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FULL-SCALE CO-COMPOSTING OF SEWAGE SLUDGE AND WASTE MATERIALS AT VARIOUS MIXING PROPORTIONS AND AERATION CONDITIONS TO PRODUCE STABILIZED COMPOST

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ABSTRACT

Two different sewage sludge (SS) waste materials were commercially co-composted with wood chips (WCh) and grass to determine their conversion potential and elucidate the manner that it is affected by the various process operational parameters. A blend of three different SS streams composted more efficiently in terms of temperature attained during the process than a single SS stream when the materials were co-treated with WCh acting as the bulking agent. WCh facilitated the accomplishment of a temperature over the sanitization limit and concurrently the establishment of a high decomposition rate over a long period. Further increasing the fraction of WCh in the composting mixture had an insignificant effect on the decomposition rate of organics in SS. The presence of grass in a mixture had a negative influence on the composting temperature that retained below the sanitization limits during the entire process. Aeration led to an increment in temperature but in all cases the enhancement remained at levels lower than 20°C throughout the trials. Concentration of heavy metals in all mixtures was sufficiently reduced with composting regardless of the SS type, mixing ratio, the use of a secondary organic substrate and aeration.

1. INTRODUCTION

The contemporary industrialization and urbanization have led to an enormous increase in the generated quantities of sewage sludge (SS), a by-product deriving from the operation of wastewater treatment plants (WWTP). SS entails significant challenges emerging from its costly management and disposal and the fact that its land application encompasses a high ecological risk emerging from a potential accumulation of toxic elements (Sreesai et al., 2013; Romero et al., 2024). Composting is the biological decomposition of an organic material under aerobic conditions through the enzymatic activity of several microorganisms and could be implemented for the treatment of SS prior to disposal mitigating any potential hazard for environment and human health (Temel, 2023). During the process, organic matter is converted into stabilized humic substances whereas any pathogens are eliminated (Muscarella et al., 2023). Legislation defines that a sludge sustained a composting stabilization under certain conditions can be safely disposed on land and implemented as a soil conditioner (Peltre et al., 2015). Nevertheless, certain characteristics of SS such as an unpleasant odour and a heavy metal content among others, are not always disappeared by the process restricting the broad utilization of compost (Smith et al., 2009).

Composting is considered one of the most acceptable and economically feasible technologies for recycling SS (Villaseñor et al., 2011). Nevertheless, due to the low-quality of SS as a composting substrate, co-composting with other types of organic materials is frequently employed for the improvement of its conversion (Ammari et al., 2012; Yousefi et al., 2013). Some physicochemical characteristics of SS may complicate even the co-composting process. For instance, SS contains a high moisture proportion and low porosity that would require the presence of a bulking agents during composting in order to absorb the moisture, enhance aeration and improve the quality of the final product (Yañez et al., 2009). Garden waste, which is commonly characterized by a loose structure, low water content and high C/N ratio is a typical substrate frequently applied to advance the composting of SS (Albrecht et al., 2010; El Fels et al., 2014).

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In the current study, five composting piles with two types of SS were prepared in an industrial environment to recycle the waste into valuable and usable compost. Wood chips (WCh) were the bulking agent that used to prevent compaction and enable oxygenation as well as to balance the moisture content and C/N ratio of the SS. The compost was characterized for its physicochemical properties after 110 days of processing. The objective of the full-scale trials was to: (i) investigate the impact of the SS type, the presence of a bulking agent and of a secondary substrate on the conversion of SS, (ii) determine the effect of aeration of composting pile on the maturity of the produced compost.

2. MATERIALS AND METHODS

2.1 Compostable substrates

Four organic materials, namely a SS waste, a blend of three SS streams of different origin (SSmix), WCh and grass, were selected for the composting trials. SS streams were obtained from four individual municipal WWTPs located in Moravian-Silesian Region (Czech Republic) with mean capacities at 3.000 tonne per month. In all cases, SS waste were received in a dewatered state. SSmix was prepared by mixing in equal proportions three of the SS streams. WCh was obtained from a commercial wood processing unit and were used as the bulking agent to prevent compaction of SS during composting. Grass was obtained from an urban greenery maintenance corporation, received in 1-2 cm long particles due to lawn mower operation and easily mixed with SS and WCh.

2.2 Composting mixtures

Five piles weighted approximately 10 tons were prepared for the composting trials. Each pile contained organic substrates mixed in different proportions. Two mixtures, M1 and M2, prepared by mixing SS and WCh in ratios 1:1 and 1:2, respectively. M3 was made by mixing SSmix and WCh in a ratio of 1:2. To examine the effect of aeration on composting, two mixtures, namely M4 and M5, were prepared so as to have the same composition containing SSmix, WCh and grass at a ratio 1:1:1. Mixtures composition is outlined in Table 1.

2.3 Industrial composting trials

Full-scale experiments were carried out in a composting plant in Moravian-Silesian Region, Czech Republic with capacity close to 2000 t/year. For this study, five piles (5 m × 5 m × 2 m) were prepared within a period of June to September and with the weather being remarkably stable during the entire period. Once the fresh dewatered SS was delivered by a WWTP to the composting plant, it was directly used for the trials.

Composting piles were regularly subjected to turning, which was carried out using a compost turner approximately once per week during the intensive microbiological degradation stage (hydrolytic phase) and on average once per two weeks during the maturity stage. This operation mode allowed the process to be more intensive. Composting trials were carried out for 110 days. Pile temperature was measured using a PT-100 type thermometer in three depths (30, 80 and 110 cm) from the pile surface. Sampling was also conducted at the same depths (30, 80 and 110 cm) after the turning procedure was finalized, gathering a total of 10 kg mixed in a single homogenous sample. To examine the influence of aeration on the composting process, fresh air was pumped into the pile of one mixture (M5) through a perforated plastic tube over the entire composting process. For the analysis of the results, the composting mixture consisted of SSmix and WCh at a ratio 1:2 (M3) was considered as the reference.

