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ASSESSMENT OF THE EVOLUTION AND EFFICIENCY OF DIFFERENT INDUSTRIAL COMPOSTING TECHNOLOGIES

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TABLE OF CONTENTS

1. INTRODUCTION ... 1
1.1. Foreword ... 1
1.2. Objectives ... 3
2. LITERATURE REVIEW ... 4
2.1. Composting process and phases ... 4
2.2. Organic matter transformation during composting ... 6
2.2.1 Organic matter fractions ... 6
2.3. Chemical processes in composting ... 7
2.3.1 Carbon and nitrogen ... 7
2.3.2 Phosphorus ... 8
2.3.3 Sulfur ... 9
2.3.4 Heavy metals ... 9
2.4. Effect of using organic wastes on soil ... 10
2.4.1 Benefits ... 10
2.4.2 Risks ... 13
2.5. Composting feedstocks ... 15
2.6. Factors affecting decomposition of waste and composting process ... 17
2.6.1 Turning frequency ... 17
2.6.2 Temperature ... 18
2.6.3 C/N ratio ... 19
2.6.4 Moisture content ... 20
2.6.5 Electrical conductivity ... 21
2.6.6 Aeration ... 22
2.6.7 pH ... 23
2.6.8 Particle size ... 24
2.6.9 Porosity ... 24
2.7. Composting processes used industrially ... 25
2.7.1 Windrow process ... 25
2.7.2 Aerated static windrow ... 26
2.7.3 Rotating drum reactor ... 27
2.7.4 Two-stage composting technology with with GORE® Cover membrane ... 27

2.7.5 Aerated static windrow with GORE® Cover membrane 28

2.8. Compost stability evaluation 30

2.8.1 Colour 32

2.8.2 Temperature 32

2.8.3 Loss of organic matter 32

2.8.4 pH 32

2.8.5 Electrical conductivity 33

2.8.6 C:N ratio (solid phase) 33

2.8.7 C:N ratio (water extract) 33

2.8.8 Water soluble carbon 33

2.8.9 Nitrification index 34

2.8.10 Cation exchange capacity 34

2.8.11 Humification 34

2.8.12 Germination test 35

2.8.13 Oxygen and CO_2 respirometry 35

2.8.14 Microbial population count 35

2.8.15 Enzyme activities 36

2.9. Humification 36

3. MATERIALS AND METHODS 38

3.1. Composting setup and description of encapsulated lifting system with GORE® Cover membrane 38

3.1.1 Composting materials 38

3.1.2 Composting process 38

3.1.3 Ventilation system 40

3.1.4 Sampling strategy and collection 40

3.1.5 Temperature monitoring 40

3.2. Composting setup and description of aerated static windrow with GORE® Cover membrane 41

3.2.1 Composting materials 41

3.2.2 Composting process 41

3.2.3 Ventilation system 42

3.2.4 Sampling strategy and collection 42

3.2.5 Temperature monitoring.. 43

3.3. GORE® Cover membrane 43

3.4. Physico-chemical analyses of compost samples 44

3.4.1 Moisture content and dry matter 44

3.4.2 Bulk density.. 44

3.4.3 Ash content and organic matter.. 44

3.4.4 Loss of organic matter 45

3.4.5 pH and electrical conductivity.. 45

3.4.6 Total phosphorus and total potassium 45

3.4.7 Oxidizable organic carbon 45

3.4.8 Total nitrogen and total carbon 46

3.4.9 Cation Exchange capacity 46

3.4.10 Ammoniacal nitrogen and nitrate nitrogen... 46

3.4.11 Water soluble carbon.. 46

3.4.12 Macro-nutrients, micro-nutrients and metal contents 47

3.5. Germination assay.. 47

3.6. UV-visible spectroscopic analysis 48

3.7. FTIR spectroscopic analysis... 48

3.8. Statistical analyses.. 48

4. RESULTS AND DISCUSSION.. 50

4.1. Encapsulated lifting system with GORE® Cover membrane 50

4.1.1 Temperature evolution 50

4.1.2 Moisture and dry matter 52

4.1.3 Bulk density.. 53

4.1.4 pH variation.. 53

4.1.5 Variation in electrical conductivity 55

4.1.6 Losses of organic matter and ash content... 56

4.1.7 Water soluble carbon.. 57

4.1.8 Total carbon and oxidisable organic carbon 58

4.1.9 C/N ratio 59

4.1.10 NH_4^+/NO_3^- ratio.. 60

4.1.11 Cation exchange capacity... 61

4.1.12 UV-visible spectroscopic analysis 62
4.1.13 Total phosphorus and total potassium 63
4.1.14 Seed germination index 63
4.1.15 Principal components analysis 64
4.1.16 FTIR spectroscopic analysis 65
4.2. Aerated static windrow with GORE® Cover membrane 68
4.2.1 Temperature evolution 68
4.2.2 Moisture and dry matter 70
4.2.3 Bulk density 71
4.2.4 pH variation 72
4.2.5 Variation in electrical conductivity 74
4.2.6 Losses of organic matter and ash content 74
4.2.7 Water soluble carbon 76
4.2.8 Total carbon and oxidisable organic carbon 77
4.2.9 C/N ratio 78
4.2.10 NH_4^+/NO_3^- ratio 79
4.2.11 Cation exchange capacity 80
4.2.12 $E_4/_6$ 81
4.2.13 Total phosphorus and total potassium 82
4.2.14 Principal components analysis 83
4.2.15 FTIR spectroscopic analysis 84
4.3. Comparison of FTIR spectra 86
4.4. Characteristics of the final compost 86
5. NEW SCIENTIFIC RESULTS 88
6. CONCLUSIONS 89
7. SUMMARY 90
8. SCOPE FOR FUTURE RESEARCH 92
9. ÖSSZEFOGLALÁS 93
10. BIBLIOGRAPHY (LIST OF WORKS CONSULTED) 95
11. ACKNOWLEDGEMENTS 96
12. REFERENCES 98

LIST OF TABLES:

Table 1. Maturity parameters established for composts of different sources (adapted from Antil et al. 2014) 31
Table 2. Characteristics of the raw materials (DSSS-GW) used for composting 38
Table 3. Characteristics of the raw materials used for composting 41
Table 4. Summary of the characteristics of final compost obtained by using ELS with GORE® Cover membrane and ASW with GORE® Cover membrane, in comparison with guidelines for acceptable quality of compost 87

LIST OF FIGURES:

Figure 1. Scheme showing the temperature profile during the composting process (solid line for static pile; dashed line for turning pile), indicating composting phases and pH evolution (dashed-dots line). Adapted from Bernal et al. (2017) 4
Figure 2. Schematic diagram of the encapsulated lifting system with GORE® Cover membrane 39
Figure 3. Schematic diagram of the aerated static windrow with GORE® Cover membrane 42
Figure 4. Evolution of temperature during DSSS-GW composting 51
Figure 5. Moisture content and dry matter evolution during DSSS-GW composting 52
Figure 6. Changes in BDwet and BDdry the during DSSS-GW composting 53
Figure 7. Evolution of pH and EC during DSSS-GW composting 55
Figure 8. Evolution of OM_{loss} and ash content during DSSS-GW composting 56
Figure 9. Evolution of WSC during DSSS-GW composting 57
Figure 10. Changes TC and OXC content during DSSS-GW composting 58
Figure 11. Evolution of the C/N ratio during DSSS-GW composting 59
Figure 12. Evolution of the NH_4^+/NO_3^- ratio during DSSS-GW composting 60
Figure 13. CEC evolution during DSSS-GW composting 61
Figure 14. Evolution of the $E_{2/4}$ (280/472 nm), $E_{2/6}$ (280/664 nm), and $E_{4/6}$ (472/664 nm) ratios at different stages of composting DSSS-GW 62
Figure 15. Macronutrients (TP and TK) evolution during DSSS-GW composting 63
Figure 16. Evolution of germination index during DSSS-GW composting 64
Figure 17. Principal component analysis of the physico-chemical parameters measured during DSSS-GW composting 65
Figure 18. FTIR spectra of DSSS-GW at various stages of composting. T0: day 0; T7: day 7; T35: day 35; and final compost 66
Figure 19. Evolution of the temperature during GW composting 68
Figure 20. Moisture content and dry matter evolution during GW composting 70
Figure 21. Changes in BDwet and BDdry the during GW composting 72
Figure 22. Evolution of the pH and EC during GW composting 73
Figure 23. Evolution of OM_{loss} and ash content during GW composting 75
Figure 24. Evolution of WSC during GW composting 76
Figure 25. Changes TC and OXC content during GW composting 78

Figure 26. Evolution of the C/N ratio during GW composting 79

Figure 27. Evolution of the NH_4^+/NO_3^- ratio during GW composting 80

Figure 28. CEC evolution during GW composting 81

Figure 29. Changes in $E_{4/6}$ ratio during GW composting 82

Figure 30. Macronutrients (TP and TK) evolution during GW composting 82

Figure 31. Principal component analysis of the physico-chemical parameters measured during GW composting 83

Figure 32. FTIR spectra of GW at various stages of composting (D 0: day 0, D 14: day 14, and final compost) 84

LIST OF EQUATIONS:

Equation 1. Dry matter 44

Equation 2 Dry bulk density (BD_{dry}) 44

Equation 3 Wet bulk density (BD_{wet}) 44

Equation 4. Organic matter 44

Equation 5. Losses of organic matter 45

Equation 6. Germination index 48

LIST OF ABBREVIATIONS

ASW	AERATED STATIC WINDROW
BD_{dry}	DRY BULK DENSITY
BD_{wet}	WET BULK DENSITY
CEC	CATION EXCHANGE CAPACITY
C_{EX}	ALKALI-EXTRACTABLE ORGANIC-C
C_{FA}	FULVIC ACID-LIKE ORGANIC CARBON
C_{HA}	HUMIC ACID-LIKE ORGANIC CARBON
DM	DRY MATTER
DRIFT	DIFFUSE REFLECTANCE INFRARED FOURIER TRANSFORM
DSSS	DEWATERED STABILIZED SEWAGE SLUDGE
EC	ELECTRICAL CONDUCTIVITY
ELS	ENCAPSULATED LIFTING SYSTEM
ePTFE	EXPANDED POLYTETRAFLUOROETHYLENE
ESR	ELECTRON SPIN RESONANCE
FTIR	FOURIER TRANSFORM INFRARED
GC–MS	GAS CHROMATOGRAPHY-MASS SPECTROMETRY
GHGs	GREENHOUSE GASES
GI	GERMINATION INDEX
GW	GREEN WASTE
HA	HUMIC ACIDS
HM	HEAVY METAL
ICP-OES	INDUCTIVELY COUPLED PLASMA-OPTICAL EMISSION SPECTROSCOPY
MC	MOISTURE CONTENT
MMSW	MIXED MUNICIPAL SOLID WASTE
MSW	MUNICIPAL SOLID WASTE
NH_4^+-N	AMMONIUM NITROGEN
NMR	NUCLEAR MAGNETIC RESONANCE
NO_3^--N	NITRATE NITROGEN
OM	ORGANIC MATTER
OXC	OXIDIZABLE ORGANIC CARBON
PC	PRIMARY COMPOSTING
PCA	PRINCIPAL COMPONENTS ANALYSIS
PCB	POLYCHLORINATED BIPHENYLS
PCDD/Fs	POLYCHLORINATED DIBENZO-P-DIOXINS, DIBENZOFURANS
SC	SECONDARY COMPOSTING
SS	SEWAGE SLUDGE
SSOW	SOURCE SEPARATED ORGANIC WASTE
TC	TOTAL CARBON
TK	TOTAL POTASSIUM
TN	TOTAL NITROGEN
TOC	TOTAL ORGANIC CARBON
TP	TOTAL PHOSPHORUS
WSC	WATER SOLUBLE CARBON

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I dedicate this thesis to
my family
for their constant support and unconditional love.
I love you all dearly.

1. INTRODUCTION

1.1. Foreword

The momentous rise of populations on a worldwide scale due to advanced levels of industrialization and urbanization has led directly rise in the generation of solid waste (Singh et al., 2011). Globally, it has been estimated that 1.3 billion metric tons of waste are generated annually, which will rise to 38-67 % by the close of 2025. It includes sewage sludge (SS), industrial, agricultural, and municipal waste (García-Sánchez et al., 2017). Wastes, such as animal wastes, garden wastes, municipal solid waste (MSW), food and kitchen waste, agricultural wastes, agro-industrial wastes, SS, etc. can be classified as solid organic wastes with an organic biodegradable fraction and a moisture level of less than 85-90 % (Mata-Alvarez et al., 2000). The majority of developing countries globally produce massive volumes of solid waste that is handling and managed badly owing to the absence of waste segregation and suitable disposal facilities (Ngoc and Schnitzer, 2009). Moreover, inappropriate waste management practices such as open dumping or burning of waste can lead to pollution and destruction of the urban landscape (Amritha and Kumar, 2019), but also make sustainable means of waste management, which is a great challenge. There is greater risk of soil pollution in regions adjacent to open waste dumping sites due to leaching of contaminants such as heavy metals. Furthermore, erosion, runoff and leaching from these sites can also result in pollution of surface water and ground water (Mor et al., 2006). These can all lead to many water-borne diseases that are transmitted via water e.g. diarrhea, cholera and typhoid. Burning or incineration of MSW can also lead to the releases of noxious, toxic gases and polychlorinated dibenzo-p-dioxins, dibenzofurans (PCDD/Fs), and putrefaction of MSW due to open dumping results in release of greenhouse gases (GHGs) such as CO_2, CH_4, and N_2O causing global warming and air pollution (Vaish et al., 2019). The biodegradable organic fraction predominates the MSW stream in developing nations and is one of the biggest causes of anthropogenic GHGs emissions (Narayana, 2009; Tian et al., 2013). Finally, open dumping of wastes affects also the esthetic look of a region by becoming a collection point for flies, insects and pathogens and lead to the release of bad odors. Such unscientific conventional waste disposal methods are a danger to the quality of the environment and to public health, moreover, they lead to loss of nutrients present in the organic waste and economic loss (Yadav and Garg, 2011).

One means of ensuring the eco-friendly and sustainable recycling of organic waste (compared to conventional methods of waste disposal and energy recovery) is through use of integrated waste management hierarchy (Sharma et al., 2017). Keeping to the rules of resource/nutrient recovery and recycling, scientific methods of conversion (via. such as composting, vermicomposting) and uses of organic wastes for agriculture purposes can provide useful nutrients for improving plant growth and soil fertility. The purpose of uses organic wastes in agronomy is to sustain a sustainability cycle in which the organic fraction of biodegradable waste is transformed into fertilizer or beneficial organic manure via vermicomposting, composting or anaerobic digestion to produce slurry and compost for agricultural purpose. Today's European policy aims to reduce, recycle and revalorize waste to ensure a circular economy. Therefore, in this light, organic wastes constitute a reservoir of nutrients with a large amount of inorganic and organic substances that must be recycled (Kacprzak et al., 2017).

Composting is one such management strategy that has been gaining global popularity due to its ability to hygienization of waste, cost effectiveness and conversion of waste to value-added products (Sharma et al., 2019). Composting imply biological conversion of heterogenous solid organic wastes (e.g. crop residues, garden waste, SS, kitchen and food waste, paper, wood) into humus-like substances with the help of microorganisms like fungi, bacteria and actinomycetes under controlled conditions of aeration, temperature and moisture (Atalia et al., 2015). The primary objectives of composting are to: (1) decreasing both volume and mass of the solid organic wastes, (2) destruction of pathogens and weed seeds present in the substrate, (3) decrease and or even eliminate inorganic and organic contaminants, and (4) obtain organic materials that are both stable and mature (Senesi et al., 2007).

However, to obtain a high-quality compost product, efficient composting technologies must be used (Cesaro et al., 2015). Currently, various composting technologies are available; the choice of technology depends on the operational cost, the time needed to reach compost maturity, the land availability, and the source of raw materials. In recent years, composting technologies have been subjected to considerable modifications (e.g., cocomposting, vermicomposting, and windrow composting) to make composting technology more cost effective (Zhou et al., 2018).

1.2. Objectives

Many previous studies have focused on evaluating the effects of composting process using a semi-permeable cover, in combination with an air-insufflation system on GHGs and odor emissions (González et al., 2016; Komilis et al., 2004; Ma et al., 2017, 2018b; Robledo-Mahón et al., 2019a; Sun et al., 2016, 2018; Turan et al., 2009; Van Haaren et al., 2010). On the other hand, other studies focused on assessment of microbial community succession in a full-scale composting with a semi-permeable membrane under slight positive pressure (Robledo-Mahón et al., 2020, 2018). However, no studies so far have analyzed in detail the physico-chemical properties of composts and provided insight into the mechanisms of how these properties changed while composting through using relatively new composting technologies that are aerated static windrow (ASW) with GORE® Cover membrane and encapsulated lifting system (ELS) with GORE® Cover membrane. Therefore, this study tested the hypothesis that these two relatively new composting technologies will produce mature compost within a shorter time than conventional composting process. This research has three main objectives: (1) characterize the changes in the physico-chemical properties during composting, (2) monitor the composting process using UV-visible and Fourier transform infrared (FTIR) spectroscopy to evaluate the humification degree, and (3) evaluate the quality of the final compost product.

The results obtained can help to safely use dewatered stabilized sewage sludge (DSSS) and green waste (GW) while avoiding their secondary contamination resulting from the improper treatment of these wastes. In addition, the results can improve and develop these two relatively new composting technologies to produce a compost product that is useful as a soil amendment or fertilizer.

2. LITERATURE REVIEW

2.1. Composting process and phases

The composting process occurs through the aerobic biothermal oxidation of the organic matter (OM) within the biowaste by chemoheterotrophic microorganisms. **Figure 1** shows the four phases of composting.

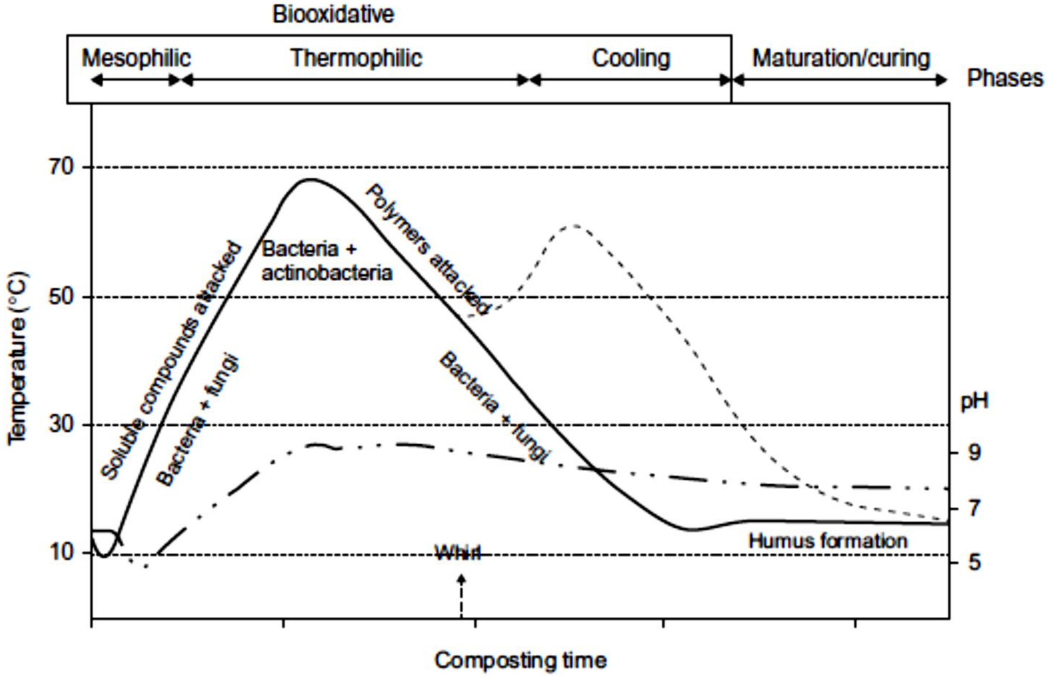

Figure 1. Scheme showing the temperature profile during the composting process (solid line for static pile; dashed line for turning pile), indicating composting phases and pH evolution (dashed-dots line). Adapted from Bernal et al. (2017)

I. Phase 1 is the initial *mesophilic phase* during which the biowaste is colonized by bacteria. The colonizing microorganisms can actively adapt to the created environment, whether that is within a compost container or as a stacked biowaste pile. This phase lasts between 1-3 days, during which simple substances, such as sugars, proteins, and amino acids are degraded by the mesophilic microorganism. The temperature within the waste pile rapidly increases until it reaches a self-limiting temperature. Then, there is a high rate of microbial transformation of the OM, with the heat being generated by its oxidation (Bernal et al., 2017).

II. Phase 2 is the *thermophilic phase* which lasts between 2-4 weeks. During this phase, the production of heat exceeds any heat lost from the waste pile resulting in a temperature increase of 60-70 ^{o}C. At this time, bacteria are the most active colonizing microorganisms and together with other thermophilic microorganisms, they are able to degrade cellulose, hemicellulose, fats and lignin. OM degradation is at its maximum during this phase, together with the elimination of weed seeds and pathogens. The organic components in the waste are oxidized by the microorganisms to generate energy for growth and cell functioning, whilst carbon is used to generate new cell material. The degradation of OM also generates the carbon dioxide and water (Bernal et al., 2017).

III. Phase 3 is the *mesophilic phase* or *cooling phase* during that time the activity of the thermophilic microorganisms decreases, and the temperature of the compost pile lowers to that of the ambient surroundings. This reduction in activity is correlated with the depletion of organic components upon which they can feed. With the reduction of thermophilic bacteria and associated lowering of temperature, the compost pile is recolonized by actinobacteria, fungi and other mesophilic microorganisms. According to Insam and de Bertoldi (2007), these organisms are able to degrade the remaining cellulose, hemicellulose and lignocellulosic materials. Hubbe et al. (2010) explained that these microorganisms are present as spores or other dormant forms during the thermophilic phase at the temperature levels above 50 °C, but when temperature drop to level below 50 ^{o}C , these microorganisms become active and are able to hydrolyze recalcitrant lignified materials, which encourages further bacterial growth on the released decomposable substrates. The combination of these activities and the increased microbial activity generate heat within the compost pile which is maintained above that of ambient temperature. Cáceres et al. (2018) explained that it is likely that NH_4^+ nitrification occurs at this stage, which can result in notable emission of N_2O if it occurs simultaneously with denitrification.

IV. Phase 4 is the *maturation* or *curing phase*. During this phase the temperature drops to that of ambient and remains at that temperature, and humification of the OM and stabilization of the microbes occurs (Hubbe et al., 2010). In this phase the low quantity of remaining degradable organic components and the low activity of the microbes results in slow decomposition the recalcitrant components by fungi (Tuomela, 2000). There is a stabilization of the compost and nitrification of the remaining mineralized nitrogen to nitrate. Bernal et al. (2017) explained that the longer this phase lasts the higher the level of OM humification; however, he claimed that the phase typically lasts for around 4 weeks and the optimum moisture content (MC) during this phase between 45-50 %.

2.2. Organic matter transformation during composting

Biomass composed of organic compounds containing phosphorus, hydrogen, sulfur, oxygen, nitrogen, and carbon. Water soluble organic compounds include amino acids, sugars, organic acids, soluble carbohydrates, and proteins and serve as readily available C source for microorganisms. In the case of insoluble OM, this plays a key role in integrating polymers, such as lignin, hemicellulose, cellulose (i.e., the main elements of plant material), aromatic compounds, chitin, and fats (Bernal et al., 2017).

Lignin has been identified as an insoluble polymer containing an aromatic phenolic group, with which in the presence of cellulose and hemicellulose establishes the rigid structure of lignocellulose biopolymer, which prevent microbial transformation of the biomass and safeguards cellulose (Tuomela, 2000). The lack of structural homogeneity in lignin makes it particularly recalcitrant to microbial decomposition; therefore, degradability of biomass decreases with the concentration of lignin or lignocellulose (Monlau et al., 2013). Lignin binds cellulose and hemicellulose together to provide strength and structure in plant cell walls and these components constitute a large proportion of the OM in the biomass employed for composting. However, a portion of the cellulosic fraction of lignocelluloses is not tightly bound in a close way to lignin and its availability for microbial decomposition is quite high.

2.2.1 Organic matter fractions

Lignocellulose concentration in agricultural plant residues, which are commonly employed as bulking agents, typically composes of 50–90 % of total OM. In the case of grass and maize, the lignocellulose concentration is low, while for straw stover, wood, and tea waste is a high concentration (Bernal et al., 2017). The lignocellulose structure differs between different biomass types, i.e., paper waste, corn stalks, plant stems, and timber, so biodegradability of biomasses that have similar lignocellulose concentration may differ based on the lignin structure. Regarding livestock manure, the proportion of lignocellulose content ranges between 30-80 %, the content is affected by animal’s diet and the manure’s straw content (e.g., for bedding), i.e., manure obtained from piglets tends to have a low lignocellulose concentration due to the low fiber feed, whereas the manure of buffaloes and dairy cows tends to have a high concentration because these livestock fed on the high-fiber diet (Hubbe et al., 2010). Consequently, the biodegradability of manure ranges from 0.38 - 0.93 % based on the animal's diet (Triolo et al., 2011). Moreover, manure from cattle tends to be associated with lower biodegradability compared to pig manure, principally due to its higher lignocellulose and lignin content.

At the start of the composting, the degradation of the soluble organic molecules takes place as a result of microorganism activity. But at later stage during composting, hydrolysis of cellulose and hemicellulose (complex organic molecules) occurs, which leads to the emergence of soluble organic molecules (Hue and Liu, 1995). The OM degradation in composting is related in an inverse way to the starting lignin concentration of the composted biomass (Eklind and Kirchmann, 2000a). In the case of organic waste originating from households, food industry, and so on, the lignin concentration is known to be low (Mirabella et al., 2014) and the OM therefore degrades readily. Contrastingly, straw and wood are associated with an elevated lignin concentration, and the composting process for these biomasses takes a comparatively long time. For all these waste products, cellulose enzyme activity plays a fundamental role in composting.

2.3. Chemical processes in composting

2.3.1 Carbon and nitrogen

The nature of the feed materials that being processed is a key determinant of the elemental composition of the material processed at a composting operation. However, N and C play a crucial role in the composting process. In the case of C, provides the primary energy source, while N plays a vital role in the growth of microbial populations. The composting process can be conceptualized as the bioconversion of biodegradable materials into carbon dioxide and water. As a result, one might expect that the C concentration in the compost material is reduced over the composting process, thereby reducing the C/N ratio. However, studies such as conducted by Bernal et al. (2017) indicated that, rather than increasing in the N concentration, this concentration actually decreases in the composting process. Although it is generally accepted in the literature that the C/N ratio declines as a result of composting, it is possible for ammonium nitrogen (NH_4^+-N) and nitrate nitrogen (NO_3^--N) concentrations to experience changes.

During the composting process, the mineralization of organic N takes place, and it is released as soluble NH_4^+ ions. The following competing pathways that removed NH_4^+ ions from the compost's liquid phase: firstly, immobilisation by microorganisms; secondly, adsorption of NH_4^+ (becoming exchangeable ions) to organic and mineral surfaces; thirdly, volatilization loss as NH_3; fourthly, nitrification to NO_3^-; and finally, runoff and leaching. Most of the N loss occurs through volatilization (Cáceres et al., 2018).

The volatility of ammonium is negligible prior to its dissociation to NH_3. The NH_4^+ conversion rate to NH_3 is controlled by factors such as pH, NH_4^+ concentration, and temperature (McCrory and Hobbs, 2001). Almost all NH_3 loss takes place during the initial days and weeks of the composting process, when temperature and pH are high (Hao and Benke, 2008).

Under aerobic conditions, the oxidization of NH_4^+ takes place to generate NO_2^-, after that NO_2^- oxidizing into NO_3^-. This is referred to as a two-stage nitrification process. As nitrification is taking place, N_2O may be generated as a by-product (Czepiel et al., 1996). According to Zeng et al. (2012), nitrogen assimilation may occur in the initial months of the composting process, but it is also possible to be carried out throughout the curing stage. Once nitrates are generated, denitrification may lead to the loss of nitrogen under anoxic circumstances (Wang et al., 2015). Additionally, Meng et al. (2017) noted that, during the composting of SS, nitrogen transformation reactions may occur at the same time, including ammonification, nitrification, denitrification, and NH_3 assimilation. However, the dominant reaction will change based on the environmental conditions during composting as well as to the substrate involved.

Elevated temperature and high NH_3 concentration have an inhibitory effect on the growth and activity of nitrifying bacterial species (Angelidaki and Ahring, 1994; Bustamante et al., 2008), thus the observed low NO_3^- content in compost particularly during thermophilic composting phase (Hao et al., 2004). However, when temperature declines and the maturing phase begins, nitrification process increases, lowering NH_4^+ and increase NO_3^- contents in the end product of composting, particularly when compared with the initial composted material (Hao et al., 2004; Tiquia et al., 2002).

2.3.2 Phosphorus

Based on the assumption that C loss takes place during composting, whereas phosphorus loss does not occur via volatilization, it is reasonable to expect that the proportion of phosphorus in the compost will increase as the composting process proceeds. Several studies have documented these effects (Cooperband and Middleton, 1996; Grebus et al., 1994; Mato et al., 1994), resulting in compost with 0.2-0.7 % P content (Canet and Pomares, 1995; Fricke and Vogtmann, 1994; He et al., 1995).

