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Estimating biogas production of biologically treated municipal solid waste

Barbara Scaglia *, Roberto Confalonieri, Giuliana D'Imporzano, Fabrizio Adani *

RICICLA Group, DiProVe, Università degli Studi di Milano, Via Celoria 2, 20133 Milano, Italy

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ABSTRACT

In this work, a respirometric approach, i.e., Dynamic Respiration Index (*DRI*), was used to predict the anaerobic biogas potential (*ABP*), studying 46 waste samples coming directly from MBT full-scale plants. A significant linear regression model was obtained by a jackknife approach: $ABP = (34.4 \pm 2.5) + (0.109 \pm 0.003) \cdot DRI$. The comparison of the model of this work with those of the previous works using a different respirometric approach (*Sapromat-AT_4*), allowed obtaining similar results and carrying out direct comparison of different limits to accept treated waste in landfill, proposed in the literature. The results indicated that on an average, MBT treatment allowed 56% of *ABP* reduction after 4 weeks of treatment, and 79% reduction after 12 weeks of treatment. The obtainment of another regression model allowed transforming *Sapromat-AT_4* limit in *DRI* units, and achieving a description of the kinetics of *DRI* and the corresponding *ABP* reductions *vs.* MBT treatment-time.

1. Introduction

Municipal solid waste (MSW), when landfilled, causes several environmental problems such as the biogas production, volatile organic compounds (VOC) emission, leachate, the presence of vectors (e.g., insects, rodents, and birds), public health hazard, risk of explosions, and plants toxicity (Thorneloe and Pacey, 1994), because of the presence of the biodegradable organic fraction (BMSW). All these negative impacts and the long time required to stabilize the materials (after care period) are the major issues that make landfills unsustainable. Nevertheless, landfill is the most diffused strategy to manage MSW in the world. The IPCC (2006) reported that about 67%, 69%, 61%, 63%, and 75% of the total MSW generated are landfilled in Europe, Africa, America, Asia, and Oceania, respectively. Among the different landfill impacts, biogas emission is the one with the highest environmental impact because of its greenhouses effect and the related impact on climate change (IPCC, 2006). According to USEPA (2003), landfills have been responsible for 13% (36.2×10^6 mg) of the global anthropogenic biogas emission (282.6×10^6 mg) during 2000.

To reduce the MSW impacts in landfill, the European Commission emanated a Landfill Directive (European Union, 1999) to drive the member states to draw up strategies for progressively reducing the amount of BMSW in landfill over a 15-year period, to reach 35% of the amount produced in 1995.

Three main strategies are currently available to reduce MSW in landfill: (i) MSW burning; (ii) source separation of BMSW; and (iii) Mechanical Biological Treatment (MBT). The MBT is extensively used in Europe to treat MSW (e.g., Italy, Germany, and Austria). Many other countries, such as United Kingdom, France and Spain, have now started adopting MBT as their main strategy to reduce the organic fraction of MSW in waste (Robinson et al., 2005; Lornage et al., 2007; Ponsá et al., 2008). The MBT consists of mechanical pre-treatment of the MSW, followed by an aerobic (composting-like) process or anaerobic/aerobic degradation (Wiemer and Kern, 1995; Bockreis and Steinberg, 2005; Fricke et al., 2005; Martens, 2005; Robinson et al., 2005). In particular, MBT involves primary mechanical screening (grid holes of 40–80 mm) to obtain two fractions. The upper-grid fraction (35–50% of wet weight, w.w.), which consists mainly of plastic and paper, is used as a refuse-derived fuel or is landfilled without further treatment (Wiemer and Kern, 1995; El-Fadel et al., 2002). The lower-grid fraction (50–65% w.w.) is biologically treated to reduce its biological reactivity prior to landfill disposal (Norbu et al., 2005).

The measurement of the biological reactivity, i.e., biological stability (Scaglia and Adani, 2008a), has become important, as it allows the classification of treated and untreated MSW and derived products, prior to their disposal in landfill, on the basis of their potential impact, with particular reference to the biogas production.

