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Investigation on greenhouse gas emissions and compost dynamics during in-vessel co-composting of septage and mixed organic wastes

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Abstract

Greenhouse gas emission during composting process is inevitable, although it relies on natural degradation. A good balance on carbon-to-nitrogen (C/N) ratio as well as the aeration mode could significantly control the emissions. The C/N ratio can be adjusted using optimal feedstock composition. The aeration can be done using passive and active modes. In the present study, greenhouse gas emissions during in-vessel co-composting of septage and organic waste with active and passive modes of aeration were investigated. This fills the gap in implementing suitable septage treatment practice. Retention of high temperature ($> 55\ °C$) for prolonged period ensured the compost hygienization. The active aeration system showed significantly lower greenhouse gas emission compared to that of passive system. The contribution of CH_4 and N_2O to the total greenhouse gas was estimated to be 2.5% and 7.4% in the passive and 0.9% and 3.8% in the active aeration modes while $> 90\%$ was CO_2. The resulted compost maturity was confirmed by seed germination index ($> 80\%$) and NO_3^-/NH_4^+ index (> 6.25). Our study highlight that in-vessel co-composting with active aeration is a suitable option to convert waste to a nutrient-rich fertilizer in a shorter period compared to other septage treatment options.

Keywords Compost dynamics · Fertilizer · Greenhouse gas emissions · In-vessel co-composting · Septage

Introduction

Septage management is a major problem in many low- and middle-income nations. Septage is the material that accumulates in septic tanks and is not transported through sewers. It is a host for many pathogens like bacteria, parasites and viruses that can cause serious health threats (Yen-Phi et al. 2010). Globally, 2.7 billion people rely on onsite sanitation systems like septic tanks for wastewater treatment (Strande et al. 2014). In India, 38.1% of the total

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sanitation is served by septic tanks (Census of India 2011) and the number of septic tanks will grow steeply in the next few years. The estimated rate of septage generation in India is 0.12 million tonnes per day (Census 2011; Radford and Sugden 2014). However, there is no proper septage management strategy adopted in India. The existing practice of septage management in India is co-treatment, i.e., combined treatment of sewage and septage in sewage treatment plants (STP). Such practice is applicable in places where there is an access to properly managed STP. The major drawback of co-treatment is that the treatment efficiency of the STP might deteriorate in the long run. Hence, there is a pressing need to adopt a suitable treatment technology for septage management in peri-urban and rural areas where properly managed STPs do not exist.

Though septage is detrimental, it has got substantial nutrient value (Prabhu et al. 2015). However, no efforts have been made in the past to recover and convert it into useful resource. Among the different waste treatment options, composting has achieved wide acceptance due to its many advantages like economically viability, waste sanitization and value-added product generation (Onwosi et al. 2017). Composting is a biological

process which converts organic wastes into stable end products like humus (can be used as soil conditioner or fertilizer) and carbon dioxide (CO_2). Since septage has low C/N ratio, it needs to be co-composted with complementary wastes. Fernando et al. (2014) explored the possibility of co-composting of solid waste and fecal sludge at field scale in Sri Lanka. They produced good quality compost and demonstrated that the combined treatment of septage and mixed organic waste not only resolves major threats of management problems but also produces value-added products out of wastes. In many developing countries where septage treatment is not carried out, it is simply disposed into the environment. The untreated septage then undergoes partial degradation, releases greenhouse gases (GHGs) and contaminates the water bodies (Diaz-Valbuena et al. 2011). Similarly, the unsegregated organic fraction of municipal solid waste ends up mostly in poorly managed landfills, which are potential hot spots for GHG emission. Yoshida et al. (2012) suggested that the GHG emissions will be reduced to a great extent if the waste disposal takes a diversion from landfills to composting. However, significant amount of GHG emission is reported even from well-aerated composting systems (Sánchez et al. 2015). Therefore, it is essential to monitor and quantify the GHG emission during septage co-composting. To the authors' knowledge, very few studies have been undertaken to assess GHG emission during septage treatment. The major sectors influencing GHG emissions during composting include initial feed stocks, composting process and compost end use. The storage of feedstocks and feedstocks not decomposed during composting can lead to GHG emissions (Sánchez et al. 2015). Ermolaev et al. (2014) revealed that several process parameters like temperature, pH, free air space (FAS), carbon-to-nitrogen (C/N) ratio, moisture content and density affect the compost operation significantly and influence the GHG emission from the systems. The end use of compost has many direct and indirect benefits on total GHG emission avoidance. The major one among them is carbon sequestration potential of final compost. However, this potential is often difficult to assess as it depends on the difference in initial total carbon content and total GHG emitted in CO_2–C equivalents (DeLonge et al. 2013). Hence, an overall understanding of GHG emission in all these steps of composting is essential for assessing the sustainability of composting process for septage treatment.

The global warming potential (GWP) of N_2O and CH_4 is 310 and 21, respectively. Since CO_2 is of biogenic origin, it is not accounted in the total GHG budget (IPCC 2006). Methane is produced by methanogens under anaerobic conditions while nitrous oxide is produced by nitrifiers and denitrifiers under aerobic and anoxic conditions, respectively (Ermolaev et al. 2014). The establishment of these composting microbial communities is dependent on the various process parameters such as temperature, pH and moisture content (Sánchez et al. 2015). Good composting practices that balance the C/N ratio and provide aeration could minimize the GHG emission significantly. Even though the optimum C/N ratio and moisture content range reported for composting are 25:1–30:1 and 55–65%, respectively, recent studies reported the feasibility of composting at low C/N ratio and high moisture content to obtain good quality compost (Kumar et al. 2010). The main advantages of composting at low C/N ratio include treatment of large quantity of septage and mixed organic waste which are rich in nitrogen content and reduce the bulking agent addition. As a result, the overall cost for composting can be reduced. Though composting at low C/N ratio yielded quality compost, the related GHG emissions likely to occur at extreme conditions need to be addressed. Many studies have been conducted in the past to comprehend the influence of aeration rate and aeration method on GHG emissions during composting (Guo et al. 2012; Jiang et al. 2011). However, seldom studies were carried out to understand the effect of aeration mode (passive and active mode) on GHG emission as it can significantly affect the aerobic microenvironment for composting. The aerobic microenvironment of the compost matrix can be determined by measuring the porosity and FAS.

In the present study, co-composting of septage and mixed organic waste was investigated using a laboratory-scale in-vessel reactor. The complete process dynamics during co-composting were studied. Experiments were conducted at a low C/N ratio of initial compost mix to understand the maximum possible GHG emissions from the compost matrix. Moreover, the effect of aeration mode on GHG emissions from in-vessel system and the carbon sequestration potential of the final compost were investigated. Maturity indices, namely seed germination index and nitrate to ammonium (NO_3^-/NH_4^+) index, were determined to ensure the quality of ripened compost so that it can be used as a plant fertilizer.

Materials and methods

Septage pre-treatment

The septage was collected from septage collection facility at Nesapakkam STP, Chennai city, India. The physicochemical parameters such as total solids (TS), total suspended solids (TSS), volatile solids (VS) and volatile suspended solids (VSS), chemical oxygen demand (COD),

biochemical oxygen demand (BOD), ammonia–nitrogen (NH_4^+–N), total nitrogen (TN) and total phosphorous (TP) were analyzed. The analyses were carried out as per standard methods for the examination of water and wastewater (APHA 2012). TS, VS, TSS and VSS were determined using gravimetric method (2540 D); BOD was analyzed by Winkler's method (5210 B); COD was determined by closed reflux method (5220 C); NH_4^+–N was analyzed using ion chromatography (Dionex, USA); TN was measured using total organic carbon analyzer V600 series (Shimadzu, Japan); and TP (4500-P-B) was measured using UV–Vis spectrometer 800 (Shimadzu, Japan) (APHA 2012). The detailed procedure is given in Krithika et al. (2017). During pre-treatment, septage was dewatered using sand drying bed to increase the concentration of solids. The laboratory-scale dewatering setup of a sand drying bed consisted (Fig. S1) of a 100-L-capacity overhead septage storage tank, inlet pipes, drying bed, under-drain pipe (1% slope) and a filtrate collection tank. The experimental procedure used for dewatering in the present study was adopted from Cófie et al. (2009).

