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Screening biological methods for laboratory scale stabilization of fine fraction from landfill mining

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ABSTRACT

Increasing interest for the landfill mining and the amount of fine fraction (FF) in landfills (40–70% (w/w) of landfill content) mean that sustainable treatment and utilization methods for FF are needed. For this study FF (<20 mm) was mined from a municipal solid waste (MSW) landfill operated from 1967 to 1989. FF, which resembles soil, was stabilized in laboratory scale reactors in two phases: first, anaerobically for 101 days and second, for 72 days using four different methods: anaerobic with the addition of moisture (water) or inoculum (sewage sludge) and aerobic with continuous water washing, with, or without, bulking material. The aim was to evaluate the effect on the stability of mined FF, which has been rarely reported, and to study the quality and quantity of gas and leachate produced during the stabilization experiment. The study showed that aerobic treatment reduced respiration activity (final values 0.9–1.1 mg O_2/g TS) and residual methane potential (1.1 L CH_4/kg TS) better than anaerobic methods (1.8–2.3 mg O_2/g TS and 1.3–2.4 L CH_4/kg TS, respectively). Bulking material mixed in FF in one aerobic reactor had no effect on the stability of FF. The benefit of anaerobic treatment was the production of methane, which could be utilized as energy. Even though the inoculum addition increased methane production from FF about 30%, but the methane production was still relatively low (in total 1.5–1.7 L CH_4/kg TS). Continuous water washing was essential to remove leachable organic matter and soluble nutrients from FF, while increasing the volume of leachate collected. In the aerobic treatment, nitrogen was oxidized into nitrite and nitrate and then washed out in the leachate. Both anaerobic and aerobic methods could be used for FF stabilization. The use of FF, in landscaping for example, is possible because its nutrient content (4 g N/kg TS and 1 g P/kg TS) can increase the nutrient content of soil, but this may have limitations due to the possible presence of heavy metal and other contaminants.

1. Introduction

Waste has been disposed of in landfills for decades, and, as a result, there are hundreds of thousands of landfills around the world. In Europe, there are an estimated 150000–500000 active and closed landfills (Hogland et al., 2010). In the USA, there were nearly 6500 active landfills until 1991 when requirements became stricter, after which small landfills were closed, leaving 1767 active landfills (Themelis and Ulloa, 2007). In many developing countries, waste is most commonly disposed of in open, uncontrolled dumpsites (Eawag, 2008), the number and content of which have been poorly documented, but the 50 largest dumpsites contain 258–368 million tons of waste (Waste Atlas Partnership, 2014). Landfills are sources of air, soil, and water pollution, and they contributed approximately 20% of anthropogenic methane emissions in 2013 (EEA, 2015).

Municipal solid waste (MSW) landfills contain biowaste, plastics, paper, cardboard, metals, glass, and various industrial wastes. Landfill materials may turn out to be valuable material and energy resources, which could be mined (meaning excavation and processing of landfilled waste) for further use (Krook et al., 2012). In addition, material recovery landfills may also be mined to provide space for additional landfilling, for other purposes, or to reduce local and global emissions. In general, landfills filled after the 1960s and before the beginning of increasing source separation practices (in Europe since the 1990s and 2000s) are considered the most suitable for landfill mining as they may contain a higher percentage of reusable materials (metals and materials with high energy content) (Eurostat, 2016; Hermann et al., 2014). Studies on landfills have shown that landfills also contain fine fraction (FF), which has been found (<10–25.4 mm) to make up 40–70% (w/w) of landfills' content (Hull et al., 2005; Kaartinen et al.,

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2013; Quaghebeur et al., 2013). FF can best be described as a soil-like material containing less than 10% (w/w) of other materials (Kaartinen et al., 2013). Especially in large landfills (>100 000 m^3), considered more cost-effective for landfill mining (Hermann et al., 2014), there is a need for feasible FF treatment and utilization methods, which, so far, have rarely been reported.

FF has a low organic content and low heating value (Kaartinen et al., 2013; Mönkäre et al., 2016; Quaghebeur et al., 2013); thus, its utilization as an energy resource might not be profitable. For other utilization purposes, such as soil improvement for example in landscaping (Rong et al., 2015; Zhou et al., 2015), construction material (Quaghebeur et al., 2013), or final disposal, FF may need processing to ensure stability to minimize emissions, so that FF presents no risk to human health or the environment. Stabilization can be conducted before and during landfill mining for example with aerobic stabilization (Raga et al., 2015) or after landfill mining separately for FF. As the biological stability of organic waste material indicates the material's emission potential and environmental impact, it has been proposed that stability measurements should be based on a combination of parameters, such as methane potential, respiration activity, and leachate quality (Ritzkowski et al., 2006). The use of such multiple parameters for stability assessment has been proposed because in some cases biological activity may be inhibited by toxic compounds (Cossu and Raga, 2008). However, the point of biological stability based on any different parameters has not been clearly defined (Kelly et al., 2006; Valencia et al., 2009). The biological stability of organic waste material can be increased by biological anaerobic and aerobic conversion processes, which in turn are affected by material characteristics, process conditions or by pretreatment of the material. In the anaerobic process, methane is produced which can be used for energy production. Methane yields and methane production rates are affected e.g. by the material composition and the moisture content of the material/process (Šan and Onay, 2001) or by the supply of inoculum, for example in the form of digestated sewage sludge to increase microbial activity as studied with fresh MSW (Mali et al., 2012). While aerobic stabilization of organic material requires energy-consuming aeration, it has been found to be a more rapid stabilization method than anaerobic stabilization e.g. for landfilled waste in bioreactor applications as reviewed by Reinhart et al., 2002. In aerobic processes, moisture is evaporated (Reinhart et al., 2002), and addition of water needs to be considered to prevent the material from drying out. Bulking material can be mixed with material to increase porosity and thus the gas mobility within the material e.g. in oil-contaminated soil remediation resulting in reduced remediation time (Rhykerd et al., 1999). In both anaerobic and aerobic methods, stabilization generates gaseous emissions and leachates, which need to be controlled and treated.