The possibility to directly measure the biogas producible by MSW may allow for a more accurate estimation of the possible impact of waste in producing greenhouse gases, when MSW is landfilled.

Analytical methods capable of measuring the potential biogas production from MSW have been proposed in the past (Binner

* Corresponding authors. Tel.: +39 02 503 16545; fax: +39 02 503 16521.
E-mail addresses: barbara.scaglia@unimi.it (B. Scaglia), fabrizio.adani@unimi.it (F. Adani).

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and Zach, 1999). Biogas laboratory tests have been classified into fermentative and incubation tests (Adani et al., 2001); the latter being widely preferred, as they are carried out under conditions that avoid toxic phenomena (Adani et al., 2001; Scaglia and Adani, 2008b).

Regardless of the test used, biogas detection is not practicable as a routine analysis, as it is expensive and time-consuming, requiring 21 (partial biogas production) to 100 days (total biogas production) (Binner and Zach, 1999; Adani et al., 2001; Hansen et al., 2004). Hence, these tests are not routinely used to characterize waste prior to their disposal in landfill, and other approaches have been suggested (Scaglia and Adani, 2008b). Among them, the measurement of respirometric activity is one of the most applied methods. Respirometric approach consists of the measurement of O_2 uptake or CO_2 production under standardized conditions, by microorganisms degrading the readily degradable fraction of the organic matter under standardized aerobic conditions (Adani et al., 2004). Thus, biological stability measures the degradability of the readily available organic matter contained in the waste, in a short period of time (1–4 days) (Adani et al., 2004).

Nevertheless, respirometric approaches have been routinely used to describe the long-term impact of waste, i.e., potential biogas production, and some relationships between respiration indices and potential biogas production have been reported (Anonymous, 2001; Cossu and Raga, 2008; Ponsá et al., 2008; Wagland et al., 2009).

The Dynamic Respiration Index (*DRI*) represents a respirometric approach for the evaluation of waste reactivity, presenting some advantages with respect to other methods, as it allows a large mass (up to 13 kg) to be tested under simulated full-scale conditions. This indicates that waste can be tested under standardized conditions as such, approaching conditions close to reality (Adani et al., 2001). Recently, *DRI* had received official recognition from the European Committee for Standardization (CEN, 2007) and the Italian Organization for Standardization (UNI) (2006). Moreover, the precision of the method (i.e. repeatability and reproducibility values) (ISO, 1994), has been provided at national level (Scaglia and Adani, 2009) and it is currently under consideration at an international level (CEN, 2008).

Therefore, owing to the significance of *DRI* to measure the waste impact in landfill, it seems important to carry out a correct evaluation of its ability to describe the potential biogas production (*ABP*) of waste prior to their disposal in landfill. Previous work indicated, as average, *DRI* of 1675 ± 1110 mg O_2 kg DM^{-1} h^{-1} and *ABP* of 308 ± 71 l kg DM^{-1} for untreated MSW, and *DRI* of 300 ± 216 mg O_2 kg DM^{-1} h^{-1} and *ABP* of 108 ± 24 l kg DM^{-1} for biostabilized MSW (Scaglia and Adani, 2008b).

This paper aims to fill this lack of knowledge by studying the relation between *DRI* and *ABP* on a large number of samples coming from full-scale plants for the treatment of MSW by MBT, and proposing a model to predict *ABP* by *DRI* measurement.

2. Methods

2.1. Wastes studied

Untreated and aerobically treated samples (#1–27 and #28–46, respectively) were tested in this study. The samples represented the under-sized fractions (particle size <60 mm) of MSW after the mechanical screening of MSW performed in full-scale MBT plants, in Italy.

