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Department of Geography

Bachelor thesis in Environmental technology

BIOGAS PRODUCTION UNDER CO-DIGESTION OF FOOD WASTE WITH SEWAGE SLUDGE

Kärt Kanger

Supervisors: Professor Jaak Truu, PhD (University of Tartu)

Christian Ebner, PhD (University of Innsbruck)

Hiie Nõlvak, PhD (University of Tartu)

Accepted for defence .....

Supervisor .....

Head of the Department .....

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Table of contents

List of abbreviations

AMPTS II – automatic methane potential test system

BMP_{15} – biological methane potential

CCHP – combined cooling, heating and power

CHP – combined heat and power

COD – chemical oxygen demand

EC – electric conductivity

EMAS – eco-management and audit scheme

GP_{21} – biogas potential

MCC – microcrystalline cellulose

N – total nitrogen content

P – total phosphorus content

SM – sludge mixture

TS – total solids content

VS – volatile solids content

WWTP – wastewater treatment plant

Introduction

The global growth in energy demand has induced active search for alternative energy sources. Renewable sources, such as biomass, have been under constant examination. A more efficient deployment of renewable energy sources will facilitate a reduction in greenhouse gas emissions and air pollution (Poeschl et al 2010). Therefore, renewable energies offer an environmentally sound alternative to fossil fuels and account for a lesser contribution to climate change.

Biomass represents a sustainable source of renewable energy. It is characterised by its abundance and offers a secure energy supply (Weiland 2010). Several organic substances have been used for anaerobic digestion. Anaerobic digestion of biomass is a multi-stage microbial process, which produces biogas and digestion residues as the final products. Biogas is an energy-rich mixture of primarily methane and carbon dioxide and can be used for energetic purposes. Digestion residues are characterised by high nutrient content and can be efficiently applied for soil fertilisation (Weiland 2010).

The production of biogas has been evaluated as one of the most energy-efficient and environmentally beneficial technology for bioenergy production (Fehrenbach et al 2008). Typical substrates for anaerobic digestion include animal manure, sewage sludge from wastewater treatment and energy crops (Weiland 2010). Co-digestion of several substrates increases biogas yield and improves process efficiency (Iacovidou et al 2012). Additionally, the utilisation of organic waste as a substrate for biogas production accounts for waste stabilisation and a reduced amount of landfilled waste.

The following thesis gives an overview of anaerobic digestion and biogas production. The study aimed at evaluating biogas production under co-digestion of food waste with sewage sludge in lab-scale reactor systems. The specific aims included substrate and digestate characterisation, detection of biogas potential of the substrates and biological methane potential measurement. The experiments were conducted at the Institute of Microbiology of the University of Innsbruck (*Leopold-Franzens-Universität Innsbruck,* Tirol, Austria) as well as in the wastewater treatment plant of Zirl (*Abwasserverband Zirl und Umgebung*, Tirol, Austria) in May-June 2012.

1. Literature review

1.1. Essence of biogas

Biogas is a mixture of methane (CH_4), carbon dioxide (CO_2) and a small amount of trace gases, which is produced by the microbial degradation of organic substances under anaerobic conditions (Tretter 2002). Anaerobic degradation occurs naturally in oxygen deficient habitats, such as sediments, water-logged soils, and intestinal tracts (Insam et al 2010). However, the same microbial process, referred to as anaerobic digestion, accounts for the formation of landfill gas and is widely used for the commercial production of biogas in modern biogas plants.

The composition of biogas varies depending on the degradable substrate as well as process conditions e.g. temperature (Al Seadi et al 2008). Typically, the methane content accounts for 50–75 vol% of biogas, followed by a carbon dioxide content of 20–45 vol%. Methane is the energy carrier of biogas, therefore a high methane content rather than CO_2 content is desirable for energy production. The concentration of water vapour varies from 2–7 vol% depending on the temperature of digestion. Biogas may also include traces of nitrogen (N_2), ammonia (NH_3), oxygen (O_2), hydrogen (H_2) and hydrogen sulphide (H_2S), the latter one acting corrosively to metals. A composition of biogas as suggested by Al Seadi et al (2008) is given in Table 1.

Table 1. Composition of biogas (Al Seadi et al 2008).

Compound	Chemical symbol	Content (vol%)
Methane	CH_4	50–75
Carbon dioxide	CO_2	25–45
Water vapour	H_2O	2–7
Oxygen	O_2	< 2
Nitrogen	N_2	< 2
Ammonia	NH_3	< 1
Hydrogen	H_2	< 1
Hydrogen sulphide	H_2S	< 1

The methane content of biogas is determined by the biochemical composition of the degradation substrate. Raw protein has the highest theoretical methane yield (70–71 vol%),

similarly, raw fat is characterised by a high theoretical methane yield (67–68 vol%). However, raw fat shows a significantly greater theoretical biogas yield than raw protein: 1200 Nm^3/t-TS compared to 700 Nm^3/t-TS, respectively (Baserga 1998). Considering a standard methane content of 50 vol%, the energetic value of biogas is 21 MJ/Nm^3 (Al Seadi et al 2008).

1.2. Biochemical process of anaerobic digestion

Anaerobic digestion of organic matter is a four-phase process accomplished by the co-operation of several microbial groups (Insam et al 2010). The four stages include depolymerisation of organic substances (hydrolysis), acidogenesis, acetogenesis and methanogenesis (Weiland 2010), which produce biogas and digestate as the final products of anaerobic digestion (Figure 1). A close co-operation between different microbial groups is essential for the vitality of the microbes due to the lower energy yield of the anoxic degradation of organic matter as compared to the thermodynamically more favourable aerobic degradation in oxygen-rich environments (Schink 1997).

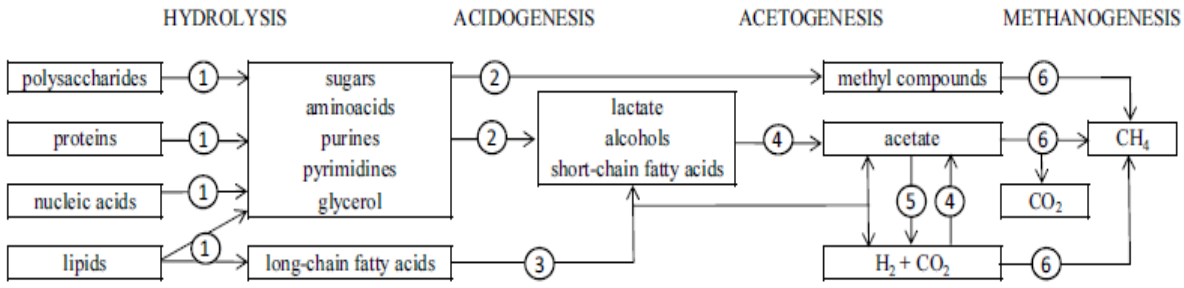

Figure 1. Stages of anaerobic digestion of the organic matter: 1) hydrolysis of biopolymers, 2) acidogenesis, 3) β-oxidation of long-chain fatty acids, 4) acetogenesis, 5) acetate oxidation, 6) methanogenesis. Modified from Insam et al (2010).