In-vessel composter experimental design and sample analysis

The composting experiments were carried out in vertical drums (50 L) having a diameter of 350 mm and height of 600 mm. Passive aeration was achieved by providing 5-cm-diameter holes (eight numbers) along the perimeter at the bottom side of the drum, while in active

system, intermittent aeration was provided using an air blower (Bosch GBL 800 E, Germany) with an air flow rate of 0.5 vvm (volume of aeration per working volume of composter per minute). The quantity of air required for composting was estimated based on stoichiometric calculations (Haug 1993), inclusive of dilution. In both actively and passively aerated composting systems, the compost matrix was supported by perforated mesh and insulated using glass wool with a thickness of 40 mm in order to retain the heat generated during the process. The schematic representation of experimental setup of in-vessel systems is shown in Fig. 1. Two temperature probes were inserted (one at 10 cm below compost surface and one at 10 cm above the porous mesh) in each composting system, and data were recorded using data logger (Engineering Gallery, India). Compost sampling was done once in 2 days, whereby approximately 200 g sample was collected from top, middle and bottom layers, respectively, and was homogeneously mixed for physicochemical testing. All the studies were done in triplicate to reduce the error percentage. The fresh compost samples were analyzed for pH and electrical conductivity (1:10 w/v compost: water extract) using Eutech cyberscan PCD 650 multiparameter kit (Thermo Scientific, Singapore), and moisture content was determined after drying the mixture to a constant weight at 105 °C in a forced hot air oven (Technico, India). Organic matter (OM) was analyzed using loss-on-ignition method by keeping oven-dried samples at 550 °C for 8 h. The OM concentration of the compost was computed from the ash
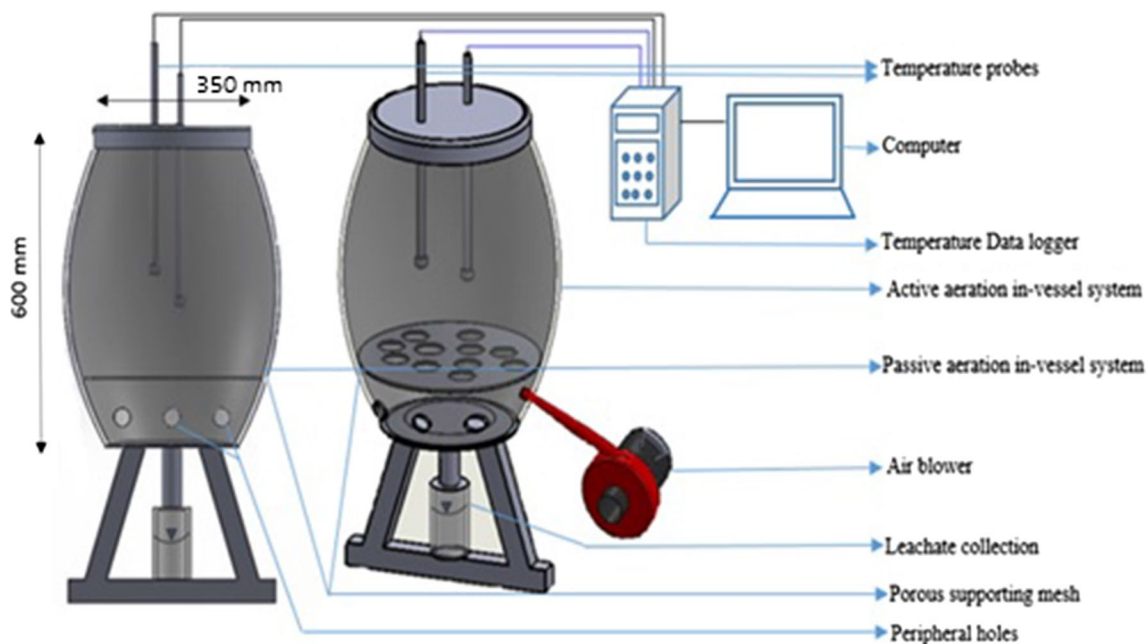

Fig. 1 Schematic representation of in-vessel composting experimental setup with passive and active aeration

content $[1 - (\text{ash content}/1000)]$. Subsamples were used for the analysis of total phosphorous (TP) and total potassium (TK) after acid digestion and were determined using stannous chloride method and atomic absorption spectrophotometry, respectively (APHA 2012). The NH_4^+–N and nitrate–nitrogen (NO_3^-–N) were extracted using 2M potassium chloride (KCl) solution and analyzed using colorimetric method (Nayak and Kalamdhad 2013). The total carbon (TC) and total nitrogen (TN) were analyzed using CHNS analyzer (Vario EL cube, Elementar Analysensysteme GmbH, Germany). Bulk density of the compost samples was determined by dividing the total mass of sample by the volume occupied. The particle density was measured using helium gas pycnometer (AccuPyc II 1340 Series, Germany). The porosity was calculated using the following equations (Eqs. 1, 2). Bacteriological quality of the compost was analyzed based on procedure given by Yen-Phi et al. (2010). The unit was expressed as number of colony forming units (CFU) per g dry weight (d.w.) of compost. Phytotoxicity test was carried out using radish seeds (*Raphanus sativus*) based on the procedure given by Kumar et al. (2010), and the seed germination index was calculated based on Eq. (3):

$$\text{Porosity, } n_p(\%) = \left(1 - (\rho_d/\rho_s)\right) \times 100 \tag{1}$$

$$\text{Free Air space, FAS (\%)} = (n_p - ((\rho_d \cdot W_g)/\rho_w \cdot (100 - W_g)) \times 100 \tag{2}$$

where ρ_d = dry density (g/cm³), ρ_w = water density (g/cm³), ρ_s = particle density (g/cm³) and W_g = water content (%).

Table 1 Characteristics of feedstocks and initial compost mix

Feedstocks/mixture	Moisture content (%)	C/N ratio	Mass used (kg)
Mixed organic waste	87 ± 2.2	18.1 ± 1.1	13.0
Dewatered septage	60.2 ± 2.7	8.2 ± 0.8	6.5
Coir pith	33.3 ± 1.2	34.5 ± 2.7	1.0
Wood chips	55 ± 0.9	40.9 ± 1.8	2.5
Cow dung	94.3 ± 2.2	9.4 ± 0.9	1.0
Initial compost mix	76.8 ± 1.2	15.7 ± 0.8	24.0

might not facilitate compost operation. Hence, cow dung was used as the booster for composting operation and was procured from neighborhood of IIT Madras campus. Hence, the major substrates need to be considered are mixed organic waste and septage which are mixed in the ratio 2:1 (wet basis) as per the study conducted by Cófie et al. (2009). The concept of co-composting is to treat both organic waste and septic tank waste streams and enrich the nutrient content in the final compost. The feedstock formulation used in the present study is given in Table 1. The total quantity of initial compost used in both systems was 24 kg with organic matter content of 69.5% (dry weight basis). In our previous studies, different initial C/N ratios 15, 20 and 25 were made on feedstock composition in order to understand the compost dynamics and final compost quality (Thomas et al. 2018). Results revealed that compost dynamics and quality were optimum at C/N 20. However, an initial C/N of

$$\text{Germination index} = \frac{(\text{Number of seeds germinated in compost extraxt} \times \text{Mean root length of seeds in compost extraxt} \times 100)}{(\text{Number of seeds germinated in control} \times \text{Mean root length of seeds in control})} \tag{3}$$

