



Composting of pig manure and forest green waste amended with industrial sludge



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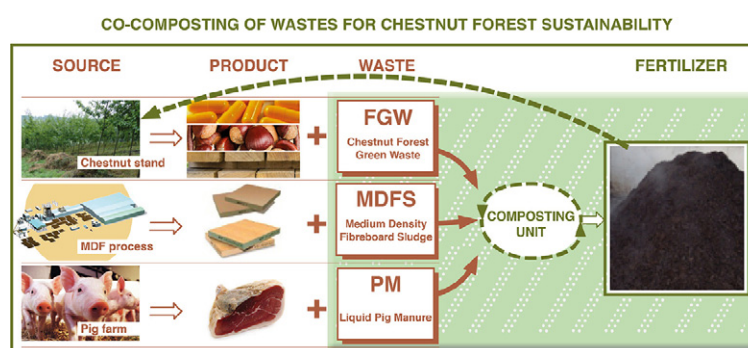
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HIGHLIGHTS

- Successful co-composting of chestnut forest green waste and MDF sludge was achieved.
- Pig manure decreased the C/N ratio and increased moisture content of the mixture.
- Active composting process was sustained for about two months.
- The addition of MDF sludge strongly reduced nitrogen losses during composting.
- The final product was rich in nutrients and low in salinity and heavy metals.

GRAPHICAL ABSTRACT



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ABSTRACT

The aim of this research was to study the composting of chestnut forest green waste (FGW) from short rotation chestnut stands amended with sludge resulting from the manufacture of Medium Density Fibreboard (MDFS) and pig manure (PM). Both FGW and MDFS presented low biodegradation potential but different characteristics in granulometry and bulk density that make its mixture of interest to achieve high composting temperatures. PM decreased the C/N ratio of the mixture and increased its moisture content (MC). Three mixtures of MDFS:FGW at volume ratios of 1:1.3 (M2), 1:2.4 (M3) and 0:1 (M4) were composted after increasing its MC to about 70% with PM. A control with food waste (OFW) and FGW (1:2.4 in volume) (M1) was run in parallel. Watering ratios reached 0.25 (M1), 1.08 (M2) 1.56 (M3) and 4.35 (M4) L PM/kg TS of added solids wastes. Treatments M2 and M3 reached a thermophilic phase shorter than M1, whilst M4 remained in the mesophilic range. After 48 days of composting, temperature gradients in respect to ambient temperature were reduced, but the mineralization process continued for around 8 months. Final reduction in total organic carbon reached 35–56%, depending mainly on the content in MDFS. MDFS addition to composting matrices largely reduced nitrogen losses, which range from 22% (M2) to 37% (M3) and 53% (M4). Final products had high nutrient content, low electrical conductivity and low heavy metal content which make it a valuable product for soil fertilization, right to amend in the chestnut forests and as a pillar of their sustainable management.

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1. Introduction

Nowadays the economies of European countries are approaching the limits of the ecological system in which they are embedded. The effects of the scarcity of resources as well as the environmental impacts from energy, mass losses and emissions are becoming apparent (EEA, 2015). Circular economy and industrial ecology provide new paradigms in which efficient use of resources can be increased and emissions reduced. Integrating the management of farm, forestry and industrial wastes and soil fertility protection can generate new sustainability schemes (Kibblewhite et al., 2012).

Because of the low density of forest biomass generated in many areas of Europe, its collection for energetic or any other kind of exploitation is not economical. Thus, it ends up creating suitable conditions for the spread of forest fires. However, collection viability changes when this biomass is used to solve other problems such as those generated by intensive livestock waste, at least in forested areas close to the location of these farms. In other cases, technological and forest health requirements force the withdrawal of biomass and its subsequent treatment.

In addition to the traditional exploitation of chestnut plantations, e.g. wood, fruit or mixed, recent research emphasize the possibility of obtaining new products with high added value from chestnut forest biomass (Aires et al., 2016; Braga et al., 2015). For this last purpose, chestnut plantations of high density and short rotation are being tested. These new strategies require a periodical collection and removal of plant biomass, during the operations of cultural maintenance of the forest (pruning and thinning of competing weeds and other plant species) and because of efficient mechanized fruit collection (Monarca et al., 2014). This creates large amounts of forest green waste (FGW) for which we must find a use. On the other hand, intensive short rotation forest plantations originate a progressive loss of nutrients and degradation of soil properties (Walmsley et al., 2009; Abbas et al., 2011), which have already been studied by Amorini et al. (1998) for the specific case of the chestnut forests (grove). Thus, FGW composting can generate a useful organic amendment to offset the loss of nutrients and organic matter in forest soils.

There are very few researches about composting of materials from the grove. Guerra-Rodríguez et al. (2001) studied the co-composting of chestnut leaves and chestnut burr mixtures with solid poultry manure, reaching thermophilic phases and positive results for the final product. More recently, Venterino et al. (2013) conducted an experiment using, in addition to leaves and chestnut burr, other components such as shrub species and tree buds. These authors composted the grove waste in piles of about 5.4 m³ on site and with no other additives, in order to simulate the natural on site degradation processes. The material to be composted presented a nitrogen content of 0.5% (dry matter basis) and a C/N ratio of 57.8. During 105 days of composting, mesophilic temperatures (27–29 °C) were reached, only about 5 °C higher than ambient temperature. Guerra-Rodríguez et al. (2001) and Venterino et al. (2013) concluded that, if other additives were not added, the composting of chestnut FGW did not reach a complete degradation because the C/N ratio was too high and the composting material never reached the thermophilic phase. In spite of these difficulties, recent researches highlight the interest and sustainability of FGW composting practices. Venterino et al. (2016) studied the lignocellulosic-adapted microbiota and microbial dynamics during the composting of chestnut residues conducted in a natural environment. These authors concluded that this compost could be considered a high-quality end product for solid-based inoculants in crop systems which contributed for a significant reduction of disease in plants. Iglesias-Díaz et al. (2009) verified the suitability of compost made from crushed gorse as a substrate for growing, peat substitute or propagation of *Thuja plicata* cuttings. López-López et al. (2016) reported that gorse composts exhibit suppressiveness against *F. circinatum* in pine seedlings.

Animal manure and other wastes can be considered as a co-substrate to improve the composting process of FGW. Intensive pig farms

produce high volumes of manure (PM) which must be correctly managed from the environmental and economic points of view. Environmental risks involve greenhouse gas emission, saturation of soils, pollution of ground and surface water by nitrates, phosphorus and heavy metals (HM) in soil (Prapasongsa et al., 2010). On the other hand, farms suffer from increasing costs when pig manure is transported to far farmlands or subjected to conventional technology treatments (Dahlin et al., 2015). The need for low cost and ecological engineered systems therefore arises. However, the high moisture content (MC) of most pig manures (>95%) makes its treatment by composting difficult, although more concentrated manures could be composted by mixing it with amendment and bulking material (Barrena et al., 2011a). Composting approaches to liquid fraction of PM with highly biodegradable organic wastes such as solid fraction of pig manure, sawdust and grape bagasse have been recently proposed (Vázquez et al., 2015a; Cherubini et al., 2015). In fact, due to the negative hydric balance of the composting process, watering the composting material is necessary, which may be carried out with PM (Barrena et al., 2011a). In this way, substantial amounts of PM can be treated by composting, allowing the recovery of nutrients and reducing the volume to be transported to distant crop fields or subjected to further treatment.