Over 80% of the total microbial diversity in anaerobic bioreactors contributing to the degradation of organic matter are bacteria (Krause et al 2008), including *Firmicutes* and *Bacteroidetes* as the dominant phyla (Zakrzewski et al 2012). Archaeal representatives commonly belong to the phylum *Euryarchaeota*, which includes all known methanogens (Insam et al 2010). Less attention has been paid to anaerobic eukaryotes contributing to the anaerobic digestion, such as fungi and protozoa (Insam et al 2010), and a significant amount

of microbial diversity in biogas reactors still remains unknown to scientists (St-Pierre and Wright 2013, Zakrzewski et al 2012).

Microbial groups contributing to the anaerobic digestion are strongly linked: the hydrolytic microbes usually coincide with acidogenic bacteria, while acetogens are often found in syntrophic relations with methanogens. For a stable degradation process, the former and the latter part of digestion must run in equilibrium. If the hydrolytic stage runs too fast, the process is inhibited by the accumulation of acids and the concurring decrease in pH-value. In case of fast acetogenesis and methanogenesis, the hydrolysis becomes limiting to methane production (Weiland 2010). Other crucial parameters affecting the process are ammonia concentration, trace element supply, fermentation temperature, and retention time in bioreactor (Braun et al 2010).

1.2.1. Hydrolysis

The first stage of anaerobic digestion – hydrolysis – accounts for the depolymerisation of biopolymers (polysaccharides, lipids, proteins, and nucleic acids) (Figure 1). A complex hydrolytic microbial community produces extracellular hydrolytic enzymes, which degrade biopolymers into their monomers (sugars, long-chain fatty acids, glycerol, amino acids, purines, pyrimidines). Bacterial as well as fungal cellulases, xylanases, proteases, amylases and lipases are the key enzymes of hydrolysis, which is often the rate limiting step of anaerobic digestion (Insam et al 2010).

Hydrolytic bacteria in biogas reactors are characterised by a diverse community, reflecting on a wide range of acceptable substrates (Insam et al 2010). The dominant phylum is *Firmicutes*, mainly represented by the genus *Clostridium* (Krause et al 2008). Other common examples of hydrolytic anaerobes found in biogas reactors include genera *Acetivibrio, Bacteroides, Selenomonas,* and *Ruminococcus* (Insam et al 2010). Most of the hydrolytic bacteria are strict anaerobes, however, facultative anaerobes, such as *Streptococci* and *Enterobacteriaceae,* have also been detected (Weiland 2010). Hydrolytic fungi are generally less abundant than bacteria, the anaerobic fungus *Neocallimastix* has been reported to contribute to the hydrolysis of organic matter (Insam et al 2010). Typically, the organisms performing hydrolysis also assimilate the resulting monomers and ferment them during the next stages of anaerobic digestion (Insam et al 2010).

1.2.2. Acidogenesis

The products of hydrolysis are microbially transformed into alcohols, volatile fatty acids, carbon dioxide and molecular hydrogen via fermentation or anaerobic oxidation if electron acceptors, such as nitrates or sulphates, are present (Figure 1). For biogas production, fermentation pathway is more desirable as it yields substrates for methanogenesis (acetate, formate, H_2, CO_2). Due to the acidic products (e.g. propionic acid) the second stage of anaerobic digestion is referred to as acidogenesis (Insam et al 2010).

Fermentative bacteria typically coincide with hydrolytic bacteria. Genera *Clostridium, Lactobacillus, Selenomonas,* and *Enterobacter* provide representatives of fermentative prokaryotes (Insam et al 2010). Fermentation products are excreted out of the cell, lowering the pH-value in the environment. The decrease in the pH-value is addressed as the most common reason for reactor failure. Therefore, equilibrium of acidogenic and acid scavenging microbes is crucial for the stable digestion process (Insam et al 2010).

1.2.3. Acetogenesis

The products of acidogenesis are further oxidised to acetate, formate, molecular hydrogen and carbon dioxide by acetogenic bacteria, producing direct substrates for methanogens (Figure 1). Acetogens mainly belong to the phylum *Firmicutes*, including typical acetogenic bacteria *Acetobacterium woodii* and *Clostridium aceticum* (Weiland 2010). The accumulation of H_2 as a product of acetogenesis, inhibits acetogens. Therefore, the maintaining of a low partial pressure of molecular hydrogen is essential for efficient anaerobic digestion (Weiland 2010).

A low partial pressure of molecular hydrogen is mainly accomplished by the syntrophic associations between H_2-producing acetogens and H_2-scavenging methanogens. These two microbial groups co-operate to perform a thermodynamically unfavourable reaction with a net energy gain due to syntrophy. This co-operation is a measure for optimal exploitation of the limited energy yield in anaerobic conditions (Schink 1997).

1.2.4. Methanogenesis

The last stage of anaerobic digestion is methanogenesis, which delivers the energy carrier of biogas – methane (CH_4). All known methanogens are archaea from the phylum *Euryarchaeota,* the orders *Methanobacteriales, Methanococcales, Methanomicrobiales,* and *Methanosarcinales* are commonly found in biogas reactors (Insam et al 2010). Methanogenesis is the rate limiting step in waste digestion due to a possible inhibition by the accumulation of ammonia (Braun et al 2010).

Methane can be produced from methyl compounds, acetate or by the reduction of carbon dioxide with molecular hydrogen. Autotrophic methanogens (e.g. *Methanoculleus* sp.) utilise CO_2 and H_2, while heterotrophic methanogens (e.g. *Methanomethylovorans*) use acetate, formate and a few other compounds, which accounts for approximately two thirds of all methane produced (Plugge et al 2010). Only a few acetoclastic methanogens have been identified, namely genera *Methanosarcina* and *Methanosaeta,* whereas all methanogens are able to use hydrogen and carbon dioxide for the formation of methane (Weiland 2010).

