Universidade de Évora - Escola de Ciências e Tecnologia
Universidade de Lisboa - Instituto Superior de Agronomia

Mestrado em Gestão e Conservação de Recursos Naturais

Dissertação

Does community scale composting produce a viable outcome? Some physical and chemical properties of green waste composts produced in the Faculty of Sciences campus

Madalena Nunes França Aires Horta

Orientador(es) | Florian Ulm
Cristina Ferreira da Cunha Queda
Gil Pessanha Penha Lopes

Évora 2021

Universidade de Évora - Escola de Ciências e Tecnologia
Universidade de Lisboa - Instituto Superior de Agronomia

Mestrado em Gestão e Conservação de Recursos Naturais

Dissertação

Does community scale composting produce a viable outcome? Some physical and chemical properties of green waste composts produced in the Faculty of Sciences campus

Madalena Nunes França Aires Horta

Orientador(es) | Florian Ulm
Cristina Ferreira da Cunha Queda
Gil Pessanha Penha Lopes

Évora 2021

A dissertação foi objeto de apreciação e discussão pública pelo seguinte júri nomeado pelo Diretor da Escola de Ciências e Tecnologia:

Presidente	Pedro R. Almeida (Universidade de Évora)
Vogais	Cristina Maria Nobre Sobral de Vilhena da Cruz Hougton (Universidade de Lisboa - Faculdade de Ciências) (Arguente) Florian Ulm (Universidade de Lisboa) (Orientador)

Évora 2021

Acknowledgments

Special thanks go to Florian Ulm for his scientific orientation, help and support during all the process as well as friendship. Many thanks also to Gil Penha-Lopes, Cristina Cunha-Queda and Cristina Cruz for their scientific orientation and support that allowed the work to be carried out. Also, I would like to thank to all volunteers and guardians of the HortaFCUL permaculture project for their help during the composting process, without them it would not be possible. Special thanks to my friends Catarina and Luís for helping me with their programming skills, Mariana for helping me to improve my text and Filipa to improve my graphs. Thanks to all my family and my friends for the phycological support during the entire thesis, especially my mother and brother.

Table of contents

Resumo

A compostagem a uma escala comunitária produz um composto viável? Avaliação de propriedades físicas e químicas de compostos de resíduos verdes produzidos no campus da Faculdade de Ciências

A compostagem pode ser definida como o processo de biodegradação de resíduos orgânicos realizado por comunidades microbianas em condições aeróbias, sendo uma forma sustentável de gerir estes resíduos no contexto de uma economia mais circular. Neste trabalho foi analisado um sistema de compostagem a uma escala comunitária. Para cada pilha, as matérias-primas e os compostos foram pesados e as temperaturas monitorizadas semanalmente. Os parâmetros físico-químicos foram analisados, bem como o teor de inertes, e foram realizados testes fitotóxicos. Os resultados mostraram que as amostras de composto cumpriam a maioria dos requisitos definidos pelas normas legais portuguesas em relação à qualidade dos mesmos, exceto no teor de humidade e de pedras. No entanto, os testes de maturação indicaram que todos os produtos finais estavam consistentemente maturados. Os resultados obtidos demonstraram que é possível gerir os resíduos orgânicos dos espaços verdes através da compostagem local, com benefícios ao nível ecológico e social.

Palavras-chave: Gestão de resíduos; Compostagem de resíduos verdes, propriedades físico-químicas, critérios de qualidade, compostagem à escala comunitária

Abstract

Does community scale composting produce a viable outcome? Some physical and chemical properties of green waste composts produced in the Faculty of Sciences campus

Composting is the biodegradation process of organic substrates carried out by microbial communities, under aerobic conditions. It is a sustainable way to manage biodegradable waste within a context of a more circular economy. In this work, a community-scale green waste composting system was under study. For each pile, feedstocks and composts were weighted and temperatures were monitored weekly. Physicochemical parameters were analysed, phytotoxic tests were performed and the inert material content was assessed. Results showed that the compost samples fulfilled the majority of the requirements set by the Portuguese statutory standards for compost quality, except for moisture and stone content. However, maturity tests indicated all final products as consistently mature. Additionally, the particle size dimensions of the final composts were suitable for both of the main uses. The results showed that is possible to manage organic waste from the green areas through local composting with ecological and social benefits associated.

Key-words: Waste management; Green waste compost, Physicochemical properties, Quality standards, Community-scale composting

List of tables

List of figures

1. Introduction

1.1. Evolution and current situation of MSW management in Portugal

1.1.1. National Context

As a reflection of diverse waste management practices, the definition of municipal solid waste (MSW) used in different countries varies. According to the European Environment Agency's 2013 report (EEA, 2013) '*municipal* [solid] *waste is mainly produced by households, though similar wastes from sources such as commerce, offices and public institutions are included. The amount of municipal* [solid] *waste generated consists of waste collected by or on behalf of municipal authorities and disposed of through the waste management system'*.

Until the implementation of the first PERSU (Portuguese National Plan for Municipal Solid Waste) in 1997, the MSW management in Portugal was based on undifferentiated collection and open-air waste disposal sites. The main objectives of PERSU I were to eliminate open dumps in a 10-year period and implementing strategies for biodegradable waste recovery through recycling and composting. The active dumpsters in Portugal were closed and replaced by incinerators, infrastructures more appropriate to the treatment of MSW (Santos, 2007). A network of multi-municipal and inter-municipal systems of MSW management was created, together with multi-material selective collection systems and MSW valorisation structures (Ministério do Ambiente, 2007).

However, there was no reduction in the amount of waste production, and recycling and composting have not reached the levels established in PERSU I (Ribeiro *et al.*, 2011). According to European Environment Agency's 2013 report (EEA, 2013), PERSU II was ratified [in 2006] in order to '*eliminate some inefficiencies observed in the implementation of the previous plan'*. The strategic guidelines presented in PERSU II (EEA, 2013) were:

- Reducing, reutilization and recycling of products;
- Promoting incineration with energy recovery and mechanical biological treatment as the solution to biodegradable waste treatment;

- Introducing separate collection of organic wastes and other measures to divert them from landfills;
- Maximizing by-products utilization;
- Applying the ‘Kyoto protocol’ as central commitment in waste management policies;
- Promoting sustainable waste management systems;

1.1.2. PERSU 2020 – Portuguese National Plan for Municipal Solid Waste 2014-2020

The PERSU 2020 maintains as a central objective the protection of the environment and human health through an appropriate use of technologies and infrastructures, although it goes further by promoting the principle of circular economy as a guideline for waste management:

"[PERSU 2020 will] *Promote residue management integrated in a life cycle of products, centred in a tendentiously circular economy and that guarantees a greater efficiency in the use of natural resources"* (APA, 2014)

The main guidelines for municipal solid waste management in PERSU 2020 are (APA, 2014):

- Waste managed as endogenous resources, minimizing their environmental impacts and valuing them socioeconomically;
- Efficiency in use and management of primary and secondary resources, dissociating economic growth from materials consumption and waste production;
- Progressive elimination of landfill disposal, and eradication of municipal waste disposal in landfill until 2030;
- Involvement of citizen in the urban waste management strategies;
- Promoting waste prevention and the reintroduction of non-use resources in new productive processes, thus reducing the need for new resources extraction and creating new value-added activities;

1.1.3. Composition of Municipal Solid Waste in Portugal

The composition of municipal waste varies according the region, the season, the socioeconomic level and consumption habits (Santos, 2007). According to the 2017 annual report on urban waste (Marçal & Teixeira, 2017) around 50.5% of total urban waste corresponded to biodegradable waste (biowaste, paper/cardboard and green waste) and 71.7% corresponded to the fraction that is recyclable (glass, paper and cardboard, metals, plastic, textiles and biowaste) and could be reintroduced in new productive process.

1.2. Biodegradable Waste

According to European Parliamentary Research Service's 2017 report on Waste Management in EU (European Parliamentary Research Service, 2017), bio-waste and residues include not only biodegradable garden and park waste, food and kitchen waste from households, but also, agricultural, forestry, marine and animal derived residues. These waste streams have always been considered a challenge. However, with the take up of new technologies, they are being re-categorised either as feedstock, raw material or energy, within the context of a more circular economy (European Parliamentary Research Service, 2017). Furthermore, waste streams have not been covered specifically in the European legislation, except for food waste, in terms of targets for separation and reduction. However, they are covered by the Landfill Directive (Council Directive 1999/31/EC, 1999), which exerts diversion of biodegradable waste from landfills and consequently impacts policies, such as landfill taxes. According to the waste management hierarchy, landfill disposal is the least preferable option and should be limited to the necessary minimum (European Parliamentary Research Service, 2017).

1.2.1. Organic Recycling

Organic recycling is defined by the EU Packaging and Packaging Waste Directive 94/62/EC (Council Directive 94/62/EC, 1994, amended in 2005/20/EC, 2005) as the aerobic treatment (composting) or anaerobic treatment (biogasification) of organic waste. The aerobic treatment is performed by microorganisms under controlled conditions and the final product of this decomposing process is a stabilized humus-like substance that can be used as soil amendment, growing medium or as mulch material. The anaerobic treatment is also performed through the action of microorganisms, although in the absence of oxygen. In this process the organic matter is converted into biogas, a gaseous mixture mainly composed of methane and carbon dioxide (Lyberatos & Skiadas, 1999). According to Mao *et al.* (2015) the anaerobic digestion cycle represents an integrated system of a physiological process of microbial and energy metabolism, as well as raw materials processed under specific conditions. The remaining sludge contains many nutrients and can be used in agriculture after an aerobic post-composting.

1.2.2. Composting

A way of processing organic waste is through composting. It consists of aerobic biological decomposition of organic substance by means of diversified microorganisms (Maheshwari, 2014). The end product of the decomposing process is a humus-like substance, which can be used as a stimulant that restore soil properties as well as an organic fertilizer (Maheshwari, 2014).

From a practical point of view, composting can be seen as a tool to recycle inputs, biomass and nutrients, available in the farm/garden and to reduce off-farm/garden inputs, such as mineral fertilizers. Compost also has beneficial effects on:

- Soil chemical and physical characteristics, such as water holding capacity and soil structure (Khaleel *et al.*, 1981);
- Soil's cation exchange capacity, which provides a buffer against acidification and improve nutrient availability to plants (Harada & Inoko, 1980; Rivero *et al.*, 2004; Feller *et al.*, 2010; Oldfield *et al.*, 2018; Mekki *et al.*, 2019);
- Soil organic matter (SOM) quantity and quality. Compost can be used as soil amendment, consequently increasing SOM concentrations while sequestering carbon (Rivero *et al.*, 2004; Mekki *et al.*, 2019);
- Soil biochemical and biological indicators such as microbial biomass and soil enzyme activities, which are potentially involved in biogeochemical cycles and can have great influence on plant productivity parameters (Pérez-Piqueres *et al.*, 2017);
- Soil microbial communities (Perucci *et al.*, 2000; Debosz *et al.*, 2002; Pérez-Piqueres *et al.*, 2006), by increasing the competition between soil native microorganisms and the ones in the compost which seems to lower the soilborne pathogens load on plants (Curl & Old, 1988; Abawi & Widmer, 2000; van Bruggen & Semenov, 2000).

Soilborne diseases are more severe when the soil conditions are poor, with an inadequate drainage, poor soil structure, low organic matter and high compaction (Curl & Old, 1988; Abawi & Widmer, 2000). Therefore, the implementation of management options for enhancing soil quality and health, such as compost application, can have a direct impact on the physical characteristics of the soil while promoting soil biota diversity (Pérez-Piqueres *et al.*, 2017).

1.2.2.1. The Microbial Transformation of Raw Materials

The composting process can be defined as a biodegradation process of a mixture of substrates carried out by a microbial community, in aerobic conditions and in the solid state (Diacono & Montemurro, 2011; Maheshwari, 2014). Under controlled conditions, this leads to the transformation of raw organic materials into biologically stable substances, which differs from the outcome of natural rotting or putrefaction (Ryckeboer *et al.*, 2003). The process includes many microorganisms and their activity is the central key of nutrient cycling, while raw materials play a functional role as microorganism feedstock. Microorganisms found in compost piles include, the ones that perform the composting process, and others that are potentially harmful for the environment as well as human, animal and plant health. However, with favourable conditions of aeration, humidity and an appropriate C/ N ratio, the process leads to the ‘inactivation’ of harmful microorganisms and the growth of beneficial ones (Fuchs, 2010). Furthermore, composting can be considered a set of processes, which entails a number of complex chemical reactions and microbiological transformations, including hydrolysis, ammonification, nitrification, C mineralization and humification (Cáceres *et al.*, 2018).

1.2.2.2. Composting phases and microbial communities

The biological decomposition of organic substrates and organic matter maturation drive the process all through different phases distinguished by time and temperature values.

The composting process includes five phases (figure 1): the latent phase, which corresponds to the time necessary for the microorganisms to acclimatize and colonize in the new environment in the compost heap; the mesophilic growth phase (up to ≈ 42 °C), which is strongly influenced by the raw materials characteristics, moisture and aeration conditions; the thermophilic phase (≈ 45–68 °C) that depends on the nature of the C compounds in the composted materials; the second mesophilic phase, which takes place when mesophilic microorganisms recolonize the substrate; and the maturation or curing phase, which follows the active phase, and is characterized by a slow and progressive temperature decrease (Maheshwari, 2014). During this final phase, the biomass loses the residual phytotoxicity, while microbial population reaches a dynamic equilibrium and the synthesis of humic substances occurs. The length of any composting phase depends essentially on the type of feedstock, moisture content and aeration conditions (Ryckeboer *et al.*, 2003). In the following paragraphs, each phase will be described in detail.

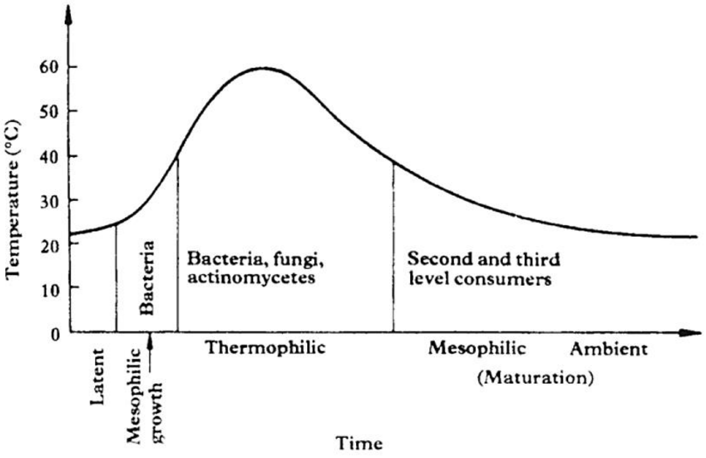

Figure 3 - Patterns of temperature and microbial growth in compost piles
(*in* Maheshwari, D.K. (Ed). 2014. Composting for sustainable agriculture. Springer)

The self-selection of microbial communities in the composting process it is based on continuous increasing of the autochthonous microorganisms, indigenous forms of soil microbial communities, to the detriment of the exogenous, introduced ones (Maheshwari, 2014).

First mesophilic phase

This initial stage starts with the activity of mesophilic zymogenous microorganisms, which are already present in raw materials, first level consumers such as bacteria, fungi and actinomycetes (Maheshwari,

2014) whose presence is transient and fluctuating. The rapid decomposition of soluble and easily degradable substrates results in the synthesis of organic acids, which are responsible for decreasing pH to acidic values during the first days of the composting process (Beffa *et al.*, 1996). Fungi and yeasts take advantage of this environmental conditions until the ammonification process increases the pH, which consequently promotes bacterial metabolism. Ammonification is one of the initial steps of N mineralization, consisting in the release of NH_4^+ from organic N (Cáceres *et al.*, 2018). The mean generation time is shorter for bacteria than for fungi, which means that bacteria can better adapt to the rapidly mutable environment than fungi. As a result, bacteria are responsible for the compost heat production and for most of the initial decomposition (Ryckeboer *et al.*, 2003).

Thermophilic phase

The thermophilic phase starts when the temperatures in the compost pile rapidly increases (to ≈ 45–68 °C within 24–72 hours after pile formation) and thermophilic microorganisms start to dominate. In this active thermophilic phase, the temperatures are high enough to eliminate pathogens, break down phytotoxic substances and devitalize weed seeds (Maheshwari, 2014). The high temperature during the active composting phase is a result of the microbial activity, together with the thermal insulation of the pile itself. At the same time, the high temperature is one of the main drivers that affects the composition of the microbial communities (Fuchs, 2010).

High temperatures in this phase accelerate the breakdown of proteins, fats, and complex carbohydrates (like cellulose and hemicelluloses). Degradation of complex organic compounds, like lignin, are mainly performed by thermophilic fungi and actinomycetes. The optimum temperature for these microorganisms is 40±50°C, which is also the optimum temperature for lignin degradation in compost (Tuomela, 2000). However, if the temperature becomes too high (greater than 65–70 °C), fungi, actinomycetes, and most bacteria become inactive and only spore forming bacteria can develop. With increase of temperature the odorous components increases (Maheshwari, 2014) so oxygen must be induced through passive, forced aeration or by turning the compost pile (Gonzalez & Cooperband, 2002).

Second mesophilic phase and maturation

When most of the degradation has taken place, the temperature gradually starts to decline (to around 40°C) and the mesophilic microorganisms of the actinomycetes become the dominant group. When they re-emerge in the process, the maturation or curing phase of compost starts (Garcia-Prendes, 2001). Organic materials continue to decompose and are converted to biologically stable humic-like substances. The low rate of oxygen consumption allows to stockpile the compost without turning. Although there is no defined time for the curing phase, common practices in commercial composting operations range from one to four months depending on the controlling conditions (Gonzalez & Cooperband, 2002). If the pile has not received enough oxygen or too little/much moisture, a long curing phase might be needed. During this phase the composting mixtures undergo different transformations, which results in changes in the composition and structure of the microbial communities (Villar *et al.*, 2016). Nitrification, the sequential oxidation process of ammonia (NH_3) to nitrate (NO_3^-), occurs during this final phase (López-Cano *et al.*, 2016; Cáceres *et al.*, 2018). Compost is considered finished when raw materials are no longer actively decomposing. When temperatures in the centre of the pile reach near-ambient levels and oxygen concentration remain greater than 10-15% for several days, compost is considered stable. According to Cooperband (2002) these measurements should be taken when the compost pile has at least 50% moisture content by weight (Cooperband, 2002). In general, the composting process can be completed in a 3 to 4-month period (Maheshwari, 2014), although it depends on the size of the pile and the conditions it was under.