Feedstock composition

In the present study, mixed organic waste includes mostly vegetable scraps and cooked waste which were obtained from students dining facilities in Indian Institute of Technology (IIT) Madras campus, Chennai, India. Wood chips used as structural material were procured from the neighborhood of IIT Madras campus, whereas coir pith waste used as bulking agent was obtained from Kancheepuram district, Chennai, India. During composting, mostly wood chips and coir pith were not degraded due to their rigid structure and lignocellulosic content. Even though septage contains millions of bacteria, many are non-beneficial organisms that

15 also resulted in the generation of good quality compost with required time–temperature criteria which are essential for effective compost. But leachate generation is the major challenge faced at low C/N ratio. However, in case of real case scenarios, a low C/N ratio and higher moisture content could be achieved easily as increasing in the C/N ratio demands higher bulking agent addition which in turn reflects the total cost for composting. Hence, in the present study, a low C/N ratio (15.7) and high moisture content (76.8%) were adopted which mimicked a real case and measured the maximum possible GHG emission from passive and active aeration systems.

Headspace greenhouse gas measurements

The headspace GHG concentrations emitted from the compost surface of the in-vessel system were measured using static chamber technique. The static chamber consisted of a plastic funnel with the cutoff stem, replaced with a removable septum. The chamber was pushed to 1 cm into the composting pile, and gas samples were taken with head space at 5 min and 30 min using disposable syringes (Chan et al. 2011). Gas samples were transferred to degasified 4-mL glass vials (vacutainer) fitted with butyl rubber stoppers and were analyzed within 24 h by using gas chromatography. The GHGs like CH_4, N_2O and CO_2 were monitored every day during the composting process. The analytical procedures and detailed calculation for GHG emission flux and emission rate in terms of kg GHG/kg compost on wet basis are given in supplementary material (section 1).

Statistical analysis

All the results reported are the mean of three replicates analyzed for each sample in order to ensure statistical reliability of the data and representativeness of the samples. To understand the significant differences of various parameters between two aeration systems, two-tailed 't' test was conducted at 95% confidence interval. These analyses are used to assess the difference between two paired measurements. The null hypothesis for this test is that the mean of the difference is zero. Pearson correlation analysis was also carried out to understand the influence of composting parameters on greenhouse gas emissions. Both analyses were conducted using IBM SPSS 22.0.

Results and discussion

Septage characterization and dewatering

A detailed septage characterization was made by collecting samples from different points in Chennai city, Tamil Nadu, India, in our previous study (Krithika et al. 2017), and the data are illustrated in Table S1. The characterization results confirmed the variable nature of septage. In addition to the seasonal and regional variation, septage characteristics can vary due to different factors including cleaning frequency, number of households, vacuum suction ratio, pumping rate, etc. Septage was found to have a maximum total solid content of 1.7%. The high water content demanded the need for dewatering before co-composting. The solid content in the septage was increased using a sand drying bed. Table S2 shows the raw septage and the filtrate quality attained after dewatering. The sand drying bed removed > 80% of the total volume of water through percolation through the bed and the remaining via evaporation. The filtrate quality suggested the need for further treatment using waste stabilization pond or constructed wetland as it contains nutrient and organics. The total solids (TS) content in septage formed on the top of sand drying bed was found to be 30–35% after the completion of dewatering, i.e., after 4 days. The dewatered septage was removed from the filter bed manually. The removal of excess moisture can significantly reduce the bulking agent consumption by improving the porosity or air space in the matrix (Haug 1993). Since septage was found to have low C/N ratio (< 8), co-substrates were added to attain an optimum C/N ratio for composting.

Compost mix characteristics

The initial compost mix characteristics are given in Table 1. In the present study, septage was the main substrate used and mixed organic waste was used as the co-substrate. Meng et al. (2017) reported that the addition of co-substrates can enhance the composting efficiency as it improves the compost matrix conditions. The water holding capacity of coir pith was 12.3 g water per g dry weight, and C/N ratio was 41:1 (Springer and Heldt 2016). Hence, it can absorb the excess moisture content prevailing in the compost mixture and increase the overall C/N ratio. The wood chips could provide structural support to resist the compaction during composting and help in maintaining the free air space FAS (> 20% throughout the compost matrix) (Haug 1993). Cow dung (4.2% wet weight) was added to the compost mixture as it acts as a booster which could result in faster degradation of organic wastes. The feedstock composition for effective in-vessel co-composting was confirmed by analyzing the elemental composition of the initial compost mixture. The obtained elemental composition of initial compost was compared with the average composition of three basic components in organic matter, i.e., carbohydrates, fats and proteins. Liwarska-Bizukojc and Ledakowicz (2003) reported that from the C-mole formulate, it is possible to predict whether it is fat, protein or carbohydrate. Since the C-mole formula of the compost feedstock [$CH_{1.7}O_{1.34}N_{0.05}$] is in agreement with that of carbohydrate [$CH_{1.8}O_{0.9}$] rather than fats [$CH_{0.75}O_{0.25}$] and proteins [$CH_{0.86}O_{0.5}N_{0.14}$], it could able to confirm that the major components in the compost mix are carbohydrates. The highest CO_2 emission observed at the initial composting period also shows the evidence of the presence of readily degradable organics. These results revealed that the selected feedstock composition was rich in readily available organic matter even though the initial C/N ratio (15.7) was slightly lower than the optimum value.
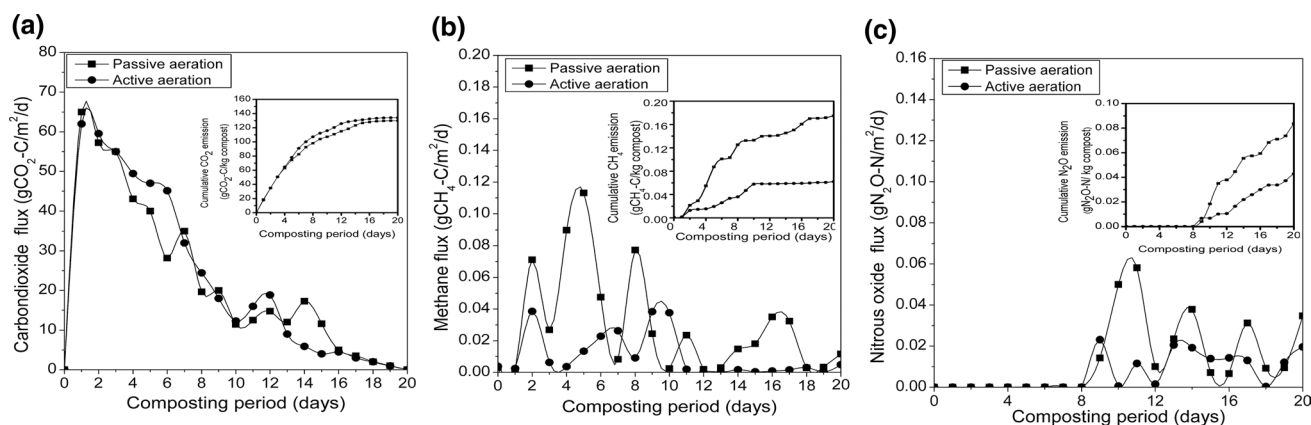
Fig. 2 Emission flux of **a** CO_2, **b** CH_4 and **c** N_2O during composting

