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Evaluation of gas removal and bacterial community diversity in a biofilter developed to treat composting exhaust gases

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Abstract

The performance of a new, but simply constructed, biofilter system, developed to purify composting exhaust air, was evaluated. The biofilter was packed with mature compost mixed with activated carbon and sludge sourced from a wastewater treatment plant. An alternating air flow system and a bioaerosol reduction device were designed to prevent pressure drop and reduce bioaerosol release. Experimental results demonstrated that satisfactory removal efficiencies of nitrogen-containing compounds, sulfur-containing compounds, fatty acids, total hydrocarbon and odor were achieved at an empty bed retention time (EBRT) of 30 s. No significant acidification or alkalinity in the biofilter was observed, and the system was characterized by a small pressure drop and a low level of bioaerosol emission. Denaturing gradient gel electrophoresis (DGGE) and fluorescence *in situ* hybridization (FISH) techniques were used to uncover the changes in the bacterial community of the biofilter during the deodorization processes. A minimum of 16 bands were observed in the DGGE profile. Phylogenetic analysis revealed the phylum of *Proteobacteria* to be predominant, followed by *Actinobacteria*, *Bacteroidetes*, and *Firmicutes*, in descending order. However, the occurrence and predominance of specific bacterial species varied with the environmental conditions of the biofilter. Our results demonstrate – from both an engineering and biological point of view – the feasibility of the biofilter system described herein in purifying the gases derived from composting food waste.
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Keywords: Composting plant; Odor; Bacterial community; Biofilter

1. Introduction

Food wastes are estimated to account for 25% of the total garbage in Taiwan, and the latest figures for 2004 indicate that the volume of recycled food wastes reached 1400 t/day. This latter value is expected to increase with continuing population growth and the implementation of the food waste recycling policy enforced by the EPA in Taiwan. In 2002, large-scale food waste disposal consisted of landfill (54%), incineration (38%), recycling (6%), and composting (2%). However, aspects of the first two methods – landfill disposal and incineration – are environmentally unfriendly in that landfill food wastes containing grease can result in landfill loading, the production of unpleasant odors, and leachate pollution of various bodies of water [1], while the incineration of food wastes with a high moisture content increases the consumption of auxiliary fuel, shortens

incinerator life, and gives rise to secondary or lethal pollutants, such as dioxin [2].

In recent years, composting has attracted much attention and has come to be regarded as an environmentally friendly and sustainable alternative for the management and recycling of organic wastes [3]. Composting converts various components in organic wastes into relatively stable substances that can be used as a soil amendment or organic fertilizer. However, composting can also impact negatively on the environment, namely through the generation of odorous gaseous emissions [4]. That this concern is taken seriously in developed countries is illustrated by the shutting down of several composting plants in Taiwan in recent years. Consequently, the problems of odor emission in composting plants require immediate attention if composting is to become a viable option for the industrial scale recycling of food waste.

More than 100 kinds of odorous gases are emitted from composting processes, of which the nitrogen-containing compounds, sulfur-containing compounds, and short-chain fatty acids have attracted the most attention due to their low threshold limits [5].

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Factors such as the composition of the raw material, temperature, oxygen concentration, and pH of the composting process all influence the concentrations of gases produced [6].

Traditional gas or odor treatment technologies include carbon adsorption, wet scrubbing, and incineration, but these involve high treatment costs and secondary waste stream problems [7]. One alternative, biofiltration is increasingly being regarded as the best available control technology (BACT) for treating odorous compounds based on its cost effectiveness, lower generation of secondary contaminated waste streams [8], and reduced ammonia emissions [9]. As such, biofiltration has the potential to simultaneously remove various odorous gases emitted from composting plants. While earlier designs of biofilters were intricate with respect to design and material [10], it has become clear that industrial-scale biofilters should be simply constructed with an equally simple biofiltration operation, and an easy access to the inoculated source; secondary pollution should also be absent. Many solid biologically active materials – such as peat, compost, soil, leaves, or wood bark – have been used as packing media, with the addition of some inert support materials to reduce any pressure drop [11]. However, in comparison with other packing media, mature compost is inexpensive, contains complex microbial communities, and is readily available. Additional nutrients are not usually required for biofilters based on compost because these already have significant amounts of organic nitrogen and other micronutrients [12].

As biofilters and composting facilities are biological systems inhabited by varied microbes, much attention has focused on the bioaerosol emitted from such facilities [13,14]. To date, however, data are scarce on the environmental and health risks of the emitted bioaerosol and on the extent at which these are reduced by biofilters [13]. To understand the diversity of the bacterial community during treatment processes, molecular ecological methods for analyzing the structure of total bacterial population have proven to be powerful tools [15]. However, few reports have demonstrated the diversity of the bacterial community in a biofilter treating the odor emitted from a composting plant by denaturing gradient gel electrophoresis (DGGE) or fluorescence *in situ* hybridization (FISH).

The aim of this investigation was two-fold: (1) to assess the operating characteristics of a readily available biofilter system, such as removal efficiency, pressure drop, and reduction in bioaerosol emission, that utilizes an advanced plant composting treatment technique; (2) to gain an insight into the dynamic diversity of the bacterial community in the biofilter during the deodorization processes by means of DGGE and FISH. The results of the analyses suggest that the biofilter system described herein is a feasible option for purifying the odors emitted from a composting plant.

