Composting Process of Olive Tree Prunings: Assessment of Nitrogen and Carbon Losses

by

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ABSTRACT

The difficulties associated with Waste Management (WM) can be diminished or even erased, if an added value can be attributed to them. Composting is one of the natural bioprocesses capable of treating organic wastes through microbial activity. Composting can destroy pathogens, converts nitrogen from unstable ammonia to stable inorganic forms, reduce the volume of waste, and satisfy the needs for fertilizer in agriculture. Thus, composting has been concerned with the efficient and economic production of a product, while functioning within a larger process-oriented approach to solid WM. The basic objective of this study is to quantify nitrogen and carbon losses during olive tree pruning composting in order to understand composting process, and to reduce potential adverse environmental impact. Nitrogen and carbon losses were determined by the difference between the amounts at the beginning and at the end of the composting. It was observed that all materials were characterized by a reduction of C from 45 -65%. The reduction of N was much higher than C loss; 62-76%. The reduction of C was due to of its transformation in CO_2 and CH_4 and their release to atmosphere. Nitrogen can be lost from the mass of materials in runoff and by nitrate leaching composting.

Keywords: agricultural residues; carbon loss; composting; greenhouse emissions; nitrogen loss; olive tree prunings; waste management.

1. Introduction

Agricultural field by-products are divided into herbaceous and woody by-products. The former, are considered to be those crop residues, which remain in the field after the crop is harvested. Their nature is diverse and depends amongst others, on the crop and the method of harvesting. Woody by-products are those produced as a consequence of pruning and regenerating orchards, vineyards and olives. Normally, herbaceous crops are cultivated on arable land, whereas woody plantations are considered permanent crops. Large differences in terms of cultivated area and types of crops and yields are observed between EU Member States (MS), due to the climate conditions, specific soil condition and farming practices. Large amounts of residues are generated from agricultural crop production and partially remain in the field after harvest. Residue production depends on a number of factors that include the types of crops, crop rotation, crop mix and agricultural practices. The availability of residues depends on the amount that can be removed from land to maintain land fertility and on their competitive use for agricultural or industrial purposes (Esteban and Carrasco, 2011).

The residual biomass is commonly estimated by the use of Residue to Product Ratios (RPR)[1]. The RPR is a key issue in every evaluation and should be used carefully, as it is typically applicable only at a regional or local level (Nikolau et al., 2003). Frequently, in making agricultural by product estimates, a constant straw to grain ratio is assumed. This assumption may not always be accurate because straw-grain ratios can vary greatly across environments and genotypes. Higher straw to product ratios have been reported for Central and Northern European countries compared to those in the South of the continent and more generally, higher ratios in wet climates than in dry ones (di Blasi, 1997). Esteban and Carrasco (2011) estimated potential resources in 12 European countries and concluded that

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[1] RPR: Residue to Product Ratios, OMW: Olive Mill Wastewater, OTP: Olive Tree Prunings, OTL: Olive Tree Leaves, OTB: Olive Tree Branches, LHV: Lower Heating Value, LCI: Life Cycle Inventory, PD: Pigeon Droppings, EC: Electrical Conductivity, OM: Organic Matter, LOI: Loss on Ignition

most of the agricultural residual biomass in Europe is to be found in Southern European countries, with 52.69% for potential and 55.18% for available resources (Table 1). However, calculating the available biomass resource is a difficult task, which has resulted in very different estimations between studies. Table 2 presents estimations from four such attempts, carried out by the European Environment Agency (EEA), the CHRISGAS and RENEW Projects and Nikolau et al. (2003) for 11 EU countries (Esteban and Carrasco, 2011). In general, the EEA and CHRISGAS provide very high estimates, while RENEW and Nikolau et al. (2003) substantially lower. However, it is not possible to know the reasons for some of the differences between the studies. This is due to lack of either enough information on the methodology followed in the compared works or the production values used for the crops involved.

A number of authors have pointed out that the overexploitation of agricultural residues is an important cause of soil degradation (e.g., Blanco-Canqui et al., 2006; Lal, 2008). Fertilizer inputs can compensate for nutrient removals connected to harvest and residue extraction, but maintenance or improvement of soil fertility, structural stability and water-holding capacity requires recirculation of organic matter to the soil (Blanco-Canqui and Lal, 2009; Wilhelm et al., 2007). Residue recirculation leading to nutrient replenishment and carbon (C) storage in soils and dead biomass not only contributes positively to climate change mitigation by withdrawing C from the atmosphere but also by reducing soil degradation and improving soil productivity.

Amongst Mediterranean countries, olive trees are a major source of agricultural residues. The largest olive oil producing countries are Spain, Italy and Greece, with 46, 17 and 11 % of the world’s total respectively (IOOC, 2012). Despite the obvious economic importance of this food product, the olive oil industry causes diverse environmental impacts in terms of resource depletion, land degradation, air emissions and waste generation. Waste is mainly in the form of Olive Mill Wastewater (OMW) and Olive Tree Prunings (OTP) however, the amounts generated vary greatly from one country to another (Salomone and Loppolo, 2012).

The amounts of OTP's produced annually in Greece are considerable, with estimates ranging from 1.4 to 3 million tons (Konstantakou et al., 2010). As most farms are small enterprises, they lack the capability and/or knowhow of treating these wastes in an environmentally friendly manner. Current practice means that OTP's are usually burned immediately after the harvest of the olive fruits and the tree pruning operations (from November to March), resulting in the loss of large amounts of energy and material recovery, and the simultaneous emission of considerable amounts of Green House Gases (GHGs). Arguably, the absence of an economically feasible waste management plan constitutes one of the most serious disadvantages in the efforts to achieve more sustainable practices in the agricultural industry.

