Improving performance of biofilters for bioaerosols and odour control in waste management facilities

by

Ikpe Emmanuel Ibanga

Submitted in accordance with the requirements for the degree of
Doctor of Philosophy

The University of Leeds
School of Civil Engineering

March, 2019

The candidate confirms that the work submitted is his own, except where work which has formed part of jointly-authored publications has been included. The contribution of the candidate and the other authors to this work has been explicitly indicated below. The candidate confirms that appropriate credit has been given within the thesis where reference has been made to the work of others.

I. **Ibanga, I. E.**, Fletcher, L. A., Noakes, C. J., King, M. F. and Steinberg, D. (2018). Pilot-scale biofiltration at a materials recovery facility: The Impact on bioaerosol control. *Waste Management*. **80** pp.154-167. https://doi.org/10.1016/j.wasman.2018.09.010

II. **Ibanga, I. E.**, Fletcher, L. A. and Noakes, C. J. (2017). Testing bioaerosol removal using pilot-scale biofilters at a material recovery facility (MRF). The Aerosol Society Focus Meeting 10 – Bioaerosols Conference, 8 June, Bristol.

In publications I and II, the candidate carried out study design, data collection and analysis, and draft of the original manuscript. The supervisors, Dr L. A. Fletcher and Prof C. J. Noakes, reviewed and provided comments on the manuscripts. Dr M. F. King provided guidance on the statistical analysis of collected data while Mr D. Steinberg was the research site supervisor who provided support and guidance for the research. Papers I and II have been reproduced in some aspects of chapters 3, 4, 5 and 9.

Dedication

This thesis is dedicated to the loving memory of my mother, late Mrs Ime Emmanuel Ibanga. I will always miss you!

Acknowledgement

To God be the glory for this dream come true. He graced me with life, strength and courage to sustain the needed motivation to see this through.

This research is the most significant academic challenge I have ever had to face, and without the support, patience and guidance of the following people, this thesis would not have come to fruition. It is to them that I owe my deepest gratitude.

- The University of Leeds PhD studentship funded by the Leeds International Research Scholarship (LIRS).
- My supervisors, Dr Louise Fletcher and Prof Catherine Noakes, who despite their busy schedules and many other academic demands pushed me in the right direction. Their skills, guidance, encouragement, constructive criticism and commitment to the highest standards really inspired and motivated me all through this study. Thank you from the bottom of my heart!
- Dr Marco-Felipe King who provided immense support with the data analysis of this project. Thank you so much!
- The management of Associated Waste Management Ltd (AWM), Leeds, for making their facility available for this study.
- Karen Stevens, Sheena Bennett and Dave Elliot, who were never tired of helping out with the laboratory aspects of this study.
- Jim Barnes and the entire University of Leeds Transport team for helping with transportation to and fro the study site.
- My wife, Edidiong Ikpe Ibanga – your love, prayers and communication kept me sane all through this journey right from the beginning. I love you babe!
- My parents, siblings and entire family, for their love, prayers, encouragement and support.
- Godwin Akpeimeh - my colleague, housemate, and now, best man - for the mutual support; indeed, you have been a loyal friend and a brother.
- Pastor and Pastor (Mrs) Ralph Ibiyeye and the entire family of The Redeemed Christian Church of God - Power Connections, Leeds for the prayers, support and encouragement.
- All my colleagues at the Faculty of Engineering especially in the School of Civil Engineering – you all made the journey bearable.

Abstract

This pilot-scale study aimed at improving the design and operation of biofilters to achieve simultaneous significant odour and bioaerosol reductions from waste air streams emitted from enclosed waste management facilities, using a materials recovery facility (MRF) as the source of contaminated air. The specific objectives included (i) to determine the key biofilter design and operating parameters required for a high level of odour and bioaerosol removal and to refine operational ranges and firmly define boundary conditions between normal and abnormal biofilter operations; (ii) to evaluate and characterise the concentrations of odour and bioaerosols that are being emitted as a result of waste management operations within the MRF; (iii) to evaluate the potential for biofilters to control bioaerosol emissions, and the potential for net emission of bioaerosols from biofilters both in terms of the overall concentration, and also the individual species; (iv) to determine the impact of gas residence time, media moisture content and media depth on simultaneous reduction of bioaerosols and odour in the process air; (v) to assess the impact of different biofilter media types (woodchips [old and new], peat and wheat straw) in terms of bioaerosol and odour emissions and removal; and (vi) to evaluate the possibility of improving a single biofilter for the removal of both bioaerosols and odour.

A pilot-scale biofiltration system was constructed for this study and comprised of four vertical up-flow plastic reactors filled with wood chips as the initial biofilter media and connected to a common plenum. Each reactor had a media volume of 181.5 L located above an air-space (for air distribution) separated by a metal mesh which supported the media. A six-stage Andersen sampler was used to measure the concentrations of four groups of bioaerosols (*Aspergillus fumigatus*, total fungi, total mesophilic bacteria and Gram negative bacteria) in the airstream before and after passing through the biofilters and these were expressed as cfu m^{-3}. Air for odour analysis was collected into air-tight Nalophan bags which were sent off to Concept Life Science odour testing laboratory for analysis within 30 hours of sampling. Olfactometry analysis was carried out on the odour samples in accordance with BS EN 13725 to determine the odour concentration of the samples in European odour units (OU_E m^{-3}). The performance of the pilot biofilters was evaluated on the basis of removal efficiency (%) for odour and bioaerosols.

The data showed that the concentration of bioaerosols in the process air (as indicated by the inlet air samples) varied from visit to visit in the range of $10^3 – 10^5$ cfu m^{-3}. The concentration of odour in the process air also varied between visits typically ranging from 94 to 489 OU_E m^{-3}. This was thought to be due to the complex interactions between the specific process operating conditions, the types of waste being processed and the configuration of the air ventilation system installed on the site. Overall, this study shows that biofilters designed and operated for odour degradation can also achieve significant bioaerosol reductions in the process air of waste treatment facilities, provided that the inlet concentration is high - which is the case for most enclosed waste treatment facilities. The biofilters achieved average removal efficiencies of 70% (35 to 97%) for *A. fumigatus*, 71% (35 to 94%) for total fungi, 68% (47 to 86%) for total mesophilic bacteria and 50% (-4 to 85%) for Gram negative bacteria, while odour reduction efficiency was in the range of 34 – 76%. Thus, biofilters can be effective for the control of potentially pathogenic species in the emissions from such treatment facilities. The performance of the biofilters was highly variable at low inlet concentration with some cases showing an increase in outlet concentrations, suggesting that biofilters had the potential to be net emitters of bioaerosols. Bioaerosol particle size distribution varied between the inlet and outlet air, with the outlet having a predominantly greater proportion of smaller size particles (3.3 μm) that represent a greater human health risk as they can penetrate the respiratory system more deeply, and even to the lung alveoli where gaseous exchange occurs. However, the outlet concentrations were low, and further reduction would be achieved by the combined effect of wind dilution and dispersal as well as exposure to environmental stress from temperature, desiccation and oxygen in full scale biofilter applications.

It appears that variations in gas residence time may not impact on bioaersosol removals; thus, gas residence time may not be critical for bioaerosol control. However, longer empty bed residence time (EBRT) delivered significant ($p < 0.05$) reductions in odour compared to shorter EBRT, implying that the longer EBRT accommodates the time required for both odorous contaminants diffusion transfer from the gas phase into the biofilm, and their subsequent biodegradation within the biofilm layer on the media materials, as established in literature. There also appears to be no media moisture content dependent differences ($p > 0.05$) in the bioaerosols reductions reported in this study. On the other hand, although not statistically

significant ($p > 0.05$), differences did exist in odour performance between the two groups, with the higher moisture content (40 – 70%) consistently showing better removals (odour RE range of 44 – 63%) than media moisture content in the range of 10 – 40%.

Furthermore, the two media depths (0.50 m and 0.25 m) investigated in this study showed potential capacity to control bioaerosols emissions from the process air of the MRF, and possibly other waste treatment facilities. Both depths achieved significant ($p < 0.05$) reductions of the inlet concentrations of bioaerosols as measured at the outlet. Although there were no statistically significant differences between the performances of both media depths, the 0.5 m media depth showed improved control of fungi than bacteria while the 0.25 m media depth had better removals of bacteria than fungi. This observation with the higher media depth has been thought to be a function of the large surface available for particles impaction; airflow rates and larger particles of fungi.

From the data, there were variations in the performance of the different media types assessed. Peat consistently delivered the highest simultaneous reduction of odour and bioaerosols; however, this was a much more expensive option. The performance of the wheat straw was the poorest both in terms of bioaerosols and odour reductions. Woodchips appeared to be the preferred choice particularly because they are relatively cost effective and offered satisfactory odour and bioaerosol removals (though not as high as peat). Nonetheless, the data indicated that the performance of woodchips may improve over time especially as the one year old woodchips indicated better removals of odour than the new woodchips which were freshly acquired for this study.

Overall, this study suggests that the ideal biofilter to simultaneously control bioaerosols and odour would be a woodchips-based reactor operated with a minimum media depth of 0.50 m and an EBRT of 16 s maintained at a moisture content level of between 40 and 70%, all of which lie within operational ranges reported in literature.

Table of Contents

List of Tables

List of Figures

List of Plates

List of Acronyms

AD	Anaerobic Digestion
ADMS	Advanced Dispersion Modelling System
AERMOD	Atmospheric Dispersion Modelling
AfOR	Association for Organic Recycling
ANOVA	Analysis of Variance
AWM	Associated Waste Management Ltd
BAT	Best Available Technology
BF	Biofilter
CEN	European Committee for Standardisation
cfu m^{-3}	Colony forming unit per cubic metre
COSHH	Control of Substances Hazardous to Health
EA	Environment Agency
EBRT	Empty Bed Residence Time
EC	Elimination Capacity
EU m^{-3}	Endotoxin Unit per cubic metre
FW	Food Waste
GAC	Granular Activated Carbon
GW	Green Waste
HSE	Health and Safety Executive
IAQM	Institute of Air Quality Management
IED	Industrial Emissions Directive
IVC	In-Vessel Composting
LATS	Landfill Allowance Trading Schemes
MBT	Mechanical Biological Treatment
MC	Moisture Content
MD	Media Depth
MRF	Materials Recovery Facility
MSW	Municipal Solid Waste
OAV	Odour Activity Value
OEF	Odour Emission Factor
OFMSW	Organic Fraction of Municipal Solid Waste
OU_E m^{-3}	European Odour Unit per cubic metre
PVC	Polyvinyl Chloride
qPCR	Quantitative Polymerase Chain Reaction
RDF	Refuse Derived Fuel
RE	Removal Efficiency
RPS	Regulatory Position Statement
SD	Standard Deviation
TWA	Time Weighted Average
USEPA	United States Environment Protection Agency
VOC	Volatile Organic Compound
WELs	Workplace Exposure Limits
WHC	Water Holding Capacity

Chapter 1
INTRODUCTION

1.1 Background

With continued emphasis on meeting the landfill diversion targets in the UK as established in the Landfill Directive (1999/31/EC) and with the launch of the Landfill Allowance Trading Scheme (LATS) in 2004 (Calaf-Forn et al., 2014), there has been an increase in the number of waste management facilities (Stagg et al., 2010; Environment Agency, 2017). Some of these facilities are enclosed, and can include mechanical biological treatment (MBT), in-vessel composting (IVC), anaerobic digestion (AD) and materials recovery facilities (MRF) or combinations of different waste management systems. These facilities, while achieving cutting edge recycling performance and value recovery from waste streams, have the potential for air pollution within the facility and externally via their extract ventilation especially due to odour and bioaerosol emissions.

Bioaerosols, which comprise predominantly plant pollen, microorganisms (viable or non-viable) and/or microbial metabolites, have the potential to cause health problems in exposed persons with symptoms such as irritation of the respiratory tract and eyes, coughing, wheezing, tiredness, rashes on skin, diarrhoea, asthma, headache, allergic rhinitis and hypersensitivity pneumonitis (Husman, 1996; Menetrez et al., 2009). Studies have shown that bioaerosol exposure can cause ill-health in exposed population (Douwes et al., 2003; Searl, 2008; Pearson et al., 2015). Lower forced vital capacity was reported in exposed compost workers (n = 190) compared to the controls (n = 38) (van Kampen et al., 2012). Hambach et al. (2012), while assessing work-related health symptoms among compost workers, reported an elevated proportion of the exposed group (n = 31) presenting with respiratory symptoms (29.0%), eye, nose and throat irritation symptoms (35.5%), gastrointestinal symptoms (29.0%) and skin rashes (20.0%) as against the control group (n = 31) who showed 3.3%, 13.3%, 6.7% and 0.0%, respectively, for these symptoms. The risk of waste workers' exposure to bioaersols may be dependent on the task they are undertaking (mostly indoors for enclosed facilities), their proximity to the source of bioaerosols and the abatement system being used on site (Stagg et al., 2010).

Odours, on the other hand, can be described as "the property of a chemical substance or substance mixture, (dependent on the concentration), to activate the sense of smell and thus [cause] an odour sensation" (The Composting Association, 2004). The odour sensation is usually perceived by a person to be offensive and/or unpleasant, thus, constituting nuisance. Blazy et al. (2014) assessed the odorous emissions for a forced aeration composting system treating sludge from a pig slaughterhouse and identified nine potential odorous compounds including Hydrogen sulphide, Trimethylamine, Dimethyl disulphide, Dimethyl sulphide, Ammonia, Dimethyl trisulphide, Acetophenone, 2-Pentanone, and 1-Propanol-2-methyl. Odour nuisance and strict enforcement of laws towards treatment of polluted air encourage operators to seek out efficient ways of treating waste gases from their facilities. Many technologies (such as wet scrubbers, incinerators, adsorption on activated carbon and masking agents) have been developed and applied to achieve this with varying degrees of success (Devinny et al., 1999; Fulazzaky et al., 2013). However, these technologies present with various pitfalls such as transfer of pollutants from the gas phase to solid adsorbents or scrubbing liquids, thus, resulting in solid waste and wastewater treatment considerations (Lin et al., 2001; Chung et al., 2007). Because the concentrations of the pollutants in the gas to be treated are relatively low, it becomes expensive to apply the aforementioned traditional air pollution control technologies (Wani et al., 1999).

In the UK, the Environment Agency (EA) is responsible for regulating waste management facilities, usually done through the granting of Permits to Operate. Part of the EA's remit is to ensure that odours and bioaerosols do not adversely impact the surrounding population (Frederickson et al., 2013), and so have included bioaerosol monitoring requirements as an environmental permit condition, and to assess the performance of abatement systems under operation in such facilities (Environment Agency, 2017). The EA gave a precautionary guidance for composting operators when applying for operating permits. This guidance stipulates that concentrations of bioaerosols (as predicted or measured directly) need to be maintained no higher than acceptable levels at 250 m from the composting site or the nearest sensitive receptor (such as a dwelling or workplace which is not part of the composting site), whichever is closer (Environment Agency, 2010). These acceptable levels have been defined as 500 cfu m^{-3}, 1000 cfu m^{-3} and 300 cfu m^{-3} for *Aspergillus fumigatus*, total bacteria and Gram-negative bacteria, respectively, as measured by the standardised monitoring

protocol (i.e. the AfOR protocol later replaced in 2017 by the M9 protocol). However, an updated regulatory position statement (RPS) on monitoring bioaerosols at regulated facilities was provided by the EA in January 2018, and excluded the reporting of Gram-negative bacteria (Environment Agency, 2018b). In the UK, there are no regulatory occupational limits for bioaersols as the acceptable levels stated above are not based on dose-response relationships (Pearson et al., 2015). However, the Control of Substances Hazardous to Health (COSHH) Regulation issued by the Health and Safety Executive (HSE) provides employers with the requirements for assessing, monitoring and controlling the exposure of employees to hazardous substances in work environments (HSE, 2013), and thus, applies to workers in waste handling facilities. In Germany, there is a regulatory occupational limit of 50 000 cfu m^{-3} for mesophilic fungi (including *A. fumigatus*) in breathable air within the workplace (BAUA, 2013 cited in Pearson et al., 2015).

Biofilters have been used as an abatement method in the waste management industry for many years with varying degrees of success. Biofilters are three phase bioreactors (gas, liquid, solid) composed of filter beds which have high porosity; high buffer capacity; high nutrient availability and high moisture retention capacity which altogether provide suitable internal environments that support the growth and attachment of a mixed-culture of pollutant-degrading microorganisms (Elias et al., 2002; Dastous et al., 2005). Biofilters offer a cost-efficient and potentially environmentally friendly alternative to traditional air treatment technologies, particularly for odour and gas treatment because of the low energy requirement; relatively low construction costs; no generation of secondary pollutants that require subsequent disposal; and capacity to treat a broad spectrum of gaseous compounds (Devinny et al., 1999; Fulazzaky et al., 2014). Biofilters are a method of biological air treatment systems that utilise populations of microorganisms to convert certain organic and inorganic pollutants into compounds and/or forms that are less toxic and/or odourless. The microbial population, which may be dominated by a single species or be composed of different interacting species, employ oxidative, and sometimes, reductive reactions to convert the airborne pollutants into CO_2, water vapour, and to increase their population using these pollutants as energy and carbon sources (Fletcher et al., 2014). The design and operation of the early biofilter systems were based on a very basic understanding of their method of operation. Although in recent years the structural materials used for biofilters have become more

sophisticated, and in the UK there is a move towards using emission stacks, the fundamental design criteria have changed very little (Fletcher et al., 2014).

Several studies have been carried out over the past two decades, in an attempt to better understand the principles of biofilter design and operation to achieve significant odour and bioaerosol removal. Some of these have looked at the microbiology of the biofilters (Juteau et al., 1999), technical characteristics(Pagella and De Faveri, 2000), performance (Jorio et al., 2000), modelling (Alonso et al., 1999), and economic viability (Gao et al., 2001). It is acknowledged that biofilters offer a versatile and cost effective option for the management of contaminated air from waste handling and treatment facilities (Devinny et al., 1999; Kummer and Thiel, 2008; Frederickson et al., 2013). However, there is a lot of contradictory data and many gaps in the knowledge which need to be addressed if biofilters are to be designed to effectively control all emissions and to perform efficiently. In particular several authors have suggested that media characteristics such as porosity, moisture content, nutrient content, temperature and water retention capacity are the most important factors governing biofilter performance, although the optimum ranges quoted in the literature vary significantly from one author to another (Devinny et al., 1999; Nicolai and Janni, 2001a; Quigley et al., 2004; Schlegelmilch et al., 2005a; Álvarez-Hornos et al., 2008; Frederickson et al., 2013). Other authors suggest that operating parameters such as empty bed residence time (EBRT), contaminant loading rate and upflow or downflow configuration are important factors but again there seems to be little consensus as to what the optimum ranges are (Leson and Winer, 1991; Lu et al., 2002; Chen and Hoff, 2009; Liu et al., 2009).

Recent studies by Frederickson et al. (2013) and Fletcher et al. (2014) have evaluated the performance of laboratory-scale and full-scale biofilters in terms of their capacity for simultaneous control of odour and bioaerosols by considering what parameters were vital in defining what design, conditions and maintenance schedules were required for optimum performance. In particular, the study by Fletcher et al. (2014), with an overall objective to determine the extent to which abatement methods incorporating either open or enclosed biofilters can reduce both bioaerosols and odour emissions from enclosed biowaste treatment operations, provided remarkable input into the current knowhow in the industry by bringing together all existing knowledge and research relating to the performance of biofilters and benchmarked the

performance of full scale biofilters operated under ideal conditions (Fletcher et al., 2014). The key findings of this study included:

(i) Well designed, operated and maintained biofilters were capable of achieving significant and sustainable reductions in biowaste odours up to 94% (with outlet concentrations in the range of 200 to 5500 OU_E/m^3);

(ii) Despite the fact that biofilters can achieve bioaerosols reductions, performance was variable with time and from site to site; and at low concentrations biofilters may be net emitters, particularly, of fungi;

(iii) The impact of the design and operating parameters of biofilters varies and the key variables do not appear to be the same for both odour and bioaerosols; which may reflect the different removal mechanisms involved;

(iv) The use of upstream scrubbers prove beneficial with regards to removal of bioaerosols, dust and potentially toxic pollutants (e.g. ammonia) that may adversely affect biofilter performance; and so biofilters either in isolation or in combination with scrubbers represent a candidate Best Available Technology (BAT) for odour control in biowaste facilities, especially if properly designed and operated.

However, these studies concluded that the literature contains apparently contradictory information regarding the impact of biofilter design and operating parameters (such as empty bed residence time, moisture content, media pH and temperature) on odour and bioaerosol emissions and removal. This is a major issue for waste management operators and regulators as there is no clear guidance in terms of design and operating parameters that would provide a robust evidence base against which to benchmark the effectiveness of existing biofilters and future abatement system proposals including biofilters. Although bioaerosols removal mechanisms by biofilter have been thought to include inertial deposition, diffusional (or Brownian) deposition and flow line interception (Ottengraf and Konings, 1991), Frederickson et al. (2013) recommended that further research is required to determine the relationship between odour and bioaerosol emissions from biofilters to determine the extent to which biofilters may be used to effectively reduce both odour and bioaerosols, and to identify best practice techniques for optimising biofilters to maximise control of both odour and bioaerosol emissions. This is especially necessary because of the differences in the removal mechanisms of odour and bioaerosols. Literature suggests that odour removal

mechanisms is dependent on sorption of the odorous compounds into the biofilm layer on the media surface where biodegradation takes place, a function which relies on long residence time; whereas bioaerosol removal is achieved via particle impaction onto the media particles, and so an extended residence time may not impact positively on removal (Devinny et al., 1999; Fletcher et al., 2014). Thus, it is imperative to develop a better understanding of biofilter design and effective performance monitoring techniques especially if they are to continue to control all emissions and achieve their full potential.

1.2 Research Questions

From the review of the existing literature, it is apparent that much has been reported on the performance (removal efficiencies and elimination capacities) of biofilters in composting facilities especially regarding odour removal. Other studies have highlighted the potential of two-stage systems for mitigating odours from swine facilities especially because of their advantage of preventing media compaction (Chen and Hoff, 2012). This has stimulated a discussion regarding the improvement of these systems and how to adapt them for control of odour and bioaersols in enclosed biowaste treatment plants, with the following questions:

(i) Can biofilters achieve significant simultaneous removal efficiencies for odours/odorous compounds and bioaerosols emitted from enclosed waste management facilities?

(ii) What design parameters and operating factors can be manipulated to achieve these high removal efficiencies for odours and bioaerosol emissions?

(iii) Can two-stage biofilters systems which are set up to prevent media compaction provide better removal efficiencies for odour and bioaerosols over single stage biofilters?

(iv) Are there specific media types or combinations of different media or media mixes that can achieve significant simultaneous reduction of odour and bioaerosols emitted from enclosed waste management facilities?

(v) Are there any similarities or differences in the inlet and outlet species composition of bioaerosols for enclosed biowaste facilities?

1.3 Research Aim

The overall aim of this research project was to characterise the design and operating parameters of biofilters with a view to improving their design and operation to achieve simultaneous significant reduction of odour and bioaerosol emissions from enclosed waste management sites.

1.4 Research Objectives

The study approach focused on a series of pilot-scale biofilter experiments at a materials recovery facility (MRF) to test design and operational parameters with a real source of typical contaminated air. The objectives were defined as follows:

(i) To determine the key biofilter design parameters and operating parameters required for a high level of odour and bioaerosol removal and to refine operational ranges and firmly define boundary conditions between normal and abnormal biofilter operations.

(ii) To evaluate and characterise the concentrations of odour and bioaerosols that are being emitted as a result of waste management operations within the MRF.

(iii) To evaluate the potential for biofilters to control bioaerosol emissions and the potential for net emission of bioaerosols from biofilters both in terms of the overall concentration, and also the individual species.

(iv) To determine the impact of gas residence time, media moisture content and media depth on simultaneous reduction of bioaerosols and odour in the process air.

(v) To assess the impact of different biofilter media types (woodchips [old and new], peat and wheat straw) in terms of bioaerosol and odour emissions and removal.

(vi) To evaluate the possibility of improving a single biofilter for the removal of both bioaerosols and odour.

1.5 Scope of the Research

This study employed pilot scale studies to achieve the aim and objectives over the study period of 14 months (May 2015 – July 2017). The pilot study was conducted in a materials recovery facility owned by Associated Waste Management Ltd (AWM), located on Gelderd road, Leeds. A pilot scale biofilter system (consisting of four biofilter units) was constructed and commissioned at the AWM site, with each unit

evaluated for odour and bioaerosols (as removal efficiencies in %) under various design and operating conditions.

1.6 Significance of the Research

According to Fletcher et al. (2014), biofilters do represent a viable abatement system for odours and bioaerosols generated from biowaste treatment facilities. Their study reported that biofilters have the capacity to achieve an effective level of odour and bioaerosol removal, and are applicable to a variety of biowaste processing operations currently in use in the UK. However the study also raised a number of key issues that still need to be addressed regarding the performance of biofilters.

This research provides a significant amount of fundamental data relating to the key design and operating parameters and their impact on the ability of biofilters to remove high concentrations of bioaerosols and odour generated at enclosed waste management facilities. It also provides data on the performance of poorly designed and operated biofilters and those that have had some kind of operational failure. This will enable operators and regulators to understand how biofilters cope under extreme operating conditions outside the established norms.

Although single biofilter systems can provide effective bioaerosol and odour reduction there is a question as to whether they can ever achieve significant reductions in bioaerosols and odour simultaneously. This research also assessed the potential for improving bioaerosol and odour removal in a single system, which has not been looked at previously. In addition the study has assessed the performance of different biofilter media types for simultaneous reductions of bioaerosols and odour emissions from these facilities.

The findings of this study will provide operators and regulators with a much more comprehensive understanding of the design, operation and performance of biofilters, which in turn will provide regulators with a more robust, evidence-based process to regulate and grant permits for biowaste treatment facilities. This will become increasingly important given that the number of enclosed biowaste treatment facilities is likely to continue to increase in response to current waste management legislation. The findings of this research will also add to the data pool for further analysis and inferences by researchers and stakeholders both in academia and industry.

Furthermore, even though the focus of this research is on treatment of waste gas emitted from enclosed biowaste treatment facilities, the knowledge gained can be extended to other facilities (such as anaerobic digestion facilities and wastewater treatment works) with huge potential for significant emissions of odour, volatile organic compounds and bioaerosols.

1.7 Organisation of Chapters

This thesis is organised in ten chapters, with Chapter 1 highlighting the study background with clearly defined research questions, aim and the objectives of the study. Chapter 1 also presents the scope of this project as well as the significance of the study especially to the operators and regulators within the waste management industry.

Chapter 2 focuses on addressing some aspects of objective 1 through an in depth literature review on odour and bioaerosols emissions at waste management facilities, mitigation of these emissions using biofilters alone and biofilter/scrubber combinations with a discussion of the key aspects of design and operation of biofilters that are considered vital for achieving optimum performance of biofilters. This chapter also presents a summary of the gaps in the knowledge which must be addressed if biofilters are to be designed and operated to effectively control all emissions. In addition to the comprehensive literature review in Chapter 2, each results chapter (Chapters 4 – 8) also presents a more specific introductory literature review to set the tone for the specific objective and results discussed in those chapters.

Chapter 3 describes the methods employed in achieving the objectives of this study, and covers the design and fabrication of the pilot-scale biofilters, the study site selection, the air sampling methods applied for bioaerosols and odour analyses, the biofilter performance assessment and data analysis carried out to fulfil the research aim. In addition to the general methodologies presented in Chapter 3, each result chapter (Chapters 4 – 8) also includes detailed methodologies that are specific to the experiments presented in those chapters.

Chapter 4 is the first result chapter, and presents elements of this research project that have been published in a paper entitled *'Pilot-scale biofiltration at a materials recovery facility: The impact on bioaerosol control'* in Waste Management journal (Ibanga et al., 2018). This chapter presents the initial biofilter performance assessment

for bioaerosol control, investigating the impacts of gas residence time and inlet bioaerosols concentration as well as the net bioaerosol emitting potential of biofilters. It also assesses the impacts of biofiltration on particle size distribution between inlet and outlet air samples.

In Chapter 5, the focus is on assessing the impact of gas residence time on the simultaneous control of bioaerosols and odour. The objectives of the study included (i) to assess the variability of the inlet concentrations of bioaerosols and odour (ii) to assess the performance of three levels of empty bed residence time (EBRT) – 11 s, 16 s and 70 s – in terms of removal efficiencies (REs), bioaerosol load removal (L) and bioaerosol removal rate (R) and (iii) to assess the effects of EBRT on particle size distribution between inlet and outlet samples of all biofilters (BFs).

Chapter 6 presents the findings of experiments which assessed the impacts of media moisture contents on the simultaneous control of bioaerosols and odour. The main objectives were focused on (i) assessing the performance of two levels of media moisture content – 10 to 40% and 40 to 70% – in terms of bioaerosols and odour reductions, bioaerosol load removal (L) and bioaerosol removal rate (R) and (ii) assessing the effects of media moisture content on particle size distribution between inlet and outlet samples of all BFs.

In Chapter 7, the objective was to assess the impact of biofilter media depth on the reduction of bioaerosols. Thus, the performance of two media depths – 0.5 m and 0.25 m – were assessed in terms of removal efficiencies, bioaerosol load removal (L) and Bioaerosol Removal Rate (R) and their effects on the particle size distribution between inlet and outlet samples.

Chapter 8 presents the findings of the experiments to determine the impact of media types on simultaneous control of bioaerosols and odour. The media assessed include old and new wood chips, peat and wheat straw. This chapter also presents the results of the assessment of the impact of media age on simultaneous bioaerosol and odour removal; as well as the assessment of the impact of media type on bioaerosol particle size distribution between biofilter inlet and outlet air samples.

Chapter 9 presents a general discussion of the data obtained from the sampling carried out as part of this study, and explores how these fit relative to the existing body of knowledge in the literature within the context of the original research questions presented in Chapter 1. Chapter 9 also comments on the variability in the data and the

benefits and limitations of using pilot scale approaches. This is followed by Chapter 10 which summarises the key conclusions of this study and the recommendations for future research.

Chapter 2
LITERATURE REVIEW

2.1 Introduction

This chapter presents an in depth review of existing literature on odour and bioaerosols emissions at waste management facilities as well as the mitigation of these emissions using biofilters alone and biofilter/scrubber combinations with a discussion of the key aspects of design and operation of biofilters that are considered vital for achieving optimum performance of biofilters. This chapter also highlights the gaps in knowledge which must be addressed if biofilters are to be designed and operated to effectively control all emissions.

2.2 Review on odours and odour emissions

In the UK, emission of odorous compounds from biowaste treatment facilities continues to be of major concern especially because of the possibility of constituting odour nuisance to residents in the immediate vicinity of such facilities; thus, generating complaints which may have a bearing on planning applications, permit determinations and/or even prosecutions, and may also lead to a lack of acceptance of such facilities (Schlegelmilch et al., 2005b; Jacobs et al., 2007). Apart from the potential for constituting odour nuisance, emissions of odour can have health implications, especially psychologically-motivated impacts as suggested by several studies (Dalton et al., 1997; Dalton, 1999; Dalton, 2002; Smeets et al., 2002; Shusterman, 2002). The releases of odours into the environment have been implicated in health complaints with symptoms including nausea, headache, dizziness, and irritation of body parts such as the eyes, nose and throat (Steinheider et al., 1998; Herr et al., 2003). Defra (2010) reports that people with health problems show a higher likelihood of undergoing odour nuisance and annoyance than healthy ones for the same level of odour exposure, and people who feel anxious that odour has potential health risks show more tendencies to experience odour-triggered annoyance.

Every biowaste has the potential for odour releases, hence it becomes imperative for site operators to give some level of guarantee to local residents and regulatory bodies that odours can be contained and minimised to as low as reasonably practicable. This avoids the cascade of public nuisance and mass rejection of these facilities which may reflect badly on the industry (Jacobs et al., 2007). This section presents information

from current literature on what odours are, their key attributes, odour generation from biowaste treatment facilities and key odorous chemical compounds generated from operations of these facilities. This is important, because for there to be effective understanding and planning of odour abatement and management at biowaste treatment sites, it is vital to have a vast knowledge on all these aspects of odours.

2.2.1 What are odours?

Odours can be defined as the stimuli from chemical compounds that are volatilised in air (Defra, 2010). The Composting Association also describe odour as "the property of a chemical substance or substance mixture, (dependent on the concentration), to activate the sense of smell and thus [cause] an odour sensation" (The Composting Association, 2004). A key distinction is set between odour and malodour by Jacobs et al. (2007) who define odour as a volatile substance or mix of volatile substances which are perceived through the sense of smell, and malodour as odour which is considered unpleasant and offensive in nature by a person. This brings about the differences in odour perception among various people because the interpretation of an olfactory sensation as odour depends on the perceiving individual who may interpret it to be pleasant or unpleasant (Defra, 2010). Thus, the key concern for odour emissions from biowaste treatment facilities is the tendency to trigger sensation that is perceived to be offensive or objectionable. In order to fully appreciate the relative contribution of odour sources from biowaste facilities and their potential to cause deplorable impacts, it is important to have an understanding of the perception of odour as well as the key odour attributes.

2.2.2 Odour Perception

Jacobs et al. (2007) stated that for odour to be perceived, three important criteria must be met; these include (a) that the chemical must be released into the air, (b) that it is then dissolved in the olfactory mucus (composed mostly of water), and (c) that receptor nerve cells must be available for detection (Figure 2.1). In humans, the odorant molecules in air are transported over the olfactory epithelium in which are embedded approximately 350 different types of receptor cells, with each containing only one type of G-protein-coupled receptor that is capable of responding to a range of chemicals (Buck, 2000; Jacobs et al., 2007). Thus, when air containing molecules of odorous chemicals or mixtures of such chemicals arrive on the receptor surface, these molecules bind to a range of different receptors.

A single chemical is capable of activating several types of receptors, and so a mixture of chemicals is capable of creating a much more complex response on many receptors; thus, resulting in the ability to differentiate between thousands of diverse smell and odour combinations (Jacobs et al., 2007). Buck (2000) noted that the signal from the receptor has a temporal component which describes the fast or slow binding of certain chemicals, and the consequence of interactions at the receptor binding sites. Signals from each of the activated G-protein-coupled receptors are relayed to a glomerulus, which is a structure located within the olfactory bulb. The signals from the glomerulus form a spatial and temporal pattern, which is then used as the input for cortical processing in higher brain centres, particularly the piriform (primary olfactory) cortex (Stevenson and Wilson, 2007).

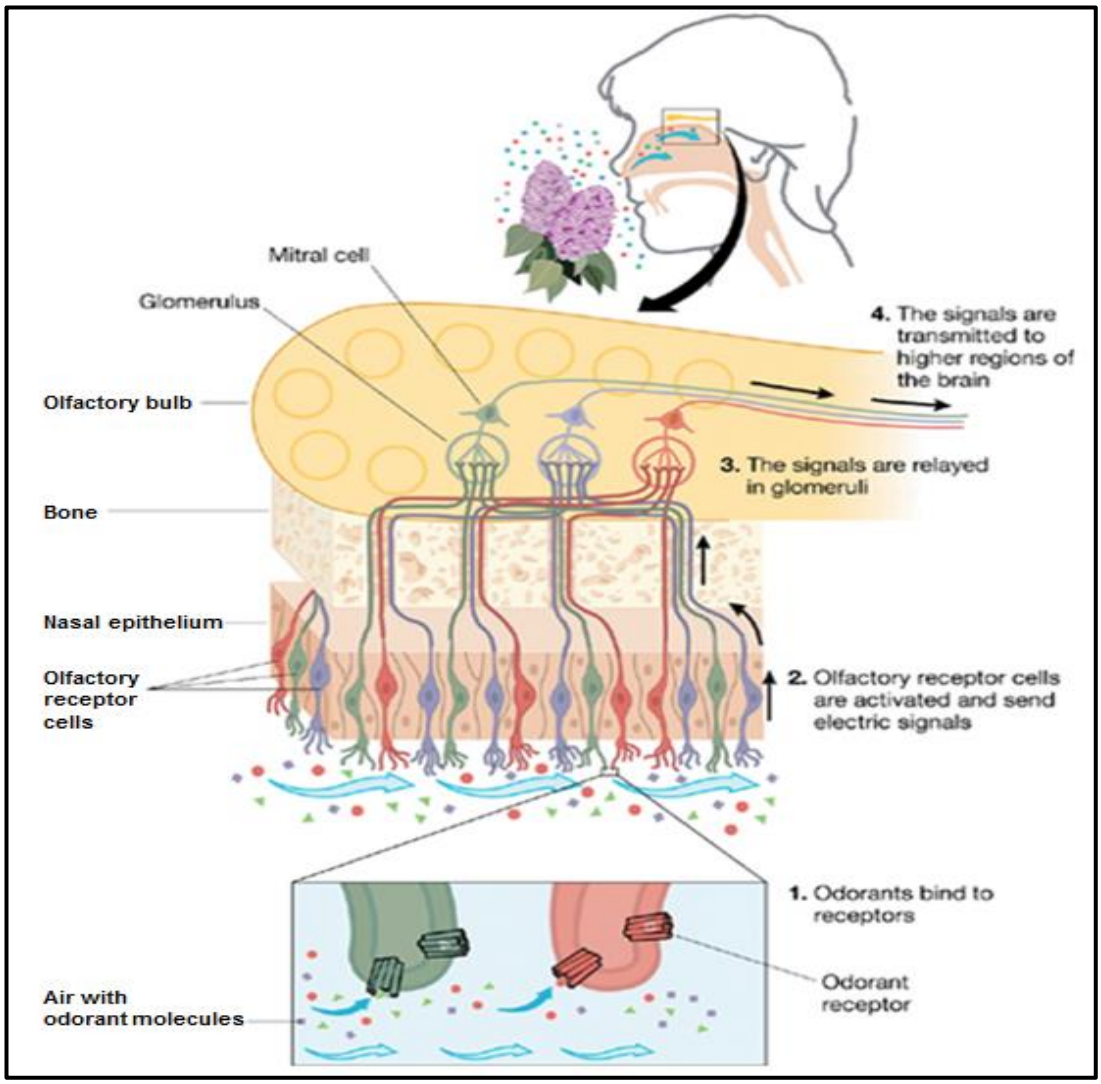


Figure 2. 1: The pathway of odour perception (Dalvi, 2013)

2.2.3 Odour attributes and characteristics

It is essential, at this point, to offer some of the descriptions of odours to aid understanding especially from the complainants' point of detection. Some of the attributes described in this section include odour units and thresholds, odour intensity, the hedonic tone and quality of odours.

2.2.3.1 Odour units and thresholds (concentration)

The odour unit (OU) is defined by Defra (2010) as a sensory measurement of the concentration of a mixture of odorous compounds in a sample of air, and which is expressed as a European odour unit or 1 OU_E by the BS EN 13725 standard. Odour threshold, on the other hand, describes the concentration at which odour is just detectable by the human nose; a concept which offers the basis for olfactometry (Defra, 2010). The European Standard (BS EN 13725:2003) stipulates the standardised methods for determining and reporting odour detectability or concentrations (Curtis, 2009).

Two types of odour thresholds have been described - the absolute and the difference thresholds (Gemert, 2011). The absolute threshold can be categorised into two,

(a) the detection threshold which is the concentration at which an odour just becomes strong enough to create a sensation within controlled conditions such as in an odour laboratory set ups e.g. the concentration that is just detected by a panel of human "sniffers" (and it is the concentration of 1 OU_E m^{-3}); and
(b) the recognition threshold which is the concentration at which an odour becomes identifiable or recognisable as a specific odour, and is generally higher than the odour detection threshold (Defra, 2010; Gemert, 2011).

The difference threshold of odour defines the smallest change in concentration of a substance required to give a perceptible change.

Table 2.1 shows a selection of regulatory limits and measurements for odour in some member states of the European Union (EU). No statutory limits have been defined for ambient odour concentrations (whether for individual or mixtures of odorous compounds) in the UK. Nonetheless, odour benchmark levels (Table 2.2) have been defined by the Environment Agency in a guidance on odour issued for use in the evaluation of potential emission impacts from plants which are regulated under the 2010 Environmental Permitting Regulations for England and Wales (and the subsequent amendments); where the benchmark levels for odours of different

offensiveness are given as the 98th percentile of hourly mean concentrations in European odour units (OU_E) over a year (Redmore, 2012). As already stated, there is a wide variation in perception and sensitivity for different people; thus, the threshold value obtained in olfactometry is based on statistics which reflects an average response from 50% of a panel of human "sniffers" (Defra, 2010).

Table 2. 1: Odour limits in some EU member states (The Composting Association, 2004)

Country	Limit and description
Austria	MBT plants: 500 OU_E m^{-3} with a minimum distance from residential property of 500m
Denmark	500-meter distance to the nearest neighbour
	5 to 10 OU_E m^{-3} at the nearest neighbour
Germany	Relative frequency of 10% (percentage of hours a year with odour hours considered to be a significant nuisance) for residential areas. For industrial areas the percentage rises to 15%. An 'odour hour' is any hour in which there is a continuous odour perception for a period of 6 minutes (Federal Standard, GIRL)
Holland	1.5 OU_E m^{-3} as 98 percentile for green waste and household
	organic waste composting
Ireland	Based on olfactometry
	• 3 OU_E m^{-3} at the 98th percentile for new facilities
	• 6 OU_E m^{-3} at the 98th percentile for existing plants
United Kingdom	No set limit. Any of the following could be applied:
	• No nuisance
	• No odour at the plant boundary
	• X ppb H_2S
	• 6 OU_E m^{-3} at the 98th percentile for existing facilities.

Table 2. 2: The Environment Agency Odour Benchmark Levels (Redmore, 2012)

Relative Offensiveness of Odour	Benchmark Level as 98th Percentile of 1-hour Means (OU_E m^{-3})*
Most offensive odours: • Processes involving decaying animal or fish • Processes involving septic effluent or sludge • Biological landfill odours	1.5
Moderately offensive odours: • Intensive livestock rearing • Fat frying (food processing) • Sugar beet processing • Well aerated green waste composting	3.0
Less offensive odours: • Brewery • Confectionery • Coffee roasting • Bakery	6.0

* Practically, this implies the 175th highest hourly average recorded within a year.

2.2.3.2 Intensity

Intensity of odour defines the perceived strength of an odour as described by a recipient of the odour (Defra, 2010). Table 2.3 shows the qualitative scoring template used by panellists for an odour sample compared to a scale for ranking odour intensity. Odour intensity increases as the concentration increases; however, it is possible for one odour to smell stronger than another odour even at the same level of concentration, and so an increase (or decrease) in odour concentration may not always result in a corresponding increase (or decrease) in the strength of odour that is perceived; this is because of the logarithmic relationship between odour intensity and concentration (Defra, 2006). This is a major point for consideration especially when addressing control because odour and odorous compounds that have strong intensities at low concentrations may bring about nuisance even at low residual levels.

Table 2. 3: Odour Intensity Scale (Defra, 2010)

Score	Intensity
0	No odour
1	Very faint odour
2	Faint odour
3	Distinct odour
4	Strong odour
5	Very strong odour
6	Extremely strong odour

2.2.3.3 Hedonic tone

This is the degree to which a person perceives odour as pleasant or unpleasant (Defra, 2006). Hedonic tone varies among people, and is greatly influenced by prior experience, and sometimes emotions (among other things), at the time the odour is being perceived. The hedonic tone of odour is typically rated using scales that range from negative numbers, which represent unpleasantness, to positive numbers, which represent pleasant odour (Fabian-Wheeler et al., 2012). The scales vary depending on research sites and typically range from -4 to +4 (e.g. coffee +2.33, cherry +2.55, alcohol -0.47, fishy -1.98, cadaverous -3.75) or -11 to +11, usually with neutral odours recorded as zero. Typical hedonic tone scales are shown in Table 2.4 and Figure 2.2. This has been developed to aid comparative judgements for the subjective nature of odour reports/complaints (Jacobs et al., 2007; Defra, 2010).

Table 2. 4: Hedonic Tone Scale for -4 to +4 (Defra, 2010)

Score	Perceived Hedonic Tone
+4	Very pleasant
+3	Pleasant
+2	Moderately pleasant
+1	Mildly pleasant
0	Neutral odour / no odour
-1	Mildly unpleasant
-2	Moderately unpleasant
-3	Unpleasant
-4	Very unpleasant

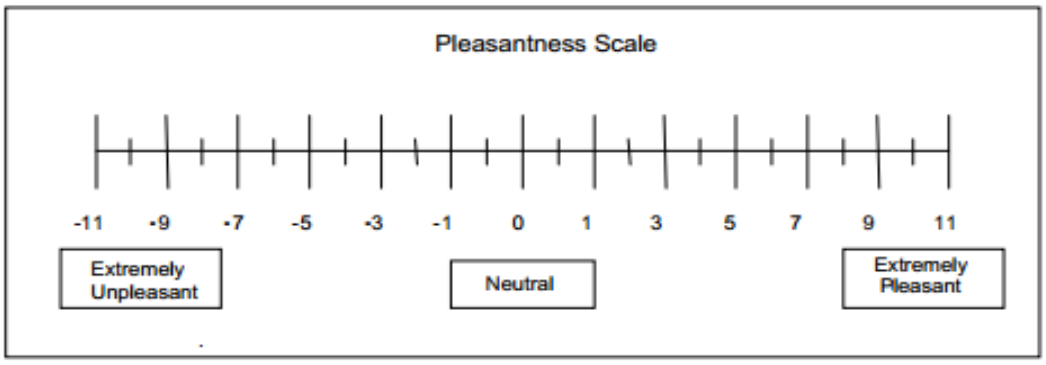


Figure 2. 2: The Hedonic Tone Scale for -11 to +11(Fabian-Wheeler et al., 2012)

2.2.3.4 Odour Quality

The quality (or character) of odour is the property that singles out or distinguishes an odour from another of equal intensity (Defra, 2006). It is the property of the odour that is described as fruity, almond, fishy, etc. Odour quality descriptors (Figure 2.3) are used on odour samples which are at or above the recognition threshold (Fabian-Wheeler et al., 2012). It is important to note that simply the presence of an odour does not necessarily imply that it is offensive. Odour pollution depends on several factors; hence, no single method can reliably assess odour pollution or measure odour, and conclusions on these parameters are based on a collection of pieces of evidence (SEPA & Natural Scotland Scottish Government, 2010; Environment Agency, 2011).

The parameters of an odour that are considered when evaluating its offensiveness are Frequency of detection, Intensity as perceived, Duration of exposure, Offensiveness, and Receptor sensitivity; factors which are at times described by the acronym FIDOR (Environment Agency, 2011). Redmore (2012) pointed out that complaints are likely to arise from frequent, intense and highly offensive odours as well as prolonged exposure to odours. Also, more sensitive areas (such as homes and hotels) are likely to have little or no tolerance for odorous emissions (Sniffer, 2013). The Environment Agency (2002) argues that odour nuisance results from long-term recurrent exposure to odours which trigger negative appraisal in the perceiving individual(s). Figure 2.4 shows the complex mechanism that leads from odorants emission into the atmosphere to actual odour nuisance and complaints. It is noted that odour nuisance problems become aggravated once the first complaint has been made (Environment Agency, 2002).

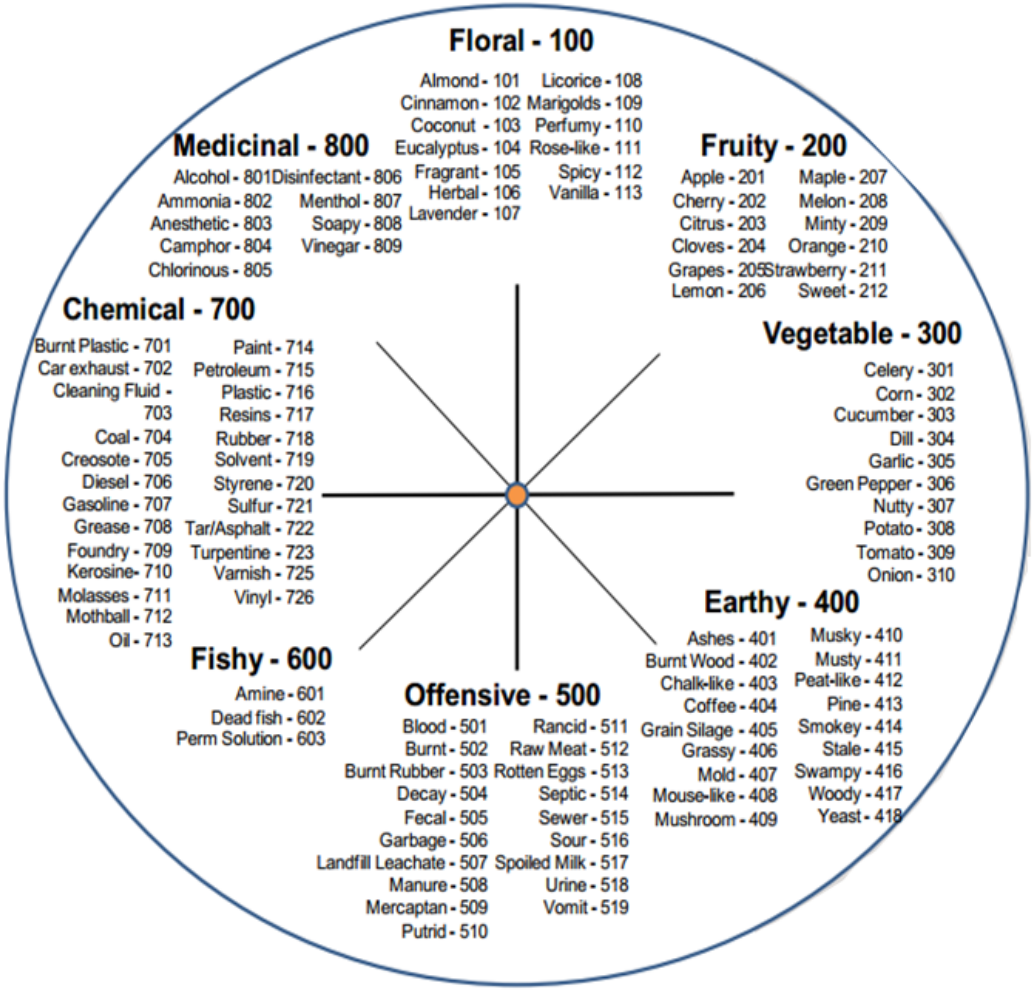


Figure 2. 3: Odour characterization - environmental odour descriptor wheel (St. Croix Sensory, 2003 cited in Fabian-Wheeler et al., 2012).

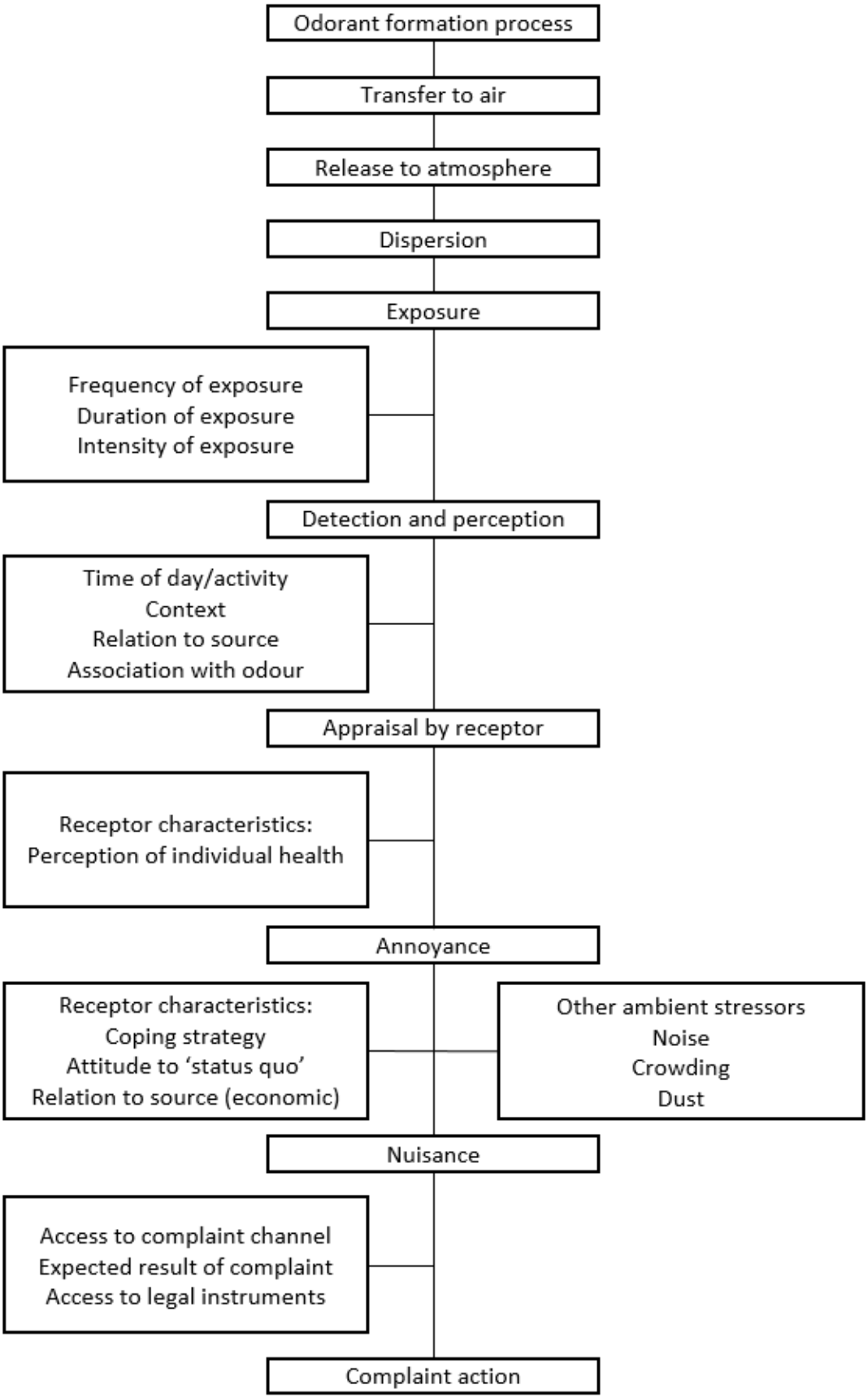


Figure 2. 4: Steps from odorant formation to complaint (Environment Agency, 2002)

2.2.4 Odour Assessment

One of the key elements of odour assessment planning is a prediction and/or observation of the likely impacts and resultant effects at sensitive receptors in the surrounding area of the odour source with the use of suitable assessment tools. This covers aspects such as determining the likely magnitude of odour emissions and the meteorological characteristics of the site of odour source, the dispersion and dilution offered along the source-pathway-receptor route, the receptor(s) sensitivity level, and the potential for increased effects of emitted odours in addition to any odours of similar character (Bull et al., 2014). It is therefore important that any odour assessment approach employed measures both the impact and effects of exposure. The Institute of Air Quality Management (IAQM) suggests a means of gauging the degree of the effects which result from the impact on a receptor for different levels of sensitivity (Table 2.5). Worthy of mention here is that this is a binary judgment system that specifies whether the likely predicted effect is 'significant' or 'not significant'.

Table 2. 5: Institute of Air Quality Management (IAQM) suggested descriptors for magnitudes of odour effects (Bull et al., 2014)

		Receptor Sensitivity		
		Low	**Medium**	**High**
Odour Exposure (Impact)	Very Large	Moderate adverse	Substantial adverse	Substantial adverse
	Large	Slight adverse	Moderate adverse	Substantial adverse
	Medium	Negligible	Slight adverse	Moderate adverse
	Small	Negligible	Negligible	Slight adverse
	Negligible	Negligible	Negligible	Negligible

Overall effects predicted as greater than 'slight adverse' are considered significant. However, in practice, this does not translate to unacceptability of the developmental proposal or refusal of planning applications; rather it is used in the detailed considerations of the consequences which would inform adequate scoping to ensure the incorporation of appropriate mitigation measures in order to make the proposed facility achieve the intended environmental, social and economic benefits (Bull et al., 2014). There are various tools used for odour assessment (Table 2.6). Bull et al. (2014) state that the combination of a number of assessment tools is the distinctive feature that sets odour assessment apart from the conventional air quality evaluation.

Table 2. 6: Summary of odour assessment tools (Bull et al., 2014)

Type	**Approach**	**Tool**		**Parameters estimated, usually as 98th percentile of 1-hour means**
Predictive	Qualitative	Risk-based assessments using Source-Pathway-Receptor concept		A relative risk score or descriptor (e.g. negligible, low, medium or high-risk impact)
	Semi-quantitative	Screening models, look-up tables and nomographs		Estimated concentration
	Modelling	Atmospheric dispersion modelling with ADMS, AERMOD, etc. vv using source terms that have been measured by Dynamic Dilution Olfactometry (DDO) or using literature values		Predicted concentrations (OU_E m^{-3}), usually as 98th percentile of 1-hour means
		CFD tools		Image representation of flow patterns
Observational /Empirical	Monitoring of odour in ambient air	Sensory	Sniff Tests	Odour exposure inferred from measurements of intensity, frequency, duration, offensiveness. Draft method pren264086 is currently going through CEN voting process.
			Field Olfactometry	Odour exposure inferred from measured concentration (Dilutions-to-Threshold, similar to OU_E m^{-3}), together with frequency, duration, offensiveness.
		Compound analysis	H_2S by gold-film analyser	Odour exposure inferred from measured concentration (µ m-3) and odour detection threshold, together with frequency, duration, offensiveness. However, it can be difficult to derive a relationship between concentrations of chemicals and odour thresholds
			VOCs, etc. analysis	
	Actively using the community as the 'sensor'	Odour diaries		Days (%) on which odour detected above a given intensity
		Community surveys		% annoyed or % experiencing nuisance
	Passively using the community as the 'sensor'	Complaints analysis		Frequency of complaints

This is done to eliminate the uncertainties that are associated with the impact estimates of these assessment tools. Two types of assessment tools have been identified to address the two main situations that are usually considered when evaluating odour sources i.e. existing odour sources or new odour sources. These are the predictive and observational/empirical assessment tools. It is noteworthy to mention that this research to employed the Empirical assessment tool, specifically olfactometry.

2.2.5 Major odour-causing compounds formed during biowaste treatment

The nature and concentration of odour-causing compounds emitted during biowaste treatment such as composting is dependent on factors such as the composition of the waste, the stage of composting, the temperature and aeration of the waste pile (Frederickson et al., 2013). SEPA (n.d.) identified four main sources of odour emissions from biowaste activities as follows:

- The inherent content of odorous compounds in most waste types which may not necessarily be offensive individually, but when combined can produce a smell of putrefaction such as limonene from citrus fruits or pinene from woody materials.
- The natural process of degradation in the presence of oxygen which can commence during storage of the waste. Typically, large molecules such as fats and proteins break down into smaller molecules, some of which are inherently highly odorous for example amines and fatty acids.
- Anaerobic conditions resulting from inadequate aeration, poor compost structure, wet materials and/or too much compaction can cause significant odour emissions. In conditions of oxygen depletion within the waste pile, some microorganisms (facultative) adapt their metabolism, while others (obligate anaerobes) become active, metabolising compounds other than oxygen, thus generating highly odorous and offensive compounds such as hydrogen sulphides.
- Mismanagement of batch temperature, due to either overly large batches or tall windrows that may hinder heat removal by aeration, can result in the generation and eventual release of compounds such as sulphur containing compounds and/or ammonia; compounds which are very offensive.

Based on these, the main odour-causing compounds formed during biowaste treatment are discussed in the following sections.

2.2.5.1 Ammonia

Ammonia has been recognised as an important contributor to fine particulate matter formation (Roe et al., 2004), and as one of the key compounds responsible for the emission of highly offensive odour from the exhaust of composting facilities, even with the relatively high detection and recognition thresholds of 17 ppmv and 37 ppmv, respectively (Busca and Pistarino, 2003). The workplace exposure limits (WELs) for ammonia have been set as 25 ppm for long-term exposure (based on an 8-hr Time Weighted Average) and 35 ppm for short-term exposure (based on a 15 minute reference period) (HSE, 2013). In composting sites, temperature, pH and the initial ammonium content of organic waste have been identified as key parameters affecting ammonia emissions in composting sites; this is because high temperatures and pH favour the volatilisation of ammonia by displacing NH_4^+/NH_3 equilibrium to ammonia (Pagans et al., 2006a). High temperatures within the compost pile have been reported to inhibit the nitrification process which promotes ammonia volatilisation (Grunditz and Dalhammar, 2001).

A study by Beck-Friis et al. (2001) revealed that ammonia emissions ensued when thermophilic temperatures (> 45°C) and high pH (about 9) co-existed within the compost pile, resulting in a loss of up to 33% of the initial nitrogen content. This was thought to be due to the fact that breakdown of large amounts of the easily biodegradable organic content of the waste at the initial composting stage caused emission of ammonia gas which was exponentially dependent on temperature, as was expected for free soluble ammonia.

Ammonia concentrations from the exhaust of a poultry manure composting facility has been reported to range from 123 ppm in Spring to 167 ppm in Summer (Zhao et al., 2008). Emission of NH_3 from composting animal manure has been shown to vary from 41 mg/kg to 458 mg/kg for dairy manure, from 323 mg/kg to 2840 mg/kg for swine manure, and from 15 mg/kg to 2740 mg/kg for poultry manure for aeration rates ranging from 0.2 to 1.4 L/(min kg) and C/N ratios from 19 to 42 for dairy, 13 to 43 for swine, and 19 to 56 for poultry, respectively (Matsusada et al., 2002 cited in Zhao et al., 2008). Indoor levels of ammonia of ≥ 25 ppm have been reported to cause reduced final body weights in poultry production (Reece et al., 1980), and exposure

to low concentrations of ammonia can induce irritation of the lungs and eyes in humans (Gay and Knowlton, 2009). Table 2.7 summarises the toxic effects in humans following acute exposure to ammonia by inhalation.

Table 2. 7: Summary of toxic effects following acute exposure to ammonia by inhalation (Public Health England, 2015)

Exposure		**Signs and Symptoms**
mg m^{-3}	**ppm**	
35	50	Irritation to eyes, nose and throat (2 hours' exposure)
70	100	Rapid eye and respiratory tract irritation
174	250	Tolerable by most people (30 – 60 minutes' exposure)
488	700	Immediately irritating to eyes and throat
>1,045	>1,500	Pulmonary oedema, coughing, laryngospasm
1,740 – 3,134	2,500 – 4,500	Fatal (30 minutes' exposure)
3,480 – 6,965	5,000 – 10,000	Rapidly fatal due to airway obstruction; may also cause skin damage

Where mg m^{-3} = ppm × gram molecular weight/24.45 (molar volume of air at standard temperature and pressure)

2.2.5.2 Hydrogen Sulphide

Hydrogen sulphide (CAS 7783-060-4) is a colourless, toxic and corrosive gas with a characteristic rotten-egg malodour that has a detection threshold as low as 0.5 ppb (WHO, 2003; Tsang et al., 2015; ATSDR, n.d.). Estimated concentrations of the gas naturally found in air range from 0.11 – 0.33 ppb (0.15 – 0.46 µg m^{-3}), and even lower levels (0.02 – 0.07 ppb; 0.03 – 0.1 µg m^{-3}) have been reported for more remote areas (Maine Department of Health & Human Services, 2006). H_2S and NH_3 have been identified as some of the chief odourous compounds constituting nuisance from the operation of biowaste facilities (Delgado-Rodriguez et al., 2011; Velusami et al., 2013). Lomans et al. (2002) stated that H_2S occurs naturally in gaseous emissions from volcanoes, sulphur springs, swamps and as a product of the biodegradation of organic matter, such as is found in decomposing biowaste. H_2S emissions usually result from anaerobic conditions, and are reduced by aeration (Derikx et al., 1991; Blazy et al., 2014).

In their study, Rosenfeld and Suffet (2004) reported that composting of biosolids emitted sulphur and nitrogen compounds while composting green waste released mainly volatile fatty acid, ketones, terpenes and aldehydes. In a study to correlate the chemical composition and the odour concentration of emissions from storage and composting of pig slaughterhouse sludge, Blazy et al. (2015) reported hydrogen

sulphide, trimethylamine and mercaptans as the three main odour compounds out of the 66 samples analysed in their study.

Inhalation has been identified as the main route of exposure to hydrogen sulphide (ATSDR, n.d.), and concentrations that are above the detection threshold often results in annoyance and uncomfortable physiological symptoms e.g. nausea and headache. The World Health Organisation recommends that 30-minute exposure to 5 ppb (7 $\mu g.m^{-3}$) should not be exceeded in order to avoid complaints and annoyance. The Health and Safety Executive has defined limits of 5 ppm (7 $mg.m^{-3}$) and 10 ppm (14 $mg.m^{-3}$) for both long and short term workplace exposure limits, respectively (HSE, 2013). There are various known health impacts of exposure to H_2S as summarised in Table 2.8. At concentrations exceeding 100 ppm (140 $mg.m^{-3}$) olfactory paralysis occurs and inhalation of air at 500 ppm (700 $mg.m^{-3}$) can be fatal (WHO, 2003).

Table 2. 8: Exposure and Effect Levels for Hydrogen Sulphide in Air (Government of Western Australia Department of Health, 2009)

Level in air (ppm)	Impacts and health effects
0.008	Odour threshold (with some individual variability)
>0.008	Increasing possibility of annoyance and headache, nausea, fatigue
2	Bronchial restriction in some asthmatics
4	Increased eye complaints
5 - 10	Minor metabolic effects
20	Neurological effects including memory loss and dizziness

2.2.5.3 Odorous Volatile Organic Compounds

Emission of odorous volatiles is also a key feature in biowaste facilities. Müller et al. (2004) stated that these volatile organic compounds (VOCs) result from the decomposing biowaste materials and from the microbes degrading the waste materials. Also, in their research to examine the emissions of volatiles from three composting facilities and the impact of process engineering on their dispersal, they observed that the volatile organic compounds detected belonged to ketones, furans, alcohols, sulphur-containing compounds and furans, with concentrations of representative members ranging from $10^2 – 10^6$ ng/m^3 depending on sampling location and process engineering. Rosenfeld and Suffet (2004) concluded in their research that aerobic composting of green wastes released aldehydes, alcohols, ketones, volatile fatty acids, terpenes and ammonia compounds which are responsible for compost odours, and turning of the pile releases odorous compounds.

With concerns over ground-level ozone formation, Kumar et al. (2011) carried out a study to characterise VOC releases from green waste compost piles of different ages, and detected over 100 VOCs including aliphatic alkanes, alkenes, aromatic hydrocarbons, biogenic organics, aldehydes, ketones, furans, acids, esters, ether, halogenated hydrocarbons, dimethyl disulphide (DMDS) and alcohols. They identified alcohols as the predominant VOC with a flux range of 2.6 – 13.0 $mg/m^2/min$. These compounds are similar to those identified by Fletcher et al. (2014) who also identified alcohol, aliphatic hydrocarbons, ketones and terpenes as the dominant compound groups. Delgado-Rodriguez et al. (2011) stated that terpenes are the dominant VOC emissions in facilities composting MSW.

A pilot study evaluating the influence of control parameters on VOCs evolution during MSW trimming residues composting identified amongst others seven key classes of compounds that have a high odour impact including limonene, *β*-pinene, 2-butanone, undecane, phenol, toluene and dimethyl disulphide (Delgado-Rodriguez et al., 2011); although the limonene and *β*-pinene may be due to woodchip and plant materials added to the biowaste processing operations. Also, a study involving online monitoring of VOCs production and emission during sewage sludge composting indicated a variation in the VOC production within the compost pile and VOC emission at the surface of the pile, with the former having a total mass of 1.09 g C kg DM^{-1} being 2.3 times higher than the total mass of emission (Shen et al., 2012).

2.2.5.4 Odorous Emissions

A study conducted in Montreal evaluating the impacts of odour from a landfill site and composting facility on the local residents revealed significant odour impacts within 500 metres of the yard waste composting facility (Heroux et al., 2004). Several factors are responsible for odour emissions from biowaste treatment facilities among which are the waste itself, the metabolic products of the aerobic degradation of the waste, some of the metabolic products of the anaerobic breakdown of the waste, all of which can be aggravated by the method of operation employed in these facilities (Bidlingmaier and Müsken, 2007).

In a study involving the full scale monitoring of pile composting of the organic fraction of municipal solid waste (OFMSW) to evaluate the impact of odour emissions, it was revealed that the odour concentration of 5224 OU_E/m^3 was reached

early in the process (during the first 2 – 3 weeks) from an odour concentration of 430 OU_E/m^3 recorded for the raw organic material before composting (Gutiérrez et al., 2015). This peak odour concentration was observed to coincide with the peak microbiological hydrolysis of organic matter and ammoniacal nitrogen. This view is supported by the study of Orzi et al. (2010) who reported that stabilisation of composted material reduces the impact of odour as measured by olfactometry. An earlier study by Sironi et al. (2006) indicated that the main source of odour emissions in a mechanical biological treatment plant (MBT) is the aerobic biological treatment which had an odour emission factor (OEF) equal to 1.4×10^8 $OU_E\ t^{-1}$, which was one order of magnitude higher than the those recorded for other process stages.

Blazy et al. (2015) were able to establish a linear regression between odour activity value (both highest and sum) and odour concentration, even though the highest odour activity value was a better predictor of odour concentration. Odour activity value (OAV), which is a dimensionless value, is defined as the ratio of the chemical concentration to the odour detection threshold of a single targeted compound within a sample. They, however, noted that odour activity values were a poor predictor of odour concentration at low concentrations (< 1000 $OU_E\ m^{-3}$).

2.2.6 Bioreactors for Odour Control

2.2.6.1 Introduction

Two categories of air emissions control are recognised; these include source control and secondary control (Devinny et al., 1999). The source control approach entails exploring the options of substitution, reduction and/or recycling of raw products in order to achieve emissions reduction; however, this approach may have cost and quality implications. The secondary control approach involves treating the off-gases after they have been generated. Depending on predetermined regulatory standards, several technologies have been developed, to be operated either singly or in combination, to treat off-gases; the choice(s) (Figure 2.5) of which is dependent on ecological and economic constraints such as the nature and concentration of the compound(s) emitted, the volumetric flow rate and the mode of emission of the off-gases (Devinny et al., 1999; Muñoz et al., 2015). Examples of these technologies include condensation and catalytic oxidation, incineration, regenerative and non-regenerative adsorption, absorption, membrane systems and biological treatment.

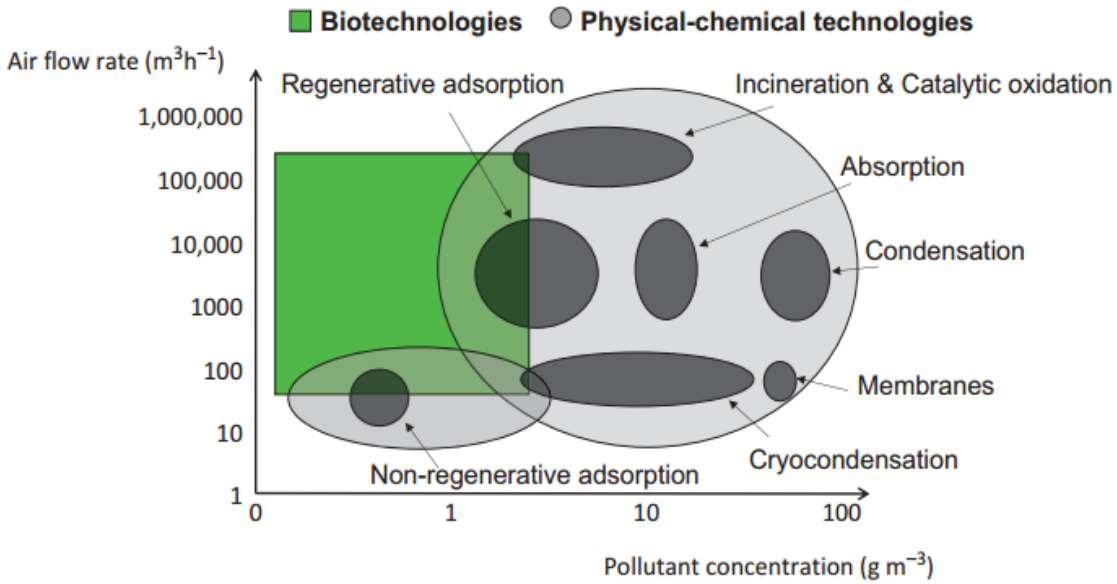


Figure 2. 5: Applicability of various air pollution control technologies based on airflow rates and concentrations to be treated (Muñoz et al., 2015)

The focus of this research is on biological treatment because of the advantage of having the capacity to treat a broad spectrum of airborne compounds (Kafle et al., 2015) without the use of chemicals which would create potentially hazardous media (e.g. spent activated carbon) requiring disposal (Singh et al., 2006). This is in addition to their low energy input requirement which means that they do not generate CO_2 from the burning of fossil fuels (Sakuma et al., 2009). Table 2.9 presents a summary of the volatile organic and inorganic pollutants that are amenable to biological air treatment.

Biological air treatment systems utilise populations of microorganisms to convert certain organic and inorganic pollutants into compounds and/or forms that are less toxic and odourless (Shareefdeen and Singh, 2008). These systems have been shown to be effective as well as economical for low concentrations of pollutants in large volumes of air (Devinny et al., 1999; Shareefdeen and Singh, 2008). They have been recommended for treating off-gases with flow rates of 100 to 500,000 m^3h^{-1} with pollutant gas concentrations in the range of <0.1 g m^{-3} to 5 g m^{-3} (Delhoménie and Heitz, 2005; Muñoz et al., 2015). The microbial population, which may be dominated by a single species or be composed of different interacting species, employ oxidative, and sometimes, reductive reactions to convert the airborne pollutants into CO_2, water vapour, and to increase their population using these pollutants as energy and carbon sources (Devinny et al., 1999).

Table 2. 9: Organic and inorganic gas pollutants commonly treated by means of biotechnologies (Muñoz et al., 2015)

Gas pollutant		Pollutant characteristics	Reference
VOCs	BTEX (benzene, toluene, ethylbenzene, xylene)	Industrial solvents commonly used in the production of petroleum derivatives	Mohammad et al. (2007)
	Acetone	Organic solvent used in cleaning products, nail polish removers and paint thinner	Chang and Lu (2003)
	Isopropyl alcohol	Solvent commonly used in the chemical, semi-conductor and opto-electronic industries	Chang and Lu (2003)
	Methanol	Industrial solvent used in the production of inks, resins, adhesives and dyes	Ramirez-Lopez et al. (2010)
	Styrene	Hazardous air pollutant emitted predominantly from industries producing polystyrene, styrene copolymers and polyester	Pérez et al. (2015)
	Isobutanol	Organic solved used as plasticiser and building block for plastics, textiles and clothing	Zhu et al. (2004)
	α-Pinene	Volatile compound typically found in waste gases from wood-related industries. Commonly used in the chemical industry and perfumery	Jin et al. (2007), Muñoz et al. (2008)
	Methane	Greenhouse gas with global warming potential 21 times higher than that of CO_2. Significant amounts of methane are emitted from waste management facilities	Rocha-Rios et al. (2009), López et al. (2014)
	Methyl tert-butyl ether	Gasoline additive for increasing the octane levels	Dupasquier et al. (2002)
	Methyl ethyl ketone	Industrial solvent used in the production of gums, resins, cellulose acetate, nitrocellulose coatings and vinyl films	Hernández et al. (2011)
VICs	H_2S	Colourless, toxic and flammable gas that has a characteristic annoying odour of rotten eggs. It is emitted from industrial activities such as petroleum refining, pulp and paper manufacturing, wastewater treatment, food processing, livestock farming and natural gas processing	Gabriel and Deshusses (2003), Ramírez et al. (2009)
	Methanetiol Dimethyl sulphide Dimethyl disulphide	Main odorous compounds found in off-gases from pulp mills and wastewater treatment plants	Ruokojärvi et al. (2001), Malhautier et al. (2015)
	Carbon disulphide	Industrial solvent widely used in the manufacture of rayon fibres, pesticides, vulcanisers and other chemicals	Hartikainen et al. (2001)
	NH_3	Emitted from intensive livestock farming and rendering facilities, sewage treatment plants and composting plants. Ammonia emissions constitute a source of severe olfactory nuisance	Malhautier et al. (2003)
	N_2O	Major GHG with a global warming potential 298 times higher than that of CO_2. It is the most important O_3- depleting substance emitted in the twenty-first century. It is emitted in waste treatment activities, nitric and adipic acid production and livestock farming	Desloover et al. (2011), López et al. (2013)

VOCs: volatile organic compounds; VICs: volatile inorganic compounds

Not all gaseous pollutants are amenable to biological treatment. For compounds to be treatable via a biological method, they must be biodegradable (which implies that they must have lower molecular weights, must be more water-soluble, and be polar) and non-toxic (Devinny et al., 1999; Adler, 2001). Alcohols, aldehydes, ketones, and some simple aromatics are some examples of the organic compounds that are readily biodegradable. Inorganic compounds such as H_2S and NH_3 are also readily biodegradable. Compounds with more complex bond structures requiring more energy for degradation by the microorganisms are not amenable for treatment by biological processes. Some of these include phenols, polyaromatic hydrocarbons, chlorinated hydrocarbons, and highly halogenated hydrocarbons (Devinny et al., 1999). Also, off-gases from some industrial activities contain chemicals that are not easily treated because the microorganisms lack the enzymes required for their metabolism. Table 2.10 shows typical industries where biological treatment of air can be applied.