2.4 Analytical methods

A WTW 340i pH-meter (SenTix 410 sensor) was used for pH measurement. Total solids (TS) content was measured according to the EN 15934:2012 standard by drying the sample at 105°C until the variation of weight was < 2.0% wt in a KERN DLB 160 3A moisture analyser. VS content was determined by measuring the loss on ignition of the material at 550°C in O_2 atmosphere in a LECO TGA 701 thermogravimetric analyser. Carbon, hydrogen, nitrogen and sulfur concentrations were determined using a LECO Truspec CHN and S628 elemental analyser operated according to the ASTM D5373-16 standard. Concentration of heavy metals including phosphorus and potassium were determined by using a multielement XRF XEPOS (SPECTRO Analytical Instruments GmbH, Germany). To increase results accuracy, measurements of heavy metals in the parent SS and SSmix were triplicated. Escherichia coli and Salmonella spp. were determined in microbiology tests performed following the EN ISO 9308-2:2012 standard method. Samples were analysed in triplicates to accurately determine the moisture, organic matter content, C, H, N, S and heavy metals.

2.5 Ecotoxicological methods

Three ecotoxicological tests were conducted to assess the ecotoxicological properties of the mixtures, namely inhibition of luminescence of marine bacteria Vibrio fischeri, root growth inhibition of Sinapis alba seeds and cultivation experiments with *Hordeum vulgare L.* The tests with V. fischeri and S. alba were performed with aqueous leachates prepared from the composting mixtures in accordance to the EN 12457-4:2002 standard method. Luminescent bacteria tests were carried out with LUMStox luminometer (Hach-Lange GmbH, Germany) in accordance to ISO 11348-1:2007 standard in two parallel determinations and two repetitions with evaluation after 15 and 30 minutes. The inhibitory effect H_t was calculated according to the equation:

$$H_t = [(I_{ct}-I_t)/I_{ct}] \times 100 \quad (1)$$

where I_{ct} is the corrected luminescence value and I_t is the luminescence intensity of the test sample after 15 min or 30 min exposure in relative units (ISO 11348-1:2007). Root growth inhibition tests with S. alba were conducted as a rapid preliminary screening prior to detailed testing during experiments with *H. vulgare*. The tests were performed in duplicates with 20 S. alba seeds and 5 mL of the leachate per treatment for 72 h. Mean root length was evaluated via ImageJ software. The root growth inhibition *I* was calculated according to the equation:

TABLE 1: Composition of blends used as substrates for composting trials.

Materials	Mixtures (t)				
	M1	M2	M3	M4	M5
Sewage sludge	5	3.3	-	-	-
Sewage sludge mixture	-	-	3.3	3.3	3.3
Grass		-	-	3.3	3.3
Wood chips	5	6.6	6.6	3.3	3.3

$$I = (L_c\text{-}L_t)/L_t \times 100 \tag{2}$$

where L_t and L_c represent the mean root length in test and control determinations, respectively. Subsequent cultivation experiments with *H. vulgare L.*, variety Bojos involved growth analyses, evaluation of the physiological state of the plants and ecotoxicological tests. The experiments were carried out with stabilized compost samples that were not further diluted with soil. The tests were conducted in 4 replicates (4 pots per compost sample) with 16 seeds in each pot.

3. RESULTS AND DISCUSSION

3.1 Properties of materials and initial mixtures

Table 2 presents the properties of all individual materials used for the formulation of mixtures. TS and carbon concentration in SSmix were comparable to those in the biodegradable waste (grass), however were much higher than those in the similar material (SS). The deviation between the two SS indicates that a different behaviour might be expected during the composting process.

Table 3 presents the initial and final properties of the mixtures formulated for the 5 trials. The moisture content of a mixture is an important parameter influencing microbial activity during composting. Moisture not only supports the metabolic process but also creates a suitable environment for the transport of micromolecules formed from the decomposition of organic matter and dissolution of oxygen. For an efficient process, moisture of the primary mixture should be between 40-60% wt at process beginning (Oshins et al., 2022, Gajalakshmi and Abbasi, 2008). In our case, moisture concentration in the piles M2, M3, M4, M5 was within the optimal range indicating that the specific mixtures had a high potential for a sufficient composting performance. On the other hand, concentration of moisture in M1 was at 64.2% wt which exceeded the upper limit of the optimal range suggesting that M1 conversion during composting might be expected to be inadequate. Comparison of the initial and final TS content of each mixture in Table 3 indicates that composting led to a reduction of the TS concentration in all cases by 5-16% wt. The reduction is considered reasonable and could be expected whereas is attributed to a high temperature attained in piles during the process that causes evaporation and loss of water from the system.

The C/N ratio has been found to affect the composting of organic substrates (Komilis et al., 2011; Ponsá et al., 2010). Carbon and nitrogen are important for the develop-

TABLE 2: Elemental analysis, pH and TS of precursors used in large-scale trials.

Parameter	SS	SSmix*	WCh	Grass
pH	7.57	7.49	5.77	8.16
TS (%wt)	21.63	26.75	57.63	28.50
VS (%wt_{TS})	54.42	58.66	92.42	70.48
Elemental analysis (%wt_{TS})				
C	27.41	31.77	48.55	33.73
H	4.53	5.22	5.56	5.13
N	3.91	4.31	0.51	2.33
S	1.31	1.02	0.05	0.59
P	5.3	3.65	0.17	0.93
K	1.2	1.11	0.87	4.98
C/N (-)	8.18	8.60	111	16.89
Heavy metals (mg/kg)				
As	17.73 ±1.07	7.65±0.80	1.7	1.59
Cd	4.43±0.46	2.00±0.30	0.1	4.50
Cr	97±25.00	100.50±23.60	32.60	16.50
Cu	134±27.80	314±42.80	23.00	36.00
Hg	1.82±0.40	2.06±0.32	ND	ND
Ni	88.8±7.80	26±2.86	9.70	23.40
Pb	55.2±9.98	25.3±6.98	6.90	2.30
Zn	252±32.00	469±39.90	248	115

**Average values from three SS used to prepare the SSmix, ND=Not detected*

TABLE 3: Properties and heavy metal content of primary mixtures and composts.