1.2.5. Factors influencing anaerobic digestion

Anaerobic digestion is a complex multi-stage process, which is sensitive to a wide range of factors. Process failures may result from technical as well as biochemical issues. Commonly reported technical problems include insufficient mixing in bioreactors, caused by inappropriate particle size or high viscosity of the substrate (Braun et al 2010). Impurities in the substrate, such as glass, plastics, and metals found in organic waste, account for mechanical problems as well as inhibition of the microbial digestion (Iacovidou et al 2012). Additionally, temperature changes as well as retention time in bioreactor influence process efficiency (Braun et al 2010).

The microbial conversion of organic matter into biogas and digestion residues is also dependent on the physico-chemical conditions in the bioreactor. Frequently reported process failures resulted from a decrease in pH-value due to the accumulation of volatile fatty acids, from the inhibitory effects of ammonia as well as H_2S, from the insufficient amount of nutrients and trace elements, and from the possible toxicity of impurities in the substrate

(Braun et al 2010). A balance between acidogenic and methanogenic microbes is essential for the stability of anaerobic digestion (Braun et al 2010).

1.3. Potential substrates for biogas production

Biogas can be produced from various types of biomass containing carbohydrates, proteins, fats, cellulose, and hemicellulose as the main components. Lignin rich substrates (e.g. wood) are unsuitable for biogas production due to their slow degradation rate in anaerobic conditions. Due to the diversity of substrates as well as variable process parameters and retention time in the bioreactor, the chemical composition and yield of biogas are subject to variations (Weiland 2010).

Historically, animal manure and sewage sludge from wastewater treatment have been used for biogas production (Weiland 2010). In contemporary bioreactors common feedstock includes manure from pigs, cattle, and chicken together with a co-substrate (Tretter 2002), which delivers a higher gas yield. Common co-substrates include energy crops such as maize, forage beet, clover, harvest residues, agricultural wastes of animal as well as vegetable origin, municipal organic waste from households, and food waste (Al Seadi et al 2008).

Table 2. Biogas yield and average methane content of different organic substrates (Normak et al 2009). VS stands for volatile solids content.

Substrate	Biogas yield (l/kg-VS)	Average CH_4 content (%)
Cattle slurry	200–500	60
Pig slurry	300–700	60–70
Municipal organic waste	150–500	58–65
Maize silage	450–700	50–55

The quantity as well as quality of biogas is strongly affected by the substrates used for anaerobic digestion. A wide range of studies have analysed the biochemical methane potential of different substrates, providing numerical data. Normak et al (2009) have summarised the biogas yield and its average methane content for different organic materials as indicated in Table 2. Luna del Risco et al (2011) studied different Estonian substrates to assess their biochemical methane potential. The results revealed herbal biomass (silages, hay) and agro-industrial residues as promising substrates for biogas production, the highest methane

potential was detected for milk wastes 458–714 l/kg-VS (volatile solids) (Luna del Risco et al 2011).

1.3.1. Sewage sludge co-digestion with food waste

In this study sewage sludge co-digestion with food waste was analysed. This combination of substrates accounts for several sustainable solutions e.g. sewage sludge stabilisation and a reduction of landfilled organic wastes (Iacovidou et al 2012). Sewage sludge is produced in municipal wastewater treatment plants, while food waste may originate from various catering institutions as well as households. Food waste is a desirable substrate for anaerobic digestion characterised by a high variability, which accrues from the origin and preparation of food (Zhang et al 2007). The reported methane yield of food waste varies from 245–525 l/kg-VS (Raposo et al 2011), while the methane yield of sewage sludge ranges from 116–318 l/kg-VS (Iacovidou et al 2012).

Sewage sludge co-digestion with food waste accounts for increased methane production compared to the mono-fermentation of sewage sludge. Synergical effects have also been reported in full-scale experiments in operating biogas plants (la Cour Jansen et al 2004). Co-substrates must be dosed in optimal proportions depending on the specific characteristics of the substrates. The addition of easily degradable organic material accelerates the hydrolosis of sewage sludge and results in a higher methanogenic potential. On the other hand, food waste may inhibit anaerobic digestion due to its variability, possible toxic substances, ammonia accumulation and acidification. Frequently, impurities such as plastic, metal and glass are found in collected food waste, which generate technical problems in the reactors (Iacovidou et al 2012).

Attention should be paid to the environmental impacts rising from the collection of food waste. Several different systems have been proposed, such as collection of household food waste in paper bags, a prior drying of the collected food waste, the use of kitchen grinders connected to settling tanks, and the use of vacuum system with subsequent central grinding (Bernstad and la Cour Jansen 2012). A comparative life cycle analysis of different collection systems showed that vacuum system results in the largest net avoidance of primary energy use, while disposal of food waste in paper bags for decentralised drying accounts for the

largest net avoidance of global warming, eutrophication and acidification (Bernstad and la Cour Jansen 2012).

1.4. Biogas and digestate utilisation

Biogas is produced in biogas plants, which employ various types of bioreactors and processing technologies. A common practice is wet fermentation in vertical continuously stirred tank reactors but horizontal digesters and dry fermentation have also been applied (Weiland 2010). The products of anaerobic digestion – digestate and biogas – have to be collected and stored in specialised facilities. For biogas storage, safety regulations, such as explosion control and safety zones, must be followed due to the flammable nature of biogas (Normak et al 2012).

The utilisation of biogas is preceded by desulphurisation and drying of collected gas. For a successful avoidance of corrosion, desulphurisation to a maximal level of 250 ppm H_2S is necessary, which is often achieved by biological means (Weiland 2010). The drying of biogas is a result of water vapour condensation and elimination in pipework due to decreased temperature as compared to the temperature in bioreactor (Normak et al 2009).

Desulphurised and dried biogas is commonly used for combined heat and power (CHP), electricity generation and upgrading to biomethane. The dominant utilisation method is CHP using gas or dual fuel engines, which allow efficiencies up to 43% (Weiland 2010). Alternatively, micro gas turbines, stirling engines, and fuel cells have been tested at pilot scale (Normak et al 2009). Biogas plant efficiency can be enhanced by combined cooling, heating and power units (CCHP) which account for seasonal variations in thermal loads (Poeschl et al 2010). The most energy efficient solution for the utilisation of biogas is upgrading it to biomethane, which can be injected into public gas grid and used as transport fuel (Poeschl et al 2010). However, the high cost of the upgrading technology restricts its deployment and the utilisation as vehicle fuel is additionally inhibited by the poor infrastructure of gas stations (Poeschl et al 2010).