2. Materials and methods

2.1. Apparatus and gas removal

Three identical units of gas production apparatus [40 cm (ID) × 60 cm (H)] were used to evaluate the performance of the biofilter. Well-mixed compost materials from the Hsinchu
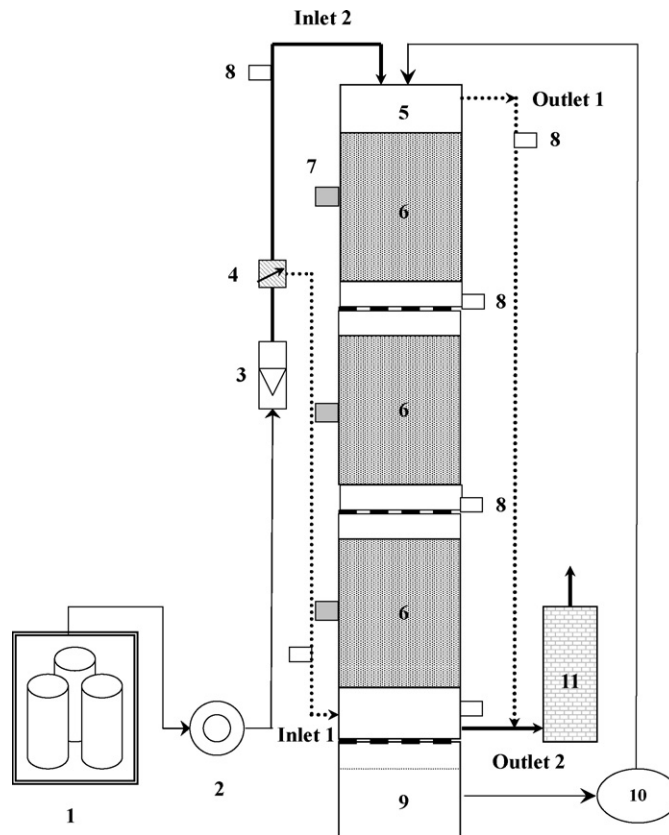

Fig. 1. Set-up and design of the laboratory-scale compost-based biofilter: (1) gas production apparatus; (2) exhaust fan; (3) flow meter; (4) switch valve; (5) sprinkling area; (6) bioreactor; (7) packing material sampling port; (8) gas sampling port; (9) nutrient tank; (10) peristaltic pump; (11) bioaerosol reduction device.

or Miaoli composting plants were introduced into the gas production apparatuses. The composting tank was 54 cm high, with a volume of 204 l (68 l × 3). The compost was turned by a labor force every 2 days, and the removal efficiencies of odorous gases by the biofilter were evaluated 30 min after turning.

The set-up and design of the laboratory-scale compost-based biofilter are illustrated in Fig. 1. Briefly, the three acrylic columns [12 cm (φ) × 30 cm (working height)] are connected in series to form the major module of the biofilter. A perforated sieve plate is fitted at the bottom of the column to allow the circulating liquid to flow out. The packed volume of the compost-based biofilter is 10.2 l. There are three sampling ports in one wall of the column, 40 cm apart, for measuring gas concentrations during the experiments and three additional sampling ports in the other column wall for collecting packing material. Odorous gases are continuously extracted from the gas production apparatus, subsequently flowing upward or downward through the biofilter at the top or bottom. The gas flow rate was maintained at 20.34 l/min, corresponding with an empty bed retention time (EBRT) of 30 s, as suggested by previous studies [11]. However, the direction of gas flow was altered weekly. An inflow medium (see medium preparation) stored in the nutrient tank was intermittently re-circulated by a peristaltic pump at 10 l/min for 4 min every 6 h. Prior to gas discharge, a bioaerosol reduction device [4 cm (ID) × 12 cm (L)] filled with pillared-type hollow ceramics was installed to

Table 1
The important operating parameters of the compost-based biofilter

Packed volume (l)	Gas flow rate (l/min)	Location	Nitrogen-containing compounds		Sulfur-containing compounds		Fatty acids	
			Concentration (ppm)	Inlet load (g-N/(m³ day))	Concentration (ppm)	Inlet load (g-S/(m³ day))	Concentration (ppm)	Inlet load (g-C/(m³ day))
10.2	20.34	Hsinchu plant	0.2–105	0.328–172.3	0.05–0.241	0.188–0.903	2.1–3.6	2.95–5.06
		Miaoli plant	0.2–68	0.328–111.5	0.13–4.62	0.488–17.33	0.2–1.0	0.28–1.41

remove the bioaerosol emissions. The ceramics were immersed in a 5% ClO_2 solution for 1 h and then dried at 60 °C for 2 h before being packed into the device. The operating characteristics of the biofilter, such as removal efficiency, pressure drop, and reduction in bioaerosol emission, were determined and the diversity of the bacterial community in the biofilter was analyzed. The important operating parameters of the compost-based biofilter are listed in Table 1.

2.2. Composting source and composition

The composting materials containing the food waste used in the experiments were collected from the Hsinchu and Miaoli composting plants in Taiwan. The food wastes sourced from the Hsinchu plant comprised mainly raw food wastes (e.g., vegetable residues, tea residues, and fruit skin), and those from the Miaoli plant consisted mainly of mixtures of raw and ripe food wastes (e.g., meat scraps, rice, and vegetable residues). The initial C/N ratios for the food wastes were in the range of 11.9–13.6; consequently, some agricultural wastes were mixed with the food wastes to achieve the optimal (C/N) composting conditions. The compost materials from the Hsinchu plant initially consisted of 50% food wastes, 18% chicken feces, 12% herbal medicine residues, 8% industrial sludge, 7% tea residues, and 5% chicken feathers. The C/N ratio, pH, and moisture content of the mixture were 32 ± 2, 8.2 ± 0.2, and $60 \pm 3\%$, respectively. The compost materials from the Miaoli plant initially consisted of 40% food wastes, 25% mushroom residues, 12% cattle feces, and 10% agricultural wastes, containing yeast, 8% pig feces, and 5% chicken feces. The C/N ratio, pH, and moisture content of this mixture were 35 ± 3, 8.4 ± 0.4, and $62 \pm 5\%$, respectively. The composting materials used in this study were considered to be representative of those standardly used in composting plants based on their physical and chemical compositions.