2.3.3 Sulfur

Cysteine and methionine, both of which are amino acids, are the major sources of sulfur in compost materials. In the context of effectively aerated conditions, sulphides are oxidized to sulfates. However, in the context of anaerobic conditions, H_2S and volatile organic sulfides, which would otherwise be oxidized and be absorbed via humic substances, are simply vaporized into the atmosphere (Stoffella and Kahn, 2001).

2.3.4 Heavy metals

The heavy metal (HM) content in compost is important for every commercial composting operator and plays an important role in determining the quality of compost. In fact, several countries are establishing, or have already established, compost quality standards that limit the acceptable contents for the metals mercury, zinc, nickel, cobalt, copper, lead, arsenic, molybdenum, chromium, and cadmium in compost (Bourque et al., 1994; Chwastowska and Skalmowski, 1997). Compost generated from animal manures may also contain excessive HM content, particularly Cu and Zn, especially when the compost has been derived from its solid fraction or pig slurry (Ko et al., 2008), but even from anaerobically digested slurry (Bustamante et al., 2013), due to the fact that these elements are present in pig slurry. Noteworthily, it is possible to supplement Cu and Zn in feeds for growth promoters, or health-related reasons (Nicholson et al., 1999). During composting trace elements availability is reduced, particularly when compared to the original waste material, due to both capacity of humic compounds in mature compost to chelate them and the formation of insoluble inorganic compounds (due to OM mineralization). However, HM minimization in the feedstocks used for composting (in the waste) is essential to obtain compost with acceptable agricultural quality for marketing.

In general, the nutrient contents of MSW compost are greater in source-separated collection when compared to organic fraction separated mechanically, and HM content is considerably reduced in the former (Huerta-Pujol et al., 2011). In the case of MSW compost, Huerta-Pujol et al. (2011) noted that source-separated collection of the organic fraction of MSW is the optimal approach, particularly when compared to post collection mechanical separation, due to HM may move from the non-compostable materials to the decomposable matrix after combining these types of materials.

2.4. Effect of using organic wastes on soil

2.4.1 Benefits

2.4.1.1 Effects on the physico-chemical properties of the soil

Frequent application of MSW compost consistently increased soil OM content and soil C/N ratio to levels above those associated with unamended soil (Crecchio et al., 2004; García-Gil et al., 2004; Montemurro et al., 2006; Walter et al., 2006). In the study conducted by Soumare et al. (2003a), the researchers reported that MSW compost displayed a high water holding capacity due to its OM content, which had improved the water holding capacity of the soil. Furthermore, Hernando et al. (1989) found that application rates of 30 and 60 Mg ha^{-1} of MSW compost contributed to a rise in the aggregate stability of soil. This was attributed to the formation of cationic bridges, which led to an improvement in the structure of the soil. Another researchers also noted that the addition of mature MSW compost, in this case to a silt loam soil type, contributed to an increase in aggregate stability (Annabi et al., 2007). Celik et al. (2004) reported that the use of MSW compost led to improving soil properties in terms of soil porosity, soil aggregation, water retention, OM content, sorption capacity, and macronutrient bioavailability. Another study noted that, when compared to unamended soil, amended soil was associated with increased in high numbers of large microaggregate (2,000-8,000 μm) and microaggregate (50-250 μm) (Nicolás et al., 2014). In the 3-year field study conducted by Weber et al. (2007), the researchers examined the effect of two different MSW composts, applied at amendment rates of 18-tonnes, 36-tonnes, and 72-tonnes dry matter (DM) per hectare, on the properties of low-fertility sandy soil. The physico-chemical properties of the soil, including organic carbon, cation exchange capacity (CEC), field water capacity, total porosity, humic acid content, plant available water, and nutrients (P, K, and Mg), were enhanced considerably due to application both MSW composts. Additionally, Calleja-Cervantes et al. (2015) reported consistent improvement in soil quality as a consequence of the long-term use (specifically, over a period of 13 years) of composted organic wastes viz. (1) pelletized organic compost, derived from animal, biosolid, and plant residues, (2) MSW compost and (3) compost derived from sheep manure. Amendment with all the compost substantially increased OM content (35 %), coupled with improvements in EC, as well as nitrogen, potassium, and phosphorous contents. Edmeades (2003), Maltas et al. (2013), and Six et al. (2002), also drew attention to the positive effect in terms of increasing the supply of nutrients, improving the water holding capacity, and reducing soil erosion and nutrient leaching, which were attributed to the higher organic carbon content in soil improves its properties, crop yields, and fertility. A commonly used approach for biological oxidation and stabilization of organic wastes, which enhances soil properties such as fertility, is known as vermicomposting (Domínguez and Gómez-Brandón, 2013). In vermicomposting, elevated surface area, porosity, and water holding capacity contributed to an increase in the retention of plant-available nutrients (Chaudhuri et al., 2000). Furthermore, as noted by Wang et al. (2008), the use of SS on agricultural land, which is an enriching source of OM and valuable plant nutrients such as nitrogen and phosphorous has a positive effect on various soil properties, including bulk density, aggregation, porosity, water holding capacity, and soil structure. Consequently, making it a good soil ameliorating agent; however, over application of SS with high HM, as well as other toxic compounds may disturb soil properties (Singh et al., 2012). In an experiment to analysis of the strengths and weaknesses associated with the use of organic waste as soil amendments in agricultural, Alvarenga et al. (2015) characterized several types of organic wastes including mixed MSW compost, agricultural waste compost, municipal sewage sludge, compost generated from agricultural waste and SS, agro-industrial waste, pig slurry digestate, paper mill waste, and municipal slaughterhouse sludge. The study reported that the high OM and nutrient content (N, P and K) in these organic waste amendments, particularly nitrogen, phosphorous, and potassium, could be beneficial for soil properties. However, certain types of organic waste like pig slurry digestate, paper mill wastes and some compost samples were identified as contributors to increased soil salinity due to higher EC; elevated concentration of $N–NH_4^+$ may result in NH_3 emissions and pathogenic microbes in case of SS; exceeding HM concentration except in compost generated from agricultural wastes and SS and groundwater pollution due to leaching of $N–NO_3^-$ from soil owing to its higher concentration pig slurry digestate.

2.4.1.2 Effects on the biological properties of the soil

Results from a field experiment used organic wastes constituted of composted plant materials (poultry litter, hay, cotton-gin trash, cattle manure, mixed yard waste) and animal manure as soil amendments, showed significant correlation between biological communities and the chemical properties of the soil (Bulluck et al., 2002). In the later study of Bailey and Lazarovits (2003), the researchers found that nitrogen-rich organic soil amendments, manures, and composts can supplement the soil with beneficial microorganisms, while suppressing pathogenic microorganisms through the release of allelochemicals. In a long-term field trial, Poulsen et al. (2013) assessed the fertilizing effects of various types of agricultural and urban waste such as human urine, composted organic municipal waste, deep litter, cattle slurry, NPK fertilizer, cattle manure, and biosolids on soil microbial activity. The researchers found that soil organic matter C content was significantly affected, along with basal CO_2 respiration and soil microbial biomass C. Specifically, elevated SMB carbon values were observed, and an abundant heterotrophic bacterial population was identified for soils fertilized with organic wastes. In 50-year-long field study conducted by Blanchet et al. (2016) involving organic and inorganic fertilizers in a Swiss traditional farming system, organic fertilizers such as crop residues and farmyard manure were found to improve soil microbial population, as reflected by the microbial biomass and phospholipid-derived fatty acid contents.

Substantial increases in soil enzyme activity have been observed for phosphomonoesterase, sucrase, phosphodiesterase, arylsulfatase, catalase, and urease after the use of vermicomposting (Albiach et al., 2000; Zuo et al., 2018). Calleja-Cervantes et al. (2015) reported a considerable increase in enzyme activity in a vineyard soil amended with various types of organic waste. The largest and most substantial increase in soil enzyme activity was associated with soil that was amended with compost generated from MSW, in which a 115 % increase was observed in phosphatase, along with 165 % and 389 % increases in β-glucosidase and protease activity, respectively, as compared to control. Pelletized organic compost increased urease and protease activity by 149 % and 275 %, respectively. Nevertheless, use of a mineral fertilizer (specifically, NPK) contributed to an overall reduction in enzyme activity, amounting to 26 % for FDA hydrolysis and 67% for β-glucosidase activity. Following the application of 75 Mg ha^{-1} MSW compost, urease, alkaline phosphodiesterase, arylsulphatase, deaminase, phosphomonoesterase, and protease enzyme activity increased, and increases above baseline levels were retained for 3 months following the initial application (Perucci, 1990). Phosphatase activity stay constant in the soil after reaching its maximum value, and so it may be concluded that MSW compost stimulates the transformation of organic phosphorous to its inorganic and available form (Perucci, 1990). For soils treated with 2.5-40 Mg ha^{-1}, acid phosphatase and urease activities increased, and these were proportional to the MSW compost application rate (Bhattacharyya et al., 2003). The activities of some enzymes decreased, when MSW compost was applied. As a case in point, protease activity decreased with the application of 24 Mg ha^{-1} MSW compost, which is likely a reflection of the product's minimal protein content (Crecchio et al., 2004). Additionally, in the study conducted by García-Gil et al. (2000) found that addition of MSW compost at 20 and 80 Mg ha^{-1} inhibited urease and protease activity. In both cases, the reduction was attributed to the potential toxic impacts of trace elements in the compost (Crecchio et al., 2004; García-Gil et al., 2000). In the more recent study undertaken by Bastida et al. (2017), the researchers analyzed the impact of organic amendments (specifically, sludge and compost) on soil microbial communities, as well as microbially mediated processes in semiarid or drought circumstances. Activity of β-glucosidase and urease enzymes, and soil microbial biomass in sludge and compost-amended soils were higher under drought circumstances than well-watered soils as compared to unamended soil following a 45-day period of application. Roig et al. (2012) noted that in long-term field study used anaerobically digested biosolids for 16 years were improved calcareous soil properties in a way that was proportional to the dose applied and/or their frequency, and the researchers also founded that properties such as microbial and enzyme activity, carbon and nitrogen mineralization, and OM and nitrogen content rose due to the biosolid amendments. The maximum dose of 40 Mg ha^{-1} year^{-1} was identified for the use of anaerobically digested biosolids as agricultural fertilizer, after which no further soil properties improvements were observed.

Field application rate of 2.5, 10, 20, and 40 Mg ha^{-1} MSW compost, resulted in an increase in soil microbial biomass carbon and soil respiration compared with the control treatment (Bhattacharyya et al., 2003). In a long-term field study undertaken by García-Gil et al. (2000), it was noted that multiple applications of MSW compost, at rates of 20 and 80 Mg ha^{-1}, contributed to a rise in microbial biomass carbon, and this increase was retained for a period of 8 years following initial application of the compost. In Pascual et al. (1999) earlier study, the researchers also noted that the soil basal respiration rate, which is a measure of microbial activity in a soil, increased for 8 years with the application of MSW compost compared to a control after compost application.

2.4.2 Risks

2.4.2.1 Potential toxic elements (heavy metals)

Long-term application of organic waste through anaerobic digestate, SS or composting/vermicomposting may results in accumulation of HM in top soil, and from soil, they can be transferred to varying trophic levels in the food chain, thereby resulting in a range of human and environmental health issues (López-Mosquera et al., 2000; Mohee and Soobhany, 2014; Wang et al., 2003).

When HM exceed the threshold permissible limits, this can have negative effect on environment and all types of organisms (Cebula et al., 1995). Additionally, they remain unaffected during the decomposition process of organic waste. According to Das et al. (2002), metal concentration increases with the volume reduction of waste during bio-degradation of organic waste. Use of MSW compost for 3 successive years was associated with positive increases in Pb, Ni, Cu, and Zn in 0-25-cm layer in the amended soil when compared to the control (Madrid et al., 2007). Additionally, the researchers found that integration of MSW compost increased the HM concentration in the soil, specifically in terms of Pb, Cu, Zn, and Cd contents in the topsoil layer Furthermore, Zn concentration was the highest, followed by Cu, Pb, Cr, Ni, and Cd (Madrid et al., 2007). In similar study undertaken by Singh and Agrawal (2010), on different organic amendments of SS showed that amendment rates exceeded 4.5 kg m^{-2} increased the rice production, but led to increases Ni and Cd concentrations above the Indian safe limits (i.e. 1.5 mg kg^{-1}). Immobilisation of metal through the formation of stable complexes with COOH or OH groups on the solid surfaces of organic polymers could be contributed by the application of organic amendments in soil (Madrid et al., 2007). OM augmentation can fix the HM in soils by facilitating the creation of organometallic complexes. In turn, this reduces HM mobility and phytotoxicity (Udom et al., 2004). Nevertheless, immature compost and fresh manure, which are characterized by high levels of soluble OM content, may promote the metal mobility (Madrid et al., 2007). In the case of the organic amendments, these have different impacts on metal availability (Walker et al., 2004). These effects are mediated by factors such as the metal in question, the soil, and the properties of the OM (particularly the level of humification).

2.4.2.2 Emerging contaminants

A range of harmful micro-pollutants in anaerobic digestate such as pharmaceuticals (Diazepam, Iopromide and Ibuprofen), detergents (like Triclosan) and pesticides are bio-transformable during the process; while other compounds, including nonylphenols, oestrogens, alkyl benzene sulphonates, and synthetic musks are resistant for degradation (Carballa et al., 2007; Hospido et al., 2010; Sheets et al., 2015). Relatively few studies have confirmed the occurrence of organic pollutants in MSW compost (Logan et al., 1999). Compounds of this kind are highly chlorinated and it is persistent in the environment (Muir and Howard, 2006). One of the most effective ways in which to lower the levels of these compounds in waste is by composting via volatilization, photolysis, and biodegradation (Epstein, 1997). Logan et al. (1999), in their survey of commercial and municipal waste, noted that, by weight, less than 0.5 % of waste is hazardous and phthalate esters are most abundant xenobiotic in MSW compost. Furthermore, in MSW compost, the researchers revealed that phthalate esters are likely to be the xenobiotic characterized by the highest level of abundance. Additionally, the researchers noted that the concentrations of dioxin/furans and polychlorinated biphenyls (PCB) were elevated in mixed MSW compost when compared to source-separated MSW compost (Logan et al., 1999). The environmental risk linked to PCB in source-separated MSW compost was identified as low (Epstein, 1997; Logan et al., 1999). Furthermore, a field study conducted in Germany found that, in MSW compost, PCDD/Fs levels were greater than those detected in adjacent soils, and exceeded Germany's regulations (Harrad et al., 1991).

2.5. Composting feedstocks

Six types of source materials commonly composted (TMECC, 2002):

1-*Food processing residuals*: compostable materials remaining after fruit, vegetables, grains, nuts, and meat are processed for consumption.

2-*Manure* and *agricultural by-products*: generated at racetracks, feedlot and other animal feeding operations, farms, nurseries, and greenhouses. Compost produced from residuals generated from farms, nurseries, and greenhouses can be readily recycled into those operations. However, great quantities of manure generated at racetracks, feedlots, and swine and poultry confinement facilities can pose a significant challenge to facility operators.

3-*Forestry* and *forest product residuals*: includes bark and sawdust, and fibre fines residue and biosolids generated by the papermaking process. Bark and sawdust can be used in the composting industry as a carbon source with other feedstock material or as a bulking material to increase porosity of the feedstock mix.

4-*Biosolids*, or *SS*: the solid material generated by the biological treatment of sewage at a wastewater treatment plant. It is possible to apply SS to soil as a solid amendment or a liquid. At present, a matter of growing concern for many researchers is the possibility that pathogenic microorganisms may exist in SS, which has prompted them for using composting as a process that can (i) eliminate human and plant pathogens, and (ii) generate valuable agricultural products. Moisture content is the critical issue currently facing the composting of SS, meaning that, prior to composting, drying or the use of a dewatering agent (e.g., lime) should take place to reduce the humidity to less than 60 %. In most cases, it is not possible for SS to be employed as the only raw material in composting, so straw litter or yard waste are commonly used as bulking agents (Comesaña et al., 2018). A final noteworthy point is that, when the concentration of specific compounds in SS (e.g., HM such as Cr, Cd, Pb, Zn, Cu, Ni, and Hg) passes a certain threshold value, this can create a risk for human health. Concentrations of HM in SS may differ considerably, depending on the origins of the SS.

5-*Leaves, brush* and *yard trimmings (yard waste)*: yard waste refers to the grass clippings, wood waste, and leaf fall that are diverted from the MSW stream. Therefore, by definition, yard waste is heterogenous, and the characteristics of yard waste depend on factors such as the weather, the geographical location, and the waste collection policy. The characteristics of the yard waste affect the collection treatment, the processing equipment and the properties of the final product. The humidity of yard waste ranges between 30-75 %, and for leaves, grass, and wood fraction, the humidity rate has been identified as 19-28 %, 54-64 %, and 17-18 %, respectively (Reyes-Torres et al., 2018). The C/N ratio of yard waste is typically high (90-125) due to the substantial levels of lignin in woody waste. To lower the initial C/N ratio to a level that is appropriate for composting (30-35), yard waste is combined with animal manure, poultry manure, SS, or any other types of waste that can provide supplementary nitrogen. Recovery of yard waste through composting is becoming an increasingly prominent approach of recycling solid organic waste. Turned windrows and static piles are common systems used for composting yard waste. Yard waste compost is convenient as a substrate in container media and unlike composts generated from other wastes such as SS and MSW by seems to be cleaner compost (Hubbe, 2014).

6-*Source separated organic waste* (SSOW): consists of the compostable and composting compatible fraction of municipal solid waste, accumulated and presorted by the generator, and collected separately from household hazardous material and sometimes non compostable material.

7-*Mixed municipal solid waste* (MMSW): includes various discards from residential, commercial, and institutional sources that are commonly taken to incinerators or landfills. The largest components of MMSW are typically paper and paper products, leaves, brush and yard trimmings, wood, food scraps, glass, plastics, and metals. The composition of MMSW varies depending on the characteristics of the waste generators in the service area, but usually from about 50-65 % is compostable when recovered by separation at a central facility. MMSW will contain relatively fewer recyclables and a relatively higher fraction of compostable material when an aggressive source-separated recycling collection program operates in conjunction with MMSW collection.

2.6. Factors affecting decomposition of waste and composting process

It is necessary to fully understand the process of composting and the factors that influence it in order to generate an optimised and high-quality product. The success of the decomposition of waste and composting process is affected by factors, such as the turning frequency, temperature, C/N ratio, MC, electrical conductivity (EC), aeration, pH, particle size, and porosity.

2.6.1 Turning frequency

In order to achieve optimum aeration during the composting process, the most frequently used procedure is that turning of the compost organic material. The mechanical turning of the composting material releases gases and provides a constant supply of degradable materials for microbial utilization (Parkinson et al., 2004). However, Awasthi et al. (2014) warned against excess aeration or turning claiming that it can result in the loss of necessary components for composting despite aeration at the early stages of OM decomposition being necessary to reduce the duration of waste stabilisation. Therefore, it is vital that the turning regime is optimised to ensure retention of the relevant nutrients or to achieve a faster rate of pathogen reduction (Kalamdhad and Kazmi, 2009).

Some physico-chemical variables can be used as compost maturity indicators, as detailed by Getahun et al. (2012); Ogunwande et al. (2008) reported a definite link between the frequency of turning and these potential indicators. Getahun et al. (2012) claimed that dry matter, pH, temperature, total carbon (TC), C/N ratio, MC, and total nitrogen (TN) are all effected by turning frequency of the compost pile. Compaction of the compost pile is also a concern as Bernal et al. (2017) explained, it can result in low air distribution and poor porosity in the compost batch. To counter this, effective mixing of the composting materials is required.

There are several turning regimes described in the literature, that vary from daily to weekly turning of the compost material. For swine manure, Li et al. (2015) reported success using a daily turning regime as this ensured adequate oxygen supply to maintain aerobic conditions in the compost feedstock. However, Ros et al. (2006b) utilised a 4-5 day turning regime for 3 months on their composting trial using pig slurry. A weekly turning regime was adopted by Mohee et al. (2015) in their composting trial using MSW. A co-composting trial using solid municipal waste and poultry manure was improved by adopting a daily mixing regime of 30-minute duration (Petric et al., 2015). Significant increase of aerobic bacterial population was reported for both three-day and weekly turning regimes, whilst daily turning made no significant increase to the bacteria population in the trial by Awasthi et al. (2014).

2.6.2 Temperature

One of the major monitoring parameters during the composting process is temperature (Gajalakshmi and Abbasi, 2008). Temperature within the composting batch is a direct indication of the organic material degradation rate as well as showing the point at which optimal conditions have been met for microbial biodegradation (Bernal et al., 2017). Simultaneously, temperature is one of the process functions as presented by Turan (2008). Kulikowska (2016) explained that the exothermic process of composting is entirely reliant on the biodegradability of the composting material and the initial temperature. Many authors detailed the rise in temperature that occurs as a result of rapid microbial biodegradation of the OM (Raut et al., 2008), but some suggested that temperature increases are actually linked to a reduction in the efficiency of the composting process (Ryckeboer et al., 2003). Microbial competition is also temperature dependent with some microbial populations gaining advantage over others at certain temperatures. Chen et al. (2015) suspected that this is a result of the influence of temperature over the compost’s physico-chemical characteristics and resultant substrate bioavailability to the composting microorganisms.

Due to the levels of pathogenic organisms in OM, including intestinal parasites and faecal coliforms, it is a significant human and animal health risk. Therefore, temperatures within the compost pile exceeding 55 ^{o}C are favoured as this ensured the elimination of pathogens and parasites and effectively sanitises the compost (Ravindran and Sekaran, 2010). Moreover, Zhang and Sun (2014) explained that by maintaining thermophilic phase for a minimum of three days will ensure complete removal of weed seeds and total sanitation, although Chan et al. (2016) suggested that the lasting of thermophilic phase for a week would have a complete elimination of pathogens. Others agree with this, especially Čempírková and Šoch (2007) who suggested that coliform bacteria, particularly *Escherichia coli* and other similar pathogens varies, are more difficult to eliminate so time and temperatures need to be extended. The optimal thermophilic activity within a composting system is maintained between 52-60 ^{o}C (Vuorinen and Saharinen, 1997). Paradelo et al. (2013) suggested that the stabilisation and decomposition of the initial organic material is determined by the thermophilic phase. Some disagree, with Rich and Bharti (2015) claimed that the optimal temperature for composting is between 40-65 ^{o}C and Miyatake and Iwabuchi (2005) stated that the highest temperature at which they observed any thermophilic activity was at 54 ^{o}C. Sundberg et al. (2004) explained that the lag phase that denotes the transition between mesophilic to thermophilic activity is associated with the way in which low pH at the initial phase of composting inhibits thermophilic activity.

During the thermophilic phase, the growth of the nitrifying bacteria can be restricted by high temperatures and ammonia levels (Huang et al., 2004). Imbeah (1998) and Cáceres et al. (2018) cautioned against elevating the temperatures too much as they claim that thermophilic microbial populations can be destroyed above 71 ^{o}C. Imbeah (1998) suggested maintaining a compost temperature below 65 ^{o}C to prevent microbial destruction and cease of the composting process, whilst Sudharsan Varma and Kalamdhad (2015) claimed that temperatures above 65 ^{o}C are also likely to destroy actinomycetes and fungi that also aid in the degradation of the OM during the thermophilic phase. Temperatures exceeding this would be optimal only for spore-forming bacteria to be developed. There are other concerns with excessive heat during the composting process, including the potential for fire risk (Chen et al., 2011).

2.6.3 C/N ratio

During composting microbes acquire nutrients, including K, N and P and energy for metabolism, from the degradation of organic compounds, in order to sustain their populations (Chen et al., 2011). K, P, N and C are the crucial nutrients required by the microorganisms take part in composting (Bernal et al., 2017). However, with C and N being the most crucial as N is utilised to create the cell structures and C is used as an energy source (Chen et al., 2011; Iqbal et al., 2015). There is a notable reduction in microbial growth and decomposition of C when the amount of N is limited. Where excess levels of N are present beyond the requirement of the microbes, the excess N volatised as ammonia gas (Igoni et al., 2008). Therefore, the C/N ratio is one of the indicators of OM decomposition efficiency; as C can also be lost during biodegradation as CO_2 (Lazcano et al., 2008). According to Yang et al. (2015), during the composting process, the C/N ratio decrease due to the lower rate of organic N mineralisation compared to that of organic C.

The optimal C/N ratio according to Igoni et al. (2008) is 25-35 as microbes need 30 units of C for each unit of N. High C/N ratios than optimal make the composting process very slow as there is an excess amount of decomposable C whilst a low C/N ratio presents an excess amount of N per degradable C and thus surplus inorganic N is either lost by leaching or ammonia volatilization from the compost batch. Therefore, low C/N ratios can be corrected with the addition of a bulking agent that provides a source of organic C (Cayuela et al., 2004; Alfano et al., 2008).

It is necessary for good composting to adjust the raw materials by creating an optimal C/N ratio of 25-35:1, although C/N ratio of 20-40:1 is optimal for composting (Bernal et al., 2017). Some authors, including Petric et al. (2015) suggested even higher initial C/N ratios values of up to 50:1 can be effective for composting.

2.6.4 Moisture content

Another crucial indicator for the composting process is MC. It controls temperature of the process, microbial activity, free air space, and rate of oxygen uptake (Petric et al., 2012). However, the optimal MC is dependent on the type or form of waste (Gajalakshmi and Abbasi, 2008). These authors suggested a MC of between 50-60 % for feedstock. Meanwhile, Razmjoo et al. (2015) claimed that biological activity is optimised during composting when the MC is between 40-70 % of the weight of the compost. The rate of gas diffusion reduces as the MC increases; this results in a lowering of the oxygen uptake rate, which becomes incapable of meeting the microbial metabolic requirements. Indeed, in these situations, the composting process can become anaerobic as a result of the activity restrictions (Mohammad et al., 2012). Some authors have indicated an inverse link between temperature and MC; Sudharsan Varma and Kalamdhad (2015) reported a decrease in MC with a correlated increase in temperature. A low temperature has also been reported when MC is high (Turan, 2008). The MC during composting is vital since the aqueous medium makes the soluble nutrients chemically and physically accessible to microbial populations (Guo et al., 2012). These authors also suggested that the biological features and specific physico-chemical properties of the compostable material dictates the optimum moisture content.

As stated by Sudharsan Varma and Kalamdhad (2015), loss of MC during the composting process is a strong indication of decomposition rate. Where MC is low, the composting process can be hindered due to early dehydration and halting of the biological activities (Makan et al., 2013). Makan et al. (2013) indicated that excess MC during the composting process can generate anaerobic conditions from the creation of water logs which will prevent effective composting activity.

For poultry manure and wheat straw composting, the optimal initial MC is around 70 % (Petric and Selimbašić, 2008). (Ros et al., 2006b) during composting pig slurry, maintained MC between 60-70 %. However, Abdullah and Chin (2010) suggested that MC of 60 % or above can impede oxygen movement due to the reduced pore spaces there by creating anaerobic conditions.

2.6.5 Electrical conductivity

The EC indicates the soluble salt content in a compost materials (Lazcano et al., 2008). The EC of mature compost needs to be at a very low level to support plant growth (Bernal et al., 2017). The literature report varying limits for EC. For example, Awasthi et al. (2014), Zhang and Sun (2016a, 2016b), and Chowdhury et al. (2015) suggested an optimal EC value of 4 mS cm^{-1} for the compost to be applied to the soil, whilst Mohee et al. (2015) claimed that the EC limit for a safe application of the compost in agriculture should be in the range of <3.5 dS m^{-1}. Additional authors have reported varying EC levels for compost to be used as organic soil amendment that range from an maximum limit 3 mS cm^{-1} (Juárez et al., 2015), whilst values lower than 2.5 mS cm^{-1} were reported by Mulec et al. (2016) as the limit value for agricultural applications. For composts, a value of 2 mS cm^{-1} or less are considered as ideal (Turan, 2008).

The literature suggested that negative impacts on plant growth will result from high EC values within compost material (Yang et al., 2015; Awasthi et al., 2014). Compost with a higher EC value can still be used for growing crops but they need to be well mixed with other substances, such as soil to reduce the EC value prior to use (Gao et al., 2010a). It is reported that there are negative impacts on seed germination or plant growth, particularly at the seedling phase, of compost with higher salinity levels (Bernal et al., 2017). During the composting process a natural mineral, known as zeolite, can be used to reduce the salinity (Chan et al., 2016). The addition of zeolite can encourage free ion exchange across its surface whilst also aiding in the adsorption of ions, thereby lowering the EC (Chan et al., 2016).

2.6.6 Aeration

Aeration is an important key factor influencing composting process (Meena et al., 2019). Principally, composting is a biochemical aerobic process during which oxygen is used for mineralized easily degradable material and H_2O, CO_2, and NH_3 are released (Awasthi et al., 2014). Therefore, oxygen levels strongly influence the efficacy of the composting process due to the oxygen requirements of the microorganisms population dynamics (Nakasaki et al., 2009). During the composting process, microbial activity and the quality of the compost, influenced by the aeration rate (Gao et al., 2010a). Aeration can also be used to maintain the temperature of the compost during thermophilic decomposition of the organic materials (Raut et al., 2008).

During the composting process, aeration can generate enough oxygen for organic material oxidization whilst also allowing for excess moisture evaporates from the substrate (Petric and Selimbašić, 2008). Compost stability is also affected by aeration (Guo et al., 2012) as this has major impacts on the physico-chemical characteristics of the compost, including temperature, OM content, MC and pH (Rich and Bharti, 2015). Where there is a reduction of optimal aeration, anaerobic conditions can be promoted, whilst too much aeration can result in a lowering of temperature, which prevents the thermophilic environment that essential for decomposing the organic materials (Gao et al., 2010a). Another impact of increased aeration is an associated increase in the evaporation rate, resulting in the compost drying at a quicker rate and thereby lowering the compost temperature (Sundberg and Jönsson, 2008). Bernal et al. (2017) suggested an optimal oxygen concentration rate of between 15-20 %, whilst Gajalakshmi and Abbasi (2008) claimed that adequate oxygen availability should be maintained at a temperature below 60-65 °C.