Table 2. 10: Industries where Biological Air Pollution Control may be applied (Shareefdeen and Singh, 2008)

General	**Specific**	
Industrial activities including production, transport and storage	• Asphalt • Chemicals • Food, feed and beverage • Foundries • Fragrance • Leather	• Petroleum and petrochemicals • Pharmaceuticals • Pulp and paper • Textile • Viscose processing
Naturally generated odours	• Compost • Farms • Food and feed • Landfill	• Sewage • Slaughter and rendering plants • Wastewater treatment
Other trades	• Paint shops • Print shops • Soil remediation	

2.2.6.2 Types of bioreactors

There are diverse configurations of biological treatment methods that can be applied to achieve pollutant elimination from off-gases. Devinny et al. (1999) stated that each configuration employs the same basic removal mechanism, but differ in the phase of microorganisms (i.e. either fixed or suspended) and the state of the liquid (i.e. either stationary or flowing) (Table 2.11). This section provides a review of four common

configurations of bioreactors – biofilters, biotrickling filters, membrane bioreactors and bioscrubbers (Figure 2.6).

Table 2. 11: Microorganism Phase and Liquid Phase differences of some Bioreactors (Devinny et al., 1999; Shareefdeen and Singh, 2008)

Reactor Type	Microorganism Phase	Liquid Phase
Biofilter	Fixed	Stationary
Biotrickling filter	Fixed	Flowing
Bioscrubber	Suspended	Flowing
Membrane bioreactor	Fixed or suspended	Flowing

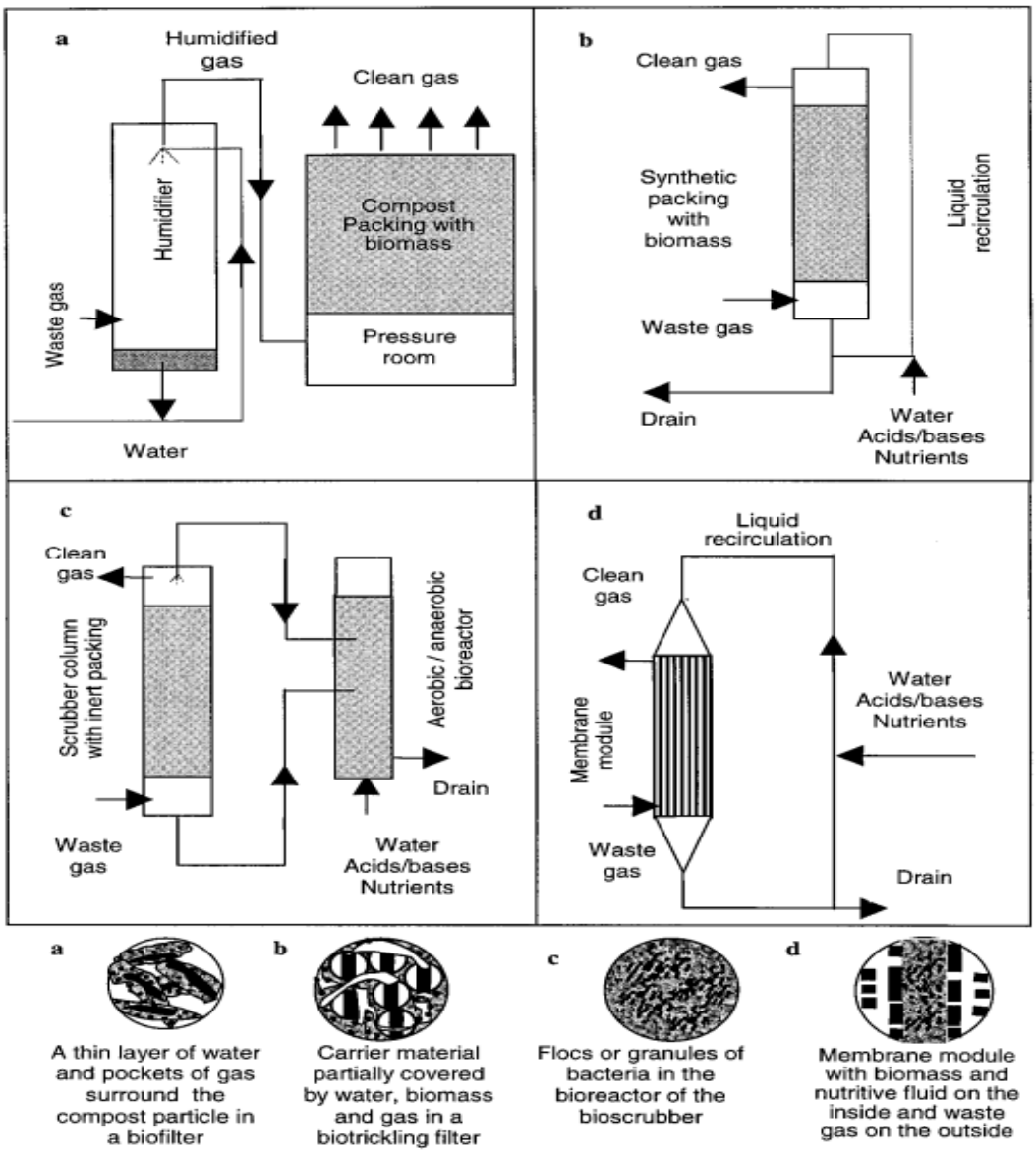


Figure 2. 6: Schematic representation of (a) biofilter, (b) biotrickling filter, (c) bioscrubber, and (d) membrane bioreactor, with close-up view of their respective microbial configurations (Waweru et al., 2005)

Biofilters

Biofilters are three phase bioreactors (gas, liquid, solid) composed of filter beds that have high porosity, high buffer capacity, high nutrient availability, and high moisture retention capacity in order to provide a suitable environment to support the growth and attachment of a mixed-culture of pollutant-degrading microorganisms (Elias et al., 2002; Dastous et al., 2005). Shareefdeen and Singh (2008) noted that the activity of the microbial population, which forms the biofilm on the surface and crevices of the media, is a function of the microbial density and environmental conditions e.g. pH, nutrient availability, moisture content, temperature and humidity.

Moisture is a key parameter which affects biofilter performance, and according to Devinny et al. (1999) this must be maintained at around 60% by mass to support the microbial population. Also, to prevent dehydration, the waste gas is generally humidified before it is passed into the biofilter bed (Waweru et al., 2005). The humid waste gas is continuously fed into the biofilter, while a nutrient solution is discontinuously added to provide nourishment to microorganisms. As the air passes through the biofilter bed, the pollutants are transferred to the biofilm that grow on the filter materials where they become degraded into carbon dioxide (CO_2), water (H_2O), mineral salts, and with the release of energy which is used with the nutrients to grow and reproduce more microbial biomass (Janni et al., 2011).

Several studies have demonstrated biofilters to be the most promising and cost-effective technology for waste gas control to meet statutory standards (Prado et al., 2009; Estrada et al., 2012). Another factor that makes this technology preferred over other air purification technologies is the environmental-friendliness as they require low energy input and do not generate secondary pollutants which would require appropriate disposal (Singh et al., 2006; Sakuma et al., 2009). Chen and Hoff (2009) and Janni et al. (2011) noted that biofilters had gained wide acceptance in animal agriculture, while Rattanapan and Ounsaneha (2011) stated that they can be used to effectively treat hydrogen sulphide (H_2S) gas generated from industrial processes such as petroleum refining, wastewater treatment, paper and pulp manufacturing, food processing, drug manufacturing, and solid waste processing.

Biotrickling filters

In a biotricking filter, the contaminated airstream is passed through a packed bed which provides surface for attachment of a contaminant-degrading microbial population and for gas-liquid contact. The packed bed is typically made of chemically inert materials e.g. plastic rings, polyurethane foam or lava rock, and because these lack nutrients (Cox and Deshusses, 1998; Waweru et al., 2005), an aqueous phase is recirculated in co- or countercurrent flow over the packed bed to provide moisture, nutrients, pH control by the addition of acids, bases and fresh medium to maintain ideal conditions for pollutant elimination, and also to allow for the elimination of products that may be inhibitory (Waweru et al., 2005; Shareefdeen and Singh, 2008; Zamir et al., 2015). Biotrickling filters are of interest because of their superiority in terms of mineralisation efficiency over other biological methods resulting from the mobile liquid phase in the reactor (Smet et al., 1998). This technology is advantageous because of the medium operating and capital cost, low pressure drop as well as the capacity to cope with acid producing compounds [Webster (1996) cited in Devinny et al. (1999)]. However, they are more complex to construct and operate, and are subject to biomass clogging.

Membrane bioreactors

In this configuration of biological waste gas treatment, a selectively permeable membrane is used as the interface between the gas and the liquid phases. The liquid phase, which is supplemented with nutrients and oxygen, is arranged such that it is in contact with one side of the membrane while the other side makes contact with the waste gas (Shareefdeen and Singh, 2008). Gaseous contaminants are transferred through the membrane to the liquid phase, where microorganisms (either suspended in the liquid phase or in the form of a biofilm attached to the membrane surface) degrade the contaminants (Reij et al., 1998; Waweru et al., 2005). The diffusion of contaminants into the liquid phase is driven by the concentration gradient. The liquid phase is kept in a reservoir which affords operational control of oxygen supply, and pH and temperature.

The basic configuration of membrane bioreactors are hollow fibres and flat sheets and membranes can be made of different materials including dense, microporous, porous or composite materials, which have diverse physical (pore size, porosity, thickness, mechanical strength) and chemical properties (selectivity, solubility) (Reij et al., 1998; Shareefdeen and Singh, 2008). The dense materials offer more selective

properties whereas the microporous materials are more permeable, even though they are prone to plugging by the microbial biomass.

Membrane bioreactors offer some advantages which include the presence of a discrete water phase which allows for optimal water supply to the biomass, and removal of the by-products of degradation; thus, preventing microbial inactivation (Kumar et al., 2008). Also, the gas and liquid flows can be controlled independent of each other without the complications of flooding, loading or foaming; however, membranes have high construction costs.

Bioscrubbers

A bioscrubber is composed of two reactors, the first of which is the absorption or scrubbing tower where pollutants in the gas phase are removed by absorption in a liquid phase (Waweru et al., 2005). This pollutant-laden liquid phase is drawn off and transfered to the second reactor, which is an activation tank where microorganisms degrade the pollutants (Potivichayanon et al., 2006); thus, regenerating the liquid phase for a co- or countercurrent re-circulation to waste gas flow within the scrubbing tower (Waweru et al., 2005). In the activation tank, the microbial population is either immobilised on packing materials (such as ceramic, plastic, metal or glass) as seen in fixed-film bioscrubber, or is suspended in the aqueous phase as seen in the suspended growth bioscrubber (Ockeleon et al., 1996; Potivichayanon et al., 2006). This configuration is particularly advantageous because of the ability to treat very high odour concentrations under severely fluctuating conditions (thereby reducing the space requirement and hence cost of construction) as well as affording better operational control over gas flow rate, pH and nutrient content.

2.2.6.3 Mechanism of Odour Biofiltration

As already stated, biological treatment of polluted air results when microorganisms within the biofilm breakdown air pollutants into carbon dioxide (CO_2), water (H_2O), mineral salts, and use the energy and nutrients to grow and reproduce more microbial biomass (Janni et al., 2011). The transformation of air pollutants to these substances with less health and environmental impacts involves a complex interplay of physical, chemical and biological processes, which must be adequately understood as they provide opportunities for improving the system, and their interruption can lead to system failure (Devinny et al., 1999; Shareefdeen and Singh, 2008). Kraakman et al. (2011) pointed out that biological gas treatment systems designed and operated based

on empirical experience without knowledge of the processes (especially the rate-limiting steps) in the system often results in problems when scaled up.

Pollutant degradation does not take place in the gaseous phase (Shareefdeen and Singh, 2008); as the various components of the polluted air (such as odorous gases, aerosols, and particulates) pass through the biofilter bed, they undergo sorption (i.e. become adsorbed on the surfaces of the media materials and/or absorbed into the biofilm on the surfaces of the media materials), from where they are degraded by microorganisms (Swanson and Loehr, 1997). Figure 2.7 summarises the processes that occur in and around the biofilm highlighting concentration gradients for oxygen and the pollutants. The pollutant first crosses the gas-biofilm interphase to meet a consortium of acclimatised microbes within the biofilm. These microorganisms derive energy from the oxidation of the pollutants as the primary substrates, or they can co-metabolise the pollutants through non-specific enzymes; thus, resulting in diffusion and uptake of nutrients (e.g. nitrogen and phosphorus) and oxygen which maintains a concentration gradient that ensures pollutant diffusion into the biofilm (Swanson and Loehr, 1997).

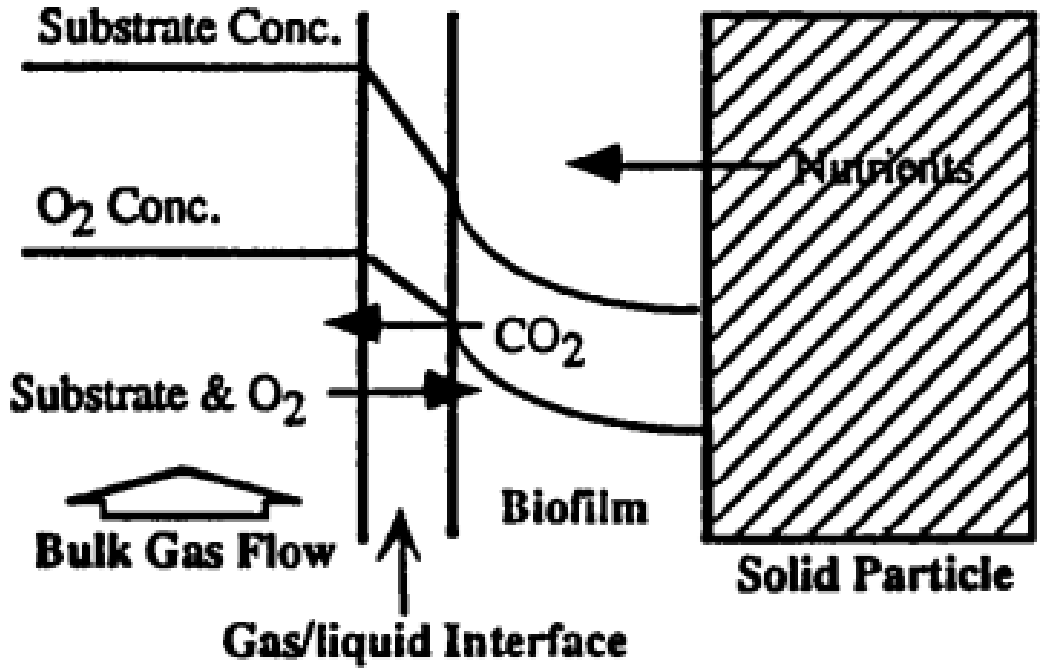


Figure 2. 7: Biofilter Pore Space Schematics (Swanson and Loehr, 1997)

2.2.6.4 Biofilter Design Parameters and Specifications

The design of biofilters is based on the nominal volume of process air that they would be expected to treat, and their effectiveness is also dependent on active monitoring and management (SEPA, n.d.). Hence, site operators are expected to assess the potential loading rates before building biofilters since the capacity for biofilters to cope with fluctuations in loads is dependent on their design characteristics, filter media and microbial composition.

Researchers have tried to understand the key design and operating parameters which enhance the performance of biofilters. One of the groups is Morgan-Sagastume and Noyola (2006) who, while studying the effect of mixing the filter media on H_2S removal in a compost biofilter, suggested that three factors determine compost biofilter performance and include (a) filter media type (including void fraction, particle size, moisture content, microbial diversity and nutrient content), (b) the operating conditions of gas flow inside the biofiltration unit (including superficial velocity, gas distribution, gas inlet pressure and temperature) and (c) the substrate concentration, solubility and biodegradability. Devinny et al. (1999) stated that medium moisture content, pH and bed temperature are the three most important factors that influence the efficiency of biofilters, pointing out that other factors influence biofilters to a lesser extent.

A study by Adler (2001) further confirmed this, noting that the design, operation, and control of biofilters are somewhat compounded by a number of sensitive and interrelated variables, including moisture content, pH, temperature, and influent air stream characteristics (such as pollutant concentrations and fluctuations in concentrations). This is because any slight change in one variable can alter the behaviour of others together with the heterogeneous nature of filter media which contributes to the complexity of modelling and controlling the behaviour of biofilters. For effective performance monitoring, SEPA (n.d.) noted that it is important to continuously monitor key performance indicators such as biofilter pressure differential, temperature, liquor pH, filter media moisture content and scrubber liquor pH/redox/flow (if used in combination with scrubbers).

Figure 2.8 presents a summary of some of the factors considered in designing a biofilter which include empty bed residence time (EBRT), specific air contaminants and their concentrations, volumetric flow rate of air to be treated, media

characteristics, biofilter size (area) constraints, moisture content/control, maintenance, and cost (Adler, 2001; Schmidt et al., 2004). These factors all interplay to either deliver the required removal efficiency or make the biofilter operation more economically viable.

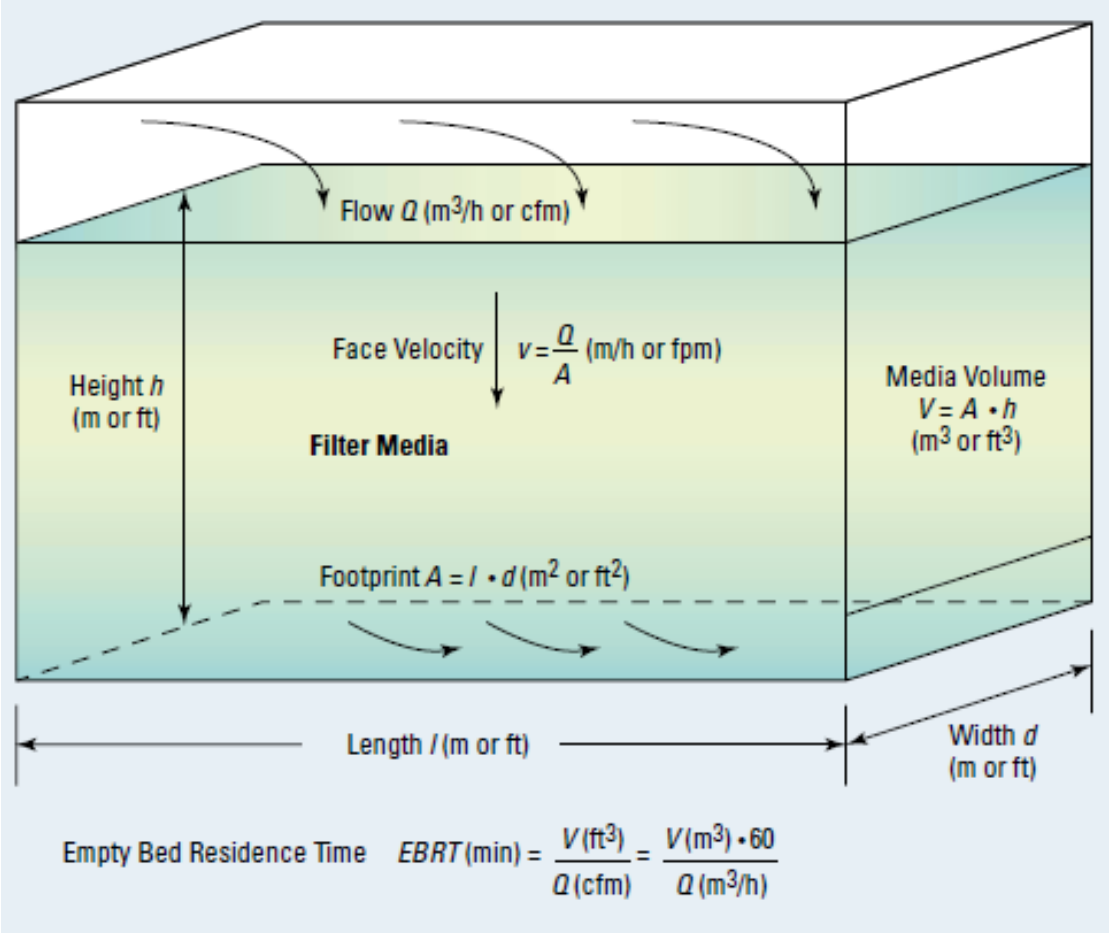


Figure 2. 8: Biofilter Design Parameters (Adler, 2001)

Microorganisms are central to the operation of biofilters and so, it is important to control these factors within limits that support optimal microbial growth to ensure biodegradation of the pollutants (Devinny et al., 1999; Fletcher et al., 2014). In addition to an understanding of these factors, it is also important to have a good characterisation of the waste air stream in order to determine the composition and to have knowledge of the actual process which generates the polluted airstream which, in turn, would inform an effective design of the biofilter (Shareefdeen and Singh, 2008).

The bulk of the information in the literature regarding the impacts of biofilter design and operation on performance have been based on odour, volatile organic compounds and N- and S- containing odour compounds with few reports regarding bioaerosols mitigation by biofilters. Fletcher et al. (2014) suggested that since the mechanisms of

removal are completely different for bioaerosols (impaction) and odour (adsorption, oxidation and biodegradation), it may be possible that the design and operating parameters that are considered critical for the removal of odour/odour-causing compounds may not necessarily be key to the removal of bioaerosols. Table 2.12 gives a summary of suggested criteria for operating a biofilter treating biowaste emissions.

Table 2. 12: Suggested operational criteria for biofilters treating biowaste emissions, based on Fletcher et al. (2014)

Operating parameter	**Typical value**
Media type	A wide variety of materials are available which are suitable for construction of biofilters. Media should be selected with reference to the following criteria: • Biologically active, but reasonably stable. • Organic matter content > 60 %. • Porous and friable with 75 – 90 % void volume. • Resistant to water logging and compaction. • Relatively low fines content to reduce gas head loss. • Relatively free of residual odour.
Media height	• 1 to 1.5 m for peat and compost biofilters. • Up to 3m for woodchip. • >2m for inorganic and synthetic media.
Surface loading	< 500 $m^3/m^2/hr$
Volumetric loading	5 – 500 $m^3/m^3/hr$
Mean effective gas residence time	40 – 100 seconds
Inlet odour concentration	500 – 350,000 OU_E/m^3
Inlet ammonia concentration	< 5 mg/m^3
Inlet hydrogen sulphide concentration	< 10 mg/m^3
Inlet air temperature	15 – 30°C
Outlet air temperature	< 50°C
Inlet air relative humidity	> 98% (Devinny et al., 1999)
Media moisture content	60% - 75%
Media pH	6 to 8.5 - Stability of pH is important. Variations should be avoided.
Air distribution	Air should be distributed uniformly through the media using a plenum chamber or distributed pipe work. Up-flow and down-flow systems can be considered.

Media Type

Media selection is a vital factor for consideration when designing and operating biofilters. According to Williams and Miller (1992), Swanson and Loehr (1997), Schmidt et al. (2004) and Rattanapan and Ounsaneha (2011), a suitable media is selected to offer (i) an optimum environment with adequate nutrient, moisture, neutral pH and unlimited supply of carbon, for the microorganisms to thrive; (ii) large surface area for maximum microbial attachment, sorption capacity, and number of reaction sites per unit volume of the media; (iii) stable compaction properties to prevent media compaction and/or gas channelling; (iv) high moisture holding capacity to ensure microorganisms remain active and to enhance higher absorption ability; (v) high enough pore space to maximise empty bed residence time and minimise pressure drop across the media depth; (vi) low bulk density to reduce media compaction; (vii) slow decomposition; and (viii) a buffer capacity towards acidification and high pollutant loads.

A wide variety of media materials have been used in biofilters, ranging from organic materials through natural inorganic materials to synthetic materials e.g. peat, compost, wood chips, bark mulch, soil, coconut fibre, lava rock, activated carbon and extruded diatomaceous earth (Devinny et al., 1999; Fulazzaky et al., 2013). It is not uncommon to find biofilters with combinations of these media materials e.g. the biological residues with inert bulking agents such as activated carbon or wood chips. A summary of some media materials is presented in Table 2.13.

There have been a lot of studies on the application of natural support media for biofiltration; however, the key problem with these media has been material deterioration and biomass accumulation (Rattanapan and Ounsaneha, 2011). Woodchips alone have been reported to be successfully used as media material for treating odour and volatile organic compounds emissions from deep pit swine facility (Chen et al., 2008b). For agricultural biofilter media, typical recommendations of 30:70 ratio of compost and wood chips mixture have been offered (Nicolai and Janni, 2001b), and a mixture of 20 – 30% compost and 70 – 80% wood chips, by weight, has been offered as ideal (Schmidt et al., 2004 cited in Chen and Hoff, 2009).

Table 2. 13: Summary of important properties of common biofilter materials (Devinny et al., 1999)

	Compost	**Peat**	**Soil**	**Activated carbon, perlite, and other inert materials**	**Synthetic material**
Indigenous microorganisms	High	Medium-low	High	None	None
Surface area	Medium	High	Low-medium	High	High
Air permeability	Medium	High	Low	Medium-high	Very high
Assimilable nutrient content	High	Medium-high	High	None	None
Pollutant sorption capacity	Medium	Medium	Medium	Low-high[a]	None to high[b], very high[a]
Lifetime	2–4 years	2–4 years	> 30 years	>5 years	>15 years
Cost	Low	Low	Very low	Medium - high[a]	Very high
General applicability	Easy, cost effective	Medium, water control problems	Easy, low-activity biofilters	Needs nutrient, may be expensive[a]	Prototype only or biotrickling filters

[a]Activated carbon; [b]Synthetics coated with activated carbon

Chen et al. (2009), in their study to examine the performance of two types of wood chips (western cedar and hardwood) as media material for the reduction of odour, H_2S and NH_3 from swine barn, reported that both materials achieved high odour removal efficiencies of 48 – 93% at a moisture content of 60% (wet basis). Other research by Nicolai and Janni (1997) showed that a media combination of compost and kidney bean straw achieved removal efficiencies of 50%, 86% and 78% for NH_3, H_2S and odour, respectively. Oyarzún et al. (2003) reported that a peat-based biofilter achieved a maximum elimination capacity of 55 g H_2S m^{-3} h^{-1} and 100% H_2S removal efficiency when fed with gas with an inlet H_2S concentration of 355 ppm at an air flow rate of 0.03 m^3 h^{-1}.

Media Moisture Content

The maintenance of optimum media moisture content is another vital consideration for biofiltration process. Media moisture has been suggested to be the most important parameter for the operation of biofilters (Sun et al., 2000), and is essential for the survival and metabolic activities of the pollutant-degrading microbial population, and also contributes to the buffering capacity of biofilters (Rattanapan and Ounsaneha, 2011). Lith et al. (1997) noted that insufficient moisture content prevents the development of a wet biofilm layer on the support material while excess moisture has the potential to reduce mass transfer of hydrophobic substances, and clog pore spaces which minimises the surface available for pollutant transfer on the biofilm. This also creates anaerobic zones, where oxygen needed for biological oxidation is exhausted (Rattanapan and Ounsaneha, 2011). Fletcher et al. (2014) pointed out that under these conditions, potentially odorous metabolic end products similar to those generated by decaying organic matter can be produced.

Biofilters achieving high elimination capacities (>50g m^3 h^{-1}) heat up, leading to moisture evaporation, thus, making it challenging to maintain an optimum moisture content even when moisture control is automated (Lith et al., 1997). Chen and Hoff (2009) suggested that the determination of the optimal moisture content range for any bioflter is dependent on the media type (Table 2.14). Drying-out of biofilter material is one of the key challenges of biofilter operation. To address this, Sakuma et al. (2009) reported that the installation of a lower irrigation system in their biofilter increased toluene elimination by a factor of 1.2 to 1.7 times higher than the control setup for a gas residence time of 13.5s. They suggested this was due to the high moisture content of that portion which supported a high density of toluene-degraders

as compared to the control biofilter. In a study to evaluate moisture effects on gas-phase biofilter ammonia removal efficiency, nitrous oxide generation and microbial communities, Yang et al. (2014) reported an improved ammonia removal efficiency when moisture level was increased from 35 to 55%, and also pointed out that a further increase to 63% only slightly improved ammonia removal.

Table 2. 14: Recommended Moisture content ranges for different media types

Media type	Moisture content range	Reference
Compost-based	55 – 55%	Goldstein (1999)
Chaff of pine and perlite	60 – 80%	Chang et al. (2004)
Mixture of compost and wood chips	35 – 65%	Nicolai and Lefers (2006b)
Wood chips	40 – 60%	Chen et al. (2008b)
Wood chips	>63%	Sheridan et al. (2002b)

Empty Bed Residence Time

An efficient transfer and subsequent degradation of gaseous pollutants requires that the gas be kept within the biofilter for a sufficient time period called the empty bed residence time (EBRT), and thus, this constitutes one of the key parameters considered when designing biofilters. The EBRT is directly related to the volume of the media bed and the flow rates of the gas to be treated as shown in Equation 2.1:

$$\mathrm{EBRT} = \frac{\mathrm{V}}{\mathrm{Q}} \qquad \textbf{Eq. 2. 1}$$

Where V is the volume of the bed (m^3,ft^3, etc) and Q is the flow rate of the gas to be treated (m^3h^{-1}, scfm, etc) (Devinny et al., 1999); and dependent on the characteristics of the pollutants in the emitted gas, usually ranging from a few seconds to several minutes (Muñoz et al., 2015).

Chen and Hoff (2009) advised that higher loading rates and a lower media moisture content requires that the gas be held within the filter bed for a longer time period, and recommended 4 – 10 s as sufficient time for odour and VOCs removal from agricultural applications provided that optimum moisture content is maintained. Kafle et al. (2015) on the other hand, recommended a slightly shorter EBRT of 2 - 3 s as optimal for the successful operation of wood bark-based biofilters (BFs). Higher EBRTs of 15 – 60 s were recommended as typical for waste air treatment by Devinny et al. (1999), while Fletcher et al. (2014) suggested 40 – 100 s as appropriate for treating biowaste emissions. Chen et al. (2008a), on the contrary, reported that there

was no significant increase in removal efficiencies (REs) of ammonia, hydrogen sulphide and odour when the EBRT was increased from 3.7 to 5.5 s.

Temperature

This is one of the key parameters affecting bioflter operation as it determines the species composition of the microbial community and their growth rates (Pagans et al., 2006b). It has been suggested that the temperature of the system depends on the temperature of the inlet gas stream and the exothermic biological reactions within the bed (Chung et al., 1998). There has been variation in the recommended optimal temperature for an efficient biofilter system. A review by Chen and Hoff (2009) suggested a range of 20 – 40°C, with 35°C as optimum for biofilter operations. Clark et al. (2004) tested the impact of biofilter operating temperature (15°C, 22.5°C and 30°C) and supplemental nutrients on the performance of two pilot scale bioflters for treating the exhaust air from a swine facility; and suggested that higher operating temperatures enhanced the establishment of pollutant-degrading microorganisms even though there was no significant difference in the overall odour removal associated with the tested treatment temperatures ($p = 0.05$).

Mann et al. (2002), investigating open biofilters used for treating the odour from a swine barn during sub-zero ambient temperatures, observed that odour removal efficiencies were inconsistent ranging from 56 – 94%. They could not ascertain whether the inconsistency was due to inadequate residence time, biofilter management problems or bed temperature. However, they suggested that uninsulated biofilters can be effective even at ambient temperatures < -20°C. Others have suggested that biofilters can be effective at temperatures >10°C (Yang and Allen, 1994; Krishnayya et al., 1999).

Media Depth

Media depth, together with air flow rate, affects pressure drop and therefore, the removal efficiency (Chen and Hoff, 2009). High media depths tend to achieve higher potential removal efficiencies but with a maximum value. However, high media depths may also result in high pressure drops across the filter bed which would cost more in terms of fan requirements to drive the flow of air. Devinny et al. (1999) recommended depths of 1 – 1.5 m for treating waste gas emissions; however, much lower values (0.3 – 1 m) have been successfully employed for on-site applications (Chen and Hoff, 2009).

Hong and Park (2004), in a study investigating the influence of woodchip biofilter properties and media depth on the control of ammonia emissions during composting of dairy manure mixed with bulking agent, revealed that media depth impacted on ammonia control and stated that a 40 – 60 cm media depth was critical to maintaining ammonia concentrations of 50 ppm. Nicolai and Janni (1999) reported that reducing the media depth below 0.15 m to achieve a reduction in gas residence time led to a reduction in the removal efficiencies for odour and H_2S below 65%, and recommended a media depth range of 0.15 – 0.3 m for compost/wood chip biofilters for livestock facilities. For agricultural biofilters, Chen and Hoff (2009) have recommended a slightly higher range of 0.25 – 0.50 m as optimal for the media depth. Fletcher et al. (2014) recommended depth depending on the type of media as 1 – 1.5 m for peat and compost, up to 3 m for wood chips and greater than 2 m for inorganic and synthetic media.

Microbial Population

Biofiltration is a complex biological process which necessitates that the conditions within the biofilter be kept within narrow ranges to support the thriving of, and efficient bio-oxidation by the pollutant-degrading microorganisms (Mann et al., 2002). The complexity of the process is somewhat due to the diversity of the microbial and chemical composition of the exhausted gas, and the interactions with the indigenous microorganisms within the biofilter bed (Chen and Hoff, 2009). Omri et al. (2011) stated that the relative abundance and diversity of the microbial population within the media is influenced by key factors such as availability of the off-gas (which provides substrate and nutrients), the prevailing environmental conditions within the filter bed (such as temperature and pH) oxygen availability, empty bed residence time and concentration of pollutants. Frederickson et al. (2013) stated that temperature, media pH, moisture content and nutrient supply are four parameters that can be managed to optimise microbial decomposition of contaminants in the waste gas. As part of this study, these conditions will be varied to evaluate which levels and combinations deliver simultaneously high removal efficiencies for odour and bioaerosols.

Devinny et al. (1999) noted that bacteria and fungi are the two main microbial groups in biofilters, even though bacteria thrive more under certain conditions. Most studies have reported inoculation of the biofilter media with activated sludge to establish the growth of the microbial population needed for bio-oxidation (Van der Heyden et al.,

2015). However, this is not really necessary especially for the organic media such as wood chips, peat and compost, which have an indigenous microbial community within the material that are usually allowed some stabilisation period (Chen et al., 2009; Omri et al., 2013; Kafle et al., 2015). Devinny et al. (1999) stated that inlet gas carries aerosols and dust which in turn harbours an initial inoculum which includes thousands of microbial species – cells, spores and cysts which may thrive or die-off depending on the prevailing conditions in the biofilter ecosystem.

Assimilation, mineralisation and sulphur oxidation are three ways of H_2S bio-degradation; and any canadidate microorganism for H_2S degradation must (i) have the capacity to convert H_2S to elemental Sulphur; (ii) require minimum nutrient inputs; and (iii) easily separate the sulphur from the biomass (Rattanapan and Ounsaneha, 2011). Several species have been identified as sulphur degrading bacteria, including *Achromatium, Acidithiobacillus, Beggiatoa, Thermothrix, Thiobacillus, Thiomicrospira, Thioplaca, Thiosphaera* and *Thiothrix.*

Ammonia biofiltration involves the conversion of ammonia to nitrate by two groups of microorganisms – chemoautotrophs and aerobic microorganisms (Shahmansouri et al., 2005). Shahmansouri et al. (2005) observed that at an inlet concentration of 236 ppm, an ammonia loading of < 9.86 g $NH_3.m^{-3}$ and an empty bed residence time of 1 minute, a removal efficiency >99.9% was achieved. Within the biofilm, NH_4^+ was converted to NO_2^-; and NO_2^- was converted to NO_3^- by nitrosomonas and nitrobacteria, respectively; both NO_2^- and NO_3^- being harmless by-products.

pH

pH is another important parameter which affects biofilter performance. This is because the greatest spectrum of bacterial activities thrives at near neutral pH (7 – 8), with exceptions such as sulphur oxidising bacteria which prefer an acidic pH of 3 (Swanson and Loehr, 1997; Omri et al., 2013). As a result biofilters must be operated and carefully monitored to keep the the pH within this narrow range. Certain components of the off-gas alter the pH of the biofilters e.g. H_2S and sulphur-containing organics cause a build-up of H_2SO_4 and NH_3 and nitrogen-containing organics cause the build-up of HNO_3, both of which lowers the pH (Swanson and Loehr, 1997). Omri et al. (2013) observed a drop in removal efficiencies when the inlet H_2S concentration was increased, and they suggested this was due to media acidification as a result of sulphate and acidic product accumulation. Thus, biofilters

must have adequate buffering capacity if they are to treat off-gases containing these compounds in order to avert accumulation of acid. Ottengraf and Van Den Oever (1983) stated that addition of crushed oyster shells, marl or limestone can buffer the pH of the biofilter media.

Chen et al. (2008a) reported that the pH of the leachate from a wood chip-based biofilter treating odour emissions from a deep-pit swine finishing facility ranged between 7.2 – 7.9 for the 12 week study period. During this period, the average reduction efficiencies they recorded were 41%, 83% and 51% for NH_3, H_2S and odour, respectively, at a moisture content of 75% and EBRT of 3.7 and 5.5 s.

Nutrient supply

Nutrient availability in the media for microbial growth and survival is also another important parameter that is considered in biofiltration processes. Rattanapan and Ounsaneha (2011) stated that carbon and energy derived from pollutant degradation and nutrients such as nitrogen, phosphorus and trace elements are necessary for the microorganisms to thrive. The need for nutrient supply is dependent on the media type. Organic media such as compost and wood chips usually have sufficient mineral nutrients, negating the need for additional supply; whereas inorganic and synthetic media do not have such nutrient content and so need supplementary nutrient supply, usually in the form of solutions containing ammonium chloride (NH_4Cl), magnesium chloride ($MgCl_2$), calcium chloride ($CaCl_2$) and dipotassium hydrogen phosphate (K_2HPO_4) (Clark et al., 2004; Chen and Hoff, 2009; Rattanapan and Ounsaneha, 2011).

2.2.6.5 Biofilter Performance Evaluation

The performance of biofilters are evaluated based on their removal efficiencies (RE) and elimination capacities (EC) (Muñoz et al., 2015). Devinny et al. (1999) described removal efficiency as the proportion of the pollutant removed by the biofilter, expressed as a percentage of the inlet concentration (Equation 2.2).

$$\text{Removal efficiency} = \left(\frac{C_{Gi} - C_{Go}}{C_{Gi}}\right) \times 100 \qquad \textbf{Eq. 2. 2}$$

Where C_{Gi} = inlet concentration (ppmv, g m^{-3}); C_{Go} = outlet concentration (ppmv, g m^{-3})

Removal efficiency (RE) does not fully describe the biofilter performance due to the fact that it only reflects the specific conditions under which it is measured, but varies with pollutant concentration, airflow and biofilter size.

Elimination capacity (EC), on the other hand, is the mass of the pollutant that is degraded per unit volume of the filter material per unit time (Equation 2.3), and it is expressed in grams of pollutant per m^3 of filter material per hour (Devinny et al., 1999).

$$\text{Elimination Capacity} = \frac{(C_{Gi} - C_{Go}) \times Q}{V_f} \qquad \textbf{Eq. 2. 3}$$

Where Q = air flow rate ($m^3\ s^{-1}$); V_f = filter bed volume.

The EC gives a more direct comparison of biofilters because the volume and flow are normalised; however, it is still a function of inlet concentration.

2.3 Review on Bioaerosols emissions

2.3.1 What are bioaerosols?

Bioaerosols can be defined as particles of biological origin (such as bacteria, fungi, virus, protozoa, algae, pollen as well as biomolecules e.g. toxins) or their cell components suspended in air, which have the capacity/potential to affect living things through the mechanisms of infectivity, allergenicity, toxicity, pharmacological and/or other mechanisms (Cox and Wathes, 1995; Sykes et al., 2011). They are minute, and vary in aerodynamic diameters ranging from 0.1 microns for viruses to 100+ microns for fungal spores, occurring either as single, unattached particles or aggregates of particles (Tisch Environmental Inc., 2015). The size range makes it possible for them to pass through the nose down to the lower respiratory system without interception by hairs and specialised cells lining the airways (SEPA, n.d.).

Human exposure to bioaerosols have been associated with some health symptoms including respiratory problems, coughs and fever, and bioaerosol inhalation has been implicated in exacerbation of respiratory conditions and gastro-intestinal symptoms, especially for immunocompromised persons who face an increased risk of an infection especially from opportunistic pathogens such as *A. fumigatus* (Prasad et al., 2004; Drew et al., 2009). Current knowledge on bioaerosol emissions from biowaste facilities have focused on culturable species such as actinomycetes, *Aspergillus*

fumigatus, and *Penicillium* (Wéry, 2014); however, it is suggested that this greatly underestimates the bioaerosol composition of emissions from such facilities especially as Albrecht et al. (2007) showed that no more than 15.3% of bioaerosols from a composting facility quantified by direct count formed countable colonies following incubation on TSA-agar.

2.3.2 Bioaerosols Generation at Biowaste Treatment Facilities

Microbiological activity is central to the operation of biowaste plants such as composting facilities (Stagg et al., 2010); thus, handling of waste materials causes the release of significant quantities of microorganisms in the air. Operations such as waste delivery, shredding, turning of compost piles and compost screening generate high levels of bioaerosols (Sanchez-Monedero et al., 2005; Taha et al., 2006; Schlosser et al., 2009), and where these operations are not confined within a building, there could be potential risk of bioaerosol dispersal to locations downwind of the site of generation, a situation that can result in complaints from people living in the vicinity of such sites. In the UK, the Environment Agency (EA) is responsible for regulating waste management facilities, usually done through the granting of Permits to Operate. Part of the EA's remit is to ensure that odours and bioaerosols do not adversely impact the surrounding population (Frederickson et al., 2013), and so have included bioaerosol monitoring requirements as an environmental permit condition, and to assess the performance of abatement systems at operation in such facilities (Environment Agency, 2017). The EA gave a precautionary guidance for composting operators when applying for operating permits. This guidance stipulates that concentrations of bioaerosols (as predicted or measured directly) need to be maintained no higher than acceptable levels at 250 m from the composting site or the nearest sensitive receptor (such as a dwelling or workplace which is not part of the composting site), whichever is closer (Environment Agency, 2010). These acceptable levels have been defined as 500 cfu m^{-3}, 1000 cfu m^{-3} and 300 cfu m^{-3} for *Aspergillus fumigatus*, total bacteria and Gram-negative bacteria, respectively, as measured by the standardised monitoring protocol (i.e. the AfOR protocol later replaced in 2017 by the M9 protocol). However, an updated regulatory position statement (RPS) on monitoring bioaerosols at regulated facilities was provided by the EA in January 2018, and excluded the reporting of Gram-negative bacteria (Environment Agency, 2018b). In the UK, there are no regulatory occupational limits for bioaersols as the acceptable levels stated above are not based on dose-response relationships (Pearson et al., 2015).

However, the Control of Substances Hazardous to Health (COSHH) Regulation issued by the Health and Safety Executive (HSE) provides employers with the requirements for assessing, monitoring and controlling the exposure of employees to hazardous substances in work environments (HSE, 2013), and thus, applies to workers in waste handling facilities. In Germany, there is a regulatory occupational limit of 50 000 cfu m^{-3} for mesophilic fungi (including *A. fumigatus*) in breathable air within the workplace (BAUA, 2013 cited in Pearson et al., 2015).

Williams et al. (2013), in a study to provide evidence on bioaerosol production, dispersion and potential exposures from four different composting facilities within England, reported peak total bacteria concentrations of $>10^6$ cfu m^{-3} immediately downwind of the sites in comparisons to the < 1000 cfu m^{-3} recorded upwind. However, the concentrations were noticed to decline at locations further downwind of the sites which is in agreement with the view that bioaerosol concentrations tend to reach background levels within 250m of their point of origin (Pankhurst et al., 2011b). In another study aimed at validating the use of three microbial groups (the bacteria genera *Saccharopolyspora* and *Thermoactinomycetaceae*, and the fungus *Thermomyces*) as indicators of composting bioaerosols, it was observed that compost-turning operations led to an increase of at least two orders of magnitude in the concentration of these three groups in comparison to the concentration measured upwind of site and of at least one order of magnitude in comparison to the concentration measured in natural environments that are not impacted by industrial activity (Le Goff et al., 2011). In research investigating critical working tasks and determinants of exposure to bioaerosols and microbial VOCs in composting facilities, Persoons et al. (2010) recorded concentrations of gram-negative bacteria up to 4×10^4 cfu m^{-3} with *Pseudomonas* as the predominant genus. The total bacteria count for this study ranged from $10^2 – 10^5$ cfu m^{-3} with *Bacillus* as the predominant genus.

Park et al. (2011) , in their study to simultaneously remove bioaerosols, odours and airborne particles from a municipal composting facility using a dielectric barrier discharge, observed average concentrations of 1.1×10^4 cfu m^{-3}, 2.1×10^8 particles/m^3, 400 ppm and 450 ppm for bioaerosols, airborne particles, amines and ammonia, respectively. Isolation and identification of the bioaerosol samples showed 11 bacteria, 4 fungi, and 2 actinomycetes as follows:

- **Bacteria:** *Bacillus licheniformis, Bacillus subtilis, Brevibacterium spp., Burkholderia cepacia, Corynebacterium glucuronolyticum, Pasteurella pneumotropica, Ralstonia pickettii, Rhodococcus spp., Staphylococcus lentus, Virgibacillus pantothenticus,* and *Weeksella virosa.*
- **Fungi:** *Aspergillus fumigatus, Aspergillus niger, Pithomyces spp.,* and *Pseudallescheria boydii.*
- **Actinomycetes:** *Nocardiopsis dassonvillei,* and *Streptomyces rochei.*

Total bacteria and *Aspergillus fumigatus* concentraions of up to 25×10^4 cfu m^{-3} and 29×10^3 cfu m^{-3}, respectively, were reported on-site and at the nearest potential receptor of a small-scale composting facility in central London (Pankhurst et al., 2011a).

With regards to fungal emissions, airborne fungal concentrations of $10^6 – 10^7$ cfu m^{-3} were reported in the air of an indoor composting plant and around the loading area with varying species composition over the year (Fischer et al., 1998). In this study, *Aspergillus fumigatus* (~ 10^6 cfu m^{-3}) was predominant in the winter and spring, while Paecilomyces variotii predominated (~ 10^6 cfu m^{-3}) in the summer samples (when *A. fumigatus* concentration ranged from $10^4 – 10^5$ cfu m^{-3}); thus, suggesting that *Aspergillus fumigatus* may not be the appropriate indicator microorganism for biowaste-related fungal exposure.

Another study conducted by Fischer et al. (2000) to compare fungal species composition to a spectrum of microbial metabolites in the air within a compost facility revealed that the highest concentrations ($10^4 – 10^7$ cfu m^{-3}) were observed in the loading area in comparison with the compost hall and storage area with $10^5 – 10^6$ cfu m^{-3} and $10^3 – 10^4$ cfu m^{-3}, respectively. Some species showed preference for either the compost pile hall (*P. variabile* and *P. verruculosum*) or the loading area (*P. crustosum, P. cyclopium, P. glabrum* and *P. roqueforti*) (Table 2.15), whereas others showed seasonal variations as was observed with *A. candidus, A. fumigatus, A. nidulans, A. niger, E. nidulans* and *Rhizopus oligosporus* which had highest spore counts during spring. Persoons et al. (2010) reported concentrations ranging from $10^2 – 10^5$ cfu m^{-3} for microscopic fungi in a green waste composting unit, with *Aspergillus fumigatus* being the predominant species (with 70 – 95% of colonies) in all composting phases except shredding which also had a predominance of *Aspergillus flavus* and *Aspergillus niger*.

Aspergillus fumigatus and actinomycetes concentrations of 9.8 – 36.8 × 10^6 cfu m^{-3} and 18.9 – 36.0 × 10^6 cfu m^{-3}, respectively, were reported in a commercial composting facility in South East England (Taha et al., 2006). A mean fungal concentration of 6.8 × 10^4 cfu m^{-3} was reported at a composting plant (Sigsgaard et al., 1994), and concentrations ranging from 5 × 10^3 – 5 × 10^7 cfu m^{-3} was reported for another composting plant (Gottlich et al., 1994).

Endotoxins and β-(1-3) glucan are other components of compost-generated bioaerosols that are of concern to human health. In a study investigating endotoxin emissions and dispersal from a commercial composting plant, it was reported that peak concentrations were recorded close to the composting activities, with a secondary peak of lesser magnitude measured at 100 – 150 metres downwind of site (Deacon et al., 2009a). The concentrations measured on-site were generally below the 50 EU m^{-3}, which is the Netherland's suggested limit for occupational exposure (Liebers et al., 2006). Endotoxin concentrations above this value are known to elicit inflammatory responses in the body, while concentrations less than 50 EU m^{-3} induce minimal cytokine release resulting in slight inflammatory response; thus, supporting 50 EU m^{-3} as the cut-off concentration for potential health effects (Liebers et al., 2008). Schappler-Scheele et al. (1998) reported concentrations of endotoxin and fungi as 0.02–304 ng m^{-3} and 10^4 –10^6 cfu m^{-3}, respectively, at German compost facilities.

In a study to characterise workers' exposure to dust, endotoxin and β-(1-3) glucan during the activities in four compost facilities, Sykes et al. (2011) observed that concentrations of inhalable dust, endotoxin and β-(1-3) glucan had a geometric mean of 0.99 mg m^{-3}, 35.10 EU m^{-3} and 0.98 ng m^{-3}, respectively; and they suggested that levels of personal inhalable dust may be a valuable predictor for personal endotoxin concentrations, particularly because of the positive correlation (r = 0.783, p<0.05) between the concentrations of both components. Concentrations of β-(1-3) glucan of ≥ 10ng/m^3 have been reported to cause health problems such as headache, inflammation of the airways and fatigue, symptoms which are similar to those experienced with endotoxin exposure (Rylander, 1999; Sykes et al., 2011). A study assessing worker and community exposure to bioaerosol generated from a yard waste composting plant in northern Illinois, USA revealed that on-site concentrations of all the bioaerosols measured were higher than off-site concentrations, and concentrations of endotoxin and β-(1-3) glucan reached up to 60 EU m^{-3} (6.06 ng m^{-3}) and 14.45 ng m^{-3}, respectively (Hryhorczuk et al., 2001).

Table 2. 15: Species with the highest spore counts found either in the loading area or in the compost pile hall at different incubation temperatures (Fischer et al., 2000).

Loading area, incubation at 22°C	**Loading area, incubation at 37°C**	**Compost pile hall, incubation at 22°C**	**Compost pile hall, incubation at 37°C**
Absidia corymbifera,	*Aspergillus flavus*	*Aspergillus candidus*	*Aspergillus versicolor*
*Aspergillus fumigatus**	*A. fumigatus*	*A. eburneo cremeus*	*Paecilomyces variotii*
Cladosporium cladosporioides	*A. nidulans (sterile)*	*Paecilomyces variotii**	*P. islandicum*
Cladosporium herbarum	*Rhizopus oligosporus*	*Penicillium fellutanum*	
Doratomyces oligosporus		*P. variabile*	
Eurotium herbariorum		*P. verruculosum*	
Mycelia sterilia			
Penicillium brevicompactum			
P. clavigerum			
P. polonicum			
P. glabrum			
P. italicum			
P. janczewskii			
P. Roqueforti			
P. spinulosum			
*Rhizopus oligosporus**			
Trichoderma citrinoviride			

Species listed here were significantly more frequent ($p \leq 0.01$) in the respective part of the facility compared to other locations when averaged over the whole period of the investigation.

*Species marked with an asterisk are thermotolerant, but have also been scored at 22°C.

2.4 Odour and Bioaerosol Removal from Biofilters or Biofilters and Scrubber Combinations

As noted earlier, biofilters were designed primarily to achieve odour control from a range of industries that generate odours, and hence, studies on its optimisation have been conducted resulting in an abundance of literature regarding performance for odour control. Few studies have been carried out regarding bioaerosol removal especially because bioaerosol removal is a recent requirement by regulators. This section provides a background to bioaerosol and odour emissions from biofilters operated as stand-alone systems or in combination with scrubbers at biowaste treatment facilities; and these are drawn from a combination of laboratory, pilot and full scale studies. Even though laboratory and pilot studies may be considered unsuitable for establishing design and sizing criteria for full scale biowaste plants, especially because of their small filter bed volume (sometimes <1 m^3) (VDI3477, 2004), these studies still prove vital for investigating the basic mechanism of biofiltration and for comparative performance analysis.

2.4.1 Hydrogen Sulphide Removal

A study conducted by Roshani et al. (2005) on performance evaluation of biofiltration in the removal of hydrogen sulphide from a gas flue confirmed that biofiltration is a highly effective, low-cost and an environmentally friendly air pollution control technology. The study which involved evaluation of the technology by assessing elimination capacity, removal efficiency, effects of sulphate accumulation, gas retention time, pressure drop across the media depth, moisture content and pH, showed that biofilters achieved a mean H_2S removal of 98% with a retention time of 60 s. However, there was a decline in the removal efficiency following accumulation of sulphates in the filter bed, which also resulted in increased pressure drop across the filter bed. Fletcher et al. (2014) suggested that H_2S tends not to be a significant constituent of biofilter emissions.

Chung et al. (1996) who worked on lab-scale operation optimisation of a *Thiobacillus thioparus* CH11 biofilter for hydrogen sulphide removal showed that removal efficiencies of >98% were achieved for the lab scale study with an optimal retention time of 28 s. A subsequent study carried out to access the removal characteristics of H_2S by a *Thiobacillus novellus* CH3 biofilter in autotrophic and mixotrophic environments showed that 99.5% of the inlet concentration of H_2S was eliminated in

mixotrophic conditions in contrast to autotrophic environments which showed a 97.5% elimination (Chung et al., 1997).

Kim et al. (2002) investigated the long-term operation of biofilters to examine the simultaneous removal of H_2S and NH_3. Two biofilters were used, one packed with wood chips and the other packed with granular activated carbon (GAC), and sprayed with a mixture of *Thiobacillus thioparus* (to aid oxidation of sulphur) and activated sludge (to provide nitrifying bacteria). They reported that before deactivation, the biofilters achieved removal efficiencies of 99.9% and 92% for H_2S and NH_3, respectively. However, these efficiencies declined following deactivation as elemental sulphur and ammonium sulphate accumulated on the packing materials resulting in removal efficiencies of 75% and 30 – 50% for H_2S and NH_3, respectively.

Elias et al. (2002), working on evaluating the performance of biofilter packing material for biodegradation of H_2S, used pig manure and sawdust as the packing material and reported that a H_2S removal efficiency of >90% was achieved with a loading rate of 45 g m^{-1} h^{-1}. They also noted that superficial gas velocity was a determining factor especially as removal efficiency decreased below 90% when the velocity of the gas was increased from 100 to 200 m h^{-1}.

Morgan-Sagastume and Noyola (2006) recommended bed mixing operation to ensure homogeneity of compost media. In their study which examined the effect of mixing the filter media on hydrogen removal, they reported that H_2S removal declined over time from 100% to 90%, but mixing the media marinated the removal capacity close to 100%.

A pilot study conducted by Omri et al. (2013) to evaluate biofilter performance for the control of H_2S from wastewater odour reported a 99% removal efficiency at an empty bed residence time of 60 s for inlet concentrations between 200 and 1300 mg H_2S/m^3. Kafle et al. (2015) evaluated the capacity of wood bark-based down-flow biofilters for mitigation of odour, ammonia and hydrogen sulphide emissions from confined swine nursery barns. They reported removal efficiencies of 95.8 – 100%, 95.2 – 97.9% and 73.5 – 76.9% for H_2S, NH_3 and odour with empty bed residence time of 1.6 – 3.1 s and a media moisture content range of 64 – 65%.

2.4.2 Ammonia Removal

Van der Heyden et al. (2015), in a review on mitigating emissions from pig and poultry housing facilities using air scrubbers and biofilters, observed that biofilters may not be suitable for direct treatment of exhaust air particularly because of their low removal efficiency for ammonia (15 – 72%), and sometimes may be net emitters of ammonia due to waste gas leakages through the filter bed or ammonia stripping. Melse and Hol (2012) stated that the low removal efficiencies are caused by such factors as poor functioning of the humidification system which allows air to pass untreated into the biofiltration system and further increasing microbial inhibition in the filter material through nitrite and ammonium accumulation.

Pagans et al. (2005) conducted a laboratory scale study to evaluate the biofiltration of ammonia from the exhaust gas of a composting plant treating the source-separated organic fraction of municipal solid wastes, animal by-products and digested wastewater sludge. They reported high ammonia removal efficiencies (up to 95.9% for a loading rate of 67,100 mg NH_3 m^{-3} h^{-1}) from the beginning of the experiment which was attributed to the high ammonia sorption (adsorption and absorption) capacity of the mature compost-based filter media. However, they observed that high NH_3 concentrations >2000 mg NH_3 m^{-3} led to a reduction in the removal efficiency. Some studies have reported that at an average loading rate of 10,180 mg $NH_3.m^{-3}$ h^{-1} and an empty bed residence time of 16 s, removal efficiency of 98% have been achieved in the biofiltration of exhaust gases from composting facilities (Park et al., 2002; Chung et al., 2003). Fletcher et al. (2014) suggested that because of the relatively high odour threshold of ammonia, it was unlikely that ammonia had a significant contribution to odour emissions from biofilters.

2.4.3 Volatile Organic Compounds (VOCs) Removal

Fletcher et al. (2014) pointed out that VOC removal efficiencies and emission concentrations were limited by such factors as concentration of air flowing through the biofilter and concentration of the VOC generated within the filter bed, and that varied from site to site which is evident in the wide range of removal efficiencies observed for many studies involving VOCs detected above their detection threshold (Table 2.16).

Table 2. 16: Indicative performance of biofilters by compound group (Fletcher et al., 2014)

Target parameter	**Inlet concentration (ug/m^3) Min – Max (Mean)**	**Outlet concentration (ug/m^3) Min – Max (Mean)**	**Removal (%) Min – Max (Mean)**
Aromatic hydrocarbons	91 – 10275 (3433)	57 – 5066 (1776)	34 - 96 (57)
Cyclic hydrocarbons	74 – 7415 (2306)	0 – 4292 (1395)	14 - 100 (53)
Aliphatic hydrocarbons	28 – 24776 (6754)	19 – 9587 (3011)	-136 – 97 (48)
Alcohols	141 – 164242 (30565)	0 – 13957 (2130)	62 - 100 (95)
Esters	0 – 10063 (2983)	0 – 2235 (233)	61 - 100 (96)
Ketones	412 – 34473 (13853)	0 – 11387 (1514)	39 - 100 (90)
Aldehydes	129 – 3672 (1476)	0 – 647 (186)	-6 – 100 (74)
Chlorinated compounds	23-3284 (910)	0 – 2988 (565)	-1481 – 100 (-93)
Organic S-compounds	220-2986 (1591)	0 – 2721 (800)	8 – 85 (50)
Furans	40-2104 (809)	0 – 1471 (337)	13 - 100 (70)
Ethers	0-356 (138)	0 – 494 (77)	-859 - 100 (-23)
Terpenes	524-50178 (12852)	79 – 8960 (3174)	25 - 99 (70)
Organic N-compounds	72-921 (291)	0 – 56 (9)	-100 – 100 (29)
Organic acids	26-16882 (2164)	0 – 700 (71)	-100 – 100 (77)

Pagans et al. (2006b) studied the emission of volatile organic compounds produced during the laboratory scale composting of different organic wastes, including the source-separated organic fraction of municipal solid wastes (OFMSW), raw sludge, anaerobically digested wastewater sludge and animal by-products, and reported removal efficiencies up to 97% depending on waste type (Table 2.17). Also worth mentioning is the fact that the biofilters emitted an estimated 50 mg C/m^3 of VOCs.

Colón et al. (2009), during full scale monitoring of a composting facility treating 14500 tons per year of organic solid wastes in Barcelona, reported that biofilters using wood chips as filter material achieved removal efficiencies of 70% for VOCs and almost 90% for ammonia, immediately after media replacement. Surprisingly, some biofilters have been shown to emit a number of VOCs that were not components of the inlet gas (Fletcher et al., 2014). This is thought to be due to factors such as incomplete oxidation of the pollutants, the presence of anaerobic zones in the filter media and/or emissions from the biofilter media.

Table 2. 17: VOC removal before and after biofiltration of exhaust gases of a lab scale composting plant (Pagans et al., 2006b)

Organic Waste Type	**VOC concentration before biofiltration (mg C m^{-3})**	**VOC concentration after biofiltration (mg C m^{-3})**
OFMSW (5:1 mix with bulking agent)	50 - 695	55 - 295
OFMSW (1:1 mix with bulking agent)	13 - 190	12 - 145
Raw sludge	200 - 965	55 - 270
Anaerobically digested wastewater sludge	43 - 2900	42 - 855
Animal by-product	50 - 465	55 - 315

A comparative assessment of a biofilter, a biotrickling filter and a hollow fiber membrane bioreactor for the treatment of odour emissions from wastewater treatment plants by Lebrero et al. (2014) evaluated the performance of these units to control trace level concentrations (0.75 – 4.9 mg/m^3) of methyl-mercaptans, toluene, alpha-pinene and hexane at EBRT of between 4 and 84 s. They reported that the biofilter was capable of achieving high removal efficiencies (> 90%) at empty bed residence time ≥ 8 s for all the compounds. The biotrickling filter achieved complete removal of methyl-mercaptan, toluene and alpha-pinene at empty bed residence time of ≥ 4 s, and 88% removal for hexane; whereas at all the tested empty bed residence time, the hollow fiber membrane bioreactor had complete removal of methyl-mercaptan and toluene, low hexane removal (38.3%) and unstable removal performance for alpha-pinene (attributed to biomass accumulation).

2.4.4 Odour Removal

Schlegelmilch et al. (2005b) carried out a bench-scale study to assess odour control at biowaste composting facilities over a 7-week period using a bioscubber/biofilter combination (Figure 2.9). They reported that maximum odour concentrations were reached in the first 2 – 3 weeks of the composting process, after which concentrations slowly declined to <1000 OU_E m^{-3}. They also showed a progressive decrease in odour concentration from the inlet of the bioscrubber to the inlet of the biofilter and outlet of the biofilter as evident within the first 20 days of the study. Also, in a study conducted by Yuwono et al. (2003) to assess the implementation of a quartz microbalance (QMB) sensor array-based instrument and olfactometer for monitoring the performance of an industrial-scale biofilter (19.0m × 6.8m × 1.5m), it was reported

that a 92% reduction in odour intensity was achieved with a loading rate of 9.5 m^3 m^{-2} h^{-1}.

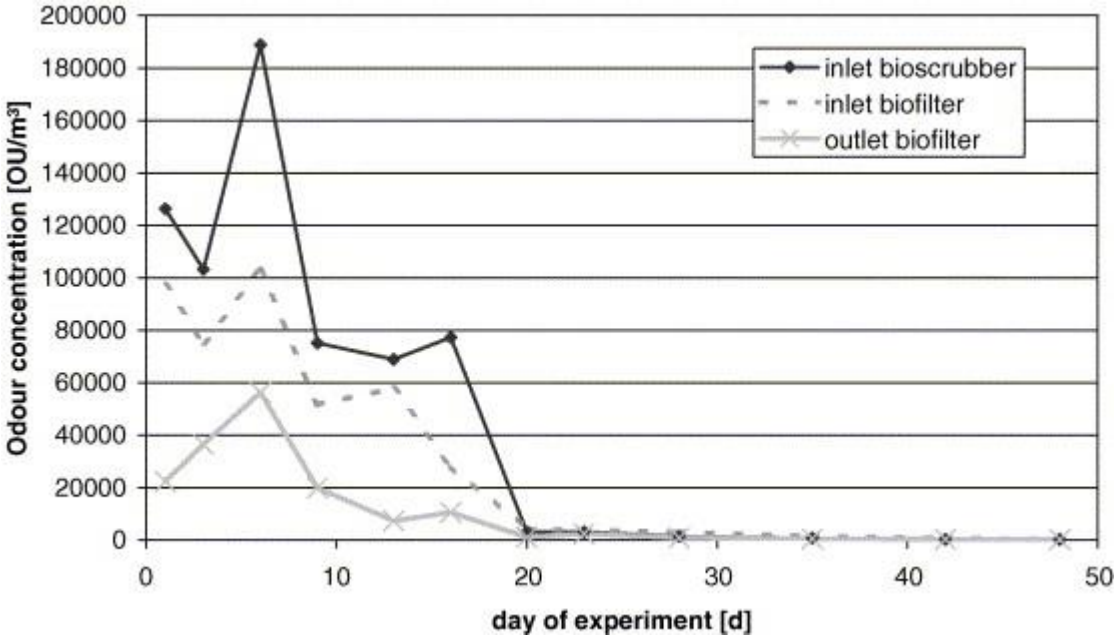


Figure 2. 9: Odour concentration at selected sampling points during a composting cycle over a period of 7 weeks (Schlegelmilch et al., 2005b)

Fletcher et al. (2014) reported an indicative performance (removal efficiencies) of 64 – 98% for biofilters with inlet odour concentrations in the range of 4856 – 145311 OU_E m^{-3} achieving outlet concentrations in the range of 212 – 5516 OU_E m^{-3}. This wide range of outlet concentrations is thought to be due to factors such as variations in the treatment capacity of biofilters for certain odorous volatiles in the waste gas, in-situ biogenic generation of odorous volatiles, and contributions from odorous volatile organics indigenous to the filter material. They also suggested 5000 OU_E m^{-3} as the upper limit that can be achieved by biofilters, and which can be used for design and regulatory purposes to guide system selection for odour abatement.

2.4.5 Bioaerosols Removal

Ottengraf and Konings (1991), in their mathematical model, suggested that biofilters may serve as systems that both capture and emit bioaerosols. The capture mechanisms, which include inertial deposition, diffusional or Brownian deposition and flow line interception, effect particle impingement on the solid bed packing material as the bioaerosol-laden gas sweeps through the filter. They also observed that at low gas velocity, the concentration of bioaerosols discharged at the outlet was more than the inlet concentrations, suggesting that biofilters are net emitters. This is thought to be due to shear stress at the gas-liquid interface which causes microorganisms to

be released from the biofilm, transported in the gas phase and eventually discharged at the outlet. However, it was observed that increasing the gas velocity resulted in a decreased outlet concentration (Ottengraf and Konings, 1991). Martens et al. (2001) suggested that the total bioaerosol emissions of biofilters is a summation of the microbial particles not deposited from the waste air and those blown off from the biofilm by the through-flowing airstream. It becomes necessary to investigate the possibility of enhancing/improving the design and operation of the system (i.e. single stage or two-stage) to achieve the capture of either the resident microorganisms that may be blown off and/or the ones that do not get impinged in the filter material.

In a study to determine the potential for the reduction of specific microbial bioaerosol, odour and ammonia from a pig facility by biofilters, Martens et al. (2001) suggested that bioflters which were excellent odour abatement systems emitted slightly more bioaerosols particles. However, they could not establish any relationships between the removal efficiencies of the odour/ammonia and bioaerosols for the five filter materials (i.e. biochips, coconut-peat, wood-bark, pellets & bark, and compost) tested. On the contrary, Tymczyna et al. (2011) observed high removal efficiencies with certain media types compared to others in their study. They noted that the use of straw as a filter media prevented the blowing out of the bacteria in the discharged air. This makes it imperative to investigate whether this is also obtainable for biowaste treatment facilities such as in-vessel composting systems employing biofilters for odour and bioaerosol control.

There is also some discrepancy in the information regarding the species composition of both the inlet and outlet air of biofilters. For instance, Martens et al. (2001) noted that biofilters emitted their own populations of microorganisms, whereas Ho et al. (2008) reported a high similarity (approximately 95%) between the inlet and outlet distribution of bioaerosol species, suggesting that the biofilter showed no selectivity in the removal of bioaerosols. In a study to assess the effect of a bioscrubber on bioaerosol emissions from a duck house, it was reported that species composition in the air within the duck house and in the purified exhaust air were different with the Enterobacteriaceae and Pseudomonadaceae being the dominant families, respectively (Scharf et al., 2004).

Sanchez-Monedero et al. (2003), in their study involving the monitoring of full scale biofilters at seven composting plants, reported that the average removal efficiencies

achieved for *Aspergillus fumigatus* and mesophilic bacteria were 90% and 39%, respectively, and suggested that this wide variation could be attributed to the size differences between fungal spores (which are larger) and bacteria resulting in the observed capture rates for these two groups. Fletcher et al. (2014) suggested that based on their full scale monitoring data the configuration of biofilters (open or enclosed) can favour the removal of one species over others, even though the odour abatement levels were comparable. They stated that open biofilters achieved higher removal efficiencies for *Aspergillus fumigatus* than enclosed systems. However, enclosed configurations achieved higher removal efficiencies for total bacteria than open configurations. However, these observations need further investigations.

2.5 Odour and Bioaerosol Reductions by Biofilters: Knowledge Gaps

The main summary of the literature regarding the performance of biofilters in controlling/reducing emissions from waste management facilities is that biofilters offer great potential as an abatement system for these facilities. However, there are still many gaps in the knowledge that need to be investigated if biofilters are to be designed and operated to achieve the highest levels of efficiency. These gaps have been summarised as follows:

- There is little quality information available on the bioaerosols concentration in the process air of waste management facilities. As Fletcher et al. (2014) suggested, there needs to be more sampling to establish these concentrations for a range of waste management facilities using robust, standardised sampling protocols to enhance data comparison across studies.
- Several studies have put forward a range of design and operating parameters which are critical for odour and VOCs control. In particular several authors have suggested that media characteristics such as porosity, moisture content, nutrient content, gas residence time, temperature and water retention capacity are the most important factors governing biofilter performance, although the optimum ranges quoted in the literature vary significantly from one author to another. However, there is little information available on whether these parameters are also vital for bioaerosol reduction by biofilters especially as the mechanisms of removal are different for these two contaminants. Thus, more research is required to investigate the criticality of these parameters in the

simultaneous biofiltration of odour/VOCs and bioaerosols in order to firmly define boundary conditions between normal and abnormal biofilter operations (Fletcher et al., 2014).

- Fletcher et al. (2014) noted that in order to define the operational limits for application of biofilter technology it was necessary to assess the performance of biofilters operated under 'abnormal' conditions since the biofilters covered in their study were actually well designed, operated and maintained.
- Frederickson et al. (2013) stated that studies were required to determine how effective different media types performed in the treatment of odour and/or bioaerosols, and to establish what maximum odour removal/particulate/bioaerosols removal rates were possible.
- Some studies have suggested that biofilters may be achieving negative removals of bioaerosols (Frederickson et al., 2013; Fletcher et al., 2014). Ottengraf and Konings (1991) suggested that this may due to extra contamination of emitted air from biofilter with microorganisms originating from the media bed. Thus, further studies is required to assess the potential for net emission of bioaerosols from biofilters both in terms of the overall concentration and also the individual species.
- Also, to date there is limited information on the bioaerosol particle size differences between biofilter inlet and outlet air samples. Deacon et al. (2009b) argued that information on particle size distribution of bioaerosols emitted from waste management operations is vital when assessing potential helath impacts of exposed persons, and it is also an important requirement for improved air dispersion modelling.

Chapter 3
MATERIALS AND METHODS

3.1 Introduction

This chapter describes the methods employed in achieving the objectives of this pilot study, including the design and fabrication of the pilot-scale biofilters. The study site was selected to simulate air contamination levels typical of waste treatment facilities. The methods employed have been chosen to reflect industry standards for evaluation and monitoring of bioaerosols and odours. In addition to the general methodologies presented in this chapter, each results chapter (Chapters 4 – 8) also includes detailed methodologies that are specific to the experiments presented in those chapters.

3.2 Pilot Study Site

3.2.1 Site selection and description

The pilot study was conducted at an enclosed MRF operated by Associated Waste Management (AWM) Ltd located in Leeds. The broad intention of this study was to generate original data on operational characteristics, levels of odour and bioaerosol emissions from the facility, and the performance of pilot woodchip-based biofilters fabricated to assess achievable emission control. This site was chosen because of the potential for significant odour and bioaerosol emissions. Stagg et al. (2013), in their study, stated that exposure to microorganisms (bacteria and fungi) in MRFs were considered to be of a medium level (between 10^4 – 10^5 cfu m^{-3}) and occasionally showed higher levels similar to those of animal houses at $>10^5$ cfu m^{-3}, and with identified species including *A. fumigatus* which is a known allergen. Hence, this site met the requirement of providing air contaminated with significant levels of bioaerosols required to test the control of bioaerosol emissions in this study.

AWM is a UK-based independent integrated waste management and recycling company. The Leeds main site (where this study was conducted) is a £12.5 million facility established in 2011 to offer services to people, businesses and companies all over West and North Yorkshire (Holland, 2011). The company is a material recovery facility (MRF) which adopts an integrated approach to waste management, combining processes such as waste collection, treatment and disposal methods with the objective of achieving environmental benefits, economic optimisation and societal acceptability

(Hester and Harrison, 2002). The facility handles approximately 200,000 tonnes of household waste per year from around 250,000 houses across Leeds, Bradford and Calderdale (Holland, 2011). Located on a 6 acre site approximately 4 miles south of Leeds city centre on the A62 (Plate 3.1), the facility accommodates the plant for processing waste and an office complex with a car park for staff and visitors (AWM, 2011).

Plate 3. 1: Satellite image of Associated Waste Management Ltd, Gelderd Road, Leeds showing the material recovery plant (top) and the office block (bottom) within the site (Google Maps, 2017).

3.2.2 Facility operation

The plant runs between 07:30 and 18:30 daily from Monday to Friday, and on Saturdays from 08:00 to 13:00 (AWM, 2011). On a daily basis, municipal and private trucks or vehicles carrying approximately 500 tonnes of waste enter the site through the A62 Gelderd Road entrance, and once on site, they are reversed into position for entry into the waste hall via Door 1 or Door 2 with the guidance of a banksman (Plate 3.2 i). With the doors shut to prevent escape of noise, odour and dust emissions, the vehicles tip the waste in the waste reception area (Plate 3.2 ii).

(i) (ii)

Plate 3. 2: The waste facility showing (i) vehicle reversing through Door 1 (AWM, 2011) and (ii) the waste reception area.

After waste ejection, the vehicles exit the plant when the doors open with air curtains around the aperture blowing air inside the reception area to keep the dust and odour emissions indoors. Loaders continuously pile the incoming waste against the back push wall in the reception area, while grab loaders feed the waste into the dinosaurus shredders that break down larger sized waste materials into smaller sized fragments in order to enhance material separation (Plate 3.3 i). The small sized fragments fall onto an inclined belt which conveys waste to the over-band magnet (Plate 3.3 ii) for exclusion of ferrous elements, which are then carried to a sorting area where they are manually sorted to remove unwanted materials such as plastics. The rest of the waste is shaken in a tumble dryer to differentiate the materials into two size ranges, those greater than 70 mm and those ≤ 70 mm (called fines), which are subsequently carried to air belt separators. The fines are conveyed via another over-band magnet to the eddy current separator (Plate 3.3 iii) where non-ferrous materials (e.g. copper and brass) are removed to a bay just underneath it (Plate 3.3 iv), while the rest of the fines move to a stretch deck screen that shakes the materials to remove the biodegradable content which is collected within the building to prevent external odour problems (these were initially conveyed to external by-product bays). The non-biodegradable portion is conveyed to the air belt separator for separation into heavy and light parts. A fraction of the heavy part is taken to the 1st picking station, and the rest get moved to the wind sifter for further separation into heavy and light items by the actions of gravity feed and air blowers, respectively. Heavy items get conveyed to the 2nd picking station while light items are conveyed to the 3rd picking station. The wind sifter also blows films through duct systems down to the films bay, from where they are loaded into vehicles going to recycling centres.