1.2.3. Key Parameters of Aerobic Composting

C/N Ratio

As mentioned above, decomposing microorganisms, bacteria and fungi, are the main actors in the composting process. They require carbon (C), nitrogen (N) and other macro and micronutrients for their growth. The C compounds provide energy for the metabolism, usually existing in excess, while the N used for building cell structure, is a limiting element for microbial growth (Maheshwari, 2014). If N is supplied in an insufficient amount, the decomposition process will slow down. However, if it is supplied in excess, it will be discharged into to the environment through leaching (when rainfall occurs, nitrates - which are present in the compost - can easily leach and might be lost through the soil profile and contribute to groundwater pollution) or volatilization (composting of organic wastes rich in nitrogen suffers from the loss of certain amount of nitrogen into atmosphere, through ammonia volatilization) (Cáceres *et al.*, 2018). Microbial respiration releases CO_2, which reduces the C content, but the C/N will only decrease if the diminishing of C is superior than N, which is possible with a negligible N rate in the leachate and of low ammonia volatilization (Maheshwari, 2014).

A study conducted by Van Gestel *et al.* (2003) suggested a theoretical optimum C/N ratio of 30 for a composting starting mix substrate (Van Gestel *et al.*, 2003), while other works have confirmed the optimum C/N ratio in the range between 25 and 35 (Larsen & McCartney, 2000; Tuomela, 2000). The type and source of input feedstock affects the microbial community and the decomposition rate during the composting process according to their starting C/N ratio. Fruit and vegetable waste, with low C/N ratios, are easily degraded as they contain mostly simple carbohydrates (sugars and starches) while green waste (e.g. leaves, nutshells, bark and wood chips), typically with high C/N ratios, decompose more slowly as they contain compounds that are very resistant to biological degradation, such as cellulose, hemicelluloses and lignin.

Temperature

Temperature is one of the major efficiency parameters, since the composting process has 3 big phases—mesophilic, thermophilic and a final maturing or cooling stage, all distinguishable by different temperature patterns. During the initial phase the active microbial population grows exponentially until the available substrate or other factors limit their growth (Marugg *et al.*, 1993). Temperature in the mesophilic phase can vary from ≈ 40°C up to 50°C. In the thermophilic stage, temperatures can reach 70 °C and above, although it has been stated that the optimum decomposition takes place between 50°C and 60°C (Maheshwari, 2014). Some studies (McGregor *et al.*, 1981; McKinley & Vestal, 1984) even suggest that with temperatures greater than 60°C decomposition can cease or be extremely reduced due to low microbial activity. The same can happen with low temperatures, which can retard or even cease the process, being an indicative of reduced microbial activity, lack of oxygen or inadequate moisture conditions. With high temperatures maintained during the thermophilic stage, there is an active destruction of pathogenic organisms, undesirable weed seeds and highest losses of volatile organic substances (Mahimairaja *et al.*, 1995). The maturing and cooling stage can proceed during many weeks and even months and comes to the end when the pile temperature reaches an ambient temperature (Paradelo *et al.*, 2011; Serramiá *et al.*, 2013; Killi & Kavdır, 2013).

pH

pH value is one of parameters of efficiency of the composting process. Metabolic activities affect the pH of compost, resulting in considerable changes during the composting process. pH of feedstock influences the type of organisms involved in the composting process. Fungi tolerate a wider pH range than bacteria do. The optimum pH range for most bacteria is between 6.0 and 7.5, whereas for fungi it can be between 5.5 and 8.0 (Atalia *et al.*, 2015). In the beginning, the formation of organic acids and carbon dioxide lower the pH value to approximately 5.0. As the process proceeds, the use of these acids as substrates by other aerobic microbes increases the pH value up to 8.0 - 8.5 (Atalia *et al.*, 2015), during the cooling and maturation phase. On average, the feedstock pH is slightly acidic while finished compost reaches a value close to neutral (Ko *et al.*, 2008).

Aeration

Aeration has an indirect effect on temperature by speeding the rate of decomposition and thus the rate of heat production. The oxygen requirement depends on the type of material and particle size, temperature of the compost and stage of the process. An oxygen level from 10 to 30% has been reported as optimal (Maheshwari, 2014). Air supply can be controlled by the use of an aeration system or by periodic turning of the pile. Alternatively, air may be actively forced into the pile, usually within a closed or in-vessel system with the aim of maximizing the rate of microbial decomposition.

Aeration is closely connected with temperature. With increase of temperature, the output of harmful compounds increases. Examples of such compounds, formed during composting, are sulphur containing substances, methane (CH_4) and methanol (CH_3OH) (Maheshwari, 2014). Aerobic decomposition, in contrast to anaerobic, is quicker and progresses at higher temperatures, and produces fewer foul odours (such compounds are produced during the rotting process as an intermediate product and can be set free into the atmosphere). Aeration conducted in excess is usually not harmful to the composting process, except that an optimum temperature is harder to maintain and excessive evaporation may cause moisture to become a limiting factor (Maheshwari, 2014).

Moisture

Moisture management requires a balance between microbial activity and adequate oxygen supply. Water is essential to the decomposition process, and water stress is one of the most common limitations on microbial activity on solid substrates (Richard *et al.*, 2002). Moisture content (MC) is also related to aeration and temperature. According to Sasaay et al. (1997) in an aerated static pile (ASP) system approximately 90% of the heat loss is due to evaporation of water (Maheshwari, 2014). The main feature that distinguishes the ASP from the windrow system of composting, is the method of aeration. Instead of periodic turning aeration is provided by forcing air through the static pile (Leton & Stentifordt, 1990). Low moisture conditions restrict the movement of bacteria, so that physical dispersal allows mixed composting systems to function at lower moisture contents than static systems (Richard *et al.*, 2002). However, mixtures can also contain too much moisture which can increase film thickness and fill the smaller pores between particles, limiting oxygen transport (Richard *et al.*, 2002).

Particle size and bulk density

Bulk density, defined has the weight per unit volume of compost, is affected by MC, particle size distribution and the degree of decomposition (Stoffella & Kahn, 2001). Particle size dictates the surface area available for microorganisms. Since larger particles have a smaller total surface, less surface will be accessible to microbes when compared to fine particles (Agnew & Leonard, 2003). Therefore, it is expected that during the composting process, the bulk density of compost would increase due to the breakdown in the particle size of the material and this results in a more compact compost (Figure 2) (Stoffella & Kahn, 2001). Although, in compost systems where substantial evaporation and loss of water is possible, the measured bulk density may decrease as the material dries out during the composting period (Stoffella & Kahn, 2001).

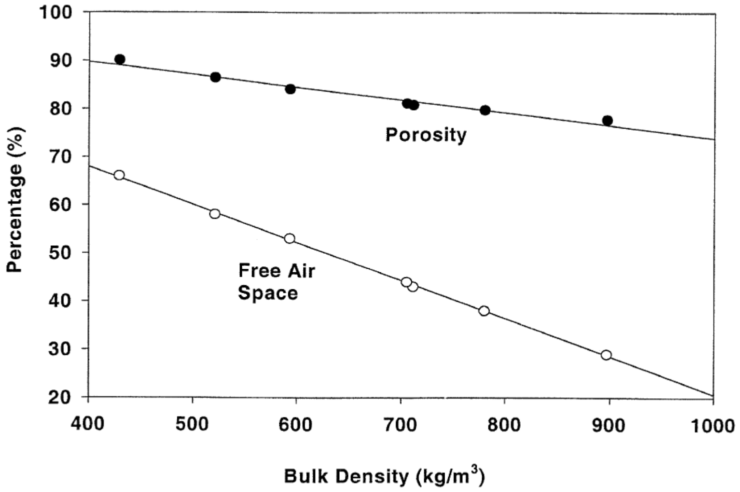

Figure 4 - Relationship between porosity, free air space, and bulk density.

(*in* Brouillette, M., L. Trepanier, J. Gallichand, and C. Beauchamp.1996. Composting paper mill deinking sludge with forced aeration. Canadian Agricultural Engineering 38(2):115–122.)

Bulking agents, such as wood chips or sawdust, should be included in the starting mixture when dealing with fine particulate feedstock (e.g. sludges and animal manures) (Stoffella & Kahn, 2001) or when necessary to improve the C/N ratio of the mix and the nutrient content on the end-product (Reyes-Torres *et al.*, 2018).

1.3. Compost Standards

Composts can be released as commercial products under the observation of national laws, that differ from one country to another. Standard sets are usually statutory in nature. These product standards are designed to regulate potentially harmful aspects of compost related to hygiene, harmful substances and impurities. According to WRAP's (The Waste and Resources Action Programme) 2002 review on Comparison of Compost Standards within the EU, North America and Australasia (WRAP Organization, 2002) *'The minimum criteria* [for statutory standards] *should probably include heavy metals, organic pollutants (as deemed necessary), and pathogens. Possibly this could be supplemented by a measure to assess when material has been sufficiently stabilised through a composting process (e.g. a stability test, ammonia concentration or similar)'.* Product standards are based on soil quality protection, and this should be the major focus of statutory standards.

Some countries regulations include parameters related to the process monitoring and the most used is the temperature-time regime. Typically, this consists in number of days over a threshold temperature regime (50°C, 55°C, 60°C or 65°C) achieved in the composting mass (Maheshwari, 2014). Some countries, like Austria, see this as not necessary, preferring instead to test end-products for the presence of pathogens (WRAP Organization, 2002). Systems tend to have other elements, beyond statutory standards, like complementary statutory standards. These usually are legal regulations influencing organic waste management indirectly. These instruments include waste laws (e.g. EU Council Directive 1999/31/EC on the landfill of waste, 1999) which have specified pre-treatment requirements of the biowaste fraction to be landfilled, or environmental and health aspects of application to land (including fertiliser legislation, soil protection and water laws). Restrictions/maximum dose rates and licensing of composts are under these laws too. Some examples are the mechanisms used by EU Member States to implement the Nitrate Directive (Council Directive 91/676/EEC, 1991), as well as legislation limiting the loading of heavy metals per unit area of land (Council Directive 2008/98/EC, 2008) (e.g. in agricultural land). As a consequence of different political and industrial contexts across the world, compost quality assessment has developed differently from place to place. Each system functions within a certain background 'policy framework' which implies that the approach undertaken in one country is not necessarily suitable for another.

1.3.1. Portuguese standards for compost production

Compost production in Portugal is under the law decree nº103/2015 (Ministério da Economia,), which provides a legislative framework applicable to products placed on the market as organic fertilizers. According to the parameters required, it is possible to classify physical-chemical parameters (granulometry, bulk density, humidity, pH, electrical conductivity, total organic matter, total nitrogen, C/N ratio, mineral elements (K, Ca, Mg, P) and heavy metal content (B, Cd, Pb, Cu, Cr, Hg, Ni, Zn), microbiological parameters (presence of *Escherichia* coli, *Enterococcus* and *Salmonella* spp), biological parameters (compost stability and phytotoxicity) and inert material content.

1.4. Decentralised Community Composting

The production of municipal solid waste (MSW) is an unavoidable consequence of our current consumption society, especially in an urban context, where the population concentration and consumption patterns are a product of great abundance of opportunities. Studies conducted by Eurostat (2010) have shown a correlation between urban population and per capita gross domestic product (GDP), as well as correlation of per capita urban organic waste (UOW) and MSW production with GDP (Eurostat, 2010). It is expected that the global urban population will reach 68% by 2050 (United Nations, 2018) and the resulting progressive growth of cities, calls for an efficient and sustainable MSW management to prevent environmental and public health risks. In 2018, 65.21% of Portugal's total population lived in urban areas (Portugal - urbanization 2008-2018, Statista, 2020).

Increasing the recovery of biowaste implies diverting biowaste from incineration or landfill, thereby reducing the emissions of greenhouse gases emitted during their combustion and transport. According to Adhikari et. al (2010), by 2025, on-site composting practices could reduce costs and greenhouse gas emissions, by 34% and 40%, respectively, when compared to maintaining landfilling practices (Adhikari *et al.*, 2010).

Large-scale centralized valorisation facilities can serve wide geographic areas and divert significant quantities of organic materials from disposal. However, when the European Parliament adopted the Waste Directive 2008/98/EC (Council Directive 2008/98/EC, 2008), measures that approach a more sustainable and circular economy vision were suggested to be set down. These imply improving the segregation of biowaste and its treatment. Composting at a local level could lead to an effective use of waste in accordance with the hierarchy imposed by European regulations, and an efficient waste management model capable of contributing to sustainable development (Comesaña *et al.*, 2017).

1.4.1. Community Composting versus Industrial Composting

Table 1 – Summary table of the main differences between decentralized and centralized composting facilities

Industrial composting/Centralized facilities	Community composting/Decentralized facilities
Transportation costs relatively high; **High operation and maintenance costs;**	Transportation costs relatively low; Comparatively less maintenance costs;
A high degree of specialized skills to operate and maintain; **Advanced technology with highly mechanized equipment;**	Low level skills required; Simple technology, manual labour intense;
Large facilities (e.g. Lipor) **material reception area - 480m^2** **primary and secondary mechanical treatment - sieves (150mm /60mm), magnetic separation, shredder for green waste** **composting - 18 tunnels (12 pre-composting and 6 post-composting tunnels); process controlled by temperature and oxygen probes** **maturation area - 2900 m^2**	Small facilities (e.g. HortaFCUL) **material reception area – 7**.5m^2 (3 compartments each one with 2.5m^2) **primary treatment -** shredder for green waste (for branches, essentially) **composting and maturation –** 10.9m^2 (3 compartments each one with 3.6m^2); temperature controlled by thermometer
Low quality of compost due to poor separation of wastes with high risk of contamination;	High quality of compost since waste is efficiently separated and risks for contamination are minimized;
Final product transported to markets and commercialized;	The final product is used by the community involved in the process.

(adapted from Bruni et al., 2020)

1.4.2. Drawbacks of Large-Scale/Industrial Composting

While municipal waste hauling companies provide a crucial service for communities, the disposal of waste from localities to distant places perpetuates an inability to separate and cycle waste streams into more valuable materials (Schlesinger, 2016). Decentralised community composting, allows people to become less dependent on the municipal waste collection service while reusing organic waste where it is generated, allowing to close the cycle of organic matter by returning the nutrients to the soil locally as well as reducing waste quantities to be transported and transport costs (Comesaña *et al.*, 2017). Large scale industrial composting does not have the same social and cultural impact as community-scale initiatives, that engage and educate communities in resource conservation and food systems thinking, while providing the compost product and keeping it as a local resource (Platt *et al.*, 2014).

1.4.3. Community Composting Case studies

"Lisboa a Compostar" project

The project began in May 2018, promoted by the Lisbon City Council and Valorsul, in the scope of the Municipal Plan for Waste Management and the European project FORCE - Cities Cooperating for Circular Economy (HORIZON 2020). The central objective is the implementation of domestic composting (food waste and green waste) in houses that have gardens, through the distribution of individual composting units. For residents who do not have a green space for the individual unit, community composting units were implemented in public spaces. To receive the composting units, or the keys to have access to the community ones, the residents must attend to a training course on composting.

Allariz (Galicia, Spain)

A study conducted by Comensanã et. al (2017) presented a project in Allariz where the local government implemented a decentralized model to manage biowaste separation and treatment though composting. With the participation of only 20% of the population of the municipality (~1,237 inhabitants), there was a reduction in the mixed fraction tonnes by 7.3%, due to the deviation of the organic fraction, biowaste and green waste, to individual composters and community composting centres, respectively. Furthermore, there was an increase in the collection of lightweight packaging, paper-cardboard and glass fraction (20.1%, 8.5% and 11.8%, respectively) due to the improvement of recycling. Through the promotion of decentralized composting, the organic fraction of MSW in Allariz was valued by producing a high-quality compost and closing the cycle of organic matter by applying it to the soil, while keeping the process and product locally and engaging the community through participation and education (Platt *et al.*, 2014).

Pyrgos and Panormos (Tinos island, Greece)

A study conducted by Panaretou et. al (2019) presented a pilot experience in Pyrgos and Panormos (~400 inhabitants), where an integrated biowaste management system has been implemented. The pilot scheme involved the separation of biowaste at source and the treatment on-site, using a prototype community-scale composting unit. During 12 months the system was monitored (physicochemical parameters and public's awareness and participation) and results have showed that the biowaste source separation was effectively practiced by the participating householders, given the low impurity level (~2%) of the collected biowaste. Also, the compost samples fulfilled the requirements proposed by EU End-of-Waste (EoW) quality criteria. It was found to contain exemplary heavy metal content (72% lower than the EoW limit value for Cd, 43% lower for Ni, 38% lower for Pd, 24% lower for Cu, and 36% lower for Zn) and phytotoxicity parameters (analyses showed that samples were free from pathogens and parasitic organisms). According to the authors, the proposed decentralized composting system could offer a sustainable solution for isolated communities similar to the ones in the experiment, in order to promote on-site nutrient recovery while improving local farming practices and reducing nutrient leaching, thus protecting their natural assets.

1.4.4. Faculty of Sciences of the University of Lisbon (FCUL) Composter

The FCUL composter is part of an initiative aimed to contribute to the carbon footprint reduction of the faculty by improving the organic waste management within the campus. It intended to transform what was considered garden waste into a high-quality compost product, while reducing the impact of exporting waste from the campus and dependency on external sources of fertilizers (Avelar *et al.*, 2017). The composter is the key element to close the organic cycle in the campus. During the first year, around 40m^3 of waste (corresponding to ~28 tons) from the green spaces of the FCUL were composted. HortaFCUL volunteers performed all the work by hand, carrying several tons of organic waste to create the piles, revolving them and taking them out.