There are a numbers of different turning regimes reported in the literature to ensure optimum aeration of the compost material. To keep aerobic environment and ensure adequate oxygen supply during the swine manure composting, Li et al. (2015) adopted a daily turning regime. However, Białobrzewski et al. (2015) reported an oxidation of carbon to CO_2 during the release of heat in forced aeration situations. They also reported nitrogen uptake, along with other biogenic elements for use as nutrients, in microbial growth, metabolic reactions and reproduction. Perforated pipes situated under the compost windrow can be used for forced aeration during windrow composting, to activate air flow throughout the organic material, thereby ensuring an adequate oxygen supply for the microorganisms (Tiquia and Tam, 2000). Other authors, such as Lazcano et al. (2008), also reported the use of a blower (forced aeration) to supply adequate oxygen to microbes during composting. They also reported that temperature impacts the operating of the air-blower. Some composting trials have been successful using natural aeration. For example, Oudart et al. (2015) reported that used natural aeration during composting of animal waste in static windrow have proven to be more cost-effective and less energy intensive operation than use forced aeration. Gaseous exchange can be improved using bulking agents as stated by Hubbe et al. (2010).

There is a definite correlation between pH levels and aeration, with the increased concentrations of oxygen causing a rapid acid decomposition within the compost, thereby resulting in a reduction in organic acid concentrations and a correlated increase in pH level (Sundberg and Jönsson, 2008). According to the literature, there is a difference in the optimal range of aeration rate as a result of variations in the composition of the composting material (Gao et al., 2010a).

2.6.7 pH

The pH level is an essential indicator during composting process (Gajalakshmi and Abbasi, 2008). It influences microbial dynamics during composting (Bernal et al., 2017). Kalemelawa et al. (2012) documented that pH level is higher during aerobic composting, compared with the anaerobic process, which is potentially a result of higher release of potassium. An optimal range for composting appears to be between a pH of 7-8 (Gajalakshmi and Abbasi, 2008). Variations in pH being during composting caused by biological activity (Turan, 2008). Some trials have shown that there is a decrease in microbial activity where pH levels fall below 5 (Sundberg et al., 2004). Paradelo et al. (2013) explained that when composting implemented at an industrial scale, the transition to the thermophilic phase from initial mesophilic can be negatively influenced by low pH levels. Hachicha et al. (2009) suggested that the degradation of proteins can lead to an accrual of ammonia which will cause an increase in pH. This author also explained that, with an increasingly alkaline pH within the composting material, pH sensitive microorganisms can be negatively affected, thereby reducing the sanitisation effect. Zhang and Sun (2016a, 2016b) indicated an optimal pH range of between 7.5-8.5 promotes microorganism degradation of organic substances. Other authors, like Rich and Bharti

(2015), have reported a wider range of between 6.7-9.0, claiming this range supports optimal microbial activity, whilst other authors have reported a range of 5.5-8.0 (de Bertoldi et al., 1983). Interestingly, Rich and Bharti (2015) founded that the stability of the compost was not influenced by pH. For mature compost, the pH range should be between 6.0-8.5 (Hachicha et al., 2009); however, this is contradicted by the trial completed by Juárez et al. (2015), who reported a range of 8.0-8.5. Sundberg et al. (2004) reported that there is potentially a correlation between pH and temperature during the composting process. This is possibly related to different microbial populations being optimal at various pH and temperature combinations. However, the pH factor is highly relevant when considering the control of nitrogen loss through ammonia volatilisation which occurs at its highest rate at pH 7.5 or above.

2.6.8 Particle size

According to Bernal et al. (2009), the particle size of the organic material influences microbial growth due to availability of surface area and can also help maintain the required porosity for aeration. Indeed, in composts with larger particle size, there is an obvious lower surface area to mass ratio and a higher porosity. In these large particle size composts, there is often incomplete decomposition of the OM as the microbes are unable to access the interior sections of the larger particles. However, very small particle sizes are also problematic for composting, as they may result in compaction and a reduction in porosity. A particle size of between 2.5 and 7.2 cm is recommended by Tchobanoglous et al. (1993), whilst Bueno et al. (2008) founded that 1 cm sized cellulosic materials achieved optimal results.

2.6.9 Porosity

Air distribution within the compost is significantly impacted by its porosity. Meena et al. (2019) reported that lower temperatures resulted from any porosity level higher than 50 % due to an exceedance of energy loss compared to heat produced. Odour generation and anaerobic conditions are prevalent where there are very low porosity levels. According to Meena et al. (2019), there should be a range of 35-50 % air-filled pore space in composting piles.

2.7. Composting processes used industrially

2.7.1 Windrow process

Windrow composting is the most common practical method of composting (Hubbe et al., 2010). In this process, the waste is laid out in parallel rows that are approximately 3-4 m wide and 2-3 m high. The length is dictated by the layout of the site and movement of materials. The shape of the windrow is naturally trapezoidal shape, with angles of repose based on the nature of utilised material.

Windrows required aeration by passive methods - convection, wind, and diffusion. Windrows are regularly turned to supplement passive aeration. Turning takes on the action of thoroughly agitation of the materials. This is often done with special windrow turning machines or bucket loaders, with the latter achieving the required turning by repeatedly lifting and dropping the materials while rebuilding the windrow. Turning with a loader can be a time-consuming, whereas the special windrow turning machines can effectively turn several thousand m^3 per hour (Rynk and Richard, 2001).

There are a number of benefits associated with turning; these include the blending and mixing of the feedstock, the homogenization of the windrow materials, the release of trapped heat and gas, the exchange of material from the cooler oxygenated outer layer of the windrow with material from the warmer O_2-poor areas near the centre, and the improved distribution of microorganisms, nutrients and water throughout the windrow (Gajalakshmi and Abbasi, 2008).

Mostly the windrow is turned approximately up to two times per week while the temperature increases to, and stand at, around 55 °C. Complete composting of the organic material is usually achieved within 3-4 weeks and the resultant compost is then left without turning for a further 3-4 weeks for curing. During this phase, residual OM are further degraded by actinomycetes and fungi (Diaz et al., 2007).

The turned windrow composting method is a flexible and simple process that can be used with a large variety of feedstocks and with a varying range of management strategies, operation scales, equipment and financial resources. It is a proven, effective system of composting, however, there is a need for a large area to position the windrows and for the turning activities. Therefore, it is unusual for such activities to be enclosed within a building due to the cost of implication. Furthermore, it is difficult to maintain aerobic conditions within the windrows resulting in anaerobic conditions generating internally with an accompanying odour. As such, the location needs to be considered prior to building the windrows as odours can be released during turning (Turovskiy and Mathai, 2005).

2.7.2 Aerated static windrow

The ASW procedure depends on fans to ventilate and aerate the composting materials. The windrow is built over an arrangement of aeration vents which supply and distribute air through the composting materials. Forced aeration supply oxygen and reduce the temperature inside the windrow, whilst also remove carbon dioxide, water vapour, and other decomposition products. The forced air allows for an increased flexibility within the operation and enable the operators to more precise control temperature and oxygen within the windrow. Krogmann et al. (2010) explained that the windrows with forced air ventilation are no longer prone to the development of anaerobic conditions and therefore the risk of odour is reduced.

There are a number of necessary components to an ASW, which include a control system and an air distribution network, the composting material, an outer layer of stable material and a coarse porous base layer. The air distribution network situated below the windrow. A number of air distribution techniques have been utilised. One technique utilises air channels cut out of a concrete pad with the slotted aeration pipe located within the air channel. This enables the distribution of air to the organic material above. Coarse materials, such as wood chips, are often used to cover the aeration pipe and embedded channel and these can prevent any clogging of the air holes. The organic material is then stacked into the usual windrow shape on top of the air distribution and porous base (Diaz et al., 2007).

Air is provided to the ASW by means of fans and associated piping and control devices-the aeration delivery and control system. Air can be supplied by negative or positive pressure, intermittently or continuously. ASW aeration systems are often managed by timer or temperature that is modified in accordance with temperature levels (Rynk and Richard, 2001).

These ASWs can be between 2-4 m in height, with the major height limitation being a result of the equipment used to build the windrow and the density of the organic materials, which can cause compaction. ASWs lengths commonly range from 40 to 80 m (Diaz et al., 2007).

The ASW composting approach is well proven and well-practiced and has the benefit of being technically simple. It also needs a lower area than the typical windrow composting technique and achieves aeration with more certainty. The only failing of the system is that it is static, which can cause compaction and a short circuiting of air, which can result in uneven decomposition within a batch of compost (Gajalakshmi and Abbasi, 2008).

2.7.3 Rotating drum reactor

Rotating drum composting reactors have been applied, mostly for large-scale MSW facilities and, at smaller scale, for backyard composting (Hubbe et al., 2010).

Drums are mounted horizontally, usually at a slight incline. These drums either continuously or intermittently rotate to tumbling the organic material inside. This tumbling action can mix and agitate the organic material whilst also aerating it. The organic material is loaded into one end of the drum with the resulting compost being emptied from the opposite end. Different loading and unloading machines are applied, based on the specific system. Loading with conveyors and unloading by gravity are popular (Rynk and Richard, 2001). The key function of the rotation of this drum is the release of gaseous decomposition products and heat and the addition of oxygen to the organic material.

One of the benefits of the rotating drum composting reactor is the very short retention time, which is typically 3-5 days. However, the drums are more often used as a primary composting stage with the organic material then being taken and finished in either aerated piles, windrows or a secondary composting system. The benefit of the drum is that there is an even and rapid composting of the feedstock in a controlled high temperature environment. Furthermore, drums are very successful at homogenizing heterogeneous mixtures like MSW (Krogmann et al., 2010).

2.7.4 Two-stage composting technology with with GORE® Cover membrane

Two-stage composting is a relatively new concept in the production of compost, which integrates two completely different technologies into a single composting process in order to improve the final compost quality and increase the cost-effectiveness of the process; it also helps to minimise the negative environmental impact of traditional composting technologies. Several two-stage systems have been examined: some systems combine two composting technologies, such as the integration of in-vessel composting with periodically turned windrow composting or the integration of vermicomposting and traditional composting methods, while other systems include some form of mechanical treatment, such as waste pre-treatment, mechanical biological treatment, or anaerobic digestion, prior to the composting process (Lim et al., 2017).

The first paper addressing the two-stage composting was reported by Kulikowska and Klimiuk (2011), who studied SS waste. The technology involves a transfer between two systems, typically a periodically turned windrow and an aerated bioreactor. The organic waste spends a short time (10 days) in the aerated bioreactor to allow decomposition and sanitation of the organic material before spending 207 days in the windrow to reach maturity. Another two-stage composting technology involves primary composting (PC) and secondary composting (SC) (Zhang et al., 2013). During PC, the thermophilic phase (50 to 60 °C) starts quickly and persists for a long period. When the temperature decreases to ambient temperature, PC is considered complete, and SC begins, during which decomposition continues, resulting in a second thermophilic phase (Zhang et al., 2013). This novel technology thus results in two thermophilic temperature peaks during the whole composting process and a longer thermophilic phase than that obtained with traditional composting (one stage with composting windrows); in addition, compared with traditional composting, the composting duration of the new method is shortened, and the quality and stability of the compost product are increased. Only 30 days are required for the production of mature compost with two-stage composting, as opposed to the 90 to 270 days required in traditional composting (Khalil et al., 2008).

2.7.5 Aerated static windrow with GORE® Cover membrane

This technology has three crucial elements: GORE® Cover membrane, controlling equipment and aeration systems. The active aeration system supplies the microorganisms that participate in the composting process with enough oxygen and the aeration is continuously controlled by the feedback information from the temperature and oxygen level of the composted material. With this technological procedure the windrows are stacked to a height of 2-3.5 m and a width of 6-9 m, while their length can vary between 20-50 meters. They are supplied with computerized operating technology that controls aeration based on pre-defined temperature and oxygen content limit values during the four week long of the composting period (Dér and Füleky, 2017). In the aerated static windrow with GORE® Cover membrane, the system is closed by a GORE® Cover, creating a complete in-vessel enclosure around the entire windrow, and heat was retained, which along with the bottom-up aeration, created a slight positive pressure. The positive pressure provided a homogeneous distribution of temperature and oxygen throughout the composting volume. These conditions enabled aerobic fermentation and accelerated the biological treatment process, therefore obviating the need for frequent windrow turning and ensuring the achievement of the necessary temperature for sanitization.

The SS composting under semi-permeable film technology at full-scale not only minimizes considerably the odour emission, but also promoted the presence of thermophilic bacteria and fungi during the hydrolytic phase of composting that results in an early biodegradation process (Robledo-Mahón et al., 2019a). In a study by González et al. (2016) demonstrated the advantages of carrying out the composting process for sewage-sludge at an industrial scale by using aerated static windrow covered by a semi-permeable film. This technology minimized gaseous and odorous emissions and accelerated the composting process. Moreover, composting under semi-permeable films is cost-effective as it facilitates the treatment of large volumes of waste. Previous studies have also shown the environmental benefits of using semi-permeable film in the composting process which reduces liquid, solid and odorous emissions (Turan et al., 2009; Komilis et al., 2004). However, the limitation of this technology is that the infrastructure is more costly and complex than open systems.

Membrane-covered compost technology has several advantages over traditional aerobic composting technology including even oxygen distribution, longer duration of high temperatures, rapid temperature increase, low energy consumption, and 30 to 40 % reduction of emissions from ammonia and methane, respectively (Sun et al., 2016; Ma et al., 2017). Van Haaren et al. (2010) demonstrated that composting with semipermeable film reduces nitrogen oxides, sulfur, and carbon dioxide emissions compared to open windrow composting. The use of an integrated system of semi-permeable cover and an air-insufflation system during SS composting at an industrial scale has showed notable shifts in bacteria communities. Microorganisms with the ability to decompose recalcitrant compounds appear in an early phase due to the high temperatures result in shorter composting time, thus the appropriateness and effectiveness of novel semi-permeable cover is confirmed as appropriate for composting SS to produce optimal compost (Robledo-Mahón et al., 2018). A semipermeable cover was an effective composting technology for excluding pathogens (Robledo-Mahón et al., 2020). Composting process with SS at an industrial scale, using semipermeable cover technology is effective for the removal of human or animal viruses, and a reduction of the majority of phytopathogen viruses (Robledo-Mahón et al., 2019b).

A recent study by Ma et al. (2018a) showed that during composting pig manure and wheat straw using an innovative semi-permeable membrane cover with a bottom-up aeration system, the abundance of anaerobic *Clostridiales* and pathogenic *Pseudomonas* declined and *Cellvibrionales* facultative anaerobe with a complex cellulase system increased. This was due to enhancement of the oxygen environment of the pile under slight micro-positive pressure (Ma et al., 2018a). In another recent study, composting pig manure and wheat straw using semi-permeable membrane with a bottom-up aeration system enabled sufficient oxygen concentration to be achieved by slight positive pressure, which decreased the anaerobic zone in the compost pile and CH_4 emission by 22.42 % (Ma et al., 2018b). In the same study the volatile solids content decreased quickly because the semi-permeable cover accelerated microbial activities which was conducive to accelerated fermentation and degradation. In addition, composting cattle manure with forced ventilation and semi-permeable film led to good reductions in GHGs (CO_2, CH_4, and N_2O) and ammonia emissions as the temperature rose quickly to 70 °C within two days and remained constant (Sun et al., 2018). Moreover, during composting biogas residue using forced ventilation composting system with semi-permeable membrane, nitrogen preservation rate was increased by 17.27 % and NH_3 and H_2S emission decreased by 58.64 % and 38.13 %, respectively (Li et al., 2020).

2.8. Compost stability evaluation

It is necessary for a compost to have a high level of maturity or stability to be safely used in soil. This requires a stable organic material content and a lack of pathogens and phytotoxic compounds. Maturity is related with phytotoxicity or plant growth potential, whereas stability is associated with the biological activity of the compost (Bernal et al., 2009). If immature compost was applied to soils, the outcome would likely be a suppressed plant growth, inhibition of seed germination and destruction of the roots.

The terms compost stability and maturity are interchangeable in the literature. The stability of compost is currently being evaluated and the composting process is being monitored with a number of parameters and criteria being proposed (Gajalakshmi and Abbasi, 2008; Bernal et al., 2009; Bernal et al., 2017; Wichuk and McCartney, 2010; Azim et al., 2018; Antil et al., 2014). These criteria and parameters, which are grouped into biological, physical or chemical, are presented in **Table 1**.

Table 1. Maturity parameters established for composts of different sources (adapted from Antil et al. 2014)

Parameter	**Value**	**Reference**
Colour (degree of darkness)	Dark brown or black colour	Lekasi et al. (2003)
Temperature	More or less constant after turning of material	Azim et al. (2018)
pH	7.1–7.7 (SS + GW)	Forster et al. (1993)
	8.4 (Organic matter)	Iglesias-Jimenez and Alvarez (1993)
	7 (GW)	Iannotti et al. (1994)
EC	≤ 3 (mS cm^{-1})	Soumare et al. (2003b)
	< 4 (mS cm^{-1})	Lasaridi et al. (2006)
C/N	< 20, preferable < 10	Bernal et al. (2009)
	< 15	Roig et al. (1988)
	7.8	Iglesias-Jimenez and Alvarez (1993)
Water soluble organic-C	≤ 1 %	Hue and Liu (1995)
	≤ 0.4 %	Zmora-Nahum et al. (2005)
	< 1.7 %	Bernai et al. (1998)
Water soluble C/N ratio	0.55	Bernai et al. (1998)
	< 0.7	Hue and Liu (1995)
NH_4^+-N	< 0.04 %	Bernai et al. (1998)
NH_4^+/NO_3^-	< 0.16	Bernai et al. (1998)
CEC	> 60 cmol/kg	(Harada and Inoko, 1980a)
	> 67 cmol/kg	Iglesias Jiménez and Pérez García (1992)
CEC/TOC	> 1.70	Roig et al. (1988)
Humic acid / fulvic acid	> 1.9	Iglesias-Jimenez and Alvarez (1993)
	3.3-7.5	Forster et al. (1993)
Humification index	> 30	Raj and Antil (2011); Bernai et al. (1998)
Germination index	> 50 %	Zucconi et al. (1981)
	> 80 %	Tiquia et al. (1996); Sellami et al. (2008)
	> 110 %	Ko et al. (2008)
CO_2 production rate	< 120 mg CO_2/kg/h	Hue and Liu (1995)
OM_{loss}	> 42 %	Raj and Antil (2011)
Microbial diversity bacterial, fungal and actinomycetes counts)	Decrease in bacterial and fungal counts, and increase in actinomycetes counts and stable at the end of composting	Antil and Raj (2012)
Micro-organism counts (total coliform, faecal coliforms and faecal enterococci)	< 500 MPN/g	Vuorinen and Saharinen (1997); Dahshan et al. (2013)
Enzyme activities cellulase, xylanase and protease)	Maximum between 30 and 60 days of composting	Goyal et al. (2005)

2.8.1 Colour

When organic waste is composted, the material becomes gradually darker, and the final product, after a satisfactory long maturation period, becomes dark brown (or, in some cases, virtually black) (Iglesias Jiménez and Pérez García, 1992). Since colour is a relatively straightforward criterion for assessing compost maturity and stability, other considerations, including physical, biological, and chemical parameters, can also be considered for confirm compost maturity and stability.

2.8.2 Temperature

Change in temperature reflects microbial activity during composting. During the initial period, which generally lasts several days, the compost heap's temperature will increased to the thermophilic range (60-70 °C), followed by an incremental decrease to a constant temperature, after which an ambient level is reached (Raj and Antil, 2012, 2011; Satisha and Devarajan, 2007). In the literature, one strategy for determining whether compost is sufficiently mature involves taking its temperature (Gajalakshmi and Abbasi, 2008). Specifically, if the compost heap's temperature has remained constant for some time, and if no change in temperature is observed with the turning over of the material, then the compost can be considered mature.

2.8.3 Loss of organic matter

Measuring the mineralization rate of OM during composting is a relatively simple procedure, which relies on the determination of weight loss. Cumulative loss of OM increases with composting time for all compost types, and it is measured as weight loss. The weight loss/OM loss was significantly correlated with C:N ratio and other stability/maturity parameters (Chefetz et al., 1996; Raj and Antil, 2012).

2.8.4 pH

As one of the initial chemical measures of compost maturity, pH has been employed in various studies (Antil et al., 2014). Iglesias Jiménez and Pérez García (1991) and Cayuela et al. (2006) have reported an increase in pH during composting process. Noteworthily, Avnimelech et al. (1996) noted that acidic pH levels are characteristic of compost that is not sufficiently mature, whereas mature composts are strongly associated with composts that have a pH of 7-9.

2.8.5 Electrical conductivity

The EC reflects the proportion of dissolved salts in the compost. It is a significant measure because it indicates the salinity of the compost, where excessively saline compost can have a negative impact on plants. The sum of soluble salts in the water extracts rises with compost maturation because of release of soluble salts and organic acids during OM decomposition suggested the stability of compost (Avnimelech et al., 1996).

2.8.6 C:N ratio (solid phase)

As decomposition progresses due to the loss of C, mainly in the form of carbon dioxide, the carbon content in the compostable material is reduced with time, and N content per unit material increases. Consequently, the C/N ratio decreased. This parameter is used to assess the maturity of compost (Azim et al., 2018). A C/N ratio of less than 20, and even 15, is indicative of mature compost (Iglesias Jiménez and Pérez García, 1991). Additionally, although a compost with a ratio of 10-15 may be considered as stable, the final ratio is strongly dependent on the initial materials that were used (Chefetz et al., 1996).

2.8.7 C:N ratio (water extract)

Given that composting involves the biochemical transformation of OM by microorganisms, in which metabolism occurs in water soluble phase. Therefore, examining the changes occurring in soluble OM can be used for evaluating compost maturity. Several groups of researchers have reported that a C_w:N_{org} ratio of 5-6 is an essential indicator of compost maturity (Garcia et al., 1991; Hue and Liu, 1995).

2.8.8 Water soluble carbon

The water soluble carbon (WSC) represents the most easily biodegradable C fraction during the composting process because it consists of amino acids, phenols, sugars and organic acids, apart from the soluble fraction of fulvic acids (Garcia et al., 1991). The WSC concentration decreased with composting time in all types of composts. Biodegradable C fractions were initially consumed by microbes, which leads to the decomposition of complex organic compounds and the subsequent release of carbon dioxide at the end and some of them polymerizes with nitrogenous compounds, thereby generating human substances. Garcia et al. (1991) established that 0.5 % value is the maximum WSC content, after which the compost can no longer be regarded as mature. Other values of this parameter were those noted by Hue and Liu (1995) (WSC < 1 %) and by Bernai et al. (1998) (WSC < 1.7 %).

2.8.9 Nitrification index

Nitrification is another criterion that has been used as an indicator of compost maturity. The so-called “nitrification index” refers to the ratio of NH_4^+-N to NO_3^--N. The literature indicates that the nitrification index declines over the course of composting, and in Bernal et al. (2009) study, 0.16 was identified as a maximum value for mature compost.

2.8.10 Cation exchange capacity

The CEC of the composts increased with composting time due to decomposition of OM. The humification process generated functional groups influenced by the increased oxidation of OM, thereby leading to increased CEC. Consequently, CEC is an indicator useful for estimating the degree of compost maturity (Harada and Inoko, 1980a). In the studies conducted by Harada and Inoko (1980a) and Iglesias Jiménez and Pérez García (1992), CEC values of 60 and 67 cmol kg^{-1}, respectively, were identified as acceptable minimum values for the maturation of city refuse compost. Additionally, Roig et al. (1988) noted that the relationship between CEC/TOC and C_t/N_t (t = total) was a valuable tool for determining the level of maturity in cattle manure, chicken, sheep, and rabbit. Furthermore, the researchers recommended a CEC/TOC value of greater than 1.7 for effective humification levels of waste manures.

2.8.11 Humification

Since humification in the composting of organic substrates leads to the decomposition of non-humic substances, as well as formation of humic substances, it is regarded as another important indicator of compost maturity (Tiquia, 2005). Therefore, humic acid fraction typically rises during composting, which reflects OM humification. The ratio of optical density of humic acid solutions at 465 and 665 nm is referred to as the E_4/E_6 ratio, and when it exceeds 1.7%, this is considered an indicator of effective compost maturity (Iglesias Jiménez and Pérez García, 1992).

2.8.12 Germination test

Germination of seeds and seedling root length may be undermined by the use of immature compost in arable soil, principally because this increases the presence of phytotoxic compounds. The application of immature compost to agriculture soil may reduce the root length of seedlings and / or inhibit germination of seeds due to the presence of phytotoxic substances in the compost. Based on this fact the germination test can be used to measure compost maturity. One plant species commonly frequently used in germination tests is Cress (*Lepidium sativum*), due to its small seed sizes, as well as their rapidly germination under optimal conditions. Noteworthily, germination tests offer an instant picture of phytotoxicity (Luo et al., 2018).

2.8.13 Oxygen and CO_2 respirometry

The aerobic respiration rate was considered as the most suitable parameters to evaluate the aerobic biological activity and, therefore, stability. The aerobic respiration can be measured using a range of approaches: self-heating, oxygen consumption and carbon dioxide evolution, which are an indication of the levels of degradable OM that are still present, and which are linked in an inverse way to stabilization (Antil et al., 2014). In compost that is not mature, the demand for oxygen is still high, and carbon dioxide generation rates are also elevated, due to the elevated microbial activity because of the abundance of easily biodegradable substances in the raw material. Due to this, oxygen consumption and carbon dioxide production reflect compost maturity and stability (Gómez et al., 2006; Hue and Liu, 1995; Iannotti et al., 1994). In the study conducted by Hue and Liu (1995), the limit for the carbon dioxide production rate for compost maturity was less than 120 mg CO_2 $kg^{-1}h^{-1}$.

2.8.14 Microbial population count

The degradation of OM is a microbial process and is linked in a direct way to total microbial count and their activity during the composting of organic wastes (de Bertoldi et al., 1983). This approach relies on the initial hypothesis that the microbial population stabilized on compost maturity. During the thermophilic phase, an increase is observed in the microbial population, followed by a decrease at the maturation phase. In turn, once the composting process has finished, the microbial population become constant, which reflects the stability of the composting material (Tiquia et al., 1996). A reduction in bacterial and fungal counts, and increase in actinomycetes counts and stable at the end of composting appear as good indicator to identify the level of biological stabilization and optimum degree of compost maturity (Antil and Raj, 2012). In the case of compost hygienization, the average number of faecal enterococci, faecal coliforms and total coliform densities should be less than < 500 MPN/g (Dahshan et al., 2013; Vuorinen and Saharinen, 1997).

2.8.15 Enzyme activities

Various hydrolytic enzymes (protease, xylanase and cellulase) are suggested to control the rate at which different organic substrates are degraded. Furthermore, it has been well-documented in the literature that enzymes play a fundamental mediating role in a range of degradative processes (Tiquia et al., 1996). Another index of compost maturity is the maximum activities of protease, xylanase and cellulase between day 30 and day 60 composting (Goyal et al., 2005). Dehydrogenase activity is the most straightforward, efficient, and cost-effective way to evaluate compost stability and maturity, which is linked to changes in humification parameters and temperature (Tiquia, 2005).

2.9. Humification

Humification process is the sum of the decomposition of lignin to aromatic units, which happen primarily during the thermophilic stage and the subsequent biosynthesis from the condensation and polymerization of aromatic units and cellular debris (Senesi and Plaza, 2007). The humification principally composed of the rearrangement and polymerization of sub-units with lower molecular weights, mediated by microorganism population growing in the composting waste (Senesi, 1989).

Within the soil's OM, the portion that is most resistant to microbial degradation is the humidified fraction. This fraction is also responsible for the soil's organic fertility function. Therefore, the degree of OM humification during composting can be an effective measure of the compost quality. In composts, where the OM has reached a high level of humification, the agricultural value increases. The humification of the OM during composting is displayed by the creation of humic acids (HA) with increasing functional groups, oxygen and nitrogen concentrations, aromatic characteristics and molecular weight, in agreement with the usual humification theory of soil OM (Guo et al., 2019). During composting, the production of humic substances (alkali-extractable organic-C, C_{EX}), is coupled with an increase in humic acid-like organic carbon (C_{HA}) production. At the same time, water extractable organic carbon and fulvic acid-like organic carbon (C_{FA}) decrease as a result of microbial degradation.

The most reliable evaluation approach of a compost's humic character is based on the determination of the structural and chemical composition, along with its functional properties, also in comparison with the humic substances present in native soil. A number of physico-chemical, chemical and spectroscopic methods have been used in the literature. These include functional and elemental group composition, molecular weight distribution, FTIR spectroscopy, fluorescence spectroscopy, electrophoresis and electrofocusing, electron spin resonance (ESR) spectroscopy, pyrolysis-gas chromatography-mass spectrometry (GC–MS), and the ratio of absorbances measured at 465 and 665 nm (E_4/E_6) (Lü et al., 2018). Amongst these methods, the more advanced techniques, including pyrolysis, FTIR and nuclear magnetic resonance (NMR), can provide a better understanding for structural change of the OM during the composting process, and thereby enhance the evaluation of the composting efficiency and maturity (Lü et al., 2018). In order to study any changes produced within the structure of humic acid, literature suggests that functional group analysis can be a highly sensitive method when compared to other methods, including infrared spectroscopy, gel permeation chromatography, and elemental analysis. The composting process generate HA with similar structural and chemical characteristics of humified soil HA (Senesi, 1989).