Parameter		Limit value	M1	M2	M3	M4	M5
pH	Initial	-	6.92	6.44	6.62	7.50	8.01
	Final	-	5.95	6.30	5.82	6.09	6.30
TS (%wt)	Initial	-	50.70	47.06	45.23	51.11	58.32
	Final	-	35. 80	41.99	40.46	40.83.	41.67
VS (%wt_{TS})	Initial	-	77.51	73.31	91.28	85.00	83.07
	Final	-	36.64	51.01	58.34	27.70	31.70
Elemental analysis (%wt_{TS})							
C	Initial	-	39.14	41.62	45.27	42.30	38.01
	Final	-	19.58	25.14	31.38	14.59	15.50
H	Initial	-	5.39	4.93	6.31	4.15	3.32
	Final	-	3.01	3.58	4.02	2.06	2.05
N	Initial	-	2.72	1.95	2.04	2.80	1.72
	Final	-	1.09	1.13	1.66	1.02	1.16
S	Initial	-	0.63	0.42	0.33	0.56	0.38
	Final	-	0.25	0.22	0.28	0.22	0.28
C/N (-)	Initial	-	16.79	24.90	25.89	17.63	25.78
	Final	-	20.96	25.96	22.05	16.69	15.59
Heavy metals (mg/kg)							
As	Initial		7.33	5.59	3.18	5.10	1.33
	Final	20	2.03	1.90	2.28	1.17	4.44
Cd	Initial		0.60	0.57	1.24	2.00	2.00
	Final	2	0.56	0.36	1.61	1.60	1.44
Cr	Initial		90.50	27.80	23.60	95.90	50
	Final	100	30	15	14.50	95	37.90
Cu	Initial		64	51	53	228	190
	Final	100	39	34	50.20	152	72.10
Hg	Initial		0.63	0.53	0.35	0.64	0.70
	Final	1	0.17	0.11	0.20	0.45	0.35
Ni	Initial		61.50	15.70	10.90	20	13.80
	Final	50	16.40	13.20	10	15	10.90
Pb	Initial		40.70	45.80	12.10	31	29
	Final	100	7.60	6.40	27.20	30	28
Zn	Initial		213	195	320	340	320
	Final	300	163	157	300	300	277

ment of microorganisms since bacteria consume nitrogen for their cellular growth and the synthesis of protein whereas use carbon to cover their energy requirements. A microorganism contains 10-15 units of carbon and one unit of nitrogen by weight and hence it is expected to consume 25 times more carbon than nitrogen to cover its needs (Temel, 2023). As a consequence, the C/N ratio of a composting sludge will significantly decrease with process evolution. Given that SS is characterized by a low C/N ratio, mixing the waste with other organic materials e.g. WCh or grass would be a proper practice for increment of its potential for efficient composting. A C/N ratio between 25 and 35 has been found to be optimal for a competent aerobic degradation (Oshins et al., 2022). The initial C/N ratio of the mixtures ranged from 16.79 to 25.89. In particular, the C/N ratio in M2 (24.90), M3 (25.89) and M5 (25.78) were close to the range that ensure a competent aerobic process. In contrast, the C/N ratio in the mixture M1 and M4 was lower than 25 (16.79 and 17.63, respectively) which is the limit above which carbon is rapidly depleted with a concurrent high loss of nitrogen and extended ammonia formation.

The way that the C/N ratio modified by the composting process was not similar in all runs. The ratio in the composts M1 and M2 was higher than that in the initial mixtures because nitrogen loss was greater than the carbon reduction during the process. In contrast, in the composts M3, M4 and M5, the carbon concentration was lower and the nitrogen proportion higher than in the primary mixtures leading to an overall lowered C/N ratio (Raj and Antil, 2011). Although the variation of C/N ratio over the composting time is considered as a stability indicator for the entire process, the parameter could not be reliable factor due to the variability in the composition of the initial mixtures (Muktadirul Bari Chowdhury et al., 2013).

The pH of the starting mixtures is shown in Table 3. The pH affects the biochemical reactions occurring during composting and its stability depends on the activity of various microorganism colonies and the buffering capacity of the system. Theoretically, pH values of a mixture at process start-up that can support the conversion of the material fall within a range of 6.5-8.0 (Temel, 2023). As can be seen from Table 3, the pH of blends (6.4 - 8.0) was precisely within the optimal interval for an efficient aerobic conversion. Moreover, according to the literature (Aycan Dümenci et al., 2021; Singh et al., 2012; Vitinaqailevu and Rajashekhar Rao, 2019; Yılmaz et al., 2022), a pH value of final compost between 5.5-7.2 signifies a successful composting together with a sufficient organic matter conversion. The low pH in compost is mostly attributable to the bioconversion of organic matter and generation of organic acids under aerobic conditions (Gigliotti et al., 2012). In our case, all final compost materials exhibited a pH value between 5.82-6.30 indicating a remarkable conversion of the compostable organic matter contained in the blends.

3.2 Large-scale co-composting of sewage sludge

3.2.1 Composting dependence on sewage sludge type

The effect of SS type on the efficiency of composting was examined by comparing the temperature profiles at three pile depths provided by a mixture of a single SS with WCh (1:2) and those provided by a mixture of an SS blend (SSmix) with WCh at the same ratio. In all cases, the temperature attained at a depth of 30 cm from piles surface was higher than the temperatures at the depths of 80 and 130 cm (Figure 1).

For M3 and at 30 cm from pile surface, the temperature increased from 35°C on the process start up to 65°C after 20 days of process. In contrast, M2 reached a maximum temperature of 53°C on the 24th day, Figure 1. For the M3 mixture, the temperature remained at high level (60-65°C) and slightly dropped after the third turning on the 24th day but kept above 55°C for a total of sixteen days. A temperature between 45 and 55°C allows a material to attain a high biodegradation rate during composting (González et al., 2019). The bulking agent that was supplied in the piles likely assisted the mixtures M2 and M3 not only to attain a high temperature that could ensure sanitization but also to acquire a high decomposition rate for a long period. Interestingly, after a turning 55th day and until the 83rd day the temperature was stable at 43°C for M3 and 38°C for M2. In general, the mixture containing the SSmix performed much better than the mixture containing a single SS in all cases. The differences in the temperature profiles were significant with the SSmix reaching higher temperature than the single SS material. Even though most chemical and physical properties of the two mixtures were similar (Table 3), M3 contained a greater amount of VS (91.28% wt_{TS}) than M2 (73.31% wt_{TS}) which might be the main reason for the better performance for the specific mixture.