2.3. Packing material, organism, and medium preparation

In this study, a mixture of substances was used as the biofilter packing materials to treat gases emitted from the composting plants. The main packing material was mature compost in situ (Hsinchu or Miaoli plant) to which 10% (w/w) granular activated carbon (5-mm diameter) was added. Additionally, the biofilter was inoculated with 5% (w/w) sludge from the aeration tank of the wastewater treatment in the field. The initial bacteria numbers in the three biofilter layers (top, middle, and bottom) were about $(3.2–8.6) \times 10^9$ CFU/g-compost. Inflow medium

(=cycling solutions; 4.08 g/l KH_2PO_4 and 5.22 g/l K_2HPO_4) was periodically supplied during the deodorization experiments and stored in the nutrient tank. The final pH of the medium was adjusted to neutral with 2N NaOH or HCl.

2.4. Bioaerosol analysis

Microorganisms emitted from the compost-based biofilter were collected by liquid impingement. The air escaping from the bottom of the filter or from the exit of the bioaerosol reduction device was forced through a 250-ml flask containing 100 ml saline (0.9% w/v NaCl) for 12 h at 4 °C. One-milliliter aliquots of the collected solution were inoculated into each of the different media, and cell numbers were determined by the serial dilution method. Potato dextrose agar (PDA) was used for culturing fungi and Luria–Bertani agar (LB) for heterotrophic bacteria. The counts were reported as colony forming units in air (CFU/m³).

2.5. Analytical methods

Gas concentrations of nitrogen-containing compounds were analyzed in a Clarus 500 gas chromatograph (Perkin-Elmer, Foster City, CA) equipped with a Porapak Q column (2.6 mm × 2 m) and a photoionization detector (PID). Gas concentrations of sulfur-containing compounds were analyzed in a Fisons-8000 gas chromatograph (Fisons, UK) equipped with a GS-Q column (0.53 mm × 30 m) and a flame photometric detector (FPD). Gas concentrations of short chain fatty acid (C_2–C_5) were analyzed in a Clarus model 500 gas chromatograph (Perkin-Elmer) equipped with a Stabilwax-DA column (0.53 mm × 30 m) and a flame ionization detector (FID). Gas concentrations of total hydrocarbon (THC) were analyzed in a Clarus model 500 gas chromatograph (Perkin-Elmer) equipped with a fused silica capillary tube (0.53 mm × 5 m) and a FID. The intensity of total odorous gases was always measured within 24 h using olfactometry.

The moisture content of the compost-based biofilter was determined by first removing a 1-g sample (approx.) of compost medium from the middle zone of the biofilter, then weighing and drying it for 24 h at 103 ± 0.5 °C. To measure the pH in the compost-based biofilter, 0.5 g of compost medium from a similar location was also removed and mixed with 5 ml distilled water. The sample was vortexed for 3 min, and the pH value was then determined using a pH meter. U-tube water manometers were used to determine the pressure drop across the biofilter, and the value obtained was expressed as mm-H_2O/m-filter height. For

cell number estimation, 0.5 g compost medium was taken from the middle zone of the biofilter and mixed with 5 ml saline (0.9% w/v NaCl). The samples were then vortexed for 3 min, and the cell numbers in the filter were enumerated by traditional plate-counting methods. The LB and PDA were used for culturing heterotrophic bacteria and fungi, respectively.

2.6. Analysis of bacterial community

The DGGE and FISH techniques were used to monitor changes in the bacterial community of the biofilter. The structure and intensity of the bacterial community were analyzed by a DGGE apparatus (Bio-Rad, Hercules, CA) and Bio-Rad's image program (Quantity One 4.5.0). The DGGE gels consisted of 8% acrylamide gel with a 45–60% denaturant gradient. Cell lysis, DNA extraction, and PCR amplification were as described by Sandaa et al. [16]. PCR primers for F968GC and R1401 were used to amplify the segment of eubacterial 16S rDNA [17]. The different bands (=strains of bacteria) were identified by excising the bands from the DGGE gel, then eluting, re-amplifying, and sequencing them [17]. Sequences were submitted for comparison to the GenBank databases using the BLAST algorithm.

FISH analysis was performed as described by Manz et al. [18] using the following probes: EUB338 (5′-GCTGCCT-CCCGTAGGAGT-3′) for domain bacteria, ALF1b (5′-CG-TTCGYTCTGAGCCAG-3′) for α subclass of *Proteobacteria*, BET42a (5′-GCCTTCCCACTTCGTTT-3′) for β subclass of *Proteobacteria*, and GAM42a (5′-GCCTTCCCACATCGTTT-3′) for γ subclass of *Proteobacteria* [18]. In the protocol used here, each probe was directed against a small ribosomal subunit and labeled with the indocarbocyanine dye Cy3. Total cell counts were determined by diamidino phenylindole (DAPI) staining. After fixation, hybridization, washing, and staining, bacterial cells were observed under an E-400 epifluorescent microscope (Nikon) equipped with optimal filter sets. Digital images were collected after FISH and DAPI staining using the CoolSNAP CF digital camera (Photometrics Roper Scientific). Image-Pro Discovery software was used to quantitatively analyze images obtained by FISH and DAPI staining.