Medium Density Fibreboard (MDF) is an engineered wood product made by refining wood into fibres and combining them with wax and resin to form panels by applying high temperature and pressure (Piekarski et al., 2017). The manufacturing industry of MDF settled sharply, with about 60 new facilities in Europe in the last two decades. In Galicia these facilities are of considerable importance with an annual production capacity of 997,000 m³ of board, approximately 60% of Spain's total (Tarling et al., 2016). This process generates wastewater which treatment by physico-chemical methods results in the generation of sludge, amounts varying from 1.5 to 16 kg of total solids (TS) per m³ MDF (EC, 2014; Mymrin et al., 2014). Besides possible recovery and application as new building material (Mymrin et al., 2014), current disposal practices of MDF sludge (MDFS) include incineration on the industry site, land application or sanitary landfill disposal (EC, 2014). In some states and regions in Europe, as is the case in Galicia, the agricultural use of MDFS requires previous stabilization treatment. To our knowledge no previous studies on the composting of MDFS have been reported. Thus, we take into consideration the use of MDFS as a co-composting substrate.

In summary, the recollection of chestnut and the maintenance of the chestnut plantations require forest biomass withdrawal. This causes a lost in fertility in the grove, being necessary to give back to the grove an appropriately processed fertilizer. Animal manures are an additional source of fertilizer elements and in many occasions a waste to be processed. The combination of forest waste and pig manure generated in the same area is formulated in this project as a comprehensive solution to solve these two problems and ensure the sustainability of the chestnut grove. A solid waste from the forestry industry is used as an amendment necessary to improve the composting process. Thus, the main objectives of this study are: a) the determination of the biodegradability potential of biomass from chestnut plantations and sludge from MDF production, b) the appropriate ratio of these materials and pig manure to prepare the composting mixture, c) the characterization of the composting process and the final product.

2. Material and methods

2.1. Solid and liquid wastes

FGW was collected from a short-rotation chestnut (*Castanea* sp., 3,333 plants/ha, 8 years old) local stand. FGW consisted of a mixture of chestnut leaves (about 20%), chestnut burr (5%), chestnut pruned branches (5%), and undergrowth vegetation (70%) composed of gorse (*Ulex* sp.), bracken (*Pteridium aquilinum*), brambles (*Rubus* sp.), brooms

(*Cytisus* sp.) and herbaceous vegetation (*Agrostis* sp., *Holcus* sp. and others). Undergrowth vegetation and pruned branches were harvested from June to August and chestnut leaves and burr from September to November 2014. FGW was shredded to get a particle size below 10 cm and completely mixed before use. MDFS was obtained from a Galician MDF factory. A fattening pig farm provided the PM, which has been stored in a tank (1000 L) provided of mechanical mixing. Organic food waste (OFW) was obtained from a university canteen and used as reference material for the composting process.

2.2. Composters and composting site

Trials were carried out from December 2014 to September 2015 in Galicia (NW Spain, 43.0438°N, 8.3268°W). 340 L composting bins (0.76 × 0.76 m base and 0.80 m height) were used in a covered shed protected from rain. Each composting bin was placed on a concrete surface of 1.20 m² delimited by a partition (15 cm height), whose base and walls were treated with an epoxy primer to prevent water infiltration in the concrete. The base had a slope of 5% allowing the drainage and manual collection of leachate that could be generated.

Four duplicate experiments were carried out. The first duplicate M1 received OFW and FGW at a ratio of 1:2.4, and was used as control. Other three different mixtures were prepared at MDFS:FGW volumetric ratios of 1:1.3 (M2), 1:2.4 (M3) and 0:1 (M4). Solid materials were loaded in layers (alternating different materials in M1, M2 and M3 systems) with a thickness estimated to give 5 layers of each material. Each layer was finely watered with PM at a rate below the respective liquid retention capacity, aiming to reach MC close to 70%. Besides, PM was added during the composting process to M2, M3 and M4 when necessary in order to maintain the adequate MC and increase the amount of PM treated.

2.3. Monitoring, sampling and analysis

The composting mixtures were aerated by turning and mixing the whole mass manually. The water content and densities of the different materials and the resulting mixtures were measured twice a month. Temperature and O₂ concentration at three points ranging from 10 to 30 cm depth were measured two to three times a week during the first month and once a week onwards. Interstitial oxygen was measured by a suction probe SON2IN and an oxygen detector ToxiRAE II (Sensotran, Spain). On the days when the material was turned, measures of temperature and O₂ were performed beforehand. The volume of the matrices was measured periodically by measuring the height of the composting material in the composting containers. In addition, the bulk density (BD) of the composting material was obtained by weighing the material occupying a representative volume (usually 10 cm in height, after turning and mixing the matrix) into the composting bin. The average BD before turning can be obtained taking into account the ratio between the total volume before and after turning. However, the differences recorded were low and we only reported here the BD values after turning.

Composite samples (obtained by mixing four grab sub-samples) from the composting matrices were taken periodically and analysed in the laboratory for MC, total solids (TS) and volatile solids (VS). The initial materials and the final composts were analysed for the same parameters as well as for pH, electrical conductivity (EC) and NH₄⁺-N. Analyses were carried out following Standard Methods (APHA, 2005). pH, EC and NH₄⁺-N in solid samples were measured in 1:5 weight:volume water extracts. Besides, elemental composition (NPK and other nutrients), total carbon (C), total organic carbon (TOC) and HM were determined in the initial and final materials. An EA1108 elemental analyser (Carlo Erba Instruments) equipped with an AS200 auto sampler was used for N, C and TOC. Phosphorus and metal concentrations were determined using inductively coupled plasma mass spectrometry (ICP-MS Element XR or Element2 from Thermo Electron) after acid digestion with

HNO₃. Previously solid samples were air-dried and the solids were finely shredded and mixed. Free air space (FAS) of composting mixtures was calculated as indicated by El Kader et al. (2007):

$$\text{FAS} = ((\text{total volume} - (\text{dry matter mass}/1.6) - (\text{water mass}/1.0))/\text{total volume}) \times 100$$

where mass is in kg and volume in L; it was assumed that the density of dry matter particles was 1.6 kg/L (because the actual value has not been measured) and density of water was 1.0 kg/L.