1.4.4.1. Entities involved

The composter is a collaboration of several entities with different functions:

- FCUL's Security, Health and Sustainability Office (G3S)

Implementation of the project, integrating it with the technical services of the campus; Coordination with gardeners (who are responsible for depositing organic waste in the reception area); Communication between entities involved and the faculty's board; Site security.

- Centre for Ecology, Evolution and Environmental Changes (cE3c)

Coordination of research projects; Monitoring the development of scientific work, coordinating it with the composter management; Project monitoring.

- HortaFCUL

Development of the project; Integrating the composter in the Permaculture Living Lab ('Permalab') plan; Keeping the space organized and give destination to the final product; Maintaining internal communication between the three entities involved; Promoting external communication about the activities and processes involving the composter; Proposing improvements whenever necessary; Training of all stakeholders.

1.4.4.2. HortaFCUL and Permaculture Living Lab ('Permalab')

HortaFCUL is a project created in 2009 by a group of biology students who were interested in permaculture and how it could provide solutions to some of the current ecological, social and economic problems. The project aims to contribute to the faculty sustainability as well as raising awareness and demonstrating more ecological practices, based on the ethics, principles, strategies, techniques and tools proposed by permaculture. Thus, in the same year, with the dedication of volunteers and support from the faculty, the first physical space of the project was created: a small food garden, with a mix of horticulture, aromatic herbs and fruit trees. This space is still considered the heart of the project.

In 2016, in collaboration with HortaFCUL group members that are part of the cE3c Research Center, the second physical space, the Permaculture Living Lab (‘Permalab’), was created. This space is dedicated to promote urban agriculture practices in a systemic way, based on ecological principles, in order to promote a more sustainable future. This space aims to integrate research and innovation processes proposed by permaculture, towards a more transdisciplinary, participatory and action-oriented research approach. Since permaculture is a science-based planning system that mimics ecological patterns, the projects to be developed in Permalab are intended to create scientific evidence of nature-based solutions, while contributing to the regeneration of the university campus.

1.5. Aims of the study

People’s consumption patterns are directly influenced by where they live. The progressive growth of cities urges for a more efficient municipal solid waste management. Composting is a sustainable and profitable alternative, compared to landfill discharge (Farrell & Jones, 2009), since it decreases environmental problems related to waste management by decreasing volumes of waste and by killing potentially hazardous organisms (Sæbø & Ferrini, 2006). The aim of the European Commission's directive on the disposal of waste in landfills (1999/31) was to “apply strict measures and processes that reduce or avoid as much as possible negative effects on the environment” (Council Directive 1999/31/EC, 1999). Consequently, in EU countries it is now seen as an objective to reduce of the waste disposal in landfills.

Green waste, together with food waste, are the most significant fractions of municipal solid waste (Wei *et al.*, 2017). Green waste includes branches, dry leaves and grass, a set of materials generated in parks and municipal gardens, as well as in domestic households (Reyes-Torres *et al.*, 2018). The management of this waste is expensive, especially the collection and transport processes for the treatment facilities (Reyes-Torres *et al.*, 2018). Composting is a suitable alternative for recycling these residues, and the product obtained can be used as a soil amendment, solving the problem of disposal of these residues while restoring soil properties. In large scale facilities it is possible to manage large amounts of organic waste, however, this approach does not have the same social impact as community-scale initiatives can have, by involving the communities and providing a product that is going to be used locally (Platt *et al.*, 2014).

The FCUL composter is part of the community-based project HortaFCUL, a project that aimed to contribute to the faculty sustainability while raising awareness and demonstrating more ecological practices. The composter initiative aims to contribute to the carbon footprint reduction of the faculty by improving the organic waste management within the campus while transforming garden waste into a high-quality compost product, thus reducing the impact of exporting waste from the campus and dependency on external sources of fertilizers (Avelar *et al.*, 2017).

The overall aim of this study is to assess whether it is possible to produce a viable compost product in a community-scale facility and context. In order to fulfil this aim, some specific objectives were defined:

- To assess if the product obtained in this system is considered viable by comparing some of the physical and chemical properties of the product with the Portuguese statutory standards and with the green waste (GW) physicochemical characteristics established in literature;
- To assess the proportion of plastic and other man-made non-biodegradable pollutants in the final composts in order to evaluate whether contamination in this context is a problem;
- To assess the particle size dimensions of the final products in order to evaluate whether they are suitable for different horticultural compost use categories;
- To assess whether the inherent variability of the system under study affects the physicochemical characteristics of the final products;
- To understand the relationships between the different variables under study;
- To analyse the final balance of the cycles, by quantifying the amount of carbon, nitrogen and total mass losses during the process as well as the biomass production;

2. Materials and Methods

2.1. Study site, compost pile procedure, in-situ measurements, sampling and physical properties

2.1.1. Study site

The composting experiment was carried out from December 2018 to January 2020, and the composter site is located within the Permaculture Living Laboratory (“PermaLab”) on the campus for Faculty of Science at the University of Lisbon (38°45’29.30” N, 9°9’30.40” W).

2.1.2. Compost pile procedure and in-situ measurements

Before composting, the organic material used as feedstock passed through a reception area, which is located near the composter and it is used for material storage and pre-treatment. The pre-treatment consists in cutting and chopping branches and separating the “green material” (e.g. grass, green leaves) from the “brown material” (e.g. dry leaves, branches). The containers keep the residues separated until the compost pile is created.

For each pile, the feedstock was weighted using a dynamometer (CR-300, Gram Precision, Barcelona, Spain) and then placed in altering layers of “green material” and “brown material” and after each two alternating layers, the feedstock was mixed using a pitchfork.

The composter has 3 compartments, so it can have several compost piles at the same time, which facilitates the turnovers. The compost was turned periodically (between 3 and 14 weeks) in order to redistribute the nutrients, homogenise the pile and maintain favourable conditions of aeration. Humidity conditions were maintained by watering the pile when necessary. The cycle began with the creation of the pile in the first compartment, which was then transferred to the second and subsequently to the third. During the composting cycles, temperatures were taken weekly from four random locations in the centre of the piles using a compost thermometer. When compost temperatures reached ambient temperatures, the pile was removed and all material was weighted using a dynamometer.

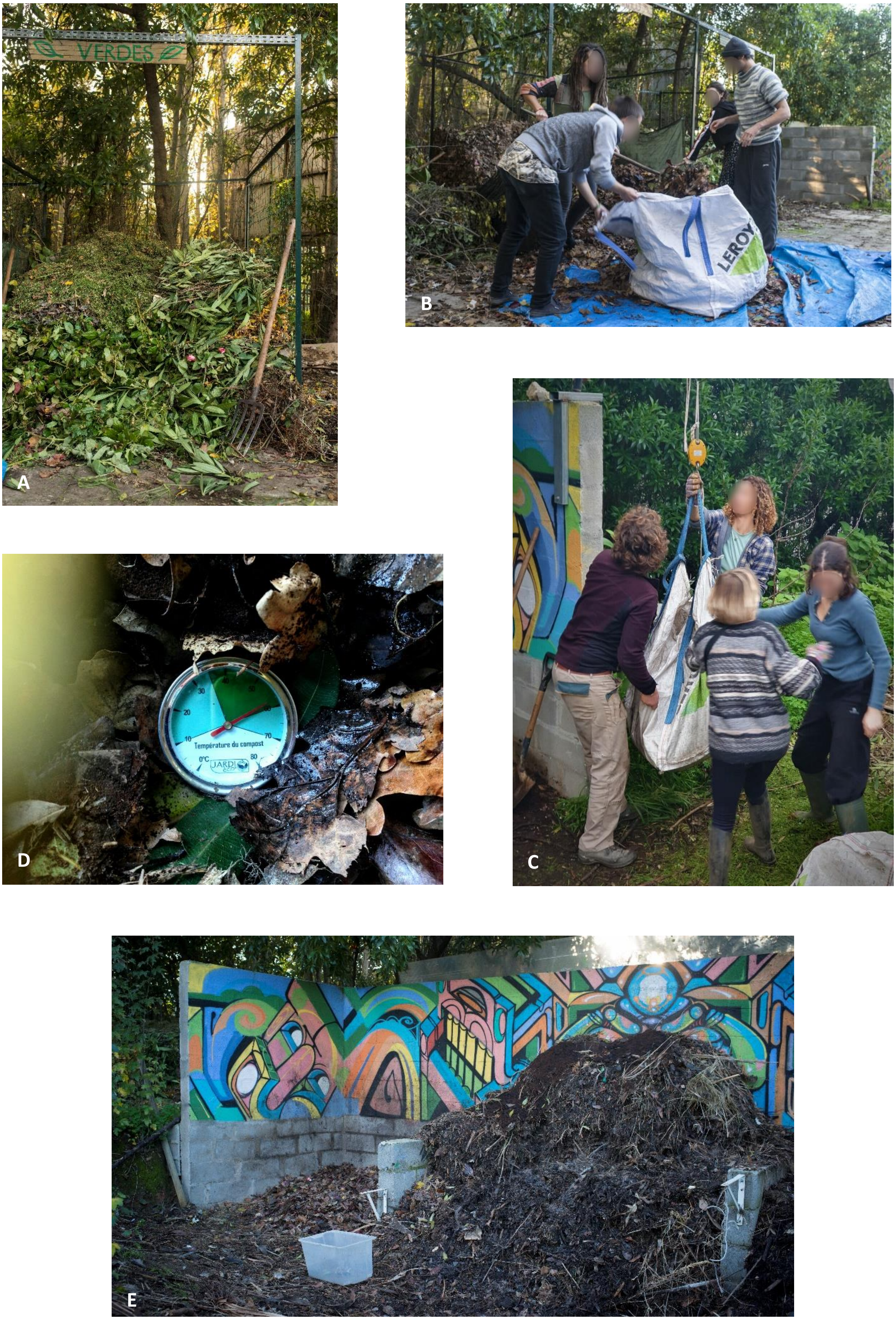

Figure 3 – Compost pile procedure; A – green material reception area; B – feedstock to be weighted; C – weight station; D – compost thermometer; E – Pile in the second compartment

2.1.3. Sampling collection, preparation and physical properties

General sampling

Composite samples were taken for each type of input material in each pile, therefore resulting n=4 replicates, with 6 sub-samples each (3 for both “green material” and “brown material”). Compost sampling was performed after sieving (12mm), resulting n=4 replicates, with 6 sub-samples each. All samples were kept at -20 °C in air tight plastic bags until analysis.

Granulometry analysis

Particle size of degradable material:

After the compost was weighted sampling was performed, thus resulting n=4 replicates, with 6 sub-samples each, for both analyses. Particle size determination was carried out by sieve analysis method (Stoffella & Kahn, 2001) where compost samples were sieved through sieves of different mesh sizes (4 and 8mm) until the amount retained became more or less constant. The sieved particles of different fractions (< 4mm, 4 to 8mm and 8 to 12mm) were weighed separately and using their weights the cumulative passing percentage (CPP) was calculated as:

$$\text{CPP} = 100 - \%\text{retention}$$

Where

$$\%\text{retention} = \frac{Wt.of\ sample\ retained\ in\ sieve}{Total\ Wt.of\ sample}$$

Inert material content:

After sampling, man-made inert determination (which included glass, plastic and metal, textiles and stones) was performed. Sieve analysis was carried out (Stoffella & Kahn, 2001), where compost samples were sieved through sieves of different mesh sizes (4 and 8mm), and separated by a visual sorting process. The particles of different fractions (< 4mm, 4 to 8mm and 8 to 12mm) and inert materials were weighted separately and using their weight the percentage inerts (g) per compost (g) was calculated.

Figure 4 – Sieve procedures; A – 12mm sieve. Compost samples were preformed after all the material passed through this sieve. B – 8mm to 12mm fraction; C – 4mm to 8mm fraction; D – under 4mm fraction

Sampling procedure, dry weight, organic matter determination

Parts of all sub-samples were weighted before and after oven-drying them for a week in order to determine dry weight. The water content (%WC) was estimated using the following formula:

$$\%WC = [(W_i - W_f)/W_i] \times 100\%$$

Where:

W_i is the initial weight, wet basis;

W_f is the final weight, dry basis;

The samples were then grounded in a ball mill (Retsch MM400, Haan, Germany) until completely homogenized. Organic matter of input material and compost was measured using a modified loss on ignition method. For this method 100mg of each dried and milled sample were transferred to a crucible and burned in a muffle furnace (L3, Nabertherm, Lilienthal, Germany) at 550° C for 4 hours, following the TMECC method (Test Methods for the Examination of Composting and Compost) for ash content determination (Matthiessen *et al.*, 2005). This method requires a temperature of a 550° C for 2 hours, however, in order not to underestimate organic matter content, the burning time was doubled. After ignition, the crucible with the ash was then left to cool for 16 hours, weighted and subsequently matter loss on ignition was termed as organic matter. The organic matter (OM) was calculated according to the following formula:

$$\%OM = (W_i - W_f) \times 100\% / W_i$$

Where:

W_i is the initial weight, before ignition (g)

W_f is the final weight, after ignition (g)

Extract preparation for pH

pH was analysed in 1:10 compost/distilled water extracts, with 30 min extraction time on a shaker at room temperature in 15 ml plastic test tubes. To obtain a clear extract, the resulting suspension was centrifuged (3600rpm, 18º, 5min) and filtered through Whatman No.1 filter papers. pH was analysed in the extract with a glass electrode (pH/mV meter 501, Crison, Barcelona, Spain).

2.2. C/N ratio and respective C (δ_{13}C) and N (δ_{15}N) isotopic fractionation

For the isotopic fractionation in input/raw and output/compost materials, the samples (dry weight basis) of all the 4 compost cycles were selected. Samples were individually grounded with a ball mill (Retsch MM400, Haan, Germany). About 4.5 ± 0.5 mg of each dried and milled sample were encapsulated and sent for elementary and isotopic analysis. The samples were weighted on a precision scale (Fisions instruments, sartorius micro, Italy) and analysed at the Stable Isotopes and Instrumental Analysis Facility (SIIAF) of the Centre of Environmental Biology (CBA), University of Lisbon. ^{13}C/^{12}C and ^{15}N/^{14}N ratios in the samples were determined by continuous flow isotope mass spectrometry (CF-IRMS) (Preston and Owens, 1983), on an Isoprime (GV, UK) stable isotope ratio mass spectrometer coupled to an EuroEA (EuroVector, Italy) elemental analyzer for online sample preparation by Dumas-combustion. The standards used were USGS-25 and USGS-35 for nitrogen isotope ratio, and Glucose BCR 657 and IAEA-CH7 for carbon isotope ratio. δ^{15}N results were referred to Air and δ^{13}C to PeeDee Belemnite (PDB). Precision of the isotope ratio analysis, calculated using values of 3 replicates of laboratory standard material interspersed among samples in every batch analysis, was ≤ 0.2 ‰

2.3. Biomass, nitrogen, carbon and total masses and losses

The calculation of the total mass (Mt) of each feedstock (green and brown) and compost, was obtained using the following formula:

$$Mt = Wf * (Wd (a) / Wf (a))\ kg$$

where Wf corresponds to the total fresh weight of the component, Wd (a) to the dry weight of the sample corresponding to the component and Wf (a) fresh weight of the sample corresponding to the component (Snowdon et al. 2002). Biomass, nitrogen mass and carbon mass in the piles were determined according to the following formula:

Organic matter, nitrogen or carbon concentration*dry weight of the windrow,

and the losses were computed as follows:

Biomass loss, nitrogen loss or carbon loss *(*%of initial*)* = (initial mass − final mass) /initial mass

2.4. Seed Germination Test

In order to evaluate the compost phytotoxicity, a seed germination test proposed by Zucconi (1981) (de Bertoldi *et al.*,; Warman, 1999; Wu *et al.*, 2000) was performed. A number 1 Whatman filter paper was placed inside a 12 by 80-mm sterilized, disposable petri dish. The filter paper was wetted with 9 mL of compost extract (1:10) and 30 lettuce seeds (*Lactuca sativa L.*) were placed on the paper. Distilled water was used as a control. The petri dishes were sealed with Parafilm to minimize water loss while allowing air penetration and then were kept in the dark for 4 days at room temperature. At the end of 4th day, the percentage of seed germination in compost extract was compared with that of the water control.

Seed germination rate (SGR) and germination index (GI) were determined for each treatment according to the following formulas:

SGR (%) = Average number of germinated seeds * 100%/Number of seeds per dish;

GI (%) = (Average number of germinated seed in the treatment * average root length in the treatment * 100%)/ (Average number of germinated seed in the control * average root length of the control);

2.5. Statistical analysis and graphic output

All data used for an analysis of variance were checked with a robust Brown-Forsythe Levene-type test for homoscedasticity and with a Shapiro-wilk test or qq-plots for normality. If the assumptions were violated, either means that a Kruskal–Wallis test was applied, followed by a multiple Mann Whitney Wilcoxon test with Holm- Bonferroni adjusted method or the data was transformed to ensure the assumptions. Linear regressions where checked with a Breusch-Pagan test for homoscedasticity and normality distribution of the residuals was verified either with Shapiro-wilk test or qq-plots. If the assumptions were violated data was transformed to ensure they were met. A principal component analysis (PCA) was employed to analyse the relationships between the parameters measured from each of the 4 GW composts produced, with the resulting graphical analysis presented in biplots. Statistical analysis was performed using R version 3.4.3 (R Core Team, 2013) and executed on RStudio version 1.1.419. Additional packages used were: “Hmisc” (Harrel *et al*, 2020), “lmtest” (Hothorn & Zeileis, 2020), “multcompView” (Graves *et al*, 2019), “dplyr” (Wickham *et al*, 2020) and “ggplot2” (Wickham *et al*, 2020). Some graphs were changed with Illustrator (Adobe, San José, CA, USA) if the output produced by R was not suitable.