3. MATERIALS AND METHODS

3.1. Composting setup and description of encapsulated lifting system with GORE® Cover membrane

3.1.1 Composting materials

The study was carried out at a composting facility at the municipal wastewater treatment plant in the city of Kecskemét, Hungary. The plant serves a population of 240,000, treats 20,000 m^3/d and generates 22,400 t/year of activated SS (ProfiKomp, 2013). The composting facility was established in 2010, with its initial production capacity being approximately 5,000 m^3/year (ProfiKomp, 2013).

The DSSS and GW were used for composting. The DSSS was obtained by centrifuging stabilized sludge from a biogas digester, while the GW (grass, wood chips, tree leaves, etc.) was collected from the parks and gardens in Kecskemét. The DSSS was mixed with the GW materials, which acted as bulking agents, at a 1:4 proportion of sludge to bulking agent to obtain optimum moisture and a C/N ratio of approximately 65 % and 25:1, respectively. The main characteristics of the raw materials used are presented in **Table 2**. The DSSS was characterised by having the highest moisture and nitrogen content, while the GW had a high OM content and the lowest moisture content. The GW materials were thus added to lower the overall moisture content, increase the C/N ratio, and enhance the final structure of the composted material.

Table 2. Characteristics of the raw materials (DSSS-GW) used for composting

Organic waste	Moisture (%)	OM (%)	TC (%)	TN (%)	C/N	EC (mS cm^{-1})	pH
DSSS	80.5 ± 0.7	71.8 ± 1.5	31.5 ± 1.2	4.1 ± 0.0	7.71 ± .4	3.1 ± .01	7.4 ± 0.0
GW	29.4 ± 1.1	97.4 ± 0.2	54.0 ± 1.0	0.7 ± 0.0	74.7 ± 2.4	0.7 ± 0.0	7.1 ± 0.2

± represents the standard deviation based on three replicates

3.1.2 Composting process

Composting processes were carried out using an ELS with GORE® Cover membrane at an industrial scale (**Figure 2**). The composting process consisted of two phases of ASW treatment under GORE® Cover membrane: an intensive phase (4 weeks) and a maturation phase (3 weeks).

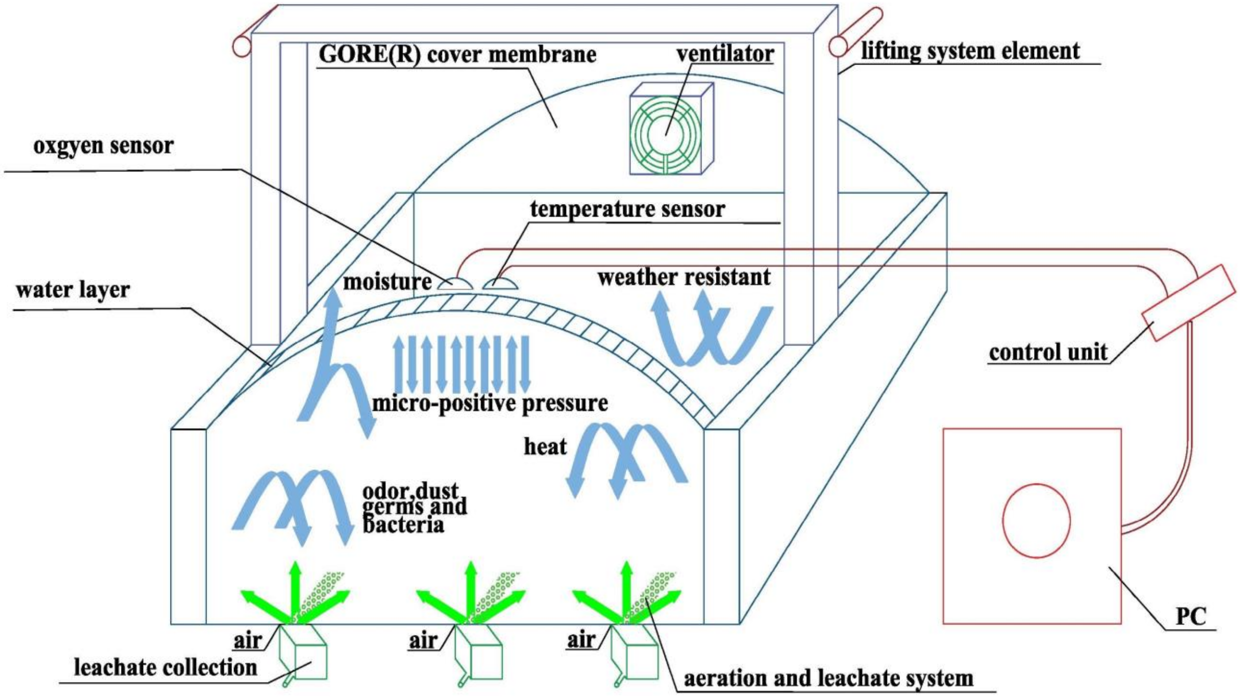

Figure 2. Schematic diagram of the encapsulated lifting system with GORE® Cover membrane

At the beginning of the intensive phase, the mixture was loaded into a composting cell on day 0. The cell (35 m long, 8 m wide, and 2.6 m high, with a 730 m^3 total capacity) had three reinforced concrete side walls, covered with GORE® Cover membrane, and had a dedicated ventilation system to ensure its oxygen supply. When the windrow temperature dropped to ambient temperature at day 28, the intensive phase was considered complete. At that time, the mixture was removed from the composting cell with an excavator and placed in another composting cell for the maturation phase. That cell (30 m long, 8 m wide, and 2.6 m high, with a 630 m^3 total capacity) had three reinforced concrete side walls, which were covered with GORE® Cover membrane and had a ventilation system to provide adequate aeration. The maturation phase began on day 29 till 49. A 4 m high glass-fibre tarpaulin was attached to the reinforced concrete walls, and the whole structure was covered with GORE® Cover membrane during both phases to avoid excessive water loss caused by wind and evaporation. The two-stage composting process was considered complete when the temperature of the windrow declined to ambient temperature. The material was then screened using a screening drum with 12 x 12 mm holes.

3.1.3 Ventilation system

Air was supplied via ventilators and trenches placed at the base of the windrow. These trenches acted as ducts to provide air to the windrow and collect leachate from the windrow. Aeration was accomplished by ventilators with a maximum flow rate of 2400 m^3/h at 2940 rpm and 80 dB(A), using a forced aeration system placed at the base of the windrow (ProfiKomp, 2013). The ventilator engine power was 2.5 kW. A ventilation system controlled by a timer was used to control the windrow temperature. The injected flow rate diffused the air intermittently. In the intensive phase, the aeration frequency was 491 min/d (491 min ON/949 min OFF), and the air flow rate was 172 m^3 air/m^3 compost/d, while in the maturation phase, the frequency was 404 min/day (404 min ON/1036 min OFF), and the air flow rate was 166 m^3 air/m^3 compost/day (ProfiKomp, 2013).

3.1.4 Sampling strategy and collection

Composting was conducted for 49 days. Homogenous 1-kg samples that represented the average conditions of the entire windrow were collected in accordance with the US Composting Council standard (TMECC, 2002) at the following intervals: 0 (first day of composting/initial non-decomposed material), 7, 14, 21, and 28 days during the intensive phase, and 35, 42, and 49 days during the maturation phase.

3.1.5 Temperature monitoring

The temperature was measured using Pt-100 type sensor housed in stainless-steel sheath that were connected to a data acquisition system linked to a standard PC. The Pt-100 sensor were placed in the windrow at measuring points located at half the windrow's height. The daily mean temperatures were calculated based on recorded hourly temperatures that were taken continuously throughout the process using a digital recorder. Another temperature sensor was placed outside to measure the ambient temperature for comparison.

3.2. Composting setup and description of aerated static windrow with GORE® Cover membrane

3.2.1 Composting materials

This research study was undertaken in a composting industrial plant located in Gödöllő city, Hungary. The main characteristics of the raw materials used are presented in **Table 3**. The GW consisted mainly of fallen leaves and branch cuttings generated by urban landscape maintenance. The GW was reduced to a particle size of about 1 cm with a shredder to increase the reactive surfaces for microorganisms, before the composting began. The MC was then adjusted to 60 % by the addition of water.

Table 3. Characteristics of the raw materials used for composting

	Moisture (%)	OM (%)	TC (%)	TN (%)	C/N	EC (mS cm $^{-1}$)	pH
GW	23.1 ± 1.9	56.9 ± 1.4	31.3 ± 0.1	0.8 ± 0.0	38.6 ± 0.6	0.9 ± 0.0	7.0 ± 0.1

± represents standard deviation based on three replicates

3.2.2 Composting process

Composting was carried out by ASW with GORE® Cover membrane at industry-scale (**Figure. 3**). The GW was stacked to 35 m in length, 8 m in width, and 2.5 m in height in trapezoidal windrow, which were covered with GORE® Cover membrane and had a ventilation system to ensure oxygen supply. The windrow had a 3-sided reinforced concrete wall to retain the material in the row on the compost pad. The composting process was considered complete when the windrow temperature dropped to ambient temperature. Then the material was screened using a screening drum with size holes that were 12 x 12 mm.

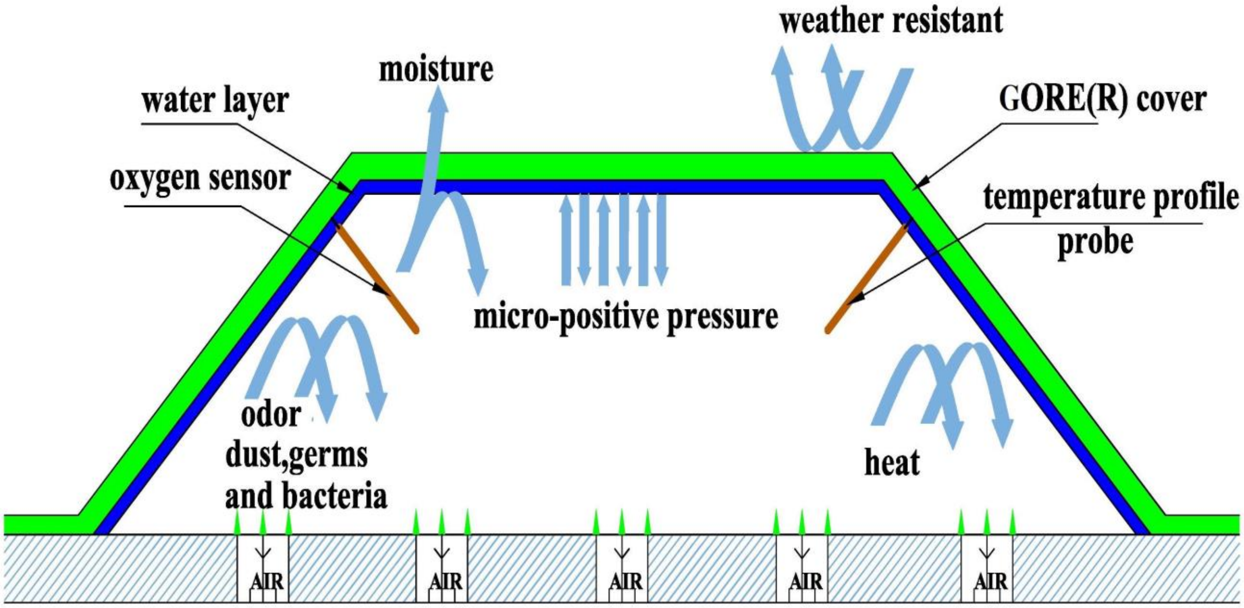

Figure 3. Schematic diagram of the aerated static windrow with GORE® Cover membrane

3.2.3 Ventilation system

Air was supplied via ventilators and trenches placed at the base of the windrow. The trenches acted as ducts to provide air to the windrow and collect leachate from the windrow. Aeration was accomplished by ventilators with a maximum flow rate of 2400 m^3/h at 2940 rpm and 80 dB(A) using a forced aeration system placed at the base of the windrow (Dér and Füleky, 2017). The ventilator engine power was 2.5 kW. A ventilation system controlled by a timer was used to control the windrow temperature. The injected flow rate diffused the air intermittently. The aeration frequency was 15 min per hour (15 min ON/45 min OFF) (Dér and Füleky, 2017).

3.2.4 Sampling strategy and collection

Composting was conducted for 30 days. Homogenous 1-kg samples that represented the average conditions of the entire windrow were collected in accordance with the US Composting Council standard (TMECC, 2002) at the following intervals: 0 (first day of composting/ the initial non-decomposed material), 7, 14, 24 and 30 days of composting.

3.2.5 Temperature monitoring

The temperature was measured using Pt-100 type sensor housed in stainless-steel sheath that were connected to a data acquisition system linked to a standard PC. The Pt-100 sensor were placed in the windrow at measuring points located at half the windrow's height. The daily mean temperatures were calculated based on recorded hourly temperatures that were taken continuously throughout the process using a digital recorder. Another temperature sensor was placed outside to measure the ambient temperature for comparison.

3.3. GORE® Cover membrane

The GORE® Cover membrane had three different layers: an expanded polytetrafluoroethylene (ePTFE) membrane/microporous film, and two protective layers made of high-robust polyester fabric that formed the top and bottom layers of the laminate. The membrane in the middle layer acted as the functional component, and the membrane parameters were as follows: water vapor permeability resistance of ≤ 19.5 m^2 Pa W^{-1}, air permeability of 1.5–6.5 m^3 m^{-2} h^{-1}, and hydrostatic pressure of $\geq$ 50 kPa (Sun et al., 2018).

The GORE® Cover membrane is waterproof, windproof, and reusable; the material is highly tolerant to biological and chemical degradation and to heat (up to 200-260°C) (Sun et al., 2018). With a micropore size of approximately ≤ 0.2 µm, the membrane effectively prohibits the release of microbes, gaseous substances, and particulate matter while remaining semipermeable to moisture (Mukhopadhyay and Vinay Kumar Midha, 2008). Most of the water vapor generated during the early composting phase due to decomposition of OM did not escape the membrane because its limited permeability; instead an aqueous condensate layer accumulated on the inner surface of the cover. This layer acted as a scrubber for odours by dissolving odour compounds and forming droplets that returned the compounds to the composting material where they could be further decomposed by microorganisms.

3.4. Physico-chemical analyses of compost samples

3.4.1 Moisture content and dry matter

The MC was determined by drying the fresh sample at 105 °C to a constant weight. The DM was estimated as follows:

Equation 1. Dry matter

$$DM\ (\%) = \left(\frac{oven\ dry\ sample\ weight}{wet\ sample\ weight}\right) \times 100$$

3.4.2 Bulk density

The wet bulk density (BD_{wet}) is the mass of fresh material occupying a particular volume. The dry bulk density (BD_{dry}) is the mass of material occupying a particular volume after oven drying (Agnew and Leonard, 2003). Both BD_{wet} and BD_{dry} was calculated using the following two equations:

Equation 2 Dry bulk density (BD_{dry})

$$\mathrm{BD}_{\mathrm{dry}} = \frac{M_{\mathrm{dry}}}{V_{\mathrm{wet}}}$$

Equation 3 Wet bulk density (BD_{wet})

$$\mathrm{BDwet} = \frac{M_{\mathrm{wet}}}{V_{\mathrm{wet}}}$$

In Equation 2 M_{dry} is the mass of dry material in kg and V_{wet} is the volume occupied by the material before it was dried in m^3. In Equation 3 where M_{wet} is the mass of undried material occupying the volume V_{wet}.

3.4.3 Ash content and organic matter

Ash content was determined by burning samples (previously dried at 105°C) at 550°C for 7 h in a muffle furnace (TMECC, 2002). The OM was estimated as follows (TMECC, 2002):

Equation 4. Organic matter

$$OM\ (\%) = 100 - ash\ (\%)$$

3.4.4 Loss of organic matter

Losses of OM due to OM mineralization were calculated based on the initial and final ash content for each time interval according to the equation (Paredes et al., 2000):

Equation 5. Losses of organic matter

$$\mathrm{OM}_{loss}(\%) = 100 - 100\left[\frac{x_1(100 - x_2)}{x_2(100 - x_1)}\right]$$

where X_1 and X_2 are the initial and final ash concentrations, respectively.

3.4.5 pH and electrical conductivity

A water extract was prepared for the determination of the *pH* and *EC*. Fresh compost samples were mixed with deionized water at a 1:10 ratio (mass ratio) and shaken for 1 h, then centrifuged at 4000 rpm for 20 min and filtered through 0.45 µm membrane filter (Guo et al., 2012).

3.4.6 Total phosphorus and total potassium

Total phosphorus (TP) was determined colorimetrically as vanadomolybdophosphate, and total potassium (TK) was analyzed using an atomic absorption spectrometer after the air-dried samples were digested with concentrated HNO_3 and 30 % H_2O_2 (Brunner, 1978).

3.4.7 Oxidizable organic carbon

The oxidizable organic carbon (OXC) was determined using air dried samples according to the Walkley-Black procedure described by Navarro et al. (1993). A 50-mg sample was placed in a 500-ml Erlenmeyer flask, and $K_2Cr_2O_7$ (10 ml) and concentrated H_2SO_4 (20 ml) were added. The mixture was then shaken vigorously for 30 s, and, after half an hour, the dichromate excess was titrated after the addition of 10 ml of concentrated H_3PO_4 with 0.5 M $Fe(NH_4)_2(SO_4)_2.6H_2O$. A blank was also run with 10 ml of $K_2Cr_2O_7$ and 20 ml of concentrated H_2SO_4.

3.4.8 Total nitrogen and total carbon

The TN and TC determinations were performed by using a CNS analyzer (Fisons NA 1500 Series II CNS analyzer). Air-dried samples were weighed in a tin capsule. The capsule was then closed and introduced into the analyzer, which had previously been calibrated with atropine. The temperature of the reaction oven was 1020°C, the reduction oven was at 600°C and the separation chromatographic column at 60°C.

3.4.9 Cation Exchange capacity

The CEC of the compost was measured by method proposed by Harada and Inoko (1980a). A 200 mg sample of air-dried samples was washed with 0.05 N HCI to replace exchangeable cations. After removal of the excess 0.05 N HCI by washing with distilled water, 1N $Ba(OAc)_2$ solution (pH 7.0) was added to the sample, and the mixture was allowed to stand overnight. After filtration, the sample was leached with another portion of 1 N $Ba(OAc)_2$ solution, and the released proton was then titrated against 0.05 N NaOH solution using a potentiometer. The amount of released proton gives the CEC.

3.4.10 Ammoniacal nitrogen and nitrate nitrogen

The NH_4^+-N and NO_3^--N were extracted by mixing air-dried samples with 0.5 M K_2SO_4 at a ratio of 1:10 (w/v), mechanically shaking for 1 h, and filtering through 0.45-μm membrane filters. The filtrate was used for NH_4^+-N and NO_3^--N determination using the colorimetric method, as described by Okalebo et al. (2002).

3.4.11 Water soluble carbon

For the analysis of WSC, 10 g of air-dried samples were extracted with 100 mL of deionized water (1:10 ratio) by shaking for 24 h. The extracts were then centrifuged (3500 rpm, 30 m) and filtered through 0.45 μm filter membranes. Thereafter, a sample of 1-8 ml of filtrate was weighted in a 250 ml Erlenmeyer flask, and 10 ml of 1 N $K_2Cr_2O_7$ and 20 ml of concentrated H_2SO_4 added. The mixture was then shaken vigorously for 30 s and, after 30 min, the excess dichromate was titrated after the addition of 10 ml of concentrated H_3PO_4 with 0.5 N $Fe(NH_4)_2(SO_4)_2$.

A blank was also run with 10 ml of $K_2Cr_2O_7$ and 20 ml of concentrated H_2SO_4 (Sánchez-Monedero et al., 1996).

3.4.12 Macro-nutrients, micro-nutrients and metal contents

The Ca, Na, Mg, Cr, Cu, Fe, Mn, Ni, Zn, Cd, Co, and Pb concentrations were measured after microwave HNO_3/H_2O_2 digestion by Inductively Coupled Plasma-Optical Emission Spectroscopy (ICP-OES) (TMECC, 2002).

3.5. Germination assay

The germination index (GI) which combines the measure of seed germination and root growth of seeds is one of the most sensitive biological parameter, and more reproducible than direct seeding tests used to evaluate toxicity and the degree of maturity and stability of compost, but require additional time for extract preparation (Luo et al., 2018; Wong et al., 2001).

A *germination test* was performed with water extracts (1:10, w/v) obtained from compost samples taken at different sampling times during composting to evaluate the phytotoxicity of the composting mixtures. The aqueous extracts were obtained by adding 100 ml of water to 10 g of compost in a flask and shaking for 1 h, centrifuging at 4000 rpm for 20 min and filtering through 0.45-μm membrane filters (Guo et al., 2012). Two sheets of sterilized filter paper were placed in a sterilized Petri dish (90 mm in diameter) and wetted with 8 ml of the compost extract. Twenty cress (*Lepidium sativum* L.) seeds were then added to the dish. Each treatment (including a deionized distilled water control) was performed in three replicate dishes. Filtered water was added as needed to maintain moisture. The Petri dishes were sealed with Parafilm to reduce water loss while allowing air penetration, and the dishes were kept at room temperature in the dark. After incubation at 25 °C for 72 h in the dark, the number of germinated seeds (after 24 h) and the lengths of root radicals (after 72 h) were determined for each dish.

The GI was calculated by the following equation (Luo et al., 2018):

Equation 6. Germination index

$$\mathrm{GI}\ (\%) = \left[\frac{(\text{mean number of germinated seeds per dish in the treatment } \times \text{ mean root length per dish in the treatment})}{(\text{mean number of germinated seeds in the control } \times \text{ average root length of the control })}\right] \times 100$$

3.6. UV-visible spectroscopic analysis

UV-visible spectroscopy analysis was performed according to the following method by (Zbytniewski and Buszewski, 2005). One gram of air-dried compost and 50 ml of 0.5 M NaOH were shaken for 2 h and then centrifuged at 3000 rpm for 25 min. The absorbance of the supernatant was measured at λ = 280 nm (A280), 472 nm (A472), and 664 nm (A664) using a UV-2000 spectrophotometer covering a frequency range of 200 to 800 nm. The following absorbance ratios were calculated: $E_{2/6}$ = A280/A664, $E_{4/6}$ = A472/A664, and $E_{2/4}$ = A280/A472. These absorbance ratios demonstrate the degree of maturation (humification) of OM. $E_{2/4}$ indicates the proportion between lignins and other materials at the beginning of humification and the content of materials at the beginning of transformation. $E_{2/6}$ denotes the relation between nonhumified and strongly humified material. $E_{4/6}$ is often called the humification index, and typical values of the $E_{4/6}$ ratio for humified materials are usually <5 (Zbytniewski and Buszewski, 2005).

3.7. FTIR spectroscopic analysis

Air-dried samples from each composting step were analyzed with a Bruker DRIFT (diffuse reflectance infrared fourier transform)-FTIR spectrometer covering a frequency range of 4000 – 400 cm^{-1}. The resolution was set to 4 cm^{-1}, and 32 scans were recorded before being averaged for each spectrum and corrected against ambient air, which acted as the background.

3.8. Statistical analyses

The data shown in the tables and figures represent the means of three replicate subsamples for each composite sample with standard deviations calculated using Microsoft Excel 2016. Each set of three subsamples was prepared from the corresponding composite sample (representing the average conditions of the entire windrow in question) and analyzed independently.

The principal components analysis (PCA), applied on the matrix of correlation between the variables, was used to study the variations occurring in the physico-chemical parameters during composting. The statistical treatments were carried out using the software SPSS Win version 25.

4. RESULTS AND DISCUSSION

4.1. Encapsulated lifting system with GORE® Cover membrane

4.1.1 Temperature evolution

The composting process is principally a biological phenomenon that relies highly on temperature fluctuations within windrows. In this case, the mixture of DSSS and GW at the beginning of the composting process was characterized by its large fermentable fraction. On day 0 (the start of the intensive phase), the temperature began to rise rapidly from 31.2 °C, reaching a peak temperature of 76.8 °C within three days (**Figure 4**). These temperatures were higher and increased faster than those observed during composting of the same substrate using classical methods (Amir et al., 2005; El Fels et al., 2014). The micropositive pressure environment resulting from the joint effects of the forced bottom-up ventilation system and the GORE® Cover membrane was conducive to providing the even supply of sufficient oxygen to microorganisms that is required to degrade OM. Ultimately, the microbial degradation of easily decomposable OM was enhanced, the decomposition rate increased, and a large amount of heat produced. After peaking during the intensive phase, the temperature decreased, remaining in the mesophilic range (below 42 °C) between days 25 and 31. The decrease in temperature can be explained by two main phenomena: the exhaustion of the initial simple molecules and the development of conditions hostile to microorganism activity.

During the composting process, a paradoxical situation was observed. As part of the maturation phase (beginning on day 29), the temperature began to increase again to produce a second thermophilic phase, with a peak temperature of approximately 75.4 °C again achieved on day 37 (**Figure 4**). Insam et al. (2010) demonstrated that temperatures from 26 to 45 °C lead to favourable conditions for the growth of actinomycetes and other heat-tolerant bacteria, an observation that explains why, during the short mesophilic stage (days 25 to 31), the conditions favoured the regrowth of microorganisms, subsequently leading to the reappearance of microorganism activity and a second increase in temperature. However, after the second thermophilic stage, the temperature progressively decreased to ambient temperature, and the covered compost did not exhibit any large variations in temperature during storage, indicating that the microbial activity and mineralization had ceased and that humification resumed, demonstrating that the compost was stable and that the process could be considered complete. This phenomenon of two thermophilic peaks has been observed by Zhang and Sun (2018b, c, a). Zhang et al. (2013) subjected raw materials to a two-phase composting process comprised of primary fermentation and secondary fermentation during which a thermophilic phase was attained twice. Tønner-Klank et al. (2007) showed that appropriate composting can eliminate many human pathogens in different types of compost, as the temperatures attained (50 to 70 °C) during treatment kill enteric microorganisms.

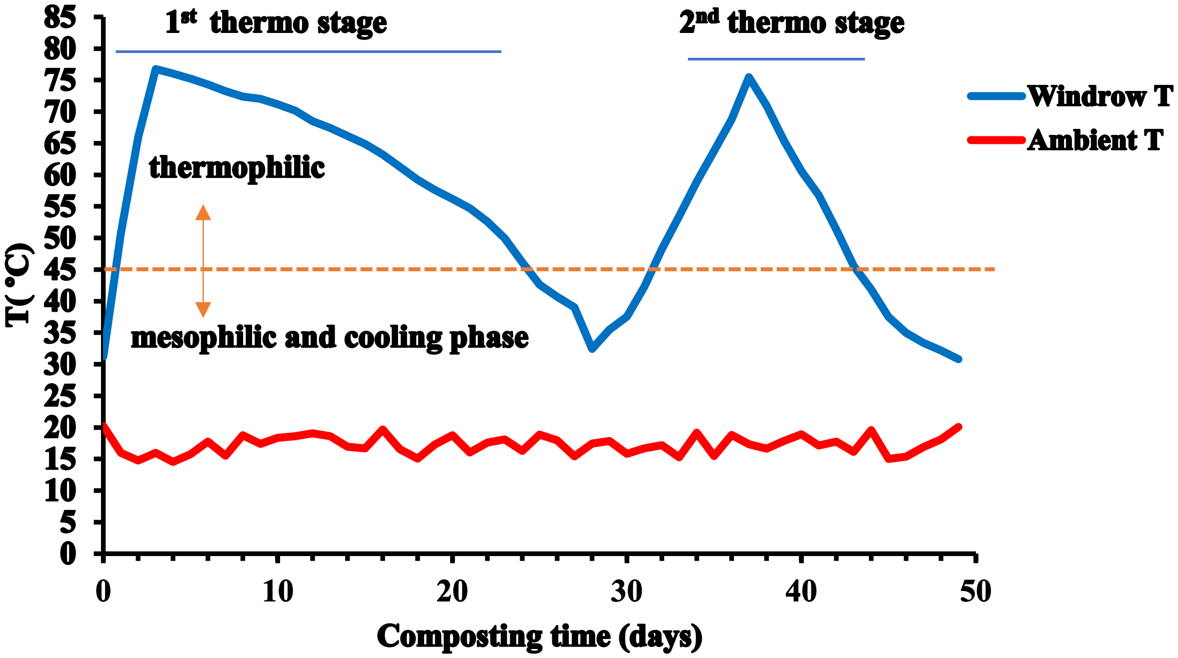

Figure 4. Evolution of temperature during DSSS-GW composting

With regard to the length of thermophilic periods, the composting temperature must exceed 45 °C for at least three consecutive days to destroy pathogenic organisms and thus meet disinfection requirements (de Bertoldi et al., 1983). The results showed that the total duration of thermophilic temperatures in this experiment (intensive phase plus maturation phase) was >3 days. This finding indicated that the duration of the thermophilic stage was sufficient to kill pathogens and meet the disinfection requirements.

By using the new technology, the temperature remained high and steady in the thermophilic temperature range for 24 days and 12 days in the intensive phase and maturation phase, respectively, as the GORE® Cover membrane completely enclosed the windrow, retaining heat. This technology resulted not only in more rapid attainment and prolongation of thermophilic temperatures but also in the creation of two thermophilic temperature peaks (one during the intensive phase and one during the maturation phase); the main consequence of this was more rapid decomposition of organic waste.