Germination assays of lettuce seeds (*Lactuca sativa* 'May Queen') were carried out as indicated by Zucconi et al. (1981) in order to determine the phytotoxicity of the final compost. Twenty seeds were placed evenly on plastic petri dishes lined with filter paper. 4.0 mL aqueous extract of compost were pipetted into each petri dish whilst distilled water was used as control. Compost extracts in distilled water (1:10 w/w) were prepared by shaking the samples mechanically for 2 h, followed by centrifugation and filtration through 0.45 µm membrane filters. Seeds were incubated in dark conditions at room temperature (20 °C) for 48 h. Germination assays were carried out in triplicate. At the end of the assays, the number of seeds germinated was counted and the root length measured. For each treatment, the Germination Index (GI) was obtained as follows:

$$\text{GI}(\%) = \frac{[(\text{average number of seeds germinated} \times \text{average root length})_{\text{treatment}}]}{[(\text{average number of seeds germinated} \times \text{average root length})_{\text{control}}]} \times 100$$

GI is a sensitive indicator of phytotoxicity (Tiquia et al., 1996). Values higher than 80% are considered an indicator of absence of phytotoxicity, whilst values below 50% indicate phytotoxicity for most crops (Zucconi et al., 1981).

Besides mean and standard deviation, statistical analysis included one-way analysis of variance (ANOVA) using IBM SPSS statistics 21.

2.4. Biodegradability and stability assays

The respirometric method and the self-heating method were used to determine the biodegradability potential of raw materials and the stability of final products. Respirometric indices have been used to classify organic materials depending on their biodegradability. Barrena et al. (2011b) proposed a three level scale of biodegradability: high (>5 mg O₂/g SV h), moderate (2–5 mg O₂/g SV h) and low (<2 mg O₂/g SV h). On the other hand, a threshold ranging from 0.3–1 mg O₂/g SV h is usually accepted to classify compost and organic wastes as stable (Barrena et al., 2006).

Respirometric assays were carried out following the method described by Lasaridi and Stentiford (1998). Approximately 8 g of the fine fraction of compost was dispersed in 500 mL of water amended with buffer, nutrient compounds and allylthiourea. This compost dispersion was subjected to intermittent aeration (45 min ON and 15 min OFF) and the oxygen consumption was determined (oxygen electrode YSI Pro ODO) during the non-aeration periods. These assays were carried out for at least 20 h. The parameters determined with this method were the maximum specific oxygen uptake rate (SOUR_{max}) and the cumulative specific oxygen consumption over 20 h (CSO_{20 h}). This procedure was adapted for raw solid materials as follows: after carrying out a standard respirometric assay with stable compost (i.e., showing a SOUR_{max} below 1 mg O₂/g SV h, as indicated by Lasaridi and Stentiford, 1998), an amount of 8–16 g of raw material (shredded to <10 mm) was added to the respirometric solution, the pH was regulated to 7.0 and the test was run for a subsequent period of 20 h. A blank was run simultaneously but without raw material addition.

Self-heating potential of compost samples were determined in Dewar flasks of 1.5 L of volume following the procedure indicated by Brinton et al. (1995). MC of samples was previously corrected to 60–65%. This procedure was adapted for raw solid materials as follows:

shredded (<10 mm) samples were mixed with stable compost at several ratios and the self-heating potential of the mixture was determined. Simultaneously, a control test was carried out with only stable compost.

3. Results

3.1. Physico-chemical characteristics of raw materials

Chemical composition and other characteristics of the raw materials used are given in Table 1. Previous studies in regard to MDFS chemical composition were not found (EC, 2014). The high Ca content could be derived from lime addition during physico-chemical treatment of generated wastewater. PM was very diluted, having only a 0.33% of dry matter and 0.16% organic matter on a wet basis. PM content in NH_4^+ -N, total suspended solids and volatile suspended solids was 514, 100 and 80 mg/L, respectively. It also has a low content in nutrient elements and heavy metals, in comparison with previous published characteristics of pig manure in the same region (Vázquez et al., 2015a). FGW was characterized by its very low BD, of only 0.06 kg/L, and relatively low MC (41%). On the other hand, MDFS and OFW had high BD and high MC. On a dry basis, MDFS was richer in Ca than FGW whilst OFW was richer in P, Ca and P than MDF and FGW but not in Mg. In regard to K, an element of interest to improve fruit production in chestnut plantations, MDFS had the lowest content and PM the highest. On dry weight basis, the content in N was higher in PM followed by OFW and lower in FGW and MDFS. C/N ratio for FGW and MDFS was approximately equal to 27, being similar to recommended values for composting (Onursal and Ekinici, 2017). HM content was low in all solid wastes.

3.2. Biodegradability of raw materials

The self-heating potential of raw materials was determined in Dewar flasks using stable compost as an inoculum and matrix base. The results are shown in Fig. 1. The maximum increase of temperature of blank assays was 1.9 °C above ambient temperature. OFW showed the highest self-heating potential reaching thermophilic temperatures for a long period of time. This was followed by MDFS and next by the mixture FGW + PM, whilst the FGW alone showed the lowest self-heating potential. Maximum temperature gradients were 23.5, 16.1,

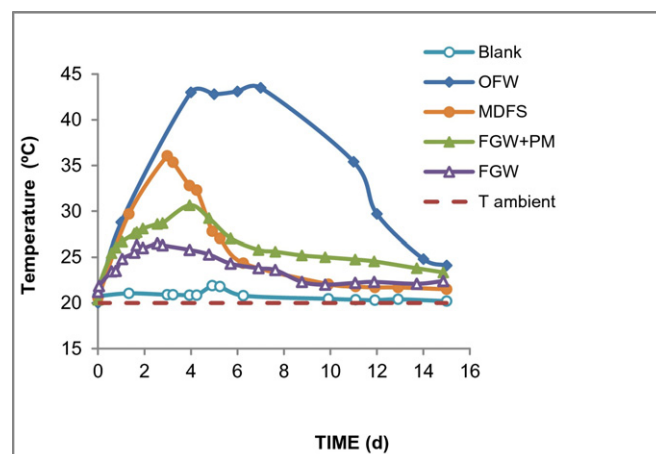


Fig. 1. The self-heating potential of raw materials in Dewar flasks (stable compost was used as inoculum and matrix base in all assays in proportions 2:1 stable compost-waste in volume).

10.7 and 6.5 °C, for OFW, MDFS, FGW + PM and FGW, respectively (Fig. 1). Thus, FGW showed a very low self-heating potential, which was increased by the addition of PM. On the other hand, MDFS showed a medium self-heating potential indicating that this waste needs treatment for further stabilization. In addition, the results suggest that the use of MDFS as a co-substrate could favour a better development of FGW composting process. Compared with the compost stability scale of Brinton et al. (1995), FGW would be class V (stable compost), whilst FGW + PM and MDF would be class IV (moderately stable) and OFW class III (active). However, we must take into account that this scale has been established for compost samples, which is not the case of the raw material analysed here. This is because we are using the self-heating method to explore the biodegradability of raw materials, but a standard procedure and biodegradability scales are not known for this application. Therefore, we must consider the results shown in Fig. 1 only as relative values for the organic residues studied and under the conditions of the tests performed.