3. Results

3.1. Feedstock general description

Composite samples of plant material used as feedstock group (green and brown) were visually inspected and a taxonomic identification performed until family, genus or species level of classification (table 2). The general description of feedstocks in the different piles can be seen in table 3. The total mass of feedstock added to each pile (dry weight (dw)) ranged from 191.13 kg in pile 4 (P4) to 343.42 kg in pile 1 (P1) in the green fraction, and from 169.58 kg in pile 3 (P3) to 492.75 kg in pile 1 (P1) in the brown fraction. In order to compare quantities in the input feedstock based on the two green waste groups, a green/brown ratio was calculated. The green/brown ratio was obtained dividing the total mass of green fraction by the total mass of brown fraction for each pile. It was possible to infer the existence of one group (pile 1 and 4, around 0.7), a high ratio (pile 3, 1.57) and an in-between (pile 2, 1.08) based on their ratio values, a reflection of the proportion between green and brown material. Total carbon (dw/dw) ranged from 73.08 g kg^{-1} (P4) to 139.31 g kg^{-1} (P1) for green material, and from 57.37 g kg^{-1} (P3) to 199.24 g kg^{-1} (P1) for brown material. Total nitrogen (dw/dw) ranged from 3.31 g kg^{-1} (P4) to 6.93 g kg^{-1} (P3) in the green fraction, and from 2.15 g kg^{-1} (P3) to 4.43 g kg^{-1} (P1) in the brown fraction. Biomass in the feedstock varied form 161.4 g kg^{-1} (P4) to 291.63 g kg^{-1} (P1) and from 122.96 g kg^{-1} (P3) to 405.15 g kg^{-1} (P1), for the green and brown fraction, respectively.

3.2. Feedstock chemical properties

Chemical properties of both feedstocks groups in the different piles can be seen in table 4. Moisture content (MC) was significant different between feedstocks in all piles (pairwise Welch T-test, $p < 0.05$) and between the four groups of each green and brown feedstock. Pile 4 had the lowest values for both fractions (13.6% and 19.1% for green and brown, respectively) and pile 1 had the highest values

(66.8% and 53.6%, green and brown). Organic matter (OM), total carbon and ash content showed the same pattern between piles with significant differences between fractions only observable in piles 2 and 4 (pairwise Welch T-test, $p < 0.05$). For OM and ash content in each feedstock between piles three significant groups were detectable. Regarding the green fraction organic matter in pile 1 and 4 was 1.2 and 1.1-fold higher when compared with pile 2 and 3, respectively (pairwise Welch T-test, $p < 0.05$). Between brown fraction values, pile 4 was 1.2-fold higher than pile 3 (pairwise Welch T-test, $p < 0.05$). Ash content showed the inverse tendency and was respectively 2.0 and 1.6-fold higher in pile 2 and 3 when compared to pile 1 and 4, between green fractions. Between brown fractions, ash content in pile 3 was 1.5, 2 and 3-fold higher than piles 1, 2 and 4, respectively. C content (%) showed significant differences between all piles regarding the green fraction, ranging from 31.9% (P2) to 40.6% (P1). Three different groups were detected between brown fractions. C content in pile 4 was 1.1-fold higher than pile 1 and 2, and 1.3-fold higher when compared with pile 3. Nitrogen (N) content (%) was significant different between feedstocks in all piles (pairwise Welch T-test, $p < 0.05$). and ranged from 1.5% (P2) and 0.9 % (P1 and P2) (green and brown, respectively) to 2.6% (P3) and 1.3% (P3) (green and brown). Between green fractions three significant groups were detected. Total N values in pile 3 were 1.6, 1.7 and 1.5-fold higher when compared with pile 1, 2 and 4, respectively. Regarding the brown fractions only two groups were observable. Values in pile 3 and 4 were respectively 1.4 and 1.3-fold higher when compared with the remaining piles. Initial C/N ratio was significant different between feedstocks in all piles (pairwise Welch T-test, $p < 0.05$), ranging from 13:1 (P3) to 22:1 (P2 and P4) between green feedstock, and from 26:1 (P3) to 46:1 (P2) between brown feedstock. For each feedstock group three significant groups were detected. No significant differences were found for $\delta^{13}C$ either between piles or feedstocks (pairwise Wilcoxon rank sum test, $p = 0.1$), although values ranged from -20.99‰ (P4) and -27.9‰ (P3 and P4) (green and brown, respectively) to -28.23‰ (P2) and -28.55‰ (P2) (green and brown). $\delta^{15}N$ was significant different between feedstocks in all piles (T-test, $p < 0.05$). Between green feedstock three groups were detected. Pile 2 and 4 were 2 and 1.4-fold higher when compared with pile 1 and 3. Regarding the brown fractions four significant groups were detected, with values ranging from 1.79‰ (P2) to 7.5‰ (P1).

Table 2 – Feedstock taxonomic description

Pile 1

Green fraction

Family	**Genus/Species**
Poaceae	-
Apocynaceae	*Nerium oleander* L.
Caprifoliaceae	*Viburnum tinus* L.

Brown fraction

Family	**Genus/Species**
Fagaceae	*Quercus* L.
Rosaceae	*Prunus* L.
Salicaceae	*Populus nigra* L.

Pile 2

Green fraction

Family	**Genus/Species**
Poaceae	-
Apocynaceae	*Nerium oleander* L.

Brown fraction

Family	**Genus/Species**
Fagaceae	*Quercus* L.
Poaceae	-

Pile 3

Green fraction

Family	**Genus/Species**
Poaceae	-
Fucaceae	*Fucus spiralis* L.
Fagaceae	*Quercus* L.

Brown fraction

Family	**Genus/Species**
Myrtaceae	*Eucalyptus globulus* Labill.
Fabaceae	*Tipuana tipu* (Benth.) Kuntze
Pinaceae	-

Pile 4

Green fraction

Family	**Genus/Species**
Fagaceae	*Quercus* L.
Poaceae	-

Brown fraction

Family	**Genus/Species**
Myrtaceae	*Eucalyptus globulus* Labill.
Fabaceae	*Tipuana tipu* (Benth.) Kuntze

Table 3 – General description of feedstocks

	P1		**P2**		**P3**		**P4**	
Properties	Green	Brown	Green	Brown	Green	Brown	Green	Brown
Total Mass (kg)	343.42	492.75	309.62	285.89	266.65	169.58	191.13	263.3
Green/Brown ratio	0.7		1.08		1.57		0.73	
Total C (g kg^{-1})	139.31	199.24	98.78	114.36	92.17	57.37	73.08	116.29
Total N (g kg^{-1})	5.61	4.43	4.54	2.48	6.93	2.15	3.31	3.16
Biomass (g kg^{-1})	291.63	405.15	214.5	244.73	201.73	122.96	161.4	239.14

Values without standard deviation did not had replicates.

Table 4 – Average chemical characteristics of feedstocks

Properties	**P1**		**P2**		**P3**		**P4**	
	Green	Brown	Green	Brown	Green	Brown	Green	Brown
***Moisture content (%)**	**66.8 ± 0.1^{d}**	**53.6 ± 0.30^{d}**	**49.1 ± 0.49^{b}**	**13.5 ± 0.31^{a}**	**57.5 ± 0.5 ^{c}**	**47.3 ± 1.01^{c}**	**13.6 ± 0.86^{a}**	**19.1 ± 0.46^{b}**
Organic matter (%)	84.9 ± 0.53^{c}	82.2 ± 1.28^{b}	**69.3 ± 1.48^{a}**	**85.6 ± 3.46^{bc}**	75.6 ± 0.52^{b}	72.5 ± 3.27^{a}	**84.4± 1.48^{c}**	**90.8 ± 0.38^{c}**
***C/N ratio**	**25 ± 0.96^{c}**	**44 ± 0.75^{c}**	**22 ± 0.9^{b}**	**46 ± 1.8^{c}**	**13 ± 0.34^{a}**	**26 ± 0.25^{a}**	**22 ± 0.47^{b}**	**36 ± 0.1^{b}**
C (%)	40.6 ± 0.15^{d}	40.4 ± 0.23^{b}	**31.9 ± 0.7^{a}**	**40 ± 1.22^{b}**	34.6 ± 0.21^{b}	33.8 ± 1.56^{a}	**38.2± 0.57^{c}**	**44.2 ± 0.29^{c}**
***N (%)**	**1.63 ± 0.06^{b}**	**0.9 ± 0^{a}**	**1.5 ± 0.06^{a}**	**0.9 ± 0.06^{a}**	**2.6 ± 0.1^{c}**	**1.3 ± 0.06^{b}**	**1.73± 0.06^{b}**	**1.2 ± 0^{b}**
Ash (%)	15.1 ± 0.53^{a}	17.8 ± 1.3^{b}	**30.7 ± 1.48^{c}**	**14.4 ± 3.46^{ab}**	24.3 ± 0.52^{b}	27.5 ± 3.27^{c}	**15.6 ± 1.48^{a}**	**9.2 ± 0.38^{a}**
δ^{13}C (‰)	-26.15± 0.2^{a}	-28.3 ± 0.1^{a}	-28.23 ± 0.1^{a}	-28.55 ± 0.1^{a}	-25.7 ± 0.2^{a}	-27.9 ± 0.1^{a}	-20.99 ± 1.1^{a}	-27.9 ± 0.2^{a}
***δ^{15}N (‰)**	**1.85 ± 0.3^{a}**	**7.5 ± 0.22^{d}**	**3.66 ± 0.18^{c}**	**1.79 ± 0.2^{a}**	**2.7 ± 0.18^{b}**	**2.26 ± 0.11^{b}**	**3.78± 0.19^{c}**	**4.62 ± 0.26^{c}**

Mean values and standard deviation of composite samples (n = 3) are shown; Data based on dry weight. The letters a, b, c and d mark significant differences between the different piles for each feedstock. The letters in bold mark significant differences between feedstock groups for each pile (ANOVA with subsequent multiple Welch T-test for MC, OM, C/N ratio, C, N, Ash and δ^{15}N; Kruskall-Wallis test with subsequent multiple Mann Whitney/Wilcoxon rank sum test for δ^{13}C with Homl – Bonferroni adjusted method; $p < 0.05$) *Measurements were log-transformed to ensure normality assumption

3.3. Input and output general description

The general description of input and output in the different piles can be seen in table 5. Total mass (dw) ranged from 436.2 kg (P3) to 836.2 kg (P1) for input, and from 277.8 kg (P4) to 491.2 kg (P1) for output. The total mass loss (dw) ranged from 54.85 kg (P3) to 344.96 kg (P1). Total C (dw/dw) ranged between 149.5 g kg^{-1} (P3) to 338.5 g kg^{-1} (P1) for input, and for output from 85.4 g kg^{-1} (P4) to 135.7 g kg^{-1} (P1). C losses were similar between pile 2 and 4 (54.3% and 54.9%) while pile 1 lost 59.9%. C loss in pile 3 was substantially lower (29.8%). Biomass (dw/dw) ranged from 324.7 g kg^{-1} (P3) to 696.8 g kg^{-1} (P1) in input, and from 175.2 g kg^{-1} (P4) to 274.8 g kg^{-1} (P1) in output. Biomass losses showed the same C loss pattern. Pile 2 and 4 with similar losses (55.6% and 56.2%), pile 1 with the highest loss (60.6%) and pile 3 with the lowest (32.15%). Total N (dw/dw) ranged from 6.47 g kg^{-1} and 5.83 g kg^{-1} (P4) (input and output, respectively) to 10.04 g kg^{-1} and 10.3 g kg^{-1} (P1) (input and output). N losses were low and only observable for pile 4 (9.8%). In the remaining piles it seems to have been an increase from the initial total N content (increase ranging from 2.7% in pile 1 to 5.7% in pile 3).

3.4. Input and output chemical properties

Chemical characteristics of input and output in the different piles can be seen in table 6. The input values are the means of both feedstock fractions. MC was significantly different between input and output for the majority of piles (pairwise Wilcoxon rank sum test, $p < 0.05$) except in pile 1. Three groups were detected between piles for both groups, input and output ranging, respectively, from 16.4% (P4) to 60.2% (P1) and from 52.2% (P2) to 66.2% (P4). OM, total C and ash content were significant different between input and output for all piles (pairwise Welch T-test, $p < 0.05$) and showed the same pattern based on significance letters with two detectable clusters, for each group, between piles. OM between inputs ranged from 74.1% (P3) to 87.6% (P4), and between outputs from 55.9% (P1) to 63.1% (P4). The C content between inputs ranged from 34.2% (P3) to 41.2% (P4). Between outputs values ranged from 27.2% (P2) to 30.75% (P4). The ash content between inputs ranged from 12.4% (P4) to 25.9% (P3), and between outputs from 36.9% (P4) to 44.06% (P1). C/N ratio was significant different between input and output for all piles (pairwise Wilcoxon rank sum test, $p < 0.05$), declining for all composts, especially in mixtures with a high initial C/N ratio. Pile 1 declined from 32:1 to 13:1. In contrast, Pile 3 only declined from 18:1 to 11:1. The N content between inputs ranged from 1.17% (P2) to 1.93% (P3). Between outputs values ranged from 2.02% (P2) to 2.52% (P3). The only significant difference between input and output for $\delta^{13}C$ was found in pile 4 (pairwise Wilcoxon rank sum test, $p < 0.05$). Between piles for each input and output groups three clusters were detected. Pile 4 had the highest values between both input and output groups (-24.5‰ and -27.02‰, respectively) and pile 2 had the lowest values (-28.39‰ and -28.17‰). The only significant difference detected for between input and output for $\delta^{15}N$ was found in pile 2 (pairwise Wilcoxon rank sum test, $p < 0.05$). Two groups were detected between piles for both groups, input and output. Pile 1 had the highest values between both inputs and outputs (4.68‰ and 5‰) and pile 3 had the lowest values (2.5‰ and 3.49‰).

3.5. Output maturity and efficiency parameters

For germination index (%), root length (cm) and pH two groups were detected has indicated by significance letters. Germination index (GI) values ranged from 162.4% in pile 1 to 180.78% in pile 2. The index was significantly higher in pile 2 when compared with the value from pile 1 (pairwise Wilcoxon rank sum test, $p < 0.05$). Root length (RL) ranged from 7.65 cm in pile 1 to 8.42 cm in pile 4, and pile 2 and 4 has significantly higher values when compared with the values from pile 1. pH values ranged from 7.78 in pile 3 to 8.13 in pile 4. Pile 2 and 4 were identified as the same group by significance letters and both piles had significantly differences when compared with pile 3. Seed germination rate (SGR) did not show any significantly difference between piles even though it ranged from 86.7% (P3) to 83.3% (P1 and P4).

Table 5 – General description of input and output

	P1		P2		P3		P4	
Properties	Input	Output	Input	Output	Input	Output	Input	Output
Total mass (kg)	836.2	491.2	595.5	357.8	436.2	381.4	454.4	277.8
Total mass loss (kg)	344.96		237.7		54.85		176.6	
C (g kg^{-1})	338.5	135.7	213.13	97.4	149.5	104.9	189.34	85.4
C loss (%)	59.9		54.3		29.8		54.9	
N (g kg^{-1})	10.04	10.3	7.02	7.21	9.08	9.6	6.47	5.83
N loss (%)	-2.7		-2.8		-5.7		9.8	
Biomass (g kg^{-1})	696.8	274.8	459.2	203.98	324.7	220.3	400.5	175.2
Biomass loss (%)	60.6		55.6		32.15		56.2	
***Degradable material (< 4mm) (%)**	-	26.1 ± 10.6^{bc}	-	9.76 ± 1.86^{a}	-	13.46 ± 1.82^{b}	-	26.28 ± 4.09^{c}
***Degradable material (4 - 8mm) (%)**	-	12.35 ± 2.29^{a}	-	9.55 ± 2.3^{a}	-	15.09 ± 4.49^{ab}	-	22.9 ± 4.44^{b}
***Degradable material (8 – 12mm) (%)**	-	42.47 ± 9.71^{a}	-	55.33 ± 7.4^{a}	-	51.49 ± 5.98^{a}	-	47.19 ± 2.97^{a}
***Stones (4-8mm) (%)**	-	0.75 ± 0.87^{c}	-	0.09 ± 0.06^{a}	-	0.08 ± 0.08^{a}	-	0.15 ± 0.05^{b}
***Stones (8-12mm) (%)**	-	16.98 ± 6^{b}	-	24.1 ± 10.7^{b}	-	18.65 ± 7.62^{b}	-	2.58 ± 0.89^{a}
***Plastic and Metal (4-8mm) (%)**	-	0.008 ± 0.005^{a}	-	0.05 ± 0.02^{b}	-	0.02 ± 0.02^{ab}	-	0.03 ± 0.01^{ab}
***Plastic and Metal (8-12mm) (%)**	-	1.05 ± 0.43^{a}	-	0.81 ± 0.51^{a}	-	0.52 ± 0.23^{a}	-	0.75 ± 0.27^{a}
***Textiles (>4mm) (%)**	-	0.04 ± 0.02^{a}	-	0.006 ± 0.008^{a}	-	0.52 ± 0.77^{a}	-	0.02 ± 0.04^{a}
***Glass (>4mm) (%)**	-	0.23 ± 0.27^{a}	-	0.27 ± 0.34^{a}	-	0.1 ± 0.16^{a}	-	0.05 ± 0.05^{a}
***Bones and shells (>4mm) (%)**	-	0.02 ± 0.02^{a}	-	0.014 ± 0.023^{a}	-	0.05 ± 0.09^{a}	-	0.02 ± 0.04^{a}

*Mean values and standard deviation of composite samples (n = 6) are shown; Data based on dry weight. The letters a and b mark significant differences between the different Piles for the Output (Kruskall-Wallis test with subsequent multiple Mann Whitney/Wilcoxon rank sum test with Homl – Bonferroni adjusted method; $p < 0.05$). Values without standard deviation did not had replicates.

