 2007; Agostinho et al., 2013), among other studies that, although not directly providing the net emergy indicator, provide numbers for its calculation (Patrizi et al., 2015; Santagata et al., 2019). Anyhow, this present work attempts to provide insights on EMI and EMS along the food recovery hierarchy, beyond the calculation of net emergy indicator for the previously established scenarios.

Emergy return index (ERI): this corresponds to the EMS/EMI ratio. It provides information about the amount of saved emergy per unit of invested emergy. Values > 1 indicates a gain, in emergy terms. It is a new index proposed in this work, modelled to facilitate the comparison among the waste management options, characterized by a different EMS and EMI, according to their peculiarities. The higher the value of this index, the higher the ability of one scenario to save emergy for each seJ of invested emergy.

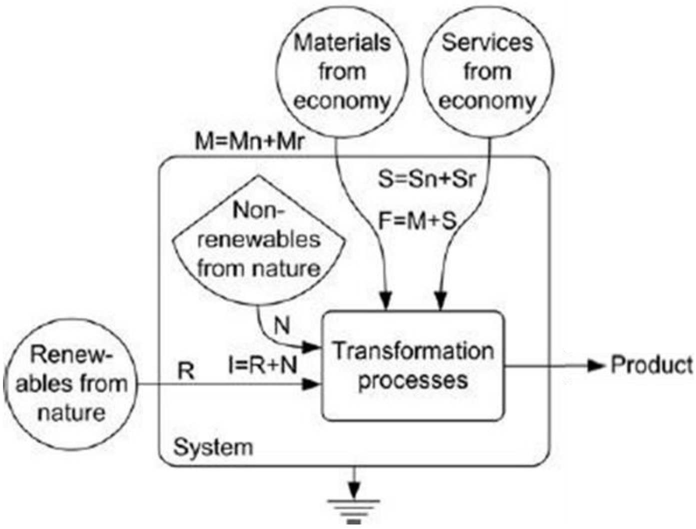

Figure 26: General diagram representing all energy sources involved in the transformation process. Adapted from Giannetti et al. (2015).

Depending on the chosen option along the food recovery hierarchy, the emergy used up during the waste treatment sometimes results in emergy benefits. According to emergy rules (Odum, 1996), a waste treatment option can be represented as an interaction between the emergy of waste (emx) with the emergy invested (EMI) in waste treatment. The sum of both (emy) represents the total emergy embodied on the assessed system, while EMS represents the emergy saved or recovered (Figure 27). Specifically, for this study, emx is the emergy of 1 ton of OBP, EMI is the invested emergy to manage 1 ton of OBP, and EMS is the saved emergy of electricity, food and/or fuel + fertilizers; this applied, for example, for scenarios #II, #V and #VIII respectively (scenario #I has no emergy recovery). It is easy to note that scenarios #I, #II, #V and #VIII represent different management levels on the FRH, although they have in common the same input emx of 1 ton OBP. According to the most recent discussions and advances on how to deal with waste in emergy synthesis, including the concepts of emergy algebra, co-products and by-products (Agostinho et al., 2013; Brown, 2015; Gala et al., 2015; Santagata et al., 2019), we have considered in this work that emergy of waste is 'lost', in other words, the emergy of OBP entering the scenario's boundaries is equal do zero. Disregarding other nomenclatures, Santagata et al. (2019) named this approach the ‘zero burden approach’. Because emx is independent of the adopted option for the waste management within the FRH, it is possible to hypothesize that saved emergy depends on EMI, leading to the following statement: the emergy used up to treat the waste (EMI) behaves as an independent variable, and the saved or recovered emergy (EMS) as a dependent variable. Here, an EMS = f(EMI) relation can be assumed, i.e., the recovered emergy EMS is a function of the invested emergy EMI. This hypothesis is discussed in this study, by considering our data and other from the literature.

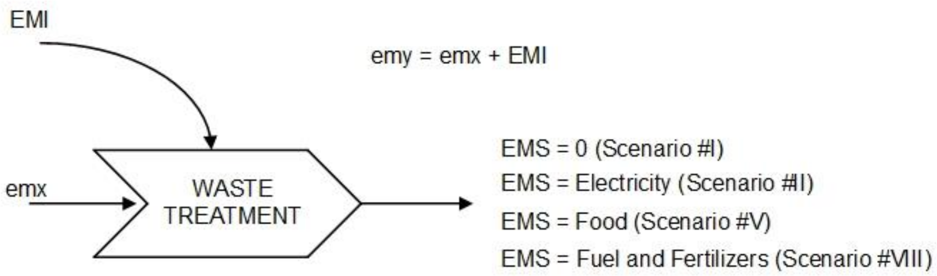

Figure 27: representation of a general waste treatment system and its dependence on emergy inputs. Legend: emx = emergy of OBP; EMI = invested emergy; EMS = saved emergy.

The resultant EMS = f(EMI) data are plotted in a x-y scatter plot graph by using a Microsoft Excel spreadsheet to identify a possible relationship between the two variables. Once the possible relation is recognized, it is important to find a function capable of describing, from a mathematical point of view, the identified trend and relative parameters through a process of curve fitting. According to Brown (2001, pg.191), “curve fitting essentially describes the experimental data as a mathematical equation in the form y=f(x), where x is the independent variable and is controlled by the experimenter; y is the dependent variable, which is measured; and f is the function that includes one or more parameters used to describe the data.” In this study, which investigates real scenarios, EMI is the variable controlled by the decision maker who chooses an option among others proposed by the food recovery hierarchy, while EMS is the result that depends on the chosen scenario. Once a probable function is recognized, the next step is to determine the goodness of a fit, how well the function describes the data. According to Brown (2001, pg.192), “the most commonly used measure of the goodness of fit is least squares. This is based on the principle that the magnitude of the difference between the data points and the curve is a good measure of how well the curve fits the data”. Figure 28 shows how the square method works, showing an example featuring a simple linear function, however, the same assumptions are also valid for non-linear functions, with few modifications that are explained in the following paragraphs. The difference between the real data and the fit is highlighted by the vertical arrows in Figure 29-A and calculated. The result is shown in Figure 29-B, where the y value of each point is replaced by the distance of that point from the linear function. To eliminate positive and negative effects of the deviation, the least squares method squares the differences, as depicted in Figure 29C and described by Equation (5).

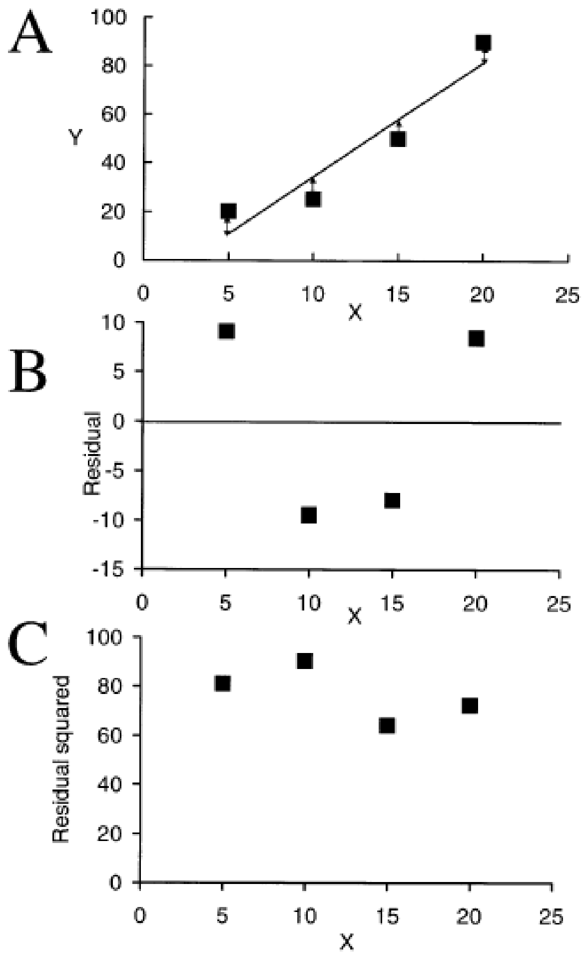

Figure 28: Least squares method.

$$SS = \sum_{i=1}^{n}\left[y - y_{fit}\right]^2 \quad (5)$$

Where: y is the data point, y_{fit} is the value of the curve at point y, and SS is the sum of the squares. In this study, Equation (5) can be rewritten as shown in Equation (6)

$$SS = \sum_{i=1}^{n}\left[EMS - EMS_{fit}\right]^2 \quad (6)$$

Where: SS is the sum of the square, EMS is the real saved emergy value found in a specific point and EMS_{fit} is the value of the theoretical model at the same point.

For data that are not described by a linear function, a method commonly used is called interactive non-linear least square fitting. This process uses the same goal as described for linear regression, i.e. minimizes the value of the squared sum of the difference between data and fit, however, it differs from linear regression as it is an iterative, or cyclical process. After a first estimation of the parameters made by the researcher, according to his/her prior experience, the first interation involves the calculation of the SS based on these initial values, followed by further interactions to calculate the SS after changing the parameters of a small amount, until the SS value is found (Brown, 2001). The coefficient of determination called $r^{2,}$ by convention, in case of linear regression, or R^2 in case of non-linear regression, are calculated to determine the proportion of variance in the dependent variable explained by the independent variable, ranging from 0 to 1. Values of 0 indicate that two variables are not correlated, while values close to 1 suggest that more observed function fits the data in a more accurate way (Brown, 2001). A simple and useful tool applied to linear and non-linear functions to calculate the SS and the coefficient of determination is the SOLVER, and it is available as a Microsoft Excel add-in. Several authors have used SOLVER and confirmed its reliability in different fields (Brown, 2001; Brown, 2006; Briones and Escola, 2019; Delgado-Aguilar et al., 2018; among others.), thus the SOLVER tool is used in this work to calculate the SS and the coefficient of determination.

5. RESULTS AND DISCUSSION

5.1. Life Cycle Assessment

5.1.1. Data Collection and modelling

The inventory data for all evaluated scenarios per ton OBP are shown in Table 5, while the main equations used in the calculation processes are depicted in Table 6. The description of modelling procedures and the main assumptions applied in this work are presented in Table 7 and in the following paragraphs. Further details regarding scenarios modelling and calculation are available in Appendix B.

Scenario #I. Data were obtained from CEAGESP's annual reports (CEAGESP Report, 2019),and during fieldwork (carried out in October 2019) through personal communication with technical staffs of CEAGESP and the 'Caieiras' Landfill. Scientific literature was also considered to fulfil dataset and check consistencies. The system boundaries include the internal waste collection at CEAGESP, the landfill, and the wastewater plant (see Figure 21). For internal OBP collection and transport, 8 diesel fueled compactor trucks with 15 m^3 capacity each are used, assuming an average consumption of 8 L/hour from Zand et al. (2019). Annual activity hours were calculated by considering two daily shifts of 2.5 hours each (5 hours/day) in 363 days/yr, according to the yearly operating days of the distribution center. For the OBP transfer, a Doosan Daewoo Solar 175 LCV excavator is used, with average diesel consumption of 217.5 g/kWh. A total of 784 hours of activity/yr were estimated, by assuming an average loading time of 30 minutes for 30 tons. For the third step, which is focused on OBP transport from CEAGESP to the 'Caieiras' Landfill, two 30-ton capacity transport trucks are used to cover a 48.4 km roundtrip, including the empty return trip from the Landfill to CEAGESP, totalizing 1,569 trips/yr. Diesel consumption of 0.28 L/km was estimated from CETESB (2019) by considering 15<ton<45 capacity transport trucks. The fourth step (OBP landfilling) includes five vehicles (1 Hyundai 220 LC excavator, 1 bulldozer, 1 soil compactor, 1 front loader, and 1 truck of about 22 tons, 30 tons, 12 tons, 23.5 tons and 14.5 tons weight respectively). Average diesel consumption was estimated as 1.11 kg/OBP ton, from Yang et al. (2014). Regarding construction materials, the demand for benthonic geocompost, HDPE, geotextile and gravel are included. In the last step, waste degradation at the 'Caieiras' Landfill, the assumptions were considered in allocating outputs, as follows. CEAGESP's OBP percentage (~2.5%) on the total organic waste landfilled in 2018 was used as a criterion to allocate landfill biogas emissions, while CEAGESP's OBP leachate fraction (0.009% in mass of the total wastewater treated at Baruerí wastewater plant in 2018, estimated by using biochemical oxygen demand values) was used as a criterion to allocate inputs and outputs of the wastewater plant. As for biogas (58% methane + 40% CO_2 + 0.6% O_2 + traces of other gases), 80% is the fraction captured and burned without electricity production, and the remaining 20% is released into the atmosphere. The leachate derived from CEAGESP's OBP in Caieiras was estimated as 16,512 m^3/yr, transported to the wastewater treatment plant (distance 78.8 km, roundtrip, including the empty return trip) by a 30 m^3 capacity tank truck in 550 trips/yr. Regarding the wastewater treatment plant, inputs of electricity, ferric chloride, and polyacrylamide were quantified, as well as the cement and steel used during the wastewater plant construction; the CH_4 released into the atmosphere and the P released to water bodies were the emissions considered. The CO_2 emissions from OBP decomposition in the landfill (directly produced or originated by the combustion of methane in flares and in the power plant) are not accounted for, as they have a biogenic origin (IPCC, 2006). Common assumptions regarding all steps are: (a) Materials used in vehicles were estimated from RICARDO AEA (2015), by considering the relative percentages of the first five components of ~14.5 ton artic truck (steel, iron, plastic, rubber and aluminium) as a reference; (b) Direct emissions from vehicles comprehend CO_2, CO, N_2O, NMVOC, CH_4, NO_x, SO_2 and PM_{10}. Due to unavailability of accurated data about vehicle models, a weighted average per year of the circulating truck fleet in São Paulo city in 2018 is assumed, to include the effects related to the age of the fleet (see appendix B, Figure B1). The report of vehicular emissions from 1999 to 2018 of São Paulo State (CETESB, 2019) was the data source, in which diesel trucks of 15<tons<45 weight were considered as a reference; (c) Diesel consumption during waste collection, transfer, transport and landfilling phases is assigned to all the OBP, despite organic content of waste being 80%. This is consistent with Buratti et al. (2015) who stated "the not separated collection of the organic fraction requires the management of a not separable fraction of inorganic waste".

Table 5: Inventory summary for the evaluated scenarios. Values per ton OBP.

Item	Unit/ton OBP	Scenarios #I	#II	#III	#IV	#V	#VI	#VII	#VIII
Inputs		**Landfilling**	**Electricity**	**Donation 80% + Landfilling 20%**	**Donation 80% + Electricity 20%**	**Avoid. pr. 80% + Landfilling 20%**	**Avoid. pr. 80% + Electricity 20%**	**Biorefinery**	**Biorefinery + Avoided pr.**
Steel	kg	3,56E-01	3,63E-01	1,69E-01	1,71E-01	1,69E-01	1,71E-01	2,33E-01	2,33E-01
Iron	kg	6,11E-02	6,11E-02	1,22E-02	1,22E-02	1,22E-02	1,22E-02	n.a.	n.a.
Rubber	kg	3,34E-02	3,34E-02	6,69E-03	6,69E-03	6,69E-03	6,69E-03	n.a.	n.a.
Plastic	kg	3,23E-02	3,23E-02	6,46E-03	6,46E-03	6,46E-03	6,46E-03	n.a.	n.a.
Aluminum	kg	2,06E-02	2,06E-02	4,12E-03	4,12E-03	4,12E-03	4,12E-03	n.a.	n.a.
Diesel	kg	5,17E+00	5,17E+00	1,03E+00	1,03E+00	1,03E+00	1,03E+00	n.a.	n.a.
GCL	kg	5,18E-01	5,18E-01	1,04E-01	1,04E-01	1,04E-01	1,04E-01	n.a.	n.a.
HDPE	kg	4,03E-01	4,03E-01	8,07E-02	8,07E-02	8,07E-02	8,07E-02	3,72E-01	3,72E-01
Geotextile	kg	1,18E-01	1,18E-01	2,36E-02	2,36E-02	2,36E-02	2,36E-02	n.a.	n.a.
Gravel	kg	1,79E+02	1,79E+02	3,59E+01	3,59E+01	3,59E+01	3,59E+01	n.a.	n.a.
Cement	kg	1,76E-02	1,76E-02	3,52E-03	3,52E-03	3,52E-03	3,52E-03	n.a.	n.a.
Electricity	kWh	3,99E-01	3,99E-01	1,66E+00	1,66E+00	1,66E+00	1,66E+00	2,57E+01	2,57E+01
Ferric chloride	kg	4,40E-01	4,40E-01	8,80E-02	8,80E-02	8,80E-02	8,80E-02	n.a.	n.a.
Polyacrylamide	kg	3,37E-02	3,37E-02	6,75E-03	6,75E-03	6,75E-03	6,75E-03	n.a.	n.a.
Concrete	kg	n.a.	2,81E-05	n.a.	5,62E-06	n.a.	5,62E-06	n.a.	n.a.
Water	m^3	n.a.	6,06E-03	n.a.	1,21E-03	n.a.	1,21E-03	1,64E-01	1,64E-01
Lubricant oil	kg	n.a.	6,71E-02	n.a.	1,34E-02	n.a.	1,34E-02	n.a.	n.a.
Lead	kg	n.a.	n.a.	1,12E-02	1,12E-02	1,12E-02	1,12E-02	1,12E-02	1,12E-02
Wooden Pallets	kg	n.a.	n.a.	1,20E-02	1,20E-02	1,20E-02	1,20E-02	1,20E-02	1,20E-02
Polystyrene	kg	n.a.	n.a.	1,75E-03	1,75E-03	1,75E-03	1,75E-03	n.a.	n.a.
Plastic Pallets	kg	n.a.	n.a.	2,28E-03	2,28E-03	2,28E-03	2,28E-03	n.a.	n.a.
Heat	kWh	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	2,39E+01	2,39E+01
Outputs									
Electricity	kWh	n.a.	1,52E+02	n.a.	3,05E+01	n.a.	3,05E+01	n.a.	n.a.
Landfill Biogas	m^3	9,46E+01	9,46E+01	1,89E+01	1,89E+01	1,89E+01	1,89E+01	n.a.	n.a.
Leachate	m^3	4,40E-01	4,40E-01	8,80E-02	8,80E-02	8,80E-02	8,80E-02	n.a.	n.a.
Donated Food	kg	n.a.	n.a.	8,00E+02	8,00E+02	8,00E+02	8,00E+02	n.a.	n.a.

Biomethane (97%)	m^3	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	3,02E+01	3,02E+01
Solid Digestate	kg	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	1,07E+02	1,07E+02
Liquid Digestate	kg	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	9,61E+02	9,61E+02
Tot. Digestate	kg	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	1,07E+03	1,07E+03
N Fertilizer	kg	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	3,43E+00	3,43E+00
P Fertilizer	kg	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	4,11E-01	4,11E-01
K Fertilizer	kg	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	1,28E+00	1,28E+00
Direct Emissions									
NO_X	kg	8,39E-02	3,17E-01	1,68E-02	6,33E-02	1,68E-02	6,33E-02	1,53E-01	1,53E-01
CH_4	kg	8,05E+00	8,05E+00	1,61E+00	1,61E+00	1,61E+00	1,61E+00	9,12E-02	9,12E-02
NMVOC	kg	2,74E-03	2,74E-03	5,49E-04	5,49E-04	5,49E-04	5,49E-04	3,95E-03	3,95E-03
CO	kg	1,41E-02	1,41E-02	2,82E-03	2,82E-03	2,82E-03	2,82E-03	7,71E-02	7,71E-02
*CO_2	kg	1,60E+01	1,60E+01	3,20E+00	3,20E+00	3,20E+00	3,20E+00	n.a.	n.a.
N_2O	kg	6,41E-04	6,41E-04	1,28E-04	1,28E-04	1,28E-04	1,28E-04	1,41E-04	1,41E-04
PM_{10}	kg	1,72E-03	1,72E-03	3,44E-04	3,44E-04	3,44E-04	3,44E-04	1,27E-04	1,27E-04
SO_2	kg	1,57E+00	1,57E+00	3,13E-01	3,13E-01	3,13E-01	3,13E-01	n.a.	n.a.
Replaced Products									
Electricity	kWh	n.a.	1,52E+02	n.a.	3,05E+01	n.a.	3,05E+01	n.a	n.a.
Food	kg	n.a.	n.a.	n.a.	n.a.	8,00E+02	8,00E+02	n.a	n.a.
Natural Gas	m^3	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	3,02E+01
N chemical Fertilizer	kg	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	3,43E+00
P chemical Fertilizer	kg	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	4,11E-01
K chemical Fertilizer	kg	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	1,28E+00

*Only CO_2 emissions derived by Diesel combustion were included. Direct CO_2 emissions from landfill (directly emitted or after CH_4 combustion) and from biodigester were not included because biogenic. n.a. = not applicable

Table 6: Overview of LCA modelling procedures applied in the evaluated scenarios

Indirect impacts	
The coefficients of the ReCipe 2008 (hierarchist; Ecoinvent version 3.6, 2019) method for eachimpact category were applied in the inventory of Table 5. Data available in Table B9.	
Direct impacts	
Diesel burned in engines (in L or kg)	
GWP ($kgCO_{2eq.}$/yr)	(L/yr * 2.603 $kgCO_2$/L) + (L/yr * 2.09E-04 $kgCH_4$/L * 22.25 $kgCO_2/kgCH_4$) + (L/yr * 1.04E-04 kgN_2O/L * 298 $kgCO_2/kgN_2O$)
PMFP ($kgPM_{10}$/yr)	(L/yr * 2.80E-04 $kgPM_{10}$/L * 1 $kgPM_{10}/kgPM_{10}$) + (L/yr * 1.37E-02 $kgNO_X$/L* 0.22 $kgPM_{10}/kgNO_X$) + (kg/yr * 3.03E-01 $kgSO_2$/kg * 0.2 $kgPM_{10}/kgSO_2$)
POFP ($kgNMVOC_{eq.}$/yr)	(L/yr * 2.09E-04 $kgCH_4$/L * 0.01 $kgNMVOC/kgCH_4$) + (L/yr * 1.37E-02$kgNO_X$/L * 1 kgNMVOC / 1 $kgNO_X$) + (kg/yr * 3.03E-01 $kgSO_2$/kg * 0.081 $kgNMVOC/kgSO_2$)+(L/yr*4.47E-04kgNMVOC/L*1 kgNMVOC/kgNMVOC)
TA ($kgSO_{2\ eq.}$/yr)	(kg/yr * 3.03E-01 $kgSO_2$/kg * 1 $kgSO_2/kgSO_2$) + (L/yr * 1.37E-02 $kgNO_X$/L* 0.56 $kgSO_2/kgNO_X$)
Landfill methane emissions to atmosphere	
GWP ($kgCO_{2eq.}$/yr)	($kgCH_4$ /yr * 22.25 $kgCO_2/kgCH_4$)
POFP ($kgNMVOC_{eq.}$/yr)	($kgCH_4$/yr * 0.01 $kgNMVOC/kgCH_4$)
Phosphorus emissions to water from wastewater treatment plant	
FEP ($kgP_{eq.}$/yr)	(kgP/yr * 1 kgP/kgP)
Emissions from landfill electricity production	
PMFP ($kgPM_{10eq.}$/yr)	(kgNOx/yr * 0.22 $kgPM_{10}/kgNO_X$)
POFP ($kgNMVOC_{eq.}$/yr)	(kgNOx/yr * 1 $kgNMVOC/kgNO_X$)
TA ($kgSO_{2eq.}$/yr)	(kgNOx/yr * 0.56 $kgSO_2/kgNO_X$)
Biogas burned at Biorefinery CHP plant (in GJ)	
GWP ($kgCO_{2eq.}$/yr)	(GJ/yr * 323 gCH_4/GJ * 10^{-3} kg/g * 22.25 $kgCO_2/kgCH_4$) + (GJ/yr * 0.5 gN_2O/GJ * 10^{-3} kg/g * 298 $kgCO_2/kgN_2O$)
PMFP ($kgPM_{10}$/yr)	(GJ/yr * 0.451 gPM_{10}/GJ * 10^{-3} kg/g * 1 $kgPM_{10}/kgPM_{10}$) + (GJ/yr * 540 gNOx/GJ * 10^{-3} kg/g * 0.22 $kgPM_{10}/kgNO_X$)
POFP ($kgNMVOC_{eq.}$/yr)	(GJ/yr * 323 gCH_4/GJ * 10^{-3} kg/g * 0.01 $kgNMVOC/kgCH_4$) + (GJ/yr * 540 gNOx/GJ * 10^{-3} kg/g * 1 kgNMVOC / 1 $kgNO_x$) + (GJ/yr * 14 gNMVOC/GJ * 10^{-3} kg/g * 1kgNMVOC/kgNMVOC)
TA ($kgSO_{2\ eq.}$/yr)	(GJ/yr *540 gNOx/GJ * 10^{-3} kg/g * 0.56 $kgSO_2/kgNO_X$)
Avoided Impacts	
Donated Food: by assuming a recovery of 800 kg/ton OBP, the contribution rate of each food type was considered as shown in table 3. Biomethane: it was assumed replacing Natural Gas production according to a ratio 1 m^3 biomethane = 1 m^3 Natural Gas Fertilizers: the production of N, P and K biofertilizers was assumed replacing the production of the conventional fertilizers N, P_2O_5 and K_2O respectively according to a ratio 1 kg : 1 kg. For all the avoided impacts, it was considered the ReCiPe 2008 (hierarchist) method for each impact category as available in the Ecoinvent version 3.6, 2019.	

Scenario #II. It was assumed that electricity generated by CEAGESP's OBP fraction corresponds to the 2.5% of total electricity annually generated at the 'Caieiras' landfill, considering CEAGESP's OBP biogas percentage as a reference. All the existing energy inputs and gas emissions of scenario #I are considered together to the direct emissions of NO_X generated by the power plant and its demand for water and lubricant oil. Indirect

Table 7: Scenarios impacts assumptions

Scenario #I	100% of impacts of landfilling
Scenario #II	(Impacts of Scenario #I) + (Impacts of electricity production) - (Impacts of electricity from the Brazilian grid being replaced by the electricity generated in the landfill)
Scenario #III	(Impacts of donation) + (20% of impacts from Scenario #I)
Scenario #IV	(Impacts of donation) + (20% of impacts from Scenario #II)
Scenario #V	(Impacts of donation) + (20% impacts from scenario #I) - (Impacts of the Brazilian food production being replaced by the donated food)
Scenario #VI	(Impacts of donation) + (20% of impacts from Scenario #II) - (Impacts ofthe Brazilian food production being replaced by the donated food)
Scenario #VII	100% impacts of Biorefinery
Scenario #VIII	(100% impacts of Biorefinery) – (Impacts of Natural Gas and fertilizers production replaced by the Biorefinery products)

emissions due to implementation of power plant are calculated from scientific literature that assessed the Brazilian São João' landfill, also located in São Paulo city and with similar characteristics as the 'Caieiras' landfill (Almeida et al., 2012; Da Silva, 2011). Avoided emissions express the emissions related to the same amount of electricity obtained if it were generated by the Brazilian matrix.

Scenario #III. Modelling the electric logistic train required information from a company that implemented similar systems in Brazil (Still Brazil, 2020), taking into account steel and lead (for batteries) as main vehicular materials, and electricity consumption for its operation phase. The logistic train route was established by including two daily shifts respecting the current OBP collection system. Three electric logistic trains are used in this scenario, each one composed by one 8 ton-capacity tow tractor + trailer constituted by three frames. Each frame can transport two trolleys with their own pallets for six pallets transported by each train. By assuming an equal distribution of the load on the existing 180 pallets, an average load of 325 kg of OBP per pallet per shift is considered, for a total of 1,950 kg per train. By assuming an average train speed of 7.5 km/hour and considering an average distance between the collection points to the checking quality of 1,500 m, to complete the loading and unloading operations, 2 hours per shift are necessary (4 hours/day), reaching a total of 1,452 hour/yr. The food bank is accounted for as its structure (basically steel), by considering a shed of 900 m^2 with measures of 30m length x 30m width x 6 m height, tables for OBP quality checking made of stainless steel, plastic pallets, and the refrigerated cold rooms (steel and polystyrene). Tectermica (2020) was the reference in modelling the refrigerated rooms.

Electricity consumption by the refrigerated rooms is estimated from Evans et al. (2014), assuming the highest provided values due to its representativeness for tropical Brazilian weather conditions and high turnover (100% in 24 hours) for the OBP stocked. Regarding OBP, 80% are considered as edible (data obtained from fieldwork and supported by Fagundes et al. (2014) and Legaspe (2006)), resulting in a recovering rate of 800 kg NMF /ton OBP managed. The remaining 20% is managed according to Scenario #I.

Scenario #IV. This scenario has the same assumptions as described in Scenario #III, but the residual fraction diverted to landfill is managed according to Scenario #II.

Scenario #V. It has the same management assumptions as for Scenario #III, but the avoided impacts related to food donation (Table 3) are included. This is important because donation will avoid food production elsewhere, and the emissions from agricultural production are assumed to be negative or avoided. Data in estimating the avoided emissions comes from Ecoinvent database version 3.6 (2019), ReCipe 2008 method, hierarchist. Open field cultivation and Brazilian values were chosen when available, since they are more representative of the Brazilian conditions, but global market values were used when Brazilian values were not available. Precisely, the following specific procedures were applied: (a) for tomatoes, data on the 'tomato open field production' was chosen because the global (GLO) option of tomato market considers that 50% of tomato is produced in greenhouses, however, tomato production in Brazil mostly occurs in open field; this could have an influence on the estimated emissions, since greenhouses demand higher amounts of resources. (b) Due to lack of data, the emissions available for onion product assumed was that for garlic since they belong to the same agro economic *Liliaceae* family. The same approach was considered for chayote, in which the values for cucumber were considered (they both belong to the *Cucurbitaceae* family); for this specific case, greenhouse-produced cucumber was used as it was the onlydata available. (c) For lettuce, the produce iceberg lettuce was chosen as it reflects what happens in Brazil: production in open fields. The option GLO lettuce considers greenhouses production and could lead to emissions overestimation. (d) For manioc, data representing carrot emissions were considered since they belong to the same *Apiaceae (umbrelliferae)* family. Details are available in Supplementary Material B, 'ecoinvent sheet'.

Scenario #VI. It was modelled based on the same assumptions as Scenario #V, however, the residual fraction diverted to landfill generates electricity, and the related avoided emissions are accounted for, as in Scenario #II.

Scenario #VII: the biorefinery scenario was modelled by considering seven steps (see Figure 22 and Table 8). It accounts for 130 tons of daily by-products as input, which become 104 tons of OBP after the separation of the inorganic fraction. All the accountability was executed on a daily basis and the results multiplied by the yearly working days at CEAGESP (363). The first step, internal collection and transport has the same assumptions as those of scenario #III,the only difference being that 100% of OBP are collected by the logistic train and there is no residual fraction going to landfill. The second step, manual separation, was modelled according to information from Uratani et al. (2014). It included a conveyor belt of 20 m length made of steel (weight 500 kg) with a lifespan of 5 years, with 7.95 kW power and by considering 9.5 daily operating hours for a daily and yearly consumption of 69.26 kWh and 25,141 kWh respectively. In the third step, mechanical grinding, two grinding machines for normal use plus one machine for emergency use in case of maintenance are considered, for a total of three machines. A maximum capacity of 7 ton/hour, a power of 30 kW (40.8CV) of each machine and an average consumption of 4.53 kWh / ton OBP (Uratani et al., 2014) were assumed, for a total daily and yearly consumption of 471 kWh and 170,973 kWh respectively. The three grinders are considered as being made of steel, weighing around 4 tons each, for a total weight of 12 tons and a lifespan of 5 years. The fourth step, anaerobic digestion, takes into account a daily input of 104,000 kg of OBP composed by a dry fraction of 11,846 kg/day and a water amount of 92,154 kg/day, with a total solid percentage of 11.39% and moisture of 88.61%, respectively. The wet digester was modelled according to Francini et al. (2020) and operates at 10% total solid (dry matter), therefore, to achieve this percentage a dilution with an amount of water equal to 14,460 kg/day for a total daily input of 118,460 kg/day is necessary. For the AD process, the amount of thermal energy (th) needed for heating the diluted mixture of OBP from an assumed initial temperature of 20 ºC to 38ºC was calculated by assuming the specific heat capacity of the feedstock as being the same as that of water, for a daily and yearly requirement of 2,478 kWh and 899,514 kWh, respectively. Regarding the Biodigester volume calculation, it was assumed a retention time (RT) of 20 days (Francini et al., 2020), an input density equal 1 ton/m^3 as the density of the water (due to 90% moisture), a biogas buffer of 15% (Uratani et al., 2014) and a security buffer of 10%, for a total volume of 3,000 m^3. The biodigester is vertical, one stage type, with an approximately cylindric shape, modelled as a tank with 18.98 m diameter, 12 m heigh, made of stainless steel and with 87,906 kg net weight. The daily and annual electricity consumption corresponds to about 188 kWh/day and 68,404 kWh/year respectively. By considering a ratio of volatile solids fraction (VS)/dry matter CEAGESP OBP equal to 0.906 (Culi, 2018) and a specific biogas production of 0.589 Nm^3 per kg/VS (Francini et al., 2020). A daily and yearly biogas production of 6,321 Nm^3/day and 2,294,523 Nm^3/year was estimated, composed by 60% CH_4, 40% CO_2, 250 ppm H_2S + traces of other gases (Francini et al., 2020), resulting in a loss of 4,172 kg VS / day and with a percentage removed VS / total VS of about 70%. The H_2S was assumed to be removed through microareation, which, according to Jeníček et al. (2017), is a mature technique, cheap and highly efficient technology to allow for the biological oxidation of H_2S to elemental sulfur up to 99% efficiency. By following Jeníček et al. (2017) information, the amount of air required was assumed to be about 1% of the raw daily biogas generation, about

64 Nm^3/day. Regarding the raw digestate generation, a daily amount of 110,786 kg digestate by assuming 100% water transfer to digestate was calculated, with a residual fraction of dry matter of 4,172 kg/day (3.77%), for a total raw digestate yearly generation of 40,215,318 kg/year. The fertilizers content per ton OBP was estimated by considering the work of Tampio et al. (2014) as reference, using a conversion factor of 45.56% related to the lower amount of dry fraction per ton OBP in this present work when compared to Tampio et al. (2014). The daily amount of recovered N, P and K was estimated in 355 kg, 43 kg and 133 kg respectively that correspond to an annual amount of 128,999 kg N, 15,480 kg P and 48,159 kg. The fifth step, water scrubbing was modelled by considering a daily input of 4,965 Nm^3 equivalent to the 78.5% of the raw biogas generation. The plant was assumed to have a 230 Nm^3/hr maximum capacity with an electricity consumption of 0.3 kWh/Nm^3 raw biogas (Bauer et al., 2013) with a daily and yearly energy requirement of 1,490 kWh/day and 540,870 kWh/year. The water consumption was estimated as 2.5 m^3/day (Bauer et al., 2013) that corresponds to a 908 m^3/year, while the equipment was supposed as made of steel, with a weight to 3,526 kg and a lifespan of 20 years (Lorenzi et al., 2018). A daily Biomethane production (with 97% concentration) of 3,128 Nm^3/day for an annual amount of 1,135,464 Nm^3/yr was estimated. The sixth step, solid – liquid separation was modelled by assuming the use of a centrifuge with an electricity consumption of 3.5 kWh/ton OBP (Tampio et al., 2014) for a daily and yearly consumption of 388 kWh/day and 140,844 kWh/year, respectively. A generation of 11,078 kg/day of solid digestate and 99,708 kg/day of liquid digestate was estimated. This digestate was assumed to be collected by the wholesalers and sent back to the countryside during the regular return trips, thus closing the nutrient cycle. By estimating a daily trucks circulation of 2,000 vehicles at CEAGESP, 2,000 drums of 50L and 2,000 plastic containers of 6 L, both made by HDPE are necessary to collect the entire amount of liquid and solid digestate, respectively. The wholesalers use the digestate at the cropland and send back the empty container during the next trip to CEAGESP, according to a circular economy management. The seventh and last step is electricity and heat production at the CHP facility associated with the biorefinery, modelled according to information from DBEIS (2021), Fusi et al. (2016) and Kelly et al. (2014). To cover Biorefinery electricity requirement (by including a 10% buffer) a quantity of biogas equal to 1,356 Nm^3/day by considering an electric efficiency of 0.36 (Probiogas, 2015) and a Low Heating Value of biogas of 6 kWh/Nm^3 (SGC, 2012) to generate 2,929 kWh/day are required. Regarding heat generation, an efficiency of 0.48 (Probiogas, 2015) with a daily heat generation of 3,905 kWh/day was assumed. To cover these requirements, two CHP plants of 100 kW power each were chosen, made of steel (4 tons each, 8 tons total) and a 25-year lifespan. Direct emissions of NO_X, CH_4, NMVOC, CO, N_2O and PM_{10} were estimated by considering average values per GJ of biogas (Kristensen et al., 2004).

Scenario #VIII: this scenario was modelled with the same assumptions of scenario #VII, the only difference being the inclusion of avoided impacts. In particular, biomethane with a concentration of 97% was modelled to able to replace natural gas production according to 1:1 coefficient substitution, while N, P and K replace conventional nitrogen N, P_2O_5 and K_2O respectively.

Table 8: Biorefinery process operative schedule per daily input (130-ton by-products / day -> 104 ton OBP).

Step Number	Step name	Duration*	Input	Output	Consumed electricity kWh_{el}	Consumed heat kWh_{th}
1	OBP collection	4 hrs	130,000 kg BP	130.000 kg BP	60	
2	Manual separation	9.5 hrs	130,000 kg BP	104,000 kg OBP; 26,000 kg RF[1]	69	
3	Grinding	7.5 hrs	104,000 kg OBP	104,000 kg OBP	471	
4	Anaerobic Digestion	20 days	104,000 kg OBP; 14,460 kg H_2O	6,321 m^3 Biogas (7,674 kg VS); 110,786 kg raw digestate	188	2478
5	Water Scrubbing	Cont.	4,965 m^3 Biogas; 2,500 kg H_2O	3,128 m^3 CH_4 (97%); 1,837 m^3 CO_2; 2,500 kg H_2O	1490	
6	Raw digestate centrifugation	1 hr	110,786 kg raw digestate	11,078 kg Solid Digestate; 99,708 kg Liquid Digestate	388	
7	Co-generation	Cont.	1,356 m^3 Biogas	2,933 kWh_{el}; 3,905 kWh_{th}		

* Estimated time for each step. VS: volatile solids; RF1 = residual inorganic fraction.

For steps details see notes in Appendix B, section B12.

5.1.2 Consistency verification for LCA inventory analysis

Since diesel demand, added to the amount and characteristics of biogas and leachate generation are the main variables involved in most options for organic waste management, their values were checked for consistency. For scenario #I, diesel consumption for internal OBP collection (3.09 L/ton OBP) is in accordance to values presented by Larsen et al. (2009). The OBP transport to landfill required 0.56 L/ton OBP, consistent with Larsen et al. (2009) and Buratti et al. (2015). Biogas production from OBP organic waste degradation (54.86 Nm^3/ton OBP; 39.16 kg/ton OBP) has shown consistent values with Buratti et al. (2015) who evaluated a landfilling scenario in Italy, Candiani and Torres (2015) who assessed biogas composition at the ‘Caieiras’ Landfill, and with Mendes et al. (2004) who have assessed the environmental impacts of São Paulo’s municipal solid waste incineration versus landfilling. The obtained value of 0.44 m^3/ton for leachate is consistent with Fernandez-Nava et al. (2014), while values for leachate’s biological oxygen demand (BOD) of 500 mg/L and chemical oxygen demand (COD) of 27,500 mg/L are typical values for mature landfills in the methanogenic phase between 10-20 years of activity (Costa et al., 2019). For scenario #VII the value of biomethane potential production modelled in this study (0.353 m^3_{CH4} /kg VS) is within the range of the values for anaerobic digestion plants that use municipal solid waste or wastewater as main input (Holliger et al., 2017), and consistent with the value (0.313 m^3_{CH4} /kg VS) found by Silva Junior et al. (2022) who analyzed the biomethane production of fruit and vegetables waste of the wholesaler food supply center of Maracanau, Ceará State, in Brazil, a similar case study.

5.1.3 Comparative analysis among scenarios: focusing on the LCA impact categories

The performance of the scenarios for the nine impact categories, as shown in Figure 29 generally matches the waste hierarchy management concept, especially when the avoided impacts are included. In particular: (i) food donation scenarios that include avoided impacts (scenarios #VI and #V) showed considerable negative values that correspond to high environmental benefits in all the categories; (ii) biorefinery scenario that considers avoided impacts (Scenario #VIII) is in an intermediate position; (iii) scenarios #III and #IV that include NMF donation without accounting for the avoided emissions related to donated food usually depict a worse performance than the biorefinery scenario without avoided emissions (scenario #VII); (iv) scenario #II was the second worst scenario with the exception of FDP, HTP, POFP and WDP impact categories, while scenario #I was the worst in all categories, except for POFP and WDP.

Regarding grouping and reciprocal positioning, Figure 29 shows that for FEP, HTP and WDP, scenarios are divided into two groups, with considerable differences between them, constituted by scenarios from #I to #IV and #VII to #VIII in the first group with higher environmental burdens, and scenarios #V and #VI as the second group with lower (negative) environmental burdens. This distribution highlights that accounting for avoided impacts of donated food in scenarios #V and #VI highly impacts the results, while the effects of the avoided impacts are less evident for biorefinery scenario (scenario #VIII) and negligible in electricity production (scenario #II). In FDP and MDP the scenarios are distributed in three areas where scenarios from #I to #IV, jointly with scenario #VII, show the highest environmental impacts, scenario #VIII has an intermediate position while scenarios #V and #VI depicted hightly negative environmental burdens. This disposition highlights again the best performance of scenarios that include avoided impacts derived by donated food (#VI and #V), but, at the same time, show a considerable effect of the avoided emissions related to conventional production replaced by biomethane and fertilizers (scenario #VIII). The same distribution in three areas is also recognizable in GWP, PMFP and TAP, but with some differences when compared to FDP and MDP. In fact, besides scenarios #VI and #V largely confirming their best environmental performance and scenario #VIII the intermediate position, a better performance by scenarios #III, #IV and #VII (intermediate position) against scenarios #I and # II is evident. These results indicate the advantages of avoiding organic waste generation and its related downstream emissions, regardless of the benefits derived from accounting for the avoided emissions. The impact category POFP shows a peculiar distribution as, besides the lowest environmental burdens depicted by scenarios #V and #VI, all the scenarios with electricity production (#II, #IV and #VI) show a worse performance, when compared with the corresponding scenarios minus electricity production (#I, #III, #V), highlighting the important role of electricity generation as a source of pollution in this impact category.

This general overview depicts donation scenarios with avoided impacts as the best options, due to the results in all categories assessed, followed by biorefinery scenario #VIII, while Electricity generation at landfill does not show a significant improvement, from an LCA perspective.

5.1.4 Comparative analysis among scenarios: focusing on specific inputs and outputs

A detailed comparative analysis for scenarios performance for each impact category is provided by the combination of results shown by Figure 29 and the role of the first contribution to the impacts in each impact category shown in Figure 30. The first contribution for each impact category is calculated by considering the worst scenario as reference and verifying which process is the first cause of the impact. Regarding FDP, scenario #I has the highest environmental burdens (8.31 kgoil/tonOBP), scenarios #V and #VI have the lowest impact, while scenario #VIII is collocated in an intermediate position, but closer to scenarios #V and #VI. Besides the previously highlighted effect of food (scenarios #V and #VI) and natural gas (scenario #VIII) replacement, it is interesting to note that when accounting for electricity generated as avoided emissions (scenario #II), a better performance for FDP than donation scenarios #III and #IV and biorefinery scenario (#VII) results. This is caused by the characteristics of the Brazilian electricity mix generation (the largest part of which being obtained from hydropower, but still using a fraction from thermoelectric plants) that is being saved because of the electricity obtained at the landfill. The first contribution in this impact category (76%) is the amount of diesel consumed during the OBP transport steps to landfill. The modelled scenarios avoid this transport, and, despite the worse performance of scenarios #III, #IV and #VI when compared to scenario #II, a better environmental performance is evident when the avoided production impacts are included (scenarios #V, #VI and #VIII).

As expected, the FEP has a similar behavior to that of the waste hierarchy management concept, since scenario #I showed the highest environmental impacts, which is very close to those in scenarios #II to #IV and scenario #VII. Scenario #VIII shows slightly better results, while scenarios #V and #VI have by far the highest performance. In the worst scenario (#I), the direct emissions of phosphorus on water have high influence on its FEP (73%) result, while for other scenarios, FEP is balanced by the avoided emissions, especially when the avoided emissions of food production are included (scenarios #VI and #V).

GWP results depicted three different groups. Again, those scenarios exclusively concerning landfilling and/or energy recovery showed the highest CO_2 equivalent emissions (203 $kgCO_2$eq./tonOBP and 173 $kgCO_2$eq./tonOBP for scenarios #I and #II respectively), in which the electricity generation shows negligible GWP reduction. Scenarios #III, #IV and #VII (41.4 $kgCO_{2eq}$./tonOBP, 35.4 $kgCO_{2eq}$./tonOBP and 4.09 $kgCO_{2eq}$./tonOBP respectively) highlight the considerable savings obtained by accounting for the avoided downstream emissions at landfill, since about 88% of CO_2eq originates from the methane released during the waste degradation at landfill and wastewater plant, therefore, simply avoiding landfill disposal generated important results also when avoided production emissions are not included. It is easy to note that Biorefinery scenario #VII shows a better performance than donation scenarios #III and #IV, differently from what is expected, according to the FRH. This is due to the residual fraction (20%) sent to landfill in these scenarios, responsible for about 98% of CO_2eq emissions. When avoided emissions are included, the substitution of natural gas and fertilizers (scenario #VIII) generates savings equal to -47.66 $kgCO_2$eq/ton OBP, on the other hand, when the avoided emissions of donated food replacement are accounted for, the saving achieves the highest values, equal to -314 $kgCO_{2eq}$./tonOBP and -320 $kgCO_{2eq}$./tonOBP in scenarios #V and #VI respectively. Focusing on MDP, scenario #I has shown again, the highest impact (1.87 $kgFe_{eq.}$/tonOBP), closely followed by scenarios #II to #IV and scenario #VII, scenario #VIII depicts an intermediate performance, while scenarios #V and #VI indicates considerable lower environmental burden in this category. Interesting to note that slight improvements for MDP in scenarios #III, #IV and #VII are related to the amount of metals (lead and steel) used in the logistic train, the infrastructure for quality checking, the refrigerated rooms or in scenario #VII the biorefinery infrastructure, that replaced the vehicles and power plant materials existing in #I and #II.

The categories PMFP and TAP show a similar trend, with scenarios distributed in three groups: scenarios #I and #II have depicted the highest environmental burdens, #III, #IV, #VII and #VIII have an intermediate position, while #V and #VI have the lowest impacts. Differently from most other categories, PMFP and TAP indicate considerable improvement for scenarios #III, #IV and #VII, when compared to #I and #II. This is the result of replacing diesel vehicles in #I and #II with electrical vehicles in #III, #IV and #VII, once diesel emissions correspond to 93% and 96% for PMFP and TAP, respectively. Among the intermediate performances, scenario #VIII shows the best results due to the sum of avoided downstream emissions and avoided Natural Gas and fertilizers production emissions.

The benefits of replacing diesel vehicles with electric ones is also observed for POFP category, but at small rates, due to a lower contribution from diesel emissions over the total (59% in #I and 36% in #II). Scenarios showed similar performance for POFP, compared to all other previous impact categories (worst case for #II and #I, intermediate #III, #IV, #VII and #VIII, best case for #V and #VI). Nevetherless, all scenarios that considered electricity production have obtained a worse performance when compared with the correspondent scenarios without electricity production. This is due to the large amount of NO_x emitted by the biogas-based power plant in the landfill, responsible for about 39% of all NO_x emitted by scenario #II.

HTP shows scenario #I with the worst performance (2.30 kg1,4-DCBeq./tonOBP), closely followed by scenarios #III, #VII, #IV and #II, with electricity generation in #II leading to lower environmental burdens than donation scenarios #III and #IV and biorefinery scenario #VII. This performance can be justified by the high demand for materials such as steel and lead by scenarios #III, #IV and #VII. Scenario #VIII shows a slightly lower environmental burden due to natural gas and conventional fertilizers avoided emissions while Scenarios #V and #VI showed the lowest impacts, in which the avoided food production emissions results in the highest HTP savings (−113 kg1,4-DCBeq./tonOBP in #VI). Finally, scenarios distribution for WDP present a slightly different behavior than those in the other impact categories. Scenario #VII shows the worst performance (0.18 m^3H_2O / ton OBP) closely followed by scenario #I, #III, #IV and #VIII. Scenario #II depicts slightly better performance (-3.08 m^3H_2O/ton OBP) while scenarios #V and #VI are by far the best ones (-72.02 m^3H_2O/ton OBP in #VI). These results are influenced by two factors: the amount of water used by the biorefinery plant, especially for OBP dilution inside the biodigester, and the Brazilian electricity matrix, which is mainly based on hydropower plants. Therefore, the biorefinery scenarios show a worse performance due to higher water demand and the scenarios that generate electricity showed better performance, since they are avoiding the demand of water by the Brazilian power plants.