Olive Tree Leaves (OTLs) accumulate as a waste at olive oil mills. Part of the fresh OTLs is used as feed for sheep and goats, but their abundance and availability on a cost-free basis make them a favorable organic material for compost production. Composting of OTLs, both in mixtures and alone, was studied using windrows (Garcia-Gomez and Bernal, 2003; Manios et al., 1989). OTLs were also used as bulking agent for composting pig manure and sewage sludge (Manios et al., 1997). The phytotoxicity of OTLs composts in relation to its maturity was assessed by measuring the germination index of lettuce seed (Manios et al., 1989). Olive Tree Branches (OTBs) are difficult and expensive to collect. However, collection could be encouraged by the development of alternative composting methods and the implementation of local legislation to prevent burning. OTBs have been used as a bulking agent after shredding for composting with other organic residues, such as cucumber plant biomass, in windrows in a 1:1 (v/v) ratio (Maniadakis and Manios, 2001). Skoulou et al.

(2008) exploited gasification of olive kernels and OTBs. It was found that gas from OTBs at 950 °C and with an air equivalence ratio of 0.42, had a higher Lower Heating Value (LHV of 9.41 MJ/Nm^3) in comparison to olive kernels (8.60 MJ/Nm^3). Olive kernels produced more char with a higher content of fixed C (16.39 w/w%) than OTBs; thus, they might be considered an attractive source for carbonaceous material production (Table 3).

In the present study, OTPs were co-composted with two different types of organic materials (grass and seaweeds). It should be noted that composting of OTPs has not been adequately studied, despite the fact that they constitute one of the most abundant agricultural residue to be found in the Mediterranean countries. The composting process was developed during two different periods (winter and summer). The basic objective of this study is to quantify nitrogen and carbon losses during olive tree lop composting in order to understand composting process, and to reduce potential adverse environmental impact.

2. Composting process

Composting is one of the few natural processes that is capable of stabilizing organic wastes (Fig. 1). This process destroys most parasites, pathogens, and viruses contained in the waste, considerably reduces odor emissions by reducing levels of biodegradable hydrocarbons, and dries up the waste making it unattractive to insects (Barrington et al., 2002). During composting, carbonaceous and nitrogenous compounds are transformed through the activities of successive microbial populations into more stable, complex organic forms, which chemically and biologically resemble humic substances (Pare et al., 1998). Nitrogen (N) losses occurring during composting are a key issue, especially in countries where ammonia (NH_3) volatilization is an important source of acid rain (Bustamante et al., 2008). Furthermore, N volatilization reduces the fertilizer value of the waste, and constitutes an important economic loss. Carbon availability, bulking agent, particle size, moisture content and aeration regime are factors affecting compost N volatilization as opposed to N immobilization into organic components (Barrington et al., 2002).

The most important factors affecting composting include temperature, moisture content, carbon to nitrogen ratio (C/N), degree of aeration, Ph level, and the physical structure of the waste material. The key factors of the bioprocess are the compost's maturity and stability. Several authors have concluded that using a single parameter as a maturity index is insufficient and that amalgamation of several parameters is usually needed (eg., Amlinger et al., 2008; Aparna et al, 2008; Colon et al., 2010). Various physical, biological and chemical parameters have been used to monitor the quality and maturity of the compost, including C/N ratio, ammonium to ammonia (NH_4/NH_3), carbon dioxide (CO_2) evolution, pH, electrical conductivity, cation exchange capacity, water-soluble C, Dewar flask self-heating capacity, oxygen uptake rate and the production of humic substances in the finished product, as well as, a germination index to measure the phytotoxicity as a reliable indirect quantification of compost maturity (Aparna et al., 2008). Monitoring of the composting conditions during the bioprocess and the changes in the physicochemical characteristics has shown that satisfactory composting occurs after approximately 12 weeks (Rashad et al., 2010). Arguably, composting has been concerned with the efficient and economic production of a product, while functioning within a larger process-oriented approach to solid waste management (Murphy and Power, 2006).

2.1 GHG emissions

The GHG emissions that may be produced by composting include: (i) methane (CH_4) generated by anaerobic decomposition, (ii) carbon storage caused by long-term C compounds, (iii) nitrous oxide (N_2O) produced by materials' initial N content, and (iv) the non-biogenic

CO_2 emissions caused by the shipping of collected organic wastes to composting facilities and mechanical turning of the compost piles.

The biogenic CO_2 emissions caused by the composting process and the use of fertilizer on soil can be discounted in accordance with the GHG inventory guidelines developed by the IPCC (2006). Additionally, composting is an aerobic biological treatment method and if perfectly carried out, CH_4 is generally not generated. The CH_4 produced at the center of the compost pile, most likely oxidizes when it reaches the oxygen-rich surface of the pile, where it is converted to CO_2 (EPA, 2002; IPCC, 2006) or may be emitted due to unintentional leakages during process disturbances (IPCC, 2006). To assess the GHG emissions created by compost, the IPCC emission factors for CH_4 (4g CH_4/$kg_{\text{waste treated}}$) and N_2O (0.3g N_2O/$kg_{\text{waste treated}}$) were used (Chen, 2008).

