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Compositional and physicochemical changes in waste materials and biogas production across 7 landfill sites in UK

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ABSTRACT

The aim of this study was to evaluate the spatial distribution of the paper and fines across seven landfill sites (LFS) and assess the relationship between waste physicochemical properties and biogas production. Physicochemical analysis of the waste samples demonstrated that there were no clear trends in the spatial distribution of total solids (TS), moisture content (MC) and waste organic strength (VS) across all LFS. There was however noticeable difference between samples from the same landfill site. The effect of landfill age on waste physicochemical properties showed no clear relationship, thus, providing evidence that waste remains dormant and non-degraded for long periods of time. Landfill age was however directly correlated with the biochemical methane potential (BMP) of waste; with the highest BMP obtained from the most recent LFS. BMP was also correlated with depth as the average methane production decreased linearly with increasing depth. There was also a high degree of correlation between the Enzymatic Hydrolysis Test (EHT) and BMP test results, which motivates its potential use as an alternative to the BMP test method. Further to this, there were also positive correlations between MC and VS, VS and biogas volume and biogas volume and CH_4 content. Outcomes of this work can be used to inform waste degradation and methane enhancement strategies for improving recovery of methane from landfills.

1. Introduction

MSW in landfills is composed of a number of different organic and inorganic materials, such as food, paper, wood, plastics, glass, metal and inert materials (Machado et al., 2009). The composition of this waste within any landfill site is highly variable, and is affected by a number of factors, including: location, climate, landfill age and local policy drivers.

Bacterial decomposition of the waste accounts for the majority of the landfill gas produced which principally consists of methane and carbon dioxide ranging between 50–55% and 40–45%, respectively (Johari et al., 2012; Donovan et al., 2011).

The process makes use of a complex, interactive network of aerobic and anaerobic microorganisms to degrade organic material (food, garden waste, wood, paper) (Godley et al., 2004). A number of chemical reactions can also account for landfill gas production on account of the combination of waste materials during disposal (Bogner and Spokas, 1993). The quantity of landfill gas along with its methane content is influenced by several factors, which include among others the types and age of the waste buried in the landfill, the quantity and types of organic compounds in the waste, and the moisture content and temperature of the waste (Emkes et al., 2015). Methane is highly combustible, and as a result can be exploited as a source of renewable energy (Rada et al., 2015). In the UK, the utilisation of landfill gas for energy has been particularly successful (Brown and Maunder, 1994; Emkes et al., 2015). The UK however have adopted a number of waste management policies aimed at reducing the amount of waste, in particular biodegradable waste, sent to landfill sites (Laner et al., 2012). The EU Landfill Directive 99/31/EC (1999) further transformed trends in MSW disposal by creating ambitious targets to shift waste away from landfill sites. This was coupled with requirements for landfill sites to install best-practise methane recovery technologies (Defra, 2006). Other policy drivers in the UK that incentivise waste diversion from landfill include: the introduction of landfill tax (1996), the Landfill Allowance Trading Schemes (LATS) (2005/06) (Defra, 2006), the EU Waste Framework Directive 2008/98/EC (European Commission, 2008) and the Renewable Energy Directive 2009/28/

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EC (European Commission, 2009). UK waste policy drivers have certainly led to a significant change in the composition of waste currently in UK landfill sites, and consequently on gas production. The effect on the future landfill gas generation is still to be fully elucidated.

A number of aerobic and anaerobic methods are currently available to assess the biodegradability of waste thus providing insight into the potential biogas production. Aerobic tests such as DR4 and ASTM, are typically more rapid than anaerobic tests but do not fully measure biodegradability (Wagland et al., 2008). On the contrary, anaerobic tests (i.e. GB21, GS90 and BM100) provide a more complete assessment of waste biodegradability but the tests take substantially longer (>30 days) and therefore are not suitable for routine monitoring application (Wagland et al., 2009; Shanmugam and Horan, 2008). Therefore, the development of a rapid, low cost test method, which can accurately assess the biodegradability of waste, will be valuable for monitoring and predictive application.

Previously an enzymatic hydrolysis tests (EHT) method was developed by Wagland et al. (2007, 2008), which followed work by Godley et al. (2004) on enzymatic hydrolysis of cellulosic material and Rodriguez et al. (2001) on the enzymatic availability of cellulose in organic waste. Studies published to date on using the EHT indicate a good correlation between EHT results and the standardised BM 100 test results (Wagland et al., 2008, 2011).