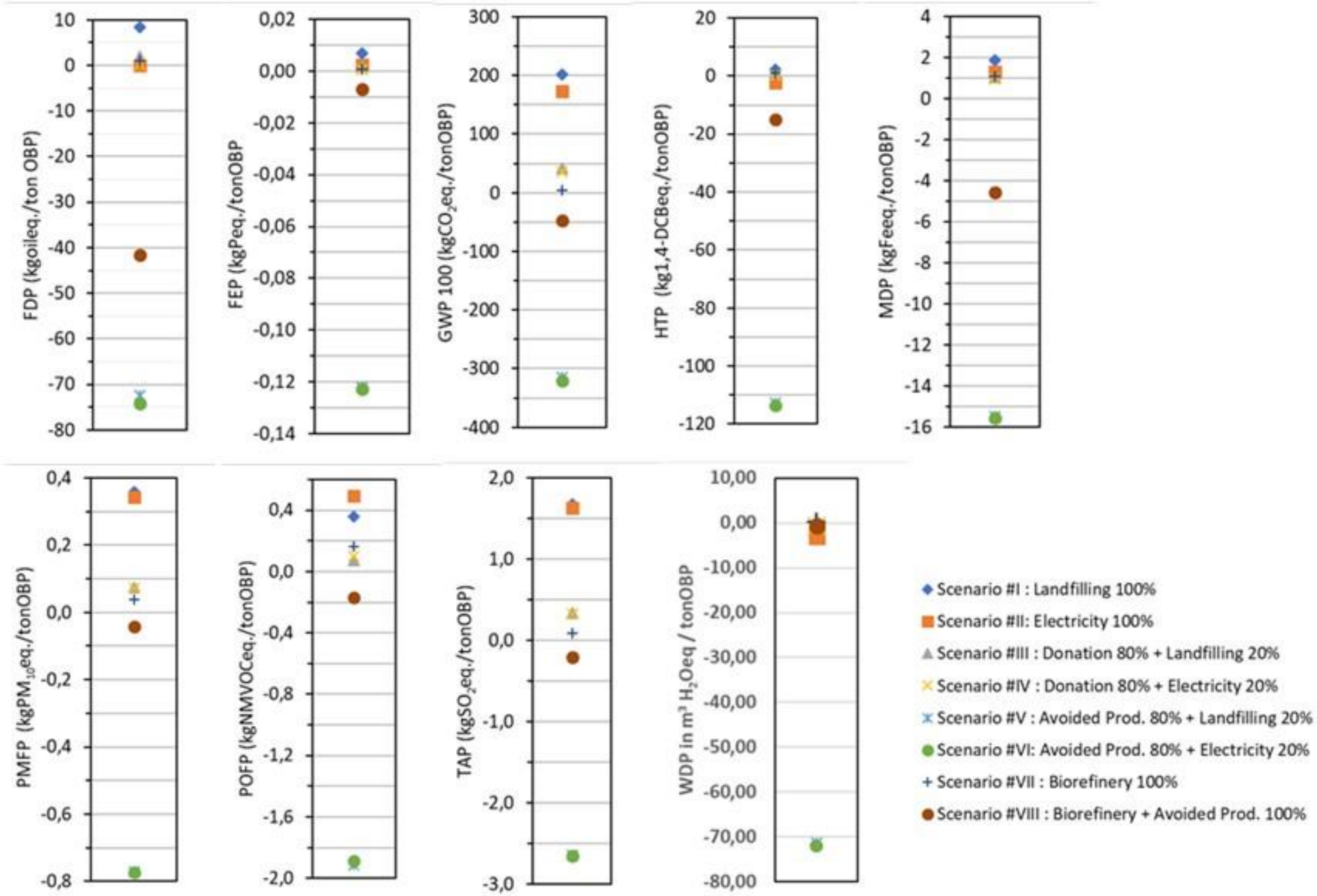

Figure 29: LCA results for the evaluated scenarios under nine impact categories

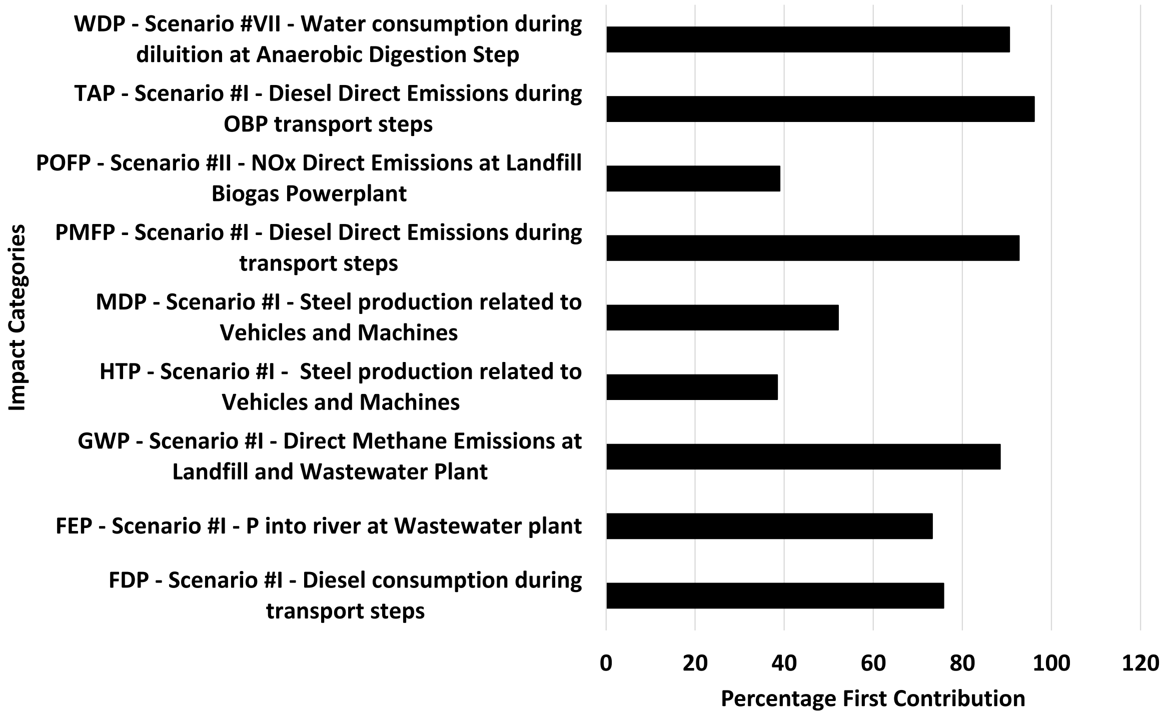

Figure 30: First contribution (%) to the impacts in the worst scenarios for each impact category

5.1.5. Relative overall comparative analysis

The environmental impacts of each scenario were compared in relation to the best result for each impact category (Figure 31), the values of which having been set as equal to ‘1’. Thus, the values in logarithmic scale show how many times the environmental burdens of one scenario is higher (i.e. it causes higher environmental impact) compared to the best-case scenario. Figure 31 shows higher differences or improvements related to GWP, in which landfilling the OBP (scenarios #I and #II) impacts approximately 500 times more than donating food with electricity recovering (scenario #VI), while biorefinery depicts an intermediate performance, showing a value of 325 and 273 times worst for scenarios #VII and #VIII, respectively. The improvements are also important in the HTP and FDP impact categories, in which scenarios #I to #IV impacts approximately 117 and 83 times more than scenario #VI, while scenario #VIII depicts an intermediate value of 34. Under an overall comparative perspective, Figure 31 indicates that scenario #VI has better environmental performance for almost all impact categories, differing in quantity among them. This result emphasizes that NMF donation and, as second choice, biorefinery scenario, should be prioritized by public policies at CEAGESP.

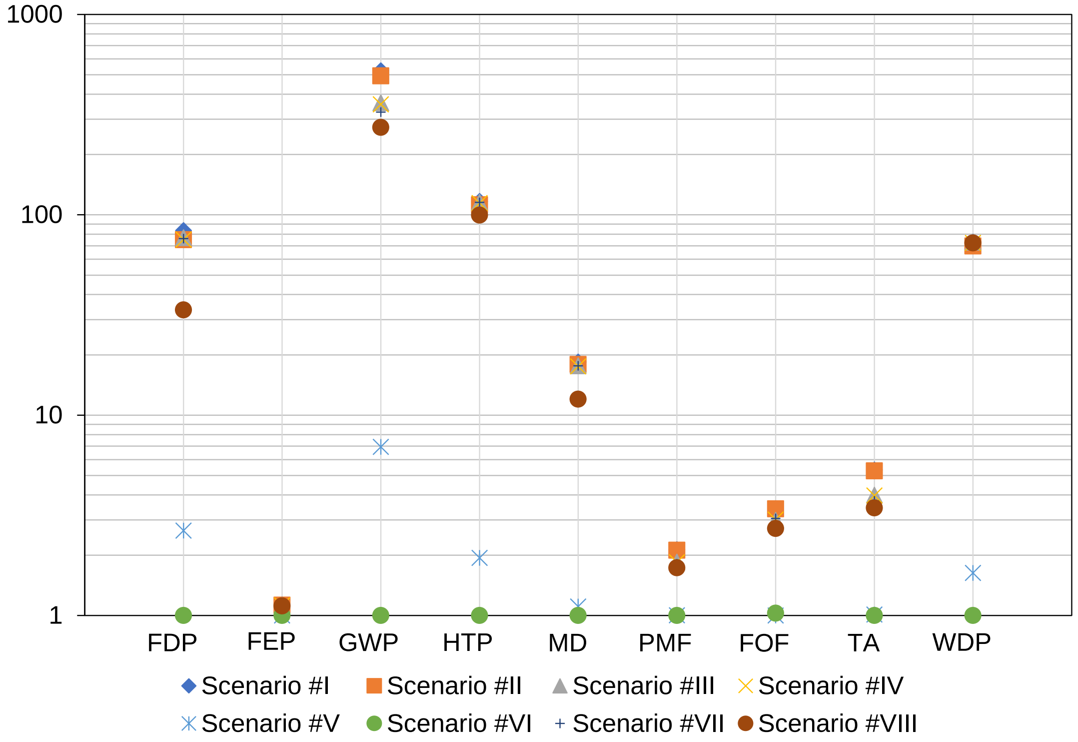

Figure 31. Comparative analysis for the environmental impacts of scenarios based on the best-case performances. The values (in logarithmic scale) indicate how many times worse an impact category is, compared to the best case.

5.1.6. Normalized comparative analysis

According to Oliveira et al. (2017), a lack of accurate information representing the Brazilian specificities is recognizable in case of normalization. Nevertheless, to allow for a more direct comparison among the different impact categories, a normalization approach is implemented by considering the values of impacts per person per year (global values), provided by the ReCiPe 2008 midpoint (Hierarchist) method (Goedkoop et al., 2009) as reference.

Figure 32 depicts the results of the evaluated scenarios, expressed in person equivalent per year (p.e.yr). Results show, in scenario #I, that most relevant impact categories are TAP, GWP, FEP and PMFP. It's interesting to note these categories are related, as previously discussed, to the direct impacts derived by emissions at landfill (GWP and FEP) and direct emissions derived by diesel combustion during transport steps (PMFP and TAP). The importance of these impact categories was also recognized in literature. For example, Ripa et al. (2017) showed that FEP, TAP and GWP were the first, the fourth and the fifth most important impact categories, respectively, in their case study, by using the same LCA method as the one used in this present study. Brogaard et al. (2013) have found that global warming, marine and terrestrial eutrophication, and particular matter were relevant in their case study, and the importance of global warming was also highlighted by Buratti et al. (2015) and Damgaard et al. (2011).

Scenario #II has not depicted important changes when compared to the baseline, while Scenarios #III, #IV and #VII present a considerable impacts reduction linked to avoided landfilling.

Scenario #VIII shows negative impacts in all the assessed categories, Negative impacts in biorefinery scenarios for biomethane and bioethanol production were also found by Ardolino et al. (2018) and Papadaskalopoulou et al. (2019).

Scenario #VI shows the best performance, with evident emissions savings especially in FEP and HTP. These savings are related to food donation that replaces Brazilian food production. The high performance in these two impact categories depends on the avoided food production and related savings regarding the use of fertilizers (FEP) and pesticides (HTP), human toxicity and eutrophication being among the most important problems of crop production, as also highlighted by Alhashim et al. (2021) and Ritchie and Roser (2022).

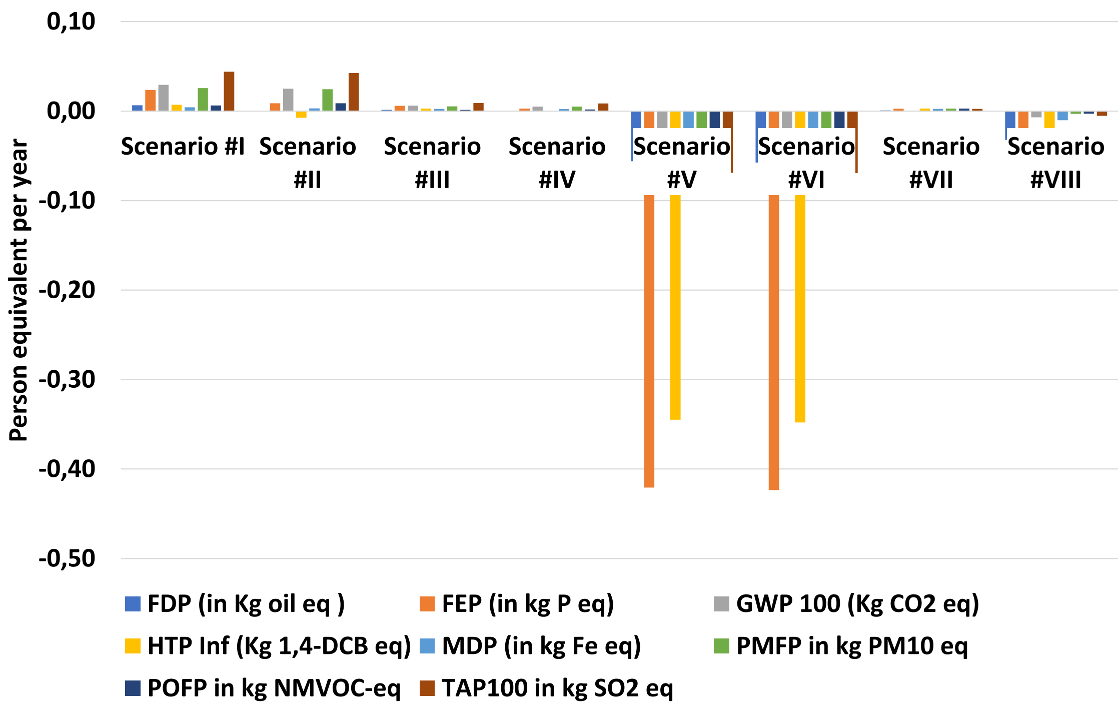

Figure 32: Normalized values for the assessed scenarios expressed in person equivalent per year (p.e.yr).

5.1.7 Comparison with previous studies

Scenarios for the OBP were modelled according to existing practical and operational real potentialities in implementing them, considering the waste hierarchy management concept as the backbone. As presented previously (Figures 29, 30, 31, 32), results show that usual practices such as landfilling, with or without energy recovery, have higher environmental impacts than all other OBP donation and biorefinery scenarios. Conversely, a better performance is obtained when considering all the avoided impacts, both in case of food donation and biorefinery scenarios, at different rates. Food donation and biorefinery showed lower impacts even when only accounting for the avoided emissions in landfilling. Electricity production from biogas usually showed lower environmental impacts than solely landfilling, but without relevant improvements. However, in scenarios with two different options jointly used (scenarios #III to #VI), electricity recovery generates a slight reduction in environmental burdens on most impact categories evaluated.

Few studies have evaluated the highest levels of FRH donation and biorefinery scenarios included, and according to the literature review developed in this work, no studies were found that include both options under an LCA perspective. This study has confirmed the consistency of FRH, in which food donation should have the highest priority, while industrial use should be the third most recommended option. Nevertheless, this work has also depicted that biorefinery showed lower environmental impacts when compared to traditional landfilling, with or without energy recovery alternatives.

Regarding the assessment of scenarios distribution along the food recovery hierarchy, and focusing on the related position of donation scenarios, the patterns found in this present study are consistent with Albizzati et al. (2019), who analyzed several food surplus scenarios in the French retail sector through a LCA perspective. Authors have found that food waste prevention was the best-case scenario, followed by the current scenario constituted by almost 100% food donation pathways with a negligible percentage recovered as animal feed, while the other waste management scenarios assessed as anaerobic digestion and incineration were clearly the worst options.

Eriksson et al. (2015) compared the outcome of a LCA-GWP of differentfood waste management scenarios available to supermarkets in Uppsala, Sweden. Six scenarios were considered according to the FRH: landfilling, incineration, composting, anaerobic digestion, feeding animals, donations, while five kinds of products were selected for the analysis: bananas, iceberg lettuce, grilled chicken, stewing beef and wheat bread. The results showed a decreasing GWP trend from higher to lower priority FRH levels. For all products, landfill was the option with the highest greenhouse gas emissions. On the other side, donation and anaerobic digestion were the alternatives with the lowest greenhouse gas emissions, with some differences related to products characteristics. Their results confirm the FRH concept and the findings of this present study. Eriksson and Spangberg (2017) also identified a similar trend of GWP increase from the highest levels (donation and conversion) to the lowest levels (incineration and anaerobic digestion) of the waste management hierarchy concept.

In Brancoli et al. (2020), authors focused on several management options for the surplus bread production in Sweden. The obtained LCA results showed that reducing bread waste was the option with lower impacts, followed by feed production, donation, beer, and ethanol production. Anaerobic digestion and incineration showed the highest environmental burdens.

Sundin et al. (2022) compared GWP performance between donation and anaerobic digestion in Sweden, and the results showed that donation has lower environmental burdens.

In Cakar (2022), who analyzed fresh food and vegetables redistribution as donation in Istanbul's supermarkets compared with landfilling, composting, and anaerobic digestion alternatives, food donation depicted better performance for GWP than landfilling and composting, but worse than anaerobic digestion. Regarding energy and water consumption, composting and anaerobic digestion depicted the worst performance, while food donation showed an intermediate performance, however, all always better than landfilling.

The environmental burdens of OBP management practices are influenced by product substitution, which is a variable that depends on regional characteristics that can influence the results, especially for some impact categories. The Brazilian electricity matrix and its influence on WDP and GWP is an example.

Focusing on WDP, disregarding the worst performance of scenario #VII derived by modelling choices, our study highlights lower performance of WDP impact category for donation (scenarios #III and #IV, when avoided food production impacts are not included) than energy recovery, an option that receives lower priority in the waste management hierarchy. This unexpected result is due to the hydropower-based electricity in Brazil (~75%), a very specific condition for the Brazilian case that affects LCA results and emphasizes the importance of implementing LCA studies in different countries to detect local specificities. The trend detected for WDP depends on the joint effect of two factors: the biogas-based electricity generated in some of our scenarios demands lower amounts of water than the Brazilian hydropower-based plant, at the same time, the donation scenarios utilize electricity from the Brazilian grid that requires a higher amount of water. A similar behavior related to marginal electricity replacement in WDP was also found by Abizzati et al. (2019) in France, since it depends on high amounts of water for cooling nuclear plants (80% of France electricity matrix).

The influence of the Brazilian electricity matrix in environmental studies was also identified by other authors. Assessing several low-prioritized municipal solid waste management alternatives in São Paulo city, Linkanen et al. (2018) found irrelevant improvements on GWP when electricity production was considered. Comparing waste incineration and landfilling alternatives in São Paulo city, Mendes et al. (2004) showed that electricity production resulted in an insignificant reduction of environmental impacts. Furthermore, Mendes et al. (2004) stated that a reduction of environmental burdens could be obtained only through a change in the solid waste management, including alternatives according to the waste hierarchy management concept; this is exactly what has been considered in the scenarios established in this present work, and confirmed by the obtained results.

Regarding biorefinery scenarios, a limited number of works (Ardolino et al., 2018; Chester and Martin, 2009; Ebner et al., 2014; Guo et al., 2021; Kalogo et al., 2007; Papadaskalopoulou et al., 2019; Stichnothe and Azapagic, 2009) have considered the biorefinery pathway as a possible alternative in OBP management, by comparing it with traditional methods.

Ardolino et al. (2018) compared biomethane production scenarios from OFMSW (with total, partial or without electrical self-sufficiency) with traditional anaerobic digestion technology with electricity and heat generation. The normalized results showed that the three most important affected impact categories are GWP, NREP (non-renewable energy potential), and RINP (respiratory inorganics potential). The total values for each impact category are negative (for GWP and NREP) or about zero. All scenarios with biomethane production are always better than that of exclusively generating energy, mainly in terms of GWP and NREP. Therefore, industrial use implemented through biogas refining to biomethane resulted in lower environmental burdens than those by traditional anaerobic digestion.

Chester and Martin (2009) assessed MSW to ethanol compared with landfilling in California. The authors conclude that ethanol production from MSW cannot be unequivocally justified from the perspective of net-GHG avoidance. It is possible that diverting feedstock from burial could avoid net GHG emissions if gas recovery at landfill is not efficient, otherwise it is not an option. Authors affirm that because the total system considers emissions that do not occur as a result of avoided landfill decomposition, it is appropriate to consider the additional emissions that result from the combustion of ethanol. This is different from the approach developed in this study, which considers CO_2 emissions neutral, since it is biogenic. Therefore, due to different approaches regarding emissions accountability, further comparison is not possible.

From a GHG-LCA perspective, Ebner et al. (2014) assessed a biorefinery with Bioethanol and animal feed production from food scrap waste of a supermarket chain, simultaneously with diluted fruit syrup derived by food processing waste. Authors compared the GHG emissions of food waste to ethanol pathway with the traditional landfilling (with and without landfill gas capturing) and composting. Results show that the biorefinery process has lower GHG emissions than all landfilling scenarios, while compost depicts a better performance than biorefinery scenarios. Therefore, despite the different type of biorefining pathway, Ebner et al. (2014) results confirm our findings regarding less GHG emissions of a biorefinery scenario, when compared to landfilling.

Guo et al. (2021) have compared the GHG emissions of different biorefineries pathway with traditional anaerobic digestion in China. Results confirm the findings of Ardolino et al. (2018) by emphasizing that a biorefinery with biogas upgraded to biomethane has higher GHG savings than traditional AD with energy production. Conversely, biowaste to bioethanol shows a worse performance.

Kalogo et al. (2007) have modelled a MSW – to ethanol facility and implemented comparisons under a life cycle energy use and air emissions perspectives. Regarding GHG, the authors affirm that emissions in landfilling waste with gas recovery (either for flaring or for energy production) result in greater net savings in GHG emissions compared to the biorefinery system, that is in contrast with the findings of our study and the work of Ebner et al. (2014).

Papadaskalopoulou et al. (2019) assessed, through an LCA perspective, a waste to ethanol biorefinery system versus conventional waste management methods in the Attica region, in Greece, comparing this pathway with landfilling with energy recovery (current method applied for mixed municipal waste in the study area); (II) windrow composting (current method applied for biowaste in the study area); (III) anaerobic digestion; (IV) incineration. The biorefinery system presented better performance against almost all impact categories compared to landfilling, confirming the findings of this present study, while composting has shown relatively higher emissions in the categories terrestrial acidification, terrestrial eutrophication and particulate matter, which are related to the air emissions from the composting process. From a general point of view, biorefinery, anaerobic digestion and incineration showed the best environmental performances, followed by composting, while landfilling is the worst one.

Stichnothe and Azapagic (2009) examined the GWP of an integrated waste management scenario for the management of MSW in UK. Two main pathways were considered: the treatment of the biodegradable fraction of MSW with combined gasification/bio-catalytic process for the production of ethanol and other byproducts (butanol, electricity) and traditional MSW management in UK. The ethanol scenario also includes the recycling of collected recyclables (15% of MSW) and the incineration/landfilling of remaining waste. The ethanol scenario is compared to the baseline situation, according to which the majority (70%) of MSW is landfilled, the collected recyclables are recycled (15% of MSW) and the remaining waste (15% of MSW) is incinerated/composted. The results depict that the scenario with ethanol production has a better GHG performance, with negative total net-emissions, when compared to traditional scenarios.

Although recognizing that quantitative comparisons among different studies could be a better way to identify advantages and disadvantages among them, most studies found in the scientific literature have considered different LCA methods, characterizations factors and units of measure, which would lead to wrong interpretations when performing direct comparative results. Moreover, most of the assessed studies are limited to GWP category, which does not allow a 360-degree evaluation of all environmental complexities. Nevertheless, as an attempt to deeply discuss the obtained results, a numerical comparison exclusively for the GWP impact category (including the avoided emissions) is provided, due to its current worldwide importance. Table 9 shows the results of the comparison among this work and similar studies in literature by considering, when applicable, donation, biorefinery (bioethanol or biomethane), anaerobic digestion and landfilling with related avoided impacts of substituted products.

Primarily, the numbers are highly dependent on the system boundaries and replaced products, highlighting the importance of authors' assumptions, LCA methods applied, and regional characteristics. The -320 kgCO_{2eq}/ton OBP of donation scenario found for CEAGESP are within the range of values related to donation found by Eriksson et al. (2015). This result is consistent with the kind of products being assessed in this present study (100% fruit and vegetables), while the higher values are related to bread, meat, and other products that demand more processes and energy in the production chain.

The GWP tends to decrease from the bottom to the top of the FRH and, interesting to note that all the values from the intermediate-low position (anaerobic digestion) to the most recommended options are, at different rates, negative, while only landfilling shows positive values. This confirms the importance of avoiding landfilling to obtain net CO_{2eq} savings. With the exception of the study by Brancoli et al. (2020), in which bioethanol has a better performance than donation (-560 kg$CO_{2\ eq}$/ton vs – 450 kg$CO_{2\ eq}$/ton), the FRH is usually respected by all other studies considered in the literature review, from higher savings for donation scenarios to no savings at all in landfilling with energy recovering. The particular behavior depicted in Brancoli et al. (2020) probably depends on the low moisture of the bread, around 40% (Ishida and Steel, 2014), when compared to other inputs as OFMSW in Guo et al (2021), which show an average moisture of 80%. The water percentage also could have an influence in the biomethane scenario carbon savings identified in this work, and by Guo et al. (2021), being around 90% and 80% respectively. Landfilling always showed positive emissions, also in the case of biogas capture and energy recovering. The relative low values found in herein (173 kgCO_2eq/ton in case of energy recovering and 203 kgCO_2eq/ton without energy recovering) are related to the high percentage of landfill biogas captured (80%) and flared, even when electricity production is not considered. Nevertheless, the numbers found in this work are close to the minimum values of Eriksson et al. (2015) and Papadaskalopoulou et al. (2019). Higher emissions correspond to landfills with lower or no gas capturing.

Overall, the obtained results indicate a high potential of food donation and biorefinery scenarios in reducing environmental burdens, with donation showing, by far, a better performance than biorefinery. Obtained results herein are consistent with other studies in the scientific literature that highlight the importance of firstly trying to implement those higher levels for waste management practices, rather than simply recovering energy in landfills. Additionally, results claim attention to the influence of local/regional specificities on LCA performance for the studied scenarios, emphasizing the need for local studies to support effective public policies.

Table 9: Net global warming potential for different OBP management pathways

Study	OBP Type	Unit	Donation	Biorefinery Bioethanol	Biorefinery Biomethane	Anaerobic Digestion	[1]Landfilling
Brancoli et al. (2020)	Bread	Kg CO_{2eq}/ton	-450	-560	n.a.	-20	n.a.
Eriksson et al. (2015)[2]	Banana, Chicken, Lettuce, Beef, Bread	Kg CO_{2eq}/ton	(-13 - 26,000)	n.a.	n.a.	(-47 - 670)	(210 to 3,100)
Eriksson and Spångberg (2017)	Banana, Tomato, Apple, Orange, Pepper	Kg CO_{2eq}/ton	(-500 - 690)	n.a.	n.a.	(-62 - 230)	n.a
Ebner et al. (2014)	Food Scrap waste and diluted fruit syrup	Kg CO_{2eq}/ton	n. a.	-30	n.a.	n.a.	(375 to 1,576)
Papadaskalopoulou et al. (2019)	OFMSW	Kg CO_{2eq}/ton	n. a.	-15	n.a.	n.a.	223
Guo et al. (2021)	OFMSW	Kg CO_{2eq}/ton	n. a.	-25	-134	-75	n.a.
This Study	Fruit and Vegetables	Kg CO_{2eq}/ton	-320	n.a.	-48	n.a.	(173 to 203)

n.a. = not applicable
1. In Eriksson et al. (2015) landfilling without energy recovery, in Papadaskalopoulou et al. (2019) landfilling with energy recovery, in Ebner et al. (2014) and this study, the minimum value corresponds to landfilling with energy recovery, the maximum to landfilling without energy recovery.
2. The authors assumed that donated food replaced bread or the original products according to different scenarios. For example, in case of beef, the CO_{2eq} saved is 310 CO_{2eq}/ton when beef replaces bread and 26,000 CO_{2eq}/ton when it replaces the original products.

5.1.8. Sensitivity and limitations

Similar to other studies in the literature (Albizzati et al., 2019; Bergström et al., 2020; Eriksson et al., 2015; Eriksson and Spångberg 2017) the obtained results of this work are sensitive to the kind of donated products. For instance, when dealing with NMF products, the higher the amount of materials and energy demanded throughout their production chain, the higher the avoided emissions will be. Although not evaluated in this study, this trend would be also applied to other LCA impact categories besides GWP.

The avoided impacts play a key role, also, in the Biorefinery scenario, where in almost all the impact categories assessed the avoided conventional N fertilizer production is responsible for, on average, 75 % of the avoided impacts, with the only exception in FDP, where the avoided natural gas production is responsible for 84% kg oil eq. saved. As shown in Figure 30, and confirmed by Eriksson et al. (2015), the amount of avoided downstream emissions in the landfill is another parameter with high influence on results (88% of GWP emissions is due to direct CH_4 emissions), which calls for actions to avoid emissions, such as the ones caused by burning it into flares to obtain CO_2. Furthermore, OBP collection and transport steps have a considerable influence in FDP, PMFP and TAP, in which diesel combustion in vehicles was responsible for 76%, 93% and 96% of the impacts, respectively (Figure 30). Replacing diesel fuel with electricity in the donation and biorefinery scenarios, allied to a reduction on total kilometers travelled, has shown a great reduction on impacts of up to 80% and 90% in scenario #III and scenario #VII for PMFP, and up to 80% for scenario #III and 95 % for scenario #VII in TAP.

The findings of this study are important to highlight the importance of considering the waste hierarchy concept in managing by-products. Although clues on this topic are available in the literature, the specific case study (the Brazilian food distribution center) and the method applied (LCA) bring insights from different perspectives that could contribute to discussions for the advancement of science in this field. It is important to emphasize that the final numbers obtained should not be used as a reference for all kinds of food distribution centers due to inherent specificities of the Brazilian case, for example, the logistic solution modelled (logistic based on the “Misuzumashi” concept) are not easily applicable elsewhere. The OBP concentration is another important aspect that allows for a different logistic operation, which is very different when compared to retail level, due to the long distances and existing complexities; in this case, donation and biorefinery scenarios would require additional costs related to transport. Another limitation of this work is related to the data obtained from Ecoinvent database used in calculating indirect and avoided impacts. For some products, due to the unavailability of data representing Brazilian conditions, global values have been chosen and thus, both indirect emissions of OBP management and the avoided emissions related to food production can be overestimated, mainly in the GWP, PMFP, POFP and TAP impact categories. This is a result of three main Brazilian peculiarities: (1) agricultural production occurs mostly in open fields, rather than in energy intensive greenhouses (Wiersinga et al., 2013); (2) the Brazilian electricity matrix is based on renewable resources (~80 % including hydropower, from biomass, wind and solar; Griebenow and Ohara, 2019); (3) the existing percentage of biodiesel in the commercial diesel reaches up to 30 % for material production chains and 10 % for liquid fuel used in the transportation sector (EPE, 2020). Also, regarding the products replaced in the biorefinery scenario, the use of global values related to natural gas extraction and transport instead of Brazilian values could have overestimated the saved impacts. Finally, the amount and kinds of OBP managed, donated food and biorefinery products established is an average value under an annual temporal analysis, which, although representative for the purposes this study, can differ from one year to another according to market demand and weather conditions influencing agricultural production.

5.2. Emergy

5.2.1. Data Collection and modelling

For the emergy synthesis, the same data previously used for the LCA inventory were considered, except for the emissions, which are not considered when using this method; other inputs that only emergy takes into account were included, such as natural renewable resources, human labor, and services. Details regarding modelling the assessed scenarios considered for emergy synthesis are shown in the following paragraphs.

Scenario #I: the allocation was the same as the one for LCA, considering CEAGESP OBP percentage in the Caieiras landfill (2.5%) and the percentage in mass of CEAGESP leachate BOD in the Baruerì wastewater plant (0.009%). In addition to the inputs considered for the LCA, the emergy contribution of the annual rainfall in the Caieiras landfill was estimated using climatological data available for the municipality of Caieiras (RIMA, 2016), rain being the most important input contributing to leachate generation. The average annual rainfall considered was 1,537 mm (see Appendix C, Figure C1). Another input of the Caieiras landfill was the local soil used to cover the waste, which was previously removed to build the landfill and later progressively reutilized to cover the waste. This input was classified as a non-renewable local resource (N) and the quantity considered was 40% in mass of the total CEAGESP´s landfilled OBP in 2018, according to information from technical visit, and the value obtained by Buranakarn (1998). The Unit Emergy Value (UEV) considered for the soil and other natural landfill materials, such as gravel, was calculated by accounting the global sedimentary cycle work (Odum, 1996), focusing on the work made by nature to generate the geological materials. Although a similar approach was considered by Marchettini et al. (2007), it is different from those by other authors, such as Almeida et al. (2012), who considered exclusively the organic fraction of the soil, or from Liu et al. (2013), who simply ignored the soil contribution. Because a landfill is a complex system that uses a huge quantity of natural resources originated by the sedimentary cycle, the work of Nature in generating the soil was accounted for, to respect the general concept of emergy. This criterion would be considered more aligned with the emergy theory, rather than focusing only on the organic fraction, which would be an anthropocentric point of view. The emergy value of the soil used in a landfill depends on Nature’s work to generate it and not on the (theoretical) soil lost for agricultural use, which reflects an anthropic perspective; this approach is commonly used in the ecological footprint method for estimating the biocapacity of urban areas.

Regarding human labor, the following assumptions were considered: for the waste collection and transport, three workers per garbage truck are considered; for the operations of waste transfer, transport and landfilling, one driver per vehicle; in the landfill, the engineer leading the operations and one worker per vehicle. For the wastewater plant: two drivers were considered, one for leachate transport and the second for sludge transport; three men for management of wastewater plant, two operatives and one engineer.

Scenario #II: besides including all inputs of scenario I, it also includes water, lubricant oil, concrete and steel for the power plant, as well as four workers: an engineer and three technicians. The saved emergy related to electricity produced by landfill biogas is estimated based on the electricity generated by the Brazilian grid (hydropower), considering the same amount of MWh generated at the Caieiras landfill.

Scenario #III: accounts for the inputs of steel for logistic trains, stainless steel equipment, shed andcold room structure, lead for batteries, wood and plastic for outdoor and indoor pallets respectively, polystyrene for insulation panels, and electricity consumption. Nineteen workers were considered (3 drivers, one for each logistic train, and 16 operatives for quality-checking and cold rooms. The emergy inputs derived by the management of the residual 20% of OBP sent to landfill are included as well, without considering electricity production.

Scenario #IV: accounts for the same inputs as scenario #III, but here the powerplant inputs and the saved emergy derived by electricity production of the residual fraction sent to landfill are included.

Scenario #V: it accounts for the same inputs of Scenario #III, but in this scenario, the saved emergy derived from food donation is included. It was estimated by considering information about crop production (Brandt-Williams, 2002; de Barros et al., 2009) and a recovery amount of 800 kg per ton of OBP treated.

Scenario #VI: similar to Scenario #V, but it includes the emergy inputs derived by electricity production from the residual fraction of OBP sent to landfill, as well as the related saved emergy.

Scenario #VII: considers all the inputs for the biorefinery construction and operation phases as for LCA (steel, lead, wooden pallets, water, HDPE for fertilizers storage in containers). Regarding labor, 38 workers were considered (3 in OBP collection and transport, 10 in manual separation, 3 for mechanical grinding, 3 for aerobic digestion, 2 for water scrubbing, 15 for solid and liquid separation and storage, and 2 for the CHP plant).

Scenario #VIII: it accounts for all the inputs of Scenario #VII and, furthermore, includes the saved emergy related to natural gas and conventional fertilizers production, according to information provided by Brown et al. (2011) and Odum (1996).

An overview of EMI and EMS for each scenario is depicted in Table 10. For all scenarios, labor and services were accounted for by using the 1.55 E+07 sej/person (emergy per person) and 8.41 E +12 seJ/USD (emergy per money ratio) values of Brazil, as published by Faria (2017); this approach is in accordance with Ulgiati and Brown (2014) proposed rules. For services calculation the monetary ratio (R$/USD) of 0.258 calculated on 31/12/2018 was considered. In this study, the most recent emergy baseline 12.00E+24 SeJ/yr (Brown et al, 2016) was chosen as a reference, and all UEV's were converted to that emergy baseline. Table 11 and its relative notes show details about this calculation procedure. Further details on scenarios elaboration are available in Appendix D, Tables D1 to D19.

Table 10 : EMI and EMS according to the evaluated scenarios

N.	Scenario	Invested emergy EMI	Saved Emergy EMS
#I	Landfilling	100% Emergy spent to landfilling	Zero
#II	Elettricity	100 % Emergy spent to landfilling by including electricity production	100 % Electricity Production from Brazilian Matrix
#III	Donation 80 + Landfilling 20	100% Emergy spent to Donation + 20% emergy spent in Scenario #I	Zero
#IV	Donation 80 + Electricity 20	100% Emergy spent to Donation + 20% emergy spent in Scenario #II	20% Electricity Production from Brazilian Matrix
#V	Avoided Production 80 + Landfilling 20	100% Emergy spent to Donation + 20% emergy spent in Scenario #I	80% Avoided Food Production
#VI	Avoided Production 80 + Electricity 20	100% Emergy spent to Donation + 20% emergy spent in Scenario #II	80% Avoided Food Production + 20% Electricity Production from Brazilian Matrix
#VII	Biorefinery	100% Emergy spent to Biorefining	Zero
#VIII	Biorefinery + Avoided Production	100% Emergy spent to Biorefining	100 % Emergy Natural Gas Production + 100% Emergy Conventional Fertilizers Production

Table 11: Unit emergy values (UEVs) used in this study

N.	Item	Unit	Type[a]	R fract. %[b]	Original UEV	Original Unit	Original Bsl.[c]	Source	Conversion[d]	UEV	Unit
1	**Rain, chemical**	kg	R	100	1.82E+04	seJ/J	9.44E+24	Odum, 1996	1.27	4.68E+06	seJ/kg
2	**Labour**	person	F	15.2	2.04E+07	seJ/person	1.58E+25	Faria, 2017	0.76	1.55E+07	seJ/person
3	**Water (River)**	kg	R	100	2.03E+05	seJ/g	9.44E+24	Buenfil, 2001	1.27	2.58E+08	seJ/kg
4	**Water (Supply System)**	kg	F	50	5.73E+11	seJ/m^3	9.44E+24	Buenfil, 2001	1.27	7.28E+08	seJ/kg
5	**Air**	kg	R	100	5.16E+07	seJ/g	1.58E+25	Wang et al., 2006	0.76	3.92E+10	seJ/kg
6	**Wood**	kg	F	82.4	9.60E+03	seJ/J	1.20E+25	De Oliveira, 2018	1	1.94E+11	seJ/kg
7	**Electricity**	kWh	F	68	1.47E+05	seJ/J	1.52E+25	Giannetti et al, 2015	0.79	4.18E+11	seJ/kWh
8	**Iron**	kg	F	0	8.55E+14	seJ/t	9.44E+24	Pan et al., 2016	1.27	1.09E+12	seJ/kg
9	**Gravel**	kg	F	0	1.00E+09	seJ/g	9.44E+24	Odum, 1996	1.27	1.27E+12	seJ/kg
10	**Geotextile (poliprop.)**	kg	F	0	2.16E+15	seJ/t	1.58E+25	Mu et al, 2012	0.76	1.64E+12	seJ/kg
11	**Soil**	kg	N	0	1.00E+09	seJ/g	9.44E+24	Odum, 1996	1.27	1.27E+12	seJ/kg
12	**Concrete**	kg	F	0	1.44E+09	seJ/g	9.44E+24	Buranakarn, 1998	1.27	1.83E+12	seJ/kg
13	**Cement**	kg	F	0	1.97E+09	seJ/g	9.44E+24	Buranakarn, 1998	1.27	2.50E+12	seJ/kg
14	**GCL (Clay)**	kg	F	0	2.00E+09	seJ/g	9.44E+24	Odum, 1996	1.27	2.54E+12	seJ/kg
15	**Steel**	kg	F	0	1.58E+15	seJ/t	9.44E+24	Pan et al, 2016	1.27	2.01E+12	seJ/kg
16	**Lubricant oil**	kg	F	0	1.73E+05	seJ/J	1.52E+25	Brown et al., 2011	0.79	4.72E+12	seJ/kg
17	**Rubber**	kg	F	0	4.30E+09	seJ/g	9.44E+24	Buranakarn, 1998	1.27	5.46E+12	seJ/kg
18	**Diesel Fuel**	kg	F	0	1.81E+05	seJ/J	1.52E+25	Brown et al., 2011	0.79	5.99E+12	seJ/kg
19	**Gasoline**	kg	F	0	1.87E+05	seJ/J	1.52E+25	Brown et al., 2011	0.79	6.18E+12	seJ/kg
20	**HDPE**	kg	F	0	5.27E+09	seJ/g	9.44E+24	Buranakarn, 1998	1.27	6.69E+12	seJ/kg
21	**Polyacrylamide**	kg	F	0	6.78E+12	seJ/kg	1.20E+25	This study	1	6.78E+12	seJ/kg
22	**Plastic (PVC)**	kg	F	0	5.87E+09	seJ/g	9.44E+24	Buranakarn, 1998	1.27	7.45E+12	seJ/kg
23	**Polystyrene**	kg	F	0	5.87E+09	seJ/g	9.44E+24	Buranakarn, 1998	1.27	7.45E+12	seJ/kg
24	**Services**	US$	F	15.2	1.11E+13	seJ/$	1.58E+25	Faria, 2017	0.76	8.41E+12	seJ/$
25	**Ferric chloride**	kg	F	0	3.86E+10	seJ/g	1.58E+25	Ingwersen, 2009	0.76	2.93E+13	seJ/kg
26	**Aluminium (Billet)**	kg	F	0	6.77E+10	seJ/g	9.44E+24	Buranakarn, 1998	1.27	8.60E+13	seJ/kg
27	**Lead**	kg	F	0	3.59E+17	seJ/t	1.20E+25	Pan et, 2019	1	3.59E+14	seJ/kg

a: Type of Emergy input. R = Local Renewable; N = local not renewable; F = purchased

b: percentage of Renewable Emergy in purchased inputs (Fr)

c: original baseline of reference paper in SeJ/yr

d: All UEVs from other authors are converted to Current Earth Baseline = 12.00E+24 SeJ/yr (Brown et al, 2016), used as reference in this study; UEVs of F are calculated without Labor and Services.

1: (1.82E+04 seJ*J^{-1})/(4.94 J/g Gibbs free Energy, from Odum, 1996)*(12E+24 SeJ*yr^{-1} / 9.44E+24 SeJ*yr^{-1} Odum, 1996 baseline)*(10^3 g/kg)

2: (1.95E+25 seJ/yr emergy BR 2018 from Faria, 2017) / ((2.09E+08 ppl BR 2018) *(3000 Kcal/day) *(365 day/yr)*(4184 J/kcal) *(12.00E+24 SeJ*yr^{-1} /15.83E+24 SeJ*yr^{-1} Odum 2000 baseline)); % Renewability of Brazilian labor from Giannetti et al. (2015).
3: (2.03E+05 seJ/g from Buenfil, 2001, p.224)*(12E+24 SeJ*yr^{-1} / 9.44E+24 SeJ*yr^{-1} Odum, 1996 baseline)*(10^{3} g/kg).
4: (68.52E+10 seJ/m^{3} - 5.98E+10 seJ / m^{3} opening and mainteinance - 3.22E+10 seJ/m^{3} chemical cost - 2.01E+10 seJ/m^{3} plant construction and upgrade = 57.31 E+10 seJ / m^{3} from Buefill, 2001, p.80) * (12E+24 SeJ*yr^{-1} / 9.44E+24 SeJ*yr^{-1} Odum, 1996 baseline)*(10^{-3} m^{3} / L) and by assuming pure water where 1 L corresponds to 1 kg; % renewability from Giannetti et al., (2015)
5: (5.16E+07 seJ/g from Lan et al., 2002 apud Wang et al., 2006) * 12.00E+24 SeJ*yr^{-1} /15.83E+24 SeJ*yr^{-1} Odum 2000 (assumed baseline)* (10^{3} g / kg)
6: (9.60E+03 seJ/J transformity of Brazilian loblolly pine (Pinus Taeda) production from De Oliveira et al., 2018, assuming bsl 12.00E24 SeJ/yr)*(2.02E+07 J/kg (HHV) of loblolly pine wood from Aquah, 2016); %R from De Oliveira et al, 2018.
7: (1.47E+05 seJ/J from Giannetti, 2015) * (3.6E+06 J/kWh) * (12.00E+24 SeJ*yr^{-1} /15.2E+24 SeJ*yr^{-1} Brown and Ulgiati 2010 baseline)
8: (8.55E+14 seJ/t from Lan et al, 2002 apud Pan et al, 2016) * (12E+24 SeJ*yr^{-1} / 9.44E+24 SeJ*yr^{-1} Odum, 1996 baseline)*(10^{-3} t/kg); Due to lack of data, it was not possible to separate labor from the total emergy amount.
9: (1.00E+09 seJ/g after Buranakarn, 1998, original source Odum, 1996 p. 310) * (12.00E+24 SeJ*yr^{-1} / 9.44E+24 SeJ*yr^{-1} Odum, 1996 baseline)*(10^{3} g/kg)
10: (2.16E+15 seJ/t from Mu et al, 2012)*(12.00E+24 SeJ*yr-1/15.83E+24 SeJ*yr-1 Odum, 2000)*(10^{-3} t/kg). Due to lack of data, it was not possible to separate labor from the total emergy amount
11: (1.00E+09 seJ/g from Odum, 1996 global sedimentary cycle p. 310)*(12.00E+24 SeJ*yr^{-1} / 9.44E+24 SeJ*yr^{-1} Odum, 1996 baseline) * (10^{3} g/kg)
12: (1.44E+09 seJ/g ready-mixed concrete from Buranakarn, 1998)*(12.00E+24 SeJ*yr^{-1} / 9.44E+24 SeJ*yr^{-1} Odum, 1996 baseline)*(10^{3} g/kg)
13: (1.97E+09 seJ/g, Buranakarn, 1998)*(12.00E+24 SeJ*yr^{-1} / 9.44E+24 SeJ*yr^{-1} Odum, 1996 baseline)*(10^{3} g/kg)
14: (2.00E+09 seJ/g from Odum, 1996 pag. 48) * (12.00E+24 SeJ*yr^{-1} / 9.44E+24 SeJ*yr^{-1} Odum, 1996 baseline) * (10^{3} g/kg)
15: (1.58E+15 seJ/t without L & S from Pan et al, 2016)*(12.00E+24 SeJ*yr^{-1} / 9.44E+24 SeJ*yr^{-1} Odum, 1996 baseline)*(10^{-3} t/kg)
16: (1.73E+05 seJ/J residual oil from Brown et al., 2011) * (3.95E+07 J/kg (LHV) from: https://h2tools.org/hyarc/calculator-tools/lower-and-higher-heating-values-fuels) * (12.00E+24 SeJ*yr-1 / 15.2E+24 SeJ/yr-1 Brown & Ulgiati, 2010)
17: (4.30E+09 seJ/g from Odum et al., 1987 apud Buranakarn, 1998)*(12.00E+24 SeJ*yr^{-1} / 9.44E+24 SeJ*yr^{-1} Odum, 1996 baseline)*(10^{3} g/kg). Due to lack of data, it was not possible to separate labor from the total emergy amount
18: (1.81E+05 seJ/J from Brown et al., 2011) * (4186 J/kcal) * (10^{4} kcal/kg from Agostinho et al, 2013) * (12.00E+24 SeJ*yr^{-1} / 15.2E+24 SeJ/yr^{-1} Brown and Ulgiati, 2010)
19: (1.87E+05 seJ/J from Brown et al., 2011) * (4186 J/kcal) * (10^{4} kcal/kg LHV gasoline) * (12.00E+24 SeJ*yr^{-1} / 15.2E+24 SeJ/yr^{-1} Brown and Ulgiati, 2010)
20: (5.27E+09 seJ/g HDPE in Europe from Buranakarn, 1998) * (12.00E+24 SeJ*yr^{-1} / 9.44E+24 SeJ*yr^{-1} Odum, 1996 baseline)*(10^{3} g/kg)
21: See Appendix F
22: (5.87E+09 seJ/g from Buranakarn, 1998)*(12.00E+24 SeJ*yr^{-1} / 9.44E+24 SeJ*yr^{-1} Odum, 1996 baseline)*(10^{3} g/kg)
23: For Polystyrene was considered the same UEV of plastic PVC (see Note 18)
24: See table services EMR calculation
25: (3.86 E+10 seJ/g from Ingwersen, 2009, related baseline from Emergy Database) * (12.00E+24 SeJ*yr^{-1} /15.83E+24 SeJ*yr^{-1} Odum 2000 baseline)*(10^{3} g/kg). Due to lack of data, it was not possible to separate labor from the total emergy amount
26: (6.77E+10 seJ/g from Buranakarn, 1998)*(12.00E+24 SeJ*yr^{-1} / 9.44E+24 SeJ*yr^{-1} Odum, 1996 baseline)*(10^{3} g/kg)
27: (3.59E+17 seJ/t from Pan et al., 2019)*(10^{-3} t/kg). Contribution of labor and services is negligible (< 1%).

5.2.2. Understanding the studied systems from energy diagrams

While the flowcharts of food and organic waste management presented in the Life Cycle Assessment section provide information about the internal pathways and their relationships, focusing on the human-side perspective, the energy diagrams provide a full picture of the system as embedded inside the natural system, highlighting the relationships with the environment as the source of resources that sustain the studied system.

Figure 33 shows the general diagram of scenarios #I and #II according to the current OBP management. As renewable (R) inputs there is rain (chemical emergy), as natural local non- renewable input (N) there is the soil, and all other inputs are classified as purchased from the larger economy (F). The food arrives at CEAGESP to be traded and sold, and the OBP (about 100% as organic waste) is sent to the Caieiras landfill; only a negligible amount is diverted to donation. External energy sources are food, diesel, materials to make vehicles (metal, rubber, plastic), human labor, services, and the outputs are the sold food, food to charity and food waste; this last one is the focus of this study. Further details about CEAGESP are provided in the CEAGESP diagram.