Table 6 – Average chemical characteristics of input and output and maturity parameters of output

	P1		**P2**		**P3**		**P4**	
Properties	Input	Output	Input	Output	Input	Output	Input	Output
Moisture content (%)	60.2 ± 7.2^{c}	59.8 ± 3.86^{b}	**31.3 ± 19.5^{ab}**	**52.2 ± 1.86^{a}**	**52.4 ± 5.6^{bc}**	**53.8 ± 1.92^{a}**	**16.4 ± 3.1^{a}**	**66.2 ± 1.28^{c}**
Organic matter (%)	**83.6 ± 1.72^{b}**	**55.9 ± 6.72^{a}**	**77.4 ± 9.25^{ab}**	**57 ± 1.84^{ab}**	**74.1 ± 2.7^{a}**	**57.8 ± 3.41^{ab}**	**87.6 ± 3.63^{b}**	**63.1 ± 2.07^{b}**
C/N ratio	**32 ± 10.6^{a}**	**13 ± 0.46^{b}**	**31 ± 13.13^{a}**	**14 ± 0.49^{b}**	**18 ± 7.2^{a}**	**11 ± 0.72^{a}**	**28 ± 7.46^{a}**	**15 ± 0.6^{c}**
C (%)	**40.5 ± 0.19^{b}**	**27.6 ± 2.81^{ab}**	**35.9 ± 4.5^{ab}**	**27.2. ± 1.02^{a}**	**34.2 ± 1.1^{a}**	**27.52 ± 1.95^{ab}**	**41.2± 3.27^{b}**	**30.75 ± 1.03^{b}**
N (%)	**1.27 ± 0.4^{a}**	**2.1 ± 0.18^{a}**	**1.17 ± 0.3^{a}**	**2.02 ± 0.04^{a}**	**1.93 ± 0.7^{a}**	**2.52 ± 0.07^{b}**	**1.47± 0.29^{a}**	**2.1 ± 0.06^{a}**
Ash (%)	**16.4 ± 1.72^{a}**	**44.06 ± 6.7^{ab}**	**22.6 ± 9.25^{ab}**	**43 ± 1.84^{b}**	**25.9 ± 2.71^{b}**	**42.2 ± 3.41^{ab}**	**12.4 ± 3.63^{a}**	**36.9 ± 2.07^{a}**
δ^{13}C (‰)	-27.2 ± 1.19^{ab}	-27.6 ± 0.46^{ab}	-28.39 ± 0.2^{b}	-28.17 ± 0.3^{b}	-26.8 ± 1.19^{a}	-27.8 ± 0.16^{b}	**-24.5 ± 3.87^{a}**	**-27.02 ± 0.2^{a}**
δ^{15}N (‰)	4.68 ± 3.1^{b}	5 ± 0.21^{d}	**2.73 ± 1.04^{a}**	**4.38 ± 0.25^{c}**	2.5 ± 0.27^{a}	3.49 ± 0.11^{a}	4.2 ± 0.5^{ab}	3.94 ± 0.21^{b}
Germination Index (%)	-	162.4 ± 6.5^{a}	-	180.78 ± 7.4^{b}	-	164.62 ± 9.6^{ab}	-	170.13 ± 10.4^{ab}
Root length (cm)	-	7.65 ± 0.2^{a}	-	8.16 ± 0.23^{b}	-	7.79 ± 0.26^{ab}	-	8.42 ± 0.35^{b}
Seed Germination Rate (%)	-	93.3 ± 3.65^{a}	-	88.0 ± 4.52^{a}	-	86.7 ± 3.65^{a}	-	93.3 ± 2.98^{a}
pH	-	7.91 ± 0.15^{ab}	-	8.06 ± 0.03^{b}	-	7.78 ± 0.05^{a}	-	8.13 ± 0.08^{b}

Mean values and standard deviation of composite samples ($n = 6$) are shown; Data based on dry weight. The letters a, b and c mark significant differences between the different Piles for each group, Input and Output. The letters in bold mark significant differences between groups for each Pile (ANOVA with subsequent multiple Welch T-test for OM, C and Ash; Kruskall-Wallis test with subsequent multiple Mann Whitney/Wilcoxon rank sum test with Homl – Bonferroni adjusted method for MC, C/N ratio, N, δ^{13}C, δ^{15}N, GI, RL, SGR and pH; $p < 0.05$)

3.6. Effect of base material

In order to compare the effect of base material in the output chemical components, a pairwise T-test between feedstock fractions (green and brown) and the final product (compost) was performed. Boxplots in figure 5 illustrate this analysis. OM and C content substantially differ between both feedstock fraction and the final product (pairwise Welch T-test, $p < 0.05$). However, the brown fraction shows higher significances for both parameters. N content and C/N ratio were significantly different between the brown fraction and compost (pairwise Welch T-test, $p < 0.05$) even though there were no significantly differences between the green fraction and compost (pairwise Welch T-test, $p = 0.21$ for N and $p = 0.11$ for C/N ratio). $\delta^{15}N$ showed no significantly different between both feedstock fractions and the final product (pairwise Welch T-test, $p = 0.97$).

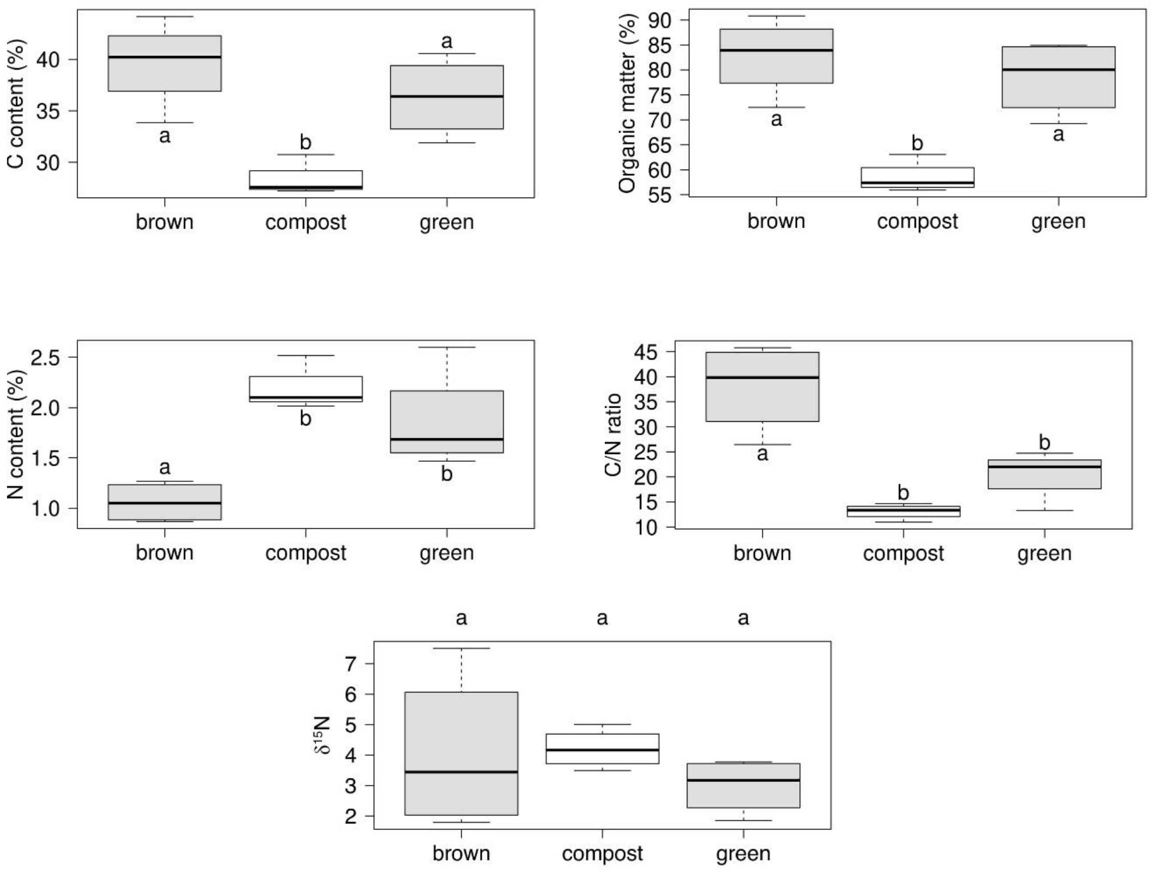

Figure 5 – Differences in C, OM, N, C/N ratio and $\delta^{15}N$ between both feedstock fractions (green and brown) and compost. Replicates are n = 4 and means from 3 and 6 sub-samples each , for feedstock and compost, respectively. The letters a and b indicate significant differences between the groups (ANOVA test with subsequent multiple Welch T-test; $p < 0.05$)

3.7. Granulometry and inert material content

The degradable/inert proportion can be seen in figure 6 (A). It ranged from 74.6% and 25.35% for Pile 2, to 96.4% and 3.6% for Pile 4, for degradable and inert fraction, respectively. The granulometric analysis of degradable and inert material can be seen in table 5. The proportion of the different inert materials and degradable material for each pile can be seen in figure 6 (B and C). The material revealed a high content of particles with sizes between 8mm and 12mm with no significant difference between piles, corresponding on average to 49.1%. The remaining fractions comprised on average 18.9% and 14.9% for under 4mm and between 4 and 8mm, respectively. Three groups were separated in the under 4mm fraction, with percentages ranging from 9.76% in pile 2 to 26.28% in pile 4. Between the 4 to 8mm fraction two groups were detected with values ranging from 9.55% (P2) to 22.9% (P4). Pile 2 had the lowest and pile 4 the highest percentages for both degradable fractions, except for the 8 to 12mm fraction. Pile 4 was significant different from the remaining piles between 4 and 8mm degradable fraction and for stones between 8 and 12 mm. A pairwise Welch T-test confirmed both of these differences ($p<0.05$). On average, the degradable fraction in pile 4 between 4 and 8mm comprised 22.9% of the total sample, in contrast with the rest of the piles, where it ranged from 9.55% (P2) to 15.09% (P3). For stones between 8 and 12mm pile 4 contained on average 2.6%, while the remaining piles ranged from 16.98% (P1) to 24.1% (P2). On average stones between 4 and 8 mm in pile 1 comprised 0.75% in contrast with the remaining piles, ranging from 0.08% (P3) to 0.15% (P4). The weights of glass, textiles and bones and shells in the different fractions (4-8mm and 8-12mm) were summed since the values were too low. The lower size limit (under 4mm) was selected as it represents a practical limit for the visual sorting process used. Between the inert categories above-mentioned, significance letters only confirm differences between piles in the percentage of plastic and metal content between 4 and 8mm. Pile 1 had the lowest content of plastic in this fraction (0.008%) and Pile 2 had the highest (0.05%). T-tests results between both plastic and metal fractions show significant differences (T-test, $p<0.01$) with the 8-12mm fraction, corresponding on average to 0.8% of the samples, in contrast with the 4-8mm fraction representing only 0.03%.

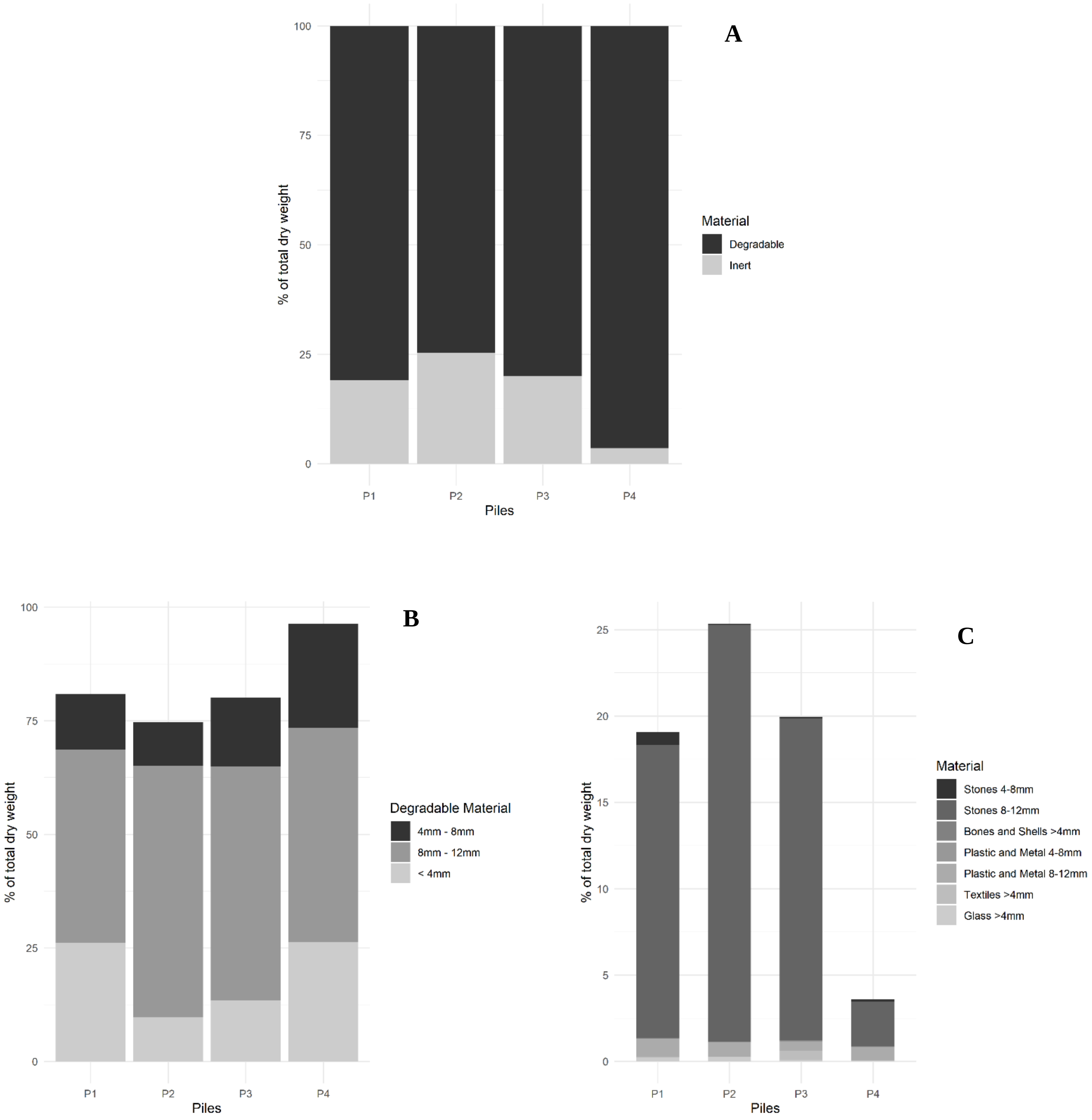

Figure 6 – Percentage of total inert material versus total degradable material (A), percentage of total dry weight of degradable materials (B), percentage of total dry weight of inert materials (C) present in the different GW composts piles (n=4)

3.8. Temperature profiling

Figure 7 presents the mean temperatures and daily precipitation along the experiment and figure 8 presents the temperature profiles of the four piles during the composting process. Pile 2 and 4 maintained thermophilic temperatures (≥ 50℃) for five weeks, while pile 1 sustained these temperatures for seven weeks and pile 3 for three weeks (table 7). Pile 1 had the lowest temperature peak (60.2˚C by day 23) and pile 4 had the highest (64˚C by day 29). The declines in temperatures took place after the day 175 for pile 1, 149 for pile 2, 75 for pile 3 and 72 for pile 4. While temperatures in pile 1 did not approach the ambient level during the 204 days (lowest temperature of 37.6˚C), pile 4 reached the lowest temperatures of the experiment in the end of the cycle (23˚C by day 113). Piles 2 and 3 lowest temperatures were 29.7℃ by day 217 and 28.5℃ by day 169, respectively. Temperatures in piles 2 and 3 had peaks of 61.6˚C by day 159 and 62.7˚C by day 7, respectively. In order to compare the temperatures in the different compartments a pairwise Wilcoxon rank sum test was performed, as it is indicated by significance letters in figure 9. The average temperatures in the 1st, 2nd and 3rd compartment were 51.6℃, 41.5˚C and 37.2˚C, respectively.

Table 7 – General description of time and temperatures during the cycles

Properties	P1	P2	P3	P4
Composting time (days)	204	223	175	120
Temperature ≥ 50˚ (weeks)	7	5	3	5
Average temperature during the cycle (˚C)	48.7 ± 6.9	44.0 ± 8.9	38.9 ± 11.3	43.4 ± 13.7

Mean values and standard deviation of temperatures during the cycles are shown;

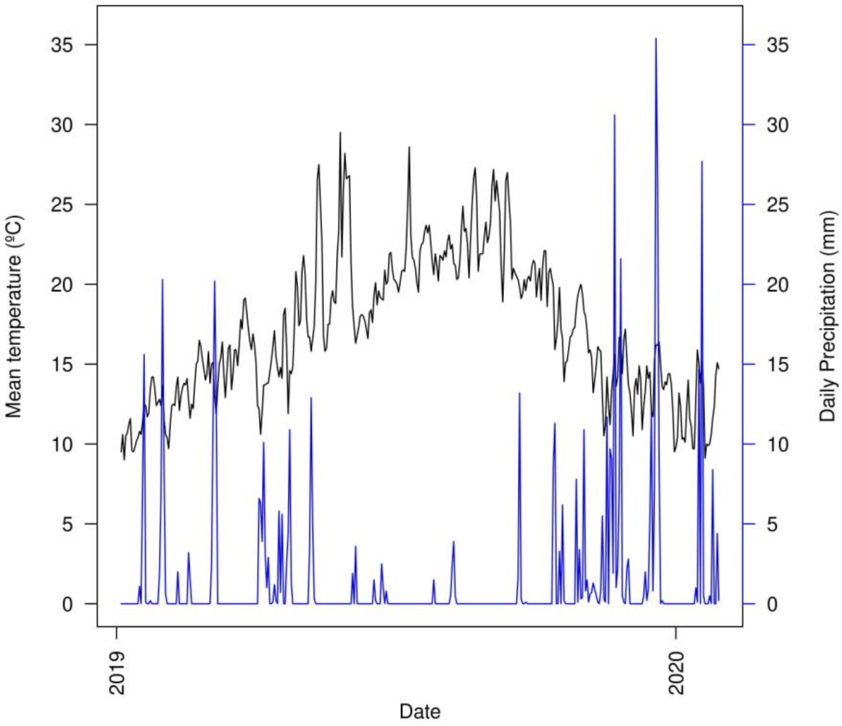

Figure 7 – Mean temperatures and daily precipitation along the experiment. Data provided by Instituto Dom Luiz (IDL) research center.