Table 2 Paired t test analysis for difference in means

Parameters	t value (P value)
pH	0.53 (0.6083)
Organic matter content	2.96 (0.0143)
Temperature	3.14 (0.0051)
Moisture content	2.99 (0.0136)
Wet bulk density	0.00 (1.00)
Particle density	-0.43 (0.6657)
Porosity	0.47 (0.6641)
Free air space	-0.43 (0.6920)
Total carbon	8.17 (9.7×10^{-6})
Total nitrogen	0.96 (0.3592)
Ammoniacal nitrogen	-1.34 (0.2111)
Nitrate nitrogen	-0.73 (0.4796)
Carbon dioxide emission	-0.57 (0.5739)
Methane emission	2.62 (0.0164)
Nitrous oxide emission	1.98 (0.0611)

Greenhouse gas emissions during in-vessel co-composting of septage

Carbon dioxide emission

The emission flux of CO_2 from the composting system is given in Fig. 2a. The CO_2 emission peaked at thermophilic phase, and more than 80% of total CO_2 emission occurred during first 10 days of composting in both passive and active aeration systems. This is attributed to the availability of readily degradable organics (especially carbohydrates) at the early stage of composting. A decreasing trend in CO_2 emission showed that the microbial activity in the system got reduced due to depletion of compost substrates. The cumulative CO_2 emission flux was 454.2 g CO_2–C/m^2/d in passive aeration system and 469 g CO_2–C/m^2/d in active aeration system. The corresponding cumulative CO_2

emission was 130 g CO_2–C/kg waste and 134 g CO_2–C/kg waste in passive and active aeration systems, respectively (Table S3). The higher CO_2 emission is due to the availability of enough free air space (FAS) in the systems in addition to the high microbial activity and rapid degradation of organic matter. The emission of CO_2 in both the systems is very similar because the same feedstock composition was used in both systems, and in aerobic composting system, the major gaseous release is CO_2 (Sánchez-Monedero et al. 2011). The statistical analysis for variation in CO_2 emission between the two systems revealed that there is no significant difference ($t = -0.57$, $P = 0.5739$) between the systems (Table 2). CO_2 is not accounted for in the total GHG emission since the CO_2 emitted is of biogenic origin (IPCC 2006).

Methane emission

The emission flux of CH_4 observed during in-vessel co-composting of septage is given in Fig. 2b. The higher moisture content (76.8%) and low C/N ratio (15.7) created conducive condition for the development of anaerobic pockets. The CH_4 was found to be emitted throughout the composting operation in both active and passive systems. This might be due to the presence of high moisture (Sánchez-Monedero et al. 2011) and due to the heterogeneity of the compost matrix (Pardo et al. 2015). In addition, the undisturbed compost matrices are more susceptible to CH_4 emission due to the reducing level of oxygen in the system during the process. During thermophilic phase, oxygen is consumed rapidly for the decomposition of readily degradable organic matter in the compost matrix. The lower CH_4 emissions in active system than those in passive system could be due to the action of methanotrophs (aerobic methane-oxidizing bacteria).

It is reported that methanotrophs get established at the compost surface when sufficient amount of CH_4 was

present in the system (Ermolaev et al. 2014). The cumulative CH_4 emission flux was 0.6 g CH_4–C/m²/d in passive aeration system and 0.21 g CH_4–C/m²/d in active aeration system. The corresponding cumulative CH_4 emission was 0.17 g CH_4–C/kg waste and 0.06 g CH_4–C/kg waste in passive and active aeration systems, respectively (Table S3). Colón et al. (2012) reported that the methane emission from an in-vessel system, treating organic waste from source-separated municipal solid waste, was 0.26 g CH_4–C/kg waste. IPCC (2006) reported that the methane emission obtained during waste composting on dry basis is 10 g CH_4/kg waste. Hence, the CH_4 emission obtained during septage co-composting was lower than the emissions reported in the previous studies conducted on in-vessel composting of sludge and other wastes. The reason for higher CH_4 emission in passive system with higher moisture content might be due to the limited oxygen supply within composting substances caused by higher moisture content. The statistical analysis for variation in GHG emissions between the two systems revealed that CH_4 emission was significantly different between the two systems ($t = 2.62$, $P = 0.0164$).

Nitrous oxide emission

Figure 2c shows the emission flux of N_2O obtained during septage co-composting in in-vessel system in both active and passive aeration systems. In the present study, the N_2O generation was not observed during the early stage of composting. During initial stage of composting, CO_2 and ammonia generation was predominant due to the degradation of easily degradable compounds like carbohydrates and proteins. N_2O generation was found to have increased during the second half of composting in both passive and active aeration systems. Normally, the nitrifying bacteria or nitrifiers are active during maturation phase when the temperature would be preferably below 40 °C and aeration conditions are conducive (Sánchez et al. 2015). The key mechanisms for N_2O generation in the present study may be due to the initiation of nitrification process (oxygen is not limited) which converts soluble NH_4^+ into NO_3^-, the final nitrification product. Moreover, the N_2O generation can also be due to incomplete denitrification process, where nitrate was used as an electron acceptor when the composting pile was devoid of sufficient FAS. Incomplete denitrification occurs when the systems are not completely in anaerobic condition, i.e., have some access to oxygen (Ermolaev et al. 2014). If oxygen limitation occurs, then N_2O generation occurs via nitrifier denitrification (Sánchez-Monedero et al. 2011; Ermolaev et al. 2015). The N_2O emission from active system was lower than that in passive system throughout the composting operation. As a

result of intermittent aeration, FAS > 30% was persistent in active composting system throughout the composting process. This resulted in lesser N_2O emission. Similar N_2O emission pattern was reported by Fukumoto et al. (2003) during swine manure composting. The cumulative N_2O emission obtained from passive system was 0.27 g/m²/d and 0.14 g/m²/d from active system. The corresponding cumulative N_2O emission was 0.08 g N_2O/kg waste and 0.04 g N_2O/kg waste in passive and active aeration systems, respectively. The N_2O emission reported by Colón et al. (2012) was 0.048 g N_2O/kg waste from an in-vessel system, treating organic waste from source-separated municipal solid waste. IPCC (2006) reported N_2O emission as 0.6 g N_2O/kg waste during waste composting on dry basis. Hence, as in the case of CH_4 emission, the N_2O emission obtained during septage co-composting was lower than the emissions reported in the previous studies conducted on in-vessel composting of sludge and other wastes (Awasthi et al. 2017). However, the passive system emitted more N_2O than the active system, due to the favorable compost environment available in the passive system for nitrifiers and denitrifiers. The statistical analysis revealed that N_2O emissions have not much significant difference ($t = 1.98$, $P = 0.0611$) between the two systems.

Influence of compost dynamics on greenhouse gas emissions

The compost microorganisms like methanogens, nitrifiers, denitrifiers, and methanotrophs are dominant microbes responsible for GHG emissions from composting (Ermolaev et al. 2015). However, the growth and activity of the microorganisms highly depend on the change in the composting environment due to changes in temperature, pH, moisture content, organic and nutrient contents.