The food waste is then transported by two trucks to the Caieiras landfill. The resources considered at this stage are diesel, vehicles materials and human labor. At Caieiras Landfill, the organic waste is discarded, and all subsequent processes involve the input of gravel and other materials to build the landfill, metals and diesel used by vehicles, the rain (involved in the process of degradation of the organic fraction), the soil to cover the various layers of waste, the human labor and the sludge (originated by the leachate generated in Caieiras) coming back from the wastewater plant. As output, there is methane, which can be directly released into the atmosphere, burned with electricity generation, and burned without electricity generation. Another output is the leachate sent to the wastewater plant by tank trucks, demanding diesel, vehicle materials, and labor inputs. Further details are explained in the Caieiras diagram.

Still regarding Figure 33, the leachate arrives at wastewater plants, which demand concrete, chemical, electricity, and labor. As outputs, the liquid effluent is released to the Tietê River, the CH_4 derived from the anaerobic sludge digestion is released into the atmosphere, and the sludge generated goes back to Caieiras Landfill.

After this overview, the functioning of each internal process of Figure 33 is described in a more detailed way. The first one is the CEAGESP food distribution center (Figure 34). CEAGESP has a trading area, a food bank and a special area for the waste transfer. Focusing on the trading area, the inputs are food, infrastructure materials, and human labor, while the outputs are food waste, food for charity, and inorganic waste for recycling. Recycling is outside of the spatial boundaries of this present study, which focuses on current organic waste fraction management.

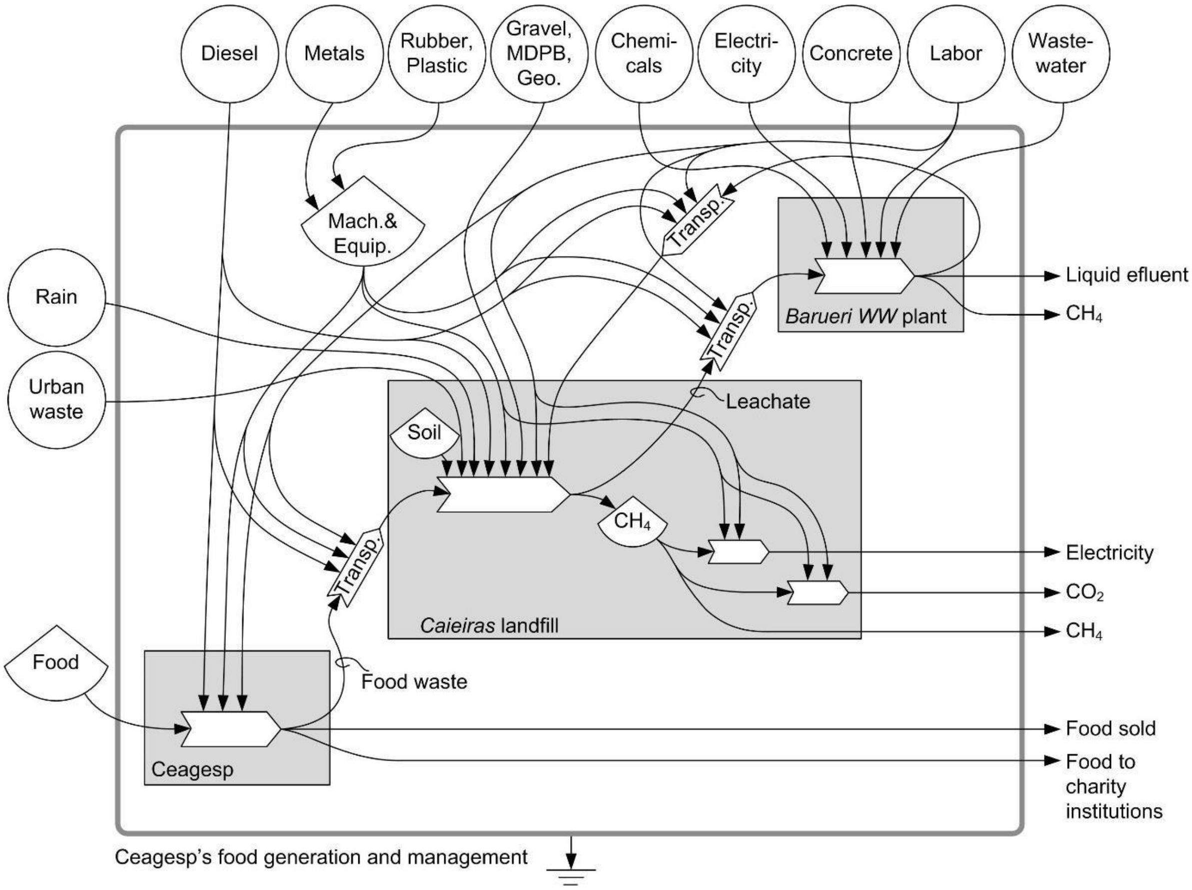

Figure 33: General energy diagram of the system assessed.

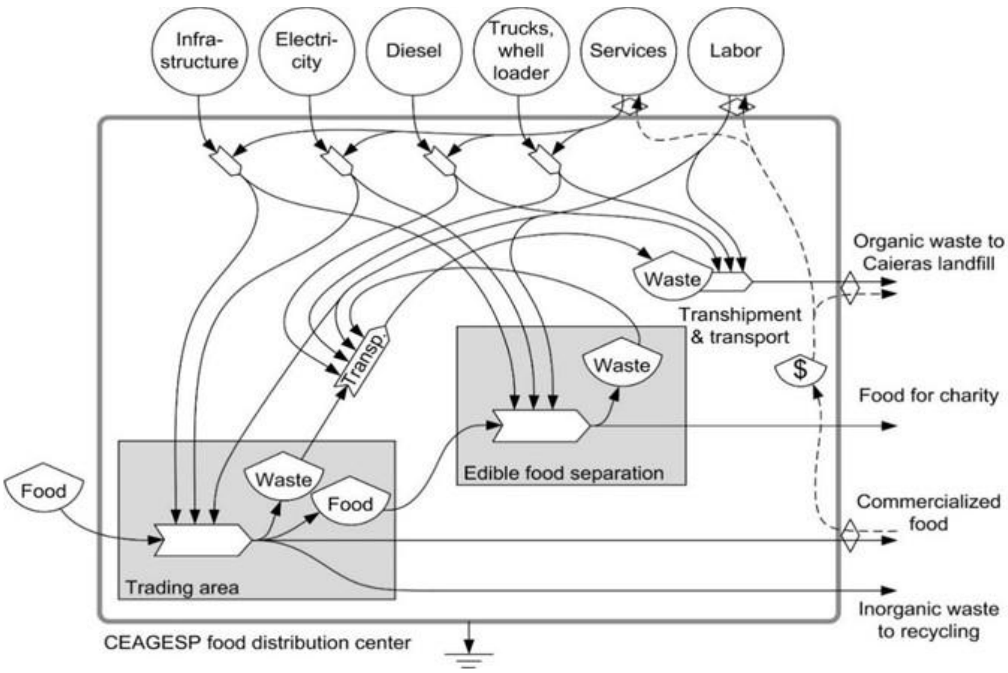

Figure 34: Detailed Energy Diagram of CEAGESP food distribution center.

The generated waste is collected by an internal system collection and transport, which demands diesel fuel, trucks materials and labor. The collected waste is accumulated and then transferred to the Caieiras landfill, demanding diesel, vehicles and human labor. From the trading area, a small percentage of OBP is sent to the food bank, where the edible food is separated. The processes include, basically, checking and whether the quality is acceptable, the food is recognized as NMF and sent to charity institutions, otherwise, it becomes organic waste and goes to the landfill.

Details about the Caieiras landfill are shown in Figure 35. The organic waste arrives from CEAGESP and is unloaded inside the landfill, which demands energy and materials for its functioning, such as diesel fuel, vehicles materials, gravel, benthonic geocomposite, high density polyethylene, geotextile, dry sludge, human labor and services. The local renewable input is the rain, while the local non-renewable input is the soil used to cover the waste. The outputs are methane and leachate generated by waste degradation. Methane has three different pathways: 20% is directly released into atmosphere, 40% is burned in flares, and 40% is burned to generate electricity. Leachate is concentrated and then loaded in tank trucks for transportation to the wastewater plant.

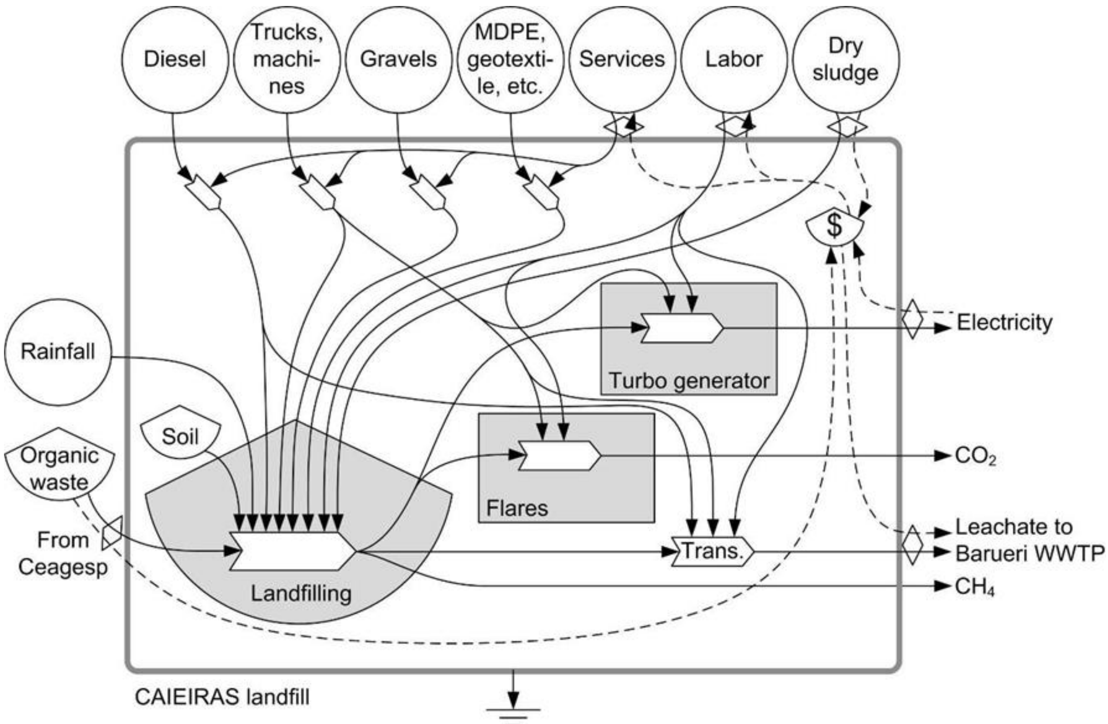

Figure 35: Detailed energy diagram of the Caieiras landfill

As for the leachate treatment in the wastewater plants (Figure 36), it is treated together with the domestic wastewater, which demands ferric chloride, calcium hydroxide, electricity, concrete, and human labor as input, and generates treated liquid effluent, methane, and dry sludge as outputs. The dried sludge is transported by trucks back to the Caieiras landfill, demanding more vehicles, diesel fuel and human labor as inputs.

In scenarios #III to #VI (Figure 37), the potential NMF (90%) is collected by a logistic train and transported to the food bank, where its quality is checked, products separated, and temporarily stocked inside cold rooms until withdrawal by charity institutions. This scenario requires material for the infrastructure (mainly steel) and vehicles (steel for the chassis and lead for the batteries), as well as electricity that has replaced the role of the fossil fuel. External labor is also needed. The food waste of scenarios #III to #VI, including both the 10% derived from mechanical injuries and the 10% generated after quality-checking operations at the food bank, follows the path of scenarios #I and #II in scenarios #III and #IV, respectively. Scenarios #V and #VI includes the saved emergy derived from food donation, but in all other aspects, they are identical to scenarios #III and #IV.

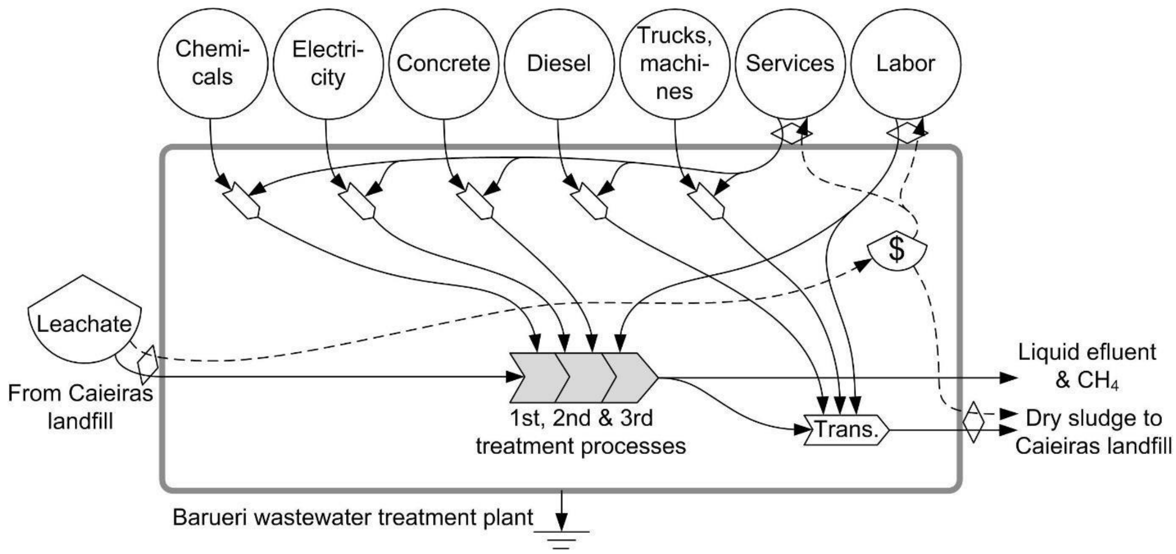

Figure 36: Detailed energy diagram of the Barueri wastewater plant.

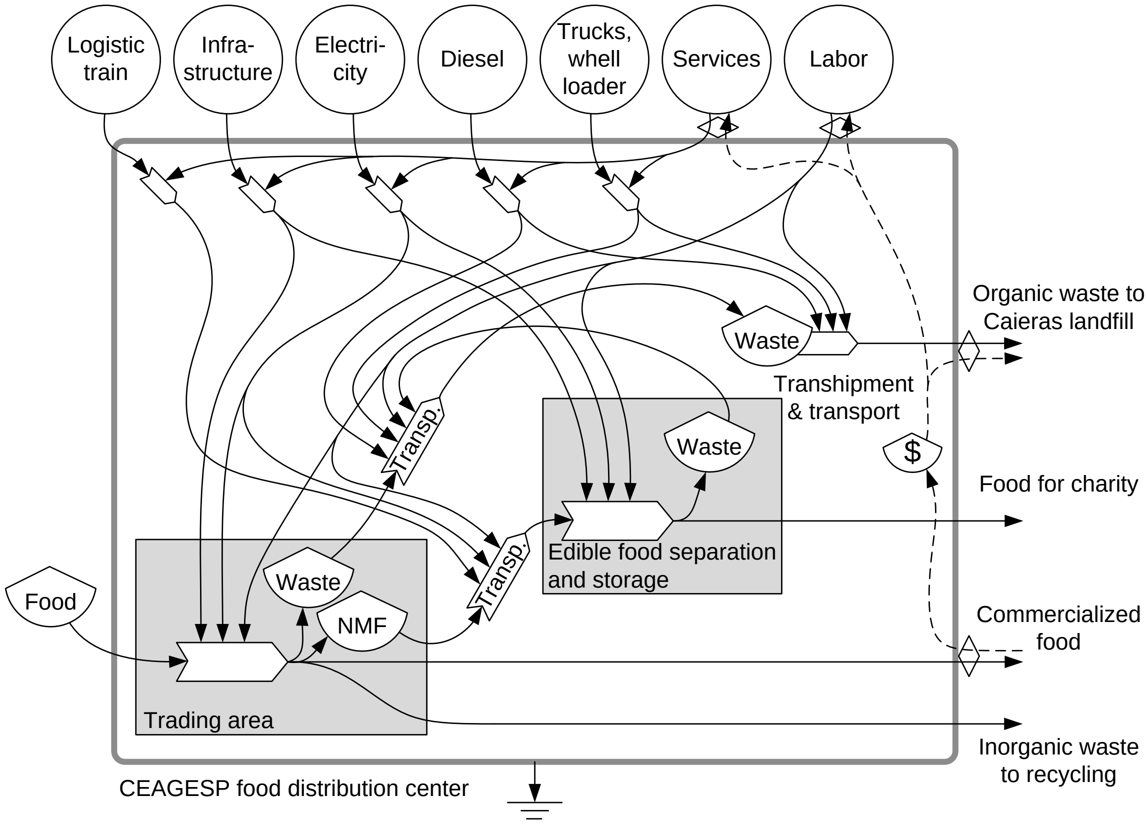

Figure 37: Energy diagram of scenarios #III to #VI.

In scenario #VII and #VIII, 100% of the byproducts generated are collected and sent to the biorefinery facility (Figure 38), with the same collection and transport systems as those of the donation scenarios. At biorefinery, the organic fraction is separated from the inorganic one, grinded and directed to a biodigester for 20 days, generating biogas and fertilizer. About 20% of this biogas is sent to a CHP to produce electricity and heat for internal use, while 80% of the biogas is refined to obtain biomethane.

According to Figure 39, which shows the biorefinery processes in a more detailed way, collection and transport are the same as for the donation scenario, therefore executed by a logistic train made by steel for chassis and lead for batteries, fueled by the electricity plus human labor. The difference is that the electricity consumed is supplied by the biorefinery (CHP plant) instead of by the Brazilian electricity grid. The subsequent steps including manual separation, grinding and fermentation requires steel for machines and infrastructure, electricity, labor; exclusively for the fermentation step, water, heat and air are also used. The anaerobic digestion generates biogas and fertilizers. The biogas splits in two pathways, where the higher percentage is collected by the upgrading tower to be purified, resulting in biomethane, CO_2 and water; the lower biogas percentage (21%) is sent to the CHP plant to produce electricity, and heat, releasing CO_2, NO_X and other gases; the plant is self-sufficient in electricity and heat. Finally, the digestate is sent to the centrifuge machine to separate solid from liquid and stocked inside drums made by HDPE to be collected by the wholesalers. Scenario #VIII includes the saved emergy derived from the natural gas being substituted by biomethane and chemical fertilizers substituted by organic fertilizers, but in all other aspects, it is identical to #VII.

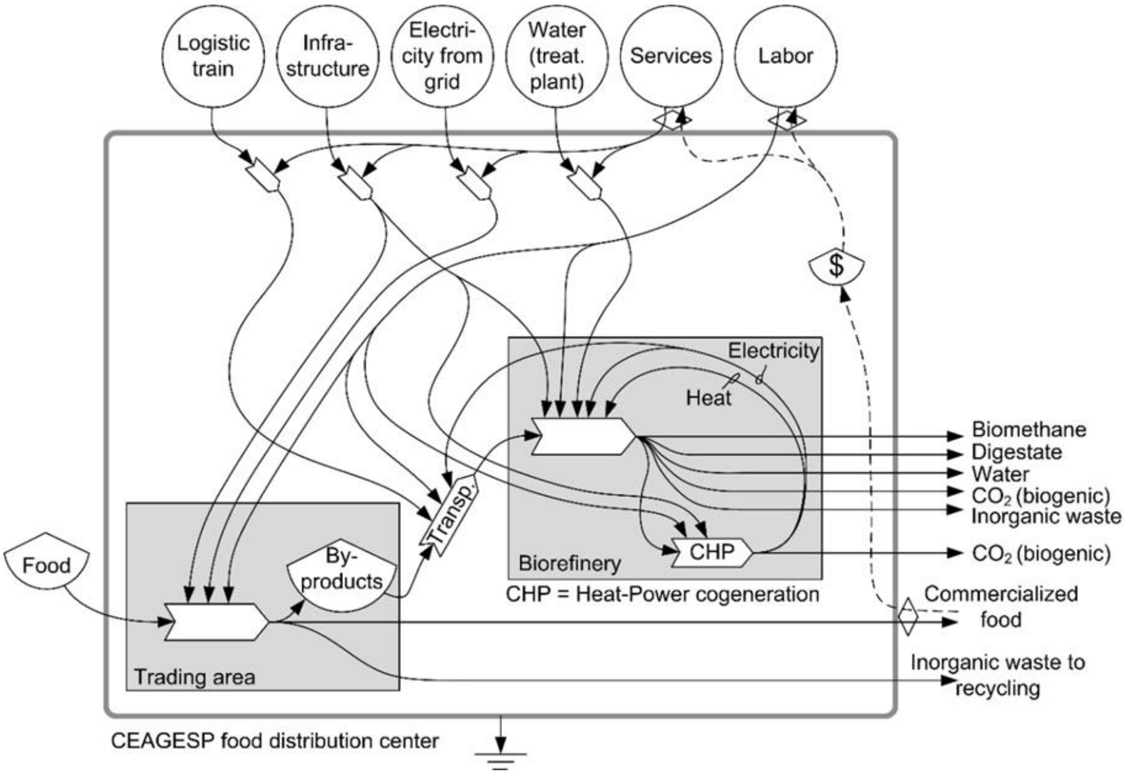

Figure 38: General energy diagram of CEAGESP with biorefinery facility (scenario VII).

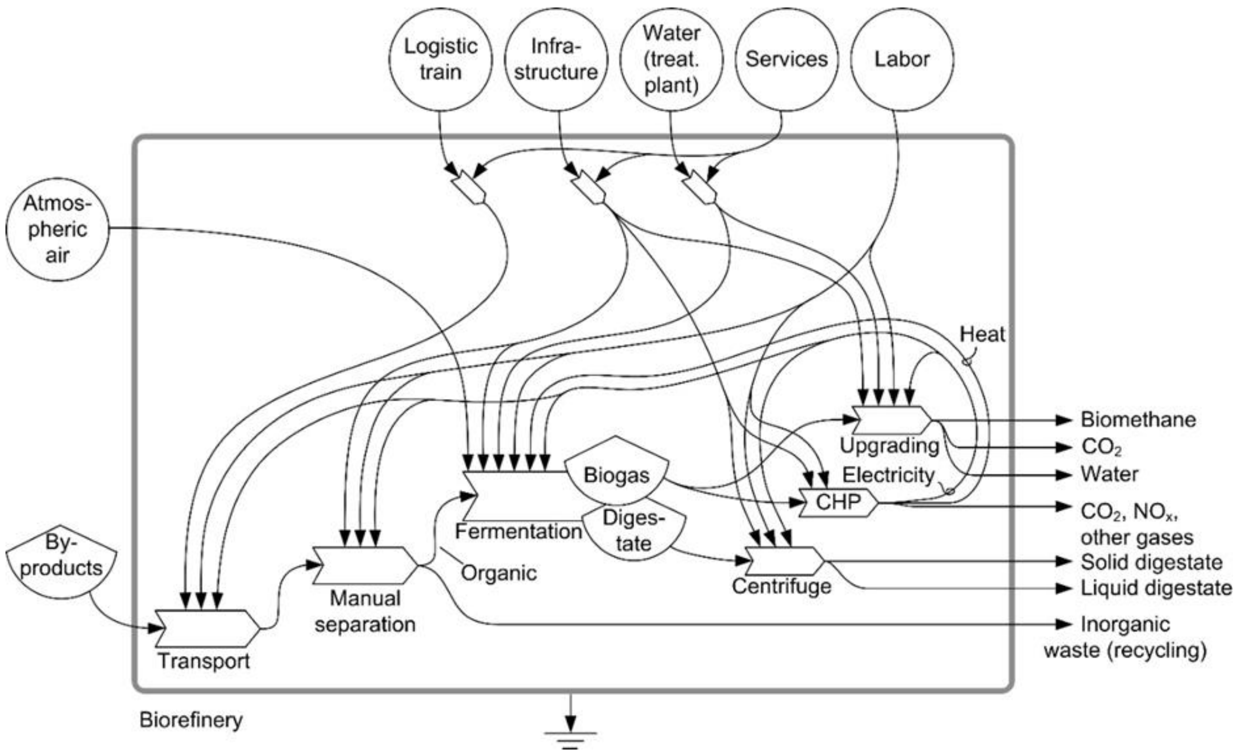

Figure 39: Detailed energy diagram of biorefinery facility.

5.2.3. Emergy performance of scenarios

The emergy inventory for all scenarios is shown in Tables 12 and 13, accounting for the annual emergy flows for each input, the total annual emergy flow U, the UEVs and the values of renewable (R), local non-renewable (N) and purchased inputs (F). Scenarios #I and #II depict the highest emergy demand (U) while scenarios #VII and #VIII have the lowest value. A total U of about nine times lower is depicted when biorefinery scenarios (#VII and #VIII) are compared with scenarios #I and #II, while this value is about five times lower when comparing donation scenarios (#III to #VI) with scenarios #I and #II. In scenarios #I and #II the main contribution for EMI is derived from the soil used to cover the waste (~60%) followed by the gravel used for the leachate drainage system (~22%). This high influence of the materials consumed by the landfill is confirmed by the study of Marchettini et al. (2007), while the different results obtained by Almeida et al. (2012) depends on the different assumptions regarding the UEV for soil. Scenario #II shows a slightly higher emergy demand, which is in accordance with Almeida et al. (2012), despite the existing differences on the UEV for soil as previously described. It is interesting to note that the emergy contribution of the residual fraction sent to the landfill on the total emergy amount in scenarios #III to #VI is equivalent to 94%, indicating that even slight improvements in OBP donation rates could have a great impact on the reduction of resources consumption. Furthermore, this emergy amount of residual fraction sent to landfill is responsible for the worse performance among donation scenarios (#III to #VI) when compared to biorefinery (#VII and #VIII). In fact, by assuming a theoretical case where the donation rate achieves 100%, it would result in an emergy demand of 5.35E+17seJ/yr, about 12% of biorefinery scenarios (#VII and #VIII).

Food donation depicts a better emergy performance even when considering the net emergy (Table 14), with a value equal to 6.33E+15 seJ/ton OBP in scenario #V. The biorefinery scenario #VIII also shows a positive net-emergy (6.23E+13 seJ/ton OBP), while the effect of the saved emergy derived from electricity production, in scenarios #II, #IV as well as for #VI is negligible. It is interesting to note that those scenarios that are the most recommended by the FRH (donation and biorefinery) show a positive net-emergy, indicating that the first options suggested by the hierarchy depicts a high capacity in emergy savings, especially the donation scenario. In fact, by calculating the emergy return index for scenarios #V and #VI, the EMS is ~29 times higher than its EMI, while for the biorefinery scenario #VIII, the EMS is about 1.5 times its EMI.

Some authors who assessed different management options for organic waste also found a positive net emergy (Figure 40). For example, Agostinho et al. (2013) assessing compost production of separated organic fraction from a municipal solid waste recycling plant in São Paulo city found a net emergy of 4.79E+13 seJ/ton of waste, Marchettini et al. (2007) assessing composting and incineration scenarios for municipal solid waste in Italy obtained 4.57E+14 seJ/ton of waste and 4.88E+14 seJ/ton of waste, respectively. Patrizi et al. (2015), assessing a biorefinery scenario for bioethanol production, showed 2.49E+14 seJ/ton of waste, while Santagata et al. (2019), assessing the electricity generated by energy recovering from slaughterhouse waste materials, depicted the second highest net emergy value of 2.68E15 seJ/ton of waste. Conversely, electricity production from landfilled OBP (scenario #II) has depicted negative values for net emergy. The numbers obtained in this work are consistent with landfill electricity production scenarios assessed by Agostinho et al. (2013), based on the information provided by the study of Almeida et al.(2012) assessing a landfill in São Paulo, and with the results obtained by Marchettini et al. (2007). It is interesting to note that those options located at the bottom of the FRH triangle have a negative net emergy while scenarios with higher priority (top of the pyramid) presented positive values. Due to the lack of emergy studies related to donation scenarios, the invested and saved emergy in the works of Eriksson et al. (2015) and Eriksson and Spangberg (2017) regarding food donation in Sweden, from information provided by their LCA inventory, was calculated in this work. The results show, by far, a positive net-emergy of 9.57E+15 seJ/ton and 5.93E+15 seJ/ton, respectively, for these studies, achieving an ERI of 75 and 295, which are very close to the ones obtained in this present work.

Numbers show that those waste management options (or evaluated scenarios) most recommended by the FRH present by far a positive net-emergy (as donation), intermediary scenarios, such as biorefinery, depict a slightly positive net-emergy (with the exception of Baral at al., 2015, due to the low-quality input - stillage), even if they are very close to the values of traditional waste management systems, such as composting and incineration. Only the energy recovery from landfill presents a negative net-emergy rate.

The results shown by the net-emergy indicator are even more evident when the Emergy Return Index (ERI) proposed in this study is applied, both to the scenarios assessed herein, and to similar scenarios provided by literature (see appendix D, table D11 for insights). In fact, energy recovery at landfill always depicts an ERI < 1, indicating that the use of landfill biogas to produce electricity is not convenient, from an emergy perspective, while donation scenarios show an ERI between 29 < ERI < 577, demonstrating that for 1 seJ of invested emergy it is possible to recover from tens to hundreds of times the invested emergy. Conversely, options at intermediate - low levels of the FRH show an ERI between 0.1 < ERI < 7.2, without any particular distinction among incineration, composting, and biorefinery alternatives.

These performances on net-emergy, and ERI along the FRH confirm, at least to some extent, the validity of the FRH concept.

Table 12: Annual Inputs emergy scenarios. UEVs converted to 12E24 SeJ/yr is the reference.

			Quantities (Unit / yr)								
N	Input	Unit	Inputs Sc.#I - Landfilling 100%	Inputs Sc. #II - Electricity 100%	Inputs Sc. #III - Donation 80% + Landfilling 20%	Inputs Sc.#IV - Donation 80% + Electricity 20%	Inputs Sc. #V - Av. Prod. 80% + Ldf. 20%	Inputs Sc. #VI - Av. Prod. 80% + ele. 20%	Inputs Sc.#VII - Biorefinery 100%	Inputs Sc.#VIII - Bioref. 100% + Av. Prod.	UEV (seJ/unit)
1	Rain, chemical	kg	4,61E+07	4,61E+07	9,22E+06	9,22E+06	9,22E+06	9,22E+06	n.a.	n.a.	4,68E+06
2	Labour	person	3,80E+01	4,20E+01	1,90E+01	1,90E+01	1,90E+01	1,90E+01	3,80E+01	3,80E+01	1,55E+07
3	Water (River)	kg	n.a.	2,28E+05	n.a.	4,56E+04	n.a.	4,56E+04	n.a.	n.a.	2,58E+08
4	Water (Supply System)	kg	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	6,16E+06	6,16E+06	7,28E+08
5	Wood	kg	n.a.	n.a.	4,50E+02	4,50E+02	4,50E+02	4,50E+02	4,50E+02	4,50E+02	1,94E+11
6	Electricity	kWh	1,50E+04	1,50E+04	6,25E+04	6,25E+04	6,25E+04	6,25E+04	n.a.	n.a.	4,18E+11
7	Iron	kg	2,30E+03	2,30E+03	4,60E+02	4,60E+02	4,60E+02	4,60E+02	n.a.	n.a.	1,09E+12
8	Gravel	kg	6,75E+06	6,75E+06	1,35E+06	1,35E+06	1,35E+06	1,35E+06	n.a.	n.a.	1,27E+12
9	Geotextile (poliprop.)	kg	4,45E+03	4,45E+03	8,90E+02	8,90E+02	8,90E+02	8,90E+02	n.a.	n.a.	1,64E+12
10	Soil	kg	1,88E+07	1,88E+07	3,77E+06	3,77E+06	3,77E+06	3,77E+06	n.a.	n.a.	1,27E+12
11	Concrete	kg	n.a.	2,65E+03	n.a.	5,29E+02	n.a.	5,29E+02	n.a.	n.a.	1,83E+12
12	Cement	kg	6,63E+02	6,63E+02	1,33E+02	1,33E+02	1,33E+02	1,33E+02	n.a.	n.a.	2,50E+12
13	GCL (Clay)	kg	1,95E+04	1,95E+04	3,90E+03	3,90E+03	3,90E+03	3,90E+03	n.a.	n.a.	2,54E+12
14	Steel	kg	1,34E+04	1,37E+04	6,37E+03	6,42E+03	6,37E+03	6,42E+03	8,76E+03	8,76E+03	2,01E+12
15	Lubricant oil	kg	n.a.	2,53E+03	n.a.	5,05E+02	n.a.	5,05E+02	n.a.	n.a.	4,72E+12
16	Rubber	kg	1,26E+03	1,26E+03	2,52E+02	2,52E+02	2,52E+02	2,52E+02	n.a.	n.a.	5,46E+12
17	Diesel Fuel	kg	1,95E+05	1,95E+05	3,90E+04	3,90E+04	3,90E+04	3,90E+04	n.a.	n.a.	5,99E+12
18	HDPE	kg	1,52E+04	1,52E+04	3,04E+03	3,04E+03	3,04E+03	3,04E+03	1,40E+04	1,40E+04	6,69E+12
19	Polyacrylamide	kg	1,27E+03	1,27E+03	2,54E+02	2,54E+02	2,54E+02	2,54E+02	n.a.	n.a.	6,78E+12
20	Plastic (PVC)	kg	1,22E+03	1,22E+03	3,29E+02	3,29E+02	3,29E+02	3,29E+02	n.a.	n.a.	7,45E+12
21	Polystyrene	kg	n.a.	n.a.	6,59E+01	6,59E+01	6,59E+01	6,59E+01	n.a.	n.a.	7,45E+12
22	Services	US$	6,44E+05	6,62E+05	1,70E+05	1,74E+05	1,70E+05	1,74E+05	4,96E+05	4,96E+05	8,41E+12
23	Ferric chloride	kg	1,66E+04	1,66E+04	3,31E+03	3,31E+03	3,31E+03	3,31E+03	n.a.	n.a.	2,93E+13
24	Aluminium (Billet)	kg	7,75E+02	7,75E+02	1,55E+02	1,55E+02	1,55E+02	1,55E+02	n.a.	n.a.	8,60E+13
25	Lead	kg	n.a.	n.a.	4,20E+02	4,20E+02	4,20E+02	4,20E+02	4,20E+02	4,20E+02	3,59E+14

Table 13: Emergy flows of the evaluated scenarios. All values are in seJ/yr. 12E24 SeJ/yr is the reference baseline R = Local Renewable; N = local not renewable; F = purchased

		Emergy flows (seJ/yr)							
N.	Input	Scenario #I Landfilling 100%	Scenario #II Electricity 100%	Scenario #III Donation 80% + Landfilling 20%	Scenario #IV Donation 80% + Electricity 20%	Scenario #V Avoided Prod. 80% + Ldf. 20%	Scenario #VI Avoided Prod. 80% + ele. 20%	Scenario #VII Biorefinery 100%	Scenario #VIII - Biorefinery 100% + Av.Prod.
1	Rain, chemical	2,16E+14	2,16E+14	4,31E+13	4,31E+13	4,31E+13	4,31E+13	n.a.	n.a.
2	Labour	5,89E+08	6,51E+08	2,95E+08	2,95E+08	2,95E+08	2,95E+08	5,89E+08	5,89E+08
3	Water (River)	n.a.	5,88E+13	n.a.	1,18E+13	n.a.	1,18E+13	n.a.	n.a.
4	Water (Supply System)	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	4,48E+15	4,48E+15
5	Wood	n.a.	n.a.	8,73E+13	8,73E+13	8,73E+13	8,73E+13	8,73E+13	8,73E+13
6	Electricity	6,28E+15	6,28E+15	2,61E+16	2,61E+16	2,61E+16	2,61E+16	n.a.	n.a.
7	Iron	2,50E+15	2,50E+15	5,00E+14	5,00E+14	5,00E+14	5,00E+14	n.a.	n.a.
8	Gravel	8,58E+18	8,58E+18	1,72E+18	1,72E+18	1,72E+18	1,72E+18	n.a.	n.a.
9	Geotextile (poliprop.)	7,30E+15	7,30E+15	1,46E+15	1,46E+15	1,46E+15	1,46E+15	n.a.	n.a.
10	Soil	2,39E+19	2,39E+19	4,78E+18	4,78E+18	4,78E+18	4,78E+18	n.a.	n.a.
11	Concrete	n.a.	4,84E+15	n.a.	9,67E+14	n.a.	9,67E+14	n.a.	n.a.
12	Cement	1,66E+15	1,66E+15	3,32E+14	3,32E+14	3,32E+14	3,32E+14	n.a.	n.a.
13	GCL (Clay)	4,95E+16	4,95E+16	9,90E+15	9,90E+15	9,90E+15	9,90E+15	n.a.	n.a.
14	Steel	2,69E+16	2,74E+16	1,28E+16	1,29E+16	1,28E+16	1,29E+16	1,76E+16	1,76E+16
15	Lubricant oil	****	1,19E+16	n.a.	2,38E+15	n.a.	2,38E+15	n.a.	n.a.
16	Rubber	6,87E+15	6,87E+15	1,37E+15	1,37E+15	1,37E+15	1,37E+15	n.a.	n.a.
17	Diesel Fuel	1,17E+18	1,17E+18	2,33E+17	2,33E+17	2,33E+17	2,33E+17	n.a.	n.a.
18	HDPE	1,02E+17	1,02E+17	2,03E+16	2,03E+16	2,03E+16	2,03E+16	9,37E+16	9,37E+16
19	Polyacrylamide	8,61E+15	8,61E+15	1,72E+15	1,72E+15	1,72E+15	1,72E+15	n.a.	n.a.
20	Plastic (PVC)	9,06E+15	9,06E+15	2,45E+15	2,45E+15	2,45E+15	2,45E+15	n.a.	n.a.
21	Polystyrene	n.a.	n.a.	4,91E+14	4,91E+14	4,91E+14	4,91E+14	n.a.	n.a.
22	Services	5,41E+18	5,57E+18	1,43E+18	1,46E+18	1,43E+18	1,46E+18	4,18E+18	4,18E+18
23	Ferric chloride	4,86E+17	4,86E+17	9,72E+16	9,72E+16	9,72E+16	9,72E+16	n.a.	n.a.
24	Aluminium (Billet)	6,66E+16	6,66E+16	1,33E+16	1,33E+16	1,33E+16	1,33E+16	n.a.	n.a.
25	Lead	n.a.	n.a.	1,51E+17	1,51E+17	1,51E+17	1,51E+17	1,51E+17	1,51E+17
	Total Emergy U in seJ / yr	3,98E+19	4,00E+19	8,50E+18	8,54E+18	8,50E+18	8,54E+18	4,44E+18	4,44E+18
	UEV, in seJ / ton OBP	1,06E+15	1,06E+15	2,26E+14	2,27E+14	2,26E+14	2,27E+14	1,18E+14	1,18E+14
	Total R	8,27E+17	8,50E+17	2,36E+17	2,40E+17	2,36E+17	2,40E+17	6,38E+17	6,38E+17
	Total N	3,25E+19	3,25E+19	6,50E+18	6,50E+18	6,50E+18	6,50E+18	0,00E+00	0,00E+00
	Total F	6,49E+18	6,64E+18	1,76E+18	1,79E+18	1,76E+18	1,79E+18	3,80E+18	3,80E+18

Table 14: Saved emergy (EMS), invested emergy (EMI) and Net – Emergy of each evaluated scenario.

	Scenario Name	EMI (seJ/yr)	EMI (seJ/ ton OBP yr)	EMS (seJ/ ton OBP yr)	Net Emergy (seJ/ ton OBP yr)
#I	**Landfilling 100%**	3.98E+19	1.06E+15	Zero	-1.06E+15
#II	**Electricity 100%**	4.00E+19	1.06E+15	6.37E+13	-9.99E+14
#III	**Donation 80% + landfilling 20%**	8.50E+18	2.26E+14	Zero	- 2.26E+14
#IV	**Donation 80% + electricity 20%**	8.54E+18	2.27E+14	1.27E+13	- 2.14E+14
#V	**Avoided Production 80% + landfilling 20%**	8.50E+18	2.26E+14	6.56E+15	6.33E+15
#VI	**Avoided Production 80% + electricity 20%**	8.54E+18	2.27E+14	6.57E+15	6.35E+15
#VII	**Biorefinery 100%**	4.44E+18	1.18E+14	Zero	-1.18E+14
#VIII	**Biorefinery 100% + Avoided Production**	4.44E+18	1.18E+14	1.80E+14	6.23E+13

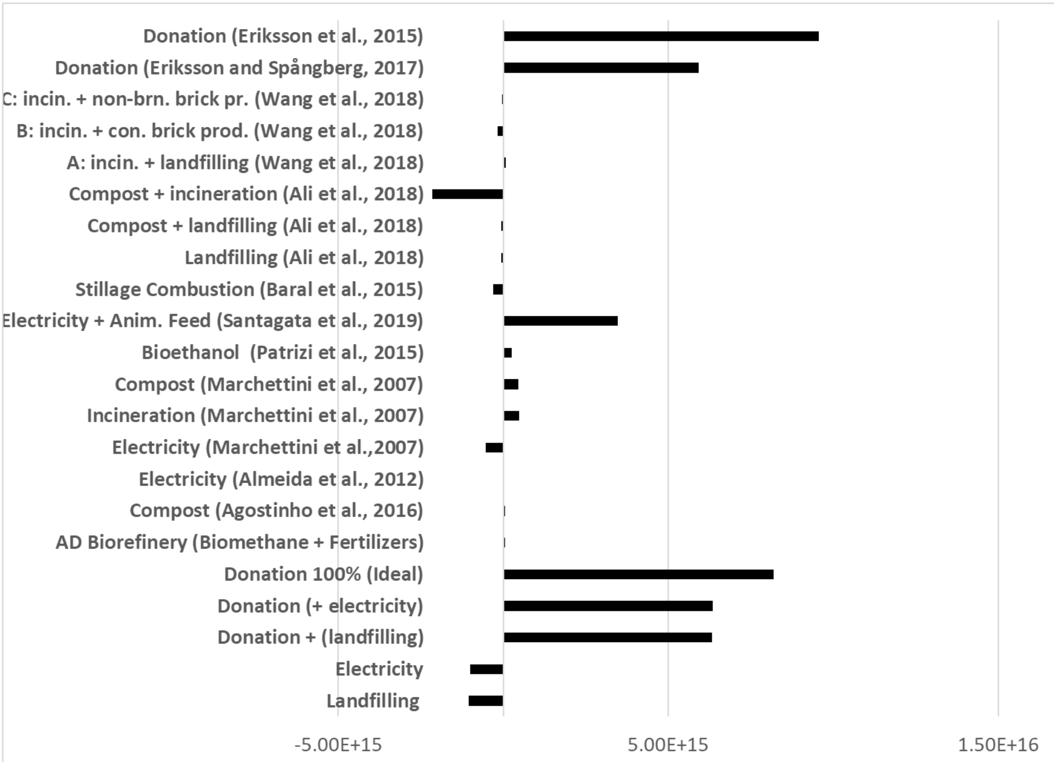

Figure 40: Net emergy (in seJ/ton) comparison among different FRH options.

5.2.4. Exploring the correlation between invested and saved emergy

Since the variation of net-emergy along the FRH is strictly correlated to the invested and saved emergy, it is important to assess potential mathematical relations for its representation. Along the FRH, each waste management option has its own characteristics that depend on international standards, physical parameters and constrains, while the choice, among the several proposed options for waste management, is on the hands of the decision maker. However, this conceptual approach also involves some physical factors that can be measured. Among others, the potential correlation between the emergy investment for a system's implementation and the emergy saved from the operational phase are still superficially explored in the emergy literature.

Figure 41 shows the EMS as the dependent variable of EMI along the FRH. The graph includes four scenarios evaluated in this work (#I, #II #VI and #VIII), as well as other scenarios from the literature, including: landfilling with energy recovery, incineration and composting scenarios in Italy (Marchettini et al., 2007), organic compost and abiotic recycling process from a municipal solid waste recycling plant in São Paulo (Agostinho et al., 2013), electricity production from Biogas generated by a landfill in Sao Paulo city (Almeida et al., 2012), EMS per ton of organic waste (animal fat) generated by a slaughterhouse to produce electricity and animal feed (Santagata et al., 2019), a biorefinery with ethanol production fed by straw from agriculture and residual geothermal heat (Patrizi et al., 2016), EMS per ton of theoretical biorefinery scenarios fed by cellulosic stillage (Baral et al., 2016), EMS and EMI of scenarios of Landfilling, Compost + Landfilling and Compost + Incineration in Pakistan (Ali et al., 2018), and finally incineration with and without bricks production in China (Wang et al., 2018). Furthermore, due to the lack of other works regarding donation scenarios, it was calculated the EMI and EMS of a theoretical donation scenario at CEAGESP with 100% of OBP (therefore without RF sent to landfill), and two other donation scenarios in Sweden using the LCA inventory from Eriksson et al. (2015) and Eriksson and Spangberg (2017).

Results of EMI versus EMS are shown in Figure 41, indicating that scenarios with low priority in the FRH (landfilling, energy recovery from landfill and incineration) have low or no ability in recovering emergy. Conversely, all the four high priority scenarios (donation) were able to save high amounts of emergy per ton OBP by demanding lower emergy investment. It is interesting to note that, for those FRH levels with less priority (including landfilling, energy recover, composting, and biorefinery) there is no considerable increase in the amount of EMS, with the exception of electricity and animal feed generated by a biorefinery scenario fed by slaughterhouse waste; this is potentially caused by the different kind of organic waste considered (100% animal fat vs mainly vegetable waste in all other cases).

Overall, the scenarios distribution according to their EMS as a function of their respective EMI (Figure 41) seems to follow the triangular shape of the waste management concept, and it suggests a non-linear decreasing trend for the saved emergy along the hierarchical levels from the highest to the lowest ones. Highest priority levels (donation in this case) are able to recover several times more emergy than its invested emergy when comparing to those options with lowest priority levels, located in the bottom of triangle. The particular disposition of the points in Figure 41 suggests a non-linear decreasing trend. The following equation (7) is herein considered to describe such graphical behavior:

$$EMS = \frac{a}{EMI^b} \qquad (7)$$

Where: EMS is the saved emergy, EMI the invested emergy, a and b are the two parameters to be determined.

By using the Solver tool provided by Microsoft Excel, and following Brown's (2001) method to determine the parameters according to the least square method and the subsequent calculation of the R-squared, the quality of the obtained mathematical model was assessed; representing how the model fits the points distribution. Although some authors like Spiess and Neumeyer (2010) understand that R^2 would be not the most adequate index to evaluate the quality of non-linear mathematical models, R^2 is used in this study due to its easy application and easy-to-understand results.

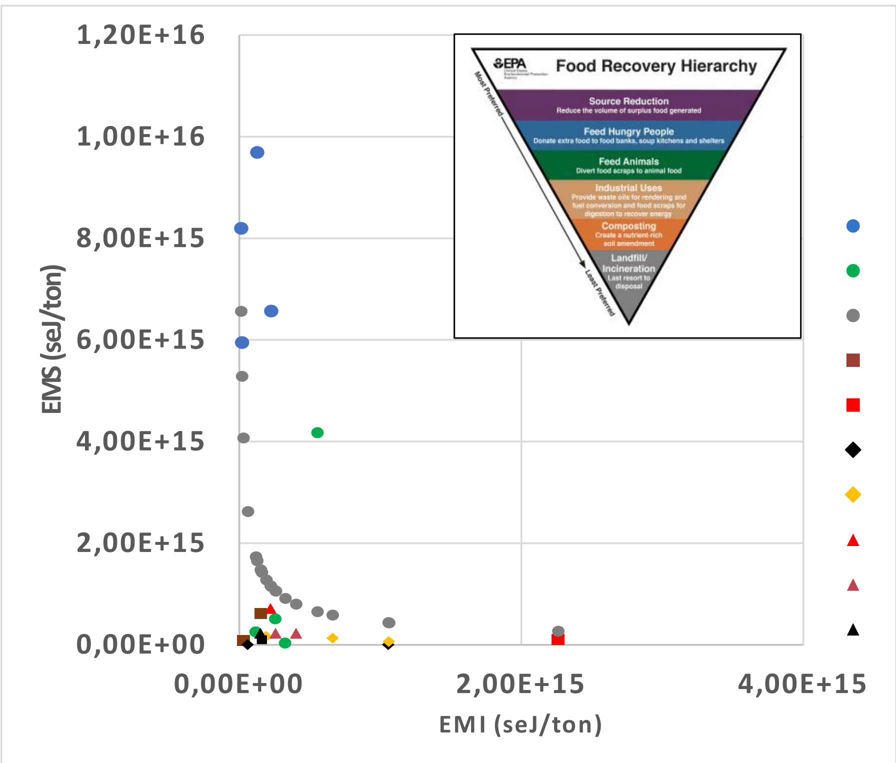

Figure 41:Saved Emergy (EMS) as a function of invested emergy (EMI). Graph obtained with results of this study. Triangle for food recovery hierarchy obtained from www.epa.gov/sustainable- management-food/food-recovery-hierarchy. Details in appendix D.

By applying the least square methods the results show that Equation (8) is the one with highest quality, with an error sum of 1.42E+32 and $R^2 = 0.25$.

$$EMS = \frac{1.43E+24}{EMI^{0.634}} \quad (8)$$

Where: EMS is the saved emergy and EMI the invested emergy.

To verify the capacity of other mathematical models to describe the points distribution, linear, logarithmic and polynomial models as available in Microsoft Excel were assessed, and the results are shown in Table 15.

Table 15: Variables, equations and R^2 for alternative mathematical models to represent point distribution in Figure 41.

