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Departamento de Ambiente e Ordenamento

Tamíris Pacheco da Costa

Avaliação de Ciclo de Vida da produção de energia a partir de resíduos florestais

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Tese apresentada à Universidade de Aveiro para cumprimento dos requisitos necessários à obtenção do grau de Doutor em Ciências e Engenharia do Ambiente, realizada sob orientação científica da Doutora Ana Cláudia Relvas Vieira Dias, Equiparada a Investigadora Auxiliar do Departamento de Ambiente e Ordenamento da Universidade de Aveiro e coorientação da Doutora Paula Sofia Gil Neto Quinteiro, Equiparada a Investigadora Júnior do Departamento de Ambiente e Ordenamento da Universidade de Aveiro.

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“O progresso é medido pela velocidade com que destroem as condições que sustentam a vida”.
George Monbiot.

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palavras-chave Avaliação do ciclo de vida, impacte ambiental, resíduos de biomassa florestal, produção de bioenergia, valorização das cinzas.

resumo A produção de bioenergia a partir de resíduos florestais tem aumentado nos últimos anos em Portugal devido às preocupações relacionadas com as alterações climáticas e os incêndios florestais. No entanto, os potenciais impactes ambientais associados à sua produção devem ser quantificados para suportar a tomada de decisão. Esta quantificação pode ser realizada com recurso à avaliação de ciclo de vida (ACV), uma metodologia que avalia todo o ciclo de vida de um produto ou processo (desde a extração das matérias-primas até ao seu fim de vida), permitindo identificar as etapas e processos mais significativos ao longo do seu ciclo. Atualmente, existe um número limitado de estudos de ACV relacionados com a produção de bioenergia a partir de resíduos florestais. Além disso, esses estudos geralmente excluem a etapa de fim de vida (gestão das cinzas) ou consideram apenas a deposição de cinzas em aterro, excluindo as alternativas de valorização. Complementarmente, dadas as restrições à disponibilidade de resíduos florestais, é necessário avaliar o melhor uso desses resíduos do ponto de vista ambiental. Assim, esta tese pretende contribuir para aumentar o conhecimento do desempenho ambiental do setor da bioenergia em Portugal. As duas tecnologias de combustão mais representativas para a produção de eletricidade a partir de resíduos florestais (fornalha de grelha e leito fluidizado) foram avaliadas e comparadas através da metodologia de ACV ao longo de todas as etapas da cadeia de valor, nomeadamente a gestão florestal, recolha, processamento, transporte, conversão em energia e fim de vida. Além disso, a ACV é aplicada na etapa de fim de vida da cinza gerada durante a combustão dos resíduos florestais para avaliar duas alternativas de valorização (materiais de construção e correção do solo) e comparar com a deposição da cinza em aterro. Nesse sentido, foram incluídos vários cenários para identificar a opção mais eficiente do ponto de vista ambiental. Adicionalmente, é usada a ACV consequencial para avaliar a melhor opção de valorização dos resíduos florestais, nomeadamente eletricidade, calor ou biocombustíveis, em comparação com um cenário de referência em que os resíduos são deixados no solo florestal e a energia é produzida a partir de combustíveis fósseis. Os resultados obtidos mostram que o leito fluidizado apresenta menores impactes ambientais do que a fornalha de grelha para todas as categorias de impacte analisadas. No que respeita à valorização das cinzas nos materiais de construção, todos os cenários avaliados apresentam um impacte ambiental inferior ao da deposição em aterro em todas as categorias de impacte. No entanto, a valorização das cinzas para correção da qualidade do solo apresenta maiores impactes ambientais do que o aterro para algumas categorias de impacte, indicando que potencialmente pode aumentar a quantidade de poluentes no solo. Finalmente, os resultados da ACV consequencial indicam que a cogeração de eletricidade e calor é a melhor opção para a valorização dos resíduos florestais, embora para algumas categorias de impacte o seu desempenho ambiental só é melhor que o do cenário de referência em determinadas condições.

keywords Bioenergy production, environmental impact, forest biomass residues, life cycle assessment, woody biomass ash valorisation.

abstract The production of bioenergy from forest biomass residues has been increasing in the last years in Portugal, mainly as a consequence of concerns related to climate change and forest fires. However, the potential environmental impacts associated with its production should be quantified to support decision-making. This quantification can be performed by using life cycle assessment (LCA), a methodology that evaluates the entire life cycle of a product or process (from the extraction of the raw materials until its end-of-life), allowing to identify the most significant stages and processes along the life cycle. Currently, there is a limited number of LCA studies concerning the production of bioenergy from forest biomass residues. In addition, those studies usually exclude the end-of-life stage (ash management) or only consider ash disposal in landfill, disregarding the valorisation alternatives. Furthermore, given the constraints on forest residues availability, it is important to assess the best use for these residues from an environmental perspective. Therefore, this thesis aims to contribute to increase the knowledge of the environmental performance of the bioenergy sector in Portugal. The two most representative combustion technologies for electricity production from forest biomass residues (grate furnace and fluidised bed) are assessed and compared using LCA throughout all stages of the value chain, namely, forest management, collection, processing, transportation, energy conversion and end-of-life. Moreover, LCA is applied to the end-of-life stage of the ash generated during the combustion of forest biomass residues to evaluate two valorisation alternatives (construction materials and soil amelioration) and compare with ash landfilling. Various scenarios are included in order to identify the most efficient option from an environmental point of view. Additionally, a consequential LCA is used to evaluate the best valorisation option for the forest biomass residues, namely, electricity, heat or bioethanol, in comparison with a baseline that entails leaving the residues in the forest soil and energy is produced from fossil fuels. The results show that fluidised bed presents smaller environmental impacts than grate furnace for all impact categories analysed. Regarding ash valorisation in construction materials, all scenarios assessed had a lower environmental impact than landfilling in all the impact categories. However, the valorisation of ash for soil amelioration presents higher environmental impacts than landfilling for some impact categories, indicating that it can potentially increase the amount of pollutants in the soil. Finally, the results of the consequential LCA indicate that the best use of the forest biomass residues is cogeneration of electricity and heat, but for some impact categories it would only perform environmentally better than the baseline under particular conditions.

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List of Acronyms, Abbreviations and Symbols

AC	Acidification
Al	Aluminium
As	Arsenic
B	Boron
Ba	Barium
BF	Bottom ash generated in a fluidised bed
BG	Bottom ash generated in a vibrating grate
Br	Bromine
C	Carbon
Ca	Calcium
CC	Climate change
CCE	Calcium carbonate equivalent
$CCE_{woody\ biomass\ ash}$	Calcium carbonate equivalent of woody biomass ash
$CCE_{liming\ product}$	Calcium carbonate equivalent of liming product
Cd	Cadmium
CH_4	Methane
Cl	Chlorine
CML	Centre of Environmental Science of Leiden University
CO	Carbon monoxide
Co	Cobalt
CO_2	Carbon dioxide
Cr	Chromium
Cu	Copper
EPA	Environmental Protection Agency
EPS	Environmental Priority Strategies
EU	European Union
FD	Fossil depletion
Fe	Iron
FEc	Freshwater ecotoxicity
FEu	Freshwater eutrophication
FF	Fly ash generated in a fluidised bed

FG	Fly ash generated in a vibrating grate
FU	Functional unit
GDP	Gross domestic product
GHG	Greenhouse gas
GLO	Global
GVA	Gross value added
H_2	Hidrogen
H_2O	Water
Hg	Mercury
HT	Human toxicity
HTc	Human toxicity carcinogenic
HTnc	Human toxicity non-carcinogenic
ILCD	International Reference Life Cycle Data System
In	Indium
IPCC	Intergovernmental Panel on Climate Change
ISO	International Organization of Standardization
JRC	Joint Research Centre
K	Potassium
k	Residue decomposition rate
K_2O	Potassium oxide
KCl	Potassium chloride
LCA	Life cycle assessment
LF	Landfilling
LM	Landfarming
LU	Land use
M(0)	Initial mass of residue on the soil
M(t)	Fraction of residue on the soil at time *t*
MAT	Marine aquatic toxicity
MEu	Marine eutrophication
MFD	Mineral and fossil resource depletion
Mg	Magnesium
$m_{liming\ product}$	Mass of liming product
Mn	Manganese

Mo	Molybdenum
$m_{woody\ biomass\ ash}$	Mass of woody biomass ash
N	Nitrogen
Na	Sodium
NH_3	Ammonia
Ni	Nickel
NMVOCs	Non-methane volatile organic compounds
N_2O	Nitrous oxide
NO	Nitric oxide
NO_2	Nitrogen dioxide
NO_3^-	Nitrate
NOx	Nitrogen oxides
Mg	Magnesium
OD	Ozone depletion
P	Phosphorus
P_2O_5	Phosphorus pentoxide
Pb	Lead
PM	Particulate matter
PM_{10}	Particulate matter 10 µm or less
$PM_{2.5}$	Particulate matter 2.5 µm or less
PNAER	National Renewable Energy Action Plan
PO_4^{3-}	Phosphate
POF	Photochemical ozone formation
RCM	Resolution of the Council of Ministers
S	Sulfur
Sb	Antimony
Sc	Scandium
SCI	Science citation index
SCR	Selective catalytic reduction
SETAC	Society of Environmental Toxicology and Chemistry
Si	Silicon
SO_2	Sulfur dioxide
SO_3	Sulfur trioxide

SO_4^{2-}	Sulphate
SOx	Sulfur oxides
Sn	Tin
Sr	Strontium
T	Transport
t	Time
TET	Terrestrial toxicity
Ti	Titanium
TSP	Triple superphosphate
USA	United States of America
V	Vanadium
Zn	Zinc

Chapter 1

General introduction

1.1 Background and motivation

Energy plays a crucial role in human well-being and economic development of a country. However, the increasing demand of energy puts significant pressure on environment and ecosystems, which remains a major challenge (IEA, 2019). Since the industrial revolution, fossil fuels have been the most widely used source of energy worldwide (WEC, 2016; Zou et al., 2016), but there are several negative factors associated with their use. They are non-renewable, limited in supply and distributed unevenly throughout the world, causing dependency of non-producer countries on energy imports from producer countries. In addition, the combustion of fossil fuels contributes to climate change due to greenhouse gas (GHG) emissions to the atmosphere (IPCC, 2017). These issues represent a strong motivation for the use of environmentally friendly and efficient renewable energy resources, encouraging the development of global policies to support the transition to a renewable energy resource economy (Muylle et al., 2015). Regarding the European Union context, the Directive 2018/2001 proposes that the Member States collectively achieve the share of energy from renewable sources in the Union's gross final consumption of energy of at least 32 % in 2030, including forest biomass (EU, 2018). Among the sources of forest biomass, forest residues from logging activities have great potential for producing thermal and electric energy (Gomes, 2006; Röser et al., 2008). The use of forest biomass residues is more advantageous than the use of fossil fuels in relation to carbon dioxide (CO_2) emissions, since a neutral CO_2 balance can be considered, i.e., CO_2 assimilated during biomass growth is equal to the CO_2 that is emitted to the atmosphere during the combustion of biomass residues to produce energy (Laganière et al., 2017; Searchinger et al., 2018; Zanchi et al., 2012). In addition, unlike other renewable energy sources, forest biomass can be stored, allowing a constant supply and does not dependent on climatic conditions, such as wind energy and hydro-energy (WEC, 2016). However, its intensive use can lead to excessive removal of carbon and nutrients from forest soils (Herbert et al., 2016; Mateos and Ormaetxea, 2019; Rafael et al., 2015), which can deteriorate soil physical (e.g. soil water holding capacity), chemical (e.g. nutrient cycling) and biological (e.g. decomposition of organic residues) properties (Bot and Benites, 2005; Tamminen et al., 2012).

Forest occupies 35 % of the Portuguese territory, making Portugal a country with high potential for the exploitation of forest biomass, including forest biomass residues (CAM, 2013; IMF, 2014). The Portuguese government has launched several policies over the last years to promote the production of energy from forest biomass residues in order to diversify the power mix and prevent forest fires (ME, 2017; RCM, 2013, 2006a, 2003). As a consequence, several power plants fuelled by forest biomass residues have been commissioned in the last few years in Portugal, intensifying the pressure for the exploitation of forest biomass residues (E2p, 2017; Nunes, 2015). Besides, it is projected the concession of additional installed capacity (60 MWe) for the construction of new power plants supplied by biomass residues (ME, 2017). Therefore, the evaluation of the potential environmental impacts from the current and future use of forest biomass residues as bioenergy resource is important to support decision-making by different actors involved in the value chains from forest management up to ash management, as well as governmental bodies (from the forest, energy and waste management sectors).

Life cycle assessment (LCA) is a tool that quantifies the environmental impacts of a product, a process, or a service through its life cycle (ISO, 2006a). The LCA has been promoted as an environmental assessment method either for government policies definition, or to support environmental management, improve productive processes or increase competitiveness in the international market (EU, 2010; UNEP/SETAC, 2017). Therefore, LCA has been applied to a great variety of products, processes and services. However, there are only a few LCA studies focusing on the environmental impacts of electricity production using forest biomass residues (e.g. Loução et al., 2019; Tagliaferri et al., 2018; Thakur et al., 2014) and some of them are limited to the assessment of the carbon footprint. Moreover, in Portugal, the dedicated power plants that use forest biomass residues have either fluidised bed furnaces or grate furnaces, but none of the mentioned studies compare the environmental performance of both technologies. Only Djomo et al. (2015) performs such a comparison but for cogeneration plants in the Belgian context and considering only the carbon footprint.

Regarding the end-of-life of woody biomass ashes generated during forest biomass combustion, it is currently poorly studied, being usually excluded from the LCA studies or, when included, only landfilling is considered (Djomo et al., 2015; González-García and Bacenetti, 2019; Whittaker et al., 2016). In addition to the environmental concerns related to woody biomass ash landfilling, the competition for limited space in disposal sites and tightening of regulations can cause significant economic burden for the energy sector (Iyer and Scott, 2001). These factors have prompted researchers to look for alternative usages for woody biomass ash, as for example the incorporation in construction materials and the application in

forest soils for fertilisation and liming (Brod et al., 2012; Hammar et al., 2019; Ukrainczyk, 2016; Wang et al., 2008). The potential impacts related to woody biomass ash valorisation need to be holistically understood because despite the positive effects of the valorisation alternatives (i.e. substitution of resources), it is also important to assess the potential environmental impacts that this practice may cause. Therefore, an LCA of the different options of valorisation is necessary to support decision-making of ash management by identifying negative and positive impacts. However, such a study is lacking for the Portuguese context.

Furthermore, given the constraints on forest residues availability and potential customers for these residues (e.g. electricity, heat and biofuels), it is imperative to evaluate the best use from an environmental perspective to support decision-making. However, such a study is missing for the Portuguese conditions. Even the LCA studies carried out for other countries are mostly limited in scope by focusing only one bioenergy service (e.g., Karlsson et al., 2014; Lindholm et al., 2011; Sandilands et al., 2009; Thakur et al., 2014). In addition, the consequences that may occur in the market from an increasing demand of bioenergy from forest residues are often disregarded (e.g., Djomo et al., 2015; Falano et al., 2014; Liang et al., 2017; Whittaker et al., 2016). This assessment calls for a consequential LCA, which describes how environmental impacts change in response to potential decisions along a product's life cycle (Zamagni et al., 2012). In consequential LCA, the system boundaries are expanded to comprise the activities contributing to any resultant changes. This enhances the complexity of the models and often means that a consequential LCA will also include additional economic concepts such as marginal costs and market trends and will look at impacts over a wider temporal and geographical range (Ekvall and Weidema, 2004).

1.2 Objectives of the thesis

The general objective of this thesis is to evaluate the environmental sustainability of using forest biomass residues for bioenergy production in Portugal, from a life cycle perspective, using LCA methodology. The specific objectives are as follows:

1) to evaluate the potential environmental impacts and associated hotspots, resulting from electricity production in Portugal using forest biomass residues from logging activities, considering the two types of technologies currently applied: fluidised bed furnaces and grate furnaces. For this purpose, a comparative LCA was carried out for the two systems considering the following stages in the system boundaries: forest

management, logging residues collection, processing and transportation and energy conversion.

2) to assess the potential environmental impacts of different end-of-life management alternatives for the woody biomass ashes generated from electricity production based on forest biomass residues. The current woody biomass ash management in Portugal is mainly the disposal in sanitary landfills. However, these ashes can be incorporated in construction materials or can be recirculated back to forest soils leading to the return of valuable nutrients (such as phosphorus and potassium) to forest ecosystems and counteract soil acidification. Therefore, four valorisation alternatives by incorporation in construction materials were evaluated, namely, incorporation in cement mortar, adhesive mortar, concrete blocks and bituminous asphalt. In addition, two valorisation alternatives for forest soil amelioration were evaluated: soil liming and soil fertilisation. In both cases, a comparison with woody biomass ash landfilling was undertaken. In this sense, the evaluation of valorisation alternatives aims to identify the most effective option to achieve the best environmental performance.
3) to evaluate the best use of forest biomass residues at national level, from an environmental point of view, in order to satisfy projected needs for three bioenergy services, namely, electricity, heat and bioethanol. Thus, the environmental consequences of changing the forest biomass residues management from the reference (left on field as soil improver) to the future bioenergy services were evaluated through a consequential LCA.

Overall, this work intends to provide knowledge regarding the bioenergy production from forest biomass residues in Portugal, contributing to support decision-making concerning bioenergy production technologies and strategies, as well as woody biomass ash management, towards a more sustainable development of this sector.

1.3 Structure of the thesis

The present thesis includes modified versions of published or submitted peer-reviewed papers from Science Citation Index (SCI) journals. The paper modifications consist in: (1) harmonisation of literature references, as the papers were submitted/published to different journals which use different reference styles and (2) format harmonisation.

The thesis is organised in 7 chapters. Chapter 1 presents the main motivations, the objectives, the structure of the thesis and the published work developed in the context of this thesis.

Chapter 2 presents the state-of-the-art regarding the environmental sustainability assessment of production of bioenergy from forest biomass and management of woody biomass ash generated from the combustion of forest residues. The relevance of the Portuguese forest, electricity sectors and the policies related to the bioenergy production from forest biomass residues are also addressed. Furthermore, an introduction to the LCA methodology, describing the main modelling and methodological issues is also presented.

Chapter 3 presents a comprehensive LCA of the electricity production in Portugal using eucalypt logging residues, comparing the environmental performance associated with two alternative combustion technologies: grate furnaces and fluidised bed furnaces.

Chapter 4 evaluates and compares the trade-offs between the environmental impacts and the benefits of woody biomass fly and bottom ash valorisation in different construction materials, namely, cement mortars, adhesive mortars, concrete blocks and bituminous asphalts, in contrast to landfilling, which is currently the main management alternative for the ashes.

Chapter 5 evaluates and compares the trade-offs between the environmental benefits and impacts of woody biomass ash valorisation in forest soil for two purposes: soil liming and soil fertilisation, comparatively to landfilling.

Chapter 6 evaluates the best use of the forest biomass residues to satisfy three bioenergy services, namely, electricity, heat and bioethanol from an environmental perspective through a consequential LCA.

Chapter 7 presents the overall conclusions of this thesis and provides suggestions for future research regarding the bioenergy sector based on forest biomass residues.

1.4 Scientific work resulting from this doctoral thesis

This work is based on the following scientific papers in peer-reviewed SCI journals:

Chapter 3 published as: da Costa, T. P., Quinteiro, P., Tarelho, L. A. C., Arroja, L., Dias, A. C., 2018. Environmental impacts of forest biomass-to-energy conversion technologies: Grate furnace vs. fluidised bed furnace. Journal of Cleaner Production. 171, 153–162. https://doi.org/10.1016/j.jclepro.2017.09.287.

Chapter 4 published as: da Costa, T. P., Quinteiro, P., Tarelho, L. A. C., Arroja, L., Dias, A. C., 2019. Environmental assessment of valorisation alternatives for woody biomass ash in construction materials. Resources, Conservation and Recycling. 148, 67–79. https://doi.org/10.1016/j.resconrec.2019.04.022.

Chapter 5 published as: da Costa, T. P., Quinteiro, P., Tarelho, L. A. C., Arroja, L., Dias, A. C., 2019. Life cycle assessment of woody biomass ash for soil amelioration. Waste Management. 101, 126-140. https://doi.org/10.1016/j.wasman.2019.10.006.

Chapter 6 submitted as: da Costa, T. P., Quinteiro, P., Arroja, L., Dias, A. C., 2019. Environmental comparison of forest biomass residues application: electricity, heat and biofuel. Renewable and Sustainable Energy Reviews.

The following oral presentations in national and international conferences also resulted from this doctoral thesis:

1. da Costa, T. P., Quinteiro, P., Tarelho, L. A. C., Arroja, L., Dias, A. C., 2017. Environmental impacts of biomass-to-energy conversion technologies: grate boilers and fluidised bed boilers. 8th International Conference on Life Cycle Management, 3-6 September 2017. Luxembourg, Luxembourg.

2. da Costa, T. P., Quinteiro, P., Tarelho, L. A. C., Arroja, L., Dias, A. C., 2018. Uso da avaliação de ciclo de vida na tomada de decisões para uma produção de energia mais sustentável: estudo de caso dos resíduos florestais em Portugal. Conferência Internacional de Ambiente em Língua Portuguesa/XI Conferência Nacional do Ambiente, 8-10 May 2018. Aveiro, Portugal.

3. da Costa, T. P., Quinteiro, P., Tarelho, L. A. C., Arroja, L., Dias, A. C., 2018 Management of biomass ash waste from energy production in building materials. Scientific Meeting on Circular Economy and Waste Management, 26 June 2018. Coimbra, Portugal.

4. da Costa, T. P., Quinteiro, P., Tarelho, L. A. C., Arroja, L., Dias, A. C., 2018. Life cycle assessment of woody biomass bottom ash valorisation in bituminous asphalt. 13th International chemical and biological engineering conference, 2-4 October 2018. Aveiro, Portugal.

5. da Costa, T. P., Quinteiro, P., Tarelho, L. A. C., Arroja, L., Dias, A. C., 2018. Environmental assessment of woody biomass ash valorisation in cement mortars. 8th Life Cycle Assessment Conference, 7-8 November 2018. Lille, France.

6. da Costa, T.P., Quinteiro, P., Tarelho, L. A. C., Arroja, L., Dias, A.C., 2018. Life cycle assessment of energy production from forest resources. Research Summit, 28-29 November 2018. Aveiro, Portugal.

7. da Costa, T.P., Quinteiro, P., Tarelho, L. A. C., Arroja, L., Dias, A.C., 2019. Life cycle assessment of woody biomass ash valorisation in construction materials. 10th International Conference on Industrial Ecology, 7-11 July 2019. Beijing, China.

Award received:

The work entitled "Environmental impacts of biomass-to-energy conversion technologies: grate boilers and fluidised bed boilers" was awarded with the prize “From PhD to apps” in the 8th International Conference on Life Cycle Management, which took place on 3-6 September 2017 in Luxembourg. (http://uaonline.ua.pt/pub/detail.asp?c=51809&lg=pt).

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Chapter 2

Literature review

2.1 Portuguese forest sector

The forest sector is important for Portugal's socio-economic development. Besides the value generated from the exploitation of honey, fruits, mushrooms or herbs and the value associated with grazing, hunting, soil and water resources protection and landscape protection, the forest provides raw materials for the pulp and paper industry and to produce bioenergy (DGAE, 2017).

The different services provided by the Portuguese forest can reach 2.6 thousand million euros per year, as well as contributes with around 10 % of the Portuguese exportations (ICNF, 2019). The gross value added (GVA) of the Portuguese forest sector represented 2.1 % of the national gross domestic product (GDP), positioning Portugal in third place in the European Union (EU), only after Finland (5.5 %) and Sweden (3.6 %) (Reboredo, 2014). In addition, the Portuguese forest sector gives direct employment to about 12,000 workers and indirectly to about 115,000 workers (2 % of the active population) (CES, 2017).

Portugal is largely covered by forest, which occupies around 3.18 million hectares, corresponding to 35 % of the mainland territory (ICNF, 2013), a percentage slightly lower than the average total forest area in the EU countries (37.9 %) (Forest Europe, 2015). The forest area in Portugal has been decreasing in the last decades. From 1990 to 2015, Portugal lost 254 thousand hectares of forest area, equivalent to a decrease of 7.4 % (Forest Europe, 2015). Portugal's losses have been mainly due to deforestation and to the devastating forest fires that have consumed several hundred thousand hectares of forest in the last years (Forest Europe, 2015; Tedim et al., 2015). Figure 1 shows the changes in forest distribution in Portugal over the last decades.

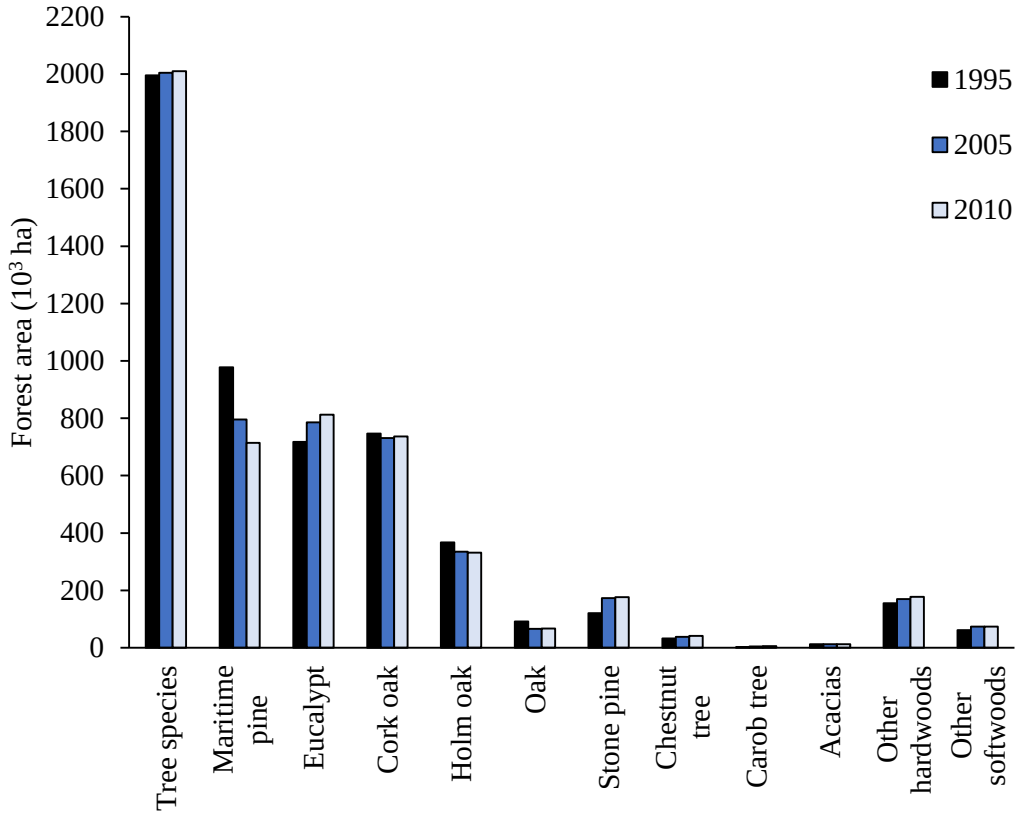

Figure 1 - Evolution of the total forest area by tree species in Portugal between 1995 and 2010. Source: adapted from ICNF (2013).

In 2010, the eucalypt was the species that occupied the largest forest area, covering 25.8 % of the total forest area in Portugal, followed by cork oak (23.4 %) and maritime pine (22.7 %) (ICNF, 2013). The total eucalypt area increased 13 % between 1995 and 2010. The considerable expansion of the eucalypt plantations in recent years reflects major competitive advantages of eucalypt especially in terms of productivity relatively to other tree species (ICNF, 2013). Cork oak has also importance due to the production of cork stoppers for the wine industry, products for the construction sector (such as flooring, insulation and coverings) and other cork products (Demertzi, 2011), while maritime pine is required to sawmill and to wood-based panels industry (Alía and Martín, 2003).

2.2 Forest biomass residues

Forest biomass residues encompasses a great diversity of ligno-cellulosic materials resulting from the forest management, wood-based industry (pulp and paper, wood panels, wood lamination, sawmills, etc.), urban tree trimmings and discarded wood after use (Ackom et al., 2010; Thoma et al., 2018). Depending on its origin, forest biomass residues can be

differentiated into primary residues, secondary residues and tertiary residues (Hoogwijk et al., 2003):

- Primary forest biomass residues arise directly from forest management activities, such as stand tending and logging (Chitawo et al., 2018; Dias et al., 2007). The residues from stand tending result from cleaning, selection of coppice stems, thinning and pruning (Dias et al., 2007). The logging residues includes branches, tree tops and leaves resulting from the felling, limbing and bucking process (Dias et al., 2007).
- The secondary forest biomass residues are related to industrial processing of wood consisting in bark, chips, shavings, sawdust and wood dust (Thiffault et al., 2018). In the pulp and paper industry, forest biomass residues include also black liquor resulting from wood cooking (Mladenov and Pelovski, 2015).
- Tertiary forest biomass residues includes tree trimmings from the urban environment and all kind of residues produced after the use of the manufactured wood-based goods by the consumer, such as the forest residues from construction, renovation and demolition debris, like unusable pallets or furniture (Röser et al., 2008).

The primary forest biomass residues are a potential important source of forest fuel, but their valorisation is strongly dependent on theoretical, technical and socioeconomic potential of collection, as schematically shown in Figure 2.

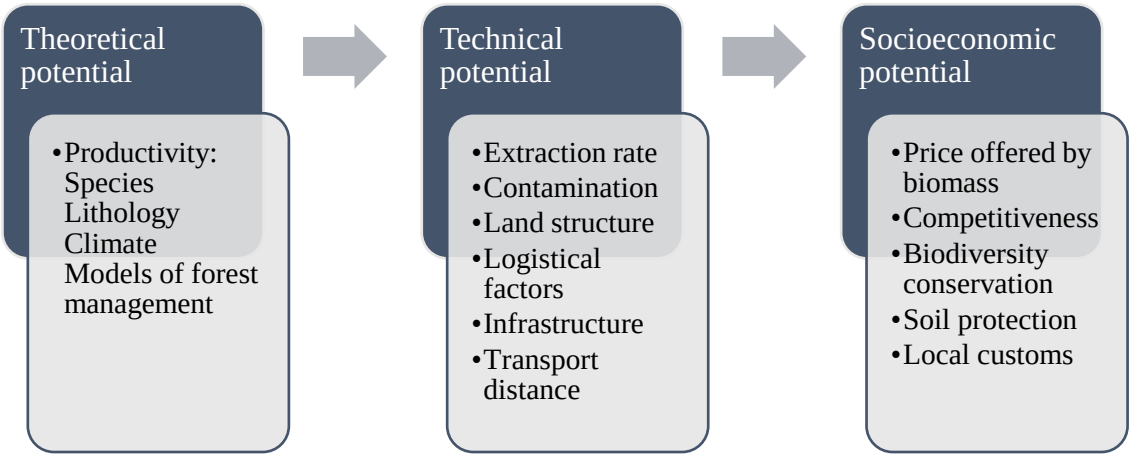

Figure 2 - Restrictions considered during the valorisation of primary forest biomass residues. Source: adapted from ACHAR (2013).

Theoretical potential for forest biomass residues valorisation refers to the forest biomass residues productivity that depends on the forest species, the site lithology, climate and models of forest management (Benavides et al., 2009; Ciccarese et al., 2014). It does not include

technological constraints, such as the collection of residues from inaccessible places, which is part of the technical potential.

Technical potential of valorisation is defined as the fraction of the theoretical potential that current technologies allow to exploit (Orecchini and Naso, 2012). This potential considers a set of factors that limit the exploitation of forest biomass residues, such as the extraction rate, the contamination and the moisture of the residues, the land structure and slope, logistical factors, transport distance and lack of infrastructure, notably roads, but also equipment (Hernández et al., 2017; Hetsch, 2008; Simas, 2013).

Another aspect for forest biomass residues valorisation is the socio-economic potential, which refers to the amount of residues that are economically and sustainably able to be used (Karaj et al., 2010). All factors that influence the total cost of forest biomass residues that are supplied to the power plants are included in this category, such as those related to collection, processing and transportation (Karaj et al., 2010). According to Viana et al. (2010), the viable area, from an economic point of view, where power plants can collect and transport the forest biomass residues is within a radius of about 35 km. Environmental standards can also influence the level of collection, e.g. through guidelines for leaving deadwood and habitat trees in the forest, or restrictions on harvesting amounts and techniques (Hetsch, 2008). Moreover, it is recommended that part of the residues be reintegrated in forest soils due to the concerns about soil fertility and nutrient losses (Hetsch, 2008).

There are few recent studies that present the availability of forest biomass residues in Portugal. Viana et al. (2010) estimated the theoretical maximum potential of forest biomass residues from logging of maritime pine and eucalypt at 1.1 Mt/yr of residues. However, the forest residues considered economically exploitable are considerably less, around 0.6 Mt/yr (Viana et al., 2010). Ferreira et al. (2017) presented the amount of forest biomass residues according to biomass type (brushwood, wood, branches, foliage, biomass from burned areas and wood industry). According to these authors, the amount of forest biomass residues available in Portugal can achieve 6.5 Mt/yr, but the amount economically available is approximately 2.2 Mt/yr. A more recent study performed by the Portuguese government estimated the amount of primary forest biomass residues in 3 Mt/yr, including the residues of maritime pine, stone pine, eucalypt, cork oak and other forest species, herbaceous wastes and brushwood (RCM, 2017). The residues from maritime pine and eucalyptus are equal to 0.54 and 0.45 Mt/yr, respectively.

Therefore, it should be noted that there is a great discrepancy between the potential availability and the effective availability of forest biomass residues in Portugal. Only a small portion of these residues have economic viability to be valorised due to the difficult orographic

conditions of a large part of Portugal, the low road network of the forest area and the high costs of extracting and transporting forest biomass residues (DNFF, 2010).

2.3 Contribution of forest biomass residues to the Portuguese electricity sector

The contribution of the different energy sources in the Portuguese electricity mix has experienced modifications through the last decades, observing an increase of renewable sources in recent years (Figure 3). In fact, in 2000, the renewable energy sources represented 30 % of the electricity production mix in Portugal, whereas in 2018 these sources increased for a total of 51 % (APREN, 2018a), which positioned Portugal in a quite favourable scenario in relation to the EU average of 31 % (EC, 2019). Hydropower and wind play a crucial role in the electricity production mix from renewable sources in Portugal, although in the last decade other renewable energy sources, such as forest biomass residues, have increased in importance.

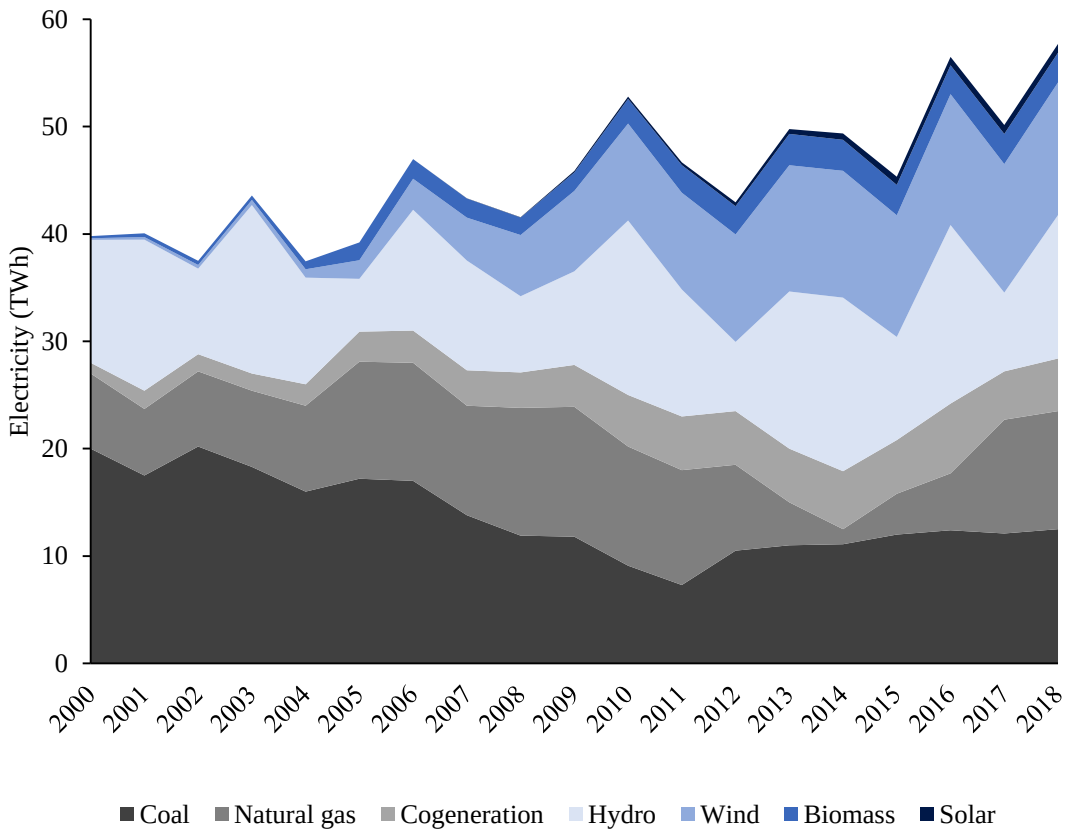

Figure 3 - Indicative trajectory of electricity production in Portugal from 2000-2018. Source: adapted from APREN (2018).

The biomass presented in Figure 3 includes forest residues, agricultural residues, black liquor and municipal solid wastes and was responsible for 10 % of the electricity produced in

Portugal from renewable sources in 2018, which represents 5 % of the total electricity produced in the country. Forest biomass residues was responsible for the production of around 1.5 % of the total electricity in the country in the same year (DGEG, 2018).

The first power plant fuelled by forest biomass residues started operating in 1998, in Vila Velha de Ródão. Initially, it had an installed capacity of 3.5 MW, but in 2007 its capacity was increased to 6 MW, with a consumption higher than 60 kt of forest biomass residues per year (Enes et al., 2007; MADRP, 2005). The second power plant was built in Mortágua in 1999 with an electricity installed capacity of 9 MW and an annual consumption higher than 100 kt of biomass residues per year (CBE, 2007; Netto, 2008).

The electricity installed capacity from forest biomass residues increased in the last years due to strategies implemented by the Portuguese government. One of the strategies taken by the Portuguese government was the elaboration of the Resolution of the Council of Ministers (RCM) 63/2003, which settled a target of 150 MW of installed power by 2010 from forest biomass residues (RCM, 2003).

In 2005, Portugal established the National Strategy for Energy through the RCM 169/2005 (RCM, 2005), which highlighted the significance of the forest biomass residues valorisation, compatible with the industries of wood and pulp and paper, for the sustainable increase of the electricity installed capacity of the country.

In 2006, the National Forest Fire Protection Plan (RCM 65/2006) planned to assure the reduction of the forest fires through structural interventions, surveillance, combat and strategic actions (RCM, 2006a). In the context of this plan, in 2006 was launched a public tendering to build 15 new power plants dedicated to forest biomass residues, with a total of 100 MW of installed capacity, to be located in areas chosen due to high forest resource availability and fire risk (RCM, 2003). The additional allocation of installed capacity promoted an increase of 67 % in relation to the previous target of 150 MW, totalling 250 MW of electricity produced from forest biomass residues (RCM, 2008; Silva, 2016).

In 2010, Portugal submitted the first National Renewable Energy Action Plan (PNAER) to the European Commission. The PNAER highlights the importance of the biomass to produce heat and electricity in the country and proposes to the new power plants in order to achieve the deadline for the construction of new units. This action plan also includes strategies regarding the forest biomass residues supply chain, the optimisation of the forest management and the sustainable increase of biomass production.

Recently, the Decree-Law 64/2017 and the RCM 163/2017 were launched, aiming at promoting the sustainability of the forest, its management and to prevent forest fires in the

country (ME, 2017; RCM, 2017). This decree-law aims to allocate 60 MW that was not fully mobilised by the private initiative in the 2006 tender to new power plants fuelled by biomass residues and the RCM estimated the availability of the biomass residues (forest, agricultural, agro-industrial, etc.), that can be valorised in bio-refineries and biomass power plants.

Although in the short-term the use of forest biomass residues in Portugal will be primarily directed to produce electricity, it is important to observe that forest biomass residues can be transformed into different types of biofuels, not only solid (e.g. briquettes and pellets) but also liquids (e.g. ethanol and methanol) or gaseous (e.g. methane), which in the future can compete directly for the same raw material increasing the pressure on this energy source (Nunes, 2015).

2.4 Technical aspects of forest biomass residues valorisation

Forest biomass residues are renewable sources, but its supply is not infinite (BERC, 2009), which distinguishes it from other renewable sources, such as wind or solar energy. Figure 4 shows the system of primary forest biomass residues valorisation for energy production, considering three components: forest management, supply chain management and energy conversion.

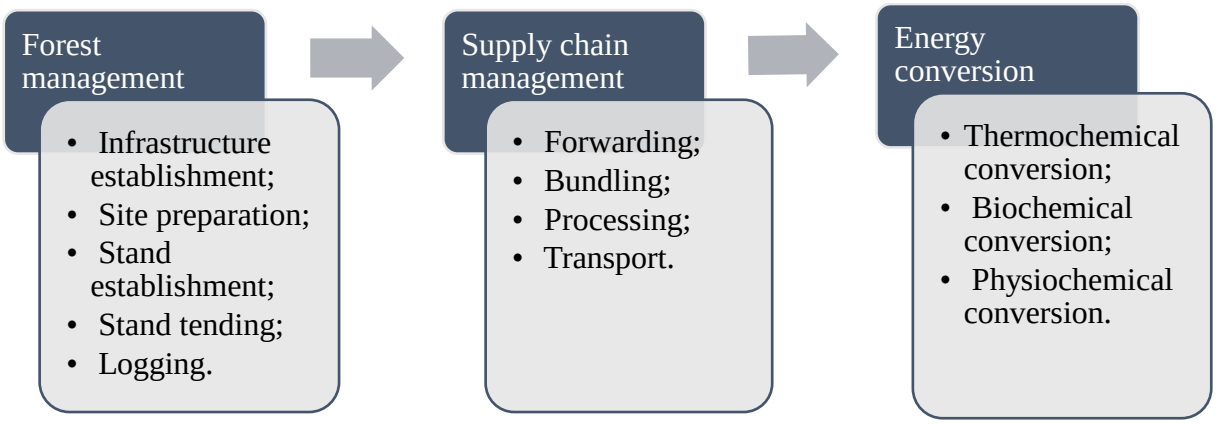

Figure 4 - General scheme of the forest biomass residues valorisation system for energy production.

Currently, there are several management practices and technologies required to valorise forest biomass residues for energy production. The sections below present in detail the most common practices for forest management, supply chain management and energy conversion.

2.4.1 Forest management

Wood productivity depends not only on climate, soil properties and water availability in the soil, but also on forest management practices. As forest management practices may differ greatly, only the most common management practices performed in Portugal for eucalypt and maritime pine stands will be presented, since maritime pine and eucalyptus have been largely used by power plants in Portugal (da Costa et al., 2018; Dias, 2014). The typical practices involved in the forest management stage are: infrastructure establishment, site preparation, stand establishment, stand tending and logging (Dias et al., 2007).

The infrastructure establishment consists on the construction and maintenance of firebreaks and roads (Dias et al., 2007). The roads are usually built with industrial tractors equipped with a front blade (Dias et al., 2007). Firebreaks are strips of land kept clear of vegetation or other combustible material and are required to protect the forest species from fires originating from beyond the property' boundaries (FAO, 2002).

The purpose of site preparation is to improve growing conditions for planted trees and to eliminate undesirable vegetation in order to reduce competition for space, light, moisture and nutrients (Johnson et al., 2009; Löf et al., 2012). The description of forest operations performed during site preparation is shown in Table 1.

Table 1 - Site preparation operations in eucalypt and maritime pine stands in Portugal. Source: adapted from Dias et al. (2007).

Operations	Description
1. Stump removal	Stumps left after tree felling are normally removed at the end of three successive coppice rotations with a mechanical digger. This operation is conducted only on eucalypt stands.
2. Clearing	Clearing is used to eliminate undesirable vegetation in order to facilitate subsequent soil scarification and planting, remove down wood or other debris and improve soil conditions for planted trees.
2.1 Disking	Disking is performed by using disk harrows pulled by a tractor to cut above- and below-ground vegetation and to break soil surface horizons, loosening the vegetation and incorporating it into the soil.
2.2 Mowing	An alternative to disking is mowing that removes the above-ground vegetation using a rotary mower pulled by a tractor.

Table 1 (cont.) - Site preparation operations in eucalypt and maritime pine stands in Portugal. Source: adapted from Dias et al. (2007).

Operations	**Description**
3. Soil scarification	Soil scarification decompress the soil, improving both infiltration and aeration, increases the uptake of nutrients and water and increases the initial growth rates of tree seedlings. In the case of being used seeds, seedling roots can penetrate faster than if they had to pass through a thick organic layer.
3.1 Excavating planting pits	A way to perform a scarification is excavating pits to aerate and loosen the soil in which the plants will grow, normally using a mechanical digger or an auger mounted on a tractor.
3.2 Ripping	Ripping is used for dry soils or for soils that have a compacted layer below the soil surface that restricts root growth and plant development. It consists in deeply breaking up compacted soils without inversion of soil horizons, using a crawler tractor equipped with one or more heavy vertical tines.
3.3 Subsoiling	Subsoiling fractures the soil structure and produces a deep scarification of the soil (up to 60 cm deep), but contrary to ripping, the tines used in this operation have horizontal wings that enhance soil fracturing and uplifting. Without subsoiling, plantations may be stagnated after a few years.
3.4 Ploughing	The purposes of ploughing are to create planting spots free from water logging and with little vegetative competition. Mounds of different sizes may be created depending on the moisture regime at the site, normally with a reversible mouldboard plough pulled by a tractor.
3.5 Furrowing and ridging	Furrowing and ridging are pattern of ridges and troughs created by a system of mounding by turning the soil in one direction, normally with a single-furrow reversible plough pulled by a tractor.
3.6 Terrace construction	Formation of horizontal platforms along the contour lines by means of a crawler tractor equipped with a front blade.

Stand establishment is an important step toward good forest management. It is a practice that ensures new healthy seedlings replace mature trees as they are harvested and includes:

- Planting is an artificial regeneration practice that consists of placing the tree seedlings on the previously prepared ground and can be performed manually or mechanically (McKell, 2012; Zaman et al., 2018). However, in Portugal, this operation is usually performed manually (Dias et al., 2007).

- Sowing consists of setting out seeds in prepared soil. It is not very common in eucalypt and maritime pine stands in Portugal (Dias et al., 2007).
- Natural regeneration consists on the creation of a new stand from natural seedfall. Natural regeneration is often used in maritime pine stands in Portugal (Dias et al., 2007).

The main objectives of the stand tending are the reduction of competition for light, water and nutrients, caused by herbaceous vegetation and brushwood, reduction of risks and damages caused by exposure to sun and wind, adequate morphological formation of plants or trees, regulation of nutrient levels, prevention against forest fires and pest and/or disease control (Unimadeiras, 2014). The stand tending encompasses the operations shown in Table 2.

Table 2 - Stand tending operations in eucalypt and maritime pine stands in Portugal. Source: adapted from Dias et al. (2007).

Operations	**Description**
Cleaning	Cleaning is used to remove unwanted vegetation in young stands. It is commonly performed either by disking or mowing between the planting lines and manually along the planting lines.
Fertilising	Fertilisation is carried out to promote tree growth on sites deficient of nutrients. It depends on the soil properties and on the forest species and is usually applied manually or simultaneously with subsoiling using a fertiliser spreader mounted on the subsoiler.
Soil loosening	Soil loosening is used to remove compaction from dense soil and is commonly done by disking.
Selection of coppice stems	It consists in is the practice of selecting certain stems and the removal of others from this regrowth, which allows the remaining stems to grow well and with a good form, giving the best yield of wood. The number of stems is reduced to two or three per stump and is exclusive to eucalypt stands.
Precommercial thinning	It removes some trees in overstocked stands to prevent tree stagnation and improve stand growth. It is performed prior to trees reaching merchantable size and can be carried out using the disking, mowing or chainsaw method, depending on the size of the trees removed.
Pruning	Pruning consists in cutting dead and broken branches to reduce the number of knots in the timber being formed, increase the health of the trees and remove potential safety hazards due to falling branches. It is normally carried out manually, exclusive to maritime pine stands.
Thinning	Cutting of some trees to accelerate diameter increment of the remaining trees. This operation is exclusive to maritime pine stands and is usually done with a chainsaw.

The logging operations include the practices presented in Table 3. The felling and limbing practices are specific to produce forest biomass residues, while the remaining practices are used to obtain wood.

Table 3 - Logging operations in eucalypt and maritime pine stands in Portugal. Source: adapted from Dias et al. (2007).

Operations	**Description**
Felling	Felling is the process of cutting down standing trees for further processing and use. It is typically performed with a chainsaw, although harvesters are also used in final cuttings.
Limbing	It is performed to remove the branches from downed trees. Limbing can be performed manually, with a chainsaw or with a harvester.
Bucking	Bucking is the operation of cutting the downed tree into desirable lengths, usually using a harvester or a chainsaw.
Debarking	Debarking is the removal of bark from downed trees, which is typically performed manually or using either a harvester or a debarker.
Extraction	Transportation of felled logs to a delivery point at the roadside. This operation is commonly made with forwarders or modified farm tractors.
Log loading onto trucks	This operation is is often accomplished with the crane of the truck or extraction equipment.

2.4.2 Supply chain management

Wood and bark are used by the wood-based industry, while logging residues and stumps can be used for bioenergy or left on the forest floor. The amount of logging residues and stumps left on the forest floor depends on the technical and logistic restrictions as well as on ecological reasons, ranging between 5 and 95 % of the total quantity produced (CBE, 2004). Due to the costs involved and the social and environmental impacts that it can cause, the supply chain management should be carefully planned, in order to contribute for a sustainable economic development. The supply chain management of the forest biomass residues can be organised in many ways (Figure 5), but usually includes:

- Forwarding of the forest biomass residues, which consists in collecting and transporting the residues from the logging site to the banking ground at roadside or to an intermediate storage park by tractor or forwarder (Dias, 2014; Stevanovic, 2018). However, the forwarders are the machines with the highest productivity and efficiency (Ranta et al., 2001).

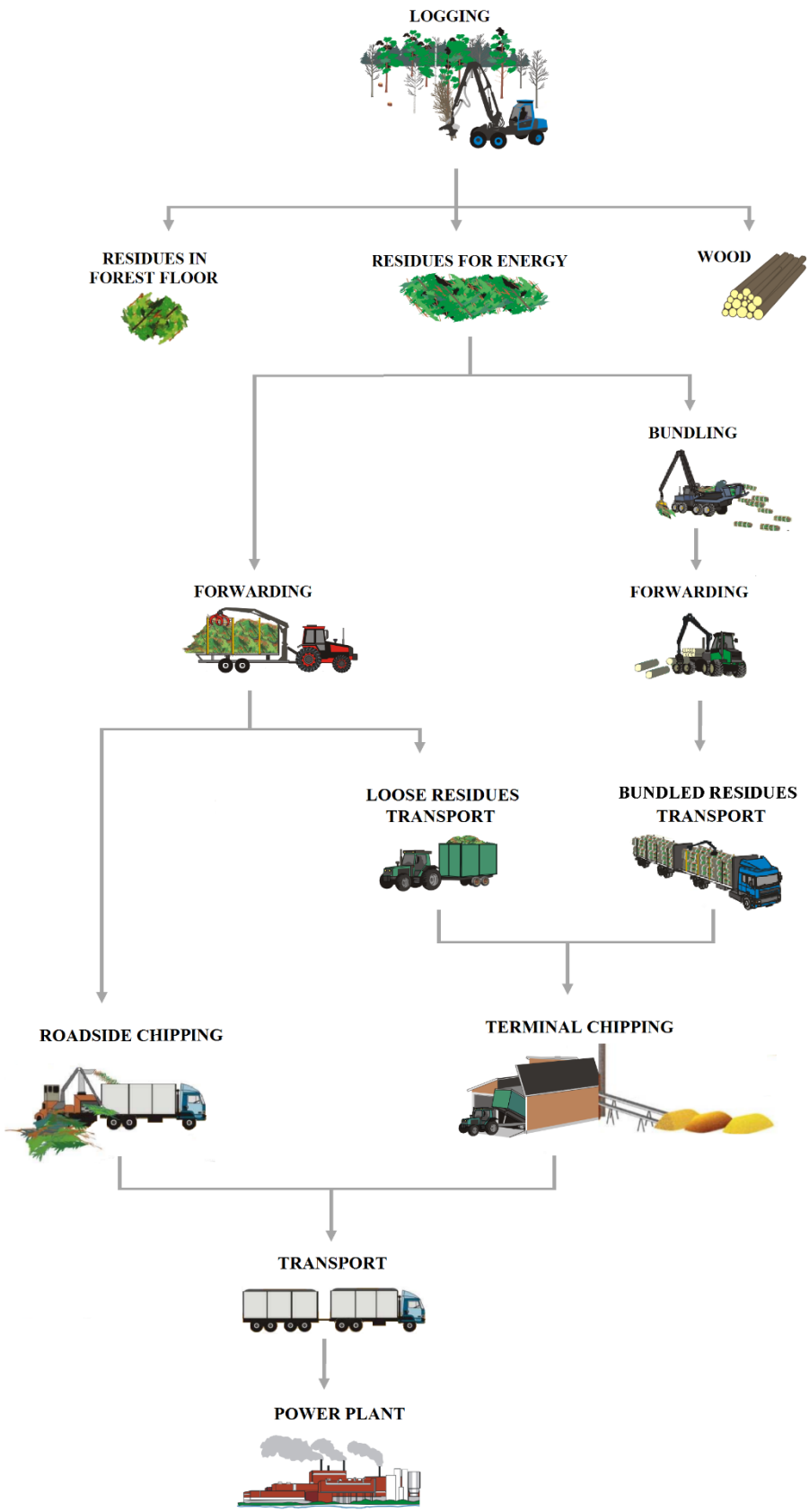

Figure 5 – General scheme of the supply chain management system of the forest biomass residues for energy production.

- Bundling is a facultative operation that can be performed in order to increase the density of forest biomass residues. In this process the forest residues are compacted into cylindrical bales in order to improve handling, transportation and storage (Hakkila, 2004). From this practice, bundlers can be stacked on the ground for later collection and the transportation by conventional trucks is possible because forest residues are compacted (Spinelli et al., 2012).
- Primary transport of the forest biomass residues is done from the intermediate storage park to the terminal chipping (Stevanovic, 2018). This transport can be performed using agricultural tractors, tractors with semitrailer or trucks in the case of bundled residues (Dias, 2014).
- Chipping is a process by which forest biomass residues are transformed into smaller particles, allowing their conversion to energy (Teixeira, 2009). Chipping can be performed in the roadside or in a terminal. When it is done at the roadside, forest biomass residues are chipped using mobile chippers (Zamora-Cristales et al., 2014). It is advantageous in two ways, namely, it allows the biomass to dry in the logging site and the transport costs are reduced, since the residues are concentrated. However, this system can only be implemented in smooth terrains and its productivity is normally lower than the chippers in the terminal due to reduced spaced. When chipping is done at a terminal, unchipped forest residues are transported to a terminal where they are chipped using mainly disc or drum (Spinelli and Hartsough, 2011). This operation is more productive and efficient, since the chips are stored and transported only when there is a greater need in the market. There is also the advantage of allowing the biomass to dry in the terminal, in order to obtain better prices. These advantages apply not only for systems based on loose residues but also for bundled residues.
- Secondary transport is related to the transport after chipping to its final destination (power plants) and is normally carried out using a truck (Stevanovic, 2018), allowing transportations between 16-26 t of residues, depending on the residues moisture (Netto, 2008).

2.4.3 Conversion technologies

Bioenergy derived from forest biomass residues comprises heat, electricity and biofuels which are obtained from three different conversion technologies: thermochemical, biochemical and physicochemical (Hurisso et al., 2018; Popa, 2018). Figure 6 summarises the various bioenergy conversion processes.

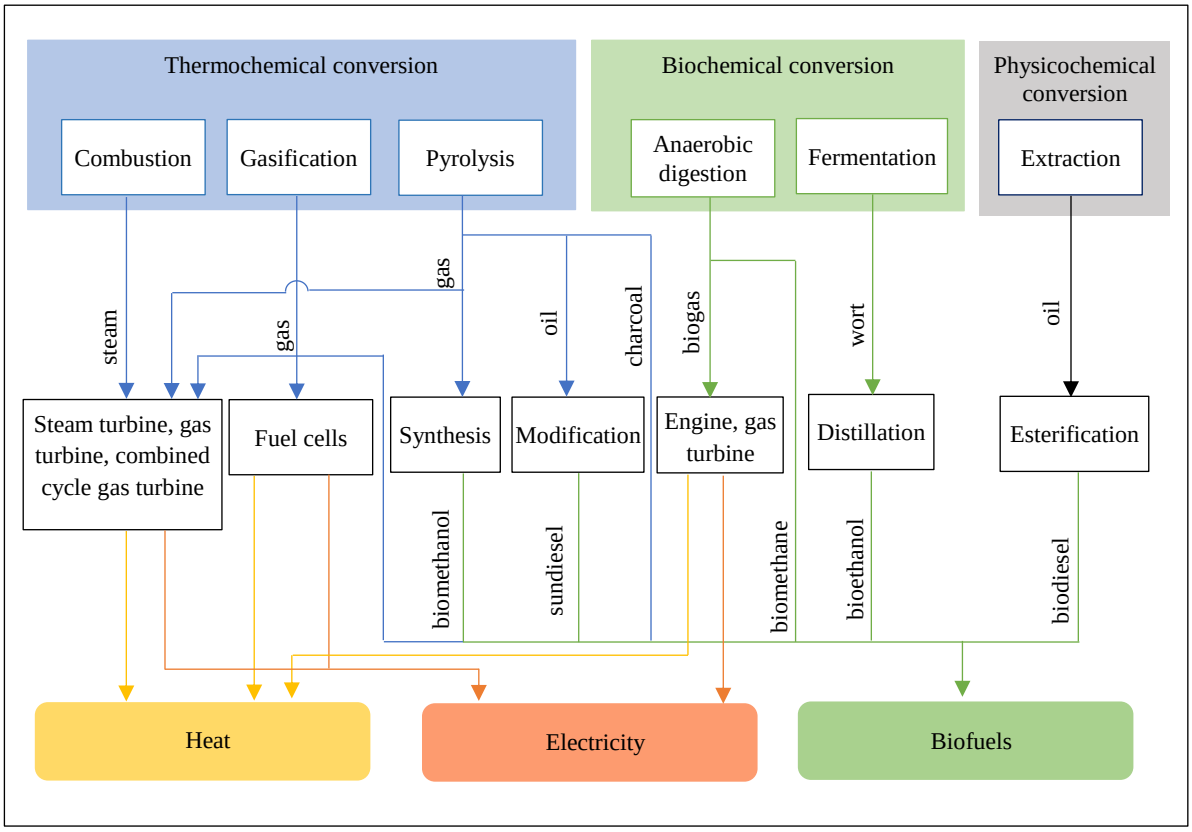

Figure 6 - Technologies of forest biomass conversion. Source: adapted from Górecki et al. (2015).

Several factors affect the choice of the conversion technology including the type, quantity and characteristics of biomass feedstock, end-use requirements, location, environmental regulations, economic aspects and project-specific factors (Ferreira et al., 2009; Fiorese et al., 2014). In general, conversion technologies used to produce energy from forest residues have to handle with a feedstock highly variable in mass and energy density, moisture content and size (Williams et al., 2016). Therefore, modern conversion technologies to produce energy from forest residues are often hybrid technologies, using fossil fuel for pre-heating and maintaining the energy production when the residues supply is interrupted, due to seasonal low availability of forest biomass residues. Thermochemical conversion encompasses mainly:

- Combustion

In Portugal, the electricity produced from forest biomass residues comes from combustion. Biomass combustion consists of series of chemical reactions in which the organic matter of the biomass is oxidised at high temperatures (700-1000 ºC, depending on the technology), using atmospheric air as oxidising agent and releasing heat, flue gases, water and ashes (Kaltschmitt et al., 2001; Ybarra et al., 2001). The combustion process consists in a

sequence of phases: water evaporation, devolatilisation, volatiles combustion and char combustion (Haseli et al., 2011; Ma et al., 2007). These phases can occur sequentially or simultaneously, depending on the particle properties (e.g. moisture content, size and density) and the surrounding conditions (e.g. temperature) (Momeni, 2012). The water evaporation and the devolatilisation are processes that expend heat (endothermic), while the volatiles combustion and char combustion in turn are processes that release heat (exothermic).

Moisture content is an important factor in the combustion process (Horvat and Dović, 2018; Rimár et al., 2016) and, typically, forest biomass residues contain high moisture content (Kizha and Han, 2017). Therefore, the combustion starts with the water evaporation phase. Depending on the forest biomass residues moisture content, water evaporation occur between 100 to 150 ºC (van Loo and Koppejan, 2008; Yuan, 2017).

The second phase is the devolatisation, i.e. the chemical decomposition of the forest biomass residues components (hemicellulose, cellulose and lignin), which occurs at 150-250 ºC (Goudriaan et al., 2008; Horvat and Dović, 2018). The most part of the solid biomass is released in the form of volatiles gases, such as methane and other hydrocarbons (Demirbaş, 2003).

The ignition occurs around 250 ºC (Eija et al., 2008), depending on fuel properties and furnace operating conditions. Then starts the volatiles combustion phase, in which the gases formed in the devolatilisation process react with oxygen in an exothermic reaction producing CO_2, water and heat.

The last phase is the char combustion, in which the char (residual black carbon material) reacts with oxygen to form mainly carbon monoxide (CO) (Shiehnejadhesar et al., 2017; Strezov and Evans, 2014). This phase occurs between 700-1000 ºC, releasing heat and light, which take the flame visible (Berton, 2007; Jones et al., 2014). After the char combustion, an inorganic ash is produced. The smaller ashes are transported by the exhaust gases, being called as fly ashes. Heavy ashes are stored at the bottom of the furnace and are called as bottom ashes.

Industrial combustion of forest biomass residues occurs mainly in plants based on steam turbines, i.e. through the Rankine cycle (Qiu et al., 2012). In the Rankine cycle, the furnace with biomass generates steam at high pressure, with a higher temperature than the boiling point of water (Qiu et al., 2012; Rout, 2013). The steam flows into a turbine designed to convert the steam's energy into kinetic energy and then to electricity in a generator (Muralikrishna et al., 2017). The cycle is completed when the condensed steam returns to the furnace to be heated again. Biomass combustion in power plants from steam turbines have an electrical efficiency of around 10-25 % (Ptasinski, 2016). Biomass combustion efficiency depends on the time required for the phases of water evaporation and devolatilisation, combustion temperature,

biomass composition and moisture and the turbulence required to the complete mixture of biomass and oxygen, ensuring complete combustion (Sims, 2002).

The typical furnaces used for industrial combustion of forest biomass residues in Portugal are grate furnace and fluidised bed furnace (E2p, 2017). The technical aspects, advantages and disadvantages of these furnaces are presented in Table 4.

Table 4 - Technical aspects, advantages and disadvantages regarding the operation of grate furnaces and fluidised bed furnaces.