El Fels et al. (2014) and El Hayany et al. (2018) stated that traditional composting requires 180 days and 105 days, respectively, to produce a mature product; two-stage composting consisting of the use of an aerated windrow under a semipermeable film and in the second stage a static open windrow requires 190 days (González et al., 2016). However, the proposed new technology shortens the SS composting time significantly, producing mature compost in only 49 days.

4.1.2 Moisture and dry matter

The loss of moisture is an index of composition rate, as the heat generated during the composting process causes vaporization (Varma et al., 2014). During the first 42 days of the composting, moisture contents obviously decreased from 57.3 % to 20.2 % (**Figure 5**). Moisture reduction during the composting process is the result of the decomposition process since an increase in temperature (heat generation) during the process leading to vaporization (Kalamdhad and Kazmi, 2009; Liao et al., 1997). In the last week of composting, the windrow was cooled, and the MC increased (37.6 %). Cooling resulted in the condensation of water vapor, which was retained on the surface of GORE® Cover membrane, moistened the compostable matter (González et al., 2016). It can be shown DM varies inversely with respect to moisture (**Figure 5**).

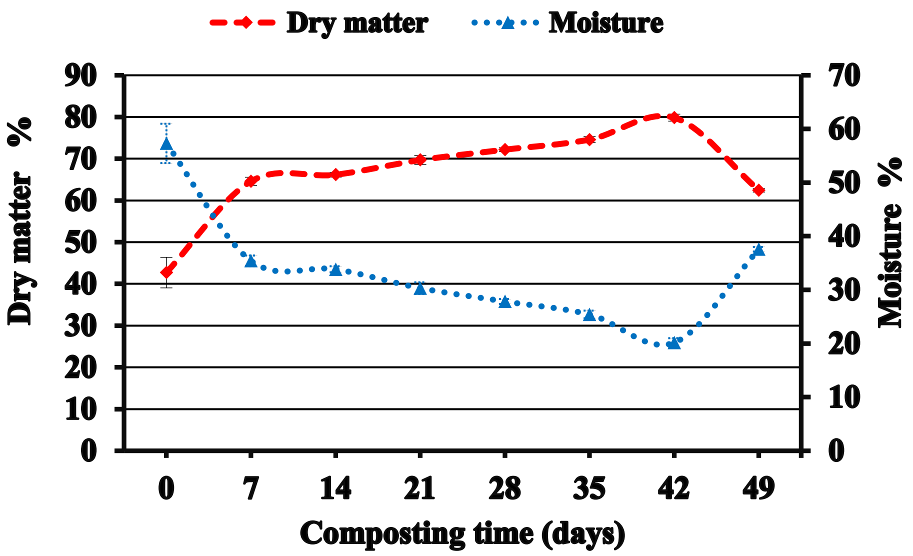

Figure 5. Moisture content and dry matter evolution during DSSS-GW composting

4.1.3 Bulk density

Bulk density evaluation is an essential tool in composting process control (Huerta-Pujol et al., 2010). Iqbal et al. (2010) indicated an importance of physical properties evaluation (bulk density, air-filled porosity, particle sizes) as it greatly influences the composting process. During the composting process, the BD_{dry} increased from an initial value of 0.19 g cm^{-3} to 0.40 g cm^{-3}, while the BD_{wet} increased from 0.36 g cm^{-3} to 0.56 g cm^{-3} (**Figure 6**). Agnew and Leonard (2003), reported that bulk density should lie between 500 and 900 kg m^{-3} at the end of the composting process. Moreover, evaluation of bulk density roughly provides degradation process of OM that can be seen from the volume of the mass under composting and its reduction during the process (Breitenbeck and Schellinger, 2004; Larney et al., 2000).

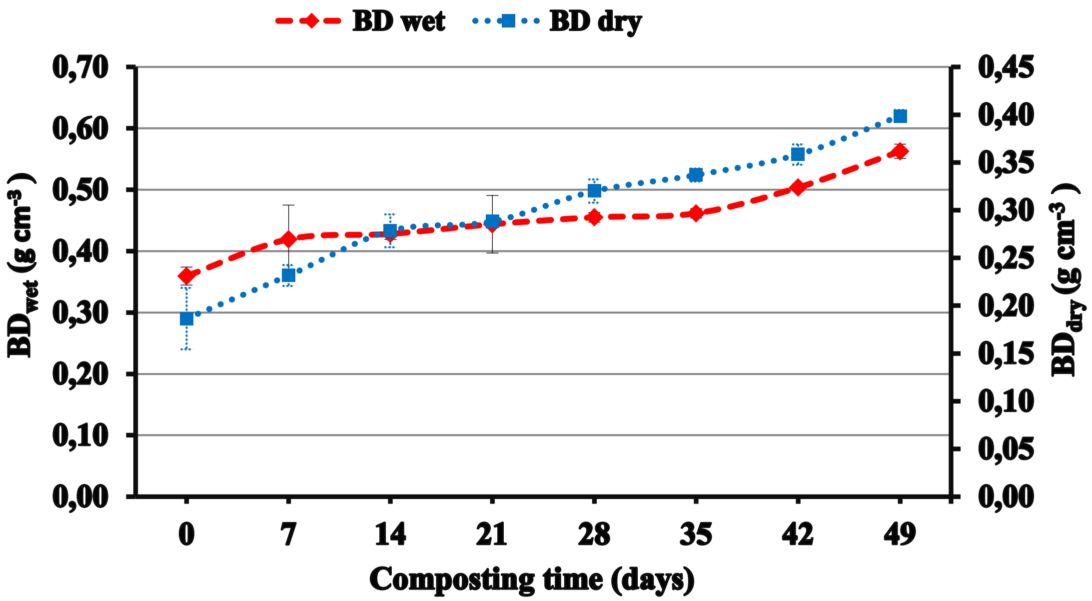

Figure 6. Changes in BDwet and BDdry the during DSSS-GW composting

4.1.4 pH variation

Changes in pH and EC were monitored during the composting of SS mixed with GW. During the first week of composting, the pH value decreased from 7.0 to 6.1 (**Figure 7**), which could be due to the production of organic acids, such as acetic acid and butyric acid from the activity of microorganism during the degradation of easily degradable compounds, or due to the loss of large quantities of carbon dioxide during the composting process (Huang et al., 2004). The decrease in pH could also be due to a reduction in the buffering capacity of the system deriving from volatilization of ammonia in the thermophilic phase (Eklind and Kirchmann, 2000b). Normally during SS composting processes, NH_3 volatilization takes place coincidentally with high temperatures and lower initial C/N ratios, thereby reducing the agronomic value of the compost and generating undesirable odours (Cáceres et al., 2018). NH_3 loss during composting ranges from 16 % to 76 % of TN loss (Barrington et al., 2002); advanced compost technology to reduce TN loss is thus needed. The GORE® Cover membrane inhibited NH_3 emissions as some NH_3 absorbed by the water layer created on the inner surface of the cover, and as a result nitrogen conservation was achieved through the transformation of reduced ammonia (Robledo-Mahón et al., 2019a; Sun et al., 2018).

The increase in pH from 6.1 to 7.3 (**Figure 7**) during the ensuing three weeks of composting (days 7 to 28 of the intensive phase) can be explained by the production of ammonia during ammonification and the mineralisation of organic nitrogen, which releases those bases already present in organic waste (El Fels et al., 2014). Sundberg and Jönsson (2008) showed that a higher oxygen concentration resulted in more rapid organic acid decomposition and thus a faster increase in pH. Baeta-Hall et al. (2005) and Hachicha et al. (2008) also showed that increase in pH may occur as a result of the mineralisation of OM and the decomposition of acids containing carboxylic and phenolic groups. After the observed increase, during days 28 to 42 of the maturation phase, the pH of the mixture stabilised at approximately 7.0 (neutral). Zenjari et al. (2006) and El Fels et al. (2014) showed that the stabilization of pH at near neutral values during composting results from the generation of humic substances, which act to increase buffering capacity. During the final week of the maturation phase (days 42 to 49), the pH decreased to 6.9, which remained constant at the end of composting; this was the result of compost maturation under aerobic conditions, the generation of HA (Hachicha et al., 2008; Michailides et al., 2011), and H^+ release as a result of nitrification by nitrifying bacteria during the later stages of composting (Cáceres et al., 2018). The final pH values of the composts were 6.9, which matches the recommended pH for matured compost products (Azim et al., 2018).

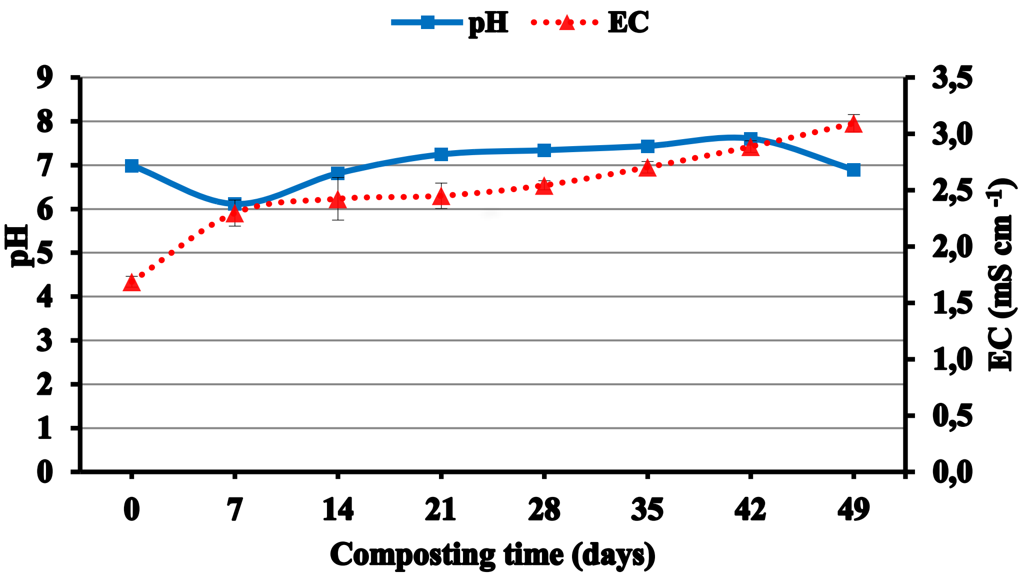

Figure 7. Evolution of pH and EC during DSSS-GW composting

4.1.5 Variation in electrical conductivity

The EC indicates the degree of compost salinity and the amount of ions in the final composting product; this parameter reflects the potential product phytotoxicity. EC increased sharply from 1.7 mS cm^{-1} to 3.1 mS cm^{-1} **(Figure 7)**. Huang et al. (2004) showed that an increase in EC could be due to the release of mineral salts, such as phosphate and ammonium ions, occasioned by the degradation of organic materials. Additionally, concentration effects resulting from OM loss and water evaporation due to high temperatures can increase EC (Cáceres et al., 2006).

A similar EC increase observed by Zhang et al. (2018) was attributed to water evaporation. The increased content of soluble salts indicated the progressive mineralization of OM (Cáceres et al., 2006). By the end of the composting process, the EC values of the final compost were 3.1 mS cm^{-1}, below 4 mS cm^{-1} which is commonly regarded as the limit for safe use on plants (Singh and Kalamdhad, 2014). This indicates that the compost is safe for most crop growth. However, due to the lack of a definite pattern in the changes in pH and EC over time, these parameters were not considered suitable for monitoring the composting process or for evaluating the stability of the composting material (Khan et al., 2009).

4.1.6 Losses of organic matter and ash content

Approximately 50 % of OM is fully mineralized (Bernal et al., 2017). The degradation of OM during composting can be estimated based on the loss of OM, and here, OM_{loss} increased sharply to 52.5 % within the first 35 days of composting (**Figure 8**) due to the greater availability of substances which are easily biodegradable by microorganisms. Then, it slowed down, reaching 56.7 % at the end of composting due to the depletion of easily biodegradable carbon and the synthesis reactions of the new complex polymerized organic substances (humification), processes which prevail over mineralisation during the maturation stage (Bernal et al., 2009). This transformation matched well with the remarkable reduction in OM, indicating a rapid decomposition rate. Expected OM loss during composting is between 30 % and 60 % and OM loss > 42 % may be accepted as an index value for mature compost (Raj and Antil, 2011). Based on this, the compost could be considered matured during 28 days. Composting is an exothermic aerobic process in which the OM in SS are decomposed and mineralized to CO_2 by microorganisms. As a result, a reduction in OM is always observed in SS composting. The higher OM_{loss} observed in this case could be due to the presence of two thermophilic temperature peaks and the longer thermophilic periods experienced during composting using this new composting technology (**Figure 4**). It has previously been found that high composting temperatures accelerate the growth of thermophilic microorganisms, allowing further biodegradation of OM (Meng et al., 2017).

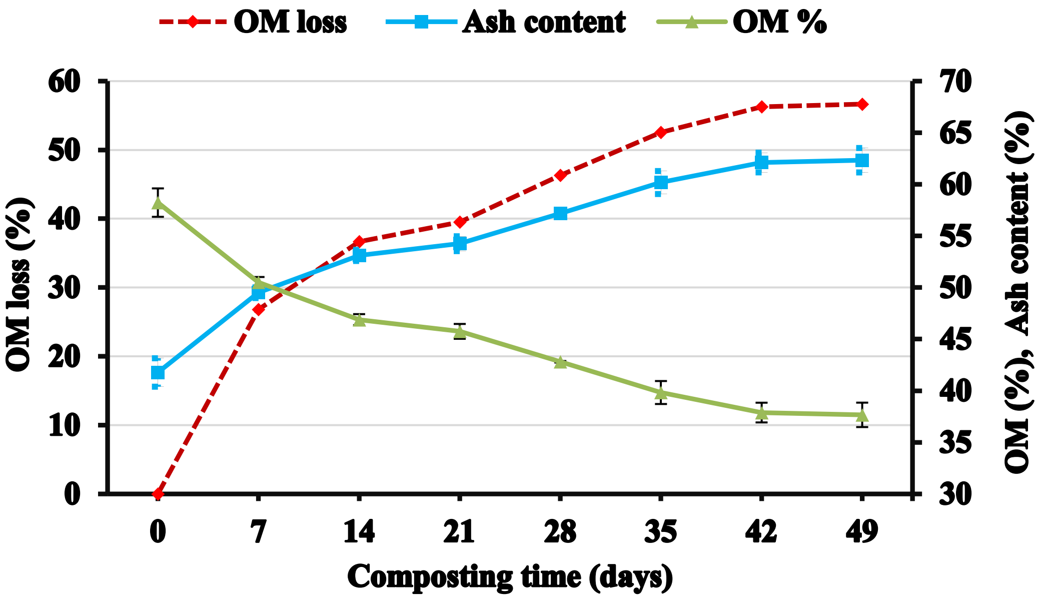

Figure 8. Evolution of OM_{loss} and ash content during DSSS-GW composting

During composting, there was an increase in ash content in the windrow. The ash content increased by 62.3 % after 49 days (**Figure 8**). Several authors have stated that increased ash content during composting results from the mineralisation of OM by microbes (Bernal et al., 2017; Rudnik, 2019), and this parameter indicates decomposition and mineralization of the OM, consequently reflecting the degree of stabilisation achieved during the process. A final compost with a higher ash content and lower OM is considered mature.

4.1.7 Water soluble carbon

WSC is one of the active parameters in defining compost stability. Sugars, organic acids, amino acids and phenols represented the major WSC in compost (Zmora-Nahum et al., 2005). The WSC substances content increased during the intensive phase and significantly decreased thereafter during the maturation phase (**Figure 9**), which is related to process of compost stabilization as suggested by several researchers (Karak et al., 2014; Said-Pullicino et al., 2007). Said-Pullicino et al. (2007) concluded that decrease of WSC in the final compost than the initial composting mixture could be due to the formation of complex organic substances produced from mineralization of soluble organic compounds and re-polymerization followed by condensation of organic compounds.

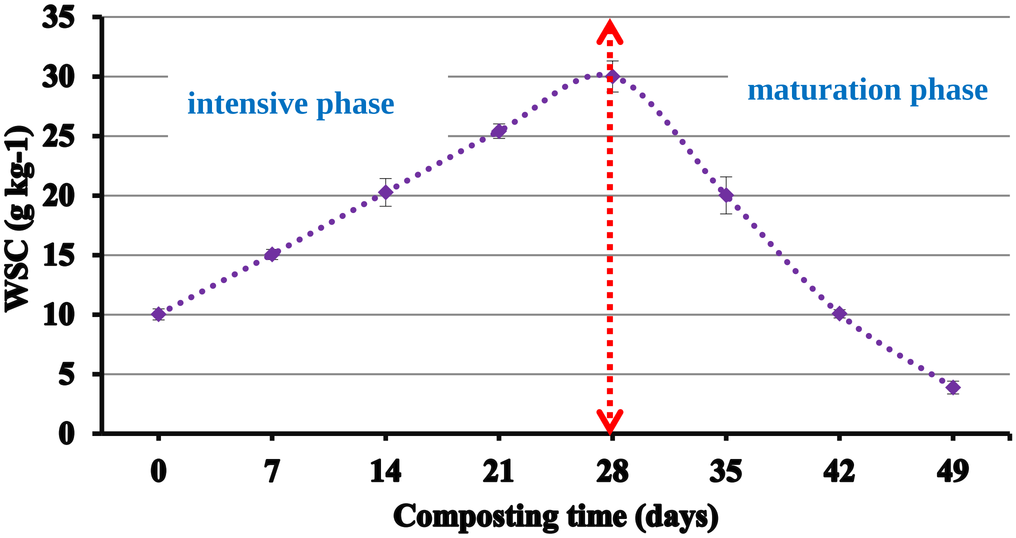

Figure 9. Evolution of WSC during DSSS-GW composting

In this context, Charest et al. (2004) suggested that the increase of WSC during the thermophilic phase would be the result of cellulose and hemicellulose biodegradation, while the posterior WSC decrease would indicated that microorganisms used it as a source of energy while acclimating to the degradation of more recalcitrant substrates. According to Bernai et al. (1998), a final WSC content <10 g kg^{-1} indicates that a compost is mature and stable. In the current study, the WSC content of the final compost were <10 g kg^{-1}.

4.1.8 Total carbon and oxidisable organic carbon

Degradation of TC depends on the availability of easily digestible material such as cellulose and hemicellulose. During the degradation process of the compost, microbes utilized 60-70 % of the carbon as CO_2 and incorporated (immobilize) only 30-40 % of the carbon into their body as cellular components (Adhikari et al., 2009; Yadav and Garg, 2009). Around 30 % of the available carbon was utilized by the microorganisms as a source of energy. Thus, changes in the TC content indicate compost maturity. Higher rate of TC content degradation indicates stabilized compost at the end of the composting. Change of TC and OXC during the progress of composting has been depicted in **Figure 10**. In general, TC decreased from 31.7 % to 20.3 % as the decomposition progressed, reflecting a notable mineralization of OM. According to Tumuhairwe et al. (2009), TC losses suggests pronounced microbial activity in the composting process. It can be shown OXC followed a similar pattern of the TC (**Figure 10**).

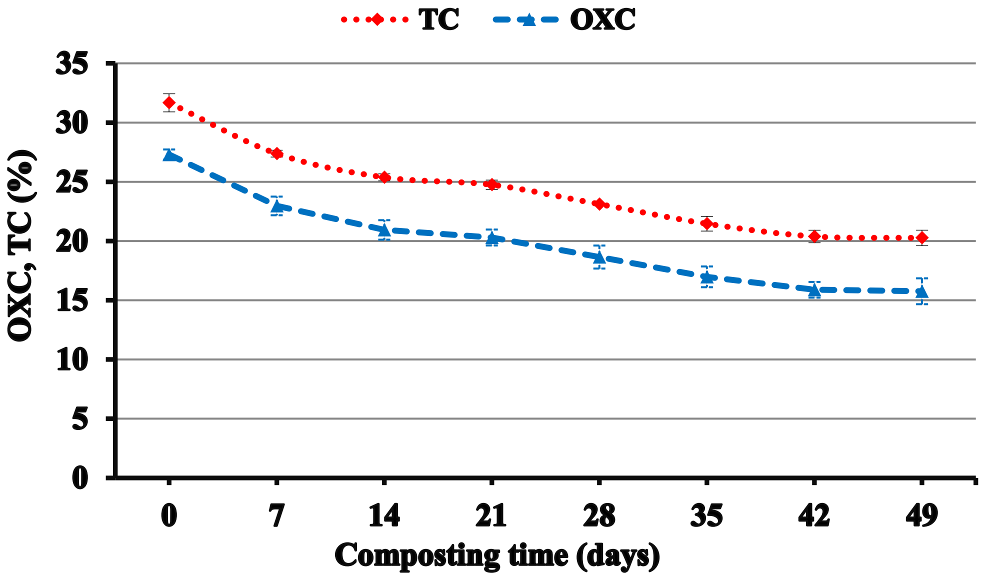

Figure 10. Changes TC and OXC content during DSSS-GW composting

4.1.9 C/N ratio

The C/N ratio is one of the most significant parameters used to evaluate the stabilization of compost. In this case, the C/N ratio decreased to reach a value of approximately 6.7 (**Figure 11**) after 49 days of composting. The continuous and rapid reduction in the C/N ratio was due to loss of TC and an increase in the proportion of TN due to the concentration effects resulting from the intense biodegradation of OM over the composting period (Huang et al., 2004). The TN value might have been also increased within the compost under the GORE® Cover membrane due to the contribution of the free-living N-fixing microorganisms that are generated during later stages of the composting process (de Bertoldi et al., 1983). Several authors have shown no agreement regarding the best C/N ratio for stable compost. However, Chefetz et al. (1996), Provenzano et al. (2001), and El Fels et al. (2014) all stated that values near to 10 indicate low microbial activity and result in a stabilized compost. According to Tomati et al. (2002), a C/N ratio of between 11 and 22 indicates stabilized compost. In the present study, the C/N ratio of the final compost reached a value of 6.7, which indicated that compost stabilization was achieved after 49 days of composting. Using this technology, the C/N ratio reduction rate was higher than the C/N ratios achieved in conventional methods of composting with the same substrate, in which C/N ratios were found reduce to 10.1 (El Fels et al., 2014), 11.7 (Raj and Antil, 2011), and 12.9 (Lahsaini et al., 2017). This is due to the forced bottom-up aeration system, which improves aeration conditions, accelerates decomposition, and results in a greater loss of carbon. The results thus agree with the findings of Sundberg and Jönsson (2008), who demonstrated that higher rates of aeration lead to faster decomposition in biowaste composting.

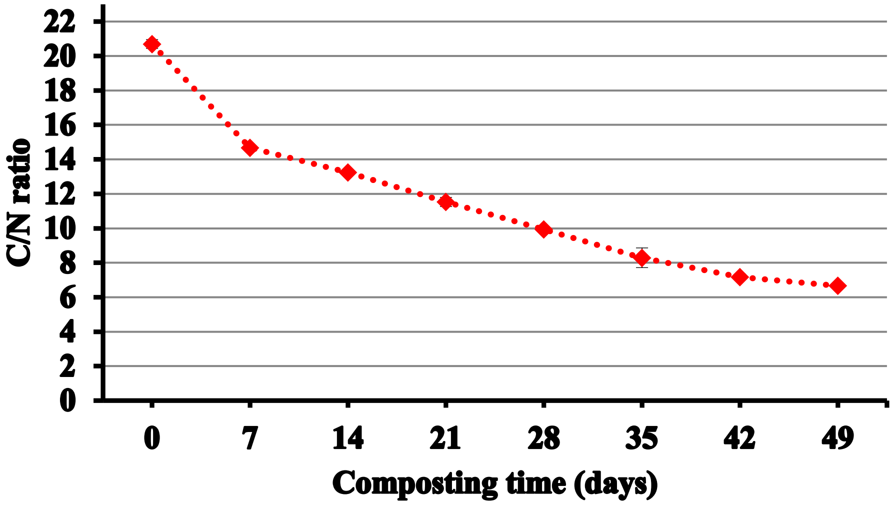

Figure 11. Evolution of the C/N ratio during DSSS-GW composting

4.1.10 NH_4^+/NO_3^- ratio

The NH_4^+/NO_3^- ratio ranged from 22.3 to 0.1 after 49 days of composting (**Figure 12**). The NH_4^+/NO_3^- ratio is used to indicate the maturity of compost; a ratio below the limit of 0.16 for composting materials such as SS, municipal solid waste, and pig slurry indicates maturity (Bernal et al., 2009). The recorded value of approximately 0.1 indicated that the final product reached maturity in this case, and that the substrate had been changed to yield a non-phytotoxic compost. Garcia et al. (1992a) and El Fels et al. (2014) showed that a ratio below 1.0 suggests that the final compost has reached maturity. For comparison, the critical limits for maturity during composting of SS by conventional composting technology have been found to be fulfilled after 6 months (El Fels et al., 2014), 220 days (Jouraiphy et al., 2005), and 135 days (Raj and Antil, 2011), while the new composting technology achieved maturity critical limits by the 49th day, implying that the conventional composting technology takes much longer to achieve the index value for maturity than the new composting technology, most likely due to the differences in aeration. These results indicate that this new technology is highly suitable for ensuring compost maturity in a shorter time frame due to improved aeration conditions and the flow of oxygen throughout the entire composting vessel using forced bottom-up aeration that increase intensity of nitrification processes (Wang et al., 2018).

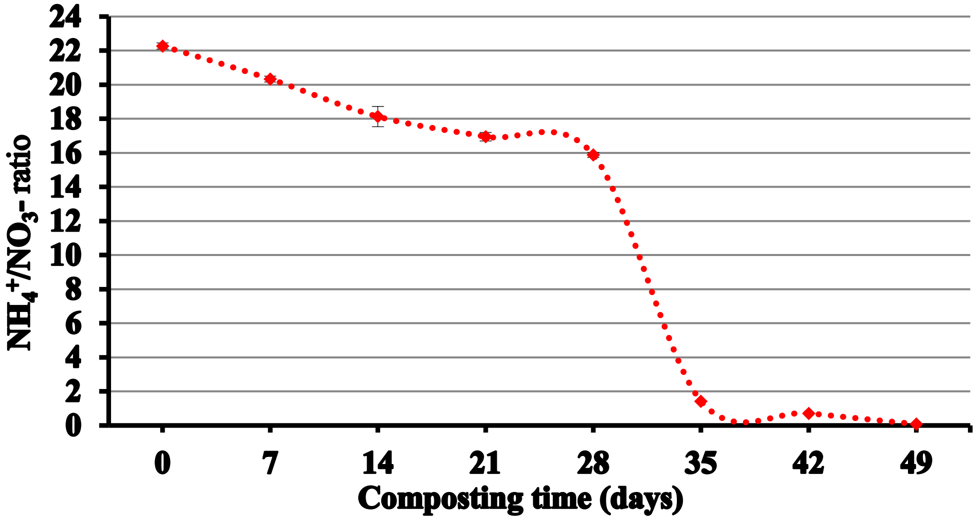

Figure 12. Evolution of the NH_4^+/NO_3^- ratio during DSSS-GW composting

4.1.11 Cation exchange capacity

During composting, CEC increased substantially from 51.3 cmol kg^{-1} to 170.3 cmol kg^{-1} after 49 days (**Figure 13**). CEC can be used to evaluate the degree of humification and the nutrient retention capacity of compost (Roig et al., 1988; Zhang and Sun, 2014a). According to Harada and Inoko (1980a), the CEC of a mature compost should be > 60 cmol kg^{-1}. In the current study, the CEC value of the final product was 170.3 cmol kg^{-1}, which indicates that the compost is mature. Zhang and Sun (2014a) confirmed that high CEC values are due to larger proportions of carboxyl functional groups in the substrates, while Lax et al. (1986) concluded that increases in CEC values during composting could be the result of the accumulation of compounds bearing a negative charge, such as lignin-derived products, carboxyl groups, and phenolic hydroxyl groups. Joseph et al. (2010) further showed that the presence of negatively charged groups caused the attraction, retention, and exchange of basic cations, which in turn could increase the CEC and nutrient-holding capacity of the compost. Any increase in CEC during composting is due to the humification process, during which carboxyl and phenolic functional groups are produced (Bernal et al., 2009). Humic compounds have high capacity to adsorb positively charged ions, which are then easily exchanged with other cations at the same adsorption sites (Azim et al., 2018).