Results of respirometric assays carried out for these materials were in the same line as those of self-heating assays. Whilst OFW and PM gave SOURmax of 5.4 and 7.1 mg O_2 /g SV h, respectively, being classified as a material of high biodegradability, MDFS and FGW showed SOURmax of 2.17 and 0.81 mg O_2 /g SV h, respectively, being of low biodegradability with reference to the scale of Barrena et al., 2011b. In addition, the respirometric index for FGW was just lower than the threshold for compost stability (Lasaridi and Stentiford, 1998; Barrena et al., 2006), anticipating a very low composting rate.

3.3. Evolution of composting processes

The amounts of raw materials used in each composting system are indicated in Table 2 which also reports the main initial characteristics of the four composting mixtures. MC ranged from 72 to 77%, being higher than the 60% recommended for the composting of pig manure solid fraction and other wastes by several authors (Huang et al., 2006; Ogunwande et al., 2008; Nolan et al., 2011). However, Vázquez et al. (2015a) reported adequate composting at MC ranging from 70 to 75%, compatible with large thermophilic periods of 3 to 4 month. Except in M1 systems, leachates were not produced in spite of the addition of PM in different amounts (Table 1) and the high MC of the mixtures. M1 leachates were about 3 L (50% of PM added) which were recirculated back to the matrix and then absorbed. The high MC of mixtures M2, M3 and M4 without leachate generation indicated a high capability of both components MDFS and FGW to absorb water.

Time profiles of temperature and oxygen in composting material are shown in Fig. 2. M1 received OFW and FGW at a ratio 1:2.4 and

Table 1
Chemical composition and other characteristics of raw materials.

| | PM | FGW | MDFS | OFW |
|------------|---------------|----------------------------|----------------------------|----------------------------|
| pH | 7.3 ± 0.1 | 6.2 ± 0.1 | 7.3 ± 0.1 | 6.3 ± 0.1 |
| BD (kg/L) | 0.990 ± 0.001 | 0.063 ± 0.019 ^a | 0.651 ± 0.040 ^b | 0.594 ± 0.017 ^c |
| MC (%) | 99.37 ± 0.01 | 40.8 ± 0.04 | 74.8 ± 0.8 | 76.7 ± 1.9 |
| VS (%) | 49.2 ± 0.0 | 85.3 ± 2.1 | 66.6 ± 0.1 | 75.2 ± 3.5 |
| C (%) | 14.45 ± 0.70 | 45.23 ± 0.11 | 36.63 ± 0.95 | 45.14 ± 0.07 |
| N (%) | 10.60 ± 0.42 | 1.56 ± 0.02 | 1.24 ± 0.02 | 3.38 ± 0.03 |
| TOC (%) | 11.79 ± 0.65 | 41.83 ± 0.14 | 34.59 ± 0.71 | 34.50 ± 0.03 |
| C/N | 1.11 ± 0.02 | 26.90 ± 0.27 | 27.88 ± 0.96 | 10.20 ± 0.10 |
| Mg (g/kg) | 9.9 ± 0.1 | 2.17 ± 0.03 | 1.81 ± 0.03 | 1.60 ± 0.06 |
| P (g/kg) | 17.1 ± 0.1 | 1.21 ± 0.07 | 2.07 ± 0.03 | 4.89 ± 0.26 |
| Ca (g/kg) | 18.1 ± 0.4 | 7.78 ± 0.23 | 21.1 ± 0.02 | 39.8 ± 0.6 |
| K (g/kg) | 67.3 ± 0.9 | 5.80 ± 0.1 | 1.94 ± 0.12 | 15.2 ± 0.8 |
| Cd (mg/kg) | 0.157 ± 0.001 | 0.060 ± 0.003 | 0.153 ± 0.004 | 0.064 ± 0.002 |
| Pb (mg/kg) | 1.41 ± 0.05 | 1.53 ± 0.01 | 3.94 ± 0.11 | 0.23 ± 0.01 |
| Hg (mg/kg) | 0.061 ± 0.002 | 0.051 ± 0.003 | <0.050 | <0.050 |
| Cr (mg/kg) | 11.4 ± 0.4 | 6.48 ± 0.11 | 25.9 ± 1.0 | 1.42 ± 0.0 |
| Co (mg/kg) | 0.80 ± 0.03 | 1.03 ± 0.04 | 3.25 ± 0.12 | 0.14 ± 0.0 |
| Ni (mg/kg) | 8.8 ± 0.4 | 3.6 ± 0.1 | 18.2 ± 0.6 | 0.7 ± 0.0 |
| Cu (mg/kg) | 165 ± 5 | 11.1 ± 0.2 | 12.2 ± 0.3 | 6.1 ± 0.1 |
| Zn (mg/kg) | 1233 ± 5 | 29.6 ± 0.7 | 77.9 ± 0.8 | 16.0 ± 0.3 |
| As (mg/kg) | 1.79 ± 0.09 | 0.39 ± 0.01 | 2.02 ± 0.14 | 0.53 ± 0.02 |
| Se (mg/kg) | <1.0 | <1.0 | <1.0 | <1.0 |

Mean values followed by standard deviation (N = 2, except for BD of solids wastes: ^aN = 10, ^bN = 8, ^cN = 4). pH measured in 1:5 weight:volume water extracts, BD and MC on wet basis, composition on dry basis.

Table 2
Initial amounts of the materials used in each composting system and total matrix characteristics.

| Composting system ^a | Solid materials added (L) | | | | PM added (L) | Overall matrix | | | | | |
|--------------------------------|---------------------------|-----|------|-------|--------------|----------------|-------------------------|------------------------|---------------------|------------------|----------------------|
| | FGW | OFW | MDFS | Total | | Mass (kg) | Volume (L) ^b | BD (kg/L) ^b | MC (%) ^c | C/N ^c | FAS (%) ^d |
| M1 | 260 | 110 | 0 | 370 | 6 | 88 | 284 ± 0 | 0.31 ± 0.00 | 71.5 | 13.9 | 72.3 |
| M2 | 200 | 0 | 160 | 360 | 32 | 149 | 285 ± 4 | 0.52 ± 0.01 | 77.2 | 26.6 | 52.3 |
| M3 | 260 | 0 | 110 | 370 | 36 | 124 | 281 ± 5 | 0.44 ± 0.01 | 77.5 | 26.4 | 59.6 |
| M4 | 340 | 0 | 0 | 340 | 29 | 50 | 294 ± 0 | 0.17 ± 0.00 | 74.5 | 25.9 | 84.6 |

^a M1: control with OFW (at 1:2.4 OFW:FGW ratio); M2, M3 and M4: mixtures at MDFS:FGW volumetric ratios of 1:1.3 (M2), 1:2.4 (M3) and 0:1 (M4).

^b Measured mean values from the two system replicates.

^c Obtained from the amounts of each material added and its characteristics (Table 1).