Model	General equation	Specific Equation	R^2
Proposed	$y= a/x^b$	$EMS = 1.43E24/EMI^{0.634}$	0.2527
Linear	y = ax + b	EMS = -1.6063EMI + 2E+15	0.0743
Logaritmic	y = aln(x) + b	EMS = - 1E+15ln(EMI) + 4E+16	0.1934
Polynomial	$y = ax^2 + bx + c$	$EMS = 2E-15(EMI)^2 - 5.071(EMI) + 3E+15$	0.1159

Although being the best option among the ones presented in Table 14, the obtained value of R^2 equal to 0.25 in the proposed model depicts a weak relation, indicating that 25% of the variability of EMS depends on EMI along the FRH. Therefore, considering the amount of data used for this analysis, the obtained mathematical representing Figure 41 could be considered as featuring a sufficient level of reliability. With more data available, along the years, from other studies, Figure 41 can be updated and a more accurate mathematical model can be obtained. One aspect can be stated: the considerable low R^2 of 0.0743 for the linear function suggests that a non-linear trend of EMS=f(EMI) along the FRH is very probable.

Although recognized as an important aspect to better understand the relationship between EMI and EMS, there are limitations on the obtained mathematical model representing Figure 41: (1) the small sample of 20 points; (2) the presence of mixed scenarios - in the ideal case, the comparison should be made only among more ‘pure’ scenarios, excluding, for instance, donation + landfilling, or compost + incineration, since this aspect can affect the EMI or EMS, as in this study, where 94% of EMI in donation scenarios is related to the residual 20% sent to landfill; (3) different authors’ assumptions related to the inputs used in landfills - some authors did not include geological materials as rocks and soil, disregarding their relevance as emergy contributors; (4) the model does not include the characteristics of the OBP considered – for example, fruit and vegetables should not be directly comparable with slaughterhouse waste - and the presence of local variables that could affect the numbers.

Without disregarding the existing limitations, the results of this mathematical approach focusing on emergy have depicted that waste management options located at the top of the FRH can save far more emergy than options located at the bottom, the latter associated with a relatively high amount of EMI. This finding scientifically supports the validity of the FRH, also from an emergy donor-side perspective.

5.3 Implications of the CEAGESP OBP management scenarios for public policies

The different OBP management options imply, at different rates, the use of natural resources (emergy perspective) and the emissions of different types of pollutants (LCA perspective). Choosing an option over another depicts a wide range of aftermaths that need to be assessed from a holistic perspective. With the goal of helping decision makers towards strategies for more sustainable OBP management, a simple multi-criteria tool that considers different connections between the chosen option and consequent public policies implications is herein developed, since not one multicriteria approach that would satisfy the specific needs of this work was found in the scientific literature.

The sustainable development goals proposed by the UN (2015) are chosen as important drivers. The 2030 UN agenda established 17 Sustainable Development Goals (SDGs) to be achieved by 2030, as shown in Figure 42. The 17 goals are an urgent call for action by all countries - developed and underdeveloped - in a global partnership, recognizing that ending poverty and other deprivations must go hand-in-hand with strategies that improve health and education, reduce inequality, spur economic growth, tackling climate change and working to preserve oceans and forests.

Figure 42: Graphic representation of the SDGs (UN, 2015).

Decisions towards a sustainable development imply the inclusion of different aspects that could be assessed under multi-criteria decision making (MCDM) based on a decision matrix. According to Triantaphyllou et al. (1998), the typical MCDM problem deals with the evaluation of a set of alternatives (options) in terms of a set of decision criteria, and involves the following steps: (1) determination of the relevant criteria and alternatives (options); (2) Attribution of numerical measures to the relative importance of the criteria and to the impacts of the alternatives on these criteria; (3) processing the numerical values to determine a ranking for each alternative; (4) the option with the highest score will be the best one, according to the chosen criteria. The flowchart in Figure 43 shows the steps of the proposed multicriteria approach in a more detailed way. In this work, the potential options are the evaluated scenarios (step 1). The factors that influence the results are the performance of the assessed scenarios (environmental from LCA and EMA indicators), the economic cost to manage 1 ton of OBP, and the associated SDGs (step 2). The score to each option is attributed according to its relative position obtained in LCA impact categories, EMI, net-emergy, and the cost to manage 1 ton of OBP, multiplied by the number of SDGs involved (step 3). Finally, the option with the highest score is considered as the one that should be firstly supported by public policies (step 4). The proposed approach is named “sustainable performance score” (SPS).

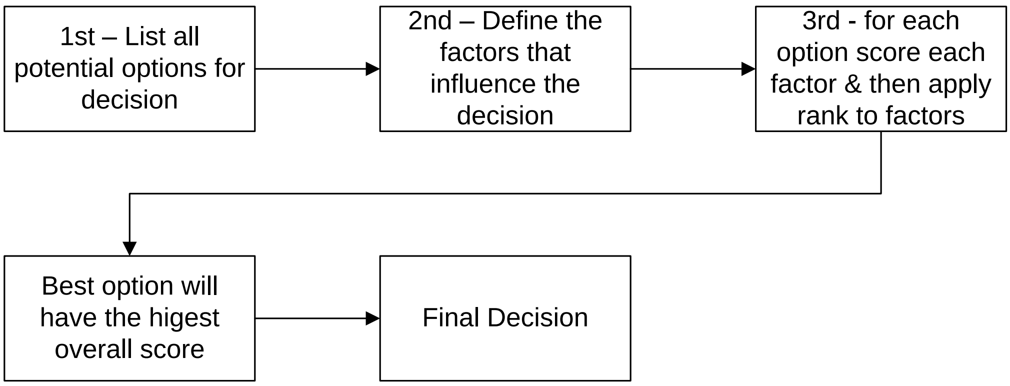

Figure 43: Flowchart for the sustainable performance scores (SPS) steps.

To apply the SPS, the 17 SDGs are initially allocated into all the waste management scenarios evaluated, resulting as in Figure 44. Scenario #I, sanitary landfilling, despite being the least recommended by the FRH, results in environmental savings when compared with open dumps. A sanitary landfill aims to reduce the number of deaths and illnesses from hazardous chemicals, air, water and soil pollution and contamination, as recommended by SDG #3 (ensure healthy lives and promote wellbeing), specific goal 3.9. Additionally, due to leachate treatment, sanitary landfill is related to SDG #6 (clean water and sanitation) since it improves water quality by reducing pollution, eliminates dumping, collaborates to halving the proportion of untreated wastewater, and increasing recycling. Finally, due to 80% of CH_4 flared, it contributes to reduce GWP, as aimed by SDG#13.

Figure 44: Allocation of the SDGs into the assessed OBP management scenarios

Scenario #II, electricity generation at landfill, in addition to all the benefits of scenario #I, has the benefits related to energy recovery, as highlighted in SDG # 7 (affordable and clean energy). In particular, the SDG # 7 recommends increasing the share of the renewable energy in the global energy mix to improve energy efficiency, especially in developing countries.

Scenarios #III and #V, related to food donation, involve the following SDGs: SDG #2 (Zero Hunger), which recommends the end of hunger and ensuring food access by all people by 2030, particularly the poorest ones, and those in vulnerable situations. This is very important in Brazil, due the chronic problems related to malnutrition, which still affects around 50 million people (Herz and Porpino, 2017), a situation that was worsened during the Covid-19 pandemic (Silva et al., 2021); SDG #3, since a healthier diet reduces diseases and mortality, especially in newborn and children under five years; SDG #6, due to the avoided leachate generation downstream and the avoided water consumption in case of product substitutions; SDG #12, which recommends reducing per-capita food waste at retail and consumer levels, and reducing food losses along the production and food supply chains; SDG #13 (climate actions), due to the avoided methane emissions at landfill and transport emissions, and, if the avoided production is included, food donation also contributes to avoided impact related to avoided agricultural production; SDG #15 (protect, restore, and promote the sustainable use of terrestrial ecosystems) due to the avoided downstream pollution derived by avoided waste landfilling, and if the avoided production is considered, the related saved natural resources.

Scenarios #IV and #VI contribute to all the same SDGs as do scenarios #III and #V, added to the benefits related to SDG #7, about clean energy production of the residual fraction.

Scenarios #VII and #VIII, related to a Biorefinery pathway, contribute to the following SDGs: SDG #3, due to avoided landfilling and related health benefits, SDG #7 due to biomethane production, which is a renewable energy resource, SDG #9 (built resilient infrastructure, promote inclusive and sustainable industrialization) due to the pivotal characteristics of a biorefinery facility within a context of circular economy pathways, SDG #12 by promoting waste recycling through a closing cycle related to fertilizers production, SDG #13, as biomethane replaces natural gas use and biofertilizers substitutes for chemical fertilizers (from fossil fuel) production, SDG #15, which promotes the conservation, restoration and sustainable use of terrestrial and inland freshwater ecosystems as well as actions to reduce the degradation of natural habitats (biorefinery avoids landfilling with consequent soil loss and leachate emissions).

Once the SDGs related to each scenario are set, the next step of SPS is the attribution of a score related to the performance of each scenario for each indicator. Indicators comprehend the nine impact categories of LCA, the invested emergy (EMI), the net-emergy, and the economic cost to manage 1 ton of OBP. This last value was obtained during fieldwork at CEAGESP and represents the costs for scenario #I achieving 87 R$/ton of OBP. For biorefinery and donation scenarios, the cost was estimated (see Appendix E) as 33R$/ton and 12 R$/ton of OBP, respectively. The scores are attributed as follows: for each indicator, the scenario with the best performance receives a score equal to 8, the second-best performance equal to 7, and so on, until the worst ranked scenario that receive the minimum score of 1 is reached. Number 8 was chosen because there are 8 scenarios being evaluated in this work. In case of equal performance for an indicator, the same score is attributed for scenarios. The highest difference among the different levels is equal to 1, except in case of *ex aequo*.

After the score attribution, the next step is the sum of the scores. The obtained value represents the relative “performance” of each scenario, when compared to others. Each value is then multiplied by the number of SDGs associated with the scenario, as expressed by Equation (9).

$$SPS = \sum_{i}^{n} iscore \sum_{SDG}^{n} SDG \qquad (9)$$

Where: SPS is the “sustainability performance score”, i represents the chosen indicator, *iscore* is the score attributed in each indicator and SDG is the number of SDGs considered.

Applying Equation 9 on all evaluated scenarios, the sustainable performance scores (SPS) are shown in Table 16. The highest score of 637 points is obtained by scenario #VI, which considers 80% of OBP as NMF donated, including the electricity production and all the avoided emissions. The second place (510 points) is obtained by scenario #V, similar to #VI but without electricity production at landfill. The third best result (438 points) is achieved by scenario #VIII (biorefinery that includes all the benefits derived by natural gas and fertilizers substitution). The worst SPS value was obtained by the landfilling scenario (45 points for scenario #I), while electricity production at landfill for scenario #II shows a slightly better result (124 points). As a result, the SPS framework indicates that donation scenarios should receive priority for public polices, followed by the biorefinery scenarios.

As highlighted by Henz and Porpino (2017), due to the issues related to food security, a constant challenge in Brazil is to find ways to reduce food waste in face of the cyclical economic and social crises, especially because Brazil has high socioeconomic inequalities (GINI index of 53.9). In this work, it was found that considering 80% for NMF redistribution on the current amount of waste by CEAGESP, and a daily consumption of horticultural products at 500 g per capita (slightly above the minimum of 400 g/capita/day, as recommended by the World Health Organization), the charity institutions that receive donation from CEAGESP's food bank could provide food for about 165,000 individuals per day. In a theoretical case where the entire diet is based on the food provided by CEAGESP's food bank, about 13,437 individuals per day could be fed under an energy requirement of 2,550 kcal/day. The hunger should be eradicated by 2030 according to SDG #2 (UN, 2015), and food donation by centralized food centers like CEAGESP plays an important role in achieving such goal, as also highlighted by Sudin et al. (2022). Furthermore, the new Brazilian legislation addressing the fight against hunger (LF - 14.016/2020) exempts the food donor and the probable intermediary from any responsibility after the first delivery of the food and may be liable for damages only if there is an intention to harm. This new law removes barriers to donation and, at the same time, ensures the prevention of food loss and waste, as recommended by Law 12305 – Brazilian National Policy on Solid Waste (NPSW, 2010).

The benefits related to food donation could interest a wider range of areas beyond the downstream impacts derived from the avoided by-products landfilling and the avoided impacts related to food substitution. When considering the rebound effect concept, figures change. Authors such as Sudin et al. (2022) have estimated a rebound effect associated with re-spending of substitution-related monetary savings. In other words, the beneficiaries of food donation can spend the money that they have not spent purchasing the equivalent amount of donated food. These alternative purchases could decrease the GWP benefits up to 50%. While acknowledging the importance of the results obtained by Sudin et al. (2022), there are probably other aspects related to the benefits of food donation that should deserve more attention. In fact, according to Gundersen and Ziliak (2015), food insecurity is negatively associated with health, and taking children as an example, it increases risks of birth defects, anemia, cognitive problems, aggression, anxiety, higher risks of being hospitalized and poorer general health. In case of adults, food insecurity is associated with decreased nutrient intakes and increased rate of mental health problems and depression, diabetes, hypertension, and hyperlipidemia. All these problems imply indirect additional economic, social and environmental costs that food donation could help to relieve. For example, a regular and healthy diet, could reduce the necessity of hospitalization and/or reduce the need of medication, receiving environmental benefits derived by the avoided hospital waste generation and avoided medication production. All these benefits, which could be considered another kind of rebound effect, could overcome the benefit loss found by Sudin et al. (2022). Therefore, this study has shown the great potential contribution of CEAGESP's NMF donation scenarios for both the environment and society.

The third position achieved by scenario #VIII (Table 15) shows that a biorefinery facility is also a desirable option for the management of CEAGESP's OBP. As also highlighted by Ardolino et al. (2018) and Guo et al. (2021), biogas upgrading to biomethane shows a better environmental performance than flaring the biogas for electricity and heat generation. This is especially true in the Brazilian context, where the electricity is mainly based on renewable resources (~80% including hydropower, biomass, wind and solar; Griebenow and Ohara, 2019), therefore, Brazilian electricity substitution with biogas electricity does not achieve important environmental benefits; this aspect was also confirmed by Linkanen et al. (2018) and Mendes et al. (2004). Conversely, biomethane production being able to replace natural gas shows interesting perspectives.

According to Probiogas (2015), the entrance of biomethane from sewage treatment plants into the market is regulated by the Brazilian National Agency of Oil, Natural Gas and Biofuels through the Resolution No. 8 of January 30, 2015 (ANP, 2015). Moreover, the chemical-physical characteristics of biomethane from agricultural and silvicultural organic waste intended for vehicular use, residential and commercial installations are also regulated throughout the national territory. The resolution also determines the obligations regarding the control of the quality to be achieved by the various economic agents that sell the product. All the standards regarding pressure, distribution and resale of biomethane are the same as for natural gas.

More recently, the Brazilian national biofuels policy instituted by law nº 13.576/2017 known as RenovaBio (2017) has established the following objectives: (1) Provide an important contribution to the fulfillment of certain compromises in Brazil within the scope of the Paris Agreement; (2) Promote the adequate expansion of biofuels in the energy matrix,

Table 16: Sustainability performance score (SPS) calculation results

Scenario	Scenario's name	FDP	FEP	GWP	HTP	MDP	PMFP	POFP	TAP	WDP	EMI	Net-EM	BRL/ ton	Sum	SDGs	Score	Position
Scenario #I	Landfilling (100%)	1	1	1	1	1	1	2	1	2	2	1	1	15	3	**45**	**8th**
Scenario #II	Electricity (100%)	5	2	2	5	2	2	1	2	6	1	2	1	31	4	**124**	**7th**
Scenario #III	(Donation 80% + Landfilling 20%)	2	3	3	2	4	3	5	3	3	6	3	8	45	6	**270**	**6th**
Scenario #IV	(Donation 80% + Electricity 20%)	4	4	4	4	5	5	4	4	4	4	4	8	54	7	**378**	**4th**
Scenario #V	(Avoided production 80% + Landfilling 20%)	7	7	7	7	7	7	8	7	7	6	7	8	85	6	**510**	**2nd**
Scenario #VI	(Avoided Production 80% + Electricity 20%)	8	8	8	8	8	8	7	8	8	4	8	8	91	7	**637**	**1st**
Scenario #VII	Biorefinery (100%)	3	5	5	3	3	4	3	5	1	8	5	5	50	6	**300**	**5th**
Scenario #VIII	Biorefinery + Avoided Production (100%)	6	6	6	6	6	6	6	6	5	8	6	6	73	6	**438**	**3rd**

Notes: Scenario #II was assumed to have the same cost as #I, as well as all the donation scenarios (#III to VI) were assumed to have the same cost, while the minor score of Scenario #VII is due to the monetary gain related to Biomethane selling not being included.

with emphasis on the regularity of the supplying; (3) the contribution of biofuels to the security of the national fuel supply, environmental preservation and the promotion of economic and social development and inclusion. This context constitutes the appropriate background for biofuel.

Finally, according to Probiogas (2015), it is possible to use biomethane in Otto cycle gasoline engines already engineered for such use, as well as in diesel cycle gas engines. In both cases, it is possible to switch from the gas to a liquid fuel mode. Biomethane is stored compressed (about 250 bar) in suitable tanks installed on the vehicle. In this context, the biorefinery scenario modelled in this work can provide its contributionThe biorefinery scenario at CEAGESP is able to generate 1,135,464 Nm^3 CH_4/year of biomethane, and considering the coefficients provided by Ardolino et al. (2018) - 4.56 Nm^3CH_4/100 km of regular car consumption -, the CEAGESP biomethane could support a 24,900,526 km trip per year. Considering an average number of km travelled by a Brazilian passenger equal to 12,900 km/vehicle year, CEAGESP biomethane could support the energy for 1,930 vehicles per year.

Fertilizers are another co-product of the AD biorefining process. Their production from CEAGESP OBP also shows interesting perspectives for the Brazilian context. According to Oliveira et al. (2019), Brazil has become the fourth largest food producer in the world, but a growing expansion of agribusiness put pressure on the national production of fertilizers. As a result, Brazil imports 80% of the fertilizers it uses (NFP, 2022), including phosphate, potassium and nitrogen fertilizers, mainly from Russia. According to Oliveira et al. (2019), the reduction in these importations should help agriculture and the domestic economy to produce food in a more profitable way and with higher sustainability in all aspects. The recent National Fertilizer Plan 2022 – 2050 (NFP, 2022) was designed to promote the domestic production of fertilizers, including aspects such as business and research, development, and innovation. In this context, the fertilizers produced by the CEAGESP OBP have great potential for contribution. For example, by considering an average annual production of 128,999 kg N, 15,840 kg P and 48,159 kg K, and an average consumption per hectare of 300 kgN/year in coffee production (Sanzonowicz et al., 2003), 160 kg P_2O_5/year and 50 kgK_2O/year in soy production (Oliveira et al., 2007), the CEAGESP OBP fertilizers can cover the yearly fertilizer needs of 430 hectares, 99 hectares and 963 hectares for N, P and K, respectively. This application of OBP fertilizers in croplands is of fundamental importance, since it corresponds to the step needed to close the cycle of nutrients (from field to field) in a circular economy perspective.

All these numbers emphasize that, as well as for the donation scenarios studied, the biorefinery scenarios could provide great benefits for both the environment and society.

6. CONCLUSIONS

Under an LCA perspective, donating the OBP edible fraction (NMF) generated by CEAGESP results, by far, in the least environmental burdens compared to all alternative evaluated scenarios. While biorefinery scenario is located in an intermediary position, landfilling (with or without energy recovery) has shown to be the worst option, as far as managing the OBP is concerned. When landfilling is compared to donation scenarios including the avoided impacts, it causes 80 times more fossil depletion, 520 times more global warming, 115 times more human toxicity, 18 times more metal depletion, and 73 times more water depletion than the latter. The biorefinery scenario causes 33, 275, 100, 12 and 72 times more the same impacts, respectively. Therefore, the applied LCA shows that implementing a smarter management for the CEAGESP's OBP under donation pathways as a first option, and considering biorefinery as a second choice, are better alternatives than the current landfilling-based management.

Local variables have shown high influence on LCA results for some impact categories, as exemplified by the hydropower-based Brazilian electricity on the WDP. This work contributes to recognizing that OBP generated by food supply centers should be seen rather as a wealth than as a problem. Rather than just organic waste disposed in landfills, options exist for OBP management to achieve better environmental, social and economic performances. Since up to 80% of CEAGESP's OBP have a potential as edible food with high nutritional quality (NMF), a smarter management to maximize efficiency is of fundamental importance, as well as a biorefinery facility, valorizing the OBP through the production of biomethane and fertilizers. This is especially important for a strategic national development oriented to reduce the imports and external dependence for energy and fertilizers.

Under an emergy perspective, the donation scenarios showed far higher emergy savings, compared to landfilling (with or without electricity recovering) and biorefinery scenarios. Donation is able to save 29 times more emergy than its emergy investment, biorefinery scenario 1.5 times, and landfilling with energy recovering achieved 6%.

Analyzing the saved emergy (EMS) as a function of the emergy invested (EMI), the existence of a non-linear decreasing trend allied with the concept of triangular hierarchy of waste management was found, from the highest to the lowest priority levels. Although additional studies are still needed to confirm this hypothesis, the evidences obtained in this study support the concept of waste management options also under a donor-side perspective.

From the proposed approach named 'sustainability performance score' (SPS) to support public policies, the donation and biorefinery scenarios have obtained the highest scores also by accounting for the number of SDGs achieved, demonstrating their potential for OBP management under a holistic perspective. Precisely, the top three scenarios that should be prioritized are donations #VI and #V, and biorefinery #VIII, with 637, 510 and 438 points, respectively.

7 SUGGESTIONS FOR FUTURE STUDIES

(a) CEAGESP OBP has huge variability, inconstancy and perishability that could affect the efficiency of donation scenarios. This could be overcome by transforming/processing the unsold products into other variations, such as soups, jams, juices, amoing others.

(b) The rebound effect related to donation scenarios (health of people receiving the food) needs deeper understanding.

(c) Donation and biorefinery scenarios in different countries and realities (including differences in the OBP) are still an undiscovered field that deserve to be assessed mainly from an emergy perspective, since LCA data can be more easily found.

(d) Regarding the EMS = (f)EMI study, more data are necessary to confirm the hypothesis discussed in this thesis. More than a higher amount of data, higher quality is also required. Emergy studies should be standardized when possible to allow for better comparisons, since studies were identified that accounted for certain items not considered in other studies, bringing uncertainties for the EMS=(f)EMI analysis.

(e) Better explore the potentialities of the proposed sustainability performance score (SPS) framework, including the establishment of weights of importance for indicators from participative meetings with experts in the topic.

8. REFERENCES

Agostinho, F., Ortega, E., 2013. Energetic-environmental assessment of a scenario for Brazilian cellulosic ethanol. Journal of Cleaner Production, 47, 474-489. Doi:10.1016/j.jclepro.2012.05.025

Agostinho, F., Almeida C. M. V. B., Bolilla, S., H., Sacomano, J., B., Giannetti B., F., 2013. Urban solid waste plant treatment in Brazil. Is there a net emergy yield on the recovered materials? Resources conservation and Recycling, 73, pp.143 – 155. Doi:10.1016/j.resconrec.2013.02.001

Ait Sair, A., Kansou, K., Michaud, F., Cathala, B., 2021. Multicriteria definition of small-scale biorefineries based on a statistical classification. Sustainability, 13(13), 7310. Doi: 10.3390/su13137310

Albizzati, P., F., Tonini, D., Chammard, C.,P., Astrup, T., F., 2019. Valorisation of surplus food in the French retail sector: Environmental and economic impacts. Waste Management, 90, 141-151. Doi:10.1016/j.wasman.2019.04.034

Albizzati, P. F., Tonini, D., Astrup, T. F., 2021. High-value products from food waste: An environmental and socio-economic assessment. Science of the Total Environment, 755, 142466. Doi: 10.1016/j.scitotenv.2020.142466

Alexander, P., Brown, C., Arneth, A., Finnigan, J., Moran, D., Rounsevell, M.D., 2017. Losses, inefficiencies and waste in the global food system. Agric. Syst. 153, 190–200. https://doi.org/10.1016/j.agsy.2017.01.014.

Alhashim, R., Deepa, R., Anandhi, A. (2021). Environmental Impact Assessment of Agricultural Production Using LCA: A Review. Climate, 9(11), 164. Doi: 10.3390/cli9110164

Ali, M., Marvuglia, A., Geng, Y., Chaudhry, N. and Khokhar, S., 2018. Emergy based carbon footprinting of household solid waste management scenarios in Pakistan. Resources, Conservation and Recycling, 131, pp.283-296. Doi: 10.1016/j.resconrec.2017.10.011

Alibardi, L., Astrup, T. F., Asunis, F., Clarke, W. P., De Gioannis, G., Dessì, P., ... Spiga, D., 2020. Organic waste biorefineries: looking towards implementation. Waste Management, 114, 274-286. Doi:10.1016/j.wasman.2020.07.010

Amiri, Z., Maghsoudi, A., Asgharipour, M.R., Nejati-Javaremi, A., Campbell, D.E., 2022. The semi-intensive production model: A strategy based on emergy and economic analyses to realize sustainability in the ecosystem of Sistani beef cattle raising in Iran. Journal of Cleaner Production, p.132304. Doi: 10.1016/j.jclepro.2022.132304

Almeida, C.,M.,V., B, Frimaio, G., S., Bonilla, S.,H., Da Silva, C., C., Giannetti, B., F., 2012. An evaluation of a MSW-to-energy system using emergy synthesis. International journal environment andsustainable development, 11 (3), pp.258 – 273.

Andersen, J.K., Boldrin, A., Christensen, T.H., Scheutz, C., 2012. Home composting as an alternative treatment option for organic household waste in Denmark: an environmental assessment using Life Cycle Assessment. Waste Management, 32, pp. 31 – 40. Doi: 10.1016/j.wasman.2011.09.014

ANP, 2015. Agência Nacional de Petróleo, Gás Natural e Biocombustíveis. Available at: http://www.anp.gov.br/

Ardolino, F., Parrillo, F., Arena, U., 2018. Biowaste-to-biomethane or biowaste-to-energy? An LCA study on anaerobic digestion of organic waste. Journal of Cleaner Production, 174, pp.462-476. Doi: 10.1016/j.jclepro.2017.10.320

Banks, C. J., Chesshire, M., Heaven, S., and Arnold, R., 2011. Anaerobic digestion of source-segregated domestic food waste: performance assessment by mass and energy balance.Bioresource technology, 102(2), 612-620. Doi: 10.1016/j.biortech.2010.08.005

Baral, R., N., Wituszynski, D., M., Martin, J., F., Shal., A. 2016. Sustainability assessment of cellulosic biorefinery stillage utilization methods using emergy analysis. Energy, 103, 16 – 28. Doi: 10.1016/j.energy.2016.04.085

Bassi, S.A., Boldrin, A., Frenna, G. and Astrup, T.F., 2021. An environmental and economic assessment of bioplastic from urban biowaste. The example of polyhydroxyalkanoate. Bioresource Technology, 327, p.124813. Doi: 10.11583/DTU.13636436

Basulto, F.S., Duch, E.S., y Gil, F.E., Plaza, R.D., Saavedra, A.L. Santamaría, J.M., 2009. Postharvest ripening and maturity indices for maradol papaya. Interciencia, 34(8), pp.583-588.

Bauer, F., Hulteberg, C., Persson, T.,Tamm, D., 2013. Biogas upgrading-Review of commercial technologies; https://www.osti.gov/etdeweb/servlets/purl/22090519

Beretta, C., Hellweg, S., 2019. Potential environmental benefits from food waste prevention in the food service sector. Resources, Conservation and Recycling, 147, 169 – 178. Doi: 10.1016/j.resconrec.2019.03.023

Bergström, P., Malefors, C., Strid, I., Hassen, J., O., Eriksson., M. 2020. Sustainability Assessment of Food Redistribution Initiatives in Sweden. Resources, 9, (27). Doi:10.3390/resources9030027

Bernstad, A. and la Cour Jansen, J., 2011. A life cycle approach to the management of household food waste–a Swedish full-scale case study. Waste management, 31(8), pp.1879-1896. Doi: 10.1016/j.wasman.2011.02.026

Bocchiglieri, M. M., 2010. O lixiviados dos aterros sanitários em estações de tratamento de sistema público de esgoto. Universidade de Sao Paulo, Faculdade de Saude pública. Doctoral thesis.

Boldrin, A., Andersen, J.K., Møller, J., Christensen, T.H. and Favoino, E., 2009. Composting and compost utilization: accounting of greenhouse gases and global warming contributions. Waste Management and Research, 27(8), pp.800-812. Doi: 10.1177/0734242X09345275

Boldrin, A., Andersen, J.K., Christensen, T.H., 2011. Environmental assessment of garden waste management in the Municipality of Aarhus, Denmark. Waste Management, 31, pp. 1560-1569. Doi: 10.1016/j.wasman.2011.01.010.

Bovea, M.D., Powell, J.C., 2006. Alternative scenarios to meet the demands of sustainable waste management. Journal of Environmental Management, 79, 115 - 132 Doi:10.1016/j.jenvman.2005.06.005

Brancoli, P., Bolton, K., Eriksson, M., 2020. Environmental impacts of waste management and valorization pathways for surplus bread in Sweden. Waste Management, 117, 136 – 145. Doi:10.1016/j.wasman.2020.07.043

Brandt-Williams, S.L., 2002. Handbook of emergy evaluation: a compendium of data for emergy computation issued in a series of Folios. Center for Environmental Policy Environmental Engineering Science. University of Floriga, Gainesville.

Briones, L., and Escola, J. M., 2019. Application of the Microsoft Excel Solver tool in the solution of optimization problems of heat exchanger network systems. Education for Chemical Engineers, 26, 41-47. Doi: 10.1016/j.ece.2018.10.003

Brogaard, L., K., Stentsoe, S., Willumsen, H., C., Christensen, T., H.,.2013. Quantifying capital goods for waste landfilling. Waste management and Research, 31 (6), pp. 585 – 598. https://doi.org/10.1177/0734242X13482032

Brown, A. M., 2001. A step-by-step guide to non-linear regression analysis of experimental data using a Microsoft Excel spreadsheet. Computer methods and programs in biomedicine, 65(3), 191-200. Doi: 10.1016/S0169-2607(00)00124-3

Brown, A. M., 2006. A non-linear regression analysis program for describing electrophysiological data with multiple functions using Microsoft Excel. Computer methods and programs in biomedicine, 82(1), 51-57.

Brown, M.T. and Ulgiati, S., 2002. Emergy evaluations and environmental loading of electricity production systems. Journal of Cleaner Production, 10(4), pp.321–334. Available at: https://doi.org/10.1016/S0959-6526(01)00043-9

Brown, M.T., 2015. Emergy and form: accounting principles for recycling pathways. Journal of Environmental Accounting and Management, 3, pp. 259-274.

Brown, M.T. and Ulgiati, S., 2004. Emergy analysis and environmental accounting. In *Encyclopedia of Energy*. Elsevier, pp. 329–354.

Brown, M, T., Protano, G., Ulgiati, S. 2011. Assessing geobiosphere work of generating global reserves of coal crude oil, and natural gas. Ecological Modelling, 222, pp. 879 – 887.

Buranakarn V., 1998. Evaluation of Recycling and Reuse of building materials using the emergy analysis method. University of Florida. Doctoral Thesis. https://ufdc.ufl.edu/AA00040382/00001

Buratti, C., Barbanera, M., Testarmata, F., Fantozzi F., 2015. Life cycle Assessment of organic waste management strategies: an Italian case study. Journal of Cleaner Production, 89, pp. 125-136. Doi:10.1016/j.jclepro.2014.11.012

Bueno, G., Latasa, I., Lozano, P.J., 2015. Comparative LCA of two approaches with different emphasis on energy or material recovery for a municipal solid waste management system in Gipuzkoa. Renew. Sustainable. Energy Rev. 51, pp. 449-459. Doi: 10.1016/j.rser.2015.06.021.

Cakar, B., 2022. Bounce back of almost wasted food: Redistribution of fresh fruit and vegetables surpluses from Istanbul's supermarkets. Journal of Cleaner Production, 132325. Doi: 10.1016/j.jclepro.2022.132325

Caldeira, C., Vlysidis, A., Fiore, G., De Laurentiis, V., Vignali, G., Sala, S., 2020. Sustainability of food waste biorefinery: A review on valorisation pathways, techno-economic constraints, and environmental assessment. Bioresource Technology, 312, 123575. Doi: 10.1016/j.biortech.2020.123575

Candiani, G., Da Silva M., R., M, 2011. Análise de biiogas no aterro de Caieiras. Ciências exatas e tecnológicas, 32 (1), pp. 69 – 77. http://dx.doi.org/10.5433/1679-0375.2011v32n1p69

Carlsson Reich, M., 2005. Economic assessment of municipal waste management systems – case studies using a combination of life cycle assessment (LCA) and life cycle costing (LCC). Journal of Cleaner Production 13, pp. 253-263. Doi: 10.1016/j.jclepro.2004.02.015

Cascini, A., Gamberi, M., Mora, C., Rosano, M., Bortolini, M., 2016. Comparative Carbon Footprint Assessment of commercial walk-in refrigeration systems under different use configurations. *Journal of Cleaner Production*, *112*, pp. 3998-4011. Doi: 10.1016/j.jclepro.2015.08.075

Castellini, C., Bastianoni, S., Granai, C., Dal Bosco, A., Brunetti, M., 2006. Sustainability of poultry production using the emergy approach: Comparison of conventional and organic rearing systems. *Agriculture, ecosystems and environment*, *114*(2-4), pp. 343-350. Doi: 10.1016/j.agee.2005.11.014

Cetrulo, T.B., Marques, R.C., Cetrulo, N.M., Pinto, F.S., Moreira, R.M., Mendiz´abal- Cortes, A.D., Malheiros, T.F., 2018. Effectiveness of solid waste policies in developing countries: a case study in Brazil. J. Clean. Prod. 205, pp. 179–187. Doi: 10.1016/j.jclepro.2018.09.094.

CEAGESP, Relatório de Gestão – 2008, São Paulo / 2008 Available at: http://www.ceagesp.gov.br/wp-content/uploads/2015/03/relatorio_gestao_2008.pdf

CEAGESP, Relatório de Gestão – 2009, São Paulo / 2009 Available at: http://www.ceagesp.gov.br/wp-content/uploads/2015/03/relatorio_gestao_2009.pdf

CEAGESP, Relatório de Gestão – 2010, São Paulo / 2010 Available at: http://www.ceagesp.gov.br/wp-content/uploads/2015/03/relatorio_gestao_2010.pdf

CEAGESP, Relatório de Gestão – 2011, São Paulo / 2011 Available at: http://www.ceagesp.gov.br/wp-content/uploads/2015/03/relatorio_gestao_2011.pdf

CEAGESP, Relatório de Gestão – 2012, São Paulo / 2012 Available at: http://www.ceagesp.gov.br/wp-content/uploads/2015/03/relatorio_gestao_2012.pdf

CEAGESP, Relatório de Gestão – 2013, São Paulo / 2013 Available at: http://www.ceagesp.gov.br/wp-content/uploads/2015/03/relatorio_gestao_2013.pdf

CEAGESP, Relatório de Gestão – 2014, São Paulo / 2015 Available at: http://www.ceagesp.gov.br/wp-content/uploads/2015/05/relatorio_gestao_2014_ceagesp.pdf

CEAGESP, Relatório de Gestão – 2015, São Paulo / 2016 Available at: http://www.ceagesp.gov.br/wp-content/uploads/2015/05/relatorio_gestao_2015.pdf

CEAGESP, Relatório de Gestão – 2016, São Paulo / 2017 Available at: http://www.ceagesp.gov.br/wp-content/uploads/2015/03/RELATORIO-DE-GEST%C3%83O-2016-FINAL-31.05.2017.pdf

CEAGESP, Relatório de Gestão – 2017, São Paulo / 2018 Available at: http://www.ceagesp.gov.br/wp-content/uploads/2015/05/RELAT%C3%93RIO-DE-GEST%C3%83O-20171.pdf

CEAGESP, Relatório de Gestão – 2018, São Paulo / 2019 Available at: http://www.ceagesp.gov.br/wp-content/uploads/2015/05/RELAT%C3%93RIO-DE-GEST%C3%83O-2018.pdf

CETESB, Emissões veicolares no estado de São Paulo no 2018. São Paulo – SP, 2019. https://cetesb.sp.gov.br/veicular/wp-content/uploads/sites/6/2020/02/Relat%C3%B3rio-Emiss%C3%B5es-Veiculares-no-Estado-de-S%C3%A3o-Paulo-2018.pdf

Cherubini, F., Bargigli S., Ulgiati S., 2009. Life cycle Assessment (LCA) of waste management strategies: Landfilling, sorting plant and incineration. Energy, 34, pp. 2116 – 2123. http://dx.doi.org/10.1016/j.energy.2008.08.023.

Cherubini, F., 2010. The biorefinery concept: using biomass instead of oil for producing energy and chemicals. Energy conversion and management, 51(7), 1412-1421. Doi:10.1016/j.enconman.2010.01.015

Chester, M., and Martin, E., 2009. Cellulosic ethanol from municipal solid waste: a case study of the economic, energy, and greenhouse gas impacts in California. Doi: 10.1021/es802788z

Coelho, L.M.G and Lange, L.C., 2018. Applying Life Cycle Assessment to support environmentally sustainable waste management in Brazil. Resources, Conservation and Recycling, 128, pp.438-450. Available at: https://doi.org/10.1016/j.resconrec.2016.09.026

Coimbra, E.A., 2009. Total Flow Management: Achieving Excellence with Kaizen and Lean Supply Chains, first ed. Publisher, Kaizen Institute, p. 277. ISBN 0473146592.

Conteratto, C., Artuzo, F.D., Santos, O.I.B. and Talamini, E., 2021. Biorefinery: A comprehensive concept for the sociotechnical transition toward bioeconomy. Renewable and Sustainable Energy Reviews, 151, p.111527.

Corrado, S., Sala, S., 2018. Food waste accounting along global and European food supply chains: State of the art and outlook. Waste management, 79, 120-131.

Costa, A. M., Alfaia, R. G. D. S. M., Campos, J. C.. 2019. Landfill leachate treatment in Brazil–An overview. Journal of environmental management, 232, 110-116.

Culi, M. J. L., 2018. Biodigestão da fração orgânica de resíduos sólidos em um reator de duas fases operado em bateladas sequenciais com imobilização da biomassa (Doctoral dissertation, Universidade de São Paulo). Available at: https://www.teses.usp.br/teses/disponiveis/3/3147/tde-08032019-134307/en.php

Damgaard, A., Manfredi, S., Merrild, H., Stensøe, S., Christensen, T. H. (2011). LCA and economic evaluation of landfill leachate and gas technologies. Waste management, 31(7), 1532-1541. Doi: 0.1016/j.wasman.2011.02.027

Daly, H. E., 1990. Toward some operational principles of sustainable development. Ecological Economics, 2, 1 – 6. 10.1016/0921-8009(90)90010-R]

Daly, H. E., 2017. Toward some operational principles of sustainable development 1. In The economics of sustainability (pp. 97-102). Routledge.

Damiani, M., Pastorello, T., Carlesso, A., Tesser, S., Semenzin, E., 2021. Quantifying environmental implications of surplus food redistribution to reduce food waste. Journal of Cleaner Production, 289, 125813. Doi: 10.1016/j.jclepro.2021.125813

Dahiya, S., Kumar, A.N., Sravan, J.S., Chatterjee, S., Sarkar, O. and Mohan, S.V., 2018. Food waste biorefinery: Sustainable strategy for circular bioeconomy. Bioresource technology, 248, pp. 2-12. Doi: 10.1016/j.biortech.2017.07.176

Da Silva, G., F., 2011. ATERRO SANITÁRIO SÃO JOÃO: ESTUDO DOS INDICADORES AMBIENTAIS EM EMERGIA. Master's Degree Thesis.

Da Silva, S. V., Pereira, R., A., Piveli R., P., Criscuolo, H., G., 2005. MONITORAMENTO DAS CARACTERÍSTICAS DOS ESGOTOS COMO INSTRUMENTO DE OTIMIZAÇÃO DO PROCESSO DE LODOS ATIVADOS E SUAS IMPLICAÇÕES. Revista Brasileira de Ciências Ambientais , 2, pp. 46 -59

Da Silva, J. R. S., Santos, M. D., Da Silva, J., K., S., Queiroga S., L., M., Neto J., C., M., 2017. COMPARAÇÃO DOS COMBUSTÍVEIS DÍESEL S10 E S500 PARA RESOLUÇÃO DE PROBLEMAS

EM MOTORES DO CICLO DÍESEL. In: Congresso Técnico Científico da Engenharia e daAgronomia, CONTECC – , Belem, PA, Brazil.

DBEIS UK, 2021 – Department for Business, Energy and Industrial Strategy: Combined Heat and Power – Technologies. London, 2021. Available at: https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/961492/Part_2_CHP_Technologies_BEIS_v03.pdf

De Campos, V.A.F., Silva, V.B., Cardoso, J.S., Brito, P.S., Tuna, C.E. and Silveira, J.L., 2021. A review of waste management in Brazil and Portugal: Waste-to-energy as pathway for sustainable development. Renewable Energy, 178, 802-820. Doi: 10.1016/j.renene.2021.06.107.

De Barros, I., Blazy, J.M., Rodrigues, G.S., Tournebize, R. and Cinna, J.P., 2009. Emergy evaluation and economic performance of banana cropping systems in Guadeloupe (French West Indies). Agriculture, Ecosystems and Environment, 129(4), pp.437-449. Doi: 10.1016/j.agee.2008.10.015

De la Torre, I., Martin-Dominguez, V., Acedos, M. G., Esteban, J., Santos, V. E., Ladero, M., 2019. Utilisation/upgrading of orange peel waste from a biological biorefinery perspective. Applied microbiology and biotechnology, 103(15), 5975-5991. Doi: 10.1007/s00253-019-09929-2

Delgado-Aguilar, M., Valverde-Som, L., Cuadros-Rodríguez, L., 2018. Solver, an Excel application to solve the difficulty in applying different univariate linear regression methods. Chemometrics and Intelligent Laboratory Systems, 178, 39-46. Doi: 10.1016/j.chemolab.2018.04.018

Deviatkin, I., Khan, M., Ernst, E., Horttanainen, M., 2019. Wooden and Plastic Pallets: A Review ofLife Cycle Assessment (LCA) Studies. Sustainability, 11 (20), 5750. Doi:10.3390/su11205750.

De Oliveira, R. K., Higa, A. R., Silva, L. D., Silva, I. C., Gonçalves, M. D. P. M., 2018. Emergy-based sustainability assessment of a loblolly pine (Pinus taeda) production system in southern Brazil. Ecological Indicators, 93, 481-489.

Ebner, J., Babbitt, C., Winer, M., Hilton, B., Williamson, A., 2014. Life cycle greenhouse gas (GHG) impacts of a novel process for converting food waste to ethanol and co-products. Applied energy, 130, 86-93. Doi: 10.1016/j.apenergy.2014.04.099

EC, 2002. Regulation (EC) No 1774/2002 of the European Parliament and of the Council of 3 October2002 Laying Down Health Rules Concerning Animal ByproductsNot Intended for Human Consumption.

EC, 2008. Directive 2008/98/EC of the European Parliament and of the Council of 19 November 2008 on Waste and Repealing Certain Directives. Official Journal of the European Union, Strasbourg. http://data.europa.eu/eli/dir/2008/98/oj

EC, 2015. Commission Recommendation of 2 December 2015 on ‘‘Closing the loop –an EU actionplan for the circular economy. Available at: https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:52015DC0614

EC. 2017. Expert Group Report: Review of the EU Bioeconomy Strategy and its Action Plan

EC. 2018. A sustainable Bioeconomy for Europe: strengthening the connection between economy, society and the environment, Brussels, Belgium.

Ecoinvent Database, version 3.6, 2019. https://www.ecoinvent.org/

Edwards, J., Othman, M., Crossin, E. and Burn, S., 2018. Life cycle assessment to compare the environmental impact of seven contemporary food waste management systems. Bioresource technology, 248, pp.156-173. Doi: 10.1016/j.biortech.2017.06.070

EPA, 2018. United States Environmental Protection Agency. Available at: https://www.epa.gov/facts-and-figures-about-materials-waste-and-recycling/national-overview-facts-and-figures-materials#Landfilling

EPE, 2020. Empresa de pesquisa Energetica. Brazilian Ministry of Mines and Energy, year 2019. Analysis of Biofuels' current outlook. Available at. https://www.epe.gov.br/sites-en/publicacoes-dados-abertos/publicacoes/Paginas/Analysis-Of-Biofuels%C2%B4Current-Outlook-2019.aspx, 14th July 2021.

EU, 2006. Review of the EU Sustainable Development Strategy (EU SDS). Available at: https://register.consilium.europa.eu/doc/srv?l=EN&f=ST%2010917%202006%20INIT

Eriksson, M., Strid, I., Hansson, P.A., 2015. Carbon footprint of food waste management options in the waste hierarchy – a Swedish case study. Journal of Cleaner Production, 93, 115–125. Doi:10.1016/j.jclepro.2015.01.026.

Eriksson, M., Spangberg., J., 2017. Carbon footprint and energy use of food waste management options for fresh fruit and vegetables from supermarkets. Waste Management, 60, 786 -789. Doi: 10.1016/j.wasman.2017.01.008

Evans, J. A., Foster, A. M., Huet, J. M., Reinholdt, L., Fikiin, K., Zilio, C., Houska, M., Landfeld A., Bond, C.,Scheurs, M., Van Sambeeck, T. W. M. (2014). Specific energy consumption values for various refrigerated food cold stores. Energy and Buildings, 74, 141-151. Doi: 10.1016/j.enbuild.2013.11.075

Eurostat, 2020. Municipal waste statistics. Available at: https://ec.europa.eu/eurostat/statistics-explained/index.php?title=Municipal_waste_statistics#Municipal_waste_treatment

Eurotankworks, 2022. Available at: https://eurotankworks.com/storage-tanks/vertical-storage-tanks/vertical-tank-vol-3000/)

Evangelisti, S., Tagliaferri, C., Clift, R., Lettieri, P., Taylor, R., Chapman, C., 2015. Life cycle assessment of conventional and two-stage advanced energy-from-waste technologies for municipal solid waste treatment. J. Clean. Prod. 100, 212 – 213. Doi: 10.1016/j.jclepro.2015.03.062

Fagundes, P. R. S., Mondini, L., Nachiluk, K., de Oliveira Pithan, R., de Oliveira Figueiredo, S. A. M., 2015. BANCO CEAGESP DE ALIMENTOS: captação e distribuição de alimentos. Informações Econômicas, 44 (4), 33-41. Available at: http://www.iea.agricultura.sp.gov.br/ftpiea/publicacoes/ie/2014/tec3-0814.pdf

FAO. 2011. Global food losses and food waste – Extent, causes and prevention., Rome. Available at: http://www.fao.org/3/mb060e/mb060e00.pdf

FAO. 2019. The State of Food and Agriculture 2019. Moving forward on food loss and waste reduction. Rome.

Faria, L., 2017. O desempenho economico, Ambiental e social do Brazil: de 2012 a 2022, sob a perspectiva dos planos do governo. Doctor's Degree thesis. Available at: https://repositorio.unip.br/dissertacoes-teses-programa-de-pos-graduacao-stricto-sensu-em-engenharia-de-producao/o-desempenho-economico-ambiental-e-social-do-brasil-de-2012-a-2022-sob-a-perspectiva-dos-planos-do-governo/

Fehr, M., Romão, D. C., 2001. Measurement of fruit and vegetable losses in Brazil: a case study. Environment, Development and Sustainability, 3(3), 253-263.

Fernandez-Nava, Y., del Río, J., Rodríguez-Iglesias, J., Castrillon, L., Maranon, E., 2014. Life cycle assessment of different municipal solid waste management options: acase study of Asturias (Spain). Journal of Cleanear Production, 81, pp. 178-189. Doi: 10.1016/j.jclepro.2014.06.008.

FL – 14.016 / 2020 - Dispõe sobre o combate ao desperdício de alimentos e a doação de excedentes de alimentos para o consumo humano. Available at: http://www.planalto.gov.br/ccivil_03/_ato2019-2022/2020/lei/L14016.htm

Folkerts, H., Koehorst, H., 1998. Challenges in international food supply chains: vertical co-ordination in the European agribusiness and food industries. British. Food Journal, 100, pp.385-388.

Francini, G., Lasagni, M., Lombardi, L., 2020. Comparison of Anaerobic Digestion Technologies: an Italian Case Study. Multidisciplinary Journal for Waste Resources & Residues, 9, pp.94-104.

Fuess, L. T., Cruz, R. B. C. M., Zaiat, M., Nascimento, C. A. O., 2021. Diversifying the portfolio of sugarcane biorefineries: Anaerobic digestion as the core process for enhanced resource recovery. Renewable and Sustainable Energy Reviews, 147, 111246.

Fusi, A., Bacenetti, J., Fiala, M., Azapagic, A., 2016. Life cycle environmental impacts of electricity from biogas produced by anaerobic digestion. Frontiers in bioengineering and biotechnology, 4, p.26. Doi: 10.3389/fbioe.2016.00026

Gala A. B., Raugei M., Ripa M., Ulgiati, 2015. Dealing with waste products and flows in life cycle assessment and emergy accounting: Methodological overview and synergies. Ecological Modelling, 315, pp. 69-76. http://dx.doi.org/10.1016/j.ecolmodel.2015.03.004

Galindo, E., Teixeira, M.A., De Araújo, M., Motta, R., Pessoa, M., Mendes, L., Renn´o, L., 2021. Efeitos da pandemia na alimentação e na situação da segurança alimentar no Brasil. Food for Justice Working Paper Series, 4. Berlin: Food for Justice: Power, Politics, and Food Inequalities in a Bioeconomy. https://doi.org/10.17169/refubium-29554.

Gerdau, 2012. Manual Galpões em Pórticos com Perfis Estruturais Laminados. Available at: http://www.skylightestruturas.com.br/downloads/manual-galpoes-em-porticos-perfis-estruturais-laminados.pdf

Giannetti, F., B., Agostinho F., Moares, L., C., Almeida C., M., B., Ulgiati, S., 2015. Multicriteria cost–benefit assessment of tannery production: The need for breakthrough process alternatives beyond conventional technology optimization. Environmental Impacts Assessment Review, 54, pp. 22-38. http://dx.doi.org/10.1016/j.eiar.2015.04.006

Goedkoop, M., Heijungs, R., Huijbregts, M., Schryver, A., Struijs, J., Zelm, R., 2009. ReCiPE 2008: A life cycle impact assessment method which comprises armonized category indicators at the midpoint and the endpoint level. https://www.researchgate.net/publication/230770853_Recipe_2008

Griebenow, C., Ohara, A., 2019. Agora energiewende & instituto E+ diálogos energéticos, 2019: report on the Brazilian power system. Available at: https://www.agoraenergiewende.de/fileadmin2/Projekte/2019/Brazil_Country_Profile/155_CountryProf_Brazil_EN_WEB.pdf, 14th July 2021

Gruppomini, 2022. Available at: https://www.gruppominitop.it/prodotti/nastro-trasportatore-bst-220v-m1-da-3mt/.