Aspects	**Grate furnace**	**Fluidised bed furnace**
Utilisation	It was the first combustion system used for solid fuels and was considered the most versatile combustion technology in the 1980s (Werther et al., 2000).	It has been used since 1960 for combustion of residues (van Loo and Koppejan, 2008). Currently, it is the most popular type of combustion technology used for the conversion of biomass into energy (Burton, 2016).
Capacities	Capacities can range from 4 to 300 MW (mainly in the range of 20-50 MW) and operates as fixed and moving grates (Yin et al., 2008). The last one can be divided in traveling grates, rotating and vibrating grates (van Loo and Koppejan, 2008; Yin et al., 2008).	Fluidised beds can be bubbling fluidised beds or circulating fluidised beds. Bubbling fluidised beds are used for plants with a nominal furnace capacity of over 20 MW, while circulating fluidised beds are used in power plants with more than 30 MW (Lee and Shah, 2013; Vamvuka, 2010).
Operation temperature	It operates at temperatures between 900 and 1100 ºC (Obernberger and Biedermann, 2012; Yin et al., 2008).	The combustion temperature is kept usually in the range 750-950 ºC in order to prevent ash sintering in the bed (Dahlquist, 2013; van Loo and Koppejan, 2008).
Excess air	The overall excess air is typically set to 25 % or above (Yin et al., 2008).	Low excess air (20 %) is used to increase combustion efficiency and to reduce the flue gas volume flow (Obernberger and Biedermann, 2012).

Table 4 (cont.) – Technical aspects, advantages and disadvantages regarding the operation of grate furnaces and fluidised bed furnaces.

Aspects	**Grate furnace**	**Fluidised bed furnace**
Configuration	It is constituted by four key elements: a fuel feeder, a grate, an air supply system (primary and secondary) and an ash discharger	It consists of a vertical, cylindrical, refractory-lined bed with a distribution plate filled with the biomass residues and inert and a system of air supply.
Inerts	Inerts are not required.	A bed of inerts is used to improve the transfer of mass and heat and for the removal of sulphur dioxide (SO_2) formed during biomass combustion.
Handling	It requires less preparation and handling of fuel comparatively to fluidised bed (van Loo and Koppejan, 2008).	The fluidised bed combustion systems need a start-up time of approximately 8-15 hours, in which oil or gas burners are used (van Loo and Koppejan, 2008).
Control	The distribution of the biomass over the grate cannot be well controlled.	The intense transfer of mass and heat provides adequate conditions for a complete combustion.
Fuel type	It can use an extensive range of fuels of varying moisture content.	It can deal with extensive ranges of fuels due to the good mixing achieved.
Biomass particle size	It is appropriated for a vast range of particle sizes (van Loo and Koppejan, 2008; Yin et al., 2008).	It is limited regarding the particle size of the biomass. Usually a particle size below 40 mm is recommended for circulating fluidised beds and below 80 mm for bubbling fluidised beds (Lee and Shah, 2013).
Costs	The investment and operation costs are lower than fluidised bed furnaces (Obernberger and Biedermann, 2012; Wiinikka, 2005).	The investment and operation costs are usually higher than grate furnaces (Obernberger and Biedermann, 2012; Wiinikka, 2005).

Regardless of the conversion technology to produce energy from forest biomass residues, there is the release of pollutants both from incomplete and complete combustion (Silva, 2016). Regarding emissions from incomplete combustion, they include emissions of CO, hydrocarbons (e.g. methane), non-methane volatile organic compounds (NMVOCs), dioxins, furans and other, as well as unburned carbon content in the ashes (Demirbas, 2003; Khan et al., 2009; Obernberger and Thek, 2004). These emissions are generated as the result of the low combustion temperature, poor biomass/air mixture, short time of residence, high biomass particle size and high content of inert in biomass (Obernberger et al., 2006; van Loo and Koppejan, 2008). Among the emissions released from complete combustion, stand out CO_2, particulate matter, nitrogen oxides [NOx, such as nitric oxide (NO), nitrogen dioxide (NO_2) and nitrous oxide (N_2O)], sulphur oxides [SOx, such as SO_2, sulphur trioxide (SO_3) and alkali sulphates], acid gases (e.g. hydrochloric acid), heavy metals condensed in fly ash (such as lead, zinc, cadmium, etc.) and aerosols (Demirbas, 2003; Vassilev et al., 2013a).

- Gasification

The aim of forest biomass residues gasification is to reacted the residues with limited amount of oxygen to convert them to a mixture of gases called syngas, which includes CO and hydrogen (H_2) (Grassi, 2015). Compared to a solid fuel, a gaseous fuel is easier to combust and is better suited to be used in modern energy conversion equipment, such as fuel cells, turbines, furnaces, etc. (van Swaaij and Kersten, 2015). Gasification takes place at high temperatures (600-1500 ºC) and to achieve this temperature the fuel can come from outside (reforming) or from inside the gasification reactor by partial combustion of the feedstock (partial oxidation) (van Swaaij and Kersten, 2015).

Typically, an energy efficiency of around 75-80 % can be achieved in the gasification process with heat-integrated operation (Grassi, 2015). From the mixture of the gases formed, a wide range of final uses can be obtained (heat, electricity, biofuels). Among the liquid fuels, it is possible to produce biodiesel, biomethanol, bioethanol, etc. Methane and a broad range of chemicals can also be produced (van Swaaij and Kersten, 2015).

- Pyrolysis

Pyrolysis is one of the oldest process used to convert biomass into other fuels and materials (van Swaaij and Kersten, 2015). Pyrolysis has become of major interest due to the versatility of the technology, flexibility in operation and adaptability to a large variety of feedstocks and products (Bridgwater, 2015). The conversion of biomass to gaseous or liquid fuel gives it a much greater commercial value, since it becomes easy to handle, store and

transport (Grassi, 2015). Pyrolysis operates in anaerobic conditions, heat is usually provided from external sources and requires a carefully controlled reaction temperature of approximately 500 ºC (Palz and Zibetta, 2015). The constituents of biomass are thermally cracked to produce solid (charcoal), liquid (pyrolysis oil) or gaseous fuels, which usually undergo secondary reactions, thereby giving a broad spectrum of products (Bridgwater, 2015).

The biochemical conversion routes encompass mainly:

- Anaerobic digestion

The anaerobic digestion of biomass occurs naturally in wet environments in the absence of oxygen by microbial decomposition of biomass, producing mainly methane (CH_4) and CO_2 (van Swaaij and Kersten, 2015). Forest biomass contain both digestible (cellulose and hemicellulose) and non-digestible components (lignin). Therefore, the polymeric components of biomass have to be broken down into smaller molecules in a process stage called hydrolysis before they can be further converted into biogas. Several microorganisms influence the conversion of biomass via enzymatic activity (van Swaaij and Kersten, 2015). The resulting biogas is collected, dried, compressed and stored and can be used as a fuel to produce heat and electricity.

- Fermentation

The fermentation of forest biomass residues represents an attracting chance to produce bioethanol (Pandey et al., 2011). Compared to first generation biofuels, the production of bioethanol from forest biomass residues requires additional processing steps. The cellulose and hemicellulose, that constitute the forest biomass, have structures highly insoluble. In addition, the lignin provide a protective structure around the cellulose, which must be removed before the fermentation (Branco et al., 2018; Bright and Strømman, 2009). The carbohydrate polymers of cellulose and hemicellulose are transformed into fermentable monomeric sugars by hydrolysis using acids or commercial enzymes in order to make sugars contained in the forest biomass accessible for fermentation (Giordano et al., 2015). For the time being, the main hydrolysis technology for biofuel production is based on enzymes due to environmental and economic reasons (Bernardes et al., 2019; Valeriano et al., 2018).

The fermentation stage occurs using a wide range of microorganisms (yeasts, bacteria and fungi) to convert the wort into ethanol and CO_2. At industrial scale, the fermentation uses one of two major technologies, batch and continuous fermentation (Giordano et al., 2015). In batch fermentation, the process occurs in a single reactor, which is filled with the sugars formed and the microorganisms. The major advantage is that it affords less contamination between trials.

In continuous fermentation, the process occurs in a tank cascade where the liquid continuously flows. The major advantage of the later is the greater productivity, but much more operating care is needed to prevent contamination.

Distillation is the last stage and is used to separate the obtained ethanol and the water formed during the fermentation (Branco et al., 2018; Giordano et al., 2015). It is a technique based upon differences in volatility between the two substances. It is performed in two different steps: vaporisation and dehydration. Through subsequent vaporisation of the mixture, condensation, re-vaporisation and re-condensation, the mixture becomes higher in ethanol content. When the mixture reaches 95.6 % of ethanol, the ethanol and water cannot be further fractionated by vaporisation and, thus, a dehydratation is performed in which a third solvent is introduced to break the azeotrope formed (Giordano et al., 2015).

The physicochemical conversion encompasses mainly:

- Esterification

The oil produced from pressing the biomass has a structure formed by esters created from glycerol and three long-chain fatty acids (van Swaaij and Kersten, 2015). In the esterification, the oil is transformed into monoalkyl esters and glycerol. These monoalkyl esters have the right properties to allow them to be mixed with diesel from fossil fuel in different proportions or to be used as 100 % biodiesel (van Swaaij and Kersten, 2015).

2.5 Woody biomass ash

Woody biomass ash is generated from the thermochemical conversion of forest biomass and is being produced in increasing amounts in Portugal due to the emerging incentives and opportunities regarding bioenergy production (RCM, 2017, 2013, 2006b, 2003). With the power plants fuelled by forest biomass currently in operation in Portugal, between 100 and 200 thousand tonnes of ash are produced annually (Cruz et al., 2017). Worldwide annual biomass ash production is estimated at 480 million tonnes (Vassilev et al., 2013a).

Woody biomass ash production occurs in two ways namely, bottom and fly ash, with distinct characteristics and collection sites. Bottom ash corresponds to the coarse particles collected directly from the bottom of the furnace. In grate furnaces, this type of ash is composed of exogenous materials (soil) and slag synthetised or agglomerated (Silva, 2016). In fluidised beds, bottom ash consists of an agglomeration with bed materials (sand and adsorbents) and also exogenous contaminants mixed with the forest biomass residues (James et al., 2012; Rafael et al., 2015; Silva, 2016).

Fly ash corresponds to the fraction of fine particles that are carried by the air from the furnace. It can be collected in the economiser, in the air heater or in the dusting removal equipment (cyclone, electrostatic precipitators, bag filter). Normally, the fly ash that is collected in cyclones has diameters greater than 5 μm, while in downstream technologies (filters and precipitators), the particles present diameter of less than 1 μm (Silva, 2016; Vos, 2005). The particles with submicron size that are not retained are emitted into the atmosphere together with the exhaust gases (Nussbaumer, 2003; van Loo and Koppejan, 2008).

In industrial fluidised beds, the larger mass fraction corresponds to fly ash, while in grates, it corresponds to the bottom ash (Dahl et al., 2010; Girón et al., 2013; Llorente et al., 2006). The most important characteristics that should be considered when the ashes are being analysed are granulometry, density and composition. The characteristics of the ashes generated depend of many factors, such as properties of the forest biomass, logistic operations of collection, contamination with inert present in the soil and presence of bed sand in the case of fluidised beds, processing and transportation, thermochemical process (combustion or gasification), conversion technologies (fluidised beds or grates furnaces), dusting removal technology and operating conditions (temperature, air flow, additives, residence time of the ashes in the furnace) (Dahl et al., 2009; Vassilev et al., 2013a, 2013b).

Regarding granulometry of typical ashes from fluidised bed, fly ash presents diameter less than 0.1 mm and bottom ash presents diameter of about 0.2 mm (Dahl et al., 2010, 2009; Silva, 2016). However, in Portugal the bottom ashes generally present particle sizes from 0.5 to 4 mm, evidencing the contamination of the forest biomass with inert during collection and processing (Coelho, 2010; Modolo et al., 2014).

Bottom ash from grate furnace has a variable granulometry, which can achieve particle sizes above 4 mm (Coelho, 2010; Dahl et al., 2010). Fly ash from grate furnace has its granulometric distribution normally below 0.2 mm (Silva, 2016). However, in Portugal, the particle size of fly ashes from grate furnace triples due to the significant presence of unburned material (Coelho, 2010).

The density of the fly ash particles can vary from 2.2 to 2.7 g/cm^3, depending on the particle size and state (fragmented or agglomerated) and the presence of unburned material (Coelho, 2010; Lanzerstorfer, 2015). In Portugal, fly ash from grate furnace and fluidised bed typically present a density of 2.2 g/cm^3, while bottom ash ranges between 2.4 and 2.6 g/cm^3 (Coelho, 2010). Bottom ashes generally have a larger particle size and density than fly ash (Dahl et al., 2009; Stiernström et al., 2014; van Loo and Koppejan, 2008). On the other hand,

fly ashes present higher specific surface area when compared to bottom ashes (Girón et al., 2013; Sahu et al., 2017; Yao et al., 2015; Zhao et al., 2013) .

Regarding composition, although carbon is mostly oxidised during combustion and nitrogen is emitted in the form of gaseous compounds, most other elements present in the forest biomass are retained in the ash (Insam and Knapp, 2011). Woody biomass ash mainly consists of calcium (Ca), potassium (K), magnesium (Mg), sulphur (S), phosphorus (P), manganese (Mn), silicon (Si) and aluminium (Al) (Lanzerstorfer, 2015; Vassilev et al., 2013a; Wedepohl and Simon, 2010). It also can be found in less quantity in woody biomass ash: zinc (Zn), arsenic (As), chromium (Cr), lead (Pb), iron (Fe), nickel (Ni), vanadium (V), barium (Ba), sodium (Na), cadmium (Cd), mercury (Hg), copper (Cu), boron (B) and molybdenum (Mo) (Demeyer et al., 2001; Pitman, 2006). The behaviour patterns of these elements vary substantially, as some are partially or completely volatilised during combustion (fly ashes), whereas others are retained at the bottom ashes (Miller et al., 2002).

The presence of heavy metals and organic pollutants influence the quality of the ashes, compromising their valorisation and increasing the risk of dangerous emissions to the environment. The use of forest biomass contaminated with inert present in the soil also causes significant increases in the amount of heavy metals present in woody biomass ash (Vassilev et al., 2013a, 2014). Forest fires are also responsible for the increased accumulation of heavy metals in the soil (Núñez-Regueira et al., 2004, 2001).

In Portugal, the most common practice of woody biomass ash management is landfilling (Coelho, 2010; Silva, 2016, 2012). However, this practice has economic and environmental drawbacks and neglects the valorisation potential of the ashes (Insam and Knapp, 2011; Tarelho et al., 2015). In 2015, the Portuguese power plants paid 5.50 €/t of residue to deposit ashes in landfills (Assembleia da República, 2014). However, this fee will increase 1.1 €/yr as a result of the new "General Regime of Waste Management", until achieve 11 €/t of residue in 2020 (Assembleia da República, 2014). This cost of disposal, the need of large areas for landfilling and the fact that woody biomass ash disposal in landfills increases the risk of groundwater contamination with heavy metals fostered the demand for alternative solutions to valorise woody biomass ashes.

These solutions include the valorisation of woody biomass ashes in civil construction, soil amelioration, closure of mines and quarries, use as adsorbents and absorbents in wastewater treatment, use as additive in fluidised beds, use for synthesis of minerals, vermicomposting, etc. (Huotari et al., 2015; Pitman, 2006; Silva, 2016; Sklivaniti et al., 2017). The rate of woody biomass ash valorisation in Portugal is less than 10 % (Williams, 2013) and encompasses

mainly the utilisation in construction materials, soil amelioration, vermicomposting and physicochemical treatment of effluents produced in the pulp and paper sector (Heleno et al., 2012; Silva, 2016), as shown in Figure 7.

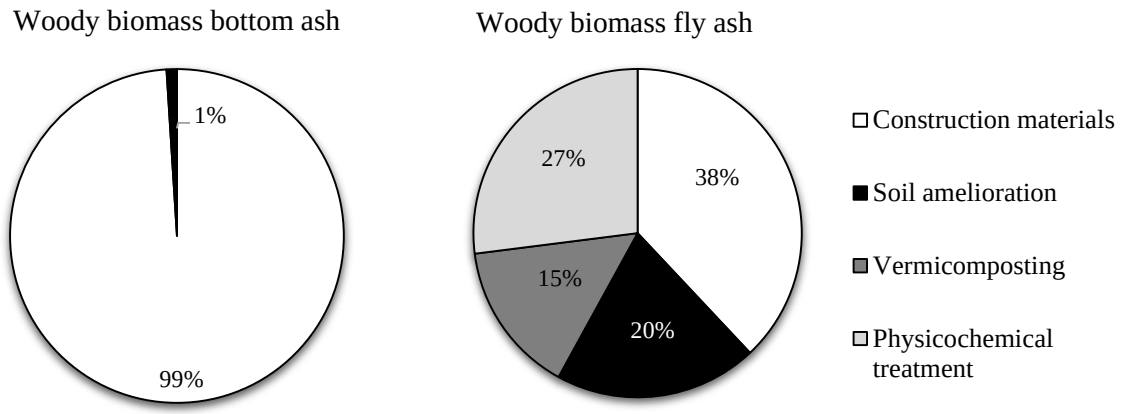

Figure 7 - Valorisation alternatives of woody biomass fly and bottom ashes most used in Portugal. Source: adapted from Silva (2016).

Woody biomass ash valorisation in construction material represent 99 % of the alternatives for bottom ash. For woody biomass fly ashes, the valorisation in construction materials and its use to soil amelioration, together represent 58 % of the valorisation alternatives. The present study focuses on these two alternatives, since valorisation in construction materials and soil amelioration has been growing in Portugal (Heleno et al., 2012).

Woody biomass ash valorisation in construction materials can allow the reduction of extraction of virgin raw materials to produce the conventional construction products. According to previous studies, the use of woody biomass ashes, in particular to produce cement mortar, adhesive mortar, concrete and asphalt, seems to be technically feasible (Barbosa et al., 2013; Dias, 2012; Modolo et al., 2015, 2013; Raheem et al., 2014; Rajamma et al., 2015; Subramania et al., 2015; Supancic and Obernberger, 2011). In literature, there are various studies evaluating the potential application of ashes in construction materials, some examples are presented in this section.

Regarding cement mortar, Rajamma et al. (2015) evaluated different cement substitution rates (0, 10, 20 and 30 %) by woody biomass fly ash to produce cement mortar. The results showed that the addition of ash up to 10 % does not influence the consistency of cement mortars, but when the substitution rate increases above 20 % the water demand increases and the flexural and compressive strength of the cement mortar decreases. In samples containing 10 % of ash, the decrease of strength was between 5-10 % compared to the conventional cement

mortar, while in samples containing 20 % of ash, the decrease of strength was equal to 18-20 %. However, the authors concluded that the introduction of woody biomass fly ashes affects the characteristics of the cement mortar but do not compromise them.

Modolo et al. (2013) studied the application of woody biomass bottom ashes from a fluidised bed furnace to produce cement mortar substituting the aggregate (sand). The aggregate was substituted by an equivalent mass of 50 and 100 % of woody biomass ash. The authors observed that compressive strength of the cement mortar increased with the use of woody biomass ashes due to a better compactness of the ash compared with sand.

Esteves et al. (2012) assessed the production of cement mortar with woody biomass fly ash used from fluidised bed and grate furnace as partial substitute of cement with substitution rates ranging from 20 to 30 %. They concluded that the incorporation of woody biomass fly ash is useful to resist the alkali-silica reaction in concretes, which is responsible for the deterioration of mortar structures. The less alkalinity is favoured by the production of cement mortars with woody biomass fly ash from grate furnace compared to that produced with ashes generated in fluidised beds.

Concerning the application of woody biomass ashes for producing adhesive mortar, Modolo et al. (2015) evaluated the substitution of sand by bottom ashes from a fluidised bed furnace. The results showed similar water demand, tensile adhesion and strength in the samples containing ash. However, the incorporation of woody biomass ash in the adhesive mortar was found to be suitable, with no detrimental implications in chemical or physical properties of the product.

Regarding the production of concrete using woody biomass ashes, Barbosa et al. (2013) assessed the possibility of using woody biomass fly ash as substitute for cement and bottom ash as substitute for aggregates. Formulations were prepared with different percentages of fly ashes substitution for cement (10, 20 and 30 %) and bottom ash substitution for sand (9, 18 and 36 %). According to these authors, the formulations in which woody biomass fly ashes were used to replace 10 % of the cement presented similar to slightly higher compressive strength values than those with cement, corroborating its application potential. The substitution level of 18 % of aggregates by bottom ashes promoted the highest compressive strength. In addition, the formulations presented similar or even lower chemical and toxicity-related emissions than those observed for the conventional cement.

Lessard et al. (2017) showed that woody biomass fly ash has the potential to substitute 10 % of the cement in concrete block production, leading to a concrete block with similar properties to the traditional concrete. Moreover, the application of woody biomass ash appears

to improve long-term permeability and showed slight changes in the water absorption and permeable voids content. Chowdhury et al. (2015) also studied the production of concrete blocks with woody biomass fly ash substituting cement. However, the study showed that woody biomass ash incorporation above 10 % decreases the concrete strength due to pozzolanic reactions and tensile strength also followed the same trend, which impairs its incorporation in concretes.

Regarding asphalt production, Dias (2012) studied the production of asphalt using woody biomass bottom ashes from grate and fluidised bed furnaces, substituting limestone in ranges between 10 to 30 %. The author observed that the substitution of the limestone by 10 and 20 % of ashes from grate furnace presented similar results to the conventional asphalt in terms of tensile strength and strain deformation. However, for the ashes from fluidised bed it was concluded that substitution rates above 10 % do not present conditions of applicability due to a decrease on the tensile strength.

Supancic and Obernberger (2011) studied the use of woody biomass fly and bottom ashes from fluidised bed as substitute for burned limestone in asphalt production. The author concluded that fly ash is not suitable for incorporation in asphalt because its high content of volatile heavy metals would lead to high concentrations of heavy metals in the soil. However, the utilisation of bottom ash can provide a suitable production of asphalt, while maintaining environmental safety.

Woody biomass ash can also contribute to the sustainable management of forest ecosystems given that its application in forest soils is an opportunity to close the biomass cycle and to reduce nutrient depletion and soil acidification (Arshad et al., 2012; Gómez-Rey et al., 2012; Wiklund, 2017).

Some soils in Portugal are naturally acid (APA, 2019), which limits their ability to vegetation growth, since normally vegetation respond better to neutral soils than to acid soils (Demeyer et al., 2001; James et al., 2012; Lin et al., 2007). Woody biomass ash appears to be a good option for liming because it is alkaline, contains high values of Ca and Mg carbonates and readily reacts with acidic components in soil (Demeyer et al., 2001). Woody biomass ashes from fluidised bed and grate furnaces generate a leachate with a pH of 11-12 for bottom ash and above 12 for fly ash (Dahl et al., 2009; Nurmesniemi et al., 2012; Vassilev et al., 2015).

Extensive research has been conducted for woody biomass ash valorisation by application in acid soils substituting liming products (Arshad et al., 2012; Cruz et al., 2017; Dvořák et al., 2017; Park et al., 2012), demonstrating that this practice is a good opportunity to valorise this residue, since woody biomass ash brings a more rapid soil pH change than using traditional

liming products (Pitman, 2006). This effect can be attributed to its fine structure and chemical composition (Arshad et al., 2012; Lickacz, 2002). Besides, the effects of woody biomass ash on soil acidity and extractable Ca and Mg concentrations were found to last for many years, since ash components bind to organic substances in the soils (Mandre et al., 2006; Saarsalmi et al., 2004, 2001).

Woody biomass ash has also potential to be used as fertiliser, since it contains important nutrients for plants (K, Ca, P, Mg, B, etc.), that had been removed from the soil during plant's growth, expect nitrogen (Gómez-Rey et al., 2012; Scheepers, 2014). This would potentially reduce the quantities of traditional fertilisers added to soils.

However, it is also necessary to consider the potential negative impacts on the environment resulting from woody biomass ash application in soils (Insam and Knapp, 2011; van Loo and Koppejan, 2008). Fly ash concentrates the largest fraction of heavy metals, since most heavy metals have high volatilisation rates and are carried along with the fly ashes during the thermochemical conversion of biomass, while bottom ash typically presents low concentrations of heavy metals either in grate or fluidised bed furnaces (Ingerslev et al., 2011; Nurmesniemi et al., 2012; van Loo and Koppejan, 2008). Heavy metals in soils can contaminate surface and groundwater resources and affect fauna and flora, which compromises ash valorisation by application in soil (Silva, 2016; Stiernström et al., 2014). Therefore, it is important to evaluate and compare the trade-offs between the environmental benefits and impacts of woody biomass ash valorisation in soil, in order to provide information to support decision-making regarding the best management option for ash.

2.6 Life cycle assessment

2.6.1 Brief description of life cycle assessment development

Concerns about environmental issues, resource efficiency and fossil fuel depletion have encouraged the development of life cycle thinking tools for environmental evaluation of products (Dincer and Rosen, 1998; EPA, 1998). The study of environmental impacts based on a life cycle approach was suggested in the 1960s (Guinée et al., 2011). The methods precursors of LCA developed in the 1960s in the United States of America (USA) could be described as material and energy flow accounting, since they focused on inventorying resource and energy use, emissions and generation of wastes, from each process in the entire life cycle of a product (Hunt et al., 1992). Most of the early studies analysed packaging and were typically motivated by industries producing and using the packaging, such as Coca Cola in a pioneering study in

1969 (Hauschild et al., 2017). The Environmental Protection Agency (EPA) also supported a study with the aim of informing regulation on packaging in 1974, demonstrating the interest of government on environmental evaluation (EPA, 1974). Energy and especially the fossil fuels also gained relevance in the 1970s. During the 1980s, life cycle-related methods received little attention in North America, but in Europe an increased interest in the impacts of milk packaging inspired a number of studies (Franke, 1984; Habersatter and Widmer, 1991; Lundholm and Sundström, 1985; Mekel and Huppes, 1990; Pommer et al., 1991). Although they compared more or less the same packaging technologies, they obtained different conclusions, which made the methodology standardisation imperative.

In 1990, the Society of Environmental Toxicology and Chemistry (SETAC) organised the first workshop to discuss a technical framework for LCA, which was followed by a series of workshops (SETAC, 1991). The workshop of 1993 held in Sesimbra (Portugal) culminated in one of the most important reports of SETAC, the "Code of practice" (Consoli et al., 1994), an effort made by LCA practitioners, users and scientists to cooperate with harmonisation of LCA methodology, framework and terminologies. Alongside to SETAC, the International Organisation for Standardisation (ISO) developed a formal standardisation of LCA methodology (Guinée et al., 2011), resulting in the publication of four standards between 1997 and 2000. The standards addressed the principles and framework (ISO 14040) (ISO, 2006b), the goal and scope (ISO 14041) (ISO, 1998), the life cycle impact assessment (ISO 14042) (ISO, 2000a) and the life cycle interpretation (ISO 14043) (ISO, 2000b). In 2006, the former standard was updated and the latter three standards were compiled in the ISO 14044 standard (ISO, 2006b).

In the 1990s, a number of life cycle inventory databases have been developed by different institutes and organisations (Hauschild et al., 2017). However, the differences in data standards and quality caused substantial inconsistencies in the results. This situation was improved with the release of the first version of the Ecoinvent database (v 1.01) in 2003, covering a wide range of sectors (Ecoinvent, 2018). The early 1990s also saw the birth of a number of impact assessment methods. The first impact assessment method that covered an ample set of midpoint impact categories was developed by the Centre of Environmental Science of Leiden University (CML) and became known as CML92 (Heijungs, 1992). The Environmental Priority Strategies (EPS) method (Steen, 1999a, 1999b) took a different approach focusing on the damages to ecosystems and human health, rather than midpoint impacts, an approach that was followed by the Eco-Indicator 99 method (Goedkoop and Spriensma, 2000). As inventories became more complex, a need for dedicated LCA software emerged and the first versions of both SimaPro

and GaBi, two widely used software, appeared in the beginning of the 1990s (PRé, 2019; Thinkstep, 2015)

Nowadays, the importance of LCA continues increasing due to the growing interest in environmental and climate change issues. LCA is a useful tool for making holistic comparisons among competing systems, as well as identifying opportunities to improve the environmental performance of an existing system (Curran, 2017). Additionally, LCA is also used in the decision-making process in industry, government or non-government organisations for strategic planning, priority setting and product design, as well as for marketing (e.g. implementing eco-labelling schemes or producing environmental product declarations) (ISO, 2006b).

2.6.2 Life cycle assessment methodology

The structure of the LCA methodology (Figure 8), according to the ISO 14044 standard, comprises four phases:

- Goal and scope definition is the first phase of an LCA and should include the definition of the goal, functional unit, system boundaries, multifunctionality procedures, impact assessment methods, data quality requirements, assumptions and limitations among others. The goal should specify the reasons for carrying out the study, the intended audience (i.e. to whom the results will be communicated) and the intended application (i.e., if the results are going to be used in comparative assertions disclosed to the public).

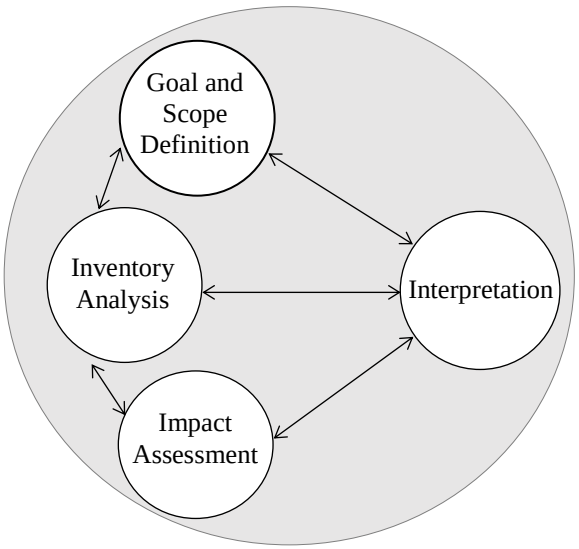

Figure 8 - Phases of an LCA. Source: adapted from ISO (2006).

The purpose of the functional unit is to provide a reference to which the input and output data are normalised, i.e. to describe the product or process for which the assessment is performed and the basis of defining the reference flow of a product. The choice of functional unit requires additional attention when the objective of the LCA is to compare products, as the basis of comparison should be defined on the basis of the same function, quantified by the same functional unit in the form of their reference flows (ISO, 2006b).

The system boundary defines where the analysis begins and where it ends and decides which processes belong to the life cycle of the product under study. Selection of the geographical and temporal boundaries should also be performed. Spatial boundaries are important because consumer behaviours differ by location, as well as the physical realities, e.g. freshwater availability. Temporal boundaries are important for similar reasons, since life cycle inventories consist of large amounts of data, generally collected from different periods, being necessary to evaluate if the data are still representative or more recent data are needed (Curran, 2017).

Multifunctionality occurs in processes or systems that deliver more than one function. For example, processes with more than one product as output (e.g. cogeneration of electricity and heat) or with more than one waste treated jointly (e.g. municipal solid waste landfill) or when recycling (including reuse, material recovery and energy recovery) results in material or energy being used in different systems (e.g. open-loop recycling, incineration of post-consumer plastics with energy-recovery). To handle multifunctional processes, the ISO 14044 standard presents a hierarchy of solutions:

1) Subdivision: the multifunctional unit process is subdivided into minor units to separate the production of the product from the production of the co-product(s) and the sub-processes that provide the additional functions are excluded from the product system.

2) System expansion: it means expanding the product system to include the additional functions related to the co-products. For example, in the comparison of a power plant with dedicated production of electricity and a power plant with cogeneration of electricity and heat, this means expanding the system of the dedicated plant to include the impacts of alternative production of heat. It is mathematically identical to credit the second power plant with the avoided production of heat that would alternatively have been produced somewhere else in the technosphere.

3) Allocation: when allocation cannot be avoided, the inputs and outputs of the system should be divided between its different products or functions in a way that reflects the underlying physical relationships between them (e.g. mass, energy). When physical

relationships alone cannot be used as the basis for allocation, another relationship should be identified, such as the economic value of the products.

The impact assessment method to be applied should be defined, identifying the inherent impact categories and characterisation models, consistently with the goal of the study. In common LCA practice, the impact assessment methods include a set of impact categories, based on specific characterisation models and the ISO 14044 provides recommendations for these choices.

A description of data quality requirements is important to understand the consistency of the study results and properly interpret the outcome of the study and should address the time-related coverage, geographical coverage, technology coverage, precision, completeness, representativeness, consistency, reproducibility and uncertainty. During an LCA study it is possible that some data are not available and, therefore, it is needed to make assumptions that must be declared in a clear way.

- Inventory analysis consists in identify and also quantify the inputs and outputs of resources and materials, energy and emissions to air, water and soil for all the unit processes within the system boundaries. Primary data, i.e. data determined by direct measurements at the operated processes, are normally collected for the foreground system, while secondary data, i.e. data collected from literature, including inventory databases, are normally used for the background system (EC/JRC/IES, 2010a). The result of the inventory analysis is a list of quantified flows for the product system that is connected with the provision of the function described by the functional unit (Bruijn et al., 2004).

- Impact assessment translates the inventory data in terms of environmental impacts. According to the ISO 14044 standard, the impact assessment consists of six elements of which the first three are mandatory (ISO, 2006b):

1) Selection of impact categories and related category indicators and characterisation models. There are several impact assessment methods that can be used in an LCA study, considering different environmental impact categories, such as, climate change, stratospheric ozone depletion, photochemical ozone formation, acidification, eutrophication, etc.

2) Classification, which consists in assigning the inventory results to different impact categories according to their characteristics. For example, CO_2 emissions are assigned to the climate change category.

3) Characterisation, which is performed by converting the different inventory inputs into common units, which allows the aggregation of all contributions within the same impact

category into one score, representing the total impact that the product system has for that category. For example, mass of CO_2-eq is the common metric for climate change category.

4) Normalisation is used to scale the data by a reference factor in order to clarify the relative magnitude of each impact category in a given context. The result is the normalised impact profile in which all impact categories are expressed in the same metric. It is useful when one impact is condemnatory and another is positive.

5) Grouping is the sorting and possibly ranking of impact categories into one or more sets according to their characteristics such as inputs and outputs or global, regional and local spatial scales or apparent severity.

6) Weighting supports comparison between the impact categories by weighting them using weighting factors that for each impact category gives a quantitative expression of how severe the impact is. It is useful when the results of the LCA are used for decision support together with other condensed information like the economic costs of the alternatives.

Additional data quality analysis can be used during the impact assessment to identify negligible inventory analysis results and to guide the iterative impact assessment process.

- Interpretation is used to identify the significant issues based on the results of the inventory analysis and the impact assessment phases (ISO, 2006b). In addition, a set of conclusions and recommendations for the study should be made, including, the identification of hotspots (processes or life cycle stages with the highest environmental impact), the completeness, sensitivity and consistency analysis of the impact results, the identification of limitations to the study and recommendations to improve the study.

The objective of the completeness analysis is to ensure that all relevant information and data needed for the interpretation are available and complete. Sensitivity analysis and uncertainty analysis are applied to guide the conclusions, to appraise the robustness and to further strengthen the conclusions. Recommendations can then be made based on a clear understanding of how the LCA was performed and the results developed in line with the goal of the study.

2.6.3 Attributional and consequential life cycle assessment

There are two different modelling principles in LCA: attributional and consequential. The attributional LCA is the most applied and best established LCA methodology (Curran, 2017) and aims to answer the question “what environmental impact product X is responsible for?”. The impact results from an attributional LCA are normally based on average data from existing process and the studied product system is separated from the rest of the economy, i.e. market

trends are not considered. However, the assumption of isolated product systems might be subject to discussions, since product systems interact with other products systems through multifunctional processes (Curran, 2017; Hauschild et al., 2017). Consequential modelling was developed to deal with the consequences of a change in response to a decision or action. Therefore, the question addressed by consequential LCA can be "what are the environmental consequences of consuming X?". The overall approach of the consequential modelling is to repetitively ask this question for each step upstream and downstream from the reference system until all changes have been covered.

Understanding the difference between these two modelling principles and when to use one or another are problematic aspects of LCA and there is still no agreement on this issue within the LCA community (Ekvall, 2012; McManus and Taylor, 2015; Weidema et al., 2018; Yang, 2016). There are several studies comparing the use of attributional and consequential LCA that conduct to different results and conclusions (Rajaeifar et al., 2017; Rehl et al., 2012; Schmidt et al., 2011; Searchinger et al., 2008; Venkatachalam et al., 2018). For example, Searchinger et al. (2008) evaluated the use of bioethanol from corn compared to the use of conventional gasoline. The results showed a decrease of 20 % in GHG emissions with bioethanol compared to conventional gasoline in the attributional modelling. However, in the consequential perspective it was observed an increase of 47 % in the GHG emissions when bioethanol is used instead of gasoline. This occurs due to land use changes induced by higher prices of corn, soybeans and other grains as a consequence of additional demand for corn starch to produce bioethanol. However, care should be taken to not get the impression that only one of these modelling approaches is the "right one", since both attributional and consequential LCAs are important to support decision-making and should be seen as complementary approaches (Curran, 2017).

The International Reference Life Cycle Data System (ILCD) handbook (EC/JRC/IES, 2010a) identifies situations in which LCA can be used and provides methodological guidance and provisions linked to each situation, such as the choice between attributional and consequential LCA and how to handle with the system multifunctionality. Figure 9 shows the four major types of situations: situation A (micro scale decision support), situation B (meso/macro scale decision support) and situations C1 and C2 (accounting with no decision support).

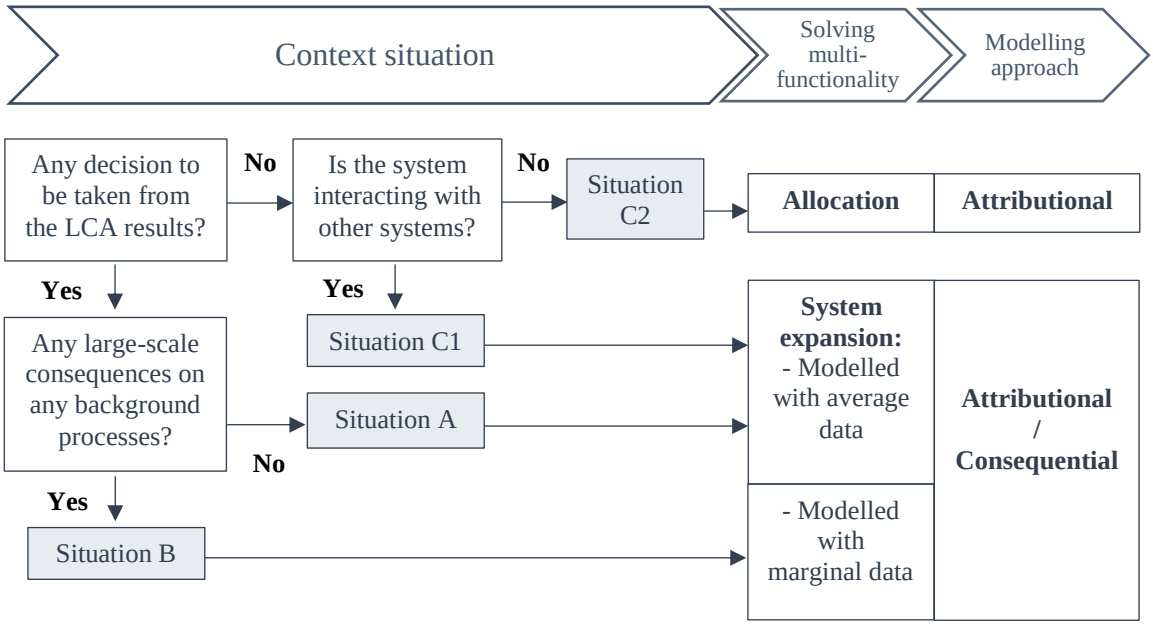

Figure 9 - Classification of the decision-context in LCA and orientation for modelling provisions. Source: adapted from EC/JRC/IES (2010).

Situation A covers all studies that are intended to support any kind of micro level comparisons, as for example, the decision support to select five photocopier models according to the environmental performance over its life cycle. Situation B refers to decision support with extensive consequences that result in additionally installed capacity. An example is a study analysing the mandatory substitution by 2025 of 50 % of diesel fuel in the USA by crop-based biodiesel. In Situations C, no direct decisions are made based on the results of the LCA, the study is only documenting what has happened (or is going to happen in future). The situation C1 is used for monitoring of a certain product group that are produced in a certain time-frame. An example is the monitoring time-series of the life cycle inventory of all cars produced annually in France. The situation C2 is used for monitoring product groups with a system boundary that is strictly referring to a certain time-frame. An example is the monitoring time-series of all car-related activities for the total amount of cars operated in a given year in France.

According to the definitions above, consequential LCA is more appropriate for situations A and B, since it states that, when LCA is used as decision-support, the model should reflect the consequences of the decision on the background system (Ekvall et al., 2016). The ILCD handbook recommends attributional modelling for micro-level decisions and for most of the studies when meso/macro scale decisions do not affect in a significant way the environmental performance of the background systems (EC/JRC/IES, 2010a).

In addition, attributional LCA is associated with the use of average data in the background system, which means that a market mix is considered, while consequential LCA is modelled with the use of marginal data (Finnveden et al., 2009). However, while background systems are modelled differently in attributional and consequential LCA, the foreground system is modelled in the same way, with the exception on how multifunctional processes are handled. Attributional modelling is usually associated with allocation as the approach to solve the multifunctionality of a product system, while in the consequential modelling the system expansion is frequently performed (Hauschild et al., 2017). However, in attributional modelling there are some cases in which the multifunctionality can be solved by a system expansion.

To perform an attributional LCA it is necessary technical knowledge about the product, how it is made, used and discharged. However, to perform a consequential LCA, besides this technical knowledge, it is also necessary knowledge about how the market reacts to an increase or decrease in its demand and supply (Weidema, 2003). According to Hauschild et al. (2017) four different steps are necessary for performing a consequential LCA:

1) to identify whether an increase or decrease in demand for a product will lead to changes in supply for that product. When performing a consequential LCA, a full elasticity of supply is normally supposed. This implies that a change in demand will lead to a change in the products supply that can satisfy this demand, but a change in supply will not lead to a change in demand;

2) to identify constraints in the market when the demand for a product is changed;

3) to identify the product that will be affected by a co-product from a multifunctional process;

4) to identify which production technology will be affected by the change in product supply.

Consequential LCA is conceptually complex because it includes additional concepts of market, such as marginal production, elasticity of demand and supply, etc. (Ibenholt, 2002). In general, the market trend is defined by extrapolating historical data of prices and consumption of the suppliers, which increases the risk of uncertainty and inadequate assumptions in the LCA modelling. These uncertainties are one reason why many LCA users prefer an attributional modelling and why ILCD recommends an attributional modelling for supporting a decision in micro-level scale (EC/JRC/IES, 2010a). To reduce the uncertainties in the consequential modelling, the results regarding different assumptions and consequences should be clearly described (Curran, 2017).

2.6.4 LCA of bioenergy systems from forest biomass residues

Forest biomass residues are becoming increasingly more important as feedstock to produce bioenergy (Lago et al., 2018), with a significant potential contribution to climate change mitigation (Woo and Turner, 2019). However, there are other environmental impacts that should be considered in the assessment of the environmental sustainability of bioenergy. An increasing scientific interest regarding the potential environmental impacts of bioenergy production from forest biomass residues using a life cycle perspective has been observed.

Table 5 presents papers focusing on LCA and carbon footprint of forest biomass residues valorisation to produce electricity and heat from combustion and bioethanol from biochemical conversion, which are the applications assessed in this thesis. The papers were selected based on a Boolean search of Web of Science and Scopus databases. Only peer-reviewed journal papers written in English published between 2000 and April 2019 were included in this analysis. This time coverage was chosen because the first set of ISO standards on LCA was completed in 2000. The keywords “forest biomass”, “life cycle assessment, “LCA”, “carbon footprint”, “energy”, “electricity”, “heat”, “ethanol” and “bioethanol” were combined and a total of 71 papers were found after the elimination of duplicates. A screening was performed to remove papers not focusing specifically on electricity and heat production from combustion and bioethanol production from biochemical conversion. It was found that 57 studies did not fulfil the inclusion criteria; therefore, 14 potentially relevant studies were analysed.

The topics analysed in these studies include the product under study, the country for which the study was performed (geographical coverage), the functional unit, the technology used for energy conversion, the modelling approach, the procedure used to handle multifunctionality, the system boundaries, the woody biomass ash destination, the impact assessment method and the impact categories chosen.

Regarding the product under study, five studies focus on electricity production, three studies on heat production, one study on cogeneration (combined heat and power) and five studies on bioethanol production.

Table 5 - LCA studies of bioenergy production (electricity, heat and bioethanol) from forest biomass residues.

Product	Reference	Country	Functional Unit	Conversion technology	Modelling approach	System boundaries	Ash management	Multifunctionality	Categories	Impact assessment method
Electricity	Lindholm et al. (2011)	Sweden	1 MJ of electricity	NA	Attributional	Residues collection to energy conversion	Not considered	Energetic allocation at energy conversion stage[(1)]	CC	IPCC (2006)
	Thakur et al. (2014)	Canada	1 kWh of electricity	Grate furnace	Attributional	Forest management to energy conversion	Not considered	NA	CC	GREET 1 and GREET 2
	Tagliaferri et al. (2018)	United Kingdom	1 MWh of electricity	Moving grate	Attributional	Residues collection to waste end-of-life	Returned to soil and landfill	System expansion at energy conversion stage[(1)]	CC, AC, FEc, FEu, HT, MFD, OD, POF, TET	CML 2 (2001)
	Loução et al. (2019)	Portugal	1 MJ of electricity	Grate furnace	Attributional	Residues collection to energy conversion	Not considered	100 % to main product at forest management stage	CC	IPCC (2006)
	González-García and Bacenetti (2019)	Italy	1 kWh of electricity	Grate furnace	Attributional	Residues collection to waste end-of-life	Landfill	100 % to main product at forest management stage	CC, AC, FD, FEu, HT, MEu, POF, PM	ReCiPe v1.12
Heat	Kimming et al. (2015)	Sweden	1 GJ of heat	Grate furnace	Consequential	Residues collection to waste end-of-life	Returned to soil	System expansion at energy conversion stage[(1)]	CC	IPCC (2006)
	Whittaker et al. (2016)	United Kingdom	1 Gwh of heat	Grate furnace	Attributional	Forest management to energy conversion	Not considered	Not considered	CC	IPCC (2006)
	Hammar et al. (2019)	Sweden	1 MJ of heat	NA	Attributional	Forest management to waste end-of-life	Returned to soil	Energetic allocation at energy conversion stage[(1)]	CC	IPCC (2006)
Cogeneration	Djomo et al. (2015)	Belgium	0.86 MJ of heat and 0.14 MJ of electricity	Stoker, grate furnace and fluidised bed	Attributional	Forest management to waste end-of-life	Landfill	NA	CC	IMPACT 2002+
Bioethanol	Sandilands et al. (2009)	New Zealand	1 km driven with bioethanol	Enzymatic hydrolysis	Attributional	Forest management to bioethanol combustion	Not considered	Mass allocation at forest management stage	CC, AC, FEu, POF	CML 2 (2001)
	McKechnie et al. (2011)	Canada	1 km driven with bioethanol	NA	Attributional	Forest management to bioethanol combustion	Not considered	Mass allocation at forest management stage	CC	IPCC (2006)
	Karlsson et al. (2014)	Sweden	1 MJ of bioethanol	Enzymatic hydrolysis	Attributional	Forest management to waste end-of-life	Returned to soil	System expansion and energetic allocation at energy conversion stage	CC	IPCC (2006)
	Falano et al. (2014)	United Kingdom	1 L of bioethanol	Acid hydrolysis	Attributional	Forest management to waste end-of-life	Landfill	System expansion and economic allocation at energy conversion stage	CC, AC, FEc, FEu, HT, LU, MAT, MFD, OD, POF, TET	CML 2 (2001)
	Liang et al. (2017)	USA	1 t of forest residue	Enzymatic hydrolysis	Attributional	Energy conversion stage	Not considered	NA	CC, AC, MEu, POF	TRACI 2.0

Notes: [(1)] Although the product under study is electricity or heat, data were collected from cogeneration facilities.

Abbreviations: Not available (NA); acidification (AC); climate change (CC); fossil depletion (FD); freshwater ecotoxicity (FEc); freshwater eutrophication (FEu); human toxicity (HT); land use (LU); marine aquatic toxicity; (MAT); marine eutrophication (MEu); mineral and fossil resource depletion (MFD); ozone depletion (OD); particulate matter (PM); photochemical ozone formation (POF); terrestrial toxicity (TET).

2.6.4.1 Electricity production

Regarding the geographic coverage of the studies on electricity production, most of them were produced in a European context (4 out of 5). North America is also represented with one study. The functional units defined in the studies are related to energy outputs (1 MJ, 1 kWh and MWh of electricity). Regarding the technology used for energy conversion, none of the studies of electricity production compared different types of technologies. Most of them assessed the combustion of forest residues in grate furnaces (4 out of 5), while only one study defined the type of grate utilised. More specifically, Tagliaferri et al. (2018) assessed the combustion in a moving grate. A common aspect of the studies was the modelling approach, as all studies performed an attributional LCA.

The life cycle of electricity production from forest biomass residues includes the following stages: forest management, forest residues collection, forest residues chipping, forest residues transportation, energy conversion and waste end-of-life. However, none of the studies included all these stages in the system boundary. Two studies, Lindholm et al. (2011) and Loução et al. (2019), considered the system boundaries from the residues collection to the energy conversion. Tagliaferri et al. (2018) and González-García and Bacenetti (2019) assessed the system from the residues collection to the waste end-of-life. Finally, Thakur et al. (2014) included forest management up to energy conversion. Most of the studies assumed that forest biomass residues are free of environmental burdens except with regard to forest residues collection, chipping and transportation and all impacts of forest management stage were allocated to the main product (wood). The only study that considered the system from the forest management was Thakur et al. (2014).

The management of the woody biomass ashes generated during forest biomass combustion has been poorly studied, being excluded from most of the studies (Lindholm et al., 2011; Loução et al., 2019; Thakur et al., 2014) as ash disposal is assumed to have a minor environmental impact. In the studies that included ash disposal, González-García and Bacenetti (2019) considered ash landfilling, while Tagliaferri et al. (2018) assessed the return of bottom ashes to the soil and the disposal of fly ashes in a landfill. However, the study does not define clearly how the assessment was conducted.

Another topic commonly discussed in the bioenergy LCA studies is the variety of methods to handle with the multifunctionality of the systems. Bioenergy production from forest biomass residues is a multifunctional process, since in the forest management stage different biomass outputs are produced, such as wood, logging residues, bark and stumps. Wood and

bark are normally used by the wood-based industry, while the logging residues and stumps can be used for bioenergy or left on the forest floor. All studies allocated 100 % of the impacts during forest management stage to the main product (wood), except Thakur et al. (2014). In addition, in some studies, inventory data for electricity production were collected from cogeneration plants where both electricity and useful heat are generated. In these studies, to handle multifunctionality of the energy conversion stage, Lindholm et al. (2011) performed an energetic allocation and Tagliaferri et al. (2018) performed a system expansion.

Regarding the impact categories, climate change was assessed in all studies analysed. Acidification, freshwater eutrophication, human toxicity and photochemical ozone formation impact categories were found in two studies (González-García and Bacenetti, 2019; Tagliaferri et al., 2018). Other impact categories found included freshwater ecotoxicity, mineral and fossil resource depletion, ozone depletion, terrestrial toxicity, fossil depletion, marine eutrophication and particulate matter. Finally, as shown in Table 5, the studies consider different methods for the evaluation of the environmental impacts, namely, Thakur et al. (2014) used GREET 1 (Wang, 2001) and GREET 2 (Wang et al., 2018), Tagliaferri et al. (2018) used CML 2 (Guinée, 2002), while González-García and Bacenetti (2019) used ReCiPe v1.12 (Goedkoop et al., 2013). Two of the studies do not specify the impact assessment method.

2.6.4.2 Heat production

There are three LCA studies focusing on the production of heat from forest biomass residues, of which two from Sweden (Hammar et al., 2019; Kimming et al., 2015) and one from the United Kingdom (Whittaker et al., 2016). The two studies with information available regarding the conversion technology assessed the combustion of forest residues in grate furnaces.

Regarding the modelling approach, two studies performed an attributional LCA and one study performed a consequential LCA. The objective of the latter was to analyse a change in the heat production system in Sweden from natural gas to forest biomass residues and, thus, the consequential LCA appears suitable in this case.

Two studies started the system boundaries at the forest management activities (Hammar et al., 2019; Whittaker et al., 2016) and one started on the residues collection (Kimming et al., 2015). Two studies go until waste end-of-life (Hammar et al., 2019; Kimming et al., 2015), while one study stops at the energy conversion stage (Whittaker et al., 2016). Regarding the

management of woody biomass ash, Kimming et al. (2015) and Hammar et al. (2019) assumed that the ashes returned to the forests and agricultural fields to avoid nutrient deficiency.

The multifunctionality of the energy conversion stage was handled in different ways in the studies analysed. Kimming et al. (2015) performed a system expansion, while Hammar et al. (2019) adopted an energetic allocation between the heat and the electricity generated. It has to be noted that the choice of different procedures to deal with multifunctionality can influence significantly the final environmental impact results. All studies evaluated the climate change impact category only and two of them specify that used global warming potentials from the Intergovernmental Panel on Climate Change (IPCC).

2.6.4.3 Cogeneration production

As shown in Table 5, for the cogeneration of heat and electricity there is only one LCA study from Djomo et al. (2015), which considered the cogeneration from forest biomass residues in Belgium. The functional unit used was 0.86 MJ of heat and 0.14 MJ of electricity and the modelling approach used was attributional. The study evaluated the forest biomass combustion with different technologies, namely, stoker, grate furnace and fluidised bed. The boundaries of the system considered a cradle-to-grave LCA approach and included the processes involved in the forest management, forest residues collection, forest residues transport, forest residues processing (chipping), conversion of forest residues to heat and electricity and the disposal of ash in landfill. The climate change was the impact category assessed and the impact assessment method used was the IMPACT 2002+ (Jolliet et al., 2003).

2.6.4.4 Bioethanol production

In literature there are five studies focusing on the LCA of bioethanol production from the biochemical conversion of forest residues: two studies from North America (Liang et al., 2017; McKechnie et al., 2011), two studies from Europe (Falano et al., 2014; Karlsson et al., 2014) and one from Oceania (Sandilands et al., 2009).

The functional units defined in these studies are quite heterogeneous, being related to distance (1 km driven with ethanol), energy (1 MJ of bioethanol), volume (1 L of ethanol) or mass (1 t of residues) outputs. This lack of uniformity presents a challenge because equivalent functional units are important for comparative purposes.

The bioethanol production assessed in these studies is basically performed through an enzymatic hydrolysis (3 out of 5 studies) followed by fermentation and distillation. One of the

studies considered acid hydrolysis instead of enzymatic hydrolysis and one study does not specify the production technology. All studies used an attributional approach to assess the environmental impacts related to the production of bioethanol. The stages normally included in the life cycle of bioethanol are forest management, forest residues collection and transportation, bioethanol production (hydrolysis, fermentation and distillation), bioethanol combustion and ash management. All studies started the assessment with the forest management, expect Liang et al. (2017) that assessed only the production of bioethanol itself. The end-of-life in two studies was the bioethanol combustion stage, while two studies assessed the ash management. Karlsson et al. (2014) assumed that the ash generated from the non-degraded fraction of forest biomass residues combusted to produce electricity and heat consumed by the facility returned to soil to recover some of the nutrients (particularly phosphorus and potassium, but also magnesium and micronutrients). This study considered nutrient compensation associated with the nitrogen lost during combustion. Falano et al. (2014) considered that the ashes were landfilled.

Bioethanol production is a multifunctional system that can deliver bioethanol, acids, gypsum, heat and electricity. Falano et al. (2014) used system expansion, in which the system was credited only for the products that leave the system to be sold, i.e. the acids and electricity. Therefore, no credits are given for the heat as all of it is used by the refinery. Furthermore, an economic allocation was also performed between ethanol and its co-products in proportion to their respective market prices. When economic allocation is used, around 85 % of the impacts are allocated to bioethanol. Karlsson et al. (2014) also used two different approaches to allocate emissions between co-products at the energy conversion stage: system expansion and energetic allocation. However, the results did not present significant differences for the net impact. The remaining studies did not identify the approach used to handle multifunctionality at the energy conversion stage, but Sandilands et al. (2009) and McKechnie et al. (2011) performed allocation at the forest management, adopting mass allocation.

Regarding the impact categories considered, climate change was assessed in all studies and acidification and photochemical ozone formation were found in three studies (Falano et al., 2014; Liang et al., 2017; Sandilands et al., 2009). Other impact categories found include freshwater eutrophication, freshwater ecotoxicity, human toxicity, land use, marine aquatic toxicity, marine eutrophication, mineral and fossil resource depletion, ozone depletion and terrestrial toxicity. Sandilands et al. (2009) and Falano et al. (2014) used CML 2 impact assessment method (Guinée, 2002) and Liang et al. (2017) used the method TRACI 2.0, whereas the remaining studies do not specify the impact assessment adopted.

2.6.5 LCA of woody biomass ash management

As shown in Table 5, the end-of-life management of the woody biomass ash is poorly covered in LCA studies focused on bioenergy production and most of them considered that ashes are disposed in landfills (Djomo et al., 2015; Falano et al., 2014; González-García and Bacenetti, 2019). Besides, there is a small number of LCA studies which address specifically the management of woody biomass ash (Deviatkin et al., 2017; Teixeira et al., 2019, 2015), as well as a model that allows to obtain the inventory data from woody biomass ash landfilling (Doka, 2003), that are described below in more detail.

Doka (2003) established a model to predict inventory data for the disposal in landfills of different types of wastes, which can be applied for woody biomass ash. The model is available in Excel spreadsheets, allowing the LCA user to specify the waste compositions, defined as concentrations of 41 chemical elements and parameters like heating values, water content, overall degradability and burnability. In addition to the inventories of the waste treatment itself, the model also considers the downstream burdens of all subsequent wastes generated. For example, in a sanitary landfill, the leachate is collected and treated in a wastewater treatment plant, which generates a sludge, which in turn is incinerated and produces ashes that need to be handled. The model is based on general statistical data on waste in Europe and supports waste disposal inventories in the Ecoinvent database (Ecoinvent, 2017).

Although there are several LCA studies that evaluate the valorisation in construction materials of coal ashes (Blankendaal et al., 2014; Celik et al., 2014; Chen et al., 2010; Hossain et al., 2017; Marinković et al., 2017) and ashes from municipal solid waste (Colangelo et al., 2018; Huang et al., 2017), there are only two LCA studies performed with woody biomass ash (Teixeira et al., 2019, 2015).

Teixeira et al. (2015) studied the environmental performance of concrete production with incorporation of woody biomass fly ash, coal fly ash or a blend of the two ashes (with equal mass content) in three percentages of cement substitution (20, 40 and 60 %). The woody biomass fly ash is originated by the combustion of forest residues at a pulp and paper mill. The functional unit chosen was 1 m^3 of concrete and the system boundaries include the production of different concrete compositions, as well as the transportation of the materials to the concrete plant and their mixing. The authors limited the study to a cradle-to-gate perspective because the use and disposal of the new concrete will result in environmental impacts that are similar to those from the traditional concrete with cement. The study performed an allocation based on economic values between the electricity and the fly ashes. The study assessed the following

impact categories: climate change, ozone depletion, acidification, eutrophication, photochemical ozone formation and fossil depletion. The impact assessment method used was the CML 2 (Guinée, 2002). The authors concluded that with the increase of the percentage of woody fly ash incorporation, the environmental impact of concrete production decreased. Besides, the concrete produced with woody fly ash had a better environmental performance than the concrete produced with coal fly ash and the reference concrete (with traditional cement).

Teixeira et al. (2019) studied the production of cement mortar using woody biomass fly ash, coal fly ash or a blend of the two ashes, substituting 50 % of the cement. The biomass fly ash was obtained from a Portuguese cogeneration power plant which used forest residues, such as bark from eucalyptus and pine as fuel to produce heat and power. The functional unit was 1 m^3 of cement mortar and the system boundaries include extraction of raw materials for cement mortar production (cement, aggregates, admixtures, coal fly ash, woody biomass fly ash and hydrated lime), preparation processes, transportation of materials, mixing of raw materials and ends at the mortar plant with the final product. The impact categories and impact assessment method was the same used in Teixeira et al. (2015). The authors concluded that cement mortar production with coal ash presented the best environmental performance, whereas the reference cement mortar (without ashes) presented the worst performance. Cement mortars with woody biomass ash presented higher environmental impacts than those with coal ash due to the fact that they require a higher amount of superplasticiser.

Regarding the studies focused on the application of woody biomass ash as soil ameliorant, there is only one LCA study (Deviatkin et al., 2017), but the woody biomass ash was mixed with other ashes. Deviatkin et al. (2017) studied the environmental impact of forest fertilisation and liming with ashes generated by thermal conversion of a mixture of peat, wood, wood bark, municipal solid waste, sewage sludge and packaging in Finland and compared with landfilling. Fly and bottom ashes generated from the combustion of this mixture were granulated and spread on forest soil. The functional unit was 17,600 ha of forest to be fertilised and neutralised. Only processes that were affected by the operations required for ash valorisation were included in the system boundaries, i.e. woody biomass ash granulation, ash transport and spreading in the soil. The authors expanded the system and the ashes were considered to substitute traditional fertilisers (from rock phosphate and potassium chloride) and liming product (limestone). The study assessed climate change, ozone depletion, human toxicity carcinogenic, human toxicity non-carcinogenic, photochemical ozone formation, acidification, terrestrial eutrophication, freshwater eutrophication, freshwater ecotoxicity and mineral and fossil resource depletion.

The impact assessment method used was the ILCD (EC/JRC/IES, 2012). The results indicated that the application of these ashes in the soil had lower environmental impact on non-toxic impact categories than landfilling, with reductions between 10 and 92 %. However, the use of these ashes presented largest impacts for freshwater ecotoxicity, human toxicity carcinogenic and human toxicity non-carcinogenic (increased by 10-24 %), which is strongly affected by the leaching of heavy metals. The content of V, Ba, Zn, antimony (Sb) and cobalt (Co) in these ashes contributed to 70-90 % of the total impact in freshwater ecotoxicity, while Cr dominated the human toxicity carcinogenic and Zn was responsible for most of the human toxicity non-carcinogenic impact. In addition, the study concluded that the uncertainty related to the leaching data was significant for the overall toxic results of the study.

From the literature review, it can be seen that there is a limited number of LCA studies regarding the environmental impacts of woody biomass ash management and that there is a need of enlarging this number and the primary data available.

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Chapter 3

Attributional life cycle assessment of energy production from forest residues

Environmental impacts of forest biomass-to-energy conversion technologies: grate furnace vs. fluidised bed furnace

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Abstract

Electricity production from forest biomass residues has the potential to significantly contribute to the power mix in Portugal with lesser environmental impact than non-renewable resources. This study focuses on electricity production from the combustion of forest biomass residues from eucalypt logging activities in Portugal using life cycle assessment. In Portugal, several power plants fuelled by forest biomass residues have been commissioned in the last few years. Most of the installations use fluidised bed furnaces and the others use grate furnaces. This study aims to compare the environmental impacts associated with these two alternative combustion technologies. System boundaries include the stages of forest management, collection processing and transportation and energy conversion. The default impact assessment method used is that suggested in the International Reference Life Cycle Data System. In a sensitivity analysis, calculations are performed using the ReCiPe method. For all of the impact categories analysed, the fluidised bed presents the smallest environmental impact. Even when the grate furnace efficiency increases and the fluidised bed efficiency decreases in the sensitivity analysis, the fluidised bed has lower impacts than the grate technology and can be the best alternative in the implementation of new power plants.

Keywords: bioenergy policies, forest biomass residues, combustion, fluidised bed, grate.