3.2.2 Effect of mixing ratio on composting process

The temperature in the composting piles M1 and M2 at depth 80 cm increased for three weeks (Figure 2), which is in agreement with previous studies investigated similar blends (Oviedo-Ocana et, al 2015). The temperature profiles indicate that the active composting phase started even by the first day (Cáceres et al., 2018). The maximum temperatures attained by the piles M1 (55°C) and M2 (54°C) were significantly close whereas appeared only for

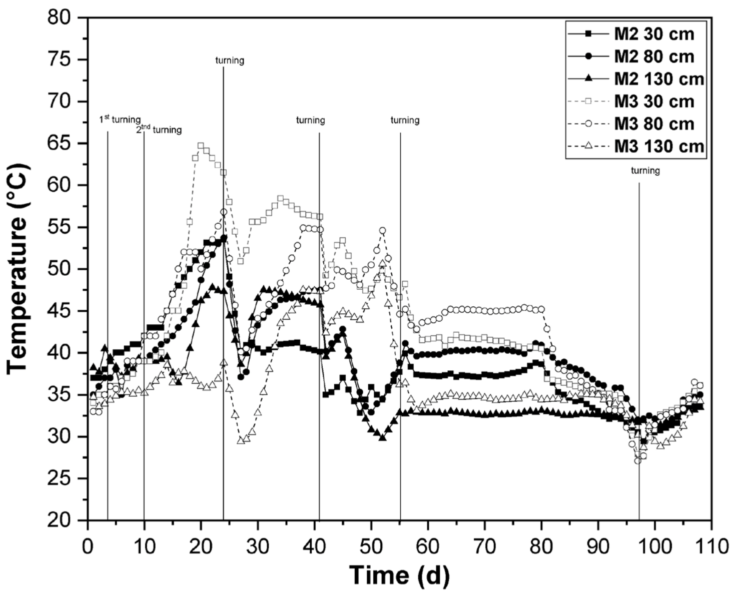

FIGURE 1: Temperature evolution at depths of 30, 80 and 110 cm from pile surface during composting of M2 (SS:WCh 1:2) and M3 (SSmix:WCh 1:2). Vertical lines indicate turning events.

a single day in the third week in both cases. This means that the high fraction of WCh had insignificant effect on the rate of decomposition of the organics in SS. The temperature in both piles dropped at lower levels after the third week. Theoretically, destruction of pathogens and sanitization of the materials in a mixture can be realized when the temperature in pile reaches values greater than 55°C and remains at those levels for at least three days (Oshins et al., 2022). It seems, therefore, that the piles M1 and M2 did not satisfy all the required temperature and time conditions that can ensure an efficient sanitization. On the other hand, according to the theory, thermophilic fungi that are responsible for the decomposition of lignin grow at 50°C and composting is competent when the temperature in the thermophilic phase of the process (>50°C) varies between 52-60°C (Miller, 1992; Tuomela et al., 2000). In our case, the temperatures reached in the piles impose that a sufficient composting could be achieved regardless of the WCh proportion in a mixture.

In spite of the moderate temperatures, microbiological evaluation, shown below, suggested that both final composts were properly sanitized by the process even though WCh did not significantly influence the maximum temperature in the thermophilic phase. In general, WCh contain a low fraction of biodegradable carbon and hence the quantity of the bulking material in the mixture would not be expected to provide any advancement in the temperatures gained during composting (Figure 2) (Tang et al., 2024). Nevertheless, when a low proportion of WCh participated in the mixture, the temperature was slightly higher from the 30th day and throughout the rest of composting process. In more detail, within the cooling phase, the mean temperature difference between the two mixtures was about 5-10°C. In any case, the difference was small enough that cannot be regarded as a consequence of the physicochemical characteristics of the materials or of any other parameters involved in composting. Therefore, it would be reasonable to assume that easily degradable organic compounds like carbohydrates, fats and amino acids that were presumably contained in SS degraded early in composting process, whilst cellulose, hemicellulose and lignin that were enclosed in WCh partially degraded and transformed slowly affecting almost negligibly the process evolution (Haug, 1993).

3.2.3 Effect of a secondary substrate and aeration process on temperature profiles

Figure 3 illustrates temperature profiles for the mixtures M3 and M4 consisted of SS:WCh (1:1) and SS:WCh:Grass (1:1:1). Temperature in M3 rapidly increased from 34 to 54°C within 20 days, which is a typical rise for the early days of composting. The thermophilic phase quickly emerged due to the generation of heat released by microbial degradation of any readily available and highly decomposable organic substances included in the mixture.

Temperature also increased after each turning of M3, but remained below the temperature securing sanitization (55°C). Nevertheless, the temperature remained over 40°C from the 15th to 80th day. Mixture M4 that contained amount of grass exhibited lower temperatures over the entire process. The profile was characterized by an increment from 40 to 45°C within 26 days and then a continuous drop that was not reversed by any turning event. In this case, green waste that replaced part of SS and WCh contained a significant proportion of lignocellulose that likely affected the activity of bacteria leading to a shorter thermophilic and a greater cooling phase. A similar effect of the lignocellulosic fraction of biomass on the composting process has been reported and attributed to the fact that a green waste is slightly more resilient than SS to microbial degradation due to the complicate structure of lignin and cellulose (Bustamante et al., 2012; Zhang 2023). Moreover, the fact that the mixture M3 contained a greater amount of VS (91.28% wt_{TS}) than M4 (85.00% wt_{TS}) (Table 3) may