(i)

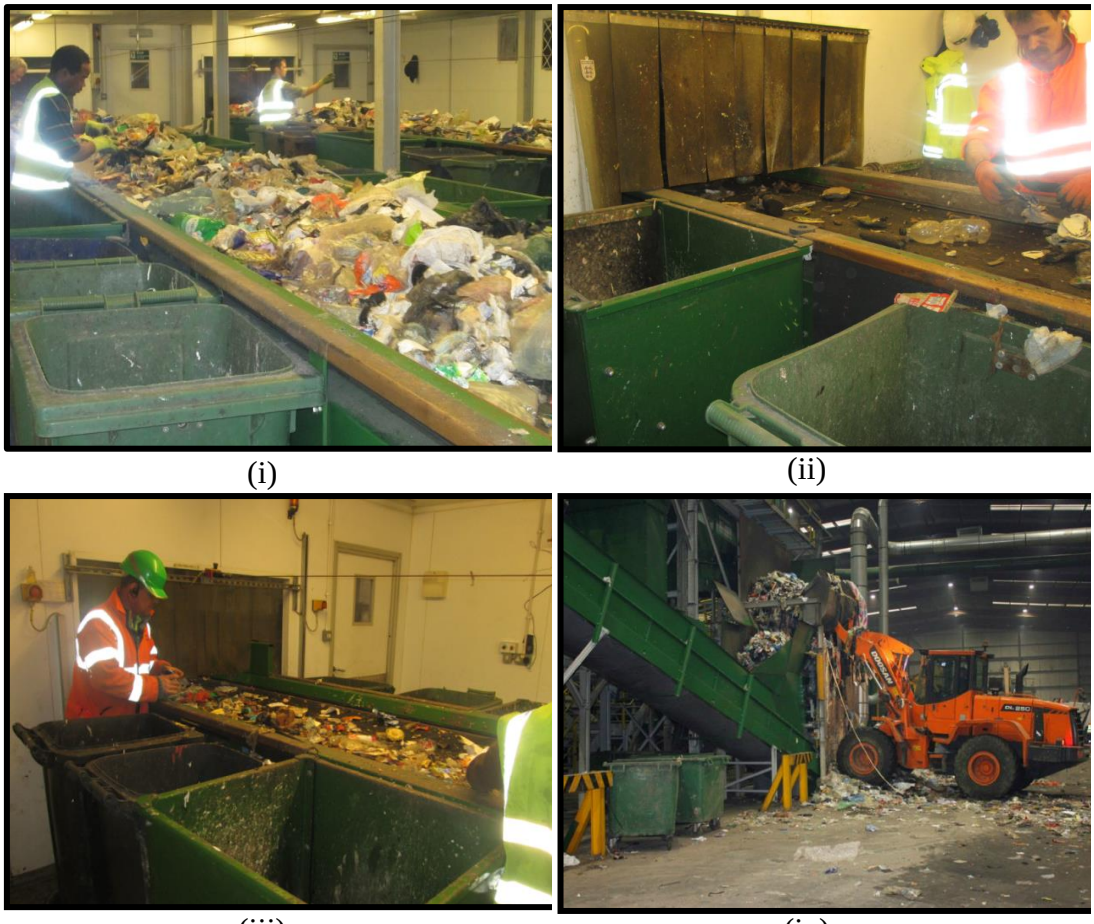

(ii)


(iii)


(iv)

Plate 3. 3: The waste facility showing (i) the grab loader feeding the shredder, (ii) the over-band magnet, (iii) waste in the eddy current separator, and (iv) the non-ferrous bay(AWM, 2011).

The three picking stations process different materials. In the first picking station (Plate 3.4 i), larger biodegradable materials (such as carrots, potatoes, and other fruits) are picked and dropped into composting bins and to a bay directly beneath the station. In the second picking station (Plate 3.4 ii), materials such as metals, aluminium cans, electrical wires and wood are sorted out and dropped into bins. The remaining wastes are conveyed to a bay located within the building for further processing. Materials such as plastics and films, aluminium cans, hard plastics, paper and cards are selected in the 3rd picking station (Plate 3.4 iii); and these materials are dropped directly into bays underneath the station and are later collected by vehicles for transport to their processing centres (AWM, 2011). The collecting vehicles access the building by reversing through Door 3 (operated by the same sensor mechanism as Doors 1 and 2) with the guidance of the banksman. Shovel machines load the materials into the vehicles after which they are directed to the weighbridge by the banksman.



(i) (ii)

(iii) (iv)

Plate 3. 4: The waste facility showing (i) Picking station 1 (ii) Picking station 2 (iii) Picking station 3 and (iv) shovel machine feeding the hopper (AWM, 2011).

Shovel machines are also used to gather materials scattered around the floor of the waste hall into their respective bays. They are also used to load residual wastes and light fraction items into the hopper (Plate 3.4 iv) that compresses the waste into bale cubes (each of approximately 1.2 tons) which are conveyed over a short distance to the baler wrapper (Plate 3.5 i) which wraps the bales in 4 layers of 30 micron plastic material. The hopper is continuously loaded with materials such that the baler continually produces Refuse Derived Fuel (RDF) bales (Plate 3.5 ii). These are distributed to power generation plants in Denmark, Holland, Norway and Sweden where they are used to derive energy in the form of heat and power. A bale generates approximately 850kWh of heat and power (AWM, 2011). The plant also has an annex, which is a separate hall that houses the finished bales before they are transported out.

(i) (ii)

Plate 3. 5: The waste facility showing (i) RDF conveyed to the bale wrapper and (ii) RDF bales for export (AWM, 2011).

3.2.3 Existing odour abatement system evaluation and odour monitoring on site

The existing odour abatement technology on the Gelderd Road facility are 11 MetalCraft Modular units containing pre-filters and impregnated activated carbon cartridges for dust and odour removal, respectively. Together, the units deliver an extract ventilation rate of 2.64 air changes per hour (surpassing the recommended industry average of 1.5 air changes per hour), treating a total air volume of 158,400 m^3 per hour (Varley, 2013). This number of units were installed following expert recommendation to allow for an increased factor of safety and increased negative pressure to contain any fugitive emissions. To assess the performance of the abatement system, odour monitoring was carried out on this facility by Gair Consulting on behalf of TerraConsult in October 2013 (Gair, 2013). Results obtained indicated that outlet concentrations (4878 – 5562 OU_E m^{-3}) at both filter outlets monitored were substantially higher than would be expected (approximately 500 OU_E m^{-3}) for an activated carbon system. Also, inlet concentrations measured (19340 OU_E m^{-3}) were not particularly high for an activated carbon system. The odour removal efficiency of 71% achieved by the filter was substantially lower than would be expected for an activated carbon system which usually have efficiencies in the range of 95 – 99%. This lower removal efficiency was thought to be due to a number of reasons including:

- high particle loading, although upstream filters are included to minimise particle loading;
- very large volume of air requiring treatment resulting in a large number of systems; and

- frequent high humidity and temperature conditions.

3.2.4 Previous bioaerosol and airborne chemical contaminants monitoring study on site

A study conducted by Ibanga (2013) to assess bioaerosol exposure and occupational health risks in the Gelderd Road facility in the summer of 2013 revealed that all operations evaluated generated significant concentrations of bioaerosols (up to 10^5 cfu m^{-3}), and these concentrations varied from operation to operation depending on the degree of agitation involved. It was further concluded that there was a high risk to the health of workers within the facility because the observed concentrations exceeded the 5000 – 10000 cfu m^{-3} limit for an 8 hour working period recommended in the literature; and this was further aggravated by the high proportion (>50%) of the respirable fraction of inhalable particles of the indicator microorganisms (*Aspergillus fumigatus* and mesophilic bacteria) measured indoors.

Nahawi (2015) conducted an assessment of occupational health risks related to airborne chemical contaminants within the facility in the summer of 2015, and concluded that site operations generated significant amounts of inhalable chemical contaminants. However, measured concentrations did not exceed the Work Exposure Limits (WELs) set out by the HSE (Appendix). The study further suggested that ethanol and limonene were the main constituents of the VOCs measured within the facility (consistently ranking 1 and 2, respectively, out of the top 10 VOCs); and argued that these resulted from emissions of waste decomposition while emissions from vehicles particularly diesel contributed to the levels of other VOCs detected within the facility.

3.3 Pilot-scale Biofilter Construction

3.3.1 Introduction, aim and objectives of construction

Pilot-scale investigation of biofilter performance allows for more controlled experimental conditions with real time polluted air, and prevents much of the uncertainties and practical difficulties associated with site operations and sampling methods in full-scale studies. Delhoménie and Heitz (2005) noted that both laboratory and pilot scale studies allow for the generation of experimental data which contribute to the understanding of the complex biofiltration mechanisms, and which allow for the development of process models that are useful for biofilter performance

extrapolation and prediction studies. Fletcher et al. (2014) also pointed out that valid results can only be obtained from pilot plants testing the actual waste gas stream to be treated. Thus, it became imperative to construct a pilot-scale biofilter test rig to assess a range of parameters identified as critical to removal of odour and bioaerosols from waste air streams. The specific objectives were:

- to design, fabricate, commission and operate a pilot-scale biofiltration test rig with the capacity for multi-level temperature monitoring, media moisture content measurement, airflow rate monitoring and leachate collection and assessment; and
- to determine the nature and characteristics of the indoor air quality in terms of bioaerosols and odour composition in order to establish the baseline concentration of inlet air to be fed to the pilot-scale biofilters.

3.3.2 Description of Pilot-scale Biofilter System

The biofilter system (Figure 3.1 and Plate 3.6) for this study was designed to meet odour treatment specifications as recommended by Fletcher et al. (2014). It was modelled after the one-stage pilot-scale biofilters in the study by Chen and Hoff (2012), and shows some of the key features of a full-scale system as described by Janni et al. (2011) and Fletcher et al. (2014). The system comprised of four vertical up-flow plastic reactors filled initially with wood chips as media. Fletcher et al. (2014) noted that both open and enclosed biofilters can achieve comparable odour abatement levels provided that the key operating parameters are ensured. For this study, the biofilters were covered between sampling days to prevent surface contamination, during which treated air was released through a 20 mm diameter exhaust provided at the top of each biofilter cover. Other authors have opined that vertical down-flow configuration is advantageous because it allows water application to the air entrance surface - the section of the biofilter reported as being prone to media drying (Kafle et al., 2015). However, the biofilters covered in the study by Fletcher et al. (2014) had the up-flow configuration and were all reported to deliver good performance. Hence, the up-flow configuration was adopted for this study as it allowed for ease of fabrication to suit the experimental site.

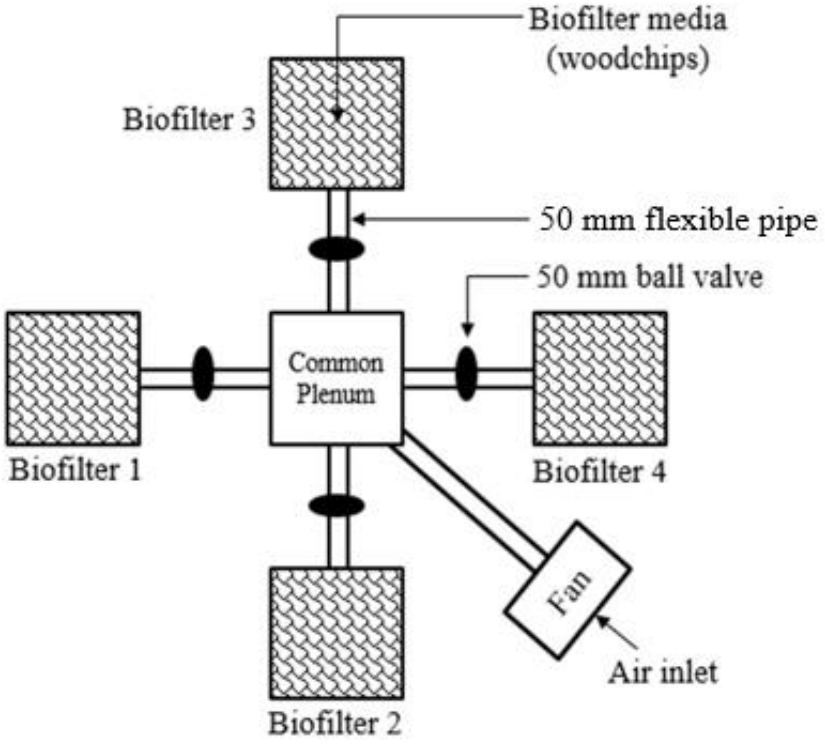


Figure 3. 1: Layout of the pilot-scale biofiltration system

Each reactor (length: 66 cm, breadth: 55 cm, depth: 99 cm) was designed with a 20 cm air space at the bottom, with an initial 50 cm biofilter media depth (giving a total biofilter media volume of 181.5 L) located above the airspace separated by a metal mesh which supports the media (Figure 3.2). All four reactors were connected to a central plenum by means of 50 mm flexible polyvinyl chloride (PVC) pipes. A centrifugal fan (Secomak 575/1 High Velocity fan) was used to pump contaminated air from the waste hall into the plenum, from where each biofilter was air-fed. Lee and Lin (2007) noted that this type of fan had the capacity to handle dirtier air streams with higher system resistance. Airflow into each biofilter was measured using an Alnor Balometer capture hood EBT731 (Plate 3.7), and regulated by means of 50 mm ball valves to obtain the range of empty bed residence times tested (9 – 109 s corresponding to airflow rates of 1210 L min^{-1} – 100 L min^{-1}, respectively). Water was supplied to the top of each biofilter by a combination of manual watering and an automatic irrigation system (Hozelock Ltd, Midpoint Park, Birmingham, Model 2756: 36419-001) connected to a peristaltic pump (Watson Marlow, model 624S) and socket timer. Irrigation was controlled by the look and feel method suggested by Janni et al. (2011) whereby moisture levels were monitored to ensure dampness across ½ to ¾

way through the media depth. Leachate from each biofilter was collected once a week throughout the study period; leachate pH was measured using a digital calibrated pH-meter.

Plate 3. 6: Fabricated pilot-scale biofilter system in the laboratory.

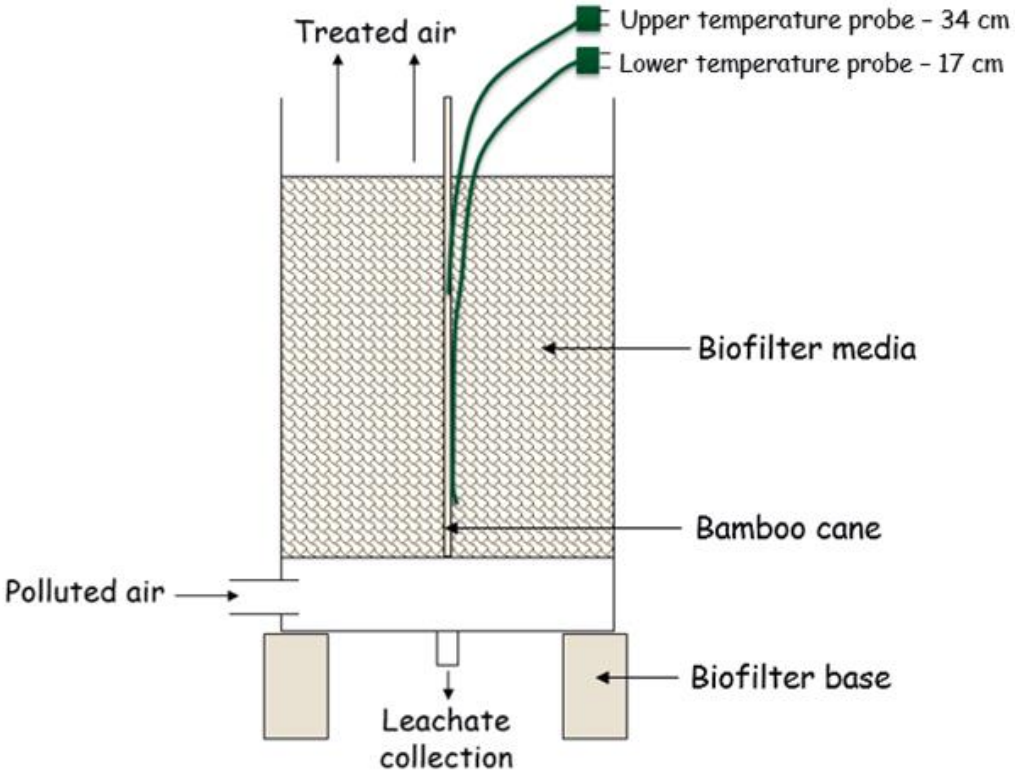


Figure 3. 2: Schematic of the pilot biofilter used in this study.

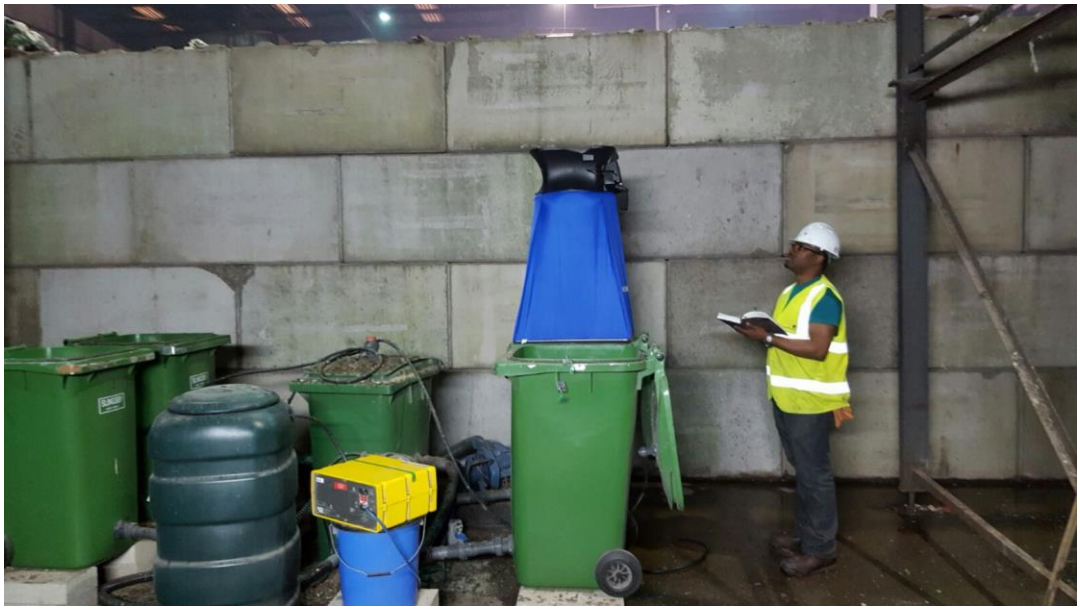


Plate 3. 7: Airflow rate measurement using an Alnor Balometer (blue cone) through the pilot-scale biofilter.

3.3.3 Media selection and characterisation

Based on the study by Fletcher et al. (2014), it was decided that woodchips be used as biofilter media for this study (except for experiments in chapter 8 which assessed the performance of various media types) because it is easily available and can be sourced locally; is relatively cost effective; and has inherent content of nutrients (Devinny et al., 1999) and naturally harbours microbial population (Hellenbrand and Reade, 1992; Tymczyna et al., 2011); thus, eliminating the need for supplementary nutrient supply and microbial inoculation, respectively. Woodchips were also selected because of their wide usage as media material in most biofilters in the UK. Devinny et al. (1999) argued that they (i) prevent bed compaction; (ii) allow for homogenous air flow through the bed and; (iii) can act as a reservoir for water which can compensate for media moisture content fluctuations resulting from poor reactor control or excess heat generated within the biofilter. Woodchips have been reported to achieve significant odour removal efficiencies (in excess of 90%), although there have been variable performance in terms of bioaerosol control (Fletcher et al., 2014).

The woodchips for this study were purchased from a local supplier (Garforth Log Supplies, Peckfield House Farm, Garforth, Leeds, UK). Preliminary laboratory tests were conducted to determine the woodchip characteristics including appropriate sizing, moisture content (MC), water holding capacity (WHC), porosity and bulk density (Table 3.1).

Table 3. 1: Characteristics of wood chips used for this study

Characteristics	Units	Values for this study	Values for Kafle et al. (2015)	
			MWB	SWB
Density	kg/m^3	225	244.3	200.8
Porosity	%	61.4	59.9	68.4
Water holding capacity	g/g dry weight	1.16	0.84	1.58
Moisture content	% (w.b.)	30	11	14

3.3.3.1 Woodchips sizing

Biofilter media size is an important factor vital for performance especially as it affects parameters such as total biofilm surface area and resistance to air flow; woodchips typically in the size range of 1 to 5 cm have been found to be ideal (Devinny et al., 1999). For this study, the woodchips (as-received) were sized by sieving using the Retsch AS200 Analytical Sieve Shaker (Plate 3.8) operated at an amplitude of 60 and a vibration height of 1.8 mm for three minutes. Sieve mesh size of 4.75 mm was used to separate out the oversize fractions (used for this study) from the undersize fractions (Plate 3.9).

Plate 3. 8: The Retsch AS200 (basic) Analytical Sieve Shaker.

Plate 3. 9: Undersize and oversize fractions of woodchips obtained with the 4.75 mm mesh size sieve.

3.3.3.2 Woodchips bulk density

Woodchips bulk density was determined according to the method of Valter Francescato et al. (2008). The procedure is as follows:

- A bucket of known volume and pair of scales were obtained.
- A representative sample of the wood chips was used to fill the bucket without compressing the woodchips.
- The sample of wood chip was weighed and their mean value was divided by the known volume (Equation 3.1).

$$\text{Bulk density} = \frac{\text{Mean weight of woodchips (kg)}}{\text{Volume of bucket (m}^3\text{)}} \qquad \textbf{Eq. 3. 1}$$

The mean weight of woodchips was 2.7 kg and the volume of bucket used was 12 litres.

3.3.3.3 Woodchips porosity (voids) measurement

Woodchips porosity (voids) was determined by the Bucket Method (Nicolai and Janni, 2001c) as follows:

- Two identical 12-litre buckets were obtained.

- One of the buckets was filled one-third full with media and dropped ten times from a height of six inches (15 cm) onto a concrete floor.
- To the same bucket, more media was added to two-thirds full and the bucket was dropped ten times from a height of six inches (15 cm) onto the concrete floor.
- More media was added to fill the same bucket to the top, and again the bucket was dropped onto the concrete floor from a height of six inches (15 cm).
- The bucket was filled with media to the top edge of the bucket.
- The second bucket was filled to the top with clean water. The water was slowly poured from the second bucket into the first bucket containing the media until the water reached the top of the media-filled bucket.
- The volume of water added to the media-filled bucket was recorded, and the procedure repeated three times.
- The Percent Void (Wood chip porosity) was determined using Equation 3.2.

$$\text{Percent voids} = \frac{\text{Volume of water added (litres)}}{12\ \text{litres}} \times 100 \qquad \textbf{Eq. 3. 2}$$

The average volume left in the bucket after pouring into the media-filled bucket was 4.63 litres.

3.3.3.4 Woodchips water holding capacity measurement

Water holding capacity (WHC) of the woodchips was determined by soaking the woodchips in water for 24 hours followed with analysis by gravimetric method which entails oven-drying the three woodchip samples for 48 hours at a temperature of 105°C (Kafle et al., 2015). The initial (before oven-drying) and final (after oven-drying) weights of the woodchip samples were measured and the water holding capacity of the woodchip (g/g dry weight) was computed using Equation 3.3.

$$\text{WHC} = \frac{\text{initial (wet)weight} - \text{final (dry)weight}}{\text{final (dry)weight}} \qquad \textbf{Eq. 3. 3}$$

Average initial (wet) weight = 114.23 g, and average final (dry) weight = 52.78 g.

3.3.3.5 Woodchips moisture content measurement

Woodchips moisture content was determined using the oven drying method which entails computing the weight loss following oven drying overnight at 105°C (Kafle et al., 2015).

Procedure:

- The oven was preheated to the point marked during calibration for an internal temperature of 105°C. The thermometer used during calibration was used to double check this temperature.
- An air-tight heatproof container was weighed to obtained initial weight, $\mathbf{W_1}$
- Woodchip samples were weighed into the air-tight heatproof container before opening to obtain $\mathbf{W_2}$. This ensured accurate weight of the samples before any material or water was lost from the samples.
- The heatproof container with the woodchip samples was put in the oven. Sample weight was logged every two hours and when the weight of the sample remained unchanged (to within 10g) for two consecutive measurements it was considered to be oven-dry, and the weight of container and oven dry woodchips, $\mathbf{W_3}$, was obtained.

Woodchips moisture content (%) (wet basis) was computed using Equation 3.4.

$$\text{Woodchips Moisture Content} = \frac{\mathbf{W_2 - W_3}}{\mathbf{W_2 - W_1}} \times 100 \qquad \textbf{Eq. 3. 4}$$

$\mathbf{W_1}$**,** Average weight of heatproof container = 137.41 g;
$\mathbf{W_2}$**,** Average weight of heatproof container with wet woodchips = 213.31 g;

$\mathbf{W_3}$**,** Average weight of heatproof container with oven-dry woodchips = 190.19 g.

This procedure was done with three samples and the average moisture content computed from the results.

3.3.4 Water supply and moisture content calibration

The maintenance of adequate media moisture content is vital to the odour control function of biofilters. For this study, water was supplied to the top of each biofilter by a combination of manual watering and an automatic irrigation system (Hozelock Ltd, Midpoint Park, Birmingham, Model 2756: 36419-001). The irrigation system was connected to a peristaltic pump (Watson Marlow, model 624S) and socket timer to control the supply volume and duration of watering to the biofilter system. Before

deployment on site, the peristaltic pump was calibrated in the laboratory to establish the water supply setting achievable for the reactors (Table 3.2). Pump calibrations showed that at 5% pump setting, no flow was observed; and at settings ≥ 40%, there was so much agitation of the pump to the extent of disconnecting the biofilter supply pipes attached to the pump. Hence, pump settings 10 – 35% were used to evaluate the target moisture content levels of 20%, 40%, 60% and 70%. The peristaltic pump was connected to a 250L water tank to accommodate the maximum daily requirement of 240L at 35% pump setting every 3 hours. Schnelle and Brown (2002) noted that water can be supplied to biofilters in two ways, including humidification of the inlet air and direct application of water to the surface of the biofilters provided that water droplet diameter is relatively small (typically maintained at less than 1 mm during winter). However, for this study manual application was also done with a 9-L garden watering can. During the experiments, irrigation was controlled by the look and feel method suggested by Janni et al. (2011) whereby moisture levels were monitored to ensure dampness across ½ to ¾ way through the media depth; this was followed by the gravimetric method as described in section 3.3.3.5. An operator was required two days a week for approximately one to three hours per day to manually supply water and for routine adjustments of the operating parameters and mechanical maintenance. The moisture content was measured weekly and the rate of water addition was adjusted by the operator until the water content was maintained at the chosen value as recommended by Devinny et al. (1999).

Table 3. 2: Water Supply setting achievable with the peristaltic pump

Peristaltic pump setting (%)	**For all four reactors**				**For each reactor**	
	Flow rate (L min^{-1})	V_{15} (L)	V_{15} every 3 hours per day (L)	V_{15} every 6 hours per day (L)	Flow rate (L min^{-1})	V_{15} (L)
5*	-	-	-	-	-	-
10	0.63	9.50	76.00	38.00	0.16	2.36
15	0.87	13.10	104.80	52.40	0.22	3.26
20	1.20	18.00	144.00	72.00	0.30	4.50
25	1.40	21.00	168.00	84.00	0.35	5.25
30	1.70	25.50	204.00	102.00	0.43	6.38
35	2.00	30.00	240.00	120.00	0.50	7.50
40*	2.30	34.50	276.00	138.00	0.58	8.63
50*	2.40	36.00	288.00	144.00	0.60	9.00

* excluded from the calibrations

V_{15} = volume supplied for 15 minutes duration

3.3.5 Media pH and Conductivity Measurement

Monitoring of the level of acidity /alkalinity of the biofilter media is essential especially since pH fluctuations have the potential to stress the resident microbial population required for pollutant degradation (Devinny et al., 1999). For this study, media pH and conductivity was measured weekly by taking a 30-g sample of the media (approximately 0.25 m from the top) from each biofilter and adding to 300 ml of distilled water. These were mixed by shaking with a Stuart Flask Shaker. After settling, the media pH and conductivity were determined by measuring the supernatant using a digital calibrated pH-meter (HI98100 Checker® Plus pH Tester supplied by HANNA Instruments) (Plate 3.10) which also had capability for conductivity measurement. This method is in agreement with the recommendation of Devinny et al. (1999) who noted the precise determination of media pH can be achieved by mixing a sample of the media with distilled water. Wani et al. (1997) argued that monitoring of biofilter media alkalinity is necessary in order to prevent any biofiltration process upsets.

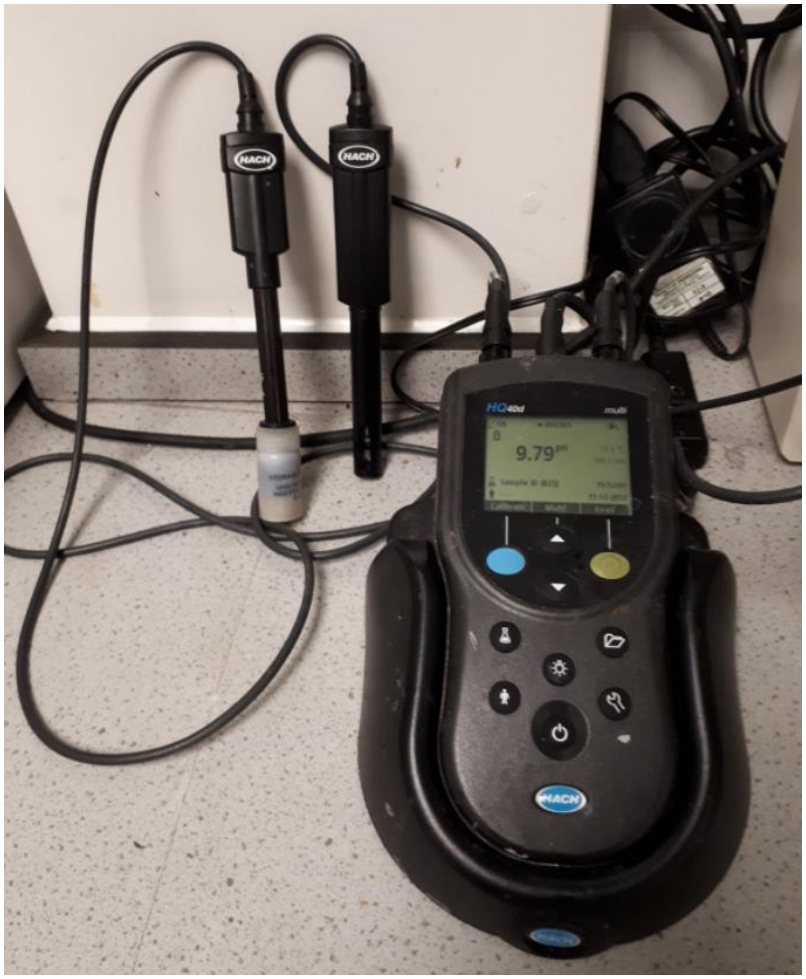


Plate 3. 10: HI98100 Checker® Plus pH Tester used for pH and Conductivity testing.

3.3.6 Media temperature and relative humidity measurement

Biofilters are dependent on the action of the rich culture of microorganisms resident in the media bed to achieve their odour control functions. These microorganisms have been reported to be most active at mesophilic temperatures of 10 – 40ºC (Schnelle and Brown, 2002; Janni et al., 2011). However, for this study the temperature was not adjusted to reflect the range suitable for microbial action. Instead, the pilot-scale biofilters were operated at the existing ambient temperatures within the facility to simulate typical indoor temperature levels obtained at waste management facilities.

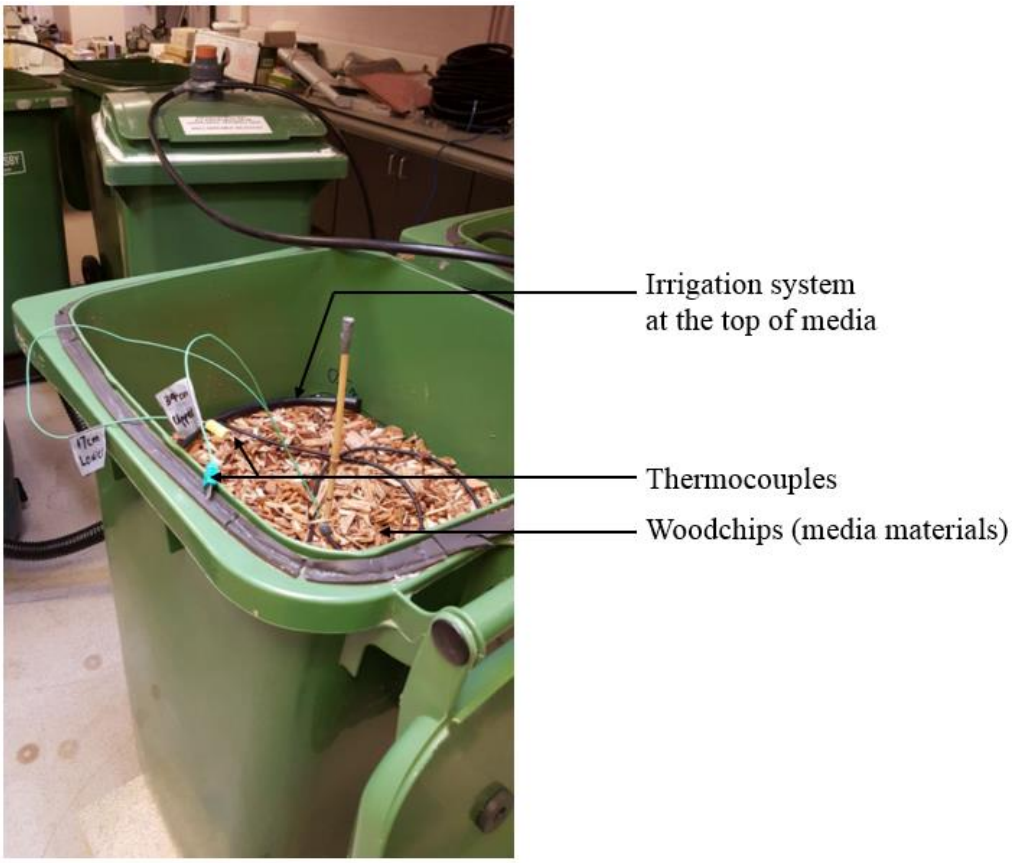


Plate 3. 11: Pilot-scale biofilter showing the thermocouples, irrigation system and woodchips.

As shown in Figure 3.2, media temperature was measured at two levels (17 cm and 34 cm from the metal mesh bases that support the media beds) within the biofilter beds using thermocouples which penetrate the media materials to the set depths. The thermocouples were calibrated in the laboratory (Plate 3.11). Temperatures were read off by connecting the thermocouples to the RS Digital Thermometer (2 Input Handheld, K Type Input No 615-8212) (Plate 3.12). Ambient temperature was also measured using the portable thermo-hygrometer (HANNA Instruments Model HI 8564) (Plate 3.13). This instrument was also used to measure relative humidity (RH)

of the inlet and outlet air at the plenum and the outlet of each of the reactors, respectively. Monitoring was done in accordance with the user's manual with the probe held in such a way to expose the head of the humidity detector directly to the air at each sampling point. For accurate reading, it is expected that the probe must be in contact with air stream of minimum velocity of 0.5 m s^{-1}. In cases where this was not possible, the probe was moved around to achieve accurate reading.

Plate 3. 12: RS Digital Thermometer.

Plate 3. 13: Hanna Instruments HI 8564

3.3.7 Airflow rate and residence time measurement and calibration

The gas EBRT is a function of the biofilter media volume and the contaminated air flow rate through the filter bed (Equation 2.1). However, this parameter tends to overestimate the actual time required for air treatment because it assumes that the whole bed volume is available to the incoming polluted air in contrast to the true residence time (τ) which is also a function of media porosity, and given by the relationship shown in Equation 3.5:

$$\tau = \frac{V_f \times \theta}{Q} \qquad \textbf{Eq. 3. 5}$$

where θ = media porosity (which is volume of void space/volume of filter material) (Devinny et al., 1999). For this research, the EBRT was adopted due to the simplicity of measurement and because it is widely used as one of the design criteria for sizing biofilters by engineers in the field of odour control technologies (James, 2015). Even though the difference (being the porosity factor) between both forms of residence time can be substantial, the effects on biofilter performance are parallel (Devinny et al., 1999).

A Secomak 575/1 High Velocity centrifugal fan was used to move contaminated air from the waste hall into the biofilter system through the common plenum. The manufacturer claims that their high velocity fans are designed to provide low volume, high pressure air flow up to at 900m³/hr (Secomak, 2017). Air flow measurement through each pilot-scale biofilter was achieved using the Alnor Balometer EBT731. The balometer has a capture hood (hood size: 610 mm × 610 mm) as the pressure tool for measuring flow from grilles and diffusers and displays the results either as l/s, m^3/hr, m^3/s or CFM depending on setting. The balometer was placed over the top of each biofilter outlet (Plate 3.7) and operated in accordance with the direction in the user's manual to take readings. 50 mm ball valves were installed on the flexible pipe on the inlet to each pilot-scale biofilter unit; these, were used to regulate the flow rates (Tables 3.3 and 3.4) to each biofilter in order to obtain the range of EBRT (9 – 109 s corresponding to airflow rates of 1210 L min^{-1} – 100 L min^{-1}, respectively) assessed in the study.

Table 3. 3: Achievable average EBRTs with the different valve settings using Secomak 575/1 Fan

Valve setting	Achievable average EBRT (s)
All 4 fully open	11.4
All 4 half open	16.4
All 4 quarter open	30.6
2 half open, 2 fully open	19.3 (half), 8.9 (full)
2 quarter open, 2 fully open	44.4 (quarter), 7.4 (full)
2 half open, 2 quarter open	9.4 (half), 46.7 (quarter)

Table 3. 4: Comparison of some test EBRTs for this study and those from Chen et al. (2008b)

Present study				Study by Chen et al. (2008b)			
EBRT (s)	**Volume (L)**	**Flow rate ($L\ s^{-1}$)**	**Flow rate ($L\ min^{-1}$)**	**EBRT (s)**	**Volume (L)**	**Flow rate ($L\ s^{-1}$)**	**Flow rate ($L\ min^{-1}$)**
9.3	181.5	19.52	1170.97	3.3	125.6	38.06	2265
10.9	181.5	16.65	999.08	5.3	125.6	23.70	1410
12.3	181.5	14.76	885.37	7.3	125.6	17.21	1025
13.3	181.5	13.65	818.80				
13.4	181.5	13.54	812.69				
22.1	181.5	8.21	492.76				
26.4	181.5	6.88	412.50				
46.8	181.5	3.88	232.69				
59.9	181.5	3.03	181.80				

3.4 Pilot-scale Biofilter Operation

After calibration in the laboratory, the biofilter system was taken on site and operated for a total of 15 months from May 2016 to July 2017. Before sampling commenced, the media in each reactor was allowed to stabilise for four weeks following the recommendations in the literature (Cabrol et al., 2012; Ralebitso-Senior et al., 2012). Stabilisation was achieved by passing the waste gas through each biofilter unit maintained at a moisture content of approximately 60% (wet basis), media depth of 0.5 m and empty bed residence time of 11.4 s in all four biofilters. The stabilisation or acclimatisation period is defined as the time necessary for pollutant-degrading microorganisms to reach high and stable biodegradation capacity for higher performance, usually ranging from 10 days to more than 10 weeks (Cabrol et al., 2012; Ralebitso-Senior et al., 2012). Muñoz et al. (2015) stated that this is a period that ensures microbial community specialisation and competition exclusion, and allows for microorganisms to be better adapted for the potential toxic effects of the pollutants.

All research objectives (except experiments evaluating net bioaerosol emitting potential of biofilters) required that the pilot-scale biofilter system be set up inside the facility just behind the back-push wall (Plate 3.14) in the waste reception area so that it was as close to the waste materials as possible (without exposing the researcher to the hazards and risks associated with waste tipping and loading operations), therefore ensuring a constant supply of odorous air contaminated with bioaerosols which was essential for this study. Another reason for selecting an indoor point for setting up the biofilter was so that any potential biofilter emissions be contained especially as the building was maintained at negative air pressure to prevent fugitive emissions. To assess the net bioaerosol emitting potential of biofilters, the pilot-scale biofilter system was relocated outside the waste hall to an external bay that was temporarily out of use (Plate 3.15). This was done to ensure that the biofilters were fed with air which has a relatively lower concentration of bioaerosols.

The biofilters were randomly selected to operate at the various levels of identified operating parameters of interest. In order to avoid media compaction and clogging (which could lead to the formation of preferential flow paths for air) during the experiments, the media was mixed with a shovel once every three weeks (Sanchez-Monedero et al., 2003). It should be noted at this point that this section presents a general operation of the pilot scale biofilters; a more detailed and specific discussion on the operation of the system to achieve the specific objectives are presented in each of the result chapter.

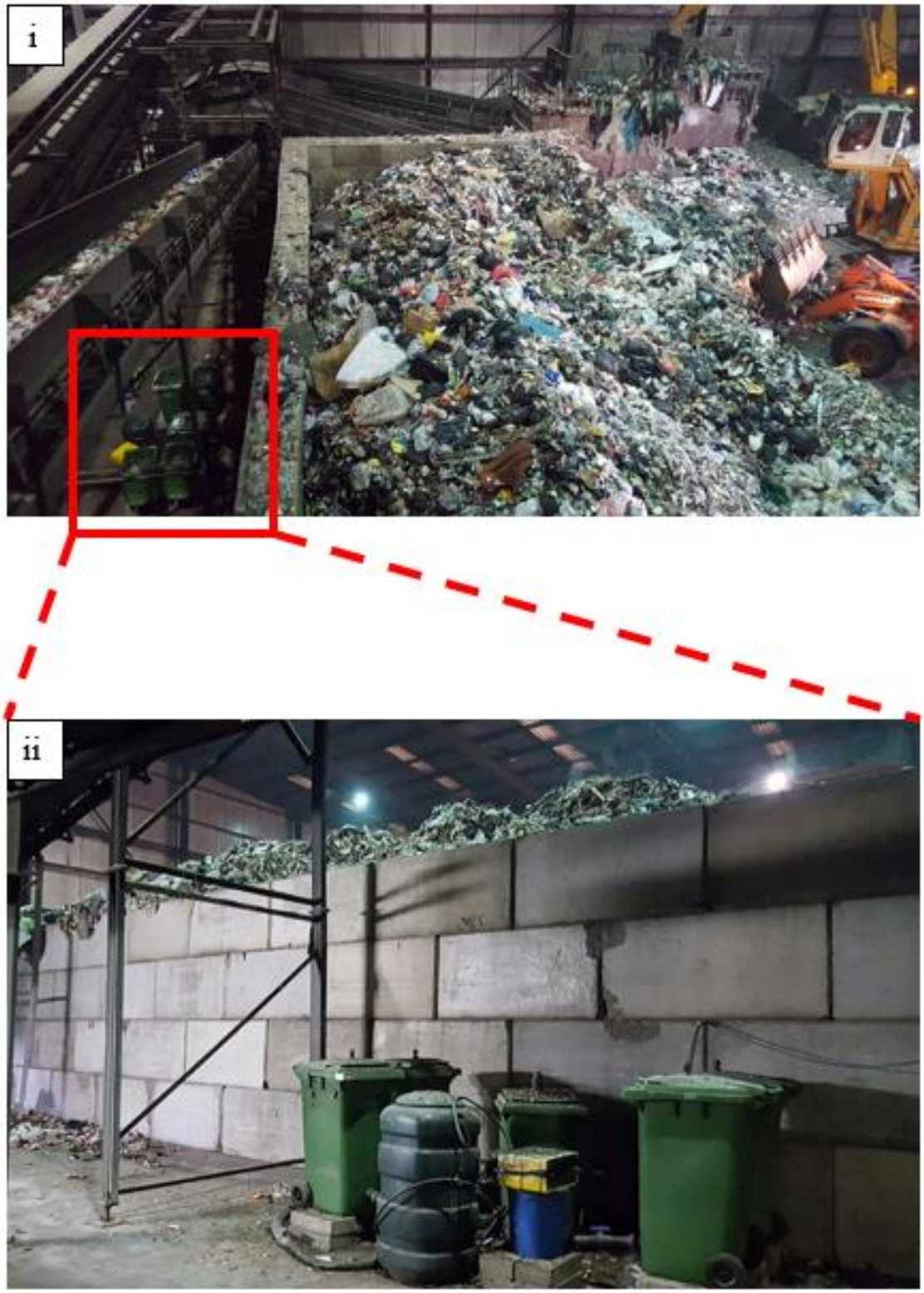


Plate 3. 14: Pilot-scale biofiltration system with four pilot-scale bioreactors located in the waste hall (i) just behind the back-push wall (ii).

Plate 3. 15: External bay (i) used to house the pilot-scale biofiltration system (ii) during the experiments to test net bioaerosol emitting potential of biofilters.

3.5 Odour and Waste Gas Composition Analysis

At the outset, the scope of this research included waste gas characterisation in terms of levels of ammonia, hydrogen sulphide, volatile organic compounds (VOCs) as well as the perceivable odour levels. These parameters were selected to enable biofilter performance assessment for simultaneous capacity to control bioaerosols as well as odours and odorous volatiles. The sampling and analysis for each group is presented in the subsequent sections.

3.5.1 Ammonia and Hydrogen Sulphide Analysis

Ammonia (NH_3) and hydrogen sulphide (H_2S) have been identified in the literature as some of the main constituents of waste gases from waste management facilities. Preliminary investigation to establish the presence and concentration of both gases on site was done using the Gastec Detector Tube System (Plate 3.16). The choice of this sampling method was based on a review of previous studies by Tsai et al. (2008) and Omri et al. (2011). This system consists of the Gastec pump (GV-110S) and Gastec Standard Detector Tubes (Elite Measurement Solutions Limited, n.d.; Gastec Corporation, 2013).

The waste reception area was selected for testing the concentrations of both gases potentially emitted from the decomposing waste heaps and/or other sources. The pump was used with two different tubes (for each type of gas), each tube selected on the basis of anticipated concentrations of contaminants to be found within the facility based on the literature. The detector tubes selected and their detection limits are shown in plate 3.17.

No humidity correction factor was required for both types of tubes. However, both required correction factors for pressure given by Equation 3.6:

$$Pressure = \frac{Tube\ Reading\ (ppm)\ \times 1013\ (hPa)}{Atmospheric\ Pressure\ (hPa)} \qquad \textbf{Eq. 3. 6}$$

Only NH_3 required a correction factor for temperature as shown in table 3.5 based on the manufacturer's guide.

Plate 3. 16: The Gastec Detector Tube System.

(i) **NH_3 detector tube # 3L**

Detection limit: 0.5 – 78 ppm

(ii) **H_2S detector tube # 4LT**

Detection limit: 0.05 – 4.0 ppm

Plate 3. 17: Gastec detector tubes for ammonia (i) and hydrogen sulphide (ii)

The tubes are usually sealed at both ends, and before sampling, the tips are broken off using a specially designed tube tip breaker fitted in the pump body. The pump connects to one end of the gas detector tube, and depending on the volume of air to be sampled; the system can be used to draw in 100 mL (at full stroke) or 50 mL (at half stroke) of gas into the tube. The full stroke and the half stroke positions are marked exactly by a red line on the pump shaft, and also at these positions the handle becomes precisely locked. Following a stroke, the drawn gas sample moves in the detector tube towards the pump, and as it moves it reacts with the chemical reagent within the tube, thus producing a colour change that is proportional in length to the concentration. With reference to a calibration scale printed on the tubes, the sample

gas concentration are easily read off as soon as sampling is completed (Plate 3.17), eliminating the need for laboratory analysis. The required correction factor can then be applied to the reading to get the actual concentration of the target gases (Table 3.5).

Table 3. 5: Ammonia temperature correction factor

Temperature ºC (ºF)	0 (32)	5 (41)	10 (50)	15 (59)	20 (68)	25 (77)	30 (86)	35 (95)	40 (104)
Correction Factor	1.25	1.25	1.15	1.07	1.0	0.95	0.9	0.86	0.83

NH_3 and H_2S monitoring was done for two consecutive days (6 & 7 August, 2015) with results shown in table 3.6. Despite high indoor odour levels (19340 OU_E/m^3) recorded for this facility through monitoring conducted by a private consultant, the levels of NH_3 and H_2S measured within the waste hall did not exceed the Work Exposure Limits (WELs) set out by the Health and Safety Executive (HSE, 2013). The observation for H_2S was thought to be due to the fact that waste piles were not allowed enough time for anaerobic decomposition to set in which triggers the release of hydrogen sulphide as is the case with the composting processes (Smet et al., 1999). Also, the low emission rates for NH_3 were due to the fact that the piles containing biological waste were not subjected to high amounts of heat and were transferred or moved on a regular basis. For these reasons, the scope of the pilot study was redefined to exclude assessment of biofilter performance in terms of ammonia and hydrogen sulphide removal.

Table 3. 6: Two-day monitoring results of Ammonia and Hydrogen Sulphide

		Day 1		**Day 2**	
Parameter		NH_3 (ppm)	H_2S (ppm)	NH_3 (ppm)	H_2S (ppm)
Preliminary results for this study		0.86	LOD	0.90	LOD
HSE WELs	Long term exposure limit (8-hr TWA reference period)	25	5	25	5
	Short term exposure limit (15 minute reference period)	35	10	35	10

LOD: Below Limit of Detection

3.5.2 VOC sampling and analysis

Volatile organic compounds (VOCs) were monitored in accordance with the methods for the determination of hazardous substances – volatile organic compounds in air (MDHS 96) as recommended by HSE (2000). Tenax tubes (supplied by Gradko International Limited) were used as the method of sampling VOCs within the facility. The tubes were designed for both passive and active monitoring of semi-volatile and volatile compounds ranging from C7 – C28 contained in the air. For the purposes of the preliminary assessment of the levels within the facility, active sampling was conducted with the use of a diffusion pump (Plate 3.18) which was calibrated to a flow rate of 50 ml/m and operated for a total of 10 minutes at the sampling location in accordance with guidance received from the tubes supplier (Table 3.7). Discussion with the operators revealed that high levels of elemental carbon were detected within the facility; however, access to the report was not possible. Pumped sampling has been recommended especially when the purpose is to compare results against workplace exposure limits (WELs) and a sampling time of 10 mins was selected because of the suspected high levels within this facility.

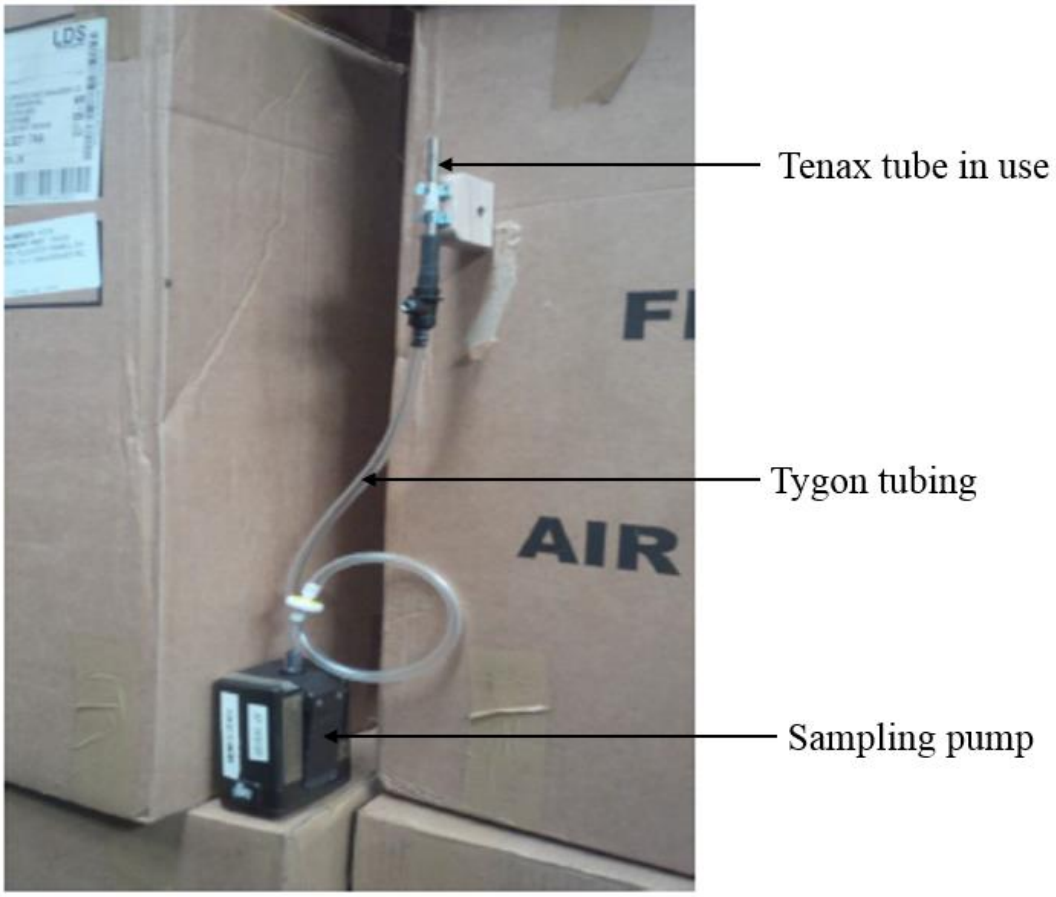


Plate 3. 18: VOC sampler setup in use on site.

Table 3. 7: Recommended monitoring durations for active samples (Gradko International, 2012)

Pollution level	Recommended pumping time
High (if you can smell it)	5 minutes
If you suspect it is high	5 - 10 minutes
No idea	50 minutes
Low	60 – 100 minutes
Expected clean air	100 minutes

Each tube (Figure 3.3) was made of stainless steel (which contain the tenax) 9 cm tall and 0.5 cm wide (excluding the cap). A total of two tubes were exposed each day for two days of preliminary assessment of VOC levels. Once sampling was completed, the exposed tubes and a travel blank (which was not exposed) were sealed and sent back to Gradko laboratory for analysis. Analysis involved sample extraction from tubes by thermal desorption followed by Gas Chromatography – Mass Spectroscopy (GC-MS) analysis for VOCs identification and concentration determination.

The tubes were analysed for BTEX and top 10 VOCs with results reported in micrograms per metre cubed (µg m^{-3}) (Tables 3.8 & 3.9). Ethanol ranked highest followed by limonene for VOCs. The identified levels were suitable for the purpose of this study; however, due to the high cost of analysis it was impossible to continue with Gradko International for identification and quantification of inlet and outlet VOCs for this study – especially as funds had to be redirected to other aspects of the project that would not compromise the achievement of the research objectives. Another reason for suspending evaluation of biofilter performance for VOC control was that during the experimental period, the laboratory within the school had not developed full capability for VOCs analysis with TD-GCMS which is the recommended VOC analysis protocol.

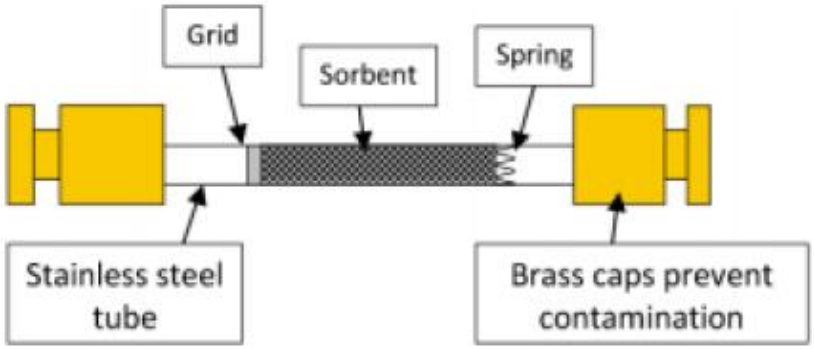


Figure 3. 3: Components of a sorbent (tenax) tube (Gradko International, 2012)

Table 3. 8: VOC Monitoring Results for Day 1

Parameter	**Results ($\mu g\ m^{-3}$)**	**Results ($mg\ m^{-3}$)**	**WEL 8-Hour TWA ($mg\ m^{-3}$)**
BTEX			
Benzene	40.84	0.04	3.25
Toluene	1412.60	1.41	191.00
Ethylbenzene	1017.50	1.02	441.00
m/p-Xylene	2137.50	2.14	220.00
o-Xylene	601.05	0.60	220.00
Top 10 VOCs			
Ethanol	12701.00	12.70	1920.00
Limonene	10629.00	10.63	300.00
Decane	7347.50	7.35	-
Butane	5106.70	5.11	1450.00
Ethyl Acetate	2598.70	2.60	200.00
Undecane	2228.40	2.23	-
Hexane	2107.20	2.11	72.00
Nonane	2034.10	2.03	1050.00
Acetic acid, butyl ester	1704.00	1.70	710.00
Decane, 4-methyl-	1361.03	1.36	208.00

Table 3. 9: VOC Monitoring Results for Day 2

Parameter	**Results ($\mu g\ m^{-3}$)**	**Results ($mg\ m^{-3}$)**	**WEL 8-Hour TWA ($mg\ m^{-3}$)**
BTEX			
Benzene	143.16	0.14	3.25
Toluene	1692.20	1.69	191.00
Ethylbenzene	620.05	0.62	441.00
m/p-Xylene	2050.10	2.05	220.00
o-Xylene	664.96	0.66	220.00
Top 10 VOCs			
Ethanol	11819.00	11.82	1920.00
Limonene	11059.00	11.06	300.00
Decane	7373.40	7.37	-
Butane	6340.60	6.34	1450.00
Benzene, 1,4-dichloro-	3453.00	3.45	153.00
Ethyl Acetate	2669.60	2.67	200.00
Cyclohexane, propyl	2270.20	2.27	350.00
Nonane	2255.10	2.26	1050.00
Undecane	2207.20	2.21	-
Hexane	2056.30	2.06	72.00

3.5.3 Odour measurement by Olfactometry

The procedure for sampling odour from the pilot biofilters was developed to optimise the collection of data which would be considered representative especially as the entire biofilter outlet could be sealed off with a lid and provided with a sampling port. The protocol drew upon guidance provided by Concept Life Science's existing UKAS accredited odour sampling procedures. Gas from each biofilter outlet headspace (and from the common plenum) was collected into bespoke 10-Litre Nalophan sampling bags (Plate 3.19) supplied by Concept Life Sciences who also carried out the analysis. Each pilot biofilter and the plenum was provided with a sampling port and during gas sampling, the top of the biofilters and plenum were completely sealed off to prevent any leakage. Gas was collected by placing the open end of the air-tight Nalophan bags over the provided sampling port and allowing the bag to be inflated by air inflow from the biofilter or common plenum, respectively (Figure 3.4). The average sampling time was 10 minutes (against the VDI 3880 recommendation of 30 minutes) and no background odour samples were taken due to cost limitations.

Plate 3. 19: Inflated Nalophan sampling bags

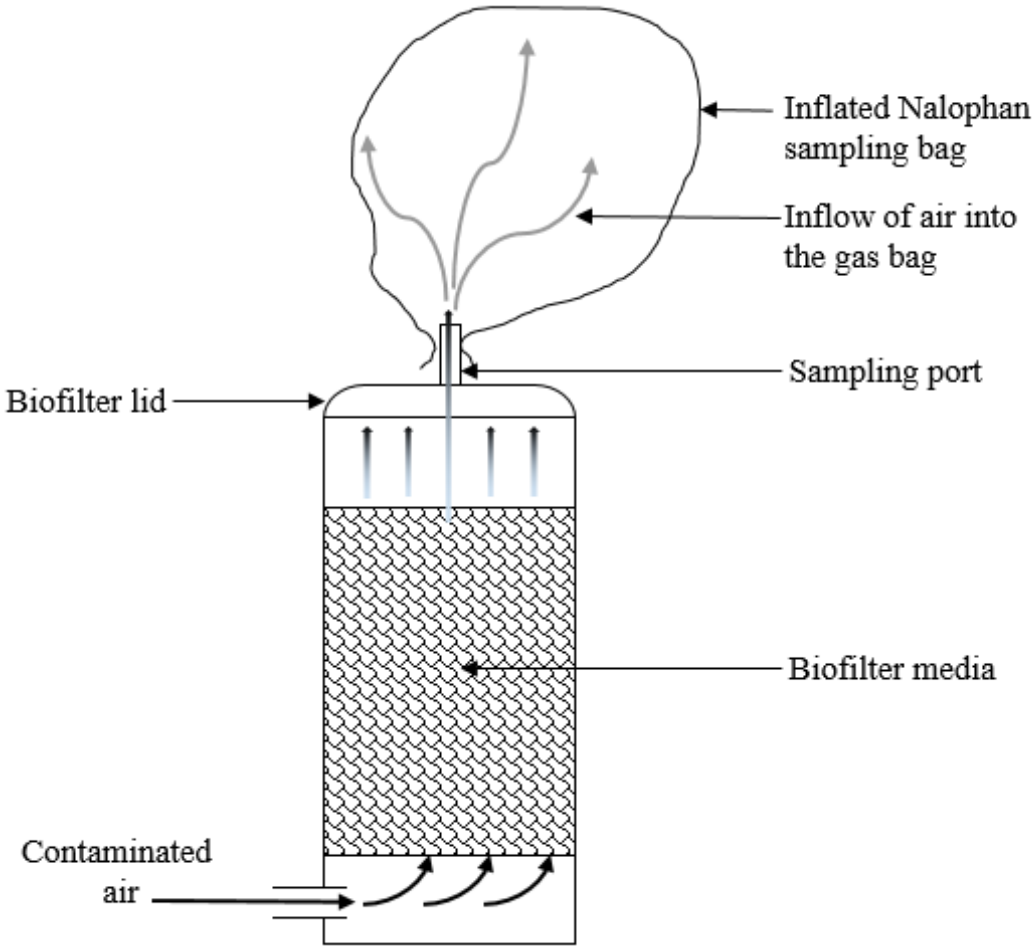


Figure 3. 4: Schematic of the gas sampling from each biofilter

The samples were sent off to Concept Life Science odour testing laboratory for analysis within 30 hours of sampling. Olfactometry analysis was carried out on the samples in accordance with BS EN 13725 to determine the odour concentration of the samples in European odour units (OU_E m^{-3}). To establish the concentration, an olfactometer test was used and this employed a panel of human noses as sensors (Concept Life Sciences, 2017). In the olfactometry testing procedure, a diluted odorous mixture and an odour-free gas (as a reference) are presented separately from sniffing ports to a group of panel members within an odour-neutral room. The panel then get asked to compare the gases emitted from each sniffing port and to report the presence of odour. The gas-diluting ratio is then decreased by a factor of two (implying that the chemical concentration is increased by a factor of two) after which the panel is asked to repeat their judgement. This procedure is repeated several times over different dilution levels. The panel's responses over a range of dilution levels are then used to calculate the concentration of the odour in terms of European odour units (OU_E m^{-3}).

This method is based on dilution of an odour sample down to the odour threshold (the point at which the odour is only just detectable to 50 % of the test panel) (Concept Life Sciences, 2017). Olfactometry was selected as the only method to assess biofilter performance in terms of simultaneous control of odour and bioaerosols. Page (2010) noted that odour measuring strategy should be directly related to the defined objectives of any study. With olfactometry, the key advantage is the direct correlation of odour and the sensitivity of the detector used (i.e. the human nose); thus affording a practical approach to measure odours directly, in order to objectively quantify the perception of odours.

3.6 Bioaerosol Measurement and analysis

Another component essential to the objectives of this study was the determination of the concentrations of various groups of viable airborne microorganisms. During each sampling visit, a six-stage Andersen sampler was used to collect air samples at the inlet (central plenum) and from the headspace of each biofilter outlet (Plate 3.20). The choice of this sampler was informed by the need to obtain both concentration and particle size data as with the study by Stagg et al. (2010). Two replicate samples were collected at each point for each of the bioaerosol groups studied. Air was pumped through the sampler at a rate of 28.3 L min^{-1} with a sampling time of 1 min to avoid overloading the Petri dishes containing the selective media for the bioaerosols. The Environment Agency (2017) recommended that sampling time should reflect the likelihood of overloading plates (>300 colonies). Preliminary sampling on this site indicated plate overload even with sampling times of 3 to 5 mins; hence, the decision for further reduction to 1 min.

Four groups of bioaerosols were measured including *Aspergillus fumigatus*, total fungi, total mesophilic bacteria and Gram negative bacteria. The choice of these microorganisms was informed by the need to reflect the range covered in the Sniffer report (ER36) on understanding biofilter performance and determining emission concentrations under operational conditions (Fletcher et al., 2014) as well as those specified in the Technical Guidance Note (M9) for monitoring of bioaerosols at regulated facilities (Environment Agency, 2017). Bioaerosols detection and quantification were achieved by selective agar and visual identification. The specific agar type, supplements added, incubation temperatures and times for the bioaerosols are shown in Table 3.10. The growth media (agar) was prepared using a combination

of a Masterclave 09 (supplied by Don Whitley Scientific) and an automated pourer stacker, APS One (supplied by AES Blue Line™) (Plate 3.21).

Bioaerosol concentrations are known to fluctuate dramatically within a short time (Searl, 2008), and also depending on the activities within the waste hall (Stagg et al., 2013). Thus, the results of this study should be interpreted with caution as there may be uncertainties in the representativeness of the measured concentrations relative to actual exposure conditions due to periodic differences in activities. Moreover, it is estimated that < 10% of bioaerosols are viable and can grow on media (Blomquist, 1994; Swan et al., 2003) the remainder being composed of non-viable components; thus, there may be an underestimation of actual bioaerosol components which may have potential health implications.

For each sampling visit, inlet concentrations corresponded to bioaerosol samples taken from the common plenum; this was considered representative of the concentrations delivered directly to each biofilter. The outlet concentrations from each biofilter was taken from the top of each biofilter (Figure 3.5). In order to ensure the integrity of samples taken, all outlet measurements were conducted using methods which isolated treated air exiting the biofilters from the effects of ambient contamination within the waste hall. This was done by completely covering the outlet (open) end of the biofilters using plastic sheets (Fletcher et al., 2014). The biofilters were covered between sampling days to prevent surface contamination, during which treated air was released through 20 mm exhaust provided at the top of each biofilter cover. On sampling days, it was assumed that the headspace air was the treated air, isolated from ambient contamination and so sampling was done immediately after sheeting the biofilters.

To assess whether outlet bioaerosol concentrations were comparable to the background levels surrounding the site, bioaerosol concentrations were measured upwind (i.e. outdoors just at the boundary of the site) at a height of 1.8 m above the ground (Environment Agency, 2017). Stagg et al. (2013) reported that the concentration of bacteria and fungi within MRFs were ten times the upper levels measured in ambient air. Thus, upwind (background) sampling was necessary to give information on the concentration of bioaerosols in the air blowing onto the site (Environment Agency, 2017) which would then form the basis to assess biofilter performance in terms of achieving background (ambient) concentrations. After the

incubation period, the number of colonies were counted and a positive-hole correction was done to adjust colony counts in accordance with the recommendations of Macher (1989). The results were expressed as means of replicate samples taken in colony forming units per cubic metre of air (cfu m^{-3}). The limit of detection of the sampler was less than 10^2 cfu m^{-3}.

Plate 3. 20: Bioaerosol sampling from (i) the outlet and (ii) the inlet (central plenum) of biofilter

Table 3. 10: Incubating conditions for specific bioaerosols tested

Bioaerosol Group	**Agar**	**Supplements**	**Incubation Temperature**	**Incubation Time**
Aspergillus fumigatus	20 g L^{-1} each of malt extract agar and bacteriological agar	Streptomycin, 50 mg L^{-1}; Novobiocin, 10 mg L^{-1}	40°C	48 hours
Total fungi	20 g L^{-1} each of malt extract agar and bacteriological agar	Streptomycin, 50 mg L^{-1}; Novobiocin, 10 mg L^{-1}	40°C	48 hours
Total mesophilic bacteria	14 g L^{-1} nutrient agar and 10 g L^{-1} bacteriological agar	Cycloheximide, 100 mg L^{-1}	37°C	48 hours
Gram negative bacteria	52 g L^{-1} MacConkey agar	Cycloheximide, 200 mg L^{-1}	37°C in the dark	3 – 7 days

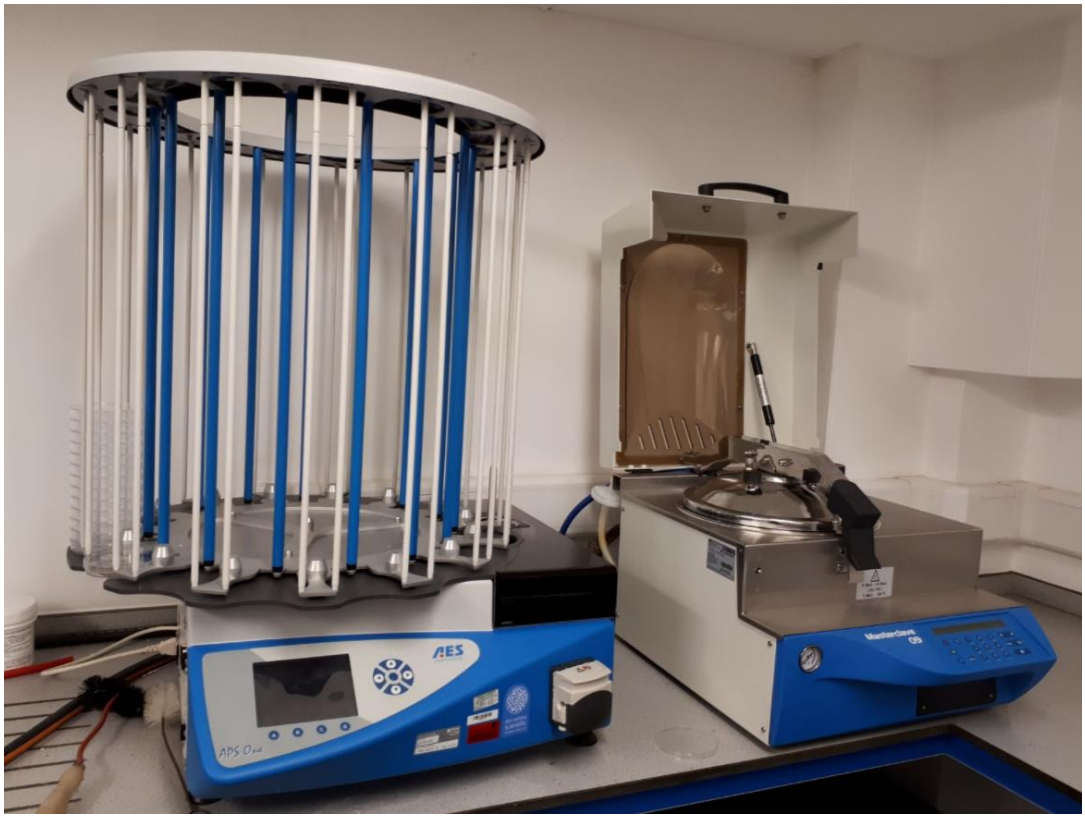


Plate 3. 21: Media preparation system consisting of the automated pourer stacker (APS One) on the left and the Masterclave 09 on the right.

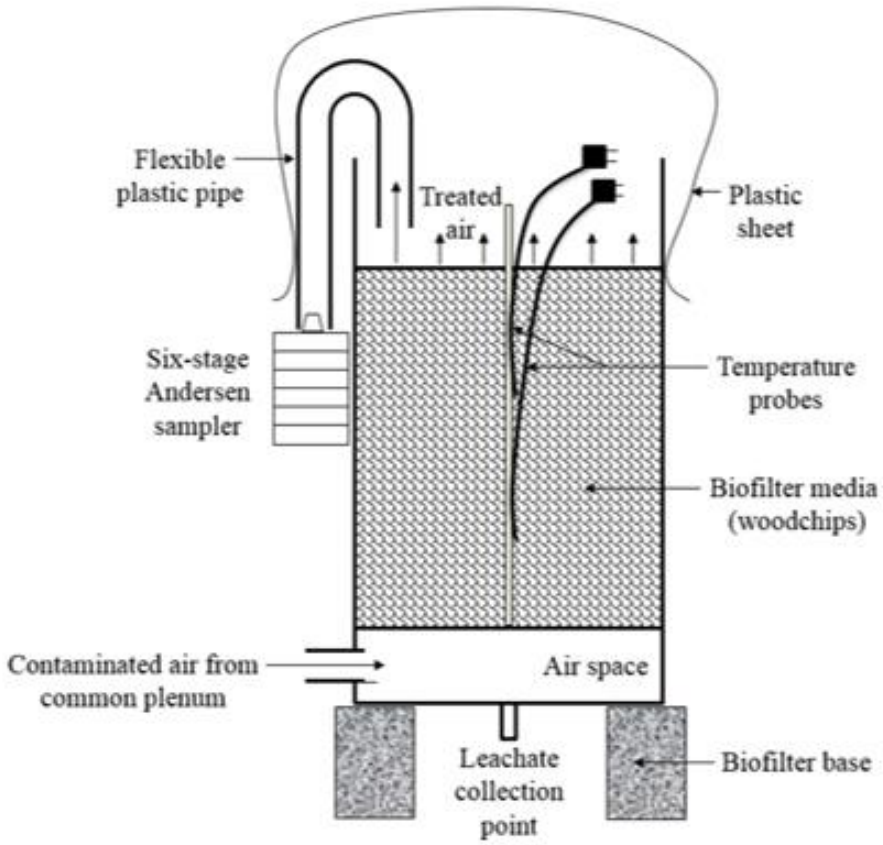


Figure 3. 5: Pilot-scale biofiltration system - schematic of each reactor showing bioaerosol sampling with the six-stage Andersen sampler

3.7 Biofilter Performance Evaluation and Data Analysis

The performance of the pilot biofilters was evaluated on the basis of removal efficiency (%) for odour and bioaerosols (Devinny et al., 1999). Removal efficiency (RE) was calculated using Equation 2.2 (section 2.2.6.5). For bioaoerosols, the air sampler design also allowed for size distribution of the collected bioaerosols according to their aerodynamic behaviour. This was obtained by summing up the corrected colony counts on each stage of the sampler and grouping according to the manufacturer's aerodynamic information for stages 1 (sampler inlet) to 6 (sampler outlet) as 7.0, 4.7, 3.3, 2.1, 1.1 and 0.65 μm, respectively.

All statistical analysis were carried out in the IBM SPSS Statistics for Windows (Released 2015. Version 23.0. Armonk, NY: IBM Corp., USA) and graphs generated using Origin (OriginLab, Northampton, MA, USA). The specific statistical analyses conducted for each objective are presented in the respective result chapters.

3.8 Leachate collection and analysis

It is good practice to measure leachate pH as this reflects the media conditions near the bottom of the media especially for up-flow biofilter configurations (Devinny et al., 1999). Leachate from each pilot biofilter collected within the airspace below the media bed and was drained twice a week through the leachate collection port (Figure 3.2). The collected leachate volumes were measured and analysed for pH, conductivity and microbial composition. The pH and conductivity of the leachate were taken to be indicative of the media health at the air inlet and were measured weekly using a digital calibrated pH-meter (HI98100 Checker® Plus pH Tester supplied by HANNA Instruments) which also had capability for conductivity measurement.

Leachate microbial composition analysis was based on identification of the same four groups of bioaerosols targeted for control with the biofilter. The specific agar type, supplements used, incubation temperatures and duration for the microorganisms were as outlined in Table 3.10. For each leachate sample, a four-fold serial dilution of the raw sample was done and 0.1 mL was transferred onto the prepared agar plate using a sterile pipette. The plates were then incubated according to the specific incubation conditions for each group of microorganisms after which colonies were counted (Garrido-Cardenas et al., 2017).

3.9 Meteorological Measurements

During each sampling visit, the environmental conditions around the waste hall and close to the pilot biofilter system were assessed by measuring the relative humidity, temperature and wind speed. Temperature and relative humidity values were measured using the portable thermo-hygrometer (HANNA Instruments Model HI 8564) (Plate 3.13). Monitoring was carried out according to the user's manual with the probe held out at head height with the head of the humidity detector exposed directly to ambient air around the biofilter system. Wind speed was measured using the Kestrel ® 1000 Pocket Wind ™ meter (Plate 3.22), which was operated according the user's manual by holding the device out in the air and reading off the values displayed on the screen.

Plate 3. 22: The Kestrel ® 1000 Pocket Wind ™ meter.

Chapter 4
PILOT-SCALE BIOFILTRATION AT A MATERIALS RECOVERY FACILITY: THE IMPACT ON BIOAEROSOL CONTROL

4.1 Introduction

The objective of this study was to investigate the performance of pilot-scale biofilters for removal of bioaerosols from waste airstreams from a materials recovery facility (MRF) which acted as a source of bioaerosols. The sub-objectives were: (i) to assess the impact of empty bed residence time (EBRT) on the performance of pilot-scale biofilters in terms of bioaerosol reductions; (ii) to evaluate the net bioaerosol emitting potentials of biofilters and to assess the effect of inlet concentration on bioaerosol control; and (iii) to assess size distribution of bioaerosol particles in biofilter exhausted air and to relate these to the tidal volume inhaled by humans.

4.2 Biofilter operation to evaluate performance

Section 3.4 presents a general discussion of the operation of the biofilters throughout the study. However, this section presents more detailed information on how the biofilters were operated and maintained and the data analyses that were carried out to fulfill the specific objectives of this particular section of the study.

4.2.1 Biofilter Operation

The biofilter system was operated continuously for 11 months from May 2016 to March 2017. A total of 16 sampling visits were completed; visits 1-6 (summer 2016) and 13-16 (winter 2017) were conducted inside the building while visits 7-12 (winter 2016) were conducted outside the building. Before sampling commenced, the media in each reactor was allowed to stabilise for four weeks following recommendations in literature (Cabrol et al., 2012; Ralebitso-Senior et al., 2012). To assess the impact of EBRT on bioaerosol removal, the system was set up inside the facility just behind the back-push wall in the waste reception area as discussed in section 3.4.