^d Calculated as indicated by El Kader et al. (2007).

had a C/N ratio of 13.9, lower than recommended values for composting. In spite of this, M1 mixture reached thermophilic temperatures which indicated a successful development of the composting process. The high intensity of biodegradation caused medium oxygen content in the matrix, which progressively increased from 10% to 16% during the first two weeks of composting. After this period, the temperature suddenly decreased to about 15 °C, due to the exhaustion of the readily biodegradable substrate. Besides, ambient temperatures near 0 °C (about day 20) probably contributed to the rapid temperature decrease.

M2 and M3 mixtures received MDFS and FGW at ratios of 1:1.3 and 1:2.4, respectively, showing C/N ratios of 26.6 and 26.4, respectively (Table 1). Profile temperatures were very similar for both mixtures, although with a maximum slightly higher for M3 (49 °C) than for M2 (44 °C). Adequate oxygen concentrations were observed during the thermophilic phase, with values of 14% (M2) and 16% (M3). The thermophilic phase lasted only about 10 days, afterwards the temperature decreased to mesophilic values for another 10 days and then to temperatures about 15 °C, in a similar way to the M1 mixture. Temperature and oxygen profiles for M2 and M3 mixtures suggest a better behaviour of the less concentrated one in MDFS (M3) in comparison to that of higher MDFS concentration (M2). M2 mixture presented a lower oxygenation rate and reached lower maximum temperature.

M4 matrix contained only FGW and PM and reached only mesophilic temperatures, with maximum values of about 22 °C, and decreasing to about 10 °C after 20 days of composting. This occurred in spite of a C/

N ratio in the optimal range. In this way, the low temperature reached could be related to the low biodegradability of the substrates as well as to the low bulk density and the large particle size which favoured the existence of high porosity and heat loss (Gea et al., 2007; Bernal et al., 2009).

After 3 weeks of composting, during the period from 21 to 56 days (Fig. 1), temperatures were low in all mixtures, ranging from 10 to 15 °C (M1 > M2 > M3 > M4), whilst the gradient above ambient temperature ranged from 2 to 8 °C. Oxygen ranged from 20 to 21% in all mixtures, in spite of the high MC. This high oxygen concentration indicated both good aeration conditions provided by the FGW material and low degradation activity. The stability of the mixtures at day 48 was measured in controlled conditions (Dewar flasks assay in lab at 20 °C). These assays gave maximum temperature gradients of 36.0, 9.0, 6.6 and 6.9 °C for M1, M2, M3 and M4, respectively. Thus, the M1 material was not stable at that time, corresponding to Rottegrade Class IV (Brinton et al., 1995), whilst the other three materials can be considered completely stabilized (Class I). This was congruent with the temperature profile shown in Fig. 2.

Since the first days, M1 matrix showed a strong and progressive reduction in volume reaching 59% reduction at day 48 of composting (Table 3). In comparison, the other matrices showed lower volume reduction of 4%, 8% and 20% for M2, M3 and M4, respectively. BD remained practically constant during this period (Table 3). Thus, the reduction of total mass followed the trend of volume reduction. The MC was high, showing values of 74–75%, and remained practically constant. In spite of the high MC, FAS was high in all mixtures but decreasing in proportion to its content in MDFS. The high FAS value was due to the low BD which is a characteristic of the FGW present in all the mixtures. These high FAS values were maintained throughout the entire composting process (Tables 2, 3 and 5) and explained the fact that aeration was good in all matrices, avoiding anaerobic conditions in spite of the high MC. The variation of organic matter content (as percentage of dry matter) was not detectable during the initial 48 days composting period.

Between days 60 and 150 of composting, matrix temperature followed ambient temperature (ranging from 7 to 16 °C) with average gradients of 5.1, 3.3, 2.8 and 0.9 for mixtures M1, M2, M3 and M4, respectively (data not shown). These gradients indicate that biodegradation continued but at a very low rate. Temperature gradients

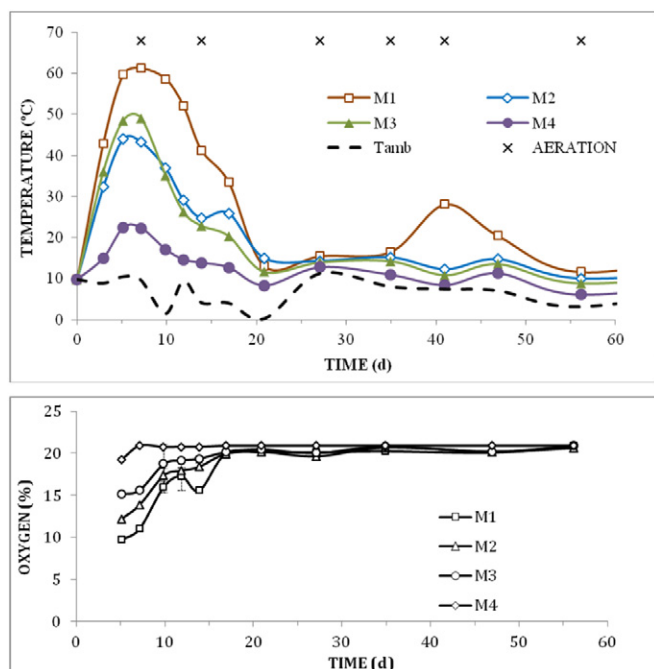


Fig. 2. Evolution of temperature and of interstitial oxygen in composting matrices.

Table 3
Evolution of matrix characteristics during the first 48 days of composting.

| Composting system | M1 | M2 | M3 | M4 |
|--|-------------|-------------|-------------|-------------|
| Mean matrix characteristics from 0 to 48 composting das (n = 8) ^a | | | | |
| Mean BD (kg/L) | 0.30 ± 0.04 | 0.38 ± 0.03 | 0.31 ± 0.02 | 0.13 ± 0.05 |
| Mean MC (%) | 75.0 ± 2.3 | 75.6 ± 2.5 | 74.2 ± 4.9 | 73.7 ± 2.2 |
| Mean VS (% TS) | 75.2 ± 1.7 | 71.6 ± 2.1 | 73.4 ± 4.3 | 86.0 ± 2.1 |
| FAS (%) | 67.7 | 64.2 | 72.7 | 82.6 |
| Volume and mass reduction at day 48 (n = 2) ^b | | | | |
| Volume (%) | 58.9 ± 0.4 | 3.9 ± 0.8 | 8.1 ± 0.8 | 20.3 ± 0.2 |
| Mass (%) | 49.7 ± 0.2 | 6.2 ± 0.2 | 7.6 ± 0.1 | 5.5 ± 0.3 |

^a Measurements each two weeks approximately (n = 4) for each replicate (n = 2) of composting system.