3.1 Introduction

The demand for electricity is increasing in both developed and developing countries (EIA, 2016). Finding ways to provide less expensive and more environment-friendly electricity from renewable sources, namely, biomass, is becoming increasingly necessary to mitigate climate change (Nunes et al., 2016). Therefore, electricity production from forest biomass is one of the most important future markets for biomass worldwide. The primary energy supply of forest biomass used worldwide is estimated at about 56 EJ, thus indicating that woody biomass is the source of over 10 % of all energy supplied annually. Overall, forest biomass provides about 90 % of the primary energy annually sourced from all forms of biomass (WEC, 2016).

In Portugal, electricity production from forest biomass is being encouraged with the objectives of decreasing GHG emissions, promoting the development of forest residue harvesting and reducing wildfire hazard (RCM, 2010). In Portugal, approximately 3.1 million hectares of land is covered by forests, which represent 35 % of the national territory (ICNF, 2013), making Portugal a country with great potential for the exploitation of forest biomass. The annual production of forest biomass residues from forest logging in this country is estimated at 0.8–1.2 million dry tonnes per year (IMF, 2014; Viana et al., 2010) and about 47–58 % of these residues come from eucalypt (Dias and Azevedo, 2004; Viana et al., 2010).

In 2016, forest biomass residues was responsible for the production of around 1 % of the total installed capacity of electricity in the country (corresponding to 234 MWe) (DGEG, 2016). However, this percentage is projected to increase through the new strategy implemented by the Portuguese government in 2017, which involves the concession of additional installed electricity production capacity for forest biomass residues plants (corresponding to an additional installed capacity of 60 MWe) (ME, 2017).

The technologies used in Portugal for forest biomass residues combustion are fluidised beds (13 plants in operation with an installed capacity of 192 MWe) and grate furnaces (8 plants in operation with an installed capacity of 42 MWe) (E2p, 2017). Grate furnaces have been used for forest biomass combustion for heat and electricity production for years (Liszka et al., 2013; Onovwiona and Ugursal, 2006). However, fluidised bed furnaces have been proposed as the most suitable option for large-scale biomass combustion because of the growing awareness of environmental impacts linked to forest biomass combustion, energy efficiency and fuel flexibility, which are mostly related to the highly heterogeneous characteristics of biomass (Calvo et al., 2013; Frey et al., 2003; Tarelho et al., 2015, 2011).

The environmental impacts of such alternative forest biomass combustion technologies can be evaluated using LCA methodology. LCA has been applied to quantify the environmental impacts of bioenergy production chains based on forest resources. However, most LCA studies evaluated the environmental performance of forest biomass to produce liquid biofuel or pellets (Gerbrandt et al., 2016; González-García et al., 2011; Jonker et al., 2016; Laschi et al., 2016), whereas only a few considered the environmental performance of using forest biomass for electricity production (Cambero et al., 2015; González-García et al., 2014; Perilhon et al., 2012). Most of these studies focused on the climate change impact category and disregarded other environmental impacts. Moreover, studies based on actual operating data, comparing fluidised bed furnaces with grate furnaces from an environmental perspective, are limited (Lombardi et al., 2015; Nussbaumer, 2003; Rummel and Paist, 2016) and not based on LCA methodology. Therefore, performing an objective and comprehensive comparison between the two technologies and their respective environmental impacts is crucial to provide support for decision-making (Morin, 2014).

This work evaluated the environmental impacts resulting from electricity production in Portugal using eucalypt logging residues (composed of branches, foliage and tops) and considering two types of technologies: grate furnaces and fluidised bed furnaces. This assessment was performed using LCA methodology.

3.2 Methodology

3.2.1 Functional unit and system boundaries

The functional unit (FU) is the production of electricity from the combustion of eucalypt logging residues equivalent to 1 kWh delivered by the power plant to the Portuguese grid. Figure 10 shows the system boundaries divided into three stages: (1) forest management (including site preparation, planting, stand tending and logging); (2) collection, processing and transportation (including bundling, forwarding, chipping, loading and unloading operations); and (3) energy conversion (including forest biomass combustion as well as treatment and final destination of wastes). The transport of workers and machinery as well as the production of capital goods (buildings, machinery and equipment) are excluded.

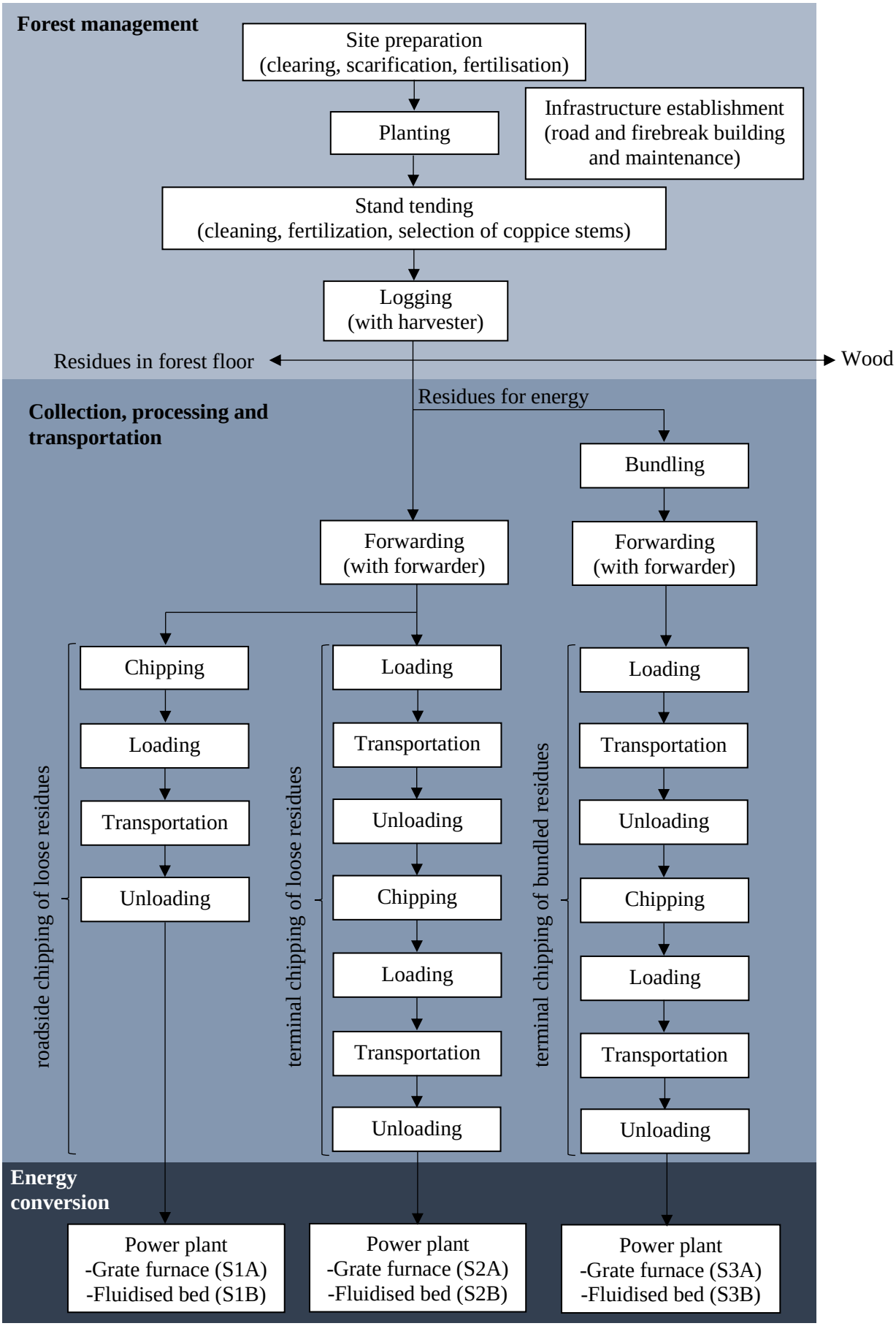

Figure 10 - System boundaries of the biomass supply chain to produce electricity.

At the collection, processing and transportation stage, the processing of eucalypt logging residues has different configurations, namely, roadside chipping of loose residues, terminal chipping of loose residues and terminal chipping of bundled residues, as indicated in Spinelli et al. (2007) and Dias (2014). At the energy conversion stage, two different scenarios are considered: combustion in grate furnaces and combustion in fluidised bed furnaces. Therefore, six different scenarios are evaluated:

- S1A: Loose eucalypt logging residues are collected with a forwarder and chipped at the roadside using a mobile chipper. The chipped biomass is then loaded onto trucks and transported to the power plant. The conversion process of eucalypt logging residues into bioenergy is performed through combustion in a grate furnace.

- S1B: The loose eucalypt logging residue processing is similar to S1A. Energy conversion is conducted through combustion in a fluidised bed furnace.

- S2A: Loose eucalypt logging residues are collected with a forwarder and then transported by tractors with a semi-trailer to a terminal to be chipped. The chipped biomass is then loaded onto trucks and transported to the power plant. Energy conversion is conducted through combustion in a grate furnace.

- S2B: The loose eucalypt logging residue processing is similar to S2A, but energy conversion takes places through combustion in a fluidised bed furnace.

- S3A: Eucalypt logging residues are bundled into cylindrical bales in the clear-cut area with a bundler attachment mounted on a forwarder. The bundles are transported to a terminal by truck. Chipping takes place at the terminal before the transportation of chipped biomass to the power plant also by truck. Energy conversion is conducted through combustion in a grate furnace.

- S3B: The eucalypt logging residue processing is similar to S3A, but the energy conversion takes place by combustion in a fluidised bed furnace.

3.2.2 Allocation

Mass allocation is used at the forest management stage to allocate the environmental burdens among the different biomass outputs, namely, wood, logging residues, bark and stumps, removed from forest. Wood and bark are used by the wood-based industry and the logging residues and stumps can be used for bioenergy or left on the forest floor. The amount of logging residues and stumps left on the forest floor depends on the technical and logistic restrictions as well as on ecological reasons; it ranges between 5 % and 95 % of the total

quantity produced (CBE, 2004). In the current study, half of the eucalypt logging residues and stumps are considered to be left on the forest floor and no environmental burden is allocated to these residues because they are not an output of the system.

The mass proportion of each biomass component is obtained on the basis of typical values (Tomé et al., 2006): 75.3 % for wood, 10.3 % for logging residues, 10.0 % for bark and 4.4 % for stumps.

3.2.3 Inventory analysis

3.2.3.1 Forest management stage

The operations included at the forest management stage are site preparation, planting, stand tending, eucalypt logging and infrastructure establishment.

Site preparation consists of clearing undesirable vegetation by disking, scarifying the soil (ripping followed by subsoiling) and fertilising with a ternary fertiliser [15 % nitrogen (N), 12 % phosphorus pentoxide (P_2O_5) and 9 % potassium oxide (K_2O)] and superphosphate (21 % P_2O_5) applied together with subsoiling. Planting is a manual operation. Eucalypt stands are managed as coppiced stands in three successive rotations, each one at 12 years (Dias and Arroja, 2012; Dias et al., 2007).

Stand tending comprises cleaning, fertilisation and selection of coppice stems. Cleaning is conducted by disking at eight times per revolution (i.e. three successive rotations). Fertilisation is conducted with an N-based fertiliser (30 % N) and a ternary fertiliser (15 % N, 8 % P_2O_5 and 8 % K_2O) applied once per rotation. Coppice stems are selected with a chainsaw once in the second and third rotations.

Eucalypt logging in Portugal is usually performed with a harvester (when large areas are harvested) or a chainsaw. Harvesters are assumed to be used in eucalypt logging. Spinelli et al. (2009) also made a similar assumption.

Infrastructure establishment includes road and firebreak building (once per revolution) and road and firebreak maintenance (six times per revolution).

A neutral biogenic CO_2 balance is considered once the CO_2 released into the atmosphere during the logging residues combustion is re-absorbed by eucalypt stands in their growing cycle. The inventory data for the production of eucalypt wood up to logging with a harvester are taken from Dias and Arroja (2012).

3.2.3.2 Collection, processing and transportation stage

At this stage, forwarders are considered for the collection of eucalypt logging residues. The eucalypt logging residues are loaded onto trucks using the crane of the truck and extraction equipment. During residue storage and chipping, the logged residues suffer a dry matter loss of 2 % (Forsberg, 2000; Whittaker et al., 2011). The inventory data for residues collection and chipping are taken from Dias (2014), with the only difference being the average moisture content of the eucalypt logging residues that arrived at the power plant. Whereas the study of Dias (2014) assumes an average moisture content of 35 %, the present study considers an average moisture of 40 % based on the new measurements undertaken by Silva (2016).

The transport of eucalypt logging residues to the power plant is established according to the optimal distances, from an economic point of view, calculated based on Viana et al. (2010). The truck is considered to return empty in each transport operation. In S1A and S1B, the chipped biomass is transported by a truck to the power plant (payload 20 t) over a distance of 35 km. In S2A and S2B, the forest biomass residues are transported to an intermediate storage park by a tractor with a semi-trailer (payload 10 t) for 10 km and then by a truck (payload 20 t) for more 25 km to the power plant. In S3A and S3B, the bundles are transported to a terminal by a truck (payload 24 t) for 10 km and then the chips are transported for more 25 km by a truck (payload 20 t) to the power plant.

3.2.3.3 Energy conversion

The inventory data for biomass to energy conversion stage are shown in Table 6.

An average lower heating value of the eucalypt logging residues of 17.5 MJ/kg biomass dry basis is assumed (Tarelho et al., 2015). Energy conversion takes place in a power plant with a moving grate (with thermal efficiency for power production equal to 20 %) and in another one with a bubbling fluidised bed (with thermal efficiency for power production equal to 25 %), both containing electrostatic precipitators to reduce particulate emissions.

Table 6 - Inventory data of the production of 1 kWh of electricity from the combustion of forest residues using the grate furnace and the fluidised bed.

	Grate	Fluidised bed
Inputs:		
Biomass (kg-dry)	1.10	0.914
Natural gas (Nm^3)	0.0173	0.00319
Sand (g)		12.6
Water (L)	2.02	2.84
Outputs:		
Product:		
Electricity (kWh)	1.00	1.00
Air emissions:		
CO_2 fossil (g)	37.4	7.18
SO_2 (g)	0.576	0.353
CH_4 (g)	1.29	0.0162
CO (g)	4.98	0.698
N_2O (g)	0.0605	0.1410
NO_x (g)	3.22	2.68
NMVOC (g)	0.141	0.117
PM_{10} (g)	0.481	0.400
$PM_{2.5}$ (g)	0.399	0.281
Wastes:		
Wastewater (L)	0.101	0.141
Ashes (g)	112	94

In the grate furnace [Figure 11 (a)], the biomass is fed from the top of the furnace and the biomass is moved and burned from one side to the other of the furnace to spend proper fuel residence time in the combustion region. The biomass distribution over the grate and the form it travels from one side to the other of the furnace create an uneven distribution of fuel and the subsequent uneven burning of the biomass. This uneven combustion promotes a high emission and increases the unburned content in ashes, thus decreasing the process efficiency. Flue gases leave the furnace in the upper part of the back wall. The typical gas velocity in the furnace is 2.4–3.0 m/s. The temperature in the furnace is controlled at 900–1100 °C and relies mostly on water cooling (boiler tubes). The typical combustion efficiency is 94–97 % (Peña, 2011).

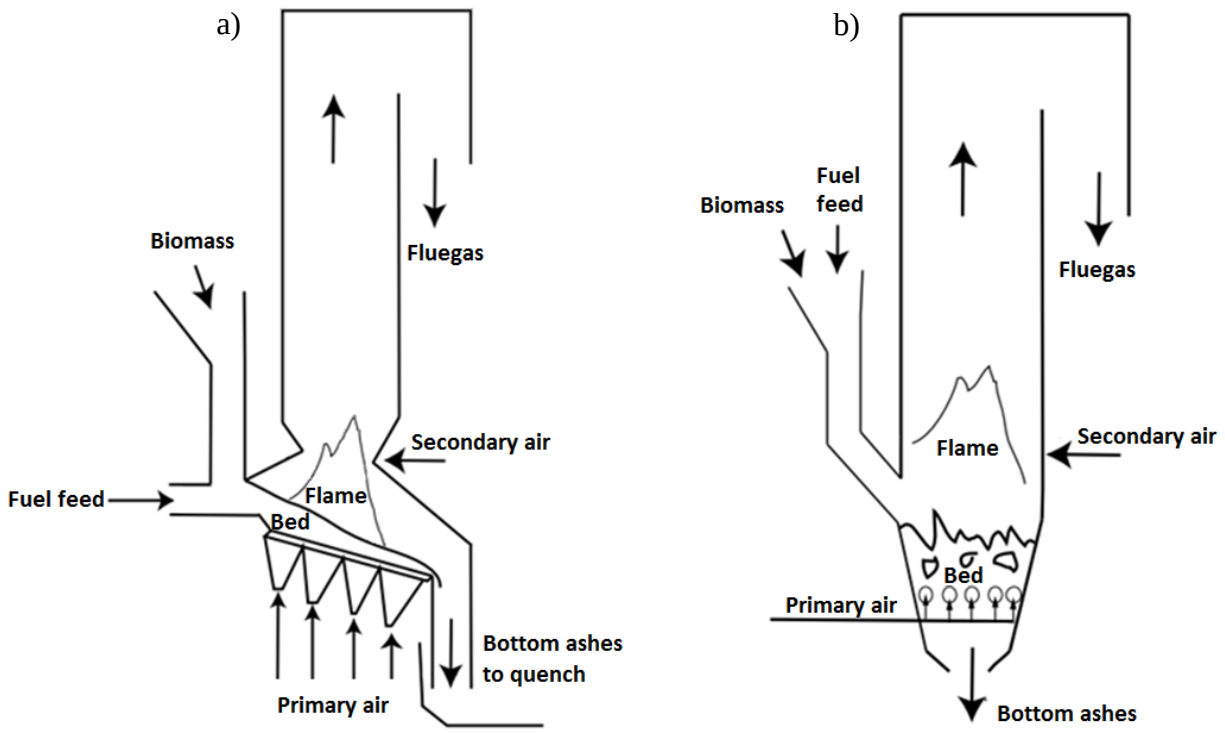

Figure 11 - Biomass-to-energy conversion technologies, (a) grate furnace, (b) fluidised bed. Source: adapted from Leckner (2015).

In the bubbling fluidised bed, the biomass is directly fed to the surface of the hot turbulent bottom bed surface composed mostly of sand and a minor amount of biomass ash. The primary combustion air is injected from the bottom of the fluidised bed furnace for bed fluidisation [Figure 11 (b)]. Fuel combustion takes place in an almost uniformly distributed high-temperature region guaranteed by the high thermal inertia of the bottom bed and the bottom bed gas/solid mix exhibits fluid-like properties. In this way, high mass and energy transfer occur, thus ensuring a proper oxygen–fuel contact and high fuel conversion efficiency. Sand (with high quartz content) is commonly used as bottom bed material (Basu, 2013). The temperature in the furnace is maintained at 750–950 ºC and the gas velocity is between 1-6 m/s. The furnace is water cooled (boiler tubes) and the typical combustion efficiency is close to 99 % (Peña, 2011). One of the key issues that have driven the continuous growth of installed fluidised bed technology plants is their availability to use efficiently a variety of biomass with heterogeneous properties and their higher performance to meet the required emission levels.

In both grate furnace and fluidised beds, the start-up fuel is natural gas until the operating temperatures for stable biomass combustion are reached. Thereafter, the combustion system is operated by using biomass as fuel.

In this work, the average data on biomass, natural gas, sand and water consumed for energy conversion were retrieved from two thermal power plants fuelled by forest biomass residues located in Portugal. The average data from two years (2011–2012) and the average data from eight years (2000–2007) were adopted for the plant with the fluidised bed and that with the grate furnace, respectively.

Aside from the main products of the combustion process (CO_2 and water), flue gas emissions from the combustion of the forest biomass residues also include some pollutants associated either with incomplete combustion (CO, hydrocarbons and carbon particles) or with operating conditions and fuel properties (ashes, SO_2, N_2O, NO and NO_2, often lumped as NOx).

Data on CO_2, SO_2, CO, NOx and particulate matter 2.5 µm or less ($PM_{2.5}$) from eucalypt logging residues combustion were retrieved from the measurements performed by both thermal power plants. The other gaseous pollutant emissions that were not measured at the power plants were calculated based on emission factors for grate furnaces and fluidised bed furnaces with a capacity of less than 50 MW. Specifically, CH_4 emission factors were taken from van Loo and Koppejan (2008), N_2O emission factors were taken from Tarnawski (2004) and NMVOCs emission factors and particulate matter 10 µm or less (PM_{10}) emission factors were both taken from EEA (2016b).

The data used on ash generation during biomass combustion were retrieved from the measurements performed in both thermal power plants. The major mass fractions are bottom ash and fly ash during grate furnace combustion and fluidised bed combustion, respectively (Leckner, 2015). The ashes generated during fluidised bed combustion are also composed of particles from the original sand bed aside from the inorganic material (ash) from the forest biomass residues. The ashes may also contain unburned organic material caused by the incomplete conversion of fuel during the combustion process. Ash and sand were considered to be landfilled. The water consumed and the wastewater produced from the cooling circuit system and the steam generation process were estimated based on data collected from both power plants under analysis.

3.2.3.4 Background processes

The production of ancillary materials, water and energy carriers consumed in the three stages was also considered and the data were taken from the Ecoinvent database (Ecoinvent, 2017). This database was also the source of the inventory data for ash landfilling and wastewater process associated with the energy conversion stage.

3.2.4 Impact assessment

The characterisation factors and the impact categories used in this study were those suggested in the ILCD (EC/JRC/IES, 2010a). The impact categories selected for the analysis were those for which sufficient inventory data were available, namely, climate change (CC), particulate matter (PM), photochemical ozone formation (POF), acidification (AC), freshwater eutrophication (FEu), marine eutrophication (MEu) and mineral and fossil depletion (MFD). The other impact categories, such as ozone depletion and human toxicity, were not addressed because of the lack of inventory data for the foreground system.

3.3 Results and discussion

3.3.1 Impact assessment

Table 7 presents the total impact obtained for the production of 1 kWh of electricity from the combustion of eucalypt logging residues. Figure 12 shows the relative contribution of each stage to the total impact.

Table 7 - Total results of the impact assessment associated with the production of 1 kWh of electricity from the combustion of forest residues.

Impact category	Scenario					
	S1A	S1B	S2A	S2B	S3A	S3B
CC (g CO_2-eq)	152.6	98.30	162.3	106.4	163.1	107.1
PM (g $PM_{2.5}$-eq)	0.7998	0.4371	0.8053	0.4416	0.8021	0.4390
POF (g NMVOC-eq)	4.048	3.166	4.163	3.262	4.144	3.246
AC (molc H^+-eq)	0.004043	0.003162	0.004138	0.003241	0.004132	0.003236
FEu (mg P-eq)	17.71	13.86	17.77	13.91	17.85	13.98
MEu (g N-eq)	1.771	1.473	1.812	1.508	1.799	1.497
MFD (mg Sb-eq)	0.8731	0.7155	0.8760	0.7179	0.8761	0.7180

The forest management stage had a low contribution to the total impact in all impact categories. The only exception is the impact category of MFD, in which forest management is mainly responsible and which accounts for 92–94 % of the total impact (Figure 12). At this stage, phosphate and nitrogen fertilisation was the process that contributed the most to the total impact because of the depletion of indium (In), Cd and Pb.

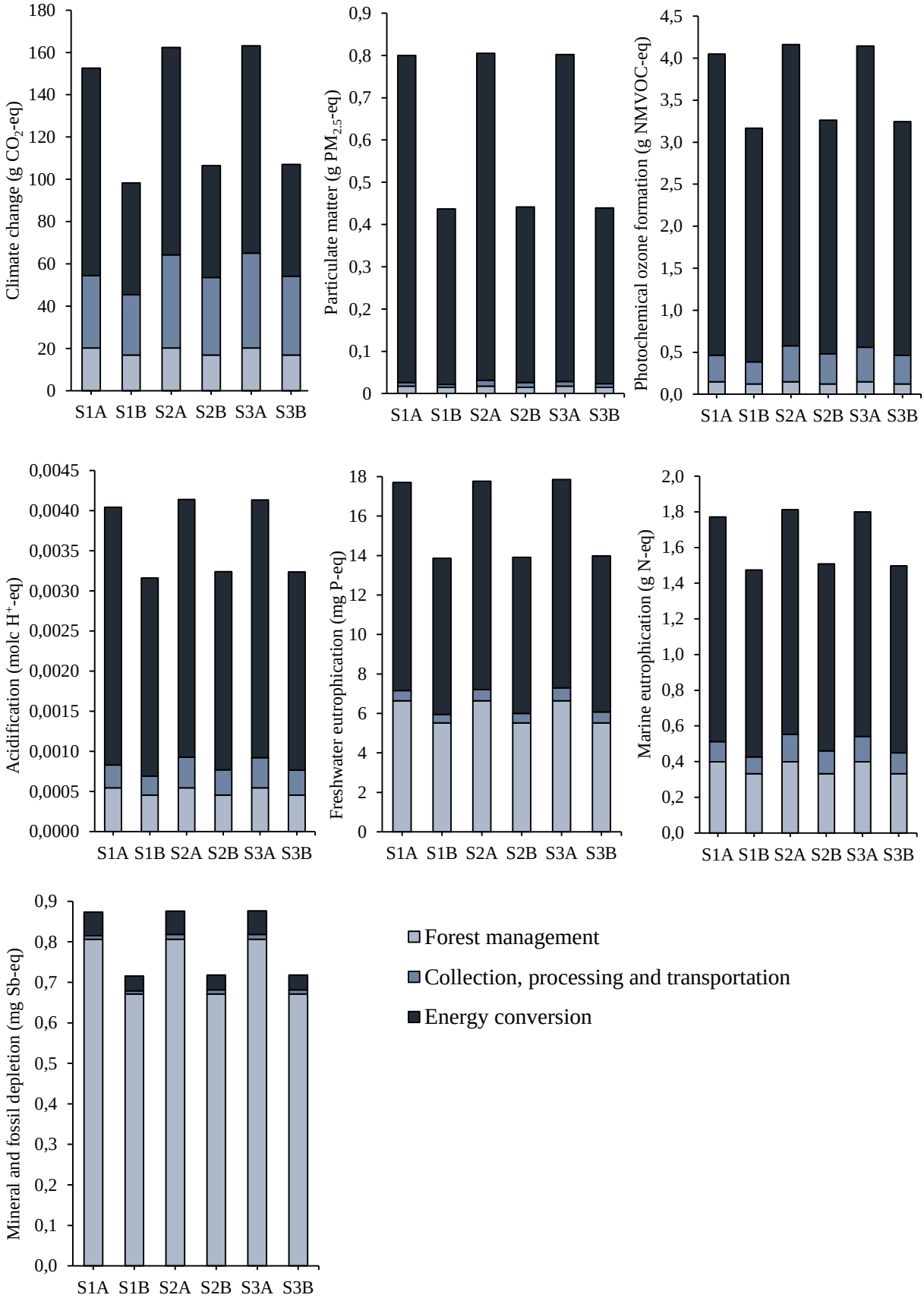

Figure 12 - Environmental impact assessment results associated with each stage of the production of 1 kWh of electricity from the combustion of forest residues.

The collection, processing and transport stage had a low contribution to the total impact in all impact categories. The impact category of CC had the greatest contribution and this stage was responsible for only 22–29 % of the total impact. Among all the impact categories analysed, the system of roadside chipping of loose residues was the biomass supply chain with the smallest impact because of the low diesel requirement in the transport and loading/unloading operations. However, this option is not always feasible. The insufficient space to install a chipper, the difficult conditions of the working field or the small amount of forest biomass residues favours the use of terminal chipping (Spinelli et al., 2007). The terminal chipping of loose residues had lesser impact than the terminal chipping of bundled residues. Spinelli et al. (2012) also found the same result in the low efficiency of bundling. However, bundling residues presents an advantage in logistics. Whereas chippers generally require a truck by the side to receive the chips, bundlers can be stacked on the ground for later collection by conventional log trucks and the transportation of large quantities is possible because forest residues can be compressed (Eriksson and Gustavsson, 2008). Nevertheless, the difference between the two scenarios was low (lower than 6 % of the total impact), as the collection, processing and transport stage had a low contribution to all impact categories.

The contribution of the energy conversion stage (Figure 12) was particularly relevant for the impact categories of CC (49–63 % of the total impact), PM (94–95 % of the total impact), POF (85–88 % of the total impact), AC (76–79 % of the total impact), FEu (56–58 % of the total impact) and MEu (70–71 % of the total impact).

Other studies have indicated the major role of the energy conversion stage in the total environmental impact, as in the present study. For example, González-García et al. (2014) assessed the electricity production from eucalypt logging residues but did not specify the technology used to convert these residues into energy. Using the characterisation method of CML, the study found that energy conversion was mainly responsible for the impact categories of AC, FEu and POF. For CC impact category, forest management contributed the most impact at 58 % and energy conversion contributed 35 %.

Cambero et al. (2015) assessed the utilisation of forest and wood residues as feedstock in small-scale technologies (0.5, 2 and 5 MW) for the production of heat and electricity in Canada. The attributional LCA approach was used to estimate the net GHG emissions of each bioenergy system. The study found that energy conversion was mainly responsible for the CO_2 emissions in a 5 MW power plant. However, in the other plants, the CO_2 emissions were mainly due to the disposal of unused biomass.

Note that a direct comparison with the absolute impacts obtained in earlier studies should be considered with caution because of the different methodological choices (e.g. system boundaries and impact assessment methodology).

In the present study, the results showed that the fluidised bed furnace had the smallest impact among all the impact categories. The relative differences between the grate furnace and the fluidised bed furnace were on average 18 % for MFD, 19 % for MEu, 21 % for AC, 21 % for POF, 23 % for FEu and 34 % for CC. The relative difference for PM among the scenarios was high at 36 %.

The CC impact resulted mostly from the emission of fossil CO_2 and N_2O throughout all the system boundaries, with a special role in the energy conversion stage. The impact of the grate furnace exceeded that of the fluidised bed furnace because CO_2 emission from the production of electricity (from natural gas burning) in the grate furnace (37.4 g/kWh) was higher than that in the fluidised bed (7.18 g/kWh) (Table 6). Currently, no existing technologies have been developed to completely capture the CO_2 formed during the combustion (Nohlgren et al., 2014). In terms of N_2O emissions, the environmental impact was higher in systems using fluidised bed technology than in those using the grate furnace, with emissions corresponding to 0.1410 g/kWh and 0.0605 g/kWh, respectively (Table 6). This finding is due to the fact that temperature in the furnace is higher in the grate than in the fluidised bed and that the lower combustion temperatures observed during fluidised bed combustion are in the range where N_2O formation can be noticed (Tsupari et al., 2007). Nevertheless, N_2O emission during biomass combustion is relatively lower than that of other fuels (Tarelho et al., 2011).

The differences in the PM impact category for the technologies under analysis (grate furnace vs. fluidised bed) were mainly due to the biomass-to-energy conversion efficiency, as the emissions factor of $PM_{2.5}$ per energy unit of each fuel considered was the same in both technologies. The energy conversion (fuel to electricity) efficiency was higher in the fluidised bed (25 %) than in the grate furnace (20 %), as supported by literature in the field. Consequently, the amount of biomass and natural gas needed to produce 1 kWh of electricity was higher in the grate furnace.

The POF originates mostly from the emission of CO and NMVOCs from the energy conversion stage. The impact of the grate furnace exceeds that of the fluidised bed because CO emission for biomass combustion in the grate furnace (4.98 g/kWh) was higher than that in the fluidised bed (0.698 g/kWh). The reasons for this phenomenon were the better and more efficient fuel conversion conditions observed in the fluidised bed than in the grate and the more efficient gas solid contact and uniform temperature distribution guaranteed by the existence of

a high temperature bottom bed in the former (Hasan and Ahsant, 2015; van Loo and Koppejan, 2008). Therefore, high rates of fuel conversion are observed in the fluidised bed and they result in a nearly complete combustion of organic matter, thus minimising the emissions of unburned organic compounds (EPA, 2001). This occurrence also explains the difference in NMVOC emissions in the two energy conversion systems; that is, 0.141 g/kWh was emitted in the grate furnace and 0.117 g/kWh was emitted in the fluidised bed (Table 6).

In AC, NOx emissions (about 62 % of the acidification total impact), which are mostly emitted at the energy conversion stage (95–96 %), contributed the most. NOx emissions were higher in the grate furnace than in the fluidised bed, consistent with the data from Peña (2011) and Basu (2013b). According to these studies, the NOx emissions were inherently higher in the grate furnace than in an equivalent bubbling fluidised bed because the combustion temperatures were significantly higher in the grate furnace, thus increasing the formation of thermal NOx.

The FEu is a nutrient pollution in the form of phosphorous from agricultural fertilisers, sewage effluent and urban storm water runoff. This impact category refers to the excessive growth of aquatic plants or algal blooms because of high levels of nutrients in freshwater ecosystems, such as lakes, reservoirs and rivers (EC/JRC/IES, 2012). The stage responsible for the highest phosphate and phosphorus emissions was energy conversion (56–58 % of the total impact), in which 94 % was due to landfilling of ash and sand, followed by the forest management stage due to fertilisation (37–40 % of the total impact). The impact of grate furnace exceeded that of fluidised bed because the amount of ash deposited in landfills was larger in the former than in the latter. The amount of biomass required to produce 1 kWh of electricity was also higher and the combustion process was not as efficient.

In MEu, N compounds are the limiting factor. The characterisation of the impact of N compounds emitted into rivers, which could subsequently reach the sea, is the target of this characterisation factor (EC/JRC/IES, 2012). The emission that mostly contributed to this impact category was NOx (about 73 % of the marine eutrophication total impact), which is usually emitted in the energy conversion stage (95–96 %). The NOx emission was 3.22 g/kWh in the grate furnace and 2.68 g/kWh in the fluidised bed.

In MFD, the forest management stage mostly contributed to the total impact, between 92–94 %. In this stage, the use of phosphate and nitrogen as fertilisers was predominant, thus causing the depletion of In, Cd and Pb. Although the energy conversion stage had a low contribution to the total impact in this category, the amount of biomass used to produce 1 kWh of energy from the fluidised bed was smaller than that from the grate furnace, thus influencing the use of the mineral and fossil resources during the forest management stage.

3.3.2 Sensitivity

Three sensitivity analyses were conducted to evaluate the effect of technology and methodology assumptions on the impact results.

First, the effect of changing the conversion efficiency into electricity in each power plant was evaluated in relation to the base scenario (20 % for grate plants and 25 % for fluidised bed plants). Selecting this analysis was based on the growing concern around the efficient use of biomass. In the future, the successful use of biomass in high-efficiency combustion systems can reduce the costs of renewable energy and produce power at efficiencies of about 40 % compared with the current combustion-based biomass efficiencies of 16–28 % (IEA, 2007; Jossart, 2015). The main factor affecting the electricity generation efficiency of biomass is fuel quality, as fuels with a high heating value and a low moisture content can yield higher efficiencies than fuels with a low heating value and a high moisture content. In addition, some design and operational changes may be needed to maximise furnace efficiency. Without these adjustments, boiler efficiency and performance can decrease in older and poorly maintained furnaces (Jossart, 2015). In this sensitivity analysis, the conversion efficiency was increased and decreased by 2.5 % to the typical efficiency ranges of the grate furnaces (17.5–22.5 %) and the fluidised bed (22.5–27.5 %). Figure 13 illustrates the results obtained for CC, as this category follows the same trend as the results obtained for the remaining impact categories.

The decrease in the conversion efficiency increases the CC impact to 12 % in the grate furnace and 6 % in the fluidised bed in terms of the base scenario and the increase in the conversion efficiency decreases the results to 10 % in the grate furnace and 5 % in the fluidised bed. When the efficiency increases, the need for biomass to produce 1 kWh decreases from 1.10 to 0.976 kg in the grate furnace and from 0.914 to 0.831 kg in the fluidised bed. Therefore, the impacts associated with air emissions decrease. At the same time, when efficiency decreases, the need for natural gas to support the production of 1 kWh increases from 0.0173 to 0.0198 Nm^3 in the grate furnace and from 0.00319 to 0.00354 Nm^3 in the fluidised bed. In the scenarios in which the grate furnace is used as the conversion technology (S1A, S2A and S3A), the CC results are the most affected ones. S3A has the highest increase in CC impact when efficiency decreases by 2.5 %, from 161 to 184 g CO_2-eq. When the efficiency increases by 2.5 %, climate change decreases from 161 to 143 g CO_2-eq. Therefore, the control of the conversion efficiency into electricity is relevant to the reduction of impacts related to air emissions and resource consumption. Note that even when the grate furnace efficiency

increases to 22.5 % and the fluidised bed efficiency decreases to 22.5 %, the fluidised bed remains the technology with lower impact.

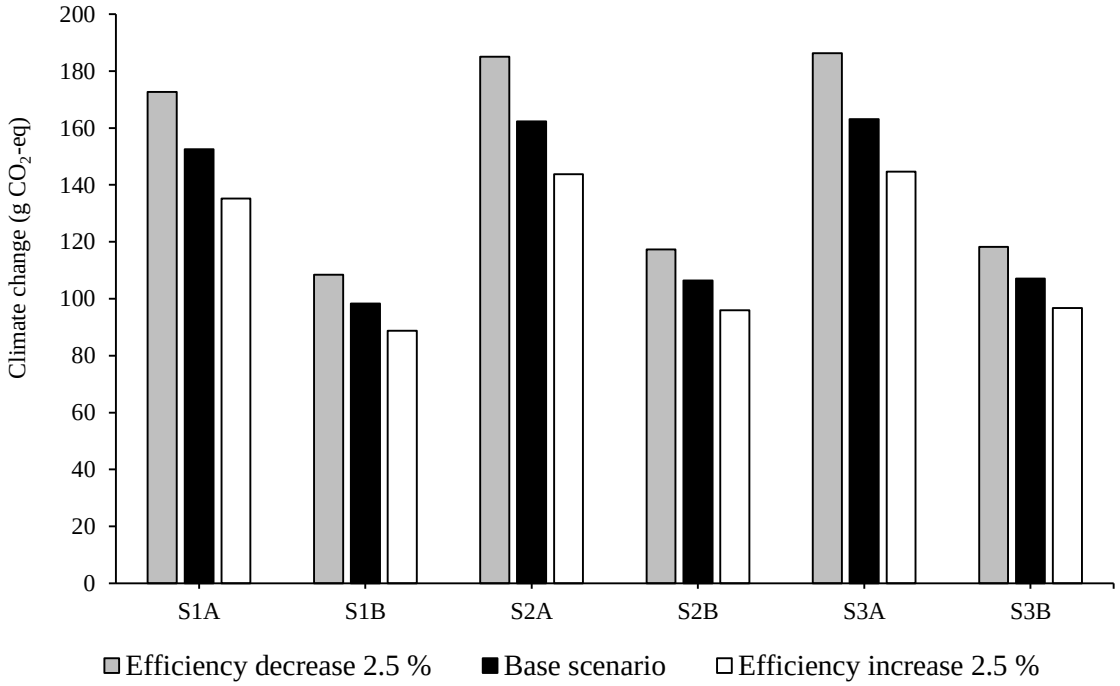

Figure 13 - Results of the sensitivity analysis for climate change impact category: effect of changing the energy conversion efficiency.

Currently, the Portuguese emission standard of NOx for biomass combustion is 600 mg/Nm^3 in the studied power plants, but it will be reduced in 2019 to a maximum of 300 mg/Nm^3 (APA, 2014). Both power plants are expected to comply with new legislation in the future and the selective catalytic reduction (SCR) is an alternative that can be used to reduce NOx emissions in the grate furnace and fluidised bed. The SCR technology alone can achieve NOx reduction up to 90 %; it converts NOx into nitrogen, water and tiny amounts of CO_2 (Can et al., 2012). Therefore, the second sensitivity analysis assessed the effects of using an additional emission control (SCR) in the NOx reduction of 90 %. Figure 14 presents the results obtained for the impact categories affected by NOx emission reduction according to the ILCD method.

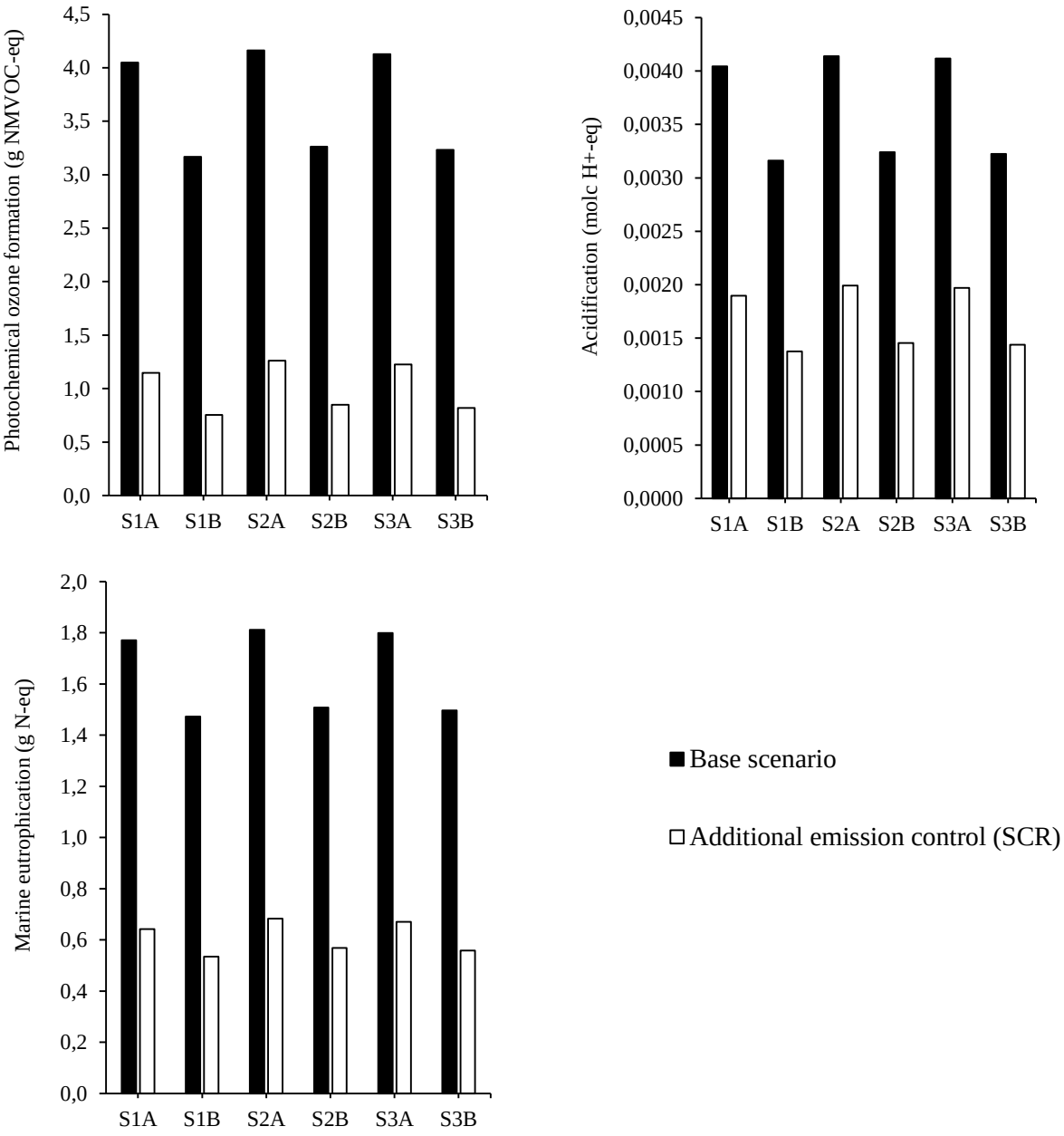

Figure 14 - Results of the sensitivity analysis: effect of the addition of an emission control system (SCR).

The NOx control system contributed the following reduction to the total impact of the grate furnace: 70 % in the POF, 63 % in the MEu and 52 % in the AC impact categories. The NOx emissions were slightly relevant for the other impact categories, such as CC, PM, FEu and MFD. The NOx emission control was moderate in the fluidised bed, contributing a reduction of 59 % in the POF, 52 % in the MEu and 44 % in the AC impact categories. In all cases, the environmental impacts associated with the NOx emissions will be reduced and will comply with the new legislation using SCR.

Thirdly, the effects of using a different impact assessment method are illustrated in Figure 15, which presents the environmental impact results obtained by applying the ReCiPe method (Goedkoop et al., 2013). The results obtained for CC, POF and FEu are the same as those for the base scenario (Figure 12) and are therefore not shown in Figure 15.

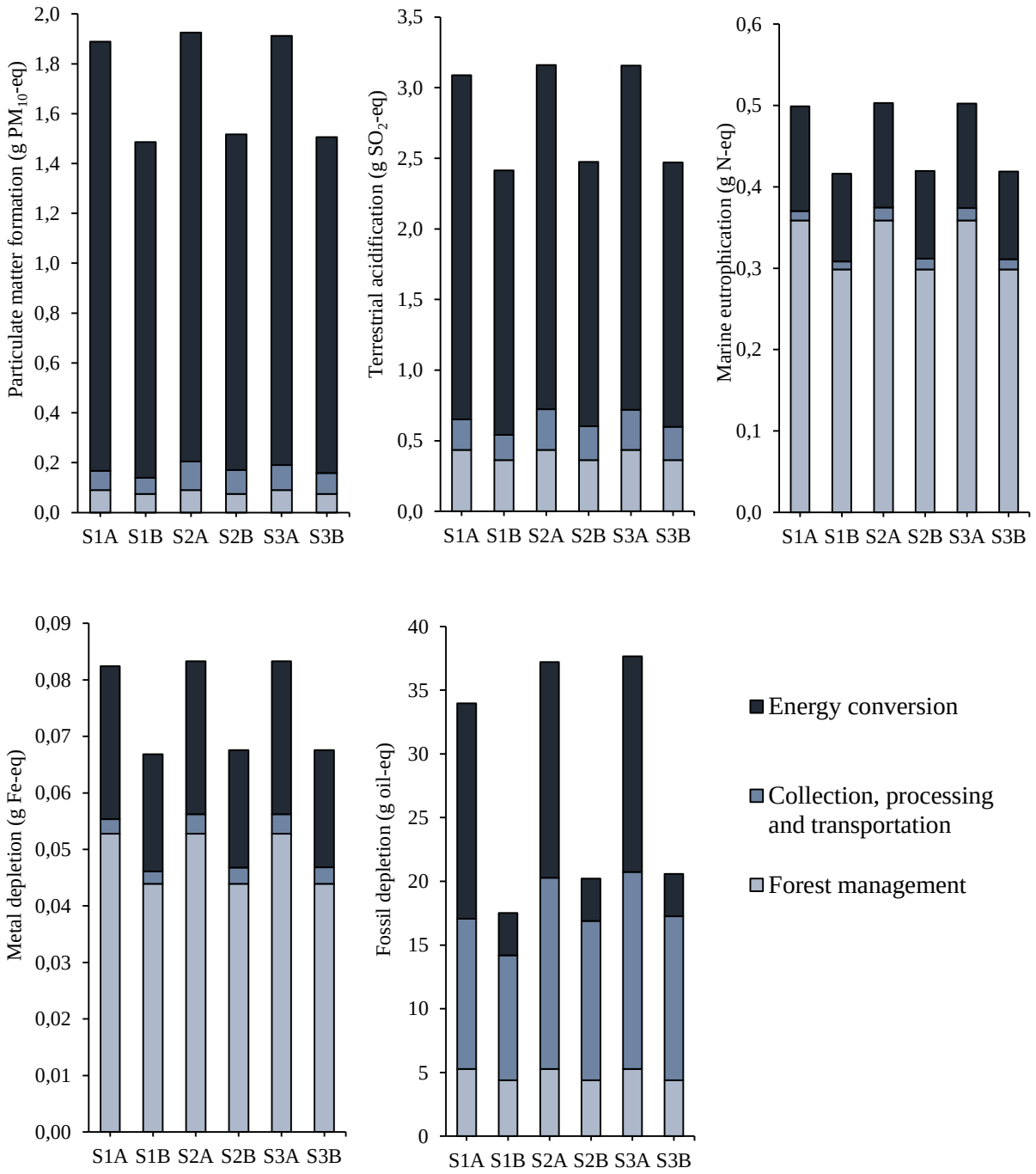

Figure 15 - Results of the sensitivity analysis: effect of changing the impact assessment method (ReCiPe).

For the remaining impact categories, the absolute results obtained with the ILCD method (Figure 12) were not directly comparable with those obtained with the ReCiPe method (Figure 15) applied in the base scenario because of the differences in the category indicators and in the characterisation factors. In addition, depletion was divided into metal and fossil depletion in the ReCiPe method, which was not done in the ILCD method. Despite these differences, the results obtained with the ReCiPe method for the six systems showed the same trends as evidenced by the ILCD method for each impact category chosen. The impacts related to energy conversion using the grate furnace were higher than those obtained using the fluidised bed.

For PM, the difference between the values occurs because the results obtained through the ILCD are expressed in $PM_{2.5}$ and those obtained through the ReCiPe are expressed in PM_{10}. However, the same trend was observed for the relative importance of each stage and for the conversion technology variation.

For terrestrial acidification, the trend confirmed the results obtained for the impact category AC in the ILCD method.

In MEu in the ILCD method, the emissions from the energy conversion stage were the most relevant. In marine eutrophication in the ReCiPe method, the emissions from forest management stage were more relevant than those from the energy conversion stage. This finding is due to the fact that the ILCD method considers the emission of NOx to be more relevant to this impact. NOx emissions represent 83 % of the total impact in the ILCD method and only 26 % in the ReCiPe method. The ReCiPe method considers more relevant the emissions caused by nitrates, which are usually emitted in the forest management stage.

In MFD in the ILCD method, the same trend of the total impacts as that in metal depletion and fossil depletion in the ReCiPe method was observed. However, in fossil depletion in the ReCiPe method, the discrepancy between grate furnace and fluidised bed was high. In this impact category, the most relevant stage depends on the combustion technology. In the grate furnace, the emissions from the energy conversion stage were the most relevant (45–50 % of the total impact) because the combustion of natural gas to produce 1 kWh was high in the grate furnace (0.0173 Nm^3), whereas the fluidised bed only required 0.00319 Nm^3. In the fluidised bed, the emissions from the collection, processing and transportation stage were the most relevant (54–56 % of the total impact).

In metal depletion in the ReCiPe method, the forest management stage was the most relevant, similar to mineral and fossil depletion in the ILCD method. However, its contribution to the total impact was lower (63–65 % in the ReCiPe method vs. 92–94 % in ILCD method). The reason is that the contribution of In and Cd represented 91 % of the total impact in the

ILCD method, whereas the contribution of iron and Cu was the most relevant at only 52 % of the total impact in the ReCiPe method.

3.4. Conclusions

This study evaluates the environmental impacts associated with electricity produced from forest biomass residues. Moreover, it identifies the production electricity stages with the largest environmental impact. This analysis is particularly important given the increasing demand for forest biomass residues in the last few years in the existing biomass power plants in Portugal.

The results show that the configuration of the supply chain has some importance (lower than 6 % of the total impact) in the impact categories of CC, FEu and MFD (from the ILCD method), but it is irrelevant for the remaining impact categories. The energy conversion stage plays a major role in most of the impacts studied; the exceptions are the impact categories of MFD (from the ILCD method), MEu, metal depletion and fossil depletion (from the ReCiPe method). Therefore, this is the stage for which improvement measures, such as the ones proposed in the sensitivity analysis, should be primarily established.

The study shows that the fluidised bed technology presents a smaller impact than the grate furnace technology and can be a good alternative for implementing new power plants. Moreover, an analysis using other indicators (e.g. economic indicators) should be conducted as a complementary tool to predict what technology is promising for electricity production from forest biomass residues. Further research is also needed to analyse the effects of converting the grate technology in Portugal to fluidised bed technology.

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Chapter 4

Life cycle assessment of woody ash valorisation in construction materials

Environmental assessment of valorisation alternatives for woody biomass ash in construction materials

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Abstract

Ash generated during the combustion of forest biomass for energy production is an industrial waste that should be properly managed in order to minimise the respective environmental impacts. Woody biomass ash valorisation in construction materials to replace conventional materials is an alternative that has proven to be technically feasible. On the other hand, it can present potential environmental impacts that should be evaluated to support the choice of more sustainable alternatives. Thus, the purpose of this work is to assess the potential impacts of woody biomass ash valorisation through the incorporation in construction materials using life cycle assessment. The valorisation alternatives consider the incorporation of woody biomass ash in cement mortars, adhesive mortars, concrete blocks and bituminous asphalts. In addition, a base scenario consisting of ash landfilling is also assessed. Four types of woody biomass ashes were considered, namely, fly and bottom ashes generated in a moving grate and fly and bottom ashes generated in a fluidised bed. The impact assessment method used is that suggested in the International Reference Life Cycle Data System for selected impact categories. The results showed that the impacts of ash transport and pre-processing and the impacts of producing the avoided materials are factors that affect the net impact of the valorisation scenarios and should be considered in the end-of-life management of biomass ash. Nevertheless, all valorisation scenarios had a lower impact than landfilling in all the impact categories under study, resulting in a significant decrease of the environmental impacts in some scenarios.

Keywords: adhesive mortar; ash management; bituminous asphalt; cement mortar; concrete blocks; life cycle assessment.

4.1 Introduction

In the past few years, several power plants burning forest biomass residues have been constructed in Portugal as a strategy to reduce forest fire risk and to mitigate climate change (RCM, 2010). With the power plants fuelled by forest biomass currently in operation in Portugal, between 100,000 and 200,000 t of ash are produced annually (Cruz et al., 2017). However, this percentage is projected to increase with the construction of new power plants based on forest biomass residues and the concession of more 60 MWe of installed capacity (ME, 2017). In view of the transition to a circular economy, adequate end-of-life management of ash should be selected, as ash from the combustion of forest biomass residues is typically disposed in landfills (Modolo et al., 2015; Tarelho et al., 2015).

Alternatives means of woody biomass ash valorisation should be pursued, not only due to the rising cost of landfill disposal, which, in turn, is reflected in the cost of the energy produced, but also as a consequence of policies aiming to achieve 'zero waste' (Maschio et al., 2011). The rate of woody biomass ash valorisation in European countries varies. For example, in countries such as Germany, the Netherlands, Denmark, France and the United Kingdom, between 50 and 90 % of biomass ash is valorised, while countries such as Austria, Switzerland, Portugal, Italy and Norway valorise less than 10 %, with the majority being sent to landfill (Williams, 2013).

A possibility for woody biomass ash valorisation includes its incorporation in construction materials through the addition in cement mortar, adhesive mortar, concrete blocks and bituminous asphalts (Ahmaruzzaman, 2010; Chowdhury et al., 2015; Dahl et al., 2010; Ribbing, 2007; Vassilev et al., 2013a). The incorporation rate depends on the biomass feedstock, the ash type (e.g., fly or bottom ash) and the technology of the combustion process (e.g. fixed-bed or fluidised bed).

Woody biomass ash valorisation appears to be technically feasible for the manufacture of cement mortars. In some studies, cement mortars were produced by substituting part of the cement with woody biomass ash (Esteves et al., 2012; Rajamma et al., 2015, 2009; Ramos et al., 2013), while in other studies, cement mortars were produced with woody biomass ash fully or partially substituting the aggregates (Coelho, 2010; Modolo et al., 2015, 2013; Ukrainczyk, 2016). The results showed that woody biomass ash used as additive to produce cement mortars has a number of positive effects, such as decreasing both water demand and expansion due to the alkali-silica reaction (Esteves et al., 2012). Modolo et al. (2015, 2014) also assessed the use of woody biomass ash in adhesive mortars and observed results for water demand that were

equivalent to those obtained for traditional cement mortars. Furthermore, an increase in the tensile adhesion strength was observed. However, a deleterious effect can be observed when the ash incorporation rate is high (Coelho, 2010; Esteves et al., 2012; Tarelho et al., 2015).

Several studies were also performed regarding the use of woody biomass ash in concrete production (Barbosa et al., 2013; Bastos, 2014; Beltrán et al., 2014; Berra et al., 2015; Dias, 2011; Garcia and Sousa-Coutinho, 2013; Lessard et al., 2017). This valorisation alternative is partly based on economic aspects, since the cost of concrete is reduced due to the partial substitution by ash and partly based on the technical benefits, such as lower water sorptivity, reduced bleeding, lower chloride permeability and reduced porosity (Chowdhury et al., 2015; Gunaseelan and Ramalingam, 2016). In addition, Barbosa et al. (2013) showed that the compressive strength of concrete made with woody biomass ash was higher than that of traditional concrete. However, in a similar way to cement mortar, the positive effects of ash incorporation are highly dependent on the amount of ash incorporated in the concrete (Barbosa et al., 2013; Lessard et al., 2017).

Valorisation in bituminous asphalt is also a good alternative for the end-of-life management of woody biomass ash (Scheetz and Earle, 1998). Woody biomass ash can be used for soil stabilisation, subgrade base course material, bituminous pavement additive and mineral filler (Alves, 2013; Dias, 2011; Melotti et al., 2013; Pasandín et al., 2016; Pinho, 2014; Yoshitake et al., 2016). The use of woody biomass ash along roadways has proven to be a beneficial practice because cover materials are scarce in some areas and ash can be used, since the pollution due to its handling in road works is negligible (Ahmaruzzaman, 2010).

On one hand, woody biomass ash valorisation minimizes the demand for traditional raw materials, but on the other hand it can entail environmental impacts (Vassilev et al., 2013a). This trade-off can be evaluated using LCA methodology. The LCA has been widely applied to quantify the potential environmental impacts of waste disposal (e.g. Borghi et al., 2018; Demertzi et al., 2015; Huang et al., 2017; Huber et al., 2017; Xuan et al., 2016). There are LCA studies focusing on ash valorisation in construction materials (Habert et al., 2016; Huang and Chuieh, 2015; Huber et al., 2017; Ondova and Estokova, 2014; Schepper et al., 2014; Seto et al., 2017), but they focus on the application of ash from coal or municipal solid waste.

The goal of this study is to evaluate and compare the trade-offs between the environmental impacts and the benefits of woody biomass fly and bottom ash valorisation in construction materials to support future decision-making regarding the best management options for woody biomass ash. The valorisation alternatives selected were the production of

cement mortar, adhesive mortar, concrete blocks and bituminous asphalt. Besides, ash landfilling was also evaluated as a base scenario.

4.2 Methodology

4.2.1 Functional unit, multifunctionality and system boundaries

The functional unit used in all scenarios evaluated in this study is the management of 1 t (on a dry basis) of each type of woody biomass ash. The ashes under study include both fly and bottom ashes generated from the combustion of mixed forest biomass residues from logging activities, namely from eucalypt (*Eucalyptus globulus* Labill.) and maritime pine (*Pinus pinaster* Ait.). Given that the properties of woody biomass ash are also influenced by the combustion technologies, ashes from two distinct technologies existing in Portugal were considered, more specifically vibrating grate (typical furnace temperatures of around 900-1100 ºC) and bubbling fluidised bed (typical furnace temperatures of around 750-950 ºC). Therefore, four types of woody biomass ashes are evaluated, as follows:

- FG – fly ash generated in a vibrating grate.
- FF – fly ash generated in a fluidised bed.
- BG – bottom ash generated in a vibrating grate.
- BF – bottom ash generated in a fluidised bed.

The current end-of-life management of woody biomass ash produced in Portugal entails disposal in landfill. The alternative scenarios assess four different valorisation alternatives by incorporation in construction materials: (1) cement mortar, (2) adhesive mortar, (3) concrete blocks and (4) bituminous asphalt.

In the valorisation scenarios, a new product is formed (construction material), which represents another function of the system besides ash management. Therefore, to deal with the multifunctionality of the system in the case of the valorisation scenarios, the environmental burdens avoided due to the substitution of traditional raw materials (cement, sand, gravel, or lime) by the ashes are modelled through the system expansion by substitution (EC/JRC/IES, 2010b). Accordingly, a "credit" is given based on the reduced requirement for traditional raw materials according to the rate of incorporation of ashes. An ideal substitution rate of 1:1 was considered given that the construction materials have similar quality regardless of whether they are produced with traditional raw materials or ashes.

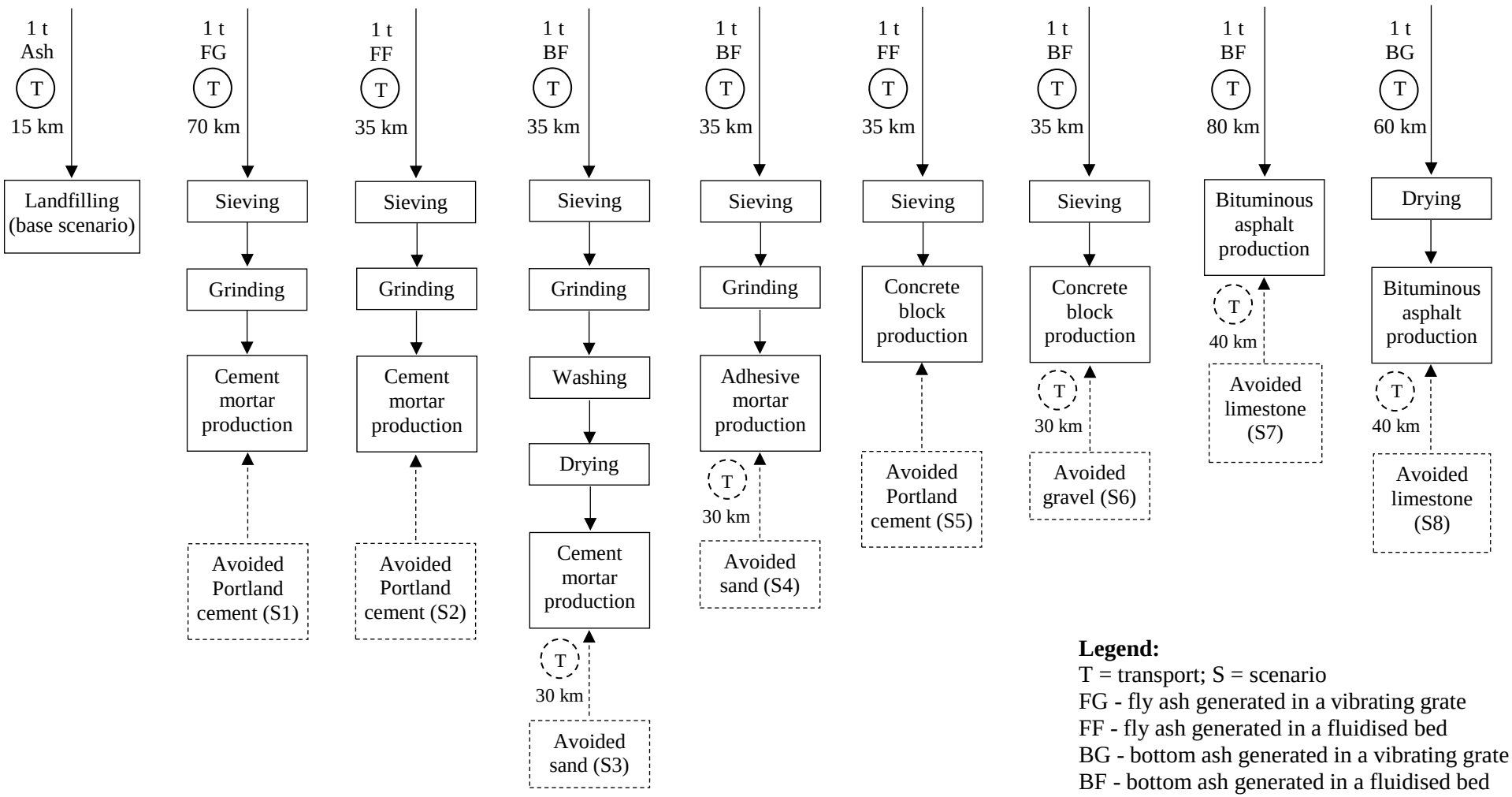

Figure 16 - System boundary for woody biomass ash end-of-life alternatives.