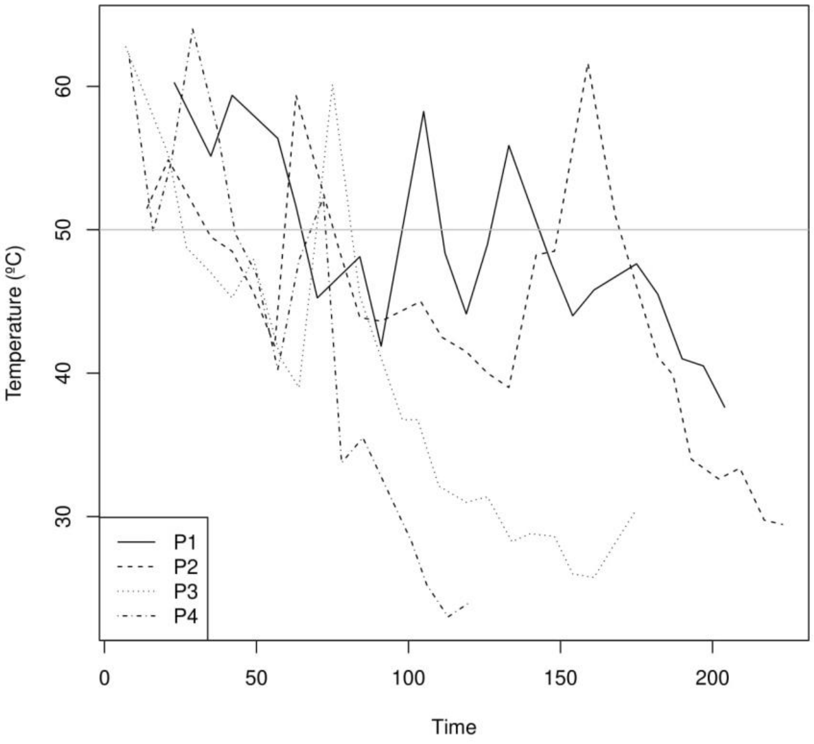

Figure 8 – Temperature development of piles along the experiment. Time expressed in days.

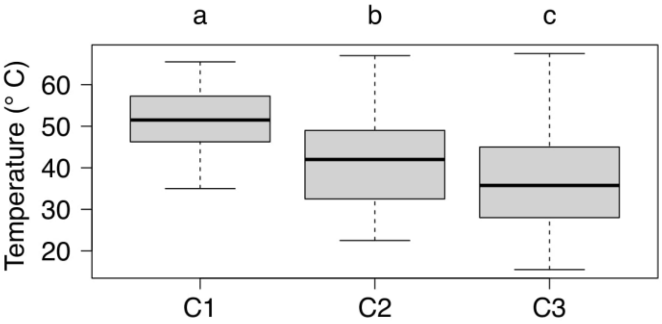

Figure 9 – Differences in temperatures between the compartment 1 (C1), 2 (C2) and 3 (C3). The letters a, b and c indicate significant differences between the compartiments (Kruskall-Wallis test with subsequent multiple Mann Whitney/Wilcoxon rank sum test with Homl – Bonferroni adjusted method; $p < 0.05$)

3.9. Multivariate analysis

The multivariate relationships between green waste compost (GWC) physical and chemical composition, inert content, GI, root length (RL) and seed germination rate (SGR) was determined by a principal component analysis (PCA) (figure 10) where it is possible to distinguish four separated groups (P1, P2, P3 and P4). The first two axes accounted for 44% of the overall variation and are graphically displayed in figure 10. The first axis differentiates between samples analysed from P2 and P3 (left of the diagram) and P4 (right of the diagram) and details 27% of the total variation between GWC samples. The second axis, which explicates 17% of the total variation of GWC, differentiates P1 according to the $\delta^{15}N$ and stones between 4 and 8mm.

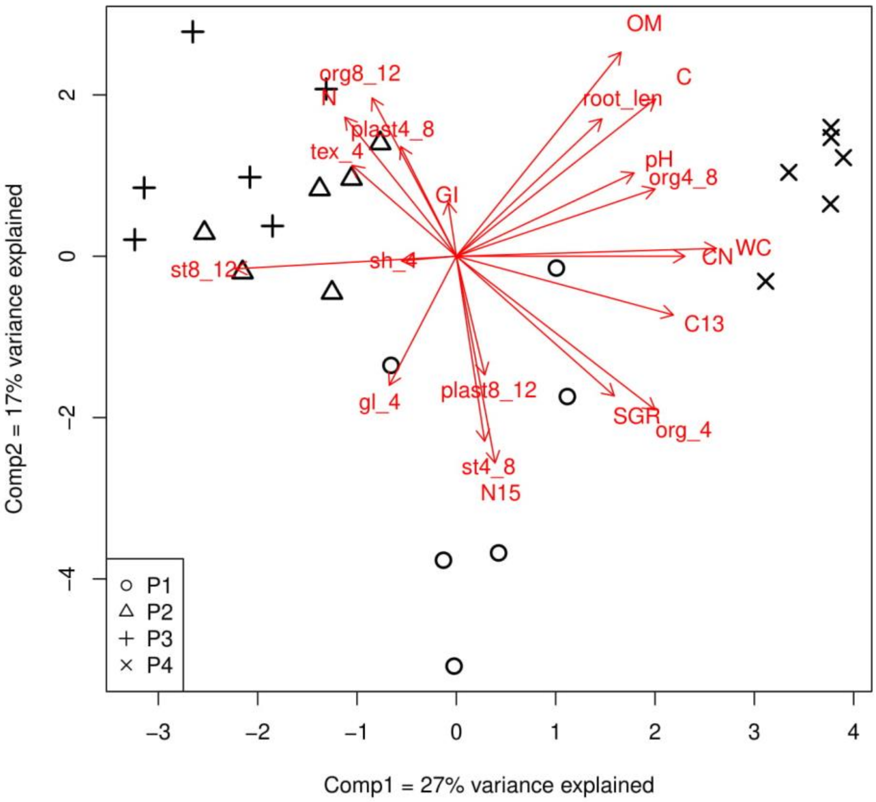

Figure 10 – PCA ordination biplot of physical and chemical composition, inert content, GI, root length (RL) and seed germination rate (SGR) from the different GWCs (P1, P2, P3, P4)

4. Discussion

4.1. Viability of the final product

Organic matter and moisture content

When the composting process is correctly performed, a stabilized form of OM ready to use as a stimulant to restore soil properties (Maheshwari, 2014) is produced. It can improve physical and chemical properties of soil, such as soil structure and water holding capacity (Khaleel *et al.*, 1981) as well as increasing pH in acidic soils and nutrient availability to plants (Feller *et al.*, 2010; Oldfield *et al.*, 2018). It has also been shown that applying compost to soil has beneficial effects on microbial biomass and soil enzyme activities, with a positive influence on plant productivity parameters (Pérez-Piqueres *et al.*, 2017).

Compost production in Portugal is regulated by the law decree nº103/2015 (Ministério da Economia, 2015) that provides legal limits applicable to products to be placed on the market as organic fertilizers. It states that fertilizer materials from organic waste must contain a minimum of 30% of OM (to dry matter). In the system described here, OM levels were measured in the initial fresh feedstocks, as well as in the final products. The initial values ranged from 74.1% to 87.6% (table 4) and the final products ranged from 55.9% to 63.1% (table 6), which can be considered high OM values. OM in soil is an important source of nutrients, a stabilizer of soil structure and a substrate for biological activity. These improvements are directly related to soil productivity and could be helpful in urban green areas by improving growing conditions of trees and shrubs that are usually cultivated in poor quality substrates, often compacted and with reduced drainage and aeration (Sæbø & Ferrini, 2006). This often happens in many urban green sites as a consequence of the lack of natural topsoil layer, where organic matter and microbial life can be found (Sæbø & Ferrini, 2006).

Moisture is essential for OM biodegradation, as it affects microbial activity on solid substrates (Richard *et al.*, 2002). Some authors (Belyaeva *et al.*, 2012; Zhang & Sun, 2014, 2016) have proposed an optimum initial MC for GW composting between 60% and 70%, although this range will vary depending on the nature and the porosity of the material. Initial feedstocks with low porosity require lower MC than a substrate with high porosity (Diaz LF & Savage GM, 2007).

In this research, moisture levels of inputs were measured within a range of 16.4% to 60.2% (table 6). Comparing these values with literature data referent to GW composting systems, the lower value was under the minimum value reported (21.1%) although the remaining initial water contents were within the values commonly reported (Reyes-Torres *et al.*, 2018). In the final product, MC ranged between 52.2% and 66.2% (table 6) which is considered high when compared with the allowed maximum moisture levels for organic fertilizers (40%) and was probably due to daily precipitation events (figure 7). Therefore, it illustrates the difficulty to maintain optimal MCs during outdoor composting and in this case the use of covers or letting the compost dry might have result in lower moisture contents in the final products.

pH and germination index

pH is an efficiency indicator of the process of composting as well as an indicator of maturity of the final product. It relates to the type of microorganisms involved in the biodegradation of the organic compounds. According to Atalia *et al.* (2015) the optimum pH range that favours the growth of bacteria is between 6.0 and 7.5, whereas fungi tolerate a wider pH range (5.5 to 8.0). Several studies found that pH during GW composting is in the range of 7.5 to 8.5 (Zhang & Sun, 2016).

In this experiment the initial pH values of raw materials were not measured, however the values reported in GW related publications (Zhang & Sun, 2014, 2016) are in the range of 7.01 to 7.33. In the final compost samples, the pH ranged between 7.8 and 8.1 which is in accordance with standard legal limits (between 5.5 and 9.0) although the values are higher than the ones found in GW related literature (between 6.62 and 7.00) (Zhang *et al.*, 2013; Zhang & Sun, 2014, 2016).

The germination test can be used to gauge compost maturity. Bernal *et al.* (2009) suggested that composts could be considered mature with germination indexes between 80% and 90% and highly mature above 90%. Given the proposed limit (90%) the final products can be considered highly mature and non-phytotoxic, with GIs ranging from 162.4% to 180.78%.

The GI values obtained were especially high comparing with the ones in literature. Zhang & Sun (2016) obtained compost with a high GI (between 90% and 160%) by adding 25% to 35% of mature GW compost to the GW initial mixture. It also has been reported by Tai & He (2007) that by adding mature compost to the initial mixture, microbial diversity was provide, the length of composting was reduced and the OM degradation rate and GI on the final product was improved. In the system described here, part of the substrate would stay in the bottom of the previous compartment when turning the piles to the next compartment, thus, the substrate that remains might function as a microbial inoculate for the following piles which might have led to these especially high GIs. Even though the PCA biplot (figure 10) showed that GI was not a good indicator of response, an interesting positive corelation was visible between OM and root length.

Granulometry, horticultural usage and stones content

Compost usage is strongly dependent on particle size. Stoffella and Kahn (2001) suggested particle sizes adequate for different horticultural applications, recommending > 25mm for soil amendment, vegetable crops or planting beds, > 13 mm for potting media and > 10mm for landscape mulch. However, any compost with particles sized under 10mm in usually considered marketable (Brinton, 2005). According to the granulometry criteria under the Portuguese standards, 99% of the material that constitutes the product must pass through a mesh screen of 25 mm. In the system under study all the material passed through a 12mm sieve in order to retain part of the matter that did not decompose during the cycle. Very often, these materials are branches with large dimensions and high content of lignin, with a consequently low microbial degradability rate due to the decrease of the available surface area (Komilis & Ham, 2003). The granulometric analysis of degradable material revealed a high content of particles (dw/dw) with sizes between 8mm and 12mm, corresponding on average to 49.1%, and remaining fractions comprised on average 18.9% and 14.9% for under 4mm and between 4 and 8mm, respectively. These dimensions make the compost under study suitable not only for potting media, which is one of the main uses of this product, but also for mulch. Organic mulching is an important practice in permaculture since it helps to prevent soil drying and rainfall erosion, by retaining water, providing habitats for useful insects and microorganisms and moderating soil temperatures (Sæbø & Ferrini, 2006).

The compost produced in this system had an excessive percentage of stones between 8 and 12mm for piles 1, 2 and 3 (17% to 24.1%) which is above the limit of 5% (dw/dw) imposed by Portuguese statutory standards (Ministério da Economia, 2015). As indicated in table 5, pile 4 was significantly different from the remaining piles either in the degradable fraction between 4 and 8mm as well as for stones between 8 and 12 mm. The principal components analysis (PCA) biplot (figure 10) projections suggested a negative correlation between stones (8 to 12mm) and degradable material between 4-8mm and under 4mm and these relationships were confirmed (Pearson's correlation, rp = -0.768, $r^2 = 0.57$, $p < 0.01$ and Spearman's correlation, rs = -0.617, $r^2 = 0.33$, $p < 0.01$, respectively). Moreover, at a multivariate level (figure 10), pile 2 and 3 stand out indicating a higher stone content and a higher degradable fraction between 8 and 12 mm, suggesting that these piles had the coarser substrate fraction. In contrast, this analysis indicated that pile 4 had a higher degradable fraction between 4 and 8mm

4.2. The challenge of man-made non-biodegradable pollution on campus

The man-made non-biodegradable content (dw/dw) in this experiment was divided in three categories: plastic and metal, textiles and glass. The maximum values for the total man-made inert material content under the legal limits applicable to organic fertilizers varies according to the class of usage: Class I and II – between 0.5 and 1% for agriculture, Class IIA – up to 2% for arboreal and shrub crops in particular orchards, olive groves and vineyards and forest species, Class III up to 3% for soils where crops planted are not for human or animal consumption (e.g. landscape restoration, fertilization of forestry soil). Taking into account these legal limits, the final products from pile 4 are in accordance with the criteria for all the defined classes (average value of 0.85%) and Pile 1, 2 and 3 (average values of 1.3%, 1.1% and 1.2%) fit in the criteria defined for Classes IIA and III. Moreover, the PCA projections (figure 10) indicate that pile 4 was the group with low impurities.

Although the man-made non-biodegradable fraction was under the values required by the statutory standards, contamination is still a problem, however, a significant amount of pollutants was retained in the 12mm sieve. Also, a visual source separation is performed before and during each composting cycle by volunteers that collect any observable contaminant. Source separation in GW composts was found to have also a positive influence on the reduction of heavy-metals content (Brinton, 2005). Capturing clean organic materials from the green spaces of the university might be difficult due to the variety of non-biodegradable disposable packaging available in the commerce nearby. As mentioned above, in the system described here, tests indicated significant differences between plastic and metal fractions. The fraction between 8 and 12mm corresponded on average to 0.8% of the samples, in contrast with the fraction between 4 and 8mm that comprised only 0.03%. In order to reduce the amount of plastics and metal in the final product, it is useful to know the fraction where they are more prevalent to assess where to intervene, which in this case is would be between 8 and 12mm. However, it is not a solution for the main problem of campus contamination with disposable products while education and advocacy for more sustainable habits towards a campus free of single use products might be.

4.3. Temperature profiling during composting

According to figure 8, temperature profiles obtained in this study represent the thermophilic phase, indicated by temperature peaks at the beginning of the four composting cycles, and the cooling phase, as the maturation occurs and temperatures start decreasing until the lowest values. There was a rapid establishment of the thermophilic phase, since temperatures above 50 ˚C were reached between day 7 and day 23, probably due to the decomposition of easily degradable compounds. However, the mesophilic phase (up to 42 ˚ C) was not clearly defined, as piles reached new peaks with temperatures typical of the thermophilic phase. These increases in temperature were observed after the turning events and it might be an indication that some of the material was undecomposed even after being exposed to high thermophilic temperatures for several weeks, suggesting that either the access to N rich materials or oxygen might have been a limiting factor. GW contains raw materials that are especially resistant to biodegradation, such as lignin compounds (Zhang & Sun, 2016; Reyes-Torres *et al.*, 2018) what might support this hypothesis. Also, while analysing the overall temperature values for each compartment, there were statistical differences between the three sections (figure 9) which might be an indication for the three main stages of the composting process. The average temperature in the 1st compartment was 51.6℃, in the 2nd was 41.5 ° C and in the 3rd was 37.2 ° C, corresponding to thermophilic, mesophilic and cooling temperatures, respectively (Maheshwari, 2014).

Effect of ambient temperature on compost temperature

Pile 1 was the only one that never reached temperatures typical of substrates in the maturation phase, although the maturity parameters like GI or pH had suitable values. This might be a consequence of the fact that pile 1 was removed in the beginning of July, with relatively high ambient temperatures (figure 7). Furthermore, according to Raj (2011) the criteria to consider the substrate a mature compost is that the temperature in the center of the pile should be ± 5 ° C when compared with the surrounding temperature. This might also be an indication of the capacity of preservation of heat emitted by the process and induced by the surrounding ambient temperature, even in the cooling phase (Oviedo-Ocaña *et al.*, 2015). Spearman's rank correlations were found between ambient and pile temperatures during the cycle for the majority of piles expect for pile 3. Interestingly, while for pile 1 (Spearman's correlation, rs = -0.55, $p < 0.01$) and 2 (Spearman's correlation, rs = -0.37, $p = 0.06$) are negative monotonic relationships, pile 4 exhibits a positive relationship (Spearman's correlation, rs = 0.61, $p = 0.01$). However, since pile temperatures are higher than ambient temperatures for the most part of the cycle, this was probably the result of a coincidence between the time ambient and pile temperatures start to decrease due to the beginning of the winter and the beginning of the maturation phase.

4.4. End of the cycle: The final balance

The experiment described here followed the composting cycle in the campus of the Faculty of Sciences for almost a year, since pile 1 was created in December 12, 2018, and pile 4 was removed in November

21, 2019. During the year, 2322 kg (dw) of waste from the 15 000m^2 green areas in the campus were composted (1211 kg (dw) of brown fraction, 1111 kg (dw) of green fraction) and transformed into 1508 kg (dw) of substrate.