3. Results and discussion

3.1. Gas and odor removal by the compost-based biofilter

Field studies have indicated that the gases released during the composting process consist primarily of nitrogen-containing and sulfur-containing compounds, short chain fatty acids, and some volatile organic compounds. As such, the efficiencies at which the biofilter being tested removed such gases were the primary considerations in this study. Given that industrial composting systems are primarily intended to run in a continuous and long-term mode, this experiment was conducted for 150 days, which were divided into three equal time periods (rounds). As the removal efficiency profiles of each of these three rounds did not vary significantly, data from the first round was selected to be representative of all three rounds and used for subsequent discussion. The removal efficiency profiles of the nitrogen-
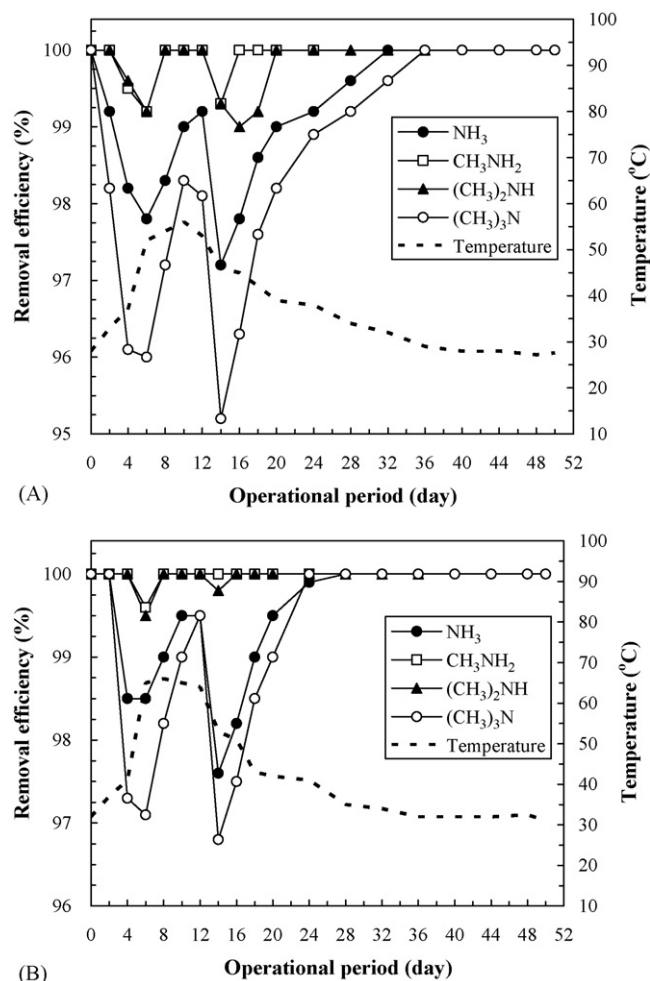

Fig. 2. Removal efficiency profiles of nitrogen-containing compounds and the change in temperature of the biofilter during the deodorization processes: (A) compost materials sourced from the Hsinchu composting plant; (B) compost materials sourced from the Miaoli composting plant.

containing compounds and the change in temperature of the biofilter are shown in Fig. 2. During the feeding of composting materials collected from the Hsinchu plant, the inlet concentrations of nitrogen-containing compounds to the biofilter were in the range of 0.2–105 ppm, with trimethylamine [(CH₃)₃N] at 53.2 ppm (average) and methylamine (CH₃NH₂) at 5.4 ppm (average) being the most and least prevalent of the N-containing gases, respectively. The composting materials collected from the Miaoli plant contained relatively low concentrations (0.2–68 ppm) of nitrogen-containing compounds. The high concentration of nitrogen compounds in the composting materials originating from the Hsinchu plant may be attributed to the relatively high proportion of N-rich chicken feces. At an EBRT of 30 s, the compost-based biofilter seemed to perform better in removing CH_3NH_2 and $(CH_3)_2NH$ (>99%) as well as ammonia. However, $(CH_3)_3N$ gas was difficult to degrade because of its complex structure and high concentration. Notwithstanding, more than 95.2% (Fig. 2A) and 96.8% removal efficiencies of $(CH_3)_3N$ (Fig. 2B) were achieved by this biofilter system for the composting material from Hsinchu and Miaoli plant, respectively. In addition, no acclimation period was required for
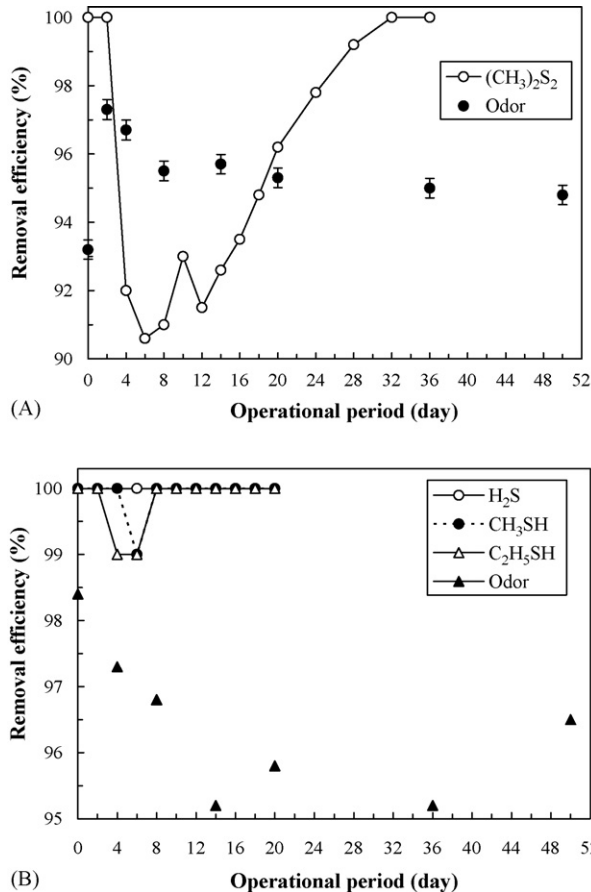
Fig. 3. Removal efficiency profiles of sulfur-containing compounds and odor in the biofilter during the deodorization processes: (A) compost materials sourced from the Hsinchu composting plant; (B) compost materials sourced from the Miaoli composting plant.
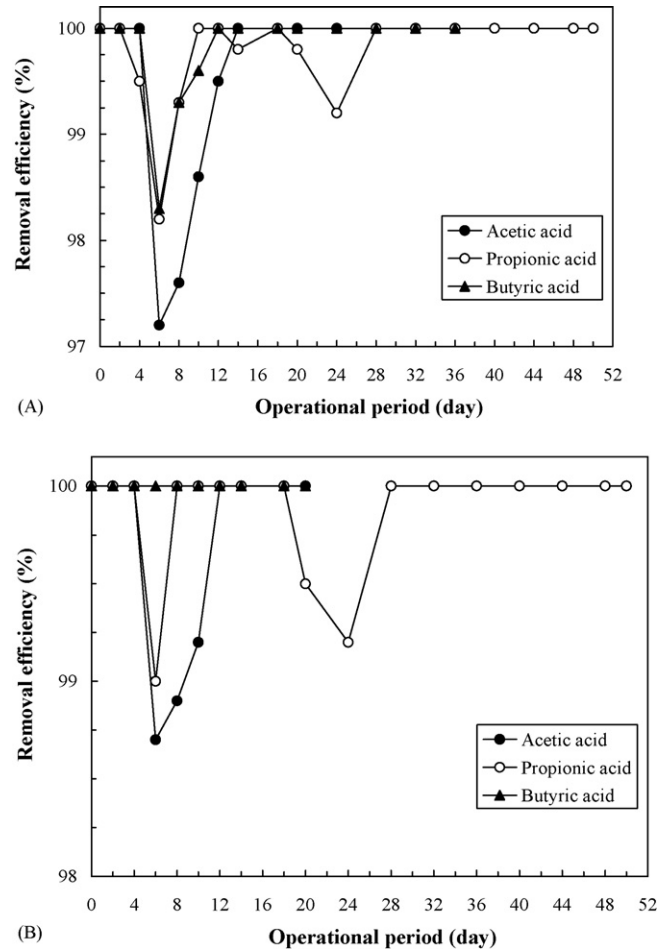