Figure 16 shows the system boundaries divided into the base scenario (landfilling) and the valorisation alternatives in construction materials. The system boundaries of the base scenario include the ash transportation from power plant to landfill, the landfill operation, leachate generation and treatment, sludge incineration and slag disposal. The system boundaries of the valorisation scenarios include the ash transportation from power plant to industrial facility, ash pre-processing and the whole production of construction materials. The system boundaries also encompass the production of the avoided raw materials, upstream activities and intermediate transports. The use and end-of-life stages of the construction materials produced with ashes are not included in the system boundaries, since they are assumed to be analogous to those of the construction materials produced with traditional raw materials and to lead similar impacts.

4.2.2 System description and inventory data

The fly ashes (FG and FF) are collected from the flue gas of the furnace with an electrostatic precipitator, as described by Tarelho et al. (2015). The bottom ash generated in a vibrating grate (BG) is discharged directly into a tank with water for rapid cooling, while the bottom ash from the fluidised bed (BF), composed of a mixture of sand and inorganic components, is collected in dry state. Table 8 shows the average contents of chemical elements and heavy metals in the ashes under study.

Table 8 - Average composition of the woody biomass ash under study.

Composition (g/kg)	FG	BG	FF	BF
H_2O	4.00	213	3.50	1.50
C	54.2	64.4	28.65	3.80
Al	49.4	35.4	60.2	27.2
As	0.0240	0.0125	0.0170	0.0130
Ba	0.470	0.500	0.510	0.210
Br	-	-	0.0700	0.0040
Cd	0.00350	0.00350	0.003333	0.00350
Ca	146	109	135	93.5
Cl	17.5	1.00	14.0	1.00
Cr	0.0385	0.0525	0.0672	0.0442
Cu	0.0514	0.0609	0.0787	0.0657
Fe	22.8	11.5	23.4	8.92
Hg	0.000500	0.000030	0.000200	0.000030

Table 8 (cont.) - Average composition of the woody biomass ash under study.

Composition (g/kg)	FG	BG	FF	BF
K	47.0	15.6	41.4	18.2
Mg	21.9	13.6	17.3	7.21
Mn	3.10	1.75	1.83	0.950
Mo	-	-	0.003	0.00100
Na	9.09	39.9	9.28	5.98
Ni	0.0861	0.0743	0.0394	0.0170
P	6.47	2.01	9.02	2.34
Pb	0.0406	0.0148	0.0776	0.0219
S	6.50	1.00	11.0	1.00
Sc	-	-	0.0150	0.00500
Sn	-	-	0.00500	0.00400
Si	149	220	203	331
Sr	0.340	0.220	0.265	0.0950
Ti	2.25	3.00	3.83	1.67
V	0.0600	0.0700	0.0640	0.0125
Zn	0.207	0.148	0.193	0.132
References	[a, b, c]	[a, b, c]	[a, b, c, d]	[a, b, c, d]

[a] Coelho (2010)
[b] Freire et al. (2015)
[c] Silva (2016)
[d] Tarelho et al. (2015)

4.2.2.1 Landfilling

The base scenario for each type of woody biomass ash evaluated the disposal in a sanitary landfill according to the detailed system boundary presented in Figure 17. The life cycle inventory data for woody ash disposal were built upon the existing model available for waste disposal in Ecoinvent (Doka, 2003b; Ecoinvent, 2017). The model allows the calculation of waste disposal inventories for wastes specified by the user using the waste's composition as a starting point. In this study, the inventory was built based on the different composition of the ashes FG, BG, FF and BF shown in Table 8.

The ashes are spread and compacted by special loaders with an average diesel consumption of 1.3 L/t waste (Doka, 2003). The other energy demand during landfill are related to the electricity and heating oil of the administrative facilities. During landfilling, two main pollutant flows can be generated, namely, landfill gas and leachate. The landfill gas is produced by decomposition of organic substances under the influence of bacteria naturally present in the landfill under anaerobic conditions, generating mainly CH_4 and CO_2. However, the carbon

contained in woody biomass ashes is mostly elemental or carbonate carbon (Bjurström et al., 2014; Straka et al., 2014) for this reason, it remains permanently stored in the landfill and does not generate landfill gas.

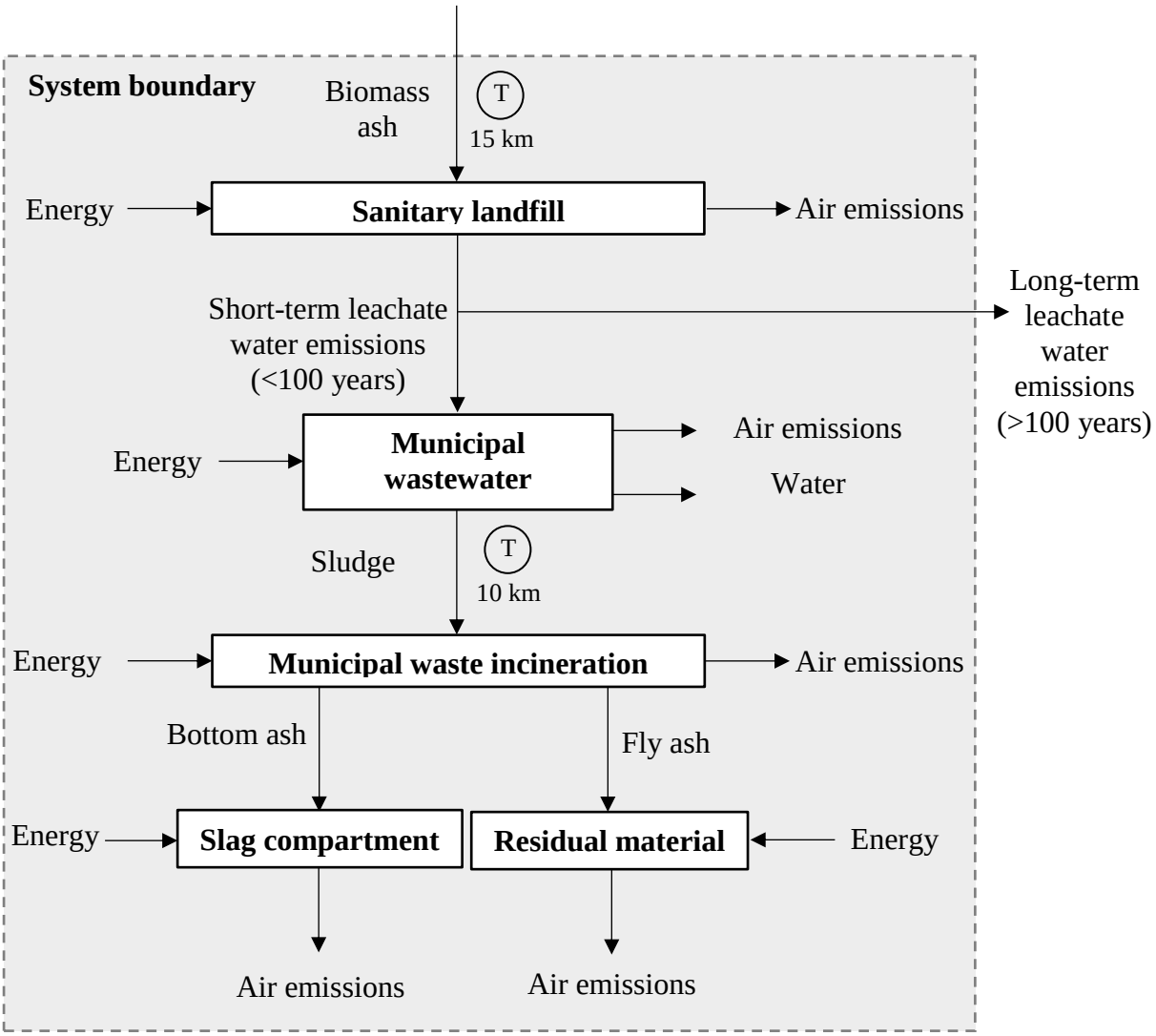

Figure 17 - Detailed systems boundary of landfilling scenarios.

The landfill leachate results from precipitation that falls directly on the landfill and percolates through the waste. The emissions during the first 100 years after waste deposition (short-term) are given from the decomposition rate of waste and the concentration of each element in the waste fraction. The decomposition rate is considered to be 5 % during the first 100 years and homogenous, i.e. all elements are liberated from the waste matrix with the same rate, as assumed by Doka (2003). The model also assesses the long-term emissions occurring 100 to 60,000 years after waste deposition. However, these emissions were not considered in this study, since modelling such long time spans inherently introduces large uncertainties. The

landfill leachate is assumed to be treated in a municipal wastewater treatment plant (volume capacity of 5 million m^3/yr) with mechanical, biological (activated sludge) and chemical (with consumption of iron chloride, iron sulphate, aluminium sulphate, sodium hydroxide, quicklime and hydrochloric acid) treatments. The electricity consumption of the wastewater treatment plant covers the electricity used to aerate the biological tank, for pumps, illumination, etc. In addition, fuel oil is consumed for general space heating in the plant. The sludge produced is further dewatered and transported by lorry to a municipal waste incinerator over 10 km, where it is incinerated in a grate furnace. After incineration, the bottom ashes are collected and quenched in water. The flue gas is passed into an electrostatic precipitator for fly ash separation. After the electrostatic precipitator, a SCR technology is employed to reduce NOx of the flue gas. The model assumes that ammonia is used as reducing agent, titanium dioxide and chromium are used as catalysts and natural gas is used for ancillary energy. The bottom ashes are landfilled in a slag compartment and the fly ash are landfilled in a residual material landfill.

The slag compartment is a separated part of sanitary landfills, used for slags with less than 3 % of organic carbon. Loaders are used to distribute the slag, with an average consumption of diesel of 0.75 L/t of waste (Doka, 2003). The energy demand (electricity and heating oil) is also inventoried for the administrative operations of the slag compartment.

The fly ashes are solidified with cement before being landfilled in the residual material landfill. Loaders are used to place the solidified residual material, with the same average consumption of loaders used in the slag compartment. A similar energy demand as for slag compartment are inventoried for the administrative operations, but its energy demand is distributed to a smaller total mass of waste in the landfill.

The life cycle inventory data for woody ash disposal are shown in Table 9. Air emissions from fuels (light fuel oil, diesel and natural gas) combustion were taken from the Ecoinvent database (Ecoinvent, 2017). This was also the source of data on the production of the inputs listed in Table 9 (the processes are shown in Table S1 of the Appendix A)

Table 9 - Inventory data for the landfilling of 1 t of woody biomass ash (base scenario), according to the type of ash.

	Unit	Base scenarios			
		FG	BG	FF	BF
Inputs:					
Sanitary landfill					
Electricity	kWh	0.0177	0.0177	0.0177	0.0177
Light fuel oil	kg	0.0377	0.0377	0.0377	0.0377
Diesel	kg	1.09	1.09	1.09	1.09
Lubricating oil	kg	0.0240	0.0240	0.0240	0.0240
Wastewater treatment					
Electricity	kWh	0.789	0.775	0.903	0.927
Light fuel oil	kg	0.0217	0.0213	0.0250	0.0257
Iron sulphate	kg	0.0857	0.0266	0.120	0.0310
Aluminium sulphate	kg	0.0232	0.00720	0.0323	0.00838
Iron chloride	kg	0.118	0.0367	0.164	0.0426
Sodium hydroxide	g	0.0288	0.00894	0.0401	0.0104
Quicklime	mg	5.21	1.62	7.28	1.89
Hydrochloric acid	mg	2.90	1.41	2.60	1.00
Municipal waste incineration					
Electricity	kWh	2.87E-04	2.81E-04	3.33E-04	3.42E-04
Natural gas	m^3	2.65E-03	2.59E-03	3.07E-03	3.15E-03
Ammonia	g	1.20	1.18	1.39	1.43
Chromium	mg	0.702	0.688	0.813	0.836
Titanium dioxide	g	0.0343	0.0336	0.0397	0.0408
Water	L	2.38	2.33	2.76	2.84
Slag compartment					
Electricity	kWh	1.57E-05	1.66E-05	1.89E-05	2.14E-05
Light fuel oil	g	0.0335	0.0354	0.0403	0.0454
Diesel	g	0.844	0.892	1.02	1.15
Lubricating oil	g	0.0185	0.0196	0.0223	0.0252
Residual material landfill					
Electricity	kWh	1.02E-05	9.74E-06	1.18E-05	1.17E-05
Light fuel oil	g	0.0217	0.0208	0.0251	0.0249
Diesel	g	0.0968	0.0928	0.112	0.111
Lubricating oil	mg	2.13	2.04	2.47	2.44
Cement	kg	0.0613	0.0587	0.0711	0.0704

Table 9 (cont.) - Inventory data for the landfilling of 1 t of woody biomass ash (base scenario), according to the type of ash.

	Unit	Base scenarios			
		FG	BG	FF	BF
Outputs:					
Water emissions (after leachate treatment)					
Al	g	7.42	5.32	9.04	4.08
As	g	0.190	0.0989	0.135	0.103
Ba	g	1.62	1.72	1.76	0.723
Br	g	0	0	8.81	0.504
Cd	g	0.0156	0.0156	0.0149	0.0156
Ca	kg	0.858	0.638	0.792	0.549
Cl	kg	2.28	0.150	1.87	0.154
Cr, ion	mg	0.0344	0.0469	0.0601	0.0395
Cr, VI	g	0.0115	0.0157	0.0200	0.0132
Cu	mg	3.26	3.87	5.00	4.17
Fe	g	7.96	4.00	8.19	3.12
Hg	mg	0.542	0.0325	0.217	0.0325
K	kg	1.72	0.571	1.51	0.666
Mg	kg	0.611	0.379	0.481	0.201
Mn	g	90.1	50.9	53.3	27.6
Mo	mg	0	0	9.36	3.12
Na	kg	1.88	8.25	1.92	1.24
Ni	g	0.152	0.131	0.0695	0.0301
PO_4^{3-}	g	19.5	6.06	27.2	7.06
Pb	mg	1.32	0.480	2.51	0.708
SO_4^{2-}	kg	0.443	0.0798	0.729	0.0836
Sc	g	0	0	0.0363	0.0121
Sn	mg	0	0	0.616	0.493
Si	g	22.5	33.1	30.6	49.8
Sr	g	0.502	0.325	0.391	0.140
Ti	g	2.85	3.81	4.86	2.11
V	g	0.161	0.187	0.171	0.0335
Zn	g	0.151	0.108	0.141	0.0965
Air emissions (sludge incineration)					
CO	g	0.530	0.520	0.614	0.632
CH_4	g	0.0152	0.0149	0.0176	0.0181
NH_3	g	0.0185	0.0181	0.0214	0.0220
NO_x	g	0.756	0.742	0.877	0.902
NMVOC	g	0.149	0.146	0.173	0.177
$PM_{2.5}$	g	0.0142	0.0139	0.0165	0.0169

Table 9 (cont.) - Inventory data for the landfilling of 1 t of woody biomass ash (base scenario), according to the type of ash.

	Unit	Base scenarios			
		FG	BG	FF	BF
PM_{10}	mg	0.0714	0.0700	0.0827	0.0851
Al	g	0.187	0.132	0.229	0.102
As	mg	0.0607	0.0316	0.0430	0.0329
Ba	g	0.0253	0.0269	0.0275	0.0113
Ca	g	0.155	0.116	0.144	0.100
Fe	mg	2.66	0.875	3.61	0.969
Mg	g	0.0915	0.0567	0.0720	0.0301
Mo	mg	0	0	0.0156	0.00519
P	g	0.0115	0.00357	0.0160	0.00415
Sc	mg	0	0	0.0166	0.00555
Sn	mg	0	0	0.00114	0.000909
Si	g	0.818	1.20	1.11	1.81
Sr	mg	0.0487	0.0315	0.0379	0.0136
Ti	mg	2.77	3.69	4.72	2.05
V	mg	0.0156	0.0182	0.0166	0.00325

4.2.2.2 Cement mortar production

Cement mortar is a mixture of binders, aggregates and water. It is suitable for walling, plastering and binding together bricks or stones. It is distinguished from concrete by the maximum size of its aggregates. In general, cement mortar contains only aggregates with a maximum grain size of 2–4 mm (Locher and Kropp, 2001), while concrete can withstand sizes up to 10 mm. In this study, Portland cement (CEM I 42.5 R) is used as binder and is produced in the mortar facility. Silica sand is used as aggregate and is mixed with cement and water in a ratio of 6:2:1 (Modolo, 2006).

Woody biomass ash can be used in cement mortar production to substitute for aggregate or binder, depending on the type of ash. The type of ash, the avoided material and the substitution rate were established based on previous studies, shown in Table S2 of the Appendix A. Based on the physical, chemical and morphological properties, it is reported that fly ash has substantial potential for use as a binder in cement mortar materials and that BF has potential to substitute sand (Coelho, 2010; Modolo et al., 2013; Rajamma et al., 2015). Regarding fly ashes, binder substitution by up to 10 wt.% fly ash ensures the maintenance of the flexion and compression strength of the cement mortar at a curing time of 28 days (Coelho, 2010). Substitution by BG revealed a great decline in mechanical strength due to the high

porosity and poor compaction of this type of ash (Coelho, 2010). With regard to BF, it is possible to incorporate up to 100 wt.% ash in cement mortars as sand substitute (Modolo, 2006; Modolo et al., 2013). However, due to the occurrence of the internal expansive phenomenon responsible for the deterioration of structures (Esteves et al., 2012), a scenario in which the ash was washed several times to remove its soluble components (alkalis, chlorides and sulphates) was considered in order to minimize this phenomenon (Modolo et al., 2013). The alternative scenarios for woody biomass ash valorisation through integration in cement mortar production are as follows:

- Scenario 1: production of cement mortar with FG (substituting for cement).
- Scenario 2: production of cement mortar with FF (substituting for cement).
- Scenario 3: production of cement mortar with washed and dried BF (substituting for silica sand).

The use of ash in mortars is conditioned by additional operations of grinding and sieving to match the sizes of the materials that will be substituted. An average energy consumption of 11.9 kWh/t is expected for ash grinding and 2.2 kWh/t is expected for ash sieving (Gentil and Vale, 2015; Sabedot et al., 2011).

In scenario 3, the ash was previously washed with deionised water (liquid/solid ratio of 2 L/kg) with a running capacity equal to 10 ton/h (Modolo et al., 2013). The average weight loss of fly ash on washing is equal to 12 % (Berra et al., 2015). The ash is dried to remove excess moisture, with an average energy consumption of 0.2 MJ/kg (Kasser and Pöll, 1998).

Table 10 presents the inventory data for the production of cement mortar using woody biomass ash. The inventory data of the production of inputs and wastewater treatment were taken from the Ecoinvent database (Ecoinvent, 2017) and their respective processes are described in Table S3 of the Appendix A.

4.2.2.3 Adhesive mortar

Adhesive mortar is marketed to ensure adhesion between tiles (ceramic or ornamental stones) and the concrete wall where they are applied. It is produced from a combination of cement, aggregates, water and an additive adhesive, which can vary depending on the use. Since BF is composed mostly of silica (quartz) sand particles, with particle sizes from a few micrometres up to some millimetres and a mean particle size of around 0.950 mm, it is very similar to conventional silica sand used as aggregate in adhesive mortar (Modolo, 2014; Modolo et al., 2015). Therefore, in scenario 4, the valorisation of BF in adhesive mortar was considered with substitution for silica sand (Modolo et al., 2015). Replacement percentages of up to

50 wt.% have been reported without negative impact on the chemical or physical properties of the adhesive mortar, as concluded by the studies presented in Table S2 of the Appendix A.

Table 10 presents the inventory data for the production of adhesive mortar using BF ash. The Ecoinvent database was the source of inventory data for the production of the inputs and wastewater treatment (Ecoinvent, 2017).

Table 10 - Inventory data for the valorisation scenarios.

	Valorisation scenarios							
	1	**2**	**3**	**4**	**5**	**6**	**7**	**8**
Inputs:								
Portland cement (t)	9.0	9.0	0.33	7.2	9.0	0.79		
Sand (t)	30	30		1.0			25	12
Gravel (t)					63	4.0		
Copolymer (t)				0.77				
Limestone (t)				5.3			9.0	4.0
Bitumen (t)							3.1	1.5
Ash (t)	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0
Water (m^3)	5.0	5.0	0.17	5.5	7.9	0.63		
Electricity (kWh)	1126	1126	107	593	147	14	1068	590
Diesel (MJ)					756	60	830	415
Heat (MJ)					555	44	57692	28846
Avoided materials:								
Portland cement (t)	1.0	1.0			1.0			
Sand (t)			1.0	1.0				
Gravel (t)						1.0		
Limestone (t)							1.0	1.0
Product:								
Cement mortar (t)	40	40	1.3					
Adhesive mortar (t)				21				
Concrete block (t)					79	6.3		
Bituminous asphalt (t)							38	19
Outputs:								
Air emissions								
NMVOC (kg)							7.3	3.7
Benzo(a)pyrene (mg)							17	8.9
Wastewater (m^3)	4.2	4.2	0.14	4.7	6.7	0.53		

The sand manufacturing process and all upstream activities were taken from the Ecoinvent database (Ecoinvent, 2017). In adhesive mortar production with woody biomass ash, additional operations of grinding and sieving are required to achieve a particle size similar to that of the material that will be substituted. The electricity consumption of these processes was described in Section 4.2.2.2.

4.2.2.4 Concrete blocks production

Concrete is one of the most widely used building materials and is usually found in bridges, buildings, roads and other infrastructures. It is produced from a combination of Portland cement, aggregates (sand and gravel) and water. Table S2 of the Appendix A shows the studies concerning concrete production with woody biomass ash. An analysis of the available studies was performed to define the materials to be replaced and the substitution rate without changing the quality of the concrete. Those studies were conducted using ashes from fluidised beds only and for this reason, scenarios for concrete production with ash from a grate furnace will not be evaluated in this study. In terms of workability and compressive strength, it was found that compositions containing up to 10 wt.% FF lead to the best results in the replacement of cement (Dias, 2011; Barbosa et al., 2013; Lessard et al., 2017). When the objective is to substitute for the coarse aggregate, the incorporation of BF up to a limit of 20 wt.% does not jeopardise the concrete workability, compressive strength, or permeability (Barbosa et al., 2013; Bastos, 2014; Lessard et al., 2017). Therefore, the valorisation scenarios for woody biomass ash through the production of concrete are as follows:

- Scenario 5: production of concrete blocks with FF (substituting for cement).
- Scenario 6: production of concrete blocks with BF (substituting for gravel).

In concrete production with woody biomass ash, a previous sieving process is needed to achieve the same particle size as the avoided cement and gravel. The electricity consumption of this process was described in Section 4.2.2.2. Inventory data for the production of concrete blocks using ash are shown in Table 10, while inventory data for the production of the inputs and wastewater treatment were obtained from Ecoinvent (Ecoinvent, 2017).

4.2.2.5 Bituminous asphalt pavement production

The raw materials of bituminous asphalt pavement are asphaltic bitumen (which consists of asphaltite), filler (limestone) and aggregate. An analysis of the studies presented in Table S2 of the Appendix A shows that, typically, the production of bituminous asphalt with woody

biomass ash is done by substituting bottom ash for the filler. The reason for this substitution is the particle size similarity with the limestone, since bottom ash constitutes a coarser fraction of ash. For this reason, the scenarios for asphalt production with fly ash will not be evaluated. According to the results found in Dias (2012), the replacement of the filler by up to 20 wt.% BG resulted in a mixture with similar properties to the reference mixture. However, with regard to BF, it was found that mixtures incorporating up to 10 wt.% ash preserve the product applicability, unlike mixtures with 20 wt.% ash. Therefore, the valorisation scenarios selected for woody biomass ash through bituminous asphalt production are as follows:

- Scenario 7: bituminous asphalt with BF (substituting for limestone).
- Scenario 8: bituminous asphalt with dried BG (substituting for limestone).

In the bituminous asphalt production with BF, the ash is mixed with the remaining aggregates without pre-processing. In the production with BG, the ash is dried since its moisture content is high (21.3 %). The electricity consumption of the drying process is 0.2 MJ/kg of ash (Kasser and Pöll, 1998).

Inventory data for the production of bituminous asphalt using ash are displayed in Table 10. Inventory data for the production of the inputs and wastewater treatment were taken from Ecoinvent (Ecoinvent, 2017).

4.2.2.6 Avoided materials

In the valorisation scenarios, avoiding the use of Portland cement, sand, gravel and limestone is considered due to their substitution by woody biomass ash. Inventory data on the production of these avoided materials were taken from Ecoinvent (Ecoinvent, 2017).

The Portland cement production consists, basically, of the acquisition of raw materials (limestone, clay, gypsum and sand), transport to the plant, clinker production and final blending (Kellenberger et al., 2007). The raw materials pass through a variety of sizing operations and are transformed into clinkers by the calcination process. During clinker production in rotary kilns at temperature of up to 1500 °C, there is the consumption of different fossil fuels (Kellenberger et al., 2007).

Sand and gravel result from the process of mining of different types of rocks (Kellenberger et al., 2007). After being transported to the industrial plant, the sand and gravel are directed to a crusher for size reduction and then to a screener for sizing differentiation.

Limestone production includes mining, crushing, washing, drying and sieving (Kellenberger et al., 2007). Limestone is mined by opencast quarrying and broken in a crusher

to a maximum size of 200 mm. After reaching the desired size, the rock is conveyed to the washing and classifying installation (Kellenberger et al., 2007).

4.2.2.7 Transportation

Table 11 presents the transport profiles in the different systems and the distances considered for woody biomass ash and the avoided materials. It was assumed that neither ash nor the avoided materials suffer losses during transportation. Inventory data for diesel consumption and air emissions during transportation were taken from the Ecoinvent database (Ecoinvent, 2017).

Table 11 - Transport profile.

Material transport	Distance (km)	Type of transport	Load (t)
Power plant to landfill			
BG and FG	15[a]	Freight lorry, EURO 3	7.5-16
BF and FF	15[a]	Freight lorry, EURO 3	7.5-16
Power plant to concrete and mortar facility			
FG	70[b]	Freight lorry, EURO 3	7.5-16
BF and FF	35[b]	Freight lorry, EURO 3	7.5-16
Power plant to asphalt facility			
BG	60[b]	Freight lorry, EURO 3	7.5-16
BF	80[b]	Freight lorry, EURO 3	7.5-16
Avoided materials to concrete and mortar facility			
Cement	0	-	-
Sand	30[b]	Freight lorry, EURO 3	16–32
Gravel	30[b]	Freight lorry, EURO 3	16–32
Avoided materials to asphalt facility			
Limestone	40[b]	Freight lorry, EURO 3	16–32

[a]The transport distances of woody biomass ash from the power plant to landfill were based on expert judgment, since the power plants under study deposit their ashes in a nearby landfill or in its own landfill.

[b]The transport distance based on distances of existing manufacturing units in Portugal.

4.2.3 Impact assessment

The characterisation factors and the impact categories used in this study were the ILCD as recommended by the Joint Research Centre (JRC) of the European Commission (EC/JRC/IES, 2012). The impact categories selected for the analysis were those for which sufficient inventory data were available, namely, CC, PM, POF, MFD, FEu, AC, human toxicity carcinogenic (HTc), human toxicity non-carcinogenic (HTnc) and freshwater ecotoxicity (FEc).

4.3 Results and discussion

Figure 18 presents the environmental impact and the relative contribution of each stage obtained for the base scenario of woody biomass ash landfilling.

The emissions during the sanitary landfill of woody biomass ash are the main hotspot (52-56 % of the total impact) in the case of CC due to CO_2 emissions from diesel combustion during ash spreading, followed by the transport (35-40 % of the total impact). The sanitary landfill is also the main hotspot for PM, POF and AC impact categories. In the case of PM, sanitary landfill represents 75-77 % of the total impact and is mainly related to the emissions of $PM_{2.5}$ due to diesel combustion. In the case of POF, the contribution of sanitary landfill represents 69-72 % of the total impact and in the case of AC it represents 64-67 % of the total impact. The emission of NOx due to diesel combustion is the main contribution to the total impact for POF and AC impact categories.

Sanitary landfill operation is also the main hotspot for the bottom ashes in the MFD impact category (41-44 % of the total impact), while the leachate treatment is the main hotspot for the fly ashes (58-64 % of the total impact).

The leachate treatment is also the main hotspot for FEu, HTc, HTnc and FEc impact categories. In the case of FEu, the leachate treatment represents 95-99 % of the total impact, due to the emission of phosphate to the water. Regarding HTc, the leachate contributed with 93-95 % to the total impact, due to the emission to water of Cr VI, which is present in woody biomass ash. Regarding HTnc, the leachate contribution was 87-93 % of the total impact and was mainly associated with the emission of As to water. With regard to FEc, the leachate contributions were 71-85 % of the total impacts and were essentially due to leaching of V and Zn to water.

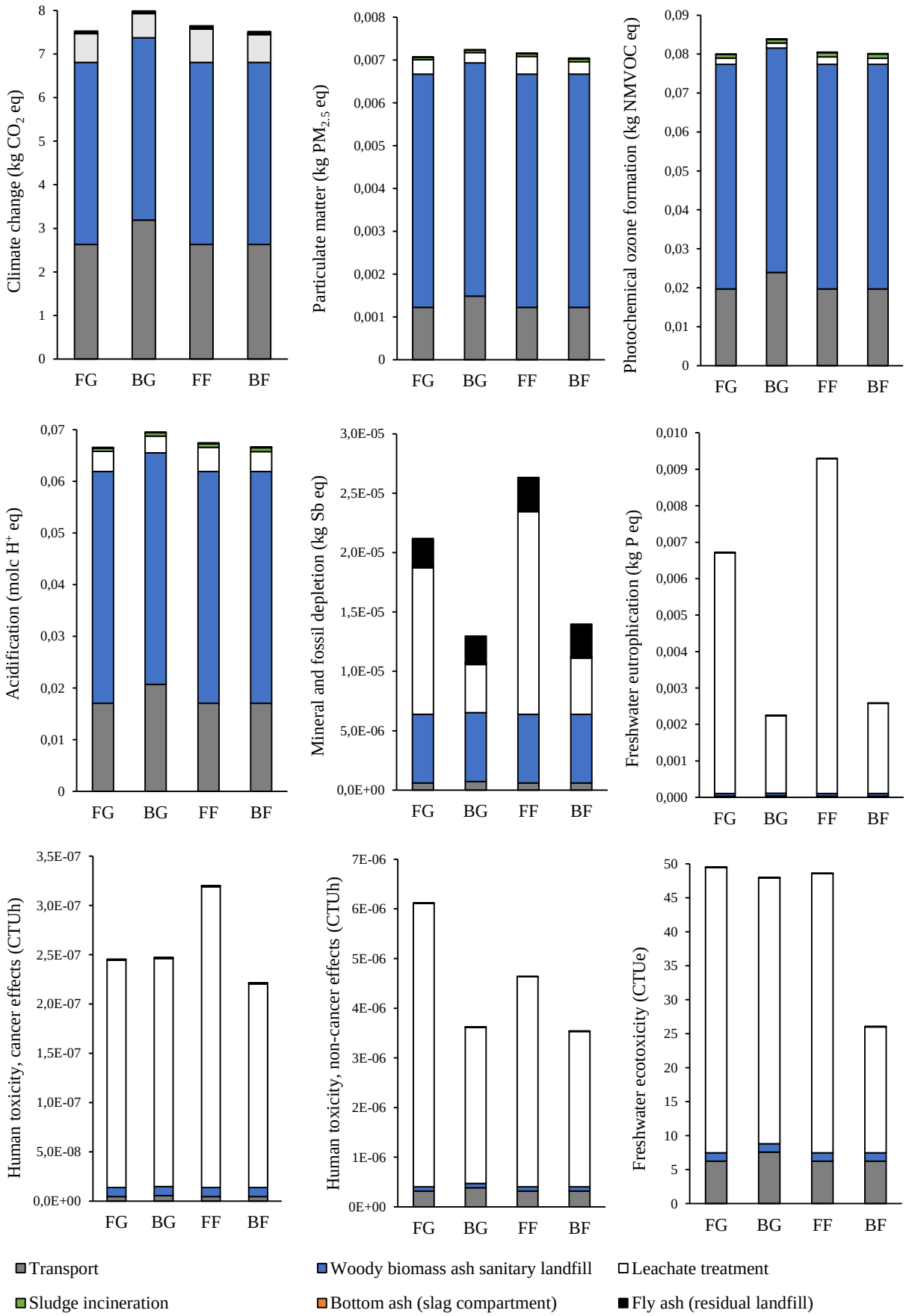

Figure 18 - Impact assessment results for the base scenario (ash landfilling).

Figure 19 presents the environmental impacts of the production of construction materials with woody biomass ash (on the positive axis) and with traditional raw materials (on the negative axis) obtained for the various end-of-life scenarios. The valorisation of woody biomass ash instead of using traditional raw materials in construction materials produces additional impacts related to ash processing (sieving, grinding, washing and drying) and transportation. However, the primary production of cement, sand, gravel and limestone is avoided, as well as the corresponding upstream impacts (transports and energy consumptions).

In the production of cement mortar (scenarios 1, 2 and 3), cement production represents a large part of the total impact in all impact categories except for the PM impact category. Environmental issues related to cement production mainly derive from the manufacture of the intermediate product called clinker due to the large quantities of raw materials (limestone, clay, gypsum, iron ore, bauxite, etc.) and the high fossil fuel consumption required to calcine the calcium carbonates into calcium oxide (Kellenberger et al., 2007). In the CC impact category, the production of cement represents 80–86 % of the total impact, due to the emissions of CO_2 during the calcination process. In the PM impact category, sand production and transportation can represent up to 60 % of the total impact (scenarios 1 and 2) while cement production represents about 33–60 %. In general, PM emissions come from the transfer of sand and cement to silos, truck loading and wind erosion of storage piles (Kellenberger et al., 2007). In the POF impact category, fuel combustion during calcination releases NOx, SO_2 and NMVOCs, leading to the relevant contribution of cement (67–74 % of total impact). In the MFD impact category, the production of cement represents 48–71 % of the total impact, mainly due to the depletion of uranium and In, followed by the consumption of sand (up to 46 % of the total impact in scenarios 1 and 2). The impact in the FEu category mostly originates from cement production (49–55 % of the total impact), mainly due to emissions of phosphate. In the AC impact category, cement production represents 52–61 % of the total impact, as a result of NOx and SO_2 emissions arising from the combustion of fossil fuels (natural gas, heavy fuel oil and hard coal) used to heat up the kiln during the calcination process. The impact in the HTc category mostly originates from Cr VI emission to water due to the production of natural gas consumed during the calcination process (47–57 %). In the HTnc impact category, the production of cement represents 57–60 % of the total impact, mainly due to the emission of Hg to air during the combustion of fossil fuels in the calcination process. In the FEc impact category, cement production represents 46–56 % of the total impact, mainly due to the emissions of silver and Zn to water during the production of natural gas.

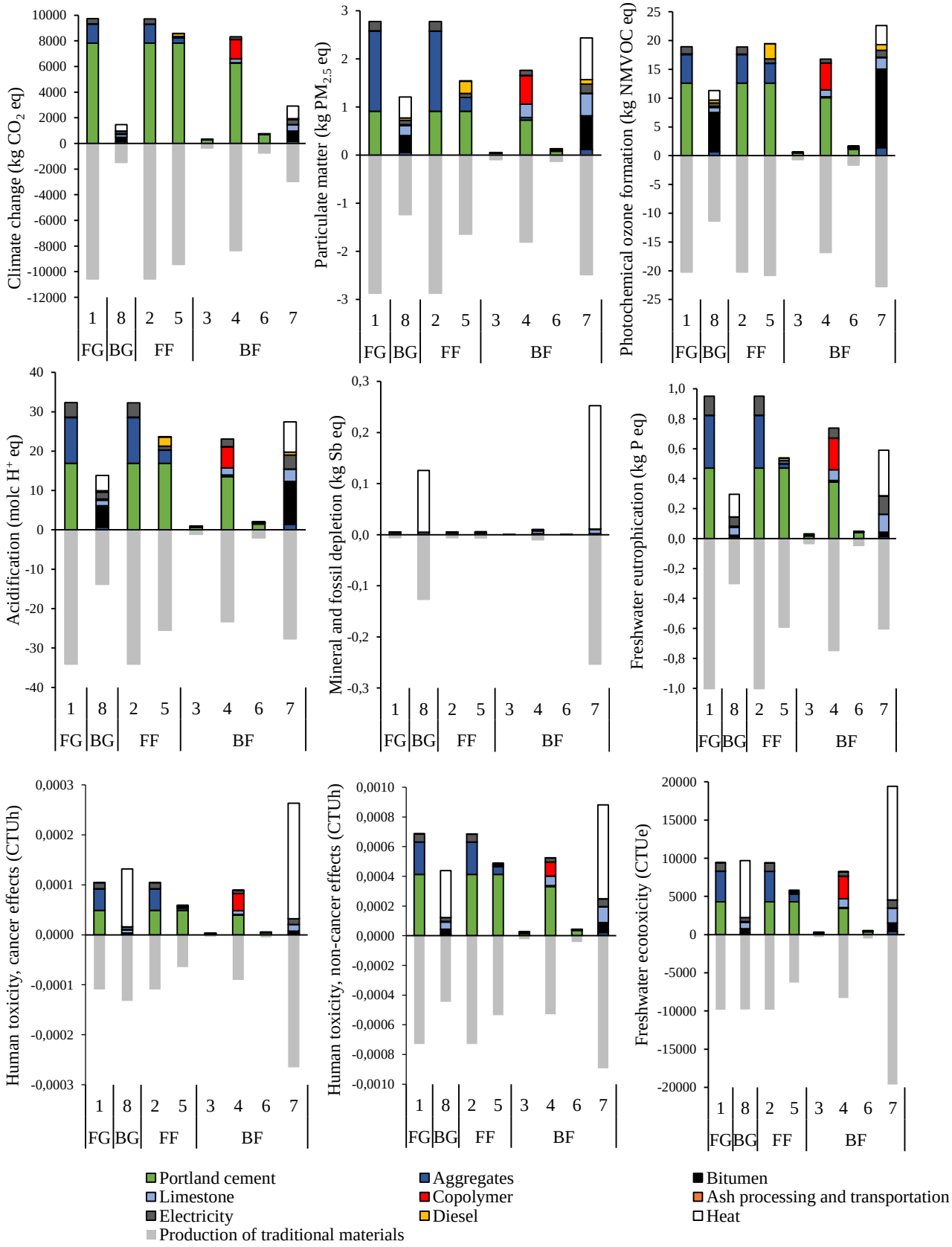

Figure 19 - Impact assessment results for the production of construction materials with woody biomass ash (positive axis) and with traditional raw materials (negative axis), grouped according to the type of ash (1 to 8).

The impacts of cement production can be partially avoided through the substitution by FG or FF (scenarios 1 and 2, respectively) and the impacts of sand production and transportation can be totally avoided through substitution by BF (scenario 3).

In the production of adhesive mortar (scenario 4), cement production is the main hotspot (with contributions ranging from 41 to 75 % of the total impact) in all impact categories, except for MFD, where the extraction, processing and transport of limestone is the hotspot (representing 51 % of the total impact). Although sand may not present the most deleterious environmental effects during the production of adhesive mortar (up to 6 % of the total impact in the categories studied), the extraction process can have an undesired impact on the local flora and fauna, can cause landscape disruption and so on (Dan Gavriletea, 2017). Given the magnitude of environmental problems associated with this activity, substitution by BF appears to be a suitable alternative to avoid sand extraction to produce adhesive mortar.

In the production of concrete blocks (scenarios 5 and 6), the production of cement makes the largest contributions to all impact categories, ranging from 44 to 92 % of the total impacts. However, the impacts of cement production can be minimised through substitution by FF (scenario 5). The impacts of gravel production can be reduced through substitution by BF (scenario 6).

In the production of bituminous asphalt (scenarios 7 and 8), the need for heating is responsible for a large part of the total impact in most impact categories. As bitumen is a semi-solid material, it needs to be heated through an oil-heater (at 130–180 ºC) to become fluid so that it can be mixed with the aggregates. This process is the most intensive one in bituminous asphalt production regarding the use of resources and fuel. Thus, heat production is the main hotspot in the CC, PM, MFD, FEu, HTc, HTnc and FEc impact categories, with contributions ranging from 34 to 96 % of the total impacts. In the POF and AC impact categories, bitumen production represents the main source of impacts (60–61 % and 39–40 %, respectively). Despite this, limestone production requires many processes (e.g. mining, crushing, transportation), which require considerable energy (Oates, 1997) and can have negative consequences for the environment that can be minimised through substitution by BF (scenario 7) and BG (scenario 8).

Figure 20 presents the net impact of the valorisation scenarios and the impact associated with the base scenarios (landfilling). The net impact is the difference between the environmental impact and the avoided burdens. If the net impact is positive, the environmental impacts of construction materials with woody biomass ash are higher than the avoided burdens associated with the utilisation of traditional raw materials; if the net impact is negative, the absolute value

of the avoided burdens is higher than the environmental impacts (i.e. environmental savings are achieved).

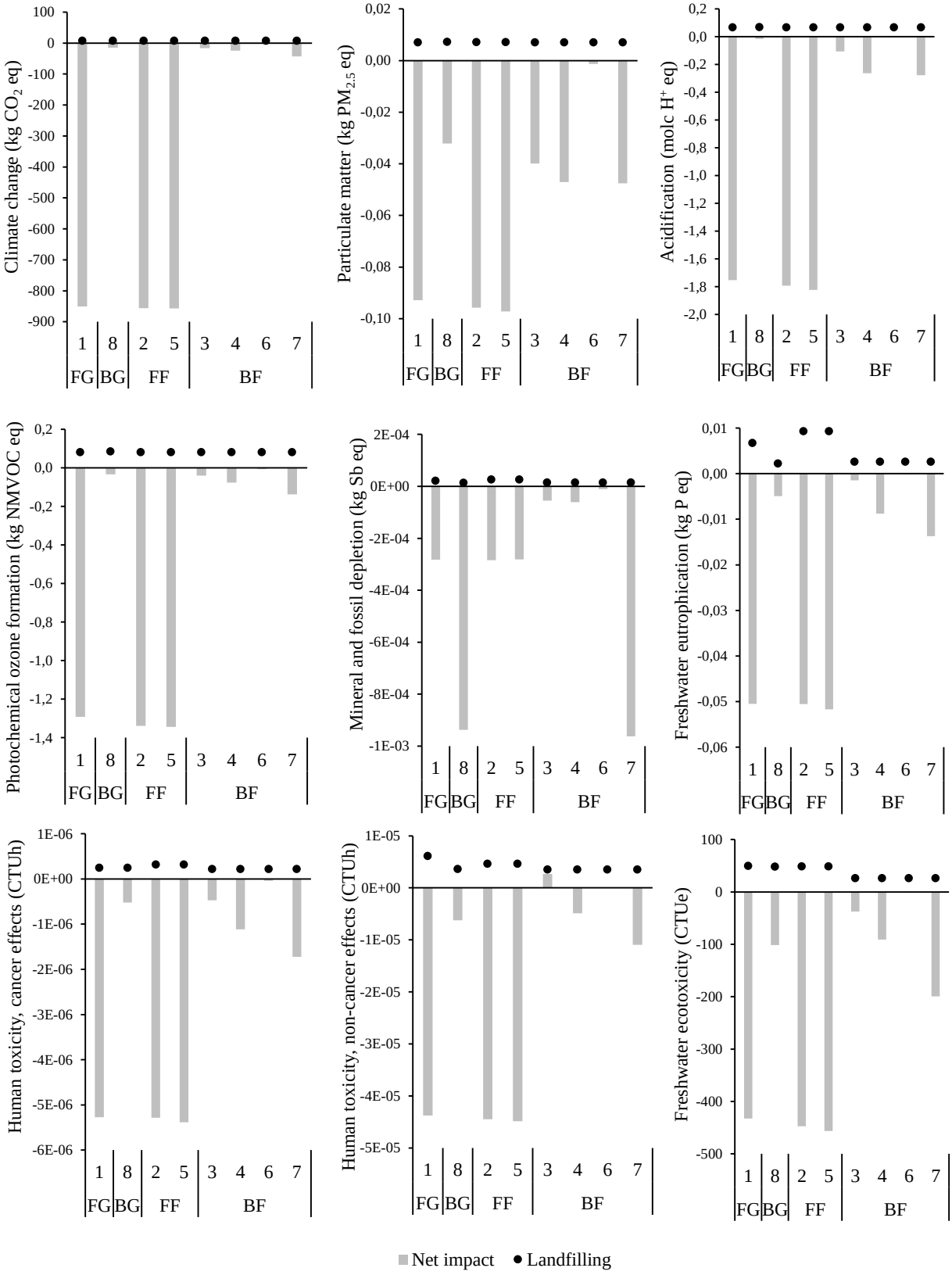

Figure 20 - Net impact results in each scenario (1 to 8). The dots marked in each scenario depicts the impacts from the ash disposal in landfills (base scenario).

The production of construction materials with woody biomass ash (positive axis in Figure 19) entails processes equivalent to the production with traditional raw materials (negative axis in Figure 19). Excluding the common processes, the ash handling (transportation and pre-processing) remains on the positive axis (Figure S1 of the Appendix A), while on the negative axis only the avoided materials (cement, sand, gravel or limestone) and their transportation remain. Therefore, the net impact is the difference between the impacts of ash handling and the impacts of the material avoided in each valorisation scenario.

Regarding FG, there is an environmental saving of all the impacts due to the valorisation of the ashes in cement mortars (scenario 1). Although there are impacts related to ash handling, the impacts avoided due to substitution of cement by ash are higher. An environmental saving of about 850 kg CO_2-eq was achieved in the CC impact category. In addition to cement substitution, the energy required for its production is also minimised. For this scenario and type of ash, cement mortar production has an environmental advantage over landfill disposal, since the impacts do not exceed the baseline scenario.

With regard to the BG, the net impact of the bituminous asphalt production substituting ash for limestone (scenario 8) is negative in all impact categories. Asphalt production with BG presents impacts related to ash transport and pre-processing. The drying process demands a large amount of energy and the transport distance of the ashes from the power plant to the factory is greater than the distance the limestone must be transported from the manufacturing unit to the factory, increasing the diesel consumption and the related impacts. However, the avoided burdens related to the substitution of limestone by ash are higher than the impacts of ash processing and additional transportation. An environmental saving of about 14 kg CO_2-eq was obtained in the CC impact category. Besides that, the net impact is lower than the impacts of ash disposal in landfill, indicating that the valorisation of BG in asphalt production is an appropriate alternative for the end-of-life management of this type of ash.

Regarding FF, two valorisation scenarios for construction materials were evaluated: cement mortar production (scenario 2) and concrete blocks production (scenario 5). In both scenarios the ashes are used to substitute for cement and therefore the avoided burdens are the same. However, the pre-processing required to incorporate the ashes into the construction materials is different. In cement mortar production, the ashes are ground and sieved to match the size of the materials that will be substituted, with an average electricity consumption of 14 kWh/t, while in concrete blocks production, only sieving is necessary to achieve the desired particle size, with an average electricity consumption of 2.2 kWh/t. However, the difference between the two scenarios in terms of the environmental impact results is less than 2 %,

indicating that the scenarios analysed for FF have similar benefits. The environmental saving in both scenarios was about 857 kg CO_2-eq in the CC impact category.

The BF is mostly formed from sand added to the fluidised bed and allows several final end-of-life scenarios (3, 4, 6 and 7), due to the greater similarity with aggregates. The results showed environmental savings in all impact categories in these scenarios. The only exception was in the CC impact category in scenario 6, as the contribution of gravel is small in this impact category and the substitution by ashes does not minimize the effects of this impact. Scenario 7 (bituminous asphalt production substituting ash for limestone) presented the largest benefit in all impact categories, followed by scenario 4 (adhesive mortar production substituting ash for sand), scenario 3 (cement mortar production substituting ash for sand) and finally scenario 6 (concrete blocks substituting ash for gravel). In addition to the differences between the materials avoided, the net impact results also reflect the type of pre-processing of the ash. In scenario 7, pre-processing is not required. In scenario 6, ash pre-processing is carried out by sieving. In scenario 4, the ash is ground and sieved, since the size of the ash particles needs to be reduced to produce adhesive mortar. Finally, in scenario 3, the ash is ground and sieved to match the size of the materials that will be substituted and then washed and dried to remove the soluble components responsible for the deterioration of structures, as in this scenario a high amount of ash is incorporated. The impacts of BF valorisation are always lower than those of BF landfilling. Therefore, BF valorisation in construction materials appears to improve the environmental performance in all scenarios, but the improvement is more pronounced if the ashes are used to produce bituminous asphalt, with an environmental saving of about 43 kg CO_2-eq in the CC impact category.

In this study, the stages of use and end-of-life of the construction materials were left out of the system boundaries as they were assumed to be similar and lead to similar impacts in both construction materials produced with ashes and traditional construction materials. However, this may be not completely true as some studies (Allegrini et al., 2015; Di Gianfilippo et al., 2016; Tosti et al., 2018) point out that potentially metals could leach from construction materials during use and end-of-life, affecting negatively the environmental performance of the construction materials with ashes. However, this assessment is still under development, since leaching data are characterised by significant variability due to the intrinsic uncertainty surrounding the sample materials and the definition of the use scenario (Allegrini et al., 2015). Besides, those studies rely on leaching tests (column percolation tests) and may not represent the real leaching effect in the environment. Therefore, further research on this topic should be carried out.

4.4 Conclusions

In this study, the environmental impacts of different valorisation alternatives for woody biomass ash through its incorporation in construction materials were evaluated. Four types of woody biomass ashes were considered: fly and bottom ashes generated in a vibrating grate and fly and bottom ashes generated in a fluidised bed. Besides that, the current end-of-life management of woody biomass ash in Portugal (disposal in landfill) was also considered.

The results showed that the ash transport and pre-processing (drying, loading, unloading and transport) affect the net impact of the valorisation alternatives and should be considered in the end-of-life management of biomass ash. Besides that, woody biomass ash valorisation in construction materials reduces the extraction and processing of virgin materials, the energy consumption required to manufacture new materials and transportation, which in turn avoids air and water emissions and contributes to improving environmental performance.

In detail, the production of cement mortar with fly ash from vibrating grate presented environmental savings compared to the production of cement mortar with traditional raw materials. For instance, regarding the CC impact category, the fly ash valorisation avoided about 850 kg CO_2-eq/t ash.

The valorisation of bottom ash from vibrating grate through the production of bituminous asphalt also revealed benefits in comparison with the traditional production of bituminous asphalt. However, the environmental benefits are limited (e.g., 14 kg CO_2-eq/t ash), since the valorisation of bottom ash from vibrating grate is only adequate to produce bituminous asphalt substituting ash for aggregates or fillers.

Regarding fly ash from fluidised bed, the two valorisation alternatives analysed (production of cement mortar and concrete blocks) have similar benefits, since the difference between the scenarios is smaller than 2 %. The ashes are used to substitute cement in both valorisation alternatives. In comparison with the traditional production of cement mortar and concrete blocks, production with fly ash from fluidised bed avoided about 857 kg CO_2-eq/t ash.

With regard to the bottom ash from fluidised bed, four valorisation alternatives were assessed due to the high similarity of this ash with aggregates. The production of bituminous asphalt presented more benefits than the other valorisation alternatives in all impact categories. In the CC impact category, the production of bituminous asphalt with bottom ashes from vibrating grates avoided about 43 kg CO_2-eq/t ash.

Finally, the results show that all valorisation scenarios have a better environmental performance than ash landfilling, resulting in a significant decrease of the environmental impacts in some scenarios.

Chapter 5

Life cycle assessment of woody ash valorisation for soil amelioration

Life cycle assessment of woody biomass ash for soil amelioration

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Waste Management

Abstract

The increasing use of forest biomass as fuel for power plants due to environmental concerns will certainly increase the amount of woody biomass ash produced. Because of the environmental problems derived from woody biomass ash disposal, an important aspect for the sustainable development of the energy sector is the implementation of effective ash management strategies. The purpose of this study is to assess the environmental impacts of woody biomass ash landfarming for soil amelioration through a life cycle assessment. The baseline scenario corresponds to the current most common practice of woody biomass ash management (landfilling) and two different landfarming alternatives were assessed: liming and fertilisation. Credits were given to the system due to the substitution of three traditional liming products and five traditional fertilisers. Woody biomass ash landfarming presented satisfactory performance in five impact categories under study in comparison to landfilling. When woody biomass ash was used for liming, the environmental savings were more pronounced when substituting hydrated lime. For potassium supply, the substitution of potassium nitrate by woody biomass ash presented the best environmental performance; while for phosphorus supply, the environmental savings were more pronounced substituting single superphosphate. However, in four impact categories, the environmental impacts of ash landfarming exceeded the impacts of ash landfilling, due to the emission of nutrients and trace elements to soil. But this does not necessarily imply increased risks for the environment, as the potential pollutants leaching depends on their bioavailability in the soil.

Keywords: ash management, end-of-life, fertilisation, liming, residue valorisation.

5.1 Introduction

Ash production and management are among the most relevant environmental issues during thermochemical conversion of biomass to energy (James et al., 2012; Modolo et al., 2014; Wang et al., 2012). Woody biomass ash is being produced in increasing amounts and contains a variety of macronutrients and micronutrients, which requires an appropriate recycling strategy (Huotari et al., 2015; Insam and Knapp, 2011). The amount of woody biomass ash produced during energy production varies significantly according to the type of biomass feedstock, combustion technology and operating conditions of the combustion process (James et al., 2012). These aspects have a strong influence in the quality of the resulting ash, therefore affecting its potential use in further applications (Freire et al., 2015).

Woody biomass ash disposal in landfills is still a common practice, that in addition to neglecting the ash valorisation potential, has considerable costs for biomass power plants (Insam and Knapp, 2011). The recirculation of woody biomass ash in forest soils reduces the need for landfills, leads to the return of valuable nutrients to forest ecosystems and counteracts soil acidification, making energy production from forest biomass combustion more sustainable (Bang-Andreasen et al., 2017; Brännvall et al., 2014; Freire et al., 2015). Extensive research has been conducted regarding the application of woody biomass ash in acid soils substituting liming products (Arshad et al., 2012; Cruz et al., 2017; Dvořák et al., 2017; Park et al., 2012). In general, crops respond better to neutral soils than to acid soils and, therefore, the addition of woody biomass ash is a good option for liming, since woody biomass ash is alkaline, contains high values of oxides and hydroxides of calcium and readily reacts with acidic components in the soil (Demeyer et al., 2001; James et al., 2012; Lin et al., 2007). Field experiments showed a fast increase in the pH of the soil when woody biomass ash is used as a liming material compared to using limestone (Arshad et al., 2012; Lickacz, 2002), with this effect being attributed to the chemical composition and particle size of the ashes (Cruz et al., 2017; Lickacz, 2002).

Woody biomass ash has also the potential to be used as fertiliser in forest soils to compensate for the nutrients removed by harvesting and other forest management activities, since it contains macro and micro elements needed by the trees, such as potassium K, Ca, P, Mg, B and Zn and can therefore contribute to reduce the use of traditional fertilisers (Gómez-Rey et al., 2012; Pereira et al., 2016; Scheepers, 2014; Wiklund, 2017). The exception is nitrogen, which is oxidised during combustion. However, the addition of other amendments

together with ash, such as sewage sludge, can supply the nitrogen required by the plants (Ochecova et al., 2014; Pesonen et al., 2016).

Although woody biomass ash valorisation for soil amelioration seems to be a good ecological solution, it is of crucial relevance to assess the potential environmental impacts resulting from this practice (Insam and Knapp, 2011; van Loo and Koppejan, 2008). One of the key factors in this assessment is the release of contaminants to the environment that affect human health and environmental safety (Dung et al., 2018). Traces of element contaminants, such as As, Cr, Cd, Cu, etc., can be found in woody biomass ash in variable concentrations (Augusto et al., 2008; Demeyer et al., 2001; Ohno and Erich, 1990; Pitman, 2006). These elements influence the quality of the ashes, compromising their valorisation and promoting changes in the soil, vegetation and consequently in the composition of groundwater, which could potentially harm the environment as well as humans and several animal and plant species living nearby (Khan et al., 2009; Röser et al., 2008; van Loo and Koppejan, 2008; Vassilev et al., 2013b, 2014).

The evaluation of the potential environmental impacts associated with woody biomass ash valorisation can be performed by using an LCA methodology. LCA has been applied to assess the use of residues for soil improvement, mostly for residues such as urban sewage sludge (Hospido et al., 2010; Sablayrolles et al., 2010; Willén et al., 2017), pig slurry (de Vries et al., 2012; Lijó et al., 2014) and ash from coal or municipal solid waste (Kalmykova et al., 2015; Wang et al., 2017), but no attention has been given to woody biomass ash.

The aim of this study is to evaluate and compare the trade-offs between the environmental benefits and impacts of woody biomass ash valorisation in the soil, in order to provide information to support future decision-making regarding the best management option for the ashes under study. The two alternatives selected were soil improvement as an agent for buffering the pH of acidic soils and adding nutrients through fertilisation. Besides, the alternative of woody biomass ash landfilling was also evaluated as a base scenario.

5.2 Methodology

5.2.1 Functional unit and system boundaries

The FU of the scenarios evaluated in this study is 1 t (dry basis) of each type of woody biomass ash that is disposed by landfilling or landfarming. The ashes under study result from the combustion of woody biomass residues from logging activities of eucalypt (*Eucalyptus globulus* Labill.) and maritime pine (*Pinus pinaster* Ait.) in Portugal. Two different types of

ashes were considered: FG and FF. This study did not consider the valorisation of bottom ash because, according to Silva (2016), the bottom ash does not meet the Portuguese normative requirements (IPQ, 1990) regarding the particle size distribution of liming materials and fertilisers. Figure 21 shows the system boundaries for the woody biomass ash end-of-life scenarios considered.

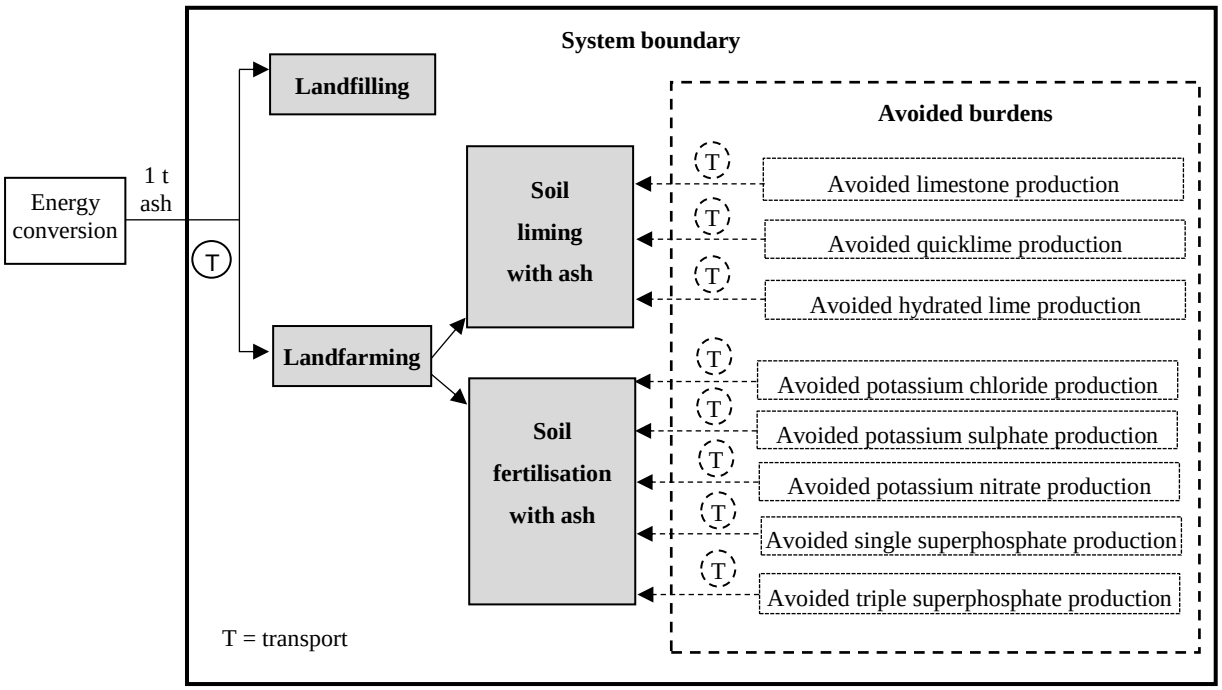

Figure 21 - System boundary for woody biomass ash end-of-life.

This study follows a comparative gate-to-grave LCA that starts with the transport of woody biomass ash from the power plant to landfilling or landfarming. The base scenario concerns the current most common practice of woody biomass ash management in Portugal: the disposal in landfill. The valorisation scenarios assess the ash landfarming for two purposes, namely: (1) liming and (2) soil fertilisation.

The environmental burdens avoided due to the valorisation of woody biomass ash were modelled through the system expansion by substitution (EC/JRC/IES, 2010b). When determining the environmental impact of soil liming/fertilising with woody biomass ash, it was given a "credit" based on the reduced requirement for traditional material production (liming and fertiliser products) and all related upstream activities, such as extraction, transport and energy required to manufacture these products.

In the case of liming, woody biomass ash was applied substituting three traditional liming products, namely, limestone, quicklime and hydrated lime, since limestone is the most used

liming material in the world (Goulding, 2016) and quicklime and hydrated lime are also generally used for soil application in Europe (EuLA, 2017).

In the case of soil fertilisation, woody biomass ash was applied substituting five traditional fertilisers (potassium chloride, potassium sulphate, potassium nitrate, single superphosphate and triple superphosphate), since they are the most applied, representing a consumption of approximately 200 thousand tonnes of fertilisers in Portugal in 2014 (FAO, 2017). The substitution by nitrogen fertilisers was not assessed, since the amount of nitrogen in woody biomass ash is insufficient to allow its use as a substitute for nitrogen fertilisers (TIFAC, 2001).

The scenarios assessed for woody biomass ash landfarming are:

- Scenario 1: Liming with FG substituting limestone.
- Scenario 2: Liming with FG substituting quicklime.
- Scenario 3: Liming with FG substituting hydrated lime.
- Scenario 4: Liming with FF substituting limestone.
- Scenario 5: Liming with FF substituting quicklime.
- Scenario 6: Liming with FF substituting hydrated lime.
- Scenario 7: Fertilisation with FG substituting potassium chloride.
- Scenario 8: Fertilisation with FG substituting potassium sulphate.
- Scenario 9: Fertilisation with FG substituting potassium nitrate.
- Scenario 10: Fertilisation with FG substituting single superphosphates.
- Scenario 11: Fertilisation with FG substituting triple superphosphates.
- Scenario 12: Fertilisation with FF substituting potassium chloride.
- Scenario 13: Fertilisation with FF substituting potassium sulphate.
- Scenario 14: Fertilisation with FF substituting potassium nitrate.
- Scenario 15: Fertilisation with FF substituting single superphosphates.
- Scenario 16: Fertilisation with FF substituting triple superphosphates.

5.2.2 Inventory data

5.2.2.1 Landfilling

The life cycle inventory data for woody biomass ash landfilling were built upon the Ecoinvent calculation tool for municipal solid waste landfill (Doka, 2003a; Ecoinvent, 2017),

taking into account the ash composition. The composition of woody biomass ash under study and the inventory data of ash landfilling can be found in da Costa et al. (2019).