One major concern with biofilters, especially for regulators and operators, is their potential to act as net emitters of bioaerosols at low inlet concentrations due to extra contamination by the filtration process (Ottengraf and Konings, 1991; Fletcher et al., 2014). To investigate this, the biofilters were relocated outside the waste hall to an

external bay (previously used to collect fines – soils, glass, small wood, small stones, ferrous and non-ferrous materials – 0 < 10mm) during visits 7 – 12. This location simulated ambient conditions as the biofilters were fed with air that had relatively lower concentrations of bioaerosols. The understanding was that biofilters would be considered net emitters if outlet concentrations were higher than inlet concentrations.

The biofilters were randomly selected to operate at the tested EBRT during which average moisture content of 64.7% (40.2 to 70.3%), 62.4 (38.8 to 70.3%), 55.2% (43.3 to 68.9%) and 59.2% (41.2 to 70.5%) were maintained in BF1, BF2, BF3 and BF4, respectively, all within the range recommended by Janni et al. (2011). In order to avoid media compaction and clogging, which could lead to the formation of preferential flow paths for air, the media was mixed with a shovel once every three weeks on days other than the sampling days (Sanchez-Monedero et al., 2003).

4.2.2 Data Analysis

Table 4.1 presents a summary of mean counts and standard deviations of measured concentrations of bioaerosols. Normality of bioaerosol concentrations was assessed using the Shapiro-Wilk test. All statistics were carried out on original bioaerosol concentrations (Appendix A) rather than the calculated RE. Differences in mean bioaerosol concentration for the background, BF inlet and all BF outlets were assessed using the one-ANOVA/Welch ANOVA, regardless of whether or not the assumption of normality was met. In all cases, the assumption of homogeneity of variances was violated, as assessed by Levene's test for equality of variances ($p < 0.05$) for all groups of bioaerosols.

For visits 1-6, Welch ANOVA followed by Games-Howell post hoc analysis indicated statistically significant differences ($p < 0.05$) between the inlet samples and all outlet samples as well as background concentration of *A. fumigatus*, total fungi and total mesophilic bacteria. There was no statistically significant difference between the inlet and outlet concentration of Gram negative bacteria ($p = .178$). For visits 7-12, there was no statistically significant difference between the mean concentrations of background, inlet and all outlets samples of *A. fumigatus* ($p = 0.054$) and Gram negative bacteria ($p = 0.776$) as assessed by Welch ANOVA.

Table 4. 1: Mean bioaerosols counts and standard deviations (SD) in cfu m^{-3}

Visit	*Aspergillus fumigatus*						Total fungi						Total mesophilic bacteria						Gram negative bacteria					
	Background*		Inlet*		Outlet**		Background*		Inlet*		Outlet**		Background*		Inlet*		Outlet**		Background*		Inlet*		Outlet**	
	Count	SD	Count	SD	Count	SD	Count	SD	Count	SD	Count	SD	Count	SD	Count	SD	Count	SD	Count	SD	Count	SD	Count	SD
1	3.3×10^2	26	9.7×10^3	79	1.4×10^3	532	4.5×10^2	32	1.2×10^4	380	1.8×10^3	584	8.8×10^2	75	1.1×10^4	1284	3.8×10^3	1317	7.0×10^2	46	6.9×10^3	1050	1.3×10^3	602
2	1.2×10^2	12	1.2×10^4	1045	2.7×10^3	1032	2.2×10^2	10	1.5×10^4	1466	3.2×10^3	1251	6.0×10^2	65	5.3×10^3	4089	8.2×10^3	3260	3.6×10^2	55	2.4×10^3	188	2.5×10^3	620
3	1.5×10^2	80	1.0×10^4	1412	4.2×10^3	6661	1.9×10^2	99	1.3×10^4	663	4.8×10^3	7464	6.2×10^2	18	5.9×10^3	76	2.9×10^3	1937	3.9×10^3	323	2.5×10^4	438	3.6×10^3	1543
4	1.8×10^2	71	7.4×10^3	1981	1.7×10^3	1675	1.9×10^2	18	9.3×10^3	2159	2.2×10^3	1764	7.4×10^2	141	2.3×10^4	896	4.6×10^3	3467	2.1×10^3	281	6.4×10^3	131	3.4×10^3	945
5	9.8×10^2	231	8.1×10^3	383	1.0×10^3	237	1.1×10^3	214	1.0×10^4	935	1.6×10^3	224	2.6×10^3	113	1.3×10^4	6627	1.9×10^3	940	2.4×10^3	44	1.9×10^4	9698	6.1×10^3	1912
6	3.0×10^2	18	3.8×10^3	1094	1.1×10^3	317	3.5×10^2	35	4.8×10^3	1544	1.3×10^3	326	1.8×10^4	10226	1.3×10^4	832	3.2×10^3	1681	1.5×10^4	12633	5.6×10^3	765	4.4×10^3	2564
7	1.1×10^3	398	1.1×10^3	286	1.9×10^2	116	1.5×10^3	800	1.5×10^3	413	4.1×10^2	306	2.6×10^3	396	5.0×10^3	2399	1.8×10^3	917	4.8×10^3	1406	3.6×10^3	822	1.7×10^3	565
8	6.2×10^2	548	1.4×10^3	288	93	53	6.2×10^2	548	1.8×10^3	382	1.1×10^2	62	1.2×10^3	253	9.3×10^3	719	1.6×10^3	478	3.6×10^3	1979	2.2×10^3	565	1.7×10^3	732
9	7.1×10^2	35	9.6×10^2	106	57	35	1.1×10^2	71	1.2×10^3	198	66	37	4.0×10^3	2670	2.6×10^3	330	1.6×10^3	818	1.4×10^3	737	5.0×10^2	359	1.1×10^3	598
10	7.1×10^2	35	6.8×10^2	216	6.8×10^2	139	7.8×10^2	35	9.1×10^2	201	7.1×10^2	138	1.4×10^3	120	6.5×10^2	159	1.5×10^3	513	1.5×10^3	910	1.7×10^3	636	1.3×10^3	374
11	1.4×10^2	35	1.1×10^2	106	44	66	1.9×10^2	18	1.6×10^2	124	80	82	2.9×10^3	443	1.6×10^3	30	1.6×10^3	735	5.7×10^2	143	1.0×10^3	323	2.3×10^3	1155
12	1.6×10^2	53	53	18	97	66	1.9×10^2	88	1.8×10^2	71	97	66	1.3×10^3	35	1.6×10^3	472	9.8×10^2	404	1.5×10^3	201	1.2×10^3	198	7.8×10^2	394
13	9.1×10^2	164	1.1×10^4	4488	3.8×10^2	481	1.3×10^3	147	1.5×10^4	6115	6.5×10^2	983	3.7×10^3	1076	1.4×10^4	2949	2.4×10^3	1120	2.0×10^3	528	5.9×10^3	1756	2.1×10^3	632
14	2.1×10^2	35	2.1×10^4	479	1.1×10^4	8454	7.3×10^2	194	4.1×10^4	3827	1.6×10^4	12982	8.2×10^2	216	6.0×10^4	3767	3.0×10^4	12969	1.7×10^3	230	1.0×10^4	7943	2.9×10^4	17064
15	8.9×10^2	37	1.0×10^4	1988	4.2×10^3	1090	1.1×10^3	94	1.2×10^4	2072	5.7×10^3	1408	6.9×10^2	53	4.7×10^4	1276	2.2×10^4	10116	1.2×10^3	104	2.6×10^4	14846	2.1×10^4	13616
16	6.0×10^2	35	5.3×10^4	9154	3.4×10^4	4122	8.9×10^2	111	6.3×10^4	11843	4.0×10^4	5448	5.2×10^3	300	9.6×10^4	25016	5.1×10^4	22975	2.3×10^3	945	2.8×10^4	4620	1.3×10^4	5011

SD: Standard Deviation; * n = 2; ** n = 8

However, Games-Howell post hoc analysis showed statistically significant differences between the inlet concentration and outlet concentrations of BF2 (p = 0.05) and BF4 (p = 0.047) for total fungi as well as between inlet and outlet samples of BF1 (p = 0.01) and BF3 (p = 0.021) for total mesophilic bacteria. For visits 13-16, there were significant differences only between inlet and background concentrations of total fungi (p = 0.048) and total mesophilic bacteria (p = 0.028).

4.3 Results and Discussion

4.3.1 Operating Conditions

The operating conditions of each biofilter for the period have been summarised in table 4.2. The impact of EBRT was assessed during the first six sampling visits by comparing the outlet bioaoerosol concentration for the four biofilters. During the first three sampling visits the biofilters were all adjusted to run at an average EBRT of 16 s. An assessment of the outlet bioaerosol concentrations showed that there were no outliers and the data was normally distributed for each group as assessed by Shapiro-Wilk test (p < .05). However, there was heterogeneity of variances for *A. fumigatus* (p = .003) and total fungi (p = .004) as assessed by Levene's test of homogeneity of variance; there was no statistically significant differences in the outlet concentrations of *A. fumigatus* (p = .433) and total fungi (p = .482) from all four biofilters as assessed with Welch ANOVA. One way ANOVA also indicated that there was no statistically significant difference in the outlet concentrations for total bacteria (p = .670) and Gram negative bacteria (p = .594).

For visits 4-6, BF1 and BF4 were randomly selected to operate at an average EBRT of 70 s while BF2 and BF3 had an average EBRT of 11 s. This was done to assess whether there were contact time dependent significant differences in the measured outlet bioaerosol concentrations between the two groups of biofilters. Welch ANOVA indicated that there was no statistically significant difference between all outlet concentrations for *A. fumigatus* (p = .407), total fungi (p = .425) and total bacteria (p = .243). For Gram negative bacteria, one way ANOVA also showed no statistically significant difference (p = .148) in the outlets from the four biofilters. In summary, there was no significant difference in the performance of the biofilters when operated under varying conditions of EBRT. Limited statistical power due to the modest sample size in this study (n = 64) may have played a role in limiting the significance of some of the statistical comparisons carried out (Cornish, 2006). Post hoc power analysis,

with power $(1 - \beta)$ set at 0.90 and $\alpha = 0.05$, indicated that sample size would have to increase up to 95 samples for group differences to reach statistical significance at the 0.05 level.

Table 4. 2: Operating conditions of the biofilters (BF) when operated within (visits 1-6, 13-16) and outside (visits 7-12) the building

Parameter		Visits		
		1-6	**7-12**	**13-16**
Mean inlet air temperature (ºC)		23.8	15.8	15.2
BF1	Leachate pH range	5.19 - 6.52	6.52 - 7.0	6.83 - 7.04
	Mean EBRT (s)	16, 70	16	16
	Mean Airflow rate (L min^{-1})	681, 156	681	681
	Mean media temperature (ºC)	19.5	15.1	13.8
	Mean outlet air temperature (ºC)	21.8	13.1	14.5
BF2	Leachate pH range	5.12 - 6.64	6.62 - 7.52	6.56 - 7.38
	Mean EBRT (s)	16, 11	16	16
	Mean Airflow rate (L min^{-1})	681, 990	681	681
	Mean media temperature (ºC)	20.3	13.9	13.6
	Mean outlet air temperature (ºC)	21.4	13.8	14.5
BF3	Leachate pH range	6.17 - 7.04	6.98 - 7.34	6.77 - 7.37
	Mean EBRT (s)	16, 11	16	16
	Mean Airflow rate (L min^{-1})	681, 990	681	681
	Mean media temperature (ºC)	21.2	14.5	15.3
	Mean outlet air temperature (ºC)	21.4	13.0	14.6
BF4	Leachate pH range	5.55 - 6.53	6.43 - 7.44	6.95 - 7.21
	Mean EBRT (s)	16, 70	16	16
	Mean Airflow rate (L min^{-1})	681, 156	681	681
	Mean media temperature (ºC)	21.2	14.6	16.1
	Mean outlet air temperature (ºC)	21.1	13.5	14.3

Odour control function of biofilters is dependent on the activity of microbial population within the media. These microorganisms thrive at a pH range of 6.5-8 which must be maintained within the internal environment of the biofilter (Wani et al., 1997; Schnelle and Brown, 2002). However, to evaluate the performance for bioaerosol control, the biofilters were operated without any supplementary attempts to alter the pH which was in the range of 5.12 – 7.52 for all four biofilters. Also, no adjustments were made to alter the media temperature especially as these were within

the optimal levels (10 – 40 ºC) recommended for biological treatment systems (Schnelle and Brown, 2002).

4.3.2 Removal Efficiency

The first set of results considers the removal efficiency of the biofilters under conditions of high inlet bioaerosol concentrations under summer (visits 1-6) and winter (visits 13-16) conditions. Figure 4.1 shows the RE and concentrations of each group of bioaerosols sampled at the different sampling points (background, biofilter inlet and outlets) plotted against the site visits conducted for this study. There was no significant difference between the performances of the four pilot-scale biofilters (Section 3.1), hence the REs were computed using the mean outlet concentrations from the four reactors for each visit.

During visits 1 – 6, inlet *A. fumigatus* concentration ranged from 3.8×10^3 to 1.2×10^4 cfu m^{-3} for which the biofilters achieved RE of 60 – 88%, giving outlet concentrations between 1.0×10^3 to 4.2×10^3 cfu m^{-3}. Similarly, the biofilters achieved RE of 65 – 85% for total fungi with inlet concentration in the range of 4.8×10^3 to 1.5×10^4 cfu m^{-3} and delivering outlet concentrations between 1.3×10^3 to 4.8×10^4 cfu m^{-3}. *A. fumigatus* particles constituted approximately 80% of the total fungi particles, comparable to the study of Millner et al. (1977) who reported that *A. fumigatus* made up 75% of the total viable mycoflora captured on the compost site studied. For this same period, slightly lower RE of 52 – 86% was recorded for the total mesophilic bacteria with outlet concentration of 1.9×10^3 to 8.2×10^3 cfu m^{-3} from inlet concentration 5.9×10^3 to 5.3×10^4 cfu m^{-3} while the biofilter achieved a much lower RE of -4.1 to 86% for Gram negative bacteria, treating inlet concentration between 2.4×10^3 to 2.5×10^4 cfu m^{-3}. The data suggest that variation of EBRT (between 11 s, 16 s and 70 s for this study) did not influence RE for the four groups of bioaerosols measured. This observation is supported by data presented by Sanchez-Monedero et al. (2003) which showed that RE for *A. fumigatus* did not appear to be related to the gas phase residence times of biofilters which operated in the range of 29 - 97s, and achieved RE > 90%. Similarly, no relationship was found between gas phase residence time and the RE for mesophilic bacteria (highest: 89.6% at 36s, and lowest: 39.1% at 37s), suggesting that gas phase residence time may not play a significant role in the capture of aerosolised bacteria and fungi.

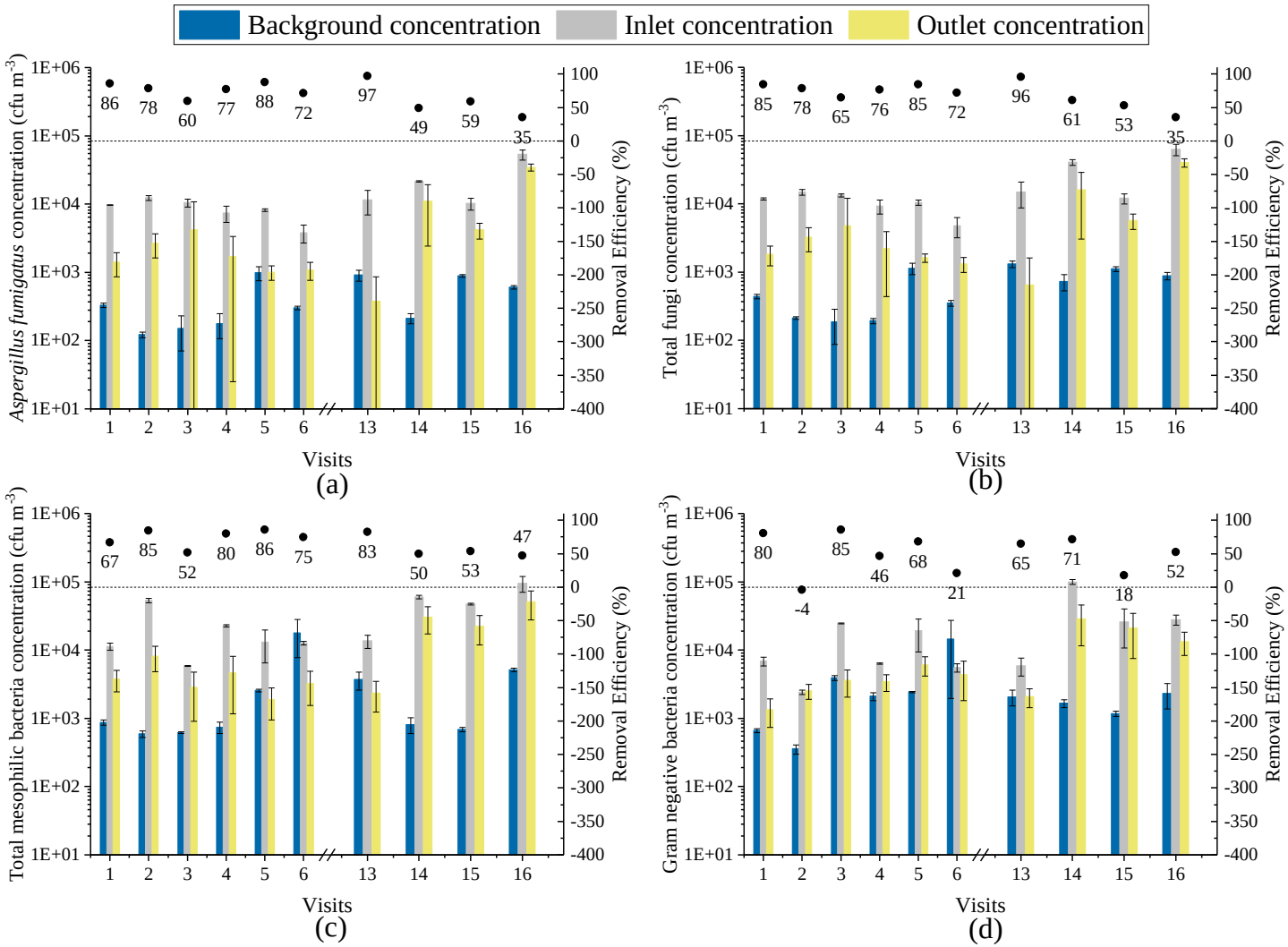


Figure 4. 1: Removal efficiency and corresponding background, inlet and outlet concentrations of (a) *A. fumigatus*, (b) total fungi, (c) total mesophilic bacteria and (d) Gram negative bacteria in cfu m^{-3} when biofilters were operated within the building. (Error bars = standard deviation; n = 2)

Leson and Winer (1991) recommended typical residence times of 25 - 60s for commercial or industrial biofilter applications for odour and low volatile organic compound (VOC) abatement, and gas phase residence times less than 23s have been shown to cause resistance of the transfer of hydrogen sulphide from the gas phase into the biofilm layer of the media (Yang and Allen, 1994). It therefore suggests that significant bioaerosol RE is achievable across a range of EBRTs that can deliver both poor and optimum odour control. Martens et al. (2001) in their research suggested that bioflters which were excellent odour abatement systems emitted slightly more bioaerosols particles. However, they could not establish any relationships between the removal efficiencies of the odour/ammonia and bioaerosols for the five filter materials (i.e. biochips, coconut-peat, wood-bark, pellets & bark, and compost) tested. Bioaerosol capture mechanisms include inertial deposition, diffusional or Brownian deposition and flow line interception (Ottengraf and Konings, 1991); and these combine to effect bioaerosol impingement on the solid media material such that as bioaerosol-laden air sweeps through the media bed, the particles get deposited within the media, a function which may not be dependent on gas contact time. This further suggests that a low EBRT biofilter which may not favour odour control may actually achieve significant bioaerosol control. However, this observation may have been influenced by the small sample size and the variability in the dataset, and so valid conclusions would require an extensive study with a larger sample size. Nonetheless, Fletcher et al. (2014) argued that it may not be possible to achieve simultaneous significant control of odour and bioaerosols within a single biofilter as the mechanisms involved in the removal of these two pollutants are different. They also noted that bioaerosol removal may be enhanced by increasing airflow which decreases the EBRT.

In winter conditions (visits 13 – 16), the sampling yielded REs of 60% (35 – 97%), 61% (35 – 96%), 58% (47 – 83%) and 51% (18 – 71%) for *A. fumigatus*, total fungi, total mesophilic bacteria and Gram negative bacteria, respectively. It was observed that the inlet concentrations during visits 13 – 16 (winter) were significantly higher ($p < .05$) than during visits 1 – 6 (summer), up to 5.3×10^4 cfu m^{-3}, 6.3×10^4 cfu m^{-3}, 9.6×10^4 cfu m^{-3} and 1.0×10^5 cfu m^{-3} for *A. fumigatus*, total fungi, total mesophilic bacteria and Gram negative bacteria, respectively. It is unclear why this was so, especially as bioaerosols concentrations tend to be higher in summer for most waste management facilities (Stagg et al., 2010). However, it was observed that the volume

of waste heap in the waste reception area were greater in the winter than in summer, thus, there were increased activity of the front loaders and dinosaurus machine to feed the conveyors while clearing the area for incoming loads. Searl (2008) noted that bioaerosol concentrations can fluctuate over short periods, and increased activity levels within the waste facility may be associated with higher bioaerosol exposure. Thus, the higher winter concentrations in this study could be a function of the increased activities due to huge volume of waste being processed. Furthermore, Nasir and Tyrrel (2017) concluded that bioaerosol emissions from waste treatment facilities can be highly variable and characterisation based on snapshot and infrequent sampling may not give a true reflection of the magnitude of emissions. Most waste management facilities have as part of their permit condition the need to demonstrate that they can meet required emissions limit values. In this study, in spite of the high REs achieved during summer and winter (> 80%), the measured outlet concentrations still exceed background (upward) concentration, and are often in excess of the guideline provided in the EA position statement and so might be of concern to site workers and members of public living in the vicinity of site if these were operated at full scale.

4.3.3 Potential for emissions from biofilters

As noted in 4.2.1, one of the key concerns with biofilters has been their potential to act as net emitters of bioaerosols – this being one of the major concerns for regulators and operators (Fletcher et al., 2014). During sampling visits 7 - 12, the biofiltration system treated less polluted air with inlet concentrations in the range of 53 to 1.4×10^3 cfu m^{-3}, 1.6×10^2 to 1.8×10^3 cfu m^{-3}, 6.5×10^2 to 9.3×10^3 cfu m^{-3} and 5.0×10^2 to 3.6×10^3 cfu m^{-3} for *A. fumigatus*, total fungi, total mesophilic bacteria and Gram negative bacteria, respectively. The results in Figure 4.2 show that REs drop significantly and in some cases become negative with values as low as -83% (*A. fumigatus*), -122% (total mesophilic bacteria) and -128% (Gram negative bacteria).

The negative removal efficiencies are indicative of a greater concentration leaving the biofilter than entering and are thought to result from microorganisms (a) passing through the media, and/or (b) growing within the media and being released from it (Sanchez-Monedero et al., 2003). Fletcher et al. (2014) stated that approximately 10^7 microorganisms/g colonise media surfaces; and some of these could become mobilised as air passes through the biofilter and so may result in higher concentration of bioaersols in the treated air compared to the untreated air (Rabe and Becker, 2000).

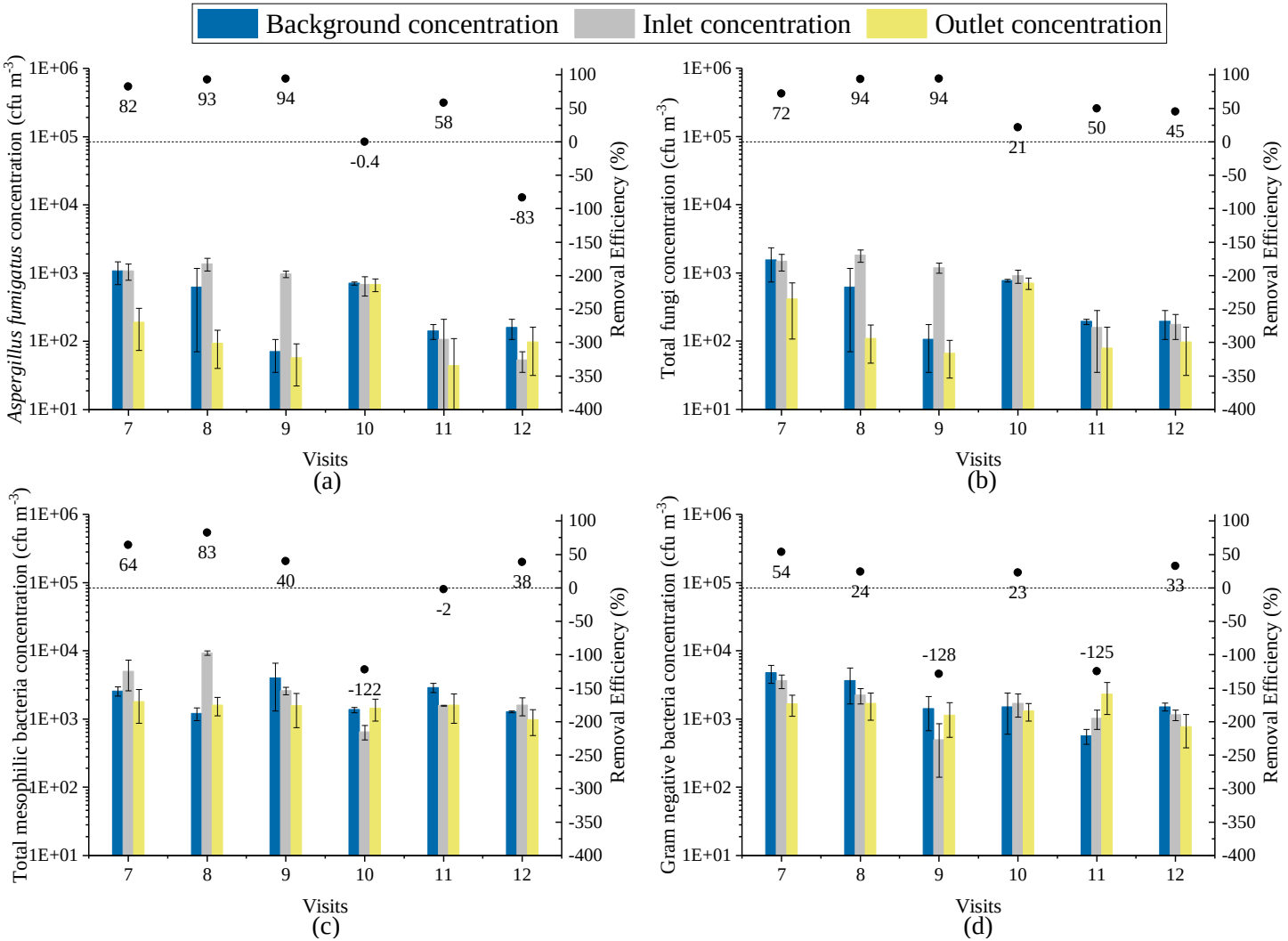


Figure 4. 2: Removal efficiency and corresponding background, inlet and outlet concentrations of (a) *A. fumigatus*, (b) total fungi, (c) total mesophilic bacteria and (d) Gram negative bacteria in cfu m^{-3} when biofilters were operated outside the building. (Error bars = standard deviation; n = 2).

Martens et al. (2001) also added that this may still occur even if the packing material can somewhat contain the bioaerosol in contaminated; thus, suggesting some contribution to the emitted bioaerosol concentration from the media microorganisms. Laboratory scale studies by Frederickson et al. (2013) also suggested that woodchips and peat based biofilters could be net emitters of total mesophilic bacteria and gram-negative bacteria.

4.3.4 Relationship between RE and inlet concentration

The relationship between the log10 of inlet bioaerosol concentration and the removal efficiency was investigated through a linear regression analysis using data from all visits (Figure 4.3). A statistically significant relationship was found for total mesophilic bacteria and Gram negative bacteria where, p <.0005 was found for both intercept and slope coefficient; log10 of inlet concentration accounted for 35.5% and 37.0% of the explained variability in the RE for total mesophilic bacteria and Gram negative bacteria, respectively. On the other hand, a statistical relationship could not be obtained for *A. fumigatus* (intercept [p = .213]; slope coefficient [p <.0005]) and total fungi (intercept [p <.0005]; slope coefficient [p = .290]) where log 10 of inlet concentration accounted for only 15.6% and 1.8% of the explained variability for *A. fumigatus* and total fungi, respectively. This indicates a much better reliability of the regression model for total mesophilic bacteria and Gram negative bacteria removal when compared to *A. fumigatus* and total fungi. This also suggests that differences exist between the ability of the biofiltration system to deal with fungi and bacteria, and these may be related to particle size (Sanchez-Monedero et al., 2003; Frederickson et al., 2013). Figure 4.3 also shows a higher variability in performance at low inlet concentration than at high inlet concentration especially for *A. fumigatus*, total mesophilic and Gram negative bacteria. It may be that biofilters receiving low inlet concentrations perform more poorly compared to when they receive waste gas with high inlet concentrations. However, it may also be the case that there is always a small emission rate from a biofilter, but this only becomes apparent when the inlet concentration is low; when inlet concentrations are high the removal may be the dominant process, with any emissions masked by this high removal rate. Martens et al. (2001), in their study on biofiltration of a pig facility, explained that microbial loads emitted from biofilters are a summation of non-impacted microorganisms retained in the treated process air and those blown off from the surface of the media particles by the passing airstream, thus, suggesting the possibility that the species

composition of the outlet air may be different from those of the inlet even for this study. Nonetheless, this is a promising result since the reality for most facilities would be high inlet concentrations, unless they have an upstream scrubber which reduces the concentration in the air before entering the biofilter bed (Fletcher et al., 2014).

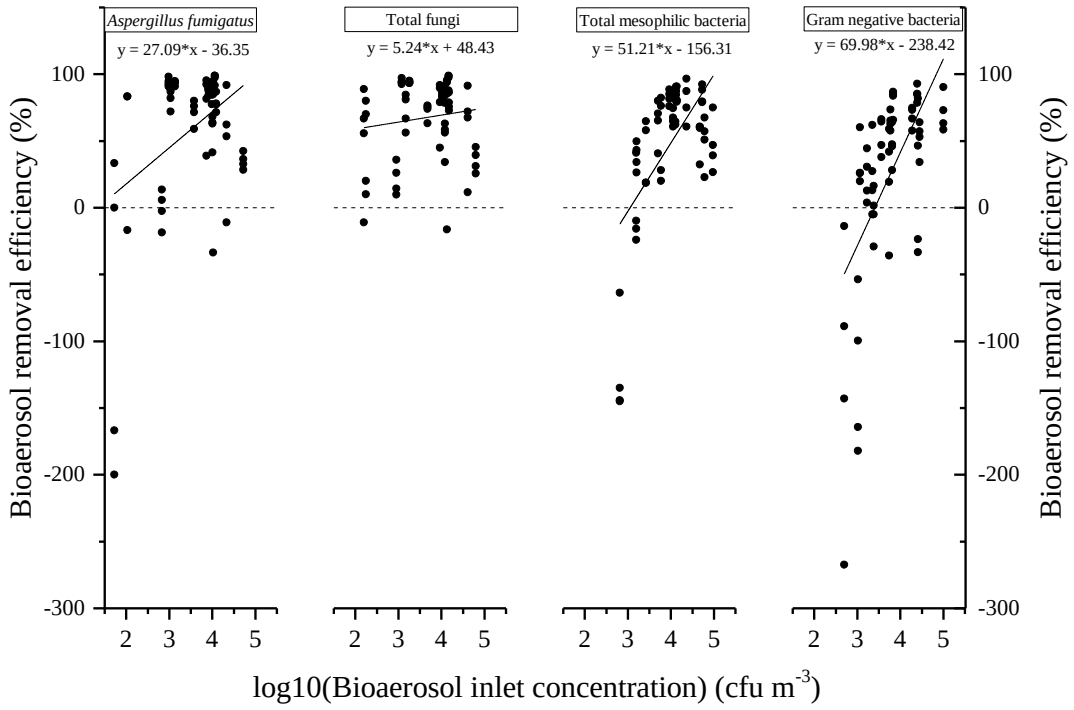


Figure 4. 3: Linear regression between log of inlet bioaerosol concentration and removal efficiency. In each case n = 64 and R^2 = 0.16 (*A. fumigatus*); 0.02 (Total fungi); 0.36 (Total mesophilic bacteria); 0.37 (Gram negative bacteria).

4.3.5 Size distribution of bioaerosols

To further evaluate the potential impact of the outlet air in a real life scenario, it was imperative to assess the size distribution of bioaerosols in biofilter exhaust air and to relate these to the tidal volume inhaled by humans. Particles collected on the various stages of the Andersen sampler represent a profile of their lung penetration potential, and so is indicative of the location of their deposition in the human respiratory tract (Andersen Instruments, 1984). Stages 1 and 2 of the sampler collect particles with an aerodynamic diameter > 4.7 μm, which equates to nasal deposition, stages 3 and 4 collect particles with an aerodynamic diameter 2.1 to 4.7 μm, which equates to bronchial deposition, and stages 5 and 6 collect particles < 2.1 μm, which equates to alveolar deposition. These correspond to the inhalable, thoracic and respirable fractions, respectively, described in TSI Incorporated (2013).

Figure 4.4 shows the variation in size distribution of bioaerosol particles collected at the different stages of the six-stage Andersen sampler (Table A.4). The size distribution was computed by taking into account all the samples taken during sampling visits 1 - 6 and 13 - 16 (when the biofilters were located indoors) from the background, inlet and all four biofilters' outlets. More than 60% of background *A. fumigatus* and total fungi particles were sized > 2.1 μm in aerodynamic diameter. On the other hand, background Gram negative bacteria had ~ 50% of particles in this range, but when considering total mesophilic bacteria, the proportion of particles in this range was slightly < 50% of a concentration of 6.2×10^2 to 1.8×10^4 cfu m^{-3}. All four biofilter outlets had ~ 40% of *A. fumigatus* (outlet concentration: 3.8×10^2 to 3.4×10^4 cfu m^{-3}) and total fungi (outlet concentration: 6.5×10^2 to 4.0×10^4 cfu m^{-3}) particles with an aerodynamic diameter < 2.1 μm, similar to their background composition. This is in contrast to the inlet samples that had ~ 20% of *A. fumigatus* particles (inlet concentration range: 3.8×10^3 to 5.3×10^4 cfu m^{-3}) and total fungi (inlet concentration range: 4.8×10^3 to 6.3×10^4 cfu m^{-3}) particles < 2.1 μm, respectively. For total mesophilic bacteria (with inlet concentration range of 5.9×10^3 to 9.6×10^4 cfu m^{-3}), the inlet samples had ~ 50% particles < 2.1 μm while the outlet samples were composed of ~ 70% of particles in this range (outlet concentration 1.9×10^3 to 5.1×10^4 cfu m^{-3}). Inlet and outlets particle size distributions for Gram negative bacteria were comparable with ~ 60% of particles < 2.1 μm, except for biofilter 4 that was slightly less than 60%.

Overall, the exhaust (outlet) air appears to have smaller particles than the air entering the system even with significantly high REs recorded when the biofiltration system was operated indoors. This could possibly result from the filter bed preferentially trapping the larger sized particles from the gas flow, and/or these may just be the size range emitted from the biofilters (Sanchez-Monedero et al., 2003)**.** However, as these pilot-scale biofilters achieved outlet concentrations predominantly in the range of 10^2 – 10^3 cfu m^{-3}, these concentrations would further be reduced (by wind dilution) downwind in full-scale applications. Williams et al. (2013), in a study to provide evidence on bioaerosol production, dispersion and potential exposures from four different composting facilities within England, reported peak total bacteria concentrations of > 10^6 cfu m^{-3} immediately downwind of the sites in comparisons to the < 10^3 cfu m^{-3} recorded upwind. However, the concentrations were observed to decline at locations further downwind of the sites which is in agreement with the view

that bioaerosol concentrations levels tend to reach background levels within 250m of their point of origin (Pankhurst et al., 2011b). Nonetheless, it may also not be possible to make this generalisation especially as these concentrations were measured at pilot scale within the waste hall, and so impact of fugitive emissions and other outdoor sources (Taha et al., 2004; Parry, 2018) were not assessed downwind of site and/or close to sensitive receptors.

For all sampling points, both *A. fumigatus* and total fungi showed a maximum particle size distribution at stage 4 corresponding to an average aerodynamic diameter of between 2.1 and 3.3μm, according the specification of the sampler. Total mesophilic bacteria and Gram negative bacteria size distribution both showed a maximum at stage 5 corresponding to an average aerodynamic diameter between 1.1 and 2.1 μm. These observations are in comparison to those of Sanchez-Monedero et al. (2003) who stated that this would imply a much better RE for the fungal spores; however, that size difference alone would not be sufficient to explain the observed difference in REs of fungi and bacteria measured in their study as well as in this study. Particle shape has also been suggested as having a key influence on particle retention (Willeke et al., 1996); and for particles with aerodynamic diameter < 1 μm, Sanchez-Monedero et al. (2003) suggested that electrostatic charge on the particles may also influence particle deposition on the biofilter media.

Bioaerosol particle size plays a key role in their dispersion in air and subsequent potential risk upon exposure via inhalation (Ferguson et al., 2017). Kell et al. (1998) argued that the potential for harmful effects by bioaerosols, upon deposition, is dependent on the number of culturable organisms, and not the culturable particles. Ferguson et al. (2017) reported that bacterial community structure and abundance were size related. They argued that since viable bacterial bioaerosols could exist either as single cells, small aggregates of cells or conglomerates of cells, then bacterial bioaerosols in stages < 3.3 μm were single cells while those in stages > 3.3 μm were either conglomerates of bacterial cells or cells attached to larger particles e.g. water droplets or dust. This latter group also showed more abundance and diversity with the highest levels found in the largest (>7 μm) size class. Thus, with predominantly lower size class in the outlet air, the potential to cause ill health from exposure would be determined by a knowledge of the species composition of the samples which was beyond the scope of this study.

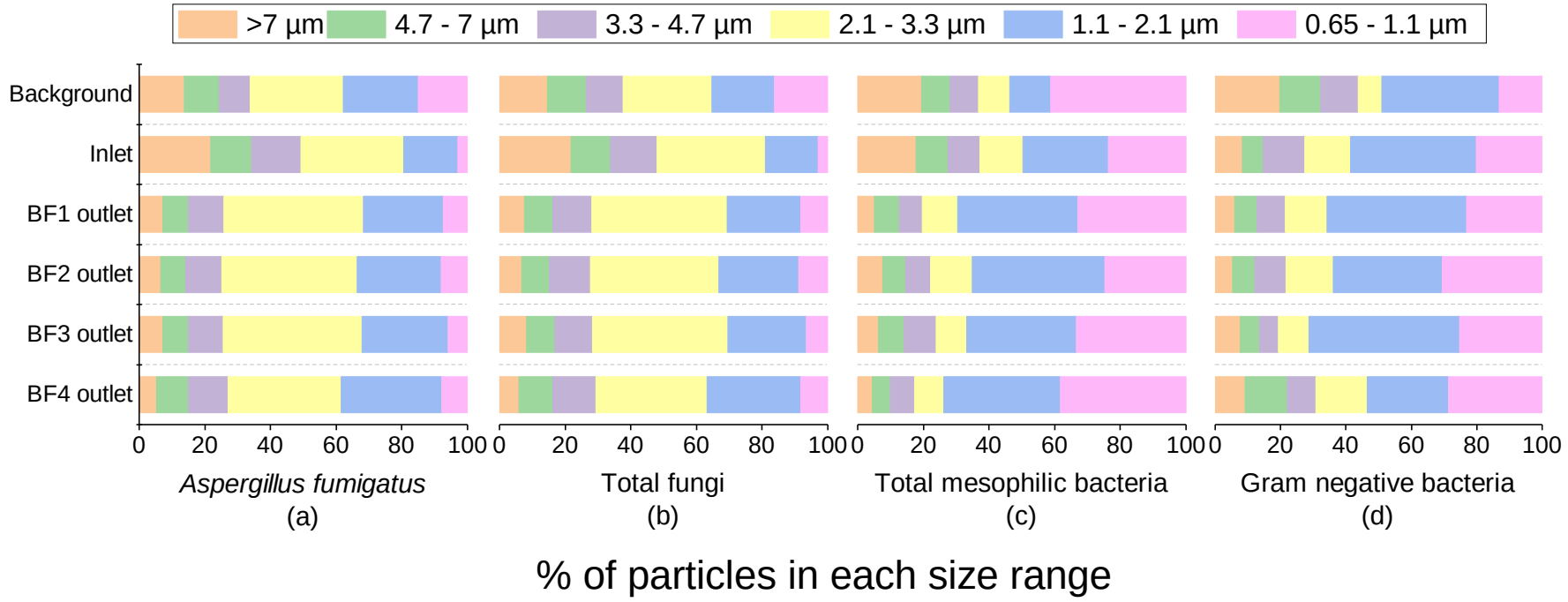


Figure 4. 4: Background, inlet and outlet percentage particle size distribution for (a) *A. fumigatus*, (b) total fungi, (c) total mesophilic bacteria and (d) Gram negative bacteria. Data based on the ten indoor sampling visits 1-6, 13-16. Outlet composition represented by BF1, BF2, BF3, BF4.

Comparison of maximum outlet to inlet respirable fractions of bioaerosols shows a ratio of 1:0.8 for both *A. fumigatus* and total fungi. This implies that more fungi particles in this size class were released from the biofilters than received with outlet concentrations of 1.4×10^4 cfu m^{-3} and 1.6×10^4 cfu m^{-3} for *A. fumigatus* and total fungi, respectively. On the contrary, total mesophilic bacteria showed a 1:1.3 ratio of outlet to inlet respirable particles while Gram negative bacteria had a 1:3.4 outlet to inlet respirable faction ratio. This indicates that the biofilters were better at controlling this fraction of bacterial particles which represent a greater human health risk as they can penetrate the respiratory system more deeply and even to the lung alveoli where gaseous exchange occurs.

Currently, there are no occupational exposure limits for bioaerosols in the UK; comparisons are usually made with other studies and publications on typical concentrations for similar facilities (Stagg et al., 2013). Malmros et al. (1992) suggested that waste workers should not be exposed to concentrations of total bacteria exceeding 5000 to 10000 cfu m^{-3} for an 8 hour working period; thus, the concentrations reported for this facility ($10^3 – 10^5$ cfu m^{-3}) present potential health risks to the workers on this site. The study by Stagg et al. (2013) on seven materials recycling facility within the UK indicated similar concentrations ($10^2 – 10^5$ cfu m^{-3}) to those observed in this study, and at those concentrations several health problems were triggered including skin symptoms, respiratory symptoms, and gastrointestinal symptoms. However, workers' health impact assessment was outside the scope of this study. Nonetheless, the respiratory-related symptoms observed in the study by Stagg et al. (2013) can be a function of the lung penetrability of the bioaerosol particles generated at the various operational activities within the waste hall. In this current study, approximately 20%, 20%, 50% and 60% of indoor concentrations of *A. fumigatus*, total fungi, total mesophilic bacteria and Gram negative bacteria, respectively, were respirable fractions (with aerodynamic diameter < 2.1μm), and so could penetrate deep into the lungs.

Tidal volume, which is the volume of air inspired or expired during a respiratory cycle (Quanjer et al., 1993), is approximately 500ml and at rest a normal human being has 12 breaths per minute (Meka and Van Oostrom, 2004). For an 8-hour working period a normal person working continuously in the vicinity of the biofilters may beinhaling 2.88 m^3 of air containing approximately 3.9×10^4 cfu of *A. fumigatus*, 4.6×10^4 cfu of total fungi, 1.0×10^5 cfu of total mesophilic bacteria and 5.0×10^4 cfu of Gram

negative bacteria respirable fractions. However, these values represent the maximum concentrations recorded during this study, and do not typify the outlet concentration ranges. Nonetheless, it is estimated that < 10% of all bioaerosols may be culturable (Blomquist, 1994; Swan et al., 2003), the remainder possibly being composed of either viable non-culturable cells or dead but intact cells which may still pose health concerns (Pearson et al., 2015). Thus, the reality might be that the actual bioaerosol concentration emitted by the biofilters may be higher than measured, and may contain species or cell components that are not detected, which still require consideration in health impact assessment (Eduard et al., 2012). Even with the measured outlet concentrations, it is expected that further reduction by microbial inactivation due to environmental stresses (such as desiccation, temperature and oxygen) (Hurst et al., 2007), and wind dilution and dispersion (as they are blown off the site) would be achieved in full-scale applications.

4.4 Section Summary

- This study shows that biofilters designed and operated for odour degradation can also achieve significant bioaerosols reduction in waste gas - 70% (35 to 97%) for *A. fumigatus*, 71% (35 to 94%) for total fungi, 68% (47 to 86%) for total mesophilic bacteria and 50 (-4 to 85%) for Gram negative bacteria - provided that the inlet concentration is high which is the case for most waste treatment facilities. Thus, they can be effective for the control of potentially pathogenic species in the emissions from these treatment facilities.
- Despite the high REs achieved, the emitted concentrations from the pilot biofilters exceeded background concentrations and the EA guideline. However, from the analysis differences may exist between the ability of the biofiltration system to deal with fungi and bacteria, as there is much more confidence with the performance for bacteria than fungi; these may be related to size differences.
- Furthermore, RE may deteriorate at low inlet concentration resulting in a net bioaerosol emitting potential of biofilters, and a proportion of the emitted bioaerosols may be originating from the microbial population colonising the media surfaces, resulting in differences in species composition between contaminated process (inlet) and treated (outlet) air samples.

- The results also suggest that gas contact time may not play a significant role in bioaerosol removal as there was no established statistical relationship over the range of EBRTs tested; however, this requires a more extensive investigation.
- Particle size distribution vary between the inlet and outlet air, with the outlet having predominantly greater proportion of smaller size particles that represent greater human health risk as they can penetrate the respiratory system more deeply and even to the lung alveoli where gaseous exchange occurs. However, the outlet concentrations were low, and further reduction would be achieved by the combined effect of wind dilution and dispersal as well as exposure to environmental stress from temperature, desiccation and oxygen in full scale applications.
- Further research with quantitative polymerase chain reaction (qPCR) and next-generation sequencing (NGS) is required to compare the species composition of both inlet and outlet air to determine whether or not new microbial populations were being emitted. Research is also required to assess the simultaneous control of odour and bioaerosols by biofilters.

Chapter 5
IMPACT OF GAS RESIDENCE TIME ON SIMULTANEOUS CONTROL OF BIOAEROSOLS AND ODOUR

5.1 Introduction

One of the design and operating parameters considered as critical for odour biofiltration is gas residence time. This is the length of time the contaminated process air is in contact with the biofilm layer on the media material necessary to allow for microbial degradation of the odorous compounds (Janni et al., 2011). This parameter is frequently assessed by the indicator empty bed residence time (EBRT) which is a function of the media cross sectional area, media depth and airflow rate across the BF, and is calculated by dividing the empty bed filter volume by the airflow rate (Devinny et al., 1999). In a review of air biofiltration, Delhoménie and Heitz (2005) stated that contaminants diffusion transfer from the gas into the biofilm, and the biodegradation reaction are the two physicochemical mechanisms that determine the efficiency of biofiltration; and that the former is slower than the latter. This implies that for there to be efficient biofiltration, EBRT must be greater than the time required for the diffusion transfer. Evidence in the literature suggest that longer EBRT results in better odour and VOCs removals (Jorio et al., 2000; Christen et al., 2002; Delhoménie et al., 2002; Martin et al., 2002; Yoon and Park, 2002), while shorter EBRTs (which imply high airflows) results in poor removals due to incomplete degradation of contaminants by the microbial population within the biofilm layer, and media desiccation by the high flow rates which strip water from the media surface, thus affecting the resident microorganisms (Delhoménie and Heitz, 2005).

From the review of literature, it is obvious that a lot of studies have focused on the criticality of EBRT on the biodegradation of odour and odorous volatiles by BFs. However, only a few studies have provided some insights into the criticality of EBRT on bioaerosol control. Although no direct comments were made, data presented by Sanchez-Monedero et al. (2003) included gas phase residence time for the BFs they studied. *A. fumigatus* REs (90 – 99%) and mesophilic bacteria REs (39 – 94%) did not appear to be related to the range of EBRTs (29 – 97s) they reported. Fletcher et al. (2014), while recommending an EBRT range of 40 s to 100s for biofiltration, argued that data from their study indicated that there was no relationship between

EBRT and bioaerosol removals especially with variabilities showing net increases up to 315% and REs close to 100% over the range of EBRTs (41 s – 84 s) in the study.

Thus, there is uncertainty in the literature especially regarding the impact of EBRT on simultaneous biofiltration of air microbial and odorous chemical contaminants from the process air of waste management facilities. Frederickson et al. (2013) agreed that there is not enough research to comment in detail on the impact of operating parameters, such EBRT, on the removal of bioaerosols. Thus, this objective investigated the impact of biofilter EBRT on the simultaneous mitigation of bioaerosol and odour emissions using a materials recovery facility (MRF) as the source of the contaminated process air. The sub-objectives were as follows: (i) to assess the variability of the inlet concentrations of bioaerosols and odour (ii) to assess the performance of three levels of EBRT – 11 s, 16 s and 70 s – in terms of REs, bioaerosol load removal (*L*) and bioaerosol removal rate (*R*) and (iii) to assess the effects of EBRT on particle size distribution between inlet and outlet samples of all BFs.

5.2 Biofilter operation to evaluate performance

Section 3.4 presents a general discussion of the operation of the biofilters throughout the study. However, this section presents more detailed information on how the biofilters were operated, maintained and assessed, and the data analyses that were carried out to fulfill the specific objectives of this particular section of the study.

5.2.1 Biofilter Operation

A total of four sampling visits were completed for this study – 13 February, 20 February, 27 February and 6 March, 2017 (Appendix B). Prior to the current study, the BF system had been in operation for nine months (3 May 2016 to 3 February 2017) during which biofiltration of bioaerosols as well as impact of inlet bioaerosols concentrations on biofiltration were assessed. Before sampling commenced, the media (Plate 5.1) in each reactor was allowed to stabilise for 10 days after the end of the previous experiments as recommended in the literature (Cabrol et al., 2012; Ralebitso-Senior et al., 2012). For this study, the woodchips used as the media were derived from previous experiments testing BF performance at the same location on the site, operated for nine months prior to the current study. Laboratory tests were

conducted to determine media characteristics (as described in section 3.3.3). Table 5.1 presents a comparison of the media characteristics in February 2017 (before commencing this study) and values obtained in May 2016.

Plate 5. 1: Woodchips used as media for this study (derived from previous experiment).

Table 5. 1: Characteristics of wood chips used for this study

Characteristics	**May 2016**	**February 2017**
Average Bulk Density (kg/m^3)	225	239.17
Average Porosity (%)	61.4	60.3
Average Water Holding Capacity (g/g dry weight)	1.16	1.12
Moisture Content (as received) (%)	30	30

For the first two visits all the pilot-scale BFs were operated with an average EBRT of 16 s (corresponding to an average airflow rate of 681 L min^{-1}). For visits 3 and 4, BF1 and BF2 were randomly allocated to the experimental groups (Coolican, 2017), such that BF1 and BF2 operated with an EBRT of 70 s (corresponding to an average airflow rate of 156 L min^{-1}) while BF3 and BF4 operated with an EBRT of 11 s (corresponding to an average airflow rate of 990 L min^{-1}). During these tests, the media moisture levels were maintained within the range 40 to 70% (wet basis). In

order to avoid media compaction and clogging, which could lead to the formation of preferential flow paths for air, the media in each BF was mixed with a shovel once every two weeks on days other than the sampling days (Sanchez-Monedero et al., 2003).

5.2.2 Biofilter Performance Assessment

Air was simultaneously sampled for odour analysis while bioaerosol sampling was being carried out. Biofilter performance was evaluated on the basis of removal efficiency as as discussed in section 3.7. In addtion, biofilter performance was also evaluated on the basis of bioaerosol load removal, *L* (Equation 5.1), and bioaerosol removal rate, *R* (Equation 5.2) (Sanchez-Monedero et al., 2003). *L* represents the number of bioaerosols removed, according to a single measurement taken at a biofilter or the maximum load of bioaerosols that could be eliminated by the biofilter, expressed in cfu m^{-3}, while *R* represents the number of bioaerosols removed per cubic metre of bed medium per unit time (hour), expressed as cfu m^{-3} h^{-1} (McNevin and Barford, 2000; Sanchez-Monedero et al., 2003).

$$L = C_{in} - C_{out} \qquad \textbf{Eq. 5. 1}$$

$$R = QL/V \qquad \textbf{Eq. 5. 2}$$

where, C_{in} = inlet concentration; C_{out} = outlet concentration.; Q = flow rate (m^3 h^{-1}) and V = volume (m^3) of the biofilter.

5.2.3 Data Analysis

All statistics were carried out on original bioaerosol and odour concentrations rather than the calculated RE. The normality of the bioaerosol concentrations was assessed using the Shapiro-Wilk test. Data for analysis was considered as mean and standard deviation. Differences in mean bioaerosol concentration for the background, BF inlet and all BF outlets were assessed using the Welch ANOVA, regardless of whether or not the assumption of normality was met. In all cases, the assumption of homogeneity of variances was violated, as assessed by Levene's test for equality of variances ($p < 0.05$) for all groups of bioaerosols assessed. For odour, there were no outliers in the data set and normality of odour concentrations was assessed using the Shapiro-Wilk test. Differences in mean odour concentration for the BF inlet and all BF outlets were assessed using ANOVA. There was homogeneity of variance as assessed by Levene's test for equality of variances ($p > 0.05$).

5.3 Results and discussion

5.3.1 Operating Conditions

Table 5.2 shows the operating conditions of the pilot scale BFs for the study period. The inlet (central plenum) temperature varied from visit to visit, and ranged from 13.0°C to 17.1°C. The BF media temperatures were left unaltered as these were within the range recommended for biofilter effectiveness (Schnelle and Brown, 2002; Chen and Hoff, 2009). Media moisture content is another key factor responsible for effective biofiltration; 75% of BF malfunctioning cases have been attributed to moisture levels (Heslinga, 1994; Morales et al., 1997), and in this study moisture levels in all BFs were carefully monitored and maintained between 50 and 70%.

Table 5. 2: Operating conditions of the biofilters (BF) during the study period

Parameter	**BF1**	**BF2**	**BF3**	**BF4**
Inlet air temperature (ºC)	13.0 – 17.1	13.0 – 17.1	13.0 – 17.1	13.0 – 17.1
Outlet air temperature (ºC)	11.8 - 16.4	11.5 – 17.1	11.7 – 16.9	11.3 – 16.5
Media temperature (ºC)	11.1 – 16.8	11.6 – 17.3	11.7 – 17.1	11.3 – 17.9
Mean EBRT (s)	16*; 70**	16*; 70**	16*; 11**	16*; 11**
Mean Airflow rate (L min^{-1})	681*; 156**	681*; 156**	681*; 990**	681*; 990**
Leachate pH range	6.83 – 7.04	6.56 – 7.38	6.77 – 7.37	6.95 – 7.21
Media moisture content (%)	58 - 68	61 - 69	54 - 64	52 - 60

* All BF operated at 16 s; ** BF1, BF2 at 70 s and BF3, BF4 at 11 s.

5.3.2 Inlet concentrations of bioaerosols and odour

Figure 5.1 shows the mean inlet concentrations of the four bioaerosol groups (error bars = standard deviations) and odour during the four sampling visits. Bioaerosol inlet concentrations varied for all four sampling visits, ranging from 1.0×10^4 to 5.3×10^4 cfu m^{-3} for *A. fumigatus*, 1.2×10^4 to 6.3×10^4 cfu m^{-3} for total fungi, 1.4×10^4 to 9.6×10^4 cfu m^{-3} for total mesophilic bacteria, 5.9×10^3 to 1.0×10^5 cfu m^{-3} for Gram negative bacteria. These concentrations are similar to those (10^3 – 10^4 cfu m^{-3}) of Fletcher et al. (2014) who reported that concentrations of bioaerosols in the process air of the full-scale facilities in their study varied from site to site and between sampling visits. They attributed these variations to the complex interactions between the specific process operating conditions, the types of waste being processed and the configuration of the air ventilation system installed on the sites.

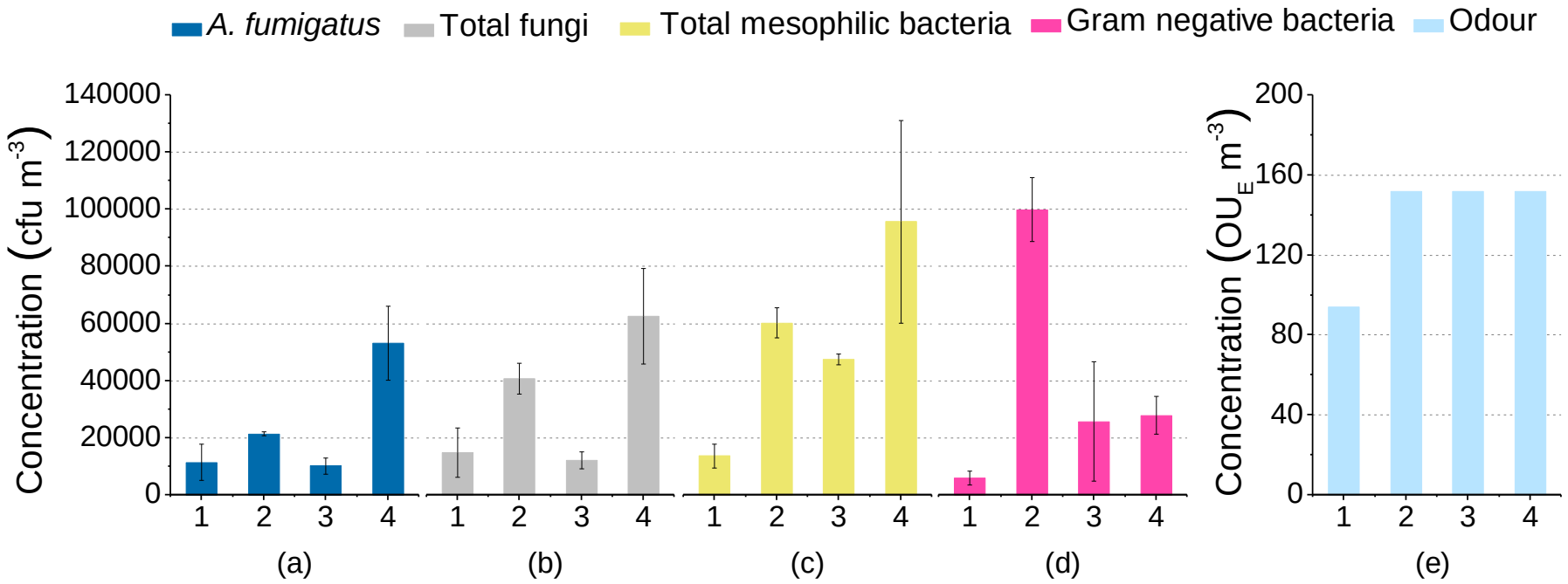


Figure 5. 1: Inlet concentrations of (a) *A. fumigatus*, (b) total fungi, (c) total mesophilic bacteria, (d) Gram negative bacteria, and (e) odour measured during the four sampling visits (n = 2 for bioaerosols; n = 1 for odour). 1 – Visit 1 (13/02/17); 2 – Visit 2 (20/02/17); 3 – Visit 3 (27/02/17); 4 – Visit 4 (06/03/17).

On the other hand, inlet odour concentrations measured for the four sampling visits were relatively low and fairly constant at 152 OU_E m^{-3}, except on visit 1 when it was 94 OU_E m^{-3}. It does appear as though variations in the process operating conditions and types of waste being processed had no obvious impact on odour concentrations. Odour control has been identified as one of the challenges of MRF managers especially when the waste materials have high amounts of organics and/or may have been left in a hot or damp environment for an extended period allowing time for decomposition and release of foul smells (Duffy, 2017). However, the odour measurements of this study are low in comparison to previous odour measurement on the exact same site by an independent consultant who reported concentrations up to 19 340 OU_E m^{-3} within the waste reception area (Gair, 2013). This variance may be attributed to the location of the pilot-biofiltration system (behind the back push wall away from the waste materials) and/or the sampling method employed. Nalophan™ bags have been reported to have non-negligible diffusion coefficients with respect to specific odorous components in air, and thus can lead to a decrease in measured odour concentration over time (Capelli et al., 2013). Capelli et al. (2013) also argued that there may be some experimental bias related with this method of sampling possibly due to the release over time of odorous substances from the polymeric film used in making the sampling bag, thus affecting the samples to be analysed.

A paired t-test was used to assess the differences in the inlet concentrations of both bioaerosols and odour when the BFs were all operated with EBRT of 16 s and when they were adjusted to run at 11 s and 70 s. Except for total bacteria ($p = 0.014$), there were no statistically significant differences ($p > 0.05$) between the inlet concentrations of all groups of bioaerosols measured between the period when the biofilters were operated at 16s (13 and 20 Feb) and when they were split to run at 11s and 70s (27 Feb and 6 March). Also, there was no statistically significant difference ($p = 0.825$) in the inlet concentrations of odour measured between both periods.

5.3.3 Effect of gas residence time on bioaerosol and odour concentration reduction

Figure 5.2 shows the boxplots (left) of the background (bioaerosols only), inlet and outlet concentrations of bioaerosols and odour from the four BFs when operated at EBRT of 16 s. It also presents the corresponding REs (right) achieved by the BFs; computed using the mean values of the inlet and outlet concentrations for the same period. The results indicate that bioaerosols were detected in the outlet of all pilot-scale BFs assessed. At this EBRT, the outlet concentration of *A. fumigatus* ranged from 1.1×10^2 to 8.5×10^3 cfu m^{-3} for BF1, 3.2×10^2 to 2.8×10^4 cfu m^{-3} for BF2, 1.1×10^2 to 1.2×10^4 cfu m^{-3} for BF3, and 1.4×10^2 to 1.9×10^3 cfu m^{-3} for BF4; translating to mean *A. fumigatus* REs of 75%, 25%, 69% and 93%, respectively, for BF1, BF2, BF3 and BF4 from a mean inlet concentration of 1.6×10^4 cfu m^{-3}.

Total fungi showed a similar trend with outlet concentrations ranging from 1.4×10^2 to 1.2×10^4 cfu m^{-3} for BF1, 3.9×10^2 to 4.5×10^4 cfu m^{-3} for BF2, 1.1×10^2 to 1.6×10^4 cfu m^{-3} for BF3, and 2.5×10^2 to 3.9×10^3 cfu m^{-3} for BF4 giving mean REs of 79%, 32%, 76% and 93%, respectively. This indicates that the BFs were capable of achieving one to two log unit reductions of fungi. On the other hand, the outlet concentrations of bacteria were much higher ($10^3 – 10^4$ cfu m^{-3}) with total mesophilic bacteria achieving REs of 61% (outlet range: 1.6×10^3 to 3.7×10^4 cfu m^{-3}) for BF1, 35% (outlet range: 1.3×10^3 to 5.6×10^4 cfu m^{-3}) for BF2, 56% (outlet range: 1.3×10^3 to 3.4×10^4 cfu m^{-3}) for BF3, and 70% (outlet range: 2.3×10^3 to 2.6×10^4 cfu m^{-3}) for BF4 from a mean inlet concentration of 3.7×10^4 cfu m^{-3}. Mean REs achieved for Gram negative bacteria were 73% for BF1, 58% for BF2, 63% for BF3, and 89% for BF4 from amean inlet concentration of 5.3×10^4 cfu m^{-3}. Welch ANOVA followed by Games-Howell post hoc analysis indicated that there were no statistically significant differences ($p > 0.05$) between the outlet concentrations of bioaerosols from the BFs operated at 16 s. Although the inlet concentrations were higher than the outlet, this was not statistically significant ($p > 0.05$). Odour concentrations measured at each BF outlet varied from visit to visit. REs achieved by the BFs were comparable and were as follows: BF1 – 51%, BF2 – 50%, BF3 – 55%, BF4 – 48%. One way ANOVA followed by Tukey post hoc analysis showed that the inlet odour concentration was significantly ($p < 0.05$) higher than the outlet of all BFs when the BFs were operated at 16 s. However, the differences in outlet odour concentrations between the four BFs were not statistically significant ($p > 0.05$).

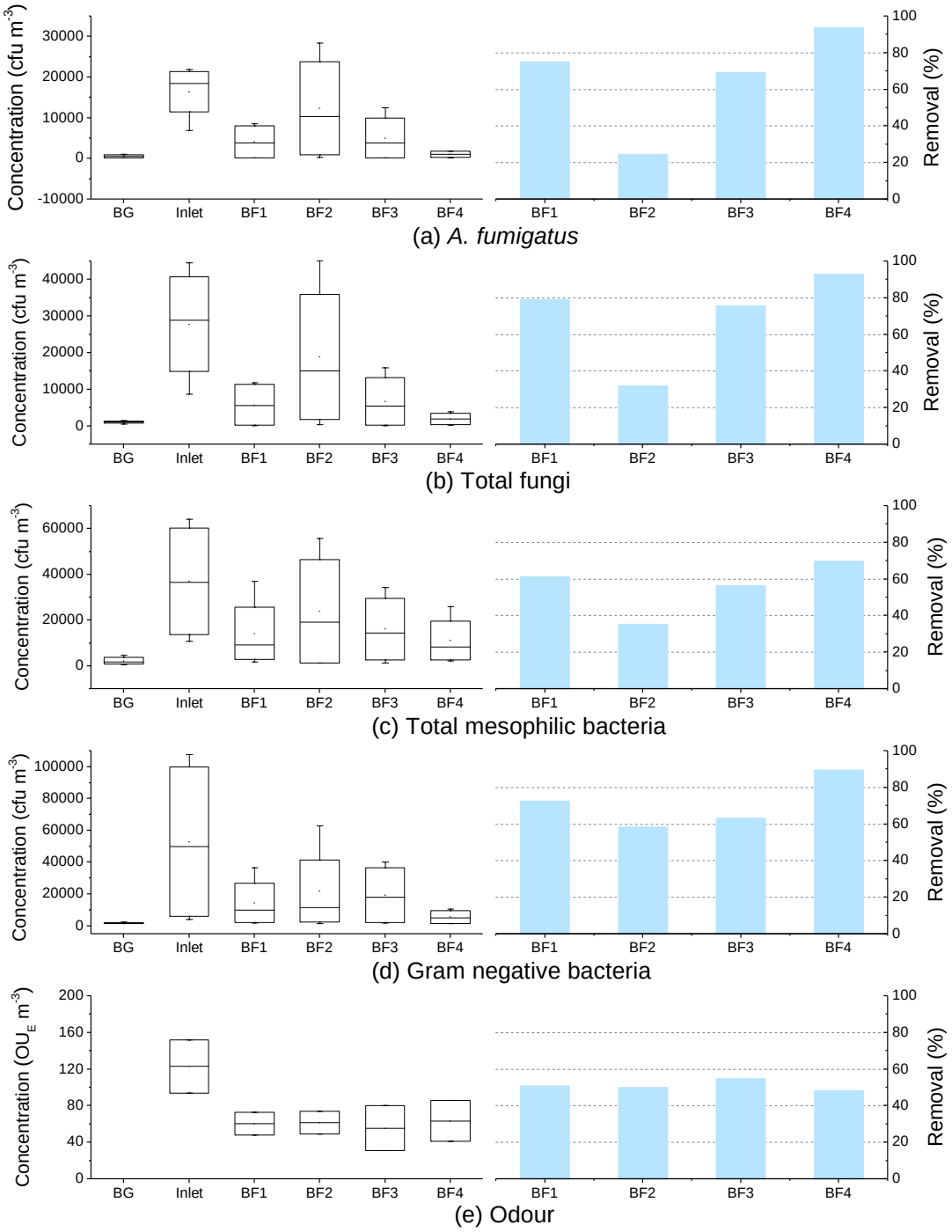


Figure 5. 2: Inlet and outlet bioaerosols and odour concentrations, n = 8 (left); and removal efficiencies (right) achieved by the biofilters when operated at 16 s. Removals computed from mean concentrations. BG = Background; BF1 = Biofilter 1, BF2 = Biofilter 2; BF3 = Biofilter 3; BF4 = Biofilter 4.

Figure 5.3 shows similar plots as Figure 5.2 but for the second part of this study when BF1 and BF2 were operated with an EBRT of 70 s while BF3 and BF4 were operated at 11 s. Generally, the achieved REs by all BFs were relatively lower compared to when they were all operated at 16 s. BF1, BF2, BF3 and BF4 achieved 34% (outlet range: 3.5×10^3 to 4.0×10^4 cfu m^{-3}), 34% (outlet range: 5.9×10^3 to 3.7×10^4 cfu m^{-3}), 46% (outlet range: 3.3×10^3 to 3.4×10^4 cfu m^{-3}), and 42% (outlet range: 3.2×10^3

to 3.9×10^4 cfu m^{-3}) mean reductions of *A. fumigatus*, respectively. Similarly, mean reductions of total fungi were 31% (outlet range: 4.6×10^3 to 4.7×10^4 cfu m^{-3}) for BF1, 32% (outlet range: 7.8×10^3 to 4.4×10^4 cfu m^{-3}) for BF2, 48% (outlet range: 4.7×10^3 to 3.7×10^4 cfu m^{-3}) for BF3, and 43% (outlet range: 4.2×10^3 to 4.2×10^4 cfu m^{-3}) for BF4 from a mean inlet concentration of 7.2×10^4 cfu m^{-3}. Similar outlet concentrations ($10^3 – 10^4$ cfu m^{-3}) were achieved by the BFs for the bacteria: BF1, BF2, BF3 and BF4 achieved mean total mesophilic bacteria removals of 46% (outlet range: 1.3×10^4 to 7.1×10^4 cfu m^{-3}), 51% (outlet range: 1.2×10^4 to 6.2×10^4 cfu m^{-3}), 70% (outlet range: 9.9×10^3 to 2.7×10^4 cfu m^{-3}), and 28% (outlet range: 2.1×10^4 to 9.6×10^4 cfu m^{-3}), while REs of 40%, 18%, 17% and 67% were respectively achieved by BF1, BF2, BF3 and BF4 for Gram negative bacteria. It does appear as though BF1 and BF2 operated with a longer EBRT (70 s) delivered slightly poorer removals of both fungi and bacteria. Fletcher et al. (2014) suggested that the BF, coded UOL07 in their study, which had pine woodchips as media and operated with the highest EBRT of 84 s performed poorest among all BFs studied, and indicated net removals of bacteria. This is in contrast to the findings of Miaskiewicz-Peska and Lebkowska (2012) that suggested that the capacity of filters to collect bacterial particles tends to decrease with increases in airflow rate (which implies decreasing EBRT). However, for this period, assumption of homogeneity of variances in bioaerosol concentrations was violated in all cases, as assessed by Levene's test for equality of variances ($p < 0.05$) for all groups of bioaerosols. Welch ANOVA indicated that there were no statistically significant differences ($p > 0.05$) between the inlet and outlet concentrations, and between the outlet concentrations for all groups of bioaerosols assessed. However, inlet concentration differed significantly ($p < 0.05$) from the background concentrations for all bioaerosols.

For the same period, mean outlet odour concentrations achieved by the BFs were 55 OU_E m^{-3} (mean RE: 64%) for BF1, 36 OU_E m^{-3} (mean RE: 76%) for BF2, 81 OU_E m^{-3} (mean RE: 47%) for BF3, and 76 OU_E m^{-3} (mean RE: 50%) for BF4. One way ANOVA followed by Tukey post hoc analysis indicated that mean inlet concentration (152 OU_E m^{-3}) differed significantly from all BF outlets ($p < 0.05$). Comparing the performance of the BFs in terms of mean outlet concentrations, BF1 showed no significant difference with BF2 ($p = 0.466$), and BF3 was not significantly different from BF4 ($p = 0.996$). However, BF2 differed significantly from BF3 ($p = 0.004$) and BF4 ($p = 0.12$); thus, suggesting that the longer EBRT of BF2 (which achieved mean

outlet concentration of 36 OU_E m^{-3}) favours odour biodegradation as established in the literature (Devinny et al., 1999; Delhoménie and Heitz, 2005; Chen and Hoff, 2009; Fletcher et al., 2014), in contrast to the short EBRT of BF3 and BF4 which showed mean outlet concentrations of 81 OU_E m^{-3} and 76 OU_E m^{-3}, respectively. However, BF1 did not differ significantly ($p > 0.05$) from BF3 and BF4.

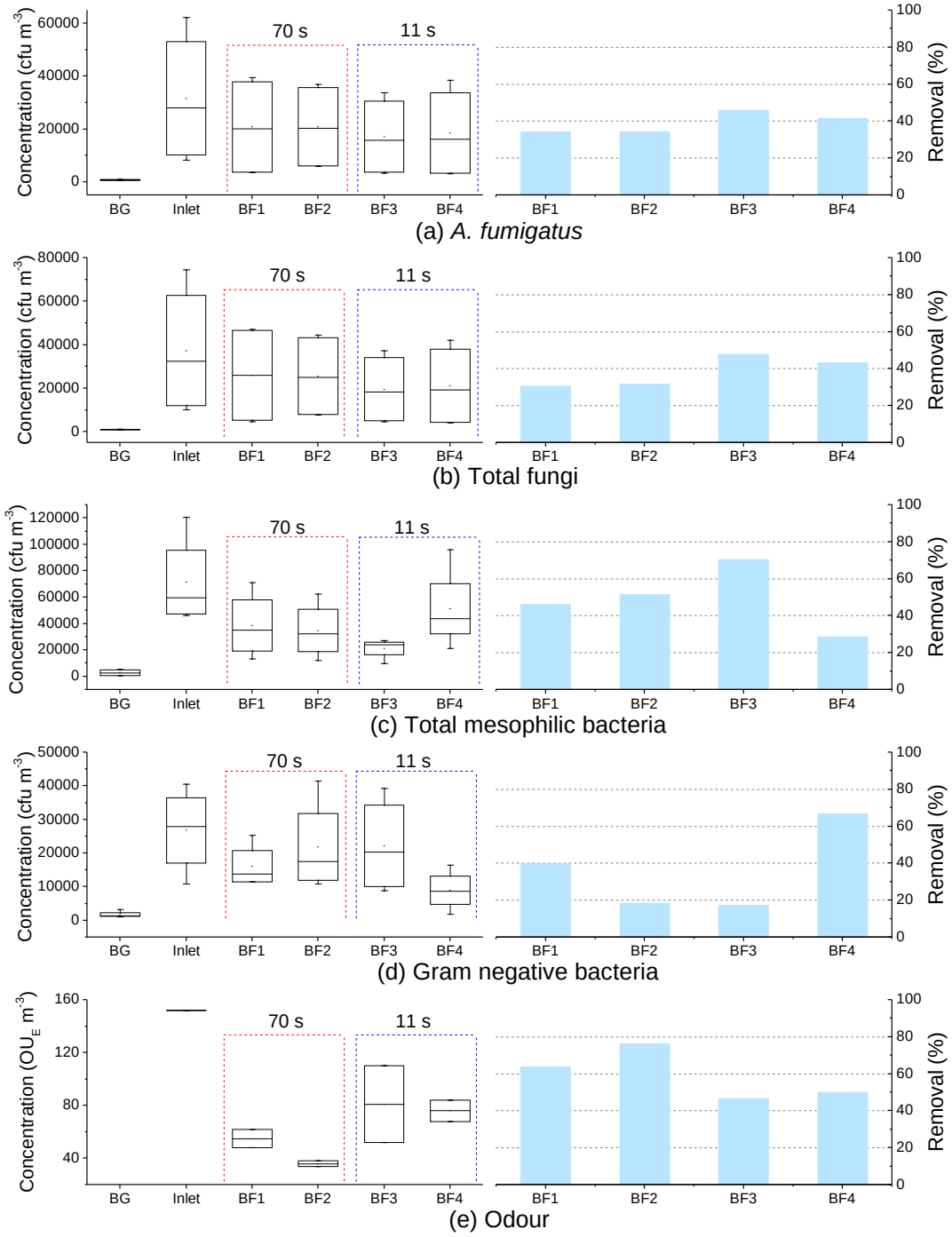


Figure 5. 3: Inlet and outlet bioaerosols and odour concentrations, n = 8 (left); and removal efficiencies (right) achieved by the biofilters – BF1 & BF2 at 70 s; BF3 & BF4 at 11 s. Removals computed from mean concentrations. BG = Background; BF1 = Biofilter 1, BF2 = Biofilter 2; BF3 = Biofilter 3; BF4 = Biofilter 4.