The other product of anaerobic digestion – the digestate – is predominantly used as agricultural fertiliser. The digestion process results in a mineralisation of organic nutrients, reduction of odours, enhanced flow properties, and a potential inactivation of weed seeds,

bacteria, viruses, fungi, and parasites depending on the process temperature and retention time (Weiland 2010). Alternatively, the digestate may be separated into solid and liquid fraction: the former is suitable as fertiliser as well as for energy production by incineration, the latter may be treated in wastewater treatment plants (Poeschl et al 2010).

1.5. Future perspectives of biogas production

Biogas production by anaerobic digestion has a multipurpose value, including improvements in environmental, agricultural, sanitary and waste reduction aspects besides supplying energy (Holm-Nielsen et al 2009). It is a significant contributor to a better utilisation of renewable energies and therefore highly valued in Europe and of increasing interest in many parts of the world. In the context of limited fossil fuel resources and tightening environmental policies, anaerobic digestion is a fast-growing market (Weiland 2010).

However, the expanded utilisation of biogas has to be accompanied by further improvements of the process efficiency, management and infrastructure. A more sustainable feedstock supply enhances the economic security of biogas plants, while more attention should be directed to a larger diversity of the substrates and their pre-treatment (Poeschl et al 2010). Possible new feedstock types include bio-slurries from biofuels processing industries and organic wastes from pharmaceutical industries (Holm-Nielsen et al 2009). A better process control and improved online measurements will account for the optimisation of anaerobic digestion and increase the biogas yield (Holm-Nielsen et al 2009). A more detailed analysis of microbial population helps to provide process stability and higher efficiency (Weiland 2010). Furthermore, socio-economic issues such as utilisation of locally available resources and job creation should be addressed to expand the sustainable production of biogas (Poeschl et al 2010).

2. Materials and methods

2.1. Feedstock

The study aimed at evaluating biogas potential of food waste under co-digestion with sewage sludge. The samples were collected in Tirol, Austria in May 2012. The sewage sludge samples together with the microbial inoculum were provided by the wastewater treatment plant of Zirl (*Abwasserverband Zirl und Umgebung*), whereas food waste was collected from the state hospital of Innsbruck (*Landeskrankenhaus Innsbruck – Universitätskliniken*) by an automatic vacuum system.

2.1.1. Sewage sludge of the wastewater treatment plant Zirl

The activated sludge treatment system in wastewater treatment plant (WWTP) Zirl, operating since 1996, processes wastewater from 14 communities nearby Zirl, which account for 42 000 population equivalents. 60% of the organic load in the wastewater inflow derives from the households, 40% derives from the industry (Häusler et al 2010). In 2005 the plant was expanded by a sludge treatment facility for the anaerobic stabilisation of wastewater treatment products. The anaerobic biogas reactor accompanied by a combined heat and power station (CHP) initially digested sewage sludge solely. However, in 2008 co-digestion of variable substrates was introduced, which significantly increased energy production. WWTP Zirl follows the EMAS regulations for eco-management and audit scheme (Häusler et al 2010).

WWTP Zirl cleans 6 000 to 20 000 m^3 water daily and the process consists of several stages. In the mechanical stage wastewater passes through a screen, grit chamber, and grease trap. This is followed by primary sedimentation and biological treatment with nitrification and pre-denitrification in two parallel activated sludge lines. As the last stage, secondary settlement tanks are used to remove all sedimentary material from the water. Process products are partly reused (return activated sludge), partly digested anaerobically for biogas production (grease, primary sludge, excess activated sludge). The outflow is regularly controlled by analytical determination of relevant wastewater components and discharged to the river Inn.

Biogas is produced in a 1350 m^3 mesophilic anaerobic reactor and used for combined heat and power production on spot. The digestate is collected for deposition due to local regulations which prohibit agricultural utilisation of the digestate (Tiroler Feldschutzgesetz 2000). In 2012 a total of 712 570 Nm^3 biogas was produced, which accounted for 1074 MWh electricity (personal communications from Christian Ebner, head of the laboratory in WWTP Zirl). Since 2009 the energy production in WWTP Zirl has exceeded the consumption of the plant, which enables electricity sale and heat transfer to the neighbouring facilities (Häusler et al 2010).

In the present study primary activated sludge and excess activated sludge from WWTP Zirl were used as digestion substrates. Inoculum was collected from the anaerobic reactor in Zirl during regular control sampling from the pipeline, and degassed for 11 days at 36ºC as suggested by Angelidaki et al (2009).

2.1.2. Food waste of the state hospital of Innsbruck

Food waste analysed in the present study was collected from the state hospital of Innsbruck (*Landeskrankenhaus Innsbruck – Universitätskliniken*). The hospital serves 54 000 meals weekly, which produces 13,4 tons of food waste (personal communications from Kornelia Giersig, the head of the *Abfall- und Gefahrgutbeauftragte* department of the state hospital of Innsbruck). The food waste in the hospital is collected by an automatic vacuum collection system "WasteStar" (MEIKO, Offenburg, Germany), which was implemented in full-scale in February 2012 (personal communications from Cornelia Giersig).

The food waste in the hospital originates from several sources, which are described in detail in Appendix 1. All waste is collected to a closed primary tank (1 bar underpressure, 1500 l), where it is homogenised and mixed. After size reduction in the primary tank, the waste is directed to a closed storage tank (25ºC, 14 m^3), which is emptied twice a week by a liquid waste collecting car and transported to the wastewater treatment plant, where it is used as a co-substrate for anaerobic digestion (personal communications from Cornelia Giersig).

In the present study food waste from three different sampling points was used (Bio1 from 05.04.2012; Bio2 from 16.04.2012; Bio3 from 23.04.2012). The samples were taken from the tank of the liquid waste collecting car, which delivered food waste to the WWTP. The initial material was homogenised in a mixer for 20 seconds to remove bigger fractions, such as bread

crumbs, rice grains, maize beads etc. A size reduction and the resulting enlargement of the available specific surface account for an improved biological digestion of the substrate (Raposo et al 2011).

2.2. Substrate characterisation

To assess the biogas potential of the substrates, sewage sludge as well as food waste samples were characterised in terms of chemical oxygen demand (COD), total solids content (TS), volatile solids content (VS), total nitrogen and total phosphorus concentrations, as well as electric conductivity (EC), and the pH-value. All measurements were conducted in triplicate, statistical outliers deviating more than 20% from the average were excluded from the results.