Temperature variation

Temperature of composting was considered as a crucial parameter that determines the success of composting process. Figure 3a shows the variation in temperature with time during in-vessel co-composting of septage in passive and active aeration systems. Both systems showed a sudden rise in the temperature soon after the start of composting due to high concentration of readily available biodegradable organic matter in the compost matrix. Higher temperatures (55–60 °C) lasted up to 5th day of composting and then slightly declined to 40–50 °C, indicating the termination of thermophilic phase and the establishment of mesophilic phase. More than 63% of the total emission of CO_2 occurred in both the systems during thermophilic phase of composting due to the conversion of majority of
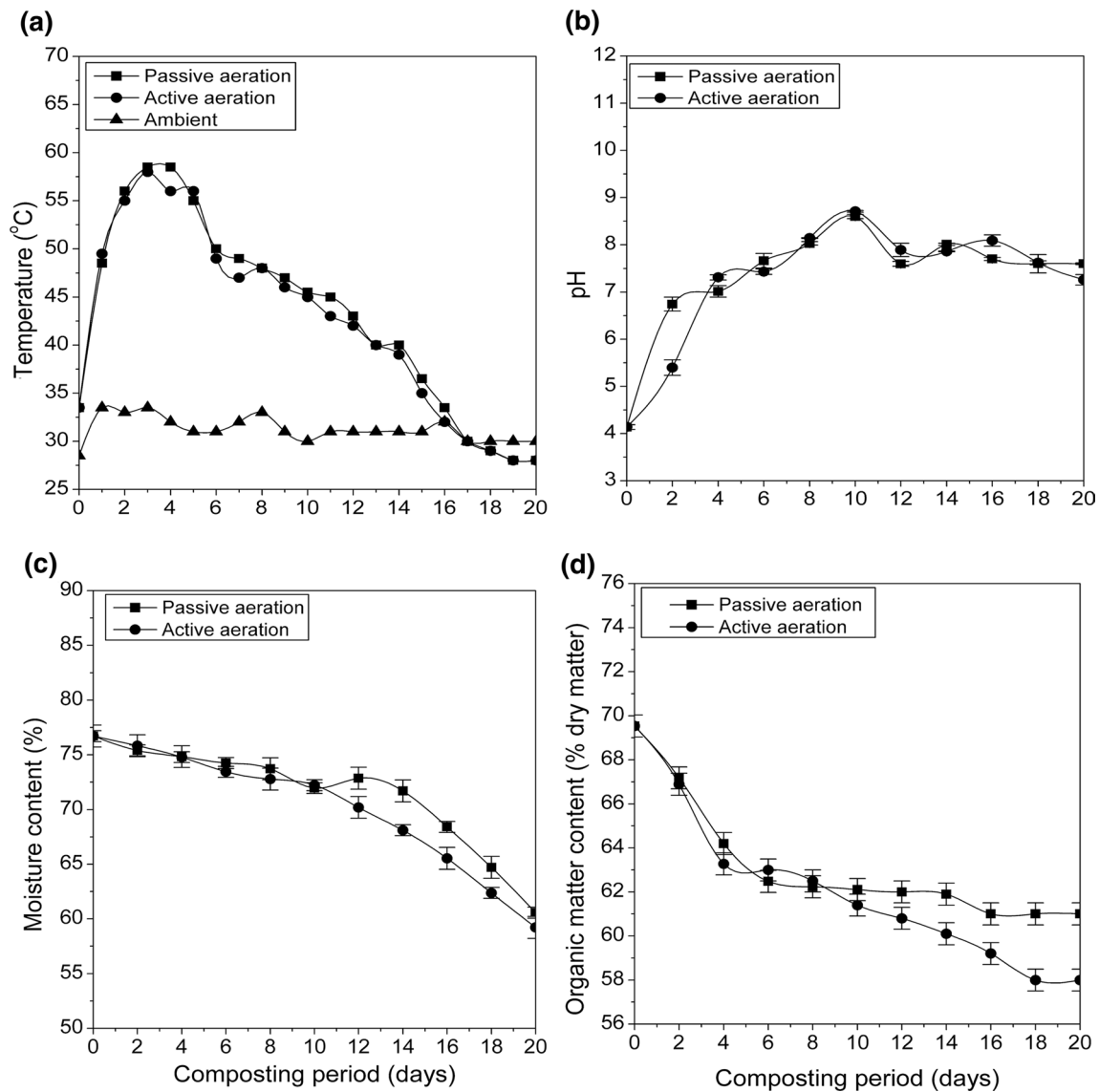
Fig. 3 Change in composting parameters during composting in passive and active aeration systems: **a** temperature profile, **b** pH pattern, **c** moisture content and **d** change in organic matter

organic matter. In passive system, the temperature showed significant correlation ($P < 0.05$) with CH_4 ($r = 0.761$) and CO_2 ($r = 0.909$) emissions (Table S4). Jiang et al. (2011) suggested that CH_4 emission is temperature sensitive, even though the CH_4 emission depends mainly on the oxygen level in the system. High temperature results in low oxygen solubility, which thereby creates anaerobic pockets leading to CH_4 formation. Accordingly, the CH_4 emission peaked in the passive system on the 5th day and in active system on the 2nd day during the course of composting. However, the amount of gas emitted depended on the prevailing size of anaerobic pockets (Sánchez-Monedero et al. 2011). More than 50% of the CH_4 was emitted during thermophilic phase while in active system,

the CH_4 emission was 41.8% during this phase. Unlike in the passive system, the temperature in active system did not correlate well with CH_4 emission ($P > 0.05$; $r = 0.539$) as given in Table S6. However, it correlated significantly with CO_2 emission ($P < 0.05$; $r = 0.915$). This is due to the supply of enough oxygen in the system through forced aeration.

Fukumoto et al. (2003) reported that the N_2O emission relies mainly on composting process temperature as the nitrifiers (responsible for N_2O production via nitrification) grow in mesophilic range (< 40 °C). Hence, the N_2O generation was observed only after 10 days of composting which was characterized predominantly by a mesophilic temperature range. However, in the present study,

the correlation of N_2O emission with temperature during composting is found to be insignificant ($P > 0.05$) in both active and passive systems. Similar findings were made by Amlinger et al. (2008) in which N_2O emission was found to be related to waste temperature. These differences occurred perhaps because of the varied size of the compost heaps used. However, the findings and correlation reported by Chan et al. (2011) were in agreement with the current study.

Wong et al. (2001) reported that the temperature development to an optimum range of 50–60 °C ensures effective composting. Further maturation period was attained and prolonged up to 18th day of composting. A maximum temperature of 45 °C was reported by Nayak and Kalamdhad (2013) during sewage sludge composting at C/N ratio of 15. Similarly, during co-composting of food waste and green waste at C/N ratio of 16.2, the temperature did not increase above 45 °C (Kumar et al. 2010). As far as composting is concerned, the thermophilic phase is vital for rapid and effective waste degradation and hygienization. The present study showed that the selected feedstock mixture is appropriate for proper septage treatment even at low C/N ratio as it met the regulatory requirement for effective composting. The paired t test analysis results (Table 2) showed that the variation in temperature between the two aeration systems during the composting period is significant ($t = 3.14$, $P = 0.0051$) at 95% confidence interval.