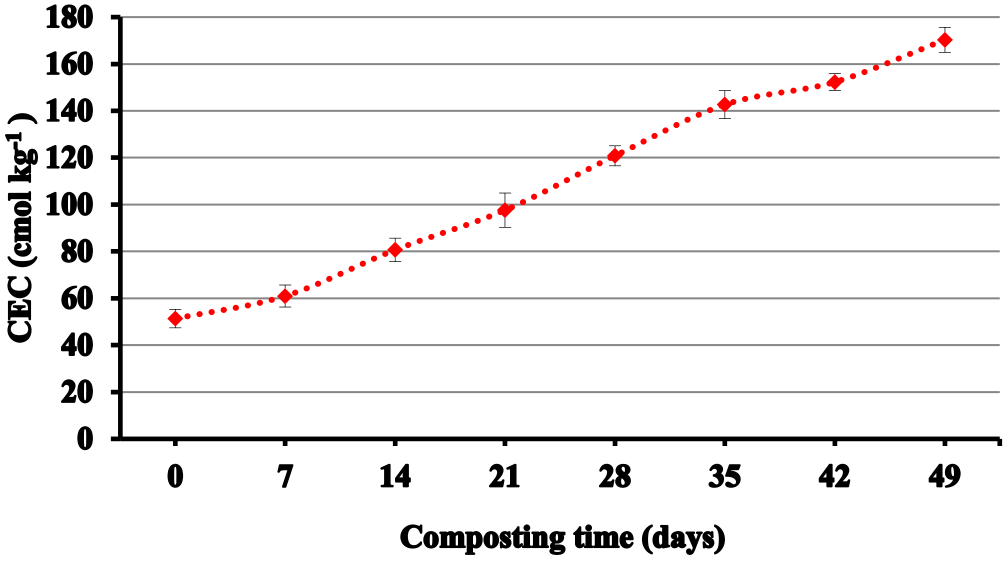

Figure 13. CEC evolution during DSSS-GW composting

4.1.12 UV-visible spectroscopic analysis

The evolution of the $E_{2/6}$, $E_{2/4}$, and $E_{4/6}$ ratios provides information on the degree of OM maturity during composting. The $E_{2/4}$, $E_{2/6}$ and $E_{4/6}$ ratios decreased sharply to 5.1, 25.4, and 3.8, respectively, after 49 days of composting (**Figure 14**). These decreases indicate the occurrence of the humification process. The decreases in the $E_{2/6}$ and $E_{4/6}$ ratios at the end of composting demonstrated a strong increase in the humification of OM (Lahsaini et al., 2017). Chen et al. (1977) used the $E_{4/6}$ ratio as a humification index, and Zbytniewski and Buszewski (2005) reported that low $E_{2/6}$ or $E_{4/6}$ ratios reflected high degrees of aromatic condensation and indicated a higher level of organic material humification. $E_{4/6}$ can also be used as an estimator of molecular weight (Chen et al., 1977). Guo et al. (2019) showed that during the thermophilic stage, rapid breakdown of lignin and quinine moieties occurred as the $E_{2/4}$ ratio decreased, indicating a higher degree of aromatic condensation, as well as a higher level of organic material humification. Therefore, a continuous decrease in $E_{2/4}$ reflects a high degree of aromatic condensation.

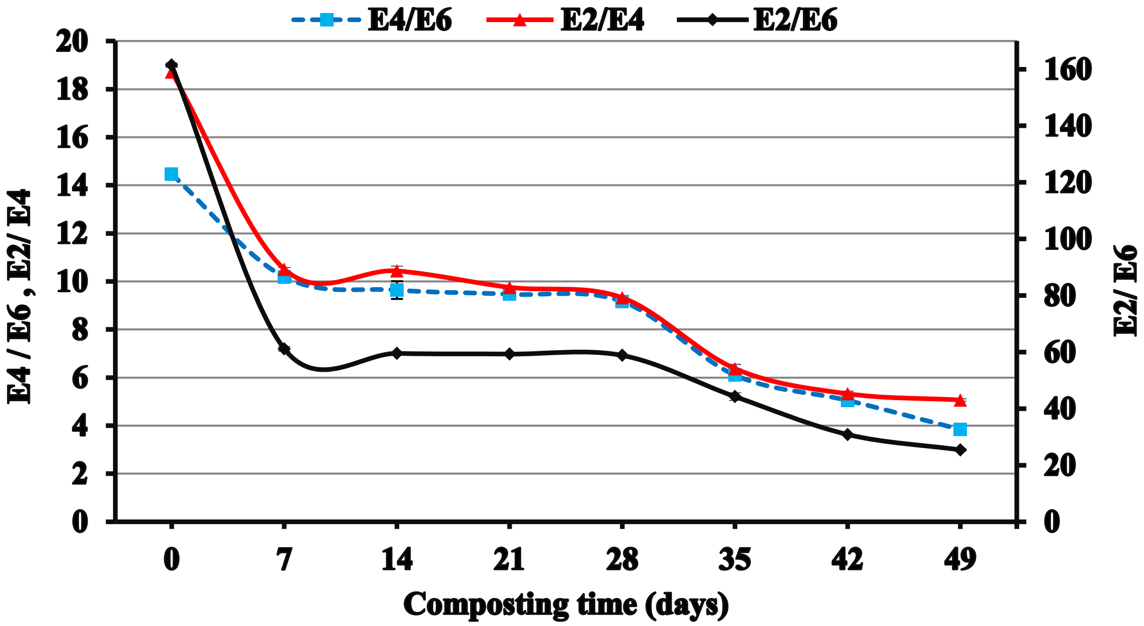

Figure 14. Evolution of the $E_{2/4}$ (280/472 nm), $E_{2/6}$ (280/664 nm), and $E_{4/6}$ (472/664 nm) ratios at different stages of composting DSSS-GW

4.1.13 Total phosphorus and total potassium

TP increased from 8.5 g kg^{-1} to 17.0 g kg^{-1}, and TK increased from 5.4 g kg^{-1} to 15.0 g kg^{-1} after 49 days (**Figure 15**) due to the decomposition of OM during the composting period. Phosphorous and potassium in OM are used by microorganisms for both growth and body metabolism. Singh and Kalamdhad (2012) showed that an increase in the potassium concentration is due to potassium assimilation and immobilization by microorganisms. Sharma and Yadav (2017) demonstrated that the increase in phosphorous concentration during composting is attributed to the concentration effect due to the higher rate of carbon loss that happens when OM is decomposed or mineralized. Zhou et al. (2018) showed that phosphorus and potassium are not volatile, so increased concentrations of these elements are caused by a loss of composting mass due to the decomposition of OM. Phosphorus and potassium are crucial nutrients for plant growth and, as a result, are important indicators for assessing compost maturity and the nutritive value of compost as an organic fertilizer (Chang et al., 2007).

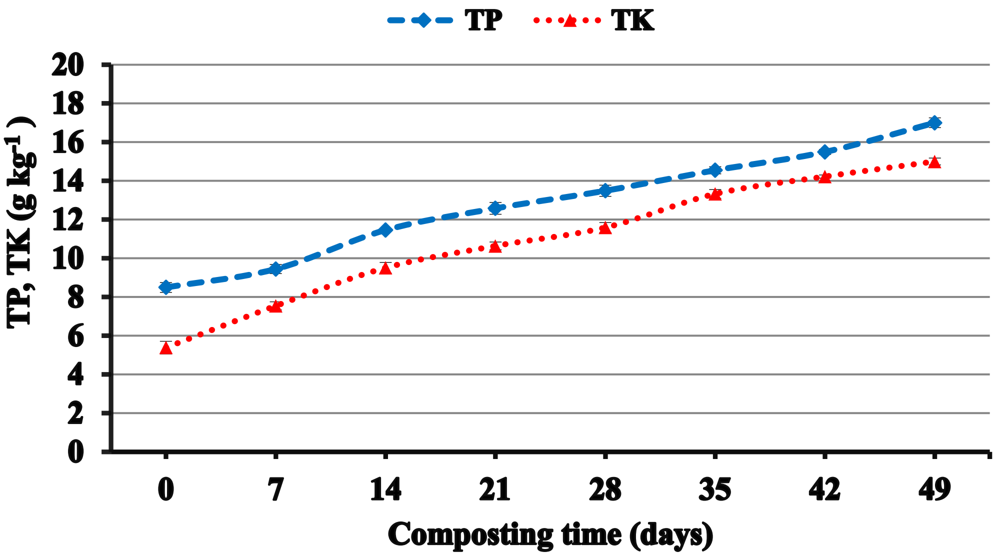

Figure 15. Macronutrients (TP and TK) evolution during DSSS-GW composting

4.1.14 Seed germination index

The results of the germination tests are depicted in **Figure 16**. The GI decreased (48.7 % to 44.4 %) in the first week of composting because of the fast degradation of OM, leading to the production of low-molecular-weight short-chain volatile fatty acids and the production of ammonia (NH_3) from microbial ammonification, which can inhibit seed germination (Barje et al., 2013). This result is similar to those reported by Awasthi et al. (2017) and Zhang and Sun (2014), who found that the GI decreased in the early phase of composting. After one week, GI increased to 125.5 % at the end of composting (**Figure 16**). The increase was caused by the degradation of toxic materials by microorganisms (Yang et al., 2013). At a GI value <25, the substrate is considered to be very phytotoxic; if the GI value is 26 < GI < 65, then the substrate is indicated as phytotoxic; if the GI value is 66 < GI < 100, then the substrate is indicated as nonphytotoxic; and if GI > 101, then the substrate is stable and can be used as a fertilizer and phytonutrient-phytostimulant (Aggelis et al., 2002). Tiquia et al. (1996) demonstrated that a GI of more than 80 % indicated that the compost was nonphytotoxic and mature. El Fels et al. (2014) showed that GI values exceeding 100 % can generally be explained by a substantial reduction in phytotoxic compounds and the enrichment of compost by stable OM and humic substances during the maturation phase. The phytotoxicity of compost is due to the presence of high levels of NH_4^+-N, soluble salts, organic acids, or pH (Wang et al., 2017). Zhang and Sun (2018) showed that the use of an additive substrate could induce an increase in GI due to the adsorptive and buffering capacity of the additive and the enhancement of OM degradation, which could help reduce the concentrations of NH_4^+-N and volatile fatty acids. Meng et al. (2017) showed that an increase in GI could be caused by the degradation of toxic materials.

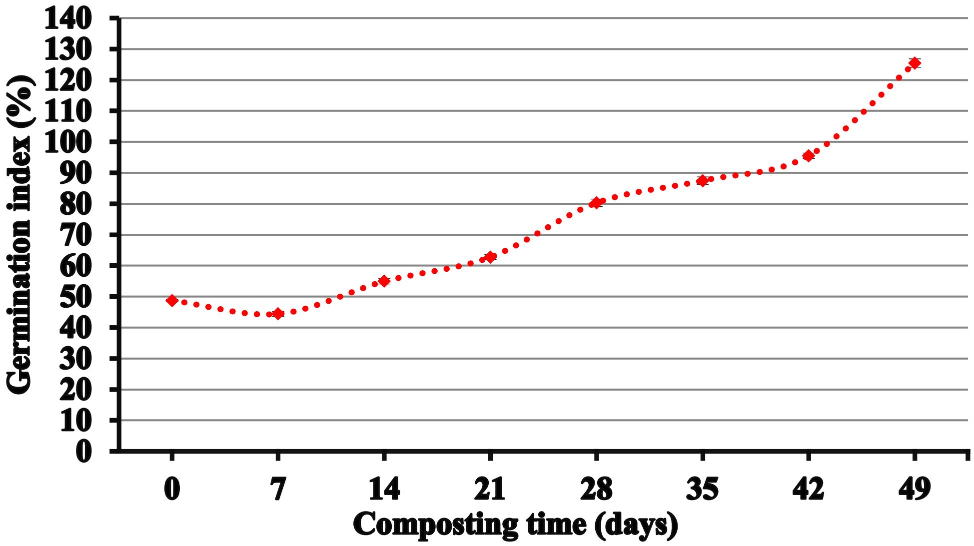

Figure 16. Evolution of germination index during DSSS-GW composting

4.1.15 Principal components analysis

PCA is one of the most powerful and common techniques used for data compression, to capture the main features in the data set and to extract information from multivariate data sets, where patterns can be hard to find (Campitelli and Ceppi, 2008). The results of the PCA run between the physico-chemical parameters during composting are reported in **Figure 17**. The PCA analysis reduces multidimensional relationships between parameters to two principal components that explain 88 % of the total system variability. The first component (PC 1) explains 74 % of the total variability and separates two groups of parameters. The first group has $E_{4/6}$, $E_{2/4}$, $E_{2/6}$, moisture, C/N ratio, TC, OM, OXC, and NH_4^+/NO_3^- ratio which are closely correlated by evolving similarly, i.e. they decrease toward the end of composting. The second group contained parameters such as pH, EC, Ash, TK, TP, CEC, TN, GI, NO_3^-, dry matter, BD_{dry}, and BD_{wet}, which are also correlated by evolving similarly, but increasing during composting. The second component (PC 2) explains 14 % of the total variability and consists of NH_4^+, and WSC, which they have same patterns as they increase during the intensive phase and decrease in maturation phase. These results reveal and visualize in a clearer way the relationship between these different physico-chemical parameters during composting and indicate that parameters that were grouped into the same PC exhibited similar behavior.

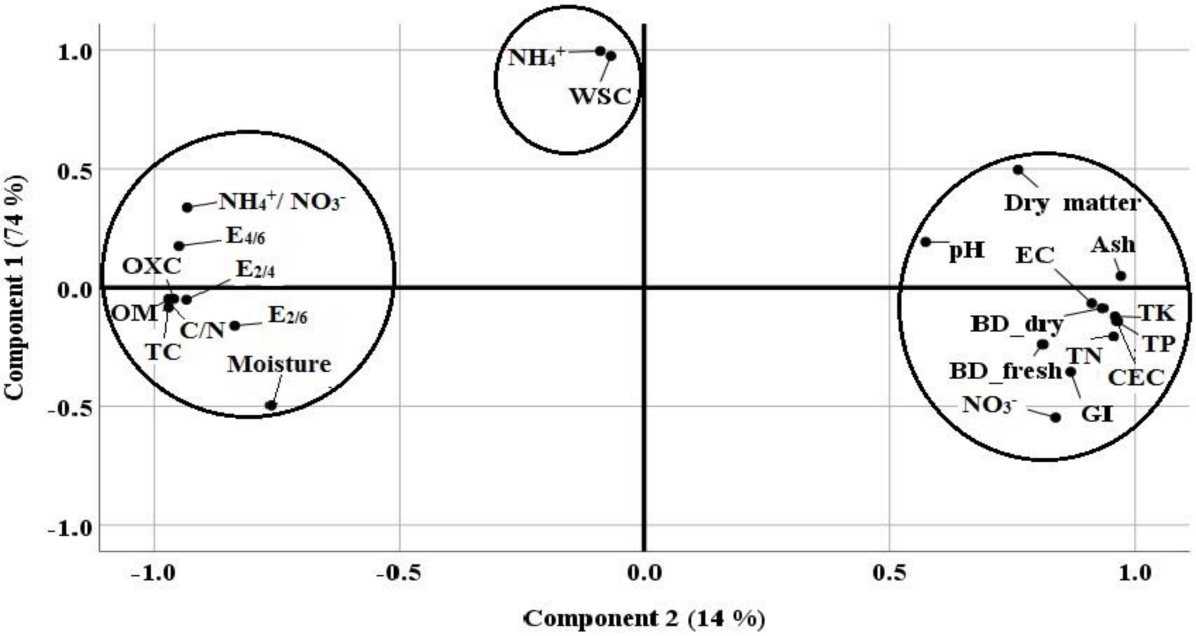

Figure 17. Principal component analysis of the physico-chemical parameters measured during DSSS-GW composting

4.1.16 FTIR spectroscopic analysis

FTIR spectrometry is a non-destructive qualitative or finger printing method used to characterize the principal classes of chemical groups that make up OM (Chen, 2003), in order to determine the changes seen in OM during composting (El Mezouari El Glauoi et al., 2018). The FTIR spectra thus derived are presented in **Figure 18**. A very wide band located at 3460 cm $^{-1}$ was detected during the process that can be attributed to the stretching vibrations of OH in hydroxyl groups. El Mezouari El Glauoi et al. (2018) attributed this band to the hydrogenic vibrations of OH groups of alcohols and to phenols or to the OH of the carboxyl groups (COOH). The interaction of hydroxyl groups with hydrogen bonds explains the wide profile of this band and the fact that the wave number is lower than that of a free OH hydroxyl (El Fels et al., 2014). El Fels et al. (2015) reported that the 3000–2800 cm^{-1} region reflects the hydrophobic properties of aliphatic OM. The peak observed at 2950 cm^{-1} is due to methylene C-H aliphatic groups. The intensity of these bands decreases during composting, which is a good indication of the microbial oxidation of OM, especially the aliphatic compounds, during composting (El Fels et al., 2014). The peak intensity ratio of the bands at 2920 and 1640 cm^{-1} (designated as aliphatic methylene and aromatic C, respectively) is suggested as a meaningful parameter for tracking the biological stability of OM during landfill stabilization (Lü et al., 2018; Wu et al., 2011).

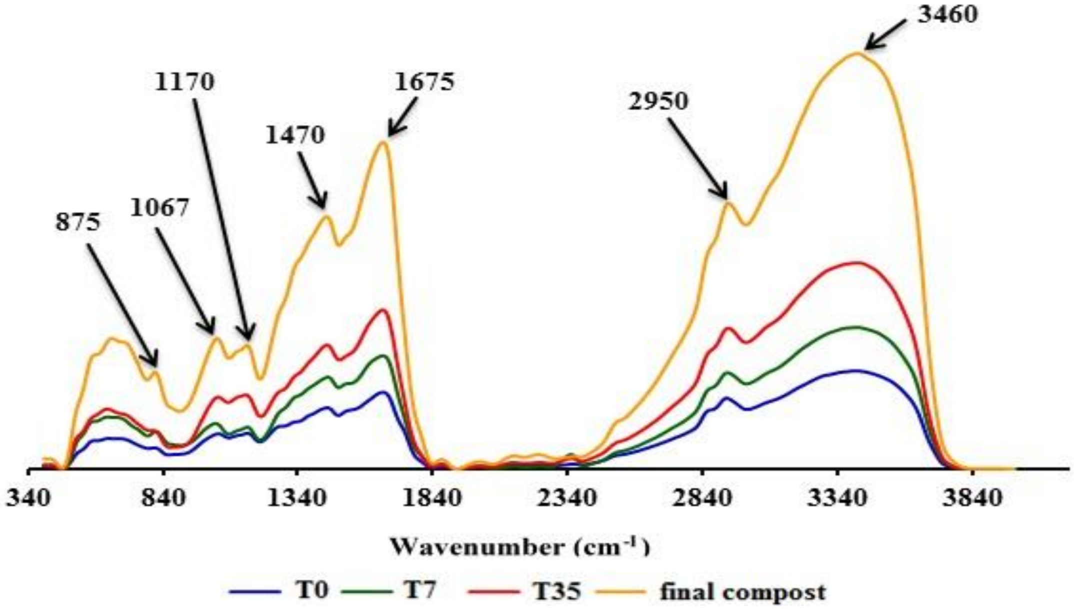

Figure 18. FTIR spectra of DSSS-GW at various stages of composting. T0: day 0; T7: day 7; T35: day 35; and final compost

The band located at approximately 1675 cm^{-1} showed an increase during composting. El Fels et al. (2014) attributed the region from 1750-1700 cm^{-1} to νC=O stretching of the carboxyl group in its nonionized form, and the peaks that appear at approximately 1650 cm^{-1} to νC=O of ionized carboxyl (COO-). According to Kaiser and Ellerbrock (2005), the band at 1650 cm^{-1} is associated with the vibrations of νC=C aromatic structures and the antisymmetric vibrations of COO-. In a biological specimen study, Maquelin et al. (2002) further attributed the 1675 cm^{-1} band to the amide I band components resulting from the antiparallel pleated sheets and β–turns of proteins. Pajączkowska et al. (2003) observed absorbance bands between 1680 and 1630 cm^{-1} and attributed these to νCOO- in the case of HA extracted from sludge at different stages of treatment. An increase in this band thus provides information on the evolution of humification of OM during composting. Smidt et al. (2005) showed that HA display a strong band at 1640 cm^{-1} that increases in intensity during the composting process. The continuous increase in the humic acid content has a positive impact on this band. The bands at 1470 cm^{-1} are attributed to aromatic benzene compounds. Droussi et al. (2009) attributed the region from 1442–1472 to C=C stretching vibrations arising from aromatic components. An increase in the intensity of the aromatic absorbance indicates progress in the humification process (El Fels et al., 2014; El Mezouari El Glauoi et al., 2018). The 1170 cm^{-1} band can be attributed to C-O vibrations. While analysing fulvic acids, Zhou et al. (2014) attributed the band at 1170 cm^{-1} to C-O stretching in alcohols, and El Fels et al. (2014) reported that the change in intensity of the observed bands at 1740 and 1155 cm^{-1} indicated the presence of a carboxyl functional group in the first, while that seen in the second is probably a characteristic of an ester bond. The 1160 cm^{-1} band is assigned to the asymmetric stretching of C–O–C of glycosidic linkages (Bower and Maddams, 1989). Ouatmane et al. (2000) and Wu et al. (2011) attributed the 1165 cm^{-1} band to C–O–C stretching; El Mezouari El Glauoi et al. (2018) showed that the latter band could also be used as an indicator of the biodegradation process. The 1067 cm^{-1} band remained constant during composting. The region at 1080–1030 cm^{-1} was attributed to the C–O stretching of polysaccharides or polysaccharide-like substances and the Si–O asymmetric stretch of silicate impurities (Filip and Bielek, 2002; Smidt et al., 2002). Aguelmous et al. (2016) observed bands located at 875 and 713 cm^{-1}, which suggest the presence of calcium carbonates. The band at 875 cm^{-1} could be attributed to the C–O out of plane vibration of carbonates. Reig et al. (2002) attributed the 875 cm^{-1} band to the C-O out-of-plane bending of carbonate. The evolution of each functional group reflects the advanced stage of humification of OM during composting.

4.2. Aerated static windrow with GORE® Cover membrane

4.2.1 Temperature evolution

Temperature monitoring is widely recognized as an excellent indicator for evaluating the degree of composting success and compost stability, since compost windrow temperature is related to the rate of decomposition and to microbial activity during composting (Tiquia, 2005). Temperature variation during composting (**Figure 19**) showed a classic pattern exhibited by composting systems. Three distinct phases were observed: short initial mesophilic-heating phase (< 1 day), thermophilic (stabilization) phase (1–24 days), and cooling/maturing phases (25–30 days):

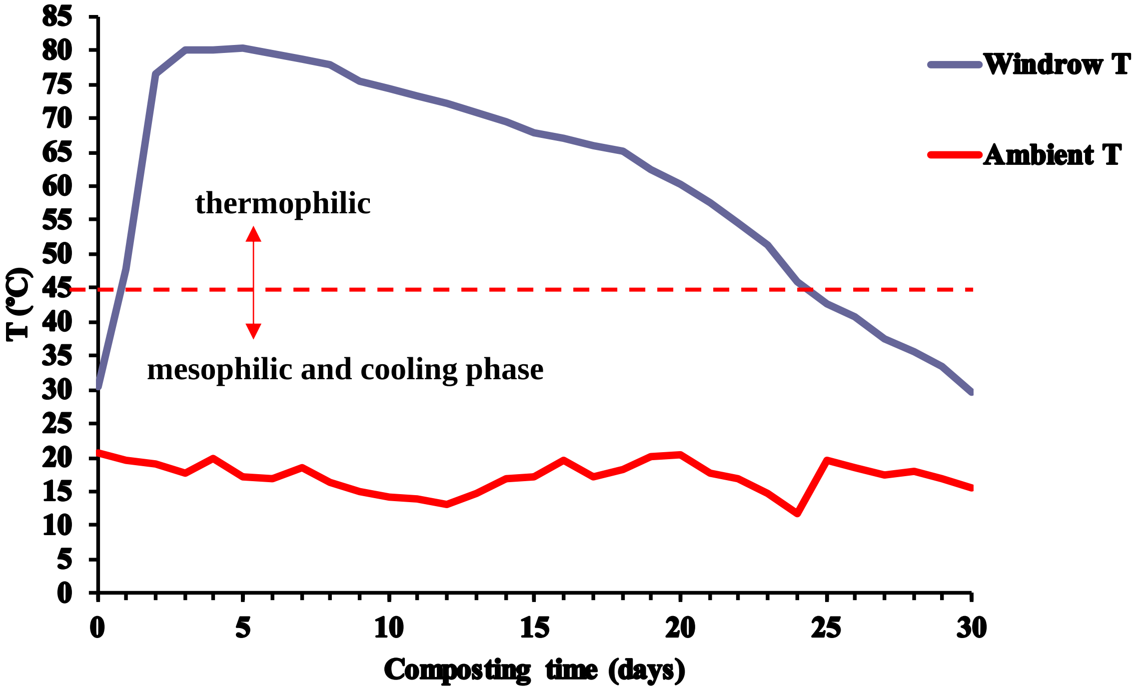

Figure 19. Evolution of the temperature during GW composting

(i) Short initial *mesophilic-heating phase* (25–45 °C), where the temperature increased rapidly from ambient values to approximately 45°C and lasted less than one day. During the initial mesophilic phase, mesophilic bacteria and fungi broke down the easily degradable organic compounds and produced H_2O, NH_3, CO_2, organic acids and heat (Bernal et al., 2009).

(ii) *Thermophilic phase* (> 45 °C), during which the temperature remains above 45°C, exceeded the tolerance limit of mesophilic microorganisms and replaced by the thermophilic microorganisms and rapidly attained maximum value about 80.2 °C within three day.

Temperatures were higher and increased more quickly than those observed during composting of the same substrate when a traditional method was used (Khalil et al., 2008). The micropositive pressure environment that resulted from the combination effects of the forced bottom-up ventilation system and the GORE® Cover membrane on top of the windrow contributed to an even supply of sufficient oxygen for microorganisms to degrade OM. Ultimately, the microbial degradation of easily-decomposable OM was enhanced, the decomposition rate increased, and a large amount of metabolic heat was produced. Combined, these resulted in temperature increases inside the windrows, and attained the peak temperatures more quickly and maintained for a long time.

The application of this innovative technology resulted in higher thermophilic temperatures that are characteristic of the most active phase of composting. The temperature remained high and steady in the thermophilic temperature range for 24 days because the GORE® Cover membrane completely enclosed the windrow and retained heat, which was consistent with results (González et al., 2016). The maintenance of high temperature for several consecutive days favors the growth of microorganisms that are responsible for the decomposition of lignocelluloses and formation of humus precursors which indicate an efficient composting process (Senesi, 1989). In fact, to produce compost free from harmful pathogens organisms and weed seed, the composting temperature must remain in the thermophilic phase (>45 °C) for at least three consecutive days (Bernal et al., 2009). In this study, the thermophilic phase lasted 24 days, thus composts generated by this technology met the disinfection requirements.

(iii) The *cooling* or *maturing phase* began on day 25 when the temperature of the windrow began to decline regularly below 45°C to reach ambient temperature on the day 30th to indicate that the process was complete. The falling temperature was due to the depletion of easily degradable OM in the mesophilic and thermophilic phases and remaining compounds that were resistant to degradation (e.g., lignin). This phase was dominated by the humification process that consisted of the condensation and polymerization reactions of the organic compounds which were present to yield more stabilized compounds known as humic-like compounds (Senesi, 1989).

This technology enabled more rapid decomposition of organic waste because of higher peak temperatures in addition to more rapid attainment and prolongation of thermophilic temperatures. Thus, high decomposition rate, accelerated GW degradation and reduced composting period result in lower operating costs for composting plants. As stated by Khalil et al. (2008), conventional composting requires 90–270 days to produce a mature product, however, the proposed new technology shortened the GW composting time and produced mature compost in 30 days only.

4.2.2 Moisture and dry matter

The MC is crucial parameter during the composting process and plays an important role in microorganism metabolism, which directly influences the composting progress and compost quality. Loss of moisture during the composting process is recognised as an index of the decomposition rate as the heat production that accompanies decomposition causes vaporisation (Liao et al., 1997). After the two weeks of composting, the MC decreased from the initial 60.5 % to 51.3 % (**Figure 20**) due to a rapid temperature increase during the composting process. Thereafter, the moisture remained relatively stable at an approximately optimum value of 50 % until the end of the composting process. This is most likely because the composting technology used was a closed system and much of the metabolic water generated as a result of decomposition, condensed on the inner surface of the GORE® Cover membrane, dropped back into the compost and replaced the water that was initially lost through evaporation. The DM varies inversely with respect to moisture. Jiang et al. (2011) found that a high initial MC reduces the NH_3 emissions but delays the compost's maturation.

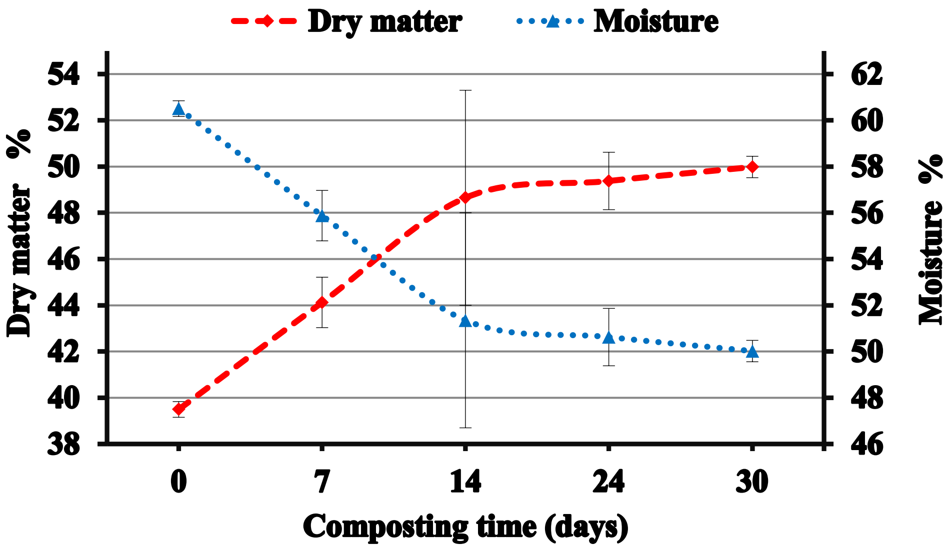

Figure 20. Moisture content and dry matter evolution during GW composting

During composting, the water content tends to increase due to the release of metabolic water by microorganisms that decompose OM in the presence of oxygen. However, Zvomuya et al. (2005) reported that during forced ventilation, the heat and high temperature was taken by the air flow causes greater water removal within the matrix pores and the moisture decreases. However, Shan et al. (2018) showed that a great loss of water could be caused by intensive microbial decomposition activities. The optimal moisture value of approximately 50-60 %, depending on the physical structure and the particle size change during composting, is necessary to increase the rate of decomposition by microorganisms which use water for nutrient solubilisation and cell protoplasm (El Fels et al., 2014). Liang et al. (2003) reported that MC of approximately 50 % was a minimal condition for rapid microorganism activity and should be kept in the range of 30–60 % MC for better microorganism activity. The results obtained in this study indicate that using an ASW with GORE® Cover membrane kept the moisture in optimal conditions by retaining the water within the compost itself and, consequently, increasing the microbial activity and the rate of OM decomposition by microorganisms. Moreover, the vapor generated by the rise in temperature inside the windrow due to microorganism activity is responsible for the breakdown of tannin and/or lignin (tough organic material) (Sudharsan Varma and Kalamdhad, 2014). At the end of the composting process, the compost had MC value which exceeded 30 %, which indicates its stability (Rich et al., 2018).