Guo, H., Zhao, Y., Damgaard, A., Wang, Q., Wang, H., Christensen, T.H. and Lu, W., 2021. Quantifying global warming potential of alternative biorefinery systems for producing fuels from Chinese food waste. Waste Management, 130, pp.38-47. Doi: 10.1016/j.wasman.2021.05.004

Hanfeng, M., Xiao, F., Khim, H., C., 2012. Calculation of emergy flows within complex chemical production systems. Ecological Engineering, 44, pp. 88 – 93.

Henz, G.,P., Porpino, G., 2017. Food losses and waste: how Brazil is facing this global challenge? Horticultura Brasileira 35: 472-482. doi:10.1590/S0102-053620170402

Holliger, C., Fruteau de Laclos, H., Hack, G., 2017. Methane production of full-scale anaerobic digestion plants calculated from substrate's biomethane potentials compares well with the one measured on-site. Frontiers in Energy Research, 5, p.12. Doi:

Huang, J., Khan, M. T., Perecin, D., Coelho, S. T., Zhang, M., 2020. Sugarcane for bioethanol production: potential of bagasse in Chinese perspective. Renewable and Sustainable Energy Reviews, 133, 110296

Hultman, J., Corvellec, H., 2012. The European waste hierarchy: From the sociomateriality of waste to a politics of consumption. Environment and Planning A, 44(10), 2413-2427.. Doi:10.1068/a44668.

IEA, 2009. IEA bioenergy Task 42 on biorefineries: adding values to the sustainable utilization of biomass. https://www.ieabioenergy.com/blog/publications/biorefineries-adding-value-to-the-sustainable-utilisation-of-biomass/

IEA, 2012. Bioenergy - Task 42 Biorefinery, Bio-based Chemicals - Value Added Products from Biorefineries.

Imperiodoplastico, 2022. Available at: https://www.imperiodoplastico.com.br/bombona-plastica/bombona-50-litros-homologada

Indiamart, 2022a. Available at: https://www.indiamart.com/proddetail/plastic-modular-conveyor-belt-10120920197.html

Indiamart, 2022b. Available at: www.indiamart.com/proddetail/10-liter-plastic-bucket-13399333633.html

Ingwersen, W.W., 2009. Emergy of Gold Mined at Yanacocha Based on an LCA Model. In: Proceedings from the Fifth Biennial Emergy Conference, Gainesville, Florida. The Center for Environmental Policy Department of Environmental Engineering Sciences, University of Florida

IPCC, 2006. 2006 IPCC guidelines for national greenhouse gas inventories e vol. 5:waste. In: Eggleston, H.S., Buendia, L., Miwa, K., Ngara, T., Tanabe, K. (Eds.), National Greenhouse Gas Inventories Programme. IGES, Japan.

Ishida, P.M.G., Steel, C.J., 2014. Physicochemical and sensory characteristics of pan bread samples available in the Brazilian market. Food Science and Technology, 34, pp.746-754.

ISO 14040, 2006. Environmental management — Life cycle assessment — Principles and

Framework.https://www.iso.org/standard/37456.html

ISO 14044, 2006. Environmental management – Life cycle assessment – Requirements and guidelines.

IUCN, 1980. International Union for Conservation of Nature, & World Wildlife Fund. World conservation strategy: Living resource conservation for sustainable development (Vol. 1). Gland, Switzerland.

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

Joglekar, S. N., Pathak, P. D., Mandavgane, S. A., Kulkarni, B. D., 2019. Process of fruit peel waste biorefinery: a case study of citrus waste biorefinery, its environmental impacts and recommendations. Environmental Science and Pollution Research, 26(34), 34713-34722. Doi: 10.1007/s11356-019-04196-0

Jeníček, P., Horejš, J., Pokorná-Krayzelová, L., Bindzar, J., Bartáček, J., 2017. Simple biogas desulfurization by microaeration–Full scale experience. Anaerobe, 46, pp.41-45. Doi: 10.1016/j.anaerobe.2017.01.002

Jones, R.E., Speight, R.E., Blinco, J.L. O'Hara, I.M., 2022. Biorefining within food loss and waste frameworks: A review. Renewable and Sustainable Energy Reviews, 154, p.111781. Doi: 10.1016/j.rser.2021.111781

JRC, 2010. International Reference Life Cycle Data System (ILCD) Handbook – General guide for Life Cycle Assessment – Detailed guidance European Commission & Institute for Environment and Sustainability, eds., Luxembourg: Publications Office of the European Union. https://op.europa.eu/en/publication-detail/-/publication/a47e524c-acc4-4a27-98d7- 78211e6aa59b/language-en

Jun D., Yong, C., Daoan, Z., Chao F., Qunxing, H., Minjiang, N., et al., 2014. Energy–environment–economy assessment of waste management systems from a life cycle perspective: Model development and case study. Applied Energy, 114, pp.400–408. Available at: https://doi.org/10.1016/j.apenergy.2013.09.037

Kalogo, Y., Habibi, S., MacLean, H. L., Joshi, S. V., 2007. Environmental implications of municipal solid waste-derived ethanol. Doi: 10.1021/es061117b

Karthikeyan, O. P., Visvanathan, C., 2013. Bio-energy recovery from high-solid organic substrates by dry anaerobic bio-conversion processes: a review. Reviews in Environmental Science and Bio/Technology, 12(3), 257-284. Doi: 10.1007/s11157-012-9304-9

Kelly, K.A., McManus, M.C., Hammond, G.P., 2014. An energy and carbon life cycle assessment of industrial CHP (combined heat and power) in the context of a low carbon UK. Energy, 77, pp.812-821. Doi: 10.1016/j.energy.2014.09.051

Kirchherr J., Reike D., and Hekkert M., 2017. Conceptualizing the circular economy: analysis of 114 definitions. Reduce, Conservation & Recycling, 127, 221 – 32. Available at: https://doi.org/10.1016/j.resconrec.2017.09.005

Kristensen, P.G., Jensen, J.K., Nielsen, M., Illerup, J.B., 2004. Emission factors for gas fired CHP units< 25 MW. Vancouver: IGRC.

Larsen, A., W., Vroc, M., Christensen, T., H., 2009. Diesel consumption in waste collection and transport and its environmental significance. Waste management and research, 00, pp. 1 – 8. Available at: https://doi.org/10.1177/0734242X08097636

Legaspe, L.R., 2006. Cap. II: Resíduo orgânico urbano e seu (re)aproveitamento como fonte de alimentação humana, animal, compostagem e energia térmica. In: Spadotto, C.A., Ribeiro, W.C., 2006,Gestão de residuos na agricultura e agroindústria. Botucatu: FEPAF. Available at: https://www.bibliotecaagptea.org.br/administracao/agroindustria/livros/GESTAO%20DE%20RESIDU OS%20NA%20AGRICULTURA%20E%20AGROINDUSTRIA.pdf, accessed on January 18th 2021 (inPortuguese language).

Liikanen, M., Havukainen, J., Ednilson V., Horttaninen, M., 2018. Steps towards more environmentally sustainable solid waste management – A life cycle assessment study of São Paulo, Brazil. Journal of cleaner production, 196 pp. 150 – 162.

Lipinski, B., Hanson, C., Lomax, J., Kitinoja, L., Waite, R. Searchinger, T., 2013. “Reducing Food Loss and Waste.” Working Paper, Installment 2 of *Creating a Sustainable Food Future*. Washington, DC: World Resources Institute. Available online at http://www.worldresourcesreport.org.

Liu, G.; Yang, Z.; Chen, B.; Zhang, Y.; Su, M.; Zhang, L., 2013. Emergy Evaluation of the Urban Solid Waste Handling in Liaoning Province, China. Energies, 6, 5486-5506 https://doi.org/10.3390/en6105486

Liu, G., Hao, Y., Dong, Y., Yang, Z., Zhang Y., Ulgiati S., 2017a. An emergy-LCA analysis of municipal solid waste management. Resources, conservation and recycling, 120 pp. 131 – 143. http://dx.doi.org/10.1016/j.resconrec.2016.12.003

Liu Y. L., P.X. Xing, J.G. Liu, 2017b. Environmental performance evaluation of different municipal solid waste management scenarios in China. Resources. Conservation and Recycling, 125 (2017), pp. 98-106. https://doi.org/10.1016/j.resconrec.2017.06.005

Lohrasbi, M., Pourbafrani, M., Niklasson, C., Taherzadeh, M. J., 2010. Process design and economic analysis of a citrus waste biorefinery with biofuels and limonene as products. Bioresource technology, 101(19), 7382-7388. Doi: 10.1016/j.biortech.2010.04.078

Lorenzi, G., Gorgoroni, M., Silva, C. and Santarelli, M., 2019. Life Cycle Assessment of biogas upgrading routes. Energy Procedia, 158, pp.2012-2018.

Lyu, Y., Raugei, M., Zhang, X., Mellino, S. and Ulgiati, S., 2021. Environmental cost and impacts of chemicals used in agriculture: An integration of emergy and Life Cycle Assessment. Renewable and Sustainable Energy Reviews, 151, p.111604. Doi: 10.1016/j.rser.2021.111604

Marcos, A. et al., 2010. Combustible gas production (methane) and biodegradation of solid and liquid mixtures of meat industry wastes. *Applied Energy*, 87(5), pp.1729–1735. Available at: https://doi.org/10.1016/j.apenergy.2009.09.037

Marguti, A. L., Ferreira Filho, S. S., Piveli, R. P., 2008. Physical-chemical process optimization for phosphorus removal from domestic wastewater by chemical precipitation with ferric chloride. *Engenharia Sanitaria e Ambiental*, *13*(4), 395-404.

Mattos, J. S., Medeiros, B. J., 2008. Densidade de polpas de frutas tropicais: banco de dados e determinação experimental. Revista Brasileira de Engenharia de Biossistemas, 2(2), 109-118. 10.18011/bioeng2008v2n2p109-118

Marchettini, N., R. Ridolfi, M. Rustici, 2007. An environmental analysis for comparing waste management option and strategies. Waste management, 27, pp. 562-571. https://doi.org/10.1016/j.wasman.2006.04.007

Martinez-Sanchez, V., Kromann, M.A., Astrup, T.F., 2015. Life cycle costing of waste management systems: overview, calculation principles and case studies. Waste management (New York, N.Y.), 36,pp.343–55. Available at: https://doi.org/10.1016/j.wasman.2014.10.033

Mcintosh, A., Pontius, J., 2017. Science and the global Environment. Elsevier, Amsterdam.

Menard, J., F., Lesage, P., Deschenes, L., Samson, R. 2004. Comparative Life Cycle assessment of Two Landfill Technologies for the treatment of Municipal solid waste. International Journal of LCA, 9 (6), 371 – 378.

Mendes M. R., Aramaki, T., Hakai, K. 2004. Comparison of the environmental impact of incineration and landfilling in São Paulo City as determined by LCA. Resources, Conservation and Recycling, 41, pp. 47-63. https://doi.org/10.1016/j.resconrec.2003.08.003

Mirbelting, 2022. Available at: https://www.mir-belting.com/wp-content/uploads/2019/04/Guide-to-conveyor-maintenance-2.pdf

Miki, M., K.,1998. Utilização de polímeros para condicionamento de lodo de ETE para desidrataçãoem filtro de prensa de placa. São Paulo, Master's Degree Thesis. https://www.teses.usp.br/teses/disponiveis/3/3147/tde-09052001-151514/publico/Mestrado.pdf

Monier, V., Mudgal, S., Escalon, V., O'Connor, C., Gibon, T., Anderson, G., Morton, G., 2010. Preparatory study on food waste across EU 27. Report for the European Commission. Technical Report – 2010 – 054.

Moraes, B. S., Junqueira, T. L., Pavanello, L. G., Cavalett, O., Mantelatto, P. E., Bonomi, A., Zaiat, M., 2014. Anaerobic digestion of vinasse from sugarcane biorefineries in Brazil from energy, environmental, and economic perspectives: Profit or expense? Applied Energy, 113, 825-835.

Moreno, A. D., Magdalena, J. A., Oliva, J. M., Greses, S., Lozano, C. C., Latorre-Sánchez, M., Negro, M., J., Susmosas, A., Iglesias, R., Llamas, M., Tomás-Pejó, E., González-Fernández, C. (2021). Sequential bioethanol and methane production from municipal solid waste: An integrated biorefinery strategy towards cost-effectiveness. Process Safety and Environmental Protection, 146, 424-431.

Moult, J., A., Allan, S., R., Hewitt, C., N., Berners-Lee, M., 2018. Greenhouse gas emissions of food waste disposal options for UK retailers. Food Policy, 77, 50 – 58. Doi: 10.1016/j.foodpol.2018.04.003

Mu, H., Feng, X., Chu, K. H., 2012. Calculation of emergy flows within complex chemical production systems. Ecological Engineering, 44, 88-93. Doi: 10.1016/j.ecoleng.2012.04.017

Muthu, S. S., Li, Y. (2016). Assessment of environmental impact by grocery shopping bags. Springer Verlag, Singapor. Available at: https://link.springer.com/book/10.1007/978-981-4560-20-7

NFP, 2022 - Plano Nacional de Fertilizantes 2022-2050 - Decreto Nº 10,991/2022. Available at: https://www.in.gov.br/en/web/dou/-/decreto-n-10.991-de-11-de-marco-de-2022-385453056

NPSW, 2010 - National Policy on Solid Waste - Federal Law 12,305/2010. Available at: http://www.braziliannr.com/brazilian-environmental-legislation/law-no-12305-brazilian-national-policy-solid-waste/

Odum, H.T. & Odum, E.C., 1980. Energy Systems and Environmental Education. In T. S. Bakshi & Z. Naveh, eds. *Environmental Education – Principles, Methods and Applications*. Boston, MA: Springer US, pp. 213–231.

Odum, H.T., 1996. Environmental Accounting: Emergy and Environmental Decision-Making, John Wiley, ed., New York. http://eu.wiley.com/WileyCDA/WileyTitle/productCd-0471114421.html

Oliveira, F.D., Sfredo, G.J., Castro, C.D. and Klepker, D., 2007. Fertilidade do solo e nutrição da soja. Londrina: Embrapa.

Oliveira L. S. B. L., Oliveira D. S. B. L, Bezerra B. S., Pereira B. S., Battistelle R. A. G., 2017. Environmental analysis of organic waste management focusing on composting scenarios. Journal of cleaner production, 155, pp. 229 – 237. http://dx.doi.org/10.1016/j.jclepro.2016.08.093

Oliveira, B., Alves, A., C., Carneiro, P., Ferreira, A., C., 2018. Lean Production and Ergonomics: a synergy to improve productivity and working conditions. International journal of occupational and environmental safety, 2 (2), 1-11. Doi: 10.24840/2184-0954_002.002_0001

Oliveira, M.P., Malagolli, G.A. and Cella, D., 2019. Mercado de fertilizantes: dependência de importações do Brasil. Revista Interface Tecnológica, 16(1), pp.489-498.

Pan, H., Zhang, X., Wu, J., Zhang, Y., Lin, L., Yang, G., Deng. S., Li, L., Yu, X., Qi, H., Peng, H., 2016. Sustainability evaluation of a steel production system in China based on emergy. Journal of cleaner production, 112, 1498-1509. Doi: http://dx.doi.org/10.1016/j.jclepro.2015.05.019

Pan, H., Geng, Y., Tian, X., Wilson, J., Chen, W., Zhong, S., Song, X. (2019). Emergy-based environmental accounting of one mining system. Environmental Science and Pollution Research, 26(14), 14598-14615 Doi: 10.1007/s11356-019-04793-z

Papadaskalopoulou, C., Sotiropoulos, A., Novacovic, J., Barabouti, E., Mai, S., Malamis, D., Kekos, D., Loizidou, M. 2019. Comparative life cycle assessment of a waste to ethanol biorefinery system versus conventional waste management methods. Resources, Conservation and Recycling, 149, 130-139. Doi: 10.1016/j.resconrec.2019.05.006

Papargyropoulou, E., Lozano, R., Steinberger, J. K., Wright, N., bin Ujang, Z., 2014. The food waste hierarchy as a framework for the management of food surplus and food waste. Journal of cleaner production, 76, 106-115. Doi: 10.1016/j.jclepro.2014.04.020

Paritosh, K., Yadav, M., Mathur, S., Balan, V., Liao, W., Pareek, N. and Vivekanand V., 2018. Organic Fraction of Municipal Solid Waste: Overview of Treatment Methodologies to Enhance Anaerobic Biodegradability. Frontiers in Energy Research. 6, 75. https://doi.org/10.3389/fenrg.2018.00075

Pathak, P. D., Mandavgane, S. A., Puranik, N. M., Jambhulkar, S. J., & Kulkarni, B. D., 2018. Valorization of potato peel: a biorefinery approach. Critical Reviews in Biotechnology, 38(2), 218-230. Doi: 10.1080/07388551.2017.1331337

Patterson., M., McDonald, G., Hardy, D., 2017. Is there more in common than we think? Convergence of ecological footprinting, emergy analysis, life cycle assessment and other methods of environmentalaccounting. Ecological modelling, 362, 19 – 36.

Patrizi, N., Pulselli, F., M., Morandi F., Bastianoni, S., 2015. Evaluation of the emergy investment needed forbioethanol production in a biorefinery using residual resources and energy. Journal of Cleaner Production, 96, 549-556.

Pennington, D.W. et al., 2004. Life cycle assessment part 2: current impact assessment practice. Environment international, 30 (5), pp.721–39.

Pires, A., Martinho, G., 2019. Waste hierarchy index for circular economy in waste management. Waste Management, 95, 298 - 305. Doi: 10.1016/j.wasman.2019.06.014

Pizzigallo, A.C.I., Granai, C. and Borsa, S., 2008. The joint use of LCA and emergy evaluation for the analysis of two Italian wine farms. Journal of Environmental Management, 86(2), 396-406. Doi: 10.1016/j.jenvman.2006.04.020

Porter, S.D., Reay, D.S., Higgins, P., Bomberg, E., 2016. A half-century of production phase greenhouse gas emissions from food loss & waste in the global food supply chain. Science of the total Environment, 571, 721–729.

Probiogas, 2015. BRASIL. Secretaria Nacional de Saneamento Ambiental. Probiogás, 2015. Guia técnico de aproveitamento energético de biogás em estações de tratamento de esgoto. Available at: https://antigo.mdr.gov.br/images/stories/ArquivosSNSA/probiogas/probiogas-guia-etes.pdf

Raugei M., Rugani B., Benetto E., Ingwersen W. W., 2014. Integrating emergy into LCA: Potential added value and lingering obstacles. Ecological modelling, 271, pp. 4 – 9 http://dx.doi.org/10.1016/j.ecolmodel.2012.11.025

RenovaBio, 2017. Política Nacional de Biocombustíveis – LEI Nº 13.576, DE 26 DE DEZEMBRO DE 2017. Available at: http://www.planalto.gov.br/ccivil_03/_ato2015-2018/2017/lei/l13576.htm

Rezende, D., Melo, J. W., Oliveira, J. E., Gondim, M. G., 2016. Estimated crop loss due to coconut mite and financial analysis of controlling the pest using the acaricide abamectin. Experimental and Applied Acarology, 69(3), 297-310.

RICARDO AEA, 2015. Light weighting as a means of improving Heavy Duty Vehicles' energy efficiency and overall CO2 emissions. https://ec.europa.eu/clima/sites/clima/files/transport/vehicles/heavy/docs/hdv_lightweighting_en.pdf

Ritchie, A., & Roser, M., 2022. - "Environmental Impacts of Food Production". Published online at OurWorldInData.org. Retrieved from: 'https://ourworldindata.org/environmental-impacts-of-food' [Online Resource]

RIM, 2013 – Revista Ideias na Mesa – vol. 1 - publicação periódica da Rede Virtual Ideias na Mesa, e uma parceria entre o Observatório de Políticas de Segurança Alimentar e Nutrição (Opsan/UnB) e o Ministério do Desenvolvimento Social e Agrário (MDSA).

Ripa, M., Fiorentino, G., Vacca, V., Ulgiati, S., 2017. The relevance of site-specific data in Life Cycle Assessment (LCA). The case of the municipal solid waste management in the metropolitan city of Naples (Italy). Journal of cleaner production, 142, pp. 445-460

RIMA, 2016. Relatorio de impacto Ambiental. Ecodue, ESSENCIS. Available at: http://www.sigrh.sp.gov.br/public/uploads/events/CBH-AT/4333/volivrimactvacaieiras.pdf

Rossi, V., Cleve-Edwards, N., Lundquist, L., Shenkern, U., Dubois, C., Humbert, S., Jolliet, O., 2015. Life cycle assessment of end-of-life options for two biodegradable packaging materials: sound application of the European waste hierarchy. Journal of Cleaner Production, 86, pp. 132 – 145. Doi: 10.1016/j.jclepro.2014.08.049

Rugani, B. & Benetto, E., 2012. Improvements to Emergy Evaluations by Using Life Cycle Assessment.

Environmental Science & Technology, 46 (9), pp.4701–4712

Rockström, J. et al., 2009. A safe operating space for humanity. Nature, 461(7263), pp. 472–5. Available at: http://dx.doi.org/10.1038/461472a

SABESP, 2018. Relatório de Sustentabilidade, São Paulo. http://site.sabesp.com.br/site/uploads/file/relatorios_sustentabilidade/sabesp_rs_2018_portugues(1).p df

Salem, Z., Hamouri, K., Djeema, R., Allia, K., 2008. Evaluation of landfill leachate pollution andtreatment. Desalination 220, 108–114. https://doi.org/10.1016/j.desal.2007.01.026

Salemdeeb, R., Zu Ermgassen, E. K., H. J., Kim, M. H., Balmford, A., Al-Tabbaa, A., 2017. Environmental and health impacts of using food waste as animal feed: a comparative analysis of food waste management options. Journal of cleaner production, 140, 871-880. Doi: 10.1016/j.jclepro.2016.05.049

Salhofer, S., Obersteiner, G., Schneider, F., Lebersorger, S., 2008. Potentials for the prevention of municipal solid waste. Waste Management, 28, 245-259. Doi:10.1016/j.wasman.2007.02.026.

San Martin, D., Ramos, S., Zufía, J. 2016. Valorisation of food waste to produce new raw materials for animal feed. Food chemistry, 198, 68-74. Doi: 10.1016/j.foodchem.2015.11.035

Santagata R., Viglia S., Fiorentino G., Gengyuan L., Ripa M., 2019. Power generation from slaughterhouse waste materials. An emergy accounting assessment. Journal of cleaner production, 223, pp. 536-552. http://dx.doi.org/10.1016/j.jclepro.2019.03.148.

Santagata, R., Zucaro, A., Fiorentino, G., Lucagnano, E., Ulgiati, S., 2020. Developing a procedure for the integration of Life Cycle Assessment and Emergy Accounting approaches. The Amalfi paper case study. Ecological Indicators, 117, 106676. Doi: 10.1016/j.ecolind.2020.106676

Sanzonowicz, C., Toledo, P.M.D.R., Sampaio, J.B.R., Guerra, A.F. and Silva, D.T.M.D., 2003. Adubação nitrogenada em café decotado num latossolo de cerrado. Available at: http://www.sbicafe.ufv.br/handle/123456789/7765

Sawatdeenarunat, C., Nguyen, D., Surendra, K.C., Shrestha, S., Rajendran, K., Oechsner, H., Xie, L. and Khanal, S.K., 2016. Anaerobic biorefinery: current status, challenges, and opportunities. Bioresource technology, 215, pp.304-313.

Scaglia, B., D'Incecco, P., Squillace, P., Dell'Orto, M., De Nisi, P., Pellegrino, L., Botto, A., Cavicchi, C., Adani, F., 2020. Development of a tomato pomace biorefinery based on a CO2-supercritical extraction process for the production of a high value lycopene product, bioenergy and digestate. Journal of Cleaner Production, 243, 118650.

Schneider, F., 2013. The evolution of food donation with respect to waste prevention. Waste Management, 33, 755-763. Doi: 10.1016/j.wasman.2012.10.025

Schiopu, A.-M., Gavrilescu, M., 2010. Options for the treatment and management of municipal landfill leachate: common and Specific Issues. Clean. – Soil, Air, Water 38, 1101–1110. https://doi.org/10.1002/clen.200900184

SGC, 2012. Basic Data on Biogas. Swedish Gas Technology Centre. Available at: http://www.sgc.se/ckfinder/userfiles/files/BasicDataonBiogas2012.pdf

Shanes, K., Doberning, K., Gozet, B., 2018. Food waste matters – A systematic review of household food waste practise and their policy implications. Jounrnal of Cleaner Production, 182, pp 978 – 991. Available at: https://doi.org/10.1016/j.jclepro.2018.02.030

Sharma, S.S., 2011. Determinants of carbon dioxide emissions: Empirical evidence from 69 countries. Applied Energy, 88(1), pp.376–382. Available at: https://doi.org/10.1016/j.apenergy.2010.07.022

Singh, B., Szamosi, Z., Siménfalvi, Z. (2019). State of the art on mixing in an anaerobic digester: A review. Renewable Energy, 141, 922-936.

Sigolo, J., B., Pinheiro, C., H., R., 2010. Lodo de Esgoto da ETE Barueri - SP: Proveniência do Enxofre Elementar e Correlações com Metais Pesados Associados. Geol. USP, Sér. cient., São Paulo, v. 10, n. 1, pp. 39-51. https://doi.org/10.5327/Z1519-874X2010000100004

Silva, D. E. W., da Silva César, A., Conejero, M. A. 2021. Prevention of food waste and alternative destinations for unused food in Brazil. Journal of Cleaner Production, 318, 128545. Doi: 10.1016/j.jclepro.2021.128545

Silva Júnior, F.D.C.G.D., Menezes, C.A.D., Cavalcante, W.A., Aragão, O.P., Zaiat, M., Leitão, R.C., 2022. Characterization of Fruits and Vegetables Waste Generated at a Central Horticultural Wholesaler: A Case Study for Energy Production Via Biogas. Industrial Biotechnology, 18(4), pp.235-239.

Slagstad, H., Brattebø, H., 2012. LCA for household waste management when planning a new urban settlement. Waste Management, 32, pp. 1482-1490. http://dx.doi.org/10.1016/j.wasman.2012.03.018.

Slack, R.J., Gronow, J.R., Vouvoulis, N., 2005. Household hazardous waste in municipal landfills: contaminants in leachate. Science. Total Environment. 337, 119–137. 10.1016/j.scitotenv.2004.07.002

Socas-Rodríguez, B., Álvarez-Rivera, G., Valdés, A., Ibáñez, E., Cifuentes, A., 2021. Food by-products and food wastes: are they safe enough for their valorization? *Trends in Food Science & Technology*, 114, 133 – 147. Doi: 10.1016/j.tifs.2021.05.002

Sofokleous, M., Christofi, A., Malamis, D., Mai, S. and Barampouti, E.M., 2022. Bioethanol and biogas production: an alternative valorisation pathway for green waste. Chemosphere, 296, p.133970. Doi: 10.1016/j.chemosphere.2022.133970

Souto, G.B.; Povinelli, J, 2007. Características do lixiviado de aterro no Brasil. In: 24º Congresso Brasileiro de Engenharia Sanitária e Ambiental, Belo Horizonte, Minas Gerais.

Spiess, A.N. and Neumeyer, N., 2010. An evaluation of R^2 as an inadequate measure for nonlinear models in pharmacological and biochemical research: a Monte Carlo approach. BMC pharmacology, 10(1), pp.1-11

Stenmarck, A., Jensen, C., Quested, T., Moates, G., 2016. Estimates of European Food Waste Levels. IVL Swedish Environmental Research Institute.

Stichnothe, H., Azapagic, A., 2009. Bioethanol from waste: Life cycle estimation of the greenhouse gas saving potential. Resources, Conservation and Recycling, 53(11), 624-630. Doi: 10.1016/j.resconrec.2009.04.012

Still Brazil, 2020. Available at https://www.stil

Sun, Q., Li, H., Yan, J., Liu, L., Yu, Z., Yu, X., 2015. Selection of appropriate biogas upgrading technology-a review of biogas cleaning, upgrading and utilisation. Renewable and Sustainable Energy Reviews, 51, pp.521-532.

Sundin, N., Osowski, C.P., Strid, I. and Eriksson, M., 2022. Surplus food donation: Effectiveness, carbon footprint, and rebound effect. Resources, Conservation and Recycling, 181, p.106271. Doi: 10.1016/j.resconrec.2022.106271

Tampio, E., Marttinen, S., Rintala, J., 2016. Liquid fertilizer products from anaerobic digestion of food waste: mass, nutrient and energy balance of four digestate liquid treatment systems. Journal of Cleaner Production, 125, pp.22-32. Doi: 10.1016/j.jclepro.2016.03.127

TBCA, 2020. Tabela Brasileira de Composição de Alimentos (TBCA). Universidade de São Paulo (USP). Food Research Center (FoRC). Versão 7.1. São Paulo, 2020. Available at: http://www.fcf.usp.br/tbca.

Tectermica, 2020. Available at: https://tectermica.com.br

Teigiserova, D. A., Hamelin, L.,Thomsen, M., 2019. Review of high-value food waste and food residues biorefineries with focus on unavoidable wastes from processing. Resources, Conservation and Recycling, 149, 413-426. Doi:10.1016/j.resconrec.2019.05.003

Traven, L., 2019. Circular economy and the waste management hierarchy: Friends or foes of sustainable economic growth? A critical appraisal illustrated by the case of the Republic of Croatia. Waste Management & Research, 37 (1), pp. 1 – 2.

Themelis, J., L., Ulloa, P., A., 2007. Methan Generation in landfills. Renewable Energy, 32, pp. 1243 – 1257. Doi: 10.1016/j.renene.2006.04.020

Tisserant, A., Pauliuk, S., Merciai, S., Schmidt, J., Fry, J., Wood, R., Tukker, A., 2017. Solid waste and the circular economy: a global analysis of waste treatment and waste footprints. J. Ind. Ecol. 21 (3), 628–640. https://doi.org/10.1111/jiec.12562.

Triantaphyllou, E., Shu, B., Sanchez, S.N., Ray, T., 1998. Multi-criteria decision making: an operations research approach. Encyclopedia of electrical and electronics engineering, 15(1998), pp.175-186.

Triturtec, 2022. Available at: https://www.tecscan.com.br/equipamentos/trituradores/tritutec/

Tsegaye, B., Jaiswal, S., Jaiswal, A. K., 2021. Food Waste Biorefinery: Pathway towards Circular Bioeconomy. Foods, 10(6), 1174. Doi:10.3390/foods10061174

Tunesi, S., 2011. LCA of local strategies for energy recovery from waste in England, applied to a large municipal flow. Waste Management. 31, pp. 561-571. Doi: 10.1016/j.wasman.2010.08.023

Ubando, A. T., Felix, C. B., Chen, W. H., 2020. Biorefineries in circular bioeconomy: A comprehensive review. Bioresource technology, 299, 122585. Doi: 10.1016/j.biortech.2019.122585

Ulgiati, S., Brown, M., T., 2014. Labor and Services as Information Carriers in EMergy-LCA Accounting. Journal of Environmental Accounting and Management, 2 (2), 163-170. DOI:10.5890/JEAM.2014.06.

Uratani, C.L.T., de Palma, J.B., Schultze, R.F., 2014. Estudo de concepção de um sistema de tratamento para os resíduos sólidos orgânicos do CEAGESP. Universidade de São Paulo.

UN, 2015. UN General Assembly, Transforming our world : the 2030 Agenda for Sustainable Development, 21 October 2015, A/RES/71, available at: https://www.refworld.org/docid/57b6e3e44.html

Van Ewijk, S., Stegemann J., A., 2016. Limitations of the waste hierarchy for achieving absolute reductions in material throughput, Journal of Cleaner Production, 132, 122-128. Doi:10.1016/j.jclepro.2014.11.051

Vandermeersch, T., Alvarenga, R. A. F., Ragaert, P., Dewulf, J., 2014. Environmental sustainability assessment of food waste valorization options. Resources, Conservation and Recycling, 87, 57-64.

Viana, 2015. CÁLCULO DA VIDA ECONÔMICA DE CAMINHÕES DE TRANSPORTE EM UMA INDÚSTRIA. Trabalho de conclusão de curso de graduação.

Vujanac, R., Djordjevic, M., Miloradovic, N., 2017. Mechanical tugger trains system for internal material handling. In: The Sixth International Conference, Transport and Logistic (Til 2017), pp. 73–76. Available at. http://til.masfak.ni.ac.rs/images/til-pedja/2.6.pdf. July 14th, 2021.

Wang, P., Hu, Y., Cheng, H., 2019. Municipal solid waste (MSW) incineration fly ash as an important source of heavy metal pollution in China. Environmental pollution, 252, 461-475. Doi: 10.1016/j.envpol.2019.04.082

Wang, Y., Zhang, X., Liao, W., Wu, J., Yang, X., Shui, W., Deng, S., Zhang, Y., Lin, L., Xiao, Y. and Yu, X., 2018. Investigating impact of waste reuse on the sustainability of municipal solid waste (MSW) incineration industry using emergy approach: A case study from Sichuan province, China. Waste Management, 77, pp.252-267. Doi: 10.1016/j.wasman.2018.04.003

Wiersinga, R.C., Vermeulen, T., Snels, J.C.M.A., da Silva, R.F., Wiersema, S.G., van der Mei, M., 2013. Sustainable vegetable chain development in Brazil: Grape tomatoes from Holambra to the higher market segment. Rapport GTB-1262. Wageningen UR Glastuinbouw. Available at. https://edepot.wur.nl/288815. July 14th, 2021

Yang, N., Damgard, A., Lu, F., Shao, L., M., Brogaard, L., K.,S., He, P.,J., 2014. Environmental impact assessment on the construction and operation of municipal solid waste sanitary landfills in developing countries: China case study. Waste Management, 34, pp 929 – 937, 2014. https://doi.org/10.1016/j.wasman.2014.02.017

Yang, X., Choi, H. S., Park, C., Kim, S. W., 2015. Current states and prospects of organic waste utilization for biorefineries. Renewable and Sustainable Energy Reviews, 49, 335-349. Doi: 10.1016/j.rser.2015.04.114

Zand A., D., Abyaneh, M., R., Hoveidi, H. Environmental and Economic Evaluation of Municipal Solid Waste Management uding WAGS Model – Air Pollutant Emission and Fuel Economy in Waste Collection Sector. Environmental Energy and Economic Research, 3 (1), pp. 37 – 44, 2019.

Zanotti, 2014. Termoverde Caieiras Biogas Thermoelectric Plant BRAZIL. Harward University. https://research.gsd.harvard.edu/zofnass/files/2016/08/12_TermoverdeCaieiras_EN_Final-version.pdf

Zhang, J., & Ma, L., 2020. Environmental sustainability assessment of a new sewage treatment plant in China based on infrastructure construction and operation phases emergy analysis. *Water*, *12*(2), 484.

Ziegler-Rodriguez, K., Margallo, M., Adalco, Vazquez-Rowe, I., Kahhat, R., 2019. Transitioning from open dumpster to landfilling in Peru: Environmental benefits and challenges from a Life-cycle perspective. Journal of cleaner production, 229, pp. 989-1003.

Appendix A: overview products traded, and waste generated at CEAGESP.

Table A1: Overview of product traded, and waste generated in CEAGESP from 2007 to 2018. Source: CEAGESP reports from 2008 to 2019.

Year	Volume Traded	Waste generated	% waste gen/vol	Waste Recycled	Waste Discarded	% waste rec/gen	% waste disc/gen
y	ton/yr	ton/yr	%	ton/yr	ton/yr	%	%
2007	3.033.812	39.486	1,30	9.485	30.001	24,02	75,98
2008	3.113.765	43.630	1,40	5.271	38.359	12,08	87,92
2009	3.155.052	47.399	1,50	20.907	26.492	44,11	55,89
2010	3.159.383	52.927	1,68	17.420	35.507	32,91	67,09
2011	3.234.362	55.585	1,72	14.778	40.807	26,59	73,41
2012	3.401.122	55.349	1,63	11.561	43.788	20,89	79,11
2013	3.371.034	56.387	1,67	10.731	45.656	19,03	80,97
2014	3.412.821	59.783	1,75	13.004	46.779	21,75	78,25
2015	3.371.803	60.195	1,79	14.608	45.587	24,27	75,73
2016	3.198.227	51.499	1,61	11.266	40.233	21,88	78,12
2017	3.301.049	54.259	1,64	8.514	45.745	15,69	84,31
2018	3.063.798	51.767	1,69	4.702	47.065	9,08	90,92
Average	**3.234.686**	**52.356**	**1,61**	**11.854**	**40.502**	**22,69**	**77,31**

Appendix B: LCA calculation procedures

Calculation details scenarios #I to #VIII. Indirect impacts were calculated multiplying the annual input by the characterization factor of each impact category as available in table B9. Direct impacts were calculated multiplying the annual input according to the equations shown in table 7.

Table B1: Direct and Indirect Impacts calculation of scenario #I. All values in the impact categories are per ton of OBP.

Description	Annual input	Unit	Annual Input per 1 ton OBP	FDP (in Kg oil eq)	P (in kg Peq)	GWP 100 (Kg CO_2 eq)	HTP (in Kg 1,4-DCB eq)	MDP (in kg Fe eq)	PMFP (in kg PM10 eq)	POFP (in kg NMVOC-eq)	TAP100 (kg SO_2 eq)	WDP (in m^3 H_2O eq)
Steel	1.34E+04	kg	3,56E-01	3,75E-01	8,64E-04	1,62E+00	8,86E-01	9,76E-01	5,47E-03	5,68E-03	5,70E-03	1,16E-02
Iron	2.30E+03	kg	6,11E-02	2,51E-02	3,72E-05	1,14E-01	1,12E-01	2,52E-03	3,95E-04	4,46E-04	3,75E-04	6,07E-04
Rubber	1.26E+03	kg	3,34E-02	6,36E-02	2,98E-05	9,14E-02	3,25E-02	5,11E-02	1,87E-04	4,34E-04	4,11E-04	1,48E-03
Plastic	1.22E+03	kg	3,23E-02	6,25E-02	1,36E-05	7,18E-02	1,33E-02	2,36E-03	9,45E-05	2,52E-04	2,36E-04	6,23E-04
Aluminum	7.75E+02	kg	2,06E-02	7,51E-02	8,79E-05	2,66E-01	1,67E-01	5,93E-01	5,73E-04	8,94E-04	1,47E-03	9,69E-03
Diesel	1.95E+05	kg	5,17E+00	6,30E+00	2,23E-04	2,42E+00	4,40E-01	6,89E-02	6,93E-03	1,80E-02	2,22E-02	8,43E-03
GCL	1.95E+04	kg	5,18E-01	7,64E-03	4,62E-06	2,43E-02	6,91E-03	3,25E-03	7,22E-05	1,88E-04	1,63E-04	2,20E-04
HDPE	1.52E+04	kg	4,03E-01	7,62E-01	1,86E-04	9,09E-01	1,75E-01	2,90E-02	1,25E-03	3,24E-03	2,97E-03	8,91E-03
Geotextile	4.45E+03	kg	1,18E-01	2,53E-01	9,57E-05	3,37E-01	8,43E-02	1,32E-02	5,56E-04	1,24E-03	1,23E-03	3,12E-03
Gravel	6.75E+06	kg	1,79E+02	2,51E-01	1,07E-04	9,22E-01	1,57E-01	7,17E-02	9,70E-03	3,40E-02	2,78E-02	1,84E-02
Cement	6.63E+02	kg	1,76E-02	1,53E-03	3,95E-07	1,23E-02	9,08E-04	9,86E-04	1,73E-05	4,43E-05	4,02E-05	5,66E-05
Electricity	1.50E+04	kWh	3,99E-01	2,19E-02	1,14E-05	7,83E-02	1,24E-02	1,53E-03	1,77E-04	2,57E-04	4,77E-04	8,28E-03
Ferric chloride	1.66E+04	kg	4,40E-01	5,51E-02	1,57E-04	1,88E-01	1,95E-01	5,27E-02	5,31E-04	6,72E-04	1,10E-03	3,72E-03
Polyacrylamide	1.27E+03	kg	3,37E-02	5,68E-02	1,78E-05	9,68E-02	2,19E-02	4,98E-03	1,54E-04	2,56E-04	5,19E-04	1,23E-03
Total Indirect Emissions				**8,31E+00**	**1,83E-03**	**7,15E+00**	**2,30E+00**	**1,87E+00**	**2,61E-02**	**6,56E-02**	**6,47E-02**	**7,64E-02**
Diesel direct emissions		kg	5,17E+00			1,62E+01			3,33E-01	2,13E-01	1,61E+00	
Methane to atmosphere		kg	8,05E+00			1,79E+02				8,05E-02		
Phosphorus to wastewater		kg	5,03E-03		5,03E-03							
Total Direct Emissions					**5,03E-03**	**1,95E+02**			**3,33E-01**	**2,94E-01**	**1,61E+00**	
Percentage First Contributor				76	73	88	38	52	93	59	96	24
IMPACTS SCENARIO 1				**8,31E+00**	**6,86E-03**	**2,03E+02**	**2,30E+00**	**1,87E+00**	**3,59E-01**	**3,60E-01**	**1,68E+00**	**7,64E-02**

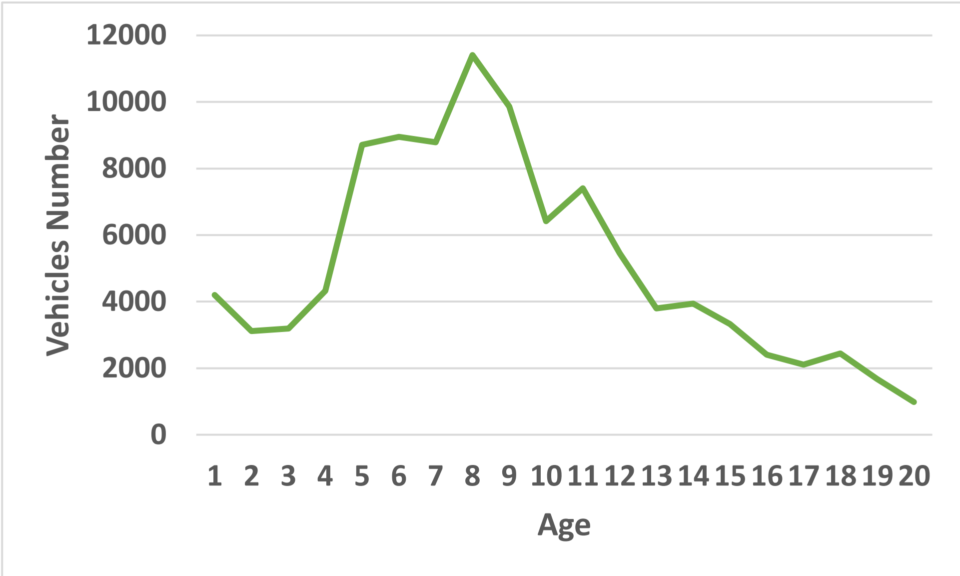

Figure B1: Number of Diesel trucks (weight 15 < t < 45) circulating in SP state in 2018 per age (CETESB, 2019)

Table B2: Direct and indirect impacts of scenario #I – STEP I. Annual Inputs raw materials per 1-ton OBP. For vehicles it was considered a lifespan of 10 years (Viana, 2015; CETESB – 2018). Percentages of vehicular materials form Ricardo AEA (2015) of 60.14, 10.60, 5.80, 5.60, 3.57 for steel, iron, rubber, plastic aluminum respectively.

Description				**Fossil Depletion**	**Freshwater Eutrophication**	**Global Warming**	**Human Toxicity**	**Metal Depletion**	**Particular Matter formation**	**Photochemical Oxidant Formation**	**Terrestrial Acidification**	**Water depletion**
1 Excavator Doosan Daewoo Solar 175 LCV[b]	**Material weight**	**Unit**	**Annual input raw material**	**FDP (in Kg oil eq)**	**FEP (in kg P eq/kg)**	**GWP 100 (Kg CO_2 eq/kg)**	**HTP Inf (Kg 1,4-DCB eq/kg)**	**MDP (in kg Fe eq /kg)**	**PMFP (in kg PM_{10} eq/ kg)**	**POFP (in kg NMVOC-eq)**	**TAP100 in kg SO_2 eq**	**WDP (in m^3 H_2O eq/kg)**
Steel	2.79E+04	kg	7.41E-02	7.80E-02	1.80E-04	3.37E-01	1.85E-01	2.03E-01	1.14E-03	1.18E-03	1.19E-03	2.42E-03
Iron	4.92E+03	kg	1.31E-02	5.36E-03	7.95E-06	2.44E-02	2.40E-02	5.38E-04	8.44E-05	9.54E-05	8.03E-05	1.30E-04
Rubber	2.69E+03	kg	7.15E-03	1.36E-02	6.36E-06	1.95E-02	6.94E-03	1.09E-02	3.99E-05	9.27E-05	8.79E-05	3.15E-04
Plastic	2.60E+03	kg	6.90E-03	1.34E-02	2.91E-06	1.53E-02	2.84E-03	5.04E-04	2.02E-05	5.39E-05	5.05E-05	1.33E-04
Aluminum	1.66E+03	kg	4.40E-03	1.61E-02	1.88E-05	5.69E-02	3.58E-02	1.27E-01	1.23E-04	1.91E-04	3.14E-04	2.07E-03
Machine Use												
Diesel fuel consumption[c]	9.79E+04		2.60E+00	3.16E+00	1.12E-04	1.22E+00	2.21E-01	3.46E-02	3.48E-03	9.03E-03	1.11E-02	4.23E-03
Diesel combustion Emissions to air						8.14E+00			1.67E-01	1.07E-01	8.10E-01	

a: Annual Input raw material per ton: Vehicles = (Material weight / Lifespan) / 37652 tons (organic fraction); Fuels: Material weight / 37652 tons (organic fraction)

b: From Zand et al., 2019; Average consumption of compactor trucks with 15 m^3 capacity is 8 L / h; Calc. 8 L/h *8 trucks * 1815 hrs/ yr (5 hrs day * 363, excl. 25 dec. and 1st of Jan) = 116159 L/yr * 0,8425 kg/L (dens. Diesel Br, source Da Silva, 2017).

TABLE B3: Direct and indirect impacts of SCENARIO #I – STEP II. Annual input raw materials per 1-ton OBP. Lifespan excavator 14 years**.** Percentages of vehicular materials form Ricardo AEA (2015) of 60.14, 10.60, 5.80, 5.60, 3.57 for steel, iron, rubber, plastic aluminum respectively.

Description				**Fossil Depletion**	**Freshwater Eutrophication**	**Global Warming**	**Human Toxicity**	**Metal Depletion**	**Particular Matter formation**	**Photochemical Oxidant Formation**	**Terrestrial Acidification**	**Water depletion**
1 Excavator Doosan Daewoo Solar 175 LCV[b]	**Material weight**	**Unit**	**Annual input raw material**	**FDP (in Kg oil eq)**	**FEP (in kg P eq/kg)**	**GWP 100 (Kg CO_2 eq/kg)**	**HTP Inf (Kg 1,4-DCB eq/kg)**	**MDP (in kg Fe eq /kg)**	**PMFP (in kg PM_{10} eq/ kg)**	**POFP (in kg NMVOC-eq)**	**TAP100 in kg SO_2 eq**	**WDP (in m^3 H_2O eq/kg)**
Steel	1.05E+04	kg	1.99E-02	2.09E-02	4.82E-05	9.03E-02	4.94E-02	5.44E-02	3.05E-04	3.17E-04	3.18E-04	6.48E-04
Iron	1.84E+03	kg	3.50E-03	1.43E-03	2.13E-06	6.54E-03	6.43E-03	1.44E-04	2.26E-05	2.56E-05	2.15E-05	3.48E-05
Rubber	1.01E+03	kg	1.91E-03	3.64E-03	1.70E-06	5.23E-03	1.86E-03	2.93E-03	1.07E-05	2.48E-05	2.36E-05	8.45E-05
Plastic	9.74E+02	kg	1.85E-03	3.58E-03	7.80E-07	4.11E-03	7.59E-04	1.35E-04	5.41E-06	1.44E-05	1.35E-05	3.57E-05
Aluminum	6.21E+02	kg	1.18E-03	4.30E-03	5.03E-06	1.52E-02	9.58E-03	3.40E-02	3.28E-05	5.12E-05	8.40E-05	5.55E-04
Machine Use												
Diesel fuel consumption[c]	2.16E+04	kg/yr	5.73E-01	6.97E-01	2.47E-05	2.68E-01	4.87E-02	7.63E-03	7.67E-04	1.99E-03	2.45E-03	9.33E-04
Diesel combustion Emissions to air	2.16E+04	kg/yr	5.73E-01			1.79E+00			3.69E-02	2.36E-02	1.79E-01	

a: Annual Input raw material per ton: Vehicles = (Material weight / Lifespan) / 37652 tons (organic fraction); Fuels: Material weight / 37652 tons (organic fraction)

b: Produced between 2003 - 2006, equipped with Diesel engine DB58TIS 126,5 kW, consumption 217.5 g/kW.h (160 g/PS.h), source Daewoo DB58 T/TI/TIS operation and maintenance manual, 2013.

c: Consumption: (217.5 g/kW.h * 126.5 kW) *(784 h/y) *(10^{-3} kg/g) = 21571 kg/yr * (1.187 L/kg Brasilian Diesel) = 25,605 L/yr

TABLE B4: Direct and indirect impacts of SCENARIO #I – STEP III. Annual input raw materials per 1-ton OBP. Lifespan Trucks 10 years. Percentages of vehicular materials form Ricardo AEA (2015) of 60.14, 10.60, 5.80, 5.60, 3.57 for steel, iron, rubber, plastic aluminum respectively.