^b Mean from the two system replicates.

practically disappeared ($<1^{\circ}\text{C}$) after day 150 (ambient temperature ranging from 17 to 23°C), indicating the finalization of the biodegradation process. Oxygen remained high, above 20%, and matrix volume showed a progressive reduction at nearly constant rates with time of $0.09\%/d$ for M1 and 0.15 – $0.16\%/d$ for the other matrices ($R^2 > 0.95$). Other parameters were not monitored during this period, but MC was observed qualitatively. During this period, PM was added to M2 (5 L), M3 (8 L) and M4 (27 L) distributed as necessary in order to maintain MC adequate for the composting process. Total watering ratios reached 0.25 (M1), 1.08 (M2), 1.56 (M3) and 4.35 (M4) L PM/kg TS of added solid wastes. Except for M4, total watering ratios were lower than those reported by Vázquez et al. (2015a) for pig manure and biowastes co-composting. The high watering ratios of Vázquez et al. (2015a) were attributed to the large heat production from the highly biodegradable biowastes used. However, in the conditions of the present study, other factors such as initial MC of solid wastes and composting time appeared to have a higher effect.

At the end of the process (day 268), the reduction in volume, total mass and other parameters (Table 4) largely varied from one matrix to another. M1 matrix, containing the food waste presented the higher loss in all parameters, which was related to the high biodegradability of the substrate and the low C/N ratio. N losses reached in this matrix 71% of its initial content, which is in the upper range of reported values in literature. Typical percentage nitrogen losses during manure composting vary from 5 to 62% of the initial nitrogen content (Bernal et al., 2009) although higher values have also been reported (Ogunwande et al., 2008; Silva et al., 2009). M4 matrix containing only forestry waste and pig manure followed M1 in percentage losses, reaching 55–58% total solids and organic carbon losses, and 53% nitrogen losses. Losses decreased for all parameters in M3 and mainly in M2 in proportion to its content in MDFS. This means that MDFS was biodegraded in lower extent than the others wastes. Of great interest is the observed reduction of nitrogen losses as the amount of MDFS in the matrix increased. Both M2 and M3 mixtures achieve very good nitrogen conservation, taking into account the long duration of the process.

After several weeks of composting, it was observed that MDFS in composters M2 and M3 originated small aggregates of spherical or ovoid shape (up to 3 cm in diameter), which were more frequent and of bigger size in M2 than in M3. These particles were resistant to deformation and breakage and consisted of an external dark layer and a light brown core, the last similar to the raw MDFS colour. Respirometric assays of material obtained separately from both layers indicated that the external layer was completely stabilized (SOURmax 0.40 and $0.35\text{ mg O}_2/\text{g VS h}$ for M2 and M3, respectively) whilst the internal core showed a higher biodegradability (SOURmax 1.73 and $1.13\text{ mg O}_2/\text{g VS h}$ for M2 and M3, respectively). At the end of the process, these aggregates accounted for $<10\%$ (M2) and 5% (M3) of the MDFS added to the composters, thus affecting in a minor extension the stability of final products.

3.4. Characteristics of final products

Chemical characteristics of final composts are shown in Table 5. Final composts showed very different characteristics. M1 has a higher content in nutrients (N, P, K, Mg and Ca) than the other mixtures, a lower

Table 5
Characteristics of final products.

| Compost | M1 | M2 | M3 | M4 |
|--|---------------------|----------------------|-----------------------|----------------------|
| BD (kg/L) | 0.291 ± 0.008^a | 0.454 ± 0.015^b | 0.346 ± 0.078^b | 0.138 ± 0.004^c |
| MC (%) | 62.0 ± 0.3^a | 69.2 ± 0.4^b | 69.7 ± 0.7^b | 66.5 ± 0.5^c |
| pH | 6.9 ± 0.1^a | 6.8 ± 0.1^a | 6.6 ± 0.2^a | 5.7 ± 0.1^b |
| EC (mS/cm) | 5.48 ± 0.35^a | 1.00 ± 0.06^b | 1.09 ± 0.15^b | 3.34 ± 0.30^c |
| OM (%) | 55.9 ± 0.5^a | 60.1 ± 0.0^a | 59.9 ± 0.4^a | 80.5 ± 7.1^b |
| FAS (%) | 75.0 ± 0.9^a | 61.0 ± 2.0^a | 69.3 ± 9.3^a | 87.9 ± 0.5^a |
| SOURmax ($\text{mg O}_2/\text{g VS h}$) | 0.25 ± 0.08^a | 0.56 ± 0.15^a | 0.49 ± 0.17^a | 0.21 ± 0.12^a |
| GI (%) | 53.9 ± 3.1^a | 97.8 ± 2.6^b | 87.7 ± 1.4^c | 34.2 ± 1.0^d |
| NH ₄ ⁺ (mg N/L) | 114.7 ± 8.5^a | 13.6 ± 0.05^b | 16.3 ± 1.3^b | 72.0 ± 0.5^c |
| C (%) | 36.1 ± 0.54^a | 33.89 ± 0.11^b | 35.11 ± 0.70^{ab} | 41.21 ± 0.46^c |
| N (%) | 3.26 ± 0.11^a | 1.51 ± 0.04^b | 1.61 ± 0.05^{bc} | 2.04 ± 0.05^c |
| TOC (%) | 33.5 ± 0.25^a | 31.21 ± 0.03^b | 31.85 ± 0.56^{ab} | 38.67 ± 0.44^c |
| C/N | 10.3 ± 0.67^a | 20.62 ± 0.48^b | 19.77 ± 0.31^b | 19.0 ± 1.0^b |
| Mg (g/kg) | 4.3 ± 0.1^a | 2.5 ± 0.3^b | 2.6 ± 0.1^b | 3.0 ± 0.2^b |
| P (g/kg) | 6.25 ± 0.03^a | 2.12 ± 0.03^b | 1.99 ± 0.08^b | 1.74 ± 0.18^b |
| Ca (g/kg) | 60 ± 1^a | 21 ± 1^b | 19 ± 1^b | 9.0 ± 0.5^c |
| K (g/kg) | 28 ± 4^a | 4.5 ± 0.5^b | 5.2 ± 0.1^b | 9.5 ± 1.1^b |
| Na (g/kg) | 9.6 ± 0.1^a | 1.4 ± 0.0^b | 1.5 ± 0.1^b | 1.1 ± 0.2^b |
| Cd (mg/kg) | 0.15 ± 0.01^a | 0.152 ± 0.005^a | 0.147 ± 0.001^a | 0.071 ± 0.006^b |
| Pb (mg/kg) | 2.28 ± 0.24^a | 4.87 ± 0.79^{bc} | 5.73 ± 0.91^c | 2.48 ± 0.14^{ab} |
| Hg (mg/kg) | <0.05 | <0.05 | <0.05 | <0.05 |
| Cr (mg/kg) | 13.7 ± 0.4^a | 32.6 ± 0.3^b | 33.5 ± 1.4^b | 17.3 ± 0.1^c |
| Co (mg/kg) | 1.51 ± 0.05^a | 3.66 ± 0.01^b | 4.13 ± 0.22^c | 1.70 ± 0.02^a |
| Ni (mg/kg) | 5.7 ± 0.1^a | 19.5 ± 1.1^{ab} | 24.2 ± 8.2^a | 6.8 ± 0.0^a |
| Cu (mg/kg) | 18.0 ± 2.3^a | 16.3 ± 1.0^a | 17.1 ± 0.5^a | 19.8 ± 0.2^a |
| Zn (mg/kg) | 53 ± 1^a | 102 ± 5^b | 102 ± 13^b | 67 ± 1^a |
| As (mg/kg) | 1.10 ± 0.0^a | 1.93 ± 0.08^b | 4.22 ± 2.85^{abc} | 3.42 ± 0.25^c |
| Se (mg/kg) | <1.0 | <1.0 | <1.1 | <1.2 |