Fig. 4. Removal efficiency profiles of fatty acids in the biofilter during the deodorization processes: (A) compost materials sourced from the Hsinchu composting plant; (B) compost materials sourced from the Miaoli composting plant.

the biofilter. The temperature of the biofilters fluctuated with changes in the inlet gases, with the peak temperature occurring on the 10th day. Although temperature shock could affect biofilter performance, the system can apparently acclimate to the problem.

As raw food wastes with low sulfur-containing compounds (e.g., vegetable residues and fruit skins) were the main composting materials in the Hsinchu plant, dimethyldisulfide (DMDS) was the only gas detected. The peak concentration of inlet DMDS and the intensity of odor were measured at 0.241 ppm and 1200 OU, and the removal efficiency of the biofilter system was 90.6% and 95.3%, respectively (Fig. 3A). Sulfur-containing compounds, such as hydrogen sulfide (H_2S), methanethiol (CH_3SH), and ethanethiol (C_2H_5SH), were detected at the Miaoli composting plant, which contained more complicated compost mixtures of raw and ripe food wastes. However, the concentrations of sulfur-containing compounds were relatively low (<5 ppm) compared to the levels of nitrogen-containing compounds. The peak odor intensity was 2800 OU on the eighth day, corresponding to a measured removal efficiency of 96.8% (Fig. 3B). The concentration of sulfur-containing compounds was below the detection limit of the analysis after 21 days; however, odorous gases were still detectable within the range of

420–50 OU at this time. One possible source of the odor was nitrogen-containing or specific compound emissions. At the end of the experimental period, odor intensity in the outlet of biofilter was 5 OU with the compost materials sourced from the Miaoli plant.

In addition to the sulfur- and nitrogen-containing compounds, low concentrations of fatty acids were detected in the compost materials from the Hsinchu and Miaoli composting plants. As the compost materials at the Hsinchu plant were primarily vegetable residues and fruit skins, they contained higher concentrations of fatty acids (2.1–3.6 ppm) than those sourced from the Miaoli plant (0.2–1.0 ppm). Fig. 4 indicates that the system successfully removed 97% of the fatty acids from the composting tanks. However, the removal characteristics of other volatile organic compounds possible emitted during the composting processes had to be monitored constantly. Variations in the concentrations of total hydrocarbons (THCs) were insignificant at both the Hsinchu and Miaoli plants. During the deodorization process, the measured concentrations of THCs were in the range of 3.5–28 ppm. The biofilter tested here achieved a 96% removal of THC at the maximal outlet concentration of 0.8 ppm (data not shown). Based on the gas emission inventory (data not shown) and a comparison of the removal efficiency of fatty acids and

THCs, some of the more difficult to degrade compounds, such as BTEX, may have been present in the THCs.

3.2. Change in pH, moisture, and pressure drop of the compost-based biofilter

Following the treatment of the gases from the gas production apparatus packed with compost materials from the Miaoli and Hsinchu plants, the changes in pH values of the biofilters were in the range of 7.2–8.3 and 7.6–8.9, respectively (data not shown). pH variations depend mainly on the concentrations and compositions of gases emitted as well as on the buffering capacity of the cycling solution. Because high concentrations of alkali nitrogen-containing compounds were introduced into the compost-based biofilter on the 6th and 14th days, the dual peak of pH in the biofilter occurred simultaneously, in accordance with the high concentrations of gases (data not shown). Our results indicate that the pH in the biofilter remained at a near-neutral pH range during the deodorization period, thereby providing further support for the use of compost-based biofilter in composting organic matter.