Table 5.3 shows the the mean bioaerosols load removal, *L* and mean removal rate, *R*, for all BFs for the two EBRT periods studied. For the period when all BFs were operated at 16 s and with the exception of BF2 which reduced by a log unit, fungal bioaerosols *L* was similar among the BFs typically in the range of 10^4 cfu m^{-3}. Similar range (10^4 cfu m^{-3}) of *L* was achieved for both groups of bacteria assessed for the same period. *L* was also very similar among all BFs evaluated for the second period with EBRTs of 70 s (BF1, BF2) and 11s (BF3, BF4), and was in the range of 10^4 cfu m^{-3} for all groups of bioaerosol assessed, except Gram negative bacteria which was one log unit lower for BF2 and BF3. The removal rate (R) was also similar among the four BFs, mostly in the range of 10^6 cfu m^{-3} h^{-1} with the bacteria showing the maximum values (10^7 cfu m^{-3} h^{-1}) for this parameter.

Table 5. 3: Biofilter performance in terms of mean bioaerosol load removal, *L* (cfu m^{-3})* and removal rate, *R* (cfu m^{-3} h^{-1})**

EBRT	**16 s (All BFs)**		**70 s (BF1 & BF2), 11 s (BF3 & BF4)**	
	L	***R***	***L***	***R***
A. fumigatus				
BF1	1.20E+04	2.75E+06	1.08E+04	5.50E+05
BF2	3.80E+03	9.35E+05	1.06E+04	5.55E+05
BF3	1.10E+04	2.55E+06	1.47E+04	4.75E+06
BF4	1.55E+04	3.45E+06	1.30E+04	4.30E+06
Total fungi				
BF1	2.20E+04	4.95E+06	1.14E+04	5.85E+05
BF2	8.85E+03	2.00E+06	1.16E+04	6.05E+05
BF3	2.10E+04	4.75E+06	1.81E+04	5.80E+06
BF4	2.55E+04	5.80E+06	1.63E+04	5.30E+06
Total mesophilic bacteria				
BF1	2.30E+04	5.10E+06	3.25E+04	1.70E+06
BF2	1.30E+04	2.95E+06	3.70E+04	1.90E+06
BF3	2.10E+04	4.70E+06	5.05E+04	1.62E+07
BF4	2.60E+04	5.80E+06	2.00E+04	6.65E+06
Gram negative bacteria				
BF1	3.85E+04	8.44E+06	1.08E+04	5.50E+05
BF2	3.07E+04	6.89E+06	4.95E+03	2.55E+05
BF3	3.34E+04	7.42E+06	4.70E+03	1.50E+06
BF4	4.72E+04	1.05E+07	1.80E+04	5.85E+06

* m^3 of inlet air; ** m^3 of media bed volume

Sanchez-Monedero et al. (2003) investigated biofiltration as a method to control airborne microorganisms released at composting facilities, and although they did not comment directly, the data they provided indicated that all the BFs in their study, with gas-phase residence times ranging from 29 s to 97 s, achieved similar *L* of 4.7×10^3

to 2.2×10^5 cfu m^{-3} for *A. fumigatus* and 9.0×10^2 to 2.4×10^5 cfu m^{-3} for mesophilic bacteria (from an inlet concentration in the range of $10^4 – 10^6$ cfu m^{-3}). Thus, it may be that the variations in EBRT may not impact on bioaerosols particle reductions in BFs especially as the range is similar between both study periods, and that the observed reductions may be the combined result of inertial deposition, diffusional (Brownian) deposition and/or flow line interception as suggested by Ottengraf and Konings (1991).

5.3.4 Effect of gas residence time on bioaerosol particle size distribution

Impact of EBRT on particle size distribution was also of interest in the current study because of the lung penetrability of certain size ranges and the potential to cause ill health in humans (Fröhlich-Nowoisky et al., 2016). In this study, each stage of the Andersen sampler used for sampling represent the part of the respiratory system where inhaled bioaerosol particles will be deposited once they penetrate the human respiratory tract (Tisch Environmental Inc., 2015). Stages 1 and 2 simulate nasopharyngeal deposition and collect particles with aerodynamic diameter > 4.7 μm (inhalable fraction); stages 3 and 4 simulate tracheobronchial deposition and collect bioaerosol particles with aerodynamic diameter 2.1 to 4.7 μm (thoracic fraction); while stages 5 and 6 simulate pulmonary or alveolar deposition, collecting < 2.1 μm (respirable fraction) (TSI Incorporated, 2013; Thomas, 2013).

A comparison of bioaerosol particle size distribution for the background, inlet and outlets of the four pilot-scale BFs assessed in this study is shown in figure 5.4. The size distributions were computed by considering all samples taken at the various points (a) when all BFs were operated with an EBRT of 16 s and (b) when the BFs were operated at 70 s and 11 s (Tables B.2 and B.3). In this study, the reference size was 3.3 μm because this is the size which differentiates particles that exist as single cells from those which exist as conglomerates of cells (Ferguson et al., 2017). Generally, the distribution of the fungal particles appear to be similar between both periods for all BFs. *A. fumigatus* and total fungi had approximately 68 – 75% and 66 – 75%, respectively, of particles in the range less than 3.3 μm when all BFs where operated at 16 s. The composition of this size range was slightly higher in the second period (EBRT 70 s and 11 s) with a range of 74 – 76% and 71 – 74%, for *A. fumigatus* and total fungi, respectively, for all BFs. Maximum proportion of particles (28 – 42%)

collected at stage 4 (corresponding to an aerodynamic diameter 2.1 to 3.3 μm). This distribution was similar to those obtained for samples taken at the inlet and at the background for both sampling periods under consideration. A Pearson's product-moment correlation was run to assess correlations of bioaerosol size distribution for the background, inlet and outlet of all BFs for the two EBRT regimes investigated (Tables 5.4 and 5.5). The strong positive correlations recorded further confirms the similarities of the distributions of bioaerosol particles at the various points sampled.

Bacterial particle size distribution was also similar for all sampling points considered between both sampling periods, although background samples indicated a slightly lower proportion of particles in the range <. 3.3 μm. Total mesophilic bacteria and Gram negative bacteria showed higher proportions of particles in the range < 3.3 μm than fungi, with 78 – 86% and 78 – 91% composition, respectively, for both periods. Contrary to the observations with fungal particle distributions, the maximum proportion of total mesophilic bacteria particles collected at stages 5 (18 – 51%) and 6 (26 – 50%). Gram negative bacteria also had maximum particles collected at stages 5 (28 – 51%) and 6 (18 – 44%) of the sampler.

The particle size distribution in this study agrees with those of Sanchez-Monedero et al. (2003) who reported maximum *A. fumigatus* and mesophilic bacteria collection on stages 4 and 5 of the six stage Andersen used in their study. This is indicative of the potential risks to exposed persons especially with the high proportions of single cell fungal and bacterial particles in the high outlet air samples which can penetrate the respiratory system more deeply and even to the lung alveoli where gaseous exchange occurs. It does appear as though variation of EBRT has no obvious impacts on both bioaerosols particle size distribution between inlet and outlet air samples collected. However, apart from *A. fumigatus* and Gram negative bacteria, it was not possible to establish whether or not the emitted particles were composed of the same species as those entering the BFs; thus, this requires a more extensive investigation with quantitative polymerase chain reaction (qPCR) and next-generation sequencing (NGS) to compare the species composition of both inlet and outlet air to determine whether or not new microbial particles are being emitted in the BF outlet samples.

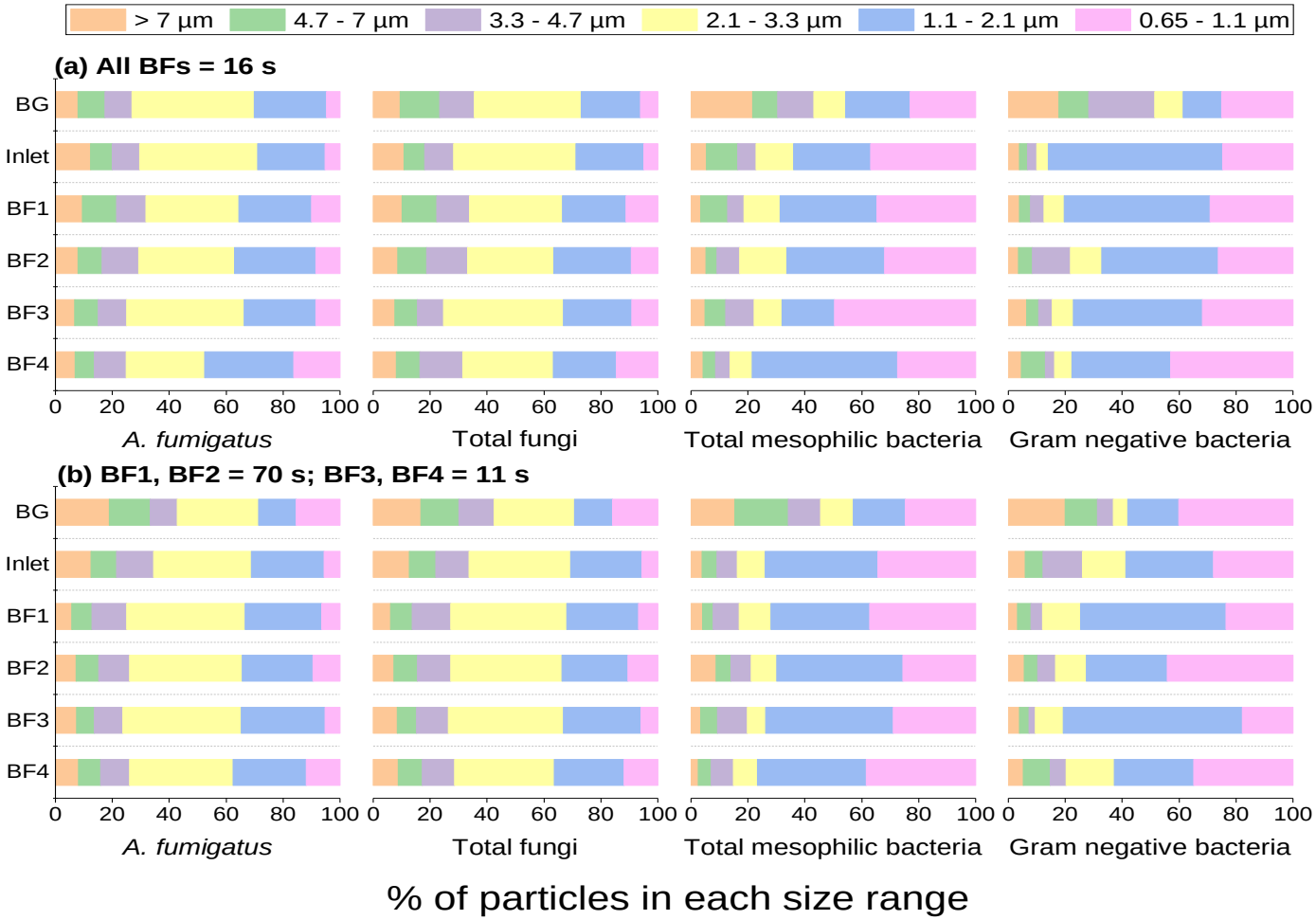


Figure 5. 4: Comparison of bioaerosol particle size distribution (a) when all biofilters were operated with an EBRT of 16 s (Data based on visits 1 & 2), and (b) when BF1 and BF2 were operated at EBRT of 70 s; BF3 and BF4 operated at EBRT of 11 s (Data based on visits 3 & 4). (Visit 1 = 13/02/17; Visit 2 = 20/02/17; Visit 3 = 27/02/17; Visit 4 = 06/03/17). BG = Background; BF – Biofilter.

Table 5. 4: Pearson correlations of the particle size distribution of the four groups of bioaerosols with all BFs operated at 16 s.

	BF1	**BF2**	**BF3**	**BF4**	**Inlet**
A. fumigatus					
BF2	.982**				
BF3	.984**	.969**			
BF4	.877*	.911*	.840*		
Inlet	.958**	.939**	.977**	.759	
Background	.982**	.961**	.993**	.792	.989**
Total fungi					
BF2	.947**				
BF3	.996**	.932**			
BF4	.946**	.932**	.956**		
Inlet	.981**	.925**	.986**	.920**	
Background	.971**	.902*	.966**	.880*	.975**
Total mesophilic bacteria					
BF2	.968**				
BF3	.804	.750			
BF4	.899*	.909*	.510		
Inlet	.973**	.921*	.910*	.780	
Background	.600	.646	.554	.654	.586
Gram negative bacteria					
BF2	.975**				
BF3	.992**	.962**			
BF4	.860*	.834*	.913*		
Inlet	.988**	.953**	.963**	.779	
Background	.068	.161	.147	.329	-.013

*.Correlation is significant at the 0.05 level (2-tailed); **.Correlation is significant at the 0.01 level (2-tailed).

Table 5. 5: Pearson correlations of the particle size distribution of the four groups of bioaerosols with BF1 & BF2 at 70 s and BF3 & BF4 at 11 s.

	BF1	**BF2**	**BF3**	**BF4**	**Inlet**
A. fumigatus					
BF2	.994**				
BF3	.993**	.988**			
BF4	.984**	.993**	.985**		
Inlet	.974**	.959**	.987**	.947**	
Background	.627	.688	.634	.661	.630
Total fungi					
BF2	.991**				
BF3	.991**	.983**			
BF4	.983**	.993**	.986**		
Inlet	.970**	.956**	.990**	.956**	
Background	.734	.791	.729	.746	.731
Total mesophilic bacteria					
BF2	.888*				
BF3	.926**	.979**			
BF4	.995**	.911*	.949**		
Inlet	.985**	.950**	.971**	.994**	
Background	.656	.501	.540	.686	.643
Gram negative bacteria					
BF2	.691				
BF3	.982**	.563			
BF4	.777	.962**	.650		
Inlet	.886*	.886*	.798	.905*	
Background	.333	.829*	.232	.698	.510

*.Correlation is significant at the 0.05 level (2-tailed); **.Correlation is significant at the 0.01 level (2-tailed).

5.4 Section Summary

- The inlet concentrations of the four bioaerosol groups studied varied from visit to visit (in the range of 10^3 – 10^4 cfu m^{-3}) possibly due to the complex interactions between the specific process operating conditions, the types and amounts of waste being processed and the configuration of the air ventilation system installed on the sites. However, such variation was not apparent with odour with concentrations remaining fairly stable (152 OU_E m^{-3}) for a greater part of the sampling period.
- The pilot scale BFs studied all achieved removals of bioaerosols in the two EBRT regimes assessed. REs of 25 – 94%, 31 – 93%, 28 – 70% and 17 – 89% were achieved for *A. fumigatus*, total fungi, total mesophilic bacteria and Gram negative bacteria, respectively, for both EBRT regimes under consideration. Although the inlet concentrations were higher than the outlet, these were not statistically significant ($p > 0.05$) especially as outlet concentrations were also in the range of 10^3 – 10^4 cfu m^{-3}.
- There were no statistically significant differences ($p > 0.05$) between the outlet concentrations of bioaerosols from the BFs for the three levels of EBRT investigated. This suggests that variations in gas residence time may not impact on bioaersosol removals by biofiltration in this study, and thus, may not be critical for bioaerosol control.
- All the BFs achieved significant reductions in odour concentrations (which were rather low in this current study), delivering REs in the range of 48 – 55% and 47 – 76 % in the first (16 s) and second (70 s, 11 s) EBRT regimes. The results demonstrated that longer EBRT (70 s) delivered significantly higher ($p < 0.05$) reductions of odour than the shorter (11 s) EBRT, implying that the longer EBRT accommodates the time required for both odorous contaminants diffusion transfer from the gas phase into the biofilm, and their subsequent biodegradation within the biofilm layer on the media materials, as established in the literature.
- Variation in EBRT does not seem to have any obvious effect on bioaerosol particle size distribution for the two EBRT regimes studied. Not only are the distributions similar for all outlet concentrations, but also for the inlet samples for all groups of bioaerosols investigated. In all cases, the outlet fungal and

bacterial particles had 66 – 76% and 78 - 91% of particles in the size range (< 3.3 μm) which represent significant risks to human health because they can penetrate the respiratory system more deeply and even to the lung alveoli where gaseous exchange occurs, and so have the potential to trigger infection.

Chapter 6
IMPACT OF BIOFILTER MEDIA MOISTURE CONTENT ON SIMULTANEOUS CONTROL OF BIOAEROSOLS AND ODOUR

6.1 Introduction

Biofiltration is a microorganism-driven degradation of odorous pollutants in process air when passed through a filter bed which harbour microorganisms in a biofilm layer (Devinny et al., 1999; Xue et al., 2018). The biofilm layer, which is supported on the filter bed material, is a critical element of biofilters as it contains the mass of microorganisms required for the degradation of odorous volatile organic compounds (VOCs) (Mudliar et al., 2010). Morales et al. (2003) stated that 90 – 95% of the biofilm layer is water which is readily available to the inherent microbial population growing on the BF media surface. Thus, the successful application of BFs for air pollution control is dependent, among other factors, on the maintenance of an optimum moisture level within the filter bed, necessary for media microorganisms to carry out normal microbial activities within the biofilm layer (Mudliar et al., 2010).

Malfunctioning of BFs have been attributed to inadequate management of media moisture content (MC) (Lith et al., 1997), and some authors have argued that up to 75% of problems encountered in biofiltration were due to poor humidity control (Heslinga, 1994; Morales et al., 1997). Excess MC have been reported to cause a reduction of media porosity which inhibits transfer of O_2 and pollutants to the biofilm, thereby development of anaerobic zones within the media and increasing pressure drop across the media bed, ultimately limiting odour biodegradation rate (Delhoménie and Heitz, 2005; Mudliar et al., 2010; Rattanapan and Ounsaneha, 2011). Fletcher et al. (2014) pointed out that under these conditions, potentially odorous metabolic end products similar to those generated by decaying organic matter can be produced. Low MC, on the contrary, causes media drying and development of paths which lead to gas flow channelling with negative impact on the microflora. After long periods of dryness, some media have been reported to become increasingly hydrophobic, and hard to re-moisten (Thompson et al., 1996). In an attempt to address the challenge of media drying, Sakuma et al. (2009) reported that the installation of a lower irrigation system in their biofilter increased toluene elimination by a factor of 1.2 to 1.7 times higher than the control setup for a gas residence time of 13.5s. They suggested this was due to the high moisture content of that portion which supported a high density

of toluene-degraders as compared to the control biofilter. In a study to evaluate moisture effects on gas-phase biofilter ammonia removal efficiency, nitrous oxide generation and microbial communities, Yang et al. (2014) reported an improved ammonia removal efficiency when moisture level was increased from 35 to 55%, and also pointed out that further increase to 63% only slightly improved ammonia removal. Several optimum ranges have been suggested in the literature depending on media type as follows; 50 – 55% for compost-based media (Goldstein, 1999), 60 – 80% for chaff of pine and perlite (Chang et al., 2004), 35 – 65% for mixture of compost and woodchips (Nicolai and Lefers, 2006b), 40 – 60% for woodchips (Chen et al., 2008b) and > 63% for woodchips (Sheridan et al., 2002b).

However, the process air from these waste management facilities also contain bioaerosols (airborne microorganisms and/or their cell components, fragments and metabolites) for which there is a growing public concern due to potential health risks (Searl, 2008; Menetrez et al., 2009; Hambach et al., 2012; Pearson et al., 2015). Unfortunately, the majority of the studies reported in the literature have focussed on the impact of media MC on odour and VOC removal and only a few on possible reduction of bioaerosols by BFs. The mechanism of odour removal (adsorption, oxidation and biodegradation) is entirely different from that of bioaerosols removal (impaction), thus it may be that the design and operating parameters vital for odour removal may not be as important for bioaerosols removal (Fletcher et al., 2014). Ottengraf and Konings (1991) argued that two mechanisms were at play in determining bioaerosol emissions from BFs – capture of bioaerosols particles due to impingement on media materials and emission of microorganisms from the biofilm layer on the media materials. They suggested that these mechanisms were impacted by the air velocity, size of biofilter media particle and bioaerosol particle size. Other studies focusing of bioaerosols (Sanchez-Monedero et al., 2003), or bioaerosols and odour removals (Martens et al., 2001) have mentioned media MC as part of the operating parameters of the BFs studied without necessarily investigating the impact of media MC on the simultaneous control of both types of air pollutants. Data provided by Fletcher et al. (2014) did not provide a clear picture on the impact of parameters (such as media MC, BF temperature, absorptivity, process air temperature and media porosity) on control of both bioaerosols and odour emissions by BFs; thus, they recommended further research to investigate the criticality of these parameters,

in order to refine operational ranges, and firmly define boundary conditions between normal and abnormal BF operation.

Thus, it is obvious that data is scarce on the effect of media MC on simultaneous biofiltration of airborne microbial and odorous chemical contaminants emitted from these facilities. This objective investigated the impact of BF media MC on the mitigation of bioaerosol and odour emissions using the MRF as the source of contaminated process air. The sub-objectives of this section of the study included (i) to assess the variability of the inlet concentrations of bioaerosols and odour within the period (ii) to assess the performance of two levels of media MC – 10 to 40% and 40 to 70% – in terms of bioaerosols and odour REs, bioaerosol load removal (L) and bioaerosol removal rate (R) and (iii) to assess the effects of media MC on particle size distribution between inlet and outlet samples of all BFs.

6.2 Biofilter operation to evaluate performance

A general discussion of the operation of the biofilters has been provided in section 3.4. However, this section presents a more detailed information on how the biofilters were operated, maintained and assessed, and the data analyses that were carried out to achieve the specific objectives of this particular section of the study.

6.2.1 Biofilter Operation

A total of four sampling visits were completed for this study - 20 March, 27 March, 3 April and 10 April, 2017 (Appendix C). Prior to the current study, the BF system had been in operation for 10 months (May 2016 to February 2017) during which the impact of EBRT and inlet concentrations on biofiltration were tested. Before sampling commenced, the media in each reactor was allowed to stabilise for two weeks after the end of the previous experiments as recommended in literature (Cabrol et al., 2012; Ralebitso-Senior et al., 2012). For this study, the woodchips used as the median were derived from previous experiments testing BF performance at the same location on the site, operated for 10 months prior to the current study. Laboratory tests to determine media characteristics (as described in section 3.3.3) indicated that the characteristics remained fairly constant as shown in table 5.1.

The BFs were randomly allocated to the media MC experimental groups (Coolican, 2017), such that BF1 and BF2 operated with a media MC range of 40 – 70% while

BF3 and BF4 operated with a media MC range of 10 – 40%. Irrigation was controlled by the look and feel method suggested by Janni et al. (2011) whereby moisture levels were monitored to ensure dampness across ½ to ¾ way through the media depth. Water supply was varied among the BFs such that less was supplied to BF3 and BF4 to achieve the desired range (10 – 40%). Throughout the study period, the airflow rates into each biofilter were regulated by means of 50 mm ball valves to ensure that all the biofilters were maintained at an average empty bed residence time (EBRT) of 16 s ($p > 0.05$) corresponding to an average airflow rate of 681 L min^{-1}. In order to avoid media compaction and clogging, which could lead to the formation of preferential flow paths for air, the media in each BF was mixed with a shovel once every two weeks on days other than the sampling days (Sanchez-Monedero et al., 2003).

6.2.2 Data Analysis

For this study, biofilter performance assessments were done according to section 5.2.2. All statistics were carried out on original bioaerosol and odour concentrations rather than the calculated RE. For bioaerosols, the data set contained outliers and these were included in the analysis because attempts in transforming or using alternative tests (e.g. Kruskal-Wallis H test) still showed these outliers. The normality of the bioaerosol concentrations was assessed using the Shapiro-Wilk test. Data for analysis was considered as mean and standard deviation. Differences in mean bioaerosol concentration for the background, BF inlet and all BF outlets were assessed using the Welch ANOVA, regardless of whether or not the assumption of normality was met. In all cases, the assumption of homogeneity of variances was violated, as assessed by Levene's test for equality of variances ($p < 0.05$) for all groups of bioaerosols assessed. For odour, there were no outliers in the data set and normality of odour concentrations was assessed using the Shapiro-Wilk test. Differences in mean odour concentration for the BF inlet and all BF outlets were assessed using ANOVA. There was homogeneity of variance as assessed by Levene's test for equality of variances ($p > 0.05$).

6.3 Results and Discussion

6.3.1 Operating Conditions

A summary of the operating conditions during the sampling visit is presented in table 6.1. Average media MCs of 60%, 60%, 23% and 19% were measured in BF1, BF2, BF3 and BF4, respectively, in order to fulfil the objective of this study. The temperature at the central plenum was taken as the inlet temperature, and varied from visit to visit within 14.7ºC to 18.5ºC; well within the range recommended by Fletcher et al. (2014) for inlet air temperature. The media temperature also varied between the BFs and from visit to visit with a mean of 15.4ºC, 15.8ºC, 16.4ºC and 16.7ºC for BF1, BF2, BF3 and BF4, respectively. No adjustments were made to alter the media temperature especially as these were within the optimal levels (10 to 40ºC) recommended for biological treatment systems (Schnelle and Brown, 2002). Clark et al. (2004), in a study to assess the impact of media temperature and supplemental nutrients in the performance of a pilot-scale BF, reported that differences in treatment temperature (range of 15ºC to 30ºC) had no apparent influence on odour removal ($p = 0.05$). The impact of waste hall temperature was not considered as Liu et al. (2017) observed that there was no obvious influence of indoor temperature on the performance of biofilters in their study. The EBRT was adjusted to 16 s in all BFs, giving an average flow rate of 681 L min^{-1} in each BF. Devinny et al. (1999) recommended EBRT range of 15 – 60 s for BFs used for waste air treatment. These pilot-scale BFs were operated without any supplementary attempts to alter the pH which were well within the recommended range (6.5 – 8) for most of the operation (Wani et al., 1997; Schnelle and Brown, 2002)

Table 6. 1: Operating conditions of the biofilters (BF) during the study period

Parameter	**BF1***	**BF2***	**BF3****	**BF4****
Inlet air temperature (ºC)	14.7 – 18.5	14.7 – 18.5	14.7 – 18.5	14.7 – 18.5
Outlet air temperature (ºC)	13.8 – 16.6	13.3 – 17.7	13.7 – 17.5	13.0 – 17.4
Media temperature (ºC)	12.9 – 17.5	12.2 – 17.2	15.5 – 17.9	15.0 – 18.0
Mean EBRT (s)	16	16	16	16
Mean Airflow rate (L min^{-1})	681	681	681	681
Leachate pH range	6.18 – 7.01	6.29 – 7.00	6.41 – 7.12	6.64 – 7.15
Media moisture content (%)	46 - 69	52 - 70	12 - 38	13 - 31

* with media MC range of 40 – 70%; ** with media MC range of 10 – 40%

6.3.2 Inlet concentrations of bioaerosols and odour

Figure 6.1 shows the inlet concentrations of the four bioaerosol groups and odour concentrations (mean and standard deviation) during the four sampling visits. Inlet concentrations varied from visit to visit, ranging from 9.2×10^3 to 1.9×10^4 cfu m^{-3} for *A. fumigatus*, 1.3×10^4 to 2.5×10^4 cfu m^{-3} for total fungi, 1.1×10^4 to 4.3×10^4 cfu m^{-3} for total mesophilic bacteria, 1.4×10^4 to 2.2×10^4 cfu m^{-3} for Gram negative bacteria and 179 to 489 OU_E m^{-3} for odour. This observation is consistent with the observation of Fletcher et al. (2014) who reported that concentrations of bioaerosols and odour in the process air of the full-scale facilities in their study varied from site to site and between sampling visits. They attributed these variations to the complex interactions between the specific process operating conditions, the types of waste being processed and the configuration of the air ventilation system installed on the sites. The volume of waste processed at the tipping area was greatest on visit 4, requiring two front-end loaders (usually one in operation at the tipping area) and one dinosaurus shredder to move and clear out the waste materials. This may have had a bearing on the highest concentrations measured for *A. fumigatus* ($1.9 \times 10^4 \pm 17198$ cfu m^{-3}), total fungi ($2.5 \times 10^4 \pm 21206$ cfu m^{-3}) and total mesophilic bacteria ($4.3 \times 10^4 \pm 14005$ cfu m^{-3}) for the same visit. Searl (2008) opined that there was increased waste handling activities were associated with increased bioaerosol exposures. However, this trend was not observed with Gram negative bacteria and odour (which was least) on visit 4. Overall, the concentration of bacteria (especially total mesophilic bacteria) in the inlet air were higher than the concentrations of the fungi. Fletcher et al. (2014) also made this observation, noting that the concentrations of bacteria (total and Gram negative) in the process air were significantly higher than those of *A. fumigatus*. As with the previous study (Chapter 5), the inlet odour concentrations were considered relatively low especially for this particular MRF when compared to concentrations reported by an independent consultant (Gair, 2013). This difference is thought to be due to the same reasons as outlined in section 5.3.2.

A Pearson's product-moment correlation was run to assess the relationship between the four groups of bioaerosols and odour (Table 6.2). All correlations were not statistically significant ($p < 0.05$). Inlet odour concentrations showed negative correlations with all groups of bioaerosols – *A. fumigatus* (-0.690), total fungi (-0.390), total mesophilic bacteria (-0.836) and Gram negative bacteria (-0.129).

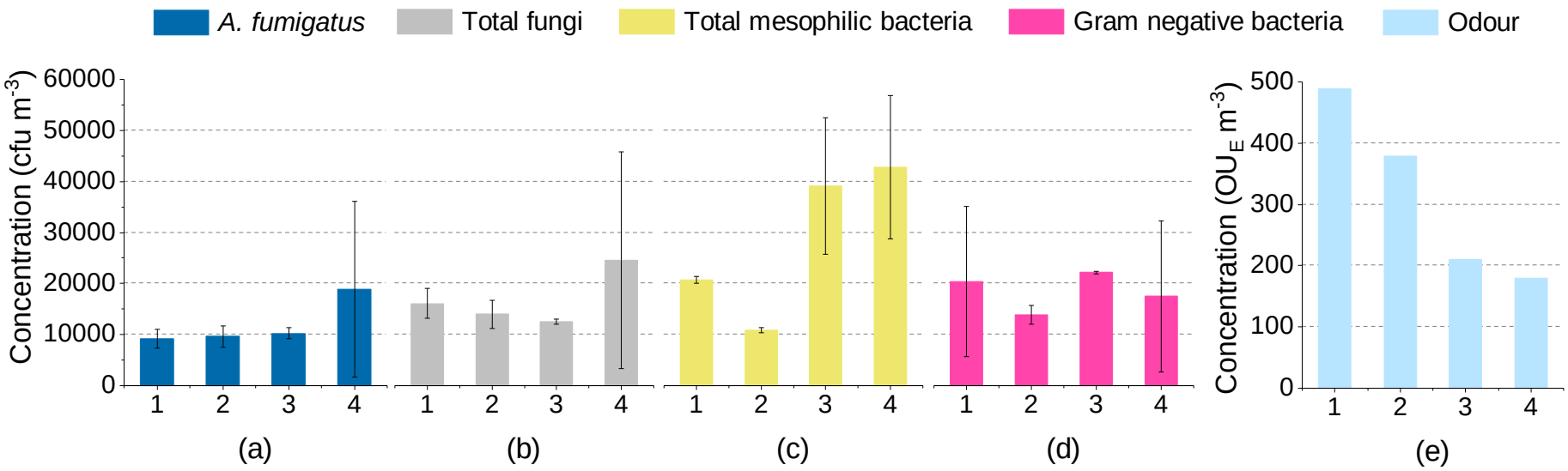


Figure 6. 1: Inlet concentrations of (a) *A. fumigatus*, (b) total fungi, (c) total mesophilic bacteria, (d) Gram negative bacteria, and (e) odour measured during the four sampling visits (n = 2). 1 – Visit 1 (20/03/17); 2 – Visit 2 (27/03/17); 3 – Visit 3 (03/04/17); 4 – Visit 4 (10/04/17).

Comparing the bioaerosol groups, *A. fumigatus* showed a strong positive correlation with total fungi and with mesophilic bacteria, and a weak negative correlation with Gram negative. Gram negative bacteria also indicated a moderate positive correlation with total mesophilic bacteria. However, due to the limited samples collected, it cannot be concluded whether or not these correlations are substantial. These findings are contrary to data reported by Fletcher et al. (2014) who found no relationship (positive or negative) between the concentrations of *A. fumigatus*, total bacteria or Gram negative bacteria in the process air of the full-scale biowaste treatment facilities investigated in their study.

Table 6. 2: Pearson correlations for the concentrations of the four groups of bioaerosols and odour

	A. fumigatus	**Total fungi**	**Total mesophilic bacteria**	**Gram negative bacteria**
Total fungi	.934			
Total mesophilic bacteria	.686	.496		
Gram negative bacteria	-.155	-.203	.550	
Odour	-.690	-.390	-.836	-.129

6.3.3 Effect of media moisture content on bioaerosol and odour concentration reduction

Figure 6.2 shows the boxplots (left) of the background (bioaerosols only), inlet and outlet concentrations of bioaerosols and odour from the four BFs. It also presents the corresponding REs (right) achieved by the BFs; computed using the mean values of the inlet and outlet concentrations. Contrary to the study by Tymczyna et al. (2007) who reported no detection of bioaerosols for two of the three media materials investigated, bioaerosols were detected in the outlet air of all BF assessed in this study.

The results of this study showed that all pilot-scale BFs in this study achieved some odour reduction, and that odour concentrations emitted by each pilot-scale BF varied between sampling visits. BF1 and BF2, operated with a media MC of 40 – 70%, delivered higher mean odour removals of 63% and 44%, respectively. The odour REs of BF3 and BF4 (both operated at media MC 10 – 40%) were 34% and 42%, respectively. One way ANOVA followed by Tukey post hoc analysis indicated significant differences ($p < 0.05$) between the inlet concentration (315 OU_E m^{-3}) and the outlet concentration of BF1 (116 OU_E m^{-3}), BF2 (175 OU_E m^{-3}) and BF4 (181 OU_E m^{-3}). There was no significant difference ($p = 0.098$) between the inlet and outlet

concentrations of BF3 (208 OU_E m^{-3}). Even though the outlet concentrations of BF1 and BF2 showed lower outlet concentrations, these were not significantly different ($p > 0.05$) from the outlet concentrations of BF3 and BF4. Also, there were no significant differences ($p > 0.05$) between BF1 and BF2 as well as between BF3 and BF4. As with the upper media MC range in this study, Leson and Winer (1991) recommended that the media MC of BFs should be maintained at 40 – 60% using inlet air humidifiers or spray irrigation at the BF surface, and that non-optimum levels may cause the media to dry out and may result in breakthroughs of incompletely treated process air. These agree with the study by Pinnette et al. (1994) who reported loss of biological degradation of odorous compounds when MC was below 40%. Also, Hong and Park (2004) suggested that BFs operated most effectively when media MC was within a slightly higher range of 50 to 70%. Ottengraf and Van den Oever (1983) cited in Fletcher et al. (2014) maintained their BFs at MC range of 50 to 70% and reported that the packing media lost its microbial activity at lower water levels, while higher levels promoted the development of anaerobic zones within the bed.

Generally, all BFs were able to achieve one log unit reduction in the concentration of fungi (except BF3 which showed one log unit higher for total fungi). Outlet concentration of *A. fumigatus* ranged from 1.2×10^3 to 3.8×10^3 cfu m^{-3} for BF1, 1.3×10^3 to 2.8×10^3 cfu m^{-3} for BF2, 2.7×10^3 to 8.7×10^3 cfu m^{-3} for BF3, and 1.8×10^3 to 4.7×10^3 cfu m^{-3} for BF4. These translated to mean *A. fumigatus* REs of 79%, 85%, 65% and 78% for BF1, BF2, BF3 and BF4, respectively, from a mean inlet concentration of 1.2×10^4 cfu m^{-3}. For total fungi, the outlet concentrations ranged from 1.7×10^3 to 5.9×10^3 cfu m^{-3} for BF1, 2.0×10^3 to 4.5×10^3 cfu m^{-3} for BF2, 4.0×10^3 to 1.1×10^4 cfu m^{-3} for BF3, and 2.6×10^3 to 7.8×10^3 cfu m^{-3} for BF4 giving mean REs of 76%, 83%, 67% and 76%, respectively.

The outlet concentrations of bacteria were slighter higher than fungi, and were in the range of 10^3 to 10^4 cfu m^{-3}. For total mesophilic bacteria, the achieved mean REs were 71% (outlet concentration range: 2.3×10^3 to 1.7×10^4 cfu m^{-3}) for BF1, 76% (outlet concentration range: 1.6×10^3 to 1.2×10^4 cfu m^{-3}) for BF2, 74% (outlet concentration range: 2.2×10^3 to 1.9×10^4 cfu m^{-3}) for BF3, and 76% (outlet concentration range: 1.6×10^3 to 1.4×10^4 cfu m^{-3}) for BF4. Mean REs achieved for Gram negative bacteria were 68% (outlet concentration range: 1.8×10^3 to 1.3×10^4 cfu m^{-3}) for BF1, 67% (outlet concentration range: 1.8×10^3 to 1.6×10^4 cfu m^{-3}) for BF2, 66% (outlet concentration range: 3.9×10^3 to 8.2×10^3 cfu m^{-3}) for BF3, and 77% (outlet

concentration range: 1.7×10^3 to 9.7×10^3 cfu m^{-3}) for BF4. Welch ANOVA followed by Games Howell post hoc test indicated that the inlet bioaerosols concentrations were significantly ($p < 0.05$) higher than the outlet concentrations for all groups of bioaerosols assessed.

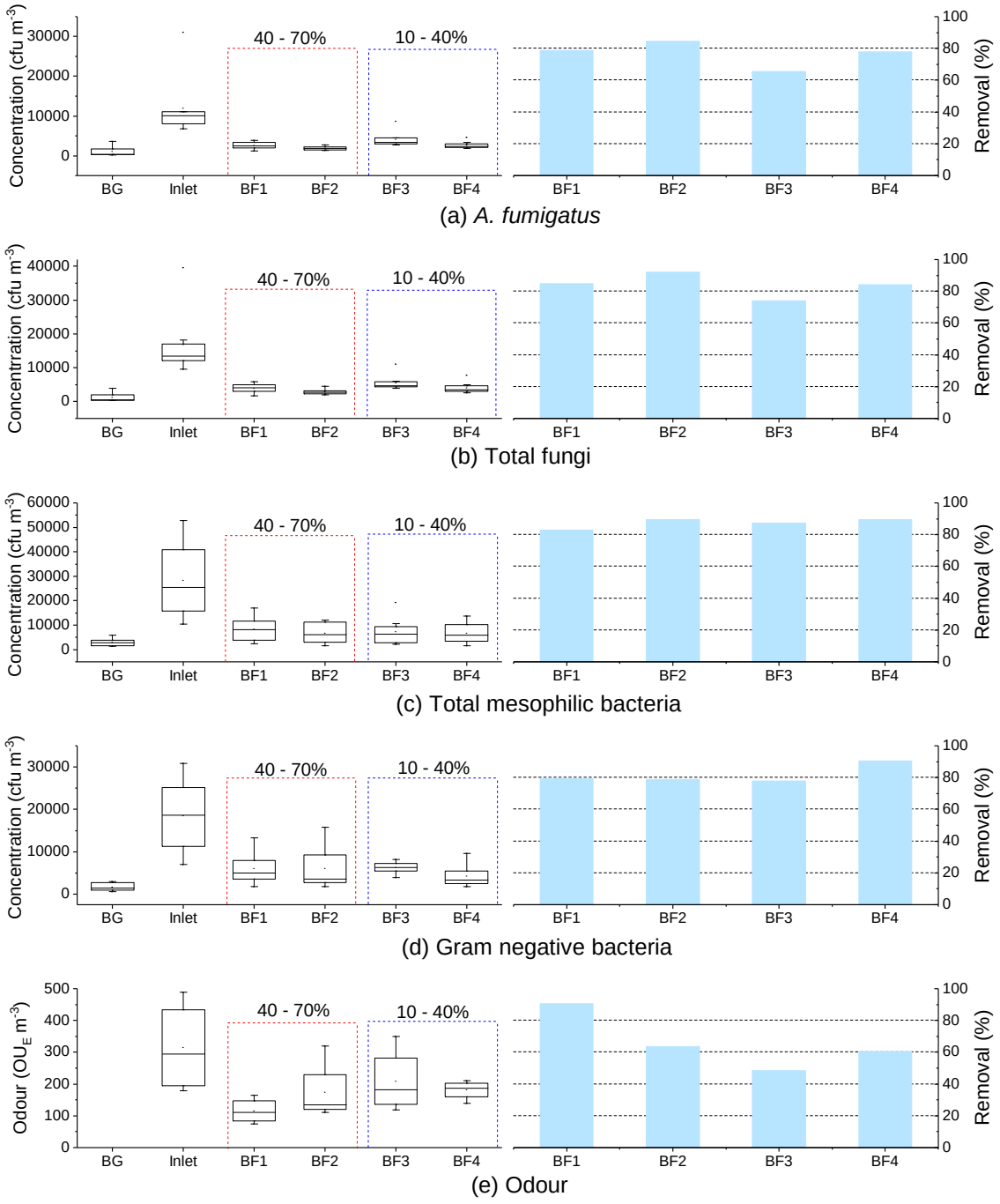


Figure 6. 2: Inlet and outlet bioaerosols and odour concentrations n = 8 (left); and removal efficiencies (right) achieved by the biofilters – BF1 & BF2 at MC of 40 – 70%; BF3 & BF4 at MC of 10 – 40%. Removals computed from mean concentrations. BG = Background; BF1 = Biofilter 1, BF2 = Biofilter 2; BF3 = Biofilter 3; BF4 = Biofilter 4.

However, there were no significant differences ($p > 0.05$) among the outlet concentrations of all groups of bioaerosols. Nevertheless, the mean *A. fumigatus* outlet concentrations of BF3 (4.1×10^3 cfu m^{-3}) and BF4 (2.7×10^3 cfu m^{-3}) were higher than those of BF1 (2.6×10^3 cfu m^{-3}) and BF2 (1.9×10^3 cfu m^{-3}). This trend was also observed for total fungi with 4.0×10^3 cfu m^{-3}, 2.9×10^3 cfu m^{-3}, 5.6×10^3 cfu m^{-3} and 4.1×10^3 cfu m^{-3} for BF1, BF2, BF3 and BF4, respectively. This trend did not appear to be the case for the bacteria (both total mesophilic and Gram negative bacteria). With the exception of BF4 which indicated mean RE of 77% for Gram negative, all other BFs achieved 71 – 76% for total mesophilic bacteria and 66 – 68% for Gram negative bacteria.

Table 6.3 summarises the mean bioaerosol load removal, *L* and mean removal rate, *R*, for all BFs. Load removal was very similar among all BFs evaluated, and was in the range of 10^4 cfu m^{-3} for all groups of bioaerosol assessed, except *A. fumigatus* which as one log unit lower. The removal rate (*R*) was also similar among the four BFs, typically in the range of 10^6 cfu m^{-3} h^{-1} with total mesophilic bacteria showing the maximum values for this parameter. Although no direct comments were made, data provided by Sanchez-Monedero et al. (2003) showed that all BFs in their study, with media MC not less than 50%, achieved similar *L* of 4.7×10^3 to 2.2×10^5 cfu m^{-3} for *A. fumigatus* and 9.0×10^2 to 2.4×10^5 cfu m^{-3} for mesophilic bacteria (from an inlet concentration in the range of $10^4 – 10^6$ cfu m^{-3}). *R* was also similar for both groups of bioaerosols in their study. Thus, it may be that the degree of dampness of a media may not confer any obvious advantage in terms of the media capacity to trap bioaerosol particles (especially as both low and high media MC in this study achieved similar *L* and *R*), and that the observation may just be the interplay of forces which effect particle capture (i.e. inertial deposition, diffusional (Brownian) deposition and/or flow line interception) as suggested by Ottengraf and Konings (1991). Sanchez-Monedero et al. (2003) further pointed to this fact by arguing that *R* depended more on *L* rather than on biofilter design (and in this case media MC) which could be more important with high inlet bioaerosol concentration.

Hong and Park (2004) argued that woodchips have a porous structure which allows for air movement, however that this also makes it a poor material for odour biofiltration. This suggests that the poor odour performance of BF3 and BF4 may be the combined effect of this poor biofiltration capability of woodchips and the low media MC which inhibits microbial degradation of odorous pollutants in the process;

while at the same time providing surfaces for bioaerosols particles impaction within the bed.

Table 6. 3: Biofilter performance in terms of mean bioaerosol load removal and removal rate

	Load removal, *L* (cfu m^{-3})*	**Removal rate, *R* (cfu m^{-3} h^{-1})****
A. fumigatus		
BF1	9.55E+03	2.13E+06
BF2	1.03E+04	2.25E+06
BF3	7.78E+03	1.75E+06
BF4	9.45E+03	2.08E+06
Total fungi		
BF1	1.28E+04	2.88E+06
BF2	1.39E+04	3.15E+06
BF3	1.13E+04	2.50E+06
BF4	1.28E+04	2.88E+06
Total mesophilic bacteria		
BF1	1.99E+04	4.50E+06
BF2	2.15E+04	4.85E+06
BF3	2.10E+04	4.75E+06
BF4	2.15E+04	4.85E+06
Gram negative bacteria		
BF1	1.25E+04	2.83E+06
BF2	1.25E+04	2.80E+06
BF3	1.21E+04	2.78E+06
BF4	1.43E+04	3.20E+06

* m^3 of inlet air; ** m^3 of media bed volume

6.3.4 Effect of media moisture content on bioaerosol particle size distribution

Apart from the interest on achieving simultaneous reduction of odour and bioaerosols concentration, it was also important to assess the impact of media MC on particle size distribution between inlet and outlet air samples. This interest is because of the relationship of between particle size and lung penetration. Fröhlich-Nowoisky et al. (2016) argued that the inhalation and deposition of bioaerosols in various regions of the human respiratory tract may cause allergic or toxic responses in humans, and that particle deposition is a function of particle properties, airway morphology and breathing characteristics. The six stages of the Andersen sampler used for this study indicate the location where inhaled bioaerosol particles will be deposited once they penetrate the human respiratory tract (Tisch Environmental Inc., 2015). Stages 1 and 2 simulate nasopharyngeal deposition and collect particles with aerodynamic diameter

> 4.7 μm (inhalable fraction); stages 3 and 4 simulate tracheobronchial deposition and collect bioaerosol particles with aerodynamic diameter 2.1 to 4.7 μm (thoracic fraction); while stages 5 and 6 simulate pulmonary or alveolar deposition, collecting < 2.1 μm (respirable fraction) (TSI Incorporated, 2013; Thomas, 2013).

Figure 6.3 shows a comparison of bioaerosol particle size distribution for the background, inlet and outlets of the four pilot-scale BFs assessed in this study. The size distributions were computed by considering all samples taken at the various points indicated throughout the four sampling visits (Table C.2). The focal size was 3.3 μm because this differentiates particles which exist as single cells from those which exist as conglomerates of cells (Ferguson et al., 2017). Generally, the inlet samples of *A. fumigatus* and total fungi had approximately 39% of the particles less than 3.3 μm while total mesophilic bacteria and Gram negative bacteria had approximately 56% and 51% of particles, respectively, in this size range. The size distribution of *A. fumigatus* and total fungi particles in the outlet samples of all BFs was similar; with 64% (BF1), 64% (BF2), 68% (BF3), 68% (BF4) of *A. fumigatus* particles, and 60% (BF1), 63% (BF2), 65% (BF3), 64% (BF4) of total fungi particles in the range < 3.3 μm. For both fungal groups assessed, the maximum proportion of particles (43 – 50%) collected at stage 4 (corresponding to aerodynamic diameter 2.1 to 3.3 μm); thus, agreeing with the findings of Sanchez-Monedero et al. (2003) who also reported maximum collection of *A. fumigatus* particles at stage 4 of the six stage Andersen sampler used in their study.

With regards to the bacteria, outlet air samples from all BFs were also similar and indicated a much higher proportion of particles in the range less than 3.3 μm. BF1, BF2, BF3 and BF4 had approximately 75%, 79%, 75% and 75%, respectively, of total mesophilic bacteria particles in this reference range. Outlet samples of BF1, BF2, BF3 and BF4 also indicated approximately 78%, 80%, 73% and 85%, respectively, of Gram negative bacteria particles <3.3 μm. As with the study of Sanchez-Monedero et al. (2003), maximum bacteria collection was on stage 5 (corresponding to an aerodynamic diameter 1.1 to 2.1 μm) containing 31 – 40% of total mesophilic bacteria particles and 35 – 51% of Gram negative bacteria particles.

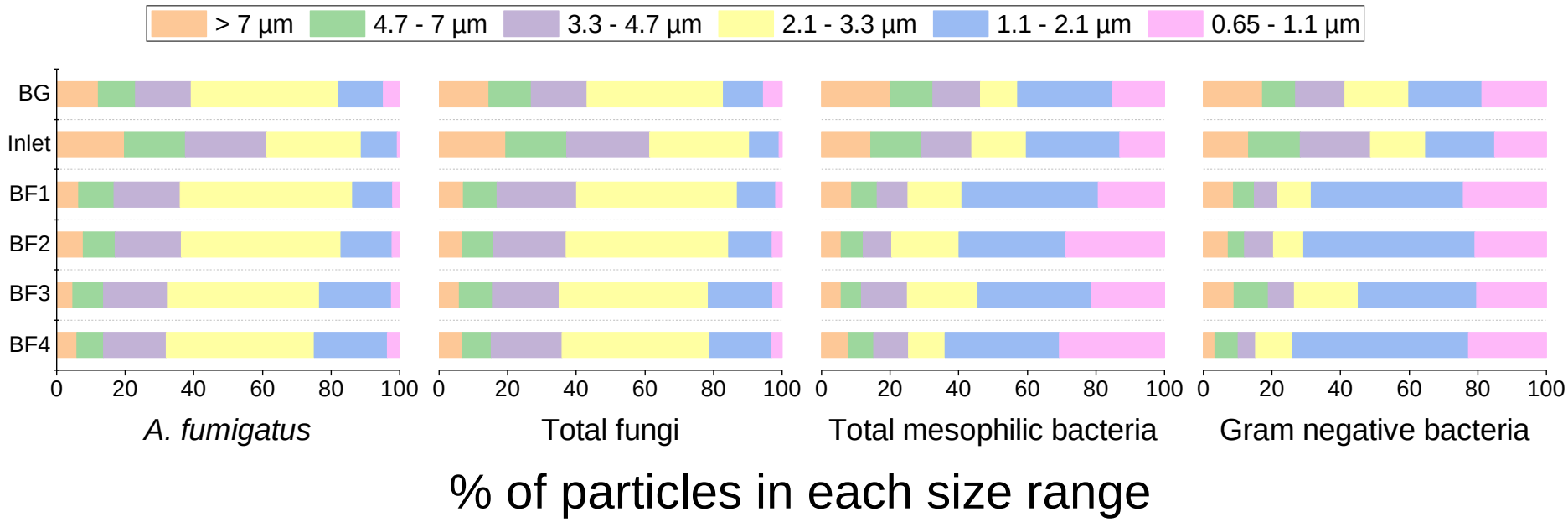


Figure 6. 3: Comparison of bioaerosol particle size distribution for samples obtained at the background, inlet and outlet of all four biofilters (Data based on all four visits). BF1 and BF2 operated with media moisture range of 40 – 70% while BF3 and BF4 operated with media moisture range of 10 – 40%. BG = Background; BF – Biofilter.

Despite the high REs measured in this study, the outlet concentrations for all groups of bioaerosols were still in the range of 10^3 to 10^4 cfu m^{-3} which is higher than the EA's precautionary guidance of 500 cfu m^{-3}, 1000 cfu m^{-3} and 300 cfu m^{-3} for *Aspergillus fumigatus*, total bacteria and Gram negative bacteria, respectively (Environment Agency, 2010). This implies that these levels may still be of health concern and especially because they are composed of high proportions of particles that can penetrate deep into the lower respiratory tract where they can cause infectivity (Andersen, 1958; Fröhlich-Nowoisky et al., 2016; Ferguson et al., 2017).

Overall, there are differences in the particle size composition between the inlet and outlet air samples in this study. However, not only are the outlet bioaerosols concentrations similar among the pilot-scale BFs studied, their compositions are also similar, suggesting that media dampness may only be vital for odour removal and not so much for bioaerosols removal. The high composition of small particles in the outlet air could possibly result from the filter bed preferentially trapping the larger sized particles from the gas flow, and/or these may just be the size range emitted from the biofilters (Ottengraf and Konings, 1991; Sanchez-Monedero et al., 2003). It may also be that the larger particles, which are conglomerates, become disintegrated upon impaction on the media particles resulting in smaller sized particles which remain in the airstream and become detected at the outlet (Jankowska et al., 2000; Miaskiewicz-Peska and Lebkowska, 2012).

6.4 Section Summary

- Inlet concentrations of the four bioaerosol groups and odour concentrations varied from visit to visit possibly due to the complex interactions between the specific process operating conditions, the types and amount of waste being processed and the configuration of the air ventilation system installed on the sites.
- The two groups of BFs studied achieved removals of both odour and bioaerosols. Although not statistically significant ($p > 0.05$), differences did exist in odour removal performance between the two groups, with BF1 and BF2 operated with media MC of 40 – 70% consistently showing better removals (odour RE range of 44 – 63%) than BF3 and BF4 operating with media MC of 10 – 40% (with odour RE range of 34 – 42%).

- There appears to be no media MC dependent differences ($p > 0.05$) in bioaerosol reductions especially as the outlet concentrations from all BFs were in the range of $10^3 – 10^4$ cfu m^{-3}, with REs of 65 – 85%, 67 – 83%, 71 – 76% and 66 – 77% achieved for *A. fumigatus*, total fungi, total mesophilic bacteria and Gram negative bacteria, respectively.
- Media MC also does not appear to have any obvious effect on bioaerosol particle size distribution between the two groups of BFs studied. Outlet samples had similar composition; all BF outlet air samples had 64 – 68% of A. fumigatus, 60 – 65% of total fungi, 75 – 79% of total mesophilic bacteria and 73 – 85% of Gram negative bacteria particles in the range less than 3.3 μm. These further suggest that the media MC may only be vital for odour reduction, and not so much for bioaerosols reduction.

Chapter 7
IMPACT OF BIOFILTER MEDIA DEPTH ON BIOAEROSOL CONTROL IN A MATERIALS RECOVERY FACILITY

7.1 Introduction

The successful application of biofilters as an air pollution control system for waste management and animal housing facilities is dependent on certain key operating parameters. Parameters such as media types (Tymczyna et al., 2007), empty bed residence time(EBRT) (Nicolai and Janni, 1999), moisture content (Nicolai and Janni, 2001a; Chen et al., 2009; Liu et al., 2017), media porosity (Nicolai and Janni, 2001a), pH (Barzgar et al., 2017), inlet concentration, temperature (Yoon and Park, 2002) and media depth (MD) (Liu et al., 2017) have all been shown to play keys roles in biofilter performance in these facilities.

The role of MD in the control of odour and odorous volatile organic compounds emissions has been well studied. In a study to investigate biofiltration in a pig unit, Sheridan et al. (2002a) showed that a 0.5 m depth biofilter containing >20 mm woodchips as a filter material could deliver odour removal efficiencies (REs) of 85%, 92.5% and 91.3% when the moisture content (wet basis) was maintained at 64 ± 4%, 69 ± 4%, and 69 ± 4%, respectively. The same biofilter also achieved ammonia REs of 73%, 85% and 87% at the same moisture contents, respectively. Lim et al. (2012) argued that higher MD achieved higher REs for H_2S and NH_3; however, they also noted that doubling the media depth from 0.127 m to 0.254 m did not necessarily double the REs for these gases. Furthermore, Kafle et al. (2015) investigated the effectiveness of two down-flow wood bark-based biofilters in mitigating odour, NH_3 and H_2S from a confined swine nursery barn, and concluded that for successful mitigation of these gases a minimum MD of 0.254 m and EBRT of 2 to 3 s are required. They further suggested that for a MD of 0.127 m, a high moisture content and EBRT greater than 3 s are required to achieve high REs.

There is a general lack of data on the impact of biofilter MD in controlling bioaerosol emissions. A study by Tymczyna et al. (2007) evaluated the impact of three biofilter media types on the removal of bioaerosols from the ventilation system exhaust from a chicken hatchery. The biofilters had depths ranging from 1.2 to 1.4 m. The study concluded that all biofilter media were highly effective in removing Gram negative bacteria (RE >99%), moderately effective in removing dust (RE = 81.6 – 87.4%), and

only slightly effective in removing endotoxins (RE = 11.1 – 51.5%). It is still unclear how MD impacts on biofiltration of airborne microbial contaminants emitted from waste management facilities.

One of the objectives of this study was to investigate the impact of biofilter MD on the mitigation of bioaerosol emissions using a materials recovery facility (MRF) as the source of the contaminated process air. The sub-objectives included (i) to assess the variability of the inlet concentrations (ii) to assess the performance of two media depths – 0.5 m and 0.25 m – in terms of REs, bioaerosol load removal (L) and Bioaerosol Removal Rate (R) and their effects on the particle size distribution between inlet and outlet samples. The two media depths were selected to reflect levels frequently reported in literature and because these were technically feasible for this pilot study.

7.2 Biofilter operation to evaluate performance

Section 3.4 presents a general discussion of the operation of the biofilters throughout the study. This section presents a more detailed information on how the biofilters were operated and maintained, and the data analyses that were conducted to fulfil the specific objectives of this particular study.

7.2.1 Biofilter Operation

A total of four sampling visits were completed - May 2, May 8, May 15 and May 22 (Appendix D). Prior to the current study, the BF system had been in operation for a year (May 2016 to April 2017) during which the impacts of EBRT, media moisture content and inlet concentrations on biofiltration were tested. Before sampling commenced, the media in each reactor was allowed to stabilise for three weeks after the end of the previous experiments as recommended in literature (Cabrol et al., 2012; Ralebitso-Senior et al., 2012). For this study, the woodchips used as the media were derived from previous experiments testing BF performance at the same location on the site, operated for a year prior to the current study. Laboratory tests to determine media characteristics (as described in section 3.3.3) indicated that they remained fairly constant as shown in table 5.1.

The biofilters (Figure 7.1) were randomly allocated to the media depth experimental groups (Coolican, 2017), such that BF1 and BF2 operated with a media depth of 0.5

m (total media volume of 0.1815 m^3 per reactor) while BF3 and BF4 operated with a media depth of 0.25 m (total media volume of 0.09075 m^3 per reactor). The air flow rates into each biofilter were adjusted using 50 mm ball valves to ensure that all the biofilters were maintained at a mean EBRT of approximately 16 s ($p > 0.05$) corresponding to air flow rates in the range of 668.1 to 684.9 L min^{-1} for BF1 and BF2, and 336.1 to 342.5 L min^{-1} for BF3 and BF4. Throughout the study period, the moisture contents were maintained within the range of 40 – 60%.

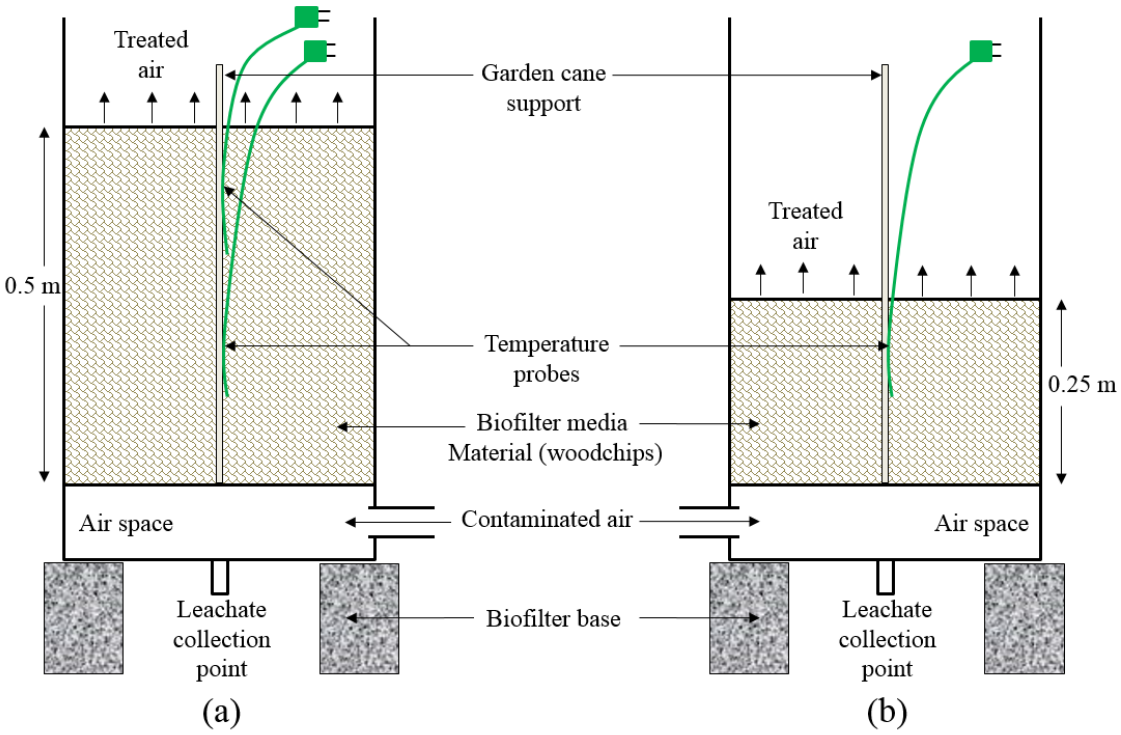


Figure 7. 1: Schematic of the pilot-scale biofiltration system (a) with 0.5 m media depth, and (b) with 0.25 m media depth (not drawn to scale).

The stabilisation (acclimation) period was 3 weeks (10 April to 2 May) at the end of the previous experiments. During this period the biofilters were adequately monitored and maintained to ensure the media moisture contents were restored to the range of 40 – 60% and the centrifugal fan was not clogged, thus ensuring continuous feed of air into the pilot-biofilters which were in continuous operation from 10 April to 2 May, 2017. The biofilters were monitored twice a week during which visual inspections were done to ensure the filter beds were uniform within the two groups for optimal performance.

7.2.2 Data Analysis

Comparison of inlet versus outlet concentrations of both 0.5 m and 0.25 m media depths BFs as well as outlet concentrations of 0.5 m depth BFs versus outlet concentrations of 0.25 m depth BFs were conducted using paired t-test and a probability of 0.05 or less was considered significant.

7.3 Results and Discussion

7.3.1 Operating Conditions

Table 7.1 provides a summary of the operating conditions during each sampling visit. The measured temperature within the waste hall varied between days with the highest of 20.7ºC measured on the fourth sampling visit. The temperature at the central plenum was taken as the inlet. The impact of waste hall temperature was not considered as Liu et al. (2017) observed that there was no obvious influence of indoor temperature on the performance of biofilters in their study. Although, some authors have acknowledged the difficulty with achieving homogenous moisture distribution in biofilter media (Akdeniz et al., 2011; Lim et al., 2012; Liu et al., 2017), media moisture content in this study was maintained within the 40 – 60% range (wet basis) as recommended for optimum biofilter performance (Lim et al., 2012).

Biofilters are a method of biological air treatment systems that utilise populations of microorganisms to convert certain organic and inorganic pollutants into compounds and/or forms that are less toxic and/or odourless. These microorganisms thrive at a pH range of 6.5-8 which must be maintained within the internal environment of the biofilter (Wani et al., 1997; Schnelle and Brown, 2002). However, to evaluate their performance for bioaerosol control, in this study the biofilters were operated without any supplementary attempts to alter the pH which was well within the recommended range. Also, no adjustments were made to the media temperature especially as these were within the optimal levels (10 – 40 ºC) recommended for biological treatment systems (Schnelle and Brown, 2002).

Table 7. 1: Operating conditions of the biofilters (BF) during the study period

Parameter	Visit 1		Visit 2		Visit 3		Visit 4	
	0.5 m	0.25 m	0.5 m	0.25 m	0.5 m	0.25 m	0.5 m	0.25 m
Mean temperature in waste hall (ºC)	16.9	16.9	14.8	14.8	15.7	15.7	20.7	20.7
Mean inlet air temperature (ºC)	18.4	18.4	16.6	16.6	18.5	18.5	22.0	22.0
Mean outlet air temperature (ºC)	16.9	17.0	14.0	14.5	16.8	16.8	21.0	21.1
Mean media temperature (ºC)	16.8	16.1	14.9	13.6	18.1	17.7	20.3	21.0
Mean EBRT (s)	16.2	16.0	15.9	16.2	16.0	16.1	16.3	15.9
Mean Airflow rate (L min^{-1})	672.2	340.3	684.9	336.1	680.6	338.2	668.1	342.5
Leachate pH range	6.60 – 6.64	6.84 – 6.85	7.02 – 7.14	7.14 – 7.28	7.24 – 7.39	7.16 – 7.28	7.27 – 7.29	7.19 -7.28
Media moisture content (%)	41 - 52	41 - 46	41 -55	42 -51	43 - 57	44 - 48	41 - 59	45 - 48

7.3.2 Inlet concentrations of bioaerosols

There were variabilities in the inlet concentrations measured on site from visit to visit, and this is consistent with the findings of Fletcher et al. (2014) for full scale biofilters. Generally, bacteria concentrations were higher than fungi concentration. The mean inlet concentrations ± standard errors (calculated using the all values for the four sampling visits) were $3.6 \times 10^4 \pm 8069$ cfu m^{-3} for *A. fumigatus*, $4.8 \times 10^4 \pm 11729$ cfu m^{-3} for total fungi, $7.8 \times 10^4 \pm 18475$ cfu m^{-3} for total mesophilic bacteria, and $3.3 \times 10^4 \pm 5720$ cfu m^{-3} for Gram negative bacteria as shown in Figure 7.2. Searl (2008) argued that bioaerosol emissions from waste processes vary over time in composition and in release rate even at the exact same location, and that even parallel comparisons of sampling devices may demonstrate considerable variability in performance, particularly in respect to viable microorganisms which may be destroyed to greater or lesser degrees by different sampling devices and protocols.

For enclosed composting facilities where waste reception is handled within a hall where material is accepted, processed and composted within vessels, bioaerosol concentrations sometimes exceed $10^{7\text{-}8}$ cfu m^{-3} (Schlegelmilch et al., 2005a). However, MRFs such as this are known to be fairly clean relative to other type of waste management facilities, and do not have the levels of organic dust and odour found in facilities such as in-vessel composting (IVC) (Surrey County Council, 2017). Stagg et al. (2013) confirmed that concentrations of bacteria and fungi within MRFs may be between $10^4 – 10^5$ cfu m^{-3}, and occasionally may measure as high as $>10^5$ cfu m^{-3} similar to levels reported for animal housing facilities, with identified species including *A. fumigatus*, certain bacteria and endotoxins, agents which are known to have harmful effects on human health. The inlet concentration range reported by Stagg et al. (2013) is in agreement with the findings of this study. Provided there is adequate supply of nutrients and water, microorganisms are known to grow on any materials where they aid to breakdown organic materials (Frederickson et al., 2013). This facility receives large volumes of household and municipal waste most of which are delivered as black bags containing items such as plastic bags, plastic films, fruit nets, animal waste and food wastes. Some of these waste may have been stored for longer periods allowing time for growth of microorganisms which become airborne during waste handling (Gladding and Gwyther, 2017).

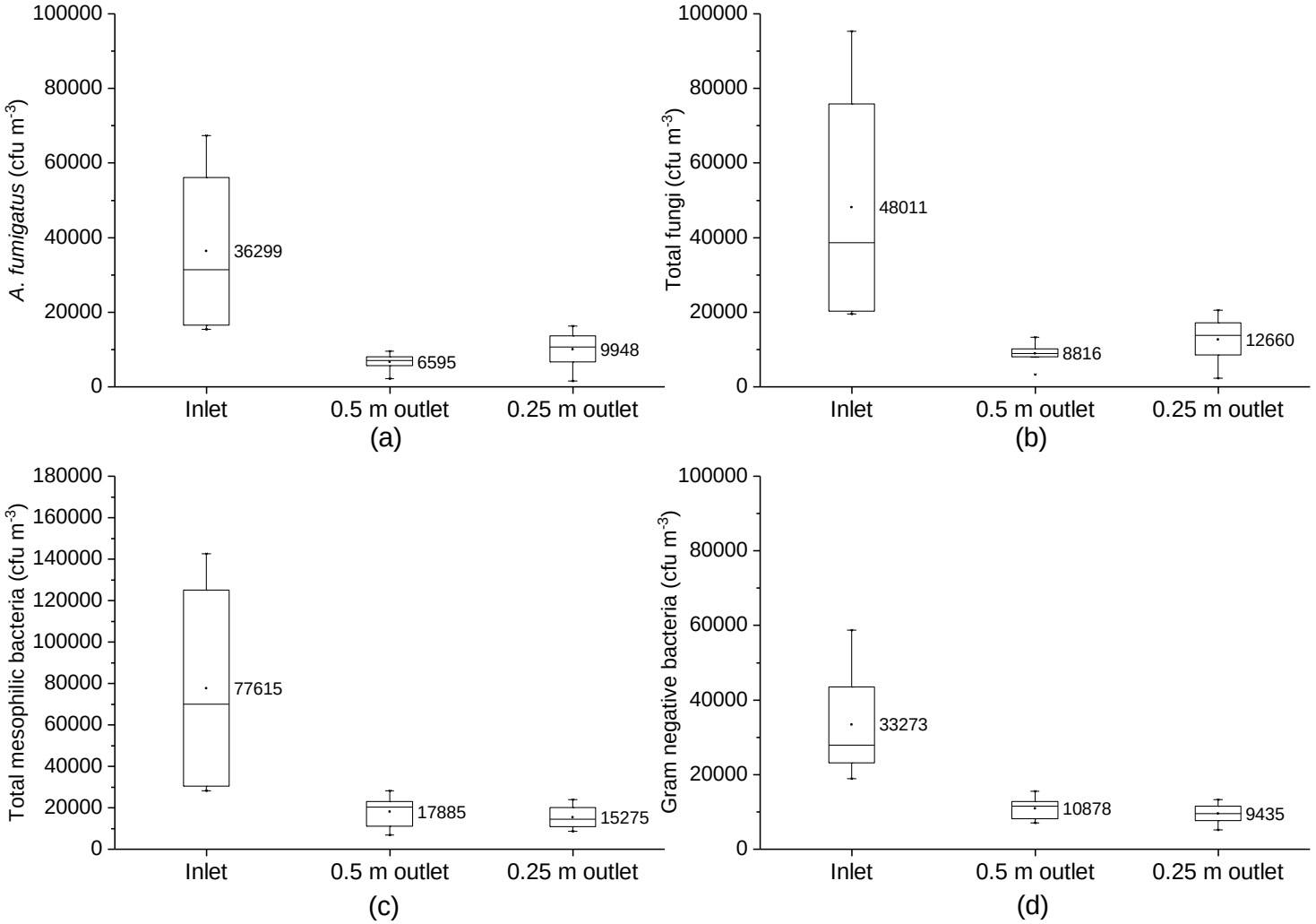


Figure 7. 2: Comparison of mean bioaerosol concentrations at the inlet and outlet of 0.5 m and 0.25 m media depth biofilters; (a) *A. fumigatus*, (b) Total fungi, (c) Total mesophilic bacteria, and (d) Gram negative bacteria. (Mean values indicated in figures; n = 8).

Fletcher et al. (2014) suggested that there may be no relationship between waste type or treatment system and the levels of bioaerosols emitted, and that the concentration emitted may be due to a complex mix of specific activities being carried out and the waste characteristics at the time of sampling. Thus, the combination of advanced mechanical separation techniques and manual sorting in this MRF agitate these waste leading to aerosolisation of these microorganisms.

7.3.3 Media depth effects on bioaerosol removals and particle size distribution

An assessment of the outlet concentrations from the two groups of biofilters showed that there were no statistically significant differences between the outlet concentrations of BF1 and BF2, and between BF3 and BF4 for all the groups of bioaerosols. There were also no statistically significant differences ($p > 0.05$) when comparing BF1 to BF3 and BF4 as well as when comparing BF2 to BF3 and BF4. Thus, the mean of the two BFs in each group were used for further analysis. Figure 7.2 also shows the mean outlet bioaerosol concentrations achieved by the 0.5 m and 0.25 m depth BFs for the study. For the 0.5 m BFs, the mean concentrations emitted were $6.6 \times 10^3 \pm 790$ cfu m^{-3} for *A. fumigatus*, $8.8 \times 10^3 \pm 998$ cfu m^{-3} for total fungi, $1.8 \times 10^4 \pm 2669$ cfu m^{-3} for total mesophilic bacteria, and $1.1 \times 10^4 \pm 1049$ cfu m^{-3} for Gram negative bacteria. On the other hand, the 0.25 m depth BFs showed mean outlet concentrations of $1.0 \times 10^4 \pm 1752$ cfu m^{-3} for *A. fumigatus*, $1.3 \times 10^4 \pm 2124$ cfu m^{-3} for total fungi, $1.5 \times 10^4 \pm 2026$ cfu m^{-3} for total mesophilic bacteria, and $9.4 \times 10^4 \pm 968$ cfu m^{-3} for Gram negative bacteria.

A paired t-test to compare the inlet and outlet concentrations indicated that inlet concentrations were statistically significantly higher ($p < 0.05$) than the outlet concentrations of all bioaerosol groups for both media depths 0.5 m and 0.25 m, indicating that all four biofilters had an effect on the concentration of bioaerosols. Differences in the outlet concentrations of the 0.5m and 0.25 media depth BFs were also assessed using the paired t-test. The mean outlet concentration from BF1 and BF2 with media depth 0.5 m were lower than those from BF3 and BF4 with media depth of 0.25 m for *A. fumigatus* ($p = 0.067$) and total fungi ($p = 0.096$). There were no statistically significant difference between the outlet concentrations of both 0.5 m and 0.25 m media depth BFs for both total mesophilic bacteria ($p = 0.434$) and Gram negative bacteria ($p = 0.428$), although the 0.5 m outlet concentrations were higher.

These suggest that the different biofilter media depths appear to only have an impact on the concentration of fungi which are bigger than bacteria.

Figure 7.3 shows the mean REs achieved by 0.5 m and 0.25 m media depth BFs for all four sampling visits. Generally, 0.5 m depth BFs showed higher removal of both *A. fumigatus* (58 – 89%) and total fungi than (56 – 89%) in comparison to the 30 - 83% and 34 - 83%, respectively, achieved by the 0.25 m depth BFs. This is contrary to the REs achieved with the bacteria. For total mesophilic bacteria, the 0.25 m depth BFs indicated removals of 71%, 87%, 87% and 33% for visits 1, 2, 3 and 4, respectively in comparison to the 0.5 m depth which achieved REs of 68%, 76%, 90% and 22%, respectively, for the same visits. For Gram negative bacteria, 0.25 m media depth BFs also indicated REs in the range of 59 – 80% while the 0.5 m media depth BFs recorded a range of 39 – 82%.

Sanchez-Monedero et al. (2003) conducted a study to assess *A. fumigatus* and mesophilic bacteria REs achieved by biofilters installed at seven commercial composting plants. Although they did not comment directly, the data provided includes the media depths of the biofilters studied. Even though the REs reported seem to be overestimated due to the sampling method employed, it can be seen that there appears to be no clear relationship between MD and the achieved REs for both groups of microorganisms. Similarly, there appears to be no clear relationship between MD and the REs for *A. fumigatus*, total bacteria and Gram negative bacteria in the study by Fletcher et al. (2014). This was even further compounded by the presence of upstream scrubbers (which helped in bioaerosol concentration reduction) in some sites and none in others, making this assessment somewhat difficult. While there may not be appropriate literature on bioaerosols to compare the findings of the present study to, Liu et al. (2017) investigated the combined impact of MD and media moisture on the removal of NH_3. They reported that increased MD and moisture resulted in improved removals; however, that there were no significant difference in NH_3 REs between the three levels (0.17 m, 0.33 m and 0.50 m) of media studied, as observed for bioaerosols in this study. Thus, it may be that increasing MD may improve odour and fungal biofiltration, and not bacterial biofiltration.