Mean value from duplicate composting bins followed by standard deviation. Values in each row followed by the same letter do not differ significantly ($p < 0.05$, one-way ANOVA with pairwise comparison). EC and pH measured in 1:5 weight:volume water extracts, BD and MC on wet basis, composition on dry basis.

C/N ratio, lower content in HM, but higher EC. In fact, EC in M1 compost exceeded the recommended values for agronomic use whilst it was suitable in M2 and M3 mixtures and M4 was close to this threshold of 3 mS/cm (Oviedo-Ocaña et al., 2015). Banegas et al. (2007) reported that composts with EC of 8 mS/cm caused negative effects on soil microorganisms activity. All compost showed low respirometric indices but C/N ratio remained high in M2, M3 and M4 due to the low carbon reduction as well as to the losses of nitrogen, as discussed in Section 3.3. However, a clear reduction of C/N ratio occurred in all systems from the starting materials (Table 2) to the final materials (Table 5). In fact, the C/N ratio for M2, M3 and M4 final products was lower than that reported by other authors for gorse based composts (Iglesias-Díaz et al., 2009; Tighe-Neira et al., 2014).

Bernal et al. (2009) indicated C/N ratios below 20, preferably below 10, as established maturity indices for composts from different sources. However, it is clear that the C/N ratio cannot be considered as an absolute maturity index, since the M1 mixture already had a C/N ratio below 20 before the composting process. Furthermore, a relation between maturity (measured as GI) and C/N ratio did not exist. Therefore, the reduction of the C/N ratio for mixtures M2, M3 and M4 from about 26 to <20 should only be considered as a suggestion of compost stabilization but not confirmation. Confirmation of final compost stability was obtained from respirometric assays for M1, M2 and M3 products which showed SOURmax below the stable compost threshold of $1\text{ mg O}_2/\text{g VS h}$ (Lasaridi and Stentiford, 1998).

The content in macronutrients in M2, M3 and M4 final products was high in general but varied depending on the considered element. The content in N, P and K was above 1.5% N, 0.17% P and 0.45% K for M2, M3 and M4 composts. Macronutrients content was in the range of that in composts from other materials such as seaweed and fish waste (Illera-Vives et al., 2015), natural casings waste with sheep manure and dead leaves (Makan, 2015), but lower than that reported by Morales et al. (2016) for agri-food sludge compost and by Vázquez et

Table 4
Percentage reduction for several parameters at the end of the composting process.

| Reduction (%) ^a | M1 | M2 | M3 | M4 |
|----------------------------|----------------|----------------|----------------|----------------|
| Volume | 80.7 ± 1.5 | 37.0 ± 0.1 | 43.0 ± 1.3 | 57.0 ± 0.8 |
| Mass | 81.9 ± 0.5 | 45.2 ± 0.2 | 55.3 ± 0.9 | 65.2 ± 0.1 |
| TS | 75.7 ± 3.1 | 25.9 ± 1.6 | 40.0 ± 2.3 | 54.6 ± 2.7 |
| TOC | 78.2 ± 1.5 | 35.1 ± 0.2 | 48.3 ± 1.8 | 57.5 ± 1.0 |
| N | 70.6 ± 3.5 | 21.7 ± 2.0 | 37.3 ± 2.3 | 52.7 ± 3.6 |

^a Mean from the two system replicates.

al. (2015a) for solid fraction of manure and other biowastes. In Galician forests, the productivity of both the understory and the trees can be limited by low soil fertility as a result of the acidic soil conditions. In order to improve productivity, several studies recommended the use of organic fertilizers such as composted sludge (Mosquera-Losada et al., 2016). Although pH in final products is slightly lower than 7, it is clearly above the pH of most Galician soils and thus a beneficial effect can be expected. On the other hand, the high content in K in final products makes them very interesting to increase fruit productivity in chestnut groves.

HM was low in all mixtures, appearing below the limits of Class A compost for agricultural use in Spain and thus compatible with European Commission organic agriculture (Vázquez et al., 2015b) and Class A⁺ of the 2001 Austrian Compost Ordinance (Fernández-Delgado et al., 2015). On the other hand, HM content was far below that reported by Guerra-Rodríguez et al. (2006) for compost from forest wastes and liquid poultry manure and by Mosquera-Losada et al. (2016) for anaerobically digested, composted and pelletized sewage sludge used for forest fertilization. Ni is the metal that reached a high level in comparison with Spanish Class A. This occurred in M2 and M3 composts because of the high content of MDFS in Ni. In general, the content in HM appeared to be in proportion to the content in MDFS of the mixtures although PM significantly contributed to the content of Cu and Zn in M4 compost.

GI values above 80% are considered an indicator of phytotoxicity disappearance (Zucconi et al., 1981) and maturity (Tiquia et al., 1996). M2 and M3 extracts gave 100% seed germination and root elongation of 98% and 88% of control, respectively. Thus, GI of M2 and M3 were 98% and 88% (Table 5), respectively, indicating the absence of phytotoxic compounds in these composts. On the other hand, GI for M4 compost was below 50%, indicating that this compost would be phytotoxic for the majority of cultures (Zucconi et al., 1981). M4 extract showed a toxic effect on both germination (60.5%) and root elongation (56.7%). M1 compost showed GI of 53.9, thus exerting moderate phytotoxicity, mainly due to reduced root elongation (57.7%) than seeds germination (93.4%).

Many factors simultaneously influence germination, thus being difficult to determine which factor causes the greatest influence (Tiquia, 2010). Huang et al. (2004) reported GI lower than 50% for compost from pig manure and sawdust and suggest that it may be due to a high EC of about 2.5 mS/cm. However, Tiquia (2010) obtained a higher threshold for EC toxicity in hog manure composts with values under 3.4 mS/cm being in the non-phytotoxic or marginally phytotoxic range. Besides, Tiquia (2010) also revealed that ammonium nitrogen was the most important factor causing phytotoxicity of hog manure compost when present at concentrations above 2 g NH₄⁺-N/kg. M1 and M4 composts showed higher values of EC and ammonia nitrogen than M2 and M3, but ammonium concentration (0.57 and 0.36 g NH₄⁺-N/kg for M1 and M4 composts, respectively) was below the threshold of toxicity indicated by Tiquia (2010), whilst EC was close to the toxicity threshold (M4) or clearly higher (M1). Thus, EC could be the factor causing phytotoxicity in M1 compost whilst other factors must contribute to phytotoxicity in M4 compost. Low pH in M4 compost (5.7, Table 5) was the main difference among the characteristics of this compost and the other three composts. The optimum pH for lettuce germination is near 7 (Gascó et al., 2016), so the moderately acidic pH might contribute to reduction of the GI of M4 compost. Total heavy metals content in the final products seems not to have a direct relation with GI since M2 and M3 composts present higher values than M1 and M4 composts.