COD was determined with *NANOCOLOR*® tube test "COD 1500" using the *NANOCOLOR*® UV-Vis spectrophotometer from MACHEREY-NAGEL (Düren, Germany). The substrates were diluted with destilled water and homogenised with MICCRA D-8 homogeniser (*ART Prozess- & Labortechnik*, Müllheim, Germany) to achieve a proper consistence for the tube test. The same protocol was applied for the detection of total nitrogen content and total phosphorus content using *NANOCOLOR*® tube tests "*total* Nitrogen TN_b 220" and "*ortho*- and *total*-Phosphate 15", respectively. Additionally, chloride concentration in food waste samples was measured using *NANOCOLOR*® tube test "Chloride 200" to detect possible inhibitory factors. All measurements were conducted according to the manufacturer's manual.

The measurement of TS and VS together with the following calculations were conducted according to the protocols described by Kroiss (2007). Total solids content (TS) was determined by drying the sample overnight at 105°C until weight constancy. TS was expressed in weight percentage according to the formula (1),

(1) $$TS\ [\%] = \frac{dried\ sample\ [g]}{original\ sample\ [g]} \times 100\%.$$

Volatile solids content (VS) was determined by an additional ignition of the dried samples in a muffle furnace at 550°C for two hours. VS was firstly expressed in percentage of the TS according to the formula (2) and subsequently in percentage of the total sample according to the formula (3),

(2) $$VS(1)\ [\%] = 100\% - \frac{burned\ sample\ [g]}{dried\ sample\ [g]} \times 100\%,$$

(3) $$VS\ [\%] = \frac{TS \times VS(1)}{100\%}.$$

The pH-value and electric conductivity (EC) were measured with calibrated pH/dissolved oxygen/conductivity measuring instrument WTW Multi 340i (WTW *Wissenschaftlich-Technische Werkstätten*, Weilheim, Germany).

2.3. Gas production

The biogas potential of the substrates was assessed in a 21-day fermentation experiment using liquid displacement system. The experiment followed the guidelines recommended by *Verein Deutscher Ingenieure* (VDI 2006). Additionally, a 15-day biological methane potential detection was conducted using automatic methane potential test system AMPTS II (Bioprocess Control, Lund, Sweden). The substrates were used in various mixtures to evaluate the gas production potential under mono- as well as co-digestion.

2.3.1. Experimental lines

The biogas production experiment consisted of 7 experimental lines, each conducted in triplicate. The experimental lines included:

- Negative control,
- Positive control with microcrystalline cellulose (MCC),
- Sludge mixture from primary and excess activated sludge (SM),
- Food waste 3 (Bio3),
- Food waste 1 with sludge mixture (Bio1+SM),
- Food waste 2 with sludge mixture (Bio2+SM),
- Food waste 3 with sludge mixture (Bio3+SM).

The composition of the fermenters comprised of degassed inoculum and the substrate. For negative control, inoculum was used alone; for positive control, microcrystalline cellulose was added to the inoculum as suggested by Angelidaki et al (2009). The substrates were used in various mixtures as described below.

2.3.2. Substrate mixtures

The composition of the substrate mixtures was defined according to the working conditions of the full-scale anaerobic reactor in WWTP Zirl. Based on the data from 2011, primary activated sludge, excess activated sludge and the co-substrate were dosed in equal amounts of chemical oxygen demand (COD) in WWTP Zirl. This principle was taken as the basis for the batch experiment in the current study. The substrates were used in concentrations similar to the operating conditions of the full-scale plant considering COD per reactor volume (kg*COD/m^3).

A mixture of primary and excess activated sludge was prepared on the principle of equal COD concentrations as seen in Table 3. The final COD concentration in the sludge mixture (SM) was 62 484 mg-O_2/l, 50% of it derived from the primary activated sludge and 50% of it derived from the excess activated sludge.

Table 3. Composition of the sludge mixture (SM).

	Average COD (mg-O_2/l)	**Volume taken for SM (ml)**
Primary activated sludge	69 533	81,59
Excess activated sludge	56 733	100

Mixtures from the SM and food waste were prepared in a way that 2/3 of the final COD concentration derived from the SM (1/3 from the primary activated sludge and 1/3 from the excess activated sludge) and 1/3 of the final COD concentration derived from a food waste sample (Bio1, Bio2 or Bio3). The final characteristics of substrate mixtures considering chemical oxygen demand (COD), total solids content (TS), and volatile solids content (VS) are given in Table 4.

Table 4. Characteristics of the substrate mixtures.

Substrate	**COD (mg-O_2/l)**	**TS (%)**	**VS (%)**
Bio1+SM	78 962	6,68	5,51
Bio2+SM	78 948	6,28	5,11
Bio3+SM	79 205	6,59	5,40

2.3.3. Composition of the reactors

Each reactor used for liquid displacement system was filled with 500 ml (equals to 500 g) degassed inoculum with the following characteristics: COD=34 710 mg-O_2/l, TS=3,34%, VS=2,01%, which resulted in the final amounts of COD=17 355 mg-O_2, TS=16,7 g and VS=10,05 g for inoculum in the reactors.

Negative control consisted of 500 g pure inoculum, positive control consisted of 500 g inoculum and 1,67 g microcrystalline cellulose. The remaining experimental lines consisted of 500 g inoculum and 40 g substrate (various mixtures or Bio3). The exact composition of the reactors is given in Table 5.

Table 5. Composition of the reactors used for liquid displacement system.

Reactors	**Volume of inoculum (ml)**	**Amount of MCC (g)**	**Amount of substrate (g)**	**COD from substrate (mg-O_2)**	**TS from substrate (g)**	**VS from substrate (g)**
Negative control	500	0	0	0	0	0
MCC	500	1,67	0	Not determined	1,67	1,67
Bio1+SM	500	0	40	3158	2,69	2,2
Bio2+SM	500	0	40	3158	2,51	2,04
Bio3+SM	500	0	40	3168	2,64	2,16
SM	500	0	40	2499	2,12	1,64
Bio3	500	0	40	6816	5,43	4,99

The same composition of the reactors was later used for AMPTS II experiment. Due to the limited number of reactors available for automatic methane detection, negative control, positive control, SM and Bio3 were conducted as single determinations. The total reactor volume in AMPTS II accounted for 300 ml: 275,2 ml of inoculum and 24,8 ml of substrate were used. The $VS_{inoculum}/VS_{substrate}$ ratio in the reactors for liquid displacement system and AMPTS II was held constant to guarantee equal conditions for both methods.