Moreover, Amlinger et al. (2008) reported emission factors (EFs) of 14–41 kg CO_2-equivalents (eq.) Mg^{-1} wet waste (ww) for windrow composting of biowaste and 9– 68 kg CO_2-eq. Mg^{-1} ww for windrow composting of garden waste, while Andersen et al. (2010a) reported an EF of 111 ± 30 kg CO_2-eq. Mg^{-1} ww for windrow composting of garden waste. The EFs for home composting as reported by Andersen et al. (2010b) was 100–239 kg CO_2-eq. Mg^{-1} ww depending on the mixing frequency. Martínez-Blanco et al. (2010) reported an EF of 207 kg CO_2-eq. Mg^{-1} ww for home composting.

2.2 Application of compost to land

A review of the literature was unable to identify any primary data related to long-term soil C storage resulting from compost application. Metherell et al. (1993) simulating the long-term dynamics of various plant–soil ecosystems, evaluated the amount of long-term soil C storage when composted organics are applied to soils. The analysis concluded that when compost with 30% moisture content is applied, 56 kg per tone is sequestered over a 10-year time scale. The study also deducted the quantified enhanced C sequestration benefit that resulted from compost application to land from the anthropogenic emission burden, resulting in anthropogenic C emissions in the form of CO_2 that were swamped by the calculated benefits from C sequestration.

Blengini (2008), evaluated the impacts and resource conservation potential of composting household biowaste in the Asti District of Italy and calculated that total GHG emissions from composting, predominantly CO_2, was 156 g/kg of input material. Linzner and Mostbauer (2005), assumed that C sequestration of finished compost is 48 g/kg of input waste. Fabrizio et al. (2009) calculated that, for the application of compost with a C content of approximately 200 kg/tone, the equivalent of 369 kg and 520 kg of C per ton of soil subject to maize growing and cropping was retained in the soil for 150 days after the application of compost at the rate of 50 and 85 tones/ha, respectively. The calculated amount of retained C in this latter study is considerably larger than that in the other studies cited, and may be due to the shorter period of time over which sequestration was measured. Longer term C retention would depend on the subsequent land management practices. In general, the studies reviewed demonstrate a considerable variation in reported C sequestration which may be partly explained on the dependence on feedstock type, application rates and the period over which the C sequestration is measured.

2.3 LCA studies on composting

A review of the composting Life Cycle Analysis (LCA) literature and LCA software and Life Cycle Inventory (LCI) databases, indicates that, whilst environmental outputs and emissions from processes are often reported in great detail, mass balances are either not calculated or not readily visible in the data presented. In many cases, the lack of mass balances for each life cycle stage arises from the LCA practitioner not recognizing the objectives and limitations of the software and databases, rather than from problems inherent in the program and database suites being used.

A characteristic of compost production not present in most LCIs is that of biogenic emissions, and their corollary, C sequestration. A review of the composting literature reveals that there are many diverse ways of assessing and allocating these (Butler and Hooper, 2010). The issue for composting LCIs is how to strike a balance between over or understating the biogenic emissions from the composting process, the effects on C sequestration from applying compost to land, and, possibly, the raw material procurement stage.

The composting process affects biogenic C emissions in three ways. Firstly, at the raw materials acquisition stage, it is necessary to 'harvest' biomass. The effect is to end any net short-term C sequestration potential of the 'harvested' material. Secondly, the composting process itself produces biogenic CO_2, and other gaseous emissions. Thirdly, the effect of applying compost to land may stimulate plant growth, thereby enhancing the rate of C fixed in net photosynthesis, and the C transferred below ground (Nguyen, 2003), though over a long period of time, C uptake through photosynthesis and loss through respiration will be in balance (Rees et al., 2005).

Colon et al. (2010) performed LCA of home composting including all upstream processes (environmental loads associated with the production of the composting units and the tools used during composting). Some of the most important factors such as GHG emissions, leachate production and the downstream processes (the substitution of peat in growth media and fertilizer) were poorly investigated or even left out of the assessment. Martínez- Blanco et al. (2010) did a more complete LCA study, comparing home composting of Organic Household Waste (OHW) with central composting (tunnel composting) in Spain. In the study by Martínez-Blanco et al. (2010), the assessment of GHG emissions was more complete than the one by Colón et al. (2010) (in which the detection limits on the measuring instruments were too high), but the downstream processes were still not included. It was concluded that home composting can be an accompaniment to central composting, especially in areas with low density population.

The lack of reliable environmental assessments with consistent data was emphasized in the studies by Weidema et al. (2006) and Lundie and Peters (2005), where the authors modeled home composting (in a LCA context) as an intermediate between aerobic and anaerobic digestion (Weidema et al., 2006) and as two scenarios with aerobic (no methane) and anaerobic home composting (Lundie and Peters, 2005).

3. Materials and methods

3.1 Composting process

The composting process studied herein and referred to as the “home composting”, was undertaken in plastic containers with a volume of 400 lt, which were filled completely at the beginning of the experiment. The container’s bottom was open, allowing the direct contact of the compost with the soil. The main raw material used was OTP’s, either on their own or in combination with grasses and seaweeds (co-composting). More specifically, four mixtures were prepared (% by weight):

- C1: 100% OTPs;
- C2: 50% OTPs and 50% grass;
- C3: 50% OTPs and 50% seaweeds, and
- C4: 100% OTPs.

Mixtures C1, C2 and C3 were prepared and studied during the winter period, while C4 was prepared and studied during the summer period, in order to delineate the influence of temperature on the composting process (Fig. 2). Apart from Pigeon Droppings (PD), no specific chemicals were used as activators of the composting process (2 kg of PD were added in each mixture).