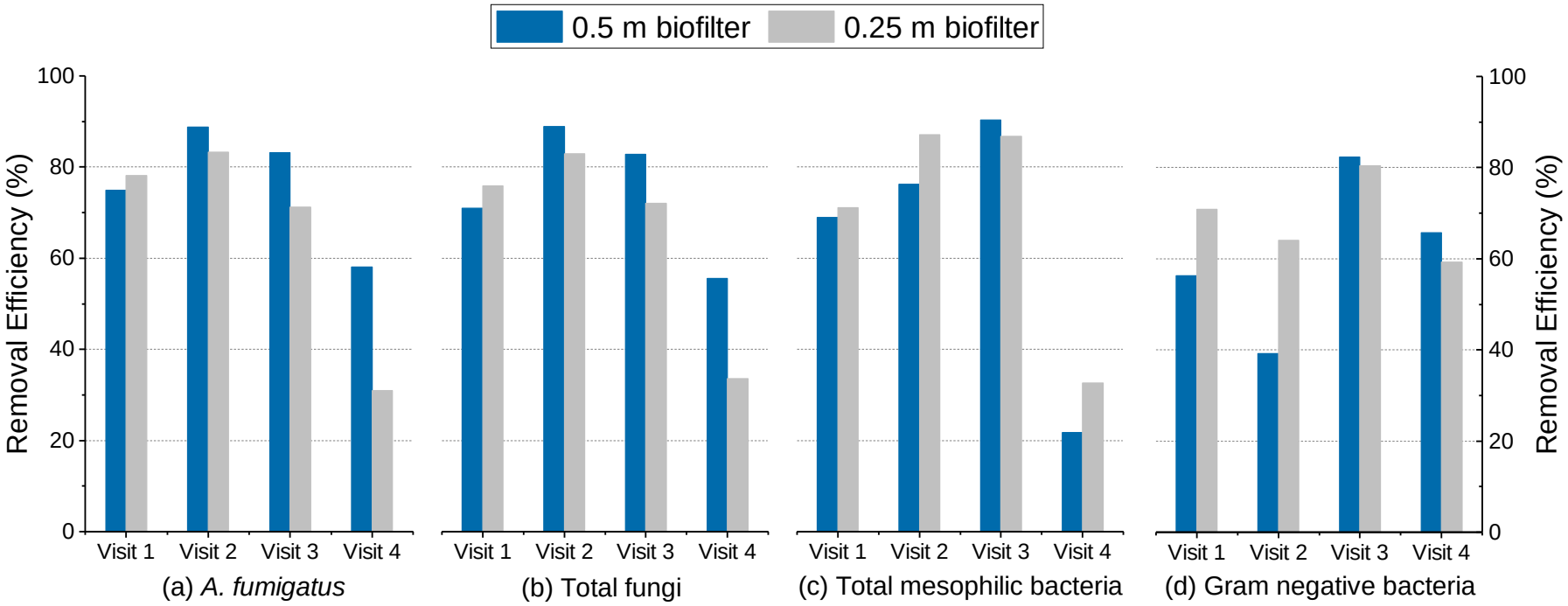


Figure 7. 3: Removal efficiencies measured for each bioaerosol group for the four sampling visits. REs computed using mean values of replicates BFs. (Visit 1 = 02/05/17; Visit 2 = 08/05/17; Visit 3 = 15/05/17; Visit 4 = 22/05/17)

Fletcher et al. (2014) stated that the biofilters in their study were particularly effective in controlling *A. fumigatus* and less effective in controlling total bacteria. A similar trend was observed with this study where the pilot biofilters achieved one log reduction of both *A. fumigatus* and total fungi. And although there were no statistically significant differences between the outlet concentrations from both 0.5 m and 0.25 m MD BFs, there appears to be an improved removal of fungi with the 0.5 m depth. Tables 7.2 and 7.3 summarise bioaerosol load removals and removal rates achieved by both groups of BFs for the four sampling visits. Again, it is evident that 0.5 m, being the thicker bed, traps more *A. fumigatus* ($1.0 \times 10^4 – 6.0 \times 10^4$ cfu m^{-3}) and total fungi ($1.2 \times 10^4 – 8.5 \times 10^4$ cfu m^{-3}) compared to $5.5 \times 10^3 – 5.6 \times 10^4$ cfu m^{-3} and $7.0 \times 10^3 – 7.9 \times 10^4$ cfu m^{-3}, respectively, trapped by 0.25 m MD BFs. This leads to higher fungi removal per hour for the 0.5 m depth BFs than the 0.25 m depth BFs (Table 5). On the other hand, the reverse was observed for bacteria with 0.25 m depth BFs trapping more total mesophilic bacteria ($9.2 \times 10^3 – 9.3 \times 10^4$) and Gram negative bacteria ($1.2 \times 10^4 – 4.7 \times 10^4$) than $6.1 \times 10^3 – 8.2 \times 10^4$ cfu m^{-3} and $7.4 \times 10^3 – 4.8 \times 10^4$ cfu m^{-3}, respectively, trapped by 0.5 m MD BFs.

Table 7. 2: Biofilter performance in terms of Bioaerosol Load Removal, *L* (cfu m^{-3})*

	Visit 1	**Visit 2**	**Visit 3**	**Visit 4**
A. fumigatus				
0.5 m BF	1.1E+04	6.0E+04	3.7E+04	1.0E+04
0.25 m BF	1.2E+04	5.6E+04	3.2E+04	5.5E+03
Total fungi				
0.5 m BF	1.4E+04	8.5E+04	4.7E+04	1.2E+04
0.25 m BF	1.5E+04	7.9E+04	4.1E+04	7.0E+03
Total mesophilic bacteria				
0.5 m BF	2.2E+04	8.2E+04	1.3E+05	6.1E+03
0.25 m BF	2.3E+04	9.3E+04	1.2E+05	9.2E+03
Gram negative bacteria				
0.5 m BF	1.5E+04	7.4E+03	4.8E+04	1.8E+04
0.25 m BF	1.9E+04	1.2E+04	4.7E+04	1.7E+04

* m^3 of inlet air

From these results, it is obvious that there are MD-dependent differences in the performance of the BFs for fungal and bacterial aerosol reduction. These differences may be due to a number of factors including airflow rate, media thickness, particle size and shape/morphology. Miaskiewicz-Peska and Lebkowska (2012) compared the collection efficiencies of air filters for non-biological aerosols and bioaerosols. They argued that increasing air flow rates had a tendency to decrease filter capacity to

collect bacterial cells. In this study, the airflow rates in the 0.5 m MD BFs (approximately 680 L min^{-1}) were approximately twice those in the 0.25 m MD BFs (approximately 340 L min^{-1}) in order to obtain an average EBRT of 16 s in all BFs. Thus, the increased airflow rates tend to work against bacteria biofiltration in contrast to the decreased airflow rates in the 0.25 m MD BF. Conversely, the high airflow rates in the 0.5 m MD BFs tend to favour fungi biofiltration. This is unclear why this is but is thought to be a function of the combined effect of bioaerosol particle size and media thickness.

Table 7. 3: Biofilter performance in terms of Bioaerosol Removal Rate, *R* (cfu m^{-3} h^{-1})**

	Visit 1	**Visit 2**	**Visit 3**	**Visit 4**
A. fumigatus				
0.5 m BF	2.6E+06	1.3E+07	8.4E+06	2.3E+06
0.25 m BF	2.7E+06	1.3E+07	7.2E+06	1.2E+06
Total fungi				
0.5 m BF	3.1E+06	1.9E+07	1.0E+07	2.6E+06
0.25 m BF	3.3E+06	1.8E+07	9.1E+06	1.6E+06
Total mesophilic bacteria				
0.5 m BF	5.1E+06	1.8E+07	2.9E+07	1.4E+06
0.25 m BF	5.2E+06	2.1E+07	2.8E+07	2.1E+06
Gram negative bacteria				
0.5 m BF	3.5E+06	1.7E+06	1.1E+07	4.2E+06
0.25 m BF	4.4E+06	2.7E+06	1.1E+07	3.7E+06

** m^3 of media bed volume

Figure 7.4 shows the particle size distributions of the inlet and outlet of both 0.5 m and 0.25 m MD BFs for the four sampling visits (Tables D.2, D.3, D.4, D.5). It can be seen that with the exception of visit 4, the inlet concentrations of both *A. fumigatus* and total fungi were composed of high percentages of particles greater than 3.3 μm – range of 56 – 73% and 56 – 75%, respectively. Ferguson et al. (2017) investigated the structure of bioaerosol communities derived from compost with the aim of quantifying and identifying the presence of specific pathogens in different size fractions with culture independent methods. They found that bioaerosol community structure and abundance are size dependent, and that bacterial bioaerosols in the range > 3.3μm were either conglomerates of bacteria or attached to larger particles such as dust while those in the range < 3.3μm were single cells. Thus, it may be that the thicker 0.5 m MD BFs preferentially filtered more of this larger size particles than the 0.25 m MD BFs because they have larger surface presented for particle impaction and interception, resulting in the lowered outlet concentration of fungi. Sanchez-

Monedero et al. (2003) argued that the large size of *A. fumigatus* (in comparison to mesophilic bacteria) in their study would favour their impaction on the BF media. Trunov et al. (2001) also suggested that fungal spores are often aerosolised in agglomerates which have a higher inertia than single spores and therefore are more likely to be impacted onto surfaces. This further indicates that the high proportion of large fungal (> 3.3μm) may actually be composed of agglomerates of spores which have easily impacted on the filter media.

On the other hand, the 0.25 m MD BFs tend to favour bacteria removal compared to the 0.5 m MD BFs. This may be a function of the difference in media bacteria content between 0.5 m and 0.25 MD BFs. The literature suggests that high MD may contain more microorganisms which aid in gas biodegradation (Kafle et al., 2015; Liu et al., 2017). Fletcher et al. (2014) argued that BF media harbour approximately 10^7 microorganisms/g, and these may become blown off into the flowing airstream, contributing to the concentrations measured at the BF outlet (Rabe and Becker, 2000). Ottengraf and Konings (1991) also confirmed this extra contamination of the outlet air due to the filtration process at low inlet gas concentration of bioaerosols. Data presented by Sanchez-Monedero et al. (2003) showed that bacteria particles are smaller than fungal particles, and because of their fine size may tend to remain in the flowing airstream without interception (Tisch Environmental Inc., 2015; Wang et al., 2018). Furthermore, Miaskiewicz-Peska and Lebkowska (2012) explained that some bacterial species possess spiny surfaces which make it hard for them to attach to the filtering media. Unfortunately, the scope of this study did not include species identification which would have provided information on the microbial cell morphology. Nonetheless, it can be argued that the high outlet concentration of bacteria recorded for 0.5 m MD BFs may be the result of the summation of particles not intercepted (due to size- and shape-dependent tendencies to remain in the airstream), and the proportion originating from within the high concentration of bacteria colonising the thicker media. However, a confirmation of this argument would require molecular analysis (which was outside the scope of this study) to determine if the population of bacteria entering the biofilters were the same as those coming out and would help establish whether they originated from within the biofilter media.

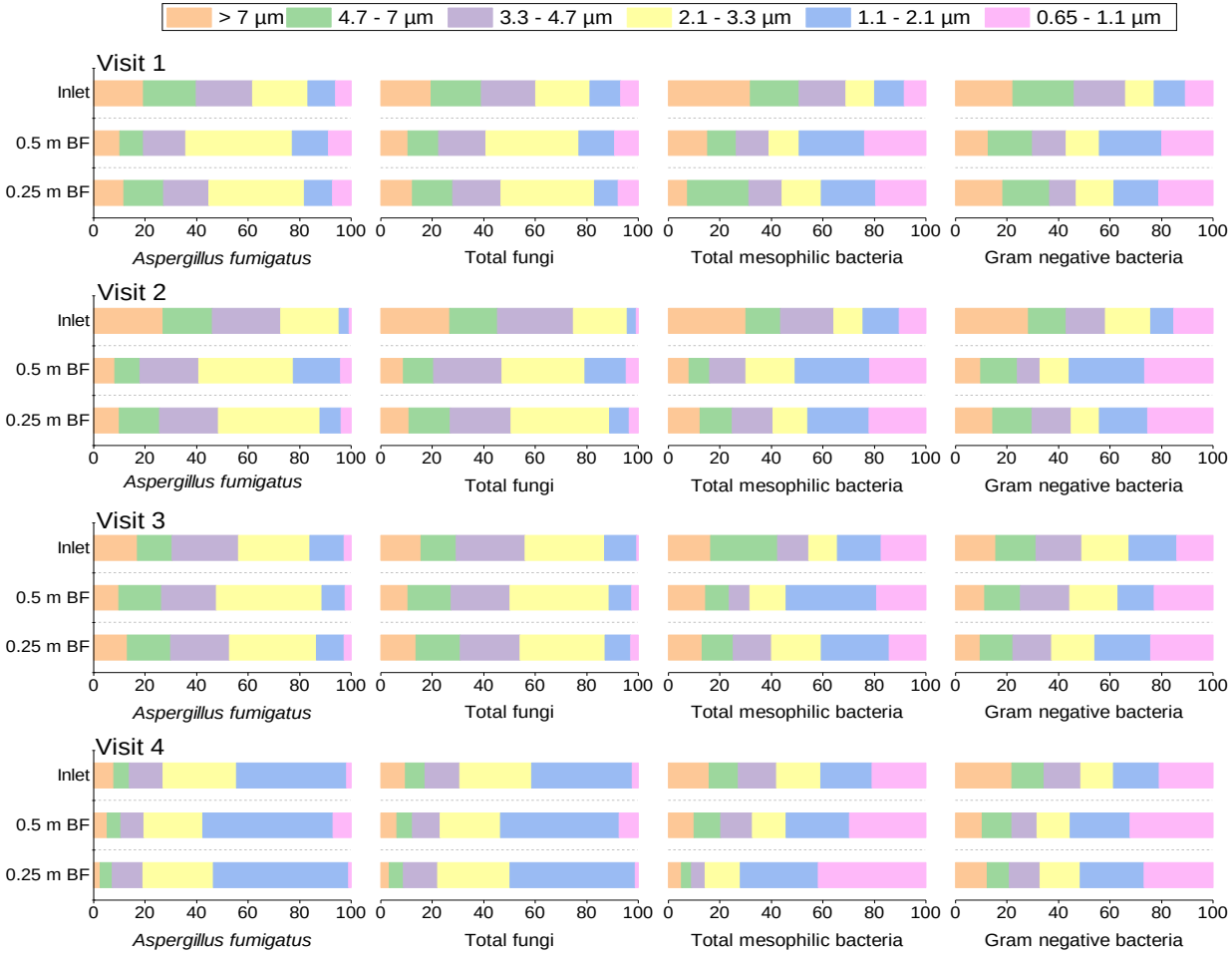


Figure 7. 4: Comparison of bioaerosol particle size distribution for the biofilter inlet and outlets of the 0.5 m and 0.25 m media depth biofilters for the four sampling visits. Data for the outlet based on summation of outlet concentration of the replicate biofilters. (Visit 1 = 02/05/17; Visit 2 = 08/05/17; Visit 3 = 15/05/17; Visit 4 = 22/05/17).

It is worth commenting on the particle distribution of the outlet samples. Table 7.4 summarises the proportion of outlet bioaerosols in the range < 3.3 μm for both MDs. The outlet concentrations from both MDs show high proportions of particles < 3.3 μm, with a slightly higher proportion for 0.5 m MD BFs especially for *A. fumigatus* and total fungi. These high counts of smaller particles sizes may be explained by the phenomenon of spore cluster break-up. Studies have shown that biological particles may be present in the air as aggregates such as pairs, chains or clusters (Wake et al., 1995; Górny et al., 1999). Miaskiewicz-Peska and Lebkowska (2012) stated that one of the challenges with bioaerosols studies is their susceptibility to change in size. This can be seen with spore cluster break up reported by Jankowska et al. (2000). They compared the collection efficiencies of fungal spores and Potassium chloride (KCl) particles (used as standard test particles) by ventilation filters. They observed that collection efficiency increased with increased aerodynamic size, but only for the KCl particles. They attributed this observation to fungal spore cluster break up in contrast to the KCl particles which had no clusters. Miaskiewicz-Peska and Lebkowska (2012) also suggested in their study that *Micrococcus luteus* cells' aggregates disintegrated upon impact on the air filter, resulting in higher counts of colony forming units in the solid media used for impaction. It is possible that the spore clusters and aggregates of bacterial and fungal particles disintegrated further to smaller particles upon impaction on media bed, thus increasing their tendency to be carried in the flowing airstream to the outlet of the BFs.

Table 7. 4: Proportion of outlet bioaerosol concentrations in the range < 3.3 μm for all four visits

	0.5 m MD BF	**0.25 MD BF**
A. fumigatus	52 – 80%	47 – 81%
Total fungi	50 – 77%	46 – 78%
Total mesophilic bacteria	61 – 70%	56 – 86%
Gram negative bacteria	56 – 68%	53 – 67%

It is important to point out that many variables may impact on the results of biofiltration studies, and a knowledge of these is required to interpret the results of any biofiltration studies. Biofilter testing often occurs within a few minutes or hours and in controlled environments; however, these are systems that are designed to run for years and so may be exposed to dozens or hundreds of environmental changes and variations (including temperatures, humidity, airflow rates and bioaerosol particle concentrations). Adding to this complexity is the technical variations in design as well

as the fluctuations in the maintenance regimes put in place by the operators. Thus, the results are useful in providing insights into the basic operation of BFs.

7.4 Section Summary

- The MDs investigated showed potential capacity to control bioaerosol emissions from the process air of MRFs, and possibly other waste treatment facilities. This is indicated by the significant ($p < 0.05$) reductions of the inlet concentrations of bioaerosols as measured at the outlet of the 0.5 m and 0.25 m MD BFs.
- The 0.5 m MD achieved higher removals in the range of 58 – 89%, 56 – 89%, 22 – 90% and 39 – 82% for *A. fumigatus*, total fungi, total mesophilic bacteria and Gram negative bacteria, respectively. In comparison, the 0.25 m MD also delivered REs almost in the same range including 30 – 83%, 34 – 83%, 33 – 87% and 59 – 80% for *A. fumigatus*, total fungi, total mesophilic bacteria and Gram negative bacteria, respectively.
- Although there were no statistically significant differences between the performances of both MDs, the 0.5 m MD shows improved control of fungi compared to bacteria while the 0.25 m MD had better removals of bacteria than fungi.
- The improved performance of 0.5 m MD for fungi is thought to be a function of the high airflow rate, bioaerosols particle size and media thickness. This MD presents high surface area for fungal particle impaction; and in addition, the fungal particles, being the larger in terms of aerodynamic size, tend to impact more on the media surface, thus effecting higher removals.
- The lower removal of bacteria by 0.5 m MD may be the result of the combined effect of particles not intercepted (due to size- and shape-dependent tendencies to remain in the airstream), and particle release from the rich abundance of bacteria colonising this thicker media.
- Nonetheless, a more extensive research is required (i) to provide insights into the various parameters that may confound the measured inlet and outlet concentrations (ii) to identify the specific species with view to understanding the interplay of cell morphology in bioaerosols removals by biofilters.

Chapter 8
IMPACT OF BIOFILTER MEDIA TYPE ON BIOAEROSOL CONTROL IN A MATERIALS RECOVERY FACILITY

8.1 Introduction

The selection of an appropriate biofilter media material is critical, especially when assessing biofilter performance. Section 2.2.6.4 presents a review of some of the media characteristics considered as vital for biofilter effectiveness; a summary of some of these media materials (Table 2.13); and a review of previous studies on the impact of different media materials on biofilter perfromance. Nonetheless, Chen et al. (2009), in their study to examine the performance of two types of wood chips (western cedar and hardwood) as media material for the reduction of odour, H_2S and NH_3 from a swine barn, reported that both materials achieved high odour removal efficiencies of 48 – 93% at a moisture content of 60% (wet basis).

It is also not uncommon to find biofilters with combinations of these media materials e.g. the biological residues (compost, soil, peat) with inert bulking agents such as activated carbon or wood chips (Devinny et al., 1999). For agricultural biofilter media, a 30:70 ratio of compost and wood chips mixture has been offered as a suitable option (Nicolai and Janni, 2001b), and a mixture of 20 – 30% compost and 70 – 80% wood chips (by weight) has also been recommended as ideal (Schmidt et al., 2004 cited in Chen and Hoff, 2009). Research by Nicolai and Janni (1997) showed that a media combination of compost and kidney bean straw achieved removal efficiencies of 50%, 86% and 78% for NH_3, H_2S and odour, respectively.

Only a few studies have assessed the use of biotreatment to control airborne microbiological contaminants, and there seems to be no clear results on the influence of various filter media on the control of bioaerosols. Some authors argue that filter material types do not have significant impact on the performance of full scale biofilters; rather the tendency is for materials with larger and more structured surfaces to deliver higher efficiency (Schlegelmilch et al., 2005a). Differences have also been reported in filter media performance in laboratory and technical scale studies. Schlegelmilch et al. (2005a) argued that while biofilter media proved to have a major influence on bioaerosol emissions in laboratory-scale studies, they seemed to have a minimal influence at technical scale. Tymczyna et al. (2007) investigated the performance of three media mixes - organic-organic medium (with 50% compost and

50% peat); organic-mineral medium (with 20% bentonite, 40% compost, and 40% peat); and organic-mineral medium (with 20% halloysite, 40% compost, and 40% peat) – and reported that all the media were highly effective in the removal of gram-negative bacteria (99.6%, 100% and 100%, respectively), moderately effective in controlling dust and only slightly effective in the removal of endotoxin. In addition, , coconut fibre has also been reported to perform well for bioaerosols control (Schlegelmilch et al., 2005a), but not so well for odour control (Dammann et al., 1996).

From an emissions mitigation viewpoint, the argument is not to have the best media to control either chemical contaminants (including odours) or bioaerosols, but a suitable media both in terms of simultaneous significant control of all emissions as well as financial sustainability. Thus, this objective assessed the impact of different biofilter media types in terms of bioaerosol and odour emissions and removal. The following sub-objectives were defined to include (i) evaluation of the comparative performance of four media types – old and new wood chips, peat and wheat straw – for simultaneous bioaerosol and odour control; (ii) assessment of the impact of media age on simultaneous bioaerosol and odour removal; (iii) assessment of the impact of media type on bioaerosol particle size distribution between biofilter inlet and outlet air samples.; and (iv) assessment of media cost analysis.

8.2 Biofilter operation to evaluate performance

Section 3.4 presents a general discussion of the operation of the biofilters throughout the study. However, this section presents a more detailed information on how the biofilters were operated and maintained, media selection and characterisation as well as on the data analyses that were carried out to achieve the specific objectives of this particular study.

8.2.1 Biofilter Operation

This study was conducted in the summer of 2017 (June and July) during which a total of four sampling visits (June 19, June 26, July 3 and July 10) were conducted (Appendix E). The biofilter system was set up inside the facility just behind the back-push wall in the waste reception area so that it was as close to the incoming waste materials as possible, therefore ensuring a constant supply of odorous air contaminated with bioaerosols which was vital for this study.

8.2.2 Biofilter Media Selection and characterisation

The range of media (Plate 8.1) selected for this study covered some of those reported in literature as a result of the advantages (Table 2.13) they offer including ease of application, cost effectiveness, low water control problems, and inherent content of nutrients and microbial population which eliminates the need for nutrient supply and microbial inoculation, respectively. Peat, wheat straw and new woodchips were freshly acquired for this study while the old woodchips were derived from a previous experiment testing biofilter performance at the same location on the site, operated for a year prior to the current study.

Plate 8. 1: Media materials used in the biofilters on site: (a) old woodchips in BF1, (b) peat in BF2, (c) wheat straw in BF3, and (d) new woodchips in BF4.

The media materials were sourced from local markets within the UK – woodchips (from Garforth Log Supplies, Leeds at £80 per m^{-3}), Irish Moss peat (from Erin Horticulture Ltd at £100 per m^{-3}), and wheat straw (from Amlico Animal Store, Wakefield at £23 per m^{-3}). Preliminary laboratory tests were conducted to determine the media characteristics including appropriate, bulk density, porosity, water holding capacity (WHC) and moisture content (MC) (Table 8.1).

Table 8. 1: Characteristics of media used for this study

Characteristics	**Old woodchips***		**Peat**	**Wheat straw**	**New woodchips**
	Before	After			
Bulk Density (kg m^{-3})	225.0	239.2	368.4	40.9	202.8
Porosity (%)	61.4	60.3	61.7	82.2	64.2
Water Holding Capacity (g/g dry weight)	1.2	1.1	8.5	5.4	1.2
Moisture Content (%), as received	30.0	30.0	75.3	1.2	15.9

*the woodchips were used continuously for one year before the current study. 'After' indicates woodchips characteristics used for this study.

Peat and wheat straw were used as received while the woodchips (as-received) were sized by sieving using the Retsch AS200 Analytical Sieve Shaker operated at an amplitude of 60 and a vibration height of 1.8 mm for three minutes. A sieve mesh size of 4.75 mm was used to obtain woodchips oversize fractions (Plate 8.2) used for this study. Media MC were determined using the oven drying method which entails computing the weight loss following oven drying overnight at 105°C while WHC were determined by soaking the media materials in water for 24 hours followed by oven-drying the media samples for 48 hours at a temperature of 105°C (Kafle et al., 2015). Media porosity (voids) was determined by the Bucket Method (Nicolai and Janni, 2001c) and bulk density was determined following the method of Valter Francescato et al. (2008).

Plate 8. 2: Size fraction of media materials used for this study: (a) old wood chips, (b) peat, (c) wheat straw, and (d) new wood chips.

8.2.3 Data Analysis

There were statistically significant differences in the mean bioaerosol concentrations at the background, inlet and outlets of all media types tested as follows, *A. fumigatus*: Welch's $F(5, 17.707) = 51.065$, $p < 0.0005$; total fungi: Welch's $F(5, 17.939) = 60.399$, $p < 0.0005$; total mesophilic bacteria: Welch's $F(5, 16.771) = 33.428$, $p < 0.0005$; Gram negative bacteria: Welch's $F(5, 18.868) = 5.875$, $p = 0.002$. For odour, there were no outliers in the data set and normality of odour concentrations was assessed using the Shapiro-Wilk test. Differences in mean odour concentration for the BF inlet and all BF outlets were assessed using ANOVA. There was homogeneity of variance as assessed by Levene's test for equality of variances ($p = 0.687$). There were statistically significant differences in the mean odour concentrations of the inlet and outlets of all media types tested, $F(4, 35) = 9.902$, $p < 0.0005$. Spearman's rank-order correlation was run to assess the association between odour and bioaerosol REs of each of the media types in this study.

8.3 Results and discussion

8.3.1 Operating conditions, bioaerosol and odour concentrations

Table 8.2 presents the operating conditions of the various media types during the sampling period. The mean inlet temperature for all BFs taken at the central plenum was 26.3ºC and apart from the peat with a media temperature of 18.4ºC, all other media had temperatures above 21ºC.

Table 8. 2: Operating conditions of the biofilters (BF) during the study period

Parameter	**Old woodchips**	**Peat**	**Wheat straw**	**New woodchips**
Mean inlet air temperature (ºC)	26.3	26.3	26.3	26.3
Mean outlet air temperature (ºC)	24.3	23.8	24.9	23.1
Mean media temperature (ºC)	21.3	18.4	22.9	23.0
Mean EBRT (s)	17.9	16.4	16.9	16.1
Mean Airflow rate (L min^{-1})	610	665	645	680
Leachate pH range	6.84 – 7.55	4.21 – 4.81	6.60 – 8.24	6.92 – 7.99
Media moisture content (%)	41 - 66	76 - 79	45 - 75	40 - 59

No attempts were made to alter the media temperature as they were within the range (10 - 40ºC) reported as optimum for effective biofilter operation (Schnelle and Brown, 2002). Figure 8.1 compares the background, inlet and outlet concentrations of the four bioaerosol groups and odour assessed under these conditions. The results show that the emitted air from biofilters containing each of the four media tested had less bioaerosols and was less odorous than the air fed into each biofilter.

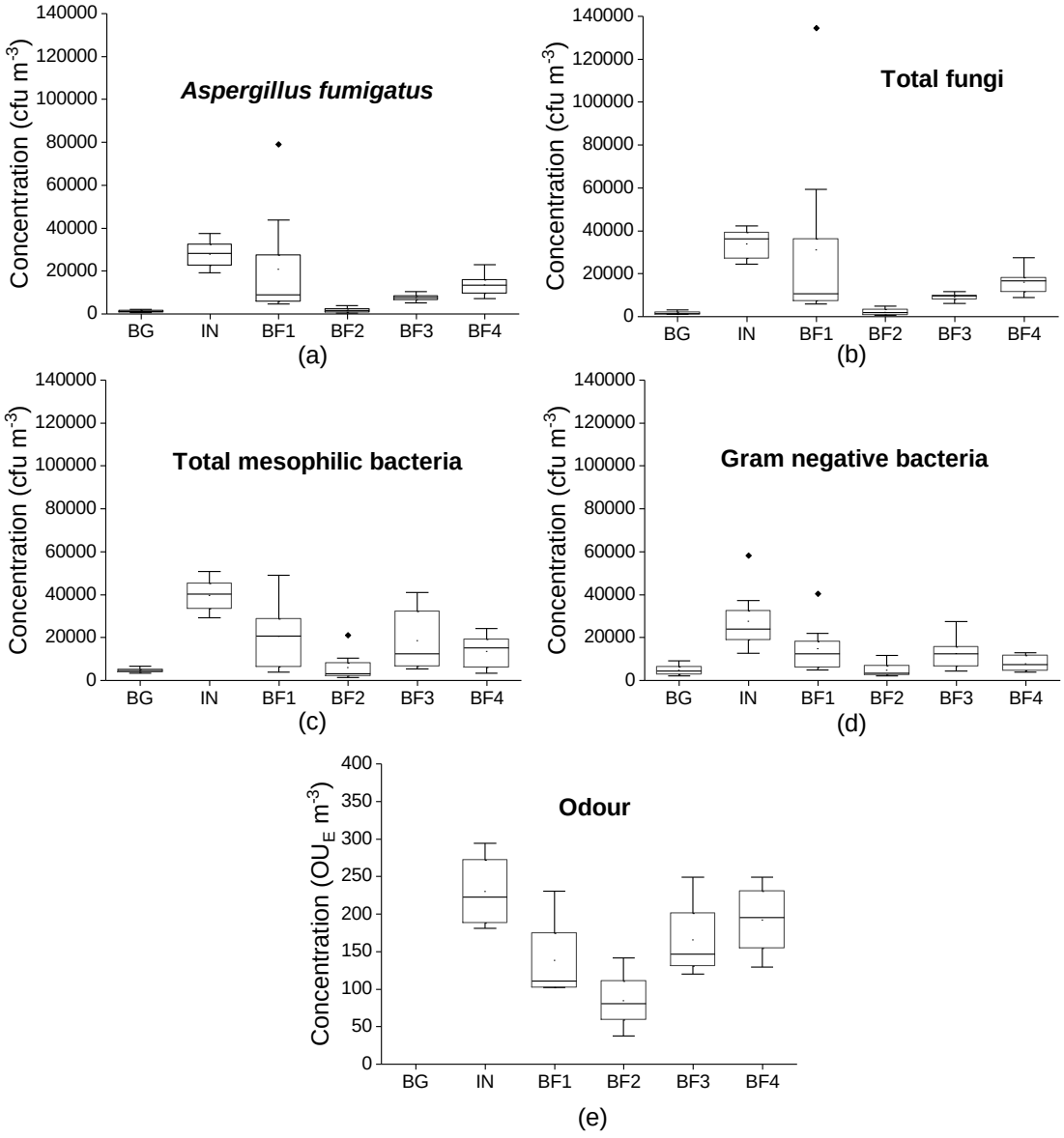


Figure 8. 1: Comparison of mean background, inlet and outlet concentrations of (a) *Aspergillus fumigatus*, (b) total fungi, (c) total mesophilic bacteria, (d) Gram negative bacteria and (e) odour for the study period (BG – Background; IN – Inlet; BF1 – Old Woodchips; BF2 – Peat; BF3 – Wheat Straw; BF4 – New Woodchips).

One-way Welch ANOVA followed by Games-Howell post hoc analysis showed that there were statistically significant differences ($p < 0.05$) between mean inlet concentrations and mean outlet concentrations from all media types assessed for the four groups of bioaerosols investigated, except the outlet concentrations of biofilters containing old woodchips ($p = 0.420$) and wheat straw ($p = 0.174$) when assessed for Gram negative bacteria removal (Figure 8.1). However, there were no statistically significant differences ($p > 0.05$) between the performance of the media types in terms of their control of both total mesophilic and Gram negative bacteria. With regards to *A. fumigatus* and total fungi removal, peat showed statistically significant differences ($p < 0.05$) in outlet concentrations in comparison to other media types, except old woodchips ($p = 0.408$ for *A. fumigatus*; and $p = 0.519$ for total fungi). Both old and new woodchips and wheat straw did not show any statistically significant differences ($p > 0.05$) in their outlet concentrations. There were no statistically significant differences ($p > 0.05$) between the outlet concentrations of all media types and background concentrations for both total mesophilic and Gram negative bacteria. Statistically significant differences were only established between background and outlet concentrations of wheat straw ($p < 0.0005$) and new woodchips ($p = 0.002$) when considering *A. fumigatus* and total fungi. These findings are consistent with those of Tymczyna et al. (2007) who reported that there was no statistically significant differences in the levels of aerosols emitted among the various media in their study. They attributed this to the high variation among sample values within the different media groups, also typical of this study.

A one-way ANOVA followed by Tukey post hoc analysis revealed that there were statistically significant differences ($p < 0.05$) in the mean odour concentrations of the inlet samples and outlet samples of old woodchips and peat media biofilters only. New woodchips and wheat straw did not show any statistically significant differences with the inlet odour concentrations. Comparing the outlet concentrations of the four media types, statistically significant differences were only shown between the concentrations from the peat biofilter and from those containing wheat straw ($p = 0.019$) and new woodchips ($p = 0.001$). There were no statistically significant differences ($p > 0.05$) established between the outlet concentrations of other media types.

8.3.2 Comparative performance of the different media materials

The performance of the pilot biofilters was assessed by the removal efficiencies they achieved throughout the study period (Figure 8.2). The REs for bioaerosols were computed using the mean values of the inlet and outlet concentrations. Figures 8.3 – 8.6 show comparisons of actual mean odour and bioaerosol concentrations at the inlet and outlets of the four biofilters investigated for each sampling day of this study. The concentration of bioaerosols and odour measured in the process air varied from visit to visit, as acknowledged by Fletcher et al. (2014) when monitoring full scale facilities. In contrast to the study of Tymczyna et al. (2007) which reported no detection of bioaerosols for two of the three media materials investigated, bioaerosols were detected in the outlet air of the all media types assessed in this study.

Generally, inlet concentrations of all bioaerosols measured were in the range of 10^4 cfu m^{-3} as follows: 2.0×10^4 to 3.6×10^4 cfu m^{-3} (*A. fumigatus*); 2.5×10^4 to 4.1×10^4 cfu m^{-3} (total fungi); 3.6×10^4 to 4.5×10^4 cfu m^{-3} (total mesophilic bacteria); and 1.5×10^4 to 3.9×10^4 cfu m^{-3} (Gram negative bacteria). The inlet odour concentrations were 250 OU_E m^{-3}, 181 OU_E m^{-3}, 294 OU_E m^{-3} and 194 OU_E m^{-3} for the 19 June, 26 June, 3 July and 10 July, respectively. These concentrations were considered low when compared to concentrations (up to 19 340 OU_E m^{-3}) reported by an independent consultant (Gair, 2013). The difference is thought to be due to the same reasons as outlined in section 5.3.2.

The old woodchips achieved REs of 57 – 86 % for both groups of fungi for all sampling days except on 3 July 2017 when the REs were reduced to -118% (outlet concentration: 6.1×10^4 cfu m^{-3}) and -150% (outlet concentration: 9.7×10^4 cfu m^{-3}) for *A. fumigatus* and total fungi, respectively. The old woodchips also achieved REs of 29 to 85% and 44 to 83% for total bacteria and Gram negative bacteria, respectively, except on 3 July 2017 when the REs dropped to 7% (outlet concentration: 3.7×10^4 cfu m^{-3}) and -14% (outlet concentration: 2.7×10^4 cfu m^{-3}), respectively, as with the fungi. This drop in performance is thought to be due to the breakthrough of biomass accumulated within the old woodchips which have been in operation for a year prior to this study. Devinny et al. (1999) argued that media clogging can lead to air channelling which has the potential to limit the amount of contaminants being treated, thus resulting in negative performance. The performance in terms of odour control was moderate, delivering REs up to 65% (on 3 July) from an inlet concentration of 294 OU_E m^{-3}.

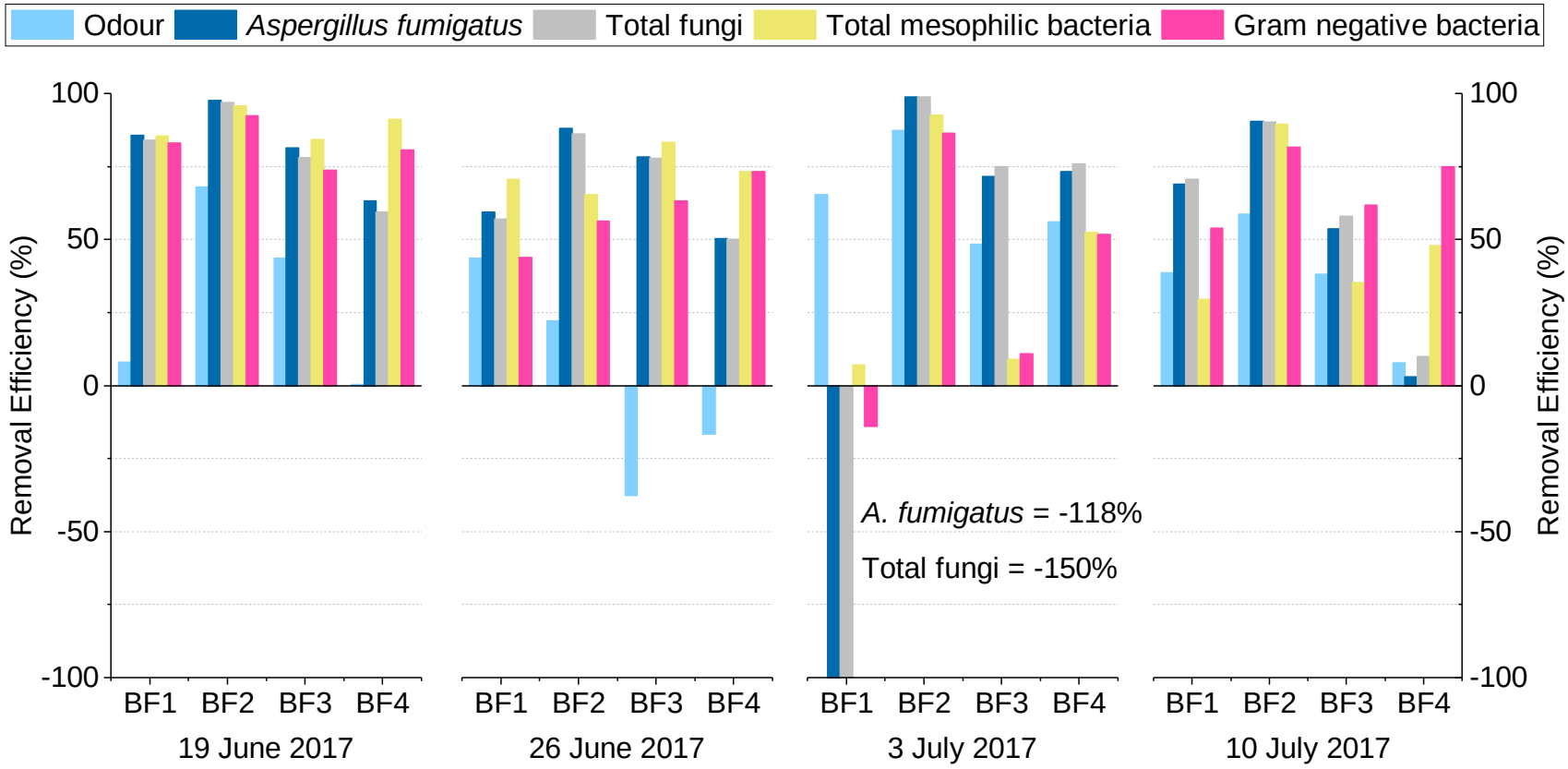


Figure 8. 2: Comparison of odour and bioaerosols removal efficiencies achieved in all four pilot-scale biofilters for the study period (BF1 – Old Woodchips; BF2 – Peat; BF3 – Wheat Straw; BF4 – New Woodchips).

The data set showed a statistically significant perfect negative correlation (r_S = - 1.0) between odour RE and REs of *A. fumigatus*, total fungi as well as Gram negative bacteria, at the 0.01 level. There was also a strong negative correlation (r_S = - 0.8) between odour RE and total mesophilic bacteria RE, but this was not statistically significant. This suggests that differences may exist in the capacity for used woodchips to achieve simultaneous removal of odour and bioaerosols.

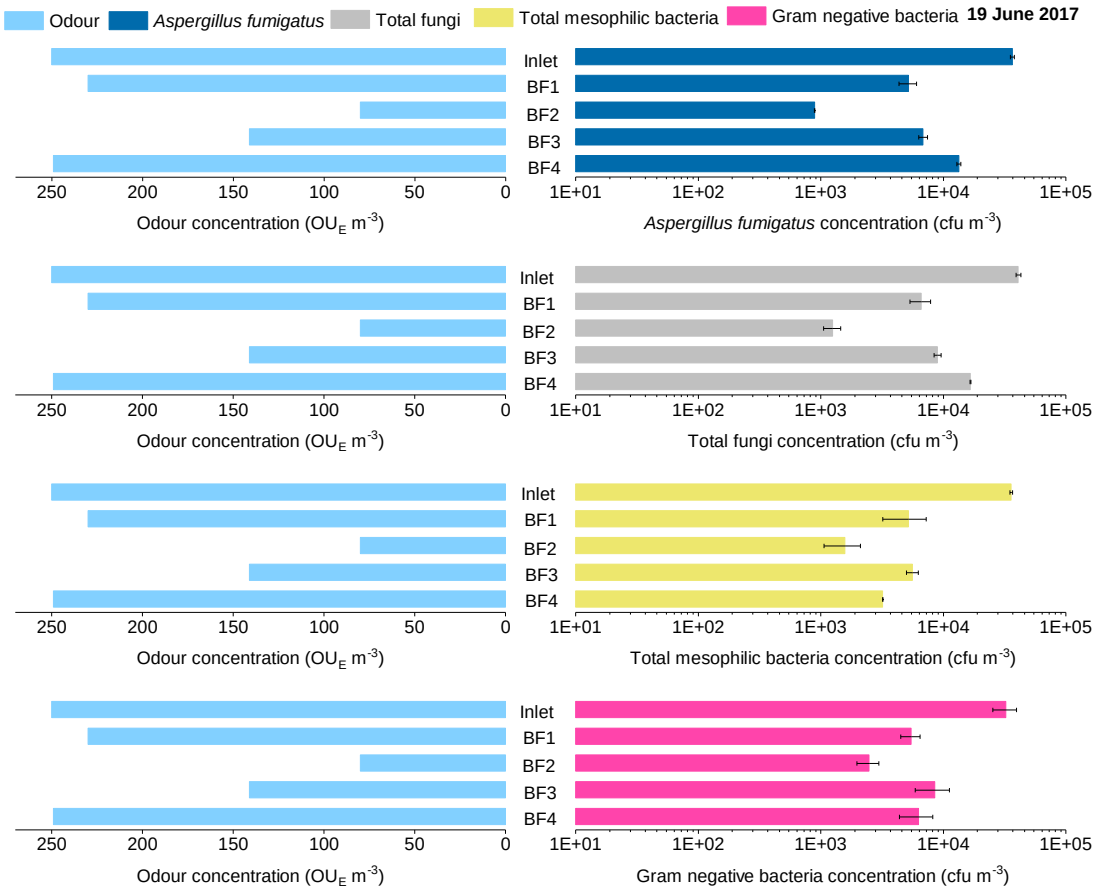


Figure 8. 3: Comparison of inlet and outlet concentrations of odour (n = 1) and bioaerosol (n = 2) on 19 June 2017 (BF1 – Old Woodchips; BF2 – Peat; BF3 – Wheat Straw; BF4 – New Woodchips).

The peat biofilter consistently showed the highest REs for both bioaerosols and odour of all the media types. REs of 88 to 99%, 86 to 99%, 65 to 96% and 56 to 92% were achieved for *A. fumigatus*, total fungi, total mesophilic bacteria and Gram negative bacteria, respectively. This brought the outlet concentrations to a range of 10^2 to 10^3 cfu m^{-3} for fungi and 10^3 cfu m^{-3} for bacteria. These high bioaerosol REs for peat could possibly result from a combination of Brownian diffusion, flow-line interception, inertial impaction of bioaerosol particles by the media as well as the low media pH (4.21 – 4.81) recorded in this study which can lead to the destruction of trapped

microbial particles especially as most aerobic bacterial activities thrive at near neutral pH (7 – 8) (Swanson and Loehr, 1997; Omri et al., 2013). Tymczyna et al. (2007), in their study, pointed out that filter effectiveness in reducing airborne contaminants was dependent on which filter media was used, as observed in this study.

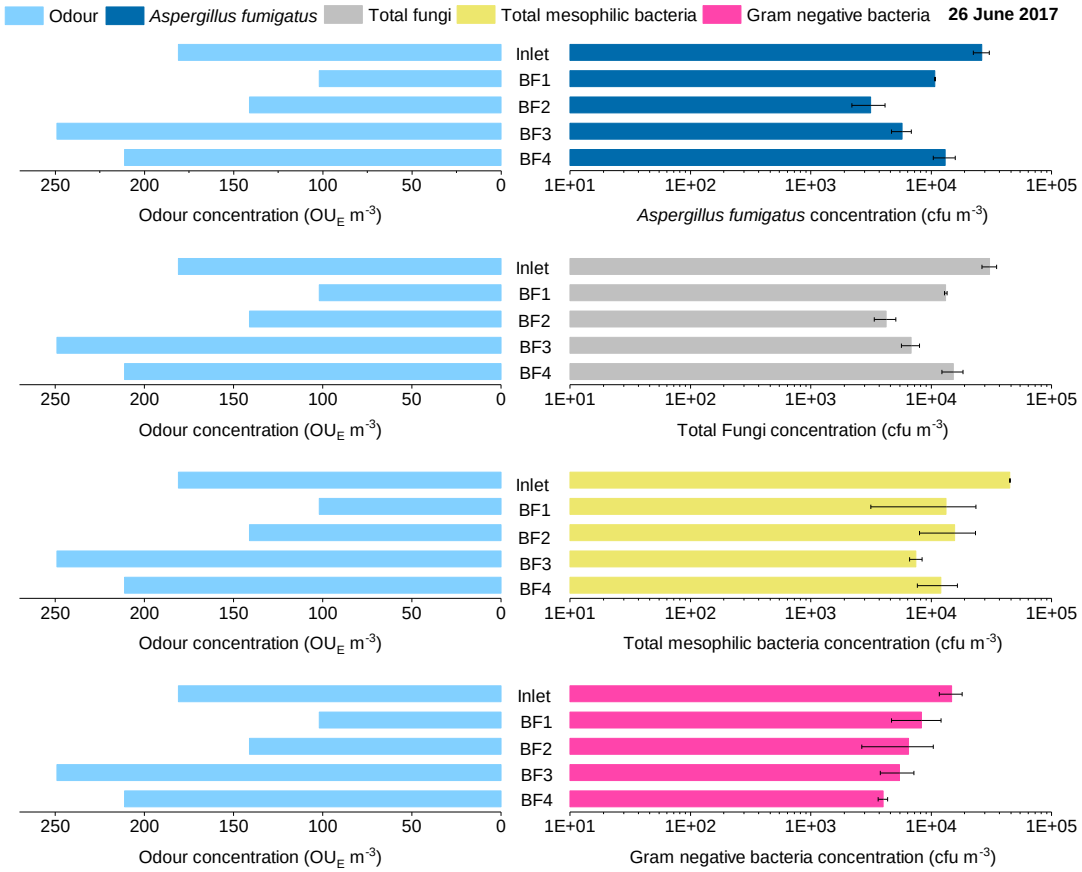


Figure 8. 4: Comparison of inlet and outlet concentrations of odour (n = 1) and bioaerosol (n = 2) on 26 June 2017 (BF1 – Old Woodchips; BF2 – Peat; BF3 – Wheat Straw; BF4 – New Woodchips).

Peat also delivered odour REs in the range of 22 to 87%. One reason for the high odour removal recorded with peat is the high water content of peat. Devinny et al. (1999) stated that fungi dominate in biofilters which are acidic and with high water content, and often have the capacity to degrade even more complex compounds. There were statistically significant perfect positive correlation between odour removal and REs of *A. fumigatus* and total fungi at the 0.01 level. A strong positive correlation of r_S = 0.8 was also established between odour RE and the REs of the bacteria. These imply that peat was capable of achieving simultaneous significant removal of bioaerosols and odour from waste airstreams emitted from waste management facilities. Devinny et al. (1999) stated the peat does not naturally harbour microorganisms and so would require microbial inoculation to be used for

biofiltration. This might mean that the reduction capacity should have been reduced especially in the early stage of biofiltration when there were supposedly few or no microorganisms within the media. Schmidt et al. (2004) supported this position by stating that biofilter efficiency is limited during the conditioning or stabilisation period. However, the results of this study contradicted this view as the peat media was not inoculated beforehand, but it consistently showed considerable odour reduction compared to other media. This shows that the acclimation period was sufficient in allowing a rich population of odour-degrading microorganisms to be developed within the media, evident in the high REs recorded for peat.

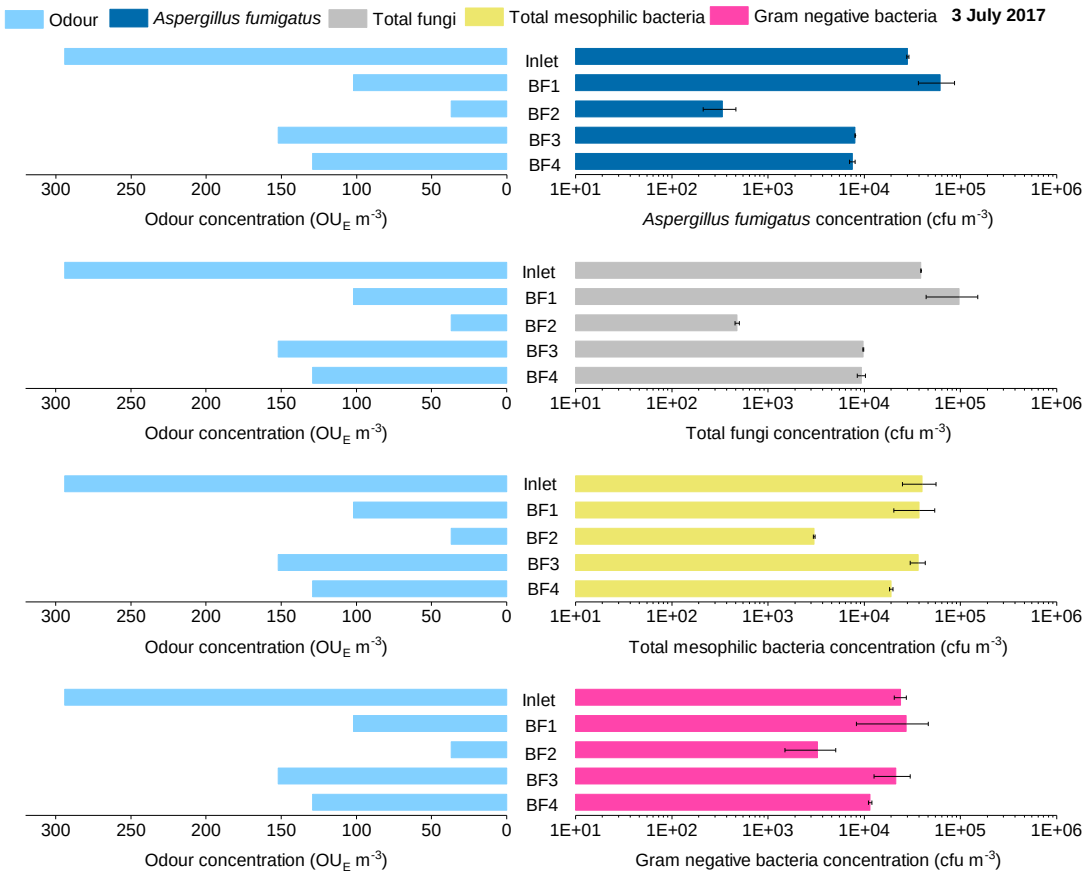


Figure 8. 5: Comparison of inlet and outlet concentrations of odour (n = 1) and bioaerosol (n = 2) on 3 July 2017 (BF1 – Old Woodchips; BF2 – Peat; BF3 – Wheat Straw; BF4 – New Woodchips).

Sheridan et al. (2002c), in comparing the performance of two media particle sizes (>20 mm and 10 to 16 mm) on odour removal, argued that the smaller sized media delivered better reduction even though they present with increased pressure drop challenges. To avoid pressure drop differences in this study the media was mixed once every two weeks to ensure no preferential air paths were created within the media. Peat had the smallest particle size of all media and hence the greatest surface area for

biofilm attachment for effective odour degradation; thus the consistent high odour performance of the peat. Hoag and Price (1997) noted that peat had a 'dual-porosity' matrix, where closed and disconnected pores may trap solutes, and particles by extension, in dead-end pore spaces, hence retarding diffusive transport. This may contribute to trapping of odour compounds and straining of bioaerosol particles from the contaminated airstream (Edelman, 2008). These immobilised bioaerosol particles can add to the resident microbial population within the media, increasing the degradation rate of odorous organic volatiles trapped within the biofilm layer on the media particles (Brincat et al., 2016), further reducing the outlet odour concentrations measured in this study.

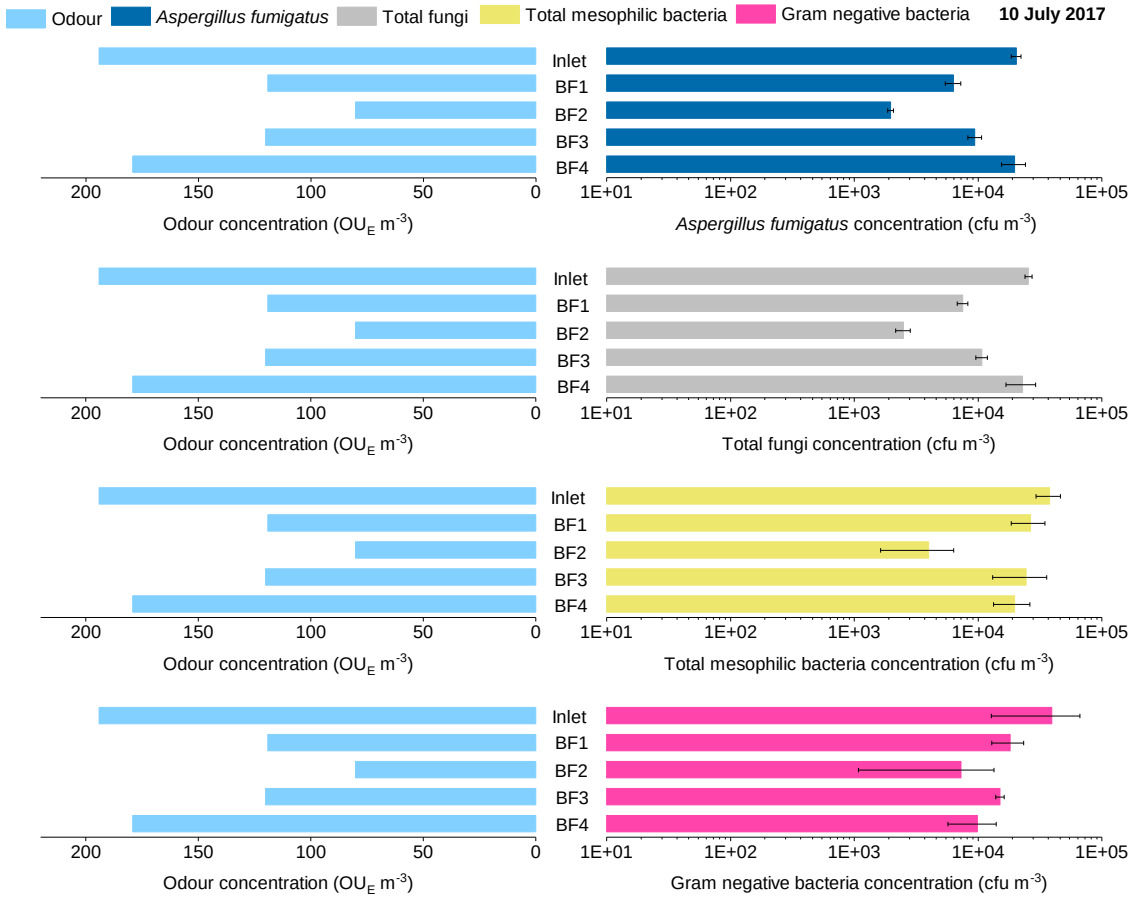


Figure 8. 6: Comparison of inlet and outlet concentrations of odour (n = 1) and bioaerosol (n = 2) on 10 July 2017 (BF1 – Old Woodchips; BF2 – Peat; BF3 – Wheat Straw; BF4 – New Woodchips).

Wheat straw showed lower removals of bioaerosols and odour in comparison to peat. Removals of 54 to 81% for *A. fumigatus*, 58 to 78% for total fungi, 9 to 84% for total mesophilic bacteria, and 11 to 74% for Gram negative bacteria were achieved in this study. The odour removals measured were all below 50%, and particularly on 26 June

outlet odour concentration (249 OU_E m^{-3}) was higher than the inlet concentration (181 OU_E m^{-3}) leading to a drop in RE to – 38%. The performance of new woodchips was somewhat comparable to that of wheat straw. New woodchips showed the poorest odour performance of 0.4 to 56% removal, and acted as a net odour emitter on 26 June with a RE of -17% as with wheat straw. Bioaerosol REs achieved were 3 to 73%, 10 to 76%, 48 to 91% and 52 to 81% for *A. fumigatus*, total fungi, total mesophilic bacteria and Gram negative bacteria, respectively. With wheat straw, there was no correlation (r_S = 0.0) between odour and fungi REs, while the bacteria only showed a moderate negative correlation (r_S = -0.4) with odour (p > 0.01). For the new woodchips, odour RE showed a moderate positive correlation (r_S = 0.4) with the REs of both fungi groups; a strong negative correlation (r_S = -0.6) with the RE of total mesophilic bacteria; and moderate negative correlation (r_S = -0.4) with Gram negative bacteria RE. However, these associations were not statistically significant at the 0.01 level.

The low odour removal performance seen with wheat straw and new woodchips may be due to the fact that the time allowed for the acclimation of the resident microbial population was not long enough, even though the acclimation period adopted in this study was informed by recommendations in literature. A report by USEPA (2003) stated that the natural method of allowing a wide variety of resident microbes to acclimatise to a particular mix of pollutants in the contaminated air may be a little longer, but the resultant microbial strains will be more adaptable in the long run. The poor odour performance of new woodchips could also be associated with the low moisture content when compared with other media types. Media moisture has a major influence on odour performance, and dry media has a potential to cause channelling which can lead to an increase in local drying along the preferential paths of airflow, allowing contaminated air to exit the biofilters untreated (Nicolai and Lefers, 2006a). This has the potential to reduce odour removal without necessarily affecting bioaerosol removal especially as bioaerosol particles can still impact on and be intercepted by the media particles.

8.3.3 Impact of media age on bioaerosols and odour removal performance

The performance of the old and new woodchips were compared in order to investigate the impact of media age on simultaneous control of odour and bioaerosols. This was possible because the woodchips were originally sourced from the same supplier who obtain their woodchips from the same stock of Larch and Spruce, thus the same properties are maintained except the age of woodchips used in the biofiltration. In this study, the concentration of bioaerosols in the outlet air from both media types varied between 10^3 to 10^4 cfu m^{-3} (up to a maximum of 10^5 cfu m^{-3} for total mesophilic bacteria emitted from the old woodchips), and these were slightly higher than the concentration encountered in open air as measured by the background concentration (10^2 to 10^3 cfu m^{-3}) for this study. Overall, comparison of the average outlet bioaerosol concentrations of both media (Table 8.3 and Figures 8.3 – 8.6) showed that the new woodchips tended to perform better than the old woodchips which were approximately 12 months old before use. This trend was reversed in terms of odour control with the old woodchips showing higher removals with outlet concentration of 102 to 230 OU_E m^{-3} compared to outlet levels of up to 249 OU_E m^{-3} for the new woodchips (Figure 8.5).

Table 8. 3: Outlet concentration of bioaerosols for old and new woodchips

		A. fumigatus	**Total fungi**	**Total mesophilic bacteria**	**Gram negative bacteria**
Old woodchips (cfu m^{-3})	Minimum	4.6×10^3	5.7×10^3	3.8×10^3	4.8×10^3
	Mean	2.1×10^4	3.1×10^4	2.1×10^4	1.5×10^4
	Maximum	7.9×10^4	1.3×10^5	4.9×10^4	4.0×10^4
New woodchips (cfu m^{-3})	Minimum	7.1×10^3	3.1×10^4	3.2×10^3	3.7×10^3
	Mean	1.3×10^4	1.6×10^4	1.3×10^4	7.9×10^3
	Maximum	2.3×10^4	1.6×10^4	2.4×10^4	1.3×10^4

The bioaerosol performance of the old woodchips is thought to be due to bioaerosol accumulation within the media over time, and eventual emission of these bioaerosols. For woodchips and other organic media, the estimated lifespan is 3 – 5 years or more (Schmidt et al., 2004). However, with time the media degrades due to microbial action; and combined with dust build-up and media settling, this can lead to pressure drop increase across the biofilter media bed (Fletcher et al., 2014). Also, as the media compacts with age, there may be an increase in particle and bioaerosols 'straining effect' from the airstream (similar to those observed for peat), thus resulting in more

microorganisms per unit mass of media available for odour biodegradation. And because there is tendency for the formation of preferential air paths when media compacts, it is possible to have some contaminated air flow through without bioaerosol impaction and interception as explained by Ottengraf and Konings (1991). Hence, the higher outlet bioaerosol concentrations measured with the old woodchips. This might also suggest that with the older woodchips the bioaerosols species going in may not necessarily be the ones that are coming out. However, this will require confirmation using molecular biology methods which is beyond the scope of this study.

Sanchez-Monedero et al. (2003) in their study assessing *A. fumigatus* and mesophilic bacteria removals by seven full-scale biofilters presented data which showed media age and the corresponding bioaerosol REs achieved. There appeared to be no clear association between *A. fumigatus* REs and media age especially as there was little variation in removal (97.9 to 99.4%) when media was between 1 and 12 months, although there was a drop to 90.4% at 18 months. Mesophilic bacteria removal showed a wider variation from biofilter to biofilter even for those with the same media age. One month old media had the lowest removal of 39.1% while other biofilters had mesophilic bacteria REs > 68%, with the oldest media (36 months) showing the highest RE of 36 month.

8.3.4 Impact of media type on bioaerosol particle size distribution

Apart from assessing bioaerosol concentration reduction by the various media biofilter types, it is also important to examine the size distribution of emitted bioaerosols especially as this ultimately plays a role in human exposure. The six stages of the Andersen sampler used for this study indicate the location where inhaled bioaerosol particles will be deposited once they penetrate the human respiratory tract (Tisch Environmental Inc., 2015).

Figure 8.7 shows a comparison of bioaerosol particle size distribution for the background, biofilter inlet and outlets of the four types of media types assessed in this study. The size distributions were computed by considering all samples taken at the various points indicated (Table E.2). Overall, considering all points sampled the distribution for *A. fumigatus* and total fungi were comparable while those of total mesophilic bacteria and Gram negative bacteria were also comparable.

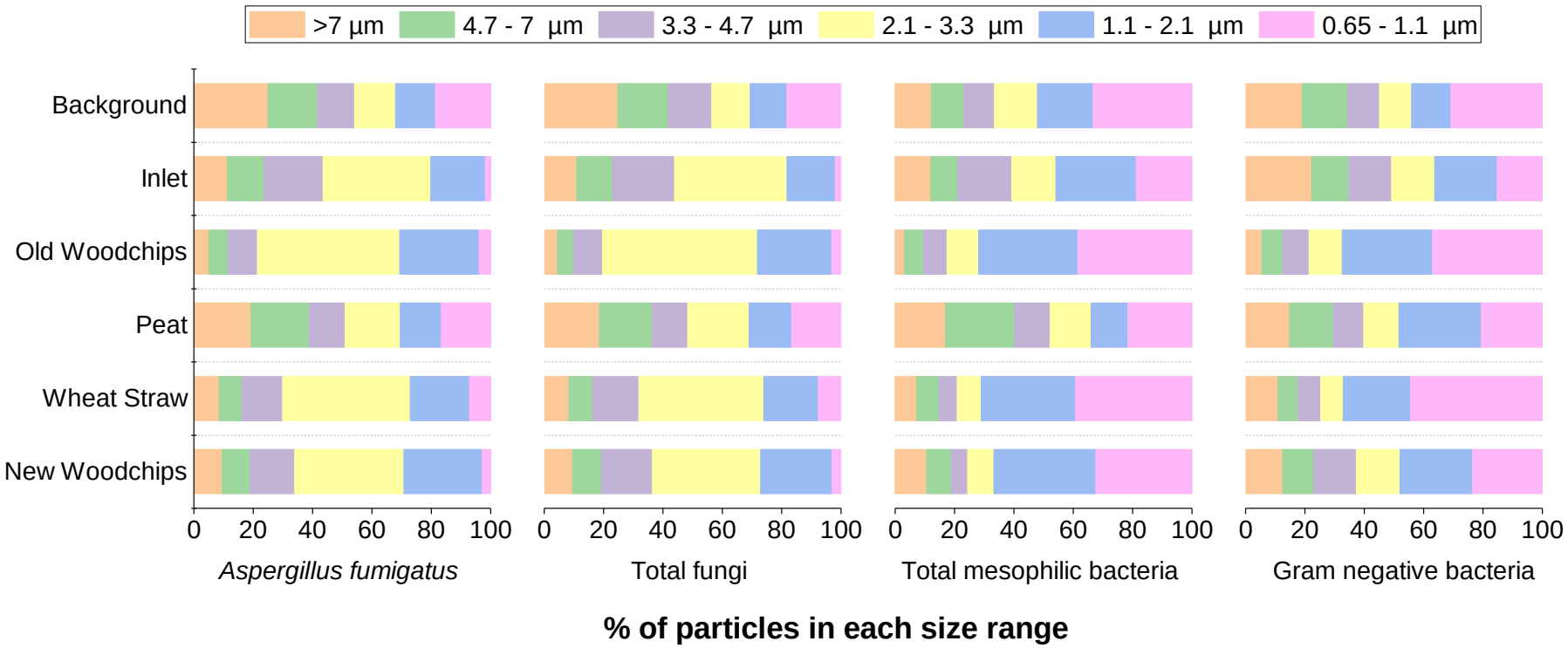


Figure 8. 7: Comparison of bioaerosol particle size distribution for the background, biofilter inlet and outlets of the four types of media. Data based on the four sampling visits.

The inlet air had approximately 20% of *A. fumigatus* particles in the range < 2.1 μm, slightly higher than the total fungi distribution with 18%. But the outlet had ~ 30% of particles in this size range for these two groups of bioaerosols. Peat delivered outlet concentrations that showed similar proportions as those of the background for both *A. fumigatus* and total fungi. This implies that peat outranks other media types in terms of its capacity to deliver outlet air that can be approximated to the background air both in terms of fungal particle concentration and size distribution. Table 8.4 shows that the outlets of old woodchips, wheat straw and new woodchips all had maximum fungal particles collected at stage 4 (aerodynamic diameter 2.1 to 3.3 μm), while peat showed maximum collection on stages 2 for *A. fumigatus* (20%) and 4 for total fungi (21%).

Table 8. 4: Stage and proportion of maximum deposition of bioaerosol particles

	A. fumigatus	**Total fungi**	**Total mesophilic bacteria**	**Gram negative bacteria**
Old woodchips	4 (48%)	4 (52%)	6 (38%)	6 (37%)
Peat	2 (20%)	4 (21%)	2 (23%)	5 (28%)
Wheat straw	4 (43%)	4 (42%)	6 (39%)	6 (44%)
New woodchips	4 (37%)	4 (36%)	5 (34%)	5 (24%)

With the exception of the peat whose outlet was composed of ~ 33% total mesophilic bacteria particles < 2.1 μm, all outlet air had ~ 46 to 71% of bacterial particles in the range < 2.1 μm. This implies that these media emitted air composed more of particles that can penetrate deep to the alveolar region, and thus present potential health concerns. Maximum bacterial particles deposited on stage 6 (for both old woodchips and wheat straw) and stage 5 for new woodchips. There was a variation with peat which showed a maximum collection on stage 2 for total mesophilic bacteria and stage 5 for Gram negative bacteria. Sanchez-Monedero et al. (2003) measured maximum deposition on stages 4 and 5 for *A. fumigatus* and mesophilic bacteria, respectively.

Given that a typical fungal cell is in the size range 2 – 10 μm and a typical size of a bacterium is 0.2 – 2 μm (Raisi et al., 2010; Miaskiewicz-Peska and Lebkowska, 2012; Qian et al., 2012), a key deduction from these results is that a large proportion of fungal particles in the outlet of the biofilters will highly likely be deposited in the upper respiratory tract while most bacterial particles are of the size range that would be deposited further down in the lower respiratory tract. Thomas (2013) argued that the size of pathogenic bioaerosol particles dictates where they are deposited within

the respiratory system; thereafter the potential to cause infection is dependent on the tissue tropism, clearance kinetics and the host immune system response.

The pulmonary or alveolar region is non-ciliated and particle clearance in this region is carried out by resident alveolar macrophages, specialised cells which carry out phagocytosis, a function which is hindered in immunocompromised persons. And because the expelled air from all the biofilters had more of the respirable particles of bacteria (< 2.1 μm), these may present health risks as they can get deep into the non-ciliated pulmonary or alveolar region. Larger particles deposited in the upper respiratory tract are cleared by the nasal and tracheobronchial escalators which is a combined mucociliary function of trapping deposited bioaerosol particles by mucus and removal by the action of cilia to the gastrointestinal tract (Thomas, 2013). Comparative studies on animal models show that greater numbers of larger particles are required to trigger infection in the upper respiratory tract in comparison the lower respiratory (Day and Berendt, 1972; Thomas et al., 2010; Thomas et al., 2012). This is possibly due to particle size dependent differences in pathogenesis between infections initiating in these two regions.

8.3.5 Media Cost

One of the advantages of adopting biofiltration is the relative low cost of treating high volumes of air containing low concentrations of a broad spectrum of chemical pollutants, in addition to low energy requirement and no generation of secondary pollutants which would require disposal (Devinny et al., 1999; Fulazzaky et al., 2014; Muñoz et al., 2015). Thus, the goal of designing a biofilter would be to control all emissions while keeping both capital and operating costs relatively low. The cost focus of this paper is on media which relates both to the capital cost (as part of the installation cost) and operating cost (due to media replacement). Assuming a typical full-scale biofilter installation with surface area 12.5 m^2 and media depth of 2.5 m, giving a total media volume of 31.25 m^3, the cost estimates of filling this volume by the various media is summarised in table 8.5.

Table 8. 5: Media cost estimates for full scale application

Media type	**Cost per m^3 (£)**	**Total volume (m^3)**	**Total cost (£)**
Woodchips	80	31.25	2500
Peat	100	31.25	3125
Wheat straw	23	31.25	719

Wheat straw has the lowest cost of all, showing a cost ratio of 1:3.5 and 1:4.3 with woodchips and peat, respectively. In selecting media, there needs to a balanced consideration of all factors because though wheat straw may appear cheap, peat has consistently delivered the highest reduction of odour and bioaerosol emissions. In this study, there was no need for microbial inoculation and nutrient addition to peat. However, Devinny et al. (1999) recommended microbial inoculation of peat media with activated sludge and nutrient addition since peat usually lacks these. These may add to media cost which must also be considered.

Media usage longevity is also another factor to consider. Media requiring frequent replacement may imply more cost spent on media as against media which last longer. However, most media including compost and woodchips have been estimated to have a lifespan of 3 – 10 years or more, but would have to be monitored for pressure drop differences (Nicolai and Lefers, 2006a). Schmidt et al. (2004) in qualifying useful lifespan of biofilter media noted that peat, heavy loamy soil and compost (yard waste) had good useful life while woodchips and straw were ranked as average and poor, respectively. Thus, wheat straw might not be a cheap choice in the long run.

8.4 Section Summary

- The four media types assessed in this study have demonstrated their potential to achieve appreciable levels of simultaneous control of bioaerosols and odour emitted from waste management facilities. The results indicated one to two log reduction for fungi and one log reduction for bacteria from inlet concentrations in the range of 10^4 cfu m^{-3} for both groups of microorganisms, while the odour removal also varied between the media, typically between -38 – 87 percent from a process air odour concentration up to 294 OU_E m^{-3}.
- Some media showed better removal of both while others show better removal either for odour or bioaerosols. Peat had the highest removal for both types of pollutants which is thought to be due to the moisture content and large surface area which favour the thriving of microbial population (particularly fungi) responsible for degrading the complex mix of chemical pollutants, thus reducing the outlet concentrations of odour recorded. The similar performance of peat for bioaerosols is thought to be due to the existence of disconnected dead-end pore spaces within the media which may act to filter bioaerosol particles from the influent air. Other media considered may not support

simultaneous control of the considered pollutants possibly because of differences in the removal mechanisms as some would favour bioaerosols removal by impaction while still having enough space for some chemical pollutants to exit the biofilters untreated, thus indicating preferentially controlling one type of pollutants against the other.

- The age of the media may actually impact on its capacity to control pollutants, and performance differences may exist with media age depending on what kinds of pollutants are considered. Old woodchips in this study may have, over time, had a build-up more microorganisms filtered from influent polluted airstream. Ultimately, these may breakthrough, adding to outlet concentrations measured and thus, presenting as reduced bioaerosol removal efficiency. However, the accumulated microbial population may contribute to the odour and chemical degrading potential of the older media.
- There seems to be no difference in the size of fungal particles emitted by the media as all showed highest deposition of fungal particles on stage 4 which is the range composed of particles (2.1 – 3.3 μm) that can be expelled by the mucocilliary clearance mechanism of the upper respiratory tract. On the other hand, the bulk of bacterial particles were of the size range (> 2.1 μm) that can penetrate deep into the alveoli which do not have this clearance mechanism. However, there are alveolar macrophages which fight off pathogenic species deposited in this region, except in immunocompromised persons in which case might result in serious health problems.
- Overall, the choice of any media has to be based on a balanced consideration of performance versus cost by operators. This is especially so as some media which offer high emission control may have huge cost implications, while the more affordable ones may not offer desired performance, and may at times prove to be more expensive to operate in the long run.

Chapter 9
GENERAL DISCUSSION

9.1 Introduction

This chapter presents a discussion of the data obtained from the sampling carried out throughout this study, and explores how these fit relative to the existing body of knowledge in the literature within the context of the original research questions presented in the introductory chapter.

9.2 Review of odour and bioaerosol emissions from waste management facilities

Biowaste treatment and management is and will continue to be a key component of the expanding waste management industry in the UK (Stagg et al., 2010; Environment Agency, 2018a). Essential to the biowaste treatment process is the role of microbial degradation of organic materials within the waste with the potential for negative consequence in the form of odour and bioaerosol emissions. Several factors have been suggested as being responsible for odour emissions from biowaste treatment facilities including the characteristics of the waste, the metabolic products of the aerobic degradation of the waste, some of the metabolic products of the anaerobic breakdown of the waste; all of which are impacted by the level of agitation of the waste by the site operations resulting in elevated ambient and point source emissions (Bidlingmaier and Müsken, 2007; Environment Agency, 2018a).

Emissions of odour and volatile organic compounds in the process air from waste management facilities have been widely studied. The choice of a MRF as the test facility in this study may not be considered ideal because they are known to be relatively clean compared to other types of facilities which have microbial decomposition as a fundamental part of their operation. However, this facility still provided process emissions with significant bioaerosols and comparatively lower concentrations of odour required to fulfil the relevant objectives of this study.

Emitted odour concentrations reported for full scale facilities in the literature vary; recent studies report figures of >2 million $OU_E\ m^{-3}$ by Frederickson et al. (2013) to 5 000 - 145 000 $OU_E\ m^{-3}$ by Fletcher et al. (2014). Gutiérrez et al. (2015) also reported peak odour concentration of 5224 $OU_E\ m^{-3}$ for a pile composting process treating the

organic fraction of municipal solid waste. They argued that this peak odour concentration was reached in parallel with peak microbiological activity. The odour concentration recorded for the inlet (used as a proxy for the indoor concentration) at this site was much lower (range: 94 – 489 OU_E m^{-3}) than those reported in the literature and were also low in comparison to concentrations reported by an independent odour consultant (mean of 19 340 OU_E m^{-3}) for the same site in a previous odour assessment (Gair, 2013). This difference has been thought to be due to factors such as sampling location, the odour sampling method employed, and the type of odour sampling bags used in this study (Capelli et al., 2013). Also, at the time of sampling all 11 modular air filtration units where in operation, and the data contained in the report of the independent odour consultant indicated that they achieved mean odour RE of 71% which was considered low for this type of filters (Gair, 2013). The reduced performance was attributed to factors including reduced residence time across the filter possibly resulting from uneven airflow; high temperatures which results in desorption of previously adsorbed compounds; high humidity which implies preference for water adsorption over adsorption of odorous compounds; clogging of filters by airborne particles; and saturation of the activated carbon. Nevertheless, data from this study agrees with the literature which observes that odour concentrations vary at the same site on different days. Cremiato et al. (2018) suggested that in MRFs odour resulted from diffused emissions of odour molecules from packaging waste such as liquids, detergents, food residues, etc. Thus, the variations in odour concentrations for different sampling days may be the result of the variation in the types and quantities of wastes processed in these facilities on a daily basis.