The profiles of the moisture content in the compost-based biofilter revealed a slight variation following the change in the biofilter temperature. The lowest moisture content occurred at the highest biofilter temperature—on the 10th day. However, the biofilter retained an average moisture content of 43% (40–46%) during the deodorization process (data not shown). Since moisture content must be maintained between 40% and 60% in the biofilter to maintain biological activity, the measured values demonstrate that the system is highly efficiency with respect to the removal of various odorous gases.

Compost is a packing material that is relatively easily degraded compared with peat and activated carbon [19]. Hence, the anticipated problem of a large pressure drop in the compost-based biofilter is a problem that must be overcome in any biofilter composting system. In this study, a switch valve was used to alternate the air flow into the biofilter on a weekly basis, thereby reducing the drop in pressure. Fig. 5 indicates the profiles of pressure drop in the biofilter using the alternating air flow mode or the downward air flow mode. The results reveal that the downward air flow system caused a large pressure drop in the biofilter, which increased with operational time, eventually reaching 80.3 mm-H_2O/m by the end of the study period. Conversely, the alternating air flow system showed a periodic variation in pressure between 20 and 30 mm-H_2O/m. Consequently, the observed pressure drop in the operating biofilter represents a satisfactory performance, with the system displaying excellent dispersion [20]. During the 91-day operation, the results on pressure drop using the alternating air flow mode were better than that those obtained in other studies using peat (74 mm-H_2O/m), rock wool (68 mm-H_2O/m), fuyolite (62 mm-H_2O/m), and ceramics (54 mm-H_2O/m) as the packing media under similar operational conditions [21,22].

3.3. Bioaerosol emission analysis and removal

Attempts to deodorize using a biofilter system have proven to be efficient and promising [21]. However, there is a potential environmental risk associated with a biofilter system containing a large concentration of microorganisms as these (or a portion thereof) may eventually be released from the system into the atmosphere, even if the packing materials can somewhat withhold bioaerosol emission [14]. In our study, the variation in bioaerosol amounts in the outlet was proportional to the microbial numbers in the biofilter regardless of the treated gases being emitted from the composting process (data not shown). In the absence of the bioaerosol reduction device in the biofilter system, the highest bioaerosol amounts reached approximately 10^6 CFU/m^3 in the outlet (Fig. 6). However, less than two orders of magnitude bioaerosol amounts were detected when the bioaerosol reduction device was in place and functioning, with the highest measured amount being 4.2×10^3 CFU/m^3. This indicates that the device had a 99.6% bioaerosol removal effi-
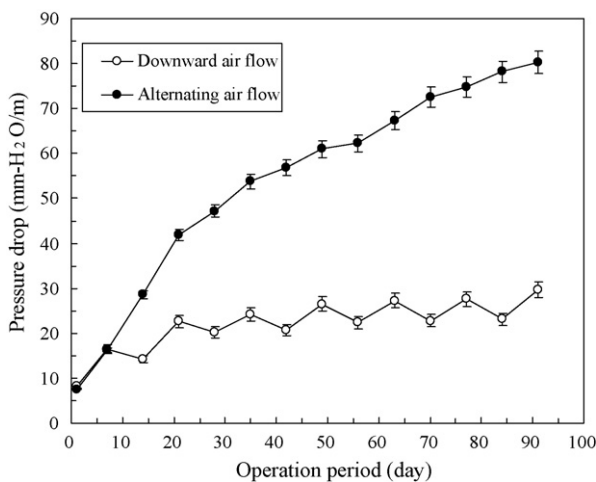
Fig. 5. Profiles of pressure drop in the biofilter when odor is introduced to the biofilter by using an alternating air flow mode or downward air flow mode. The compost materials were sourced from the Hsinchu composting plant.
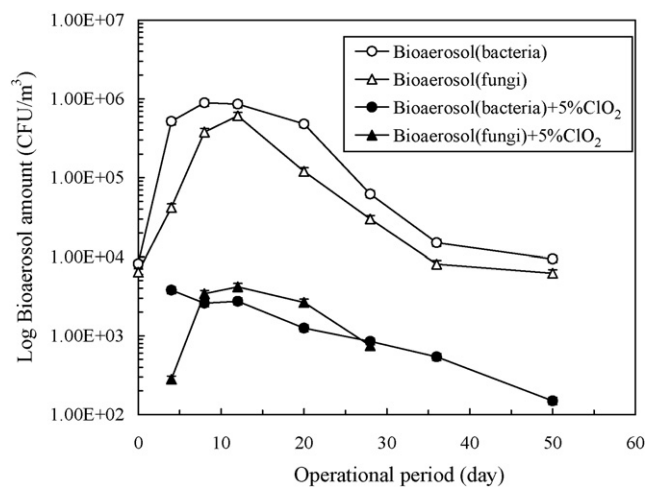
Fig. 6. Profiles of bioaerosol amounts in the outlet of the compost-based biofilter with/without the bioaerosol reduction device. The bioaerosol reduction device was filled with pillared and hollow ceramics, and these packing materials were pretreated with a 5% ClO$_2$ solution.
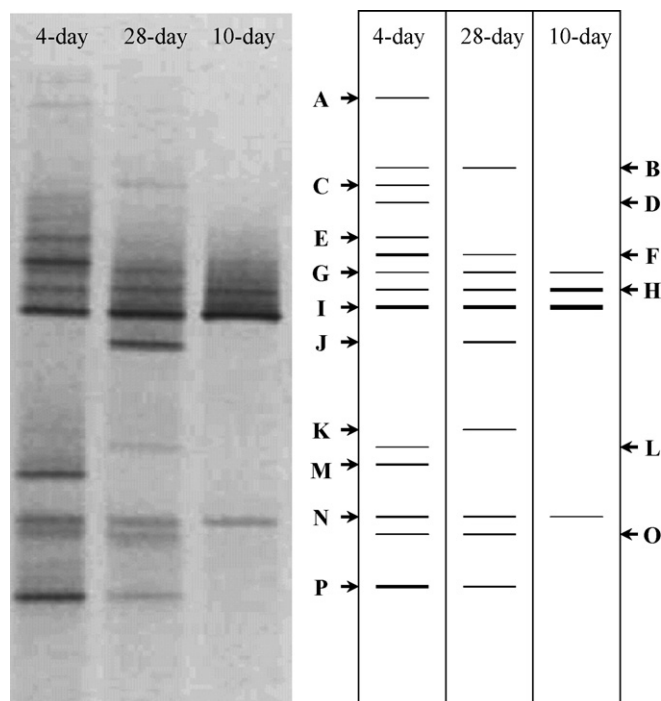
Fig. 7. DGGE profiles of bacterial community in the compost-based biofilter. Lane 1: 4 days; lane 2: 28 days; lane 3: 10 days.