The OTP's used were shredded into small pieces (a mean length of less than 2 cm) in order to facilitate their degradation (Fig. 3). A certain period after the process initiation (45 days), it was observed that the compost's mass under investigation was appreciably decreased. Thus, the containers were re-filled by the same mixture, and with the same proportion, so that the compost's temperature reached the levels that are necessary for the process.

As is well documented, the composting process requires adequate conditions for temperature, moisture, oxygenation and nutrients, in order to secure the development-action of the microbial population (Garcia-Gomez and Bernal, 2003; Gigliotti et al., 2012; Konstantakou and Papadakis, 2010; Manios, 2004; Zmora-Nahum et al., 2005). In the home compost, operational parameters (oxygen, temperature, and humidity) were monitored with a frequency of 4-5 days and measured with the same apparatus - probe (EMS 6/3 Series Data Logger).

This apparatus has a sensor - probe OTH 1 (Oxygen/Temperature/ Humidity Sensor), that was placed in the middle of the material's mass and after approximately 5 min the reading was recorded. The ventilation of the home compost was performed manually, with the use of an iron stirrer. Turning and watering were not automatically controlled.

The frequency of both turning and watering was undertaken by a predetermined protocol. More specifically, if any one of the following occurred: (i) $RH < 60\%$ or, (ii) $T > 55$ ^{o}C, or $O_2 < 15\%$, then turning was carried out to achieve the temperature reduction and the proper aeration of the system. Furthermore, if $RH < 60\%$ then watering was carried out in order to enhance the microorganisms viability in the compost's mass and thus, aid in the appropriate evolution of the degradation process. It was observed that it was necessary to turn the materials approximately every 10 days and to water them once a week. It should be pointed out that the mixture that contained grasses, due to its natural moisture, was watered less frequently (every 12-13 days); however it required turning more often than any of the other mixtures.

3.2 Compost analyses

During the composting process, the following physicochemical parameters were measured: (i) Moisture content, which was determined by heating a sample at 105 ^oC until constant weight, (ii) pH and Electrical Conductivity (EC) in the water extract, by diluting 1 part of compost by volume, with 1.5 parts of distilled water (Zmora-Nahum et al., 2005), (iii) Total N, using the micro-Kjeldahl method (Manios, 2004), (iv) Organic matter (OM), which was measured in a muffle furnace through the Loss On Ignition (LOI) at 550 ^oC for 2 hr and expressed on a dry weight (d.w.) basis, (v) Organic carbon. The chemical analyses of the four composts were performed approximately every two weeks. In order to estimate the total C and N quantity at the four buckets, the weight of all materials was recorded both at the beginning and the end of the experiment, when the compost was stable and mature. Additionally, the moisture content of materials was measured and the weight values above were converted into values of dry weight.

4. Results and discussion

4.1 Carbon balance

The carbon content was assayed every 15 days. The C content was found to decrease during the composting process reaching approximately 30% of the initial value in the four final composts (Fig. 4). This organic C content value in the final composts is in agreement with other results reported in the literature with final product values ranging between 30 and

50% (Manios, 2004; Komilis and Tziouvaras, 2009). It should be noted that the available C is primarily lost as CO_2 during the immobilization process (Barrington et al., 2002).

C and N values were measured from the beginning to the end of the composting process, for a period of about 200 days. The materials which were added to the containers were rich in both N (grass) and C (OTP) and therefore the C:N ratio was optimum for all mixtures.

The results of the chemical analysis about the C losses for each container are shown at Table 4. It should be emphasized that the C losses reported are total losses (and not only C content decrease) taking into account the overall mass losses. It was observed that all materials were characterized by an absolute reduction of C from 45 to 65%. The greatest losses recorded were for the containers with OTPs and OTPs plus seaweeds (C1 & C3), while the smallest C losses were recorded for the containers with OTPs plus grass (C2).

It's worth noting that the same material (OTPs), both during winter and summer experiments, showed a similar fluctuation in C values, while the total C content in both final composts was identical. It was also observed that the high N content of materials (OTP and grass) contributes to the reduction of C losses from the mass of composting materials. Therefore, lower C losses during the composting process seem to have benefited by the presence of high N content at the studied materials.

It should be also noted that the C content was decreased in all containers due to the degradation of the materials by micro and macro-organisms. These organisms use C as an energy source for their metabolic activities. The 55-75% of all plant residues released into the atmosphere as CO_2, although there are different rates of release from various organic materials incorporated into the soil. Generally, larger amounts of C are lost to the atmosphere from the organic substrate; the less C is stabilized in the humus of the final compost.

4.1.1 Carbon dioxide emissions

CO_2 accounts for over 99% by mass of total gaseous biogenic emissions from aerobic composting, excluding water vapor. White et al. (1995) concluded that 323 kg of CO_2 (88 kg of carbon) is released for every tone of fresh feedstock composted. This value assumes that the weight of the dry biomass (50% of the fresh feedstock) is subject to a 40% loss of cellulose, 44% of which is C respired in the form of CO_2. However, an analysis of the changes in the organic matter resulting from the composting grape marc, conducted by Inbar et al. (1991), demonstrated that only about 40% of the cellulose mass contained in the initial dry matter is lost during composting which, assuming a cellulose C content of 40% would result in 198 kg of CO_2 being released for 1 tone of fresh feedstock. Other emission measurement and LCA studies quantifying the C emissions from the aerobic composting of various types of feedstock are compatible with these findings (Riffaldi et al, 1986;Jackson & Line, 1997; Jakobsen, 1994; Komilis and Tziouvaras, 2006; Martino-Blanco et al., 2009). Summary data from five of these studies is presented in Table 5.