With regards to bioaerosols emissions from full scale waste management facilities, the information has been rather sparse, and available data reported in the literature are indicative of variability in sampling techniques. A summary of these is provided in Table 9.1 and shows that the inlet bioaerosols concentrations obtained in this study compare to the concentrations reported in the literature. The EA produced a Technical Guidance note (M9) which provides a standardised approach for bioaerosol monitoring at regulated facilities (Environment Agency, 2018a); it has been argued that as of February 2017, 106 bio-waste treatment facilities were affected by the revised monitoring requirements because they had houses, people or businesses within 250 m of their site boundaries (Regulatory Policy Committee, 2017).

The results of this study indicate that the concentration of *A. fumigatus* was one order of magnitude lower than concentrations previously quoted in the literature for these facilities, with a maximum value of 10^4 cfu m^{-3} compared to 10^5 cfu m^{-3}. The bacteria concentrations in this study were similar to the concentrations quoted in literature with a maximum value of 10^5 cfu m^{-3}, there are very few cases quoted in literature in which concentrations were higher at 10^6 cfu m^{-3}. Overall, it was observed that regardless of the types and volume of waste being processed, the concentrations of total mesophilic bacteria and Gram negative bacteria were higher than those of *A. fumigatus* and total fungi. This observation is consistent with the findings of Frederickson et al. (2013) and Fletcher et al. (2014) who suggested that exhaust air from the facilities they monitored contained more bacteria than fungi. MRFs are not expected to emit high concentrations of bioaersosols because they do not have organic decomposition as part of their process (Surrey County Council, 2017). Nonetheless, Stagg et al. (2013) stated that exposure to microorganisms (bacteria and fungi) in MRFs were considered to be of a medium level typically between $10^4 – 10^5$ cfu m^{-3}, and occasionally showed higher concentrations similar to those of animal houses at $>10^5$ cfu m^{-3}, and with identified species including *A. fumigatus* which is a known allergen – similar to concentrations reported in this study.

Table 9. 1: Concentration of bioaerosols in the process air reported in literature (updated from Fletcher et al. 2014)

System	**Waste**	**Bioaerosols**	**Concentration (cfu m^{-3})**	**Authors**
Various	Various	*A. fumigatus* Mesophilic bacteria	$10^2 – 10^5$ $10^3 – 10^5$	Sanchez-Monedero et al. (2003)
-	-	Mesophilic bacteria	10^5 - 10^6	Fischer et al. (2008)
-	-	*A. fumigatus*	$10^2 – 10^5$	Kummer and Thiel (2008)
Various	GW/FW	Bacteria Gram negative bacteria Fungi	$10^3 – 10^5$ $10^4 – 10^5$ $0 – 10^4$	Frederickson et al. (2013)
Various	Various	*A. fumigatus* Total bacteria Gram negative bacteria	$9 – 10^3$ $10^3 – 10^4$ $10^2 – 10^3$	Fletcher et al. (2014)
MRF	MSW	*A. fumigatus* Total fungi Total mesophilic bacteria Gram negative bacteria	$10^3 – 10^4$ $10^3 – 10^4$ $10^3 – 10^5$ $10^3 – 10^5$	This study

FW – Food waste, GW – Green waste, MSW – Municipal Solid Waste

9.3 Review of achievable odour and bioaerosol removals and the criticality of design and operating parameters

As previously established, the EA is responsible for regulating commercial waste management facilities and this is done through the granting of Permits to Operate. Part of the EA's remit is to ensure that odours and bioaerosols emitted from waste facilities do not adversely impact the workers and the surrounding population. When making decisions as to whether a permit should be granted the EA has to consider whether the proposed abatement system can effectively control emissions. The adoption of the Industrial Emissions Directive (IED) which was transposed into UK legislation by The Environmental Permitting (England and Wales) (Amendment) Regulations in 2013 has meant that regulators such as the EA need to fully understand the performance of such systems and the impact of design and operating parameters.

Table 9. 2: Achievable bioaerosols and odour removals with the range of parameters assessed

Parameters	**Removal Efficiencies achieved (%)**				
	Aspergillus fumigatus	**Total fungi**	**Total mesophilic bacteria**	**Gram negative bacteria**	**Odour**
EBRT					
11s	42 – 46	43 – 48	28 – 70	17 – 67	48 – 50
16s	25 – 94	32 – 93	35 – 70	58 – 89	48 – 55
70s	34	30 – 32	46 – 51	18 – 40	64 – 76
Moisture content					
40 – 70%	79 – 85	76 – 83	71 – 76	67 – 68	44 – 63
10 – 40%	65 - 78	67 – 76	74 – 76	66 – 77	34 - 42
Media depth					
0.50 m	58 – 89	56 – 89	22 – 90	39 – 82	-
0.25 m	30 – 83	34 – 83	33 – 87	59 – 80	-
Media type					
Old woodchips	57 – 86	57 – 86	29 – 85	44 – 83	8 – 65
Peat	88 – 99	86 – 99	65 – 96	56 – 92	22 – 87
Wheat straw	54 – 81	58 – 78	9 – 84	11 – 74	-38 – 48
New woodchips	3 – 73	10 – 76	48 – 91	52 – 81	-17 – 56

There is evidence in the literature supporting the potential to apply biofilters as abatement systems to reduce odour and VOCs emissions, and more recently bioaerosols emissions from commercial waste management facilities. However, to date only a few literature have assessed the criticality of the design and operating parameters of biofilters in delivering simultaneous control of odour and bioaerosols in the emissions from these facilities. Overall, this pilot study shows that biofilters

can achieve simultaneous reductions in the concentrations of odour and bioaerosols over the range of media design and operating parameters assessed (Table 9.2). Thus, the best combinations of these parameters will depend on a critical assessment of all parameters together since they are not mutually exclusive.

Empty bed residence time is one of the key parameters reported as being crucial for the design of biofilters targeted at odour control. From table 9.2 it can be seen that in this study a longer EBRT (70 s) delivers the best odour REs confirming the information presented in the literature. This is because there is sufficient time for the odorous contaminants to diffuse from the gas phase into the biofilm where the resident microorganisms can carry out biodegradation. However, at this EBRT the biofilters in this study were found to achieve the lowest bioaerosols REs and the shorter EBRT appears to favour bioaerosols removals over odour removals, leaving an EBRT of 16 s as ideal for both groups of contaminants based on this study. This is contrary to Fletcher et al. (2014) who recommended a much higher EBRT range of 40 – 100 s as appropriate for biofiltration of process air from biowaste treatment facilities. It has been suggested that different odorous pollutants have different characteristics which affect the time required for their absorption, adsorption and degradation (Chen and Hoff, 2009), thus it is imperative to have knowledge of the odour contaminants present in the process air against which to base selection of an adequate EBRT if simultaneous control of odours and bioaerosols is the goal.

Central to the operation of biofilters is the role of the microorganisms which must be maintained within the biofilm layers on the surface of the media bed particles. Morales et al. (2003) have argued that a layer of biofilm is made of 90 – 95% water; this indicates the criticality of water to the biodegradation function of biofilters. Although the two levels of moisture content assessed in this study achieved similar levels of bioaerosols reduction, clearly the higher moisture range (40 – 70%) delivered the best reduction for odour. Thus, it is recommended from this study that the maintenance of moisture levels critical for odour control will simultaneously achieve significant removals of bioaerosols which may not necessarily be moisture content related. It would seem that irrespective of the moisture levels, there will always be the interplay of the inertial deposition, diffusional (or Brownian) deposition and flow line interception – forces which are responsible for bioaerosol particles impaction within biofilter media (Ottengraf and Konings, 1991).

Media depth is another key design parameter which determines the odour biofiltration efficiency of a biofilter. Chen and Hoff (2009) recommended a media depth of 0.25 – 0.5 m as optimal for biofilter applied in agriculture, and argued that though higher media depth would deliver higher odour REs they may result in higher pressure drop across the biofilter media. In this study, these two (0.25 m and 0.50 m) levels of media depth were assessed for bioaerosols removals only. Although there were no statistically significant differences between the performances of these two depths, the 0.50 m media depth showed improved control of fungi compared to bacteria while the 0.25 m MD had better removals of bacteria than fungi. This observation regarding the higher media depth has been thought to be a function of the large surface area available for particle impaction; airflow rates and larger particles of fungi. Although Fletcher et al. (2014) recommended media depths up to 3 m for woodchips-based biofilters, the depths in this study which also employed woodchips as media have demonstrated significant reduction of bioaerosol particles in the process air from the MRF. This study agrees with the submissions by Fletcher et al. (2014) that media surface should be kept level devoid of undulations and weed growth which has the capacity to influence biofilter functionality be manually removed during routine maintenance. These conditions were maintained all through the study and have proven to be vital to the health of biofilters.

The last part of this study assessed the impact of media types on odour and bioaerosols biofiltration. It was observed that the peat media was consistent in delivering the highest simultaneous reduction of odour and bioaerosols possibly because of high content of moisture and large surface area which favour the support and growth of the odour degrading microbial population and the existence of disconnected dead-end pore spaces which may help in filtering bioaerosol particles from the process air. Nevertheless, peat may not be an economical option for operators because of the relative high cost per m^3 of biofilter bed in comparison to other media types assessed in this study. The performance of the wheat straw was the poorest both in terms of bioaerosols and odour concentration reductions. Data from this study indicate that the performance of woodchips may improve over time especially as the one year old woodchips indicated better removals than the new woodchips which were freshly acquired for this study. Data presented by Sanchez-Monedero et al. (2003) tend to support this view as one month old media showed the lowest removal of 39.1% for

mesophilic bacteria while other biofilters had mesophilic bacteria REs > 68%, with the oldest media (36 months) showing the highest RE of 94.2%.

Another vital aspect of this study that has not been extensively reported in literature is the impact of biofilter design and operating parameters on the size distribution of bioaerosol particles between the untreated process air (biofilter inlet) and treated air (outlet). The reference size used in this study was 3.3 μm because this is the cut off size which distinguishes between the larger conglomerates of cells and the smaller single cells which can penetrate deep into lower respiratory tract where they can pose health concerns (Fröhlich-Nowoisky et al., 2016). Variations of both ERBT and media moisture content do not appear to have obvious impacts on size distribution especially as the outlet samples tended to mirror the inlet samples in size distribution. With the media depth, the data shows that the higher media depth preferentially filtered more of the fungal particles; an observation which is attributed to the larger surface presented for particle impaction and interception, resulting in the lowered outlet concentration of fungi. In addition, Trunov et al. (2001) also suggested that fungal spores are often aerosolised in agglomerates which have a higher inertia than single spores and therefore are more likely to be impact on surfaces. This trend seems to be reversed for bacteria as the high media depth tends to release more bacterial particles. This is thought to be size-related as the smaller, lighter and spiny bacterial cells tend to remain in the flowing airstream without interception (Andersen Instruments, 1984; Wang et al., 2018). In addition to this is the contribution from the proportion of bacteria blown off from the media materials as the air flows through the filter bed which has high composition of microorganisms – up to 10^7 microorganisms/g of media material (Fletcher et al., 2014). Also, the particle size distribution of the outlet air samples from the four types of media assessed were comparable with maximum deposition of fungal particles on stage 4 and maximum deposition of bacterial particles on stages 5 and 6 of the six stage Andersen sampler. Overall, all outlet samples assessed against the various parameters assessed in the study had at least 50% of particles size in the range less than 3.3 μm. In spite of the high REs achieved in this study (up to 94% for *A. fumigatus*, up to 93% for total fungi, up to 90% for total mesophilic bacteria and up to 89% for Gram negative bacteria), the measured outlet concentrations still exceed background (upwind) concentration, and are often in excess of the concentration guideline provided in the EA position statement. They tend to be mainly composed of a size range that can penetrate deep into the lungs and

therefore, might be of concern to site workers and members of public living in the vicinity of site if these biofilters were operated at full scale.

9.4 Review of suggested candidate biofilter for bioaerosol and odour abatement at enclosed waste management facilities

The literature review has identified a range of design and operating parameters considered as critical to biofilter performance, including media pH and alkalinity, operating temperature, waste air flow, bed void volume, moisture content, nutrients, gas residence time, media moisture content. Table 9.3 shows a summary of some of these parameters considered to represent indicative best available technologies (BAT) and the operational ranges as defined by two key studies (Devinny et al., 1999 and Fletcher et al., 2014), and how they compare to the range reported in the current study. Fletcher et al. (2014), however, did point out that these parameters have been mainly defined on the basis of research on odour and VOC removal and so may not be critical for bioaerosol removal since the mechanisms of removal of both types of pollutants are different.

Nevertheless, data from this study has provided more insights into aspects of biofilters that have not been well studied. Although some parameters in table 9.3 were outside the scope of this study, values reported agree with some of the operational ranges reported by Devinny et al. (1999) and Fletcher et al. (2014) especially for surface loading rates, volumetric loading rates, media pH and operational temperature. It would seem, therefore, that based on the data from this study the ideal biofilter to simultaneous control bioaerosols and odour would be a woodchips-based reactor operated with a minimum media depth of 0.50 m and an EBRT of 16 s maintained at a moisture content of between 40 and 70%, all of which lie within operational ranges reported in literature. This further affirms the view that some parameters critical for odour reduction may not be as critical to bioaerosols reduction.

It is important to note that many parameters may impact on the results of biofiltration studies, and in some cases direct correlations have been established. However, a knowledge of all of these is required to interpret the results of any biofiltration studies. The ranges presented in table 9.3 reflect recommended values based on this study; however, it is not impossible to achieve biofilter effectiveness for both bioaerosols and odour reductions outside the ranges reported in this study.

Table 9. 3: Suggested design and operational criteria for biofilters

Design and Operating Parameters	**Devinny et al. (1999)**	**Fletcher et al. (2014)**	**This study**
Media type		Biologically active; organic matter content >60%	Woodchips
Media porosity	50%	75 – 90%	Minimum of 60%
Media bulk density	-	-	225 Kg m^{-3}
Media moisture content	60%	60 – 75%	40 – 70%
Media depth (height)	1 – 1.5 m	Up to 3 m for woodchips	> 0.50 m
Surface loading	5 - 500 m^3 m^{-2} hr^{-1}	<500 m^3 m^{-2} hr^{-1}	26 - 164 m^3 m^{-2} hr^{-1}
Volumetric loading	5 - 500 m^3 m^{-3} hr^{-1}	5 - 500 m^3 m^{-3} hr^{-1}	52 - 327 m^3 m^{-3} hr^{-1}
Inlet air temperature	15 - 30ºC	15 - 30ºC	13 - 27ºC
Outlet air temperature	15 - 30ºC	<50ºC	11 - 22ºC
Media temperature	15 - 30ºC	-	11 – 23ºC
Media pH	6 - 8	6 – 8.5	5.5 – 8.0
Mean effective gas residence time	15 – 60 seconds	40 – 100 seconds	10 – 70 seconds
Air distribution	Top or bottom loaded using plenum	Using plenum chamber or distributed pipe work; up-flow or down-flow configuration	Through a plenum, up-flow configuration

Also, biofilter testing often occurs within a few minutes or hours and in controlled environments; however, these are systems that are designed to run for several months (and even years), and so may be exposed to a range of environmental changes and variations (including temperatures, humidity, airflow rates, odour/VOCs concentrations and bioaerosol particle concentrations). Adding to this complexity is the technical variations in design as well as the fluctuations in the maintenance regimes put in place by operators. Thus, the results are useful in providing insights into the basic operation of biofilter systems.

As already stated, before the EA grants Permits to Operate to new facilities, it considers the kind of abatement system to be installed and whether these have the capacity to effectively control all emissions. However, many bio-waste treatment facilities across the UK are already using biofilters with varying degrees of success. Thus, there was the notion as to whether existing biofilters needed to be modified to reflect parameters which allow for the highest levels of performance or whether entirely new systems were required to effectively deal with odour and bioaerosols emissions. This study has demonstrated that the existing range of parameters recommended in the literature, particularly those of Devinny et al. (1999) and Fletcher et al. (2014) have proven to achieve significant reductions of both bioaerosols and odour. Thus, there is no need to alter or modify the design and operating parameters of existing biofilters if these conform to the recommendations. This is because biofiltration of bioaerosols seems to be a passive function of biofilters during active biofiltration of odours. Hence, even though design and operating parameters critical for odour biofiltration may not be as critical for bioaerosols, there will always be simultaneous reduction of bioaerosols. However, the caveat with this is that there may be extra contamination of the outlet air with microbial populations resident within the biofilter which may be completely different from the species entering it. Nonetheless, this is a significant contribution to the existing literature and insight especially for waste management regulators and operators who have been looking forward to clear guidance on the key design and operating parameters together with maintenance requirements for effectively removing odour and bioaerosols from process air.

9.5 Applicability of results

The application of a pilot scale biofilter in this study has provided new insights into bioaerosols removal including relationships between operating parameters, removal efficiency, size distribution of microorganism in air and potential for emissions. The application of the pilot scale biofilter within an MRF facility provided an opportunity to collect data that is closer to real-world biofilter operation than a highly controlled laboratory study. However it should be noted that this brings with it some uncertainties and limitations and hence the results of this study may not allow for generalisation of conclusions for various reasons.

Although biofilters have been applied to MBT plants (Stagg et al., 2013), they are less common in MRF plants. As stated in section 9.2, these facilities are generally fairly clean and do not have the levels of organic dust and odour found in facilities such as in-vessel composting (IVC) (Surrey County Council, 2017) where biofilters are likely to be better suited. The location of the pilot system (away from the waste heap behind the back-push wall) is another factor which could have impacted on the results especially as this area was relatively cleaner compared to other areas within the waste hall. For these reasons, the measured bioaerosol concentrations have to be considered relative to those levels typical in facilities that generate much higher concentrations, and it is not clear whether the same findings would be apparent under much higher bioaerosols loads. Nonetheless, this study was based in this MRF to enable the evaluation of the system with the real source of bioaerosols associated with this type of waste being processed, and hence the findings are likely to be applicable to other comparable MRF facilities.

Secondly, the bioaerosol sampling methods employed in this study were targeted at assessing total bioaerosol loads removal by biofilters, and bioaerosols size distributions, rather than specifically identified pathogenic species which would have been more relevant for occupational exposure risk assessment. Literature suggests that some of the species released at the outlet may in fact have originated from within the biofilter (Martens et al., 2001; Frederickson et al., 2013) and so techniques such as this that focus more on the general microbial concentrations may miss out important trends that should be studied complementarily. It was also technically infeasible to collect inlet and outlet samples concurrently; this would have given a better prediction of bioaerosol removal. Although the time difference between inlet and outlet sample

collection was minimised as far as practical, some of the bioaerosol concentration variations may be due to fluctuating levels of activities within the waste hall on a short timescale.

Also, it is recommended that air samples for odour analysis be collected using the Sampling Lung Technique which entails direct air collection through Teflon lined tubing into a gas bag placed within a rigid, leak proof container using a leak-free pump (Bokowa, 2008; Bokowa, 2009). This technique allows the gas bag to be filled up at a rate that equals the rate at which the vacuum in the container is evacuated, and in the process only the Teflon tube and the gas bag are in contact with the air sample (da Rocha Carmo Junior et al., 2010). However, it was not economically feasible to adopt this sampling method because of the cost associated with acquiring the sampling lung kit. Hence, the improvisation by direct sampling into the gas bags at the top of each biofilter/plenum. A minimum of duplicate samples is expected for odour analysis (Hove et al., 2016), however, this study was limited to single samples due to cost. This research takes into consideration that smaller sample sizes have been reported to result in a much wider confidence interval and more variability (McGinley and McGinley, 2006).

All pilot-scale biofilters in this study were carefully and regularly monitored to ensure they were operating optimally during the investigations, especially regarding water content and prevention of media compaction. As such the results may not allow conclusions to be drawn regarding poorly maintained systems. Although the biofilters were fed with air containing significantly high concentrations of bioaerosols, there were occasional operational interruptions (e.g. waste hall cleaning, conveyor shut down, decreased machine/vehicle activities) during sampling which could have varied the measured inlet concentrations (Stagg et al., 2013). Thus, the data presented in this thesis especially for bioaerosols are indicative of a high degree of variability between the replicate samples taken for each bioaerosol group at each point as shown by the error bars. This suggests that there is a high degree of measurement uncertainty, which may have led to the variable results recorded in this study. As such, the data presented here gives an insight into the likely influence of operating parameters, however further research is required to make more specific conclusions on the performance of biofilters, and particularly the mechanisms for bioaerosols removal.

Chapter 10
CONCLUSIONS AND RECOMMENDATIONS FOR FUTURE RESEARCH

10.1 Conclusions

This study was conducted to provide answers to some key questions and to address the gaps in knowledge required to define a robust set of design and operating criteria for biofilters to optimise the simultaneous removal of odour and bioaerosols from the process air of waste management facilities as presented in chapter one. In this study, a pilot biofiltration system consisting of four plastic reactors was constructed and tested at a live MRF which acted as a source of polluted process air with detectable levels of odour and bioaerosols concentrations.

The biofiltration systems were sampled over a period of approximately 14 months (May 2016 to July 2017) during which the impacts of the parameters of interest were tested. Analysis was undertaken either for bioaerosols alone or bioaerosols and odour (in OU_E m^{-3}). Four groups of bioaerosols were selected to cover the range specified in literature; this include *Aspergillus fumigatus*, total fungi, total mesophilic bacteria and Gram negative bacteria. This site was chosen because of the potential for significant odour and bioaerosols emissions based on previous unpublished study and a report of independent odour consultant on a study conducted on site. The key parameters that were considered were the impact of gas residence time, media moisture content and media types on simultaneous reduction of bioaerosols and odour. The impact of media depth on the reduction of bioaerosols was also assessed. It is important to state that apart from the parameters which were investigated, all of the pilot scale biofilters sampled as part of this study were maintained within operational ranges reported in literature, especially with regards to media condition and particle size, surface loading rates, volumetric loading rates, media pH and operational temperature.

Although a number of specific observations and conclusions have been made in each of the result chapters, the key conclusions of this study are arranged under three themes and presented below.

Emissions of bioaerosols and odour from waste management facilities

- The concentration of bioaerosols in the process air (as indicated by the inlet air samples) varied from visit to visit in the range of $10^3 – 10^5$ cfu m^{-3}. This was thought to be due to the complex interactions between the specific process operating conditions, the types of waste being processed and the configuration of the air ventilation system installed on the site.
- The concentration of odour in the process air also varied between visits typically ranging from 94 to 489 OU_E m^{-3}, although it remained fairly stable during the EBRT impact assessment study at 152 OU_E m^{-3}. It would appear that the low inlet odour concentration is a function of factors such as sampling location, the odour sampling method employed, and the type of odour sampling bags used. It could also be due to the fact that the ventilation systems were fully operational throughout the sampling period.
- Overall, the data shows that regardless of the types and volume of waste being processed, the concentrations of total mesophilic bacteria and Gram negative bacteria were higher than those of *A. fumigatus* and total fungi. The results of this study indicate that inlet *A. fumigatus* concentration was one order of magnitude lower than the concentrations quoted in the literature for these facilities, with a maximum value of 10^4 cfu m^{-3} compared to 10^5 cfu m^{-3}. The inlet bacteria concentrations in this study was similar to concentrations quoted in literature with a maximum value of 10^5 cfu m^{-3}, only in few cases were figures quoted in literature higher at 10^6 cfu m^{-3}.
- Inlet odour concentrations showed negative correlations with the four groups of bioaerosols assessed. Comparing the bioaerosol groups, *A. fumigatus* showed a strong positive correlation with total fungi and with mesophilic bacteria, and a weak negative correlation with Gram negative bacteria. However, due to the limited samples collected in the study, it cannot be concluded whether or not these correlations are substantial.

Biofilter emissions and reduction efficiencies of bioaerosols and odour

- The concentration of bioaerosols and odour emitted from the pilot scale biofilters varied from visit to visit, and between the biofilters. For most cases, the inlet bioaerosol concentrations were statistically significantly higher than the outlet concentrations.

- Overall, this study shows that biofilters designed and operated for odour degradation can also achieve significant bioaerosol reductions in the process air of waste treatment facilities, provided that the inlet concentration is high - which is the case for most waste treatment facilities. Thus, biofilters can be effective for the control of potentially pathogenic species in the emissions from such treatment facilities.
- Based on data from this study, the performance of these pilot scale biofilters in terms of bioaerosols reduction efficiency was variable from visit to visit and between the biofilters. Overall, the data suggests that bioaerosol removal efficiency may not always be a good indicator of biofilter performance and should be evaluated in combination with other biofilter performance indicators, such as bioaerosol emission concentration. This is because even though the biofilters achieved high bioaerosol removals, the emitted concentrations still exceeded background concentrations and the EA guideline limits.
- The data showed that differences may exist between the ability of biofiltration systems to deal with fungi and bacteria, as there is much more confidence with the performance for bacteria than fungi.
- Bioaerosols removals may deteriorate at low inlet concentration resulting in a net bioaerosol emitting potential of biofilters, and a proportion of the emitted bioaerosols may be originating from the microbial population colonising the media surfaces, resulting in possible potential differences in species composition between contaminated process (inlet) and treated (outlet) air samples.
- Overall, particle size distribution varied between the inlet and outlet air, with the outlet having predominantly greater proportion of smaller size particles (3.3 μm) that represent greater human health risk as they can penetrate the respiratory system more deeply, and even to the lung alveoli where gaseous exchange occurs. However, the outlet concentrations were low, and further reduction would be achieved by the combined effect of wind dilution and dispersal as well as exposure to environmental stress from temperature, desiccation and oxygen in full scale applications.

Impact of biofilter design and operating parameters on bioaerosols and odour reductions

- Based on the data from this study, it appears that variations in gas residence time may not impact on the bioaersosols removals; thus, gas residence time may not be critical for bioaerosol control. However, longer EBRT delivered significant ($p < 0.05$) reductions of odour than the shorter EBRT, implying that the longer EBRT accommodates the time required for both odorous contaminants diffusion transfer from the gas phase into the biofilm, and their subsequent biodegradation within the biofilm layer on the media materials, as established in literature.
- There appears to be no media moisture content dependent differences ($p > 0.05$) in the bioaerosols reductions reported in this study. On the other hand, although not statistically significant ($p > 0.05$), differences did exist in odour performance between the two groups, with the higher moisture content (40 – 70%) consistently showing better removals (odour RE range of 44 – 63%) than media moisture content in the range of 10 – 40%.
- The two media depths (0.50 m and 0.25 m) investigated showed potential capacity to control bioaerosols emissions from the process air of the MRF in this study, and possibly other waste treatment facilities. Both depths achieved significant ($p < 0.05$) reductions of the inlet concentrations of bioaerosols as measured at the outlet. Although there were no statistically significant differences between the performances of both MDs, the 0.5 m media depth showed improved control of fungi than bacteria while the 0.25 m MD had better removals of bacteria than fungi. The observation with the high media depth has been thought to be a function of the large surface available for particles impaction; airflow rates and larger particles of fungi.
- From the data, there was variation in the performance of the different media types assessed. Peat consistently delivered the highest simultaneous reduction of odour and bioaerosols; however, this was a much more expensive option. The performance of the wheat straw was the poorest both in terms of bioaerosols and odour concentration reductions. Woodchips appeared to be the preferred choice particularly because it is relatively cost effective and offered satisfactory odour and bioaerosol removals (though not as high as peat).

- Data from this study also indicate that the performance of woodchips may improve over time especially as the one year old woodchips indicated better removals of odour and bioaerosols than the new woodchips which were freshly acquired for this study.
- Overall, data from this study suggest that the ideal biofilter to simultaneously control bioaerosols and odour would be a woodchips-based reactor operated with a minimum media depth of 0.50 m and an EBRT of 16 s maintained at a moisture content level of between 40 and 70%, all of which lie within operational ranges reported in literature.

10.2 Knowledge gaps and future research

This research has produced some important contributions to the current knowledge on gaseous emissions from enclosed waste management facilities, particularly MRFs, and has also generated valuable insights regarding the criticality of key parameters on the performance of biofilters in terms of emissions control potential. Nonetheless, some knowledge gaps have been identified which require further research to provide a more comprehensive understanding of the workings of biofilters to effectively deliver simultaneous control of all emissions. These are presented in this section.

- Information from this study indicated that biofilters demonstrated the potential to simultaneous control odour and bioaerosols. However, it is still unclear whether the species of bioaerosols at the inlet are the same species leaving the biofilters at the outlet. Thus, it becomes imperative to conduct research employing quantitative polymerase chain reaction (qPCR) and next-generation sequencing (NGS) to compare the species composition of both inlet and outlet air to determine whether or not resident microbial populations are being emitted from biofilters.
- A key challenge in this study was the limited sample size which played a role in limiting the significance of some of the statistical comparisons conducted. Also, this study was conducted at pilot scale which has the bias of the controlled environment. Thus, a more extensive study is required with full scale biofilters in a live enclosed biowaste treatment facility to assess whether or not the results obtained in this study can be reproduced.

- This study compared the performance of three media types – peat, wheat straw and woodchips (one month old and one year old). However, further research is required to evaluate the performance of media mixes and combinations of different media on the simultaneous reduction of bioaerosols, odour and VOCs.
- This study also provided significant information regarding the performance of some design and operating parameters – gas residence time, media moisture content, media depth, media types as well as inlet bioaerosols concentration. However, the data did not provide clear insights into the performance of parameters such as biofilter temperature, media pH, absorptivity and oxygen limitation, contaminant load and surface load. Thus, further research is required to assess how critical these parameters are to biofiltration of bioaerosols, odour and VOCs. This will ensure that boundary conditions between optimally operated and poorly operated biofilters are clearly established.
- As identified in the study by Fletcher et al. (2014), there seems to be a contradiction in information on ammonia toxicity within biofilters. Unfortunately, it was not possible to investigate this aspect of biofilter operation which is of concern especially at composting sites. Thus, further research is required to establish how robust biofilters are with respect to dealing with elevated ammonia concentrations and to determine how critical ammonia toxicity is to biofilter performance.
- Data obtained in this study suggest that biofilters can be net emitters of biofilters at low inlet concentration, and even with the high removals achieved at high inlet concentrations the emitted concentration still exceed the guideline provided by the UK Environment Agency. Further research is required to evaluate the potential of using post biofiltration scrubbers to remove bioaerosols.

References

Adler, S.F. 2001. *Biofiltration – a Primer* [Online]. [Accessed 16 Jan]. Available from: http://www.d.umn.edu/~rdavis/courses/che4601/notes/biofiltration.pdf

Akdeniz, N., Janni, K.A. and Salnikov, I.A. 2011. Biofilter performance of pine nuggets and lava rock as media. *Bioresource technology.* **102**(8), pp.4974-4980.

Albrecht, A., Witzenberger, R., Bernzen, U. and Jackel, U. 2007. Detection of airborne microbes in a composting facility by cultivation based and cultivation-independent methods. *Annals of Agricultural and Environmental Medicine.* **14**(1), p81.

Alonso, C., Zhu, X., Suidan, M.T., Kim, B.R. and Kim, B.J. 1999. Mathematical model for the biodegradation of VOCs in trickle bed biofilterss. *Water Science and Technology.* **39**(7), pp.139-146.

Álvarez-Hornos, F., Gabaldón, C., Martínez-Soria, V., Martín, M., Marzal, P. and Penya-Roja, J. 2008. Biofiltration of ethylbenzene vapours: influence of the packing material. *Bioresource technology.* **99**(2), pp.269-276.

Andersen, A.A. 1958. New sampler for the collection, sizing, and enumeration of viable airborne particles. *Journal of Bacteriology.* **76**(5), p471.

Andersen Instruments. 1984. Operating Manual for the Andersen Sampler: Viable (Microbial) Particle Sizing Sampler. *Andersen Samplers, Inc, Atlanta, GA.*

ATSDR. n.d. *Medical Management Guidelines for Hydrogen Sulfide (H_2S). Agency for Toxic Substances & Disease Registry General Information: Hydrogen Sulfide (H_2S) CAS 7783-060-4; UN 1053*

AWM. 2011. *Associated Waste Management Media* [Online]. [Accessed 11 Jun]. Available from: http://www.awm.uk.com/media#/1/zoomed

Barzgar, S., Hettiaratchi, J.P., Pearse, L. and Kumar, S. 2017. Inhibitory effects of acidic pH and confounding effects of moisture content on methane biofiltration. *Bioresource Technology.* **245**, pp.633-640.

Beck-Friis, B., Smårs, S., Jönsson, H. and Kirchmann, H. 2001. SE—Structures and Environment: Gaseous Emissions of Carbon Dioxide, Ammonia and Nitrous Oxide from Organic Household Waste in a Compost Reactor under Different Temperature Regimes. *Journal of Agricultural Engineering Research.* **78**(4), pp.423-430.

Bidlingmaier, W. and Müsken, J. 2007. Chapter 11 Odor emissions from composting plants. In: L.F. Diaz, M.d.B.W.B. and Stentiford, E. eds. *Waste Management Series.* Elsevier, pp.215-324.

Blazy, V., de Guardia, A., Benoist, J.C., Daumoin, M., Guiziou, F., Lemasle, M., Wolbert, D. and Barrington, S. 2015. Correlation of chemical composition and odor concentration for emissions from pig slaughterhouse sludge composting and storage. *Chemical Engineering Journal.* **276**, pp.398-409.

Blazy, V., de Guardia, A., Benoist, J.C., Daumoin, M., Lemasle, M., Wolbert, D. and Barrington, S. 2014. Odorous gaseous emissions as influence by process condition for the forced aeration composting of pig slaughterhouse sludge. *Waste Management.* **34**(7), pp.1125-1138.

Blomquist, G. 1994. Sampling of biological particles. *Analyst.* **119**(1), pp.53-56.

Bokowa, A. 2008. *Guidelines for odor sampling for point, area and ambient sources with evaluation using the dynamic dilution olfactometry by AWMA EE 6 Subcommittee on the Standardization of Odor Measurement.* ASTM E18 Sensory Committee.

Bokowa, A.H. 2009. Ambient odour assessments: Comparison of two techniques used for assessing ambient odours. In: Frechen, F.-B. ed. *Odours and VOCs: Measurement, Regulation and Control* Kassel: Kassel University Press GmbH (kup).

Brincat, J.-P., Sardella, D., Muscat, A., Decelis, S., Grima, J.N., Valdramidis, V. and Gatt, R. 2016. A review of the state-of-the-art in air filtration technologies as may be applied to cold storage warehouses. *Trends in Food Science & Technology.* **50**, pp.175-185.

Buck, L.B. 2000. The Molecular Architecture of Odor and Pheromone Sensing in Mammals. *Cell.* **100**(6), pp.611-618.

Bull, M., McIntyre, A., Hall, D., Gordon, A., Redmore, J., Pullen, J., Caird, L., Stoaling, M. and Fain, R. 2014. *IAQM Guidance on the assessment of odour for planning* London: Institute of Air Quality Management. www.iaqm.co.uk/text/guidance/odourguidance-2014.

Busca, G. and Pistarino, C. 2003. Abatement of ammonia and amines from waste gases: a summary. *Journal of Loss Prevention in the Process Industries.* **16**(2), pp.157-163.

Cabrol, L., Malhautier, L., Poly, F., Lepeuple, A.-S. and Fanlo, J.-L. 2012. Bacterial dynamics in steady-state biofilters: beyond functional stability. *FEMS microbiology ecology.* **79**(1), pp.260-271.

Calaf-Forn, M., Roca, J. and Puig-Ventosa, I. 2014. Cap and trade schemes on waste management: A case study of the Landfill Allowance Trading Scheme (LATS) in England. *Waste Management.* **34**(5), pp.919-928.

Capelli, L., Sironi, S. and Del Rosso, R. 2013. Odor sampling: techniques and strategies for the estimation of odor emission rates from different source types. *Sensors.* **13**(1), pp.938-955.

Chang, D., Lee, S., Choi, W. and Lee, S. 2004. A pilot-scale biofilter system to reduce odor from swine operation. In: *ASAE Meeting Paper*.

Chang, K. and Lu, C. 2003. Biofiltration of isopropyl alcohol and acetone mixtures by a trickle-bed air biofilter. *Process Biochemistry.* **39**(4), pp.415-423.

Chen, L. and Hoff, S. 2009. Mitigating Odors from Agricultural Facilities: A Review of Literature Concerning Biofilters. *Agricultural and Biosystems Engineering.* **25**(5), pp.751 - 766.

Chen, L., Hoff, S., Cai, L., Koziel, J. and Zelle, B. 2009. Evaluation of wood chip-based biofilters to reduce odor, hydrogen sulfide, and ammonia from swine barn ventilation air. *J Air Waste Manag Assoc.* **59**(5), pp.520-530.

Chen, L. and Hoff, S.J. 2012. A Two-Stage Wood Chip-Based Biofilter System to Mitigate Odors from a Deep-Pit Swine Building. *Applied Engineering in Agriculture.* **28**(6), pp.893 - 901.

Chen, L., Hoff, S.J., Cai, L. and Koziel, J.A. 2008a. Odor reduction during biofiltration as affected by air flow rate and media moisture content.

Chen, L., Hoff, S.J., Koziel, J.A., Cai, L., Zelle, B. and Sun, G. 2008b. Performance evaluation of a wood-chip based biofilter using solid-phase microextraction and gas chromatography–mass spectroscopy–olfactometry. *Bioresource Technology.* **99**(16), pp.7767-7780.

Christen, P., Domenech, F., Michelena, G., Auria, R. and Revah, S. 2002. Biofiltration of volatile ethanol using sugar cane bagasse inoculated with Candida utilis. *Journal of hazardous materials.* **89**(2-3), pp.253-265.

Chung, W.J., Jin, C.S., Lee, T.J., Kwon, E.M., Jeong, S.G., Park, J.K. and Galera, M.M. 2003. Development of a two-step biofiltration system for the control of odorous emission from composting process of food waste. In: IWA ed.

Proceedings of the Second IWA International Workshop and Conference on Odour and Volatile Organic Compounds. Singapore. pp 14 - 17, pp.14 - 17.

Chung, Y.-C., Ho, K.-L. and Tseng, C.-P. 2007. Two-Stage Biofilter for Effective NH3 Removal from Waste Gases Containing High Concentrations of H2S. *Journal of the Air & Waste Management Association.* **57**(3), pp.337-347.

Chung, Y.-C., Huang, C. and Tseng, C.-P. 1996. Operation optimization of Thiobacillus thioparus CH11 biofilter for hydrogen sulfide removal. *Journal of Biotechnology.* **52**(1), pp.31-38.

Chung, Y., Huang, C., Pan, J. and Tseng, C. 1998. Comparison of Autotrophic and Mixotrophic Biofilters for H2S Removal. *Journal of Environmental Engineering.* **124**(4), pp.362-367.

Chung, Y.C., Huang, C. and Li, C.F. 1997. Removal characteristics of h2s by thiobacillus novellus ch 3 biofilter in autotrophic and mixotrophic environments. *Journal of Environmental Science and Health . Part A: Environmental Science and Engineering and Toxicology.* **32**(5), pp.1435-1450.

Clark, O., Edeogu, I., Feddes, J., Coleman, R. and Abolghasemi, A. 2004. Effects of operating temperature and supplemental nutrients in a pilot-scale agricultural biofilter. *Canadian Biosystems Engineering.* **46**(6), pp.7-16.

Colón, J., Martínez-Blanco, J., Gabarrell, X., Rieradevall, J., Font, X., Artola, A. and Sánchez, A. 2009. Performance of an industrial biofilter from a composting plant in the removal of ammonia and VOCs after material replacement. *Journal of Chemical Technology & Biotechnology.* **84**(8), pp.1111-1117.

Concept Life Sciences. 2017. *Odour.* [Online]. [Accessed 14 November]. Available from: https://www.conceptlifesciences.com/capability/odour/

Coolican, H. 2017. *Research methods and statistics in psychology.* Psychology Press.

Cornish, R. 2006. *Statistics: An introduction to sample size calculations. Mathematics Learning Support Centre.* [Online]. [Accessed 23 January 2017]. Available from: http://www.statstutor.ac.uk/resources/uploaded/sample-size.pdf

Cox, C.S. and Wathes, C.M. 1995. *Bioaerosols handbook.* crc press.

Cox, H.H.J. and Deshusses, M.A. 1998. Biological waste air treatment in biotrickling filters. *Current Opinion in Biotechnology.* **9**(3), pp.256-262.

Cremiato, R., Mastellone, M.L., Tagliaferri, C., Zaccariello, L. and Lettieri, P. 2018. Environmental impact of municipal solid waste management using Life Cycle Assessment: The effect of anaerobic digestion, materials recovery and secondary fuels production. *Renewable Energy.* **124**, pp.180-188.

Curtis, D. 2009. *Standards and Methods for Environmental Monitoring in the UK - Selection of Standards for Emission Monitoring. Annual Buyers Guide.* Herts.

da Rocha Carmo Junior, G.N., Belli Filho, P., de Melo Lisboa, H., Nagel Schirmer, W. and Quadros Lacey, M.E. 2010. Odor assessment tools and odor emissions in industrial processes. *Acta Scientiarum. Technology.* **32**(3).

Dalton, P. 1999. Cognitive influences on health symptoms from acute chemical exposure. *Health psychology : official journal of the Division of Health Psychology, American Psychological Association.* **18**(6), pp.579-590.

Dalton, P. 2002. Odor, irritation and perception of health risk. *International Archives of Occupational and Environmental Health.* **75**(5), pp.283-290.

Dalton, P., Wysocki, C.J., Brody, M.J. and Lawley, H.J. 1997. The influence of cognitive bias on the perceived odor, irritation and health symptoms from chemical exposure. *Int Arch Occup Environ Health.* **69**(6), pp.407 - 417.

Dalvi, P. 2013. Making sense of smells. *Duke Research Blog.* [Online]. Available from: http://sites.duke.edu/dukeresearch/2013/03/18/making-sense-of-smells/

Dammann, B., Heining, B. and Stegmann, R. 1996. Weitergehende Elimination von Gerüchen aus Kompostwerken. *Neue Techniken der Kompostierung, Dokumentation des.* **2**, pp.459-476.

Dastous, P.A., Soreanu, G., Nikiema, J. and Heitz, M. 2005. Biofiltration of three alcohols on a mature bed compost. *2005 A&WMA Annual Conference Proceedings CD-ROM, Paper #1038.* Pittsburgh, PA: Air & Water Management Association.

Day, W.C. and Berendt, R.F. 1972. Experimental tularemia in Macaca mulatta: relationship of aerosol particle size to the infectivity of airborne Pasteurella tularensis. *Infection and Immunity.* **5**(1), pp.77-82.

Deacon, L., Pankhurst, L., Liu, J., Drew, G., Hayes, E., Jackson, S., Longhurst, J., Longhurst, P., Pollard, S. and Tyrrel, S. 2009a. Endotoxin emissions from commercial composting activities. *Environmental Health.* **8**(Suppl 1), pS9.

Deacon, L.J., Pankhurst, L.J., Drew, G.H., Hayes, E.T., Jackson, S., Longhurst, P.J., Longhurst, J.W.S., Liu, J., Pollard, S.J.T. and Tyrrel, S.F. 2009b. Particle size distribution of airborne Aspergillus fumigatus spores emitted from compost using membrane filtration. *Atmospheric Environment.* **43**(35), pp.5698-5701.

Defra. 2006. *Code of Practice on Odour Nuisance from Sewage Treatment Works. PB 11833.* London: Department for Environment, Food and Rural Affairs.

Defra. 2010. *Odour Guidance for Local Authorities.* London: Department for Environment, Food and Rural Affairs.

Delgado-Rodriguez, M., Ruiz-Montoya, M., Giraldez, I., Lopez, R., Madejon, E. and Diaz, M.J. 2011. Influence of control parameters in VOCs evolution during MSW trimming residues composting. *J Agric Food Chem.* **59**(24), pp.13035-13042.

Delhoménie, M.-C., Bibeau, L., Bredin, N., Roy, S., Broussau, S., Brzezinski, R., Kugelmass, J.L. and Heitz, M. 2002. Biofiltration of air contaminated with toluene on a compost-based bed. *Advances in Environmental Research.* **6**(3), pp.239-254.

Delhoménie, M.-C. and Heitz, M. 2005. Biofiltration of Air: A Review. *Critical Reviews in Biotechnology.* **25**(1-2), pp.53-72.

Derikx, P.J.L., Simons, F.H.M., Op den Camp, H.J.M., van der Drift, C., Van Griensven, L.J.L.D. and Vogels, G.D. 1991. Evolution of Volatile Sulfur Compounds during Laboratory-Scale Incubations and Indoor Preparation of Compost Used as a Substrate in Mushroom Cultivation. *Applied and Environmental Microbiology.* **57**(2), pp.563-567.

Desloover, J., Puig, S., Virdis, B., Clauwaert, P., Boeckx, P., Verstraete, W. and Boon, N. 2011. Biocathodic Nitrous Oxide Removal in Bioelectrochemical Systems. *Environmental Science & Technology.* **45**(24), pp.10557-10566.

Devinny, J.S., Deshusses, M.A. and Webster, T.S. 1999. *Biofiltration for Air Pollution Control.* Florida: Lewis Publishers.

Douwes, J., Thorne, P., Pearce, N. and Heederik, D. 2003. Bioaerosol health effects and exposure assessment: progress and prospects. *Annals of Occupational Hygiene.* **47**(3), pp.187 - 200.

Drew, G., Deacon, L., Pankhurst, L., Pollard, S. and Tyrrell, S. 2009. Guidance on the evaluation of bioaerosol risk assessments for composting facilities. *Bristol, UK: Environmnt Agency.*

Duffy, D.P. 2017. *What's That Smell? The causes of odor and dust at MRFs and how to eliminate them.* [Online]. [Accessed 6 August]. Available from: https://foresternetwork.com/msw-management-magazine/ms-waste/waste-processing/whats-that-smell/

Dupasquier, D., Revah, S. and Auria, R. 2002. Biofiltration of Methyl tert-Butyl Ether Vapors by Cometabolism with Pentane: Modeling and Experimental Approach. *Environmental Science & Technology.* **36**(2), pp.247-253.

Edelman, L. 2008. Air filtration: Air filter media — an industry evolves. *Filtration & Separation.* **45**(9), pp.24-26.

Eduard, W., Heederik, D., Duchaine, C. and Green, B.J. 2012. Bioaerosol exposure assessment in the workplace: the past, present and recent advances. *Journal of Environmental Monitoring.* **14**(2), pp.334-339.

Elias, A., Barona, A., Arreguy, A., Rios, J., Aranguiz, I. and Peñas, J. 2002. Evaluation of a packing material for the biodegradation of H2S and product analysis. *Process Biochemistry.* **37**(8), pp.813-820.

Elite Measurement Solutions Limited. n.d. *Gastec GV-100S and GV-110S Sampling Pump Manual.* [Online]. [Accessed 14 Jun]. Available from: http://shop.elitemeasurement.com/WebRoot/Store/Shops/es136873/4D59/41BE/5205/04A1/DBD2/0A0F/110C/F012/Elite_Gastec_Pump_Manual.pdf

Environment Agency. 2010. *Composting and potential health effects from bioaerosols: our interim guidance for permit applicants. Position Statement* [Online]. [Accessed 14 May 2018]. Available from: http://www.organics-recycling.org.uk/uploads/article1822/Composting_&_bioaerosols_position_statement_-_fina_2010l%5B1%5D.pdf

Environment Agency. 2017. *Environmental monitoring of bioaerosols at regulated facilities. Technical Guidance Note (Monitoring) M9, January 2017, version 1.*

Environment Agency. 2018a. *Environmental monitoring of bioaerosols at regulated facilities. Technical Guidance Note (Monitoring) M9, July 2018, version 2.*

Environment Agency. 2018b. *Guidance - Bioaerosol monitoring at regulated facilities - use of M9: RPS 209.* [Online]. [Accessed 12 August]. Available from: https://www.gov.uk/government/publications/bioaerosol-monitoring-at-regulated-facilities-use-of-m9-rps-209/bioaerosol-monitoring-at-regulated-facilities-use-of-m9-rps-209

Environment Agency. 2002. *Assessment of Community Response to Odorous Emissions. R&D Technical Report P4-095/TR , July 2002.*

Environment Agency. 2011. *Additional guidance for H4 Odour Management: How to comply with your environment permit.* Bristol: Environment Agency

Estrada, J.M., Kraakman, N.J.R., Lebrero, R. and Muñoz, R. 2012. A sensitivity analysis of process design parameters, commodity prices and robustness on the economics of odour abatement technologies. *Biotechnology Advances.* **30**(6), pp.1354-1363.

Fabian-Wheeler, E., Hile, M. and Schmidt, D. 2012. Air Quality: Odor Measurement for Animal Agriculture. *Air Quality Education in Animal Agriculture.* pp.1 - 21.

Ferguson, R.M.W., Neath, C.E., Dumbrell, A.J., Whitby, C. and Colbeck, I. 2017. Novel insights into the size distribution of bacterial bioaerosols at

composting sites. In: Focus Meeting 10 - Bioaerosols, 8 June 2017, University of Bristol, UK. North Somerset: The Aerosol Society, pp.25 - 28. In.

Fischer, G., Albrecht, A., Jäckel, U. and Kämpfer, P. 2008. Analysis of airborne microorganisms, MVOC and odour in the surrounding of composting facilities and implications for future investigations. *International Journal of Hygiene and Environmental Health.* **211**(1), pp.132-142.

Fischer, G., Müller, T., Schwalbe, R., Ostrowski, R. and Dott, W. 2000. Exposure to airborne fungi, MVOC and mycotoxins in biowaste-handling facilities. *International Journal of Hygiene and Environmental Health.* **203**(2), pp.97-104.

Fischer, G., Schwalbe, R., Ostrowski, R. and Dott, W. 1998. Airborne fungi and their secondary metabolites in working places in a compost facility. *Mycoses.* **41**(9-10), pp.383-388.

Fletcher, L.A., Jones, N., Warren, L. and Stentiford, E.I. 2014. *Understanding biofilter performance and determining emission concentrations under operational conditions.* Edinburgh, Scotland,: Sniffer.

Frederickson, J., Boardman, C.P., Gladding, T.L., Simpson, A.E., Howell, G. and Sgouridis, F. 2013. *Evidence: Biofilter performance and operation as related to commercial composting.* Environment Agency, Bristol.

Fröhlich-Nowoisky, J., Kampf, C.J., Weber, B., Huffman, J.A., Pöhlker, C., Andreae, M.O., Lang-Yona, N., Burrows, S.M., Gunthe, S.S. and Elbert, W. 2016. Bioaerosols in the Earth system: Climate, health, and ecosystem interactions. *Atmospheric Research.* **182**, pp.346-376.

Fulazzaky, M.A., Talaiekhozani, A., Ponraj, M., Abd Majid, M., Hadibarata, T. and Goli, A. 2014. Biofiltration process as an ideal approach to remove pollutants from polluted air. *Desalination and Water Treatment.* **52**(19-21), pp.3600-3615.

Fulazzaky, M.A., Talaiekhozani, A., Ponraj, M., Abd Majid, M.Z., Hadibarata, T. and Goli, A. 2013. Biofiltration process as an ideal approach to remove pollutants from polluted air. *Desalination and Water Treatment.* **52**(19-21), pp.3600-3615.

Gabriel, D. and Deshusses, M.A. 2003. Performance of a full-scale biotrickling filter treating H2S at a gas contact time of 1.6 to 2.2 seconds. *Environmental Progress.* **22**(2), pp.111-118.

Gair, A. 2013. *Results of Olfactometry at Associated Waste Management (AWM) Facilities at Bradford and Leeds. P69-P01-R02/C66-P01-R02*. Unpublished.

Gao, L., Keener, T., Zhuang, L. and Siddiqui, K. 2001. A technical and economic comparison of biofiltration and wet chemical oxidation (scrubbing) for odor control at wastewater treatment plants. *Environmental Engineering and Policy.* **2**(4), pp.203-212.

Garrido-Cardenas, J.A., Polo-López, M.I. and Oller-Alberola, I. 2017. Advanced microbial analysis for wastewater quality monitoring: metagenomics trend. *Applied Microbiology and Biotechnology.* **101**(20), pp.7445-7458.

Gastec Corporation. 2013. *Gas Detection Syetem: Detector Tube List,* 21st ed. [Exhibition catalogue]. Japan: Gastec.

Gay, S.W. and Knowlton, K.F. 2009. Ammonia Emissions and Animal Agriculture. *Virginia Cooperative Extension.* **Publication 442-110**, pp.1-5.

Gemert, L.J.v. 2011. *Odour Thresholds: Compilations of odour threshold values in air, water and other media. (second enlarged and revised edition).* The Netherlands: Oliemans Punter & Partners BV.

Gladding, T.L. and Gwyther, C.L. 2017. A study of the potential release of bioaerosols from containers as a result of reduced frequency residual waste collections. *Science of The Total Environment.* **576**, pp.481-489.

Goldstein, N. 1999. Longer life for biofilters. *BioCycle* **40**(7), p62.

Google Maps. 2017. *AWM Gildersome Leeds, West Yorkshire LS27. 53°45'54.3"N, 1°36'35.3"W.* [Online]. [Accessed 14 October]. Available from: https://www.google.co.uk/maps/place/Leeds+LS27+7NA/@53.7647023,-1.6095594,316a,35y,270h/data=!3m1!1e3!4m5!3m4!1s0x48795fa943e441ed:0xa4f444f354126e7f!8m2!3d53.7649257!4d-1.6134715

Gottlich, E., Engresser, K. and Bardtke, D. 1994. Emission von Pilzsporen in Mullerarbeitungslangen. *Forum Stadte Hygiene.* **45**(6), pp.321 - 325.

Government of Western Australia Department of Health. 2009. *Environmental Health Guide: Hydrogen Sulphide and Public Health.* Government of Western Australia. 1 - 12.

Gradko International. 2012. *Volatile Organic Compound Measuring.* [Online]. [Accessed 13 November]. Available from: http://www.gradko.com/environmental/products/volatile-organic-compounds.shtml

Grunditz, C. and Dalhammar, G. 2001. Development of nitrification inhibition assays using pure cultures of Nitrosomonas and Nitrobacter. *Water Res.* **35**(2), pp.433-440.

Gutiérrez, M.C., Martín, M.A., Serrano, A. and Chica, A.F. 2015. Monitoring of pile composting process of OFMSW at full scale and evaluation of odour emission impact. *Journal of Environmental Management.* **151**(0), pp.531-539.

Górny, R.L., Dutkiewicz, J. and Krysinska-Traczyk, E. 1999. Size distribution of bacterial and fungal bioaerosols in indoor air. *Ann Agric Environ Med.* **6**, pp.105-113.

Hambach, R., Droste, J., François, G., Weyler, J., Van Soom, U., De Schryver, A., Vanoeteren, J. and van Sprundel, M. 2012. Work-related health symptoms among compost facility workers: a cross-sectional study. *Archives of Public Health.* **70**(1), p13.

Hartikainen, T., Ruuskanen, J. and Martikainen, P.J. 2001. Carbon Disulfide and Hydrogen Sulfide Removal with a Peat Biofilter. *Journal of the Air & Waste Management Association.* **51**(3), pp.387-392.

Hellenbrand, K.E. and Reade, A.E. 1992. Microorganisms associated with fuel wood chips and their impact on indoor air quality: a review. *International Biodeterioration & Biodegradation.* **29**(1), pp.19-43.

Hernández, M., Muñoz, R. and Daugulis, A.J. 2011. Biodegradation of VOC mixtures of different hydrophobicities in two-phase partitioning bioreactors containing tailored polymer mixtures. *Journal of Chemical Technology & Biotechnology.* **86**(1), pp.138-144.

Heroux, M., Page, T., Gelinas, C. and Guy, C. 2004. Evaluating odour impacts from a landfilling and composting site: involving citizens in the monitoring. *Water Sci Technol.* **50**(4), pp.131-137.

Herr, C.E.W., zur Nieden, A., Bödeker, R.H., Gieler, U. and Eikmann, T.F. 2003. Ranking and frequency of somatic symptoms in residents near composting sites with odor annoyance. *International Journal of Hygiene and Environmental Health.* **206**(1), pp.61-64.

Heslinga, D. 1994. Kommission reinhaltung der luft im vdi und din. *Biologische Abgasreinigung: Heidelberg, Germany.*

Hester, R.E. and Harrison, R.M. 2002. Environmental and Health Impact of Solid Waste Management activities. *The Royal Society of Chemsitry.* **Thomas Graham House, Cambridge, UK**.

Ho, K.-L., Chung, Y.-C., Lin, Y.-H. and Tseng, C.-P. 2008. Microbial populations analysis and field application of biofilter for the removal of volatile-sulfur compounds from swine wastewater treatment system. *Journal of Hazardous Materials.* **152**(2), pp.580-588.

Hoag, R.S. and Price, J.S. 1997. The effects of matrix diffusion on solute transport and retardation in undisturbed peat in laboratory columns. *Journal of Contaminant Hydrology.* **28**(3), pp.193-205.

Holland, C. 2011. *AWM's new recycling facility's power plan. Bradford Telegraph and Argus* [Online]. [Accessed 6 June 2015]. Available from: http://www.thetelegraphandargus.co.uk/business/9326078.AWM_s_new_recycling_facility___s_power_plan/

Hong, J. and Park, K. 2004. Wood chip biofilter performance of ammonia gas from composting manure. *Compost science & utilization.* **12**(1), pp.25-30.

Hove, N.C.Y., Van Langenhove, H., Van Weyenberg, S. and Demeyer, P. 2016. Comparative odour measurements according to EN 13725 using pig house odour and n-butanol reference gas. *Biosystems Engineering.* **143**, pp.119-127.

Hryhorczuk, D., Curtis, L., Scheff, P., Chung, J., Rizzo, M., Lewis, C., Keys, N. and Moomey, M. 2001. Bioaerosol emissions from a suburban yard waste composting facility. *Annals of Agricultural and Environmental Medicine.* **8**(2), pp.177-185.

HSE. 2000. *MDHS 96: Methods for the Determination of Hazardous Substances - Volatile organic compounds in air. Health and Safety Laboratory, March 2000.*

HSE. 2013. *EH40/2005 Workplace exposure limits - Containing the list of workplace exposure limits for use with the Control of Substances Hazardous to Health Regulations (as amended).* Norwich.

Hurst, C.J., Crawford, R.L., Garland, J.L. and Lipson, D.A. 2007. *Manual of environmental microbiology.* American Society for Microbiology Press.

Husman, T. 1996. Health effects of indoor-air microorganisms. *Scandinavian journal of work, environment & health.* pp.5-13.

Ibanga, I.E. 2013. *Air Quality Monitoring for Bioaerosols: A Review of the Occupational Health Risks in a Materials Recovery Facility (MRF)*. thesis, University of Leeds.

Ibanga, I.E., Fletcher, L.A., Noakes, C.J., King, M.F. and Steinberg, D. 2018. Pilot-scale biofiltration at a materials recovery facility: The impact on bioaerosol control. *Waste Management.* **80**, pp.154-167.

Jacobs, J., Sauer, N. and Gilbert, E.J. 2007. *An industry guide for the prevention and control of odours at biowaste processing facilities.* Northamptonshire: The Composting Association.

James, J. 2015. *Using Empty-Bed Contact Time (EBCT) To Design Biological Odor Control Technologies.* [Online]. [Accessed 7 November]. Available from: https://www.wateronline.com/doc/using-empty-bed-contact-time-ebct-to-design-biological-odor-control-technologies-0001

Jankowska, E., Reponen, T., Willeke, K., Grinshpun, S.A. and Choi, K.-J. 2000. Collection of fungal spores on air filters and spore reentrainment from filters into air. *Journal of Aerosol Science.* **31**(8), pp.969-978.

Janni, K.A., Nicolai, R.K., Hoff, S. and Stenglein, R.M. 2011. Air Quality Education In Animal Agriculture: Biofilters for Odor and Air Pollution Mitigation in Animal Agriculture. *Agricultural and Biosystems Engineering Extension and Outreach Publications. Book 3.* pp.1 - 9.

Jin, Y., Guo, L., Veiga, M.C. and Kennes, C. 2007. Fungal biofiltration of α-pinene: Effects of temperature, relative humidity, and transient loads. *Biotechnology and Bioengineering.* **96**(3), pp.433-443.

Jorio, H., Bibeau, L., Viel, G. and Heitz, M. 2000. Effects of gas flow rate and inlet concentration on xylene vapors biofiltration performance. *Chemical Engineering Journal.* **76**(3), pp.209-221.

Juteau, P., Rho, D., Larocque, R. and LeDuy, A. 1999. Analysis of the relative abundance of different types of bacteria capable of toluene degradation in a compost biofilter. *Applied Microbiology and Biotechnology.* **52**(6), pp.863-868.

Kafle, G.K., Chen, L., Neibling, H. and Brian He, B. 2015. Field evaluation of wood bark-based down-flow biofilters for mitigation of odor, ammonia, and hydrogen sulfide emissions from confined swine nursery barns. *Journal of Environmental Management.* **147**(0), pp.164-174.

Kell, D.B., Kaprelyants, A.S., Weichart, D.H., Harwood, C.R. and Barer, M.R. 1998. Viability and activity in readily culturable bacteria: a review and discussion of the practical issues. *Antonie van Leeuwenhoek.* **73**(2), pp.169-187.

Kim, H., Kim, Y.J., Chung, J.S. and Quan, X. 2002. Long-term operation of a biofilter for simultaneous removal of H2S and NH3. *J Air Waste Manag Assoc.* **52**(12), pp.1389-1398.

Kraakman, N.J.R., Rocha-Rios, J. and van Loosdrecht, M.C.M. 2011. Review of mass transfer aspects for biological gas treatment. *Applied Microbiology and Biotechnology.* **91**(4), pp.873-886.

Krishnayya, A.V., Agar, J.G., Wong, T.T., Nevokshonoff, B. and Warren, R. 1999. Temperature effects on biofiltration of off-gases. *Proc. on the Assessment and Remediation of Contaminated Sites in Arctic and Cold Climates.* pp.186-195.

Kumar, A., Alaimo, C.P., Horowitz, R., Mitloehner, F.M., Kleeman, M.J. and Green, P.G. 2011. Volatile organic compound emissions from green waste composting: Characterization and ozone formation. *Atmospheric Environment.* **45**(10), pp.1841-1848.

Kumar, A., Dewulf, J. and Van Langenhove, H. 2008. Membrane-based biological waste gas treatment. *Chemical Engineering Journal.* **136**(2–3), pp.82-91.

Kummer, V. and Thiel, W.R. 2008. Bioaerosols–sources and control measures. *International Journal of Hygiene and Environmental Health.* **211**(3-4), pp.299-307.

Le Goff, O., Godon, J.-J., Steyer, J.-P. and Wéry, N. 2011. New specific indicators for qPCR monitoring of airborne microorganisms emitted by composting plants. *Atmospheric Environment.* **45**(30), pp.5342-5350.

Lebrero, R., Gondim, A.C., Pérez, R., García-Encina, P.A. and Muñoz, R. 2014. Comparative assessment of a biofilter, a biotrickling filter and a hollow fiber membrane bioreactor for odor treatment in wastewater treatment plants. *Water Research.* **49**(0), pp.339-350.

Lee, C. and Lin, S. 2007. *Handbook of Environmental Engineering Calculations 2nd Ed.* McGraw-Hill Prof Med/Tech.

Leson, G. and Winer, A.M. 1991. Biofiltration: An Innovative Air Pollution Control Technology For VOC Emissions. *Journal of the Air & Waste Management Association.* **41**(8), pp.1045-1054.

Liebers, V., Bruning, T. and Raulf-Heimsoth, M. 2006. Occupational endotoxin exposure and possible health effects on humans. *American Journal of Industrial Medicine.* **49**, pp.474 - 491.

Liebers, V., Raulf-Heimsoth, M. and Bruning, T. 2008. Health effects due to endotoxin inhalation (review). *Arch Toxicol.* **82**(4), pp.203-210.

Lim, T.-T., Jin, Y., Ni, J.-Q. and Heber, A.J. 2012. Field evaluation of biofilters in reducing aerial pollutant emissions from a commercial pig finishing building. *Biosystems Engineering.* **112**(3), pp.192-201.

Lin, C.-Y., Hesu, P.-H. and Yang, D.-H. 2001. Removal of Hydrogen Sulfide Gas and Landfill Leachate Treatment Using Coal Bottom Ash. *Journal of the Air & Waste Management Association.* **51**(6), pp.939-945.

Lith, C.v., Leson, G. and Michelsen, R. 1997. Evaluating Design Options for Biofilters. *Journal of the Air & Waste Management Association.* **47**(1), pp.37-48.

Liu, Q., Li, M., Chen, R., Li, Z., Qian, G., An, T., Fu, J. and Sheng, G. 2009. Biofiltration treatment of odors from municipal solid waste treatment plants. *Waste Management.* **29**(7), pp.2051-2058.

Liu, T., Dong, H., Zhu, Z., Shang, B., Yin, F., Zhang, W. and Zhou, T. 2017. Effects of biofilter media depth and moisture content on removal of gases from a swine barn. *Journal of the Air & Waste Management Association.* **67**(12), pp.1288-1297.

Lomans, B.P., Pol, A. and Op den Camp, H.J. 2002. Microbial cycling of volatile organic sulfur compounds in anoxic environments. *Water Sci Technol.* **45**(10), pp.55-60.

López, J., Quijano, G., Souza, T.O., Estrada, J., Lebrero, R. and Muñoz, R. 2013. Biotechnologies for greenhouse gases (CH4, N2O, and CO2) abatement: state of the art and challenges. *Applied Microbiology and Biotechnology.* **97**(6), pp.2277-2303.

López, J.C., Quijano, G., Pérez, R. and Muñoz, R. 2014. Assessing the influence of CH4 concentration during culture enrichment on the biodegradation kinetics and population structure. *Journal of Environmental Management.* **146**, pp.116-123.

Lu, C., Lin, M.-R. and Chu, C. 2002. Effects of pH, moisture, and flow pattern on trickle-bed air biofilter performance for BTEX removal. *Advances in Environmental Research.* **6**(2), pp.99-106.

Macher, J.M. 1989. Positive-hole correction of multiple-jet impactors for collecting viable microorganisms. *The American Industrial Hygiene Association Journal.* **50**(11), pp.561-568.

Maine Department of Health & Human Services. 2006. *Ambient Air Guidelines for Hydrogen Sulfide CAS Registry Number: 7783-06-4.* Augusta: Maine Center for Disease Control & Prevention, Maine Department of Health & Human Services

Malhautier, L., Gracian, C., Roux, J.-C., Fanlo, J.-L. and Le Cloirec, P. 2003. Biological treatment process of air loaded with an ammonia and hydrogen sulfide mixture. *Chemosphere.* **50**(1), pp.145-153.

Malhautier, L., Soupramanien, A., Bayle, S., Rocher, J. and Fanlo, J.-L. 2015. Potentialities of coupling biological processes (biotrickler/biofilter) for the

degradation of a mixture of sulphur compounds. *Applied Microbiology and Biotechnology.* **99**(1), pp.89-96.

Malmros, P., Sigsgaard, T. and Bach, B. 1992. Occupational health problems due to garbage sorting. *Waste Management & Research.* **10**(3), pp.227-234.

Mann, D., DeBruyn, J. and Zhang, Q. 2002. Design and evaluation of an open biofilter for treatment of odour from swine barns during sub-zero ambient temperatures. *Canadian Biosystems Engineering.* **44**(6), pp.21-26.

Martens, W., Martinec, M., Zapirain, R., Stark, M., Hartung, E. and Palmgren, U. 2001. Reduction potential of microbial, odour and ammonia emissions from a pig facility by biofilters. *International Journal of Hygiene and Environmental Health.* **203**(4), pp.335-345.

Martin, R.W., Li, H., Mihelcic, J.R., Crittenden, J.C., Lueking, D.R., Hatch, C.R. and Ball, P. 2002. Optimization of biofiltration for odor control: model calibration, validation, and applications. *Water environment research.* **74**(1), pp.17-27.

McGinley, M.A. and McGinley, C.M. 2006. Precision of olfactometry and odor testing results. *Proceedings of the Water Environment Federation.* **2006**(3), pp.657-666.

McNevin, D. and Barford, J. 2000. Biofiltration as an odour abatement strategy. *Biochemical Engineering Journal.* **5**(3), pp.231-242.

Meka, V. and Van Oostrom, J. 2004. Bellows-less lung system for the human patient simulator. *Medical and Biological Engineering and Computing.* **42**(3), pp.413-418.

Melse, R.W. and Hol, J.M.G. 2012. *Measures to reduce fine dust emissions from poultry houses: Biofiltration of exhaust air of a manure (in Dutch). Report 498. Wageningen UR, the Netherlands: Livestock Research.*

Menetrez, M., Foarde, K., Esch, R., Schwartz, T., Dean, T., Hays, M., Cho, S., Betancourt, D. and Moore, S. 2009. An evaluation of indoor and outdoor biological particulate matter. *Atmospheric Environment.* **43**(34), pp.5476-5483.

Miaskiewicz-Peska, E. and Lebkowska, M. 2012. *Comparison of aerosol and bioaerosol collection on air filters.*

Millner, P., Marsh, P., Snowden, R. and Parr, J. 1977. Occurrence of Aspergillus fumigatus during composting of sewage sludge. *Applied and Environmental Microbiology.* **34**(6), pp.765-772.

Mohammad, B.T., Veiga, M.C. and Kennes, C. 2007. Mesophilic and thermophilic biotreatment of BTEX-polluted air in reactors. *Biotechnology and Bioengineering.* **97**(6), pp.1423-1438.

Morales, M., Acuna, M., Perez, F., Revah, S., Frere, G. and Auria, R. 1997. *Influence of mixing on the removal rate of toluene vapors by biofiltration.* Air and Waste Management Association, Pittsburgh, PA (United States).

Morales, M., Hernández, S., Cornabé, T., Revah, S. and Auria, R. 2003. Effect of drying on biofilter performance: modeling and experimental approach. *Environmental science & technology.* **37**(5), pp.985-992.

Morgan-Sagastume, J.M. and Noyola, A. 2006. Hydrogen sulfide removal by compost biofiltration: Effect of mixing the filter media on operational factors. *Bioresource Technology.* **97**(13), pp.1546-1553.

Mudliar, S., Giri, B., Padoley, K., Satpute, D., Dixit, R., Bhatt, P., Pandey, R., Juwarkar, A. and Vaidya, A. 2010. Bioreactors for treatment of VOCs and odours – A review. *Journal of Environmental Management.* **91**(5), pp.1039-1054.

Müller, T., Thißen, R., Braun, S., Dott, W. and Fischer, G. 2004. (M)VoC and Composting Facilities Part 1: (M)VOC Emissions from Municipal Biowaste and Plant Refuse. *Environmental Science and Pollution Research.* **11**(2), pp.91-97.

Muñoz, R., Chambaud, M., Bordel, S. and Villaverde, S. 2008. A systematic selection of the non-aqueous phase in a bacterial two liquid phase bioreactor treating α-pinene. *Applied Microbiology and Biotechnology.* **79**(1), pp.33-41.

Muñoz, R., Malhautier, L., Fanlo, J.-L. and Quijano, G. 2015. Biological technologies for the treatment of atmospheric pollutants. *International Journal of Environmental Analytical Chemistry.* **95**(10), pp.950-967.

Nahawi, H. 2015. *Indoor Chemical Air Quality Monitoring: An Assessment of Occupational Health Risks Related to Airborne Chemical Contaminates Within a Materials Recovery Facility*. thesis, University of Leeds.

Nasir, Z.A. and Tyrrel, S. 2017. Advances in bioaerosols emissions characterization from green waste composting. In: Focus Meeting 10 - Bioaerosols, 8 June 2017, University of Bristol, UK. North Somerset: The Aerosol Society, pp.12 - 15. In.

Nicolai, R. and Janni, K. 2001a. Biofilter media mixture ratio of wood chips and compost treating swine odors. *Water science and technology.* **44**(9), pp.261-267.

Nicolai, R. and Lefers, R. 2006a. Biofilters used to reduce emissions from livestock housing–A literature review. In: *Proceedings of the Workshop on agricultural air quality: State of the science*, pp.5-8.

Nicolai, R. and Lefers, R. 2006b. Biofilters used to reduce emissions from livestock housing–A literature review. In: *Workshop on Agricultural Air Quality*, pp.952-960.

Nicolai, R.E. and Janni, K.A. 1997. Development of a Low Cost Biofilter on Swine Production Facilities. Presented at the American Society of Agricultural Engineers Annual International Meeting, Minneapolis, MN, August 1997; Paper 974040.

Nicolai, R.E. and Janni, K.A. 1999. *Effect of biofilter retention time on emissions from dairy, swine, and poultry buildings. ASAE Annual Meeting.*

Nicolai, R.E. and Janni, K.A. 2001b. Biofilter media mixture ratio of wood chips and compost treating swine odors. *Water Sci Technol.* **44**(9), pp.261-267.

Nicolai, R.E. and Janni, K.A. 2001c. Determining pressure drop through compost-wood chip biofilter Media. ASAE Paper No. 014080. St. Joseph, Mich.: ASAE.

Ockeleon, H., Overcamp, T. and Grady Jr., C. 1996. Engineering Model for Fixed-Film Bioscrubbers. *Journal of Environmental Engineering.* **122**(3), pp.191-197.

Omri, I., Aouidi, F., Bouallagui, H., Godon, J.-J. and Hamdi, M. 2013. Performance study of biofilter developed to treat H2S from wastewater odour. *Saudi Journal of Biological Sciences.* **20**(2), pp.169-176.

Omri, I., Bouallagui, H., Aouidi, F., Godon, J.-J. and Hamdi, M. 2011. H 2 S gas biological removal efficiency and bacterial community diversity in biofilter treating wastewater odor. *Bioresource technology.* **102**(22), pp.10202-10209.

Orzi, V., Cadena, E., D'Imporzano, G., Artola, A., Davoli, E., Crivelli, M. and Adani, F. 2010. Potential odour emission measurement in organic fraction of municipal solid waste during anaerobic digestion: Relationship with process and biological stability parameters. *Bioresource Technology.* **101**(19), pp.7330-7337.

Ottengraf, S. and Konings, J. 1991. Emission of microorganisms from biofilters. *Bioprocess Engineering.* **7**(1-2), pp.89-96.

Ottengraf, S. and Van Den Oever, A. 1983. Kinetics of organic compound removal from waste gases with a biological filter. *Biotechnology and Bioengineering.* **25**(12), pp.3089-3102.

Oyarzún, P., Arancibia, F., Canales, C. and Aroca, G.E. 2003. Biofiltration of high concentration of hydrogen sulphide using Thiobacillus thioparus. *Process Biochemistry.* **39**(2), pp.165-170.

Pagans, E., Barrena, R., Font, X. and Sánchez, A. 2006a. Ammonia emissions from the composting of different organic wastes. Dependency on process temperature. *Chemosphere.* **62**(9), pp.1534-1542.

Pagans, E., Font, X. and Sánchez, A. 2006b. Emission of volatile organic compounds from composting of different solid wastes: Abatement by biofiltration. *Journal of Hazardous Materials.* **131**(1–3), pp.179-186.

Pagans, E.l., Font, X. and Sánchez, A. 2005. Biofiltration for ammonia removal from composting exhaust gases. *Chemical Engineering Journal.* **113**(2–3), pp.105-110.

Page, T. 2010. *Odor Metrology | Olfactometry vs Chemical Analysis.* [Online]. [Accessed 14 November]. Available from: https://www.odotech.com/en/odor-metrology-olfactometry-vs-chemical-analysis/

Pagella, C. and De Faveri, D.M. 2000. H2S gas treatment by iron bioprocess. *Chemical Engineering Science.* **55**(12), pp.2185-2194.

Pankhurst, L.J., Akeel, U., Hewson, C., Maduka, I., Pham, P., Saragossi, J., Taylor, J. and Lai, K.M. 2011a. Understanding and mitigating the challenge of bioaerosol emissions from urban community composting. *Atmospheric Environment.* **45**(1), pp.85-93.

Pankhurst, L.J., Deacon, L.J., Liu, J., Drew, G.H., Hayes, E.T., Jackson, S., Longhurst, P.J., Longhurst, J.W.S., Pollard, S.J.T. and Tyrrel, S.F. 2011b. Spatial variations in airborne microorganism and endotoxin concentrations at green waste composting facilities. *International Journal of Hygiene and Environmental Health.* **214**(5), pp.376-383.

Park, C.W., Byeon, J.H., Yoon, K.Y., Park, J.H. and Hwang, J. 2011. Simultaneous removal of odors, airborne particles, and bioaerosols in a municipal composting facility by dielectric barrier discharge. *Separation and Purification Technology.* **77**(1), pp.87-93.

Park, K.J., Choi, M.H. and Hong, J.H. 2002. Control of Composting Odor Using Biofiltration. *Compost Science & Utilization.* **10**(4), pp.356-362.

Parry, G. 2018. *Bioaerosol Risk Assessment - Alne Material Recycling Facility.* [Online]. [Accessed 12 June 2018]. Available from: https://consult.environment-agency.gov.uk/psc/yo61-1tu-allium-helmsley-ltd/supporting_documents/9f%20Supporting%20Documents%20%20Bioaerosol%20risk%20assessment.pdf

Pearson, C., Littlewood, E., Douglas, P., Robertson, S., Gant, T.W. and Hansell, A.L. 2015. Exposures and health outcomes in relation to bioaerosol emissions from composting facilities: a systematic review of occupational and community studies. *Journal of Toxicology and Environmental Health, Part B.* **18**(1), pp.43-69.