pH variation

The compost pH has crucial role in the growth and establishment of compost microorganisms as many of them are pH sensitive. In particular, methanogens accountable for CH_4 production can be active within certain pH and temperature ranges (Amlinger et al. 2008). However, growth of methanogens can be inhibited by acidic pH. Mesophilic temperatures and low pH existed in the beginning of the compost period, and therefore, a slow CH_4 production was observed (Ermolaev et al. 2015). The pH values for both passive and active aerated composting systems were monitored and are presented in Fig. 3b. The pH value at the start of composting was 4.1 and gradually increased and reached a maximum of 8.6 in passive system and 8.7 in active system. The pH then dropped during the later stages of composting and reached final values of 7.3 and 7.6 in active and passive systems, respectively. The reason for the initial low pH in the composting mixture was the presence of acidic components in the mixed organic waste. Since CO_2 is the major end product in aerobic composting, the CO_2 emission peaked during the thermophilic phase and triggered the increase in pH with the escape of CO_2 from the system. The protein

degradation resulted in ammonia release to the system and also contributed to the net increase in compost pH (Petric et al. 2012). The slight decrease in pH in both systems at the end of composting was due to the ammonia conversion as a result of the nitrification process by nitrifying bacteria. Similar to methanogens, the nitrifiers are both temperature and pH sensitive. Due to the presence of unfavorable conditions like high temperature and low pH, nitrification did not occur during the initial phase of composting (Sánchez-Monedero et al. 2011). Hence, N_2O was not emitted during the initial phase of composting. The pH observed throughout the composting process falls within the optimal range for the growth of different microorganisms which triggers the emission of GHGs during in-vessel co-composting of septage. However, in the present study, the compost pH was not significantly correlated ($P > 0.05$) with GHG emissions in both systems (Table S4). The pH of the final compost obtained in the study was similar to the pH ranges of 7.1–8.6 reported by Bernai et al. (1998). This makes the compost suitable for further use as a soil conditioner. Paired 't' test results (Table 2) showed no significant differences in pH between the two systems ($t = 0.53$, $P = 0.6083$) at 95% confidence interval.

Moisture content and loss in organic matter

Moisture loss is considered as an index of decomposition rate in composting process. It occurs due to the vaporization by the heat produced during organic matter degradation (Haug 1993). The presence of minimum moisture content in the composting matrix is essential for the micro-organisms to survive. The initial low C/N ratio (15.7) and high moisture content (76.8%) in the present study resulted in the development of anaerobic pockets. Wet feed stocks are more susceptible to CH_4 emissions. The high moisture and heterogeneity of feedstock materials resulted in CH_4 emission during the course of composting in both the systems (Sánchez et al. 2015; Pardo et al. 2015). The high moisture content restricts the air passage, creating oxygen-limited spots that lead to further N_2O emissions. Figure 3c shows the change in moisture content in both passive and active aeration systems. A decreasing trend was obtained during the progress in septage co-composting. Even though the initial feedstock has N-rich organic material, the emission of N_2O was not observed initially in both systems. The decrease in moisture favors the occupancy of oxygen during the course of composting which later resulted in N_2O emission. (Jiang et al. 2011; Sánchez-Monedero et al. 2011).

The efficiency of composting can be calculated based on the reduction in the organic matter (OM). The loss in OM is also an indicator for the significant emission
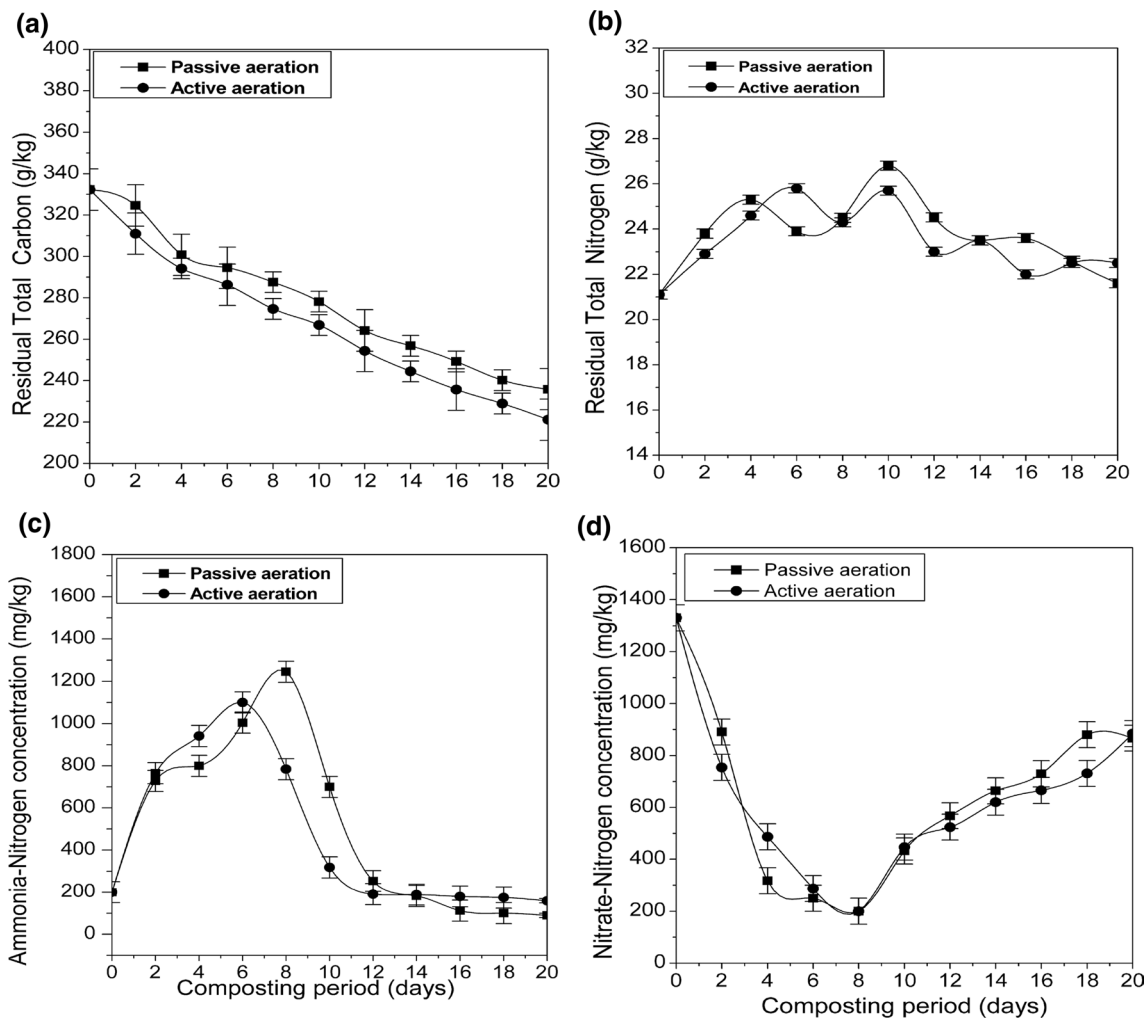
Fig. 4 Change in **a** residual TC, **b** residual TN, **c** ammonia–nitrogen and **d** nitrate–nitrogen during composting

from the compost matrix. The degradation of OM in passive and active aeration systems which occurred during the composting period is shown in Fig. 3d. The OM loss in the present study was 32.5% in the passive system and 36.3% in the active system. The maximum OM loss reported for dewatered sludge composting was 29.5% (Ponsá et al. 2009). The maximum OM loss observed by Kalamdhad et al. (2009) during mixed vegetable waste composting at C/N ratio of 16 was 76.5%. Since the septage was composted with mixed organic waste, the OM loss was found to be 30–40%, which is in between the values reported during composting of sludge and mixed organic waste treated separately. The *t* test analysis results showed that the difference in OM degradation between the two aeration systems during the composting period is significant ($t = 2.96$, $P = 0.0143$) at 95% confidence interval. Although there was a significant loss of organic matter during the course of composting, the organic matter content in the compost of either system is not correlated with

the GHG emissions ($P > 0.05$). For the statistical analyses, the number of points available was only 20. This might have affected the results.