2 Transport Trucks (2*14550 kg)	Material weight	Unit	Annual input raw material[a]	FDP (in Kgoil eq)	FEP (in kg P eq/kg)	GWP 100 (Kg CO_2 eq/kg)	HTP Inf(Kg 1,4-DCB eq/kg)	MDP (in kgFe eq /kg)	PMFP (in kg PM_{10}eq/ kg)	POFP (in kg NMVOC-eq)	TAP100 in kg SO_2 eq	WDP (in m^3 H_2O eq/kg)
Steel	1.75E+04	kg	4.65E-02	4.89E-02	1.13E-04	2.12E-01	1.16E-01	1.27E-01	7.15E-04	7.42E-04	7.44E-04	1.52E-03
Iron	3.08E+03	kg	8.19E-03	3.36E-03	4.98E-06	1.53E-02	1.51E-02	3.38E-04	5.29E-05	5.98E-05	5.03E-05	8.14E-05
Rubber	1.69E+03	kg	4.48E-03	8.52E-03	3.99E-06	1.23E-02	4.35E-03	6.85E-03	2.50E-05	5.81E-05	5.51E-05	1.98E-04
Plastic	1.63E+03	kg	4.33E-03	8.38E-03	1.83E-06	9.62E-03	1.78E-03	3.16E-04	1.27E-05	3.38E-05	3.17E-05	8.36E-05
Aluminum	1.04E+03	kg	2.76E-03	1.01E-02	1.18E-05	3.57E-02	2.24E-02	7.95E-02	7.68E-05	1.20E-04	1.97E-04	1.30E-03
Vehicles Use												
Diesel fuel Consumption[b]	1.79E+04	kg	4.76E-01	5.79E-01	2.05E-05	2.23E-01	4.04E-02	6.34E-03	6.37E-04	1.65E-03	2.04E-03	7.75E-04
Diesel combustion Emissions to air	1.79E+04	kg	4.76E-01			1.49E+00			3.07E-02	1.96E-02	1.48E-01	

a: Annual Input raw material per ton: Vehicles = (Material weight / Lifespan) / 37652 tons (organic fraction); Fuels: Material weight / 37652 tons (organic fraction)

b: Dist. CEAGESP - Caieiras 24,2 km; Consumption truck 0.28 L/km (Source: CETESB, 2019. Trucks: 15 < t < 40); Number annual trips CEAGESP - CAIEIRAS = 47065 t waste / 30 tons Truck Capacity = 1,569 trips; One trip consumption: 0,28 L/km * 24,2 km* 2 = 13,552 L/ trip; Total yearly consumption 13,552 L / trip*(annual trips) = 13.552 * 1569 = 21263 L/yr; Conversion factor Brazilian Diesel = 1 L = 0,8425 kg/L = 0,8425 kg; Total yearly consumption in kg = 21263 L/yr * 0,8425 kg/L = 17,914 kg/y

TABLE B5: Direct and indirect impacts of SCENARIO #I – STEP IV. Annual input raw materials per 1-ton OBP. Lifespan Trucks and machines 10 years. Percentages of vehicular materials form Ricardo AEA (2015) of 60.14, 10.60, 5.80, 5.60, 3.57 for steel, iron, rubber, plastic aluminum respectively.

Description				Fossil Depletion	Freshwater Eutrophication	Global Warming	Human Toxicity	Metal Depletion	Particular Matter formation	Photochemical Oxidant Formation	Terrestrial Acidification	Water depletion
Vehicles Use	**Material weight**	**Unit**	**Annual input raw material[a]**	**FDP (in Kg oil eq)**	**FEP (in kg P eq/kg)**	**GWP 100 (Kg CO_2 eq/kg)**	**HTP Inf (Kg 1,4-DCB eq/kg)**	**MDP (in kg Fe eq /kg)**	**PMFP (in kg PM_{10} eq/ kg)**	**POFP (in kg NMVOC-eq)**	**TAP100 in kg SO_2 eq**	**WDP (in m^3 H_2O eq/kg)**
Diesel fuel Consumption[b]	5.22E+04	kg	1.39E+00	1.69E+00	5.98E-05	6.50E-01	1.18E-01	1.85E-02	1.86E-03	4.82E-03	5.94E-03	2.26E-03
Diesel combustion emissions to air	5.22E+04	kg	1.39E+00			4.35E+00			8.94E-02	5.73E-02	4.33E-01	
Machines Materials[c]												
Steel	6.12E+04	kg	1.63E-01	1.71E-01	3.95E-04	7.40E-01	4.05E-01	4.46E-01	2.50E-03	2.60E-03	2.60E-03	5.31E-03
Iron	1.08E+04	kg	2.86E-02	1.17E-02	1.74E-05	5.35E-02	5.26E-02	1.18E-03	1.85E-04	2.09E-04	1.76E-04	2.85E-04
Rubber	5.90E+03	kg	1.57E-02	2.98E-02	1.40E-05	4.29E-02	1.52E-02	2.40E-02	8.75E-05	2.03E-04	1.93E-04	6.92E-04
Plastic	5.70E+03	kg	1.51E-02	2.93E-02	6.39E-06	3.36E-02	6.22E-03	1.10E-03	4.43E-05	1.18E-04	1.11E-04	2.92E-04
Aluminum	3.63E+03	kg	9.65E-03	3.52E-02	4.12E-05	1.25E-01	7.84E-02	2.78E-01	2.69E-04	4.19E-04	6.88E-04	4.54E-03
Landfill Capital Goods (Construction and Operation)												
Total materials												
GCL[d]	1.95E+04	kg	5.18E-01	7.64E-03	4.62E-06	2.43E-02	6.91E-03	3.25E-03	7.22E-05	1.88E-04	1.63E-04	2.20E-04
HDPE[e]	1.52E+04	kg	4.03E-01	7.62E-01	1.86E-04	9.09E-01	1.75E-01	2.90E-02	1.25E-03	3.24E-03	2.97E-03	8.91E-03
Geotextile	4.45E+03	kg	1.18E-01	2.53E-01	9.57E-05	3.37E-01	8.43E-02	1.32E-02	5.56E-04	1.24E-03	1.23E-03	3.12E-03
Gravel[g]	6.75E+06	kg	1.79E+02	2.51E-01	1.07E-04	9.22E-01	1.57E-01	7.16E-02	9.69E-03	3.39E-02	2.78E-02	1.84E-02

a: Annual Input raw material per ton: Vehicles = (Material weight / Lifespan) / 37652 tons (organic fraction); Fuels: Material weight / 37652 tons (organic fraction)

b: Average consumption per ton 1.11 kg/ / ton waste from Yang et al. (2014).

c: The step considers 5 machines (1 Hyundai 220 LC excavator, 1 bulldozer, 1 soil compactor, 1 front loader, and 1 truck of about 22 tons, 30 tons, 12 tons, 23.5 tons and 14.5 tons weight respectively)

d: Average value of Yang et al. (2014) and Menard et al. (2004). Menard et al. (2004): 257000 kg /600000 tons = 0.428 kg/t (bentonite excluded.)

e: Average value of Yang et al. (2014), Brogaard et al. (2013), Cherubini et al. (2009) and Menard et al. (2004). Yang et al. (2014) including HDPE geomembranes, pipes and geonets; Cherubini etal. (2009) including landfill walls and pipes. Menard et al (2004): geom + pipes = (223000 kg + 265000 kg)/600000 t = 0.813 t/kg (PVC excluded)f: Average value of Yang et al. (2014) and Menard et al. (2004)

g: Average value of Yang et al. (2014), Brogaard et al. (2013), and Menard et al. (2004).

TABLE B6: Emissions from waste degradation SCENARIO #I – STEP V.

				Fossil Depletion	Freshwater Eutrophication	Global Warming	Human Toxicity	Metal Depletion	Particular Matter formation	Photochemical Oxidant Formation	Terrestrial Acidification	Water depletion
Emissions Biogas[a] Caieiras Landfill	**Material weight**	Unit	**Annual input raw material**	**FDP (in Kg oil eq)**	**:P (in kg Peq/kg)**	**GWP 100 (Kg CO_2 eq/kg)**	**HTP Inf (Kg 1,4-DCB eq/kg)**	**MDP (in kg Fe eq /kg)**	**PMFP (in kg PM_{10}eq/ kg)**	**POFP (in kg NMVOC-eq)**	**TAP100 in kg SO_2 eq**	**WDP (in m^3 H_2O eq/kg)**
Total Biogas emitted[b]	3.56E+06	Nm^3	9.45E+01									
CH_4 total[c]	1.47E+06	kg	3.91E+01			8.71E+02						
CH_4 emissions. Atm.[d]	2.95E+05	kg	7.83E+00			1.74E+02				7.83E-02		
CH_4 burnt flares	5.90E+05	kg	1.57E+01									
CH_4 elettricity	5.90E+05	kg	1.57E+01									
Direct Emissions from wastewater Treatment[e]												
P to water[f]	1.89E+02	kg	5.03E-03		5.03E-03							
CH_4 to air[d, g]	8.44E+03	kg	2.24E-01			4.99E+00				2.24E-03		

a: Biogas emissions all CAIEIRAS landfill (80% captured); 58% CH_4 (Fieldwork, confirmed by Candiani and Torres, 2015). Measured (captured): 13,000 Nm^3/hr; Total hourly CAIEIRAS Emissions = 13,000 Nm^3/ hour: 80 % = x: 100 %; x = 13,000 Nm^3/hour *100%: 80% = 16,250 Nm^3 / hr. Daily Biogas CAIEIRAS Emissions 16,250 Nm^3/ hr * 24 = 390,000 Nm^3/hr; Yearly Biogas CAIEIRAS Emissions: 390,000 Nm^3/ d * 365 d = 142,350,000 Nm^3/ yr; Conversion m^3 to kg CH_4: 1 m^3 = 1000 L ; Numbers of moles of CH_4 in 1 m^3 = Mol CH_4 = 1000 L / 22.414 L mol^{-1}= 44,61 mol; Molar mass CH_4 = 16 g/mol; 44,61 mol * 16 g/mol = 713,76 g/m^3 *10^{-3} kg/g = 0.714 kg/m^3 ;

b: Biogas emission CEAGESP organic fraction: Organic fraction CAIEIRAS = 43 %. Organic fraction CAIEIRAS = 3,500,000 t/yr * 43% org fr = 1,505,000 t/ yr org; Organic fraction CEAGESP = 80%; Organic waste CEAGESP 2018 = 47,065 t/yr * 80% = 37,652 t org/ yr; CEAGESP org waste fr in CAIEIRAS = 37,652 t/yr org 1505000 t/yr org = x : 100; x = 37652 t/yr org * 100 % / 1,505,000 t / yr org = 2,50 % CEAGESP org waste fr; Biogas org waste CEAGESP fr in m^3 = 142,350,000 Nm3*yr^{-1} * 2.50 % = 3,558,750 Nm^3/yr

c: Hourly CH_4 CAIEIRAS emissions in kg = 9425 m^3/ hr * 0.714 kg/ m^3= 6,729.45 kg/hr; Daily CAIEIRAS CH_4 emissions = 6729.45 kg/hr * 24 hr = 161,507 kg/d; Yearly CAIEIRAS CH_4 emissions in kg= 161,507 kg/d * 365 d = 58,949,982 kg/ yr. Yearly emis. CH_4 CAIEIRAS per ton org waste (kg): 58,949,982 kg*yr-1/1505000 t/yr org = 39.16 kg/ ton; Percentage Organic fraction CAIEIRAS = 43 %.Organic fraction CAIEIRAS = 3,500,000 t/yr * 43% org fr = 1,505,000 t/ yr org; Organic fraction CEAGESP = 80 %; Organic waste CEAGESP 2018 = 47,065 t/yr * 80% = 37,652 t org/ yr; CEAGESPorg waste fr in CAIEIRAS = 37652 t/yr org : 1505000 t/yr org = x : 100; x = 37652 t/yr org *100 % / 1505000 t/yr org = 2,50 % CEAGESP org waste fr; Methane org waste CEAGESP fr in kg = 58,949,982 kg/yr * 2.50 % = 1,473,750 kg / yr.

d: Only these ones were accounted for GWP.

e: For methane emissions at Baruerì it was considered a CEAGESP fraction BOD percentage of 0.009%. For details see table B 7.1

f: Phosphorus in wwater Barueri (mg/l). P Leachate concentration (from Souto and Povinelli, 2009): 25.05 mg/L; Leachate inflow 1.65E+07 L/yr. P inflow and outflow Baruerì: from Marguti et al., 2008. Inflow 9 mg/L; Outlow 5 mg/L. Details in table B 6.1. (Conc P CEAGESP*flow CEAGESP) : (Conc P tot*inflow tot) = x : 100; 4.14E+08 mg/yr : 3.41E+12 mg/yr = x : 100; x = 4.14E+08 mg/yr *100 / 3.41E+12 mg/yr = 0.0121; P CEAGESP fract = 0.01 %; Conc. P outflow* flow tot = 5 mg/l * 3.78E+11Ll/yr = 1.89E+12; Mass P CEAGESP fract = 1.89E+12 mg/yr * 0,01% =1.89E+08; Mass P CEAGESP fract = 1.89E+08mg/yr*1.00E-06 kg/mg = 189 kg/yr

Table B 6.1: Allocation Phosphorus leachate CEAGESP in Wwat SABESP using Mass

Values CEAGESP		Val. Wastewater inflow	Val. Total	Unit
Concentration	2.51E+01	9.00E+00	3.41E+01	mg/l
Flow	1.65E+07	3.78E+11	3.78E+11	l/yr
Conc.*Flow	4.14E+08	3.41E+12	3.41E+12	mg/yr

g: Total emissions GHG Sabesp: 2,223,172 t CO_2 eq. Source SABESP, 2019; Emission GHG Sabesp Wastewater = 2,223,172 * 90.4% (perc. Emissions wwat plant SABESP) = 2,009,749 t CO_2 eq;Emission GHG CEAGESP leachate fraction: (2,009,747 t CO2 eq/yr) * (0,009% BOD mass fr. CEAGESP) = Emissions GHG CEAGESP leach fract = 177.21t CO_2 eq/yr; Changing conversion factor CH_4 in t CO_2 eq from Sabesp (21 times) to Recipe midpoint (H) method (22.25 times). Conversion CO_2 eq in CH_4 = (177.21 t CO_2 eq / 21 CH_4 / CO_2 eq) = 8.439 t CH_4/yr *10^3 kg/t = 8,439 kg CH_4 /yr; Conversion with new characterization factor = 8,439 kg CH_4 * 22.25 CO_2 eq = 187,757 kg CO_2 eq/yr

Table B7: Indirect Impacts of Waste Degradation – SCENARIO #I Step V. Annual input raw materials per 1-ton OBP. Lifespan Trucks and machines 10 years. Percentages of vehicular materials form Ricardo AEA (2015) of 60.14, 10.60, 5.80, 5.60, 3.57 for steel, iron, rubber, plastic aluminum respectively

Description				**Fossil Depletion**	**Freshwater Eutrophication**	**Global Warming**	**Human Toxicity**	**Metal Depletion**	**Particular Matter formation**	**Photochemical Oxidant Formation**	**Terrestrial Acidification**	**Water depletion**
Indirect Emissions Barueri[a]	**Material weight**	Unit	**Annual input raw material**	**FDP (in Kg oil eq)**	**FEP (in kg P eq/kg)**	**GWP 100 (Kg CO_2 eq/kg)**	**HTP Inf (Kg 1,4-DCB eq/kg)**	**MDP (in kg Fe eq /kg)**	**PMFP (in kg PM_{10} eq/ kg)**	**POFP (in kg NMVOC-eq)**	**TAP100 in kg SO_2 eq**	**WDP (in m^3 H_2O eq/kg)**
Electricity Consumption[b]	1.50E-04	kWh	3.99E-01	2.19E-02	1.14E-05	7.83E-02	1.24E-02	1.53E-03	1.77E-04	2.57E-04	4.77E-04	8.28E-03
Wastewater treatment chemical consumption[c]												
Ferric chloride	1.66E+04	kg	4.40E-01	5.51E-02	1.57E-04	1.88E-01	1.95E-01	5.27E-02	5.31E-04	6.72E-04	1.10E-03	3.72E-03
Polyacrylamide	1.27E+03	kg	3.37E-02	5.68E-02	1.78E-05	9.68E-02	2.19E-02	4.98E-03	1.54E-04	2.56E-04	5.19E-04	1.23E-03
Materials Barueri Sewage Plant[f]												
Cement	6.63E+02	kg	1.76E-02	1.53E-03	3.95E-07	1.23E-02	9.08E-04	9.86E-04	1.73E-05	4.43E-05	4.02E-05	5.66E-05
Gravel	7.42E+03	kg	1.97E-01	2.76E-04	1.17E-07	6.04E-10	1.73E-04	7.88E-05	1.07E-05	3.73E-05	3.06E-05	2.02E-05
Steel	3.45E+02	kg	9.16E-03	9.64E-03	2.22E-05	4.17E-02	2.28E-02	2.51E-02	1.41E-04	1.46E-04	1.47E-04	2.99E-04
Total Machine Materials[g]												
Steel	1.64E+04	kg	4.37E-02	4.60E-02	1.06E-04	1.99E-01	1.09E-01	1.20E-01	6.72E-04	6.98E-04	6.99E-04	1.43E-03
Iron	2.90E+03	kg	7.70E-03	3.16E-03	4.68E-06	1.44E-02	1.42E-02	3.17E-04	4.98E-05	5.62E-05	4.73E-05	7.65E-05
Rubber	1.59E+03	kg	4.21E-03	8.01E-03	3.75E-06	1.15E-02	4.09E-03	6.44E-03	2.35E-05	5.47E-05	5.18E-05	1.86E-04
Plastic	1.53E+03	kg	4.07E-03	7.87E-03	1.72E-06	9.04E-03	1.67E-03	2.97E-04	1.19E-05	3.17E-05	2.97E-05	7.86E-05
aluminum	9.76E+02	kg	2.59E-03	9.46E-03	1.11E-05	3.35E-02	2.11E-02	7.48E-02	7.22E-05	1.13E-04	1.85E-04	1.22E-03
Diesel Consumption[h]												
Diesel fuel Consumption	5.19E+03	kg	1.38E-01	1.68E-01	5.94E-06	6.46E-02	1.17E-02	1.84E-03	1.85E-04	4.79E-04	5.91E-04	2.25E-04
Direct Diesel emissions	5.19E+03	kg	1.38E-01			4.32E-01			8.89E-03	5.69E-03	4.30E-02	

a: Calculated by accounting for 16,512 m^3 leachate fr diluted in wwater plant. Details: 55000 m^3 leachate/ month * 12 months = 660000 m^3/ year; Leachate CEAGESP = 37652 tons org CEAGESP : 1505000 t org Caieiras = x : 660,000 m^3/yr; X= 37,652 t org CEAGESP*660,000 m^3 * yr^{-1}/1,505,000 t org CAIER = 16,512 m^3 leachate CEAGESP/yr. Percentage leachate CEAGESP /Leach total = 16,512m^3/yr : 660,000 m^3/yr = x : 100; Percentage CEAGESP leach x = (16512 m^3/yr*100)/660,000 m^3/yr = 2,5 %. Allocation details leachate CEAGESP fr in wwater plant by using BOD (see table 7.1)

Table B 7.1 Allocation leachate CEAGESP in Wwat SABESP using Mass

	Val. Leachate	Val. Wastewater	Val. Total	Unit
BOD	5.00E+02	2.47E+02	7.47E+02	mg/L
Flow	1.65E+07	3.78E+11	3.78E+11	L/yr
BOD*Flow	8.26E+09	9.36E+13	9.36E+13	mg/yr

BOD Wastewater input: 247,4 mg/l (Source Da Silva et al, 2005); BOD leach CAIEIRAS: 500 mg/l (Fieldwork); (BOD*flow leach) : (BOD*flow tot) = x : 100; 8.26E+09 mg/yr : 9.36E+13 mg/yr = x :100 ; x = 8.26E+09 mg/yr * 100 / 9.36E+13 mg/yr; Perc. Mass. CEAGESP 0.009%. For COD calculation, same process of BOD, but were considered in the Table above 27500 mg/L for leachate and 473.2 mg/L for wastewater, with a result of 0.25% of Leachate Mass CEAGESP in Baruerì. COD value was used for sludge allocation.

b: Sabesp: electricity/ m^3 wastewater treated: 0,45 kWh / m^3 (Source SABESP, 2019); Annual wwat flow Barueri = 12 m/s*60sec *60min. *24 hrs *365days = 378,432,000 m^3 / yr; Annual Wastewater flow tot (Leachate + wwat) = 16512000 l/yr + 378432000000 L/yr = 378448512000 L/yr; Percentage of organic mass CEAGESP in SABESP = 0,009%; Consumption electricity CEAGESP fract in Barueri: 0,45 kWh / m^3 * 3,78E+08 m^3/yr = 1.70E+08 kWh/yr = consumption to treat 9,36E+13 mg/yr mass BOD; Mass CEAGESP BOD fract = 0,009 % total mass; kWhconsumption CEAGESP fraction = 1,70E+08 kwh/yr*0,009% = 15016 kWh / yr

c: Daily sludge production 2007: 228 t/d with flow 9.5 m^3 /s (Sigolo et al., 2009); Flow 2018 = 12 m^3 / s; Daily flow 2007= 9500 l/s*60 sec* 60 min*24 hr = 820,800,000 L/d; Daily flow 2018 = 12000L/s*60 sec* 60 min*24 hr = 1,036,800,000; Estm. Daily Production 2018 with proportion (228 t / d : 820,800,000 l / d 2007) = (x : 1,036,800,000 l/d 2018); x = (228 t / d * 1,036,800,000 l / d)/ (672,192,000 l / d) = 288 t/d; Annual sludge production barueri = 288 t/d * 365 d = 105120 t/y (2018); Annual sludge production Barueri CEAGESP fraction (COD allocation): 105120 t/y * 0.25% = 263 tons = 263000 kg

d: Treatment with $FeCl_3$ and synthetic cationic polimer of acrylamide (source Sigolo et al, 2009; Miki, 1998); 63 kg $FeCl_3$ / sludge ton * 263 t CEAGESP fr / yr = 16,569 kg/ yr $FeCl_3$ used.

e: (Source Sigolo et al, 2009; Miki, 1998); 4,83 kg synthetic cationic polymer / sludge ton * 263 t CEAGESP fr = 1270 kg synthetic cationic polymer used.

f: For Cement, Gravel and Steel values in kg/m^3/yr respectively of 1.99 E-02, 2.22E-01 and 1.03E-02 estimated from Zhand and Ma, 2020 and multiplied by 33,354 m^3eq CEAGESP leachate fraction in wwater plant (0.009% BOD).

g: Sum of 1 Tank truck capacity 30 m^3 (12800 kg weight) for leachate transport + 1 Transport Truck of 14550 kg for sludge transport.

h: Leachate transport Caieiras Landfill –to Wastewater Baruerì plant: 60 trips / day x 365 days - year= 21900 trips year; Truck capacity: 660000 m^3 / 21900 = 30,14 m3 -> 30 m^3; 16,512 : 30 = 550 trips; Dist. Caieiras - Attend Ambiental Barueri: 39,4 km; Consumption 1 Trip Leachate Transport: 0,28 L/ km *39,4 km*2 (round trip) = 11,03 L/ trip; Total consumption CEAGESP leachate Transport:11.03L /trip * 550 trip/yr = 6,067 L/ yr; Annual diesel consumption in kg = 6067 L/yr * 0,8425 kg/L = 5111 kg/yr; - Sludge transport: Dist. Caieiras - Attend Ambiental Barueri: 39,4 km; Truck Capacity 30 t; Trips Number sludge CEAGESP fraction = 263 t / 30 t/ trip = 9 trips; Consumption 1 Trip Sludge Transport: 0,28 L/ km *39,4 km*2 = 11,03 L/ trip; Diesel consum. Sludge transp. to Caieiras = 11,03 L/trip * 9 trips/ yr = 99 L/yr; Diesel consumption Sludge Transp. In kg = 99 L/yr*0,8425 kg/L = 83 kg/ yr. Total = 5111 kg/yr + 83 kg/yr = 5194 kg / yr.

Table B8: Impacts Scenario #II: electricity production

Description	Material weight	Unit	Annual input raw material	Fossil Depletion FDP (in Kg oil eq)	Freshwater Eutrophication FEP (in kg Peq/kg)	Global Warming GWP 100 (Kg CO_2 eq/kg)	Human Toxicity HTP Inf (Kg 1,4-DCB eq/kg)	Metal Depletion MDP (in kg Fe eq /kg)	ParticulaterMatter formation PMFP (in kg PM_{10} eq/ kg)	PhotochemicalOxidant Formation POFP (in kgNMVOC-eq)	Terrestrial Acidification TAP100 in kg SO_2 eq	Water depletion WDP (in m^3 H_2O eq/kg)
Impacts Scenario #I				8.31E+00	6.86E-03	2.03E+02	2.30E+00	1.87E+00	3.59E-01	3.60E-01	1.68E+00	7.64E-02
Concrete[a]	1.06E+00	m^3	2.81E-05	1.06E-03	5.49E-07	5.68E-03	1.08E-03	1.88E-03	1.21E-05	3.16E-05	2.68E-05	4.08E-05
Steel[b]	2.79E+02	kg	7.42E-03	7.81E-03	1.80E-05	3.38E-02	1.85E-02	2.03E-02	1.14E-04	1.18E-04	1.19E-04	2.42E-04
Water[c]	2.28E+02	m^3	6.06E-03									6.06E-03
Lubricant Oil[d]	2.53E+03	kg	6.71E-02	1.01E-01	2.67E-05	9.14E-02	3.20E-02	1.21E-02	2.18E-04	1.83E-03	5.29E-04	6.52E-04
Biogas combustion NOxdirect Emissions[e]	8.76E+03	kg	2.33E-01						5.12E-02	2.33E-01	1.30E-01	
Sum Electricity production impacts				1.10E-01	4.52E-05	1.31E-01	5.16E-02	3.44E-02	5.15E-02	2.35E-01	1.31E-01	6.99E-03
TOT. IMPACTS PER F.U.ELECTR. PROD.				**8.42E+00**	**6.91E-03**	**2.03E+02**	**2.36E+00**	**1.91E+00**	**4.11E-01**	**5.94E-01**	**1.81E+00**	**8.34E-02**
Gross avoided emissionsCEAGESP FR. ElectricityProduction	5.75E+06	kWh	1.53E+02	8.40E+00	4.38E-03	3.00E+01	4.75E+00	5.84E-01	6.78E-02	9.84E-02	1.83E-01	3.17E+00
Emissions Baruerì plant	1.50E+04	kWh	3.99E-01	2.19E-02	1.14E-05	7.83E-02	1.24E-02	1.53E-03	1.77E-04	2.57E-04	4.77E-04	8.28E-03
Net avoided emissions CEAGESP FR. Electricitypr.		**kWh**	**1.52E+02**	**8.38E+00**	**4.37E-03**	**2.99E+01**	**4.74E+00**	**5.83E-01**	**6.77E-02**	**9.81E-02**	**1.82E-01**	**3.16E+00**
TOT. IMPACTS LESS AVOIDED ELECTR. EMISSION (SCENARIO 2)				**3.46E-02**	**2.54E-03**	**1.73E+02**	**-2.39E+00**	**1.32E+00**	**3.43E-01**	**4.96E-01**	**1.63E+00**	**-3.08E+00**

a: Average Yearly Production Electricity São João landfill: 4.90E+09 kWh (from Da Silva, 2011); Average Yearly Concrete use : 2.25E+09 g/yr (from Almeida et al, 2012)*10-3 kg/g = 2.25E+06 kg/yr; Calc. concrete per kWh electricity São João: 2.25E+06 kg/yr : 4.90E+09 kWh/yr; Concrete per kWh electricity generated = 4.59E-04 kg/kWh; Electricity generated CEAGESP fr in CAIEIRAS : 5.75E+06 kWh/yr; Estimation concrete use in one year in CAIEIRAS electr. Prod. CEAGESP fr. in kg = 5.75E+06 kWh/yr * 4.59E-04 kg/kWh; Concrete used CEAGESP fr. CAIEIRAS electricity production in kg = 2.64E+03 kg/yr; Average Concrete Density used in São João landfill (from Da Silva, 2011) = 2500 kg/m^3 . Concrete used CEAGESP fr CAIEIRAS electricity production in m^3 = 1.06 m^3 / yr

b: Average yearly steel use: 2.38E+08 g/yr (from Almeida et al., 2012) * 10^{-3} kg/g = 2.38E+05 kg/yr; Calc. steel per kWh electricity São João: 2.38E+05 kg/yr : 4.90E+09 kWh/yr; Steel per kWhelectricity generated: 4.86E-05 kg/kWh; Electricity generated CEAGESP fr in CAIEIRAS : 5.75E+06 kWh/yr; Estimation steel use in one year in CAIEIRAS electr. Product. CEAGESP fr. = 5.75E+06 kWh/yr* 4.86E-05 kg/kWh; Steel use CEAGESP fr. CAEIRAS electr. Prod. = 2.79E+02 kg/yr

c: Annual water used at Termoverde 25 m^3/ day (from Zanotti, 2014. Source https://research.gsd.harvard.edu/zofnass/files/2016/08/12_TermoverdeCaieiras_EN_Final-version.pdf) * 365 days/yr = 9125 m3/yr; Water consumed by CEAGESP fr. at Termoverde = 9125 m^3/yr * 2.5% = 228 m^3/yr

d: Annual Lubricating oil used by TERMOVERDE = 101 t/yr (Zanotti, 2014); Lubricating oil consumed by CEAGESP fr. at Termoverde = 101 t/yr * 2.5% = 2.525 t/yr = 2525 kg/yr;

e: NOx annual emissions at Termoverde = 350.4 t/yr (Zanotti, 2014); NO_X emitted by electricity generation of Biogas CEAGESP fr. In CAIEIRAS = 350.4 t/yr * 2.5% = 8.76 t/yr = 8760 kg/

Table B9: Ecoinvent table for Scenarios #I and #II: Data source: ecoinvent database (https://www.ecoinvent.org/login-databases.html), Version 3.6 (2019); Allocation at the point of substitution; Recipe Midpoint (H) V1.13;

Item	Ref. Weight	Fossil Depletion FDP (in Kg oil eq)	Freshwater Eutrophication FEP (in kg P eq/kg)	Global Warming GWP 100 (Kg CO2 eq/kg)	Human Toxicity HTP Inf (Kg 1,4-DCB eq/kg)	Metal Depletion MDP (in kg Fe eq /kg)	Particular Matter formation PMFP in kg PM_{10} eq/ kg	Photochemical Oxidant Formation POFP in kg NMVOC-eq/ kg	Terrestrial Acidification TAP100 in kg SO_2 eq	Water depletion WDP (in m3 H_2O eq/kg)
Aluminum Liquid Pr.	1 kg	2.88E+00	2.07E-03	1.02E+01	3.31E+00	2.39E-01	1.86E-02	3.01E-02	5.13E-02	3.84E-01
Aluminum alloy slab pr.	1 kg	7.67E-01	2.20E-03	2.78E+00	4.82E+00	2.86E+01	9.28E-03	1.34E-02	2.00E-02	8.66E-02
Aluminum Total[a]	1 kg	3.65E+00	4.27E-03	1.29E+01	8.13E+00	2.88E+01	2.79E-02	4.34E-02	7.13E-02	4.71E-01
Cement	1 kg	8.68E-02	2.25E-05	6.98E-01	5.16E-02	5.60E-02	9.85E-04	2.52E-03	2.28E-03	3.21E-03
Concrete	1 m^3	3.78E+01	1.95E-02	2.02E+02	3.84E+01	6.67E+01	4.30E-01	1.13E+00	9.52E-01	1.45E+00
Diesel	1 kg	1.22E+00	4.31E-05	4.68E-01	8.50E-02	1.33E-02	1.34E-03	3.47E-03	4.28E-03	1.63E-03
Electricity	1 kWh	5.50E-02	2.87E-05	1.96E-01	3.11E-02	3.83E-03	4.44E-04	6.44E-04	1.20E-03	2.08E-02
Ferric Chloride	1 kg	1.25E-01	3.56E-04	4.27E-01	4.43E-01	1.20E-01	1.21E-03	1.53E-03	2.50E-03	8.45E-03
Geosint. Clay Liner (GCL)	1 kg	1.48E-02	8.92E-06	4.70E-02	1.34E-02	6.27E-03	1.40E-04	3.64E-04	3.16E-04	4.26E-04
Geotextile	1 kg	2.14E+00	8.10E-04	2.85E+00	7.13E-01	1.12E-01	4.71E-03	1.05E-02	1.04E-02	2.64E-02
Gravel	1 kg	1.40E-03	5.96E-07	5.15E-03	8.77E-04	4.00E-04	5.41E-05	1.89E-04	1.55E-04	1.03E-04
HDPE	1 kg	1.89E+00	4.61E-04	2.25E+00	4.33E-01	7.19E-02	3.09E-03	8.03E-03	7.38E-03	2.21E-02
Iron	1 kg	4.10E-01	6.08E-04	1.87E+00	1.84E+00	4.12E-02	6.46E-03	7.30E-03	6.14E-03	9.94E-03
Lubricating Oil	1 kg	1.51E+00	3.98E-04	1.36E+00	4.77E-01	1.81E-01	3.25E-03	2.73E-02	7.89E-03	9.72E-03
Plastic	1 kg	1.94E+00	4.22E-04	2.22E+00	4.11E-01	7.30E-02	2.93E-03	7.80E-03	7.31E-03	1.93E-02
Polyacrylamide	1 kg	1.68E+00	5.28E-04	2.87E+00	6.48E-01	1.48E-01	4.58E-03	7.59E-03	1.54E-02	3.65E-02
Rubber	1 kg	1.90E+00	8.90E-04	2.73E+00	9.71E-01	1.53E+00	5.58E-03	1.30E-02	1.23E-02	4.41E-02
Primary Steel Production	1 kg	4.67E-01	1.59E-03	2.42E+00	1.59E+00	2.36E+00	1.01E-02	9.91E-03	8.69E-03	1.64E-02
Steel Metal Working	1 kg	5.86E-01	8.43E-04	2.13E+00	9.03E-01	3.81E-01	5.28E-03	6.06E-03	7.33E-03	1.63E-02
Steel Total[a]	1 kg	1.05E+00	2.43E-03	4.55E+00	2.49E+00	2.74E+00	1.54E-02	1.60E-02	1.60E-02	3.27E-02

a: for aluminum and steel it was considered the total value.

TABLE B10: Processes details of materials shown in table B9

Item	Ref. Weight	Process Name	Product
Aluminum Liquid Pr.	1 kg	Aluminum production, primary, liquid, prebake, IAI Area, South America, (101)	Aluminum, primary, liquid (kg)
Aluminum alloy slab pr.	1 kg	Aluminum production, primary, cast alloy slab from continuous casting, RoW (82)	Aluminum, primary, cast alloy slab from continuous casting
Aluminum Total	1 kg	Aluminum Production, total process (liquid production + cast alloy slab production)	Aluminum, primary, bars, (kg)
Cement	1 kg	Cement, all types to generic market for cement, unspecified, BR (139)	Cement, 1 kg
Concrete	1 m^3	Concrete, all types to generic market for concrete, normal strength, BR (104)	Concrete (m^3)
Diesel	1 kg	Market for diesel, BR, (69)	Diesel [kg]
Electricity	1 kWh	Electricity, high voltage, production mix, BR (2213)	Electricity, high voltage 1 kWh
Ferric Chloride	1 kg	iron (III) chloride production, without water, in 14% solution state, RoW (4)	Chlorite Ferric, 1 kg
Geosint. Clay Liner(GCL)	1 kg	Market for bentonite, GLO (8)	Bentonite (kg)
Geotextile**	1 kg	Market for textile, nonwoven polypropylene, GLO (2)	textile, non-woven polypropylene [kg]
Gravel	1 kg	Gravel Production, crushed, BR (5)	Gravel, crushed
HDPE	1 kg	Polyethylene production, high density, granulate, RoW (2)	Polyethylene, high density, granulate (kg)
Iron	1 kg	Cast iron production, RoW (34)	Cast Iron (kg)
Lubricating Oil	1 kg	Market for Lubricant Oil, RoW	Lubricant Oil, 1 kg.
Plastic	1 kg	Market for polypropylene, granulate, GLO (1)	Polypropylene, granulate
Polyacrylamide	1 kg	Market for polyacrylamide, GLO, (1)	Polyacrylamide, (kg)
Rubber	1 kg	Market for synthetic rubber, GLO (16)	Synthetic rubber [kg]
Primary Steel Prod.	1 kg	Steel production, converter, low-alloyed, RoW (320)	steel, low-alloyed (kg)
Steel Metal Working	1 kg	Metal working, average for steel product manufacturing, RoW (271)	metal working, average for steel product manufacturing [kg]
Steel Total	1 kg	Steel production and working, total process (Production + metal working)	Steel, bars, 1 kg

Table B11: Donation Scenarios (from #III to #VI) – All Steps - Annual input raw materials per 1-ton OBP.

STEP 1: FOOD BANK COLLECTION SYSTEM[a]	Material weight	Lifespan		Annual Input Raw Material	FDP (in Kg oil eq)	FEP (in kg P eq/kg)	GWP 100 (Kg CO_2 eq/kg)	HTP Inf (Kg 1,4-DCB eq/kg)	MDP (in kg Fe eq /kg)	PMFP (in kg PM_{10} eq/ kg)	POFP (in kg NMVOC-eq)	TAP100 in kg SO_2 eq	WDP (in m^3 H_2O eq/kg)
Steel	17985	8	kg	5.97E-02	6.29E-02	1.45E-04	2.72E-01	1.49E-01	1.64E-01	9.18E-04	9.54E-04	9.56E-04	1.95E-03
Lead	1680	4	kg	1.12E-02	8.03E-03	2.22E-05	2.92E-02	1.22E-01	4.06E-01	1.14E-04	1.23E-04	2.48E-04	3.70E-04
Wooden pallets	4500	10	unit	1.20E-02	3.84E-02	2.91E-05	9.09E-02	3.20E-02	8.16E-03	2.92E-04	6.82E-04	4.46E-04	8.73E-04
Electricity consumption	21693		kWh	5.76E-01	3.17E-02	1.65E-05	1.13E-01	1.79E-02	2.20E-03	2.56E-04	3.71E-04	6.90E-04	1.20E-02
STEP 2: CHECK QUALITY[b]													
Structure steel	21150	45	kg	1.25E-02	1.31E-02	3.03E-05	5.68E-02	3.11E-02	3.42E-02	1.92E-04	1.99E-04	2.00E-04	4.08E-04
Steel Roof tiles	3600	45	kg	2.12E-03	2.24E-03	5.16E-06	9.67E-03	5.29E-03	5.83E-03	3.27E-05	3.39E-05	3.40E-05	6.94E-05
Steel tables	3478		kg	8.40E-03	8.84E-03	2.04E-05	3.82E-02	2.09E-02	2.30E-02	1.29E-04	1.34E-04	1.34E-04	2.74E-04
STEP 3: STORAGE[c]													
Wall panels Steel	5733	10	kg	1.52E-02	1.60E-02	3.70E-05	6.93E-02	3.79E-02	4.18E-02	2.34E-04	2.43E-04	2.44E-04	4.97E-04
Polystyrene	659	10	kg	1.75E-03	4.12E-03	1.11E-06	7.77E-03	9.89E-04	2.04E-04	1.04E-05	2.71E-05	2.56E-05	1.37E-04
Plastic pallets	1289	15	kg	2.28E-03	4.42E-03	9.63E-07	5.07E-03	9.37E-04	1.67E-04	6.68E-06	1.78E-05	1.67E-05	4.41E-05
Electricity consumption	37812		kWh	1.00E+00	5.53E-02	2.88E-05	1.97E-01	3.13E-02	3.84E-03	4.46E-04	6.47E-04	1.20E-03	2.08E-02

a: **STEP 1 FOOD BANK COLLECTION SYSTEM: Collection system composed by 3 three logistic trains, each one constituted by 1 Tow Tractor LTX 80 + trailer of 3 C - type frame + 6 Trolleys + 6 Pallets. Details Mizusumashi system implementation**: **Electric tow Tractor LTX 80 (with cab):** Max speed: 20 km/h without load; 10 km/h with load; Max towing capacity 8 tons. Kerb weight 1515 kg; Battery (Lead - Acid Type model DIN 43531 12 - 1998) weight 560 kg; Chassis weight (steel) = kerb weight - Battery weight = 1515 kg - 560 kg = 955 kg; Energy consumption according to VDI cycle 2012: 4.98 kW / h; **Tugger train C - Type frame:** Net weight: 960 kg; Load Capacity 1600 kg; Max speed: 15 km/h; It can transport two trolleys of 1200 mm x 800 mm x 280 mm; **Pallet trolley:** Size: 1200 mm x 800 mm x 280 mm; weight: 36 kg; Max speed variable, in this work 15 km/h, the same of tugger train C - Type frame; Wooden pallets: 25 kg weight; 1200 mm X800 mm; load capacity 1 t; 1 euro pallet can hold four standard 600 X 400 perforated containers or 6 standard 400 x 300 containers per level; Total pallets: 180 considering an average maximum distance of 100 m among the waste collection points; Each pallet is hold by one trolley: total 180 trolleys; Each C - Type frame can load two euro pallets until 1600 kg (2 x 800 kg); Each Tow Train can tow up to 8 tons. **Standard configuration T - train in this system: 1 - T train LTX 80 + trailer (3 C - type frame + 6 Trolleys + 6 Pallets):** Weight of standard configuration trailer without load: (960 kg x 3) + (36 kg x 6) + (25 kg x 6) = 3246 kg; T-train max products load capacity = 8000 kg - 3246 = 4754 kg; Max products load per frame (2 pallets) = tot capacity -2 x (empty pallet weight + empty trolley weight); Max products load per frame (2 pallets) = 1600 kg - 2 x (25 kg + 36 kg) = 1600 kg - 122 kg = 1478 kg; Max product load per pallet = 1478 kg / 2 = 739 kg; Real max load capacity (net capacity)= 739 kg x 6 = 4434 kg; Daily CEAGESP by-products generation 47065 t * yr^{-1} / 363 d * yr-1 (excl. Christ. and new year) = 130 t/ day * 90% (10% discarded before transport = 117 t/ day); Maximum theoretical load of loaded pallets = Daily Waste CEAGESP 117 t / 180 pallets = 0.65 tons = 650 kg; Real average load per pallet = 650 / 2 = 325 kg (There are two collection shifts); Real average load per T - train = 325 kg x 6 = 1950 kg = 1.95 tons; Average Realistic T train speed 7.5 km/h = 2.08 m/s; Maximum path length between Food Bank and farther CEAGESP box = 3000 m; Average path length between Food Bank and farther CEAGESP box = 1500 m; Average time collection per T - Train (1 return trip) = 1500 m : 2.08 m/s = 721 sec (12 min. and 1 sec)= ~ 12 min.; Average total load per shift (2 daily shifts) = 325 kg * 180 pallets = 58500 kg (58.5 tons); 58500 kg products per shift / 1950 kg per Train = 30 trips of 1950 kg load each one. 30 trips * 12 min average trip duration = 360 min per shift with one train (720 min = 12 hrs per day); Considering two trains : 360 min / 2 = 180 min per shift (360 min = 6 hrs per day); **Considering three trains: 360 min/ 3 = 120 min per shift (240 min = 4 hrs per day)*363 = 1,452 hrs/yr.**

Material summary in Table B 11.1

Table B 11.1

Mizusumashi Table Considering Three Tow Tractors 3 x (1 Tow Tractor LTX 80 + trailer (3 C - type frame + 6 Trolleys + 6 Pallets))						
Item	**Quantity**	**Weight per unit**	**Unit**	**Sum**	**Lifespan[6]**	**Total**
Still electric Tow Tractor LTX80[1]	3	955	kg	2865	8	358
Tow tractor Battery (Pb)[1]	3	560	kg	1680	4	420
C Type frame[2]	9	960	kg	8640	8	1080
Pallet Trolley[3]	180	36	kg	6480	8	810
Wooden pallet[4]	180	25	kg	4500	10	450
Electricity Consumption[5]	21693		kWh	21693		21693

Notes tab B 11.1

1. Source https://www.still.com.br/ltx-70-manual-br.0.0.html;

2. Source https://www.still.com.br/tugger-train-c-manual-br.0.0.html

3. Source https://www.still.com.br/21780.0.0.html

4. Source European pallet association.https://www.epal-pallets.org/fileadmin/user_upload/ntg_package/images/Produktdownloads/Produktdatenbla__tter/GB/EPAL1_Produktdatenblatt_GB.pdf; For Brasilhttps://www.chep.com/pt/pt-pt/consumer-goods/product/wooden-pallet-1200-x-800-mm-03. Wooden pallet lifespan was assumed 10 years, according to Deviatkin et al., (2019).

5. (4.98 kW/h according to VDI cycle 2012 source: https://www.still.com.br/ltx-70-manual-br.0.0.html)*(4 hrs/ day)*(3 tractors)*(363 days/yr Exl. Christmas an 1st of year)

6. For tow tractors and trailer system, due to lack of data, we have considered the average lifespan of forklifts expressed in hrs that work less than 2000 hrs per year with a heavy application that correspond to ~ 8 years in our study considering ~1452 operative hrs per year, according to information from Adaptalift Group: https://www.adaptalift.com.au/blog/2012-10-24-the- optimal-time-to-replace-your-forklift; For battery lifespan it was chosen an average value of 4 years. by considering the minimum value from of the range provided by Powerthru (http://power-thru.com/documents/The%20Truth%20About%20Batteries%20-%20POWERTHRU%20White%20Paper.pdf) diminished by one year due to usual Brazilian High working temperature (Tropical Climate).

Annual material inputs logistic train is summarized in table B10.

Table B11.2 Mizusumashi logistic train material table (1 Year)

Input	Quantity	Unit	Annual Input[b]
Steel[I]	17985	kg	2248
Lead[II]	1680	kg	420
Pallet[III]	4500	kg	450
Electricity	21693	kWh	21693

I: Sum of Tow Tractors + C-Type frames + Pallet trolleys, annual input considered Lifespan of 8 yearsII: considering Lifespan battery of 4 years

III: wooden pallet lifespan 10 years

b: STEP 2 CHECK QUALITY: Shed structure estimation (from GERDAU, 2012); 900 m^2; 30m x 30m (5 modules, 6 m distance. Height 6 m). Steel structure 23.5 kg/m^2 x 900 m^2 = 21150 kg; SteelRoof Tiles = 4 kg/m^2 x 900 m^2 = 3600 kg/m^2 ; Lifespan Steel 45 years (Buranakarn, 1998**).** Checking quality tables: Height 90 cm, length 160 cm, width 70 cm; weight 32,2 kg; Capacity 300 kg; 1 table of 1.6m x 0.7m = 1.12 m^2 can load 300 kg; Considering 32500 kg (1/2 shift average load) are necessary: equation -> 1 table : 300 kg = x : 32500 kg; x = 1 * 32500 kg / 300kg = 108 (121 m^2);Total steel used = 32.2kg * 108 tables = 3478 kg; Steel Table average lifespan 11 year; Yearly steel table inputs = 3478 kg/11 years = 316 kg / year.

c: STEP 3 STORAGE: cold room size calculation -> Waste CEAGESP 47065 t/yr (2018); Daily flux (stock room time 24 hrs, from Fagundes et al, 2014); 47065 t * yr^{-1} / 363 d * yr^{-1} = 130 t / d; 90% is collected by the Train, 10% directly discarded (130 t/d : 100) * 90 = 117 t/d; Food bank rejected another 10 % (from CEAGESP REPORTS, Fagundes et al, (2014) and tech. visit); Daily products donated (130/100)*80 = 104 tons/ day (packaging included); 83.2 tons/ day net amount donated (80%); Cold room Tectermica model CFP 66 RF version (+ 1ºC < + 10ºC), capacity 20000 kg; Cold room size (Volume): width 4.60 m x 5.75 m depth (length) x 2.50 m height = 66.125 m^3 ; Average food stock per m^3 = 20000 kg/66.125 = 302 kg/ m^3 (within the range 250 - 500 kg/m3 of Evans et al., (2014); Cold room floor surface = 4.60 m x 5.75 m = 26.45 m^2 ; **Estimation electricity consumption one cold room:** It was considered the max value within the range found by Evans, 2014 (excluded 10% extreme values), due to tropical climate, high differences between food loaded temperature and store temperature, daily turnover 100%; 95.3kWh / m^3 * yr^{-1} (from Evans et al., 2014)*66.125 m^3 = 6.302 kWh /yr per cold room (20 tons capacity); Daily products quantity stocked = 104 tons, therefore are necessary 6 cold rooms of 20 tons capacity each one; Average consumption cold room= 6302 kWh/yr * 6 cold rooms = 37812 kWh/yr; **Materials cold room calculation:** Cold room wall tickness 75 mm, filled up with polystyrene 14 kg/m^3; Floor and roof surface = 26.45 m^2*2 = 52.90 m^2; wall surfaces = 2*(width 4.60m x height 2.50 m) + 2*(length 5.75m x height 2.50 m) = 23 m^2 + 28.75 m^2 = 51.75 m^2 ; Total surface (floor + roof + walls) = 52.90 m^2 + 51.75 m^2 = 104.65 m^2 ; Weight m^2 Steel panel with 75 mm tickness filled up with polystyrene = (weight panel 100 mm + weight panel 50 mm) / 2 = (9.82 kg/m^2+10.54kg/m^2) / 2 = 10.18 kg/m^2 ; Tickness volume calculation per m^2 = 1 m x 1m x 0.075 m = 0.075 m^3 ; For each m^2 of steel are used 0.075 m^3 of Polystyrene; Weight estimation of polystyrene used per m^2 = 14 kg/m^3 * 0.075 m^3/m^2 = 1.05 kg/m^2 ; Total polystyrene used per cold room = 1.05 kg/m^2 * 104.65 m^2 = 109.8825 kg/ room; Weight estimation of Steel used per m^2 = Total weight m^2 - Polystyrene weight m^2 = 10.18 kg/m^2 - 1.05 kg/ m^2 = 9.13 kg/m^2 ; Steel used for one cold room = 9.13 kg/m^2 * 104.65 m^2 = 955.4545 kg; Total polystyrene used considering 6 cold rooms = 109.8825 *6 = 659 kg; Total Steel used considering 6 cold rooms = 955.4545 * 6 = 5733 kg; Average cold room lifespan is supposed to be 10 years, from Cascini et al., (2015) Steel input 573.3 kg - polystyrene input 65.9 kg. **Plastic pallet 15 x 100 x 120**; Plastic pallet weight 17.9 kg; 3000 kg capacity; 1.2 m^2 area; Considering 12 pallets in each cold room there are 14.4 m^2 used by pallets and 12.05 m^2 free; Total plastic pallets = 72; Each pallet has a max. average load of 117000 kg / 72 = 1625 kg; Total plastic pallet weight = 17.9 x 72 =1288.8; Plastic pallets lifespan ~ 15 years (Deviatkin et al, 2019); yearly input = 1288.8 / 15 = 85.92 kg.