2.3.4. Gas potential measurement with liquid displacement system

Over a 21-day biogas production experiment, the volume of produced biogas was measured daily using a liquid displacement system. The system was based on the eudiometer unit

described in the international standard ISO/DIS 14853 (ISO/DIS 14853 1999). An overview of the working mechanism of the eudiometer unit is given in Appendix 2.

Several studies have reported the importance of the barrier solution to avoid errors in the measurement of the produced biogas due to solubilisation of the gas in the barrier solution (Raposo et al 2011, Walker et al 2009). In the current study acidified saturated alkaline solution was used as suggested by Walker et al (2009). A detailed description of the barrier solution composition can be found in Appendix 4 of the German Landfill Act (Deponieverordnung 2009).

In addition to the volume of the produced biogas, the percentage of its main components was detected with Biogas Check BM 2000 Instrument (Geotechnical Instruments, Warwickshire, UK). The hydrogen sulphide concentration in the biogas was measured using H_2S detector tubes (*Dräger*, Lübeck, Germany). Concurrently to the daily measurements, the air pressure in the region was noted from the homepage of *Zentralanstalt für Meteorologie und Geodynamik* (www.zamg.ac.at). The room temperature was examined regularly and held constant at 36°C. The reactors were mixed manually on a regular every-day basis.

2.3.5. Methane potential measurement with AMPTS II

The methane potential of the substrates was measured using an automatic methane potential test system AMPTS II (Bioprocess Control, Lund, Sweden), which is described in more detail in Appendix 3. AMPTS II uses a real-time data recording system, which also allows the analysis of the results (Browne and Murphy 2013).

In the current study the reactors were incubated at 37°C and mixed automatically every four minutes. The detection of air pressure and the following calculations of methane potential were conducted automatically.

2.3.6. Data validation

The experimental data gathered from the biogas potential experiment in the liquid displacement system was corrected to standard conditions at 273 K temperature and 1013 mbar air pressure according to the formula (4),

(4) $$\mathrm{Vo} = \mathrm{V(measured)} \times \frac{[p(air) - p(water)] \times To}{po \times T},$$

where Vo stands for corrected gas volume in Nml, V(measured) stands for the measured gas volume in ml, p(air) is the air pressure of the detection time in mbar, p(water) is the water vapour pressure at the given temperature in mbar, To and po stand for the standard temperature and air pressure, respectively, and T is the temperature at the detection time in K (Deponieverordnung 2009). The correction of the gas measurement data in AMPTS II was conducted automatically.

Three replicates of each experimental line were characterised by an arithmetical average and standard deviation. Data points deviating for more than 20% from the average were excluded from the analysis.

The corrected average of daily production of each experimental line was added to gain a cumulative production value, from which the cumulative production of inoculum was subtracted. Thus, the results of negative control were not evaluated individually, but used for the evaluation of the remaining experimental lines. The cumulative results of the remaining experimental lines were divided by the volatile solids content from the substrate in the corresponding reactor as suggested in Deponieverordnung (2009). The biogas potential of the substrate was characterised by a specific production value GP_{21} in Nml/g-VS.

2.4. Digestate characterisation

The digestion residues from liquid displacement system were characterised in terms of chemical oxygen demand (COD), total solids content (TS), volatile solids content (VS), the pH-value and electric conductivity (EC). The measurements were conducted using the same test methods and protocols described previously for substrate analysis (Chapter 2.2).

3. Results and discussion

3.1. Characteristics of the substrates

Primary sludge, excess sludge and three food waste samples (Bio1, Bio2, Bio3) were characterised in terms of total solids content (TS), volatile solids content (VS), chemical oxygen demand (COD), total nitrogen content (N), total phosphorus content (P), electric conductivity (EC), and the pH-value. Additionally, the chloride content (Cl^-) in food waste samples was measured to detect possible inhibitory factors. The results with standard deviations are depicted in Figure 2 (TS and VS), Figure 3 (COD), and Figure 4 (total N, total P, Cl^-) as well as in Table 6 (pH and EC).

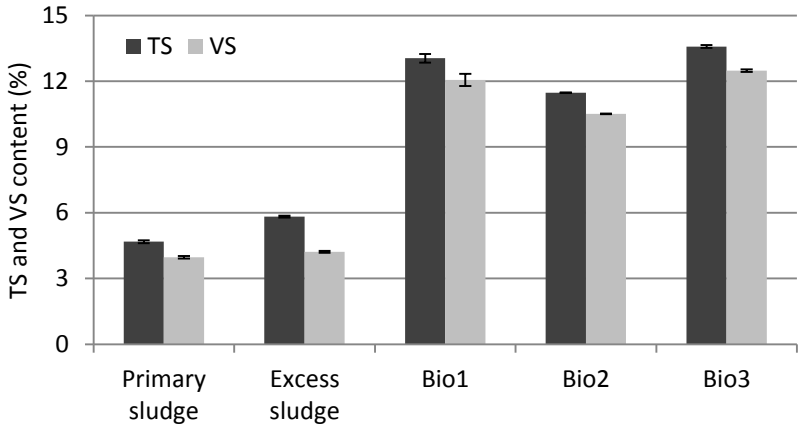

Figure 2. Total solids (TS) and volatile solids (VS) content in the substrates. Bars represent standard deviations. Bio1, Bio2 and Bio3 stand for food waste samples.

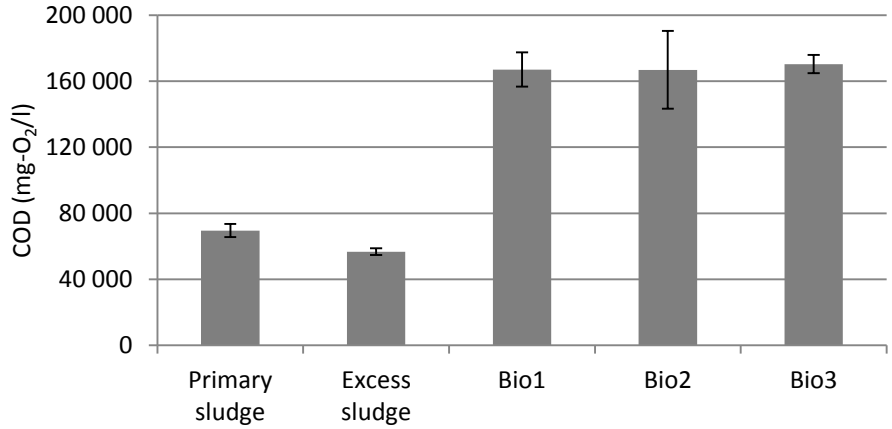

Figure 3. Chemical oxygen demand (COD) of the substrates. Bars represent standard deviations. Bio1, Bio2 and Bio3 stand for food waste samples.