Carbon and nitrogen decomposition during composting

The total carbon (TC) content and nitrogen (TN) content available in the compost are considered as the major energy source and cell building block for the compost microorganisms, respectively. They not only impact the compost quality but also the gaseous emissions. Figure 4a–d shows the change in residual TC, TN, NH_4^+–N and NO_3^-–N during the composting. The initial compost mixture had a low C/N ratio of 15.7, which is attributed to the large quantity of high N-containing wastes such as green vegetable waste, food waste, cow dung and septage. Majority of carbon was consumed by microbes for their energy and released as CO_2. The total carbon (TC) levels reduced from an initial 332.3

to 235.8 g/kg (passive) and 221.1 g/kg (active) after 20 days of composting (Table S5). Majority of carbon was lost during the initial active phase of composting, in both systems. This was due to the presence of readily available carbonaceous material, and as decomposition progressed, the TC content got decreased. This pattern of decreasing TC content was reported during mixed organic waste composting in rotary drum (Kalamdhad et al. 2009) and fecal sludge co-composting in windrow pile (Cófie et al. 2009). Even though the previous studies reported significant carbon and nitrogen decomposition, the potential GHGs that could be emitted during decomposition were seldom studied, especially during septage co-composting. In the present study, the initial decrease in TC content in the system is confirmed with the emission of CO_2 and CH_4 during the active phase of composting. The TC content in compost of both aeration systems is found to have insignificant correlation ($P > 0.05$) with GHG emissions during composting. This is due to lack of data points obtained during composting to execute the correlation study. During composting, the carbon loss was more than that of nitrogen. As a result, the percentage nitrogen increased in both systems. Consequently, the overall C/N ratio got significantly reduced. The C/N ratios of final composts were 9.8 and 10.9 in active and passive systems, respectively (Table S4). The TN increased from an initial 21.1 to 21.6 g/kg in passive system and 22.5 g/kg in active system at the end of composting. This increase is due to the net reduction in carbon in both the systems. The net loss in TC and TN during composting is given Table S5. The NH_4^+–N concentration was found to be higher during the early stages of composting. Degradation of organic matter was high during thermophilic period, which resulted in the production of NH_4^+–N by the mineralization of organic nitrogen. The NH_4^+–N

concentration reached a maximum of 1245.7 mg/kg in passive system and 1100.3 mg/kg in active system on 8th day and 6th day of composting, respectively. After thermophilic period, the NH_4^+–N content decreased due to NH_3 stripping and nitrification (Jiang et al. 2011). As mentioned earlier, the key mechanisms for N_2O generation observed in the present study were: (1) the nitrification process and (2) the incomplete denitrification process. The presence of ammonia, lower temperature and pH > 5 are favorable conditions for nitrifiers, which are responsible for nitrification that triggers N_2O emissions from the system. The final NH_4^+–N concentration was 89.7 mg/kg and 158.8 mg/kg in passive system and active system, respectively. The decrease in ammoniacal nitrogen concentration indicated a high quality of ripened compost (Wong et al. 2011). The initial high NO_3^-–N content (1330 mg/kg) was due to the higher NO_3^-–N concentration in cattle manure (Hao et al. 2004) and green vegetables such as spinach and carrot (Nabrzyski and Gajewska 1994). A decreasing trend was observed during active phase of composting in both systems due to high temperature and high NH_4^+–N content which inhibited the growth of nitrifying bacteria (nitrifiers). The nitrification process was observed mainly during the maturation period in both the systems. This is attributed to the conducive condition for nitrification which includes low temperature (< 40 °C) and less amount of NH_4^+–N in the system. The paired t test analysis results (Table 2) for TN, ammonia and nitrate content showed no significant differences between the two systems ($P > 0.05$), while a significant difference was observed for TC content between the two systems ($P < 0.05$). The TN, NH_4^+–N and NO_3^-–N contents in compost of both aeration systems did not correlate significantly ($P > 0.05$) with GHG emissions during septage co-composting.

Table 3 Physical properties and total phosphate contents during composting

Composting period (days)	Wet bulk density (kg/m^3)	Particle density (kg/m^3)	Porosity (%)	FAS (%)	Total phosphate (g/kg)
Passive aeration					
0	720	1650	89.8	34.6	3.1
5	740	1730	87.9	34.5	2.8
10	740	1670	88.9	33.5	3.3
15	790	1730	87.1	30.5	3.5
20	890	1760	80.0	26.1	3.6
Active aeration					
0	720	1650	89.8	34.6	3.1
5	770	1700	88.6	33.2	2.6
10	790	1720	87.3	30.2	3.2
15	780	1730	85.6	32.2	3.4
20	820	1770	81.1	32.5	3.7

Bulk density, particle density, porosity and free air space

The physical properties of compost such as wet bulk density, free air space (FAS), particle density and porosity play significant role in the maintenance of aerobic condition in the system during composting. Wet bulk density is the mass of wet material per unit volume of the material while particle density reflects the mineral solids in the material. Porosity is the volume of voids over the total volume of the material. FAS is a direct indication of oxygen availability in the system. FAS depends on the structural characteristics of compost materials such as wet bulk density, particle density and water content. Ruggieri et al. (2009) suggested that the correct level of FAS in the system ensures proper aeration of the compost matrix in both active and passive aeration systems and prevents anaerobiosis, thereby preventing the GHG emissions. The change in wet bulk density, free air space, particle density and porosity during composting is given in Table 3. In the present study, wet bulk density of compost mix increased from an initial value of 720 to 890 kg/m^3 and 820 kg/m^3 in passive and active systems, respectively. Similarly, the particle density increased from an initial value of 1650 to 1760 kg/m^3 and 1770 kg/m^3 in passive and active systems, respectively. These changes could be due to the volume reduction by natural compaction. Springer and Heldt (2016) reported that a wet bulk density of nearly 700 kg/m^3 of the initial compost mixture would be able to withstand the vertical load and able to preserve sufficient FAS in the compost matrix. Though the moisture content in the present study was higher than the preferable range, i.e., 65%, the wet bulk density of the initial compost mixture is in agreement with the reported value of the previous study. Though the presence of higher moisture content in the compost matrix could hinder the air passage and thereby result in compaction, the addition of wood chips in the initial feed stocks created a pathway for air penetration in the present study (Ermolaev et al. 2015). Mohee and Mudhoo (2005) reported that the aerobic decomposition reduces particle size and this results in both settlement and volume reduction. Hence, the porosity in the compost matrix reduced significantly in both the systems due to the moisture loss and compaction. In active system, though there were a decrease in porosity and increase in wet bulk density, a FAS of above 30% and corresponding oxygen supply of above 5% were maintained in the system. This is attributed to the forced aeration that was provided intermittently to the system. However, in passive system, a FAS of above 20% was observed. Even though the FAS in passive system was found to be lower than that of active system, the level was enough to sustain the aerobic

ambience in the compost matrix. The main reason for the maintenance of minimum FAS in these systems could be due to the presence of mixture of different wastes as feedstocks that resulted in varying particle size, which in turn contributed to higher porosity. Haug (1993) reported that a target FAS of $\geq 20\%$ was essential for adequate natural draft ventilation in windrows during biosolids composting. As explained earlier, both the passive and active in-vessel composting systems in the present study were able to maintain a FAS > 20%. On analyzing the results by t test, the variations in FAS ($t = -0.43$, $P = 0.6920$), porosity ($t = -0.47$, $P = 0.6641$), wet bulk density ($t = -0.00$, $P = 1.0000$) and the particle density ($t = -0.46$, $P = 0.6657$) were found to be insignificant between the two aeration systems during the composting period. The correlation of FAS, wet bulk density, porosity and particle density with GHG emissions was found to be insignificant ($P > 0.05$) in the present study. (This can be even due to the limited number of data available for correlation analysis in the present study.)