Section B12: Notes of Table 8 – Biorefinery process operative schedule per daily input

1. OBP collection and transport -> same as in Donation Scenario, see table B11, step 1.

2. Manual Separation: Information from Uratani et al. (2014): 16 hours / day to check 223 tons / day; Conveyor belt: length 20 m, power 7.29 kW, 16 working hours (two shifts of 8 hours each) to treat 223 ton/day; conveyor belt working hours calculation in this study: 16 hours/day : 223 tons/day = x : 130 ton/day; x = (16 hours/day * 130 tons/day) / 223 tons/ day = 9.5 hours/day; Outputs = 104 ton OBP, 26 ton residual inorganic fraction (20%); conveyor belt electricity consumption calculation: 7.29 kW * 9.5 hours / day = 69.26 kWh / day * 363 working days / year = 25,141 kWh; Weight: Information from Gruppomini (2022), Indiamart (2022a) and Mirbelting (2022): conveyor belt has approximately a weight of 25 kg per meter, therefore the weight for 20 m was estimated as 500 kg. It was assumed a conveyor belt made by steel and a lifespan of 5 years.

3. Grinding: 960 kWh / day : 212 ton / day (Uratani et al., 2014) = 4.53 kWh/ ton * 104 ton / day = 471 kWh / day * 363 Wdays / year = 170,973 kWh / year. By assuming a max capacity of 7 ton / hour and a power of 30 kW (40.8CV) of each machine (Uratani et al., 2014), two grinding machines were considered for normal use plus one machine for emergency use in case of maintenance, for a total of three machines. The step duration is 104 ton / 14-ton hr^{-1} = about 7.5 hr. It was considered an equipment formed by three grinders Triturtec (Triturtec, 2022) made by steel with an average weight around 4 t each one, for a total weight of 12 tons. The lifespan was assumed to be the same of the conveyor belt (5 years).

4. Anaerobic Digestion: Daily Input dry mass calculation: Dry matter = 11.39 g dry matter / 100 g OBP (from TBCA, 2020) -> 113,900 g dry matter / ton OBP * 10^{-3} kg / g = 113.9 kg dry matter/ ton OBP * 104 ton OBP / day = 11,846 kg dry matter / day; Moisture = 104,000 kg OBP/ day - 11,846 kg/ dry fraction / day = 92,154 kg water / day; Dry faction percentage calculation through proportion: 11,846 kg dry matter / day : 104,000 kg OBP/day = x : 100; x = 11,846 kg dry matter * 100 / 104,000 kg OBP day = 11.39%; Moisture = 100% - 11.39% = 88.61%; The wet digester was modelled according to Francini et al. (2020) and operates at 10% total solid (dry matter), therefore to achieve this quantity is necessary a dilution with an amount of water equal to 14,460 kg / day for a total daily input of 118,460 kg / day. Yearly water amount: 14,460 kg / wday * 363 wdays / year = 5,248,980 kg / year. For the AD process, the amount of thermal energy (th) needed for heating the diluted mixture of OBP (118,460 kg / day) from an assumed initial temperature of 20 ºC to 38ºC (from 293.15 K to 311.15 K) was calculated assuming the specific heat capacity of the feedstock to be the same as that of water. The calculation was executed according to the equation $\Delta E = c \times m \times \Delta t$ where ΔE is the energy needed for heating the feedstock mixture, c is the specific heat capacity of water (c_{water} = 4.18 kJ/kgºK), m is the mass of the mixture (kg) and Δt is the change of the temperature (from 293.15 K to 311.15 K). Daily energy to heat 118,460 kg mixture = 4.18 kJ/kgºK * 118,460 kg * 18ºK = 8,912,930 kJ / day * 2.78 * 10^{-4} kJ/ kWh = 2,478 kWh / day * 363 wdays / year = 899,514 kWh / year; Biodigester volume calculation - considered parameters: retention time (RT) of 20 days (Francini et al., 2020); density of the input equal 1 ton / m^3 as the density of the water (due to 90% moisture); Being RT = V / Q (from Karthikeyan and Visvanathan, 2013) where V is the volume of digester in m^3 an Q the daily flow in m^3, V = Q * RT = 118.460 m^3 / day * 20 day = 2,369.2 m^3 + 15% biogas buffer (Uratani et al., 2014) = 2,725 m^3 + 10% security buffer = 2,997.5 m^3 ~ 3,000 m^3; Biodigester dimensioning: by assuming a one stage vertical biodigester with a approximately cylindric shape, for a volume of about 3,000 m^3 it was considered as model a tank with 18.98 m diameter and 12 m heigh, made by stainless steel, with 87,906 kg net weight (Eurotankworks, 2022). Electricity consumption Biodigester: Stirring electricity consumption: 5.22 kWh / 100 m^3 / day active digester (Singh et al., 2019) * (~ 2400 m^3 active digester this study/100 m^3 by assuming a linear correlation) = 125.28 kWh/day. Regarding loading and unloading pumps electricity consumption all the values used by Ebner et al. (2014) modelling or 4.7 tons were multiplied by a (118.46 t this study / 4.7 t) = 25.2 factor for loading process and (110.78 t this study / 4.7 t) = 23.6 for unloading process, to obtain the electricity consumption of loading and unloading processes this study/ day by assuming a linear correlation. Pump loading tank consumption = 1.5 kWh * 25.2 = 37.2 kWh. Pump to centrifugation filter = 1.1 kWh * 23.6 = 25.96 kWh; Total anaerobic digestion daily electricity consumption = 125.28 kWh stirring + 37.2 kWh loading process + 25.96 kWh unloading process = 188.44 kWh/day. Yearly total electricity consumption: 188.44 kWh / day * 363 wday / year = 68,404 kWh / year. Biogas generation estimation: ratio Volatile Solids (VS) / Dry Matter CEAGESP OBPs = 0.906 (90.6 % from Culi, 2018); VS = 113.9 kg * 90.6% = 103.19 kg per t OBP. Volatile Solids

Daily Input biodigester (after dilution): VS daily input = Total Daily Input * Dry Matter Fraction * Volatile Solid fraction (Uratani et al., 2014) = 118,460 kg / day * (1 - Moisture) * (VS fraction) = 118,460 kg/day * (1 - 0.90) * (0.906) = 118,460 kg / day * 0.10 * 0.906 = 10,732 kg VS / day. By considering a average biogas composition of ~ 60% CH_4 and 40% CO_2; a concentration of 250 ppm of H_2S with negligible percentages of other components and a specific biogas production of 0.589 Nm^3 per kg / SV (Francini et al., 2020) the daily biogas generation was estimated as follow: Daily Biogas generation: Total daily VS input in kg * 0.589 Nm^3 / kg VS; Daily biogas generation: 10,732 kg VS / day * 0.589 Nm^3 / kg VS = 6,321 Nm^3 / day; early raw biogas generation: 6,321 Nm^3 / day * 363 day / year = 2,294,523 Nm^3 / year. Theoretical Daily Methane Generation: 10,732 kg VS/ day * 0.589 Nm^3 / kg VS * 60% CH^4 = 3,793 Nm^3 CH_4/ day. Theoretical Yearly Methane generation: 3,793 m^3 CH_4/ day * 363 operative days / year = 1,376,859 m^3 CH_4 / year. H_2S generation by considering a concentration of 250 ppm (Francini et al., 2020): Conversion ppm to mg / Nm^3 -> 250 ppm means 250 mol of H_2S in 10^6 mol gas; it is necessary to convert 250 mol to mg and 10^6 mol to Nm^3; Conversion 250 ppm to mg -> 250 mol * 34 g / mol H_2S * 10^3 mg / g = 8.50 * 10^6 mg; Conversion 10^6 mol gas to Nm^3 -> being the ideal gas equation pv= nRT, the volume V = nRT / p where p is the pressure equal to 101,325 Pa; n = 10^6 mol; R = 8.314 J / mol*K and T = 273 K; Therefore V = 10^6 mol * 8.314 J / mol*K 273 K/ 101,325 Pa = 22,400 m^3; Concentration in mg/ Nm3 = 8.50*10^6 mg / 22,400 Nm^3 = 379 mg / Nm^3. Daily H_2S generation: 379 mg/Nm3 * 6,321 Nm^3/ day = 2,395,659 mg / day * 10^{-6} kg / mg = 2.39 kg / day. Output Anaerobic digestion = CO_2 density: 1.963 kg / Nm^3; CH_4 density = 0.714 kg/Nm^3 in normal condition (0 C, 1 atm). Average density VS removed by considering a Biogas composition in percentage of ~ 60% CH_4 and 40 % CO_2 = ((0.714 kg / m^3 *60) + (1.963 kg / m^3 *40) / 100) = 1.214 kg/ m^3; by adding the small amount of H_2S equal to 0.0034 kg / m^3, the total VS removed per m^3 is 1.214 kg/ Nm^3 (CH_4 and CO_2) + 0.0034 kg /Nm^3 H_2S = 1.2174 kg / Nm^3. Desulfurization with microareation:$H_2S + 0.5O_2$ -> S + H_2O. Calculation of the amount of elemental sulfur generated: 1 mol H_2S + 0.5 mol O_2 -> 1 mol S + 1 mol H_2O. Molar mass: H_2S = 34 g/ mol; O_2 = 32 g/ mol; S = 32 g / mol; H_2O = 18 g/mol. Therefore: 34 g H_2S + (0.5 *32 g) O_2 -> 32 g S + 18 g H_2O; 34 g H_2S + 16 g O_2 -> 32 g S + 18 g H_2O. Being the mass of elemental S ~ 94% of H_2S, the daily deposit of elemental sulfur is estimated in 2.39 kg/ day * 0.94 = 2.25 kg / day. Regarding the amount of air is consumed by the desulfurization process, it was considered the minimum value as proposed by Jenicek et al. (2017), equal to about 1% of the row biogas flow (64 m^3 / day in our study). Calculation daily VS loss in kg: 6,321 Nm^3 biogas / day * 1.214 kg / m^3 = 7,674 kg VS / day. Percentage Removed VS on total VS = 7,674 kg VS loss / day : 10,732 kg VS / day = x : 100 -> x = (7,674 kg VS loss / day * 100)/10,732 kg / day = 71.50 %. Total dry matter per ton OBPs after anaerobic digestion: Total Dry matter before AD - Removed VS = 113.9 kg dry matter/ ton OBP - 73.79 kg VS loss / ton OBP = 40.11 kg / ton OBP; Daily Dry matter after AD = 11,846 kg dry fraction / day - 7,674 kg VS loss / day = 4,172 kg / day. Total daily digestate generation: 118,460 kg diluited input / day - 7,674 kg VS loss = 110,786 kg digestate by assuming 100% water transfer to digestate. Total yearly digestate generation: 110,786 kg / day * 363 wday / year = 40,215,318 kg / year. Residual moisture after AD by assuming 100% water transfer during the digestion process = Total input - VS loss during AD - Residual dry matter after AD = 118,460 kg / day - 7,674 kg VS / day - 4,172 kg / day = 106,614 kg water / day. Raw Digestate dry fraction percentage (proportion) = 4,172 kg dry matter: 110,786 kg raw digestate= x : 100; x = (4,172 kg dry matter * 100)/110,786 kg raw digestate = 3.77%. Fertilizers content per t OBP: information from Tampio et al. (2014). Being the dry matter fraction 25 % (250 kg / ton FW) in Tampio et al., (2014) and 11.39% (113.9 kg / ton OBP) in this work, the total quantity of Nitrogenous (N tot), phosphorus (P tot) and Potassium (K tot) was estimated according the following proportion: 113.9 kg : 250 kg = x : 100; x = (113,9 kg * 100) / 250 kg = 45.56 % (for more details see table B12).

5. Water Scrubbing: Hourly flow rate biogas by assuming constant flow: (6,321 m^3 / day - 1,356 m^3 / day CHP plant) = 4,965 m^3 / day : 24 hrs = 207 m^3/ hr. Electricity Consumption: 4,965 m^3 / day * 0.3 kWh /m^3 (SGC, 2013) = 1,490 kWh / day; Yearly electricity consumption 1,490 kWh / day * 363 wday / year = 540,870 kWh. Water consumption 2.5 m^3 day (SGC, 2013); monthly consumption = 2.5 m^3 * 30 days / month = 75 m^3 / month; yearly consumption = 2.5 m^3 * 363 wdays = 907.5 m^3 ~ 908 m^3 * 10^3 kg / m^3 = 908,000 kg. Raw Biogas composition was assumed 60% CH_4 and 40% CO_2. The estimated daily biogas generation sent to the water scrubbing plant is 4,965 Nm^3/ day, therefore the biogas is composed by 2,979 m^3 CH_4 (60%) and 1,986 Nm^3 CO_2 (40%). To achieve a final composition of upgraded biogas of 97% CH_4 and 3% CO_2 is necessary to remove 1,837 m_3 CO2, equal to the 37% of the raw biogas and 92.5% of the inicial CO_2 percentage. The final amount of biomethane (97%) generated and ready to be sold will be equal to 4,965 m^3 Biogas - 1,837 m^3 - Removed CO_2 = 3,128 m^3 / wday; Yearly biomethane generation (97%) is equal to 3,128 m^3/ wday * 363 wday / year = 1,135,464 Nm^3 / year. Water scrubbing plant dimensioning: according to Lorenzi et al. (2018) for a water scrubbing plant with an average flowrate of 561 Nm^3 are necessary 8,600 kg of steel. By considering in our case an average flowrate of 207 Nm^3 / hr and a max flowrate of 230 Nm^3 / hr (buffer ~ 10%), the amount of steel needed for the plant was calculated by assuming a linear correlation according to the following proportion = 8,600 kg : 561 Nm^3 = x : 230 Nm^3; x = 8,600 kg * 230 Nm^3 / 561 Nm^3 = ~ 3,526 kg. Plant lifespan was assumed to be 20 years (Lorenzi et al., 2018).

6**. Raw Digestate centrifugation:** this step divides the raw digestate in solid and liquid fraction. Electricity consumption solid - liquid separation: 3.5 kWh / ton raw digestate (from Tampio et al., 2014) = 110,785 kg raw digestate * 0.0035 kWh/ kg = 388 kWh / day; Yearly electricity consumption = 388 kWh / wday * 363 wday / year = 140,844 kWh / year; the solid-liquid partition coefficients were assumed to be the same of Tampio et al. (2014), as shown in table B12, B13 and B14. Solid and liquid fertilizers plastic containers: Liquid fertilizers Plastic container 50L capacity: Unit weight: 2.2 kg (Imperiodoplastico, 2022); Material HDPE; 2,000 daily trucks Lifespan 5 years, daily trucks circulation: 2,000; by assuming a turnover of 7 days and a deposit for 7 days the HDPE yearly total amount necessary is equal to 2.2 kg / drum * 2000 drums / day * 14 days (turnover + deposit) = 61,600 kg. Solid fertilizer plastic container, material HDPE: weight 0.50 kg per 10L bucket (Indiamart, 2022b), per 6 L it's assumed the 60% of 0.5 kg necessary. Therefore 0.50 * 60% = 0.3 kg. Lifespan 5 years. By considering 2000 trucks/ day and a turnover of 7 days and 7 days of deposit the total yearly amount is 0.3 kg / bucket * 2000 bucket / day * 14 days (turnover + deposit) = 8,400 kg.

7. Co – generation: CHP unit dimensioning -> Total daily Potential kWh in Biogas = LHW biogas * daily biogas generation-> 6 kWh / m^3 (SGC, 2012) * 6,321 Nm3 / day = 37,926 kWh; CHP characteristics: electricity efficiency 0.35 (Probiogas, 2015); thermal efficiency 0.48 (Probiogas, 2015). Total daily potential electricity = 37,926 kWh * 0.36 engine efficiency = 13,653 kWh; Total daily electricity consumption biorefinery = 2,666 kWh; 10% buffer = (2,666 kWh / 100) * 110 = 2,933 kWh; To cover Biorefinery electricity requirement (by including + 10% buffer) is necessary a quantity of biogas of about 1,356 Nm^3/ day -> 1,356 Nm3/ day * 6 kWh / Nm^3 * 0.36 = 2929 kWh (~110% of total el. consume) ; Annual electricity generation: 2,929 kWh /wday * 363 wday / year = 1,063,227 kWh / year; Daily Heat generation: 1,356 Nm^3 / day * 6 kWh / Nm^3 * 0.48 thermal efficiency = 3,905 kWh / day; Yearly heat generation: 3,905 kWh / day* 363 wday / year = 1,417,515 kWh / year; Calculation daily % biogas for internal use: proportion 1,356 Nm^3 : 6,321 Nm^3 = x : 100; x = 1,356 Nm3 * 100 / 6,321 m3 = 21.45%. Calculation methane % to sell = 100 % - 21.45% = 78.55%. Calculation % thermal heat reused by the biodigester = 2,478 kWh *100 / 3,905 kWh = 63.5%. CHP plant infrastructure (Information from DBEIS, 2021); For a daily requirement of 2,929 kWh, it is necessary a power of at least 2,929 kWh / 24 h = 122 kW. To cover this requirement two CHP plants of 100 kW power were chosen, made by steel (4 tons each one for a total of 8 tons) and with a lifespan of 25 years (Average value from Fusi et al., 2016 and Kelly et al., 2014)

Table B13: Fertilizer concentration in Tampio et al. (2014) and This study

	Unit	Dry matter	N tot	P tot	K tot
Tampio et al. (2014)	kg/ ton OBP	250	7.50	0.90	2.80
This Study	kg/ ton OBP	113.9	3.42	0.41	1.28
Estimated daily input this study	kg/ day	11846	355	43	133

Table B14: Digestate Solid liquid - fraction separation (from Tampio et al., 2014)

Tampio et al., (2014)	Unit	Mass	Dry matter	N tot	P tot	K tot
Total	ton/ yr	87414	2414	450	54	168
Solid	ton/ yr	8741	1932	135	49	25
Liquid	ton/ yr	78673	483	315	5	143
% Solid	ton/ yr	10	80	30	91	15
% Liquid	ton/ yr	90	20	70	9	85
Total		100	100	100	100	100

TABLE B15: Digestate solid - liquid separation this study (Coefficients from Tampio et al., 2014)

	Unit	Mass	Dry matter	Water	N tot	P tot	K tot
Total Digestate	kg/day	110786	4172	106614	355	43	133
% Solid	kg/day	10	80		30	91	15
% Liquid	kg/day	90	20		70	9	85
Solid digestate (with moisture)	kg/day	11078	3339	7739	107	39	20
Liquid digestate (with dispersed solids)	kg/day	99708	835	98873	249	4	113
Solid digestate (with moisture)	kg / year	4021348	1212051	2809297	38700	14047	7167
Liquid digestate (with dispersed solids)	kg / year	36193970	303013	35890958	90299	1433	40993
Total Digestate	kg / year	40215318	1514436	38700882	128999	15480	48159
% solids and fertilizers in SD			30.14	69.86	0.96	0.35	0.18
% solid and fertilizers in LD			0.84	99.16	0.25	0.004	0.11
Solid digestate per ton OBP	kg/ t OBP	107	32.11		1.03	0.37	0.19
Liquid digestate per ton OBP	kg/ t OBP	959	8.03		2.39	0.04	1.09
Total digestate per ton OBP	kg/ t OBP	1068	40	1028	3.43	0.41	1.28
solid digestate per vehicle	kg/ vehicle	5.5	1.6695	3.8696	0.0533	0.0193	0.0099
Liquid digestate per vehicle	kg/ vehicle	49.9	0.4174	49.4366	0.1244	0.0020	0.0565
Solid Fertilizers % concentration					0.96	0.35	0.18
Liquid Fertilizers % concentration					0.25	0.004	0.11
Total Recovered Fertilizers per year	kg / year	192638					

SD: solid digestate; LD: liquid digestate

Table B16: Biorefinery steps 1 to 5. Values in Annual input raw materials and Impact Categories are per 1-ton OBP

STEP 1: Biorefinery OBPs collection system	**Material weight**	**Lifespan**		**Annual Input Raw Material**	**FDP (in Kg oil eq)**	**FEP (in kg P eq/kg)**	**GWP 100 (Kg CO_2 eq/kg)**	**HTP Inf (Kg 1,4-DCB eq/kg)**	**MDP (in kg Fe eq /kg)**	**PMFP (in kg PM_{10} eq/kg)**	**POFP (in kg NMVOC-eq)**	**TAP100 in kg SO_2 eq**	**WDP (in m^3 H_2O eq/kg)**
Steel	17985	8	kg	5.97E-02	6.29E-02	1.45E-04	2.72E-01	1.49E-01	1.64E-01	9.18E-04	9.54E-04	9.56E-04	1.95E-03
Lead	1680	4	kg	1.12E-02	8.03E-03	2.22E-05	2.92E-02	1.22E-01	4.06E-01	1.14E-04	1.23E-04	2.48E-04	3.70E-04
Wooden pallets	4500	10	unit	1.20E-02	3.84E-02	2.91E-05	9.09E-02	3.20E-02	8.16E-03	2.92E-04	6.82E-04	4.46E-04	8.73E-04
Electricity consumption	21693		kWh	5.76E-01									
STEP 2: Manual Separation													
Steel	500	5	kg	2.66E-03	2.80E-03	6.45E-06	1.21E-02	6.61E-03	7.28E-03	4.08E-05	4.24E-05	4.25E-05	8.67E-05
Electricity consumption	25141		kWh	6.68E-01									
STEP 3: Mechanical Grinding													
Steel	12000	5	kg	2400	2.53E+03	5.83E+00	1.09E+04	5.98E+03	6.58E+03	3.69E+01	3.83E+01	3.84E+01	7.84E+01
Electricity consumption	170973		kWh										
STEP 4: Anaerobic Digestion													
Steel	87906	25	kg	3516	3.70E+03	8.54E+00	1.60E+04	8.75E+03	9.64E+03	5.41E+01	5.62E+01	5.63E+01	1.15E+02
Electricity consumption	68404		kWh	68404									
Heating consumption	899514		kWh	899514									
Water consumption	5249		m^3	5249									1.39E-01
Biogas Generation	2294523		m^3	2294523									
Raw Digestate Generation	40215318		kg	40215318									
STEP 5: Water Scrubbing													
Steel	3526	20	kg	4.68E-03	4.93E-03	1.14E-05	2.13E-02	1.17E-02	1.28E-02	7.20E-05	7.48E-05	7.50E-05	1.53E-04
Electricity consumption	540870		kWh	1.44E+01									
Water Consumption	908		m^3	2.41E-02									2.41E-02
Biomethane (97%) Generation[a]	1135464		m^3	3.02E+01	3.59E+01	4.47E-04	7.83E+00	3.79E+00	7.62E-02	1.08E-02	4.42E-02	3.65E-02	8.44E-03

a: For Biomethane were considered the related avoided impacts of Natural Gas Substitution.

TABLE B17: Biorefinery step 6. Values in Annual input raw materials and Impact Categories are per 1-ton OBP

STEP 6: Digestate Solid-Liquid separation and Storage	Material weight	Lifespan	Unit	Annual Input Raw Material	FDP (in Kg oil eq)	FEP (in kg P eq/kg)	GWP 100 (Kg CO_2 eq/kg)	HTP Inf (Kg 1,4-DCB eq/kg)	MDP (in kg Fe eq /kg)	PMFP (in kg PM_{10} eq/ kg)	POFP (in kg NMVOC-eq)	TAP100 in kg SO_2 eq	WDP (in m^3 H_2O eq/kg)
Electricity consumption	140844		kWh	3.74E+00									
Solid Digestate Generation	4021348		kg	1.07E+02									
Liquid Digestate Generation	36193970		kg	9.61E+02									
Recovered Fertilizers													
Total Recovered N in liq fraction	90299		kg	2.40E+00	4.24E+00	4.26E-03	2.86E+01	7.61E+00	3.40E+00	4.25E-02	1.64E-01	1.58E-01	5.01E-01
Total Recovered P in liquid fraction	1433		kg	3.81E-02	2.64E-02	7.77E-05	8.24E-02	5.62E-02	3.68E-02	4.01E-04	5.70E-04	9.23E-04	4.65E-03
Total Recovered K in liquid fraction	40993		kg	1.09E+00	2.51E-01	3.96E-04	1.86E+00	3.71E-01	2.55E-01	4.42E-03	3.98E-02	1.94E-02	9.14E-02
Total Recovered N in solid fraction	38700		kg	1.03E+00	1.82E+00	1.82E-03	1.23E+01	3.26E+00	1.46E+00	1.82E-02	7.04E-02	6.79E-02	2.15E-01
Total Recovered P in solid fraction	14047		kg	3.73E-01	2.59E-01	7.61E-04	8.07E-01	5.51E-01	3.61E-01	3.93E-03	5.59E-03	9.05E-03	4.56E-02
Total Recovered K in solid fraction	7167		kg	1.90E-01	4.40E-02	6.93E-05	3.25E-01	6.48E-02	4.46E-02	7.73E-04	6.96E-03	3.39E-03	1.60E-02
Storage													
Liquid Digestate Plastic Drum (50L)													
Plastic (HDPE)	61600	5	kg	3.27E-01	6.18E-01	1.51E-04	7.38E-01	1.42E-01	2.35E-02	1.01E-03	2.63E-03	2.41E-03	7.23E-03
Solid Digestate Plastic Drum													
Plastic (HDPE)	8400	5	kg	4.46E-02	8.43E-02	2.05E-05	1.01E-01	1.93E-02	3.21E-03	1.38E-04	3.58E-04	3.29E-04	9.86E-04

TABLE B18: Biorefinery Step 7. Values in Annual input raw materials and Impact Categories are per 1-ton OBP

STEP 7: Heat and Power Generation	Material weight	Lifespan	Unit	Annual Input Raw Material	FDP (in Kg oil eq)	FEP (in kg P eq/kg)	GWP 100 (Kg CO_2 eq/kg)	HTP Inf (Kg 1,4-DCB eq/kg)	MDP (in kg Fe eq /kg)	PMFP (in kg PM_{10} eq/kg)	POFP (in kg NMVOC-eq)	TAP100 in kg SO_2 eq	WDP (in m^3 H_2O eq/kg)
CHP plant infrastructure													
Steel	8000	25	kg	8.50E-03	8.95E-03	2.06E-05	3.87E-02	2.12E-02	2.33E-02	1.31E-04	1.36E-04	1.36E-04	2.78E-04
CHP plant operation													
Electricity generation	1063227			2.82E+01									
Heat generation	1417515			3.76E+01									
CHP plant biomethane combustion direct emissions													
NO_X	5743		kg	1.53E-01						3.36E-02	1.53E-01	8.54E-02	
CH_4	3435		kg	9.12E-02			2.03E+00				9.12E-04		
NMVOC	149		kg	3.95E-03							3.95E-03		
CO	2904		kg	7.71E-02									
N_2O	5.3		kg	1.41E-04			4.21E-02						
PM_{10}	4.8		kg	1.27E-04						1.27E-04			

Table B19: Ecoinvent table for Scenarios #III to #VIII without considering RF: Data source: ecoinvent database (https://www.ecoinvent.org/login-databases.html), Version 3.6 (2019), Allocation at the point of substitution; Recipe Midpoint (H) V1.13;

						IMPACT CATEGORIES				
Item	**Ref. Weight**	**Fossil Depletion**	**Freshwater Eutrophication**	**Global Warming**	**Human Toxicity**	**Metal Depletion**	**Particular Matter formation**	**Photochemica lOxidant Formation**	**Terrestrial Acidification**	**Water depletion**
		FDP (in Kg oil eq)	**FEP (in kg P eq/kg)**	**GWP 100 (Kg CO_2 eq/kg)**	**HTP Inf (Kg1,4-DCB eq/kg)**	**MDP (in kg Fe eq /kg)**	**PMFP in kg PM_{10} eq/ kg**	**POFP in kg NMVOC-eq/ kg**	**TAP100 in kg SO_2 eq**	**WDP (in m^3 H_2O eq/kg)**
Electricity	1 kWh	5.50E-02	2.87E-05	1.96E-01	3.11E-02	3.83E-03	4.44E-04	6.44E-04	1.20E-03	2.08E-02
HDPE	1 kg	1.89E+00	4.61E-04	2.25E+00	4.33E-01	7.19E-02	3.09E-03	8.03E-03	7.38E-03	2.21E-02
Lead	1 kg	7.20E-01	1.99E-03	2.61E+00	1.09E+01	3.64E+01	1.02E-02	1.10E-02	2.22E-02	3.32E-02
Pallet (Wooden)	1 unit	3.21E+00	2.43E-03	7.60E+00	2.68E+00	6.83E-01	2.45E-02	5.70E-02	3.73E-02	7.30E-02
Plastic	1 kg	1.94E+00	4.22E-04	2.22E+00	4.11E-01	7.30E-02	2.93E-03	7.80E-03	7.31E-03	1.93E-02
Polystyrene extruded	1 kg	2.35E+00	6.34E-04	4.44E+00	5.65E-01	1.17E-01	5.91E-03	1.55E-02	1.46E-02	7.82E-02
Primary Steel Production	1 kg	4.67E-01	1.59E-03	2.42E+00	1.59E+00	2.36E+00	1.01E-02	9.91E-03	8.69E-03	1.64E-02
Steel Metal Working	1 kg	5.86E-01	8.43E-04	2.13E+00	9.03E-01	3.81E-01	5.28E-03	6.06E-03	7.33E-03	1.63E-02
Steel Total	1 kg	1.05E+00	2.43E-03	4.55E+00	2.49E+00	2.74E+00	1.54E-02	1.60E-02	1.60E-02	3.27E-02

a: for steel it was used the total value.

Table B20: Process details of items shown in table B19.

Item	Ref. Weight	Process Name	Product
Electricity	1 kWh	Electricity, high voltage, production mix, BR (2213)	Electricity, high voltage 1 kWh
HDPE	1 kg	Polyethylene production, high density, granulate, RoW (2)	Polyethylene, high density, granulate (kg)
Lead	1 kg	Market for lead, GLO	Lead, 1 kg
Pallet (Wooden)	1 unit	Market for EUR-flat pallet	Wooden Euro Pallet, 1 Unit
Plastic	1 kg	Market for polypropylene, granulate, GLO (1)	Polypropylene, granulate
Polystyrene extruded	1 kg	Market for polystyrene, extruded, GLO (13)	Polystyrene, 1 kg
Primary Steel Production	1 kg	Steel production, converter, low-alloyed, RoW (320)	steel, low-alloyed (kg)
Steel Metal Working	1 kg	Metal working, average for steel product manufacturing, RoW (271)	metal working, average for steel product manufacturing [kg]
Steel Total	1 kg	Steel production and working, total process (Production + metal working)	Steel, bars, 1 kg

Table B21: Avoided impacts Food production CF for Scenarios #V and #VI. Data source: ecoinvent database (https://www.ecoinvent.org/login-databases.html), Version 3.6 (2019). Allocation at the point of substitution; Recipe Midpoint (H) V1.13;

Item	Ref. Weight	IMPACT CATEGORIES								
		Fossil Depletion FDP (in Kg oil eq)	Freshwater Eutrophication FEP (in kg P eq/kg)	Global Warming GWP 100 (Kg CO_2 eq/kg)	Human Toxicity HTP Inf (Kg 1,4-DCB eq/kg)	Metal Depletion MDP (in kg Fe eq /kg)	Particular Matter formation PMFP in kg PM_{10} eq/ kg	Photochemical Oxidant Formation POFP in kg NMVOC-eq/ kg	Terrestial Acidification TAP100 in kg SO_2 eq	Water depletion WDP (in m^3 H_2O eq/kg)
Tomato	1 kg	2.29E-02	7.76E-05	1.65E-01	4.31E-02	1.11E-02	4.27E-04	1.00E-03	1.86E-03	9.06E-02
Oranges	1 kg	6.38E-02	6.43E-05	2.83E-01	9.44E-02	2.12E-02	8.71E-04	1.97E-03	3.03E-03	1.14E-01
Potato	1 kg	7.30E-02	2.09E-04	3.24E-01	1.26E-01	2.88E-02	1.27E-03	1.87E-03	5.83E-03	8.03E-02
Apple	1 kg	8.03E-02	9.69E-05	2.92E-01	1.01E-01	3.84E-02	1.06E-03	2.25E-03	2.62E-03	1.77E-01
Papaya	1 kg	5.79E-02	5.00E-05	2.57E-01	7.13E-02	1.98E-02	5.26E-04	1.08E-03	1.47E-03	4.48E-02
Garlic	1 kg	7.13E-02	8.53E-05	3.10E-01	1.11E-01	2.78E-02	1.32E-03	2.13E-03	6.00E-03	4.57E-02
Zucchini	1 kg	7.76E-02	4.83E-05	3.03E-01	7.29E-02	2.72E-02	8.01E-04	2.16E-03	1.87E-03	1.19E-02
Chayote (cucumber)	1 kg	6.83E-01	1.13E-03	3.31E+00	1.07E+00	2.44E-02	4.97E-03	1.59E-02	1.42E-02	2.41E-02
Lettuce	1 kg	6.40E-02	4.72E-05	2.70E-01	6.44E-02	1.77E-02	6.23E-04	1.60E-03	1.56E-03	1.11E-03
Onion	1 kg	7.13E-02	8.53E-05	3.10E-01	1.11E-01	2.78E-02	1.32E-03	2.13E-03	6.00E-03	4.57E-02
Banana	1 kg	6.13E-02	7.22E-05	2.76E-01	6.02E-02	1.49E-02	8.37E-04	1.69E-03	2.95E-03	1.87E-01
Eggplant	1 kg	8.08E-01	1.39E-03	3.87E+00	1.31E+00	4.07E-02	6.01E-03	1.77E-02	1.68E-02	3.63E-02
Peach	1 kg	1.00E-01	1.22E-04	4.17E-01	1.33E-01	3.44E-02	1.75E-03	2.75E-03	6.99E-03	2.46E-01
Cucumber	1 kg	6.83E-01	1.13E-03	3.31E+00	1.07E+00	2.44E-02	4.97E-03	1.59E-02	1.42E-02	2.41E-02
Manioc (carrot)	1 kg	5.50E-02	6.42E-05	2.55E-01	1.05E-01	1.38E-02	9.79E-04	1.47E-03	4.44E-03	6.31E-02
Carrot	1 kg	5.50E-02	6.42E-05	2.55E-01	1.05E-01	1.38E-02	9.79E-04	1.47E-03	4.44E-03	6.31E-02
Pear	1 kg	1.03E-01	1.21E-04	4.39E-01	1.37E-01	4.62E-02	2.06E-03	2.95E-03	8.46E-03	2.12E-01
Mango	1 kg	2.13E-02	2.82E-05	1.10E-01	5.04E-02	2.17E-02	3.48E-04	7.79E-04	1.33E-03	2.41E-01

TABLE B22: Process details of items shown in table B21.

Item	Ref. Weight	Process Name	Product
Tomato[1]	1 kg	tomato production, fresh grade, open field, RoW	tomato, fresh grade, 1 kg
Oranges[2]	1 kg	market for orange, fresh grade GLO	orange, fresh grade
Potato	1 kg	market for potato, GLO	potato, 1 kg
Apple	1 kg	market for apple, GLO	apple, 1 kg
Papaya	1 kg	market for papaya, GLO	Papaya, 1 kg
Garlic[3]	1 kg	market for onion, GLO	onions, 1 kg
Zucchini	1 kg	market for zucchini, GLO	zucchini, 1 kg
Chayote (cucumber)[4]	1 kg	market for cucumber, GLO	cucumber,1 kg
lettuce[5]	1 kg	market for iceberg lettuce, GLO	iceberg lettuce, 1 kg
Onion	1 kg	market for onion, GLO	onions, 1 kg
Banana	1 kg	market for banana, GLO	Banana, 1 kg
Eggplant	1 kg	aubergine production, in heated greenhouse, GLO	Eggplant, 1 kg
Peach	1 kg	Market for peach, 1 kg GLO	Peach, 1 kg
cucumber	1 kg	market for cucumber, GLO	cucumber, 1 kg
Manioc (carrot)[6]	1 kg	market for carrot, 1 kgm GLO	Carrot, 1 kg
Carrot	1 kg	market for carrot, 1 kg GLO	Carrot, 1 kg
Pear	1 kg	market for pears, 1 kg GLO	Pears, 1 kg
Mango	1 kg	market for Mango, 1 kg BR	Mango. 1 kg

1: Tomato open field production because GLO tomato market consider 50% of tomato in greenhouse, being in Brasil open field local production, it is not suitable.

2. (Tangerine 1 and 2 + Oranges)

3: Due to lack of data about Garlic were considered Onion emissions because they belong to the same family, *Liliaceae*

4: Due to lack of data about Chayote were considered Cucumber (Greenhouse production) emissions because they belong to the same family, Cucurbitaceae. Due to greenhouse production,emissions could be overestimated.

5.Iceberg Lettuce because is open field production, the system used in Brazil, general lettuce considers greenhouse that could overestimate the emissions.

6.Due to lack of data about Manioc, were considered data of Carrot emissions because they belong to the same family, Apiaceae (umbrelliferae).

TABLE B23: avoided impacts food production: percentage contribution. Original values in table B21. Final values calculated per 800 kg NMF / ton OBP.

Item	Ref. Weight	Percentage contribution	Fossil Depletion FDP (in Kg oil eq)	Freshwater Eutrophication FEP (in kg P eq/kg)	Global Warming GWP 100 (Kg CO2 eq/kg)	Human Toxicity HTP Inf (Kg 1,4-DCB eq/kg)	Metal Depletion MDP (in kg Fe eq /kg)	Particular Matter formation PMFP in kg PM10 eq/ kg	Photochemical Oxidant Formation POFP in kg NMVOC-eq/ kg	Terrestrial Acidification TAP100 in kg SO_2 eq	Water depletion WDP (in m^3 H_2O eq/kg)
			IMPACT CATEGORIES								
Tomato	1 kg	35.58	8.15E-03	2.76E-05	5.86E-02	1.53E-02	3.95E-03	1.52E-04	3.57E-04	6.62E-04	3.22E-02
Oranges	1 kg	13.72	8.76E-03	8.82E-06	3.89E-02	1.29E-02	2.91E-03	1.19E-04	2.70E-04	4.16E-04	1.57E-02
Potato	1 kg	8.12	5.93E-03	1.70E-05	2.63E-02	1.02E-02	2.34E-03	1.03E-04	1.52E-04	4.74E-04	6.53E-03
Apple	1 kg	7.50	6.02E-03	7.27E-06	2.19E-02	7.60E-03	2.88E-03	7.92E-05	1.69E-04	1.97E-04	1.33E-02
Papaya	1 kg	6.12	3.54E-03	3.06E-06	1.57E-02	4.36E-03	1.21E-03	3.22E-05	6.59E-05	9.03E-05	2.74E-03
Garlic	1 kg	5.50	3.92E-03	4.69E-06	1.70E-02	6.09E-03	1.53E-03	7.25E-05	1.17E-04	3.30E-04	2.51E-03
Zucchini	1 kg	4.37	3.39E-03	2.11E-06	1.32E-02	3.18E-03	1.19E-03	3.50E-05	9.42E-05	8.18E-05	5.20E-04
Chayote (cucumber)	1 kg	3.49	2.39E-02	3.95E-05	1.16E-01	3.73E-02	8.52E-04	1.74E-04	5.54E-04	4.96E-04	8.43E-04
lettuce	1 kg	2.74	1.75E-03	1.29E-06	7.38E-03	1.76E-03	4.84E-04	1.70E-05	4.37E-05	4.26E-05	3.04E-05
Onion	1 kg	2.39	1.70E-03	2.04E-06	7.39E-03	2.64E-03	6.63E-04	3.15E-05	5.07E-05	1.43E-04	1.09E-03
Banana	1 kg	2.13	1.30E-03	1.54E-06	5.87E-03	1.28E-03	3.17E-04	1.78E-05	3.59E-05	6.27E-05	3.98E-03
Eggplant	1 kg	1.45	1.18E-02	2.02E-05	5.62E-02	1.90E-02	5.92E-04	8.74E-05	2.58E-04	2.44E-04	5.28E-04
Peach	1 kg	1.45	1.45E-03	1.77E-06	6.05E-03	1.93E-03	5.00E-04	2.55E-05	4.00E-05	1.02E-04	3.57E-03
cucumber	1 kg	1.32	9.00E-03	1.49E-05	4.35E-02	1.40E-02	3.21E-04	6.54E-05	2.09E-04	1.87E-04	3.18E-04
Manioc (carrot)[6]	1 kg	1.21	6.66E-04	7.78E-07	3.09E-03	1.27E-03	1.68E-04	1.19E-05	1.78E-05	5.38E-05	7.64E-04
Carrot	1 kg	1.16	6.39E-04	7.46E-07	2.96E-03	1.22E-03	1.61E-04	1.14E-05	1.70E-05	5.16E-05	7.34E-04
Pear	1 kg	0.88	9.06E-04	1.07E-06	3.88E-03	1.21E-03	4.08E-04	1.81E-05	2.60E-05	7.47E-05	1.87E-03
Mango	1 kg	0.87	1.85E-04	2.45E-07	9.57E-04	4.38E-04	1.88E-04	3.02E-06	6.77E-06	1.15E-05	2.10E-03
Avoided emissions Donated food	**1 kg**	**100.00**	**9.29E-02**	**1.55E-04**	**4.44E-01**	**1.42E-01**	**2.07E-02**	**1.06E-03**	**2.48E-03**	**3.72E-03**	**8.93E-02**
Avoided emission Donated food per 80 % 1 t	**80 % 1 t**	**100.00**	**7.44E+01**	**1.24E-01**	**3.56E+02**	**1.13E+02**	**1.65E+01**	**8.45E-01**	**1.99E+00**	**2.98E+00**	**7.14E+01**

TABLE B24: avoided impacts biorefinery scenario #VIII due to products replacement. Data source: Ecoinvent Database (https://www.ecoinvent.org/login-databases.html), Version 3.6 (2019), Allocation at the point of substitution; Recipe Midpoint (H) V1.13;

Item	Ref. Weight	IMPACT CATEGORIES								
		Fossil Depletion	Freshwater Eutrophication	Global Warming	Human Toxicity	Metal Depletion	Particular Matter formation	Photochemical Oxidant Formation	Terrestrial Acidification	Water depletion
		FDP (in Kg oil eq)	FEP (in kg P eq/kg)	GWP 100 (Kg CO2 eq/kg)	HTP Inf (Kg 1,4-DCB eq/kg)	MDP (in kg Fe eq /kg)	PMFP in kg PM10 eq/ kg	POFP in kg NMVOC-eq/ kg	TAP100 in kg SO_2 eq	WDP (in m^3 H_2O eq/kg)
Natural Gas	m^3	1.19E+00	1.48E-05	2.60E-01	1.26E-01	2.53E-03	3.58E-04	1.46E-03	1.21E-03	2.80E-04
Nitrogen fertilizer (N)	kg	1.77E+00	1.78E-03	1.19E+01	3.17E+00	1.42E+00	1.77E-02	6.85E-02	6.60E-02	2.09E-01
Potassium Fertilizer (K)	kg	2.31E-01	3.64E-04	1.71E+00	3.40E-01	2.34E-01	4.06E-03	3.66E-02	1.78E-02	8.40E-02
Phosphorus Fertilizer (P)	kg	6.94E-01	2.04E-03	2.16E+00	1.48E+00	9.67E-01	1.05E-02	1.50E-02	2.43E-02	1.22E-01

Table B25: Processes details of table B24. Data source: Ecoinvent Database (https://www.ecoinvent.org/login-databases.html), Version 3.6 (2019), Allocation at the point of substitution; Recipe Midpoint (H) V1.13;

Item	Ref. Weight	Process Name	Product
Natural Gas	m^3	market for natural gas, high pressure, RoW	Natural Gas, High Pressure, 1 m^3
Nitrogen fertiliser (N)	kg	market for nitrogen fertilizer, as N, GLO	nitrogen fertilizer, as N
Potassium Fertilizer (K)	kg	market for potassium fertilizer, as K2O, GLO	potassium fertilizer, as K_2O
Phosphorus Fertilizer (P)	kg	market for phosphate fertilizer, as P2O5, GLO	phosphate fertilizer, as P_2O_5

TABLE B26: All scenarios impacts per 1 ton OBP.

		IMPACT CATEGORIES								
N.	**Scenario[a]**	**Fossil Depletion**	**Freshwater Eutrophication**	**Global Warming**	**Human Toxicity**	**Metal Depletion**	**Particular Matter formation**	**Photochemical Oxidant Formation**	**Terrestrial Acidification**	**Water depletion**
		FDP (Kgoil eq)	**FEP (in kg P eq/kg)**	**GWP 100 (Kg CO_2 eq/kg)**	**HTP Inf (Kg1,4-DCB eq/kg)**	**MDP (in kg Fe eq /kg)**	**PMFP in kg PM_{10} eq/ kg**	**POFP in kg NMVOC-eq/ kg**	**TAP100 in kg SO_2 eq**	**WDP (in m^3 H_2O eq/kg)**
#I	**Landfilling 100%**	8.31	0.00686	202.52	2.30	1.87	0.36	0.36	1.68	0.08
#II	**Electricity 100%**	0.03	0.00254	172.76	-2.39	1.32	0.34	0.50	1.63	-3.08
#III	**Donation 80% + Landfilling 20%**	1.91	0.00171	41.39	0.91	1.06	0.07	0.08	0.34	0.05
#IV	**Donation 80% + Electricity 20%**	0.25	0.00084	35.44	-0.03	0.95	0.07	0.10	0.33	-0.58
#V	**Avoided Production 80% + Landfilling 20%**	-72.44	-0.12193	-314.20	-112.56	-15.46	-0.77	-1.91	-2.64	-71.39
#VI	**Avoided Production 80% + Electricity 20%**	-74.10	-0.12279	-320.15	-113.50	-15.57	-0.77	-1.89	-2.65	-72.02
#VII	**Biorefinery 100%**	0.99	0.00079	4.09	0.89	1.08	0.039	0.165	0.09	0.18
#VIII	**Biorefinery + Avoided Production 100%**	-41.55	-0.00705	-47.66	-14.81	-4.56	-0.04	-0.17	-0.20	-0.70

a: **Scenarios Impacts:** (#I) = 100% of impacts of landfilling; (#II) = (Impacts of Scenario #I) + (Impacts of electricity production) - (Impacts of electricity from the Brazilian grid being replaced by the electricity generated in the landfill); (#3) = (Impacts of donation) + (20% of impacts from Scenario #I); (#4) = (Impacts of donation) + (20% of impacts from Scenario #II); (#V) = (Impacts of donation) + (20% impacts from scenario #I) - (Impacts of the Brazilian food production being replaced by the donated food equal to 800 kg / ton OBP); (#VI) = (Impacts of donation) + (20% of impacts fromScenario #II) - (Impacts of the Brazilian food production being replaced by the donated food equal to 800 kg / ton OBP); (#VII) = 100% Impacts Biorefinery; (#VIII) = 100% Impacts Biorefinery – (Impacts of Natural Gas and Fertilizers Production replaced by Biorefinery products)

Appendix C: Barueri precipitation figure

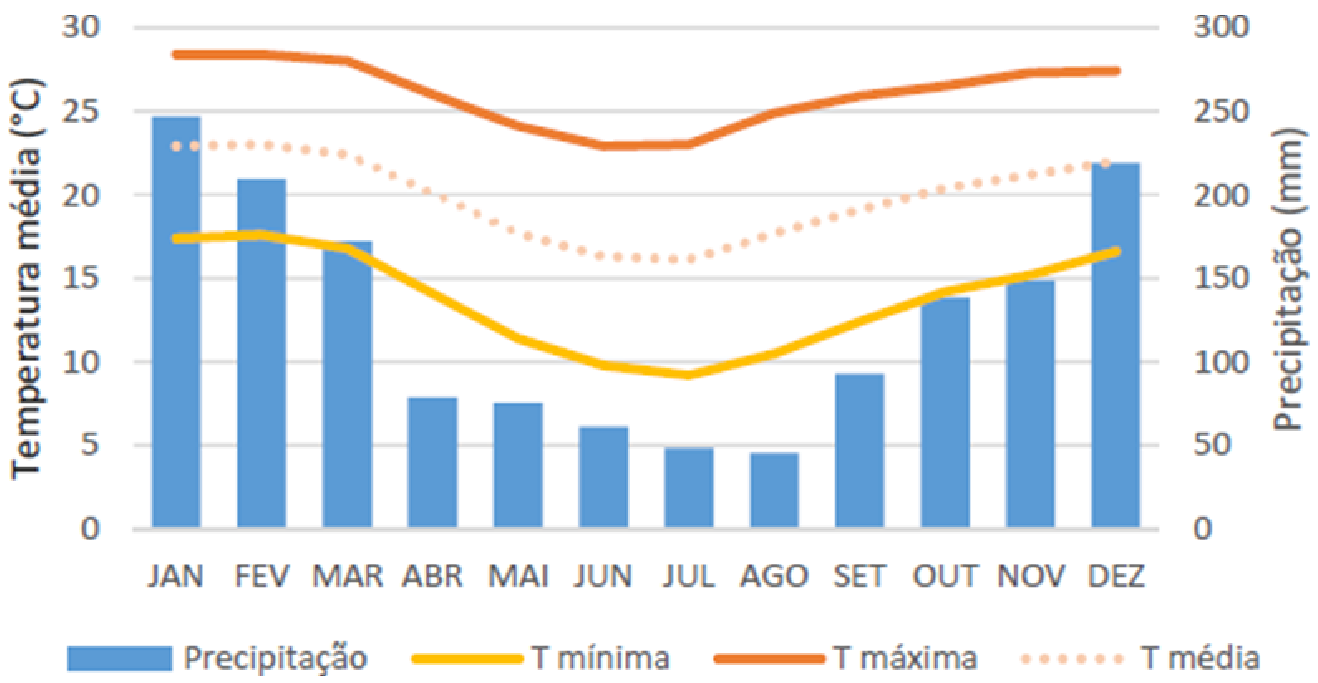

Figure C1: average annual precipitation in Baruerì (from RIMA, 2016)

Appendix D: Emergy Procedure Calculation

Emergy tables: In the following tables are shown all Inputs of Emergy synthesis. Regarding Inputs used only in emergy synthesis calculation details are provided in table notes, for Inputs in common with LCA see appendix B. UEVs calculation details are available in Table 7.