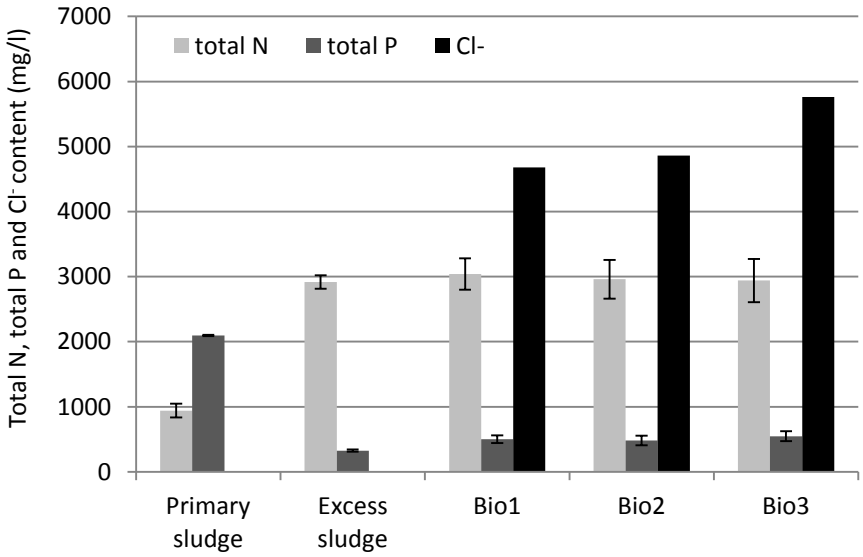

Figure 4. Total nitrogen (N), total phosphorus (P) and chloride (Cl^-) content in the substrates. Bars represent standard deviations. Bio1, Bio2 and Bio3 stand for food waste samples.

Table 6. pH-value and electric conductivity (EC) of the substrates at the given temperature. Bio1, Bio2 and Bio3 stand for food waste samples.

	pH	**EC (mS/cm)**	**Temperature (°C)**
Primary sludge	6,16	0,776	23,7
Excess sludge	6,57	1,19	22,8
Bio1	3,48	12,3	15,6
Bio2	3,20	11,8	14,5
Bio3	3,47	12,7	14,6

In general, two distinct groups of the substrates could be distinguished – the sludges and the food waste. The sludges were characterised by a lower TS as well as VS content and COD than food waste samples. TS values of the substrates ranged from 4,68% (primary sludge) to 13,6% (Bio3) as indicated in Figure 2. VS content formed 72% to 92% of the TS (Figure 2), varying between 3,96% (primary sludge) and 12,5% (Bio3). The chemical oxygen demand (Figure 3) ranged from 56 700 mg-O_2/l (excess sludge) to 170 400 mg-O_2/l (Bio3). Normak et al (2009) summarised the TS value of food waste to be in the range of 9–37% and VS value 80–98% of the TS. Thus, the findings of the current study coincide with formerly published data (Normak et al 2009, Al Seadi et al 2008, la Cour Jansen et al 2004).

The total nitrogen content of the substrates ranged from 942 mg-N/l (primary sludge) to 3040 mg-N/l (Bio1), and total phosphorus content from 326 mg-P/l (excess sludge) to 2100 mg-P/l

(primary sludge) as indicated in Figure 4. These values lie within the same range with the findings of other authors (Zhang et al 2007, la Cour Jansen et al 2004). The sludges showed slightly lower levels of total nitrogen content than food waste samples, while phosphorus content was highest in the primary sludge. This reflects on a high phosphorus removal efficiency of the primary sedimentation of the wastewater. The average chloride content (Figure 4) of Bio1, Bio2 and Bio3 was 5100 mg-Cl^-/l with a standard deviation 579 mg-Cl^-/l. Detected chloride concentration bore no inhibitory effect during the gas production experiment.

pH of the substrates varied between 3,2 (Bio2) and 6,57 (excess sludge). Electric conductivity varied between 0,776 mS/cm (primary sludge) and 12,7 mS/cm (Bio3), whereas temperature impact on the EC value has to be considered. Food waste samples distinguished from the sludges due to their higher EC and lower pH-value. The acidic pH of the food waste might have resulted from the hydrolysis of the substrates by microbial digestion, which started already during the collection of the waste.

Based on the recorded characteristics, the substrates were divided into two groups: the sludges and the food waste. The sludges were characterised by lower TS, VS and COD values, while food waste was distinguished by lower pH-value. Three food waste samples were found to be similar to one another in terms of all measured parameters indicating on low variability of hospital food waste in time.

3.2. Biogas and methane potential

The specific cumulative biogas production in liquid displacement system is depicted in Figure 5. The specific cumulative methane production in AMPTS II is depicted in Figure 6. The final values of each experimental line for biogas production as well as methane production are defined as gas potential GP_{21} and biological methane potential BMP_{15}, respectively, as illustrated in Figure 7.

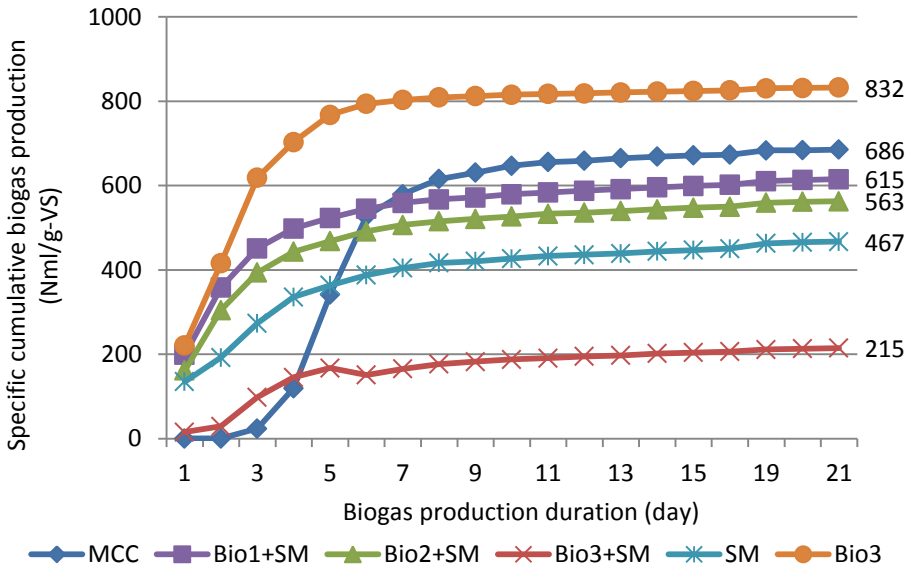

Figure 5. Specific cumulative biogas production of different experimental lines in liquid displacement system. Experimental lines include MCC (positive control with microcrystalline cellulose); SM (sludge mixture of primary and excess sludge); Bio3 (food waste from third sampling); Bio1+SM, Bio2+SM, Bio3+SM (mixtures of different food waste samples and sludge mixture).