ciency [$(10^6 - 4.2 \times 10^3)/10^6$]. Malmros (1990) suggested that an acceptable air quality in terms of bacterial bioaerosol amounts is 10^4 CFU/m^3 [23]. Previous studies have reported bioaerosol amounts of bacteria of 10^6 CFU/m^3 (peat industry), 10^5 CFU/m^3 (refuse collections), 10^5 CFU/m^3 (composting facilities) and 10^3 CFU/m^3 (homes) [24–27]. It can therefore be concluded that a bioaerosol reduction device can effectively obviate the environmental risk of bioaerosol emission and that the biofilter system may be installed in the vicinity of populated areas without any unacceptable safety concerns.

3.4. Bacterial community in the biofilter

Fig. 7 shows the DGGE profiles of the bacterial community in the compost-based biofilter during the deodorization process. The different deodorization stages of the biofilter were analyzed in separate time periods. The 4th, 10th, and 28th days represent the three purification stages of odorous gases emitted from the primary-fermentation, fermentation, and ripening stages of the composting processes, respectively. The gases emitted from the different fermentation stages caused dramatic changes in the temperature of the biofilter, with measurements of 37, 56, and 34 °C on the 4th, 10th, and 28th days, respectively. The relative diversity of the bacterial community is related to the number of bands in the gel, while the degree of abundance constituting each bacterial group is correlated to the intensity of the specific band. The appearance or disappearance of bands indicates changes in the bacterial community of the biofilter as the deodorization process proceeds.

A significant change could be observed during the deodorization period of the biofilter (Fig. 7), with the complexity of the

DGGE bands generally decreasing with increasing biofilter temperature. For example, on day 10, with a relatively high recorded temperature (56 °C), four DGGE bands were found; on day 4, with a relatively low recorded temperature (37 °C), there were 14 discriminable bands. The bacteria in the biofilter may have originated from the inoculated sludge, compost medium, and/or the composting tank. Major bands (those of highest intensity) G, H, I, and N predominated from days 4 to 28. However, the intensity of band N decreased on day 10 under a high temperature environmental condition. Bands J and K were only observed on day 28, and bands A, C, D, E, L, and M were only observed on day 4. In addition, bands B, F, O, and P appeared at days 4 and 28. Based on these observations, it is apparent that the primary-fermentation stage of the composting process on the fourth day produced the largest number of different bacterial populations in the deodorization biofilter.

To better understand the differences in bacterial diversity among samples, discriminable bands were excised and sequenced. The strain name, phylogeny classification, nucleotide sequence similarity, and relative abundance (intensity) of sequenced DGGE bands during different operating times are listed in Table 2. Sixteen discriminable bands (A–P) were individually identified as members of different eubacterial phyla, as shown in Table 2, and the closest relative was identified by comparison with the GenBank database. Four bands (H, I, M, N) were grouped with the phylum α-Proteobacteria, and their closest relative showed homology to *Aminobacter aminovorans*, *Paracoccus denitrificans*, *Caulobacter bacteroides*, and *C. fustformis*. Only the P band, which was affiliated to *Comamonas testosteroni*, belonged to β-Proteobacteria. Three bands (B–D) were clustered within the phylum γ-Proteobacteria, namely *Pseudomonas citronellolis*, *P. fluorescens*, and *P. putida*, respectively. Two bands (A and G) were clustered within the phylum *Firmicutes*, namely *Staphylococcus capitis* and *Bacillus subtilis*. Two bands (E and K) were clustered within the phylum *Bacteroidetes*, namely *Chryseobacterium scophthalmum*, and *Flavobacterium mizutaii*. Four bands (F, J, L and O) were clustered within the phylum, *Actinobacteria*, namely *Micrococcus luteus*, *Terrabacter tumescens*, *Propionibacterium acnes*, and *Arthrobacter oxydans*. These results indicate that *Proteobacteria* phylum was predominant in the biofilter, which is in agreement with earlier published results [28].