The model considers that the ashes are spread and compacted by loaders in the sanitary landfill. Leachate is generated from rainwater infiltration and the corresponding emissions are calculated from the decomposition rate of waste (5 % during the first 100 years) and the concentration of each element in the waste fraction (Doka, 2003). The landfill leachate is assumed to be treated in a municipal wastewater treatment plant with mechanical, biological and chemical treatments. The sludge produced is further dewatered and incinerated in a grate furnace. After incineration, the bottom ashes are collected and landfilled in a slag compartment and the fly ashes are landfilled in a residual material landfill. No landfill gas is produced because the carbon contained in woody biomass ashes is mostly elemental or carbonate carbon (Bjurström et al., 2014; Straka et al., 2014) and remains permanently stored in the landfill.

5.2.2.2 Landfarming

Regarding the landfarming scenarios, woody biomass ash is transported from the power plant to the landfarming place and spread in the soil previously prepared by hand hoeing. The ashes are applied in loose and unstabilised form. The ash application is made by a broadcaster with a capacity of 500 L. The density of the fly ashes is based on measurements undertaken by Silva (2016), namely, 2.35 kg/L for FG and 2.48 kg/L for FF. The diesel consumed by the broadcaster during spreading and the related emissions to air were taken from Ecoinvent (Ecoinvent, 2017). The following activities were considered part of the work process: preliminary work at the farm, such as attaching the spreader to the tractor; driving to field (with an assumed distance of 1 km); field work (for a parcel of land of 0.1 ha surface); driving to farm and concluding work, like uncoupling the machine. The emissions to the soil resulting from the ash application were based on the ash composition and built upon existing model in Ecoinvent for waste landfarming (Doka, 2003a; Ecoinvent, 2017). The model assumes that the waste constituents are inventoried as direct emissions to soil. All chemical elements (Ca, P, S, etc.) are emitted as such and no conversion (e.g. phosphorus to phosphate) is performed. Besides, it ignores the uptake of macro and micro elements by the plants and their solubility. However, the uptake of P and K were taken into account according to the nutrients' bioavailability, as described in Section 5.2.2.3.2. The life cycle inventory data for the ash landfarming are shown in Table S4 of the Appendix B.

5.2.2.3 Avoided products

In the valorisation scenarios, avoiding the use of limestone, quicklime, hydrated lime, potassium chloride, potassium sulphate, potassium nitrate, single superphosphate and triple superphosphate was considered due to their substitution by woody biomass ash. The inventory data related to the manufacturing process of traditional liming and fertiliser products were taken from Ecoinvent (Ecoinvent, 2017).

Limestone consists mainly of calcium carbonate and the basic processes to produce limestone are: mining, crushing, washing, drying and sieving (Kellenberger et al., 2007). The limestone is mined mainly from quarries in opencast. The processing consists of breaking the limestone to a maximum of 200 mm and washing to remove the impurities (Kellenberger et al., 2007).

Quicklime is produced by the thermal decomposition of limestone. The quicklime production is similar to limestone production, with the addition of the calcining at a moderate temperature of about 1,200 °C (Oates, 1997).

Hydrated lime is produced by reacting quicklime with water. The initial processes in hydrated lime production are similar to those in quicklime processing, however, after the calcination there is the addition of hydrating and drying processes (Oates, 1997). The quicklime is mixed with twice the amount of water at the hydration plant and dried in a separator circuit in which an air stream removes the finest particles (Kellenberger et al., 2007).

Potassium chloride is produced by mining and beneficiation of potash salt. The production process starts with the salt being mined from underground mines, then crushed and milled (Kongshaugh, 1998). Three different processes are applied to concentrated potassium chloride to produce a usable fertiliser: solution in hot water, flotation and electrostatic separation. The data refers to a mixture of these three methods. The final product is dried and has a K_2O content of 60 %.

Potassium sulphate is produced by reacting potassium chloride with sulphuric acid. The potassium chloride reacts during slow mixing in the furnace with sulphuric acid, producing gaseous hydrogen chloride and potassium sulphate (Davis and Haglund, 1999). The final product is cooled and crushed and has a K_2O content of 50 %.

Potassium nitrate is mainly produced by the reaction of potassium chloride with nitric acid. The raw materials are fed into a reactor at 50-100 ºC and when the reaction is completed, crystallisation takes place by cooling at 30 °C (Kongshaugh, 1998). The crystals are separated in a decanter and dried and have a K_2O content of 46 %.

Single superphosphate is produced by reacting ground phosphate rock with sulphuric acid. After the reaction, superphosphate is sent to a granulator where steam, water and acid are added to aid in granulation (Davis and Haglund, 1999). The final product is cooled and dried and has a P_2O_5 content of 21 %.

Triple superphosphate is produced by reacting 70 % of sulphuric acid with 30 % of ground phosphate rock (Davis and Haglund, 1999). The processes in triple superphosphate production are similar to single superphosphate, but the final product has a P_2O_5 content of 40 %.

The substitution rate varies for different materials and can achieve up to 1:1 if the ash and the traditional material are functionally equivalent, which is not the case of the ash assessed in this study. The equivalence is related to the amount of woody biomass ash needed to perform the same function (liming/fertilisation) as traditional amendments products. Woody biomass ash equivalence is reported in Sections 5.2.2.3.1 and 5.2.2.3.2.

In addition to the desired nutrients, traditional ameliorants may contain trace element contaminants, which are shown in Table S5 of the Appendix B. For the calculation of impacts, it was considered that these elements were emitted to the soil as such and no conversion is performed.

5.2.2.3.1 Liming equivalence

The liming equivalence of woody biomass ash compared to the traditional liming products was obtained through the calcium carbonate equivalent (CCE). The CCE is the percentage effectiveness of a particular liming product relative to a pure limestone with a CCE of 100 % (Heckman, 1998). For quicklime, the value is 179 % and for lime hydrated, the value is 135 % (Alcarde, 2005). The CCE values for the ashes under study were calculated based on the optimum dose in limestone equivalents and the optimum dose to be applied in the soil taken from Silva (2016), namely, 21.94 % for FG and 19.53 % for FF. Equation 1 was used to determine the amount of liming product avoided by the application of 1 t of woody biomass ash.

$$m_{liming\ product} = m_{woody\ biomass\ ash} * \frac{CCE_{woody\ biomass\ ash}}{CCE_{liming\ product}} \quad (1)$$

where $m_{liming\ product}$ is the amount of liming product (kg), $m_{woody\ biomass\ ash}$ is the amount of woody biomass ash (equal to 1000 kg), $CCE_{woody\ biomass\ ash}$ is the CCE of woody biomass ash (%) and $CCE_{liming\ product}$ is the CCE of the liming product (%).

The amount of liming product equivalent to 1 t of woody biomass ash calculated with Equation 1 is shown in Table S6 of the Appendix B. For example, the application of 1000 kg of FG is equivalent to using 219.4 kg of pure limestone.

5.2.2.3.2 Fertilisation equivalence

The fertilisation equivalence of woody biomass ash compared to the traditional fertilisers was obtained through the bioavailability of nutrients in the ashes and in the fertilisers. Regarding K fertilisers, the absolute amount of K in the potassium chloride is 498 g/kg, in potassium sulphate, 415 g/kg, and in potassium nitrate, 365 g/kg, while in the ash the amount of K is 47.0 g/kg in FG and 41.4 g/kg in FF. However, the K bioavailable for use by the plants is not directly correlated to the absolute amount of K present in the fertiliser and the ash, as only a part of K is extractable and bioavailable for plants (Kaminski et al., 2010). The K bioavailable for plants can range from 18 % to 51 % of the total K in woody biomass ash due to the formation of insoluble fused K compounds with insoluble elements such as silicon at high temperatures (Naylor and Schmidt, 1986). In comparison, the K bioavailable for plants in traditional fertilisers can range from 65 % to 70 % of the total K (Naylor and Schmidt, 1986). The mean values of the K bioavailability ranges were considered in this study to calculate the fertilisation equivalence, namely, 34.5 % for woody biomass ash and 67.5 % for K fertilisers.

Regarding P fertilisers, the absolute amount of P in the single superphosphate is 92 g/kg and in triple superphosphate is 210 g/kg, while in the ash the amount of P is 6.47 g/kg in FG and 9.02 g/kg in FF. However, the P bioavailable to plants can range from less than 1 % to 20 % of the total P in woody biomass ash (Clarholm, 1994; Erich, 1991; Ohno and Erich, 1990; Quirantes et al., 2016; Santalla et al., 2011; Schiemenz and Eichler-lo, 2010), from 16 % to 46 % of the total P in the single superphosphate (Plotegher and Ribeiro, 2016) and from 80 % to 93 % of the total P in the triple superphosphate (Mullins and Sikora, 1994). The difference between total P content and P bioavailable for plants is related to physical and chemical processes in the soil and also result from the increased phosphatase activity in the soil (Augusto et al., 2008; Mandre, 2006). The mean values of the ranges of P bioavailability were considered for calculation purposes, namely, 10.5 % for woody biomass ash, 31 % for single superphosphate and 86.5 % for triple superphosphate. The fertilisation equivalence of 1 t of woody biomass ash is shown in Table S7 of the Appendix B. For example, the application of 1000 kg of FG is equivalent to using 22.74 kg of single superphosphate.

5.2.2.4 Transportation

The transportation profiles considered for woody biomass ash management, as well as for the corresponding traditional amendments are shown in Table S8 of the Appendix B. It was considered that no losses of ash occur during its transportation to landfill and to landfarm. Inventory data for the transport processes were taken from the Ecoinvent database (Ecoinvent, 2017).

5.2.3 Impact assessment

The characterisation factors and the impact categories used in this study were those from ILCD (EC/JRC/IES, 2012). The following impact categories were considered: CC, PM, POF, MFD, FEu, AC, HTc, HTnc and FEc.

However, the impact results of human toxicity and freshwater ecotoxicity should be analysed with some caution due to the high uncertainties in the interim characterisation factors associated with metals, dissociating chemicals and amphiphiles in the USEtox method adopted by the ILCD (Rosenbaum et al., 2008).

5.2.4 Sensitivity analyses

Sensitivity analyses were performed to understand the influence of some parameters in the environmental impact assessment results. A sensitivity analysis was made to assess the influence of the transport distances in the results obtained for the base and valorisation scenarios. Since the assumed baseline distances presented in Table S8 of the Appendix B are related to the typical distances of Portugal, the distances were changed to make the assessment more representative of other regions too. Therefore, the transport distances of ash from the power plant to landfill and to landfarm as well as the transport distances of traditional amendments from the manufacturing unit to landfarm, were changed to half and double of the baseline distances.

Another sensitivity analysis assesses the influence of the nutrient bioavailability in the results. Selecting this analysis was based on the concern around the efficient use of woody biomass ash as amendment. In the future, the successful valorisation of woody biomass ash can reduce the costs of ash disposal and soil amendment. The main factor affecting woody biomass ash application as amendment is nutrient (P and K) bioavailability, as ashes with a high nutrient content can yield higher efficiencies. However, nutrient bioavailability can vary widely as mentioned in Section 5.2.2.3.2 adding uncertainty to the results. Therefore, in this sensitivity

analysis, the bioavailability of K and P in each fertiliser and in woody biomass ash was varied according to the ranges presented in Table S9 of the Appendix B.

The third sensitivity analysis assesses the addition of a nitrogen-based fertiliser to woody biomass ash fertilisation on soil in order to compensate for the depletion of nitrogen due to the potassium nitrate substitution in scenarios 9 and 14. In the baseline scenarios, the woody biomass ash is assumed to substitute the K bioavailable for plants contained in the potassium nitrate fertiliser, whereas the nitrogen contained in this fertiliser will not be uptaken by the plants, assuming that the soil is not deficient of this nutrient. Thus, in this sensitivity analysis, it is considered that ashes are applied on soils poor in nitrogen and, therefore, it is necessary to consider an equivalent amount of a nitrogen-based fertiliser to ensure that the functions delivered by the process of fertilisation with ash and the avoided fertilisation process are the same (i.e. providing the same amount of K and nitrogen). The nitrogen-based fertiliser added together with ash was the calcium nitrate, since the nitrogen present in this fertiliser is also in the form of nitrate, maintaining the same bioavailability of the nitrogen for the plants. There 52.8 kg of calcium nitrate added with FG and 46.5 kg with FF, considering a nitrogen content of 15.5 % in the calcium nitrate.

5.3 Results and discussion

5.3.1 Hotspots analyses

Figure 22 presents the environmental impact and the relative contribution of each source of impact, obtained for the base scenario [landfilling (LF)] and for the valorisation scenario [(landfarming (LM)] without avoided burdens. It should be noted that Figure 22 cannot be used to compare the end-of-life management alternatives, since the credits have not yet been considered.

Regarding landfilling scenarios, the emissions from diesel combustion during ash spreading at the sanitary landfill represents the largest impacts for CC (54-55 % of the total impact) mainly due to the CO_2. Diesel combustion at the sanitary landfill is also the main hotspot for PM, POF and AC impact categories. In the PM impact category, it represents 76-77 % of the total impact, mainly related to the emissions of $PM_{2.5}$. For POF and AC, it represents 71-72 % and 66-67 % of the total impact, respectively, mainly due to the emissions of NOx in both impact categories. In the MFD impact category the leachate treatment is the main hotspot (58-64 % of the total impact), as a result of the depletion of In and Cd mainly due to the consumption of aluminium sulphate as coagulating agent.

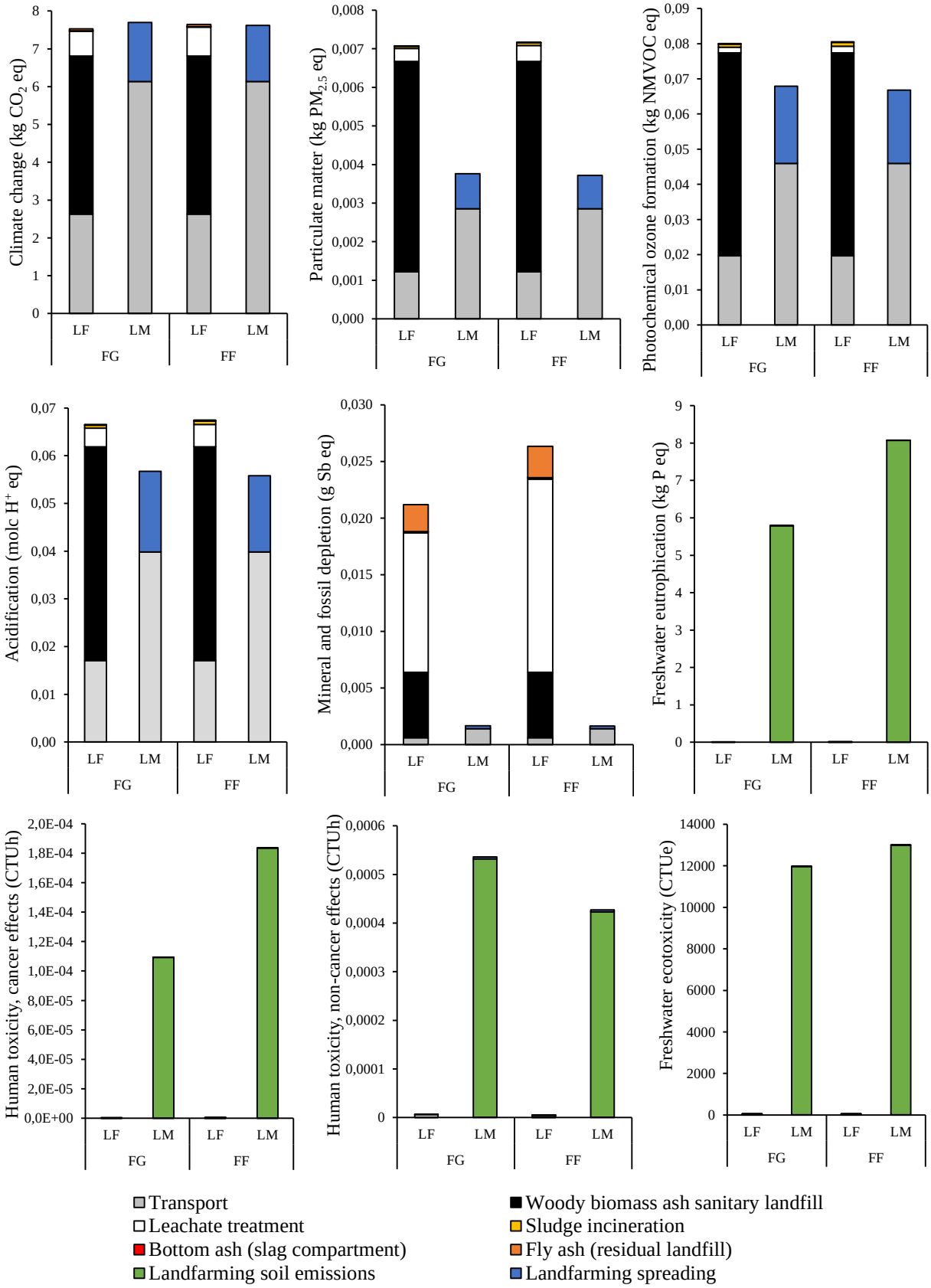

Figure 22 - Impact assessment results for the base scenario (ash landfilling) and for landfarming without avoided burdens per FU. The results are shown for the fly ash produced in a vibrating grate and in a fluidised bed.

Although it is not possible to observe in Figure 22, the leachate treatment is also the main hotspot for FEu, HTc, HTnc and FEc impact categories (with contributions ranging from 58 to 98 % of the total impact).

Regarding landfarming scenarios, woody biomass ash transportation is the main responsible for CC impact category (79-81 % of the total impact) mainly due to CO_2 emissions from diesel combustion. The transportation is also the main hotspot for PM, POF, AC and MFD impact categories. In the case of PM, transportation represents 76-77 % of the total impact and is mainly related to the emissions of $PM_{2.5}$ due to diesel combustion. In the case of POF, the contribution of transportation represents 68-69 % of the total impact and in the case of AC it represents 70-71 % of the total impact. The emission of NOx due to diesel combustion is the main contribution to the total impact for POF and AC impact categories. In the case of MFD, the transportation represents 84-85 % of the total impact and is mainly related to the depletion of oil and In.

The direct emissions to soil are the main hotspot for FEu, HTc, HTnc and FEc impact categories (with contributions ranging from 98 to 99 % of the total impact). In the case of FEu, the impacts are related to the emission of P to soil. Since only part of the P contained in woody biomass ash is extractable and used by the plants (about 10.5 %), the remaining P added to the soil can potentially contribute to freshwater eutrophication due to leaching. The magnitude of this environmental impact depends on the amount of P added to soil. The P content in FF is equal to 9.02 g/kg and in FG is 6.46 g/kg. Excessive P in freshwater ecosystems causes algae proliferation, toxin production, fish kill, altered plant species diversity and problems in the food chain (Elser et al., 2007; Justic et al., 2009). Regarding HTc, the resulting impacts are due to the emission of Cr to the soil, which is present in FF (0.067 g/kg) and in FG (0.038 g/kg). Chromium is a naturally occurring element found in the soil predominantly in its trivalent oxide form (despite other forms can be present in the woody ash), which is the least soluble among Cr compounds, making Cr very immobile and unavailable for leaching and uptake (Banks et al., 2006). Regarding HTnc, the impacts are mainly associated with the emission to the soil of As contained in the ashes (0.024 g/kg in FG and 0.017 g/kg in FF). The As accumulated in the soil can be leached from the soil, which may be detrimental mainly to human health (Chang et al., 2004). Exposure to inorganic arsenic leads to serious effects, such as circulatory disorders, neurological and respiratory complications and hepatic and renal dysfunction (Chen et al., 2009).

With regard to FEc, the impacts essentially come from the emissions of Zn to the soil, which is present in FG (0.207 g/kg) and in FF (0.193 g/kg). Zinc is an essential micronutrient

needed for normal growth in plants and animals. However, at both high and low concentrations Zn can be detrimental to organisms, which has caused great environmental concern (Cheng and Allen, 2006).

It should be noted that leaching of trace elements to the soil depends on many variables, such as rainfall and temperature regimes, and the form of the ash at application, as well as soil and surface litter factors (Pitman, 2006) and these aspects were not considered by the model used in the current study to predict emissions of trace elements to soil in landfarming.

One alternative to preventing contaminant elements from leaching out of the woody biomass ashes is the stabilisation of ash by ageing. In fact, some studies claim that ashes should be stabilised prior to their application in the soil, mainly to reduce their pH and to decrease the bioavailability of potentially toxic elements (Nilsson et al., 2016; von Wilpert et al., 2016). Another alternative to reduce the impacts of the leaching process is the agglomeration of the ashes into bigger particles, since it leads to a slow dissolution of the elements contained in the ashes (Callesen et al., 2007). However, woody biomass ash pre-treatment was not considered, since these processes (ageing and agglomeration) are still under development and are generally not common practices, at least currently in Portugal.

Deviatkin et al. (2017) studied the environmental impacts of forest fertilisation and liming with a mixture of woody biomass ashes and other types of ashes, in the granulated form, generated by thermal conversion and compared them with landfilling. The results indicated that the application of these ashes on the forest soil had larger environmental impacts than landfilling for FEc, HTc and HTnc (10 to 24 %), which was strongly affected by the leaching of heavy metals, such as V, Ba, Sb, Co, Cr and Zn. However, the study concluded that the high uncertainty related to the leaching data has contributed for these results.

5.3.2 Liming net impacts

Figure 23 presents the net impact of the liming scenarios and the impact associated with the base scenarios (landfilling). The net impact is the difference between the environmental impact and the avoided burdens. If the net impact is positive, the environmental impact of ash landfarming is higher than the avoided burdens associated with the substitution of traditional liming products; if the net impact is negative, the environmental impact is lower than the avoided burdens.

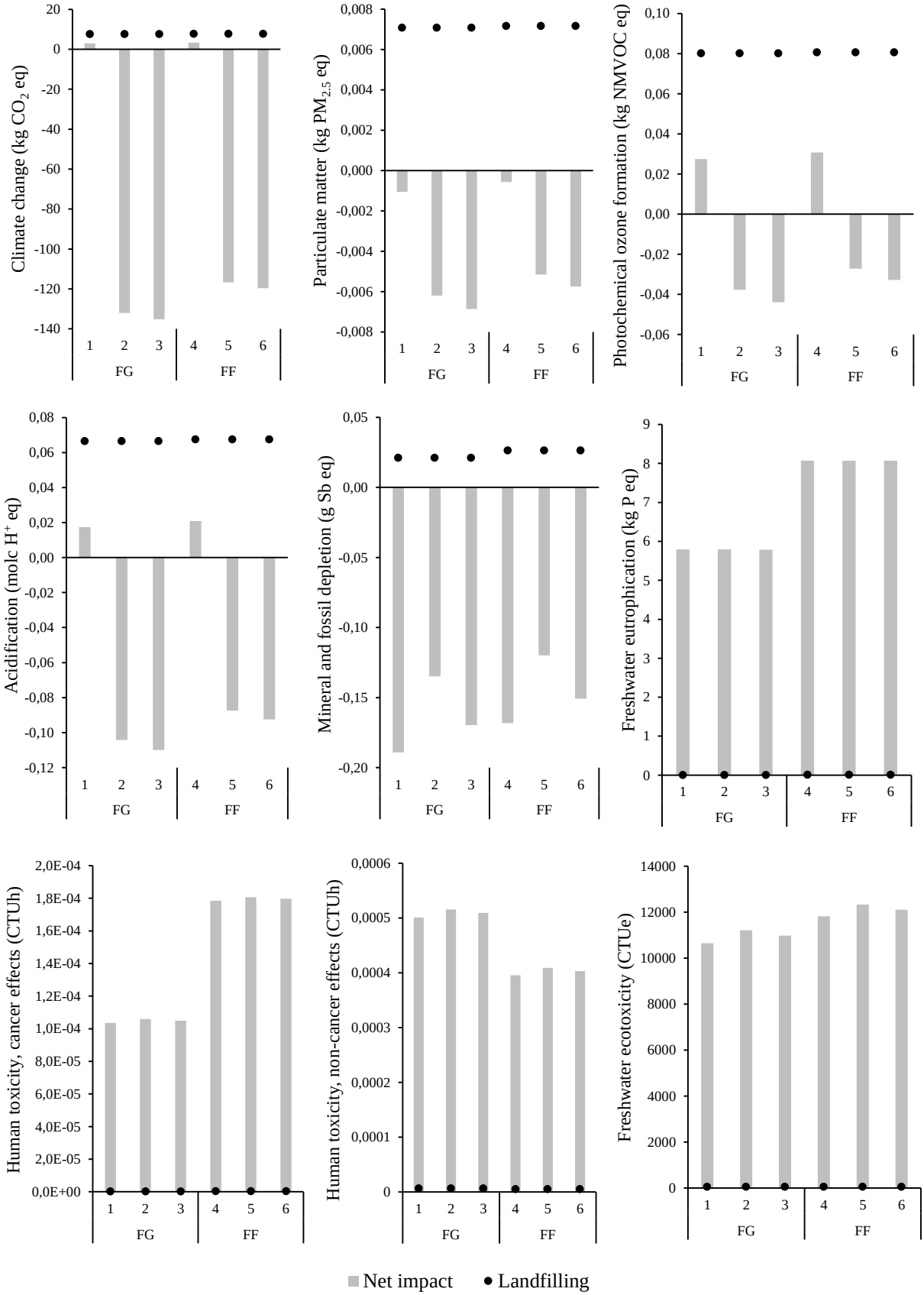

Figure 23 - Net impact results in each scenario of ash landfarming for liming (1 to 6) per FU. The dots marked in each scenario depict the impacts from the ash disposal in landfills (base scenario).

The substitution of hydrated lime (scenarios 3 and 6) presented the highest environmental saving followed, in this order, by quicklime (scenarios 2 and 5) and limestone (scenarios 1 and 4). The substitution of hydrated lime by ash avoids considerable consumption of energy and materials, since its production requires many operation processes (e.g. mixing, hydrating and drying) (Oates, 1997). The only exception is in the MFD impact category, where the highest environmental saving corresponds to the limestone substitution scenarios (1 and 4).

For the remaining impact categories (FEu, HTc, HTnc and FEc), the net impact was positive for all scenarios, which indicates that the use of woody biomass ash on land for liming has a greater environmental impact than the traditional liming products. The impacts in these categories are mainly due to P and heavy metals (Cr, As and Zn) added to the soil, which occurs to a lesser extent in liming products because the presence of these trace elements is lower than in the ashes. Besides that, the environmental impacts of ash landfarming exceed the impacts of ash landfilling for all scenarios in these categories.

Both types of ashes presented CCE value at less than half the value of traditional liming products under study, being a limiting aspect in the ash landfarming for liming. According to Silva (2016), the low CCE value is related to the high amount of silicate minerals and other crystalline and amorphous forms that retain the basic cations of the ash, substantially reducing their solubility. An alternative to improve the CCE value in woody biomass ash and, consequently, in the environmental performance for all impact categories, consists in reducing the amounts of bed sand and inerts in the biomass during the combustion process (soil, quartz, micas and other silicates), as these components contribute to the reduction of the alkaline properties of the ash (Silva, 2016).

5.3.3 Fertilisation net impacts

Figure 24 presents the net impact for soil fertilisation scenarios and the impact associated to the base scenarios (landfilling). The results show that the substitution of K fertilisers by woody biomass ash (scenarios 7, 8, 9, 12, 13 and 14) presented environmental savings for most scenarios of CC, PM, POF, MFD and AC impact categories. However, scenarios 7 and 12 presented positive environmental impacts for PM, POF and AC, since the impacts related to the production of potassium chloride are lower than those from the production of potassium sulphate and potassium nitrate and do not exceed the impacts related to the application of woody biomass ash. Despite this, the environmental impacts of ash landfarming were lower than the impacts of ash landfilling for all scenarios in these categories.

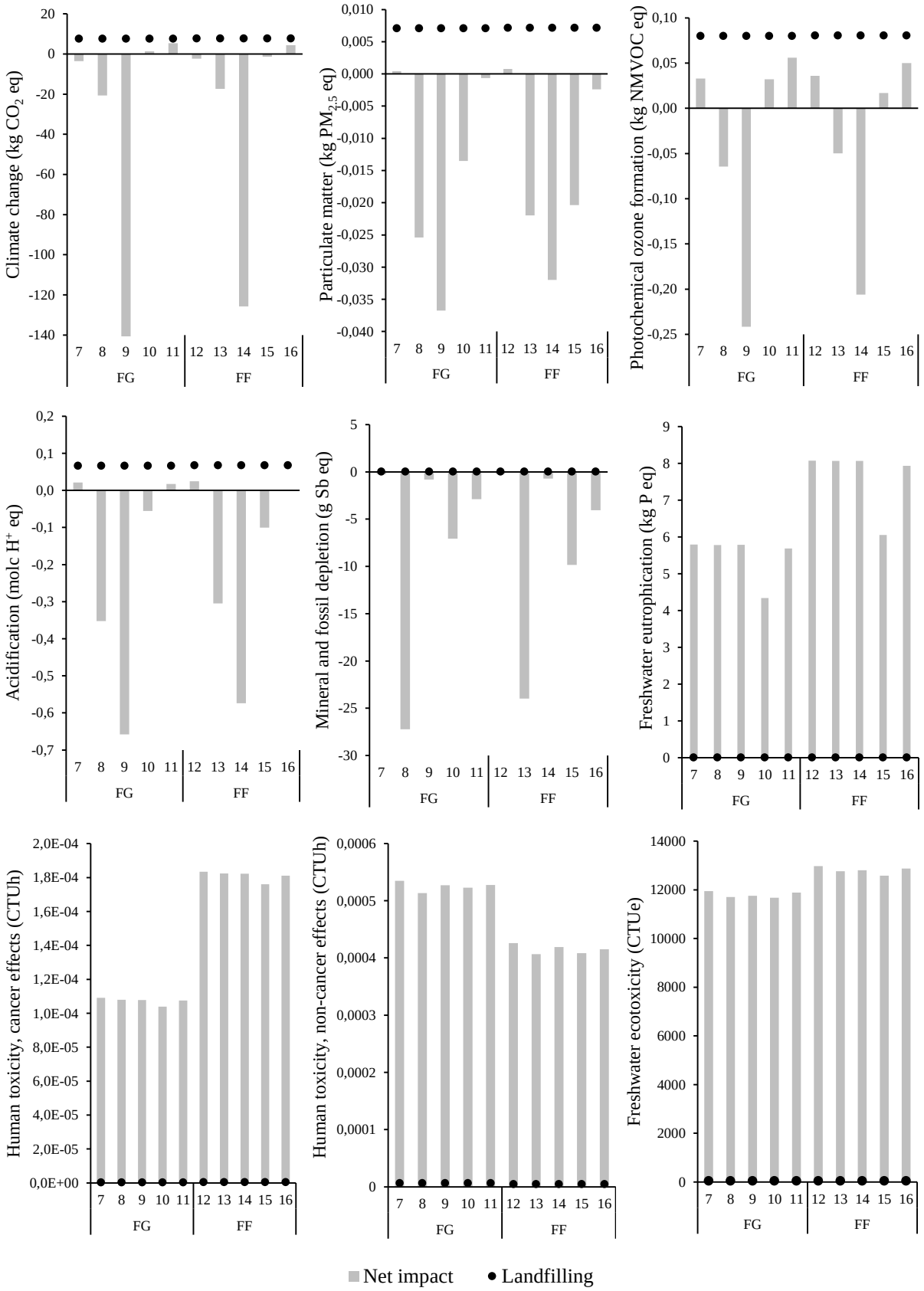

Figure 24 - Net impact results for ash landfarming scenarios for soil fertilisation (7 to 16) per FU. The dots marked in each scenario depict the impacts from the ash disposal in landfills (base scenario).

However, for the remaining impact categories (FEu, HTc, HTnc and FEc), the net impact was positive and exceeded the impacts of ash landfilling for all scenarios studied, which indicates that the use of woody biomass ash on land for fertilisation has a greater environmental impact than the disposal in landfill. The impacts in these categories are mainly due to P and heavy metals (Cr, As and Zn) from the ash added to the soil, which occurs with less intensity with the application of traditional fertilisers due to the low substitution equivalence.

The substitution of P fertilisers by woody biomass ash presented environmental savings for all scenarios of PM and MFD impact categories. The substitution of single superphosphate (scenarios 10 and 15) presented the better environmental performance for most of the impact categories followed by triple superphosphate (scenarios 11 and 16). The single superphosphate production consumes many resources to be produced (e.g. phosphate rock, sulphuric acid and energy) and the substitution by woody biomass ashes avoids the consumption of these resources.

The ash landfarming presents a better environmental performance than ash landfilling for all scenarios of CC, PM, POF, MFD and AC impact categories. However, for FEu, HTc, HTnc and FEc impact categories, the impact related to landfarming exceeded the impacts of ash disposal in landfill mainly due to the emissions of P, Cr, As and Zn to the soil.

Despite the worst environmental performance for some impact categories of ash in comparison with traditional fertilisers, Gonçalves and Moro (1995) observed positive aspects in tree's growth due to the application of woody biomass ash in the soil. These authors assessed the mineral nutrition of Eucalyptus grandis in red latosol with woody biomass ash comparatively to the application of a traditional fertiliser (NPK 10.20.10). After 72 months from ash application in the soil, it was observed that the content of macro nutrients placed in the tree's components (wood, bark, leaves and branches) increased significantly by the application of woody biomass ash, notoriously for Ca and K, compared to the traditional fertiliser. According to these authors, most of the nutrients in the traditional fertiliser are in the soluble form and, therefore, more subject to losses by leaching. In the case of the nutrients present in woody biomass ash, part is in the soluble form and another part is released with time by gradual solubilisation of the chemical compounds, which makes the nutrients less subject to leaching, promoting the best use of them by the trees.

5.3.4 Cumulative net impacts

Cumulative effects can be observed when woody biomass ash is applied on land, i.e., when woody biomass ash is used as liming material to improve soil pH, the bioavailability of K and P in the soil increases. Therefore, woody biomass ash used as an alternative liming material will also provide important quantities of nutrients as a secondary benefit.

Figure 25 presents the best and worst scenarios of the cumulative effects of ash application as a soil amendment in the CC impact category as an example, since the trend was similar for all categories. The best and worst scenarios were obtained through the sum of the impacts avoided individually. The best environmental performance was achieved by the substitution of lime hydrated, potassium nitrate and single superphosphate and the worst environmental performance was achieved by the substitution of limestone, potassium chloride and triple superphosphate. It was observed that the net impact due to the substitution of the materials indicated in the best scenario is -293 kg CO_2-eq/t for FG and -261 kg CO_2-eq/t for FF, while in the worst scenario, the net impact is only -10.7 kg CO_2-eq/t for FG and -9.8 kg CO_2-eq/t for FF.

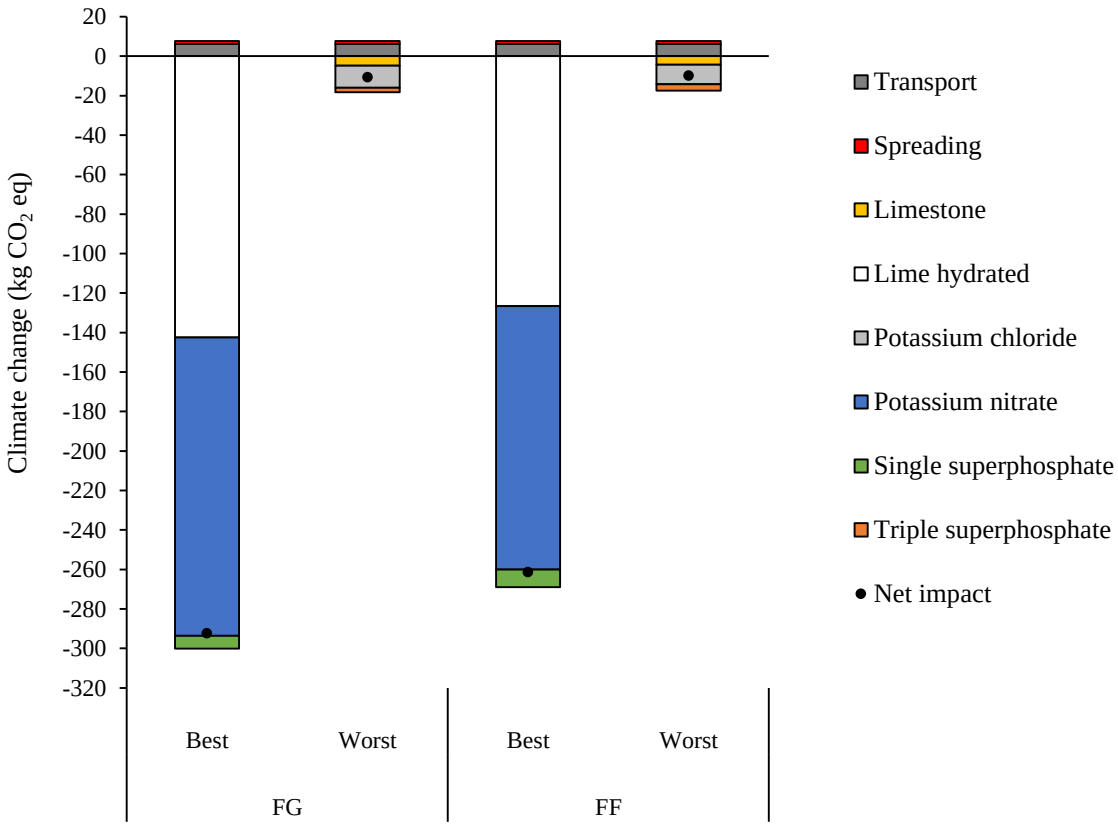

Figure 25 - Cumulative avoided burdens of the amendment materials substitution by woody biomass ash for the impact category climate change.

Despite the lack of information regarding the amount of ash produced in Portugal, it is estimated that 54 thousand tonnes of woody biomass fly ashes are produced annually by current power plants in operation, with 12 % being from a grate furnace and 88 % from a fluidised bed (Coelho, 2010; Silva, 2016). Therefore, if the total amount of FG is applied in the soil, it would be possible to supply up to the equivalent of 5 % of the potassium nitrate market and 6 % of the single superphosphate market, considering the consumption of fertilisers in Portugal in 2014 (FAO, 2017). On the other hand, the use of FF could supply up to the equivalent of 32 % of the potassium nitrate market and 64 % of the single superphosphate market in Portugal (FAO, 2017).

5.3.5 Sensitivity analyses

Regarding sensitivity analyses, Figure 26 shows the influence of transport distance in the environmental impacts of ash landfarming and landfilling.

The results obtained for the impact categories MFD, FEu, HTc, HTnc and FEc are the same as those for the base scenario and are therefore not shown in Figure 26. The transport of woody biomass ash and traditional amendments over larger distances results in a higher consumption of diesel, increasing the emissions of CO_2, CO, SO_2, NOx, $PM_{2.5}$, NMVOCs and others. These emissions are generated due to the combustion process of the diesel which affects mainly the impact categories of CC, PM, POF and AC.

When the transport distance of woody biomass ash and traditional amendments to the landfarm is changed (black line), the net impact can vary by up to 230 % in the POF impact category. When the transport distance of woody biomass ash to the landfill is changed (red line), the net impact can vary by up to 60 % in the POF impact category. The black line intercepts the red line in some scenarios of the impact categories POF and AC and can change the best end-of-life alternative for the ashes. In these scenarios, such as scenario 16 for the POF impact category, the best end-of-life alternative would no longer be landfarming. The best option would be the landfilling whether the distance from the ashes to the landfarm was increased and the distance of the triple superphosphate was decreased.

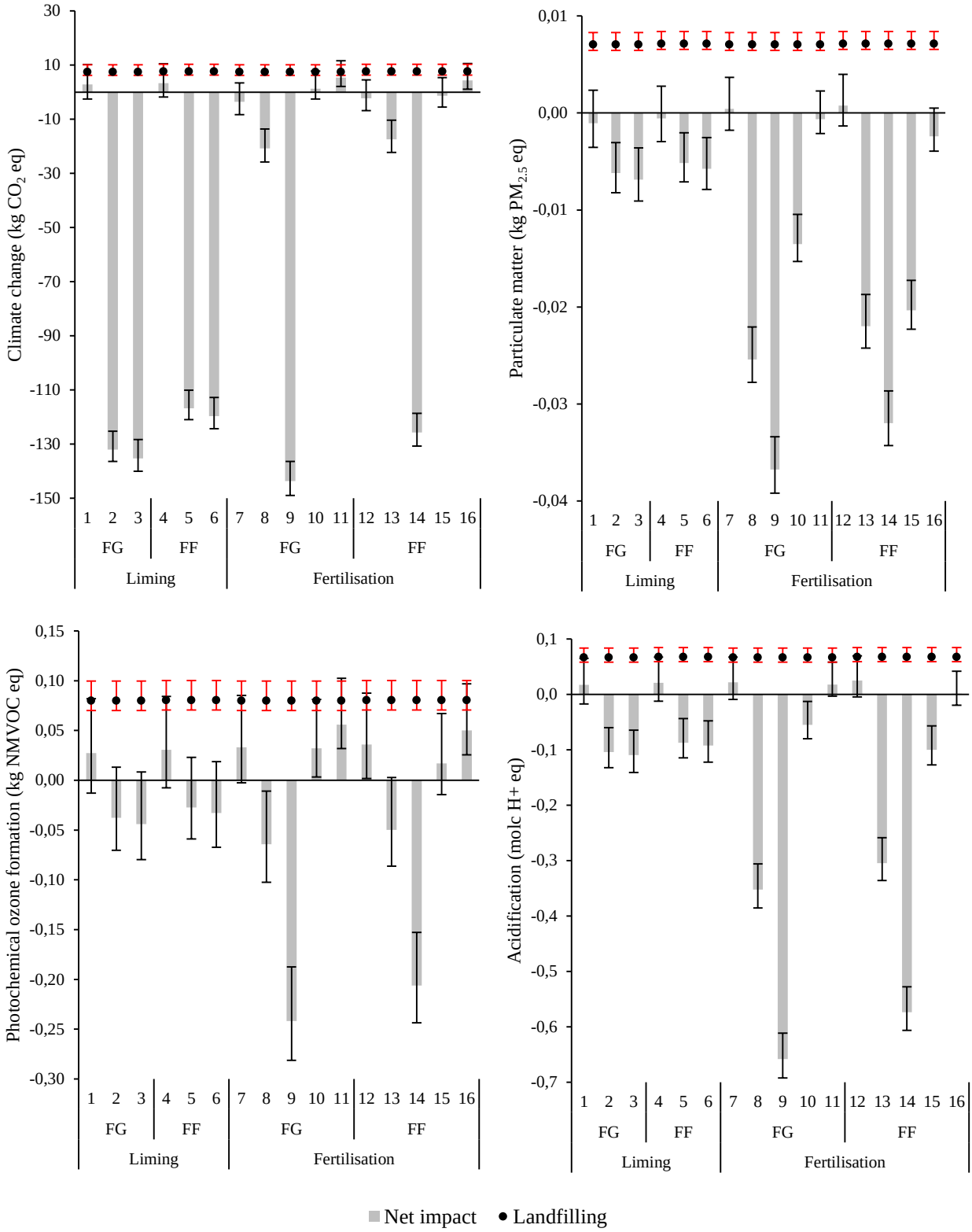

Figure 26 - Sensitivity analysis of the valorisation scenarios (1 to 16) per FU, regarding the influence of transportation distance in the environmental impact. The error bars with red line represents the variation in the impacts when the transport distance from the power plant to the landfill changes and the error bars with black line represents the variation in the impacts when the transport distance from the power plant to the landfarm changes.

When the distances of the traditional products are set at base distance (Table S8 of the Appendix B), the maximum distance in which the ash can be transported without exceeding the impacts of the landfill in any scenario of the POF impact category is 25 km for FG and 28 km for FF. Therefore, the results show that the best end-of-life scenario for woody biomass ash is highly dependent on the transport distance and the landfarms in which woody biomass ashes are spread should be close to the power plants. In addition, the influence of the fertiliser equivalence in the impact results was also assessed, as shown in Figure 27.

Effects in the environmental impacts were observed due to changes in the bioavailability of nutrients in woody biomass ash and fertilisers. The impact categories mainly affected by the fertilisation equivalence were CC, PM, POF, MFD, AC and some scenarios of FEu. The net impact can vary by up to 615 %, as in scenario 10 in the AC impact category.

When woody ash fertilisation equivalence is changed, the best end-of-life alternative for the ashes can be modified. In some scenarios, such as scenarios 10 and 15 for the FEu impact category, the increased bioavailability of nutrients in the ashes can change the best end-of-life from landfilling to landfarming. However, for HTc, HTnc and FEc, as well as even increasing the nutrients bioavailability, the best end-of-life remains the landfill. Despite that, the results show that the best end-of-life scenario for woody biomass ash depends on the bioavailability of nutrients and trace elements in the ashes.

Figure 28 shows the environmental impacts of ash landfarming with calcium nitrate to compensate for the depletion of nitrogen due to potassium nitrate substitution in scenarios 9 and 14. The impact categories mainly affected by the addition of calcium nitrate were CC, PM, POF, AC and MFD, which show higher impacts that in the baseline and, thus, the substitution of potassium nitrate no longer presents a good performance. In the case of CC, PM and AC, the use of calcium nitrate changes the best end-of-life from landfarming to landfilling and in the case of POF, the environmental impacts are similar. For FEu, HTc, HTnc and FEc the environmental performance has worsened slightly, but the best end-of-life remains the landfill.

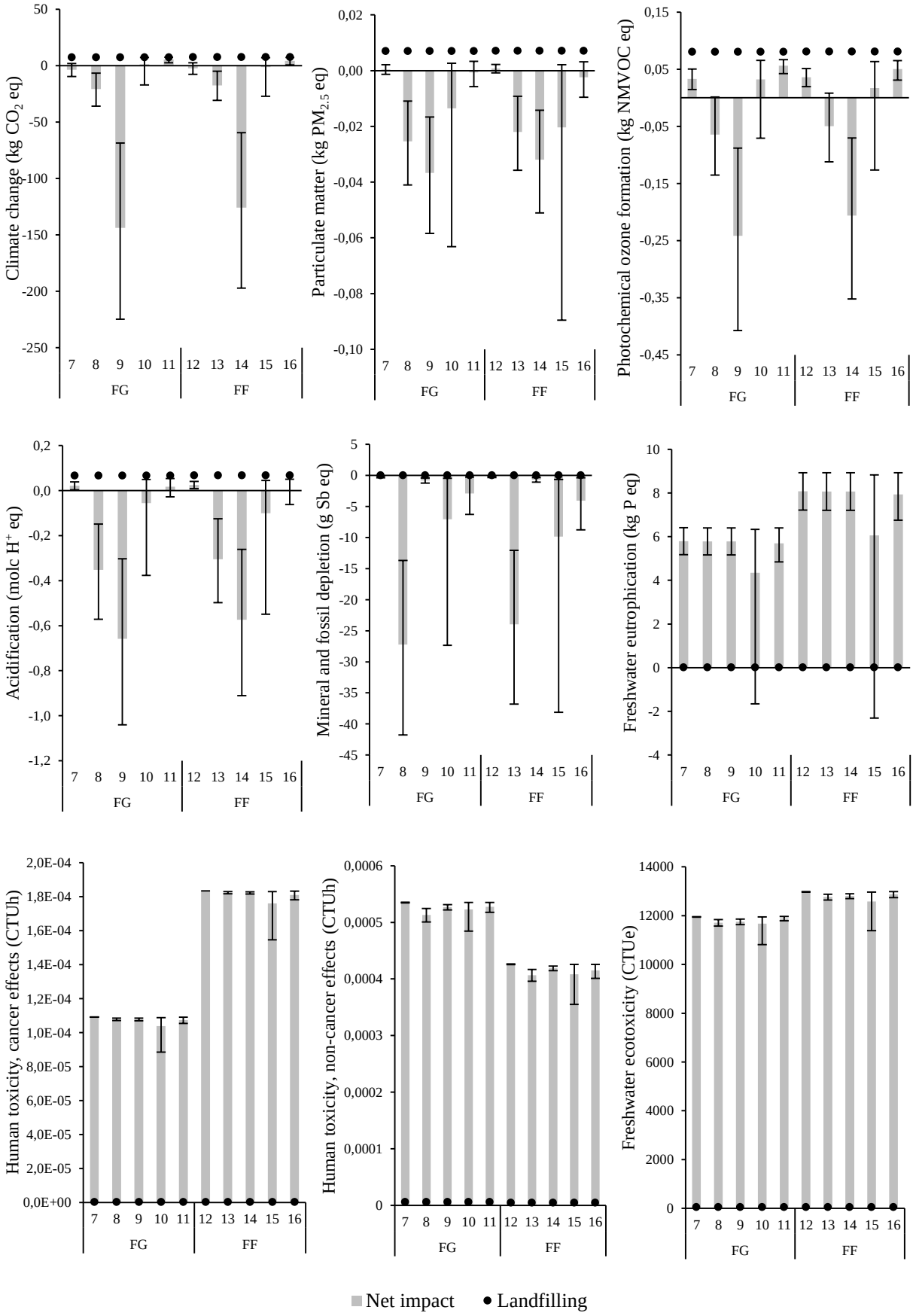

Figure 27 - Sensitivity analysis of the fertilisation scenarios (7 to 16) per FU, regarding the influence of the bioavailability of nutrients in the environmental impact.

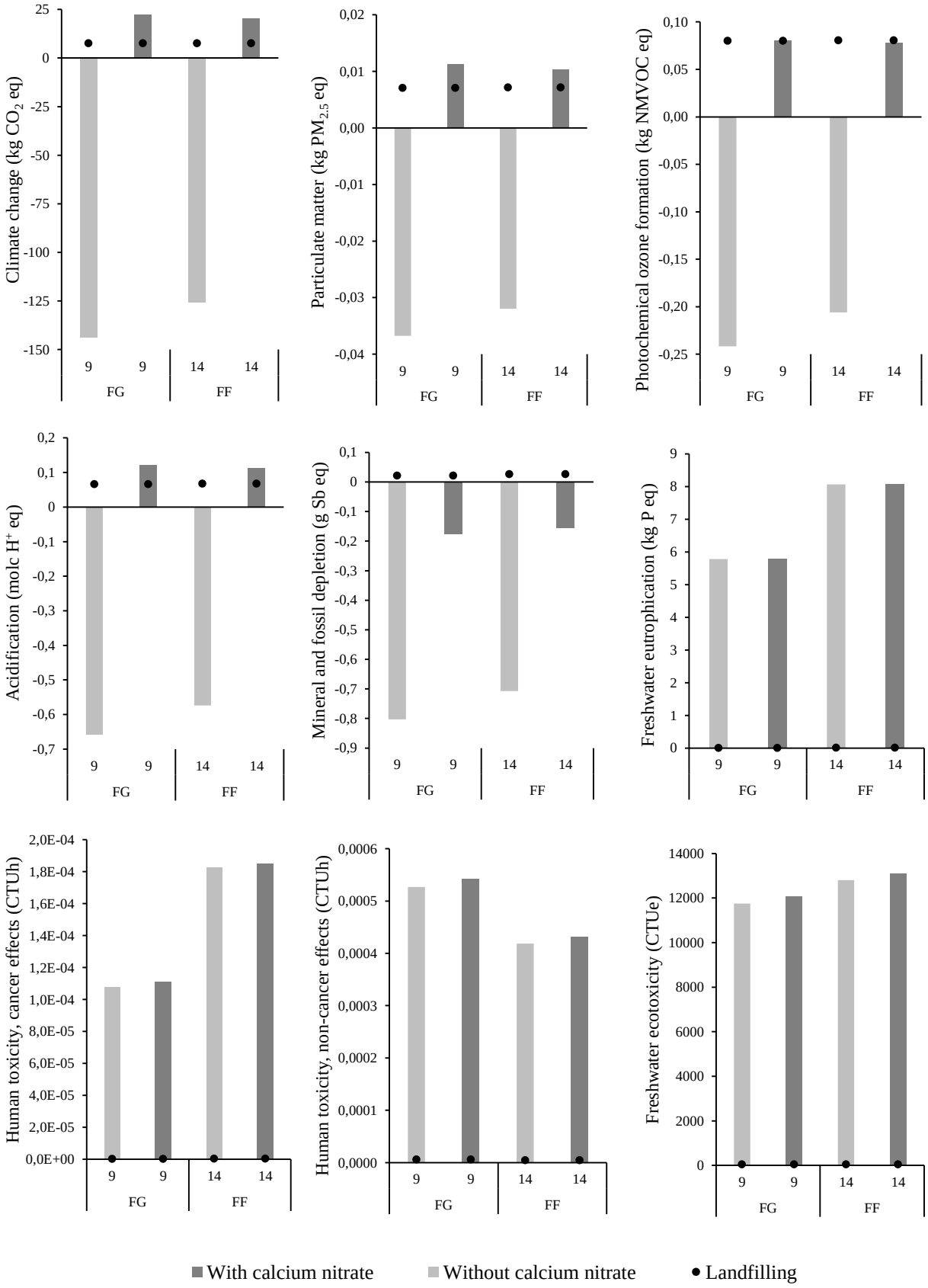

Figure 28 - Sensitivity analysis of the fertilisation scenarios with potassium nitrate (9 and 14) per FU, with and without nitrogen compensation through the application of calcium nitrate together with ash.

5.4 Conclusions

Woody biomass ash landfarming presented satisfactory performance for the impact categories CC, PM, POF, AC and MFD in comparison to landfilling. For the liming scenarios, the improvement in the environmental performance is higher when the ashes substitute hydrated lime. Regarding the application of ashes for K fertilisation, the substitution of potassium nitrate is the best alternative, while the substitution of single superphosphate is the best scenario for P fertilisation.

However, the environmental impacts of ash landfarming exceed the impacts of ash landfilling in FEu, HTc, HTnc and FEc, indicating that ash valorisation for soil amelioration depends on its composition, since ashes can contain trace element contaminants, such as Cr, As and Zn, which can be inadvertently added to the soil. However, it is noteworthy that the presence of trace elements in the ash does not directly imply increased risks for the environment, as the potential pollutants leaching depends on their bioavailability in the soil. Future research should focus on the effect of trace element solubility on the leaching process that occurs naturally in the forest soil, in order to obtain more accurate results for the toxicity-related categories.

The sensitivity analysis regarding the influence of transport distance showed a significant effect for the impact categories PM, POF and AC. Therefore, ash landfarming should be made as close as possible to the power plant to ensure the reduction of diesel consumption and the inherent environment impacts associated. The results of the sensitivity analysis also showed for ash valorisation is highly dependent on the bioavailability of nutrients.

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Chapter 6

Consequential LCA of energy production from forest residues

Environmental comparison of forest biomass residues application: electricity, heat and biofuel

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Renewable and Sustainable Energy Reviews.

Abstract

A new strategy of the Portuguese government foresees an increase of the installed electricity capacity in Portugal based on the use of biomass residues, including forest biomass residues. However, this implies that the use of forest biomass residues for electricity production will most likely happen at the expense of alternative uses, such as heat and biofuel production. Therefore, what would be the best use from an environmental perspective for the forest biomass residues available in the country? To answer this question, a consequential life cycle assessment was applied to assess three different scenarios: (1) production of electricity in dedicated power plants, (2) cogeneration of electricity and heat in combined heat and power plants and (3) production of bioethanol from biochemical conversion. The results showed that the strategy of using forest biomass residues for electricity production is advantageous in relation to the baseline (leaving the forest residues in the forest soil and using fossil fuel sources to produce electricity) only for some environmental impact categories. The scenario of cogeneration of electricity and heat production is the best alternative among the three scenarios. However, for impacts related with particulate matter and marine eutrophication it only would perform better than the baseline if measures are adopted to decrease the impacts of cogeneration or the fuels displaced are coal in electricity production and at least 12 % is fuel oil (and the rest is natural gas) in heat production. The conversion of forest biomass residues to bioethanol is the least beneficial scenario for all impacts under study.

Keywords: bioenergy production, consequential life cycle assessment, forest biomass residues.

6.1 Introduction

During the last decades, concerns regarding climate change and energy security have led to a growing demand for renewable sources in order to diversify the world's energy supply (WEC, 2016). Forest biomass residues are receiving increased attention as a response to the increasing global demand for energy at the same time that reduces the GHG emissions (EC, 2017). Indeed forest biomass has two important advantages when used for energy production: firstly, it is considered a major potential sink for atmospheric CO_2 and secondly, it contributes for reducing additional emissions of CO_2 by displacing fossil fuels (Zhang and Xu, 2003).

Given its location and climate, Portugal is a country well-suited for forest growth, where forests cover about 35 % of the territory. In this context, forest biomass residues are a potential renewable energy source for the country (ICNF, 2013). Therefore, many public policies have been launched over the years to promote the use of forest biomass residues in the country. In 2003, the Portuguese government launched the RCM 63/2003, to ensure the diversification of the national energy mix, the reduction of the fossil fuels dependence and to promote sustainable development (RCM, 2003). In 2006, the National Forest Fire Protection Plan (RCM 65/2006) was launched to assure the reduction of occurrence of forest fires through structural interventions, surveillance, combat and strategic actions (RCM, 2006a). In the context of these two actions, in 2006, a public tendering was launched to build 15 new power plants fuelled by forest biomass residues, with a total of 100 MW of installed capacity, to be located in areas chosen due to sustainability of water supply and forest resources as well as reduction of forest fire risk (RCM, 2005). This potential installed capacity was not fully mobilised by the private initiative and about 50 % of the capacity was not yet installed. Therefore, according to the Decree-Law 64/2017, the Portuguese government aims to assign 60 MW in special and extraordinary regime for the installation and operation of new power plants fuelled by biomass residues, with a maximum installed capacity of 15 MW per power plant (ME, 2017). Forest biomass residues are the most common fuels used currently in the biomass-based power plants in the country (da Costa et al., 2018; Dias, 2014). However, forest residues are limited and can be used to produce alternative bioenergy services other than electricity, such as heat and biofuels. Therefore, comparisons between the alternative energy uses of these residues become imperative to support decision-making.

Life cycle assessment has been widely applied to evaluate the environmental impacts of energy production from forest biomass residues (e.g., da Costa et al., 2018; González-García and Bacenetti, 2019; Hammar et al., 2019; Tagliaferri et al., 2018). However, there is a lack of

studies comparing the potential alternative bioenergy services (heating, electricity and biofuels) of forest biomass residues and most of studies are limited in their scope, focusing only one bioenergy service (e.g., Karlsson et al., 2014; Lindholm et al., 2011; Sandilands et al., 2009; Thakur et al., 2014). Besides, most of the studies adopt an LCA attributional approach disregarding the consequences that may occur in the market as a result of the strategic decisions taken (e.g., Djomo et al., 2015; Falano et al., 2014; Liang et al., 2017; Whittaker et al., 2016). For example, the collection of forest biomass residues for bioenergy production instead of the current use to increase soil nutrients in forest soil may induce the nutrients compensation by mineral fertilisation application.

In this context, the goal of this study is to evaluate the environmental consequences of implementing the Portuguese strategy of increasing 60 MW of installed capacity for electricity production from forest biomass residues (Decree-Law 64/2017), when held up against the consequences of losing alternative uses for the forest biomass residues, in particular, heat and biofuel production. Therefore, a consequential LCA was performed to determine the best use of the forest biomass residues to satisfy three alternative bioenergy services (electricity, heat and biofuel for transportation) from an environmental perspective.

6.2 Methodology

In LCA, depending on the goal of the study, two alternative approaches can be used: attributional and consequential. As this study aims to address the environmental consequences of changing the management of the forest biomass residues from the baseline (left on the field as soil improver) to future bioenergy scenarios (electricity, heat and biofuels), a consequential approach is the most suitable for this purpose. The overall approach followed to identify the processes to include in the consequential modelling of the systems under study was based on the ILCD handbook (EC/JRC/IES, 2010b).

6.2.1 Scenarios, functional unit and system boundaries

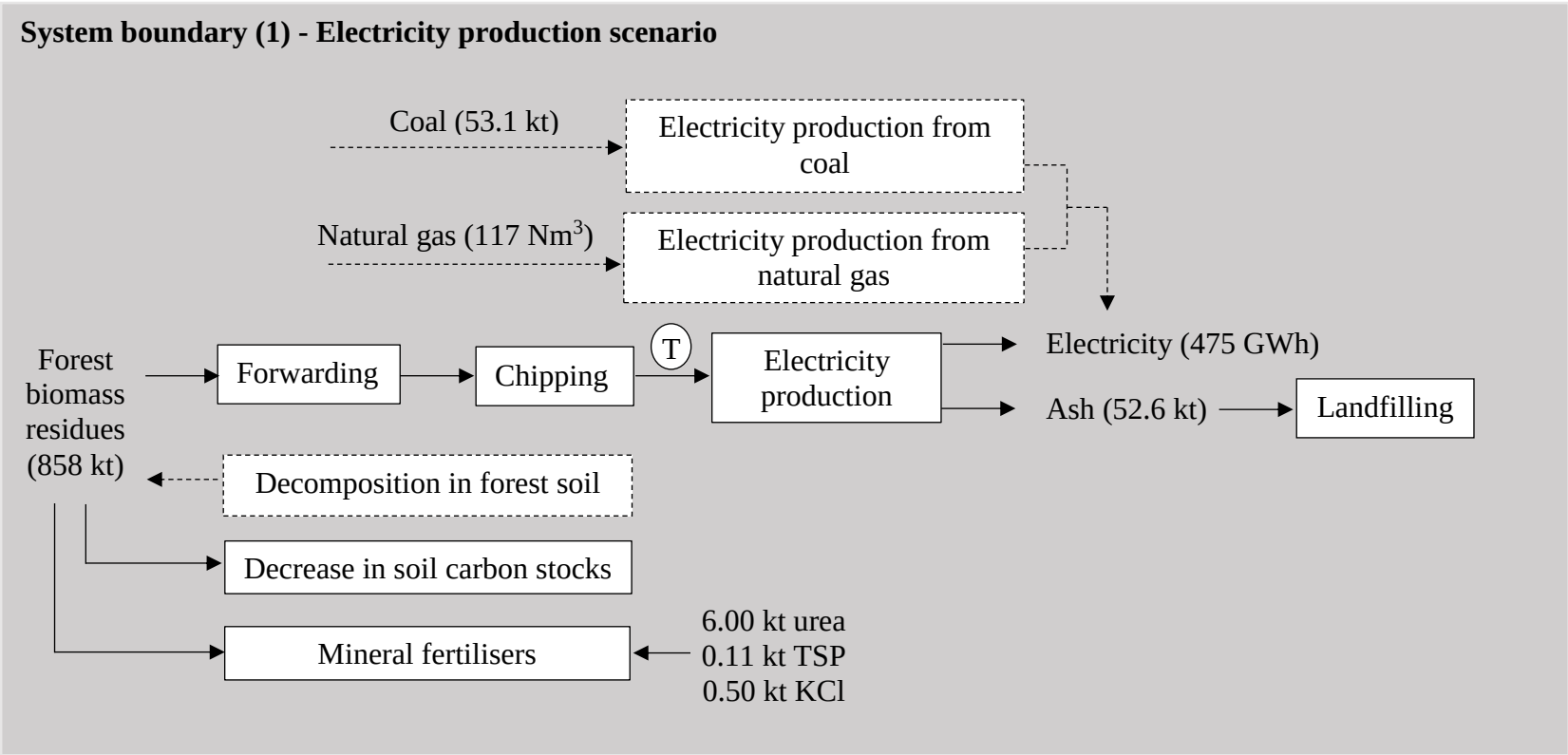
The bioenergy scenarios considered are: (1) the production of electricity in dedicated power plants, (2) the cogeneration of electricity and heat in combined heat and power plants and (3) the production of bioethanol from biochemical conversion of forest biomass residues.

The first scenario represents the implementation of the Portuguese Decree-Law 64/2017. The second scenario was selected in order to increase the energy conversion efficiency. Compared to the conventional electricity generation, cogeneration offers energy savings and

provides a more efficient use of the forest biomass resources by using the heat that would normally be lost (Bhatia, 2014). The third scenario is aligned with the goal of the Portuguese government to incorporate 10 % (v/v) of biofuels into the road transport sector by 2020 (MEID, 2010), which can potentially be produced from forest biomass residues. Bioethanol production was assessed, since cellulosic biomass has been commercially used in the European Union for bioethanol production (COM, 2017).