Pérez, M.C., Álvarez-Hornos, F.J., Portune, K. and Gabaldón, C. 2015. Abatement of styrene waste gas emission by biofilter and biotrickling filter: comparison

of packing materials and inoculation procedures. *Applied Microbiology and Biotechnology.* **99**(1), pp.19-32.

Persoons, R., Parat, S., Stoklov, M., Perdrix, A. and Maitre, A. 2010. Critical working tasks and determinants of exposure to bioaerosols and MVOC at composting facilities. *International Journal of Hygiene and Environmental Health.* **213**(5), pp.338-347.

Pinnette, J., Giggey, M. and Marcy, G. 1994. Performance of biofilters at two agitated bin composting facilities. In: *A AND WMA ANNUAL MEETING*: AIR & WASTE MANAGEMENT ASSOCIATION, pp.94-TP52. 02.

Potivichayanon, S., Pokethitiyook, P. and Kruatrachue, M. 2006. Hydrogen sulfide removal by a novel fixed-film bioscrubber system. *Process Biochemistry.* **41**(3), pp.708-715.

Prado, Ó.J., Gabriel, D. and Lafuente, J. 2009. Economical assessment of the design, construction and operation of open-bed biofilters for waste gas treatment. *Journal of Environmental Management.* **90**(8), pp.2515-2523.

Prasad, M., Van der Werf, P. and Brinkmann, A. 2004. Bioaerosols and composting—A literature evaluation. *Composting Association of Ireland TEO.*

Public Health England. 2015. *Ammonia: Toxicological Overview. PHE publications gateway number: 2014790.* PHE Centre for Radiation, Chemical and Environmental Hazards

Qian, J., Hospodsky, D., Yamamoto, N., Nazaroff, W.W. and Peccia, J. 2012. Size-resolved emission rates of airborne bacteria and fungi in an occupied classroom. *Indoor air.* **22**(4), pp.339-351.

Quanjer, P.H., Tammeling, G., Cotes, J., Pedersen, O., Peslin, R. and Yernault, J. 1993. *Lung volumes and forced ventilatory flows*. Eur Respiratory Soc.

Quigley, C., Easter, C., Burrowes, P. and Witherspoon, J. 2004. Biotechnology-based odour control: design criteria and performance data. *Water Science and Technology.* **50**(4), pp.319-326.

Rabe, R. and Becker, M. 2000. Emissionen von Pilzsporen aus Biofiltern von Kompostwerken. *Gefahrstoffe-Reinhaltung der Luft.* **60**, pp.168-170.

Raisi, L., Lazaridis, M. and Katsivela, E. 2010. Relationship between airborne microbial and particulate matter concentrations in the ambient air at a Mediterranean site. *Global NEST Journal.* **12**(1), pp.84-91.

Ralebitso-Senior, T.K., Senior, E., Di Felice, R. and Jarvis, K. 2012. Waste gas biofiltration: advances and limitations of current approaches in microbiology. *Environmental science & technology.* **46**(16), pp.8542-8573.

Ramirez-Lopez, E.M., Corona-Hernandez, J., Avelar-Gonzalez, F.J., Omil, F. and Thalasso, F. 2010. Biofiltration of methanol in an organic biofilter using peanut shells as medium. *Bioresource Technology.* **101**(1), pp.87-91.

Ramírez, M., Gómez, J.M., Aroca, G. and Cantero, D. 2009. Removal of hydrogen sulfide by immobilized Thiobacillus thioparus in a biotrickling filter packed with polyurethane foam. *Bioresource Technology.* **100**(21), pp.4989-4995.

Rattanapan, C. and Ounsaneha, W. 2011. *Removal of Hydrogen Sulfide Gas using Biofiltration - a Review.*

Redmore, J. 2012. *Odour and Air Quality Assessment: Surrey Hill Energy Anaerobic Digestion Plant. REC Report - 33276r1.* Manchester: Resource & Environmental Consultants Ltd.

Reece, F.N., Lott, B.D. and Deaton, J.W. 1980. Ammonia in the Atmosphere during Brooding Affects Performance of Broiler Chickens. *Poultry Science.* **59**(3), pp.486-488.

Regulatory Policy Committee. 2017. *Technical Guidance note M9: Environmental monitoring of bioaerosols at regulated facilities. Environment Agency - RPC rating: validated February 2017.* [Online]. [Accessed 21 August]. Available from: https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/707777/DEFRA_EA_Technical_Guidance_note_M9_Environmental_monitoring_of_bioaerosols.pdf

Reij, M.W., Keurentjes, J.T.F. and Hartmans, S. 1998. Membrane bioreactors for waste gas treatment. *Journal of Biotechnology.* **59**(3), pp.155-167.

Rocha-Rios, J., Bordel, S., Hernández, S. and Revah, S. 2009. Methane degradation in two-phase partition bioreactors. *Chemical Engineering Journal.* **152**(1), pp.289-292.

Roe, S.M., Spivey, M.D., Lindquist, H.C., Thesing, K.B., Strait, R.P. and Pechan E. H. & Associates, I. 2004. *Estimating Ammonia Emissions from Nonagricultural Sources - Draft Final Report. Prepared for: Emission Inventory Improvement Program, April 2004.*

Rosenfeld, P.E. and Suffet, I.H. 2004. Understanding odorants associated with compost, biomass facilities, and the land application of biosolids. *Water Sci Technol.* **49**(9), pp.193-199.

Roshani, B., Torkian, A., Dehghanzadeh, R. and Shahmansori, M.R. 2005. Performance evaluation of biofiltration in the removal of hydrogen sulfide from gas flue. In: *Biotechniques for air pollution control: proceedings of the international congress Biotechniques for Air Pollution Control: A Coruña, Spain, October 5-7, 2005*: Servicio de Publicaciones, pp.119-126.

Ruokojärvi, A., Ruuskanen, J., Martikainen, P.J. and Olkkonen, M. 2001. Oxidation of Gas Mixtures Containing Dimethyl Sulfide, Hydrogen Sulfide, and Methanethiol Using a Two-Stage Biotrickling Filter. *Journal of the Air & Waste Management Association.* **51**(1), pp.11-16.

Rylander, R. 1999. Indoor air-related effects and airborne (1 --> 3)-beta-D-glucan. *Environ Health Perspect.* **107 Suppl 3**, pp.501-503.

Sakuma, T., Hattori, T. and Deshusses, M.A. 2009. The effects of a lower irrigation system on pollutant removal and on the microflora of a biofilter. *Environmental Technology.* **30**(6), pp.621-627.

Sanchez-Monedero, M.A., Stentiford, E.I. and Mondini, C. 2003. Biofiltration at Composting Facilities: Effectiveness for Bioaerosol Control. *Environmental Science & Technology.* **37**(18), pp.4299-4303.

Sanchez-Monedero, M.A., Stentiford, E.I. and Urpilainen, S.T. 2005. Bioaerosol generation at large-scale green waste composting plants. *J Air Waste Manag Assoc.* **55**(5), pp.612-618.

Schappler-Scheele, H., Schurrman, W., Hartung, J., Missel, J., Benning, C., Schroederm, H. and Lieber, J. 1998. Untersuchung der gesundheitlichen Gefährdung von Arbeitnehmen der Abfallwirtschaft in Kompostierungsanlangen. *Schriftenreihe Bundesanstalt für Arbeitsschutz und Arbeitsmedizin.* **21**, p11.

Scharf, P., Zucker, B.-A. and Müller, W. 2004. Effect of a bioscrubber on emission of bioaerosols from a duck fattening unit. *International Society for Animal Hygiène. Saint-Malo.*

Schlegelmilch, M., Herold, T., Streese, J., Hensel, A. and Stegmann, R. 2005a. The potential to reduce emissions of airborne microorganisms by means of biological waste gas treatment systems. *Waste management.* **25**(9), pp.955-964.

Schlegelmilch, M., Streese, J., Biedermann, W., Herold, T. and Stegmann, R. 2005b. Odour control at biowaste composting facilities. *Waste Management.* **25**(9), pp.917-927.

Schlosser, O., Huyard, A., Cartnick, K., Yanez, A., Catalan, V. and Quang, Z.D. 2009. Bioaerosol in composting facilities: occupational health risk assessment. *Water Environ Res.* **81**(9), pp.866-877.

Schmidt, D., Jacobson, L. and Nicolai, R. 2004. *Biofilter design information* [Online]. [Accessed 18 Jan]. Available from: http://www.extension.umn.edu/agriculture/manure-management-and-air-quality/air-quality/biofilter-design-information/

Schnelle, K.B. and Brown, C.A. 2002. *Air pollution control technology handbook*. thesis, CRC Press LLC.

Searl, A. 2008. Exposure-Response Relationship for Bioaerosols Emissions from Waste Treatment Processes. *IOM, Edinburgh.*

Secomak. 2017. *High Velocity Fans: Centrifugal fans and blowers.* [Online]. [Accessed November 7]. Available from: https://www.secomak.com/product/high-velocity-fans

SEPA. n.d. *BAT for Composting. SEPA Technical Guidance.* [Online]. [Accessed 10 Sep]. Available from: http://www.sepa.org.uk/media/112766/sepa-technical-guidance-bat-for-composting.pdf

SEPA & Natural Scotland Scottish Government. 2010. *Odour guidance 2010. Version 1.* January 2010.

Shahmansouri, M., Taghipour, H., Bina, B. and Movahdian, H. 2005. Biological removal of ammonia from contaminated air streams using biofiltration system. *Iranian Journal of Environmental Health Science & Engineering.* **2**(2), pp.17-25.

Shareefdeen, Z. and Singh, A. 2008. *Biotechnology for Odor and Air Pollution Control.* Germany: Springer-Verlag Berlin Heidelberg.

Shen, Y., Chen, T.-B., Gao, D., Zheng, G., Liu, H. and Yang, Q. 2012. Online monitoring of volatile organic compound production and emission during sewage sludge composting. *Bioresource Technology.* **123**(0), pp.463-470.

Sheridan, B., Curran, T., Dodd, V. and Colligan, J. 2002a. *Biofiltration of Odour and Ammonia from a Pig Unit – A Pilot-Scale Study.*

Sheridan, B., Curran, T., Dodd, V. and Colligan, J. 2002b. SE—Structures and Environment: Biofiltration of Odour and Ammonia from a Pig Unit—Biofiltration of Odour and Ammonia from a Pig Unit—a pilot-scale Study. *Biosystems Engineering.* **82**(4), pp.441-453.

Sheridan, B., Curran, T.P. and Dodd, V. 2002c. Assessment of the influence of media particle size on the biofiltration of odorous exhaust ventilation air from a piggery facility. *Bioresource technology.* **84**(2), pp.129-143.

Shusterman, D. 2002. Review of the upper airway, including olfaction, as mediator of symptoms. *Environmental health perspectives.* **110 Suppl 4**, pp.649-653.

Sigsgaard, T., Abel, A., Donbaek, L. and Malmros, P. 1994. Lung function changes among recycling workers exposed to organic dust. *Am J Ind Med.* **25**(1), pp.69-72.

Singh, R.S., Agnihotri, S.S. and Upadhyay, S.N. 2006. Removal of toluene vapour using agro-waste as biofilter media. *Bioresource Technology.* **97**(18), pp.2296-2301.

Sironi, S., Capelli, L., Céntola, P., Del Rosso, R. and Il Grande, M. 2006. Odour emission factors for the prediction of odour emissions from plants for the

mechanical and biological treatment of MSW. *Atmospheric Environment.* **40**(39), pp.7632-7643.

Smeets, M.A.M., Mauté, C. and Dalton, Pamela H. 2002. Acute Sensory Irritation from Exposure to Isopropanol (2-Propanol) at TLV in Workers and Controls: Objective versus Subjective Effects. *Annals of Occupational Hygiene.* **46**(4), pp.359-373.

Smet, E., Lens, P. and Langenhove, H.V. 1998. Treatment of Waste Gases Contaminated with Odorous Sulfur Compounds. *Critical Reviews in Environmental Science and Technology.* **28**(1), pp.89-117.

Smet, E., Van Langenhove, H. and De Bo, I. 1999. The emission of volatile compounds during the aerobic and the combined anaerobic/aerobic composting of biowaste. *Atmospheric Environment.* **33**(8), pp.1295-1303.

Sniffer. 2013. *Odour monitoring and control on landfill sites. ER31 Final Report.* January 2013.

Stagg, S., Bowry, A., Kelsey, A. and Crook, B. 2010. *Bioaerosol emissions from waste composting and the potential for workers' exposure. RR786.* Derbyshire: Health and Safety Executive.

Stagg, S., Sandys, V., Crook, B., Wood, J. and McAlinden, J. 2013. *Occupational Hygiene implications of processing waste at Materials Recycling Facilities (MRFs).* [Online]. [Accessed 30 August]. Available from: http://www.hse.gov.uk/research/rrpdf/rr977.pdf

Steinheider, B., Both, R. and Winneke, G. 1998. Field studies on environmental odors inducing annoyance as well as gastric and general health-related symptoms. *Journal of Psychophysiology.* **12**(Suppl 1), pp.64-79.

Stevenson, R.J. and Wilson, D.A. 2007. Odour perception: an object-recognition approach. *Perception.* **36**(12), pp.1821 - 1833.

Sun, Y., Clanton, C.J., Janni, K.A. and Malzer, G.L. 2000. Sulfur and nitrogen balances in biofilters for odorous gas emission control. *Transactions of the American Society of Agricultural Engineers.* **43**(6), pp.1861-1875.

Surrey County Council. 2017. *Types of Waste Management Facilities: An Explanation Note.* [Online]. [Accessed 20 June 2018]. Available from: https://www.surreycc.gov.uk/__data/assets/pdf_file/0010/147367/2017-10-27-Types-of-waste-mgt-facilties-explanation-note.pdf

Swan, J., Kelsey, A., Crook, B. and Gilbert, E. 2003. *Occupational and environmental exposure to bioaerosols from composts and potential health effects: a critical review of published data.* HSE Books.

Swanson, W. and Loehr, R. 1997. Biofiltration: Fundamentals, Design and Operations Principles, and Applications. *Journal of Environmental Engineering.* **123**(6), pp.538-546.

Sykes, P., Morris, R.H.K., Allen, J.A., Wildsmith, J.D. and Jones, K.P. 2011. Workers' exposure to dust, endotoxin and β-(1–3) glucan at four large-scale composting facilities. *Waste Management.* **31**(3), pp.423-430.

Taha, M., Pollard, S.J., Sarkar, U. and Longhurst, P.J. 2004. The influence of process variables on bioaerosol emission flux and exposure-estimating fugitive bioaerosol releases from static compost windrows. *Proceedings of the 1st UK Conference and Exhibition on Biodegradable and Residual Waste Management.*

Taha, M.P.M., Drew, G.H., Longhurst, P.J., Smith, R. and Pollard, S.J.T. 2006. Bioaerosol releases from compost facilities: Evaluating passive and active source terms at a green waste facility for improved risk assessments. *Atmospheric Environment.* **40**(6), pp.1159-1169.

The Composting Association. 2004. *INFORMATION SHEET 22: Dealing with Odours and Biofilters* [Online]. [Accessed 24 Jan]. Available from: http://www.organics-recycling.org.uk/dmdocuments/22_odours_and_biofilters.pdf

Thomas, R., Davies, C., Nunez, A., Hibbs, S., Eastaugh, L., Harding, S., Jordan, J., Barnes, K., Oyston, P. and Eley, S. 2012. Particle-size dependent effects in the Balb/c murine model of inhalational melioidosis. *Frontiers in cellular and infection microbiology.* **2**, p101.

Thomas, R., Davies, C., Nunez, A., Hibbs, S., Flick-Smith, H., Eastaugh, L., Smither, S., Gates, A., Oyston, P. and Atkins, T. 2010. Influence of particle size on the pathology and efficacy of vaccination in a murine model of inhalational anthrax. *Journal of medical microbiology.* **59**(12), pp.1415-1427.

Thomas, R.J. 2013. Particle size and pathogenicity in the respiratory tract. *Virulence.* **4**(8), pp.847-858.

Thompson, D., Sterne, L., Bell, J., Parker, W. and Lye, A. 1996. Pilot scale investigation of sustainable BTEX removal with a compost biofilter. In: *Proc. of 89th Annual Meeting and Exhibition. A&WMA, Nashville*.

Tisch Environmental Inc. 2015. *Cascade Impactor Series 10-8XX Viable (Microbial) Particle Sizing Instruments: Operations Manual* [Online]. [Accessed 10 October 2015]. Available from: https://tisch-env.com/wp-content/uploads/2015/06/TE-10-800-Viable-Cascade-Impactor.pdf

Trunov, M., Trakumas, S., Willeke, K., Grinshpun, S.A. and Reponen, T. 2001. Collection of bioaerosol particles by impaction: effect of fungal spore agglomeration and bounce. *Aerosol Science & Technology.* **35**(1), pp.617-624.

Tsai, C.-J., Chen, M.-L., Ye, A.-D., Chou, M.-S., Shen, S.-H. and Mao, I.-F. 2008. The relationship of odor concentration and the critical components emitted from food waste composting plants. *Atmospheric Environment.* **42**(35), pp.8246-8251.

Tsang, Y.F., Wang, L. and Chua, H. 2015. Simultaneous hydrogen sulphide and ammonia removal in a biotrickling filter: Crossed inhibitory effects among selected pollutants and microbial community change. *Chemical Engineering Journal.* **281**, pp.389-396.

TSI Incorporated. 2013. *Health-Based Particle-Size-Selective Sampling.* [Online]. [Accessed 21 August 2017]. Available from: http://www.tsi.com/uploadedFiles/_Site_Root/Products/Literature/Application_Notes/ITI-050.pdf

Tymczyna, L., Chmielowiec-Korzeniowska, A. and Drabik, A. 2007. The effectiveness of various biofiltration substrates in removing bacteria, endotoxins, and dust from ventilation system exhaust from a chicken hatchery. *Poultry science.* **86**(10), pp.2095-2100.

Tymczyna, L., Chmielowiec-Korzeniowska, A., Raczynska, J. and Drabik, A. 2011. Removal of microbial contaminants from pig house air using biofilter organic media. *Ann Anim Sci.* **11**, pp.453-464.

USEPA. 2003. *Using Bioreactors to Control Air Pollution. EPA-456/R-03-003.* [Online]. [Accessed 24 April]. Available from: https://www3.epa.gov/ttncatc1/dir1/fbiorect.pdf

Valter Francescato, E.A., Bergomi, L.Z., Metschina, C., Schnedl, C., Krajnc, N., Kajetan Koscik, P.G., Nocentini, G. and Stranieri, S. 2008. *Wood Fuels*

Handbook: Production, Quality Requirements, Trading. Italy: AIEL - Italian Agriforestry Energy Association.

Van der Heyden, C., Demeyer, P. and Volcke, E.I.P. 2015. Mitigating emissions from pig and poultry housing facilities through air scrubbers and biofilters: State-of-the-art and perspectives. *Biosystems Engineering.* **134**(0), pp.74-93.

van Kampen, V., Deckert, A., Hoffmeyer, F., Taeger, D., Brinkmann, E., Brüning, T., Raulf-Heimsoth, M. and Bünger, J. 2012. Symptoms, spirometry, and serum antibody concentrations among compost workers exposed to organic dust. *Journal of Toxicology and Environmental Health, Part A.* **75**(8-10), pp.492-500.

Varley, N. 2013. *Technical Assessment Odour Abatement System for AWM Gelderd Road*. Unpublished.

VDI3477. 2004. *Biological waste gas purification Biofilters. Verein Deutscher Ingenieure e.V., Düsseldor.*

Velusami, B., Curran, T.P. and Grogan, H.M. 2013. Hydrogen sulfide gas emissions during disturbance and removal of stored spent mushroom compost. *J Agric Saf Health.* **19**(4), pp.261-275.

Wake, D., Redmayne, A., Thorpe, A., Gould, J., Brown, R. and Crook, B. 1995. Sizing and filtration of microbiological aerosols. *Journal of Aerosol Science.* (26), pp.S529-S530.

Wang, Y., Lan, H., Li, L., Yang, K., Qu, J. and Liu, J. 2018. Chemicals and microbes in bioaerosols from reaction tanks of six wastewater treatment plants: survival factors, generation sources, and mechanisms. *Scientific Reports.* **8**(1), p9362.

Wani, A.H., Branion, R.M.R. and Lau, A.K. 1997. Biofiltration: A promising and cost-effective control technology for Odors, VOCs and air toxics. *Journal of Environmental Science and Health . Part A: Environmental Science and Engineering and Toxicology.* **32**(7), pp.2027-2055.

Wani, A.H., Lau, A.K. and Branion, R.M. 1999. Biofiltration control of pulping odors–hydrogen sulfide: performance, macrokinetics and coexistence effects of organo-sulfur species. *Journal of Chemical Technology and Biotechnology.* **74**(1), pp.9-16.

Waweru, M., Herrygers, V., van Langenhove, H. and Verstraete, W. 2005. Process Engineering of Biological Waste Gas Purification. In: Jördening, H.-J. and Winter, J. eds. *Environmental Biotechnology: Concepts and Applications.* Weinheim: WILEY-VCH Verlang GmbH & Co. KGaA.

Wéry, N. 2014. Bioaerosols from composting facilities—a review. *Frontiers in cellular and infection microbiology.* **4**.

WHO. 2003. *Hydrogen sulfide : Human health aspects.* Geneva: World Health Organization.

Willeke, K., Qian, Y., Donnelly, J., Grinshpun, S. and Ulevicius, V. 1996. Penetration of airborne microorganisms through a surgical mask and a dust/mist respirator. *American Industrial Hygiene Association.* **57**(4), pp.348-355.

Williams, M., Lamarre, B., Butterfield, D., Tyrrel, S., Longhurst, P., Drew, G., Al-Ashaab, R., Nelson, A., Gladding, T., Simpson, A., Coughlin, D. and Hansell, A. 2013. *Bioaerosols and odour emissions from composting facilities. Defra Project WR 1121* Department for Environment Food & Rural Affairs.

Williams, T.O. and Miller, F.C. 1992. Odor control using biofilters: Part 1. This overview of basic design and operating criteria explains how biofilters can

be used more efficiently to control odors emanating from composting facilities. *BioCycle.* **33**(10), pp.72 - 77.

Xue, S., Chen, W., Deng, M., Luo, H., Huang, W., Han, Y. and Li, L. 2018. Effects of moisture content on the performance of a two-stage thermophilic biofilter and choice of irrigation rate. *Process Safety and Environmental Protection.* **113**, pp.164-173.

Yang, L., Kent, A.D., Wang, X., Funk, T.L., Gates, R.S. and Zhang, Y. 2014. Moisture effects on gas-phase biofilter ammonia removal efficiency, nitrous oxide generation, and microbial communities. *Journal of Hazardous Materials.* **271**, pp.292-301.

Yang, Y. and Allen, E.R. 1994. Biofiltration control of hydrogen sulfide 1. Design and operational parameters. *Air & waste.* **44**(7), pp.863-868.

Yoon, I.-K. and Park, C.-H. 2002. Effects of gas flow rate, inlet concentration and temperature on biofiltration of volatile organic compounds in a peat-packed biofilter. *Journal of Bioscience and Bioengineering.* **93**(2), pp.165-169.

Yuwono, A.S., Hamacher, T., Nieß, J., Boeker, P. and Lammers, P.S. 2003. Implementation of a quartz microbalance (QMB) sensor array-based instrument and olfactometer for monitoring the performance of an odour biofilter. *2nd IWA International Workshop & Conference on Odour & VOC's, Singapore.* pp.1 - 8.

Zamir, S.M., Babatabar, S. and Shojaosadati, S.A. 2015. Styrene vapor biodegradation in single- and two-liquid phase biotrickling filters using Ralstonia eutropha. *Chemical Engineering Journal.* **268**(0), pp.21-27.

Zhao, L., Manuzon, R.B., Darr, M.J., Keener, H.M., Heber, A.J. and Ni, J.-Q. 2008. Ammonia Emissions from a Commercial Poultry Manure Composting Facility. In: *Livestock Environment VIII, Proceedings of the International Symposium. St. Joseph Michigan: ASABE. ,31 August 2008 . ASAE Pub #701P0408.*, pp.41 - 48.

Zhu, X., Suidan, M.T., Pruden, A., Yang, C., Alonso, C., Kim, B.J. and Kim, B.R. 2004. Effect of Substrate Henry's Constant on Biofilter Performance. *Journal of the Air & Waste Management Association.* **54**(4), pp.409-418.

Appendix A
DATA SET FOR CHAPTER 4

Table A. 1: Mean inlet and outlet bioaerosol concentration (in cfu m^{-3}) for visits 1 – 6 (June to August 2016 [summer] – indoors; n = 2).

		Aspergillus fumigatus				Total fungi				Total mesophilic bacteria				Gram negative bacteria			
		Inlet	SD	Outlet	SD	Inlet	SD	Outlet	SD	Inlet	SD	Outlet	SD	Inlet	SD	Outlet	SD
Visit 1	BF1	9692.6	111.6	853.4	310.6	11888.2	538	1096	349	11342.8	1815.6	4014.2	2780.1	6894	1484.2	912.3	225.6
	BF2	9692.6	111.6	1488.2	95.7	11888.2	538	1967	13.3	11342.8	1815.6	3692	1853.2	6894	1484.2	1113.7	79.1
	BF3	9692.6	111.6	1074.2	351.4	11888.2	538	1732	617.2	11342.8	1815.6	2897.5	299.8	6894	1484.2	1004.2	125.8
	BF4	9692.6	111.6	2170.2	147.4	11888.2	538	2544.2	263.2	11342.8	1815.6	4461.1	114.1	6894	1484.2	2366.9	167.4
Visit 2	BF1	12312.8	1478.3	3510.0	682.9	14977.7	2073.9	4084.8	752.9	53434.1	5782.6	11238.5	1624.9	2432.9	265.7	2393.4	388.1
	BF2	12312.8	1478.3	2694.9	164.9	14977.7	2073.9	3177.3	247.3	53434.1	5782.6	4121.9	1928.1	2432.9	265.7	2554.2	938.6
	BF3	12312.8	1478.3	1629.0	1076.1	14977.7	2073.9	2053.6	1235.2	53434.1	5782.6	6217.9	896.2	2432.9	265.7	2039.5	878.7
	BF4	12312.8	1478.3	2811.0	1801.5	14977.7	2073.9	3667.3	2397.9	53434.1	5782.6	11026.0	1522.5	2432.9	265.7	3142.0	49.1
Visit 3	BF1	10427.6	1996.4	729.7	197.4	13412.6	938.3	847.2	263.6	5901.1	107.4	1041.6	281.1	24666.1	619.6	5275.7	1621.6
	BF2	10427.6	1996.4	1579.5	264.9	13412.6	938.3	1956.7	536.0	5901.1	107.4	4248.2	2684.7	24666.1	619.6	3720.9	1769.0
	BF3	10427.6	1996.4	564.5	41.2	13412.6	938.3	645.8	3.7	5901.1	107.4	1400.2	278.6	24666.1	619.6	1761.5	194.9
	BF4	10427.6	1996.4	13941.7	10039.4	13412.6	938.3	15580.4	11442.4	5901.1	107.4	4721.8	990.7	24666.1	619.6	3595.4	921.9
Visit 4	BF1	7376.4	2800.9	565.4	50.0	9303.9	3053.3	977.1	77.4	22913.5	1266.8	9031.8	4412.5	6420.5	184.8	2311.0	1089.4
	BF2	7376.4	2800.9	353.4	99.9	9303.9	3053.3	761.5	277.3	22913.5	1266.8	798.6	25.0	6420.5	184.8	3477.0	624.7
	BF3	7376.4	2800.9	1372.8	487.2	9303.9	3053.3	1948.8	642.1	22913.5	1266.8	2940.0	259.9	6420.5	184.8	4620.2	432.3
	BF4	7376.4	2800.9	4508.9	65.0	9303.9	3053.3	5127.2	124.9	22913.5	1266.8	5759.7	489.7	6420.5	184.8	3379.9	187.5
Visit 5	BF1	8125.4	542.2	851.6	104.9	10493.0	1321.8	1572.5	99.9	13164.3	9372.3	1671.4	1009.5	19136.1	13714.8	4812.8	2003.9
	BF2	8125.4	542.2	867.5	72.4	10493.0	1321.8	1404.6	177.3	13164.3	9372.3	2021.2	1349.3	19136.1	13714.8	5130.8	2478.6
	BF3	8125.4	542.2	1296.8	329.8	10493.0	1321.8	1750.9	357.3	13164.3	9372.3	1245.6	152.5	19136.1	13714.8	8074.2	2383.7
	BF4	8125.4	542.2	996.5	259.9	10493.0	1321.8	1772.1	232.4	13164.3	9372.3	2618.4	1479.2	19136.1	13714.8	6349.8	554.7
Visit 6	BF1	3800.4	1546.7	909.9	127.4	4763.3	2183.8	1233.2	224.9	12733.3	1176.8	1584.8	277.3	5549.5	1081.9	4478.8	1836.5
	BF2	3800.4	1546.7	1084.8	124.9	4763.3	2183.8	1174.9	202.4	12733.3	1176.8	2028.3	489.7	5549.5	1081.9	2282.7	494.7
	BF3	3800.4	1546.7	765.0	227.4	4763.3	2183.8	1121.9	482.2	12733.3	1176.8	4819.8	924.5	5549.5	1081.9	7537.1	4132.8
	BF4	3800.4	1546.7	1558.3	75.0	4763.3	2183.8	1756.2	100.0	12733.3	1176.8	4531.8	2166.3	5549.5	1081.9	3222.6	689.6

SD – Standard Deviation

Table A. 2: Mean inlet and outlet bioaerosol concentration (in cfu m^{-3}) for visits 7 – 12 (September 2016 to February 2017 - outdoors; n = 2).

		Aspergillus fumigatus				Total fungi				Total mesophilic bacteria				Gram negative bacteria			
		Inlet	SD	Outlet	SD	Inlet	SD	Outlet	SD	Inlet	SD	Outlet	SD	Inlet	SD	Outlet	SD
Visit 7	BF1	1070.7	404.7	141.4	50.0	1480.6	584.7	650.2	619.6	4996.5	3393.1	1477.0	639.6	3641.4	1161.8	1929.3	864.5
	BF2	1070.7	404.7	123.7	174.9	1480.6	584.7	282.7	349.8	4996.5	3393.1	1731.5	384.7	3641.4	1161.8	1291.6	312.3
	BF3	1070.7	404.7	300.4	174.9	1480.6	584.7	494.7	100.0	4996.5	3393.1	1001.8	862.0	3641.4	1161.8	1235.0	427.2
	BF4	1070.7	404.7	194.3	75.0	1480.6	584.7	229.7	75.0	4996.5	3393.1	2962.9	1106.9	3641.4	1161.8	2258.0	180.0
Visit 8	BF1	1358.7	407.2	88.3	75.0	1816.3	539.7	88.3	75.0	9323.3	1017.0	1733.2	227.4	2247.4	799.5	1954.1	539.7
	BF2	1358.7	407.2	123.7	75.0	1816.3	539.7	123.7	75.0	9323.3	1017.0	1067.2	404.8	2247.4	799.5	853.4	207.4
	BF3	1358.7	407.2	88.3	75.0	1816.3	539.7	106.0	49.9	9323.3	1017.0	1390.5	307.4	2247.4	799.5	2358.7	532.2
	BF4	1358.7	407.2	70.7	50.0	1816.3	539.7	123.7	124.9	9323.3	1017.0	2242.1	27.5	2247.4	799.5	1634.3	1106.9
Visit 9	BF1	961.1	149.9	88.4	25.0	1194.4	279.8	88.4	25.0	2623.7	467.3	927.6	47.4	500.0	507.3	568.9	49.9
	BF2	961.1	149.9	53.0	25.0	1194.4	279.8	70.7	50.0	2623.7	467.3	2129.0	1396.7	500.0	507.3	1215.5	714.6
	BF3	961.1	149.9	17.7	25.0	1194.4	279.8	35.4	50.0	2623.7	467.3	1102.5	50.0	500.0	507.3	1837.5	629.7
	BF4	961.1	149.9	70.7	50.0	1194.4	279.8	70.7	50.0	2623.7	467.3	2136.1	932.0	500.0	507.3	944.2	485.8
Visit 10	BF1	675.0	304.8	583.1	124.9	908.1	284.8	583.1	124.9	653.7	224.9	1070.7	459.8	1703.2	899.5	1180.2	55.0
	BF2	675.0	304.8	800.4	232.4	908.1	284.8	818.1	207.4	653.7	224.9	1535.4	102.5	1703.2	899.5	1484.1	169.8
	BF3	675.0	304.8	636.1	50.0	908.1	284.8	671.4	50.0	653.7	224.9	1597.2	179.9	1703.2	899.5	945.3	687.1
	BF4	675.0	304.8	690.9	177.4	908.1	284.8	779.2	152.4	653.7	224.9	1600.7	1209.3	1703.2	899.5	1639.6	194.9
Visit 11	BF1	106.0	149.9	17.7	25.0	159.0	174.9	17.7	25.0	1574.2	42.4	925.8	459.8	1033.6	457.3	2917.0	2805.9
	BF2	106.0	149.9	17.7	25.0	159.0	174.9	70.7	50.0	1574.2	42.4	1823.4	1019.4	1033.6	457.3	2730.6	236.1
	BF3	106.0	149.9	123.7	124.9	159.0	174.9	176.7	149.9	1574.2	42.4	1952.3	22.5	1033.6	457.3	2061.8	452.3
	BF4	106.0	149.9	17.7	25.0	159.0	174.9	53.0	25.0	1574.2	42.4	1726.2	1336.8	1033.6	457.3	1590.1	534.7
Visit 12	BF1	53.0	25.0	53.0	75.0	176.7	99.9	53.0	75.0	1591.9	667.2	906.4	72.5	1155.5	279.9	925.8	199.8
	BF2	53.0	25.0	35.4	50.0	176.7	99.9	35.4	50.0	1591.9	667.2	1171.4	702.1	1155.5	279.9	855.1	759.6
	BF3	53.0	25.0	141.4	50.0	176.7	99.9	141.4	50.0	1591.9	667.2	1051.3	427.3	1155.5	279.9	461.1	202.4
	BF4	53.0	25.0	159.0	25.0	176.7	99.9	159.0	25.0	1591.9	667.2	802.1	684.6	1155.5	279.9	856.9	562.1

SD – Standard Deviation

Table A. 3: Mean inlet and outlet bioaerosol concentration (in cfu m^{-3}) for visits 13 – 16 (February to March 2017 [winter] – indoors; n = 2).

		Aspergillus fumigatus				Total fungi				Total mesophilic bacteria				Gram negative bacteria			
		Inlet	**SD**	**Outlet**	**SD**	**Inlet**	**SD**	**Outlet**	**SD**	**Inlet**	**SD**	**Outlet**	**SD**	**Inlet**	**SD**	**Outlet**	**SD**
Visit 13	BF1	11441.7	6346.4	123.7	25.0	14878.1	8647.6	176.7	50.0	13630.8	4170.2	2841.0	1694.1	5915.2	2483.6	2060.1	489.7
	BF2	11441.7	6346.4	964.7	914.5	14878.1	8647.6	1805.7	2003.9	13630.8	4170.2	1268.6	25.0	5915.2	2483.6	2507.1	1266.8
	BF3	11441.7	6346.4	123.7	25.0	14878.1	8647.6	194.4	124.9	13630.8	4170.2	2669.7	1906.4	5915.2	2483.6	2197.9	664.6
	BF4	11441.7	6346.4	300.4	224.9	14878.1	8647.6	424.0	249.9	13630.8	4170.2	2680.2	542.2	5915.2	2483.6	1574.2	47.5
Visit 14	BF1	21404.6	677.1	8063.6	634.7	40706.8	5412.0	11379.9	517.2	60282.7	5327.1	25712.1	16033.6	99940.0	11233.7	26971.7	13547.5
	BF2	21404.6	677.1	23766.8	6611.3	40706.8	5412.0	36012.4	12825.4	60282.7	5327.1	46448.8	13122.7	99940.0	11233.7	41523.0	30423.1
	BF3	21404.6	677.1	9973.5	3485.6	40706.8	5412.0	13259.7	3650.5	60282.7	5327.1	29556.6	6723.8	99940.0	11233.7	36708.5	4869.8
	BF4	21404.6	677.1	1773.9	129.9	40706.8	5412.0	3524.8	517.2	60282.7	5327.1	19655.5	8927.5	99940.0	11233.7	9570.7	1316.8
Visit 15	BF1	10178.5	2811.0	3692.6	329.8	12083.1	2930.9	5289.8	1009.4	47473.5	1804.0	19122.0	8292.9	25722.6	20995.8	13814.5	3265.7
	BF2	10178.5	2811.0	5975.3	50.0	12083.1	2930.9	7952.3	257.4	47473.5	1804.0	18780.9	9634.6	25722.6	20995.8	31812.7	13632.5
	BF3	10178.5	2811.0	3768.6	612.1	12083.1	2930.9	5008.9	442.3	47473.5	1804.0	18616.6	12275.7	25722.6	20995.8	34302.2	7058.6
	BF4	10178.5	2811.0	3243.9	124.9	12083.1	2930.9	4466.5	334.8	47473.5	1804.0	32141.4	15516.4	25722.6	20995.8	4717.3	3987.8
Visit 16	BF1	53044.2	12945.3	37973.5	2191.3	62655.5	16748.2	46558.3	754.6	95627.2	35377.8	58199.7	18087.4	27913.5	6533.9	18390.5	9862.0
	BF2	53044.2	12945.3	35683.8	1736.6	62655.5	16748.2	43210.3	1651.6	95627.2	35377.8	50742.1	16590.8	27913.5	6533.9	11917.0	1506.6
	BF3	53044.2	12945.3	30533.6	4612.5	62655.5	16748.2	34137.8	4327.6	95627.2	35377.8	23980.6	937.0	27913.5	6533.9	10063.6	1714.0
	BF4	53044.2	12945.3	33738.5	6876.2	62655.5	16748.2	37892.3	5729.3	95627.2	35377.8	70263.3	36247.4	27913.5	6533.9	13111.3	4664.9

SD – Standard Deviation

Results of Linear Regression between log10 of bioaerosol inlet concentrations and removal efficiencies

(a) *Aspergillus fumigatus*

Model Summary[b]

Model	R	R Square	Adjusted R Square	Std. Error of the Estimate	Durbin-Watson
1	.395[a]	.156	.142	51.48355	1.228

a. Predictors: (Constant), Log10BioConc

b. Dependent Variable: Removal Efficiency for bioaerosol

ANOVA[a]

Model		Sum of Squares	df	Mean Square	F	Sig.
1	Regression	30298.121	1	30298.121	11.431	.001[b]
	Residual	164334.476	62	2650.556		
	Total	194632.598	63			

a. Dependent Variable: Removal Efficiency for bioaerosol

b. Predictors: (Constant), Log10BioConc

Coefficients[a]

Model		Unstandardized Coefficients		Standardized Coefficients	t	Sig.	95.0% Confidence Interval for B	
		B	Std. Error	Beta			Lower Bound	Upper Bound
1	(Constant)	-36.354	28.923		-1.257	.213	-94.170	21.462
	Log10BioConc	27.094	8.014	.395	3.381	.001	11.075	43.114

a. Dependent Variable: Removal Efficiency for bioaerosol

Casewise Diagnostics[a]

Case Number	Std. Residual	Removal Efficiency for bioaerosol	Predicted Value	Residual
47	-3.439	-166.70	10.3641	-177.06413
48	-4.086	-200.00	10.3641	-210.36413

a. Dependent Variable: Removal Efficiency for bioaerosol

Residuals Statistics[a]

	Minimum	Maximum	Mean	Std. Deviation	N
Predicted Value	10.3641	91.6572	58.9813	21.92995	64
Residual	-210.36412	64.77964	.00000	51.07332	64
Std. Predicted Value	-2.217	1.490	.000	1.000	64
Std. Residual	-4.086	1.258	.000	.992	64

a. Dependent Variable: Removal Efficiency for bioaerosol

(b) Total fungi

Model Summary[b]

Model	R	R Square	Adjusted R Square	Std. Error of the Estimate	Durbin-Watson
1	.134[a]	.018	.002	29.25321	1.390

a. Predictors: (Constant), Log10BioConc

b. Dependent Variable: Removal Efficiency for bioaerosol

ANOVA[a]

Model		Sum of Squares	df	Mean Square	F	Sig.
1	Regression	974.254	1	974.254	1.138	.290[b]
	Residual	53056.506	62	855.750		
	Total	54030.760	63			

a. Dependent Variable: Removal Efficiency for bioaerosol

b. Predictors: (Constant), Log10BioConc

Coefficients[a]

Model		Unstandardized Coefficients		Standardized Coefficients	t	Sig.	95.0% Confidence Interval for B	
		B	Std. Error	Beta			Lower Bound	Upper Bound
1	(Constant)	48.431	18.357		2.638	.011	11.736	85.126
	Log10BioConc	5.239	4.910	.134	1.067	.290	-4.576	15.054

a. Dependent Variable: Removal Efficiency for bioaerosol

Residuals Statistics[a]

	Minimum	Maximum	Mean	Std. Deviation	N
Predicted Value	59.9638	73.5617	67.6250	3.93247	64
Residual	-86.25451	32.44835	.00000	29.02011	64
Std. Predicted Value	-1.948	1.510	.000	1.000	64
Std. Residual	-2.949	1.109	.000	.992	64

a. Dependent Variable: Removal Efficiency for bioaerosol

(c) Total mesophilic bacteria

Model Summary[b]

Model	R	R Square	Adjusted R Square	Std. Error of the Estimate	Durbin-Watson
1	.596[a]	.355	.345	42.58910	.677

a. Predictors: (Constant), Log10BioConc

b. Dependent Variable: Removal Efficiency for bioaerosol

ANOVA[a]

Model		Sum of Squares	df	Mean Square	F	Sig.
1	Regression	61885.308	1	61885.308	34.119	.000[b]
	Residual	112457.532	62	1813.831		
	Total	174342.840	63			

a. Dependent Variable: Removal Efficiency for bioaerosol

b. Predictors: (Constant), Log10BioConc

Coefficients[a]

Model		Unstandardized Coefficients		Standardized Coefficients	t	Sig.	95.0% Confidence Interval for B	
		B	Std. Error	Beta			Lower Bound	Upper Bound
1	(Constant)	-156.310	35.474		-4.406	.000	-227.222	-85.399
	Log10BioConc	51.206	8.767	.596	5.841	.000	33.682	68.730

a. Dependent Variable: Removal Efficiency for bioaerosol

Casewise Diagnostics[a]

Case Number	Std. Residual	Removal Efficiency for bioaerosol	Predicted Value	Residual
39	-3.103	-144.30	-12.1455	-132.15453
40	-3.117	-144.90	-12.1455	-132.75453

a. Dependent Variable: Removal Efficiency for bioaerosol

Residuals Statistics[a]

	Minimum	Maximum	Mean	Std. Deviation	N
Predicted Value	-12.1455	98.7261	48.5500	31.34177	64
Residual	-132.75453	46.91557	.00000	42.24974	64
Std. Predicted Value	-1.937	1.601	.000	1.000	64
Std. Residual	-3.117	1.102	.000	.992	64

a. Dependent Variable: Removal Efficiency for bioaerosol

(d) Gram negative bacteria

Model Summary[b]

Model	R	R Square	Adjusted R Square	Std. Error of the Estimate	Durbin-Watson
1	.608[a]	.370	.360	56.38325	.916

a. Predictors: (Constant), Log10BioConc

b. Dependent Variable: Removal Efficiency for bioaerosol

ANOVA[a]

Model		Sum of Squares	df	Mean Square	F	Sig.
1	Regression	115635.213	1	115635.213	36.374	.000[b]
	Residual	197102.384	62	3179.071		
	Total	312737.597	63			

a. Dependent Variable: Removal Efficiency for bioaerosol

b. Predictors: (Constant), Log10BioConc

Coefficients[a]

Model		Unstandardized Coefficients		Standardized Coefficients	t	Sig.	95.0% Confidence Interval for B	
		B	Std. Error	Beta			Lower Bound	Upper Bound
1	(Constant)	-238.420	44.079		-5.409	.000	-326.533	-150.306
	Log10BioConc	69.982	11.604	.608	6.031	.000	46.787	93.177

a. Dependent Variable: Removal Efficiency for bioaerosol

Casewise Diagnostics[a]

Case Number	Std. Residual	Removal Efficiency for bioaerosol	Predicted Value	Residual
35	-3.866	-267.50	-49.5402	-217.95985

a. Dependent Variable: Removal Efficiency for bioaerosol

Residuals Statistics[a]

	Minimum	Maximum	Mean	Std. Deviation	N
Predicted Value	-49.5402	111.4724	24.0062	42.84250	64
Residual	-217.95985	84.18069	.00000	55.93397	64
Std. Predicted Value	-1.717	2.042	.000	1.000	64
Std. Residual	-3.866	1.493	.000	.992	64

a. Dependent Variable: Removal Efficiency for bioaerosol

Table A. 4: Bioaerosol particles (cfu) collected in each stage of the six-stage Andersen sampler (number derived from total cfu per stage for visits 1 – 6 and 13 – 16).

Stage	BG	IN	BF1	BF2	BF3	BF4
Aspergillus fumigatus						
1	47.0	2631.3	268.1	193.4	234.2	261.8
2	71.2	1464.2	296.7	220.6	247.4	492.1
3	88.8	1780	412.8	358.9	342.1	610.6
4	29.6	3595	1605.4	1258	1366.6	1734.5
5	33.1	1889.3	916.5	642.7	849.5	1543.4
6	43.3	331.9	277.4	228	190.5	392
Total	313	11691.7	3776.9	2901.6	3230.3	5034.4
Total fungi						
1	68.7	3384.2	363.2	250.1	330.8	351.6
2	81.3	1856	413.6	312.5	338.3	611.4
3	114.7	2215.3	570	487	454.6	782.4
4	47.1	4902.3	1965.1	1493.2	1632.4	2000
5	50.5	2388.5	1074.2	727.2	951.3	1697.9
6	61.6	424.5	392.3	317.3	258.6	486
Total	423.9	15170.8	4778.4	3587.3	3966	5929.3
Total mesophilic bacteria						
1	214.7	4818	486.3	541.5	406.3	477.7
2	95.2	2621.9	709	487.1	507.5	619.5
3	94.3	2632	642.6	389	624.2	791.7
4	105.4	3441.5	1022.3	702.2	602.8	971.9
5	137.0	6877.9	3431.3	1905.8	2155.2	3896.2
6	455.1	6345.9	3104.3	1093.1	2155	4198.2
Total	1101.7	26737.2	9395.8	5118.7	6451	10955.2
Gram negative bacteria						
1	237.7	1405.3	317.1	245.8	506.9	338.3
2	653.7	1057.1	357	341.8	388.8	480.4
3	131.2	2260.9	458.7	377.9	371.8	329
4	208.8	2248.3	670.2	619.6	616.6	579.9
5	228.7	5853.2	2258	811.2	3023	919.9
6	357.6	3107.9	1220.2	981.3	1667.4	1067.9
Total	1817.7	15932.7	5281.2	3377.6	6574.5	3715.4

BG – Background; IN - Inlet

Appendix B
DATA SET FOR CHAPTER 5

Table B. 1: Mean inlet and outlet bioaerosol concentration (in cfu m^{-3}) for visits 1 – 4 (February and March 2017; n = 2).

		Aspergillus fumigatus				Total fungi				Total mesophilic bacteria				Gram negative bacteria			
		Inlet	**SD**	**Outlet**	**SD**	**Inlet**	**SD**	**Outlet**	**SD**	**Inlet**	**SD**	**Outlet**	**SD**	**Inlet**	**SD**	**Outlet**	**SD**
Visit 1	BF1	11441.7	6346.4	123.7	25.0	14878.1	8647.6	176.7	50.0	13630.8	4170.2	2841.0	1694.1	5915.2	2483.6	2060.1	489.7
	BF2	11441.7	6346.4	964.7	914.5	14878.1	8647.6	1805.7	2003.9	13630.8	4170.2	1268.6	25.0	5915.2	2483.6	2507.1	1266.8
	BF3	11441.7	6346.4	123.7	25.0	14878.1	8647.6	194.4	124.9	13630.8	4170.2	2669.7	1906.4	5915.2	2483.6	2197.9	664.6
	BF4	11441.7	6346.4	300.4	224.9	14878.1	8647.6	424.0	249.9	13630.8	4170.2	2680.2	542.2	5915.2	2483.6	1574.2	47.5
Visit 2	BF1	21404.6	677.1	8063.6	634.7	40706.8	5412.0	11379.9	517.2	60282.7	5327.1	25712.1	16033.6	99940.0	11233.7	26971.7	13547.5
	BF2	21404.6	677.1	23766.8	6611.3	40706.8	5412.0	36012.4	12825.4	60282.7	5327.1	46448.8	13122.7	99940.0	11233.7	41523.0	30423.1
	BF3	21404.6	677.1	9973.5	3485.6	40706.8	5412.0	13259.7	3650.5	60282.7	5327.1	29556.6	6723.8	99940.0	11233.7	36708.5	4869.8
	BF4	21404.6	677.1	1773.9	129.9	40706.8	5412.0	3524.8	517.2	60282.7	5327.1	19655.5	8927.5	99940.0	11233.7	9570.7	1316.8
Visit 3	BF1	10178.5	2811.0	3692.6	329.8	12083.1	2930.9	5289.8	1009.4	47473.5	1804.0	19122.0	8292.9	25722.6	20995.8	13814.5	3265.7
	BF2	10178.5	2811.0	5975.3	50.0	12083.1	2930.9	7952.3	257.4	47473.5	1804.0	18780.9	9634.6	25722.6	20995.8	31812.7	13632.5
	BF3	10178.5	2811.0	3768.6	612.1	12083.1	2930.9	5008.9	442.3	47473.5	1804.0	18616.6	12275.7	25722.6	20995.8	34302.2	7058.6
	BF4	10178.5	2811.0	3243.9	124.9	12083.1	2930.9	4466.5	334.8	47473.5	1804.0	32141.4	15516.4	25722.6	20995.8	4717.3	3987.8
Visit 4	BF1	53044.2	12945.3	37973.5	2191.3	62655.5	16748.2	46558.3	754.6	95627.2	35377.8	58199.7	18087.4	27913.5	6533.9	18390.5	9862.0
	BF2	53044.2	12945.3	35683.8	1736.6	62655.5	16748.2	43210.3	1651.6	95627.2	35377.8	50742.1	16590.8	27913.5	6533.9	11917.0	1506.6
	BF3	53044.2	12945.3	30533.6	4612.5	62655.5	16748.2	34137.8	4327.6	95627.2	35377.8	23980.6	937.0	27913.5	6533.9	10063.6	1714.0
	BF4	53044.2	12945.3	33738.5	6876.2	62655.5	16748.2	37892.3	5729.3	95627.2	35377.8	70263.3	36247.4	27913.5	6533.9	13111.3	4664.9

SD – Standard Deviation

Table B. 2: Bioaerosol particles (cfu) collected in each stage of the six-stage Andersen sampler (number derived from total cfu per stage for visits 1 - 2 [all BFs operated at 16 s]).

Stage	BG	IN	BF1	BF2	BF3	BF4
Aspergillus fumigatus						
1	5.0	219.7	43.1	103.1	36.7	8.0
2	6.0	135.1	54.9	111.5	46.3	8.0
3	6.0	168.7	47.3	170.3	54.8	13.1
4	27.4	801.6	153.4	489.2	244.5	32.5
5	16.1	442.4	119.4	413.4	143.0	36.7
6	3.0	91.6	45.3	112.3	46.2	19.1
Total	63.5	1859.1	463.4	1399.8	571.5	117.4
Total fungi						
1	11.0	310.8	65.6	168.5	54.8	18.1
2	16.1	205.4	77.6	200.2	57.8	18.1
3	14.1	292.0	74.3	293.2	66.8	33.6
4	43.8	1451.3	218.1	674.2	335.2	71.8
5	24.3	746.2	146.3	618.9	180.3	49.4
6	7.0	140.4	72.2	185.5	66.6	32.5
Total	116.3	3146.1	654.1	2140.5	761.5	223.5
Total mesophilic bacteria						
1	55.0	197.9	188.1	125	79.6	46.9
2	22.2	436.6	41.9	90.1	119.8	50.7
3	32.5	242.2	136.7	192.7	165.8	57.2
4	28.2	515.7	97.1	418.0	164.2	89.0
5	58.3	1136.1	211.4	978.7	318.2	676.4
6	59.1	1655.0	563.7	896.3	976.4	344.0
Total	255.3	4183.5	1238.9	2700.8	1824.0	1264.2
Gram negative bacteria						
1	37.5	162.1	55.3	74.0	121.7	27.3
2	22.2	119.6	53.0	104.0	80.7	50.9
3	48.8	140.1	66.9	297.4	88.2	19.0
4	21.1	176.3	103.3	252.1	143	36.5
5	28.2	4086.4	906.7	1117.6	1073.8	219.6
6	53.3	1306.9	458.0	647.0	694.7	277.5
Total	211.1	5991.4	1643.2	2492.1	2202.1	630.8

BG – Background; IN - Inlet

Table B. 3: Bioaerosol particles (cfu) collected in each stage of the six-stage Andersen sampler (number derived from total cfu per stage for visits 3 - 4 [with BF1 and BF2 at 70 s, BF3 and BF4 at 11 s]).

Stage	BG	IN	BF1	BF2	BF3	BF4
Aspergillus fumigatus						
1	32.0	769.3	233.2	303.5	257.1	300.9
2	24.0	547.3	287.2	329.2	213.4	288.4
3	16.0	789.1	500.6	449.6	337.1	376.8
4	48.3	2098.4	1691.0	1650.2	1424.0	1352.8
5	22.0	1557.6	1093.8	1029.3	1006.0	956.8
6	26.0	341.7	260.5	397.1	179.9	437.5
Total	168.3	6103.4	4066.3	4158.9	3417.5	3713.2
Total fungi						
1	38.1	891.1	304.0	363.0	331.5	371.0
2	30.1	656.5	375.6	415.4	261.6	358.8
3	28.0	812.8	668.2	583.7	433.0	482.6
4	64.4	2510.4	2022.5	1958.8	1568.8	1477.6
5	30.1	1756.2	1249.7	1166.7	1059.6	1041.0
6	36.1	398.2	333.6	524.2	232.2	502.5
Total	226.8	7025.2	4953.6	5011.8	3886.7	4233.5
Total mesophilic bacteria						
1	101.3	468.8	282.9	568.8	142.4	222.3
2	122.0	624.1	261.6	347.9	246.6	414.3
3	74.7	853.7	647.8	454.6	441.2	696.1
4	74.6	1164.3	789.6	584.4	270.8	754.8
5	120.0	4731.9	2455.1	2878.5	1875.5	3434.7
6	161.8	4104.7	2627.4	1655.8	1216.5	3438.9
Total	654.4	11947.5	7064.4	6490.0	4193.0	8961.1
Gram negative bacteria						
1	78.5	320.4	102.9	241.0	156.6	101.0
2	44.2	327.0	158.1	199.2	143.6	179.3
3	22.0	745.3	131.5	269.5	86.6	107.3
4	20.0	815.8	438.6	462.2	402.6	325.2
5	70.4	1644.0	1669.4	1208.5	2568.4	533.1
6	157.5	1497.3	772.3	1882.7	726.3	667.2
Total	392.6	5349.8	3272.8	4263.1	4084.1	1913.1

BG – Background; IN - Inlet

Appendix C
DATA SET FOR CHAPTER 6

Table C. 1: Mean inlet and outlet bioaerosol concentration (in cfu m^{-3}) for visits 1 – 4 (March and April 2017; n = 2).

		Aspergillus fumigatus				Total fungi				Total mesophilic bacteria				Gram negative bacteria			
		Inlet	**SD**	**Outlet**	**SD**	**Inlet**	**SD**	**Outlet**	**SD**	**Inlet**	**SD**	**Outlet**	**SD**	**Inlet**	**SD**	**Outlet**	**SD**
Visit 1	BF1	9217.3	1881.5	2176.7	334.8	16099.0	2883.4	3901.1	449.8	20678.5	644.7	8206.7	467.3	20401.1	14694.3	10371.1	4082.8
	BF2	9217.3	1881.5	2374.6	554.7	16099.0	2883.4	3862.2	869.5	20678.5	644.7	11964.7	110.0	20401.1	14694.3	14450.6	1841.5
	BF3	9217.3	1881.5	2823.3	149.9	16099.0	2883.4	4561.9	65.0	20678.5	644.7	3318.1	1524.2	20401.1	14694.3	7222.6	699.6
	BF4	9217.3	1881.5	2404.6	397.3	16099.0	2883.4	3932.9	1449.2	20678.5	644.7	10620.2	4410.0	20401.1	14694.3	8341.0	1866.5
Visit 2	BF1	9616.6	2096.3	1551.3	524.7	13984.1	2756.0	2084.8	559.7	10837.5	554.7	2409.9	100.0	13885.2	1826.5	1989.4	324.8
	BF2	9616.6	2096.3	1450.5	187.4	13984.1	2756.0	2466.5	479.8	10837.5	554.7	1625.5	25.0	13885.2	1826.5	2061.9	387.3
	BF3	9616.6	2096.3	3270.3	332.3	13984.1	2756.0	4477.1	379.8	10837.5	554.7	2814.5	652.1	13885.2	1826.5	4567.1	902.0
	BF4	9616.6	2096.3	2300.4	339.8	13984.1	2756.0	3316.3	17.5	10837.5	554.7	1961.2	539.7	13885.2	1826.5	1909.9	257.4
Visit 3	BF1	10261.5	1049.4	3203.2	847.0	12551.3	539.7	4517.7	907.0	39113.1	13442.5	11669.6	827.0	22157.3	197.4	4809.2	49.9
	BF2	10261.5	1049.4	2190.8	299.8	12551.3	539.7	2954.1	389.8	39113.1	13442.5	9157.3	2006.4	22157.3	197.4	3572.5	69.9
	BF3	10261.5	1049.4	3842.8	877.0	12551.3	539.7	4865.8	1244.3	39113.1	13442.5	8051.3	17.5	22157.3	197.4	6277.4	57.5
	BF4	10261.5	1049.4	2549.5	1076.9	12551.3	539.7	3401.1	1191.8	39113.1	13442.5	10106.0	3912.8	22157.3	197.4	2839.3	17.5
Visit 4	BF1	18938.2	17197.9	3302.1	262.3	24591.9	21205.7	5330.4	802.0	42828.6	14004.7	11130.8	8280.4	17468.2	14739.4	6841.0	2353.7
	BF2	18938.2	17197.9	1411.7	132.4	24591.9	21205.7	2261.5	344.8	42828.6	14004.7	4443.5	112.4	17468.2	14739.4	4236.8	1529.1
	BF3	18938.2	17197.9	6641.3	2955.8	24591.9	21205.7	8521.2	3495.5	42828.6	14004.7	14973.5	6044.1	17468.2	14739.4	6977.1	1671.5
	BF4	18938.2	17197.9	3424.1	1779.0	24591.9	21205.7	5641.4	3030.9	42828.6	14004.7	4515.9	104.9	17468.2	14739.4	3874.6	137.4

SD – Standard Deviation

Table C. 2: Bioaerosol particles (cfu) collected in each stage of the six-stage Andersen sampler (number derived from total cfu per stage for visits 1 - 4 [with BF1 and BF2 operated at 40 – 70% MC, BF3 and BF4 at 10 – 40% MC]).

	Stage	BG	IN	BF1	BF2	BF3	BF4
Aspergillus fumigatus							
	1	56.0	1012.7	74.1	64.2	88.2	70.2
	2	50.0	915.5	116.6	78.1	161.4	92.3
	3	74.7	1208.3	220.2	160.9	340.9	217.7
	4	197.8	1420.2	574.7	386.6	813.4	514.2
	5	60.5	535.0	132.6	124.6	382.4	254.9
	6	22.0	36.0	24.0	18.0	44.0	42.1
	Total	461.0	5127.7	1142.2	832.4	1830.3	1191.4
Total fungi							
	1	80.3	1368.9	126.8	86.3	146.7	122.7
	2	68.2	1252.3	175.3	116.5	234.9	151.2
	3	89.0	1706.2	411.9	273.0	481.6	372.2
	4	219.2	2052.5	836.6	610.5	1069.9	775.9
	5	64.5	617	199.6	163.0	460.0	327.0
	6	30.0	54.1	34.0	36.1	68.2	54.1
	Total	551.2	7051.0	1784.2	1285.4	2461.3	1803.1
Total mesophilic bacteria							
	1	269.3	1644.8	314.9	168.0	181.7	229.0
	2	165.3	1697.5	269.2	188.0	183.7	218.4
	3	185.8	1682.0	319.8	246.8	416.7	297.5
	4	145.6	1834.1	562.8	574.6	639.1	315.6
	5	371.9	3119.3	1430.0	915.2	1036.5	977.8
	6	200.7	1491.0	689.7	843.4	665.6	900.4
	Total	1338.6	11468.7	3586.4	2936.0	3123.3	2938.7
Gram negative bacteria							
	1	131.0	1024.8	228.2	187.7	242.7	62.2
	2	74.4	1159.6	157.0	116.7	270.6	124.5
	3	108.8	1589.4	177.7	217.8	209.9	94.4
	4	141.8	1246.4	255.3	224.1	504.5	202.0
	5	161.9	1558.6	1145.6	1274.2	937.2	947.3
	6	143.2	1173.6	629.2	530.1	551.6	419.8
	Total	761.1	7752.4	2593.0	2550.6	2716.5	1850.2

BG – Background; IN - Inlet

Appendix D
DATA SET FOR CHAPTER 7

Table D. 1: Mean inlet and outlet bioaerosol concentration (in cfu m^{-3}) for visits 1 – 4 (May 2017; n = 2).

		Aspergillus fumigatus				Total fungi				Total mesophilic bacteria				Gram negative bacteria			
		Inlet	**SD**	**Outlet**	**SD**	**Inlet**	**SD**	**Outlet**	**SD**	**Inlet**	**SD**	**Outlet**	**SD**	**Inlet**	**SD**	**Outlet**	**SD**
Visit 1	BF1	15303.9	2693.5	2132.5	437.3	19515.9	3518.0	3303.9	589.7	32613.1	12355.6	6818.0	222.4	27448.8	5342.0	8485.9	1336.8
	BF2	15303.9	2693.5	5560.1	147.4	19515.9	3518.0	8047.7	1051.9	32613.1	12355.6	13459.4	919.5	27448.8	5342.0	15531.8	9487.2
	BF3	15303.9	2693.5	5125.4	277.3	19515.9	3518.0	7166.1	594.7	32613.1	12355.6	10386.9	602.2	27448.8	5342.0	9314.5	3588.0
	BF4	15303.9	2693.5	1554.8	75.0	19515.9	3518.0	2268.6	109.9	32613.1	12355.6	8432.9	752.1	27448.8	5342.0	6683.7	5494.4
Visit 2	BF1	67296.8	1464.2	5604.2	479.7	95318.0	21433.1	7885.2	417.3	107261.5	63649.6	28125.4	4684.9	18798.6	5966.7	11902.8	102.4
	BF2	67296.8	1464.2	9549.5	2341.2	95318.0	21433.1	13289.8	2628.5	107261.5	63649.6	23005.3	9147.4	18798.6	5966.7	10966.4	427.3
	BF3	67296.8	1464.2	13818.0	4824.8	95318.0	21433.1	20413.4	5162.1	107261.5	63649.6	16699.6	3298.2	18798.6	5966.7	8448.8	519.7
	BF4	67296.8	1464.2	8738.5	749.6	95318.0	21433.1	12116.6	1494.2	107261.5	63649.6	11010.6	5696.8	18798.6	5966.7	5056.5	604.7
Visit 3	BF1	44924.0	40045.2	8185.5	4724.9	56289.8	50461.9	10238.5	6064.1	142455.8	58792.3	19235.0	2820.9	58696.1	3133.3	13180.2	8185.4
	BF2	44924.0	40045.2	6925.8	1759.0	56289.8	50461.9	9151.9	2268.7	142455.8	58792.3	8441.7	249.9	58696.1	3133.3	7646.6	2163.8
	BF3	44924.0	40045.2	13507.1	4624.9	56289.8	50461.9	16280.9	4460.0	142455.8	58792.3	14470.0	7370.9	58696.1	3133.3	9777.4	839.5
	BF4	44924.0	40045.2	12411.7	4859.8	56289.8	50461.9	15245.6	4135.2	142455.8	58792.3	23312.7	16123.5	58696.1	3133.3	13249.1	1926.4
Visit 4	BF1	17673.1	4195.2	7812.7	1372.2	20922.3	4312.6	10098.7	1573.6	28130.7	834.5	22664.3	2640.5	28146.6	2281.2	12342.5	2261.3
	BF2	17673.1	4195.2	6989.4	2513.6	20922.3	4312.6	8508.8	2473.6	28130.7	834.5	21333.6	9426.5	28146.6	2281.2	6970.0	2980.8
	BF3	17673.1	4195.2	16263.3	4460.0	20922.3	4312.6	17851.6	4387.6	28130.7	834.5	23781.4	4859.5	28146.6	2281.2	12417.0	9588.8
	BF4	17673.1	4195.2	8162.5	2703.5	20922.3	4312.6	9936.4	2833.4	28130.7	834.5	14107.8	12.5	28146.6	2281.2	10531.8	1066.9

SD – Standard Deviation

Table D. 2: Bioaerosol particles (cfu) collected in each stage of the six-stage Andersen sampler (number derived from total cfu per stage for visit 1.

Stage	*Aspergillus fumigatus*			Total fungi			Total mesophilic bacteria			Gram negative bacteria		
	IN	0.50m	0.25m	IN	0.50m	0.25m	IN	0.50m	0.25m	IN	0.50m	0.25m
1	318.8	86.6	86.6	407.5	131.6	127.4	1042.6	336.6	153.8	632.2	329.0	323.3
2	336.9	78.5	115.7	405.4	150.1	163.3	618.4	247.1	490.8	675.9	440.7	318.7
3	361.2	140.0	129.8	435.1	229.4	196.2	592.4	278.7	266.0	562.4	336.8	178.8
4	354.3	351.8	276.1	439.8	452.8	378.5	363.7	261.6	315.2	315.1	333.3	256.8
5	177.4	121.1	80.6	248.6	172.4	96.8	380.8	561.5	436.3	347.1	622.8	304.7
6	99.6	74.4	54.3	141.2	115.2	80.8	273.0	525.2	401.1	302.9	512.8	371.2
Total	1648.2	852.4	743.1	2077.6	1251.5	1043.0	3270.9	2210.7	2063.2	2835.6	2575.4	1753.5

IN - Inlet

Table D. 3: Bioaerosol particles (cfu) collected in each stage of the six-stage Andersen sampler (number derived from total cfu per stage for visit 2.

Stage	*Aspergillus fumigatus*			Total fungi			Total mesophilic bacteria			Gram negative bacteria		
	IN	0.50m	0.25m	IN	0.50m	0.25m	IN	0.50m	0.25m	IN	0.50m	0.25m
1	1636.4	135.7	240.7	2153.2	201.5	369.2	2727.6	424.3	365.2	565.6	242.6	214.4
2	1173.2	160.5	378.0	1491.2	268.1	550.4	1229.0	427.1	373.4	292.3	353.3	229.0
3	1616.1	379.6	550.5	2358.0	608.5	801.8	1889.2	750.5	473.7	306.2	215.9	226.4
4	1381.1	607.1	950.0	1681.4	736.8	1306.7	1029.1	1025.2	400.1	352.6	282.8	164.3
5	234.8	306.4	201.4	278.4	367.8	259.4	1278.1	1534.9	710.8	179.4	726.8	281.4
6	50.4	66.4	93.1	68.8	108.8	117.7	938.0	1169.0	660.2	302.9	657.0	377.9
Total	6092.0	1655.7	2413.7	8031.0	2291.5	3405.2	9091.0	5331.0	2983.4	1999.0	2478.4	1493.4

IN - Inlet

Table D. 4: Bioaerosol particles (cfu) collected in each stage of the six-stage Andersen sampler (number derived from total cfu per stage for visit 3.

Stage	*Aspergillus fumigatus*			Total fungi			Total mesophilic bacteria			Gram negative bacteria		
	IN	0.50m	0.25m	IN	0.50m	0.25m	IN	0.50m	0.25m	IN	0.50m	0.25m
1	711.9	161.1	361.3	785.8	219.1	459.4	1909.6	424.6	514.3	882.6	254.3	240.7
2	557.1	272.3	469.0	694.1	352.3	578.4	3007.6	268.2	477.6	897.3	313.6	316.6
3	1076.6	350.1	634.3	1345.2	479.7	774.1	1407.1	238.8	586.9	1000.5	432.8	377.2
4	1158.6	675.9	943.2	1556.8	806.3	1111.2	1288.6	409.6	761.8	1048.6	425.4	423.0
5	553.4	147.8	297.7	625.0	184.9	331.2	1991.9	1038.4	1040.1	1055.0	318.4	545.0
6	111.1	38.1	74.5	34.1	52.2	99.1	2009.2	560.9	564.8	798.2	517.3	604.8
Total	4168.7	1645.3	2780.0	5041.0	2094.5	3353.4	11614.0	2940.5	3945.5	5682.2	2261.8	2507.3

IN - Inlet

Table D. 5: Bioaerosol particles (cfu) collected in each stage of the six-stage Andersen sampler (number derived from total cfu per stage for visit 4.

Stage	*Aspergillus fumigatus*			Total fungi			Total mesophilic bacteria			Gram negative bacteria		
	IN	0.50m	0.25m	IN	0.50m	0.25m	IN	0.50m	0.25m	IN	0.50m	0.25m
1	143.2	134.5	62.3	204.8	188.2	96.8	465.9	439.8	458.5	642.6	336.8	586.3
2	110.0	133.9	123.3	164.6	181.7	155.9	329.8	451.9	342.5	363.8	375.9	399.4
3	239.3	232.0	298.0	292.1	320.2	382.6	437.6	544.8	466.5	418.9	319.8	567.8
4	523.8	583.5	693.0	603.3	706.6	810.0	501.2	574.3	1222.5	373.1	422.7	741.1
5	782.8	1296.2	1332.9	838.9	1381.5	1398.5	586.2	1078.8	2720.4	520.5	754.1	1169.5
6	32.2	180.1	26.0	52.5	224.0	36.0	611.5	1301.7	3731.7	616.2	1052.8	1268.1
Total	1831.3	2560.2	2535.5	2156.2	3002.2	2879.8	2932.2	4391.3	8942.1	2935.1	3262.1	4732.2

IN - Inlet

Appendix E
DATA SET FOR CHAPTER 8

Table E. 1: Mean inlet and outlet bioaerosol concentration (in cfu m^{-3}) for visits 1 – 4 (June and July 2017; n = 2).

		Aspergillus fumigatus				Total fungi				Total mesophilic bacteria				Gram negative bacteria			
		Inlet	SD	Outlet	SD	Inlet	SD	Outlet	SD	Inlet	SD	Outlet	SD	Inlet	SD	Outlet	SD
Visit 1	BF1	36459.4	1264.3	5183.7	839.5	40879.9	1794.0	6586.6	1254.3	35763.3	979.5	5204.9	1998.9	32295.1	6963.6	5455.8	974.5
	BF2	36459.4	1264.3	888.7	2.5	40879.9	1794.0	1247.3	199.9	35763.3	979.5	1577.7	522.2	32295.1	6963.6	2468.2	502.2
	BF3	36459.4	1264.3	6823.3	564.7	40879.9	1794.0	8966.4	612.2	35763.3	979.5	5614.8	634.6	32295.1	6963.6	8505.3	2643.5
	BF4	36459.4	1264.3	13386.9	482.2	40879.9	1794.0	16607.8	80.0	35763.3	979.5	3187.3	0.0	32295.1	6963.6	6261.5	1903.9
Visit 2	BF1	26314.5	4027.8	10712.0	87.5	30620.1	4265.1	13176.7	319.8	45049.5	349.8	13279.2	10124.4	14839.2	3170.7	8333.9	3675.5
	BF2	26314.5	4027.8	3153.7	967.0	30620.1	4265.1	4212.0	859.5	45049.5	349.8	15613.1	7628.3	14839.2	3170.7	6500.0	3850.4
	BF3	26314.5	4027.8	5745.6	1069.4	30620.1	4265.1	6818.0	1176.8	45049.5	349.8	7482.3	907.0	14839.2	3170.7	5450.5	1691.6
	BF4	26314.5	4027.8	13081.3	2673.5	30620.1	4265.1	15261.5	3073.3	45049.5	349.8	12044.2	4395.1	14839.2	3170.7	3980.6	362.3
Visit 3	BF1	28091.9	739.6	61351.6	25003.6	38669.6	87.5	96743.8	53242.9	39924.0	15169.1	37063.6	16870.6	23821.6	3395.6	27134.3	18899.5
	BF2	28091.9	739.6	335.7	124.9	38669.6	87.5	477.0	25.0	39924.0	15169.1	3001.8	52.5	23821.6	3395.6	3250.9	1764.0
	BF3	28091.9	739.6	7987.6	22.5	38669.6	87.5	9708.5	122.4	39924.0	15169.1	36351.6	6558.9	23821.6	3395.6	21219.1	8590.2
	BF4	28091.9	739.6	7492.9	532.2	38669.6	87.5	9319.8	922.0	39924.0	15169.1	18982.3	784.6	23821.6	3395.6	11489.4	467.2
Visit 4	BF1	20330.4	1786.5	6318.0	924.5	25455.8	1654.1	7507.1	737.1	37713.8	8400.3	26590.1	8050.5	39367.5	26660.2	18113.1	5217.1
	BF2	20330.4	1786.5	1964.7	104.9	25455.8	1654.1	2489.4	332.3	37713.8	8400.3	3980.6	2356.2	39367.5	26660.2	7265.0	6186.6
	BF3	20330.4	1786.5	9404.6	1196.8	25455.8	1654.1	10706.7	1164.4	37713.8	8400.3	24438.2	11393.7	39367.5	26660.2	15040.6	1151.9
	BF4	20330.4	1786.5	19747.3	4350.1	25455.8	1654.1	22934.6	6239.0	37713.8	8400.3	19669.6	6369.0	39367.5	26660.2	9848.1	4162.7

SD – Standard Deviation

BG – Background; IN – Inlet; BF1 – old woodchips; BF2 – peat; BF3 – wheat straw; BF4 – new woodchips

Table E. 2: Bioaerosol particles (cfu) collected in each stage of the six-stage Andersen sampler (number derived from total cfu per stage for visits 1 – 4).

	Stage	BG	IN	BF1	BF2	BF3	BF4
Aspergillus fumigatus							
	1	63.8	714.4	240.8	69.3	143.3	291.4
	2	42.3	782.9	304.5	70.5	133.8	282.2
	3	32.0	1244.2	466.1	43.4	230.8	457.0
	4	35.3	2286.7	2268.8	66.8	728.3	1120.4
	5	34.2	1153.7	1263.7	49.3	340.6	804.9
	6	47.4	111.8	185.9	59.7	119.0	84.0
	Total	255.0	6293.7	4729.8	359.0	1695.8	3039.9
Total fungi							
	1	88.4	843.0	309.1	89.0	171.6	348.0
	2	60.6	928.6	383.1	85.1	162.3	352.1
	3	52.5	1597.6	686.7	56.7	317.9	623.1
	4	46.5	2901.2	3667.4	98.0	862.3	1322.1
	5	44.4	1257.7	1743.2	68.8	379.2	874.5
	6	65.0	148.3	229.7	79.3	155.6	109.6
	Total	357.4	7676.4	7019.2	476.9	2048.9	3629.4
Total mesophilic bacteria							
	1	124.8	1082.8	146.8	231.7	306.8	322.6
	2	111.6	809.8	295.7	318.4	311.7	253.0
	3	104.8	1623.5	370.5	163.5	256.8	171.5
	4	148.0	1333.3	496.0	189.1	337.2	270.0
	5	190.4	2443.4	1550.4	170.9	1324.7	1042.6
	6	339.7	1675.5	1789.6	294.6	1644.8	990.1
	Total	1019.3	8968.3	4649.0	1368.2	4182.0	3049.8
Gram negative bacteria							
	1	204.4	1388.6	185.0	164.4	313.2	224.5
	2	162.3	790.7	237.0	161.9	188.9	186.1
	3	114.7	889.5	293.6	113.3	218.9	255.6
	4	114.5	912.1	372.7	129.5	212.6	263.7
	5	142.9	1311.2	1017.4	306.8	644.9	437.1
	6	327.7	952.2	1235.8	226.9	1263.7	420.4
	Total	1066.5	6244.3	3341.5	1102.8	2842.2	1787.4

BG – Background; IN – Inlet

BF1 – old woodchips; BF2 – peat; BF3 – wheat straw; BF4 – new woodchips