Based on the presence of their DDGE bands, *B. subtilis*, *A. aminovorans*, *P. denitrificans*, and *C. fustformis* were consistently present from days 4 to 28. *B. subtilis* is usually responsible for the degradation of proteins, *A. aminovorans* is able to effectively degrade organic amine compounds [29], *P. denitrificans* has been shown to be capable of removing sulfur-containing compounds and trimethylamine compounds [30,31], and *C. fustformis* is able to remove compounds such as BTEX and PAHs, which are difficult to degrade [32]. Thus, *A. aminovorans* and *P. denitrificans*, responsible for the degradation of sulfur- and nitrogen-containing compounds, accounted for 98.6% of the total amount of bacteria in the biofilter system (Table 2) when odor production was highest on the 10th day. *T. tumescens* and *F. mizutaii* belong to the group of bacteria that was most prevalent when the composting process neared its end [33]; consequently,

Table 2
Nucleotide sequence similarity and intensity of sequenced DGGE bands

Band[a]	Phylogeny	Closest relatives	Similarity	Relative abundance of DGGE bands		
				4-day	28-day	10-day
A	*Firmicutes*	*Staphylococcus capitis*	96%	0.15%	0%	0%
B	γ-Proteobacteria	*Pseudomonas citronellolis*	98%	0.08%	0.25%	0%
C	γ-Proteobacteria	*Pseudomonas fluorescens*	97%	0.12%	0%	0%
D	γ-Proteobacteria	*Pseudomonas putida*	99%	6.21%	0%	0%
E	*Bacteroidetes*	*Chryseobacterium scophthalmum*	98%	10.23%	0%	0%
F	*Actinobacteria*	*Micrococcus luteus*	96%	15.62%	0.36%	0%
G	*Firmicutes*	*Bacillus subtilis*	98%	2.35%	15.22%	0.63%
H	α-Proteobacteria	*Aminobacter aminovorans*	96%	12.08%	19.63%	34.21%
I	α-Proteobacteria	*Paracoccus denitrificans*	98%	14.23%	27.92%	64.39%
J	*Actinobacteria*	*Terrabacter tumescens*	95%	0%	17.62%	0%
K	*Bacteroidetes*	*Flavobacterium mizutaii*	96%	0%	1.68%	0%
L	*Actinobacteria*	*Propionibacterium acnes*	98%	0.13%	0%	0%
M	α-Proteobacteria	*Caulobacter bacteroides*	96%	11.35%	0%	0%
N	α-Proteobacteria	*Caulobacter fustformis*	96%	8.52%	6.41%	0.77%
O	*Actinobacteria*	*Arthrobacter oxydans*	99%	7.63%	6.63%	0%
P	β-Proteobacteria	*Comamonas testosteroni*	97%	11.43%	4.28%	0%

[a] The bands are designated as shown in Fig. 7.

they were only observed on day 28. Partial bacterial species, such as *S. capitis*, *C. scophthalmum*, *P. acnes*, and *C. bacteroides*, which also were present on day 4, are known to be capable of degrading carbohydrates during the primary-fermentation composting stage [33]. *P. fluorescens* and *P. putida* favored BTEX and H₂S removal also during the primary-fermentation composting stage [34,35]. Among the bacteria, *C. bacteroides* was proportionally the most prevalent (11.35%), followed by *C. scophthalmum* (10.23%). *P. putida* accounted for 6.21% of all populations, and populations of the other stains accounted for less than 1%. *P. citronellolis*, *M. luteus*, *A. oxydans*, and *C. testosterone* were found on the 4th and 28th days. These species are known to favor the degradation of branched hydrocarbons, dimethylamine, ammonia, and PAHs, respectively [36–39]. The fact that these compounds are more difficult to biodegrade or/and are constantly emitted during the composting process explains why these bacterial species were simultaneously observed at the primary-fermentation and ripening stages of the composting processes. Among the bacteria, *M. luteus*, which is responsible for the degradation of dimethylamine, was predominant, especially in the primary-fermentation stage.

The FISH technique was used to test the results of the DGGE analysis and thus avoid any possible bias due to the application of only one technique. As *Proteobacteria* belong to the largest group of bacteria under widespread environmental conditions [18], the α, β, and γ subclasses of *Proteobacteria* were selected for FISH analysis. Table 3 indicates the relative abundance of α-, β-, and γ-*Proteobacteria* with respect to all eubacterial cells present in the compost-based biofilter during different operating times. As can be seen by comparing Tables 2 and 3, the relative abundance of these *Proteobacteria* approximated that indicated by the DGGE analysis. For example, in the DGGE analysis, α-*Proteobacteria*, β-*Proteobacteria*, and γ-*Proteobacteria* accounted for 46.18, 11.43, and 6.41%, respectively, of the bacteria in the compost-based biofilter on the fourth day, while FISH analysis pegged them at 45.83, 10.26, and 5.83%, respectively

Table 3
Relative abundance of α-, β-, and γ-*Proteobacteria* with respect to all eubacterial cells present in the compost-based biofilter during different operating times

Phylogenetic group	Operating times		
	4-day	28-day	10-day
α-*Proteobacteria*	45.83%	52.63%	99.16%
β-*Proteobacteria*	10.26%	4.56%	0%
γ-*Proteobacteria*	5.83%	0.31%	0%

(Table 3). The residual populations belonged to phylum *Firmicutes*, *Bacteroidetes* or *Actinobacteria* (Table 2). In all, less than 5% of the biofilter bacterial population did not belong to any of the bacteria justified by the results of DAPI staining (data not shown). These microorganisms may be fungi or protozoa.

4. Conclusions

The experimental data reported here demonstrate that this newly developed biofilter system is capable of successfully purifying composting exhaust gases. The efficiency of this system to remove various odorous gases emitted from different food waste composting plants, including nitrogen- and sulfur-containing compounds, fatty acids, and hydrocarbon compounds, was evaluated and found to be satisfactory. This characteristic combined with satisfactory operating characteristics and the simplicity of the biofilter structure suggest that this biofilter system is suitable for application in composting plants. The feasibility of this biofilter system is further supported by its design, which is aimed at preventing the pressure drop and reducing bioaerosol emissions. The results of the molecular biology analyses (DGGE and FISH) provide valuable information on the roles or functions of bacterial species in the bacterial community in the biofilter during the deodorization processes. The valuable strains screened in this study should become candidate strains for the purification of composting exhaust gases in the future.

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