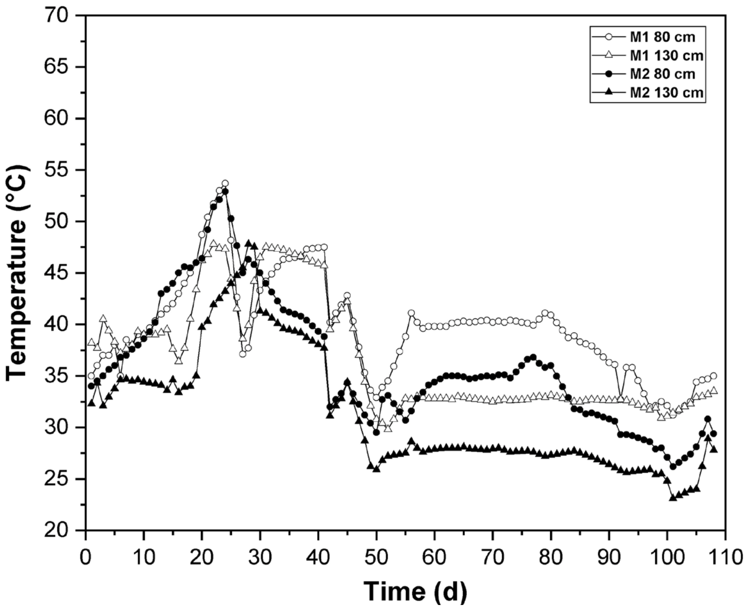

FIGURE 2: Temperature evolution at two depths (80 and 130 cm) during composting of M1 (SS:WCh 1:1) and M2 (SS:WCh 1:2).

also be reason for the superior performance of the mixture without green waste. It should be noted here that none of the mixtures reached a sufficient sanitization temperature level despite the observed advancement of the temperature after every turning. The beneficial impact of the turning procedure on the temperature was also reported by Şevik (2018).

In addition to turning, the aeration of the composting sludge was examined by preparing a pile with the mixture M5, which had the same composition with the mixture M4, and install a direct aeration system in the pile. Temperature in M5 reached a maximum at 54°C within approximately 15 days from the beginning of the trial (Figure 3). Thereafter, the temperature dropped, however, the thermophilic phase (>50°C) (de Bertoldi et al., 1983) started from the 10th and lasted for 18 days. Temperature declined at a constant rate during the mesophilic phase (40-50°C), however, from the 60th day the reduction became acute and temperature dropped from 39 to 18°C within 20 days. From this point onwards, composting process was deemed completed with the temperature maintaining close to the ambient. The fall-off in temperature denoted that most of the organic compounds were consumed during the thermophilic and mesophilic phases and that few easily degradable compounds remained to decompose within the cooling phase (Meng et al., 2017). On the other hand, the non-aeriated mixture M4 showed the highest temperature of 46°C on the 26th day and immediately entered in the cooling phase where remained until the end of the trial. The superior performance of M5 over M4 caused by the aeration, however, the temperature differences were limited, less than 20°C, over the entire process.

3.2.4 Effect of composting on mixtures contamination

Contamination of substrates with E. coli and Salmonella spp. was assessed with a determination of microbial populations before and after the composting process (Table 4). E. coli bacteria are often used as an indicative factor of the overall quality of soil and water. Colony forming units (CFU) per gram of substrate were determined in five replicates per sample with a limit at 50 CFU/g. Salmonella spp. was not detected in any case before and after the composting process. According to legislation (Decree No 474/2000) the determination of E. coli should be performed in five replicates and from the values obtained in them none should exceed 5000 CFU/g and four of them should be lower than 1000 CFU/g.

The values for E. coli before composting were counted at $5*10^4$ (CFU/g) for the mixtures M1, M2 and M3 and at $5*10^3$ (CFU/g) for the mixtures M4 and M5. However, the values considerably decreased to less than 50 CFU/g in all mixtures at the end of the process. The decrease was presumably a consequence of the temperature and the unfavourable conditions for microbial survival established during the thermophilic phase of the trials. According to de Bertoldi et al. (1983) the ideal temperature in thermophilic phase ranges between 40-65°C whereas temperatures above 55°C are necessary to destroy any coliforms. Here, in spite of the fact that the temperature was not at a critical level for an efficient coliform destruction, the concentration of E. coli reduced by at least three orders of magnitude by the composting process in all cases.

3.2.5 Heavy metals fate

Co-composting of SS with other materials can reduce the concentration of heavy metals due to dissolution, adjust the moisture of waste mixture to optimal values and provide a high-quality compost in terms of stability, maturity and agronomic and environmental parameters (Proietti et al., 2016). In particular, co-composting of sewage sludge and WCh can successfully lead to low heavy metal concentrations in the compost compared to the pre-composting

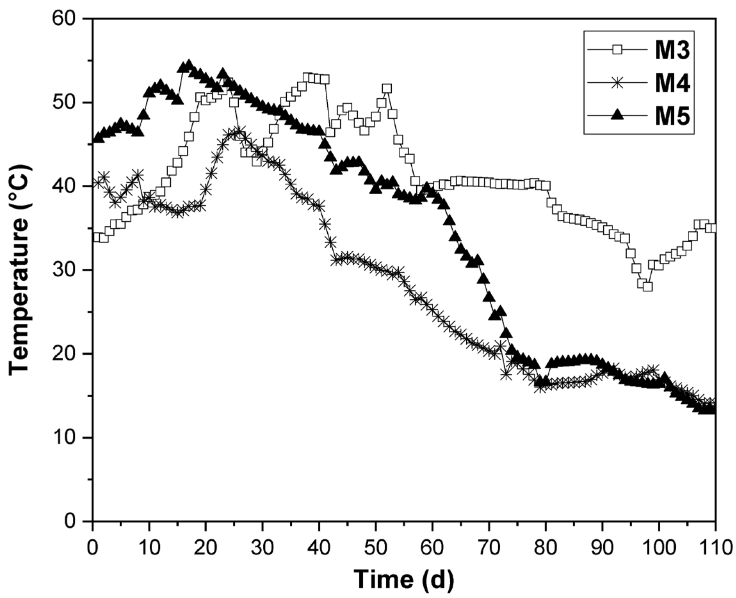

FIGURE 3: Average temperature in three depths (30, 80 and 130 cm) from pile surface during composting of mixtures M3 (SSmix:WCh 1:2), M4 (SSmix:WCh:G 1:1:1 without aeration) and M5 (SSmix:WCh:G 1:1:1 with aeration).