Carbon and mass losses during pile composting

To examine actual losses in total C, total N and biomass, mass balances were calculated (table 5). In general, total mass, total carbon and biomass declined as the organic matter degraded during the composting process, regardless of the feedstock's variability. C losses were higher in piles 1, 2 and 4 (54.3% to 60.2%) than in pile 3 (29.7%) (table 5). Total mass and biomass losses showed the same pattern (table 5). N losses were only observable in pile 4 (9.8%) (table 5). C/N ratio also declined in all piles as a result of the total carbon losses relative to total N losses. However, the decrease in C/N ratio was substantially lower in pile 3 when compared with the remaining piles (table 6).

Figure 11 illustrates the relationship between input and output of C, N, total mass, biomass and C/N ratio contents in the different piles. It clearly indicates that pile 3 had a lower biodegradation rate compared to the remaining piles, expressed in carbon, biomass and total mass values. However, the nitrogen input and output ratio show that pile 3 retained this nutrient as much as the other piles. During composting, the carbon loss occurs as a consequence of bio-oxidation of C to CO_2 (Tiquia *et al.*, 2002). Microorganisms and the enzymes they produce are responsible for the biodegradation process (Diacono & Montemurro, 2011). Therefore, microbial activity will lead to CO_2 release and heat production as well as temperature, oxygen and moisture gradients in the substrate matrix.

Several studies have modelled degradation kinetics in composting as a function of temperature (Bari *et al*. 2000), moisture (Richard *et al.*, 2002), and aeration (Richard *et al.*, 2006; de Guardia *et al.*, 2008). In the system described in this study all piles were turned twice during each cycle, however, pile 3 had the lowest total mass and carbon losses as well as the lowest average temperature during the cycle and the lowest number of weeks maintaining thermophilic temperatures (table 7), suggesting a lower oxygen level (Tiquia *et al.,* 2002). Also, pile 3 appear to have conserved more nitrogen (table 5). For some important quality criteria, such as GI and RL, pile 3 values were lower compared with the ones in piles 2 and 4 (table 6). As mentioned before, at a multivariate level (figure 10), it was clear that GI was not an adequate indicator of variance between piles while SGR seems a much more suitable one. When comparing SGR between piles, although there was not a statistically significant difference between piles, pile 3 had the lowest value, indicating that some important criteria were not readily achieved. Furthermore, pile 3 had the highest green to brown ratio (table 3). A reduced proportion of woody material (generally in the brown fraction) in the compost feedstock might have had negative effects by inhibiting aeration and consequently the composting process (Vandecasteele *et al.*, 2016).

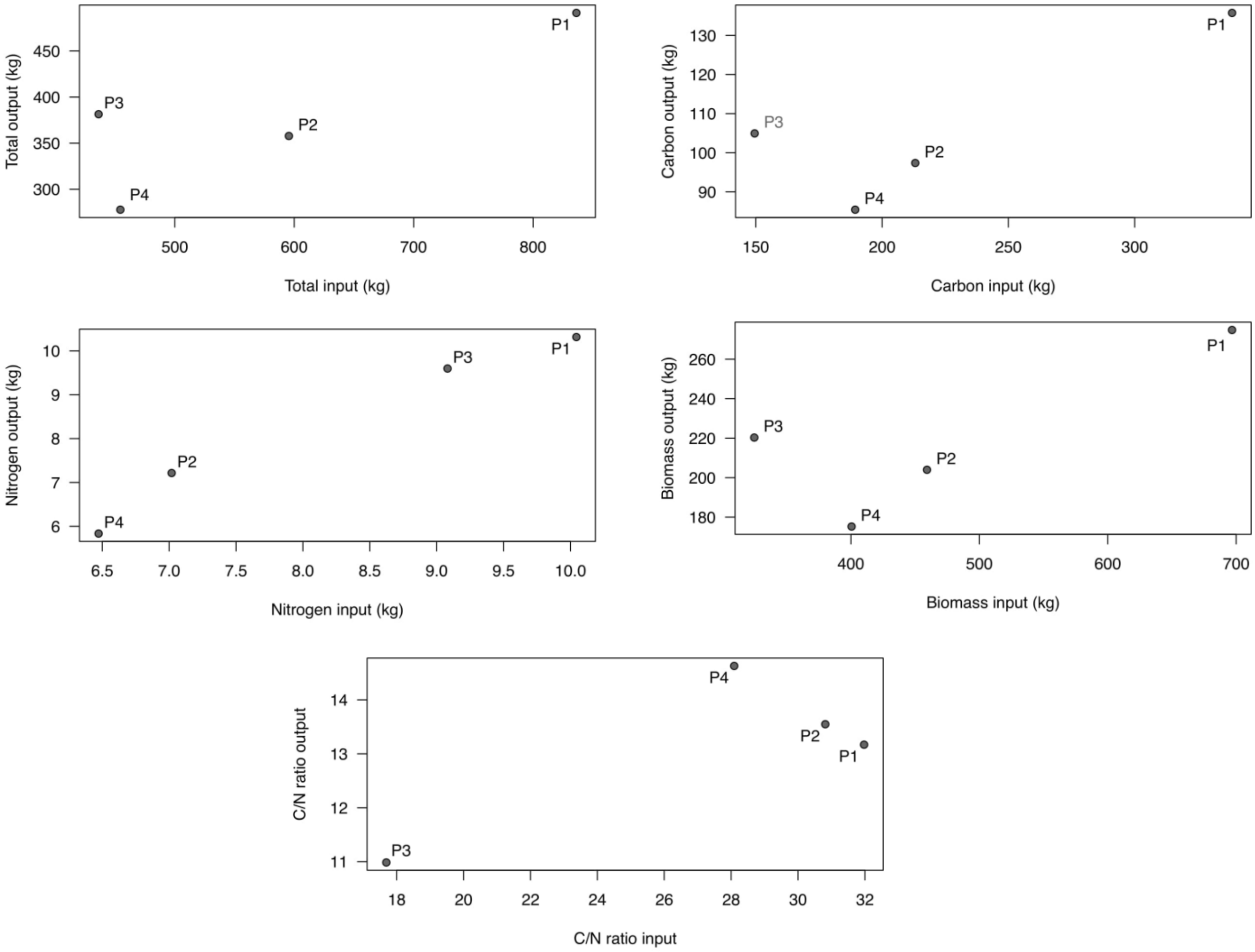

Figure 11 – Relationship between input and output of total mass, carbon, nitrogen, biomass and C/N ratio contents in the different piles (n = 4)

Nitrogen losses and conservation strategies

Nitrogen loss is an important issue during composting not only from a nutrient conservation standpoint but also from an environmental health perspective. As mentioned above, nitrogen losses in the experiment were only observable for pile 4 (table 5). In the remaining piles there seems to have been an increase from the initial nitrogen content, ranging from 2.7% to 5.7% (table 5). The increase in total nitrogen is probably due to fact that the rate of total mass loss exceeded the rate of nitrogen loss (Hao 2004) as the organic matter is oxidized and CO_2 is released. Nitrogen losses can be due to a high initial nitrogen content, a low C/N ratio, high ammonia (NH_3) losses and high oxygenation rate (Michel & Reddy, 1998; Larsen & McCartney, 2000; Raviv *et al.*, 2004). Pile 4 had the highest pH (figure 10 and 12) , which might have contributed to higher NH_3 emissions, since volatilization of NH_3 increases with pH (Hao *et al.*, 2004; Rochette *et al.*, 2013) leading to higher nitrogen losses. As mentioned in table 7, pile 4 maintained thermophilic temperatures (≥ 50 ˚C) for five weeks during the 120 composting days. This might have contributed to the high nitrogen losses since high temperatures during the thermophilic phase can lead to a rapid loss of organic matter (Raviv *et al.*, 1999). Pile 3 had the highest increase in total nitrogen content during the cycle, despite having the highest initial nitrogen content and the lowest C/N ratio of initial mixture. Previous studies (Hao *et al.*, 2004; Lynch *et al.*, 2006) reported high N conservation during composting as a result of an initial high C/N ratio and low pH. While the initial C/N ratio is seemingly inconsistent with the values obtained, the low pH values (values highlighted in figure 12) could explain the high N conservation. Since nitrification is an acid producing reaction (Fauci *et al.*, 1999) a higher nitrogen content might explain the lower pH in pile 3 (figure 12). Interestingly, while no correlations between pH and nitrogen content were found, a strong positive and linear relationship was found between pH and C/N ratio (Pearson's correlation, $r = 0.82$, $r^2 = 0.657$, $p < 0.01$) and a monotonic relationship between pH and the carbon content (Spearman's correlation, $r = 0.61$, $r^2 = 0.296$, $p < 0.01$) as indicated by the red lines in figure 13. Since C/N ratio shows a higher correlation with pH than C, and no clear relationship with N was found, it might suggests that anly C and N together explain changes in pH. The relationships between these variables are easily seen at a multivariate level, illustrated in figure 10, with pH, C/N ratio and C grouped together while N is separated by a ≈ 90 ° angle from these variables, indicating that is not likely to be correlated.

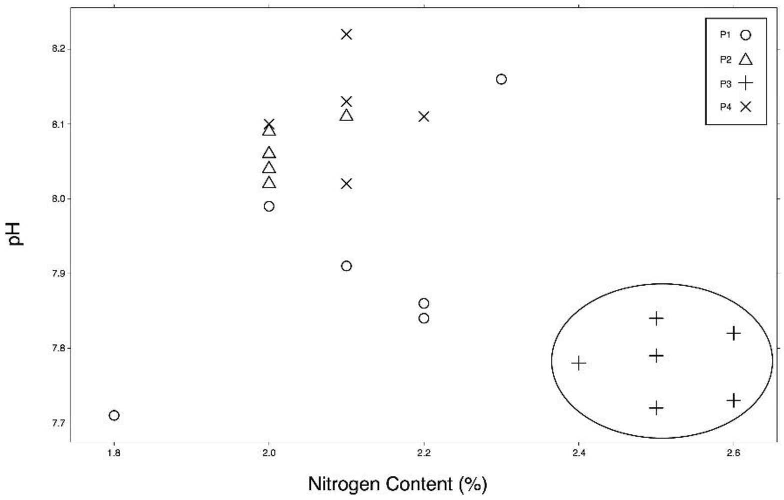

Figure 12 – Scatterplot for the relationship between pH and total nitrogen content (%) of the final products from the different piles.

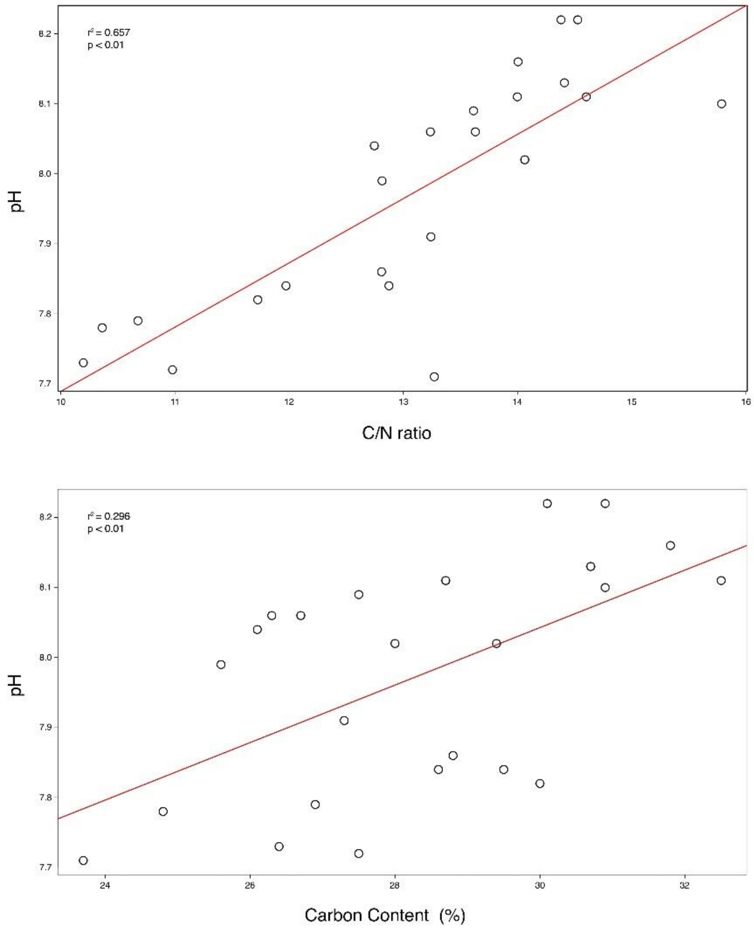

Figure 13 – Scatterplots and Pearson’s correlation coefficient for the linear relationships between pH and C/N ratio (upper) and carbon content (lower) of the final products from the different piles.

Changes in δ^{15}N during composting

The nitrogen isotope fractionation occurs in all biochemical and physical processes, including mineralization, ammonia volatilization, nitrification and denitrification (Hogberg, 1997). According to Lynch *et al.* (2006) δ^{15}N enrichment is a result of a combination of fractionation mechanisms, such as a shift to more complex nitrogen compounds and fractionation during NH_3 volatilization.

In this study, in the majority of piles there was relatively low δ^{15}N enrichment during the process, probably mainly due to nitrogen losses. Pile 4 was the exception with a slight δ^{15}N depletion (-0.26‰) which was not expected. Pile 4 had the higher nitrogen loss during the cycle which would lead to a greater ^{15}N enrichment (Hogberg, 1997). A study conducted by Kim *et al.* (2008) showed that δ^{15}N of compost is not only a consequence of nitrogen losses but also of δ^{15}N in nitrogen sources, and higher δ^{15}N in feedstocks might lead to a ^{15}N isotope dilution, by retarding the increase in δ^{15}N. This could be an explanation for the δ^{15}N depletion in pile 4, since it had the highest values of δ^{15}N in the green fraction, and the second highest in the brown fraction (table 4). The principal components analysis (PCA) biplot (figure 10) differentiates P1 in part according to the δ^{15}N, although it is more related to the second principal component. Moreover, the vector projections suggest a light negative correlation between N and δ^{15}N, which was confirmed by the negative linear correlation found between these two variables (Pearson's correlation, rp = -0.54, $r^2 = 0.263$, $p < 0.01$) as indicated by the red line in figure 14.

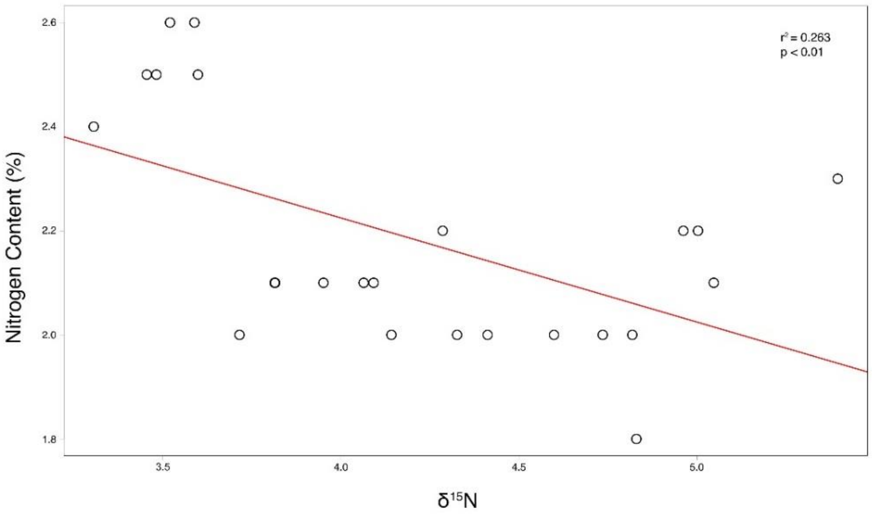

Figure 14 – Scatterplots and Pearson's correlation coefficient for the linear relationship between N content and δ^{15}N of the final products from the different piles.

Conclusion

In FCUL’s community-scale composting system, during a year, 1508 kg (dw) of substrate were produced from 2322 kg (dw) of composted green waste out of the 15000m^2 green areas of the university campus. This study demonstrates that with the development of decentralized and community managed compost facilities it is possible to reach more sustainable patterns in urban areas. Recycling organic wastes from gardens and parks, and transforming them into stabilized hummus rich-substrates that can help to improve the poor quality of urban soils, by suppling nutrients, organic matter and microbial life. Therefore, the main conclusions and recommendations resulting from the present study are:

- The majority of physicochemical parameters of the final composts fulfilled the quality criteria imposed by the Portuguese standards applied to organic fertilizers, irrespectively of the season and feedstock variability. The moisture content values were above the maximum limit, thus a recommendation during outdoor composting would be the use of covers, in order to lower MC in the final products.
- Although the man made non-biodegradable fraction did not interfere negatively with the quality of the final product, since these impurities were under the legal limits, contamination in the green areas of the campus with disposable packaging is still a problem. Significant amounts of pollutants were either collected by visual source separation before the beginning of the cycle, by volunteers during each composting cycle or even retained in the 12mm sieve. A possible solution for this problem might be more education and advocacy for sustainable habits towards a campus free of non-biodegradable single use products.
- Through the promotion of decentralized community composting in the campus, the GW fraction was valued by obtaining high-quality compost while closing the cycle of organic matter either by applying it to the soil or as growing media for new plants. Moreover, it educated the community involved in the process and kept the compost as local as possible.

The production of MSW is an unavoidable consequence of the urban expansion and densification. It is not only urgent to have a more efficient and sustainable MSW management in order to prevent environmental and public health risks, but also to foment the responsibility of the individuals for their own waste production. Local composting it is an alternative to the current MSW management models, and could not only lead to an effective use of waste by transforming it into a local resource but also to engage individuals in a community action for sustainability.

Literature

Portugal-urbanization 2008-2018 | Statista. (https://www.statista.com/statistics/455915/urbanization-in-portugal/. Accessed: 22/6/2020).

Abawi, G.S. & Widmer, T.L. 2000. Impact of soil health management practices on soilborne pathogens, nematodes and root diseases of vegetable crops. Applied Soil Ecology, 15, 37–47.