The aim of this study was to determine the current state of waste at UK landfill sites, i.e. compositional and physicochemical properties, and its relationship with landfill gas production. The specific objectives were to (1) evaluate the spatial distribution of unspent carbon at seven landfill sites, (2) determine the physicochemical properties of waste distributed between different landfill sites and a range of depths, (3) understand the relationship between organic waste distribution, waste composition and waste physicochemistry, and its effect on landfill gas production and methane yield and (4) evaluate the use of enzymatic hydrolysis as a feasible, cost-effective and rapid test method to determine potential biogas production.

2. Materials and methods

2.1. Waste origin

95 municipal solid waste (MSW) samples were obtained from 7 UK landfill sites (LFS) during drilling activities between October 2013 and June 2014. The sites are referred to as LFS 1-7. Two separate tests sites were evaluated at LFS 4, which are referred to as LFS 4.1 and LFS 4.2, respectively. Background information on the LFS is provided in Table 1. Approximately 5 kg of waste was sampled at each depth. The waste samples were then collected within a maximum of 2 weeks after drilling and stored at 4 °C until analysis.

Table 1
Background information on the tested landfill sites 1–7.

Landfill site	Age of landfill site (as of 2015) (years)	Status of landfill (as of 2015)	Capacity of site (Mt)	Tonnage received per year (kt year^{-1})
LFS 1	35	Closed	5.8	200–300
LFS 2	23	Open	6.6	300
LFS 3	22	Open	4.2	200–250
LFS 4	19	Closed	5.0	200–250
LFS 5	8	Open	0.9	100
LFS 6	35	Open	1.4	50
LFS 7	7	Closed	1.1	100–150

2.2. Waste samples preparation and characterisation

Waste was screened and separated by hand into plastics, paper and fines as defined by Quaghebeur et al. (2013) (degraded garden and food materials), textiles, glass and metal. Waste composition was reported as a percentage contribution of total weight. Dry matter (DM) or Total solids (TS) and Volatile Solids (VS) of the paper and fines were determined in triplicate following the procedure BS EN 12879:2000 (Wagland, 2008). This was done to calculate the amount of moisture and VS present in the waste samples. It should be noted though that the VS determination should not be taken as a direct correlation with the organic part of the waste as the loss of volatile organic substances (i.e. plastics) can occur and therefore giving high VS results (Section 5 of the BS EN 12879:2000 standards). The waste samples were dried at 105 °C for 24 h for the DM determination and at 505 °C for 4 h for the VS determination. The paper and fines samples were shredded to particle size of 8 mm as recommended by Wagland (2008). pH and soluble Chemical Oxygen Demand (sCOD) of the separated paper and fines were determined according to the Standard Analytical Methods published by the American Public Health Association (APHA, 1995). sCOD was conducted in duplicate due to reliability of test kits while all other tests were conducted in triplicate.

2.3. Biochemical methane potential

The BMP test were carried out as described in Garcia et al. (2016). Briefly each BMP test was conducted by mixing 20 g loss on ignition (LOI) equivalent of the paper and fines with 40 g LOI equivalent of digested primary sludge in a 1 l bottle. Sludge was obtained from the Cotton Valley wastewater treatment plant in Milton Keynes, UK and was used to introduce the active microbial community. The bottles were filled with distilled water up to 500 ml, leaving a headspace of 500 ml, and flushed with nitrogen gas to set anaerobic conditions. Bottles were thereafter sealed, and incubated at 38°C in a water bath. The volume of biogas was measured volumetrically daily until no more biogas was produced. The concentration of methane in the biogas was determined once a week using either gas chromatography or a gas analyser (Servomex 1440 GA), depending on the availability of the measuring device. Further to this, sludge alone and sludge + cellulose (both in the absence of waste) were used as control BMP tests as recommended in the WRAP guidelines (Walker et al., 2010). Cellulose was used at a concentration of 10 g kg^{-1}.

The amount of biogas produced was calculated considering the area of the columns (specific to this study) and the environmental conditions of the laboratory, according to Eq. (1) (Walker et al., 2010):

$$V_{stp} = \frac{T_{stp} \cdot A}{T_{atm} \cdot P_{stp}} \cdot [(P_{atm-}P_{H2O} - p.g.(H-h)) \cdot h] \quad (1)$$

where V_{stp} = volume of biogas (in standard conditions), T_{stp} = standard temperature, T_{atm} = ambient temperature, P_{stp} = standard pressure, P_{atm} = ambient pressure, P_{H2O} = pressure of the water, ρ = density, g = gravity, H = distance from the bottom to the top of the column, h = void distance in the column.

The biogas production of the inoculum was subtracted when calculating the amount of the biogas produced by the waste samples. The volume of biogas produced is presented as l biogas per kg total wet waste. Methane production is presented in ml CH_4/g VS waste.