4.1.2 Methane emissions

The other principal emissions to air from aerobic composting are methane (CH_4), volatile organic compounds of various kinds, ammonia (NH_3) and nitrous oxide (N_2O) (Colón et al., 2010; Martino-Blanco et al., 2009). CH_4 emissions may arise from anaerobic pockets within unturned windrow piles, but where forced aeration is accompanied by periodic windrow turning any methane will be oxidized, resulting in minimal residual emissions. A study of municipal waste composting by Martino-Blanco et al. (2009) reported CH_4 emissions equivalent to 4 g/tone of wet organic waste. By contrast, Colón et al. (2010) reported CH_4 emissions from home composting in the region of 3000 g/tone wet waste, which may be partly a function of home composting in containers with little or no aeration.

4.2 Nitrogen balance

The Nitrogen (N) content was also assayed every 15 days. The highest N content was determined for the OTPs and grass mixture (C2) and the lowest for the OTPs and seaweeds mixture (C3) (Fig. 5). Total N content in the four final composts was about 1.5%, an adequate value for their use in organic cultivations. This nitrogen content value for the four composts coincides well with results of other similar studies in which the N value in the final products ranges between 1.2-1.7% d.w. (Manios, 2004, Komilis and Tziouvaras, 2009). It has been estimated that the majority of N was lost in the form of NH_3, with volatilization accounting for over 92% of all N losses.

The experimental results for the absolute N loss shown that the N losses were much greater than C losses, since N losses were close to 62 to 76%. The most significant N decrease was observed, as for C, at the containers 1 and 4 (OTPs), while the smallest N loss was recorded for containers 2 and 3 (OTPs with grass and OTPs with seaweeds, respectively). However, the N loss did not show great variation among the four samples (Table 6).

N loss is justified by its removal from the compost mass in the form of gaseous ammonia (NH_4) to the environment, which is produced by bacteria acting during the phase of decomposition of organic matter. In addition, N loss comes from the removal of materials by watering, as the base of the experimental containers was not closed, but in direct contact with the ground. Moreover, a slight decrease may be due to the N consumption by microorganisms, for covering their metabolic activity. It should also be pointed out that for the same material (OTPs), but for different composting periods (winter-summer), different N losses were recorded.

4.3 Carbon to Nitrogen (C/N) ratio

The C/N ratio is widely used as an indicator of compost maturity. For the home or industrial composting process to work, the recommended range of C/N ratio of the input material is 20-40 g/g (de Guardia et al., 2010). However, the changes in the C/N ratio reflect the organic matter decomposition and the stabilization during composting. The C/N ratio is often used as an index of compost maturity with an optimum value between 15–25, as reported by Manios (2004). Consequently, the eventual C/N ratios of the present study showed that all produced composts are mature. However, in some cases the C/N ratio may not be a good indicator of compost maturity as it can level off before the compost stabilizes (Zmora-Nahum et al., 2005). The C/N ratio of all the compost mixtures increases substantially for about 200 days, and stabilizes thereafter (Raj and Antil, 2011). As the decomposition progresses, due to losses of C mainly as CO_2, the C content of the compostable material decreases with time, and the N content per unit of material decreases at a lower rate, which results in the increase of C/N ratio. Composts containing agro-industrial wastes show a more rapid decrease in the C/N ratio compared to farm waste composts (Bernal et al.,1998). The addition of agro-industrial wastes as organic additives might help in increasing the biological activity. Golueke (1981) reported that a C/N ratio below 20 is indicative of a mature compost, while Bernal et al. (2009) stated that a ratio of 15 or less is preferable. Sellami et al. (2008) on the other hand, reported that the C/N ratio is not a good indicator of maturity in a compost due to the large variability in raw materials. They also argued that the C/N ratio may not reflect a material which is sufficiently decomposed (Sellami et al., 2008). The C/N ratio could affect the NH_3 ($p = 0.0442$) and CH_4 ($p = 0.0246$) emissions significantly, but not the N_2O. Lower C/N ratio causes higher NH_3 and CH_4 emissions.

Rashad et al. (2010) reported that the narrowest C/N ratios were achieved in treatments that received vinasse and were inoculated with composite inoculum including cellulolytic fungi (trichoderma and phanerochate) or buffalo’s manure and composite inoculums, which accelerated the decomposition of organic matter as corroborated by biodegradability coefficients (Kb). In their study, the initial C/N ratio of co-composting mixtures was adjusted to about 30/1, the ideal value to accelerate the microbial decomposition of organic matter as recommended by Hansen et al. (1990). Moreover, specific microbial strains or combinations to enhance the degradation of relatively recalcitrant materials were used. As has been demonstrated by Diaz et al. (2002) that vinasses containing carbon and nitrogen in highly biodegradable forms, helping the proliferation of the microbial population and comprise the best solution in optimizing the composting process and obtain a high quality product only when added in a moderate amount (10–20%). Beside, its acidic nature can minimize NH_3 losses due to the immobilization of NH^+_4 in the composting material (Whitely and Pettit, 1994). In addition, total N can also be increased by the activities of N-fixing bacteria at the end of the composting process (Bishop and Godfrey, 1983).

In Table 7 literature data of initial and final C/N (g/g), C and N loss (%) for various waste materials and experimental conditions, as duration, temperature (°C) and moisture content (%) are shown. It can be observed that values are widely ranging, depending greatly on material and composting conditions.