TABLE 4: Contamination of substrates with *E. coli* and *Salmonella spp.* before and after composting.

Mixture		*Escherichia coli*	*Salmonella spp.*
		(CFU/g)	
M1	initial	$5*10^4$	NEGATIVE
	final	<50	NEGATIVE
M2	initial	$5*10^4$	NEGATIVE
	final	<50	NEGATIVE
M3	initial	$5*10^4$	NEGATIVE
	final	<50	NEGATIVE
M4	initial	$5*10^3$	NEGATIVE
	final	<50	NEGATIVE
M5	initial	$5*10^3$	NEGATIVE
	final	<50	NEGATIVE

mixture (Mohee and Soobhany, 2014). This is consistent with the results in Table 3, where the fraction of heavy metals fell with the co-composting of SS and SSmix with WCh and/or grass. The reduction was likely attributable to a dissolution of each individual metal element into the aqueous phase existing in the composting system. It is interesting to note that a decrease in the concentration of all heavy metals in SS and SSmix was primarily occurred along with the preparation of mixtures of the wastes with WCh and grass (Table 2 and 3) and was presumably ascribed to the fact that the lignocellulosic materials contained a low or negligible amount of metal substances. Hence, the composting process itself appears to assist in attaining a low concentration of heavy metals in the solid effluent of a composter.

From Table 3, the concentration of each of the heavy metals was highly variable in the various mixtures. A similar wide variation was described in earlier studies (Ščančar et al., 2000). Table 3 presents the maximum concentrations of the most important heavy metals in a compost that would allow it to be considered appropriate for agricultural use according to the legislation. Zn proportion exceeded the limit value in M3, M4 and M5 before composting (320-340 mg kg^{-1}). This is not surprizing, however, since earlier studies (Gondek et al., 2018; Wang et al., 2005) reported that Zn concentration in municipal SS is usually the highest among the monitored heavy metals and SS frequently contains Zn at proportions approaching a 1600 mg kg^{-1}. Concentrations dropped below boundaries with the treatment. In addition to Zn, Ni proportion in the mixture M1 (61.5 mg kg^{-1}) exceeded the limit of 50 mg kg^{-1}. However, the concentration of the element reduced 3.75-fold with the composting process gaining a value that was far smaller than the upper boundary (Table 3). The proportion of other elements such as Cr and Pb were inside the limits before composting and became even smaller in the final composts. Cd is one of the most mobile heavy metals in the soil and restrictive factor for any land application of SS (Zheng et al., 2020). In all mixtures, Cd ranged in the initial sludge between 0.57-2 mg kg^{-1} whereas varied between 0.36-1.61 mg kg^{-1} in the final compost.

3.3 Ecotoxicity of composts

3.3.1 Vibrio fischeri

A very low luminescence inhibition was observed for the composts M2 and M3 whereas a very low luminescence stimulation (negative values of inhibition) was found for M1, M4 and M5 (Table 5). However, the values of luminescence inhibition oscillate around zero and therefore it can be concluded that none toxic effect existed. The results are consistent with the findings of Roig et al. (2012), who assessed the ecotoxicity of sewage sludge depending on the type of sludge treatment. Therein, composting was identified as the most favourable treatment for attaining a low ecotoxicity.

3.3.2 Sinapis alba

Inhibition of root growth did not exceed a 30% in any of the M1-M5 composts (Table 5). In addition, a root growth stimulation (negative inhibition value) was observed for mixtures M4 and M5. The potential toxicity of composts may be due to the high nitrogen content (Sowinsky et al., 2023), with the source of nitrogen being sewage sludge. However, it should be stressed that the S. alba root growth inhibition values around 20% are still reported as non-toxic.

3.3.3 Hordeum vulgare L.

Cultivation experiments with H. vulgare L. were carried out for all the generated composts. Germination of the seeds was not inhibited and no statistically significant differences were found compared to the control substrate. In addition to the germination, the nitrogen index (NBI) and the chlorophyll content in fully developed leaves were determined for 4 weeks old emerged plants. Both parameters were estimated by using a Dualex (FORCE-A, Germany), whereas the chlorophyll content was assessed using the leaf transmittance ratio at far-red and near-infrared wavelengths, while the accumulation of flavonols was measured using the differential ratio of chlorophyll fluorescence. The Normalized Biomass Index (NBI) was defined as the concentration ratio of chlorophyll to UV-absorbing phenolics, primarily flavonols. The results showed a higher sensitivity of the plants to the substrates used than in the case of germination. Composts M2 and M3 did not show a significant influence on nitrogen index and chlorophyll content compared to the control. However, in the case of the mixture M1, a conclusive decrease of both parameters was observed. Finally, the dry weight of the above-ground biomass was determined and no statistically significant difference was observed for substrates M1-M3 compared to a certified control substrate.

4. CONCLUSIONS

Two different SS streams were subjected to co-composting at industrial scale with a bulking agent (WCh) and grass to determine the effect of process parameters and SS type on the conversion potential and sanitization of the waste material. The mixture that contained a blend of three different SS (SSmix) reached a higher temperature than the mixture contained only a single SS. The presence of

TABLE 5: Ecotoxicity tests with *V. fischeri* and *S. alba*.

Ecotoxicity test	Compost				
	M1	M2	M3	M4	M5
	(%)				
V. fischeri luminescence inhibition (15 min)	-0.07±2.66	2.21±2.58	1.80±5.62	-0.25 ±3.15	-1,32±4.18
V. fischeri luminescence inhibition (30 min)	-0.76±1.99	2.55±1.15	0.84±1.02	-1.12±3.45	-2.58±4.92
S. alba root growth inhibition	23	24	7	-14	-8

a bulking agent assisted the materials not only to attain a high temperature that could ensure sanitization but also to acquire a high decomposition rate for a long period. An increased fraction of WCh, however, had an insignificant effect on the rate of decomposition of the organics in SS. Likewise, the presence of grass in a mixture negatively affected the extent of the thermophilic phase and more promoted the cooling phase presumably due to the complicate lignocellulosic structure of grass that is resilient to microbial degradation. Aeration promoted the composting performance of a mixture although the increment of temperature attained was less than 20°C over the entire process. In any case, aeration appears to be an essential parameter and under conditions a sufficient solution in accomplishing the required temperature conditions for comprehensive sanitization of an SS mixture. Co-composting of SS or SS-mix with WCh and grass successfully reduced the concentration of all heavy metals in the solid fraction likely due to a dissolution in the aqueous phase. All composts provided positive results in ecotoxicological evaluation and cultivation experiments where none significant inhibitory effect on the growth of spring barley was observed.