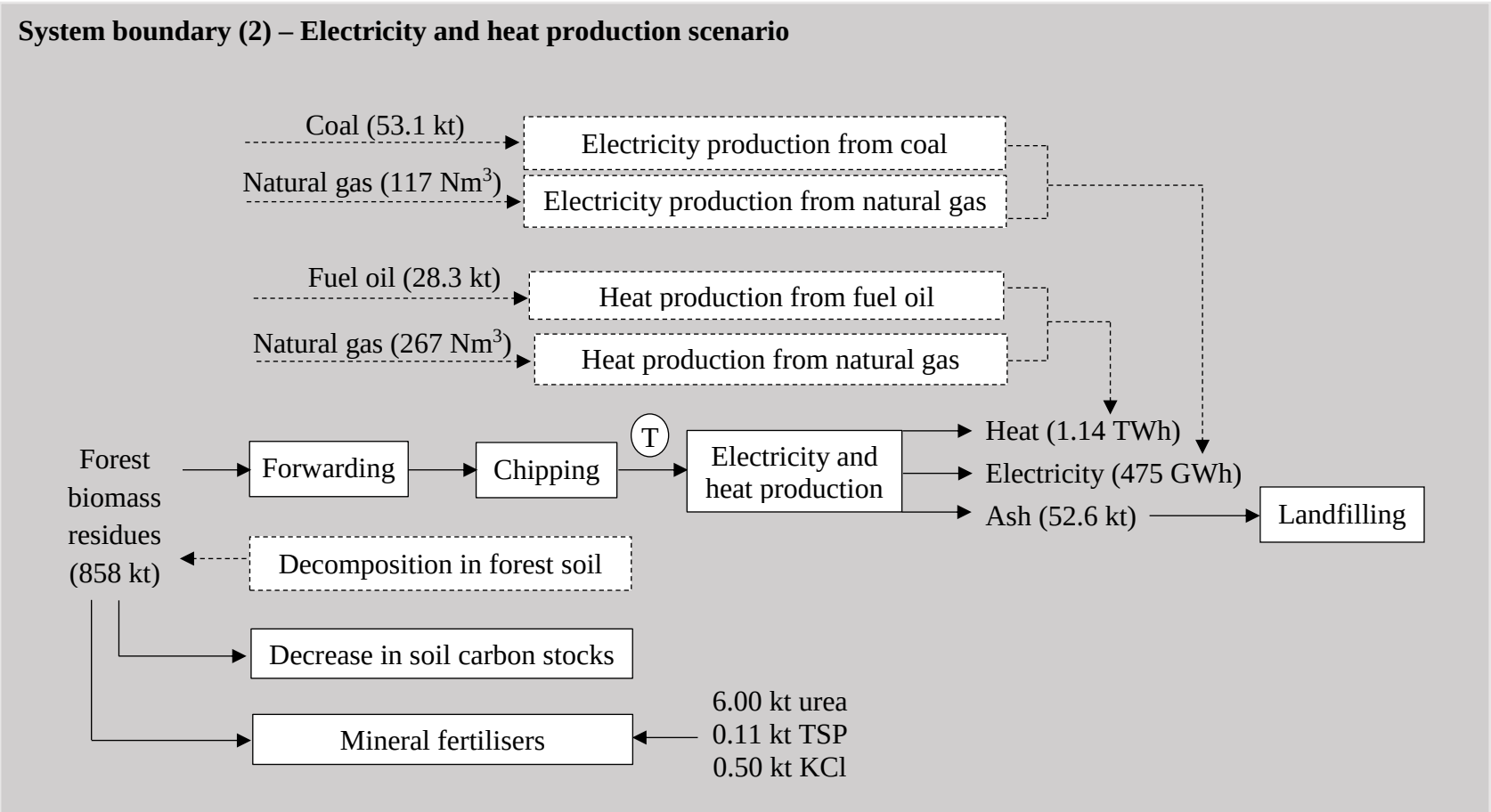
The FU selected was the increase in the consumption of forest biomass residues corresponding to a capacity of 60 MW for electricity production. This FU was kept in the three scenarios considered, in order to enable fair comparisons. Therefore, the FU is the annual consumption of 858 kt (wet basis) of forest biomass residues. This amount was calculated considering that: (1) the residues are logging residues (branches, foliage and tops) from eucalyptus and maritime pine stands with lower heating values of 17.5 MJ/kg and 17.9 MJ/kg (ECN, 2019; Tarelho et al., 2015), respectively and an average moisture content of 40 % (Silva, 2016); (2) an installed capacity of 60 MW allows the annual generation of 525.6 GWh of electricity (with continuous production over the year), but 9.6 % is self-consumed, which allows a net electricity production of 475 GWh; (3) the electricity conversion efficiency is 20 % assuming that the technology adopted will be grate furnaces (da Costa et al., 2018) as explained in Section 6.2.3.1 and; (4) forest biomass residues provide 98.4 % of the energy contained in the fuel feedstocks and the remainder is provided by natural gas (da Costa et al., 2018).

The study considers a short-term time scope (up to 5 years) and the geographical scope is Portugal. Detailed information of the size of market changes expected with the implementation of the scenarios under study can be found in Section C1 of the Appendix C. The system boundaries (Figures 29-31) encompasses the collection, processing and transportation of the forest biomass residues (including forwarding, chipping, loading and unloading operations); the bioenergy conversion (including forest biomass combustion or fermentation, as well as the final destination of ashes); the avoided forest residues decomposition in forest soil; the decrease in soil carbon stocks; the mineral fertiliser compensation due to nutrient losses; and the fuels displaced. Direct and indirect land use changes are not included in this study, since the use of forest residues does not demand extra land (Karlsson, 2018). Detailed information regarding the stages encompassed in the system boundaries are presented in Sections 6.2.2 to 6.2.7.



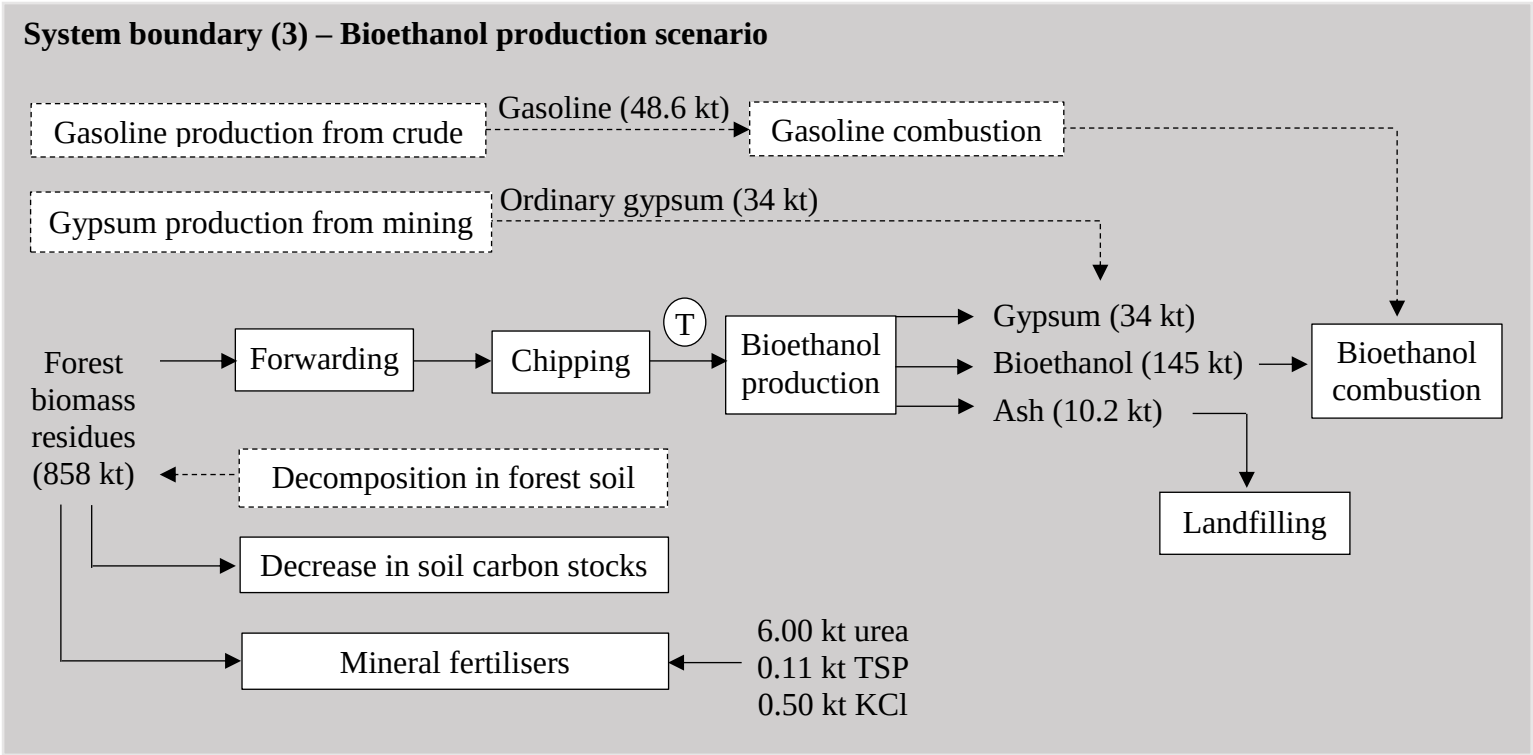
Legend: T: transport; TSP: triple superphosphate; KCl: potassium chloride.

Figure 29 - System boundaries of the forest biomass residues application for electricity production. The continuous line represents the environmental impacts and the dashed line represents the avoided impacts, both considered in relation to the baseline (biomass residues left on the field as soil ameliorant).



Legend: T: transport; TSP: triple superphosphate; KCl: potassium chloride.

Figure 30 - System boundaries of the forest biomass residues application for cogeneration of electricity and heat. The continuous line represents the environmental impacts and the dashed line represents the avoided impacts, both considered in relation to the baseline (biomass residues left on the field as soil ameliorant).



Legend: T: transport; TSP: triple superphosphate; KCl: potassium chloride.

Figure 31 - System boundaries of the forest biomass residues application for bioethanol production from the biochemical conversion. The continuous line represents the environmental impacts and the dashed line represents the avoided impacts, both considered in relation to the baseline (biomass residues left on the field as soil ameliorant).

6.2.2 Forest biomass residues

6.2.2.1 Availability

Despite in the scenarios under analysis the demand for forest biomass residues increases, it is observed that the production of those residues is not constrained. As forest biomass residues are a low value by-product from the production of wood for industrial purposes, the production of forest biomass residues follows the demand for wood and additional demand for bioenergy will not result in an additional supply of forest biomass residues. However, according to the last estimate of the Portuguese government, the forest biomass residues are not totally used and almost 3 Mt/yr of surplus forest biomass residues are available in the country (RCM, 2017). Of these residues, 543 kt are from maritime pine and 447 kt are from eucalyptus, which are enough to supply the 858 kt of forest biomass residues needed to provide the bioenergy services. Therefore, given that these residues are also the most common fuels used currently in the biomass-based power plants in the country (da Costa et al., 2018; Dias, 2014), it was assumed that these forest species will supply the bioenergy services in a proportion of 55 % of residues from maritime pine and 45 % from eucalyptus, according to its availability.

6.2.2.2 Collection, processing and transportation

According to the consequential modelling principles, only the processes that would be affected by a change in demand have to be included in the system boundaries. Therefore, the operations related with forest management were excluded, because the use of logging residues would not affect them. Hence, all scenarios start with the collection of the residues from the forest soil.

The collection and processing of forest biomass residues can have different configurations (Dias, 2014; Spinelli et al., 2007). In this study, it was assumed that loose logging residues are collected with a forwarder and chipped at the roadside using a mobile chipper. The chipped biomass is then loaded onto trucks (payload 20 t) and transported to the power plant/industry over a distance of 35 km, according to the optimal distances based on Viana et al. (2010). During residue chipping, the forest residues suffer a dry matter loss of 2 % (Forsberg, 2000; Whittaker et al., 2011). The inventory data for the eucalypt and maritime pine residues collection and chipping are taken from Dias (2014) and Quinteiro et al. (2019) adjusting the moisture of the forest biomass residues to 40 % (Silva, 2016).

6.2.3 Bioenergy conversion

6.2.3.1 Electricity production

The existing power plants in Portugal using forest biomass residues are of two types: grate furnace and fluidised bed furnace. Fluidised bed furnaces have been proposed as the most suitable option for large-scale biomass combustion because of the growing awareness of environmental impacts linked to forest biomass combustion, energy efficiency and fuel flexibility (Frey et al., 2003; Tarelho et al., 2011, 2015; Calvo et al., 2013). However, the new power plants will operate with a maximum installed capacity of 15 MW (ME, 2017) and the fluidised beds are designed for capacities above 20 MW (Lee and Shah, 2013; Vamvuka, 2010). For these reasons, the first scenario assesses the production of electricity from the combustion of forest biomass residues in grate furnace, which is the best established and most commonly used technology for converting biomass to electricity (and heat) at this scale (EPA, 2007).

The inventory data for the conversion of forest biomass residues to electricity were taken from da Costa et al. (2018). Furthermore, it is important to mention that the biogenic CO_2 emissions during forest biomass combustion do not count to the climate change impact, based on the idea that they are originated by the carbon sequestered during tree's growth (Brandão et al., 2013).

6.2.3.2 Combined heat and electricity production

In the first scenario, the electrical efficiency is 20 % with 80 % of the heat contained in the fuel being wasted. In the second scenario, the waste heat is converted into useful heat in combined heat and power plants with grate furnace technology, which in Portugal are typically used in the industrial sector (DGEG, 2019). The additional thermal energy produced is 1.14 TWh, assuming a typical thermal recovery of 60 % (Hedman, 2000; Saldanha, 2001; Zhou et al., 2015). The inventory data for the combustion of forest biomass residues are the same adopted in the first scenario (from da Costa et al. (2018)) with the additional production of heat, assuming that the heat recovery does not generate additional burdens to the environment.

6.2.3.3 Bioethanol production

In the third scenario, the inventory data of bioethanol production (biochemical conversion) from eucalypt and maritime pine are based on data from Liang et al. (2017) for bioethanol production from the fermentation of a mix of forest biomass residues from the national forests of USA. The implicit assumption that bioethanol production is equivalent for

other types of forest residues is also adopted in other studies that use wood based feedstock as raw material (Aden et al., 2002; Fu et al., 2003; Kumar et al., 2009).

The biochemical conversion includes the following steps: pre-treatment of the lignocellulosic material (to degrade lignocellulosic network into its fractions), enzymatic hydrolysis (using commercial enzymes to obtain fermentable sugars), saccharification/fermentation of sugars (to convert sugars into bioethanol using different microorganisms or bacteria) and distillation of bioethanol with dehydration (to separate and purify the bioethanol). The non-degraded lignin, cellulose and hemicellulose fraction of the feedstock is used to provide electricity and heat needed by the facility (Liang et al., 2017).

According to the inventory, 145 kt of bioethanol can be produced through the biochemical conversion of 858 kt of forest biomass residues. The bioethanol production also leads to the co-production of 34 kt of gypsum. The combustion emissions from bioethanol burned in a passenger vehicle was taken from Daystar et al. (2015) and an average fuel consumption of 0.099 kg of fuel/km was assumed (González-García et al., 2012).

6.2.3.4 Ash landfilling

Scenarios 1 and 2 generate 52.6 kt of ash from the combustion of forest biomass residues, while scenario 3 generates about 10.2 kt of ash from the non-degraded fraction of forest biomass residues combusted to produce electricity and heat needed by the facility. In all scenarios, it is assumed that 25 % of the ashes are fly ashes and 75 % are bottom ashes as in Coelho (2010). As ash valorisation is not yet well developed in Portugal, it was considered that the ashes are disposed of in sanitary landfills and the inventory data were taken from da Costa et al. (2019).

6.2.4 Avoided decomposition of forest biomass residues in the forest soil

Currently, in Portugal, part of the forest residues is sold as fuel to produce energy in power plants and another part is left in the forest soil to restore the nutrients that have been removed from the soil during the tree growth. The forest biomass residues collection avoids the decomposition of these residues in the forest soil, which in turn avoids emissions to the environment. Emissions of N_2O, NOx and ammonia (NH_3) to air, emissions of nitrate (NO_3^-) to water and emissions of P and K to soil were considered. The CO_2 emissions from forest residues decomposition are biogenic and, thus, were not considered in the climate change impact category. The release of CH_4 emissions were not considered because the conditions in

forest soils are generally aerobic (EPA, 2010). The emissions depend on the composition of the forest residues, which is presented in Table 12.

Table 12 - Composition of eucalypt and pine residues under study. Source: adapted from ECN/TNO (2019).

Composition (g/kg of dry residues)	Eucalypt	Maritime pine
C	488	513
N	3.60	4.00
P	0.35	0.23
K	2.47	1.38

The emission factors of N-related emissions are based on crop residues due to the lack of data specific for forest residues. The emission factors are: 1.25 % directly emitted as N_2O (IPCC, 2006), 10.5 % emitted as NH_3 (de Ruijter et al., 2010), 0.7 % emitted as NOx (Hamelin et al., 2012) and 10 % emitted as NO_3^- (Perrin, 2013). Indirect N_2O emissions arising from N volatilised as NH_3 and NOx and lost through leaching as NO_3^- were calculated using emission factors from IPCC (2006): 0.010 kg N_2O per kg of NH_3 and NOx volatilised and 0.025 kg N_2O per kg of NO_3^- leached. The remaining nitrogen was considered to be uptaken by the plants.

The P available for plants can range from 5 to 22 % of the total P contained in the forest biomass residues (Espinosa et al., 2017; Maltais-Landry and Frossard, 2015; Nachimuthu et al., 2009). The mean value of this range (13.5 %) was adopted and the remaining P was considered to be emitted to the soil from the decomposition of the forest residues. The K available for plants considered in this study is 17.4 % of the total K in the forest biomass residues (Liao et al., 2013; Sinha et al., 2018). The K that is not uptaken by the plants was considered to be emitted to the soil.

6.2.5 Decrease in carbon stocks

The forest biomass residues contain carbon that would have been left in the forest soil if not used for bioenergy production. The decomposition of the forest biomass residues that occurs naturally by soil microorganisms produces organic substances which are incorporated into the soil, contributing to forest soil litter, to recirculate part of the nutrients and to be slowly incorporated as soil organic carbon increasing carbon stocks. However, when the forest residues

management is changed due to increased residues collection, the dynamic equilibrium in the soil is broken and generally the carbon inputs in the soil are smaller than the carbon outputs, leading to a decrease in soil carbon stocks (Cherubini, 2010; Lal, 2008).

The decrease in soil carbon stocks due to the collection of forest biomass residues was calculated as the difference between carbon contained in the initial forest biomass residues and the carbon still present in the soil after 100 years. The latter is assumed to be permanently stored in soil (EC/JRC/IES, 2012). The mass of carbon present in the forest residues was considered to follow an exponential decomposition, as shown in Equation 2 (EIA, 2007).

$$M(t) = M(0).e^{-kt} \quad (2)$$

where *M(t)* is the remaining fraction of residue on the soil at time *t* (expressed as mass of carbon), *M(0)* is the initial mass of residue on the soil (expressed as mass of carbon) and *k* is the residue decomposition rate.

The decomposition rate (k) for mass loss under aerobic conditions of the eucalypt and maritime pine residues was considered equal to 0.050 yr^{-1} and 0.028 yr^{-1}, respectively (Russell et al., 2014). Considering a time horizon of 100 years, it is estimated that 0.67 % of the carbon contained in the eucalypt and 6.08 % of the carbon contained in the maritime pine residues stay permanently stored in soil. Therefore, the decrease in carbon stocks if forest residues are collected from forest soil to produce bioenergy is equivalent to 9.60 kt of carbon, meaning that 35.2 kt of CO_2 would not be sequestered.

6.2.6 Mineral fertilisers compensation

Forest biomass residues contain significant amounts of macro-nutrients, mainly N and K but also P. When forest biomass residues are incorporated in the soil, the nutrients are released and left available for trees. However, when the forest biomass residues are collected for bioenergy and the combustion ashes are not recirculated to the same fields, these nutrients are removed from the soil. Therefore, the amount of nutrients removed with the forest residues should be replaced by mineral fertilisers. It was considered that the new demand for nutrients in the forest soil will be fulfilled by the application of urea, triple superphosphate (TSP) and potassium chloride (KCl). This choice is supported by the trends in demand/consumption of traditional fertilisers in Portugal shown in Section C2 of the Appendix C. Inventory data on the production of fertilisers were sourced from Ecoinvent database (Ecoinvent, 2017).

The equivalence in terms of nutrients between forest biomass residues and mineral fertilisers was established based on the respective bioavailability of nutrients. The amount of N

contained in urea is 450 g/kg and 56 % is available for plants (difference between the N content in urea and the losses by volatilisation and leaching during its application). Regarding TSP, its P content is 210 g/kg, of which 86.5 % is available for plants (Mullins and Sikora, 1994). The amount of K in the KCl is 498 g/kg and 67.5 % of this K is available for plants (Naylor and Schmidt, 1986). The bioavailability of nutrients in the forest biomass residues was presented in Section 6.2.4. The amount of mineral fertilisers needed to compensate the removal of forest biomass residues is shown in Table 13.

Table 13 - Amount of mineral fertiliser added to compensate the forest biomass residues collection.

Fertiliser compensation (g/kg of dry residues) [(1)]	Eucalypt	Maritime pine
N	4.95	5.49
P	0.05	0.04
K	0.64	0.36

[(1)] The amount of fertiliser was calculated including the losses due to volatilisation and leaching.

The emissions to air related to the application of urea due to volatilisation were quantified as follows: 1.25 % directly emitted as N_2O (IPCC, 2006), 21 % emitted as NH_3 (Hamelin et al., 2012), 1.3 % emitted as NOx (Hamelin et al., 2012). The same emission factors used to predict the indirect emissions of N_2O of forest residues were used for the urea application. In addition, 20 % was emitted as NO_3^- to water due to leaching (Galloway et al., 2004). For the remaining mineral fertilisers, P and K not uptaken by the plants were considered to be emitted to the soil, i.e. 13.5 % and 32.5 %, respectively.

6.2.7 Products displaced and equivalency

Table 14 shows the products displaced by the product/service affected. For short-term changes, the immediately affected product/service are typically the least competitive (often using older technology), since these suppliers are most sensitive to price changes (Weidema, 2015).

Table 14 - Product displaced and market trend for the product/services under study.

Product affected	Decision leads to increase in	Market scope	Market trend	Product displaced	Reason
Electricity	Supply	PT	Growing	Coal and natural gas	Least competitive
Industrial heat	Supply	PT	Stable	Fuel oil and natural gas	Least competitive
Bioethanol	Supply	PT	Stable	Gasoline	Least competitive
Gypsum	Supply	GLO	Growing	Ordinary gypsum	Single supplier

PT: Portugal; GLO: Global.

The decision to produce bioenergy from forest biomass residues leads to changes in the supply of electricity, heat, bioethanol and gypsum. In consequential LCA, an additional supply/demand of a product is modelled according to the market trend: "growing, stable, or slightly declining" or "strongly declining" (EC/JRC/IES, 2010b). For a "growing, stable, or slightly declining" market, the additional supply of a product overcharges the market of that product, which can lead the suppliers to be forced out of the market, typically the least cost-efficient technology. In a "strongly declining" market, the additionally supplied product could be argued to not be used at all. The assumptions for identifying the market trend and the products displaced assumed in this study are described in detail in Section C3 of the Appendix C.

In the consequential approach, all the alternatives compared should be equivalent and provide the same services to society (Hedegaard et al., 2008). Therefore, for a product to work as substitute, it needs to fulfil the same functions for the user in terms of equivalency, technical quality, etc.

The equivalency of electricity production from biomass residues was assessed comparing the production of electricity from coal and natural gas in Portugal. To produce 1 kWh of electricity, the consumption of forest residues is 1.83 kg (wet basis) (da Costa et al., 2018), whereas the consumption of hard coal is about 0.411 kg (Ecoinvent, 2017) and the consumption of natural gas is about 0.279 m^3 (Ecoinvent, 2017). The inventory data for electricity production from coal and natural gas were taken from Ecoinvent database (Ecoinvent, 2017).

Regarding the equivalency to produce heat, to produce 1 kWh of heat it is necessary to consume 0.764 kg of forest biomass residues, while the consumption of fuel oil is 0.310 kg and that of natural gas is 0.255 m^3. The data for heat production from natural gas and fuel were obtained from Ecoinvent database (Ecoinvent, 2017).

The equivalency between gasoline and bioethanol was assessed according to the average fuel consumption of a vehicle running with both fuels. The consumption of gasoline and bioethanol was assumed to be 0.066 and 0.099 kg of fuel/km, respectively (González-García et al., 2012). The production of gasoline was taken from Ecoinvent database (Ecoinvent, 2017) and the combustion emissions of gasoline in a passenger vehicle were taken from Daystar et al. (2015).

The production of bioethanol leads to the co-production of gypsum that can be sold for use in concrete and wallboard manufacturing (Martinez et al., 2001). For calculation purposes, it was assumed that the equivalency of the gypsum from bioethanol production is equal to the ordinary gypsum production. The inventory data for ordinary gypsum production were taken from Ecoinvent database (Ecoinvent, 2017).

6.2.8 Impact assessment

The impact assessment method used in this study is the ILCD as recommended by the JRC and the European Commission (EC/JRC/IES, 2012). The following environmental impact categories were included: CC, HTnc, HTc, PM, POF, AC, FEu, MEu, FEc and MFD.

6.3 Results

Figure 32 presents the relative contribution of each stage for the total impact in the scenario of electricity production from forest biomass residues. The electricity production stage from forest biomass residues is the main hotspot for CC, HTnc, HTc, PM, POF, MEu and FEc (40-93 % of the total impact). In the case of CC, most of the impact in electricity production comes from CO_2 emissions during natural gas combustion, while for the remaining impact categories the emissions with origin in the forest residues combustion is dominant: Zn and Hg in HTnc, Cr VI in HTc, $PM_{2.5}$ and PM_{10} in PM, NOx both in POF and MEu and Zn and Sb in FEc. Despite not being the main hotspot, the electricity production stage is also important for AC, mainly due to emissions of NOx from forest biomass combustion. The main hotspot in the case of AC is the mineral fertiliser compensation (45 % of the total impact) mainly due to the emissions of NH_3. The mineral fertiliser compensation is also the main hotspot in the case of FEu, although the environmental impact is negligible compared to the impact that is avoided. For MFD, the collection, processing and transportation of forest biomass residues represents 83 % of the total impact, which is mostly due to the depletion of lubricating oil and diesel consumed during the operations of forwarding, loading, unloading, chipping and transportation.

Ash landfilling presented little importance for the categories under study, while the decrease in soil carbon stocks is relevant for CC (26 % of the total impact).

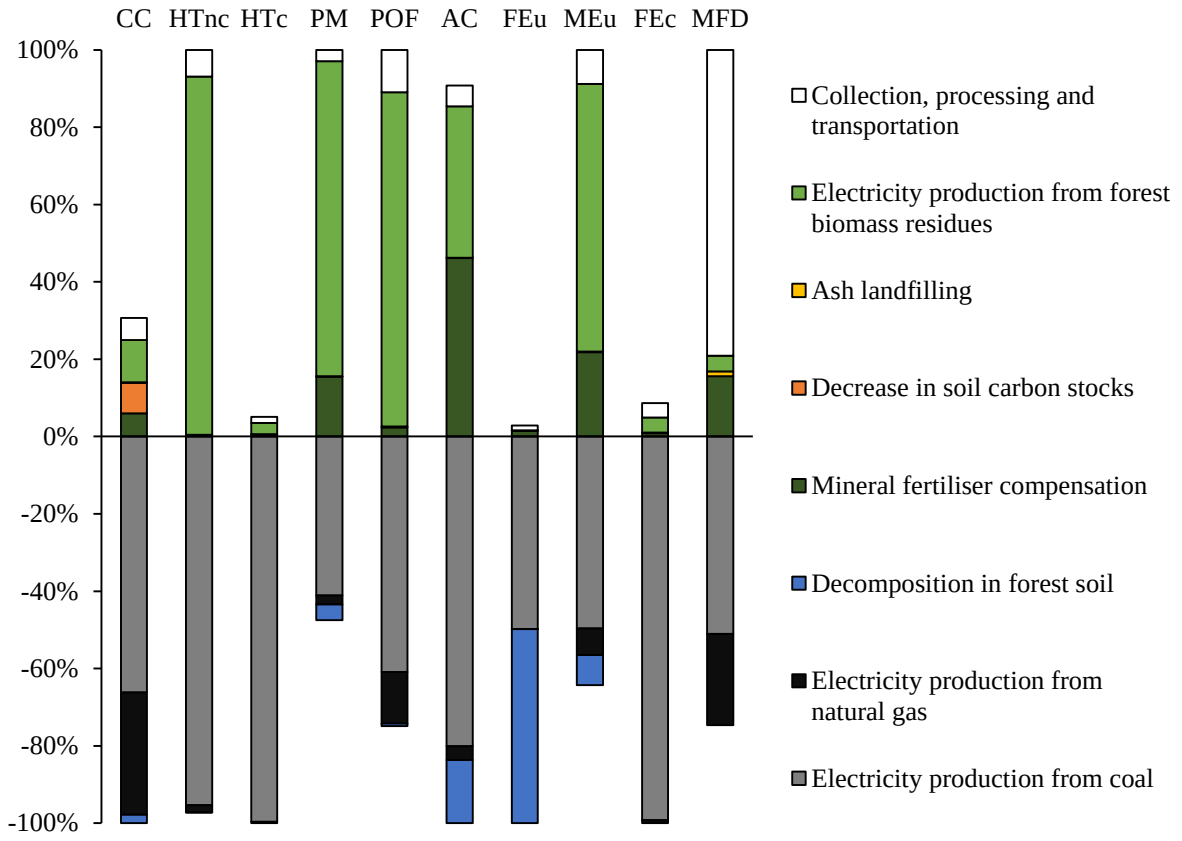


Figure 32 - Hotspot analysis for the scenario of electricity production. The environmental impacts related to the production of electricity from the forest biomass residues are shown in the positive axis and the avoided impacts are shown in the negative axis.

Regarding the avoided impacts, the electricity production from coal is the most relevant contribution for all impact categories (66-99 % of the total impact) except FEu. The emissions due to hard coal combustion are mainly responsible for the impact categories CC, PM, POF, AC and MEu. The main causes are CO_2 and N_2O in CC, SO_2 and $PM_{2.5}$ in PM, NOx and SO_2 in both POF and AC and NOx in MEu. Hard coal mine operation and hard coal preparation are mainly responsible for the impact categories HTnc, HTc, FEc and MFD. These impacts are mainly originated from emissions of Zn and As in HTnc, Cr VI in HTc, Zn and Ni in FEc and depletion of coal in MFD. In the category FEu, the forest residues decomposition in the forest soil is the main responsible for the impact avoided mainly due to P emissions, although

electricity production from coal is also relevant for the avoided impact. The contribution of electricity production from natural gas for the avoided impacts was less significant.

The attributional LCA study of da Costa et al. (2018) also assessed the production of electricity from the combustion of forest biomass residues in grate furnaces and fluidised beds. Despite that study does not take into account the fertiliser compensation and the decrease in soil carbon stocks to calculate the impacts, the same trend was observed for the relative importance of each stage for the impact categories in common with this study (CC, PM, POF, AC, FEu, MEu and MFD). The reason is that the main contribution for the total impact are related to emissions from the foreground processes and not from the background processes. The latter differ in both studies, consisting in average data in da Costa et al. (2018) and marginal data in this study. The net impact of this study cannot be compared since avoided impacts associated with replaced fuels and decomposition in the forest soil were excluded from da Costa et al. (2018).

Figure 33 presents the relative contribution of each stage for the different impact categories in the case of the scenario of cogeneration of electricity and heat from forest biomass residues. In this second scenario, the relative contribution of the stages for the total environmental impact is the same as in the first scenario. Regarding the avoided impact, electricity production from coal remains the main contribution for most of the impact categories (39-93 % of the total impact avoided), mainly due to emissions from hard coal mine operation, preparation and combustion. The exceptions are the CC and MFD, in which the emissions avoided due to heat production from natural gas are the main responsible for the impact avoided (45-47 % of the total impact avoided), mainly due to the emissions of CO_2 during natural gas combustion and the depletion of natural gas, respectively. For POF, the heat production from natural gas was also significant, being 29 % of the total impact avoided.

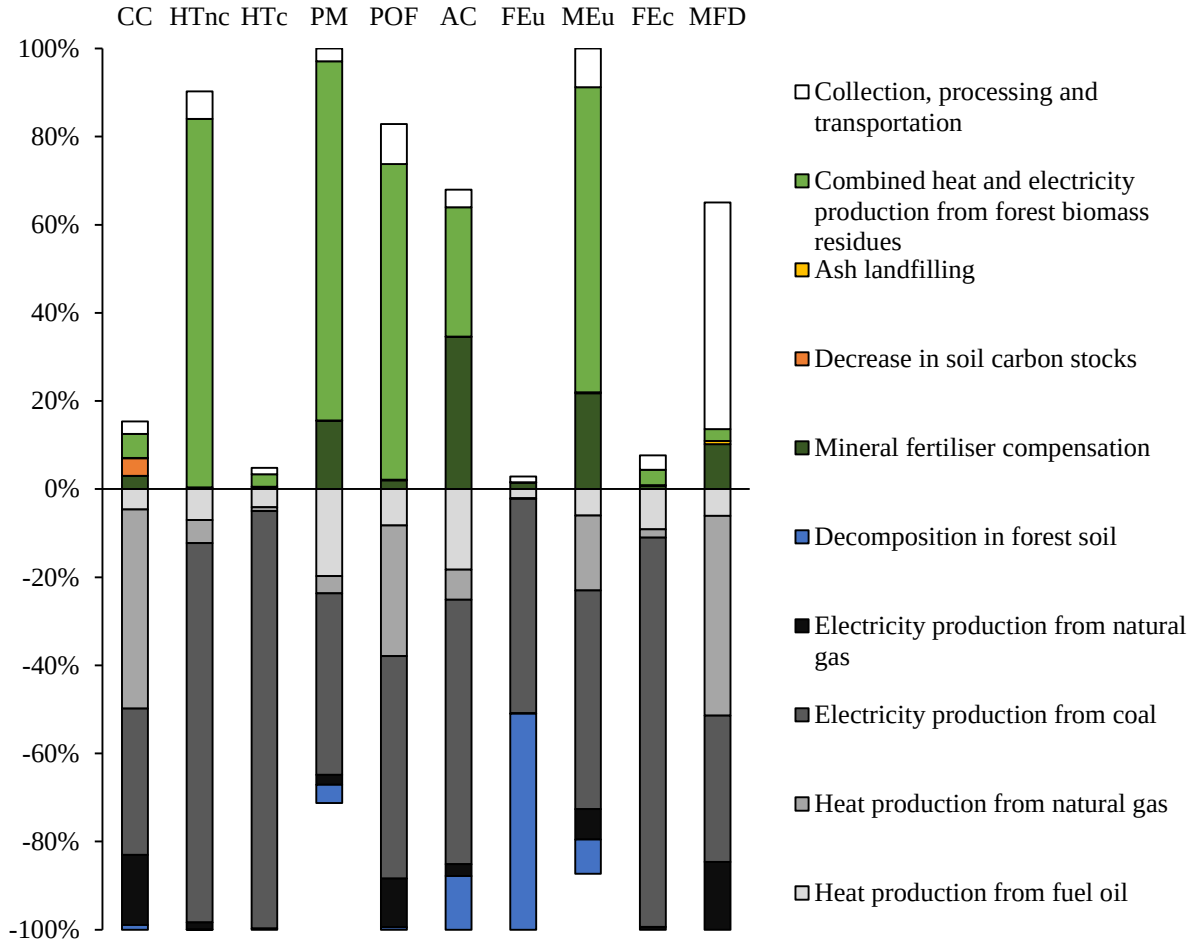


Figure 33 - Hotspot analysis for the scenario of cogeneration production. The environmental impacts related to the production of electricity from the forest biomass residues are shown in the positive axis and the avoided impacts are shown in the negative axis.

Figure 34 presents the relative contribution of each stage for the impact categories under study in the case of the scenario of bioethanol production from forest biomass residues. The hotspots depend on the impact category under analysis. The bioethanol (and gypsum) production represent the largest impacts for CC, HTnc, HTc, FEu, FEc and MFD (35-81 % of the total impact). These impacts are mainly related with emissions of CO_2 in CC, Zn in HTnc, Cr VI in HTc, Zn and Ni in FEc and depletion of In in MFD. The mineral fertiliser compensation is the main hotspot for PM, AC and MEu (38-63 % of the total impact). For PM the impact mainly results from $PM_{2.5}$ emissions, for AC is mainly from SO_2 and NH_3 emissions and for MEu is mainly from NO_3^- and NOx emissions. The bioethanol combustion in a passenger car is the hotspot for POF (41 % of the total impact, respectively), mainly due to the NOx emissions.

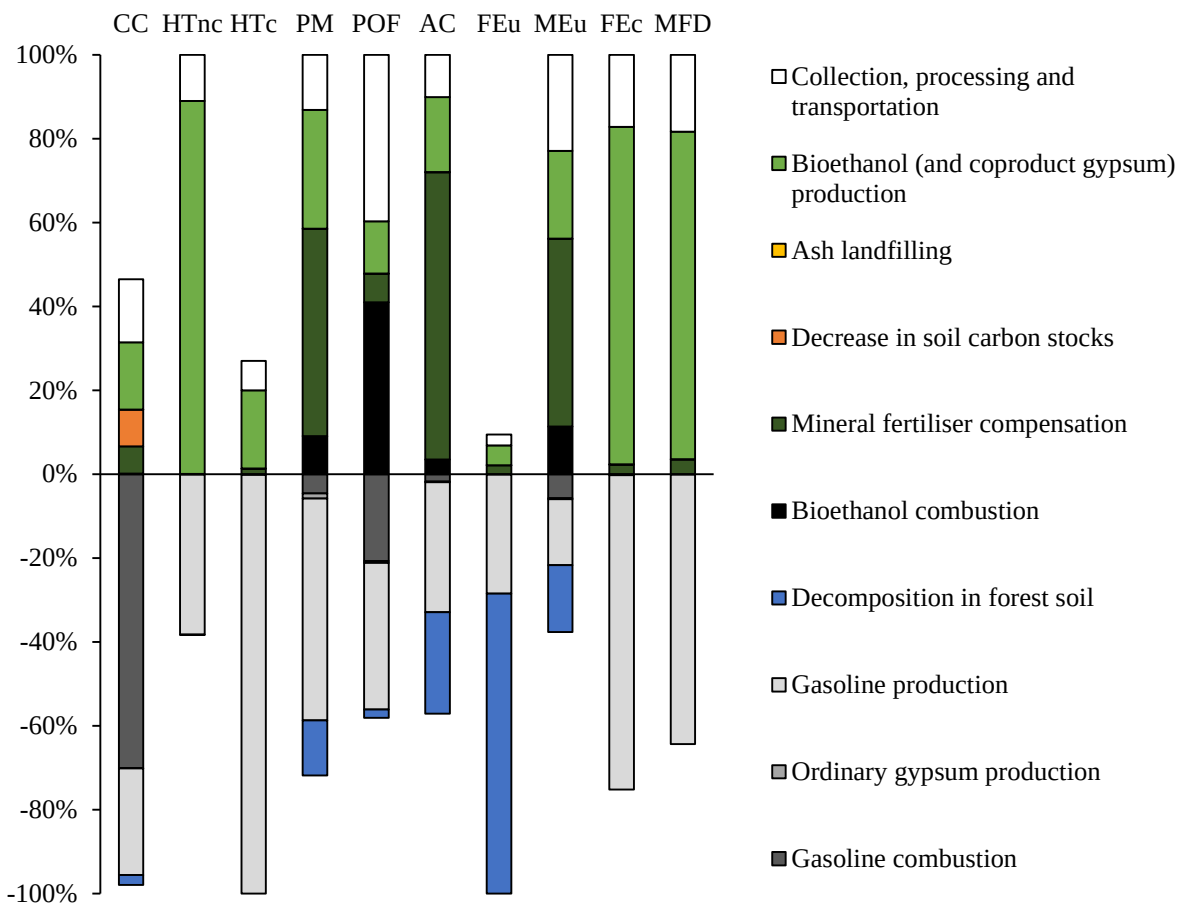


Figure 34 - Hotspot analysis for the scenario of bioethanol production from the biochemical conversion. The environmental impacts related to the production of electricity from the forest biomass residues are shown in the positive axis and the avoided impacts are shown in the negative axis.

Regarding the avoided impacts, the gasoline production is the main hotspot for all impact categories (37-99 % of the total impact), except CC and FEu. For CC, the impact avoided is mainly related to the gasoline combustion (73 % of the total impact avoided) due to CO_2 emissions. Regarding FEu, the decomposition of the forest residues in the forest soil is the main responsible for the impacts avoided (93 % of the total impact avoided), mostly due to P emissions avoided.

Table 15 presents the net impact (difference between impacts and avoided impacts) obtained for the three scenarios under study per FU. The results of the first scenario show that the strategy of the Portuguese government to increase the installed capacity for electricity production from biomass residues presents net environmental savings for CC, HTc, AC, FEu and FEc. However, the implementation of this strategy leads to environmental impacts for the

remaining impact categories (HTnc, PM, POF, MEu and MFD) that are higher than those associated with electricity production from fossil fuels (natural gas and coal).

Table 15 - Net impact results for bioenergy production from forest biomass residues per FU.

Impact category	**Unit**	**Scenarios**		
		1	2	3
CC	kg CO_2 eq	-3.05E+08	-7.42E+08	-8.68E+07
HTnc	CTUh	1.24E+00	-4.80E+00	2.54E+01
HTc	CTUh	-1.06E+01	-1.12E+01	-2.91E+00
PM	kg $PM_{2.5}$ eq	1.75E+05	9.57E+04	2.94E+04
POF	kg NMVOC eq	4.98E+05	-4.10E+05	2.91E+05
AC	molc H^+ eq	-3.54E+05	-1.64E+06	1.11E+06
FEu	kg P eq	-2.45E+05	-2.51E+05	-1.60E+05
MEu	kg N eq	3.10E+05	1.10E+05	2.64E+05
FEc	CTUe	-8.96E+08	-1.02E+09	9.38E+07
MFD	kg Sb eq	1.92E+01	-4.08E+01	1.19E+02

The net impacts calculated for the second scenario show that cogeneration of electricity and heat has a better environmental performance than production of electricity only, as net environmental savings were obtained for all impact categories, except for PM and MEu. In the latter impact categories, the environmental impacts from electricity and heat production from forest biomass residues exceed the environmental impacts of electricity production from natural gas and coal and heat production from natural gas and fuel oil.

The net impacts obtained for the third scenario show that use of forest biomass residues to produce bioethanol from biochemical conversion is the least beneficial for all impact categories other than PM and MEu for which the scenario of electricity production is the worst. The third scenario only presented net environmental savings for the impact categories CC, HTc and FEu. For the remaining categories, the environmental impacts associated with the use of bioethanol from forest biomass residues are higher than those resulting from the use of gasoline. The main reasons for this poor environmental performance of bioethanol are: the low substitution rate between the gasoline and bioethanol, since it takes around 66 % of gasoline to make the same distance as a bioethanol driven car and the amount of inputs required to produce the bioethanol, such as sulphuric acid, lime, phosphate and enzymes.

The comparison of the three scenarios under study, shows that cogeneration of electricity and industrial heat (second scenario) leads to larger net environmental savings for all categories than the other scenarios. The reduction in the environmental impacts ranges between 5 and 221 % comparatively to the scenario that represents the Portuguese government strategy (first scenario). However, for PM and MEu categories, the second scenario performs worse than using fossil fuels, as referred above. However, the share of fossil fuels displaced by using forest biomass residues has been assumed based on past trends and in practice other shares are possible. Therefore, a sensitivity analysis was performed assessing how the environmental impacts of the second scenario would change when the fuel displaced to produce electricity is only coal (instead of 60 % coal and 40 % natural gas as considered by default). This change would be enough for turning the use of forest biomass residues more favorable for the impact category MEu, but not for PM. In the latter impact category, the results would be more favorable only when the percentage of fuel oil displaced is higher than 12 % (instead of 8 % by default).

6.4 Conclusion

This study assessed the environmental consequences of implementing the strategy of the Portuguese government to increase the installed capacity for electricity production from biomass residues, in contrast to the consequences of losing alternative uses for these residues, in particular, cogeneration of electricity and heat and bioethanol production. The hotspot analysis showed that the stages that contributed most for the environmental impacts were bioenergy production and mineral fertiliser compensation. Therefore, environmental improvements in the systems analysed could be more effectively reached by improving efficiencies of bioenergy conversion or implementing more efficient flue gas treatment (e.g. for reduction of particulate matter), as well as by optimising fertilisation practices. Regarding the avoided impacts, they are mainly dominated by substitution of fossil fuels in all scenarios, but forest residues decomposition in forest soil is also important for the impact category FEu. Therefore, they should be potentially accounted for in consequential LCA studies of bioenergy systems. Other processes that are normally excluded from attributional LCA studies of bioenergy but should be considered when a consequential perspective is adopted are the decrease in soil carbon stocks (relevant for CC) and the mineral fertiliser compensation (relevant for several impact categories as mentioned above).

None of the three scenarios analysed has a better environmental performance than the baseline (leaving the forest residues in the forest soil and using fossil fuel sources to produce

energy) for all the impact categories analysed. The strategy of the Portuguese government to implement more installed capacity of electricity production from forest residues assessed in the first scenario, increases the environmental performance of CC, HTc, AC, FEu and FEc, while is worse for HTnc, PM, POF, MEu and MFD. The conversion of forest biomass residues to bioethanol is the least beneficial for all impact categories. The cogeneration of electricity and heat appears to be the best use of the forest biomass residues. However, if no measures are adopted to decrease the impacts of cogeneration, it would only be more beneficial than the baseline for all categories under study, if the fuels displaced are coal in electricity production and at least 12 % is fuel oil (and the rest is natural gas) in heat production.

Chapter 7

General conclusions and future perspectives

7.1 General conclusions

This thesis presents and analyses the potential environmental impacts of bioenergy production from forest biomass residues in Portugal following the principles of LCA methodology. The main findings of the present thesis can be summarised as follows.

Chapters 1 and 2 showed that there is a limited number of LCA studies regarding either bioenergy production from forest biomass residues or woody biomass ash management. Therefore, this thesis provides important information from these activities, that can be used for supporting decision-making by different actors from the value chains evaluated, in particular forest managers, stakeholders involved in biomass processing and logistics and industrial stakeholders (both from the energy conversion and ash users sectors), as well as governmental bodies (from the forest, energy and waste management sectors). This information is also potentially relevant for the scientific community as it can also be used in future studies.

Chapter 3 showed that the environmental impacts of electricity production from forest biomass residues is highly influenced by the energy conversion technology. For the six scenarios studied, it was found that the energy conversion was the hotspot stage in most of the impact categories studied. This result allowed suggesting future improvement actions for the power plant-related stakeholders, such as the implementation of systems of pollution control (e.g. selective catalytic reduction) and the adoption of high-efficiency combustion systems to convert the forest biomass into electricity. The forest management stage had also prominence in some impact categories (freshwater eutrophication, marine eutrophication and mineral and fossil depletion), mainly due to mineral fertilisers application. This is an important result for forest managers, who should optimise fertiliser application in order to decrease more effectively the impacts of forest management. An important conclusion for the stakeholders involved in biomass processing and logistics is that the configuration of the forest residues supply chain has a small role in the total environmental impacts and, thus, other factors such as economic and technical may prevail in the selection of the configuration chain. The results also showed that the fluidised bed furnace technology presented the best environmental performance for all impact categories assessed in comparison with the moving grates furnace technology. This is a relevant conclusion mainly for governmental bodies in charge of defining energy strategies,

which in the absence of more influencing factors should give preference to the fluidised bed instead of grates furnaces.

Chapter 4 focused on the end-of-life of woody biomass ash and its valorisation in construction materials (cement mortar, adhesive mortar, concrete blocks and bituminous asphalt) against landfilling. For the different valorisation/disposal alternatives considered, different scenarios were studied according to the type of ash (bottom and fly ashes) and the combustion technology (fluidised bed and grate furnaces). The landfilling scenarios showed that emissions generated in the sanitary landfill due to diesel combustion during ash spreading are significant for the impact categories of climate change, particulate matter, photochemical ozone formation and acidification, while leachate treatment is the main responsible for the remaining impact categories.

Regarding the valorisation scenarios, the best choice in terms of environmental performance depends on the type of ash. The production of cement mortar using fly ash from vibrating grate furnace (the only scenario assessed for this ash) presented environmental savings compared to the production of cement mortar with traditional raw materials. For bottom ash from vibrating grate furnace, the production of bituminous asphalt (the only scenario assessed for this ash) revealed benefits in comparison with the traditional production of bituminous asphalt. Regarding fly ash from fluidised bed furnace, the two valorisation alternatives analysed (production of cement mortar and concrete blocks) have similar benefits (difference between the scenarios is smaller than 2 %) and both have better environmental performance than traditional production of these products. The best valorisation alternative for bottom ash from fluidised bed furnace is the production of bituminous asphalt, which leads to environmental savings in comparison with traditional production of this product. Additionally, all valorisation alternatives provide better environmental performance than ash landfilling, which reinforces the valorisation potential of the ashes in construction materials towards a more circular economy.

Chapter 5 assessed the environmental performance of woody biomass fly ash landfarming for liming and fertilisation purposes comparatively to landfilling. For the different valorisation alternatives considered, different scenarios were studied according to the combustion technology (fluidised bed and grate furnaces) and the traditional ameliorants replaced (limestone, quicklime, hydrated lime, potassium chloride, potassium sulphate, potassium nitrate, single superphosphate and triple superphosphate). In the landfarming scenarios, the transport was the stage that contributed most for the impact categories other than those related with eutrophication and toxicity, showing that woody biomass ashes should be applied in soils

as close as possible to the power plants, in order to minimise the environmental impacts related to transportation. For these impact categories, the results also indicated that woody biomass ash landfarming has lower environmental impacts than landfilling for all scenarios studied, regardless of the purpose of ash application and type of traditional ameliorants replaced. However, for the toxicity-related impact categories and freshwater eutrophication, the direct emissions to the soil of ash application were the main responsible for the total impact, due to the emissions to soil of trace element contaminants, such as Cr, As and Zn. In these categories, ash landfarming presented higher impacts than ash landfilling. However, this does not necessarily imply increased risks for the environment, as the potential pollutants leaching depends on their bioavailability in the soil. Therefore, to decrease the bioavailability of potentially toxic elements in the soil it is recommended the stabilisation of woody biomass ash before application by ageing and ash agglomeration into bigger particles to decrease the dissolution rate of those elements. However, both ageing and agglomeration are under development and more research needs to be performed in this area.

Chapter 6 assessed the environmental consequences of implementing the strategy proposed by the Portuguese government to increase the installed capacity of electricity production from forest biomass residues, in contrast to the consequences of losing alternative uses for these residues, in particular, cogeneration of electricity and heat and bioethanol production. The results from the consequential LCA showed that the net environmental benefits would be higher if forest biomass residues would be used for combined electricity and heat production. However, for PM and MEu the impacts of electricity and heat production from forest biomass residues exceed the impacts of the baseline (leaving the forest residues in the forest floor and producing electricity from fossil fuels). The environment performance of this scenario (and also that of the electricity production scenario) could be improved by increasing the conversion efficiency (e.g. by adopting fluidised bed technology instead of grate technology), or by implementing more efficient systems of pollution control at the plants (e.g. for reduction of particulate matter), as well as by optimising fertilisation practices. Another alternative to improve the results in PM and MEu is to displace coal in electricity production and fuel oil in heat production. The conversion of forest biomass residues to bioethanol from biochemical conversion is the least beneficial for all the impact categories analysed. Therefore, it would only be environmentally viable if the bioethanol production technology becomes much more efficient in terms of materials and energy consumption and emissions to the environment.

Overall, the results presented in Chapter 6 allow to better understand a complex problem and to facilitate the decision-making process, which in this case allows to suggest actions to

improve the Portuguese strategy of increasing the installed capacity for electricity production from forest biomass. The recovery of the heat lost during the power production in dedicated power plants appears to be a good alternative to minimize the environmental impacts, implying that the new power plants should be located close to heat consumers. Otherwise, for some impact categories, the environmental impacts of this strategy would be higher than producing electricity from fossil fuels. Finally, it is noteworthy that the application of consequential LCA in the field of bioenergy systems should be fomented in order to provide a useful outcome for helping decision makers in the definition of strategies that effectively lead to a more sustainable development.

7.2 Future perspectives

Based on the findings of this thesis, suggestions for future work are provided. Considering the LCA studies of forest residues production found in literature, it is important to broad the scope to other biomass residual feedstocks besides eucalyptus and maritime pine logging residues, such as residues from the exploitation of other forest species (cork oak, holm oak, stone pine, etc.) and residues from agriculture. Eucalyptus and maritime pine residues other than those produced from logging activities can also have potential for bioenergy production, such as eucalyptus stumps and residues produced during cleaning (herbaceous and brushwood), selection of coppice stems (in eucalyptus), thinning and pruning (both in maritime pine) and should be further studied in the future. The evaluation of more feedstocks can result in more primary data that could be used for comparison and optimisation purposes, in order to find the most attractive feedstock for bioenergy production, from an environmental point of view.

In order to expand the results of this thesis, it would be interesting to assess combustion technologies other than moving grate and fluidised beds, namely stokers and pulverised fuel furnaces and differentiate the moving grates between traveling grates, rotating grates, vibrating grates and the fluidised beds between bubbling and circulating in order to improve power plants competitiveness and environmental performance. In addition to combustion technologies, other conversion systems to produce bioenergy from forest biomass residues could benefit the bioenergy sector. Although future changes in electricity production are uncertain and depend on a number of factors, including technical developments, policy measures and other external aspects, some trends can be assumed, as the promising use of gasification and pyrolysis technologies. This analysis would increase even more the supporting information for the decision-making and would allow the comparison between a wider range of technologies.

Regarding the end-of-life management of woody biomass ash, it is important to further study the influential stages. The hotspot analysis showed that in the case of ash landfarming, the critical aspect that determines its environmental viability is the leaching of trace elements contained in the ash. Therefore, it is recommended to include leaching data of trace elements and other hazardous substances in the LCA studies, since it affects the toxicity-related categories. In addition, environmental assessments should be performed to evaluate different techniques and strategies to minimize the trace elements leaching in order to recover the nutrients maintaining environmental safety.

This work also suggests another future research path where the uncertainty in consequential assessment should be analysed in detail, so that some guidelines can be given to the decision makers in terms of the best application for the forest biomass residues available in the country. The results from a consequential assessment are intrinsically linked to many assumptions, which reduces the confidence in results and may cause concerns about the utilisation of this approach to support decision-making. The robustness of the results can be evaluated through an uncertainty assessment (e.g., by using Monte Carlo simulation techniques), in order to provide confidence to the LCA results.

Moreover, an analysis using other indicators (e.g. economic and social indicators) should be conducted as a complementary tool to LCA. The sustainable use of forest biomass residues will play its part in the fossil to renewable transition. However, sustainability is considered a combination of environmental, economic and social conditions. Therefore, to assess the sustainability of using forest biomass residues, a multidimensional impact assessment in a framework considering economic, environmental and social dimensions to support decision-making and to develop policies is needed.

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References

Ackom, E.K., Mabee, W.E., Saddler, J.N., 2010. Industrial sustainability of competing wood energy options in Canada. Applied Biochemistry and Biotechnology, 162(8). p.2259–2272.

Aden, A., Ruth, M., Ibsen, K., Jechura, J., Neeves, K., Sheehan, J., Wallace, B., 2002. Lignocellulosic biomass to ethanol process design and economics utilizing co-current dilute acid prehydrolysis and enzymatic hydrolysis for corn stover, La Revue de Médecine Interne. Golde, USA. p. 154.

Ahmaruzzaman, M., 2010. A review on the utilization of fly ash. Progress in Energy and Combustion Science, 36(3). p.327–363.

Alcarde, J.C., 2005. Soil acidity correctives: characteristics and technical interpretations. (in Portuguese) Corretivos da acidez dos solos: características e interpretações técnicas., Boletim Técnico No. 6. São Paulo, Brazil. p.24.

Alía, R., Martín, S., 2003. Technical guidelines for genetic conservation and use for Maritime pine (Pinus pinaster). International Plant Genetic Resources Institute. Rome, Italy. p.6.

Allegrini, E., Butera, S., Kosson, D.S., Van Zomeren, A., Van der Sloot, H.A., Astrup, T.F., 2015. Life cycle assessment and residue leaching: The importance of parameter, scenario and leaching data selection. Waste Management, 38(1). p.474–485.

Alves, J.C.L.C.M., 2013. Environmental impact assessment of biomass ash for paving. (in Portuguese) Avaliação do impacte ambiental das cinzas de biomassa na pavimentação. Master Thesis. Department of Environment and Planning. University of Aveiro, Aveiro, Portugal. p.168.

APA, 2019. Atlas of the environment. (in Portuguese) Atlas do ambiente. Agência Portuguesa do Ambiente. URL https://sniamb.apambiente.pt/ (accessed 04.22.2019).

APA, 2014. National transition plan for combustion plants in Chapter III of Directive 2010/75/EU. (in Portuguese) Plano de transição nacional aplicável às instalações de combustão incluídas no capítulo III da Diretiva 2010/75/EU. Agência Portuguesa do Ambiente. Lisbon, Portugal. p.23.

APREN, 2018a. Evolution of the electricity generation in mainland Portugal (2000-2017). (in Portuguese) Evolução da produção elétrica em Portugal continental. Associação de Energias Renováveis. URL https://www.apren.pt/pt/energias-renovaveis/producao (accessed 04.18.2019).

APREN, 2018b. Yearbook 2018. Associação de Energias Renováveis. Lisbon, Portugal. p.18.

Arshad, M.A., Soon, Y.K., Azooz, R.H., Lupwayi, N.Z., Chang, S.X., 2012. Soil and crop response to wood ash and lime application in acidic soils. Agronomy Journal, 104(3). p.715–721.

Assembleia da República, 2014. Law No. 82-D/2014 of December 31, 2014. (in Portuguese) Lei Nº. 82-D/2014 de 31 de dezembro de 2014. Assembleia da República. Portugal. p.320-338.

Augusto, L., Bakker, M.R., Meredieu, C., 2008. Wood ash applications to temperate forest ecosystems - Potential benefits and drawbacks, in: Plant and Soil. p.181–198.

Bang-Andreasen, T., Nielsen, J.T., Voriskova, J., Heise, J., Rønn, R., Kjøller, R., Hansen, H.C.B., Jacobsen, C.S., 2017. Wood ash induced pH changes strongly affect soil bacterial numbers and community composition. Frontiers in Microbiology, 8. p.1–14.

Banks, M.K., Schwab, A.P., Henderson, C., 2006. Leaching and reduction of chromium in soil as affected by soil organic content and plants. Chemosphere, 62(2). p.255–264.

Barbosa, R., Lapa, N., Dias, D., Mendes, B., 2013. Concretes containing biomass ashes: Mechanical, chemical and ecotoxic performances. Construction and Building Materials, 48. p.457–463.

Bastos, D.R.S., 2014. Evaluation of cases of recovery of cellulose waste in building materials. (in Portuguese) Avaliação de casos de valorização de resíduos de celulose em materiais de construção. Master Thesis. Department of civil engineering. University of Aveiro, Aveiro, Portugal. p.115.

Basu, P., 2013. Combustion and gasification in fluidized beds. Journal of Chemical Information and Modeling, 53(9). p.1689–1699.

Beltrán, M.G., Agrela, F., Barbudo, A., Ayuso, J., Ramírez, A., 2014. Mechanical and durability properties of concretes manufactured with biomass bottom ash and recycled coarse aggregates. Construction and Building Materials, 72. p.231–238.

Benavides, R., Roig, S., Osoro, K., 2009. Potential productivity of forested areas based on a biophysical model. A case study of a mountainous region in northern Spain. Annals of Forest Science, 66(1). p.108–108.

BERC, 2009. North country forest energy project. Wood heating in Coos Country, New Hampshire. Biomass Energy Resources Center. Burlington, USA. p.54.

Bernardes, A., Pellegrini, V.O.A., Curtolo, F., Camilo, C.M., Mello, B.L., Johns, M.A., Scott, J.L., Guimaraes, F.E.C., Polikarpov, I., 2019. Carbohydrate binding modules enhance cellulose enzymatic hydrolysis by increasing access of cellulases to the substrate. Carbohydrate Polymers, 211. p.57–68.

Berra, M., Mangialardi, T., Paolini, A.E., 2015. Reuse of woody biomass fly ash in cement-based materials. Construction and Building Materials, 76. p.286–296.

Berton, L.P., 2007. Chemical engineering research trends. Nova Science Publishers, Inc., New York, USA. p.398.

Bhatia, S.C., 2014. Cogeneration, in: Bhatia, S.C. (Ed.), Advanced Renewable Energy SystemsWoodhead Publishing India. p.490–508.

Bjurström, H., Lind, B.B., Lagerkvist, A., 2014. Unburned carbon in combustion residues from solid biofuels. Fuel, 117. p.890–899.

Blankendaal, T., Schuur, P., Voordijk, H., 2014. Reducing the environmental impact of concrete and asphalt: A scenario approach. Journal of Cleaner Production, 66. p.27–36.

Blunden, G., Campbell, S.A., Smith, J.R., Guiry, M.D., Hession, C.C., Griffin, R.L., 1997. Chemical and physical characterization of calcified red algal deposits known as maerl. Journal of Applied Phycology, 9(1). p.11–17.

Borghi, G., Pantini, S., Rigamonti, L., 2018. Life cycle assessment of non-hazardous Construction and Demolition Waste (CDW) management in Lombardy Region (Italy). Journal of Cleaner Production, 184. p.815–825.

Bot, A., Benites, J., 2005. The importance of soil organic matter: key to drought-resistant soil and sustained food and production. Food and Agriculture Organization of the United Nations., Rome, Italy. p.80.

Branco, R., Serafim, L., Xavier, A., 2018. Second generation bioethanol production: on the use of pulp and paper industry wastes as feedstock. Fermentation, 5(1). p.4.

Brandão, M., Levasseur, A., Kirschbaum, M.U.F., Weidema, B.P., Cowie, A.L., Jørgensen, S.V., Hauschild, M.Z., Pennington, D.W., Chomkhamsri, K., 2013. Key issues and options in accounting for carbon sequestration and temporary storage in life cycle assessment and carbon footprinting. International Journal of Life Cycle Assessment, 18(1). p.230–240.

Brännvall, E., Nilsson, M., Sjöblom, R., Skoglund, N., Kumpiene, J., 2014. Effect of residue combinations on plant uptake of nutrients and potentially toxic elements. Journal of Environmental Management, 132(2). p.287–295.

Bridgwater, A. V., 2015. Pyrolysis of biomass, in: van Swaaij, W.P.M., Kersten, S.R.A., Palz, W. (Eds.), Biomass Power for the World.Jenny Stanford Publishing, Singapore. 1st. ed. p.473–513.

Bright, R.M., Strømman, A.H., 2009. Life cycle assessment of second generation bioethanols produced from scandinavian Boreal forest resources a regional analysis for middle Norway. Journal of Industrial Ecology, 13(4). p.514–531.