5. Conclusions

Home composting occurs when bacteria and fungi degrade biomass under aerobic conditions at ambient temperatures (<35°C). Compared with industrial composting, here the biomass remains at lower temperatures, and is mixed less frequently and as a result, biomass degrades more slowly. As the adding of organic matter to the soil through compost increases the soil's carbon level, this helps make up for the reduction in carbon content caused by an increase in crop yields or other soil activities. Stable carbon compounds created by the composting process include an increase in humic substances and aggregates allowing carbon to be stored for long periods of time in the soil. The C/N ratio is widely used as an indicator of compost maturity. In general, for both home and industrial composting, the recommended range of carbon to nitrogen ratio of the input material is 20-40 g/g. Care should be taken as changes in the carbon-to-nitrogen proportion reflect the organic matter decomposition and the stabilization during composting.

In the present work, the composting of OTP's, either on their own or used in mixtures with either grass or seaweeds, was studied during both the winter and summer periods. Operational parameters, physicochemical characteristics and agronomic values were measured as a function of time, elucidating the composting process of this material and offering valuable observations to potential practitioners. The study concludes that OTP's can be converted into a highly quality compost with the potential to help current agricultural practices become more sustainable. Further, the co-composting of OTP's with either grasses or seaweeds, can enhance several characteristics of the final product.

In summary, OTP's are the largest agricultural residue in Greece, amounting to several million tones per year. Arguably, the current practice of burning them in the countryside causes both energy losses and severe air pollution. In this respect, the composting of OTP's is an attractive alternative and the present study could help pave the way towards more sustainable waste management practices. The solution proposed herein could be extended to other agricultural residues and can be applied by farmers, enterprises or local authorities, enhancing both the exploitation of natural resources and agriculturists' incomes.

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Table 1. Biomass resources estimates for selected EU countries (Esteban and Carrasco, 2011)

Country	**Agricultural** **Potential**	 **Available**	 **MAPV***
	Tg yr^{-1}	Tg yr^{-1}	Mg ha^{-1} y^{-1}
Sweden	5.93	3.11	4.19
Finland	3.08	1.12	1.9
Germany	67.34	42.56	6.85
Norway	1.85	0.83	0.26
Austria	6.25	3.79	5.00
Poland	28.09	10.36	2.87
Denmark	12.99	8.17	5.80
SUBTOTAL	**125.53**	**69.94**	**-**
France	78.39	49.81	4.24
Spain	26.22	12.84	1.23
Italy	25.44	16.94	2.35
Greece	7.42	4.82	2.09
Portugal	2.38	1.70	0.96
SUBTOTAL	**139.85**	**86.11**	**-**
TOTAL	**265.38**	**156.05**	**-**

*Mean Annual Productivity Value

Table 2. Comparison of biomass resources estimates for selected EU countries (PJyr^{-1}) (Esteban and Carrasco, 2011)

Country	**EEA**	**CHRISGAS**	**RENEW**	**Nikolau et al**
Spain	296.8	228.6	128.5	126.0
France	530.9	886.6	573.4	412.0
Italy	673.0	301.5	230.4	163.3
Greece	66.9	85.8	73.0	69.0
Portugal	112.9	30.3	18.9	25.8
Sweden	372.0	55.4	14.9	5.5
Austria	125.4	67.5	42.3	9.0
Denmark	96.1	145.4	38.7	28.9
Finland	255.0	19.9	17.2	9.7
Germany	622.8	757.6	189.0	130.0
Poland	305.1	184.4	133.7	125.0
Total	**3456.9**	**2762.9**	**1460.2**	**1104.2**

Table 3. Ultimate and proximate analysis of olive kernel and OTBs (Skoulou et al, 2008)

Sample	**Olive kernels**	**OTBs**
Ultimate analysis (w/w%, dry basis)		
Carbon	48.61	47.27
Hydrogen	6.41	6.41
Oxygen	46.32	44.98
H.H.V (MJ/kg)	20.39	19.13
Proximate analysis (w/w%)		
Moisture	4.59	4.84
Volatiles	75.56	78.31
Fixed C	16.39	8.47
Ash	3.46	0.62

Table 4. Total carbon losses at materials from the different mixtures

	C1	**C2**	**C3**	**C4**
Total initial amount of materials (kg)	23.8	30.6	41.5	32.3
% TOC initially	*50.5*	*45.1*	*50.2*	*49.7*
kg C at the initial amount of materials	12.0	13.8	20.8	16.0
Total final amount of materials (kg)	15.0	24.5	22.0	24.2
% TOC finally	*28.4*	*30.9*	*35.2*	*28.4*
kg C at the final amount of materials	4.3	7.6	7.7	6.9
C loss (kg)	7.7	6.2	13.1	9.1
% loss	*64.6*	*45.1*	*63.0*	*57.2*

Table 5. Comparison of literature data on carbon and nitrogen emissions for home composting per kg carbon input assuming short-term carbon storage in compost

Study	Feedstock type	Feedstock volume	Composting process	Total processing period	Emissions (CO_2 kg/tonne)
Jackson & Line, 1997	Pulp & paper mill sludges, urea, ammonium sulfate ammonium nitrate	75m^3 windrows	Wind rowing	21 weeks	191
Jakobsen, 1994	Garden organics, food organics, sewage sludge, pig & cattle manure	n/a	n/a	n/a	182
Komilis & Tziouvaras, 2006	Yard wastes –grass clippings & leaves	Simulated composting using 25l airtight stainless-steel digesters	n/a	n/a	217
Martino-Blanco et al, 2009	Organic MSW & pruning wastes(1:1)	Not stated	Forced aeration & Wind rowing	10 weeks	165
Riffaldi et al, 1986	Paper processing, sludge, straw	11.25m^3 static pile	Forced aeration & Wind rowing	60 days composting, 80 days maturation	193