Adhikari, B.K., Trémier, A., Martinez, J. & Barrington, S. 2010. Home and community composting for on-site treatment of urban organic waste: perspective for Europe and Canada. Waste Management & Research, 28, 1039–1053.

Agência Portuguesa do Ambiente. 2014. PERSU2020 - Relatorio Ambiental.

Agnew, J.M. & Leonard, J.J. 2003. The Physical Properties of Compost. Compost Science & Utilization, 11, 238–264.

Atalia, K.R., Buha, D.M., Bhavsar, K.A. & Shah, N.K. A Review on Composting of Municipal Solid Waste. 10.

Avelar, D., Ulm, F., Alves, J., Lousa, J., Cruz, C. & Maia Alves, J. 2017. Guia do Compostor FCUL.

Bari, Q.H., Koenig, A. (2000). Kinetic analysis of forced aeration composting - II. Application of multilayer analysis for the prediction of biological degradation. Waste Management and Research, 18, 313 - 319.

Beffa, T., Blanc, M., Lyon, P.-F.O., Vogt, G., Marchiani, M., Fischer, J.L. & Aragno, M. 1996. Isolation of Thermus Strains from Hot Composts. APPL. ENVIRON. MICROBIOL., 62, 5.

Belyaeva, O.N., Haynes, R.J. & Sturm, E.C. 2012. Chemical, physical and microbial properties and microbial diversity in manufactured soils produced from co-composting green waste and biosolids. Waste Management, 32, 2248–2257.

Bernal, M.P., Alburquerque, J.A. & Moral, R. 2009. Composting of animal manures and chemical criteria for compost maturity assessment. A review. Bioresource Technology, 100, 5444–5453.

de Bertoldi, M., Vallini, G. & Pera, A. 1983. The Biology of Composting: A review. 20Waste Management & Research, 1, 157-176.

Brinton, W.F. 2005. Characterization of Man-made Foreign Matter And its Presence in Multiple Size Fractions From Mixed Waste Composting. Compost Science & Utilization, 13, 274–280.

van Bruggen, A.H.C. & Semenov, A.M. 2000. In search of biological indicators for soil health and disease suppression. Applied Soil Ecology, 15, 13–24.

Bruni, C., Akyol, Ç., Cipolletta, G., Eusebi, A.L., Caniani, D., Masi, S., Colón, J. & Fatone, F. 2020. Decentralized Community Composting: Past, Present and Future Aspects of Italy. Sustainability, 12, 3319.

Cáceres, R., Malińska, K. & Marfà, O. 2018. Nitrification within composting: A review. Waste Management, 72, 119–137.

Comesaña, I.V., Alves, D., Mato, S., Romero, X.M. & Varela, B. 2017. Decentralized Composting of Organic Waste in a European Rural Region: A Case Study in Allariz (Galicia, Spain). In: Solid Waste Management in Rural Areas (ed. Mihai, F.-C.). InTech.

Cooperband, L. 2002. The Art and Science of Composting.

Council Directive (91/676/EEC) on the concerning the protection of waters against pollution caused by nitrates from agricultural sources. 1991.

Council Directive 94/62/EC on packaging and packaging waste. 1994.

Council Directive 1999/31/EC onthe landfill of waste. 1999.

Council Directive 2008/98/EC on waste and repealing certain Directives. 2008.

Curl, E.A. & Old, K.M. 1988. The role of soil microfauna in plant-disease suppression. Critical Reviews in Plant Sciences, 7, 175–196.

Debosz, K., Petersen, S.O., Kure, L.K. & Ambus, P. 2002. Evaluating effects of sewage sludge and household compost on soil physical, chemical and microbiological properties. Applied Soil Ecology, 19, 237–248.

Diacono, M. & Montemurro, F. 2011. Long-Term Effects of Organic Amendments on Soil Fertility. In: Sustainable Agriculture Volume 2 (eds. Lichtfouse, E., Hamelin, M., Navarrete, M. & Debaeke, P.), pp. 761–786. Springer Netherlands, Dordrecht.

European Environment Agency. 2013. Managing municipal solid waste: a review of achievements in 32 European countries. European Environment Agency, Copenhagen.

European Environment Agency. 2013. Municipal waste management in Portugal.

EUROSTAT, Union européenne & Commission européenne. 2010. Energy, transport and environment indicators. Office for Official Publications of the European Communities, Luxembourg.

Farrell, M. & Jones, D.L. 2009. Critical evaluation of municipal solid waste composting and potential compost markets. Bioresource Technology, 100, 4301–4310.

Fauci, M.F., Bezdicek, D.F., Caldwell, D. & Finch, R. 1999. End Product Quality and Agronomic Performance of Compost. Compost Science & Utilization, 7, 17–29.

Federal Compost Quality Assurance Organization. 1994. Methods book for the analysis of compost: in addition with the results of the parallel interlaboratory test 1993. Federal Compost Quality Assurance Organization, Stuttgart.

Feller, C., Brossard, M., Chen, Y., Landa, E.R. & Trichet, J. 2010. Selected pioneering works on humus in soils and sediments during the 20th century: A retrospective look from the International Humic Substances Society view. Physics and Chemistry of the Earth, Parts A/B/C, 35, 903–912.

Fuchs, J.G. 2010. Interactions Between Beneficial and Harmful Microorganisms: From the Composting Process to Compost Application. In: Microbes at Work (eds. Insam, H., Franke-Whittle, I. & Goberna, M.), pp. 213–229. Springer Berlin Heidelberg, Berlin, Heidelberg.

Garcia-Prendes, R. 2001. Evaluation of dairy manure compost as a peat substitute in potting media for container grown plants. Master thesis.

Gonzalez, R.F. & Cooperband, L.R. 2002. Compost Effects on Soil Physical Properties And Field Nursery Production. Compost Science & Utilization, 10, 226–237.

de Guardia, A., Petiot, C. & Rogeau, D. 2008. Influence of aeration rate and biodegradability fractionation on composting kinetics. Waste Management, 28, 73–84.

Hao, X., Chang, C. & Larney, F.J. 2004. Carbon, Nitrogen Balances and Greenhouse Gas Emission during Cattle Feedlot Manure Composting. J. ENVIRON. QUAL., 33, 8.

Harada, Y. & Inoko, A. 1980. The measurement of the cation-exchange capacity of composts for the estimation of the degree of maturity. Soil Science and Plant Nutrition, 26, 127–134.

Hogberg, P. 1997. 15N natural abundance in soil-plant systems. New Phytol., 137.

Khaleel, R., Reddy, K.R. & Overcash, M.R. 1981. Changes in Soil Physical Properties Due to Organic Waste Applications: A Review. Journal of Environmental Quality, 10, 133–141.

Killi, D. & Kavdır, Y. 2013. Effects of olive solid waste and olive solid waste compost application on soil properties and growth of Solanum lycopersicum. International Biodeterioration & Biodegradation, 82, 157–165.

Kim, Y.-J., Choi, W.-J., Lim, S.-S., Kwak, J.-H., Chang, S.X., Kim, H.-Y., Yoon, K.-S. & Ro, H.-M. 2008. Changes in nitrogen isotopic compositions during composting of cattle feedlot manure: Effects of bedding material type. Bioresource Technology, 99, 5452–5458.

Ko, H.J., Kim, K.Y., Kim, H.T., Kim, C.N. & Umeda, M. 2008. Evaluation of maturity parameters and heavy metal contents in composts made from animal manure. Waste Management, 28, 813–820.

Komilis, D.P. & Ham, R.K. 2003. The effect of lignin and sugars to the aerobic decomposition of solid wastes. Waste Management, 23, 419–423.

Larsen, K.L. & McCartney, D.M. 2000. Effect of C:N Ratio on Microbial Activity and N Retention: Bench-scale Study Using Pulp and Paper Biosolids. Compost Science & Utilization, 8, 147–159.

Leton, T.G. & Stentifordt, E.I. 1990. Control of aeration in static pile composting. Waste Management & Research 8, 29 9-306.

López-Cano, I., Roig, A., Cayuela, M.L., Alburquerque, J.A. & Sánchez-Monedero, M.A. 2016. Biochar improves N cycling during composting of olive mill wastes and sheep manure. Waste Management, 49, 553–559.

Lyberatos, G. & Skiadas, I.V. 1999. Modelling of anaerobic digestion - a review. Global Nest, 2, 63-76.

Lynch, D.H., Voroney, R.P. & Warman, P.R. 2006. Use of 13C and 15N natural abundance techniques to characterize carbon and nitrogen dynamics in composting and in compost-amended soils. Soil Biology and Biochemistry, 38, 103–114.

MAcGREGOR, S.T., Mltler, F.C., Psarianos, K.M. & Finsteini, M.S. 1981. Composting Process Control Based on Interaction Between Microbial Heat Output and Temperaturet. APPL. ENVIRON. MICROBIOL., 41, 10.

Maheshwari, D.K. (Ed). 2014. Composting for sustainable agriculture. Springer, Cham Heidelberg New York.

Mahimairaja, S., Bolan, N.S. & Hedley, M.J. 1995. Denitrification losses of N from fresh and composted manures. Soil Biology and Biochemistry, 27, 1223–1225.

Mao, C., Feng, Y., Wang, X. & Ren, G. 2015. Review on research achievements of biogas from anaerobic digestion. Renewable and Sustainable Energy Reviews, 45, 540–555.

Marçal, A. & Teixeira, A.R. 2017. Resíduos Urbanos Relatório Anual 2017. Agência Portuguesa do Ambiente, I.P. Departamento de Resíduos.

Marugg, C., Grebus, M., Hansen, R.C., Keener, H.M. & Hoitink, H.A.J. 1993. A Kinetic Model of the Yard Waste Composting Process. Compost Science & Utilization, 1, 38–51.

Matthiessen, M.K., Larney, F.J., Brent Selinger, L. & Olson, A.F. 2005. Influence of Loss-on-Ignition Temperature and Heating Time on Ash Content of Compost and Manure. Communications in Soil Science and Plant Analysis, 36, 2561–2573.

McKINLEY, V.L. & Vestal, J.R. 1984. Biokinetic Analyses of Adaptation and Succession: Microbial Activity in Composting Municipal Sewage Sludge. APPL. ENVIRON. MICROBIOL., 47, 9.

Mekki, A., Aloui, F. & Sayadi, S. 2019. Influence of biowaste compost amendment on soil organic carbon storage under arid climate. Journal of the Air & Waste Management Association, 69, 867–877.

Michel, F.C. & Reddy, C.A. 1998. Effect of Oxygenation Level on Yard Trimmings Composting Rate, Odor Production, And Compost Quality In Bench-Scale Reactors. Compost Science & Utilization, 6, 6–14.

Ministério da Economia. Decreto-Lei n.o 103/2015 de 15 de junho.

Ministério do Ambiente. PERSU II - Plano Estratégico para os Resíduos Sólidos Urbanos.

Oldfield, E.E., Wood, S.A. & Bradford, M.A. 2018. Direct effects of soil organic matter on productivity mirror those observed with organic amendments. Plant and Soil, 423, 363–373.

Oviedo-Ocaña, E.R., Torres-Lozada, P., Marmolejo-Rebellon, L.F., Hoyos, L.V., Gonzales, S., Barrena, R., Komilis, D. & Sanchez, A. 2015. Stability and maturity of biowaste composts derived by small municipalities: Correlation among physical, chemical and biological indices. Waste Management, 44, 63–71.

Paradelo, R., Villada, A. & Barral, M.T. 2011. Reduction of the short-term availability of copper, lead and zinc in a contaminated soil amended with municipal solid waste compost. Journal of Hazardous Materials, 188, 98–104.

Pérez-Piqueres, A., Albiach, R., Domínguez, A. & Canet, R. 2017. Effect of soil preparation techniques on the biochemical properties and microbial communities of a citrus orchard after replanting and conversion into organic management. Applied Soil Ecology, 119, 8–17.

Pérez-Piqueres, A., Edel-Hermann, V., Alabouvette, C. & Steinberg, C. 2006. Response of soil microbial communities to compost amendments. Soil Biology and Biochemistry, 38, 460–470.

Perucci, P., Dumontet, S., Bufo, S.A., Mazzatura, A. & Casucci, C. 2000. Effects of organic amendment and herbicide treatment on soil microbial biomass. Biology and Fertility of Soils, 32, 17–23.

Platt, B., McSweeney, J. & Davis, J. Growing Local Fertility: A Guide to Community Composting.

Raj, D. 2011. Evaluation of maturity and stability parameters of composts prepared from agro-industrial wastes. Bioresource Technology, 6.

Raviv, M., Medina, S., Krasnovsky, A. & Ziadna, H. 2004. Organic Matter and Nitrogen Conservation in Manure Compost for Organic Agriculture. Compost Science & Utilization, 12, 6–10.

Raviv, M., Medina, S. & Shamir, Y. 1999. Cocomposting – A Method to Improve Results of Poultry Manure Composting. Compost Science & Utilization, 7, 70–73.

Reyes-Torres, M., Oviedo-Ocaña, E.R., Dominguez, I., Komilis, D. & Sánchez, A. 2018. A systematic review on the composting of green waste: Feedstock quality and optimization strategies. Waste Management, 77, 486–499.

Ribeiro, A., Castro, F., Macedo, M. & Carvalho, J. 2011. Waste management in Portugal and Europe - An overview of the past, present and future. in WASTES: Solutions, Treatments and Opportunities 1st International Conference.

Richard, T.L., (Bert) Hamelers, H.V.M., Veeken, A. & Silva, T. 2002. Moisture Relationships in Composting Processes. Compost Science & Utilization, 10, 286–302.

Richard, T.L., Walker, L.P. & Gossett, J.M. 2006. Effects of Oxygen on Aerobic Solid-State Biodegradation Kinetics. Biotechnology Progress, 22, 60–69.

Rivero, C., Chirenje, T., Ma, L.Q. & Martinez, G. 2004. Influence of compost on soil organic matter quality under tropical conditions. Geoderma, 123, 355–361.

Rochette, P., Angers, D.A., Chantigny, M.H., Gasser, M.-O., MacDonald, J.D., Pelster, D.E. & Bertrand, N. 2013. NH 3 volatilization, soil concentration and soil pH following subsurface banding of urea at increasing rates. Canadian Journal of Soil Science, 93, 261–268.

Ryckeboer, J., Mergaert, J., Vaes, K., Klammer, S., Clercq, D.D., Coosemans, J., Insam, H. & Swings, J. A survey of bacteria and fungi occurring during composting and self-heating processes. 63.

Sæbø, A. & Ferrini, F. 2006. The use of compost in urban green areas – A review for practical application. Urban Forestry & Urban Greening, 4, 159–169.

Santos, J. (2007) Caracterização físico-química e biológica em diferentes laboratórios de produtos obtidos a partir da compostagem de resíduos orgânicos biodegradáveis. Master thesis.

Schlesinger, A.P. (2016) Reframing the processes and perceptions of community composting in New Brunswick, NJ. Master thesis.

Serramiá, N., Sánchez-Monedero, M.A., Roig, A., Contin, M. & De Nobili, M. 2013. Changes in soil humic pools after soil application of two-phase olive mill waste compost. Geoderma, 192, 21–30.

Snowdon P, Raison J, Keith H, Ritson P, Grierson P, Adams M, Montagu K, Bi H, Burrows W, Eamus D (2002) Protocol for sampling tree and stand biomass. Technical report 31, 6-26

Stoffella, P.J. & Kahn, B.A. (Eds). 2001. Compost utilization in horticultural cropping systems. Lewis, Boca Raton, Fla.

Tai, H.-S. & He, W.-H. 2007. A novel composting process for plant wastes in Taiwan military barracks. Resources, Conservation and Recycling, 51, 408–417.

Tiquia, S.M., Richard, T.L. & Honeyman, M.S. (2002) Carbon, nutrient, and mass loss during composting. Nutrient Cycling in Agroecosystems, 62, 15–24.

European Parliamentary Research Service. (2017) Towards a circular economy – Waste management in the EU.

Tuomela, M. 2000. Biodegradation of lignin in a compost environment: a review. Bioresource Technology, 72, 169–183.

United Nations. 2018. World Urbanization Prospects: The 2018 Revision.

Van Gestel, K., Mergaert, J., Swings, J., Coosemans, J. & Ryckeboer, J. 2003. Bioremediation of diesel oil-contaminated soil by composting with biowaste. Environmental Pollution, 125, 361–368.

Vandecasteele, B., Boogaerts, C. & Vandaele, E. 2016. Combining woody biomass for combustion with green waste composting: Effect of removal of woody biomass on compost quality. Waste Management, 58, 169–180.

Villar, I., Alves, D., Garrido, J. & Mato, S. 2016. Evolution of microbial dynamics during the maturation phase of the composting of different types of waste. Waste Management, 54, 83–92.

Warman, P.R. 1999. Evaluation of Seed Germination and Growth Tests for Assessing Compost Maturity. Compost Science & Utilization, 7, 33–37.

Wei, Y., Li, J., Shi, D., Liu, G., Zhao, Y. & Shimaoka, T. 2017. Environmental challenges impeding the composting of biodegradable municipal solid waste: A critical review. Resources, Conservation and Recycling, 122, 51–65.

WRAP (Organization). 2002. Comparison of compost standards within the EU, North America and Australasia. Waste and Resources Action Programme (WRAP), Banbury.

Wu, L., Ma, L.Q. & Martinez, G.A. 2000. Comparison of Methods for Evaluating Stability and Maturity of Biosolids Compost. Journal of Environmental Quality, 29, 424–429.

Zhang, L. & Sun, X. 2014. Changes in physical, chemical, and microbiological properties during the two-stage co-composting of green waste with spent mushroom compost and biochar. Bioresource Technology, 171, 274–284.

Zhang, L. & Sun, X. 2016. Influence of bulking agents on physical, chemical, and microbiological properties during the two-stage composting of green waste. Waste Management, 48, 115–126.

Zhang, L., Sun, X., Tian, Y. & Gong, X. 2013. Effects of brown sugar and calcium superphosphate on the secondary fermentation of green waste. Bioresource Technology, 131, 68–75.