The waste samples were collected directly from the full-scale plants by using standard procedures (CEN, 2006) before and after the biological treatment. The weight of each sample brought to the laboratory was about 40–50 kg (w.w.). The accepted samples were stored at 4 °C and processed within 3–5 days from the date of receiving it. A homogeneous sub-sample of 1–3 kg (depending on the particle size) was taken from each biomass to determining the moisture content after drying at 105 °C, and volatile solids (VS) after burning at 550 °C.

2.2. Wastes characterization

All the 46 samples were analyzed for both *DRI* and *ABP*. The *DRI* was measured using a 30-l adiabatic respirometric reactor (Costech International, Cernusco S.N., Italy; DiProVe, Milan, Italy) (Adani et al., 2001). About 10–13 kg of the wet samples were used for the tests. The samples were optimized for the moisture content (75% w.w. of the water-holding capacity) (Adani et al., 2001), and tests were performed by setting O_2 concentration at 140 ml l^{-1} in the outlet airflow (Adani et al., 2001). This value was maintained by a feedback control that automatically adapted airflow rate as a function of the O_2 concentration in the outlet airflow.

The hourly *DRI* (DRI_h) was determined by measuring the difference in the O_2 concentration (ml l^{-1}) between the respirometer inlet and outlet airflow, and was calculated as reported by Adani et al. (2006) (Eq. (1)):

$$DRI_h\ (\text{mg O}_2\ \text{kg DM}^{-1}\ \text{h}^{-1}) = Q \cdot \Delta O_2 \cdot Vg^{-1} \cdot 31.98 \cdot DM^{-1}, \quad (1)$$

where DRI_h is the hourly *DRI*, Q (l h^{-1}) is the airflow rate, ΔO_2 (ml l^{-1}) is the difference in the O_2 concentration in the inlet and outlet air flow of the reactor, Vg (l mol^{-1}) is the volume of 1 mol of gas at the inlet air temperature, 31.98 (g mol^{-1}) is the molecular weight of O_2 and DM (kg) is the initial total dry-matter content.

The *DRI* was calculated as the average of the 24 DRI_h values taken over the 24-h period, characterized by the most intense biological activity (mobile mean), avoiding lag-phase (Adani et al., 2004) (Eq. (2)). Each sample was tested twice. On average, about 58 h was required to determine the *DRI* (Adani et al., 2004).

$$DRI\ (\text{mg O}_2\ \text{kg DM}^{-1}\ \text{h}^{-1}) = \sum_{\theta=0}^{24} (DRI_h)/24. \quad (2)$$

The *ABP* of all the samples was determined by using the method described by Schievano et al. (2008). In 500-ml serum bottles, 3 g of dried sample was added to 187.5 ml of inoculum and 100 ml of deionized water. Inoculum in stable methanogenic activity (CH_4 > 60% in biogas, v/v) was obtained using the output digestate of the post digester of plant treating the organic fraction of municipal solid waste. The pH was around 7.8; DM and VS contents were about 30–40 g kg^{-1} on wet weight basis (w/w) and 700–800 g kg^{-1} on DM basis, respectively. Digestate was incubated at 37 ± 1 °C for 15 d before use (Schievano et al., 2008). The fresh feedstock to inoculum DM ratio was 1:2. Control blanks experiments were prepared using 300 ml of inoculum.

All the batches were sealed with Teflon hermetic caps, flushed with an N_2 atmosphere, and incubated at 37 ± 1 °C, until no further biogas production was detected (normally around 90 days). The assay bottles were periodically analyzed for both quantitative and qualitative determination of biogas production. Quantitative biogas production was estimated by withdrawing the extra-pressure gas with a 120-ml syringe. The biogas production of blank control batches was subtracted from the biogas production of each sample. Qualitative characterization of biogas was performed using a gas chromatograph (Carlo Erba Megaseries 5300, capillary column 25 m × 0.32 mm diameter, and flame ionization detector [FID]). All the tests were performed in duplicate. The absence of inhibition phenomena allowed reaching the optimal production of methane (70 ± 4% v/v), starting from the first day of the test. The *ABP* experimental data were reported as normalized values (T = 273 K, P = 1 atm).