Brod, E., Haraldsen, T.K., Breland, T.A., 2012. Fertilization effects of organic waste resources and bottom wood ash: Results from a pot experiment. Agricultural and Food Science, 21(4). p.332–347.

Bruijn, H., Duin, R., Huijbregts, M.A.J., 2004. Handbook on life cycle assessment - Operational guide to the ISO standards. Kluwer Academic Publishers, New York, USA. p.692.

Burton, A.H., 2016. Bed agglomeration during biomass fast pyrolysis in a fluidized- bed reactor. Doctoral Thesis. Department of Chemical Engineering. Curtin University. p. 234.

Callesen, I., Ingerslev, M., Raulund-Rasmussen, K., 2007. Dissolution of granulated wood ash examined by in situ incubation: Effects of tree species and soil type. Biomass and Bioenergy, 31(10). p.693–699.

Calvo, A.I., Tarelho, L.A.C., Teixeira, E.R., Alves, C., Nunes, T., Duarte, M., Coz, E., Custodio, D., Castro, A., Artiñano, B., Fraile, R., 2013. Particulate emissions from the co-combustion of forest biomass and sewage sludge in a bubbling fluidised bed reactor. Fuel Processing Technology, 114.

p.58–68.
CAM, 2013. Report - Biomass working group. (in Portuguese) Relatório - Grupo de trabalho da biomassa., Comissão de Agricultura e mar. Lisbon, Portugal. p.116.
Cambero, C., Alexandre, M.H., Sowlati, T., 2015. Life cycle greenhouse gas analysis of bioenergy generation alternatives using forest and wood residues in remote locations: A case study in British Columbia, Canada. Resources, Conservation and Recycling, 105. p.59–72.
Cames, M., Helmers, E., 2013. Critical evaluation of the European diesel car boom - Global comparison, environmental effects and various national strategies. Environmental Sciences Europe, 25(1). p.1–22.
Can, F., Courtois, X., Royer, S., Blanchard, G., Rousseau, S., Duprez, D., 2012. An overview of the production and use of ammonia in NSR + SCR coupled system for NOx reduction from lean exhaust gas. Catalysis Today, 197(1). p.144–154.
CBE, 2007. Supply chains of Mortágua generating plant using forest biomass residual for energy production. Centro da Biomassa para a Energia. Miranda do Corvo, Portugal. p.10.
CBE, 2004. Optimization of forest operations, collection, transport and processing of forest residues. (in Portuguese) Optimização das operações de exploração florestal, recolha, acondicionamento, transporte e transformação primária da biomassa florestal residual. Centro da Biomassa para a Energia. Miranda do Corvo, Portugal. p.136.
Celik, K., Meral, C., Gursel, A.P., Ehta, P.K., Horvath, A., Monteiro, P.J.M., 2014. Mechanical properties, durability and life-cycle assessment of self-consolidating concrete mixtures made with blended portland cements containing fly ash and limestone powder. Cement and Concrete Composites, 56. p.59–72.
CES, 2017. Forest economics and spatial planning. (in Portuguese) Economia da floresta e ordenamento do território. Conselho Económico e Social. Lisbon, Portugal. p.187.
Chang, A.C., Page, A.L., Krage, N.J., 2004. Role of fertilizer and micronutrient applications on arsenic, cadmium and lead accumulation in California cropland soils, Department of Environmental Sciences, University of California. Riverside, USA. p.124.
Chen, C., Habert, G., Bouzidi, Y., Jullien, A., Ventura, A., 2010. LCA allocation procedure used as an incitative method for waste recycling: An application to mineral additions in concrete. Resources, Conservation and Recycling, 54(12). p.1231–1240.
Chen, Y., Parvez, F., Gamble, M., Islam, T., Ahmed, A., Argos, M., Graziano, J.H., Ahsan, H., 2009. Arsenic exposure at low-to-moderate levels and skin lesions, arsenic metabolism, neurological functions and biomarkers for respiratory and cardiovascular diseases: Review of recent findings from the Health Effects of Arsenic Longitudinal Study (HEALS) in. Toxicology and Applied Pharmacology, 239(2). p.184–192.
Cheng, T., Allen, H.E., 2006. Comparison of zinc complexation properties of dissolved natural organic matter from different surface waters. Journal of Environmental Management, 80(3). p.222–229.
Cherubini, F., 2010. GHG balances of bioenergy systems - Overview of key steps in the production chain and methodological concerns. Renewable Energy, 35(7). p.1565–1573.
Chitawo, M.L., Chimphango, A.F.A., Peterson, S., 2018. Modelling sustainability of primary forest residues-based bioenergy system. Biomass and Bioenergy, 108. p.90–100.
Chowdhury, S., Mishra, M., Suganya, O., 2015. The incorporation of wood waste ash as a partial cement replacement material for making structural grade concrete: An overview. Ain Shams Engineering Journal, 6(2). p.429–437.
Ciccarese, L., Bianco, P.M., Mandrone, S., .P, P., Vicini, C., 2014. Assessing the bionergy potential of forest and out of forest in Latium. ISPRA - Istituto Superiore per la Protezione e la Ricerca Ambientale. Rome, Italy. p.64.
Clarholm, M., 1994. Granulated wood ash and a “N-free” fertilizer to a forest soil-effects on P availability. Forest Ecology and Management, 66(1-3). p.127–136.
Coelho, A.M.S.L., 2010. Ash management produced in cogeneration plants operated with biomass. (in Portuguese) Gestão de cinzas produzidas em centrais de cogeração operadas com biomassa. Master Thesis. Department of Environment and Planning. University of Aveiro, Aveiro, Portugal. p.180.
Colangelo, F., Forcina, A., Farina, I., Petrillo, A., 2018. Life Cycle Assessment (LCA) of different kinds of concrete containing waste for sustainable construction. Buildings, 8(5). p.70.
COM, 2017. Renewable energy progress report. Commission to the European Parliament, the Council,

the European Economic and Social Committee and the Committee of the Regions. Brussels, Belgium. p.19.

Consoli, F., Allen, D., Boustead, I., Fava, J., Franklin, W., Jensen, A., De Oude, N., Parrish, R., Perriman, R., Postelthwaite, D., Quay, B., Séguin, J., Vigon, B., 1994. Guidelines for Life-Cycle Assessment: A "Code of Practice," in: The Sesimbra Portugal 31 March 3 April 1993 Meeting of the Society of Environmental Toxicology and Chemistry SETAC. p.55–55.

Cruz, N.C., Rodrigues, S.M., Carvalho, L., Duarte, A.C., Pereira, E., Römkens, P.F.A.M., Tarelho, L.A.C., 2017. Ashes from fluidized bed combustion of forest biomass residues: recycling to soil as a viable management option. Environmental Science and Pollution Research, 24(17). p.14770–14781.

Curran, M.A., 2017. Goal and scope definition in life cycle assessment, 1st. ed. Springer, New York, USA. p.170.

da Costa, T.P., Quinteiro, P., Tarelho, L.A.C., Arroja, L., Dias, A.C., 2019. Environmental assessment of valorisation alternatives for woody biomass ash in construction materials. Resources, Conservation and Recycling, 148. p.67–79.

da Costa, T.P., Quinteiro, P., Tarelho, L.A.C., Arroja, L., Dias, A.C., 2018. Environmental impacts of forest biomass-to-energy conversion technologies: Grate furnace vs. fluidised bed furnace. Journal of Cleaner Production, 171. p.153–162.

Dahl, O., Nurmesniemi, H., Pöykiö, R., Watkins, G., 2010. Heavy metal concentrations in bottom ash and fly ash fractions from a large-sized (246 MW) fluidized bed boiler with respect to their Finnish forest fertilizer limit values. Fuel Processing Technology, 91(11). p.1634–1639.

Dahl, O., Nurmesniemi, H., Pöykiö, R., Watkins, G., 2009. Comparison of the characteristics of bottom ash and fly ash from a medium-size (32 MW) municipal district heating plant incinerating forest residues and peat in a fluidized-bed boiler. Fuel Processing Technology, 90(7). p.871–878.

Dahlquist, E., 2013. Technologies for converting biomass to useful energy combustion, gasification, pyrolysis, torrefaction and fermentation, 1st. ed. CRC Press, Boca Raton, USA. p.520.

Dan Gavriletea, M., 2017. Environmental impacts of sand exploitation. Analysis of sand market. Sustainability, 9(7). p.560–579.

Davis, J., Haglund, C., 1999. Life cycle inventory (LCI) of fertiliser production. Fertiliser products used in Sweden and Western Europe. Doctoral Thesis. Chalmers University of Technology. p.112.

Daystar, J., Reeb, C., Gonzalez, R., Venditti, R., Kelley, S.S., 2015. Environmental life cycle impacts of cellulosic ethanol in the Southern U.S. produced from loblolly pine, eucalyptus, unmanaged hardwoods, forest residues and switchgrass using a thermochemical conversion pathway. Fuel Processing Technology, 138. p.164–174.

de Ruijter, F.J., Huijsmans, J.F.M., Rutgers, B., 2010. Ammonia volatilization from crop residues and frozen green manure crops. Atmospheric Environment, 44(28). p.3362–3368.

de Vries, J.W., Groenestein, C.M., de Boer, I.J.M., 2012. Environmental consequences of processing manure to produce mineral fertilizer and bio-energy. Journal of Environmental Management, 102. p.173–183.

Demertzi, M., 2011. Evaluation of the cork sector's environmental performance through Life Cycle Assessment. Doctoral Thesis. University of Aveiro. Department of Environment and Planning. p.269.

Demertzi, M., Dias, A.C., Matos, A., Arroja, L.M., 2015. Evaluation of different end-of-life management alternatives for used natural cork stoppers through life cycle assessment. Waste Management, 46. p.668–680.

Demeyer, A., Voundi Nkana, J.C., Verloo, M.G., 2001. Characteristics of wood ash and influence on soil properties and nutrient uptake: An overview. Bioresource Technology, 77(3). p.287–295.

Demirbas, A., 2003. Relationships between lignin contents and fixed carbon contents of biomass samples. Energy Conversion and Management, 44(9). p.1481–1486.

Demirbaş, A., 2003. Hydrocarbons from pyrolysis and hydrolysis processes of biomass. Energy Sources, 25(1). p.67–75.

Deviatkin, I., Havukainen, J., Horttanainen, M., 2017. Comparative life cycle assessment of thermal residue recycling on a regional scale: A case study of South-East Finland. Journal of Cleaner Production, 149. p.275–289.

DGAE, 2017. Forestry-based industries: Synopsis 2017. (in Portuguese) Indústrias de base florestal:

Sinopse 2017. Direção Geral das Atividades Económicas. Lisbon, Portugal. p.27.
DGEG, 2019. Planning and statistics. (in Portuguese) Planeamento e estatística. URL http://www.dgeg.gov.pt/ (accessed 03.28.2019).
DGEG, 2018. Renewables: Fast statistics No.168. (in Portuguese) Renováveis: Estatísticas rápidas Nº.168. Direção Geral de Energia e Geologia. Lisbon, Portugal. p.21.
DGEG, 2016. Renewables: Fast statistics No.143. (in Portuguese) Renováveis: Estatísticas rápidas Nº.143. Direção Geral de Energia e Geologia. Lisbon, Portugal. p.21.
Di Gianfilippo, M., Costa, G., Pantini, S., Allegrini, E., Lombardi, F., Astrup, T.F., 2016. LCA of management strategies for RDF incineration and gasification bottom ash based on experimental leaching data. Waste Management, 47. p.285–298.
Dias, A.C., 2014. Life cycle assessment of fuel chip production from eucalypt forest residues. Fuel, 19. p.705–717.
Dias, A.C., Arroja, L., 2012. Environmental impacts of eucalypt and maritime pine wood production in Portugal. Journal of Cleaner Production, 37. p.368–376.
Dias, A.C., Arroja, L., Capela, I., 2007. Carbon dioxide emissions from forest operations in Portuguese eucalypt and maritime pine stands. Scandinavian Journal of Forest Research, 22(5). p.422–432.
Dias, D.A.C.M., 2011. Study of biomass ash valorization in the production of materials for the construction of artificial reefs (in Portuguese) Estudo da valorização de cinzas de biomassa na produção de materiais para a construção de recifes artificiais. Master Thesis. Faculty of Science and Technology. New University of Lisbon. Lisbon, Portugal. p.130.
Dias, J., Azevedo, J.L.T., 2004. Evaluation of biomass residuals in Portugal mainland, in: Afgan, N.H., Carvalho, M.G. (Eds.), Conference on New and Renewable Energy Technologies for Sustainable Development. A. A. Balkema Publishers, Ponta Delgada, Portugal. p.215–228.
Dias, V.D.M., 2012. Application of solid waste in road pavements. (in Portuguese) Aplicação de resíduos sólidos em pavimentos rodoviários. Master Thesis. University of Aveiro. Department of Civil Engineering.
Dincer, I., Rosen, M.A., 1998. A worldwide perspective on energy, environment and sustainable development. International Journal of Energy Research, 22(15). p.1305–1321.
Djomo, S.N., Witters, N., Van Dael, M., Gabrielle, B., Ceulemans, R., 2015. Impact of feedstock, land use change and soil organic carbon on energy and greenhouse gas performance of biomass cogeneration technologies. Applied Energy, 154. p.122–130.
DNFF, 2010. Forest energy crops: First approach to the survey of the current situation. (in Portuguese) Culturas energéticas florestais: Primeira abordagem do levantamento da situação actual. Direção Nacional das Fileiras Florestais. Lisbon, Portugal. p.30.
Doka, G., 2003a. Life cycle inventories of waste treatment services. Part I: General introduction waste material compositions municipal waste collection. Ecoinvent report No. 13. Swiss Centre for Life Cycle Inventories. Dubendorf, Switzerland. p.111.
Doka, G., 2003b. Life cycle inventories of waste treatment services. Part III: Landfills - underground deposits - landfarming. Ecoinvent report No. 13. Swiss Centre for Life Cycle Inventories. Dubendorf, Switzerland. p.137.
Dung, T., Vassilieva, E., Swennen, R., Cappuyns, V., 2018. Release of trace elements from bottom ash from hazardous waste incinerators. Recycling, 3(3). p.36.
Dvořák, K., Kulísek, K., Gazdič, D., 2017. The FBC ash as a hydraulic ingredient of hydraulic lime. Procedia Engineering, 172. p.264–269.
E2p, 2017. Database of renewable energy sources. (in Portuguese) Base de dados de fontes renováveis de energia. Energias endógenas em Portugal. URL http://e2p.inegi.up.pt/?Lang=PT (accessed 04.21.2019).
EC, 2019. Estatísticas das energias renováveis. (in Portuguese) Renewable energy statistics. Eurostat, Statistics Explained. European Comission. URL https://ec.europa.eu/eurostat/statisticsexplained/ (accessed 01.09.2019).
EC, 2017. Sustainable and optimal use of biomass for energy in the EU beyond 2020. European Commission. Brussels, Belgium. p.404.
EC/JRC/IES, 2012. Characterisation factors of the ILCD recommended life cycle impact assessment methods. Database and supporting information. European Commission, Joint Research Centre, Institute for Environment and Sustainability. Luxembourg, Luxembourg. p.31.

EC/JRC/IES, 2010a. International Reference Life Cycle Data System (ILCD) handbook: Framework and requirements for life cycle impact assessment models and indicators. European Commission, Joint Research Centre, Institute for Environment and Sustainability. Luxembourg, Luxembourg. p.116.

EC/JRC/IES, 2010b. International Reference Life Cycle Data System (ILCD) handbook. General guide for life cycle assessment. Joint Research Centre - Institute for Environment and Sustainability. Luxembourg, Luxembourg. p.417.

ECN/TNO, 2019. Phyllis2 - Database for biomass and waste. Energy Research Centre of the Netherlands and Netherlands Organisation for Applied Scientific Research. URL https://www.ecn.nl/phyllis2/ (accessed 08.15.2019).

Ecoinvent, 2018. History – ecoinvent. URL https://www.ecoinvent.org/about/history/history.html (accessed 01.28.2019).

Ecoinvent, 2017. Ecoinvent Database v. 3.4. Swiss Centre for Life Cycle Inventories. Dübendorf, Switzerland.

Ecolan, 2012. Fertilizers. Edition 2011/2012. URL http://ecolan.com.ru/standard-samples-foreign/4/4-5.pdf (accessed 06.05.2019).

EEA, 2016. EMEP/EEA air pollutant emission inventory guidebook 2016: Technical guidance to prepare national emission inventories. European Environment Agency. Copenhagen, Denmark. p.124.

EIA, 2016. International energy outlook 2016. U.S. Energy Information Administration. Washington, USA. p.290.

EIA, 2007. Technical guidelines: Voluntary reporting of greenhouse gases, U.S. Energy Information Administration. Washington, USA.

Eija, A., Erkkila, A., Oravainen, H., Rubik, A., Englisch, M., Corbella, L., Floc'h-Laizet, C., Pennequin, J., Puente, F., 2008. Efficient and environmentally friendly biomass heating: firewood production and use in fireplaces and stoves. Technical Research Centre of Finland. Espoo, Finland. p.74.

Ekvall, T., 2012. Life cycle assessment, in: Getting to Green: A Sourcebook of Pollution Management Policy Tools for Growth and Competitiveness.The World Bank Group, Washington, USA. p.1–295.

Ekvall, T., Azapagic, A., Finnveden, G., Rydberg, T., Weidema, B.P., Zamagni, A., 2016. Attributional and consequential LCA in the ILCD handbook. International Journal of Life Cycle Assessment, 21(3). p.293–296.

Ekvall, T., Weidema, B.P., 2004. System boundaries and input data in consequential life cycle inventory analysis. International Journal of Life Cycle Assessment, 9(3). p.161–171.

Elser, J.J., Bracken, M.E.S., Cleland, E.E., Gruner, D.S., Harpole, W.S., Hillebrand, H., Ngai, J.T., Seabloom, E.W., Shurin, J.B., Smith, J.E., 2007. Global analysis of nitrogen and phosphorus limitation of primary producers in freshwater, marine and terrestrial ecosystems. Ecology Letters, 10(12). p.1135–1142.

Enes, P., Valle, F.D., Martínez, B.M., 2007. Promoting the use of forest biomass for energy purposes in South-western Europe (2004-2007). (in Portuguese) Promoção do uso da biomassa florestal para fins energéticos no sudoeste da Europa (2004-2007). Enersilva. Santiago de Compostela, Spain. p.44.

EPA, 2010. Methane and nitrous oxide emissions from natural sources. Environmental Protection Agency. Washington, USA. p.194.

EPA, 2007. Biomass combined heat and power catalog of technologies. Environmental Protection Agency. Washington, USA. p.122.

EPA, 2001. Background document report on revisions: Wood residue combustion in boilers. Environmental Protection Agency. Washington, USA. p.37.

EPA, 1998. Environmental labeling issues, policies and practices worldwide. Environmental Protection Agency. Washington, USA. p. 322.

EPA, 1974. Resource and environmental profile analysis of nine beverage container alternatives. Environmental Protection Agency. Washington, USA. p.185.

Erich, M.S., 1991. Agronomic effectiveness of wood ash as a source of phosphorus and potassium. Journal of Environmental Quality, 20(3). p.576–581.

Eriksson, L.N., Gustavsson, L., 2008. Biofuels from stumps and small roundwood-Costs and CO2

benefits. Biomass and Bioenergy, 32(10). p.897–902.
Espinosa, D., Sale, P., Tang, C., 2017. Effect of soil phosphorus availability and residue quality on phosphorus transfer from crop residues to the following wheat. Plant and Soil, 416(1-2). p.361–375.
Esteves, T.C., Rajamma, R., Soares, D., Silva, A.S., Ferreira, V.M., Labrincha, J.A., 2012. Use of biomass fly ash for mitigation of alkali-silica reaction of cement mortars. Construction and Building Materials, 26(1). p.687–693.
EU, 2018. Directive 2018/2001 of 11 December 2018. European Union. Brussels, Belgium. p.124.
EU, 2010. Making sustainable consumption and production a reality: a guide for businessand policy makers to life cycle thinking and assessment. European Union. Brussels, Belgium. p.32.
EuLA, 2017. 2016-2017 Activity Report. European Lime Association. Brussels, Belgium. p. 64.
Falano, T., Jeswani, H.K., Azapagic, A., 2014. Assessing the environmental sustainability of ethanol from integrated biorefineries. Biotechnology Journal, 9(6). p.753–765.
FAO, 2017. Resource statistics - Fertilizers. URL https://knoema.com/FAORSF2015/fao-resource-statistics-fertilizers?country=1001330-portugal (accessed 05.03.2018).
FAO, 2002. Guidelines on fire management in temperate and boreal forests. Food and Agriculture Organization. Rome, Italy. p.257.
Ferreira, S., Monteiro, E., Brito, P., Vilarinho, C., 2017. Biomass resources in Portugal: Current status and prospects. Renewable and Sustainable Energy Reviews, 78. p.1221–1235.
Ferreira, S., Moreira, N.A., Monteiro, E., 2009. Bioenergy overview for Portugal. Biomass and Bioenergy, 33(11). p.1567–1576.
Finnveden, G., Hauschild, M., Ekvall, T., Guinée, J., Heijungs, R., Hellweg, S., Koehler, A., Pennington, D., Suh, S., 2009. Recent developments in life cycle assessment. Journal of Environmental Management, 91(1). p.1–21.
Fiorese, G., Catenacci, M., Bosetti, V., Verdolini, E., 2014. The power of biomass: Experts disclose the potential for success of bioenergy technologies. Energy Policy, 65. p.94–114.
Forest Europe, 2015. State of Europe's forests 2015. Status and trends in sustainable forests management in Europe. Ministerial Conference on the Protection of Forests in Europe. Oslo, Norway. p.344.
Forsberg, G., 2000. Biomass energy transport. Biomass and Bioenergy, 19(1). p.17–30.
Franke, M., 1984. Environmental impact of drink packaging. System for determining the environmental impact of complex processes using the example of disposable and reusable drinks containers. (in Dutch) Umweltauswirkungen dutch Getrinkeverpackungen. Systematik zur Ermittlu. E.F.-Verlag for Energie- und Umwelttechnik GmbH.., Berlim, Germany. p. 241.
Freire, M., Lopes, H., Tarelho, L.A.C., 2015. Critical aspects of biomass ashes utilization in soils: Composition, leachability, PAH and PCDD/F. Waste Management, 46. p.304–315.
Frey, H.H., Peters, B., Hunsinger, H., Vehlow, J., 2003. Characterization of municipal solid waste combustion in a grate furnace. Waste Management, 23(8). p.689–701.
Fu, G.Z., Chan, A.W., Minns, D.E., 2003. Life cycle assessment of bio-ethanol derived from cellulose. International Journal of Life Cycle Assessment, 8(3). p.137–141.
Galloway, J.N., Dentener, F.J., Capone, D.G., Boyer, E.W., Howarth, R.W., Seitzinger, S.P., Asner, G.P., Cleveland, C.C., Green, P.A., Holland, E.A., Karl, D.M., Michaels, A.F., Porter, J.H., Townsend, A.R., Vo, C.J., 2004. Nitrogen cycles: past, present and future. Biogeochemistry, 70. p.153–226.
Garcia, M.D.L., Sousa-Coutinho, J., 2013. Strength and durability of cement with forest waste bottom ash. Construction and Building Materials, 41. p.897–910.
Garcia, R., Freire, F., 2016. Marginal life-cycle greenhouse gas emissions of electricity generation in Portugal and implications for electric vehicles. Resources, 5(4). p.41.
Gentil, L.V., Vale, A.T., 2015. Energy balance and efficiency in wood sawdust briquettes production. Floresta, 45(2). p.281–288.
Gerbrandt, K., Chu, P.L., Simmonds, A., Mullins, K.A., MacLean, H.L., Griffin, W.M., Saville, B.A., 2016. Life cycle assessment of lignocellulosic ethanol: A review of key factors and methods affecting calculated GHG emissions and energy use. Current Opinion in Biotechnology, 38. p.63–70.
Giordano, D., Tito, A.C., Tito, V., Ottonello, P., Pescarolo, S., 2015. Biomass conversion to advanced biofuels: a short review on second generation processes to produce ethanol, in: van Swaaij, W.,

Kersten, S., Palz, W. (Eds.), Biomass Power for the World.CRC Press, Boca Raton, USA. 1st ed. p.599–630.

Girón, R.P., Ruiz, B., Fuente, E., Gil, R.R., Suárez-Ruiz, I., 2013. Properties of fly ash from forest biomass combustion. Fuel, 114. p.71–77.

Goedkoop, M., Spriensma, R., 2000. The Eco-indicator 99, a damage oriented method for life cycle impact assessment. PRé - Product Ecology Consultants. Amersfoort, Netherlands. p.87.

Goedkoop, M.J., Heijungs, R., Huijbregts, M.. J., Schryver, A., Struijs, J., van Zelm, R., 2013. ReCiPE 2008: A life cycle impact assessment method which comprises harmonised category indicators at the midpoint and the endpoint level. First edition (version 1.08). Ministerie van Volkshuisvesting. Amersfoort, Netherlands. p.134.

Gomes, J.F.P., 2006. Forest fires in Portugal: how they happen and why they happen. International Journal of Environmental Studies, 63(2). p.109–119.

Gómez-Rey, M.X., Madeira, M., Coutinho, J., 2012. Wood ash effects on nutrient dynamics and soil properties under Mediterranean climate. Annals of Forest Science, 69(5). p.569–579.

Gonçalves, J.L.M., Moro, L., 1995. Uso da cinza de biomassa florestal como fonte de nutrientes em povoamentos puros de Eucalyptus grandis. Ipef, 48. p.28–37.

González-García, S., Bacenetti, J., 2019. Exploring the production of bio-energy from wood biomass. Italian case study. Science of the Total Environment, 647. p.158–168.

González-García, S., Dias, A.C., Clermidy, S., Benoist, A., Bellon Maurel, V., Gasol, C.M., Gabarrell, X., Arroja, L., 2014. Comparative environmental and energy profiles of potential bioenergy production chains in Southern Europe. Journal of Cleaner Production, 76. p.42–54.

González-García, S., Gasol, C.M., Moreira, M.T., Gabarrell, X., Pons, J.R.I., Feijoo, G., 2011. Environmental assessment of black locust (Robinia pseudoacacia L.)-based ethanol as potential transport fuel. International Journal of Life Cycle Assessment, 16(5). p.465–477.

González-García, S., Luo, L., Moreira, M.T., Feijoo, G., Huppes, G., 2012. Life cycle assessment of hemp hurds use in second generation ethanol production. Biomass and Bioenergy, 36. p.268–279.

Górecki, R., Piętak, A., Meus, M., Kozłowski, M., 2015. Biomass energy potential – Green energy for the University of Warmia and Mazury in Olsztyn. Journal of KONES. Powertrain and Transport, 20(4). p.99–106.

Goudriaan, F., van de Beld, B., Boerefijn, F., Bos, G., Naber, J., van der Wal, S., Zeevalkink, J., 2008. Progress in thermochemical biomass conversion. John Wiley & Sons, Hoboken, USA. p.1744.

Goulding, K.W.T., 2016. Soil acidification and the importance of liming agricultural soils with particular reference to the United Kingdom. Soil Use and Management, 32(3). p.390–399.

Grassi, G., 2015. Large European bioenergy network: A strategy for commercial exploitation of biomass at the regional level, in: van Swaaij, W.P.M., Kersten, S.R.A., Palz, W. (Eds.), Biomass Power for the World.Jenny Stanford Publishing, Singapore. 1st. ed. p.191–226.

Guinée, J., 2002. Handbook on life cycle assessment. Operational guide to the ISO standards. Dordrecht, Netherlands.

Guinée, J.B., Heijungs, R., Huppes, G., Zamagni, A., Masoni, P., Buonamici, R., Ekvall, T., Rydberg, T., 2011. Life cycle assessment: Past, present and future. Environmental Science and Technology, 45(1). p.90–96.

Gunaseelan, A., Ramalingam, K.M., 2016. Experimental study on performance of high volume fly ash concrete. International Journal for Innovative Research in Science & Technology, 2(12). p.187–195.

Habersatter, K., Widmer, F., 1991. Eco-balance of packaging materials as of 1990. (in German) Oekobilanz von Packstoffen Stand 1990. Dokumentationsdienst BUWAL., Bern, Switzerland. p.261.

Habert, G., Ignjatovic, I.S., Dragas, J., Tosic, N., 2016. Life cycle analysis of fly ash concrete with recycled concrete aggregate, in: Sustainable Built Environment (SBE) Regional Conference. Zurich, Switzerland. p.7.

Hakkila, P., 2004. Developing technology for large-scale production of forest chips. Technical Research Centre of Finland. Espoo, Finland. p.58.

Hamelin, L., Jørgensen, U., Petersen, B.M., Olesen, J.E., Wenzel, H., 2012. Modelling the carbon and nitrogen balances of direct land use changes from energy crops in Denmark: A consequential life cycle inventory. GCB Bioenergy, 4(6). p.889–907.

Hammar, T., Stendahl, J., Sundberg, C., Holmström, H., Hansson, P.A., 2019. Climate impact and energy efficiency of woody bioenergy systems from a landscape perspective. Biomass and Bioenergy, 120. p.189–199.

Hasan, B., Ahsant, A., 2015. A performance comparison between the grate-fired and fluidized bed combustion of municipal solid waste and RDF as a fuel for CHP plant applications. KTH School of Industrial Engineering and Management. Stockholm, Sweden. p.61.

Haseli, Y., van Oijen, J.A., de Goey, L.P.H., 2011. A detailed one-dimensional model of combustion of a woody biomass particle. Bioresource Technology, 102(20). p.9772–9782.

Hauschild, M.Z., Rosenbaum, R.K., Olsen, S.I., 2017. Life cycle assessment: theory and practice. Springer, New York, USA. p.1216.

Heckman, J.R., 1998. Agricultural liming materials., Rutgers Cooperative Research & Extension. The State University of New Jersey. New Brunswick, USA. p.1-4.

Hedegaard, K., Thyø, K.A., Wenzel, H., 2008. Life cycle assessment of an advanced bioethanol technology in the perspective of constrained biomass availability. Environmental Science and Technology, 42(21). p.7992–7999.

Hedman, B.A., 2000. The market and technical potential for combined heat and power in the commercial/institutional sector. Onsite Sycom Energy Corporation. Washington, USA. p.91.

Heijungs, R., 1992. Environmental life cycle assessment of products. Guide - October 1992. Centre of Environemental Science. Leiden, Netherlands. p.97.

Heleno, L., Tarelho, L.A.C., Silva, D.F.R., 2012. Quantitative and qualitative comparison of bottom and fly ash generated in a fluidized bed. (in Portuguese) Comparação quantitativa e qualitativa das cinzas de fundo e cinzas volantes geradas por uma caldeira de combustão de leito fluidizado., in: 1st International Congress on Water, Waste and Energy Management.Salamanca, Spain. p.2012.

Herbert, A.S., Azzaro-Pantel, C., Le Boulch, D., 2016. A typology for world electricity mix: Application for inventories in Consequential LCA (CLCA). Sustainable Production and Consumption, 8. p.93–107.

Hernández, U.F., Jaeger, D., Samperio, J.I., 2017. Bioenergy potential and utilization costs for the supply of forest woody biomass for energetic use at a regional scale in Mexico. Energies, 10(8). p.631–645.

Hetsch, S., 2008. Potential sustainable wood supply in Europe. The United Nations Economic Commission for Europe/Food and Agriculture Organization. Geneva, Switzerland. p.44.

Hoogwijk, M., Faaij, A., van den Broek, R., Berndes, G., Gielen, D., Turkenburg, W., 2003. Exploration of the ranges of the global potential of biomass for energy. Biomass and Bioenergy, 25(2). p.119–133.

Horvat, I., Dović, D., 2018. Combustion of agricultural biomass - issues and solutions. Transactions of FAMENA, 42(1). p.75–86.

Hospido, A., Carballa, M., Moreira, M., Omil, F., Lema, J.M., Feijoo, G., 2010. Environmental assessment of anaerobically digested sludge reuse in agriculture: Potential impacts of emerging micropollutants. Water Research, 44(10). p.3225–3233.

Hossain, M.U., Poon, C.S., Lo, I.M.C., Cheng, J.C.P., 2017. Comparative LCA on using waste materials in the cement industry: A Hong Kong case study. Resources, Conservation and Recycling, 120. p.199–208.

Huang, T.Y., Chiueh, P.T., Lo, S.L., 2017. Life-cycle environmental and cost impacts of reusing fly ash. Resources, Conservation and Recycling, 123. p.255–260.

Huang, T.Y., Chuieh, P.T., 2015. Life cycle assessment of reusing fly ash from municipal solid waste incineration. Procedia Engineering, 118. p.984–991.

Huber, F., Laner, D., Fellner, J., 2017. Comparative life cycle assessment of MSWI fly ash treatment and disposal. Waste Management, 73. p.392–403.

Hunt, R.G., Sellers, J.D., Franklin, W.E., 1992. Resource and environmental profile analysis: A life cycle environmental assessment for products and procedures. Environmental Impact Assessment Review, 12(3). p.245–269.

Huotari, N., Tillman-Sutela, E., Moilanen, M., Laiho, R., 2015. Recycling of ash - For the good of the environment? Forest Ecology and Management, 348. p.226–240.

Hurisso, B., Abuelyazeed, O., Elshokary, S., Ismail, M., Faraga, S., 2018. Conversion of biomass and waste to value-add products: challenges and opportunities. International Journal of Waste

Resources, 8(4). p.1–13.
Ibenholt, K., 2002. Materials flow analysis and economic modeling, in: Ayres, R.U., Ayres, L.. (Eds.), A Handbook of Industrial EcologyEdward Elgar, Cheltenham, United Kingdom. p.704.
ICNF, 2019. Portugal Market Report 2019. Lisbon, Portugal. p.21.
ICNF, 2013. 6th National forest inventory. Areas of land use and forest species of Portugal. (in Portuguese) 6º Inventário florestal nacional. Áreas dos usos do solo e das espécies florestais de Portugal. Instituto da Conservação da Natureza e das florestas. Lisbon, Portugal. p.34.
IEA, 2019. Global energy and CO_2 status report: Emissions. International Energy Agency. Paris, France. p.29.
IEA, 2007. Energy technology essentials: Biomass for power generation and CHP. International Energy Agency. Paris, France. p.4.
IMF, 2014. Study of the heat energy potential of each biomass/agricultural and vegetable residue. (in Portuguese) Estudo do potencial energético de calor de cada biomassa/resíduo agrícola e vegetal. Ibero Massa Florestal. Lisbon, Portugal. p.54.
Ingerslev, M., Skov, S., Sevel, L., Pedersen, L.B., 2011. Element budgets of forest biomass combustion and ash fertilisation: a Danish case-study. Biomass and Bioenergy, 35(7). p.2697–2704.
Insam, H., Knapp, B., 2011. Recycling of biomass ashes., 8th ed. Springer, New York, USA. p.164.
IPCC, 2017. Scoping of the IPCC 9th assessment report. Background, cross cutting issues and the AR6 synthesis report. Intergovernmental Panel on Climate Change. Geneva, Switzerland. p. 58.
IPCC, 2013. 2013 Supplement to the 2006 IPCC guidelines for national greenhouse gas inventories: Wetlands. Intergovernmental Panel on Climate Change. Geneva, Switzerland. p.354.
IPCC, 2006. Guidelines for national greenhouse gas inventories. Intergovernmental Panel on Climate Change. Geneva, Switzerland.
IPQ, 1990. Fertilizer materials. (in Portuguese) Matérias fertilizantes. Instituto Português da Qualidade. Lisbon, Portugal. p.10.
ISO, 2006a. ISO 14040. Environmental management – Life cycle assessment – Principles and framework. International Organization for Standardization. Geneva, Switzerland. p.20.
ISO, 2006b. ISO 14044. Environmental management - Life cycle assessment – Requirements and guidelines. International Organization for Standardization. Geneva, Switzerland. p.54.
ISO, 2000a. ISO 14042. Environmental managment - Life cycle assessment - Life cycle impact assessment. International Organization for Standardization. Geneva, Switzerland. p.8.
ISO, 2000b. ISO 14043. Environmental management — Life cycle assessment — Life cycle interpretation. International Organization for Standardization. Geneva, Switzerland. p.8.
ISO, 1998. ISO 14041. Environmental management — Life cycle assessment — Goal and scope definition and inventory analysis. International Organization for Standardization. Geneva, Switzerland. p.11.
Iyer, R.S., Scott, J.A., 2001. Power station fly ash - A review of value-added utilization outside of the construction industry. Resources, Conservation and Recycling, 31(3). p.217–228.
James, A.K., Thring, R.W., Helle, S., Ghuman, H.S., 2012. Ash management review-applications of biomass bottom ash. Energies, 5(10). p.3856–3873.
Johnson, P.S., Shifley, S.R., Rogers, R., 2009. The ecology and silviculture of oaks., 2nd. ed. CAB International, Oxfordshire, United Kingdom. p.580.
Jolliet, O., Margni, M., Charles, R., Humbert, S., Payet, J., Rebitzer, G., 2003. IMPACT 2002+: A new life cycle impact assessment methodology. International Journal of Life Cycle Assessment, 8(6). p.324–330.
Jones, J.M., Lea-Langton, A.R., Ma, L., Pourkashanian, M., Williams, A., 2014. Pollutants generated by the ccombustion of solid biomass fuels, 1st. ed. Springer, New York, USA. p.104.
Jonker, J.G.G., Junginger, H.M., Verstegen, J.A., Lin, T., Rodriguez, L.F., Ting, K.C., Faaij, A.P.C., van der Hilst, F., 2016. Supply chain optimization of sugarcane first generation and eucalyptus second generation ethanol production in Brazil. Applied Energy, 173. p.494–510.
Jossart, J.M., 2015. Report on conversion efficiency of biomass: Biomass availability and sustainability information system. European Commission. Brussels, Belgium. p.21.
Justic, D., Rabalais, N., Turner, R.E., Dıaz, R.J., 2009. Global change and eutrophication of coastal waters. ICES Journal of Marine Science. p.1528–1537.
Kalmykova, Y., Palme, U., Karlfeldt Fedje, K., Yu, S., 2015. Total material requirement assessment of

phosphorus sources from phosphate ore and urban sinks: Sewage sludge and MSW incineration fly ash. International Journal of Environmental Research, 9(2). p.561–566.

Kaltschmitt, M., Thrän, D., Smith, K.R., 2001. Renewable energy from biomass, in: Meyers, R. (Ed.), Encyclopedia of Physical Science and Technology. Academic Press, London, United Kingdom. p.203-228. 3rd. ed. p.203–228.

Kaminski, J., Moterle, D.F., Rheinheimer, D. dos S., Gatiboni, L.C., Brunetto, G., 2010. Potassium availability in a hapludalf soil under long term fertilization. Revista Brasileira de Ciência do Solo, 34(3). p.49–57.

Karaj, S., Rehl, T., Leis, H., Müller, J., 2010. Analysis of biomass residues potential for electrical energy generation in Albania. Renewable and Sustainable Energy Reviews, 14(1). p.493–499.

Karlsson, H., Börjesson, P., Hansson, P.A., Ahlgren, S., 2014. Ethanol production in biorefineries using lignocellulosic feedstock - GHG performance, energy balance and implications of life cycle calculation methodology. Journal of Cleaner Production, 83. p.420–427.

Kasser, U., Pöll, M., 1998. Energy of building materials: data on construction materials, building chemicals, processing and transport processes with explanations and recommendations for the construction industry. (in German) Graue energie von baustoffen: Daten zu baustoffen, bauche, 2nd. ed. Zürich, Switzerland. p. 345.

Kellenberger, D., Althaus, H.-J., Künninger, T., Lehmann, M., Jungbluth, N., Thalmann, P., 2007. Life cycle inventories of building products. Swiss Centre for Life Cycle Inventories. Dübendorf, Switzerland. p.457.

Khan, A.A., de Jong, W., Jansens, P.J., Spliethoff, H., 2009. Biomass combustion in fluidized bed boilers: Potential problems and remedies. Fuel Processing Technology, 90(1). p.21–50.

Kimming, M., Sundberg, C., Nordberg, A., Hansson, P.A., 2015. Vertical integration of local fuel producers into rural district heating systems - Climate impact and production costs. Energy Policy, 78. p.51–61.

Kizha, A.R., Han, H.S., 2017. Moisture content in forest residues: an insight on sampling methods and procedures. Current Forestry Reports, 3(3). p.202–212.

Kongshaugh, G., 1998. Energy consumption and greenhouse gas emissions in fertilizer production., in: IFA Technical Conference.Marrakech, Morocco. p.6. p.1–9.

Kratz, S., Schick, J., Schnug, E., 2016. Trace elements in rock phosphates and P containing mineral and organo-mineral fertilizers sold in Germany. Science of the Total Environment, 542. p.1013–1019.

Kumar, S., Singh, S.P., Mishra, I.M., Adhikari, D.K., 2009. Recent advances in production of bioethanol from lignocellulosic biomass. Chemical Engineering and Technology, 32(4). p.517–526.

Laganière, J., Paré, D., Thiffault, E., Bernier, P.Y., 2017. Range and uncertainties in estimating delays in greenhouse gas mitigation potential of forest bioenergy sourced from Canadian forests. GCB Bioenergy, 9(2). p.358–369.

Lago, C., Herrera, I., Caldés, N., Lechón, Y., 2018. Nexus bioenergy – Bioeconomy., in: Lago, C., Caldés, N., Lechón, Y. (Eds.), The Role of Bioenergy in the BioeconomyAcademic Press, Cambridge, USA. 1st. ed. p.3–24.

Lal, R., 2008. Crop residues as soil amendments and feedstock for bioethanol production. Waste Management, 28(4). p.747–758.

Lanzerstorfer, C., 2015. Chemical composition and physical properties of filter fly ashes from eight grate-fired biomass combustion plants. Journal of Environmental Sciences., 30. p.191–197.

Laschi, A., Marchi, E., González-García, S., 2016. Environmental performance of wood pellets' production through life cycle analysis. Energy, 103. p.469–480.

Leckner, B., 2015. Process aspects in combustion and gasification Waste-to-Energy (WtE) units. Waste Management, 37. p.13–25.

Lee, S., Shah, Y.T., 2013. Biofuels and bioenergy: processes and technologies, 1st ed. CRC Press, Boca Raton, USA. p.341.

Lessard, J.M., Omran, A., Tagnit-Hamou, A., Gagne, R., 2017. Feasibility of using biomass fly and bottom ashes in dry-cast concrete production. Construction and Building Materials, 132. p.565–577.

Liang, S., Gu, H., Bergman, R.D., 2017. Life cycle assessment of cellulosic ethanol and biomethane production from forest residues. BioResources, 12(4). p.7873–7883.

Liao, Y. lin, Zheng, S. xian, Nie, J., Xie, J., Lu, Y. hong, Qin, X. bo, 2013. Long-term effect of fertilizer

and rice straw on mineral composition and potassium adsorption in a reddish paddy soil. Journal of Integrative Agriculture, 12(4). p.694–710.

Lickacz, J., 2002. Wood ash: an alternative liming material for agricultural soils. Alberta Agriculture, Food and Rural Development. Alberta, USA. p.1-6.

Lijó, L., González-García, S., Bacenetti, J., Fiala, M., Feijoo, G., Moreira, M.T., 2014. Assuring the sustainable production of biogas from anaerobic mono-digestion. Journal of Cleaner Production, 72. p.23–34.

Lin, D.F., Lin, K.L., Hung, M.J., Luo, H.L., 2007. Sludge ash/hydrated lime on the geotechnical properties of soft soil. Journal of Hazardous Materials, 145(1-2). p.58–64.

Lindholm, E., Stendahl, J., Berg, S., Hansson, P., 2011. Greenhouse gas balance of harvesting stumps and logging residues for energy in Sweden. Scandinavian Journal of Forest Research, 26(6). p.586–594.

Liszka, M., Malik, T., Budnik, M., Ziebik, A., 2013. Comparison of IGCC (integrated gasification combined cycle) and CFB (circulating fluidized bed) cogeneration plants equipped with CO2 removal. Energy, 58. p.86–96.

Llorente, M.J.F., Cuadrado, R.E., Laplaza, J.M.M., García, J.E.C., 2006. Combustion in bubbling fluidised bed with bed material of limestone to reduce the biomass ash agglomeration and sintering. Fuel, 85(14). p.2081–2092.

Locher, F.W., Kropp, J., 2001. Cement and concrete, in: Ullmann's Encyclopedia of Industrial ChemistryWiley & Sons, London, England. p.351-380. 5th ed.

Löf, M., Dey, D.C., Navarro, R.M., Jacobs, D.F., 2012. Mechanical site preparation for forest restoration. New Forests, 43(5-6). p.825–848.

Lombardi, L., Carnevale, E., Corti, A., 2015. A review of technologies and performances of thermal treatment systems for energy recovery from waste. Waste Management, 37. p.26–44.

Loução, P.O., Ribau, J.P., Ferreira, A.F., 2019. Life cycle and decision analysis of electricity production from biomass – Portugal case study. Renewable and Sustainable Energy Reviews, 108. p.452–480.

Lundholm, M.P., Sundström, G., 1985. Resource and environmental impact of Tetra Brik carton and refillable and non-refillable glass bottles. Tetra Brik Aseptic environmental profile. Malmo, Sweden. p.174.

Ma, L., Jones, J.M., Pourkashanian, M., Williams, A., 2007. Modelling the combustion of pulverized biomass in an industrial combustion test furnace. Fuel, 86(12-13). p.1959–1965.

MADRP, 2005. Biomass and renewable energies in agriculture, fisheries and forestry. (in Portuguese) Biomassa e energias renováveis na agricultura, pescas e florestas. Ministério da Agricultura, do Desenvolvimento Rural e das Pescas. Lisbon, Portugal. p.92.

Maltais-Landry, G., Frossard, E., 2015. Similar phosphorus transfer from cover crop residues and water-soluble mineral fertilizer to soils and a subsequent crop. Plant and Soil, 393(1-2). p.193–205.

Mandre, M., 2006. Influence of wood ash on soil chemical composition and biochemical parameters of young Scots pine. Proceedings of the Estonian Academy of Sciences: Biology, Ecology, 55. p.91–107.

Mandre, M., Pärn, H., Ots, K., 2006. Short-term effects of wood ash on the soil and the lignin concentration and growth of Pinus sylvestris L. Forest Ecology and Management, 223(1). p.349–357.

Marinković, S., Dragaš, J., Ignjatović, I., Tošić, N., 2017. Environmental assessment of green concretes for structural use. Journal of Cleaner Production, 154. p.633–649.

Martinez, A., Rodriguez, M.E., Wells, M.L., York, S.W., Preston, J.F., Ingram, L.O., 2001. Detoxification of dilute acid hydrolysates of lignocellulose with lime. Biotechnology Progress, 17(2). p.287–293.

Martínez-Lage, I., Velay-Lizancos, M., Vázquez-Burgo, P., Rivas-Fernández, M., Vázquez-Herrero, C., Ramírez-Rodríguez, A., Martín-Cano, M., 2016. Concretes and mortars with waste paper industry: Biomass ash and dregs. Journal of Environmental Management, 181. p.863–873.

Maschio, S., Tonello, G., Piani, L., Furlani, E., 2011. Fly and bottom ashes from biomass combustion as cement replacing components in mortars production: Rheological behaviour of the pastes and materials compression strength. Chemosphere, 85(4). p.666–671.

Mateos, E., Ormaetxea, L., 2019. Sustainable renewable energy by means of using forest biomass residues. Energies, 12(1). p.1–16.

McBride, M.B., Spiers, G., 2001. Trace element content of selected fertilizers and dairy manures as determined by ICP-MS. Communications in Soil Science and Plant Analysis, 32(1-2). p.139–156.

McKechnie, J., Colombo, S., Chen, J., Mabee, W., MacLean, H.L., 2011. Forest bioenergy or forest carbon? Assessing trade-offs in greenhouse gas mitigation with wood-based fuels. Environmental Science and Technology, 45(2). p.789–795.

McKell, C., 2012. The biology and utilization of shrubs. Academic Press, London, United Kingdom. p.632.

McManus, M.C., Taylor, C.M., 2015. The changing nature of life cycle assessment. Biomass and Bioenergy, 82. p.13–26.

ME, 2017. Decree-Law No. 64/2017 of June 12, 2017. (in Portuguese) Decreto-Lei Nº. 64/2017 de 12 de junho de 2017. Ministério da Economia. Portugal. p.2966-2968.

MEID, 2010. Decree-Law No. 171/2010 of October 25, 2010. (in Portuguese) Decreto-Lei Nº. 171/2010 de 25 de outubro de 2010. Ministério da Economia, da Inovação e do Desenvolvimento. Portugal, p.4782-4795.

Mekel, O.C.L., Huppes, G., 1990. Environmental effects of different package systems for fresh milk. Leiden, The Netherlands. p.111.

Melotti, R., Santagata, E., Bassani, M., Salvo, M., Rizzo, S., 2013. A preliminary investigation into the physical and chemical properties of biomass ashes used as aggregate fillers for bituminous mixtures. Waste Management, 33(9). p.1906–1917.

Miller, B.B., Dugwell, D.R., Kandiyoti, R., 2002. Partitioning of trace elements during the combustion of coal and biomass in a suspension-firing reactor. Fuel, 81(2). p.159–171.

Mladenov, M., Pelovski, Y., 2015. Utilization of wastes from pulp and paper industry. Journal of the University of Chemical Technology and Metallurgy, 45(1). p.33–38.

Modolo, R.C.E., 2014. Valorisation of solid wastes from cellulose and paper industry. Valorização de resíduos sólidos da indústria de celulose e papel. Department of Environment and Planning. University of Aveiro. Aveiro, Portugal. p.306.

Modolo, R.C.E., 2006. Waste recovery from the pulp sector in construction products. (in Portuguese) Valorização de resíduos do sector de pasta de papel em produtos da construção civil. Master Thesis. Department of Materials and Ceramics Engineering. University of Aveiro. Aveiro, Portugal. p.101.

Modolo, R.C.E., Ferreira, V.M., Tarelho, L.A., Labrincha, J.A., Senff, L., Silva, L., 2013. Mortar formulations with bottom ash from biomass combustion. Construction and Building Materials, 45. p.275–281.

Modolo, R.C.E., Silva, T., Senff, L., Tarelho, L.A.C., Labrincha, J.A., Ferreira, V.M., Silva, L., 2015. Bottom ash from biomass combustion in BFB and its use in adhesive-mortars. Fuel Processing Technology, 129. p.192–202.

Modolo, R.C.E., Tarelho, L.A.C., Teixeira, E.R., Ferreira, V.M., Labrincha, J.A., 2014. Treatment and use of bottom bed waste in biomass fluidized bed combustors. Fuel Processing Technology, 125. p.170–181.

Momeni, M., 2012. Fundamental study of single biomass particle combustion. Doctoral Thesis. Aalborg University. Department of Energy Technology. p.71.

Morin, O., 2014. Technical and environmental comparison of circulating fluidized bed (CFB) and moving grate reactors. Master Thesis. Columbia University. Department of Earth and Environmental Engineering. p.47.

Mullins, G.L., Sikora, F.J., 1994. Effect of soil pH on the requirement for water-soluble phosphorus in triple superphosphate fertilizers. Fertilizer Research, 40(3). p.207–214.

Muralikrishna, M., Muralikrishna, M., Rajasekhar, S., 2017. Design, study and analysis of steam turbine blade to find its frequency and deformations. International Journal of Research and Innovative Technology, 4(1). p.1–5.

Muylle, H., Van Hulle, S., De Vliegher, A., Baert, J., Van Bockstaele, E., Roldán-Ruiz, I., 2015. Yield and energy balance of annual and perennial lignocellulosic crops for bio-refinery use: A 4-year field experiment in Belgium. European Journal of Agronomy, 63. p.62–70.

Nachimuthu, G., Guppy, C., Kristiansen, P., Lockwood, P., 2009. Isotopic tracing of phosphorus uptake in corn from ^{33}P labelled legume residues and ^{32}P labelled fertilisers applied to a sandy loam soil. Plant and Soil, 314(1-2). p.303–310.

Naylor, L.M., Schmidt, E.J., 1986. Agricultural use of wood ash as a fertilizer and liming material. Tappi Journal, 69(10). p.114–119.

Netto, C.A., 2008. Potential of forest biomass residues for energy purposes in three municipalities in the district of Santarém. (in Portuguese) Potencial da biomassa florestal residual para fins energéticos de três concelhos do distrito de Santarém. Master Thesis. New University of Lisbon. Department of Science and Environmental Engineering. p.129.

Nilsson, M. andreas, L., Lagerkvist, A., 2016. Effect of accelerated carbonation and zero valent iron on metal leaching from bottom ash. Waste Management, 51. p.97–104.

Nohlgren, I., Svard, S.H., Jansson, M., Rodin, J., 2014. Electricity from new and future facilities. (in Swedish) El från nya och framtida anläggningar. ELFORSK. Stockholm, Sweden. p.234.

Nunes, L.J.R., 2015. The use of biomass as an energy alternative for the sustainability and competitiveness of Portuguese industry. (in Portuguese) A utilização de biomassa como alternativa energética para a sustentabilidade e competitividade da indústria Portuguesa. Doctoral Thesis. University of Beira Interior. p.262.

Nunes, L.J.R., Matias, J.C.O., Catalão, J.P.S., 2016. Biomass in the generation of electricity in Portugal: A review. Renewable and Sustainable Energy Reviews, 71. p.373–378.

Núñez-Regueira, L., Proupín-Castiñeiras, J., Rodríguez-Añón, J., 2004. Energy evaluation of forest residues originated from shrub species in Galicia. Bioresource technology, 91. p.215–221.

Núñez-Regueira, L., Rodríguez-Añón, J., Proupín-Castiñeiras, J., Romero-García, A., 2001. Energetic evaluation of biomass originating from forest waste by bomb calorimetry. Journal of Thermal Analysis and Calorimetry, 66(1). p.281–292.

Nurmesniemi, H., Mäkelä, M., Pöykiö, R., Manskinen, K., Dahl, O., 2012. Comparison of the forest fertilizer properties of ash fractions from two power plants of pulp and paper mills incinerating biomass-based fuels. Fuel Processing Technology, 104. p.1–6.

Nussbaumer, T., 2003. Combustion and co-combustion of biomass: fundamentals, technologies and primary measures for emission reduction. Energy and Fuels, 17(6). p.1510–1521.

Oates, T., 1997. Lime and Limestones. In: Ullmann's encyclopedia of industrial chemistry., 5th ed. Wiley and Sons., London, United Kingdom. p.125-149.

Obernberger, I., Biedermann, F., 2012. Biomass energy heat provision in modern large-scale systems, 1st ed. Springer, New York, USA. p.1350.

Obernberger, I., Brunner, T., Bärnthaler, G., 2006. Chemical properties of solid biofuels-significance and impact. Biomass and Bioenergy, 30(11). p.973–982.

Obernberger, I., Thek, G., 2004. Physical characterisation and chemical composition of densified biomass fuels with regard to their combustion behaviour. Biomass and Bioenergy, 27(6). p.653–669.

Ochecova, P., Tlustos, P., Szakova, J., 2014. Wheat and soil response to wood fly ash application in contaminated soils. Agronomy Journal, 106(3). p.995–1002.

Ohno, T., 1992. Neutralization of soil acidity and release of phosphorus and potassium by wood ash. Journal of Environmental Quality, 21(3). p.433–438.

Ohno, T., Erich, M.S., 1990. Effect of wood ash application on soil pH and soil test nutrient levels. Agriculture, Ecosystems & Environment, 32(3). p.223–239.

Ondova, M., Estokova, A., 2014. LCA and multi-criteria analysis of fly ash concrete pavements. International Journal of Environmental and Ecological Engineering, 8(5). p.313–318.

Onovwiona, H.I., Ugursal, V.I., 2006. Residential cogeneration systems: review of the current technology. Renewable and Sustainable Energy Reviews, 10(5). p.389–431.

Orecchini, F., Naso, V., 2012. Energy systems in the era of energy vectors: Green energy and technology. Pringer-Verlag, London, United Kingdom. p.362.

Palz, W., Zibetta, H., 2015. EU biomass R&D: The beginnings (laying the ground for biomass becoming mainstream in sustainable energy supply in Europe), in: van Swaaij, W.P.M., Kersten, S.R.A., Palz, W. (Eds.), Biomass Power for the World. Jenny Stanford Publishing, Singapore. 1st. ed. p.163–189.

Pandey, A., Larroche, C., Ricke, S.C., Dussap, C.G., Gnansounou, E., 2011. Biofuels., 1st. ed. Academic Press, Cambridge, USA. p.642.

Park, N.D., Michael Rutherford, P., Thring, R.W., Helle, S.S., 2012. Wood pellet fly ash and bottom ash as an effective liming agent and nutrient source for rye grass (Lolium perenne L.) and oats

(Avena sativa). Chemosphere, 86(4). p.427–432.
Pasandín, A.R., Pérez, I., Ramírez, A., Cano, M.M., 2016. Moisture damage resistance of hot-mix asphalt made with paper industry wastes as filler. Journal of Cleaner Production, 112. p.853–862.
Peña, J.A.P., 2011. Bubbling fluidized beds: When to use this technology, in: Luckos, A., den Hoed, P. (Eds.), Industrial Fluidization South Africa. Johannesburg, South Africa. p.57–66.
Pereira, M.T.J., Silva, T.J.A. da, Bonfim-Silva, E.M., 2016. Soil water content and wood ash fertilization on the cultivation of gladiolus. Revista Brasileira de Engenharia Agrícola e Ambiental, 20(4). p.350–356.
Perilhon, C., Alkadee, D., Descombes, G., Lacour, S., 2012. Life cycle assessment applied to electricity generation from renewable biomass. Energy Procedia, 18. p.165–176.
Perrin, A., 2013. urban agricultural systems in west africa (in French) Evaluation environnementale des systèmes agricoles urbains en Afrique de l'Ouest. Doctoral Thesis. PhD thesis. L'Institut des Sciences et Industries du Vivant et de l'Environnement (AgroParisTech). p.218.
Pesonen, J., Kuokkanen, V., Kuokkanen, T., Illikainen, M., 2016. Co-granulation of bio-ash with sewage sludge and lime for fertilizer use. Journal of Environmental Chemical Engineering, 4(4). p.4817–4821.
Pinho, P.A.M., 2014. Residues valorisation in construction materials. (in Portuguese) Valorização de resíduos em materiais de construção. Department of Civil Engineering. University of Aveiro. Aveiro, Portugal. p.135.
Pitman, R.M., 2006. Wood ash use in forestry: a review of the environmental impacts. Forestry, 79(5). p.563–588.
Plotegher, F., Ribeiro, C., 2016. Characterization of single superphosphate powders – A study of milling effects on solubilization kinetics. Materials Research, 19(1). p.98–105.
Pommer, K., Hillersborg, A.S., Hirsback, S., Hoffman, L., Schimidt, A., 1991. Environmental assessment of milk pachaging. Return bottles in polycarbonate compared to disposable cartons. Environmental project N. 168. Copenhagen, Denmark. p.38.
Popa, V.I., 2018. Biomass for fuels and biomaterials, in: Popa, V., Volf, I. (Eds.), Biomass as Renewable Raw Material to Obtain Bioproducts of High-Tech Value. Elsevier, Amsterdam, Netherlands. 1st ed. p.1–37.
PRé, 2019. Company history. Product Ecology Consultants. URL https://www.pre-sustainability.com/about-pre/company-history (accessed 03.23.2019).
Ptasinski, K.J., 2016. Efficiency of biomass energy: an exergy approach to biofuels, power and biorefineries, 1st ed. John Wiley & Sons, Hoboken, USA. p.784.
Qiu, G., Shao, Y., Li, J., Liu, H., Riffat, S.B., 2012. Experimental investigation of a biomass-fired ORC-based micro-CHP for domestic applications. Fuel, 96. p.374–382.
Quinteiro, P., Tarelho, L., Marques, P., Martin-Gamboa, M., Freire, F., Arroja, L., Dias, A.C., 2019. Life cycle assessment of wood pellets and wood split logs for residential heating. Science of The Total Environment, 689. p.580–589.
Quirantes, M., Calvo, F., Romero, E., Nogales, R., 2016. Soil-nutrient availability affected by different biomass-ash applications. Journal of soil science and plant nutrition, 16. p.159–163.
Rafael, S., Tarelho, L., Monteiro, A., Monteiro, T., Gonçalves, C., Freitas, S., Lopes, M., 2015. Atmospheric emissions from forest biomass residues to energy supply chain: a casa study in Portugal. Environmental Engineering Science, 32(6). p.505–515.
Raheem, A.A., Akinteye, I.A., Lasisi, S.A., 2014. A study of thermal conductivity of wood ash blended cement mortar. Construction Materials and Structures. p.261 – 267.
Rajaeifar, M.A., Tabatabaei, M., Abdi, R., Latifi, A.M., Saberi, F., Askari, M., Zenouzi, A., Ghorbani, M., 2017. Attributional and consequential environmental assessment of using waste cooking oil- and poultry fat-based biodiesel blends in urban buses: a real-world operation condition study. Biofuel Research Journal, 4(3). p.638–653.
Rajamma, R., 2011. Biomass fly ash incorporation in cement based materials. Doctoral Thesis. Department of Materials and Ceramics Engineering. University of Aveiro. p.259.
Rajamma, R., Ball, R.J., Tarelho, L.A.C., Allen, G.C., Labrincha, J.A., Ferreira, V.M., 2009. Characterisation and use of biomass fly ash in cement-based materials. Journal of Hazardous Materials, 172. p.1049–1060.
Rajamma, R., Senff, L., Ribeiro, M.J., Labrincha, J.A., Ball, R.J., Allen, G.C., Ferreira, V.M., 2015.

Biomass fly ash effect on fresh and hardened state properties of cement based materials. Composites Part B: Engineering, 77. p.1–9.

Ramos, T., Matos, A.M., Sousa-Coutinho, J., 2013a. Mortar with wood waste ash: Mechanical strength carbonation resistance and ASR expansion. Construction and Building Materials, 49. p.343–351.

Ramos, T., Matos, A.M., Sousa-Coutinho, J., 2013b. Mortar with wood waste ash: Mechanical strength carbonation resistance and ASR expansion. Construction and Building Materials, 49. p.343–351.

Ranta, T., Halonen, P., Alakangas, E., 2001. Production of forest chips in Finland. Technical Research Centre of Finland. Espoo, Finland. p.64.

RCM, 2017. Resolution of Council of Ministers No.163/2017 of October 31, 2017. (in Portuguese) Resolução do Conselho de Ministros Nº. 163/2017 de 31 de outubro de 2017. Portugal. p.5839-5847.

RCM, 2013. Resolution of Council of Ministers No. 20/2013 of April 10, 2013. (in Portuguese) Resolução do Conselho de Ministros Nº. 20/2013 de 10 de abril de 2013. Portugal. p. 6394-6397.

RCM, 2010. Resolution of Council of Ministers No. 29/2010 of 15 April 2010. (in Portuguese) Resolução do Conselho de Ministros Nº. 29/2010 de 15 de abril de 2010. Portugal. p.1289-1296.

RCM, 2008. Resolution of Council of Ministers No. 1/2008 of January 4, 2008. (in Portuguese) Resolução do Conselho de Ministros Nº. 1/2008 de 4 de janeiro de 2008. Portugal. p. 106-141.

RCM, 2006a. Resolution of Council of Ministers No. 65/2006 of 26 May, 2006. (in Portuguese) Resolução do Conselho de Ministros N.º 65/2006 de 26 de maio de 2006. Portugal. p.3511-3559.

RCM, 2006b. Resolution of Council of Ministers No. 114/2006 of 15 September, 2006. (in Portuguese) Resolução do Conselho de Ministros Nº. 114/2006 de 15 de setembro de 2006. Portugal. p.6730-6809.

RCM, 2005. Resolution of Council of Ministers No. 169/2005 of 24 October, 2005. (in Portuguese) Resolução do Conselho de Ministros Nº. 169/2005 de 24 de outubro de 2005. Portugal. p.6168-6176.