4.2.3 Bulk density

Both BD_{wet} and BD_{dry} evaluations are important, as they influence the composting process and compaction of the compost. The BD_{dry} increased from an initial value of 0.36 g cm^{-3} to 0.75 g cm^{-3} after 30 days of composting, while the BD_{wet} increased from 0.55 g cm^{-3} to 0.90 g cm^{-3} (**Figure 21**). These results are confirmed by other authors (Karak et al., 2015, 2013). As compost matures, its organic carbon content decreases due to chemical and biological conversions of organic carbon to carbon dioxide. As this occurs, the structural support provided by various carbon compounds degrades and collapses, causing the remaining inorganic materials (salts and metals) to compact. The result is a higher unit of mass per unit volume in aged, composted material than that of the original feedstock. Khater (2015) observed a negative correlation between the bulk density and total OM of the compost (R^2 = -0.89). As particle size is reduced and ash content increases by decomposition, BD_{dry} increases with composting progress (Raviv et al., 1987). The BD_{dry} of compost is significant when the compost comprises a large proportion of the potting media and as bulk density increases, drainage and the air-filled porosity of the growing media (e.g., potting media) are reduced, and the water holding capacity is increased.

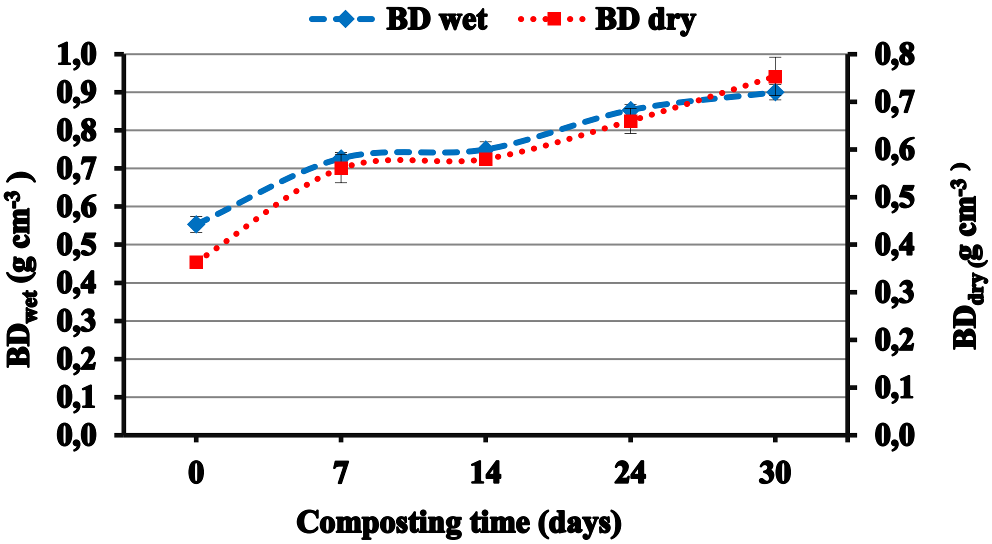

Figure 21. Changes in BDwet and BDdry the during GW composting

However, the measured bulk density may decrease as the material dries out during composting due to evaporation and loss of water (Day et al., 1998). The normal BD_{dry} is in the range of 100 kg m^{-3} to 400 kg m^{-3}, whereas BD_{wet} is normally 500 kg m^{-3} to 900 kg m^{-3} (Agnew and Leonard, 2003). Azim et al. (2018) stated that in composting, the porosity (pore space) of the windrow must allow decomposition to occur under aerobic conditions and small particle size decreases the number of large pores and increases the oxygen needed to diffuse through small pores. Higher bulk density indicates an increase in mass and a decrease of porosity and air volume. However, low BD_{wet} can imply extreme substrate aeration and, indirectly, a decrease in the available water fraction (Nappi and Barberis, 1993).

4.2.4 pH variation

The pH value of the compost is one of significant parameters used to assess compost maturity (Azim et al., 2018). Therefore, changes in pH were monitored during the composting GW (**Figure 22)**.

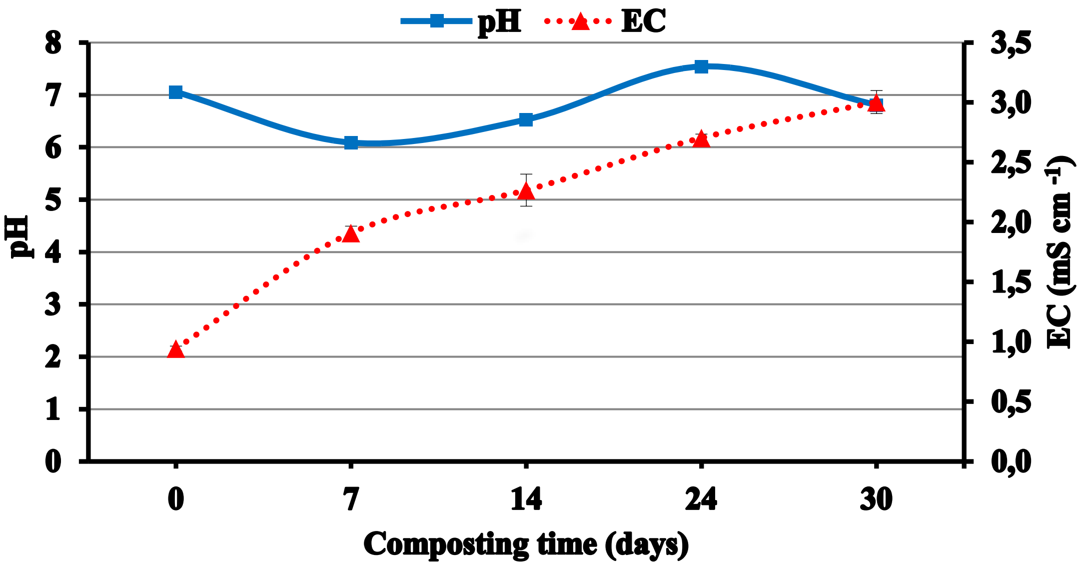

Figure 22. Evolution of the pH and EC during GW composting

In the first week of composting, the pH value declined from 7.1 to 6.1 (**Figure 22**). This can be due to intense activity of mesophilic microbes and OM decomposition with the production of organic acids (such as acetic acid and butyric acid) under high temperatures, or due to the loss of large quantities of CO_2 during the initial stage of the decomposition of lignocellulose in the compost windrow (Azim et al., 2018). In addition, initial volatilization of ammonia following an increase in temperature during the thermophilic phase reduced the buffering capacity of the system and reduced the pH (Cáceres et al., 2018).

The increase in pH from 6.1 to 7.5 (**Figure 22**) for the next 17 days (days 7–24) was attributed to the generation of ammonia during ammonification and mineralization of organic N due to microbial activities (Huang et al., 2004). Vergnoux et al. (2009) demonstrated that higher oxygen concentration causes faster decomposition of organic acids, and thus a quicker rise in pH. Alternatively, Tello-Andrade et al. (2015) and Hachicha et al. (2008) showed that the rise of pH may occur as a result of the metabolic degradation of organic acids.

At the end of the composting period, pH decreased to 6.8 because of compost maturation under aerobic conditions, synthesis of HA which worked as pH buffers (Amir et al., 2005), and hydrogen ion release from the nitrification process during later stages of composting (Eklind and Kirchmann, 2000b). Similar results were reported for organic waste composting (Gao et al., 2010b). The final pH value of the composts was 6.8, which is recommended for a matured compost (Zhang and Sun, 2014b).

4.2.5 Variation in electrical conductivity

The EC indicates potential phytotoxicity on plant growth and is a very useful parameter as it reflects the degree of compost salinity and the amount of ions in the composting material (Gao et al., 2010b). However, the EC value depends on the rate of OM decomposition which leads to accumulation of different ionic species (Chan et al., 2016). As illustrated in **Figure 22**, EC values increased sharply from 0.9 mS cm^{-1} to 3.0 mS cm^{-1} during composting, which could be attributed to the release of mineral salts, such as ammonium and magnesium ions and/or to sulphates and phosphates (Huang et al., 2004) and concentration effect due to water loss from the high temperature (Zhang et al., 2016).

High salinity level in the compost can damage plant roots, affect nutrient uptake, limit plant-available soil water, and inhibit seed germination (Arslan et al., 2011). EC values of the final compost was 3.0 mS cm^{-1}, below 4 mS cm^{-1} which is commonly regarded as the limit for safely growing plants (Singh and Kalamdhad, 2014). This indicated this compost could be applied to agricultural cultivations.

4.2.6 Losses of organic matter and ash content

In a well-managed process, approximately 50 % of the biodegradable OM becomes fully-mineralized, mostly due to the degradation of protein, cellulose and hemicellulose, which are used by microorganisms a source of energy (Chefetz et al., 1998).

During the composting process, OM is degraded progressively by microbes and converted to carbon dioxide, water, ammonia and new microbial biomass. The loss of OM at each time interval during composting was calculated in order to quantify the rate of OM mineralization (**Figure 23**). OM_{loss} increased sharply with time to 46.9 % within the first 14 days of composting (mesophilic and thermophilic phases) due to the greater availability of substances that are easily biodegradable by microorganisms. Then, it increased slowly during the cooling and maturation stages to reach 51.9 % at the end of composting. This was due to the depletion of easily biodegradable carbon and the synthesis reactions of new complex and polymerized organic compounds (humification), processes which prevail over mineralization during the maturation stage (Bernal et al., 2009). The expected losses of OM during composting are between 30 % and 60 %, and an OM loss >42 % is acceptable as a value for mature compost (Raj and Antil, 2011). According to this index, the compost matured in 14 days.

The higher OM_{loss} could be due to longer thermophilic periods (24 days) and higher temperature during composting because of this new composting technology (**Figure 19**). It has been found that high composting temperature accelerated the growth of thermophilic microorganisms for further biodegradation of lignin, cellulose and hemicellulose (Reyes-Torres et al., 2018).

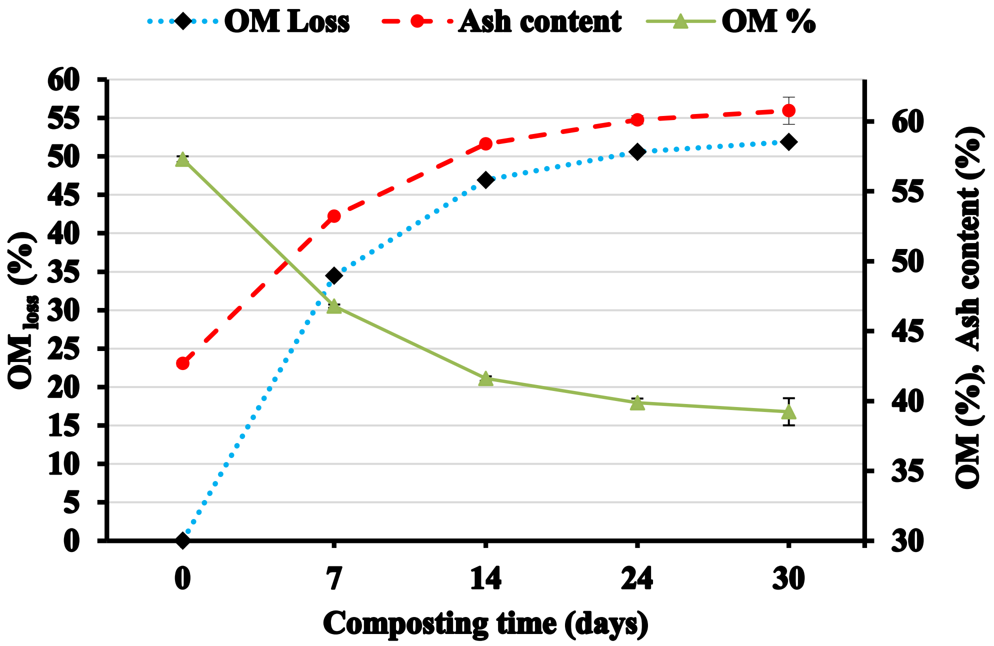

Figure 23. Evolution of OM_{loss} and ash content during GW composting

Changes in ash content as a function of time is displayed in **Figure 23**. During composting, the ash content increased by 60.8 % after 30 days of composting. Several researchers have reported that increased ash content during composting resulted from the mineralization of OM by microbial degradation (Kalamdhad et al., 2009; Rihani et al., 2010; Tognetti et al., 2007) Thus, this parameter measured the degradation and mineralization of OM and indicated stabilization during the composting process (Hsu and Lo, 1999).

4.2.7 Water soluble carbon

The WSC is one of the most readily available carbon fractions, it represents a source of carbon compounds that is easily biodegradable by microbes and consequently, it contributes to maintaining a high level of microbial activity. In this case, the WSC increased gradually from 6.2 g kg^{-1} to 33.7 g kg^{-1} during the first 14 days of composting (**Figure 24**). Thereafter, the WSC dropped sharply to reach 8.9 g kg^{-1} at the end of composting. An initial increase followed by a decline in the WSC during composting has been observed by several authors (Tumuhairwe et al., 2009; Goyal et al., 2005; Charest et al., 2004). However, a continuous decrease in WSC was reported by Leita and De Nobili (1991) and Inbar et al. (1993) for MSW and cattle manure, respectively.

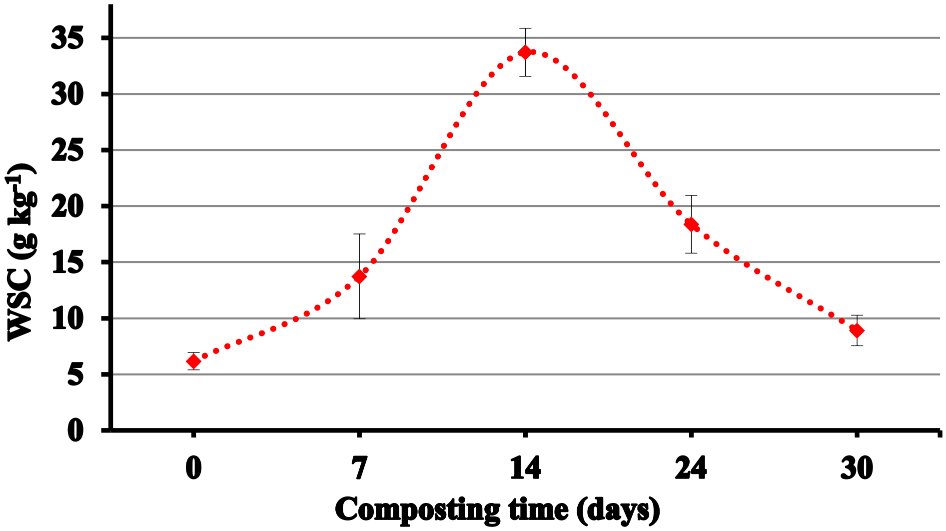

Figure 24. Evolution of WSC during GW composting

The observed increases of WSC suggest that these compounds were released regardless of compost feed mixtures. Ros et al. (2006a) stated that the increase in the content of WSC during composting indicates that new glucose or other WSC originated from microbial activity during this process, since composting is a synthesis process. Charest et al. (2004) suggested that the increase of WSC during the thermophilic phase might be the result of hemicellulose, cellulose, and lignin decomposition. The posterior drop in WSC indicated that these substances were biodegraded during composting by microorganisms, which use it as a source of energy for their growth, and the decomposition of more recalcitrant carbon fractions (Ros et al., 2006a). In addition, Karak et al. (2013) reported that the rapid drop of WSC suggests the rapid exhaustion of reserves of easily degradable organic compounds present in the composting mass to synthesise new macromolecules with properties similar to the humic substances in soils. The decrease of WSC at the end of process could be due to the formation of complex organic molecules produced by the continuous mineralisation of soluble organic compounds and the repolymerization and condensation pathways of organic compounds, which led to the production of complex organic molecules with low solubility in water (Said-Pullicino et al., 2007).

WSC is one of the active parameters in defining compost maturity and a reduction in WSC is often used as an indication of compost maturity (Bernal et al., 2009). Some authors proposed that the threshold maturity values for WSC in finished compost should be <0.4 % (Zmora-Nahum et al., 2005), <0.5 % (Garcia et al., 1992b), <1 % (Hue and Liu, 1995), and < 1.7 % (Bernai et al., 1998). Nevertheless, WSC is mainly constituted of sugars, polyphenols, amino acids and apart from the soluble fraction of fulvic acids (Ros et al., 2006a). Iannotti et al. (1994) reported that immature compost contains sugar, phenolic substances, organic acids, amino acids, peptides and other easily biodegradable substances. However, in mature compost, the major components of WSC are present as humic substances, which are resistant to further biodegradation, and thus demonstrate increased compost stability (Hsu and Lo, 1999).

The rate of decrease WSC during composting depends on the composting technique utilized and on the source material (Hsu and Lo, 1999). In the present study, the WSC of the final compost was 8.9 g kg^{-1}, which was lower than the maturity threshold of 10 g kg^{-1} as suggested by Hue and Liu, (1995), indicating that the compost became mature after 30 days of composting. Composting of pruning residues by conventional open windrow results in final a WSC of 3.6 g kg^{-1} after 18 weeks of composting (Fontanive et al., 2004), which indicates it took a longer time to mature; this is due to poor aeration conditions. This suggests that this new technology facilitates reaching a maturity in a shorter period which is due to a forced bottom-up aeration system that improved composting conditions and resulted in accelerate the decomposition of WSC during GW composting.

4.2.8 Total carbon and oxidisable organic carbon

Change of TC and OXC content during composting period has been illustrated in (**Figure 25**). In general, TC decreased within the proceeds of composting. The initial TC was 31.2 % and finally it reduced to 21.1 % after 30 days. The TC content decreases during composting because of microorganisms degradation of organic substances necessary for their metabolism, leading to their mineralization as CO_2, H_2O and energy, whereas the remaining organic carbon is transformed to stable humus like substance (Azim et al., 2018). A same result in the reduction of TC in organic waste was cited in the literature (Karak et al., 2013; Sharma and Yadav, 2017). The depletion of the TC content indicates the mineralization of the OM and the maturity of the compost. At the end of the composting period, higher percentage of carbon content reduction indicates stabilized compost.

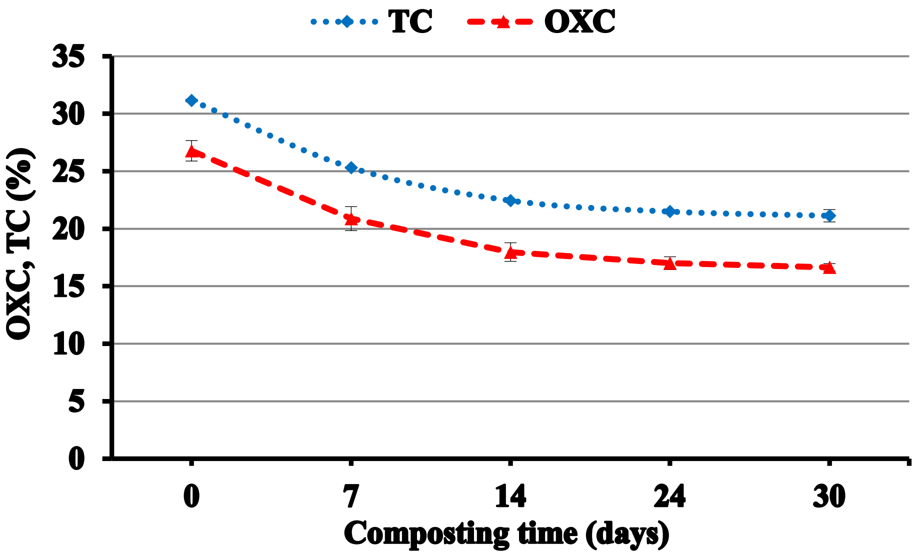

Figure 25. Changes TC and OXC content during GW composting

The initial OXC was 26.8 % then fell to reach 16.6 % after 30 days (**Figure 25**). OXC followed a similar tend with that of the TC indicated the decomposition of OM during composting. Karak et al. (2013) found linear relationship between TC and OXC. The apparent decrease in TC and OXC due to carbon volatilization as CO_2. The decrease trend of OXC content was in agreement with Karak et al. (2013) result.

4.2.9 C/N ratio

The C/N ratio, which is normally used to determine compost stabilization, is expected to decrease with the progress of composting (Bernai et al., 1998). In this case, the C/N ratio decreased substantially as composting progressed and reached 11.4 at the end of the composting period (**Figure 26**). The present study results align closely to Azim et al. (2018), who confirmed that the C/N ratio gradually decreased with composting time. The reduction in C:N ratio was due to the loss of TC in the form of carbon dioxide and the related increase in the proportion of TN resulted from the intense biological oxidation of OM during the composting period (Azim et al., 2018) and the contribution of nitrogen fixing bacteria (Garcia et al., 1992b).

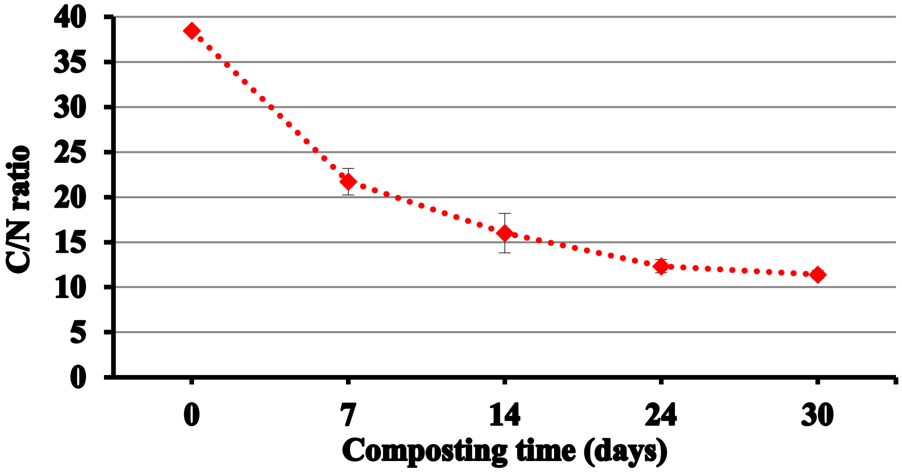

Figure 26. Evolution of the C/N ratio during GW composting

Notably, in the literature there is no general agreement regarding the exact value of the C/N ratio that indicates stabilization of compost. Some authors suggest a maturity index for C:N ratio of < 20 (Azim et al., 2018), whereas Bernai et al. (1998) considered the C:N ratio < 12 as indicative of mature composts. In the present study, the C/N ratio of the final compost reached a value of 11.4, indicating that the compost became mature after 30 days of composting. Composting of pruning residues by conventional open windrow results in a final C/N ratio of 24.8 after 18 weeks of composting (Fontanive et al., 2004), which indicates failure to achieve maturity, possibly due to poor aeration conditions. This suggests that this new technology facilitates reaching a maturity in a shorter period thanks to a forced bottom-up aeration system that improved aeration conditions, accelerated degradation and resulted in higher loss of OM. The results align with the findings of Guo et al. (2012), who demonstrated that higher rates of aeration lead to increased carbon loss.

4.2.10 NH_4^+/NO_3^- ratio

Nitrification index (NH_4^+/NO_3^- ratio) has also been used as an indicator to estimate the compost maturity with values of <1 (Jouraiphy et al., 2005) or even <0.16 (Bernai et al., 1998) denoting a mature compost. The NH_4^+/NO_3^- ratio **(Figure 27**) shows a gradual decline until the end of composting, reaching a final value of 0.1, so indicating an effective nitrification process occurring during composting and the achievement of a mature compost after 30 days. Differently, during composting of pruning residues using conventional open windrows an acceptable maturity index value was recorded after 18 weeks (Fontanive et al., 2004). Thus, the new technology results in a shortened the period to reach maturity due to improved aeration conditions by forced bottom-up aeration that led to increase of nitrification process (Hao and Chang, 2001).

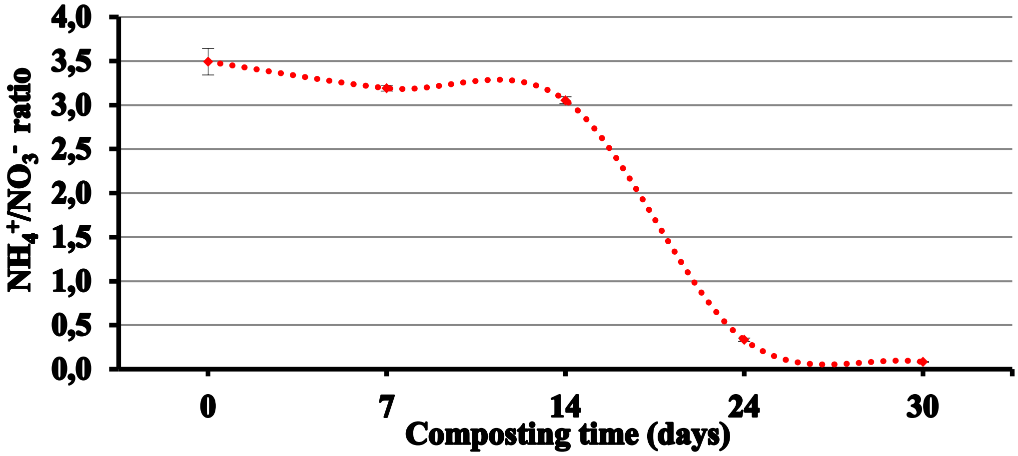

Figure 27. Evolution of the NH_4^+/NO_3^- ratio during GW composting

4.2.11 Cation exchange capacity

Several studies have found a correlation between the CEC and degree of OM degradation (Azim et al. 2018; Senesi, 1989), thus CEC is considered an important indicator of compost maturity and indicates the ability of compost to retain nutrients (Senesi, 1989). The CEC value rose substantially over time from 37.1 cmol kg^{-1} to 160.1 cmol kg^{-1} after 30 days of composting (**Figure 28**), as expected by the increasing temperature that accelerated the oxidation and humification of OM, so increasing the formation of ionized carboxyl and phenolic functional groups which contribute to enhance CEC values and nutrient conservation (Senesi and Plaza, 2007). The contribution of the phenolic groups accounted for 35 % and carboxyl groups 55 % (Lax et al., 1986). Harada and Inoko (1980a) founded an increase in the CEC value during composting of city refuse waste, which is similar to the findings of Zhang and Sun (2014a) during co-composted GW with spent mushroom compost and biochar. According to Harada and Inoko (1980b), the CEC of the final compost should be > 60 cmol kg^{-1} to be considered mature. The higher CEC value at the end of the composting period increased the capacity of the compost to maintain nutrients and increased the GI (Ameen et al., 2016). In the current study, the CEC value of the final product was 160.1 cmol kg^{-1}, which indicated that the compost was mature (Senesi, 1989).

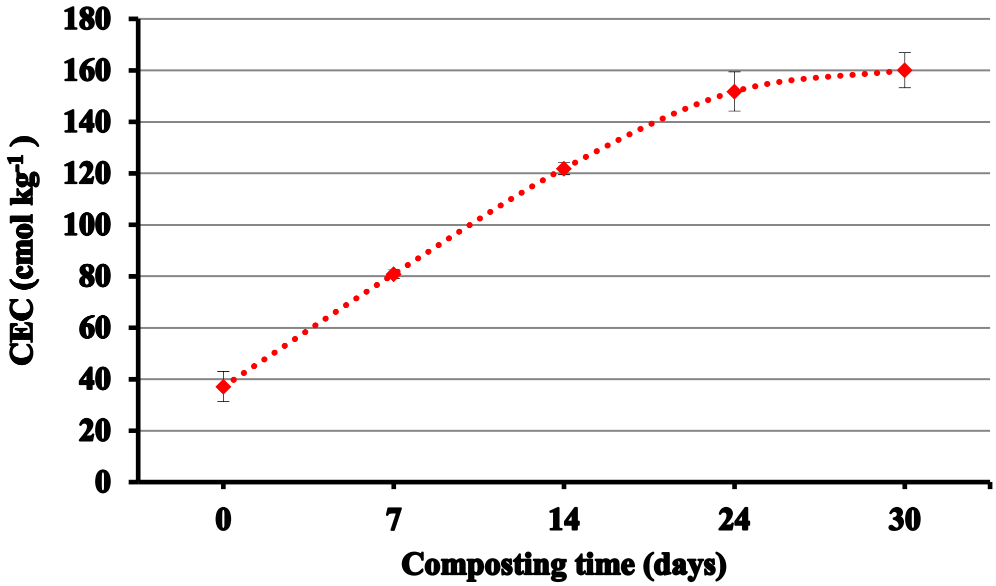

Figure 28. CEC evolution during GW composting

4.2.12 $E_{4/6}$

Another important index of compost maturity is the humification degree of OM, i.e. the generation of humic-like substances (Senesi, 1989). **Figure. 29** depicts the evolution of the absorbance ratios $E_{4/6}$ (the absorbance ratio of wavelength 465 and 665 nm), a traditional indicator of the degree of polymerization of OM and an inverse index of molecular size and aromaticity (Guo et al., 2019; Senesi, 1989). High values of $E_{4/6}$ ratio at the beginning of composting indicate the presence of smaller sized organic molecules and/or high proportions of aliphatic structures (Chen et al., 1977; Guo et al., 2019). As composting time progresses, the $E_{4/6}$ ratio decreased markedly suggesting a progressive condensation of aromatic OM constituents, mineralization of carbohydrates, and oxidization of phenolic compounds and bound to methoxyl groups and/or aliphatic side chains in humic substance; this demonstrated an increasing OM humification (Guo et al., 2019; Senesi, 1989). At the end of composting, the attainment of a value of $E_{4/6}$ lower than 5 indicated an adequate degree of OM maturity in a stable compost, which similar to native humified OM in soil (Chen et al., 1977).