4. Discussion

Forestry cleaning green waste is a very abundant material but it can show very different characteristics. Gorse dominated wastes present high nitrogen content and can undergo thermophilic composting if it is composted in fresh after gridding (Iglesias-Díaz et al., 2009).

However, achieving a thermophilic phase is not always guaranteed, particularly when the proportion of tender shoots and young plant parts is scarce, or when the material passed a long period after harvesting. Thus, it is more usual to use forestry green waste as a bulking material in co-composting of animal manures (Tighe-Neira et al., 2014; López-López et al., 2016). Guerra-Rodríguez et al. (2001) stated that the composting of chestnut burr and leaf litter did not occur because the C/N ratio was too high and the compost never reached the thermophilic phase. However, co-composting of chestnut green waste and poultry manure reached 60 °C in 14 days (Guerra-Rodríguez et al., 2001). Venterino et al. (2013) reported a maximum temperature of 29 °C during composting of chestnut cleaning green waste. Composting times can vary largely from about 3 months to 18 months (Iglesias-Díaz et al., 2009; Venterino et al., 2016). In our composting systems, the addition of MDFS to FGW clearly increased the maximum temperature and the duration of the thermophilic phase in relation to the composting of FGW. However, even with MDFS amendment, the thermophilic and mesophilic phases were short (about one month or less) and the biodegradation process during these periods was limited to the readily biodegradable fractions. Watering with PM was not sufficient to reach the thermophilic phase during the composting of FGW. However, this result must be considered to be restricted to the use of a much diluted PM, whilst the use of a high loaded PM could increase the composting temperature.

A factor determining the temperature level of FGW composting matrices could be the high FAS derived from the low BD of forestry green wastes. Grinding is a usual pre-treatment to increase BD and reduce FAS (Iglesias-Díaz et al., 2009), but it was usually insufficient because the reduction of the particle size is limited due to practical and economic reasons. Adding MDFS to FGW significantly decreased the FAS of composting matrices, but the obtained values for M2 and M3 mixtures continue to be too high. Increasing MC and physical compacting are other factors that can help in reducing FAS (El Kader et al., 2007). MC was high although below leaching conditions in our experiments and higher MC would not be positive for the composting process. Compacting was not favoured in these experiments, but it could play a positive role in improving composting conditions. A high compaction of composting matrices would increase BD and reduce FAS, thus favouring heat retention. Furthermore, El Kader et al. (2007) found that lower FAS significantly reduced ammonia volatilization and nitrogen losses.

As indicated in Section 3.3, the high FAS values throughout the entire composting process helped in avoiding anaerobic conditions. Methane emissions during aerobic composting of organic wastes mostly occur during the thermophilic phase, when oxygen consumption is higher and anaerobic microniches are more probable to be present (Fernández-Delgado et al., 2015). These authors reported that oxygen concentration in pore space should not fall below 10% in order to avoid the presence of anaerobic zones and methane formation. In our systems, oxygen content was always above 10% (Fig. 2) and thus methane formation can be considered not probable.

The FGW waste used in this study is characterized by its heterogeneity in terms of its composition, predominating the species of bush over the fraction of chestnut tree. Another aspect to consider is that it included all fractions generated in short rotation chestnut plantations. Venterino et al. (2013) also studied the composting of the total waste generated in the forest grove. However, the contents of N and TOC varied substantially as Venterino et al. (2013) reported 0.5% N and 28.9% TOC (C/N ratio of 57.8) whilst our FGW presented 1.54% N and 41.7% TOC (C/N ratio of 27.1). The last C/N ratio was more favourable for composting than the former, and it was still reduced slightly in the composting matrices of our study because of the addition of MDFS and PM. C/N ratio is a parameter that expresses the nutritional balance of the composting mixture, because microorganisms require an energy source (degradable organic-C) and N for their development and activity, being the adequate C/N ratio for composting in the range 25–35 (Bernal et al., 2009). A low C/N ratio such as that of the M1 mixture (C/N of 13.9)

required more frequent or intensive turning operations. This coincides with the findings of Onursal and Ekinci (2017) who reported that an initial C/N ratio of 29 (in the range of 12.8 to 37.4) minimized the energy requirement to supply oxygen in composting systems with forced aeration. High MC increases the specific heat of the mixture and high FAS increases the air flux, thus both factors can reduce the temperature level and retard the composting process. However, El Kader et al. (2007) reported that reducing FAS lead to lower temperature and lower mass losses. Whilst we are interested in high MC because this implies the treatment of larger amounts of PM, FAS can be reduced by mechanical compacting of the composting matrix. The potential effects of this modification on the temperature level and material conversion are of difficult prediction and will require additional research.

Although variations in nutrient content exist depending on the plant fraction (higher in leaves than in stems) and considered season (generally larger in spring), these were not considered the relevant factors to explain the differences in nutrient content between the FGW of our study and that of Ventorino et al. (2013). The differences could probably be due to a higher proportion of biomass of the legume family (*Fabaceae*) in our study, represented by species of the genera *Cytisus* and *Ulex* (gorse), as well as herbaceous species. Encouraging the presence of these species in the chestnut understory would allow us to obtain forestry wastes with higher nitrogen content and composts with higher fertilizer value.

5. Conclusions

FGW is a very abundant material which can show very different characteristics and composting patterns. FGW in the present study showed a low biodegradation rate which had been increased by amending with MDFS and PM. Mixtures of MDFS:FGW at volume ratios of 1:1.3 and 1:2.4 reached short thermophilic and mesophilic phases of about 3 weeks during which the readily biodegradable fraction of organic matter was effectively removed but the overall material conversion was reduced. After 48 days of composting, reductions were in the ranges of 4–20% (volume) and 5–8% (mass). However, the conversion process continued with reduced gradient temperatures up to 150 days of composting. At the end of the composting process, reductions reached 37–57% (volume), 45–65% (mass), 35–56% (TOC) and 22–53% (N), depending mainly on the content in MDFS. Addition of MDFS to composting matrices largely reduced nitrogen losses as well as phytotoxicity. Nutrients in the final products were above 1.5% N, 0.17% P and 0.45% K and thus similar to nutrient content in compost from other sources. EC was below recommended limit values for agronomic use and HM content was low in all mixtures, appearing below the limits for organic agriculture.

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