To the best of our knowledge, the stabilization of FF mined from landfills is rarely reported (Mönkäre et al., 2015; Raga and Cossu, 2013), while several studies on stabilization of fresh or landfilled MSW (not sieved) have been reported (e.g. Mali et al., 2012; Šan and Onay, 2001; Sponza and Ağdağ, 2004). Thus, the objective of this study was to evaluate the feasibility of four biological stabilization methods for FF mined from a landfill which operated for 22 years (1967–1989) and closed 25 years before landfill mining. Biological stabilization was studied using laboratory scale leach bed reactors (LBR) in two anaerobic reactors, with the initial addition of water or sludge, and in two aerobic reactors with and without bulking material and with continuous water addition. Methods were chosen to enhance biological stabilization, so that the more stable FF is, the wider uses it may have. The exact composition of the landfilled material in the studied landfill was unknown before the landfill mining and the initial organic content of the studied FF was found to be low (15.8 ± 5.0% of TS). Stabilization of FF was evaluated by measuring organic content, biological activity, and nutrient content before and after the stabilization experiment. The quality and quantity of gas and leachate formed were examined during the experiment.

2. Materials and methods

2.1. Fine fraction

The FF studied was mined from the Lohja landfill in Southern Finland in June of 2013. The landfill operated from 1967 to 1989, and it reportedly received unsorted MSW, industrial waste, construction waste and soil, but the composition, volume, and placement of the various waste fractions were not known or documented. The landfill area is ca 5 ha with a depth 15 m, and it has no bottom structure. The landfill is sealed with a top cover of 2 m of soil. For this study, samples were taken from the four different points (distances 25–45 m) by drilling wells with a Casagrande B 170 hydraulic piling rig to a depth of 10–13 m (details in Mönkäre et al., 2016). Samples from the three wells, with depths of 2–10 m, formed three samples, while one well was divided into three layers (2–5 m, 5–9 m, 9–13 m) to form a total of six samples. The mined landfill material (ca 600 L/sample) was manually sieved on location and the FF (<20 mm) in the six samples ranged from 39.8 to 73.6% (w/w) of the total mined landfill material. Total solids (TS) content was 59.6–81.6% and volatile solids (VS) content was 6.0–24.0% of TS (Mönkäre et al., 2016). The six samples were mixed to make one landfill FF sample that was used in this study. TS of the mixed sample was 66.4 ± 7.3% and VS/TS ratio was 15.8 ± 5.0%.

2.2. Stabilization experiment

Stabilization of FF was performed in four acryl LBRs (Fig. 1) with a height of 600 mm and diameter of 150 mm. On the bottom of each LBR was a gravel layer separated from the FF with mesh to prevent the FF from flushing out. On the bottom of reactors was a porthole used to collect leachate into 1 L bottles. The top portholes (two in each LBR) were used to inject air and/or water by pumps into the reactors and to collect gas (formed gas) from the reactors into aluminum gas bags kept at a room temperature of 20 ± 1 °C.

First, 5.9 kg FF (3.9 kg TS, 0.6 kg VS) was added to all four LBRs. All four parallel LBRs were operated anaerobically, without water addition or leachate collection, for 101 days (phase 1). Operation temperature was 35 °C to enhance bioprocess. Subsequently, all LBRs were opened, and FF was collected from all LBRs and sampled for TS and VS. Then, FF samples from the four LBRs were manually mixed and redistributed to four LBRs (5.5 kg in each, TS 66% and VS/TS 15.9%; phase 2). In phase 2, two of the LBRs were operated anaerobically and two aerobically at 35 °C (Fig. 1). In one anaerobic LBR, 800 mL of tap water (Anaerobic-LBR) was added at the beginning of the second phase (day 102), while, in the other (Sludge-LBR), 800 mL of digested sewage sludge from a wastewater treatment plant in Viinikanlahti, Tampere, Finland (TS 2.8% and VS/TS 55%.) was added as inoculum to increase microbial activity at a ratio 0.02 g VS in sludge per VS in FF. Based on methane potential assays (data not shown) 800 mL of studied sewage sludge was estimated to produce methane approximately 0.5 L during the 72 days. In one aerobic LBR (Bulking-LBR), 450 g bulking material (wood chips, TS 82% and VS/TS 83%) was added to increase air mobility, while the other LBR (Aeration-LBR) contained FF only. Both aerobic LBRs were aerated with indoor air (0.1 L/min or 2.5 L/h/kg TS). Tap water was pumped into both aerobic LBRs at a flow rate of 0.023 L/h for 1 h six times a day, resulting in a total of 1 L/week. Leachate was collected into 1 L glass bottles at 20 °C for a one week period

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