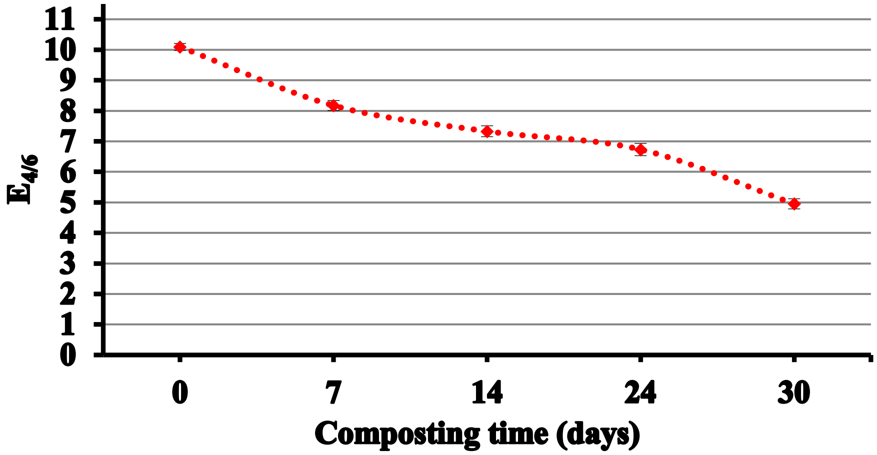

Figure 29. Changes in $E_{4/6}$ ratio during GW composting

4.2.13 Total phosphorus and total potassium

Change of TP and TK during the progress of composting has been depicted in **Figure 30**. From the figures it can be seen that TP and TK gradually increased due to the decomposition of OM during the composting period. When the organic carbon in the compost is mineralized into CO_2, it will have a higher losing rate in comparison with TP, and TK and such difference may result in a rise of TP and TK concentration (Lin, 2008). For most crops, P is the second and K the third most needed nutrients after N.

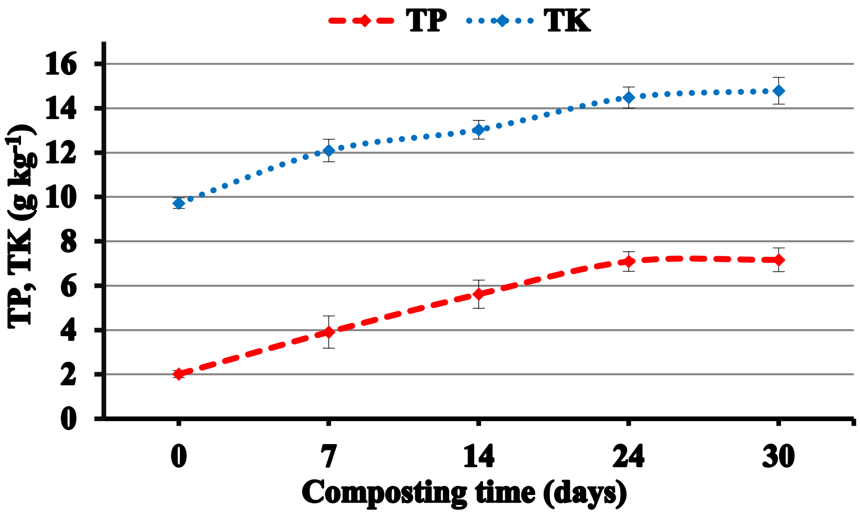

Figure 30. Macronutrients (TP and TK) evolution during GW composting

4.2.14 Principal components analysis

The PCA is a statistical method that can evaluate the relationship between responses of all experimental variables which vary concurrently. The results of PCA of physico-chemical parameters along the composting process demonstrated the existence of high correlations (positive or negative) shown in the circles (**Figure 31**). The projection on the plane of the variables shows the affinity of variables along each axis. The PCA analysis reduces multidimensional relationships between parameters to two principal components that explain 90.2 % of the total system variability. The first component (PC 1) explains 78.3 % of the total variability and separates two groups of parameters. The first group has OM, OXC, $E_{4/6}$, C/N ratio, moisture, TC, and NH_4^+/NO_3^- ratio which are closely correlated by evolving similarly, i.e. they decrease toward the end of composting. The second group contained parameters, such as pH, EC, Ash, TK, TP, CEC, TN, NO_3^-, dry matter, BD_{dry}, and BD_{wet}, which are also correlated by evolving similarly, but increasing during composting. The second component (PC 2), which consists of NH_4^+ and WSC that initially increased during the first 14 days of composting and then followed by a decline till the end of composting, explains 11.9 % of the total variability.

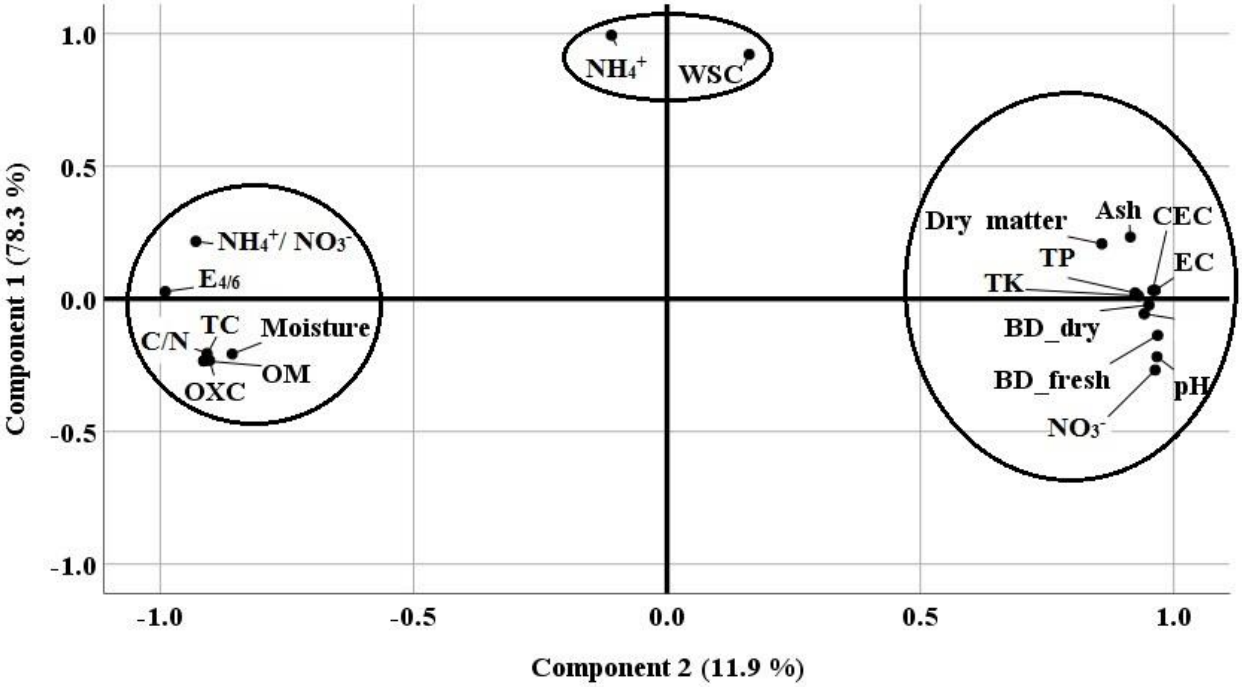

Figure 31. Principal component analysis of the physico-chemical parameters measured during GW composting

4.2.15 FTIR spectroscopic analysis

FTIR spectroscopy analysis is widely used to characterize the principal classes of chemical groups that make up OM and determine changes in the OM decomposition in waste materials during composting (Lü et al., 2018). The FTIR spectra are presented in **Figure 32**. The interpretation of the spectra is according to numerous works, notably El Fels et al. (2014), El Mezouari El Glauoi et al. (2018), Wu et al. (2011), Huang et al. (2006), Hsu and Lo (1999), Jouraiphy et al. (2005), Smidt et al. (2005), and Lü et al. (2018).

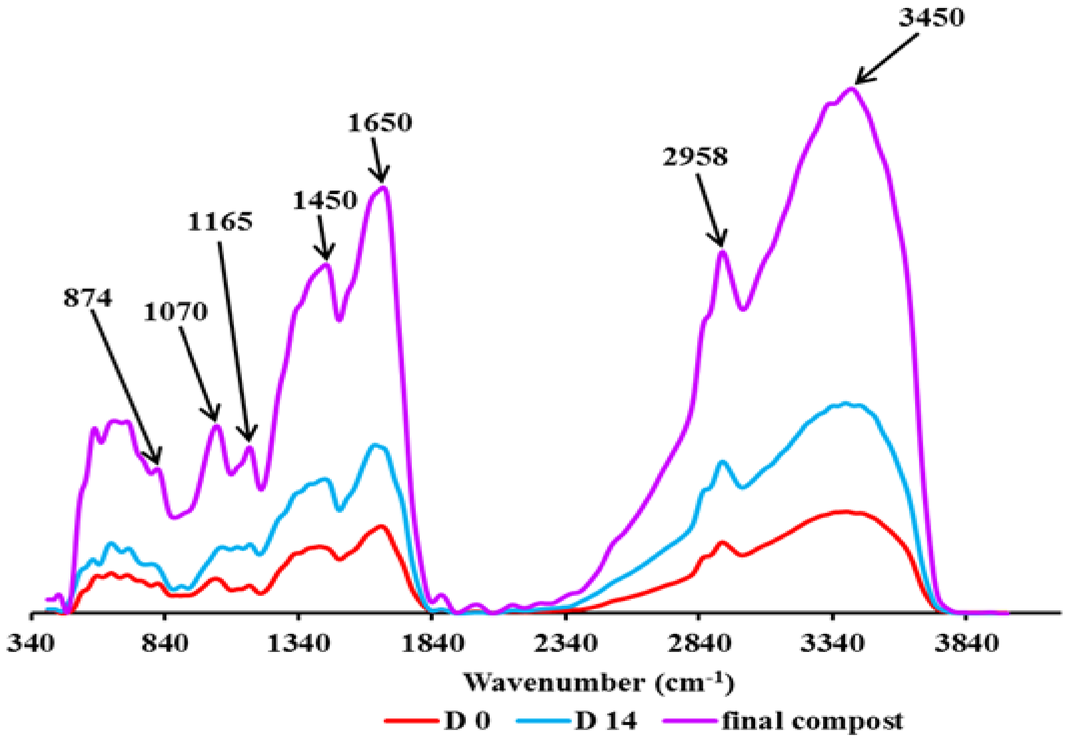

Figure 32. FTIR spectra of GW at various stages of composting (D 0: day 0, D 14: day 14, and final compost)

A very broad band from 3700 to 2800 cm^{-1} and centered on 3450 cm^{-1} was detected during the process; this band can be attributed to stretching vibrations of OH hydroxyl groups. Soobhany et al. (2017) attributed this band to the hydrogenic vibrations of the OH groups of alcohols and to the phenols or the OH of the carboxyl groups (COOH). The 3000-2800 cm^{-1} region reflects the hydrophobic properties of the aliphatic OM (El Fels et al., 2015, 2014).

The very strong peak observed at 2958 cm^{-1} is due the νC-H methyl, and methylene C-H aliphatic groups. The intensity of the bands at 2958 cm^{-1} decreased during composting and was consistent with the microbial degradation of aliphatic carbon chains and peptidic compounds, this was an indicator of the OM degradation during the composting process and can be used to evaluate the composting processes (Amir et al., 2005; Smidt and Schwanninger, 2005). The bands between 2920 and 1640 cm^{-1} designated as aliphatic methylene and unsaturated or aromatic C=C vibrations, respectively is considered an indicator of OM humification (Droussi et al., 2009).

A distinctively sharp peak at band 1650 cm^{-1} increased during composting. This band is associated with the absorption of aromatic C=C bonds (Smith, 1998) and to stretching vibration of the C=O group that is part of carboxylic acids, ketones, and aldheydes (Kaiser and Ellerbrock, 2005). Moreover, El Fels et al. (2014, 2015) attributed the peaks that appear at 1650 cm^{-1} to νC=O of ionized carboxyl (COO-). Given that the peak at 1640 cm^{-1} is attributed to aromatic C=C vibrations, and increasing intensity as the composting process progressed, this indicated an enrichment in aromatic C=C compared with aliphatic carbon (Baddi et al., 2004). Smidt et al. (2005) found that HA have a strong band at 1640 cm^{-1} that increases during the composting process. In the same study, the continuous increase in the humic acid content had a positive impact on this band and formation of carboxylates due to the release of carboxylic acids from decomposed lipids, which contributed to the rise as well. The decrease in peak at 2958 cm^{-1}, together with the appreciable increase in the intensity at bands 1650 cm^{-1}, were also observed by FTIR spectroscopy previous studies (El Fels et al., 2014; Hsu and Lo, 1999; Huang et al., 2006; Makni et al., 2010). These changes demonstrated that easily degradable OM components, such as aliphatic and peptidic compounds, are oxidized, and as a result, more aromatic compounds are contained in the mature compost.

The bands at 1450 cm^{-1} are attributed to aromatic benzene compounds. Droussi et al. (2009) attributed the region from 1442-1472 to C=C stretching vibrations from aromatic components. El Fels et al. (2014) demonstrated that the decrease in band intensities of aliphatic chains (νC-H and $\delta CH_2/CH_3$ at 1450 cm^{-1}) and the increased intensities of C=C functions favor a dynamic composting process leading to the formation of unsaturated and aromatic structures. The 1165 cm^{-1} band could be attributed to C-O-C stretching. Ouatmane et al. (2000) and Wu et al. (2011) attributed the 1165 cm^{-1} band to C-O-C stretching which could be used as an indicator of the biodegradation process. The 1070 cm^{-1} band was constant during composting. The region at 1080-1030 cm^{-1} was attributed to the C-O stretching of polysaccharides or polysaccharide-like substances, and the Si-O asymmetric stretch of silicate impurities (Filip and Bielek, 2002; Senesi et al., 2003). El Fels et al. (2015) attributed this band to the mineral phase provided by the sludge. The band at 874 cm^{-1} could be attributed to calcite.

4.3. Comparison of FTIR spectra

The FTIR spectra of DSSS-GW and GW collected during and at the end of composting (**Figures 18 and 32**) are qualitatively similar to one another. This suggests that the relatively new composting technologies that are ELS with GORE® Cover membrane and ASW with GORE® Cover membrane impacts the stabilization of OM and compost maturation.

4.4. Characteristics of the final compost

Compost product can be safely used as organic fertilizer or conditioner once it is mature or stable and does not present toxicity to plant growth (Sharma et al., 2019). **Table 4** shows the main physico-chemical properties of the final composts obtained by using ELS with GORE® Cover membrane and ASW with GORE® Cover membrane. The final compost product satisfied the threshold levels established in the literature for common maturation indices, suggesting that it can be used as an organic fertilizer. These findings indicated that composting was carried out successfully under optimized conditions.

Table 4. Summary of the characteristics of final compost obtained by using ELS with GORE® Cover membrane and ASW with GORE® Cover membrane, in comparison with guidelines for acceptable quality of compost

Physico-chemical properties	ELS compost	ASW compost	Threshold values	References
pH	7.1 ± 0.1	7.1 ± 0.0	7.1-7.7	Forster et al. (1993)
EC (mS cm^{-1})	3.0 ± 0.0	3.0 ± 0.1	< 4	Lasaridi et al. (2006)
BD$_{wet}$ (g cm^{-3})	0.6 ± 0.0	0.9 ± 0.1	≥ 0.4	Abad et al. (2001)
BD$_{dry}$ (g cm^{-3})	0.4 ± 0.0	0.8 ± 0.0		
OM %	37.5 ± 1.0	38.8 ± 0.5	33.3–55.4	Fricke and Vogtmann (1994)
C/N	6.6 ± 0.5	11.4 ± 0.4	< 20, preferable < 10	Bernai et al. (1998)
CEC (cmol kg^{-1})	171.9 ± 2.8	161.7 ± 5.8	> 60	Harada and Inoko (1980a)
TC (%)	20.2 ± 0.0	20.9 ± 0.3	> 20	Lasaridi et al. (2006)
TN (%)	3.1 ± 0.2	1.8 ± 0.1	≥ 1	Cheng et al. (2013)
TP (%)	1.7 ± 0.0	0.7 ± 0.1	0.6-1.7	Fricke and Vogtmann (1994)
TK (%)	1.5 ± 0.0	1.5 ± 0.1	1-1.5	Anthonis (1994)
WSC (g kg^{-1})	3.9 ± 0.9	8.8 ± 0.8	≤ 10	Hue and Liu (1995)
			≤ 4	Zmora-Nahum et al. (2005)
			< 17	Bernai et al. (1998)
NH$_4^+$ (mg kg^{-1})	350.2 ± 9.9	73.4 ± 7.4	< 400	Bernai et al. (1998)
NH$_4^+$/NO$_3^-$	0.1 ± 0.0	0.1 ± 0.0	< 0.16	Bernai et al. (1998)
Ca (mg kg^{-1})	43203 ± 238	30107 ±105	≥ 12000	Hungarian law of fertilizers (2006)
Na (mg kg^{-1})	434 ± 10	210 ± 24	-	
Mg (mg kg^{-1})	6953 ± 18	6734 ± 28	≥ 5000	Hungarian law of fertilizers (2006)
Cr (mg kg^{-1})	56.6 ± 0.8	80.4 ± 1.4	≤ 100	Hungarian law of fertilizers (2006)
Cu (mg kg^{-1})	144 ± 1	100 ± 3	≤ 300	Hungarian law of fertilizers (2006)
Fe (mg kg^{-1})	22248 ± 71	13648 ± 135	-	
Mn (mg kg^{-1})	327 ± 1	362 ± 3	-	
Ni (mg kg^{-1})	19.8 ± 0.2	21.1 ± 0.4	≤ 50	Hungarian law of fertilizers (2006)
Zn (mg kg^{-1})	621 ± 2	70.0 ± 0.4	-	
Cd (mg kg^{-1})	1.68 ± 0.03	1.79 ± 0.17	≤ 2	Hungarian law of fertilizers (2006)
Co (mg kg^{-1})	4.63 ± 0.04	2.54 ± 0.25	≤ 50	Hungarian law of fertilizers (2006)
Pb (mg kg^{-1})	13.5 ± 0.2	11.7 ± 0.4	≤ 100	Hungarian law of fertilizers (2006)

(-) – there is no threshold in 36/2006. (V. 18.) FVM regulation

5. NEW SCIENTIFIC RESULTS

1-The FTIR spectra showed that the chemical composition of the OM, while composting using ELS with GORE® Cover membrane and ASW with GORE® Cover membrane is characterized by a decrease in methylene C-H aliphatic groups and enrichment of aromatic compounds, which is confirmed by drastic decreases in $E_{2/4}$, $E_{2/6}$, and $E_{4/6}$ ratios. This indicates a rise in the humification degree, stabilization of the final compost, and confirms the efficiency of composting.

2-The FTIR spectra of DSSS-GW and GW collected during and at the end of composting are qualitatively similar to one another. This suggests that the relatively new composting technologies that are ELS with GORE® Cover membrane and ASW with GORE® Cover membrane impacts the stabilization of OM and compost maturation.

3-It is recommended to shorten the composting period during composting using ELS composting technology with GORE® Cover membrane and ASW with GORE® Cover membrane to 28 days and 14 days, respectively, because the DSSS-GW and GW compost product is considered to mature at 28th and 14th day, respectively, as per loss of OM maturity index >42 %.

4-In the ELS composting process with GORE® Cover membrane and ASW with GORE® Cover membrane, the DSSS-GW and GW compost satisfied the maturity by the day 49th and 30th, respectively, based on the following maturity index namely: NH_4^+/NO_3^- ratio, CEC value, and C/N ratio. This suggests that this new technology can effectively ensure compost maturity in a shorter time frame because of improved aeration conditions and oxygen flow throughout the entire composting vessel using forced bottom-up aeration that enhances the decomposition of OM.

6. CONCLUSIONS

It is observed that the ELS composting process with GORE® Cover membrane and aerated static windrow with GORE® Cover membrane is an effective waste management option. The final compost produced by these two composting technologies satisfied the threshold levels established in the literature regarding its use as an organic fertilizer. These findings indicated that composting was performed successfully under optimized conditions and that these relatively new composting technologies are a suitable choice for converting the organic waste into a useful product.

Moreover, using ELS with GORE® Cover membrane led to not only more rapid attainment and prolongation of thermophilic temperatures but also in creating two thermophilic temperature peaks. This results in more rapid decomposition of organic waste while significantly shortening the DSSS composting period for obtaining mature compost. Similarly, ASW with GORE® Cover membrane enabled GW to decompose more rapidly while reducing the composting period because of higher peak temperatures and rapid attainment and prolongation of thermophilic temperatures.

Data from the FTIR spectra and UV-visible spectroscopic analysis showed that the chemical composition of OM during composting using ELS with GORE® Cover membrane and ASW with GORE® Cover membrane, is characterized by enrichment in aromatic structures and a degradation of both aliphatic and peptide structures. These changes suggested that there was an increase in humification degree, stabilization of the final compost, and confirmed the efficiency of composting. The compost obtained, which included a substantial richness of stabilized OM, can be considered as a suitable soil amendment or organic fertilizer without risking the soil–plant system.

7. SUMMARY

Composting is one of the most effective integrated waste management approach for recycling organic waste and generating a useful product. There are various composting methods in terms of potency of stability and duration of degradation. However, traditional composting technology is generating foul smells, produces immature compost, and time consuming. An ELS and ASW along with GORE® Cover membrane and an air-floor aeration system are relatively new industrial-scale composting technologies that previously have not yet explored. Therefore, the aim of this research study was to evaluate the effectiveness of these new composting technologies. The composting process was monitored in terms of changes in the physico-chemical properties, UV-visible spectra, and FTIR spectra.

During composting by ELS with GORE® Cover membrane, the temperature was steady in the thermophilic range for 24 and 12 days in the intensive and maturation phases, respectively, thus fulfilling the disinfection requirement. Moreover, the temperature increased rapidly to 76.8°C within three days, and the thermophilic temperatures peaked twice and lasted longer than in traditional composting, thus accelerating DSSS degradation and reducing the composting period necessary to obtain mature compost. Similarly, ASW with GORE® Cover membrane, enabled organic waste decomposed more rapidly because of higher peak temperatures as well as more rapid attainment and prolongation of thermophilic temperatures. Thus, high decomposition rate, accelerated GW degradation, and reduced composting period leads to lower operating costs for composting plants.

The FTIR spectra and UV-visible spectroscopic data showed that the chemical composition of OM during composting using ELS with GORE® Cover membrane and ASW with GORE® Cover membrane is based on the disappearance of aliphatic chains (peptide and methylene C–H aliphatic groups) and a corresponding increase in aromatic chains. Such changes indicated a rise in the humification degree, stabilization of the final compost, and confirmed the efficiency of composting. The resulting compost that had a substantial richness of more aromatic structures with higher stability can be regarded as a suitable soil amendment or organic fertilizer without risking the soil–plant system.

These two composting technologies significantly improved the conditions of composting process. i.e., temperature, and enhanced the final compost quality product in terms of bulk density, pH, EC, carbon dynamics, OM degradation, nitrogen transformation, humification, CEC, and enrichment with the nutrient content (TN, TP and TK). Considering the ELS composting process with GORE® Cover membrane, the compost matured in only 49 days instead of the 105–180 days needed for traditional composting. However, the ASW with GORE® Cover membrane produced a mature compost in only 30 days compared with the standard period of 90–270 days needed for traditional composting. These relatively new composting technology can be regarded as a viable and sustainable alternative for DSSS and GW management that converts waste into compost that useful as a soil amendment.

8. SCOPE FOR FUTURE RESEARCH

The following recommendations are the scope for future research:

1- Examining enzymatic activities such as dehydrogenase, protease, ß-glucosidase, arylsulphatase, and phosphatases during composting using these relatively new industrial composting technologies.

2- Evaluate economic advantages of these relatively new industrial composting technologies and compare them with the economic costs of traditional industrial composting technology.

3- Investigate the bacterial dynamic and community structure in these relatively new industrial composting technologies using next generation sequencing techniques.

4- Investigate the humification process by using nuclear magnetic resonance spectroscopy (13C NMR/^{1}H NMR), and synchronous-scan fluorescence spectroscopy.

5- Investigate efficiency of these relatively new industrial composting technologies in reducing the human and veterinary antibiotics levels during composting.

9. ÖSSZEFOGLALÁS

A komposztálás az egyik leghatékonyabb integrált hulladékgazdálkodási módszer a szerves hulladékok újra hasznosítására és hasznos termék előállítására. Számos komposztálási eljárás létezik, a stabilitás hatékonysága, valamint a lebomlás időtartama szempontjából. A hagyományos komposztálási technológiákra jellemző, hogy a komposztálás során kellemetlen szagok, nem megfelelően érett komposzt keletkezik hosszú előállítási idő alatt. Az emelőtetős rendszer (Encapsulated Lifting System - ELS), a levegőztetett prizmás rendszer (Aerated Static Windrow - ASW), a GORE® Cover membrántakaró, valamint a levegőztető rendszer (air-floor aeration system) a nagyüzemi komposztálásban viszonylag újszerű technológiának minősülnek, amelyek részletes vizsgálata nem történt meg. Ezért ennek a kutatásnak az volt a célkitűzése, hogy ezeknek az új komposztálási technológiáknak a hatékonyságát megvizsgálja. A komposztálás folyamatát a fizikai, kémiai paraméterek, a UV-látható spektrumok, valamint az FTIR spektrumok változásának monitorozásával végeztük.

A GORE® Cover membrántakaró rendszerrel működtetett emelőtetős komposztálás során a hőmérséklet állandó volt a termofil szakaszban 24 és 12 napon át, az intenzív és az érési szakaszban, ezáltal a fertőtlenítés követelményeinek is megfelelt. A hőmérséklet gyorsan, három napon belül 76,8 °C-ra emelkedett, valamint a termofil hőmérsékleti értékek két csúcsértéket adtak és hosszabb ideig tartottak, mint az a hagyományos komposztálás esetén jellemző. Ez a víztelenített stabilizált szennyvíziszap (Dewatered Stabilized Sewage Sludge - DSSS) lebontását felgyorsította és az érett komposzt előállításához szükséges időt is csökkentette. A GORE® Cover membrántakaró rendszerrel működtetett levegőztetett statikus komposztálási rendszerben (Aerated Static Windrow - ASW) hasonlóan gyorsan lebomlott a szerves hulladék a magasabb hőmérsékleti értékek, valamint azok hosszabb időtartama miatt. Ezáltal a magas lebomlási arány, a gyorsított zöldhulladék lebomlása, valamint a csökkentett komposztálási idő alacsonyabb üzemeltetési költségeket eredményez.

A szerves anyag kémiai összetétele a GORE® Cover membrántakaró rendszerrel működtetett ELS és az ASW komposztálás során az FTIR spektrumok, valamint az UV-látható tartomány spektroszkópos adatok alapján elmondhatjuk, hogy az alifás láncok (peptid és metilén C-H alifás csoportok), valamint az aromás láncok növekvő megjelenése jellemző. Ezek a változások a humifikáció mértékének növekedését, a kész komposzt stabilizálódását, valamint a komposztálás hatékonyságát jelezték. Az így kapott komposzt, amely nagy stabilitású, sok aromás szerkezeti egységgel rendelkezik, megfelelő talajjavító anyagként, szerves tápanyag-utánpótlási anyagként használható anélkül, hogy a talaj-növény rendszert veszélyeztetnénk.

Ez a két komposztálási rendszer szignifikánsan javította a komposztálás folyamatát, pl. a hőmérséklet növelte a kész komposzt minőségét, a térfogattömeg, pH, elektromos vezetőképesség, szerves anyag lebomlás, nitrogén átalakulás, humifikáció, kationcsere-kapacitás, valamint a tápanyag tartalom növekedése (összes nitrogén, összes foszfor és összes kálium), mind pozitív változás a kiindulási anyaghoz késpest. A GORE® Cover membrántakaró rendszerrel működtetett ELS komposztáló rendszer esetében a komposzt éréséhez csupán 49 nap volt szükséges a 105-180 nap helyett, amely a hagyományos komposztálásnál volt jellemző. A GORE® Cover membrántakaró rendszerrel működtetett ASW komposztáló rendszer esetében a komposzt éréséhez csupán 30 nap volt szükséges, a hagyományos komposztálásnál jellemző 90-270 nap helyett. Ez a viszonylag új komposztálási technológia egy fenntartható alternatívát nyújt a víztelenített stabilizált szennyvíziszap és a zöldhulladék kezelésében, amely során ezeket a hulladékokat hasznos talajjavító anyaggá alakítja.

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1-Q2; impact factor: 1.087

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2-Q2; impact factor: 1.54

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