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REFERENCES

Albrecht, R., Le Petit, J., Calvert, V., Terrom, G., Perissol, C., 2010. Changes in the level of alkaline and acid phosphatase activities during green wastes and sewage sludge co-composting. Bioresour. Technol. 101(1), 228-233. https://doi.org/10.1016/j.biortech.2009.08.017.

Ammari, T.G., Al-Omari, Q., Abbassi, B.E., 2012. Composting sewage sludge amended with different sawdust proportions and textures and organic waste of food industry – assessment of quality. Environ. Technol. 33(14), 1641-1649. https://doi.org/10.1080/09593330.2011.641589.

Aycan Dümenci, N., Cagcag Yolcu, O., Aydın Temel, F., Turan, N.G., 2021. Identifying the maturity of co-compost of olive mill waste and natural mineral materials: Modelling via ANN and multi-objective optimization. Bioresour. Technol. 338, 125516. https://doi.org/10.1016/j.biortech.2021.125516.

de Bertoldi, M., Vallini, G., Pera, A., 1983. The biology of composting: A review. Waste Manag. Res. 1(2), 157-176.https://doi.org/10.1016/0734-242X(83)90055-1.

Bustamante, M.A., Alburquerque, J.A., Restrepo, A.P., de la Fuente, C., Paredes, C., Moral, R., Bernal, M.P., 2012. Co-composting of the solid fraction of anaerobic digestates, to obtain added-value materials for use in agriculture. Biomass Bioenergy 43, 26-35. https://doi.org/10.1016/j.biombioe.2012.04.010.

Cáceres, R., Malinska, K., Marfa, O., 2018. Nitrification within composting: A review. Waste Manag. 72, 119-137.https://doi.org/10.1016/j.wasman.2017.10.049.

El Fels, L., Zamama, M., El Asli, A., Hafidi, M., 2014. Assessment of biotransformation of organic matter during co-composting of sewage sludge-lignocelullosic waste by chemical, FTIR analyses, and phytotoxicity tests. Int. Biodeterior. Biodegradation 87, 128-137. https://doi.org/10.1016/j.ibiod.2013.09.024.

Gajalakshmi, S., Abbasi, S.A., 2008. Solid waste management by composting: state of the art. Crit. Rev. Environ. Sci. Technol. 38(5), 311-400. https://doi.org/10.1080/10643380701413633.

Gigliotti, G., Proietti, P., Said-Pullicino, D., Nasini, L., Pezzolla, D., Rosati, L., Porceddu, P.R., 2012. Co-composting of olive husks with high moisture contents: Organic matter dynamics and compost quality. Int. biodeterior. biodegrad. 67, 8-14. https://doi.org/10.1016/j.ibiod.2011.11.009.

Gondek, K., Mierzwa-Hersztek, M., Kopec, M., 2018. Mobility of heavy metals in sandy soil after application of composts produced from maize straw, sewage sludge and biochar. J. Environ. Manage., 210, 1-2. https://doi.org/10.1016/j.jenvman.2018.01.023.

González, D., Colón, J., Gabriel, D., Sánchez, A., 2019. The effect of the composting time on the gaseous emissions and the compost stability in a full-scale sewage sludge composting plant. Sci. Total Environ. 654, 311-323. https://doi.org/10.1016/j.scitotenv.2018.11.081.

Haug, R.T, 1993. The practical handbook of compost engineering (1st ed.). Routledge. https://doi.org/10.1201/9780203736234.

Komilis, D., Kontou, I., Ntougias, S., 2011. A modified static respiration assay and its relationship with an enzymatic test to assess compost stability and maturity. Bioresour. Technol. 102, 5863-5872. https://doi.org/10.1016/j.biortech.2011.02.021.

Meng, L., Li, W., Zhang, S., Wu, Ch., Lv, L., 2017. Feasibility of co-composting of sewage sludge, spent mushroom substrate and wheat straw. Bioresour. Technol. 226, 39-45. https://doi.org/10.1016/j.biortech.2016.11.054.

Miller, F.C., 1992. Composting as a process based on the control of ecologically selective factors. In: Metting, F.B.J. (Ed.), Soil microbial ecology, applications in agricultural and environmental management. Marcel Dekker Inc., New York, pp. 515-544.

Mohee, R., Soobhany, N., 2014. Comparison of heavy metals content in compost against vermicompost of organic solid waste: past and present. Resour. Conserv. Recycl. 92, 206-213. https://doi.org/10.1016/j.resconrec.2014.07.004.

Muktadirul Bari Chowdhury, A.K.M., Akratos, C.S., Vayenas, D.V., Pavlou, S., 2013. Olive mill waste composting: a review. Int. Biodeterior. Biodegrad. 85, 108-119. https://doi.org/10.1016/j.ibiod.2013.06.019.

Oshins, C., Michel, F., Louis, P., Richard, T.L., Rynk, R., 2022. Chapter 3 - The composting process. In: Rynk R, editor. The Composting Handbook. Academic Press (2022), 51-101. https://doi.org/10.1016/B978-0-323-85602-7.00008-X.

Oviedo-Ocana, E.R., Torres-Lozada, P., Marmolejo-Rebellon, L.F., Hoyos, L.V., Gonzales, S., Barrena, R., Komilis, D., Sanchez, A., 2015. Stability and maturity of biowaste composts derived by small municipalities: Correlation among physical, chemical and biological indices. Waste Manag. 44, 63-71. https://doi.org/10.1016/j.wasman.2015.07.034.

Peltre, C., Nyord, T., Bruun, S., Jensen, L.S., Magid, J., 2015. Repeated soil application of organic waste amendments reduces draught force and fuel consumption for soil tillage. Agric. Ecosyst. Environ. 21 1, 94-101. https://doi.org/10.1016/j.agee.2015.06.004.