RCM, 2003. Resolution of Council of Ministers No. 63/2003 of 28 April, 2003. (in Portuguese) Resolução do Conselho de Ministros Nº. 63/2003 de 28 de abril de 2003. Portugal. p.2722-2731.

Reboredo, F., 2014. Forest context and policies in Portugal, 1st ed. Springer, New York, USA. p.233.

Rehl, T., Lansche, J., Müller, J., 2012. Life cycle assessment of energy generation from biogas - Attributional vs. consequential approach. Renewable and Sustainable Energy Reviews, 16(6). p.3766–3775.

Ribbing, C., 2007. Environmentally friendly use of non-coal ashes in Sweden. Waste Management, 27(10). p.1428–1435.

Rimár, M., Fedák, M., Korshunov, A., Kulikov, A., Mizáková, J., 2016. Determination of excess air ratio during combustion of wood chips respect to moisture content. Acta Facultatis Xylologiae, 58(2). p.133–140.

Rosales, J., Cabrera, M., Beltrán, M.G., López, M., Agrela, F., 2017. Effects of treatments on biomass bottom ash applied to the manufacture of cement mortars. Journal of Cleaner Production, 154. p.424–435.

Röser, D., Asikainen, A., Raulund-Rasmussen, K., Stupak, I., 2008. Sustainable use of forest biomass for energy. A systhesis with focus on the Baltiv and Nordic Region., 1st ed. Springer, New York, USA. p.261.

Rout, I., 2013. Thermal analysis of steam turbine power plants. IOSR Journal of Mechanical and Civil Engineering, 7(2). p.28–36.

Rummel, L., Paist, A., 2016. Case study comparison of bubbling fluidised bed and grate-firing technology based small biomass combined heat and power plants. Chemical Engineering Transactions, 52(2015). p.1147–1152.

Russell, M.B., Woodall, C.W., Fraver, S., D'Amato, A.W., Domke, G.M., Skog, K.E., 2014. Residence times and decay rates of downed woody debris biomass/carbon in eastern U.S. forests. Ecosystems, 17(5). p.765–777.

Saarsalmi, A., Mälkönen, E., Kukkola, M., 2004. Effect of wood ash fertilization on soil chemical properties and stand nutrient status and growth of some coniferous stands in Finland. Scandinavian Journal of Forest Research, 19(3). p.217–233.

Saarsalmi, A., Mälkönen, E., Piirainen, S., 2001. Effects of wood ash fertilization on forest soil chemical properties. Silva Fennica, 35(3). p.355–368.

Sabedot, S., Sundstron, M.G., De Böer, S.C., Sampaio, C.H., Dias, R.G. de O., Ramos, C.G., 2011. Caracterização e aproveitamento de cinzas da combustão de carvão mineral geradas em usinas termelétricas, in: III Congresso Brasileiro de Carvão MineralGramado, Brazil. p.15.

Sablayrolles, C., Gabrielle, B., Montrejaud-Vignoles, M., 2010. Life Cycle Assessment of Biosolids Land Application and Evaluation of the Factors Impacting Human Toxicity through Plant Uptake. Journal of Industrial Ecology, 14(2). p.231–241.

Sahu, P., Sharma, S., Purohit, R., Rana, R.S., Parashar, V., 2017. Analysis of mechanical properties of fiber glass-epoxy-fly ash composites. Materials Today: Proceedings, 4(2). p.3102–3109.

Saldanha, E.S., 2001. The application of cogeneration systems to the cooling of food and buildings in East Timor. Master Thesis. Massey University New Zealand. p.94.

Sandilands, J., Kellenberger, D., Nicholas, I., Nielsen, P., 2009. Life cycle assessment of wood pellets and bioethanol from wood residues and willow. New Zealand Journal of Forestry, 53(4). p.25–33.

Santalla, M., Omil, B., Rodríguez-Soalleiro, R., Merino, A., 2011. Effectiveness of wood ash containing charcoal as a fertilizer for a forest plantation in a temperate region. Plant and Soil, 346(1). p.63–78.

Scheepers, G.., 2014. The effect of wood ash on the soil properties and nutrition and growth of Eucalyptus grandis x urophylla grown on a sandy coastal soil in Zululand. Master Thesis. University of Stellenbosch. Department of Forest and Wood Science. p.162.

Scheetz, B.E., Earle, R., 1998. Utilization of fly ash. Current Opinion in Solid State and Materials Science, 3(5). p.510–520.

Schepper, M., Van den Heede, P., Van Driessche, I., De Belie, N., 2014. Life cycle assessment of completely recyclable concrete. Materials, 7(8). p.6010–6027.

Schiemenz, K., Eichler-lo, B., 2010. Biomass ashes and their phosphorus fertilizing effect on different crops. p.471–482.

Schmidt, J.H., Merciai, S., Thrane, M., Dalgaard, R., 2011. Inventory of country specific electricity in LCA - Consequential and attributional scenarios. 2.0 LCA consultants. Aalborg, Denmark. p.26.

Searchinger, T., Heimlich, R., Houghton, R.A., Dong, F., Elobeid, A., Fabiosa, J., Tokgoz, S., Hayes, D., Yu, T.H., 2008. Use of U.S. croplands for biofuels increases greenhouse gases through emissions from land-use change. Science, 319(5867). p.1238–1240.

Searchinger, T.D., Beringer, T., Holtsmark, B., Kammen, D.M., Lambin, E.F., Lucht, W., Raven, P., van Ypersele, J.P., 2018. Europe's renewable energy directive poised to harm global forests. Nature Communications, 9(1). p.10–13.

Senesi, G.S., Baldassarre, G., Senesi, N., Radina, B., 1999. Trace element inputs into soils by anthropogenic activities and implications for human health, in: Chemosphere. p.343–377.

SETAC, 1991. A technical framework for life- cycle assessments. Society of Environmental Toxicology and Chemistry. Washington, USA. p.134.

Seto, K.E., Churchill, C.J., Panesar, D.K., 2017. Influence of fly ash allocation approaches on the life cycle assessment of cement-based materials. Journal of Cleaner Production, 157. p.65–75.

Shiehnejadhesar, A., Sommersacher, P., Scharler, R., Anca-Couce, A., Hochenauer, C., Mehrabian, R., 2017. CO/CO2 ratio in biomass char oxidation. Energy Procedia, 120. p.238–245.

Silva, D.F.R., 2016. Evaluation of biomass burning ash in liming and recycling of nutrients in soil. (in Portuguese) Valorização das cinzas de combustão de biomassa na calagem e reciclagem de nutrientes no solo. Doctoral Thesis. University of Aveiro. Department of Environment and Planning. p.640.

Simas, F.C. da S.P. e, 2013. Availability of forest biomass for combustion at the plant Sines thermoelectric plant. (in Portuguese) Disponibilidade de biomassa florestal residual para combustão na central termoeléctrica de Sines. Master Thesis. New University of Lisbon. Department of Science and Technology.

Sims, R.E.H., 2002. The brilliance of bioenergy in business and in practice. Routledge Studies in Bioenergy, London, United Kingdom. p.328.

Sinha, A.K., Deep, K.P., Minz, A., Barla, S., Alam, P., 2018. Effect of crop residue incorporation in maize on nutrient status their uptake and yield in acid soil of Ranchi. Journal of Pharmacognosy and Phytochemistry, 1. p.3246–3251.

Sklivaniti, V., Tsakiridis, P.E., Katsiotis, N.S., Velissariou, D., Pistofidis, N., Papageorgiou, D., Beazi, M., 2017. Valorisation of woody biomass bottom ash in Portland cement: A characterization and

hydration study. Journal of Environmental Chemical Engineering, 5(1). p.205–213.
Soezen, K., 2018. The future of gypsum: market forecasts to 2028. Smithers Apex. Akron, USA. p.670.
Spinelli, R., Hartsough, B., 2011. A survey of Italian chipping operations. Biomass and Bioenergy, 21(6). p.433–444.
Spinelli, R., Magagnotti, N., Picchi, G., 2012. A supply chain evaluation of slash bundling under the conditions of mountain forestry. Biomass and Bioenergy, 36. p.339–345.
Spinelli, R., Nati, C., Magagnotti, N., 2007. Recovering logging residue: experiences from the Italian Eastern Alps. Croatian Journal of Forest Engineering, 28(1). p.1–9.
Spinelli, R., Ward, S.M., Owende, P.M., 2009. A harvest and transport cost model for Eucalyptus spp. fast-growing short rotation plantations. Biomass and Bioenergy, 33(9). p.1265–1270.
Steen, B., 1999a. A systematic approach to environmental priority strategies in product development (EPS). Version 2000 – General system characteristics. Chalmers University of Technology. Göteborg, Sweden. p.67.
Steen, B., 1999b. A systematic approach to environmental priority strategies in product development (EPS). Version 2000 – Models and data of the default method. Chalmers University of Technology. Göteborg, Sweden. p.312.
Stevanovic, T., 2018. Chemistry of lignocellulosics., 1st ed. CRC Press, Boca Raton, USA. p.308.
Stiernström, S., Enell, A., Wik, O., Hemström, K., Breitholtz, M., 2014. Influence of leaching conditions for ecotoxicological classification of ash. Waste Management, 34(2). p.421–429.
Straka, P., Náhunková, J., Žaloudková, M., 2014. Analysis of unburned carbon in industrial ashes from biomass combustion by thermogravimetric method using Boudouard reaction. Thermochimica Acta, 575. p.188–194.
Strezov, V., Evans, T.J., 2014. Biomass processing technologies, 1st ed. CRC Press, Boca Raton, USA. p.408.
Subramania, P., Subasinghe, K., Fonseka, W.R.K., 2015. Wood ash as an effective raw material for concrete blocks. International Journal of Research in Engineering and Technology, 4(2). p.228–233.
Supancic, K., Obernberger, I., 2011. Wood ash utilisation as a stabiliser in road construction – First results of large-scale tests, in: 19th European Biomass Conference & Exhibition. Berlin, Germany. p.859–870.
Tagliaferri, C., Evangelisti, S., Clift, R., Lettieri, P., 2018. Life cycle assessment of a biomass CHP plant in UK: The Heathrow energy centre case. Chemical Engineering Research and Design, 133. p.210–221.
Tamminen, P., Saarsalmi, A., Smolander, A., Kukkola, M., Helmisaari, H.S., 2012. Effects of logging residue harvest in thinnings on amounts of soil carbon and nutrients in Scots pine and Norway spruce stands. Forest Ecology and Management, 263. p.31–38.
Tarelho, L.A.C., Neves, D.S.F., Matos, M.A.A., 2011. Forest biomass waste combustion in a pilot-scale bubbling fluidised bed combustor. Biomass and Bioenergy, 35(4). p.1511–1523.
Tarelho, L.A.C., Teixeira, E.R., Silva, D.F.R., Modolo, R.C.E., Labrincha, J.A., Rocha, F., 2015. Characteristics of distinct ash flows in a biomass thermal power plant with bubbling fluidised bed combustor. Energy, 90. p.387–402.
Tarnawski, W., 2004. Emission factors for combustion of biomass fuels in the pulp and paper mills. Fibres and Textiles in Eastern Europe, 12(3). p.91–95.
Tedim, F., Remelgado, R., Martins, J., Carvalho, S., 2015. The largest forest fires in portugal: The constraints of burned area size on the comprehension offire severity. Journal of Environmental Biology, 36(1). p.133–143.
Teixeira, A., 2009. Options for the use of forest biomass in the energy context of Portugal. (in Portuguese) Opções de utilização da biomassa florestal no contexto energético de Portugal. Master Thesis. University of Porto. Faculty of Engineering. p.129.
Teixeira, E.R., Mateus, R., Camões, A., Branco, F.G., 2019. Quality and durability properties and life-cycle assessment of high volume biomass fly ash mortar. Construction and Building Materials, 197. p.195–207.
Teixeira, E.R., Mateus, R., Camões, A.F., Bragança, L., Branco, F.G., 2015. Comparative environmental life-cycle analysis of concretes using biomass and coal fly ashes as partial cement replacement material. Journal of Cleaner Production, 112. p.2221–2230.

Thakur, A., Canter, C.E., Kumar, A., 2014. Life-cycle energy and emission analysis of power generation from forest biomass. Applied Energy, 128. p.246–253.

Thiffault, E., Sokhansanj, S., Ebadian, M., Rezaei, H., Bahman, E.O., Yazdanpanah, F., Asikainen, A., Routa, J., 2018. Biomass pre-treatment for bioenergy. Case study 2: Moisture, physical property, ash and density management as pre-treatment practices in Canadian forest biomass supply chains. International Energy Agency. Paris, France. p.84.

Thinkstep, 2015. A brief history of life cycle assessment (LCA). URL http://www.gabi-software.com/news/news-detail/article/a-brief-history-of-life-cycle-assessmentlca (accessed 03.23.2019).

Thoma, F., Husson, H., Nuala, N.F., Nordfejll, T., Picos, J., Prior, M., 2018. EIP-Agri Focus group sustainable mobilisation of forest biomass. European map of the regional forest-based sector. Agriculture & Innovation, European Commission. Brussels, Belgium. p.14.

TIFAC, 2001. Technology linked business opportunity publications. Non conventional sources of plant nutrient and soil conditioners to enhance agricultural productivity. Technology information forecasting and assessment council. Department of science and technology. p.

Tomé, M., Oliveira, T., Soares, P., 2006. The GLOBULUS 3.0 model: Data and equations. (in Portuguese) O modelo GLOBULUS 3.0: Dados e equações. Universidade Técnica de Lisboa e Instituto Superior de Agronomia. Lisbon, Portugal. p.26.

Tosti, L., van Zomeren, A., Pels, J.R., Comans, R.N.J., 2018. Technical and environmental performance of lower carbon footprint cement mortars containing biomass fly ash as a secondary cementitious material. Resources, Conservation and Recycling, 134. p.25–33.

Tsupari, E., Monni, S., Tormonen, K., Pellikka, T., Syri, S., 2007. Estimation of annual CH4 and N2O emissions from fluidised bed combustion: An advanced measurement-based method and its application to Finland. International Journal of Greenhouse Gas Control, 1(3). p.289–297.

Ukrainczyk, N., 2016. Reuse of woody biomass ash waste in cementitious materials. Chemical and Biochemical Engineering Quarterly, 30(2). p.137–148.

UNEP/SETAC, 2017. Life cycle management: How business uses it to decrease footprint, create opportunities and make value chains more sustainable. United Nations Environment Programme. Society of Environmental Toxicology and Chemistry. Paris, France. p.48.

Unimadeiras, 2014. Woody forest products. (in Portuguese) Produtos florestais lenhosos. Grupo Unifloresta

. Albergaria-a-Velha, Portugal. p.56.

Uprety, D., Hejcman, M., Száková, J., Kunzová, E., Tlustoš, P., 2009. Concentration of trace elements in arable soil after long-term application of organic and inorganic fertilizers. Nutrient Cycling in Agroecosystems, 85(3). p.241–252.

Valeriano, I.H., Marques, G.L.L., Freitas, S.P., Penha, E.M., Gonçalves, M.M.M., 2018. Cassava pulp enzymatic hydrolysate as a promising feedstock for ethanol production. Brazilian Archives of Biology and Technology., 61. p.1–10.

Vamvuka, D., 2010. Overview of solid fuels combustion, in: Handbook of Combustion. Wiley-VCH, Weinheim, Germany. p.3168.

van Loo, S., Koppejan, J., 2008. The handbook of biomass combustion and co-firing, 2nd ed. Earthscan, Sterling, USA. p.442.

van Swaaij, W., Kersten, S., 2015. Overview of energy from biomass for nonexperts, in: van Swaaij, W., Kersten, S., Palz, W. (Eds.), Biomass Power for the World. Jenny Stanford Publishing, Singapore. 1st. ed. p.1–116.

Vassilev, S. V, Baxter, D. andersen, L.K., Vassileva, C.G., 2013a. An overview of the composition and application of biomass ash.: Part 2. Potential utilisation, technological and ecological advantages and challenges. Fuel, 105. p.19–39.

Vassilev, S. V, Baxter, D. andersen, L.K., Vassileva, C.G., 2013b. An overview of the composition and application of biomass ash. Part 1. Phase-mineral and chemical composition and classification. Fuel, 105. p.40–76.

Vassilev, S. V, Vassileva, C.G., Vassilev, V.S., 2015. Advantages and disadvantages of composition and properties of biomass in comparison with coal: An overview. Fuel, 158. p.330–350.

Vassilev, S. V., Vassileva, C.G., Baxter, D., 2014. Trace element concentrations and associations in some biomass ashes. Fuel, 129. p.292–313.

Venkatachalam, V., Spierling, S., Horn, R., Endres, H.J., 2018. LCA and eco-design: Consequential and attributional approaches for bio-based plastics. Procedia CIRP, 69. p.579–584.

Viana, H., Cohen, W.B., Lopes, D., Aranha, J., 2010. Assessment of forest biomass for use as energy. GIS-based analysis of geographical availability and locations of wood-fired power plants in Portugal. Applied Energy, 87(8). p.2551–2560.

von Wilpert, K., Hartmann, P., Schäffer, J., 2016. Quality control in a wood ash re-cycling concept for forests. VGB PowerTech(4). p.67–72.

Vos, J., 2005. Biomass energy for heating and hot water supply in Belarus. Best practice guidelines. Part A: biomass combustion. Biomass Technology Group BV and University of Twente. Enschede, Netherlands. p.125.

Wang, L., Hustad, J.E., Skreiberg, Ø., Skjevrak, G., Grønli, M., 2012. A critical review on additives to reduce ash related operation problems in biomass combustion applications. Energy Procedia, 20(1876). p.20–29.

Wang, M.Q., 2001. Development and use of GREET 1.6. Fuel-cycle model for transportation fuels and vehicle technologies. Center for Transportation Research Argonne National Laboratory. Argonne, USA. p. 39.

Wang, M.Q., Elgowainy, A., Benavides, P.T., Burnham, A., Cai, H., Dai, Q., Hawkins, T.R., Kelly, J.C., Kwon, H., Lee, D., Lee, U., Lu, Z., Ou, L., 2018. Summary of expansions and updates in GREET 2018. Center for Transportation Research Argonne National Laboratory. Argonne, USA. p. 32.

Wang, P., Wang, J., Qin, Q., Wang, H., 2017. Life cycle assessment of magnetized fly-ash compound fertilizer production: A case study in China. Renewable and Sustainable Energy Reviews, 73. p.706–713.

Wang, S., Llamazos, E., Baxter, L., Fonseca, F., 2008. Durability of biomass fly ash concrete: Freezing and thawing and rapid chloride permeability tests. Fuel, 87(3). p.359–364.

WEC, 2016. World energy resources 2016. World Energy Council. London, United Kingdom. p.1028.

Wedepohl, K.H., Simon, K., 2010. Chemie der Erde The chemical composition of medieval wood ash glass from Central Europe. Chemie der Erde - Geochemistry, 70(1). p.89–97.

Weidema, B.P., 2015. Marginal Suppliers. Consequential-LCA. URL http://www.consequential-lca.org. (accessed 08.03.2019).

Weidema, B.P., 2003. Market information in life cycle assessment. Danish Ministry of the Environment, 863(863). p.147.

Weidema, B.P., Pizzol, M., Schmidt, J., Thoma, G., 2018. Attributional or consequential Life Cycle Assessment: A matter of social responsibility. Journal of Cleaner Production, 174. p.305–314.

Werther, J., Saenger, M., Hartge, E.-U., Ogada, T., Siagi, Z., 2000. Combustion of agricultural residues. Progress in Energy and Combustion Science, 26(1). p.1–27.

Whittaker, C., Macalpine, W., Yates, N.E., Shield, I., 2016. Dry matter losses and methane emissions during wood chip storage: the impact on full life cycle greenhouse gas savings of short rotation coppice willow for heat. Bioenergy Research, 9(3). p.820–835.

Whittaker, C., Mortimer, N., Murphy, R., Matthews, R., 2011. Energy and greenhouse gas balance of the use of forest residues for bioenergy production in the UK. Biomass and Bioenergy, 35(11). p.4581–4594.

Wiinikka, H., 2005. High temperature aerosol formation and emission minimisation during combustion of wood pellets. Doctoral Thesis. Luleå University of Technology. Department of Applied Physics and Mechanical Engineering. p.196.

Wiklund, J., 2017. Effects of wood ash on soil fertility and plant performance in southwestern Kenya. Master Thesis. Swedish University of Agricultural Sciences. Department of Soil and Environment. p.54.

Willén, A., Junestedt, C., Rodhe, L., Pell, M., Jönsson, H., 2017. Sewage sludge as fertiliser – environmental assessment of storage and land application options. Water Science and Technology, 75(5).

Williams, C.L., Westover, T.L., Emerson, R.M., 2016. Sources of biomass feedstock variability and the potential impact on biofuels production. Bioenergy Research. p.1–14.

Williams, P.T., 2013. Waste treatment and disposal, 2nd ed. John Wiley & Sons, Leeds.

Woo, H., Turner, P., 2019. A review of recent research on carbon neutrality in forest bioenergy

feedstocks. International Journal of Environmental Sciences & Natural Resources, 19. p.5.
Xuan, D., Zhan, B., Poon, C.S., Zheng, W., 2016. Carbon dioxide sequestration of concrete slurry waste and its valorisation in construction products. Construction and Building Materials, 113. p.664–672.
Yang, Y., 2016. Two sides of the same coin: consequential life cycle assessment based on the attributional framework. Journal of Cleaner Production, 127. p.274–281.
Yao, Z.T., Ji, X.S., Sarker, P.K., Tang, J.H., Ge, L.Q., Xia, M.S., Xi, Y.Q., 2015. A comprehensive review on the applications of coal fly ash. Earth-Science Reviews. p.105–121.
Ybarra, P.L.G., Granda, A.B., Muñoz, F.Y., 2001. Tecnologías energéticas e impacto ambiental. McGraw-Hill Interamericana de España, Madrid, Spain. p.728.
Yin, C., Rosendahl, L.A., Kær, S.K., 2008. Grate-firing of biomass for heat and power production. Progress in Energy and Combustion Science, 34(6). p.725–754.
Yoshitake, I., Ueno, S., Ushio, Y., Arano, H., Fukumoto, S., 2016. Abrasion and skid resistance of recyclable fly ash concrete pavement made with limestone aggregate. Construction and Building Materials, 112. p.440–446.
Yuan, Z., 2017. Bioenergy: principles and technologies, 1st ed. Walter de Gruyter, Berlin, Germany. p.348.
Zamagni, A., Guinée, J., Heijungs, R., Masoni, P., Raggi, A., 2012. Lights and shadows in consequential LCA. International Journal of Life Cycle Assessment, 17(7). p.904–918.
Zaman, M., Shahid, S.A., Heng, L., 2018. Guideline for salinity assessment, mitigation and adaptation using nuclear and related techniques. Springer, New York, USA. p.164.
Zamora-Cristales, R., Boston, K., Sessions, J., Murphy, G., 2014. Stochastic simulation and optimization of mobile chipping economics in processing and transport of forest biomass from residues. Silva Fennica, 47(5). p.1–22.
Zanchi, G., Pena, N., Bird, N., 2012. Is woody bioenergy carbon neutral? A comparative assessment of emissions from consumption of woody bioenergy and fossil fuel. GCB Bioenergy, 4(6). p.761–772.
Zhang, X.Q., Xu, D., 2003. Potential carbon sequestration in China’s forests. Environmental Science and Policy, 6(5). p.421–432.
Zhao, M., Han, Z., Sheng, C., Wu, H., 2013. Characterization of residual carbon in fly ashes from power plants firing biomass. Energy and Fuels, 27(2). p.898–907.
Zhou, X., Xu, W., Shi, Y., Wang, J., Wu, X., 2015. The economic analysis of the household combined heat and power system, in: Wang, Y. Zhao, J. (Ed.), Advances in Energy, Environment and Materials Science: Proceedings of the International Conference on Energy, Environment and Materials Science.CRC Press., Guangzhou, China. p.872.
Zou, C., Zhao, Q., Zhang, G., Xiong, B., 2016. Energy revolution: From a fossil energy era to a new energy era. Natural Gas Industry, 36(1). p.1–10.

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Appendices

Appendix A: Life cycle assessment of woody ash valorisation in construction materials

Table S1 - Ecoinvent processes used for landfilling scenarios.

Scenarios	Input material	Ecoinvent processes
Ash disposal in sanitary landfill	Electricity, high voltage	Electricity, high voltage {PT}| market for | APOS, U
	Electricity, low voltage	Electricity, low voltage {PT}| market for | APOS, U
	Light fuel oil	Light fuel oil {Europe without Switzerland}| market for | APOS, U
	Diesel	Diesel {Europe without Switzerland}| market for | APOS, U
	Lubricating oil	Lubricating oil {GLO}| market for | APOS, U
	Iron sulphate	Iron sulfate {GLO}| market for | APOS, U
	Aluminium sulphate	Aluminum sulfate, powder {GLO}| market for | APOS, U
	Iron chloride	Iron (III) chloride, without water, in 40 % solution state {GLO}| market for | APOS, U
	Sodium hydroxide	Sodium hydroxide, without water, in 50 % solution state {GLO}| market for | APOS, U
	Quicklime	Quicklime, milled, packed {GLO}| market for | APOS, U
	Hydrochloric acid	Hydrochloric acid, without water, in 30 % solution state {RER}| hydrochloric acid production, from the reaction of hydrogen with chlorine | APOS, U
	Natural gas	Natural gas, high pressure {GLO}| market group for | APOS, U
	Ammonia	Ammonia, liquid {RER}| ammonia production, partial oxidation, liquid | APOS, U
	Chromium	Chromium {GLO}| market for | APOS, U
	Titanium dioxide	Titanium dioxide {RER}| market for | APOS, U
	Cement	Cement, unspecified {Europe without Switzerland}| market for cement, unspecified | APOS, U
	Water	Water, decarbonised, at user {GLO}| market for | APOS, U

Table S2 - Woody biomass ash applications, showing the biomass origin and type, the ash type, the energy conversion technology, the ash pre-processing, the avoided material and substitution percentage.

Reference	Valorisation	Biomass origin	Biomass	Ash	Conversion technology	Ash pre-processing	Avoided material	Substitution rate
Modolo (2006)	Cement mortar	Cogeneration from a pulp and paper industry	Eucalyptus bark	Bottom ash	Fluidised bed	Sieving, washing and drying	Fine aggregate	30 and 100 %
Coelho (2010)	Cement mortar	Thermoelectric and cogeneration power plants	Eucalyptus bark and wastes from logging	Fly and bottom ash	Fluidised bed and grate	Drying and sieving	Cement and aggregates	10-30 %(1) 20-50 %
Esteves et al. (2012)	Cement mortar	Thermoelectric and cogeneration power plants from a pulp and paper industry	Eucalyptus wastes from logging	Fly ash	Fluidised bed and vibrating grate	Grinding, sieving, washing and drying	Cement	20-30 %
Modolo et al. (2013)	Cement mortar	Thermal power plant	Eucalyptus bark and wastes from logging	Bottom ash	Bubbling fluidised bed	Just sieving or washing and sieving	Coarse aggregates	50-100 %
Ukrainczyk (2016)	Cement mortar	Thermoelectric power plant	Unspecified forest waste from the timber industry	Fly ash	Moving grate	Non-processed	Cement and fine aggregate	10-20 %
Rosales et al. (2017)	Cement mortar	Thermal power plant	Wastes from olive trees and other crops	Bottom ash	Fluidised bed	Non-processed or crushed	Cement	20 - 38.5 %
Tosti et al. (2018)	Cement mortar	Not specified	Wood pellets, paper sludge, cacao husks, molasses	Fly ash	Fluidised bed	Non-processed	Cement	20-40 %
Rajamma et al., (2009)	Cement mortar	Thermoelectric and cogeneration power plants from a pulp and paper industry	Eucalyptus wastes from logging	Fly ash	Fluidised bed and vibrating grate	Drying and sieving	Cement	10-30 %

Table S2 (cont.) - Woody biomass ash applications, showing the biomass origin and type, the ash type, the energy conversion technology, the ash pre-processing, the avoided material and the substitution percentage.

Reference	Valorisation	Biomass origin	Biomass	Ash	Conversion technology	Ash pre-processing	Avoided material	Substitution rate
Rajamma (2011)	Cement mortar	Thermoelectric and cogeneration power plants from a pulp and paper industry	Eucalyptus wastes from logging	Fly ash	Fluidised bed and vibrating grate	Just sieving or sieving, washing and drying	Cement	10-30 %
Ramos et al. (2013)	Cement mortar	Thermoelectric power plant	Wood waste from the timber industry	Fly ash	Not specified	Grinding	Cement	10-20 %
Rajamma et al. (2015)	Cement mortar	Thermoelectric and cogeneration power plants from a pulp and paper industry	Unspecified forest waste from logging and wood processing	Fly ash	Fluidised bed and vibrating grate	Sieving	Cement	10-30 %
Modolo (2014)	Cement mortar and adhesive mortar	Thermal power plant	Eucalyptus bark and wastes from logging	Bottom ash	Bubbling fluidised bed	Just sieving or washing and sieving	Coarse aggregates	50-100 %
Modolo et al. (2015)	Adhesive mortar	Thermal power plant	Eucalyptus bark and wastes from logging	Bottom ash	Bubbling fluidised bed	Sieving	Coarse aggregates	25-100 %
Martínez-Lage et al. (2016)	Cement mortar and concrete	Cogeneration from a pulp and paper industry	Eucalyptus bark and wastes from logging	Fly ash	Not specified	Non-processed	Cement	10-30 % (2) 10-20 %
Dias (2011)	Concrete	Thermal power plant	Unspecified forest waste	Fly and bottom ash	Bubbling fluidised bed	Sieving and screening	Cement and aggregates	10-30 % (1) 20-80 %
Barbosa et al. (2013)	Concrete	Cogeneration from a pulp and paper industry	Eucalyptus and pines wastes from logging	Fly and bottom ash	Bubbling fluidised bed	Non-processed	Cement and aggregates	10-30 % (1) 9-36 %
Bastos (2014)	Concrete	Cogeneration from a pulp and paper industry	Unspecified forest waste	Bottom ash	Fluidised bed	Sieving	Aggregates	5-30 %

Table S2 (cont.) - Woody biomass ash applications, showing the biomass origin and type, the ash type, the energy conversion technology, the ash pre-processing, the avoided material and the substitution percentage.

Reference	Valorisation	Biomass origin	Biomass	Ash	Conversion technology	Ash pre-processing	Avoided material	Substitution rate
Berra et al. (2015)	Concrete	Not specified	Chestnut or poplar virgin wood chips	Fly ash	Not specified	Non-processed or washing	Filler	15-30 %
Lessard et al. (2017)	Concrete	Cogeneration from a pulp and paper industry	Unspecified forest waste	Fly and bottom ash	Fluidised bed	Non-processed	Cement and aggregates	5-15 %(1) 25-50 %
Dias (2012)	Bituminous asphalt	Thermoelectric power plant	Unspecified forest waste	Bottom ash	Fluidised bed and grate	Non-processed	Filler	5-20 %
Pinho (2014)	Asphalt	Cogeneration from a pulp and paper industry	Eucalyptus wastes from logging	Fly and bottom ash	Fluidised bed	Sieving	Soil	25 e 50 %

(1) First percentage for cement replacement and second for aggregate replacement.

(2) First percentage for mortars production and second for concrete production.

Table S3 - Ecoinvent processes used for the valorisation scenarios.

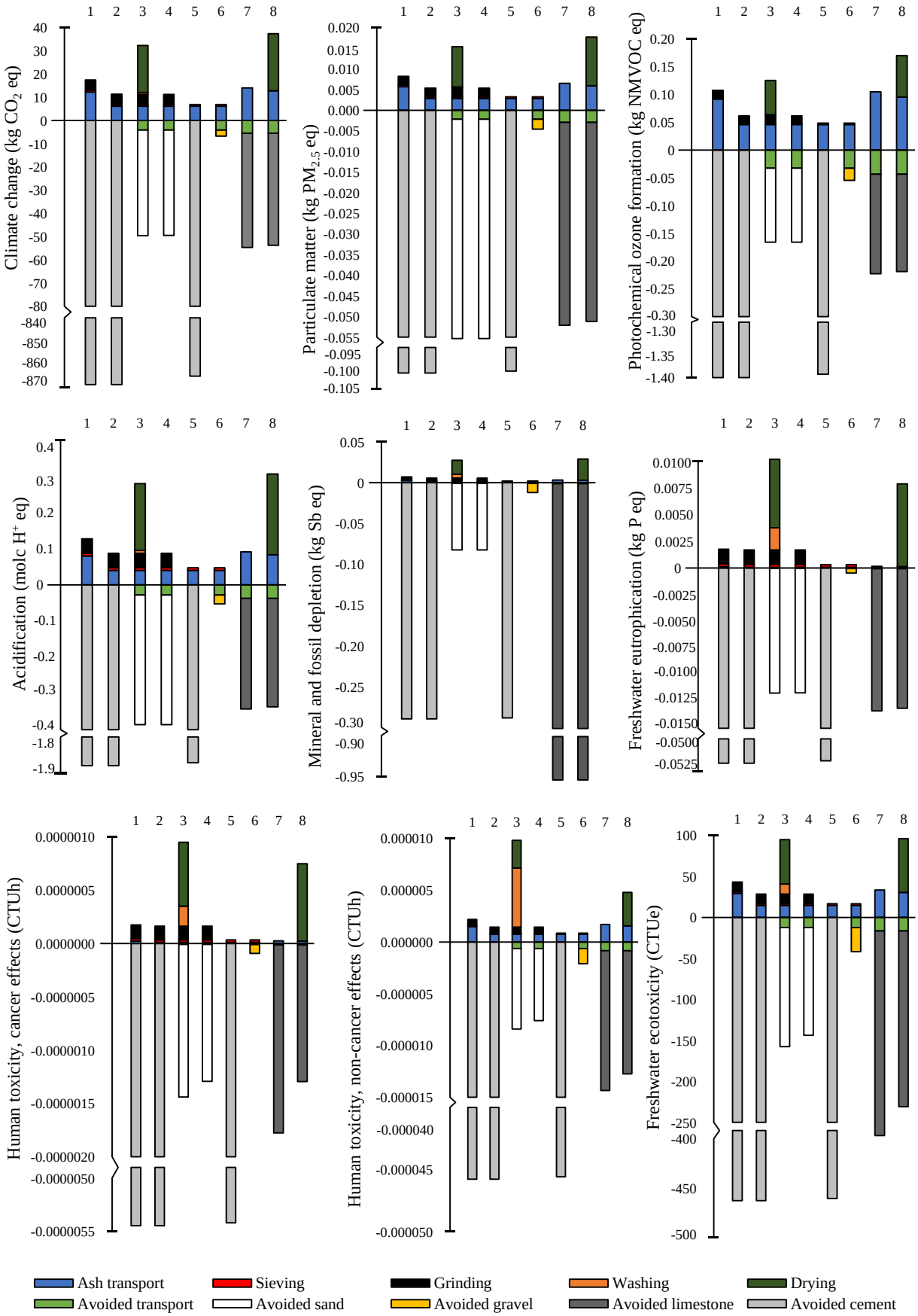
Scenarios/processes	Input material	Ecoinvent processes
Ash pre-processing and transportation	Electricity, medium voltage	Electricity, medium {PT}| market for | APOS, U
	Water	Water, unspecified natural origin, PT
	Transport, freight, lorry	Transport, freight, lorry 7.5-16 metric ton, EURO3 {GLO}| market for | APOS, U
Cement mortar with woody biomass ash	Portland cement	Cement, Portland {Europe without Switzerland}| market for | APOS, U
	Silica sand	Silica sand {GLO}| market for | APOS, U
	Electricity, medium voltage	Electricity, medium {PT}| market for | APOS, U
	Water	Water, unspecified natural origin, PT
Adhesive mortar with woody biomass ash	Portland cement	Cement, Portland {Europe without Switzerland}| market for | APOS, U
	Limestone	Lime {GLO}| market for | APOS, U
	Copolymer	Ethylene vinyl acetate copolymer {RER}| production | APOS, U
	Electricity, medium voltage	Electricity, medium {PT}| market for | APOS, U
	Water	Water, river, PT
Concrete blocks with woody biomass ash	Portland cement	Cement, Portland {Europe without Switzerland}| market for | APOS, U
	Gravel	Gravel, round {GLO}| market for | APOS, U
	Tap water	Tap water {Europe without Switzerland}| market for | APOS, U
	Heat, natural gas	Heat, district or industrial, natural gas {Europe without Switzerland}|market for heat, district or industrial, natural gas | APOS, U
	Heat, other than natural gas	Heat, district or industrial, other than natural gas {Europe without Switzerland}|market for heat, district or industrial, other than natural gas | APOS, U
	Diesel	Diesel, burned in building machine {GLO}| market for | APOS, U
	Electricity, medium voltage	Electricity, medium {PT}| market for | APOS, U
	Concrete block	Concrete block {GLO}| market for | APOS, U

Table S3 (cont.) - Ecoinvent processes used for the valorisation scenarios.

Scenarios/processes	Input material	Ecoinvent processes
Bituminous asphalt with woody biomass ash	Bitumen	Pitch {GLO}| market for | APOS, U
	Limestone	Lime {GLO}| market for | APOS, U
	Sand	Sand {GLO}| market for | APOS, U
	Electricity, medium voltage	Electricity, medium {PT}| market for | APOS, U
	Diesel	Diesel, burned in building machine {GLO}| market for | APOS, U
	Heat, other than natural gas	Heat, district or industrial, other than natural gas {Europe without Switzerland}|market for heat, district or industrial, other than natural gas | APOS, U

Within the intermediate processes, the electricity was modified with Portuguese (PT) electricity mix (37.5 % hydropower, 28 % wind, 27 % hard coal, 3.6 % natural gas, 2.3 % oil and 1.6 % wood, for the year 2014).

Figure S1 - Impact assessment results for valorisation scenarios disregarding the common processes (1 to 8).



Appendix B: Life cycle assessment of woody ash valorisation for soil amelioration

Table S4 - Inventory data for the landfarming of woody biomass ash per FU.

	FG	FF
Inputs:		
Ash (t)	1.00	1.00
Diesel (L)	0.546	0.517
Emissions to air		
CO_2 (kg)	1.41	1.34
CO (g)	2.72	2.57
CH_4 (mg)	58.7	55.6
NO_X (g)	20.7	19.6
N_2O (mg)	54.6	51.7
NM_{VOC} (g)	1.36	1.29
$PM_{2.5}$ (g)	1.93	1.83
SO_2 (g)	0.459	0.434
Emissions to soil		
C (kg)	54.2	28.7
S (kg)	6.50	11.0
Al (kg)	49.4	60.2
As (g)	24.0	17.0
Ba (kg)	0.470	0.510
Cd (g)	3.50	3.33
Ca (kg)	146	135
Cl (kg)	17.5	14.0
Cr (g)	38.5	67.2
Cu (g)	51.4	78.7
Fe (kg)	22.8	23.4
Pb (g)	40.6	77.6
Mg (kg)	21.9	17.3
Mn (kg)	3.10	1.83
Hg (g)	0.500	0.200
Mo (g)	0	3.00
Ni (g)	86.1	39.4
P (kg)	5.79	8.07
K (kg)	30.8	27.1
Na (kg)	9.09	9.28
Si (kg)	149	203
Sr (kg)	0.340	0.265
Ti (kg)	2.25	3.83
V (g)	60.0	64.0
Zn (kg)	0.207	0.193

FG – fly ash from grate furnace
FF – fly ash from fluidised bed

Table S5 - Trace elements in traditional liming and fertiliser products substituted.

Composition (g/kg)	Liming products	Potassium chloride	Potassium sulphate	Potassium nitrate	Calcium nitrate	Single superph.	Triple superph.
Al	1.6E-01	N/A	N/A	N/A	N/A	2.0E+00	2.8E+00
As	1.9E-02	9.0E-05	5.2E-03	1.0E-03	6.2E-03	1.2E-02	1.1E-01
Ba	1.4E-01	N/A	1.5E-02	1.0E-03	N/A	2.2E-01	6.8E-02
Cd	4.1E-04	2.0E-05	1.9E-03	1.0E-04	1.4E-03	9.9E-03	1.2E-02
Cl	N/A	4.8E-01	1.9E-02	N/A	N/A	N/A	N/A
Cr	9.4E-03	2.0E-05	2.4E-03	5.0E-03	1.7E-02	7.4E-02	1.5E-01
Cu	2.6E-02	4.9E-04	1.5E-03	1.0E-03	1.0E-02	5.9E-02	5.5E-02
Fe	6.5E+00	N/A	N/A	N/A	N/A	5.1E+00	2.2E+00
Hg	3.0E-05	N/A	2.0E-04	1.0E-05	1.2E-03	5.5E-04	1.6E-03
Mn	9.6E-01	1.7E-03	4.5E-03	N/A	2.8E-02	1.1E-01	1.3E-01
Mo	2.7E-02	N/A	6.5E-03	3.0E-04	6.0E-03	1.1E-02	1.4E-02
Na	1.2E-01	1.5E-02	1.1E-02	N/A	N/A	N/A	N/A
Ni	1.1E-02	1.6E-04	1.4E-02	3.0E-03	2.7E-02	2.7E-02	3.0E-02
Pb	2.8E-01	6.3E-04	9.4E-03	1.5E-03	5.2E-03	8.4E-03	7.8E-03
S	1.4E+00	N/A	N/A	N/A	N/A	N/A	N/A
Sc	1.0E-03	N/A	5.0E-06	N/A	N/A	4.1E-04	6.1E-04
Sn	6.0E-03	N/A	2.0E-03	1.0E-04	6.1E-03	5.4E-03	2.6E-03
Sr	3.0E-01	N/A	5.0E-02	1.0E-03	2.0E-02	4.8E-01	3.7E-01
Ti	1.3E-02	N/A	9.5E-04	N/A	4.5E-04	7.3E-02	8.0E-02
V	1.8E-02	N/A	9.1E-02	1.0E-03	3.7E-03	1.4E-02	5.0E-02
Zn	1.7E-01	4.4E-03	4.0E-03	1.0E-03	2.2E-02	2.1E-01	2.0E-01
References	a, b, c	d, e	c, d	b	c	b, c, f	b, c, f

N/A – Not available
[a] Blunden et al. (1997)
[b] McBride and Spiers (2001)
[c] Senesi et al. (1999)
[d] Ecolan (2012)
[e] Uprety et al. (2009)
[f] Kratz et al. (2016)

Table S6 - Amount of liming products substituted equivalent to the application of 1 t of woody biomass ash.

Liming products	Amount substituted (kg)	
	FG	FF
Limestone	219.4	195.3
Quicklime	122.6	109.1
Lime hydrated	162.5	144.7

Table S7 - Amount of traditional fertiliser substituted equivalent to the application of 1 t of woody biomass ash.

Commercial fertiliser	Amount substituted (kg)	
	FG	FF
Potassium chloride	48.24	42.49
Potassium sulphate	57.88	50.99
Potassium nitrate	65.78	57.94
Single superphosphate	22.74	31.71
Triple superphosphate	3.566	4.971

Table S8 - Transport profile.

Material transport	Baseline distance (km)	Sensitivity analysis distance (km)	Type of transport	Load (t)
Power plant to landfill				
FG	15[a]	7.5-30	Freight lorry, EURO 3	7.5-16
FF	15[a]	7.5-30	Freight lorry, EURO 3	7.5-16
Power plant to landfarm				
FG	35[b]	17.5-70	Freight lorry, EURO 3	7.5-16
FF	35[b]	17.5-70	Freight lorry, EURO 3	7.5-16
Manufacturing unit to landfarm				
Liming products	60[c]	30-90	Freight lorry, EURO 3	7.5-16
Fertilisers	200[c]	100-400	Freight lorry, EURO 3	7.5-16

[a]The transport distances of woody biomass ash from the power plant to landfill were based on expert judgment, since the power plants under study deposit their ashes in a nearby landfill or in its own landfill.

[b]Optimal distances from an economic point of view were considered for woody biomass ash transportation from the power plant to landfarming (da Costa et al., 2018).

[c]The transport distance to the landfarming was based on distances of existing manufacturing units in Portugal.

Table S9 - Amounts of traditional fertilisers substituted in the sensitivity analyses equivalent to the application of 1 t of woody biomass ash.

Commercial fertiliser	Nutrient bioavailability in the fertiliser (%)	Nutrient bioavailability in the ashes (%)	Amount substituted (kg)	
			FG	FF
Potassium chloride	0.70	0.01	24.27	21.38
	0.65	0.20	74.05	65.23
Potassium sulphate	0.70	0.01	29.12	25.65
	0.65	0.20	88.86	78.27
Potassium nitrate	0.70	0.01	33.09	29.15
	0.65	0.20	100.90	88.95
Single superphosphate	0.46	0.18	1.53	2.137
	0.16	0.51	88.13	122.90
Triple superphosphate	0.93	0.18	0.33	0.46
	0.80	0.51	7.71	10.75

Appendix C: Consequential LCA of energy production from forest residues

C.1 Size of market changes

The time scope of the decision to increase the supply of bioenergy is considered to be short-term (up to 5 years after the implementation of the decision), since this decision creates small changes, which means that the annual amount of additional supply is smaller than the average percentage of annual replacement of capacity (assumed to be 5 %) (EC/JRC/IES, 2010b). The trends in the production of electricity, industrial heat and fuels for road transportation in Portugal is shown in Figure S2.

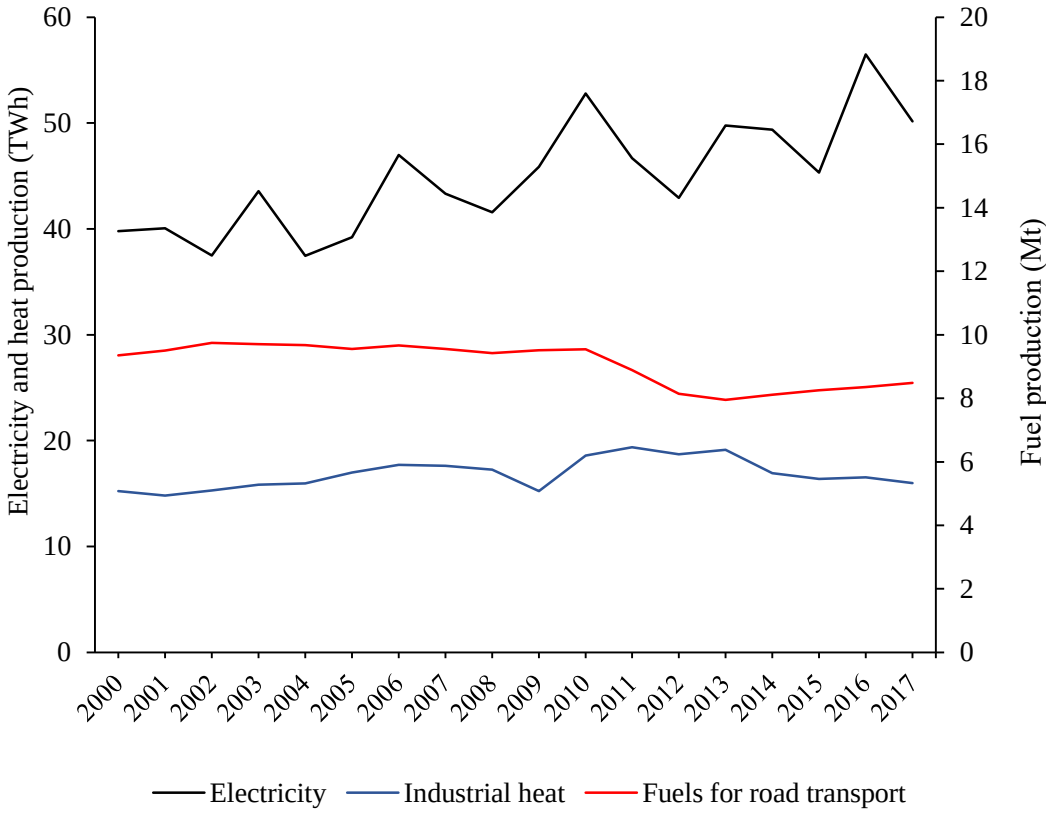


Figure S2 - Trends in the Portuguese market for the supply of electricity, industrial heat and fuel for road transportation from 2000-2017. Source: adapted from APREN (2018) and DGEG (2019).

The market for electricity production is growing. In the first scenario (electricity production from forest biomass residues) evaluated in this study, the additional supply of 475 GWh of electricity would represent an increase of 1.0 % in the average electricity produced in the country in 2017, which can be considered a small change.

The market trend for industrial heat production is relatively stable over the years. In the second scenario (cogeneration of electricity and heat), the additional supply of 1140 GWh of heat represents an increase of 4.8 % in the total industrial heating in the country in 2017, which is also a small change.

The market trend for fuels for road transportation slightly declined between 2010 and 2013, but from 2014 to 2017 the market remained stable. In the third scenario (bioethanol production), the production of 145 kt of bioethanol represents an increase of 1.5 % in the production of fuels for road transportation in the country in 2017 and, thus, can also be considered a small change.

C.2 Market trend and marginal suppliers of mineral fertilisers

A consequence of collecting the forest biomass residues to produce bioenergy leads to changes in the demand for fertilisers. In a "growing, stable, or slightly declining" market, an additional demand can be met by the most cost-efficient technology. The marginal nitrogen, phosphorous and potassium fertilisers considered were urea, triple superphosphate and potassium chloride, as these fertilisers correspond to the most consumed ones in Portugal during the period between 2008-2011 shown in Figure S3 (FAO, 2017). Latest data is not available, so it was assumed that this trend is valid until nowadays.

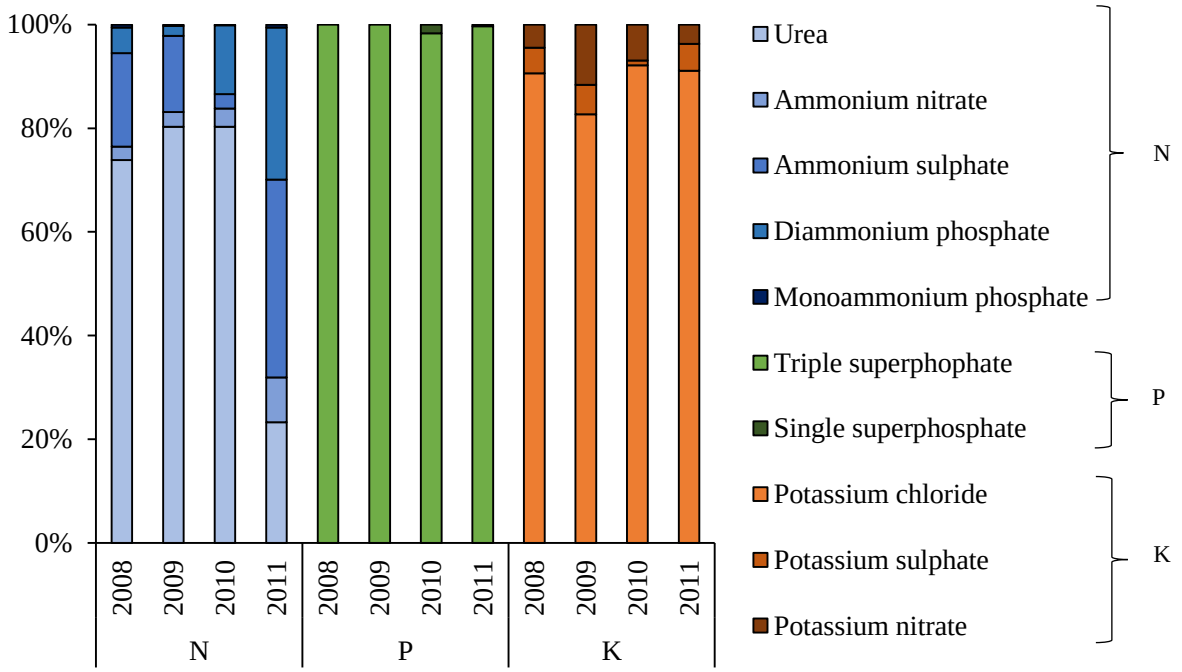


Figure S3 - Portuguese consumption of fertiliser from 2008-2011. Source adapted from FAO (2017).

C.3 Market trend and products displaced

C.3.1 Electricity production

The Portuguese electricity mix has the contribution of several energy sources, as shown in Figure S4, with renewable sources playing an increasingly role after 2003.

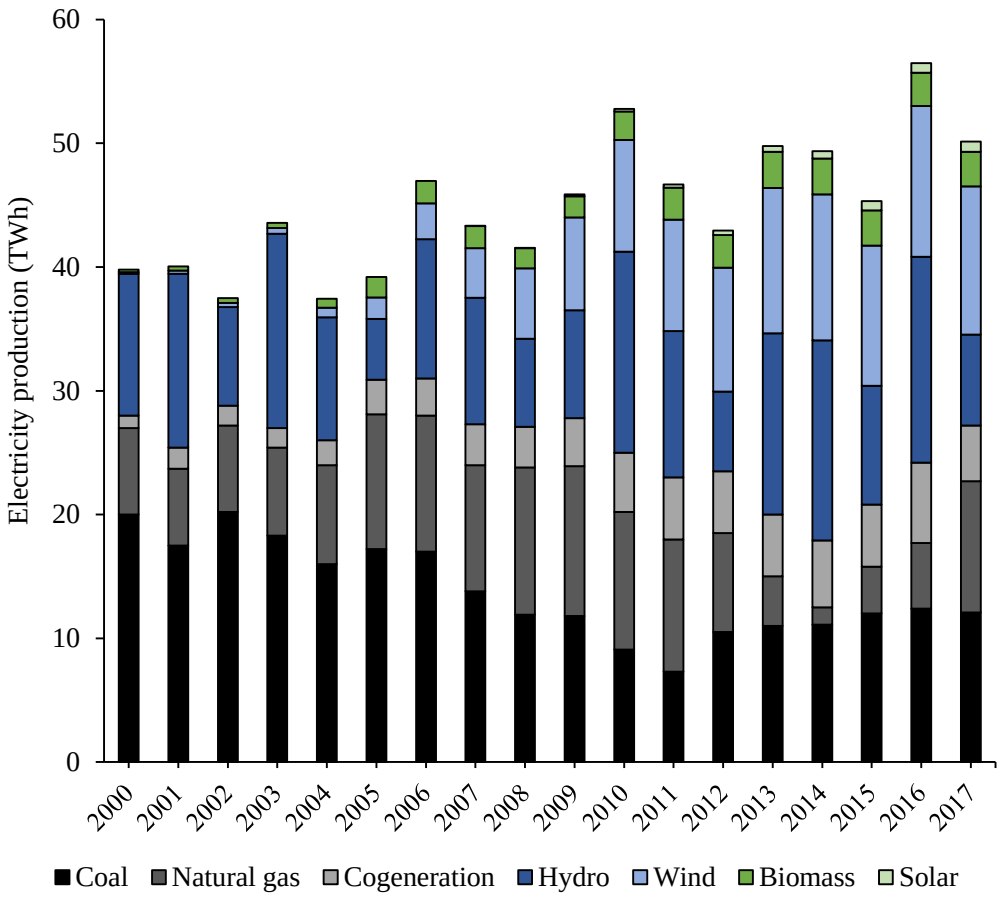


Figure S4 - Indicative trajectory of electricity production from 2000-2017. Source: adapted from APREN (2018).

The consequential modelling is characterised by excluding constrained suppliers in order to find the appropriate substitute (Schmidt et al., 2011). For short-term changes, the immediately affected suppliers are typically be the least competitive (often using older technology), since these suppliers are most sensitive to price changes (Weidema, 2015).

Garcia and Freire (2016) identified the unconstrained technologies operating in the Portuguese electricity system: coal and natural gas. The reasons for this choice are because natural gas power plants are flexible regarding adjusting power and coal power plants are used both for baseload and load-following in Portugal.

Garcia and Freire (2016) also observed that in low demand hours, coal was the dominant marginal technology in Portugal, with about 84 % of the share, whilst natural gas dominated at high demand hours with 66 % of the share. For calculation purposes, in this study it was used the average demand hours between 2012 and 2014, which means that the low demand hours represented 52 % of the time and high demand hours represented 48 %. It was assumed that this trend is valid until nowadays.

C.3.2 Industrial heating

The second scenario also leads to a change in supply and therefore, it is necessary to identify the substitute for heat supply which fulfils the same functions. Figure S5 shows the sources of heat supplied to the Portuguese industrial sector (DGEG, 2019).

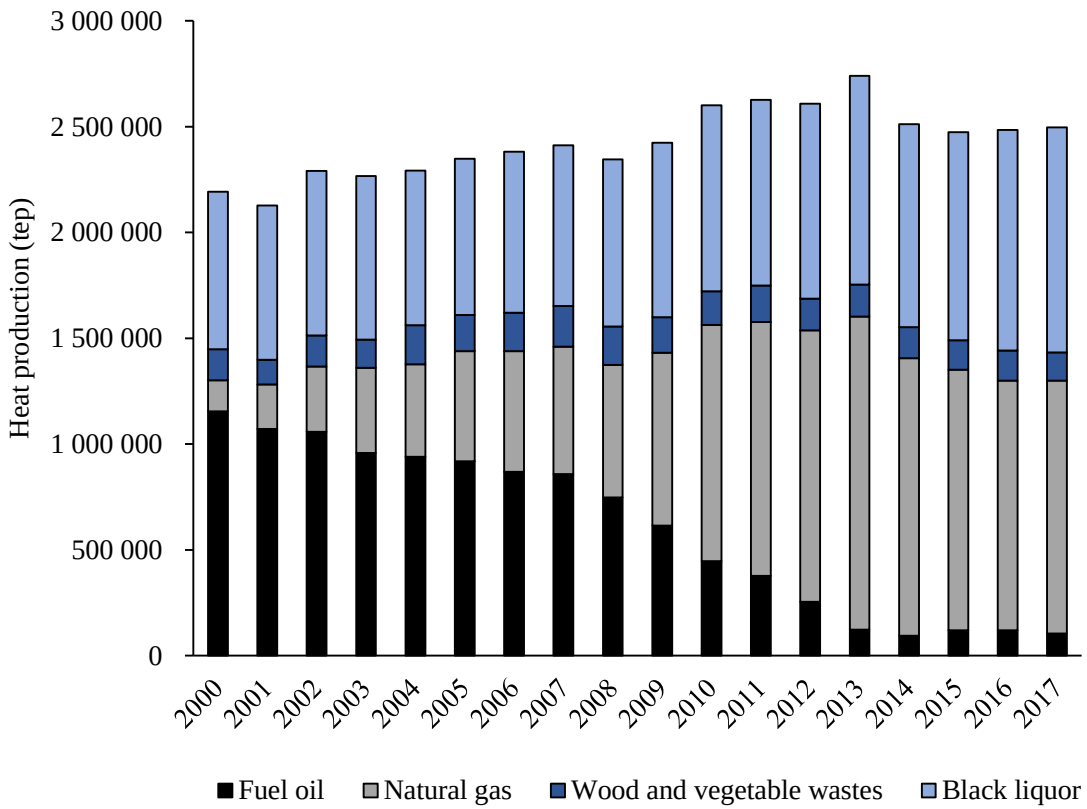


Figure S5 - Heat cogeneration in Portugal from 2000-2017. Source: adapted from DGEG (2019).

A change in heat supply does not affect all heat generators, but only those that can respond to the change, i.e. the unconstrained generators. Natural gas and fuel oil are flexible with regard to adjusting power output (i.e. can be produced without constraints) and can therefore both be

considered reasonable alternatives. Wood and vegetable wastes and black liquor are constrained sources, since they are by-products and do not respond to changes in the market. For calculations purposes, the average share of natural gas and fuel oil over the last five years (2013-2017) will be used to assume the contribution ratio of the marginal generation, since the trend has remained constant during this period. Therefore, the share between the energy sources are 92 % of natural gas and 8 % of fuel oil.

C.3.3 Marginal gasoline production

As in the other scenarios, the decision to produce bioethanol leads to changes in supply, which implies to choose replaced product(s). The Portuguese road transport has been satisfied by the forms of energy shown in Figure S6 (DGEG, 2019).

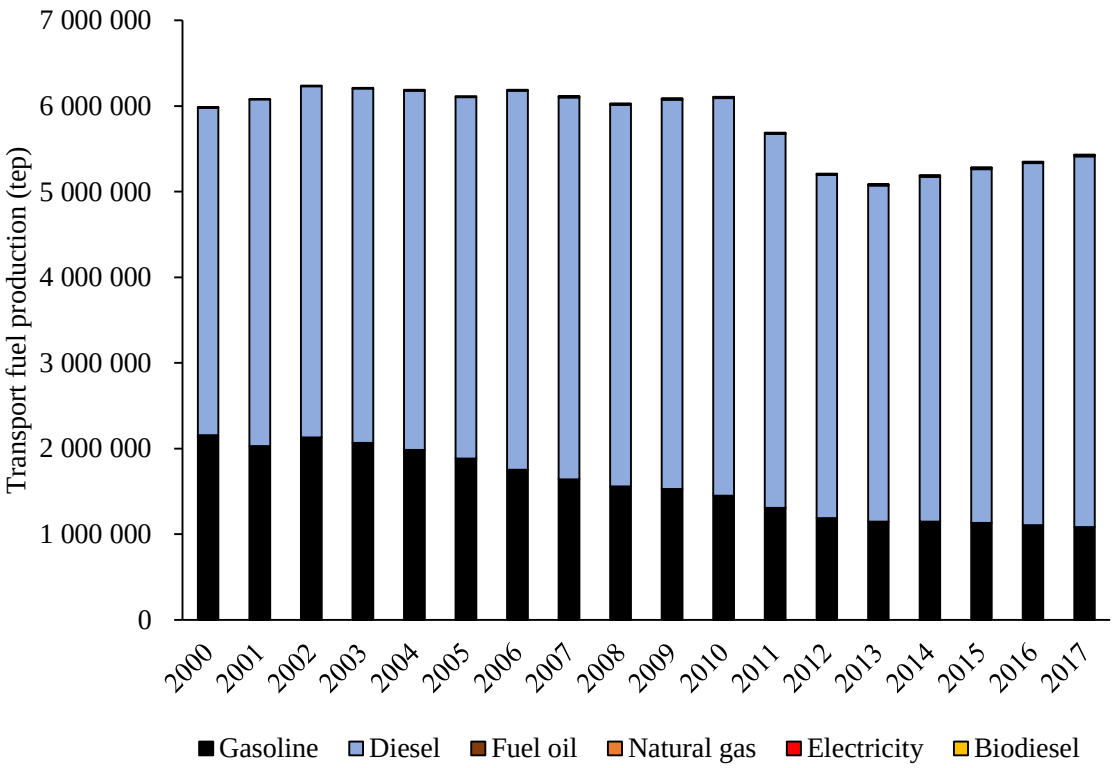


Figure S6 - Portuguese road transport balance from 2000-2017. Source: adapted from DGEG (2019).

Gasoline was assumed to be the marginal fuel (least competitive), because its consumption has been steadily decreasing since year 2000, while diesel consumption is slightly increasing. This behavior is observed due to the policies, which over the years have favored the diesel against the gasoline, e.g. paying less taxes. The strategy followed by most European governments since 1970 has been to encourage the use of diesel engines in both passenger and

commercial light vehicles, because the diesel engines are not only more efficient, but their consumption is reduced (Cames and Helmers, 2013).

C.3.4 Gypsum

The bioethanol production also leads to the co-production of gypsum. The global gypsum market shows that the consumption of gypsum is increasing (Soezen, 2018); therefore, it is necessary to identify the substitute for the gypsum which fulfils the same functions. The gypsum produced through the biochemical conversion of forest biomass residues can be used in concrete and wallboard manufacturing (Martinez et al., 2001). Therefore, it was considered that it replaces the ordinary gypsum produced from mining.