Table 6. Total nitrogen losses at materials of the four buckets

	C1	**C2**	**C3**	**C4**
Total initial amount of materials (kg)	23.8	30.6	41.5	32.3
% N initially	3.5	3.7	2.4	3.3
kg N at the initial amount of materials	0.8	1.1	1.0	1.1
Total final amount of materials (kg)	15.0	24.5	22.0	24.2
% N finally	1.5	1.7	1.5	1.5
kg N at the final amount of materials	0.2	0.4	0.3	0.4
N loss (kg)	0.6	0.7	0.7	0.7
% loss	76.0	62.8	70.0	66.3

Table 7. Comparison of literature data of initial C/N (g/g), final C/N (g/g), C loss (%), N loss (%) for various waste materials and compost experimental conditions, as duration, temperature (ºC) and moisture content (%)

Reference	Waste Material	Duration	Temperature (ºC)	Moisture content (%)	Initial C/N (g/g)	Final C/N (g/g)	C loss (%)	N loss (%)
Chikae et al., 2006	food wastes and tree cuttings	130 days Ph=5.5-8.0 →7.5	65-70 -> 35	45% water	26.7 31.5	14.9 22.3	8.5 8.3	n/a
Fernández et al., 2008	exhausted grape marc (EGM), cow manure and straw (CMS), municipal solid waste (MSW), grape stalks (GS)	120 days 19% O2 ph=7-8.3	30-60-40 25-60-40 45-60-35 30-60-35	68.3 (EGM) 71.0 (EGM+CMS) 59.1 (EGM+MSW) 49.1 (EGM+GS)	10 11 10 30	n/a	34 31 50 60	4.0 4.0 4.0 1.4
de Guardia et al., 2010	household waste, separated pig solids, food waste, pig slaughter-house sludge & green algae	27 to 50 days	15-60-20 30-65-30 30-50-20 20-70-25 10-55-20	55% -70%	80.7 (HW+WC) 12.3 (SPS) 46.6 (FW+WC) 441.3 (PSS+WC) 374.5 (GA+WC)	n/a	n/a	n/a
Rashad et al., 2010	rice straw, soybean residue enriched with rock phosphate	12 weeks (84 days) ph=7-7.5	35-65-30	70% moisture	32.8	16-22	23.82 23.60 19.53 17.10 13.64	n/a
Kaboré et al., 2010	urban wastes: dry tree leaves, household refuses as vegetables, paper from city offices, slaughterhouse wastes	12 weeks ph=5.8 (P1) ph=6.2 (P2) ph=9.0 (P3) ph=7.3 (P4)	n/a	50–70% (w/w)	37.2(SW) 26.8 (SW+HR) 34.6 (SW+HR$^+$) 24.8 (SW+HR-) 33.9 (HR) 49.8 (TM)	19.1 18.2 20.6 18.2 18.4 25.8	n/a	n/a
Sun et al., 2011	(1) potato, (2) carrot, (3) ground pork, (4) steamed rice, (5) American Elm leaves, (6) cooked soybean	42 days	50-20	56.98 56.38 62.55 66.65 64.27 50.09	14.75 (P1) 15.08 (P2) 32.63 (P3) 29.89 (P4) 31.90 (P5) 34.99 (P6)	12.70 12.70 19.43 17.15 11.79 12.84	n/a	n/a
Raj et al., 2011	farm & agro-industrial wastes	150 days	40-60-35	n/a	65 (C1) 30 (C2-C5)	36.7 (C1) 11.7 (C2) 17.6 (C3) 14.1 (C4) 14.4 (C5)	27.1 44.8 40.8 43.8 42.5	n/a
Our study	olive tree lops, grass, seaweeds	200 days	n/a	n/a	15.0 (C1) 12.5 (C2) 20.8 (C3) 14.5 (C4)	21.5 (C1) 19.0 (C2) 25.7 (C3) 17.2 (C4)	64.6 45.1 63.0 57.2	76.0 62.8 70.0 66.3

Notes: CMS: Cow Manure and Straw, FW: Food Waste, EGM: Exhausted Grape Marc, GA: Green Algae, GS: Grape Stalks, HR: Household Refuge, HW: Household Waste, MSW: Municipal Solid Waste, PSS: Pig Slaughter house Sludge, SW: Slaughterhouse Wastes, TM: Tree Leaves

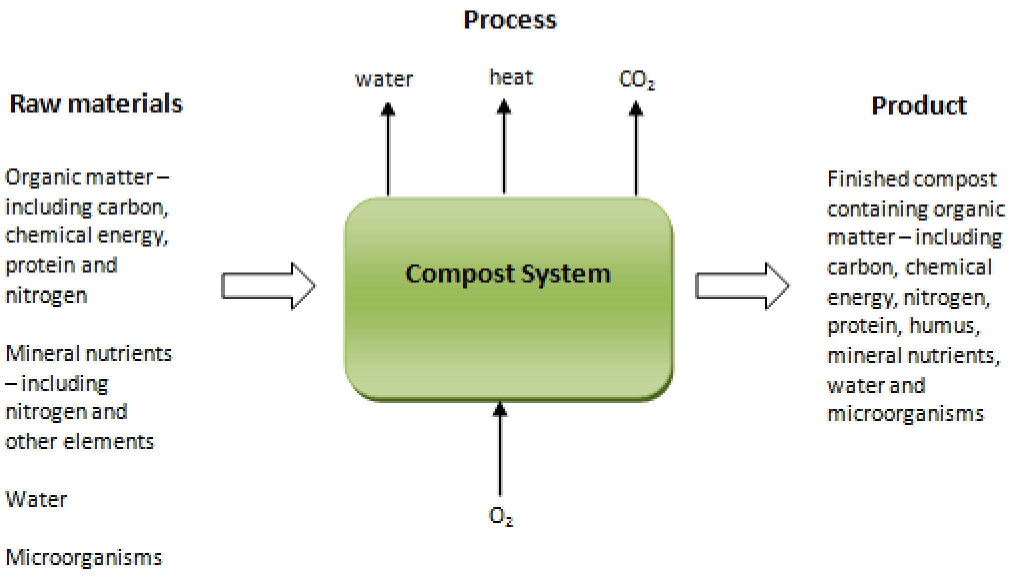

Fig. 1. The composting process (adapted from Mohee, 2007)