Total greenhouse gas emission budget and carbon sequestration potential

The total greenhouse gas (GHG) emission budget during septage co-composting is illustrated in Table S3. The contributions of CH_4 emissions to the total GHG emissions were calculated as 2.5 and 0.9% in passive and active systems, respectively. The contributions of N_2O emissions to the total GHG emission budget were 7.5 and 3.8% in passive and active systems, respectively. The N_2O emissions were found to be higher than CH_4 emissions, when considering the total GHG emissions. The N_2O emission is attributed to the presence of N-rich organic materials in the feedstock composition. Similar observation was reported by He et al. (2001) during aerobic composting of organic wastes. The total GHG emission budget revealed the significant carbon sequestration potential of final compost that has key role in the emission avoidance. Carbon sequestration is the storage of carbon for longer period and is considered as sink (Composting Council 2008). The total GHG emissions during in-vessel co-composting in terms of CO_2–C equivalent were found to be 144.2 g CO_2-e/kg in passive system and 140.8 g CO_2-e/kg in active system (Table S6). The initial total carbon content of the compost mixture was 332.3 g C/kg. Since the total emission was found to be lower than the initial carbon content, there arises a net benefit of septage co-composting in carbon sequestration (Hao et al. 2004). From the experimental results, it was observed that the forced aeration could able to maintain the aerobic environment through composting which resulted in a

Table 4 Outcomes of seed germination test

Item/parameter	Control test	Compost extract from passive aeration system	Compost extract from active aeration system
Total seeds	40	40	40
Germinated seeds	29	32	30
Mean root length (cm)	0.88	1.09	1.22
Relative seed germination (%)	–	110	103
Relative root growth (%)	–	124	139
Germination index (%)	–	136	143

reduction of 50% of CH_4 and N_2O emission in the total GHG budget with the generation of ripened compost with high carbon sequestration potential at a faster rate.

Compost hygienization

Septage contains numerous species of pathogenic organisms (Yen-Phi et al. 2010). *Escherichia coli* (*E. coli*) is the indicator organism used to identify fecal contamination in the environment. In the present study, the initial *E. coli* count was 1.3×10^4 CFU/g dry weight (d.w) of compost and 1.2×10^4 CFU/g d.w of compost in passive and active aeration system, respectively. In composting systems, the time–temperature relationship required for achieving pathogen-free compost provided by USEPA (2003) is given in Table S7. In this study, the EPA regulatory requirement for pathogen reduction was satisfied with respect to time–temperature criteria. As a result, after thermophilic phase (where the final samples were taken and analyzed before re-growth), *E. coli* count was reduced to 26 CFU/g d.w of compost and 24 CFU/g d.w of compost in passive and active systems, respectively. The present study could ensure not only the time–temperature criteria (retention of temperature > 55 °C for 3 days), but also produced hygienically improved quality compost at a faster rate.

Compost quality and maturity

The maturity level of the final compost was assessed using indices like C/N ratio, NO_3^-/NH_4^+ and phytotoxicity test or seed germination test. The C/N ratio < 10 and temperature that decreased to ambient temperature revealed the maturity and stability of the compost. NO_3^-/NH_4^+ ratio is considered as another maturity indicator for the composting process. Bernai et al. (1998) established $NO_3^-/NH_4^+ > 6.25$ as a stringent index to describe compost maturity. The ratio obtained at the end of composting 9.7 and 5.7 for passive and active systems, respectively, indicated that the final compost from both systems can be considered as matured enough for agricultural purpose. The outcomes of the germination test are given in Table 4. The prescribed germination index (GI) value for

phytotoxicity-free compost for safe application of matured compost is GI ≥ 80% (Wong et al. 2011). The GI was 136% in passive system and 143% in active system. Hence, in this study both systems produced a highly stable and phytotoxicity-free compost within 20 days of in-vessel co-composting. In addition to the stability parameters, the final compost from both systems was found to be rich in nutrient content (Table S7). The final compost has a TP and K content of 3.6 g/kg and 15.6 g/kg in passive system and 3.7 g/kg and 18.9 g/kg in active system, respectively (Table 3). According to the criteria laid by Hong Kong Organic Resource Centre, the composts obtained from both systems with a total nutrient content (N, P and K together) of above 4% d.w could be classified as organic fertilizers (Wong et al. 2011). The properties of the final composts obtained from active and passive aeration in-vessel system at the end of septage co-composting are given in Table S8.

Comparison of in-vessel co-composting with other treatment technologies

In the present study, in-vessel co-composting is evaluated as a decentralized treatment option for septage management in developing countries like India. Co-composting seems to be a less expensive and sustainable treatment option compared to other available treatment options currently practiced in India as it treats not only dewatered septage but also mixed organic waste. Hence, the overall cost will be lower for co-composting compared to the available treatment options like co-treatment with sewage or municipal solid waste and anaerobic co-digestion (Lin et al. 2000; Prabhu et al. 2015). Furthermore, resource recovery from septage and organic waste is possible and effective in co-composting than other treatment methods. It can be even carried out using home composter. The cost of an ordinary aerobic home composter used in the present study is about Rs 2500 (US$ 35.5)/unit with a capacity of 50 liters. In order to reduce the cost and to understand the feasibility of further application, locally available and less expensive materials were used to fabricate the setup. The price details of each component associated with an

in-vessel composter is given in Table S9. The total energy consumption for entire composting operation of 30 kg of raw waste is 9.7 kWh, and the cost associated with the operation is Rs 1.94 (~ 3 US cents) per kg waste treated. The compost thus obtained is found to have better quality as plant fertilizer compared to that of the commercially available organic fertilizer. The cost of processed compost/kg is around Rs 25/- (0.36 US $), while the cost of commercially available organic fertilizer is Rs 300/- (4.26 US $). The cost comparison of in-vessel co-composting and co-treatment with sewage for the septage treatment is given in Table S10. In addition, the comparison of the present study with previous studies on septage treatment and its potential application is given in Table S11. A detailed comparison was also made to understand the GHG emissions that occurred during manure composting and with other treatment technologies. The GHG emission comparison is given in Tables S12 and S13. The overall findings revealed that in-vessel co-composting is comparatively a low-cost and unique method than available other septage treatment options for conversion of septage and organic waste to fertilizer in a short span of time with least threatening emission.

Conclusion and recommendations

The present study revealed that septage and mixed organic waste can be effectively composted using in-vessel system. The influence of active and passive aeration on greenhouse gas emissions and compost dynamics during in-vessel co-composting of septage and mixed organic waste was investigated. Results revealed that the aeration pattern has significant influence on greenhouse gas emissions while the compost quality obtained from both systems was similar. The active aeration pattern was found to be better than that of passive mode in terms of emission. The ripened compost could also be used as a sink for carbon sequestration as the initial carbon content in both systems was higher than the GHG emissions in terms of CO_2 equivalent. The final compost quality obtained after 20 days of operation confirmed the maturity using indices like seed germination index and NO_3^-/NH_4^+ index. The compost generated also met the international compost quality standards.

Previous studies on septage treatment mainly focused on resource recovery as compost rather than the monitoring and mitigation of potential greenhouse gas emissions from composting systems. Our findings highlight that in-vessel co-composting is a suitable treatment option, because this method achieved hygienically improved, phytotoxic-free and nutrient-rich compost from septage and mixed organic waste in a shorter span with least

threatening emissions. Implementation of this treatment option in field scale can resolve the looming septage and organic waste management problem in developing countries in current scenario. Since the septage is highly variable in nature, a comprehensive investigation is required for the application of in-vessel co-composting in terms of feedstock composition optimization. In addition, the lifecycle assessment and the scalability of the system for field application need to be addressed.

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