Table D1: Emergy Table Scenario #I

	Input	Type	R fract.	Amount	Unit	UEVs	UEV unit	Emergy (seJ)	% Em. Contr.
1	**Rain[a]**	R	100	4.61E+07	kg	4.68E+06	seJ/kg	2.16E+14	0.00
2	**Labor[b]**	F	15.2	3.80E+01	Person	1.55E+07	seJ/person	5.89E+08	0.00
3	**Electricity**	F	68	1.50E+04	kWh	4.18E+11	seJ/kWh	6.28E+15	0.02
4	**Iron**	F	0	2.30E+03	kg	1.09E+12	seJ/kg	2.50E+15	0.01
5	**Gravel**	F	0	6.75E+06	kg	1.27E+12	seJ/kg	8.58E+18	21.53
6	**Geotextile (poliprop.)**	F	0	4.45E+03	kg	1.64E+12	seJ/kg	7.30E+15	0.02
7	**Soil[c]**	N	0	1.88E+07	kg	1.27E+12	seJ/kg	2.39E+19	60.01
8	**Cement**	F	0	6.63E+02	kg	2.50E+12	seJ/kg	1.66E+15	0.00
10	**GCL (Clay)**	F	0	1.95E+04	kg	2.54E+12	seJ/kg	4.95E+16	0.12
11	**Steel**	F	0	1.34E+04	kg	2.01E+12	seJ/kg	2.69E+16	0.07
12	**Rubber**	F	0	1.26E+03	kg	5.46E+12	seJ/kg	6.87E+15	0.02
13	**Diesel Fuel**	F	0	1.95E+05	kg	5.99E+12	seJ/kg	1.17E+18	2.93
14	**HDPE**	F	0	1.52E+04	kg	6.69E+12	seJ/kg	1.02E+17	0.26
15	**Polyacrylammide**	F	0	1.27E+03	kg	6.78E+12	seJ/kg	8.61E+15	0.02
16	**Plastic (PVC)**	F	0	1.22E+03	kg	7.45E+12	seJ/kg	9.06E+15	0.02
17	**Services[b]**	F	15.2	6.44E+05	US$	8.41E+12	seJ/$	5.41E+18	13.59
18	**Ferric chloride**	F	0	1.66E+04	kg	2.93E+13	seJ/kg	4.86E+17	1.22
19	**Aluminum (Billet)**	F	0	7.75E+02	kg	8.60E+13	seJ/kg	6.66E+16	0.17
	Total (U) Scenario 1							**3.98E+19**	**100.00**
	UEV system (Em per 1 t)							**1.06E+15**	
	TOTAL R scenario 1	R	100					**8.27E+17**	**2.08**
	Total N scenario 1	N	0					**2.39E+19**	**60.01**
	Total F scenario 1	**F**	**0**					**1.51E+19**	**37.91**

a: Calculation Rainfall Caieiras: Landfill Class II 120 ha = 1,200,000m^2; Annual Rainfall Caieiras 1537 mm/ yr (RIMA), 2016; 1 mm = 1 L / m^2 = 1 kg / m^2; AnnualRain quantity in kg per m^2 in Caieiras; 1537 kg / m^2; Tot. Annual Rain amount on Class II landfill surface; 1537 kg / m^2 * 1,200,000 m^2 = 1.84E+09 kg; Calc Rainused by CEAGESP fr; CEAGESP OF = 2.50 % tot CAIEIRAS OF; 1.84E+09 kg * 2.50% = 4.61E+07 kg / yr

b: For labor and services see table D3.

c: Calculation Soil use: 47,065 waste tons * 40% (Buranakarn, 1998) = 18,826 tons/ yr = 18,826 t/yr * 10^3 kg/t = 18,826,000 kg / yr

Table D2: emergy Table Scenario #II

N	Input	Type	R fract.	Amount	Unit	UEVs	UEV unit	Emergy (seJ)	% Em. Contr.
1	**U Scenario I**							3.98E+19	99.58
2	**Labor[a]**	F	15.2	4.00E+00	Person	1.55E+07	seJ/person	6.20E+07	0.00
3	**Water**	R	100	2.28E+05	kg	2.58E+08	seJ/kg	5.88E+13	0.00
4	**Concrete**	F	0	2.64E+03	kg	1.83E+12	seJ/kg	4.84E+15	0.01
5	**Steel**	F	0	2.79E+02	kg	2.01E+12	seJ/kg	5.62E+14	0.00
6	**Lubricant Oil**	F	0	2.53E+03	kg	4.72E+12	seJ/kg	1.19E+16	0.03
7	**Services[a]**	F	15.2	1.80E+04	$	8.41E+12	seJ/$	1.51E+17	0.38
	Tot. Emergy U scenario II							**4.00E+19**	
	UEV system scenario II							**1.06E+15**	
	TOTAL R scenario II	R						**8.50E+17**	2.13
	Total N scenario II	N						**2.39E+19**	59.76
	Total F scenario II	F						**1.52E+19**	38.11

a: For labor and services see table D3

Table D3: Table Labor and Services Calculation Scenarios #I and #II

Vehicle / material type	unit price (R$)	quantity	Unit	total price	Currency	year	Infl. 12/2018 - 08/ 2020 (%)	Valor Real 2018 eqv.	Change R$ / US$ 2018 [a]	Lifespan*	Annual cost in US$
Compactor trucks 15 m^3	93500	8	Vehicles	748000	R$	2018	0	748000	0.258	10	**19298**
Excavator (x2)	450000	2	Vehicles	900000	R$	2020	5.00	855000	0.258	14	**15756**
Transport truck 30 t	500000	4	Vehicles	2000000	R$	2020	5.00	1900000	0.258	10	**49020**
Tank truck 30 m^3	500000	1	Vehicles	500000	R$	2020	5.00	475000	0.258	10	**12255**
Bulldozer	1500000	1	Vehicles	1500000	R$	2020	5.00	1425000	0.258	10	**36765**
Soil Compactor	200000	1	Vehicles	200000	R$	2020	5.00	190000	0.258	10	**4902**
Front Loader	550000	1	Vehicles	550000	R$	2020	5.00	522500	0.258	10	**13481**
Diesel[b]	3.50	231199	L	809196	R$	2018	0	809196	0.258	1	**208773**
GCL[c]	30.22	5413	m^2	163581	R$	2020	5.00	155402	0.258	1	**40094**
HDPE[d]	34	12244	m^2	416296	R$	2020	5.00	395481	0.258	1	**102034**
Geotextile[e]	4.20	11120	m^2	46704	R$	2020	5.00	44369	0.258	1	**11447**
Gravel (Landfill + wwat plant)	67.42	4723	m^3	318425	R$	2020	5.00	302503	0.258	1	**78046**
Ferric Chloride	11.50	16569	kg	190544	R$	2020	5.00	181016	0.258	1	**46702**
Polyacrylammide	15.00	1270	kg	19050	R$	2020	5.00	18098	0.258	1	**4669**
Steel (wwplant)	4.15	345	kg	1431.75	R$	2018	0	1432	0.258	1	**369**
Cement	0.5	663	kg	331.5	R$	2020	5.00	314.925	0.258	1	**81**
Total Services Scenario I				**8363558**	**R$**			**8023312**			**643693**
Concrete power plant	270	1.06	m^3	286.2	R$	2018	0	286.2	0.258	1	**74**
Steel Power plant	4.15	279	kg	1158	R$	2018	0	1158	0.258	1	**299**
Lubricant oil	25	2879	L	71975	R$	2020	5.00	68376	0.258	1	**17641**
Sum services Scenario II electr. Prod.				73419	R$			69820	0.258		**18014**
Total Services Scenario II				**8436977**	**R$**			**8093132**	**0.258**		**661706**

a: Change BRL / USD = 1 USD /3.87 BRL = 0.258 on 31/12/2018 source: https://www.statista.com/statistics/958311/usd-brl-exchange-rate

b: Diesel price 2018 = 3.50 R$ / L (Source ANP, (Agência Nacional do Petróleo), 2019, Boletim Trimestral de Preços de Combustíveis; https://www.gov.br/anp/pt-br/centrais-de-conteudo/publicacoes/boletins-anp/btpvc-1/boletim-trimestral-1.pdf)

c: GCL price: 1 m^2 = 3.6 kg; Price 1 m^2 in 2020 = 30,22 R$ (CGC concessões, 2020); Proportion 1 m^2 : 3.6 kg = x : 19485 kg (quantity used in this study); x = 1 m^2 * 19485 kg / 3.6 kg = 5413 m^2

d: HDPE (2 mm tickness): (5 m * 80 m) = 400 m^2, total weight 495 kg (Source company ROMA http://www.roma.ind.br/ and (CGC 2020); Weight m^2 = 495 kg/400 m^2 = 1.24 kg / m^2; Proportion 1 m^2

: 1.24 kg = x : 15183 kg (quantity used in this study) ; x = 1 m^2 * 15183 kg / 1.24 kg = 12,244 m^2

e: Geotextile (400 g / m^2): 4.20 R$ m^2; Weight m^2 = 0.400 kg/ m^2; Proportion 1 m^2 : 0.400 kg = x : 4448 kg (used in this study); x = 1 m^2 * 4,448 kg / 0.400 kg = 11,120 m^2

f: Gravel (pedra, brita n. 4) specific weight = 1430 kg / m^3 ; Proportion 1 m^3 : 1430 kg = x : 6,753,403 kg (total quantity used this study); x = 1 m^3 * 6,753,403 kg / 1430 kg = 4,723 m^3 ;

Table D4: Emergy Table Donation Scenarios #III and #V

N	**Input**	**Type**	R fract.	Amount	Unit	UEVs	UEV unit	**Emergy (seJ)**	**% Em. Contr.**
1	**Labor**[d]	F	15.2	1.90E+01	Person	1.55E+07	seJ/Person	2.95E+08	0.00
2	**Wood**	F	82.4	4.50E+02	kg	1.88E+08	seJ/kg	8.45E+10	0.00
3	**Electricity**	F	68	5.95E+04	kWh	4.18E+11	seJ/kWh	2.49E+16	4.65
4	**Steel**	F	0	3.69E+03	kg	2.01E+12	seJ/kg	7.40E+15	1.38
5	**Plastic**	F	0	8.59E+01	kg	7.45E+12	seJ/kg	6.41E+14	0.12
6	**Polystyrene**	F	0	6.59E+01	kg	7.45E+12	seJ/kg	4.91E+14	0.09
7	**Services**[d]	F	15.2	4.17E+04	$	8.41E+12	seJ/$	3.51E+17	65.57
8	**Lead**	F	0	4.20E+02	kg	3.59E+14	seJ/kg	1.51E+17	28.19
	U Smart Scenario without RF[a]							**5.35E+17**	**100.00**
	20 % U RF landfilling (Scenario # I)							**7.97E+18**	
	Total U Donation Scenario[b]							**8.50E+18**	
	UEV System (Em per 1 ton OBP)[c]							**2.27E+14**	

a: Without considering RF.

b: Total eMergy U of Donation scenario, including the residual fraction RF sent to landfill without for electricity production (as in scenario II)

c: Total eMergy U of Donation scenario divided by 37,652

tons/yrd: For Labor and services see table D6.

Table D5: Emergy Table Donation Scenarios #IV and #VI

N	Input	Type	R fract.	Amount	Unit	UEVs	UEV unit	Emergy (seJ)	% Em. Contr.
1	**Labor[d]**	F	15.2	1.90E+01	Person	1.55E+07	seJ/Person	2.95E+08	0.00
2	**Wood**	F	82.4	4.50E+02	kg	1.88E+08	seJ/kg	8.45E+10	0.00
3	**Electricity**	F	68	5.95E+04	kWh	4.18E+11	seJ/kWh	2.49E+16	4.65
4	**Steel**	F	0	3.69E+03	kg	2.01E+12	seJ/kg	7.40E+15	1.38
5	**Plastic**	F	0	8.59E+01	kg	7.45E+12	seJ/kg	6.41E+14	0.12
6	**Polystyrene**	F	0	6.59E+01	kg	7.45E+12	seJ/kg	4.91E+14	0.09
7	**Services[d]**	F	15.2	4.17E+04	$	8.41E+12	seJ/$	3.51E+17	65.57
8	**Lead**	F	0	4.20E+02	kg	3.59E+14	seJ/kg	1.51E+17	28.19
	U Smart Scenario without RF[a]							**5.35E+17**	**100.00**
	20 % U RF electricity (Scenario #II)							**8.00E+18**	
	Total U Donation Scenario[b]							**8.54E+18**	
	UEV System (Em per 1 ton OBP)[c]							**2.27E+14**	

a: Without considering RF.

b: Total eMergy U of Donation scenario, including the residual fraction RF sent to landfill by including electricity production (as in scenario #II)

c: Total eMergy U of Donation scenario divided by 37,652
ton/yr **d:** For labor and services see table D6

TABLE D6: Services Donation Scenarios from #III to #VI

Vehicle / material type	Price Unit (R$)	quantity	Unit	Total price	Currency	Price year	Lifecost Var.[9]	Valor Real 2018 eqv.	Change R$ / US$ 2018[10]	Lifespan*	Annual cost in US$
Tow Tractor[1]	63000	3	Vehicle	189000	R$	**	**	189000	0.258	8	**6095**
Frame[1]	15000	9	Frame	135000	R$	**	**	135000	0.258	8	**4354**
Trolley[2]	2057	180	Trolley	370260	R$	2021	01/2021-12/2018	336937	0.258	8	**10866**
Wooden Pallet[3]	20	180	Pallet	3600	R$	2021	01/2021-12/2018	3276	0.258	10	**85**
Electricity[4]	0.8	59505	kWh	47604	R$	2021	01/2021-12/2018	43320	0.258	1	**11176**
Shed[5]	200	900	m^2	180000	R$	2021	01/2021-12/2018	163800	0.258	45	**939**
Table[6]	1259	108	Table	135972	R$	2021	01/2021-12/2018	123735	0.258	11	**2902**
Cold Room[7]	30435	6	Cold Room	182610	R$	2016	08/2016-12/2018	197219	0.258	10	**5088**
Plastic Pallet[8]	179	72	Pallet	12888	R$	2021	01/2021-12/2018	11728	0.258	15	**202**
Total				1256934				1204014			**41707**

1: Adapted from "Quanto custa percorrer" - Revista intralogistica https://www.imam.com.br/consultoria/artigo/pdf/quanto-custa-percorrer.pdf andhttps://b2b.nowak.com.br/transpaletes/transpaleteeletrico/rebocador-eletrico-4000kg-paletrans-rp40-ref-7772 for 4 tons capacity tow tractors and related frames.

2: Trolley price 315 Eur /unit (https://www.hahn-kolb.nl/All-categories/Pallet-trolley-for-commercially-available-tugger-trains/5010CL04_0508010112.cyid/5010.cgid/en/US/EUR/) Conversion in BRL: 375 Eur/unit * 6.53 R$/Eur (change at 19/02/2021) = 2057 R$/unit

3: Considering a Purchase > 50 pallets. From: https://www.viadutrapallets.com.br/palete-tipo-euro#:~:text=O%20pallet%20Euro%20pre%C3%A7o%20diferentespara,00%20a%20R%2430%2C00.

4: Average price of electricity in SP in 2020 = 0,80 R$/kWh from:https://www.ngsolar.com.br/single-post/preco-kwh-cpfl#:~:text=A%20tarifa%20A4%20comercial%20e,R%240%2C49%20por%20kWh.

5: Shed price per m^2 from: https://www.cronoshare.com.br/blog/quanto-custa-m2-construcao-galpao-precos/

6: Table price per unit; from: https://www.lojabrazil.com.br/mesa-aco-inox-industrial-bancada-de-apoio-1-6m-160x70x90cm-br-160s-brascool.html

7: Cold room prices from: https://www.tectermica.com.br/assets/camara-frigorifica-padronizada_tabela-precos.pdf

8: Plastic Pallet Prices from: https://www.pisosplasticos.com.br/pallet-plastico-fabricante-palet-leve-100x120-palete-plastico-pallets

9: Reference period 12/2018. Lifecost variation from 01/2021 to 12/2018 estimated in - 9%; From 08/2016 to 12/2018 estimated in + 8% **10:** Change BRL / USD = 1 USD /3.87 BRL = 0.258 on 31/12/2018 source: https://www.statista.com/statistics/958311/usd-brl-exchange-rate

TABLE D7: emergy table biorefinery scenarios #VII and #VIII

N	Input	Type	R fract.	Amount	Unit	UEVs	UEV Unit	Emergy (seJ) / yr	% Em.contr.
1	**Labor**	F	15.2	3.80E+01	Person	1.55E+07	seJ/Person	5.89E+08	0.00
2	**Water**	F	50	6.16E+06	kg	7.28E+08	seJ/kg	4.48E+15	0.10
3	**Air**[a]	R	100	2.86E+04	kg	3.92E+10	seJ/kg	0.00E+00	0.00
4	**Wood**	F	82.4	4.50E+02	kg	1.94E+11	seJ/kg	8.73E+13	0.00
5	**Steel**	F	0	8.76E+03	kg	2.01E+12	seJ/kg	1.76E+16	0.40
6	**HDPE**	F	0	1.40E+04	kg	6.69E+12	seJ/kg	9.37E+16	2.11
7	**Services**	F	15.2	4.96E+05	$	8.41E+12	seJ/$	4.18E+18	94.00
8	**Lead**	F	0	4.20E+02	kg	3.59E+14	seJ/kg	1.51E+17	3.39
	Total U							**4.44E+18**	**100.00**
	UEV system (eM per t OBP)[c]							**1.18E+14**	
	Total R	**R**						**6.37E+17**	**14.34**
	Total N	**N**						**0.00E+00**	
	Total F	**F**						**3.80E+18**	**85.66**

a: Due to high level of uncertainty, the emergy of Air was not included.

Table D8: Services Biorefinery Scenarios #VII and #VIII

Vehicle / material type	Price Unit (R$)	Quantity	Unit	Total price	Currency	Price year	Lifecost Var.[12]	Valor Real 2018 eqv.	Change R$ / US$ 2018	Lifespan	Annual cost in US$
Tow Tractor[1]	63000	3	Vehicle	189000	R$	**	**	189000	0.258	8	6095
Frame[1]	15000	9	Frame	135000	R$	**	**	135000	0.258	8	4354
Trolley[2]	2057	180	Trolley	370260	R$	2021	01/2021-12/2018	336937	0.258	8	10866
Wooden Pallet[3]	20	180	Pallet	3600	R$	2021	01/2021-12/2018	3276	0.258	10	85
Conveyor Belt[4]	39600	1	Machine	39600	R$	2014	12/2014-12/2018	50142	0.258	5	2587
Grinder[5]	5000	3	Grinder	15000	R$	2014	12/2014-12/2018	18993	0.258	5	980
Digester[6]	25422000	1	Digester	25422000	R$	2018	**	25422000	0.258	25	262355
Consumed Water[7]	39	908	m^3	35412	R$	2018	**	35412	0.258	1	9136
Water Scrubber[8]	4014000	1	W Scrubber	4014000	R$	2018	**	4014000	0.258	20	51781
Liquid Digestate Plastic Drum[9]	112	28000		3136000	R$	2022	10/2022 - 12/2018	2508800	0.258	5	129454
Solid Digestate Plastic Drum[10]	12	28000		336000	R$	2022	10/2022 - 12/2018	268800	0.258	5	13870
CHP plant[11]	679000	2	Generator	1358000	R$	2018	**	1358000	0.258	25	14015
Total				35053872	R$			32982359	0.258		505578

1: Adapted from "Quanto custa percorrer" - Revista intralogistica https://www.imam.com.br/consultoria/artigo/pdf/quanto-custa-percorrer.pdf and https://b2b.nowak.com.br/transpaletes/transpaleteeletrico/rebocador-eletrico-4000kg-paletrans-rp40-ref-7772 for 4 tons capacity tow tractors and related frames

2: Trolley price 315 Eur /unit (https://www.hahn-kolb.nl/All-categories/Pallet-trolley-for-commercially-available-tugger-trains/5010CL04_0508010112.cyid/5010.cgid/en/US/EUR/) Conversion in BRL: 375 Eur/unit * 6.53 R$/Eur (change at 19/02/2021) = 2057 R$/unit

3: Considering a Purchase > 50 pallets. From: https://www.viadutrapallets.com.br/palete-tipo euro#:~:text=O%20pallet%20Euro%20pre%C3%A7o%20diferentespara,00%20a%20R%2430%2C00.

4: From Uratani et al. (2014).

5: From Uratani et al. (2014)

6: From Francini et al. (2020): 11,399,000 Euro investment for the construction of a 4,000 m^3 wet mesophilic biodigester with pre and post-treatment-> in this study it was considered the 50 % of this value due to smaller biodigester (3,000 m^3, - 25%) and pre and post - treatment considered separately (-25%); therefore 11,399,000 EUR / 2 = 5,699,950 EUR ~ 5,700,000 EUR and by considering an EUR / BRL echange rate of 4.46 on 31 / 12 / 2018, it was assumed a price in BRL equal to 5,700,000 EUR * 4.46 = 25,422,000 BRL

7: by considering a monthly consumption of ~ 75.625 m^3 / month and a SABESP price in 2018 of 19.50 R$ water + 19.50 R$ wwater = 39 R$ / m^3 (https://site.sabesp.com.br/site/uploads/file/asabesp_doctos/comunicado_06_2018.pdf)

8: From Qie Sun et al. (2015): the average cost of a water scrubber of ~200 m3 / h capacity is about 700,000 - 900,000 EUR. In this study it was modelled a water scrubber of 230 m3 / hr capacity, therefore it was assumed a price of 900,000 EUR. By considering an 4.46 EUR / BRL exchange rate on 31/12/2018, it was assumed a price in BRL equal to 900,000 EUR * 4.46 BRL / EUR = 4,014,000 BRL.

9: by considering a 50L container price of 112R$ (from Plastícos Ipiranga https://loja.plasticosipiranga.com.br/bombonas-plasticas-nova-50-litros-tampa-fixa-homologada?utm_source=Site&utm_medium=GoogleMerchant&utm_campaign=GoogleMerchant&sku=BOMB-50TF&gclid=EAIaIQobChMI3_KJgOzn-gIV1RXUAR3EJAVCEAQYASABEgI_UPD_BwE) and a need of 2,000 containers / day * 14 days (turnover + deposit) = 28,000 containers

10: by considering a 10 L container price of 30 R$ (from: https://loja.plasticosipiranga.com.br/buscar?q=bombona+10+litros) and a need of 2,000 containers / day * 14 days (turnover + deposit) = 28,000 containers

11: It was considered a plant constituted by two generators of 100 kW power each one, for a total capacity of 200 kW. Being the capital cost for kW equal to 1,750 U$ per kW (from: https://www.institutodeengenharia.org.br/site/wp-content/uploads/2017/10/arqnot8956.pdf) the total capital cost was estimated in: (1,750 U$ / kW * 100 kW = 175,000 U$ * 3.88 BRL / U$ on 31/ 12/ 2018) = 679,000 R$ * 2 plants = 1,358,000 R$

12: Data from IBGE (https://www.ibge.gov.br/explica/inflacao.php): it was considered an inflation rate of about - 9% between 01 / 2021 - 12/2018, of about + 27% between 12/2014 and 12/2018 and about - 20% between 09/2022 - 12/2018

TABLE D9: Saved Emergy Table Biorefinery Scenario #VIII

N.	Replaced Product	Amount	Unit	UEVs	UEV Unit	Emergy (seJ) / yr	% Em.contr.
1	**Natural Gas**	1135464	m^3	5.30E+12	seJ / m^3	6.02E+18	88.70
2	**Potassium (K)**	48159	kg	1.40E+12	seJ / kg	6.73E+16	0.70
3	**Nitrogen (N)**	128999	kg	4.83E+12	seJ / kg	6.23E+17	6.52
4	**Phosphorus (P)**	15480	kg	4.95E+12	seJ / kg	7.67E+16	0.80
	Total Saved Emergy					6.79E+18	**100.00**
	Saved Emergy / ton OBP					1.80E+14	
	Net Emergy					2.34E+18	
	Net Emergy / ton OBP					6.23E+13	

TABLE D10: saved Emergy Literature sources and conversion factors

N.	Item	Unit	Original UEV	Original Unit	Original Bsl. (SeJ/yr)	Source	Conversion	This Study UEV	Unit Used
1	**Tomato**	g	1.60E+10	seJ/g	1.58E+25	Brandt Williams (2002)	0.76	1.22E+13	seJ/kg
2	**Orange**	g	1.92E+09	seJ/g	1.58E+25	Brandt Williams (2002)	0.76	1.46E+12	seJ/kg
3	**Potato**	g	2.80E+09	seJ/g	1.58E+25	Brandt Williams (2002)	0.76	2.13E+12	seJ/kg
4	**Bell Pepper**	g	1.68E+10	sej/g	1.58E+25	Brandt Williams (2002)	0.76	1.28E+13	seJ/kg
5	**Banana**	g	1.23E+09	sej/g	Not Available	de Barros et al. (2009)	1	1.23E+12	seJ/kg
6	**Beef**	g	1.58E+10	sej/g	1.20E+25	Amiri et al. (2022)	1	1.58E+13	seJ/kg
7	**Lettuce**	g	1.96E+10	sej/g	1.58E+25	Brandt Williams (2002)	0.76	1.49E+13	seJ/kg
8	**Chicken**	g	4.35E+09	sej/g	9.44E+24	Castellini et al. (2006)	1.27	5.52E+12	seJ/kg
9	**Bread (Grain)**	g	1.45E+10	sej/g	1.58E+25	Brandt Williams (2002)	0.76	1.10E+13	seJ/kg
10	**Natural Gas[a]**	m^3	1.78E+05	seJ/J	1.52E+25	Brown et al. (2011)	0.79	5.30E+12	seJ / m^3
11	**Potassium (K)**	kg	1.10E+09	seJ/g	9.44E+24	Odum (1996)	1.27	1.40E+12	seJ / kg
12	**Nitrogen (N)**	kg	3.80E+09	seJ/g	9.44E+24	Odum (1996)	1.27	4.83E+12	seJ / kg
13	**Phosphorus (P)**	kg	3.90E+09	seJ/g	9.44E+24	Odum, (1996)	1.27	4.95E+12	seJ / kg

a: Conversion calculation: 1.78E+05 seJ/J (Brown at al., 2011) * 37.7 MJ/m^3 (Gross Heating Value from https://www.engineeringtoolbox.com/gross-net-heating-values-d_420.html)*(12.00E+24 seJ*yr^{-1} /15.2E+24 seJ*yr^{-1}) Brown and Ulgiati 2010 baseline.

TABLE D11: Invested Emergy (EMI), saved Emergy (EMS) and ERI (EMS/EMI ratio)

	Scenarios	Used eM (seJ/yr)	Invested EMI per ton	Saved Emergy per ton (EMS)	ERI
1	**Landfilling**	3.98E+19	1.06E+15	0.00E+00	0.0
2	**Electricity**	4.00E+19	1.06E+15	6.37E+13	0.06
3	**Donation + (landfilling)**	8.50E+18	2.26E+14	6.56E+15	29.0
4	**Donation (+ electricity)**	8.54E+18	2.27E+14	6.57E+15	29.0
5	**Donation 100% (Ideal)**	5.35E+17	1.42E+13	8.20E+15	577.0
6	**Biomethane Biorefinery**	4.44E+18	1.18E+14	1.80E+14	1.5
7	**Compost (Agostinho et al., 2016)**		3.04E+13	7.83E+13	2.6
8	**Electricity (Almeida et al., 2012)**	1.22E+20	1.91E+14	1.64E+14	0.9
9	**Electricity (Marchettini et al., 2007)**		6.63E+14	1.28E+14	0.2
10	**Incineration (Marchettini et al., 2007)**		2.22E+14	7.10E+14	3.2
11	**Compost (Marchettini et al., 2007)**		1.55E+14	6.12E+14	4.0
12	**Bioethanol, (Patrizi et al., 2015)**		2.57E+14	5.06E+14	2.0
13	**Electr. + An. Feed (Santagata, et al., 2019)**		5.56E+14	4.02E+15	7.2
14	**Stillage Combustion (Baral et al., 2015)**		3.27E+14	2.95E+13	0.1
15	**Landfilling (Ali et al., 2018)**		6.11E+13	0.00E+00	0.0
16	**Compost + landfilling (Ali et al., 2018)**		1.61E+14	9.91E+13	0.6
17	**Compost + incineration (Ali et al., 2018)**		2.26E+15	9.91E+13	0.04
18	**A: incin. + landfilling (Wang et al., 2018)**		1.52E+14	2.27E+14	1.5
19	**B: incin. + conr. paving brick product. (Wang et al., 2018)**		4.04E+14	2.27E+14	0.6
20	**c: Incin. + non-burnt wall brick prod. (Wang et al., 2018)**		2.59E+14	2.27E+14	0.9
21	**Donation (Eriksson and Spångberg, 2017)**		2.01E+13	5.95E+15	296.5
22	**Donation (Eriksson et al., 2015)**		1.27E+14	9.69E+15	76.3

TABLE D12: Invested emergy from literature.

Source	U (seJ/yr)	Orig. Unit.	Orig. EMI/unit (seJ/unit)	Orig. EMI / ton	Original bsl. (SeJ/yr)	C.V.	EMI per ton upd.
Compost (Agostinho et al., 2013)		ton	4.00E+13	4.00E+13	1.58E+25	0.76	3.04E+13
Electricity (Almeida et al., 2012)	1.60E+20			2.52E+14	1.58E+25	0.76	1.91E+14
Electricity (Marchettini et al., 2007)		g	5.22E+08	5.22E+14	9.44E+24	1.27	6.63E+14
Incineration (Marchettini et al., 2007)		g	1.75E+08	1.75E+14	9.44E+24	1.27	2.22E+14
Compost (Marchettini et al., 2007)		g	1.22E+08	1.22E+14	9.44E+24	1.27	1.55E+14
Bioethanol (Patrizi et al., 2015)	7.52E+18	ton	1.98E+14	1.98E+14	9.26E+24	1.3	2.57E+14
Electricity (Santagata et al., 2019)				5.56E+14	1.20E+25	1	5.56E+14
Animal feed (Santagata et al., 2019)				5.56E+14	1.20E+25	1	5.56E+14
Stillage Combustion (Baral et al., 2015)		ton	4.13E+14	4.13E+14	1.52E+25	0.79	3.27E+14
Landfilling (Ali et al., 2018)		ton	6.11E+13	6.11E+13	1.20E+15	1	6.11E+13
Compost + landfilling (Ali et al., 2018)		ton	1.61E+14	1.61E+14	1.20E+15	1	1.61E+14
Compost + incineration (Ali et al., 2018)		ton	2.26E+15	2.26E+15	1.20E+15	1	2.26E+15
A: incin. + landfilling (Wang et al., 2018)	6.54E+19			1.52E+14	1.20E+15	1	1.52E+14
B: incin. + conr. paving brick product (Wang et al., 2018)	1.73E+20			4.04E+14	1.20E+15	1	4.04E+14
c: Incin. + non-burnt wall brick prod. (Wang er al., 2018)	1.11E+20			2.59E+14	1.20E+15	1	2.59E+14
Donation (Eriksson and Spångberg, 2017)	2.02E+14			2.01E+13	1.20E+15	1	2.01E+13
Donation (Eriksson et al., 2015)				1.27E+14	1.20E+15	1	1.27E+14

TABLE D13: Table Saved Emergy Literature

N.	Source	Original unit	EMS/ Or.Unit (sej/unit)	EMS / ton	C. v.[a]	EMS updated /ton
1	**Compost (Agostinho et al., 2013)**	ton	1.03E+14	1.03E+14	0.76	7.83E+13
2	**Electricity (Almeida et al., 2012)**	ton		2.16E+14	0.76	1.64E+14
3	**Electricity (Marchettini et al., 2007)**	g	1.01E+08	1.01E+14	1.27	1.28E+14
4	**Incineration (Marchettini et al., 2007)**	g	5.59E+08	5.59E+14	1.27	7.10E+14
5	**Compost (Marchettini et al., 2007)**	g	4.82E+08	4.82E+14	1.27	6.12E+14
6	**Bioethanol (Patrizi et al., 2015)**			3.89E+14	1.3	5.06E+14
7	**Electricity (Santagata et al., 2019)**			3.24E+15	1	3.24E+15
8	**Animal feed (Santagata et al., 2019)**			9.31E+14	1	9.31E+14
8A	**Santagata total (animal feed + electricity)**			4.17E+15	1	4.17E+15
9	**Stillage Combustion (Baral et al., 2015)**	ton		3.74E+13	0.79	2.95E+13
10	**Landfilling (Ali et al., 2018)**	ton	0	0.00E+00	1	0.00E+00
11	**Compost + landfilling (Ali et al., 2018)**	ton	9.91E+13	9.91E+13	1	9.91E+13
12	**Compost + incineration (Ali et al., 2018)**	ton	9.91E+13	9.91E+13	1	9.91E+13
13	**A: incin. + landfilling (Wang et al., 2018)**			2.27E+14	1	2.27E+14
14	**B: incin. + conr. paving brick product (Wang et al., 2018)**			2.27E+14	1	2.27E+14
15	**C: Incin. + non-burnt wall brick prod. (Wang er al., 2018)**			2.27E+14	1	2.27E+14
16	**Donation (Eriksson and Spångberg, 2017)**			5.95E+15	1	5.95E+15
17	**Donation (Eriksson et al., 2015)**			9.69E+15	1	9.69E+15

a: conversion factor baseline

Table D14: EMS = f(EMI) model table.

				Model	Error
				EMS = a/EMI^b	E = (y – Est. y)^2
N.	Scenario	EMI (x)	EMS (y)	Estimated y	Error
1	Landfilling	1.06E+15	0.00E+00	4.28E+14	1.83E+29
2	Electricity	1.06E+15	6.37E+13	4.27E+14	1.32E+29
3	Donation (+ electricity)	2.27E+14	6.57E+15	1.14E+15	2.95E+31
4	Donation 100% (Ideal)	1.42E+13	8.20E+15	6.58E+15	2.62E+30
5	Biomethane Biorefinery	1.18E+14	1.80E+14	1.72E+15	2.37E+30
6	Compost (Agostinho et al., 2016)	3.04E+13	7.83E+13	4.06E+15	1.59E+31
7	Electricity (Almeida et al., 2012)	1.91E+14	1.64E+14	1.27E+15	1.21E+30
8	Electricity (Marchettini et al., 2007)	6.63E+14	1.28E+14	5.76E+14	2.00E+29
9	Incineration (Marchettini et al., 2007)	2.22E+14	7.10E+14	1.15E+15	1.95E+29
10	Compost (Marchettini et al., 2007)	1.55E+14	6.12E+14	1.45E+15	6.97E+29
11	Bioethanol, (Patrizi et al., 2015)	2.57E+14	5.06E+14	1.05E+15	2.95E+29
12	Animal feed + Electricity (Santagata et al., 2019)	5.56E+14	4.17E+15	6.44E+14	1.14E+31
13	Stillage Combustion (Baral et al, 2015)	3.27E+14	2.95E+13	9.02E+14	7.79E+29
14	Landfilling (Ali et al., 2018)	6.11E+13	0.00E+00	2.61E+15	6.82E+30
15	Compost + landfilling (Ali et al., 2018)	1.61E+14	9.91E+13	1.41E+15	1.72E+30
16	Compost + incineration (Ali et al., 2018)	2.26E+15	9.91E+13	2.65E+14	2.74E+28
17	A: incin. + landfilling (Wang et al., 2018)	1.52E+14	2.27E+14	1.46E+15	1.52E+30
18	B: incin. + conr. paving brick product. (Wang et al., 2018)	4.04E+14	2.27E+14	7.88E+14	3.15E+29
18	c: Incin. + non-burnt wall brick prod. (Wang et al., 2018)	2.59E+14	2.27E+14	1.04E+15	6.67E+29
19	Donation (Eriksson and Spångberg, 2017)	2.01E+13	5.95E+15	5.29E+15	4.38E+29
20	Donation (Eriksson et al., 2015)	1.27E+14	9.69E+15	1.64E+15	6.49E+31
	Error Sum				1.42E+32
	R-squared				0.2527
	Parameter a				1.43E+24
	Parameter b				0.63393944

Table D15: Emergy Donation scenario inventory by considering Eriksson and Spangber (2017) LCA information

Description	Weight	%	Material Weight	Lifespan (yrs)[a]	Unit	% SU / TU[f]	Row Material Annual Input
STEP 1: FOOD BANK COLLECTION SYSTEM							
1.1 Vehicles type and materials							
1 car, Petrol E5, Euro 4, [b,c]	1460			10	kg		
Steel		49	715	10	kg	0.0957	7
Plastic		14	204	10	kg	0.0957	2
Aluminum		13	190	10	kg	0.0957	2
Iron		5	73	10	kg	0.0957	1
1.2 Vehicle use[d]							
Gasoline consumption	9				kg		9
Labor[e]							
1 Driver + 1 Operative	2				Person		2
Services							
Car Price[e]	36426			10		0.0957	349
Gasoline price[g]	18				US$		18

a: Lifespan car in Sweden provided by ACEA: https://www.helgilibrary.com/charts/age-of-car-remained-unchanged-in-sweden-in-2018/

b: Average car weight from EUROPEAN VEHICLE MARKET STATISTICS Pocketbook 2016/17 - https://theicct.org/sites/default/files/publications/ICCT_Pocketbook_2016.pdf

c: Average Car composition from: RICARDO - AEA, 2015 - The potential for mass reduction of passenger cars and light commercial vehicles in relation to future CO2 regulatory requirements

d: (5.5 L/100 km from https://www.statista.com/statistics/792869/fuel-usage-of-gasoline-and-diesel-cars-in-sweden/) /100 = (0.055 L/km) * ((4.2/100)*20 km/day = 0.84 km/day) * (5 days / week) * (52 weeks/yr) = (12.012 ~ 12 L/yr) * (0.740 kg/L gasoline density) = 8.88 kg/yr

e: Average car price 31813 Euros in 2015 * 1,1450 EUR/USD in 2018 (31 December) = 36,426 USD

f: according to Eriksson and Spångberg (2017), the collection round is on average 4.2 km/day. The car involved in the process is not exclusively used for this purpose, therefore, to allocate the resources, were considered the total kilometers annually traveled by the car during the food collection process as a fraction of the average kilometers driver per year in Sweden by a typical medium-size car with kerb weight between 1401 and 1500 kg (https://www.trafa.se/en/road-traffic/driving-distances-with-swedish-registered-vehicles/). Calculation: (4.2 km/ day) * (5 days / week)*(52 week/ year) = 1092 km/year. Average km travelled by a medium size Swedish car in one year equal to 11410 km. Therefore 1092 km/11410 km = 0.0957 (9.57 %)

g: Gasoline price in Sweden in 2016 1.46 US$/L (https://data.worldbank.org/indicator/EP.PMP.SGAS.CD?end=2016&locations=SE&start=1995&view=chart). Calculation = 1.46 $/L* 12.012 L = 18

Table D16: Emergy table donation scenarios Eriksson and Spangberg (2017)

N	Input	Type	Amount	Unit	UEVs	UEV Unit	Emergy (seJ)
1	**Labour**	F	2	Person	1.55E+07	seJ/Person	3.10E+07
4	**Gasoline**	F	9	L	6.18E+12	seJ/kg	5.49E+13
7	**Services**	F	18	$	8.41E+12	seJ/$	1.47E+14
	Total						2.02E+14
	Emi = UEV (seJ/ton)[a]						**2.01E+13**

a: by considering 10.08 tons donated food per year

Table D17: Emergy Donation scenario inventory by considering Eriksson et al. (2015) LCA information.

Description	Weight	%	Material Weight	Lifespan (yrs)[a]	Unit	Row Material Annual Input
Step 1: Cold Room Use						
Electricity Consumption[b]	129				kWh	129
Step 2: Food Bank Collection System						
1.1 Vehicles type and materials						
Pick-up Diesel Truck[c]	1837					
Steel		49	900	10	kg	90
Plastic		14	257	10	kg	26
Aluminum		13	239	10	kg	24
Iron		5	92	10	kg	9
1.2 Vehicle use			8			
Diesel consumption[d]	266				kg	266
Labor	7				person	7
Services						
Vehicle Price	58000			10	US$	5800
Diesel price	499				US$	499
Electricity	27				US$	27

a: Lifespan vehicle in Sweden provided by ACEA: https://www.helgilibrary.com/charts/age-of-car-remained-unchanged-in-sweden-in-2018/

b: 52.3 kWh/ m^3/yr : 365 day = x : 300 day -> x = (52.3 kWh m^3 yr / 365 day) * 300 days = 42.98 kWh/ m^3 /yr *3 (density 250 kg m^3) = 128.95 kWh/yr (data from Evans et al. (2014)

c: Average vehicle weight from EUROPEAN VEHICLE MARKET STATISTICS Pocketbook 2016/17 - https://theicct.org/sites/default/files/publications/ICCT_Pocketbook_2016.pdf

d: Diesel consumption: (5 L /100 km from https://www.statista.com/statistics/792869/fuel-usage-of-gasoline-and-diesel-cars-in-sweden/)/100 = (0.05 L/km) * (21.2 km/day from Eriksson et al., 2015)*(300 day/year from Eriksson et al., 2015) = 318 L/ year * (0.835 kg/L European diesel density https://dieselnet.com/standards/eu/fuel_reference.php) = 265.53 kg/ year

Donated food = (700 kg/day) * (300 day/years from Eriksson et al., 2015) = (210000 kg/yr) * 10^{-3} ton/kg = 210 tons/yr

Diesel price Sweden Eur/L = 1.37; Total price = 1.37 Eur/L * 318 L/yr * 1.1450 eur/USD = 498.83 $/yr

Electricity price Sweden 2015- 0.1851 Eur/ kWh = 128.95 kWh/yr * 0.1851 Eur / kWh * 1.1450 eur/USD

Table D18: Emergy Table Donation Scenario Eriksson et al. (2015)

N	Input	Type	Amount	Unit	UEVs	UEV Unit	Emergy (seJ / yr)
1	**Labor**	F	7	Person	1.55E+07	seJ/Person	1.09E+08
2	**Electricity**	F					0.00E+00
3	**Iron**	F	9		1.09E+12	seJ/kg	9.97E+12
4	**Steel**	F	90		2.01E+12	seJ/kg	1.81E+14
5	**Services**	F	6326		3.58E+12	seJ/$	2.27E+16
6	**Diesel Fuel**	F	266		5.99E+12	seJ/kg	1.59E+15
7	**Plastic**	F	26		7.45E+12	seJ/kg	1.92E+14
8	**Aluminum**	F	24		8.60E+13	seJ/kg	2.05E+15
	Total						2.67E+16
	UEV (EMI)[a]					**seJ/ton**	**1.27E+14**

a: by considering 210 ton / yr donated food

Table D19: Saved emergy calculation of donation Scenarios. Original UEVs from literature available in table D10.

		UEV Saved Emergy Donation Scenarios this thesis				
		%	%	UEV (seJ/kg)	conv. UEV (seJ/kg)	Saved (Recov.) EMS (seJ/t)
1	**Tomato**	35.8	62.11	1.22E+13	7.55E+12	7.55E+15
2	**Orange**	13.72	23.80	1.46E+12	3.47E+11	3.47E+14
3	**Potato**	8.12	14.09	2.13E+12	3.00E+11	3.00E+14
	Average	57.64	100.00	5.25E+12	8.20E+12	8.20E+15
		UEV Saved Emergy (M. Eriksson, J. Spångberg (2017)				
		real %	weighted %	UEV (seJ/kg)	Conv. UEV (seJ/kg)	Saved (Recov.) EMS (seJ/t)
1	**Tomato**	20	20	1.22E+13	1.22E+13	1.22E+16
2	**Orange**	20	20	1.46E+12	1.46E+12	1.46E+15
3	**Potato**	20	20	2.13E+12	2.13E+12	2.13E+15
4	**Bell Pepper**	20	20	1.28E+13	1.28E+13	1.28E+16
5	**Banana**	20	20	1.23E+12	1.23E+12	1.23E+15
	Total	100	100			
	Average					5.95E+15
		UEV Saved Emergy (Eriksson et al., (2015)				
		real %	weighted %	UEV (seJ/kg)	Conv. UEV (seJ/kg)	Saved (Recov.) EMS (seJ/t)
1	**Bananas**	20	20	1.23E+12	1.23E+12	1.23E+15
2	**Beef**	20	20	1.58E+13	1.58E+13	1.58E+16
3	**lettuce**	20	20	1.49E+13	1.49E+13	1.49E+16
4	**Grilled chicken (eggs)**	20	20	5.52E+12	5.52E+12	5.52E+15
5	**Bread (Corn, grain)**	20	20	1.10E+13	1.10E+13	1.10E+16
	Total	100	100			
	Average					9.69E+15

Appendix E: Economic Costs Calculation

Table E1: Calculation costs in Donation Scenarios:

Type	Number	Minimum wage[b]	C.F. Wage[c]	Months	Total
Operatives	18	954	1	13	**223236**
supervisors	1	954	6	13	**74412**
Total wages (BRL)					**297648**
Machines cost[a] (BRL)					**161408**
Total Cost (BRL)					**459056**
Total cost per t OBP (BRL)					**12**

a: 41,707 USD (calculation available in table D6)* 3.87 BRL per USD on 31/12/2018 = 161,408 BRL.

b: minimum wage in Brazil in 2018

c: wages are expressed as a multiple of minimum wage.

Table E2: calculation costs Biorefinery Scenarios:

Type	Number	Minimum wage	C.F. Wage	Months	Total
Operatives	34	954	1	13	421668
drivers	3	954	2.5	13	93015
supervisors	2	954	6	13	148824
Total wages					663507
Machines[a]					1956586
Total Cost (BRL / yr)					**2620093**
Total cost - Gain biomethane sold[b] **(BRL / yr)**					**1257536**
Total cost per ton / OBP (BRL)					**33**

a: 505,578 USD (calculation available in table D6)* 3.87 BRL per USD on 31/12/2018 = 1,956,586 BRL/year

b: Biomethane production: 1,135,464 m^3 / yr * 1.2 BRL / m^3 (From ensaioenergetico: https://ensaioenergetico.com.br/o-preco-de-equilibrio-do-biometano-no-estado-do-rio-de-janeiro-a-ineficacia-da-politica-estadual-de-gas-natural-renovavel/) = 1,362,557 BRL / yr

Appendix F: UEV polyacrylamide estimation

Polyacrylamide UEV calculation: from propylene production + acrylonitrile production + polyacrylamide production. Data source: Ecoinvent, Propylene production, RoW, (4), 1 kg; Ecoinvent, SOHIO process. RoW, (7), Acrylonitrile, 1 kg; Ecoinvent, Polyacrylamide production, GLO, (2), 1 kg. The three most important inputs were considered (~ 99% of total).

Propylene production:

1. Crude oil: (1.56 E+05 SeJ/J from Brown et al. 2011)*(12E+24 seJ*yr^{-1} / 15.2E+24 seJ/yr^{-1} Brown and Ulgiati, 2011)*(44.5 MJ/ kg oil from world nuclear association)*(0.937 kg) = 5.14E+12 SeJ
2. Natural Gas (1.78 E+05 SeJ/J from Brown et al. 2011)*(12E+24 seJ*yr^{-1} / 15.2E+24 seJ/yr^{-1} Brown and Ulgiati, 2011)*(52.5 MJ/kg methane from world nuclear association)*(0.584 m3 * 0.714 kg/ m3 methane) = 3.08E+12 seJ
3. Coal (hard) (1.78 E+05 SeJ/J from Brown et al. 2011)*(12E+24 seJ*yr^{-1} / 15.2E+24 seJ/yr^{-1} Brown and Ulgiati, 2011)*(25.0 MJ/kg coal from world nuclear association)*(0.0437 kg) = 1.14 E+11 SeJ
4. Total Emergy used to produce 1 kg polypropylene output = (5.14E+12 seJ) + (3.08E+12 seJ) + (1.14 E+11 seJ) = 8.33 E+12 seJ/kg)

Acrylonitrile production

1. Ammonia: (3.80E+09 seJ/g from Odum, 1996)*(12.00E+24 seJ*yr^{-1} / 9.44E+24 seJ*yr^{-1} Odum, 1996 baseline)*(103g / kg)*(0.374 kg) = 1.42E+12 SeJ
2. Electricity: (1.47E+05 seJ/J from Giannetti et al., 2015)*(12.00E+24 seJ*yr^{-1} /15.2E+24 seJ*yr^{-1} Brown and Ulgiati 2010 baseline)*(3.6E+06 J/kWh) = 1.16 E+11 SeJ)
3. This study (8.33E+12 seJ/kg)*(0.833 kg) = 7.36E+12 seJ
4. Total emergy used to produce 1 kg Acrylonitrile = (1.42E+12 seJ) + (1.16 E+11 seJ) + (7.36E+12 seJ) = 8.89 E+12 seJ / kg

Polyacrylamide production

1. Acrylonitrile: this study (8.89E+12 seJ/kg)*(0.747 kg) = 6.64E+12 seJ
2. Electricity: (1.47E+05 seJ/J from Giannetti et al., 2015)*(12.00E+24 seJ*yr^{-1} /15.2E+24 seJ*yr^{-1} Brown and Ulgiati 2010 baseline)*(3.6E+06 J/kWh)
3. Water: (1.82E+04 seJ*J-1)*(12E+24 seJ*yr^{-1} / 9.44E+24 seJ*yr^{-1} Odum, 1996 baseline)/(4.94 J/g Gibbs free Energy, from Odum, 1996)*(103 g/kg)
4. Total emergy used to produce 1 kg Polyacrylammide = (6.64E+12 seJ) + (1.39E+11 seJ) + (1.18E+06 seJ) = 6.78E+12 seJ/kg