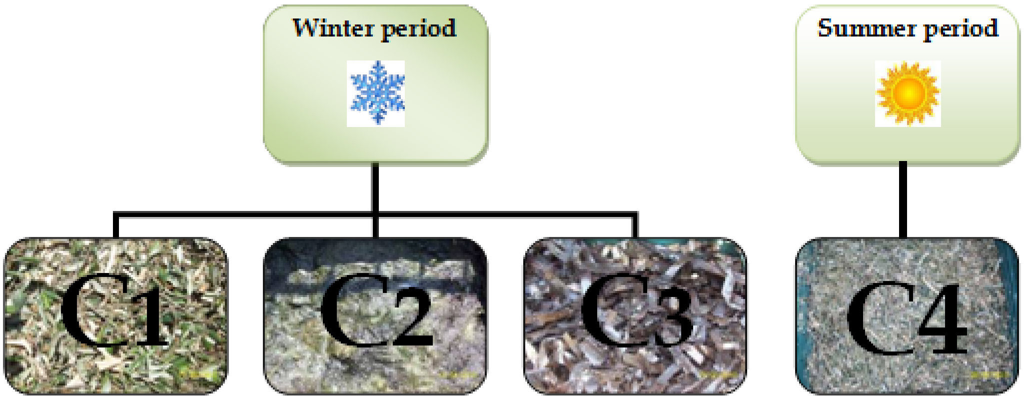

Fig. 2. Schematic representation and images of the four mixtures for composting; C1: OTP; C2: OTP and grass; C3: OTP and seaweeds and C4: OTP.

Fig. 3. (a) Experimental composting containers, (b) fresh and shredded OTPs (c) mature compost

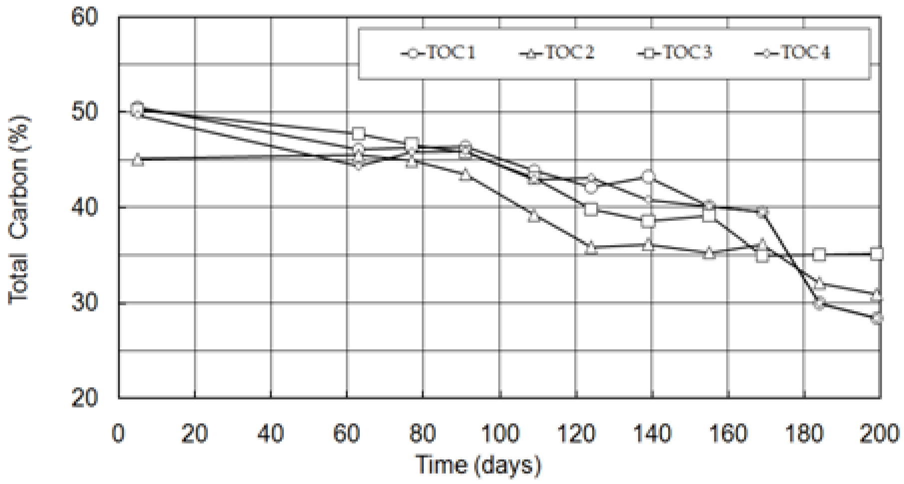

Fig. 4. Evolution in time of TOC for the four different composts

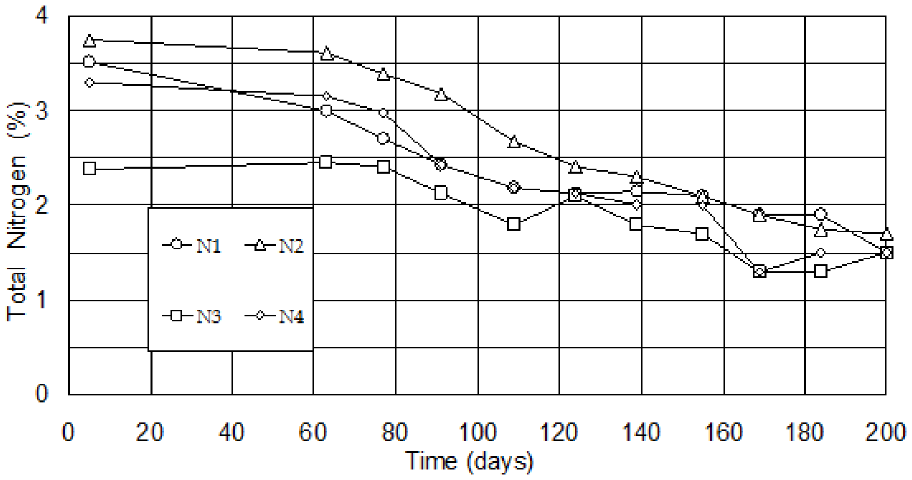

Fig. 5. Evolution in time of N for the four different composts