Ponsá, S., Gea, T., Sánchez, A., 2010. The effect of storage and mechanical pretreatment on the biological stability of municipal solid wastes. J. Waste Manag. 30 (3), 441-445. https://doi.org/10.1016/j.wasman.2009.10.020.

Proietti, P., Calisti, R., Gigliotti, G.,Nasini, L., Regni, L., Marchini, A., 2016. Composting optimization: Integrating cost analysis with the physical-chemical properties of materials to be composted. J. Clean. Prod. 137. 1086-1099. https://doi.org/10.1016/j.jclepro.2016.07.158.

Raj, D., Antil, R.S., 2011. Evaluation of maturity and stability parameters of composts prepared from agro-industrial wastes. Bioresour. Technol. 102 (3), 2868-2873. https://doi.org/10.1016/j.biortech.2010.10.077.

Romero, L., Moreno, Juan F., Oulego, P., Collado, S., Díaz, M., 2024. Surfactant-assisted thermal hydrolysis of sewage sludge: Biomolecule extraction and its potential for protease production. J. Environ. Chem. Eng. 12 (5), 113353. https://doi.org/10.1016/j.jece.2024.113353.

Roig, N., Sierra, J., Nadal, M., Martí, E., Navalón-Madrigal. P., Schuhmacher, M., Domingo, JL, 2012. Relationship between pollutant content and ecotoxicity of sewage sludges from Spanish wastewater treatment plants. Sci Total Environ. 425, 99-109. https://doi.org/10.1016/j.scitotenv.2012.03.018.

Şevik, F., Tosun, I., Ekinci, K., 2018. The effect of FAS and C/N ratios on co-composting of sewage sludge, dairy manure and tomato stalks, Waste Manage. 80, 450-455. https://doi.org/10.1016/j.wasman.2018.07.051.

Singh, S., Singh, B., Mishra, B.K., Pandey, A.K., Nain, L., 2012. Microbes in Agrowaste Management for Sustainable Agriculture. In: Satyanarayana, T., Johri, B.N., Prakash, A. (Eds.), Microorganisms in Sustainable Agriculture and Biotechnology. Springer, Netherlands.

Smith, S.R., 2009. A critical review of the bioavailability and impacts of heavy metals in municipal solid waste composts compared to sewage sludge. Environ. Int. 35, 142-156. https://doi.org/10.1016/j.envint.2008.06.009.

Sreesai, S., Peapueng, P., Tippayamongkonkun, T., Sthiannopkao, S., 2013. Assessment of a potential agricultural application of Bangkok-digested sewage sludge and finished compost products. Waste Manage. Res. 31 (9), 925-936. https://doi.org/10.1177/0734242X13494261.

Ščančar, J., Milačič, R., Stražar, M., Burica, O., 2000. Total metal concentrations and partitioning of Cd, Cr, Cu, Fe, Ni and Zn in sewage sludge. Science of The Total Environment 250, 9-19. https://doi.org/10.1016/S0048-9697(99)00478-7.

Tang, J., Zhang, S., Zheng, G., Han, Z., Wang, D., Lin, H., 2024. Role of bioavailability in compost maturity during aerobic composting of chicken manure. Sustainability, 16(24), 11122. https://doi.org/10.3390/su162411122.

Temel, F.A., 2023. Evaluation of the influence of rice husk amendment on compost quality in the composting of sewage sludge. Bioresour. Technol. 373, 128748. https://doi.org/10.1016/j.biortech.2023.128748.

Tuomela, M., Vikman, M., Hatakka, A., Itavaara, M., 2000. Biodegradation of lignin in a compost environment: a review. Bioresour. Technol. 72, 169-183. https://doi.org/10.1016/S0960-8524(99)00104-2.

Villaseñor, J., Rodríguez, L., Fernández, F.J., 2011. Composting of domestic sewage sludge with natural zeolites in a rotary drum reactor. Bioresour. Technol. 102, 1447-1454. https://doi.org/10.1016/j.biortech.2010.09.085

Vitinaqailevu, R., Rajashekhar Rao, B.K., 2019. The role of chemical amendments on modulating ammonia loss and quality parameters of co-composts from waste cocoa pods. Int. J. Recycl. Org. Waste Agric. 8, 153-160. https://doi.org/10.1007/s40093- 019-0285-3.

Yılmaz, E.C., Aydın Temel, F., Cagcag Yolcu, O., Turan, N.G., 2022. Modeling and optimization of process parameters in co-composting of tea waste and food waste: Radial basis function neural networks and genetic algorithm. Bioresour. Technol. 363, 127910. https://doi.org/10.1016/j.biortech.2022.127910.

Yousefi, J., Younesi, H., Ghasempoury, S.M., 2013. Co-composting of municipal solid waste with sawdust: improving compost quality. Clean Soil Air Water 41 (2), 185-194. https://doi.org/10.1002/clen.201100315.

Wang, C., Hu, X., Chen, M.-L., Wu, Y.-H., 2005. Total concentrations and fractions of Cd, Cr, Pb, Cu, Ni and Zn in sewage sludge from municipal and industrial wastewater treatment plants. J. Hazard. Mater. 119, 245-249. https://doi.org/10.1016/j.jhazmat.2004.11.023.

Zhang, Z.J., Duan, C.Q., Liu, Y.X., Li, A.N., Hu, X., Chen, J.K., Zhang, S., Li, X., Che, R.X., Li, S.Y., Ekelund, F., Cui, X.Y., 2023. Green waste and sewage sludge feeding ratio alters co-composting performance: emphasis on the role of bacterial community during humification. Bioresour. Technol. 380, 129014. https://doi.org/10.1016/j.biortech.2023.129014.

Zheng, G., Wanga, X., Chena, T., Yanga, J., Yanga, J, Liu, J., Shia, X., 2020. Passivation of lead and cadmium and increase of the nutrient content during sewage sludge composting by phosphate amendments. Environ. Res. 185, 109431. https://doi.org/10.1016/j.envres.2020.109431.