The results depicted in Figures 5, 6 and 7 reveal the differences in biogas production potential between the tested substrates. The results of the positive control (MCC) confirm the activity of the inoculum used. However, it can be seen in Figure 5 that positive control encountered a lag-phase, while biogas production from other substrates started at a higher production rate. This was probably due to a prior hydrolysis of the substrates and the characteristics of the microbial community, which was adapted for the digestion of the given substrates but not for the digestion of MCC. The majority of the biogas was produced in the first five to seven days; the production rates during the last two weeks of the experiment remained low as illustrated by a plateau of the cumulative curve in Figure 5 for all experimental lines. A similar biogas production pattern has also been reported by other authors (Zhang et al 2007).

In addition to the volume of produced biogas, the percentage of its main components in liquid displacement system was detected regularly. The highest methane content (72%) was measured in Bio3 at day 6 and the concentration remained around 70% until the end of the experiment. The reactors with co-substrate showed highly similar methane concentrations: the methane content remained stable around 54% after a five-day increase in the beginning of the

experiment. Additionally, H_2S concentration was measured. All detected H_2S concentrations remained below the critical limit of 250 ppm, which marks significant corrosive damage to biogas processing facilities (Weiland 2010). The highest detected H_2S concentration was 90 ppm in Bio3 usage, reflecting an appropriate composition of biogas for utilisation.

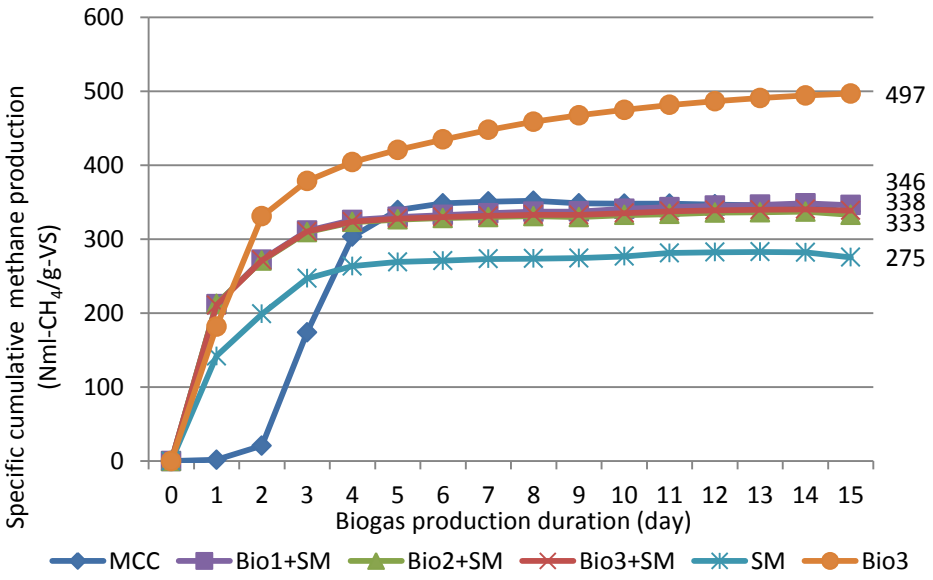

Figure 6. Specific cumulative methane production of different experimental lines in AMPTS II. Experimental lines include MCC (positive control with microcrystalline cellulose); SM (sludge mixture of primary and excess sludge); Bio3 (food waste from third sampling); Bio1+SM, Bio2+SM, Bio3+SM (mixtures of different food waste samples and sludge mixture).

The results from AMPTS II experiment are depicted in Figures 6 and 7. In general, methane production in AMPTS II followed similar trends with biogas production in liquid displacement system. The digestion was characterised by a high methane production rate during the first days of the experiment and a subsequent plateau phase in the production. Similarly to the biogas production in liquid displacement system, positive control (MCC) encountered a preliminary lag-phase. The experimental lines with co-substrates followed a highly similar methane production pattern with one another.

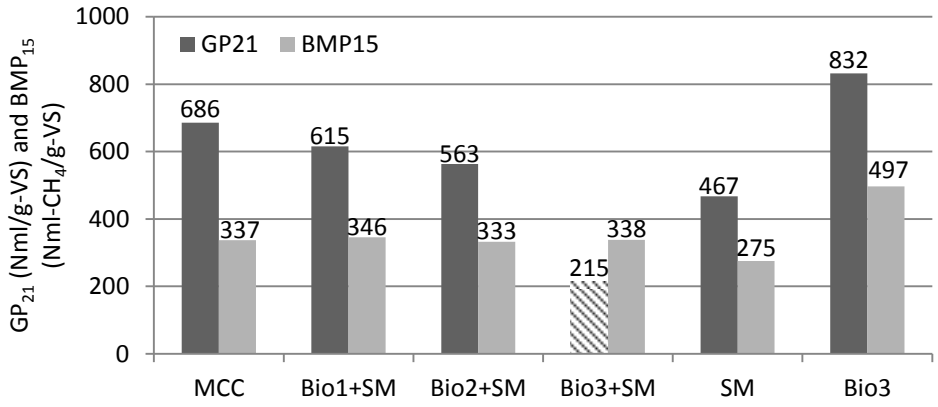

Figure 7. Biogas potential GP_{21} and biological methane potential BMP_{15} of the experimental lines. Experimental lines include MCC (positive control with microcrystalline cellulose); SM (sludge mixture of primary and excess sludge); Bio3 (food waste from third sampling); Bio1+SM, Bio2+SM, Bio3+SM (mixtures of different food waste samples and sludge mixture). GP_{21} of Bio3+SM (hatched diagonally) should not be validated due to a technical error in the experiment.

Biogas potential together with the biological methane potential of the substrates is depicted in Figure 7. It must be noted, that the significantly low GP_{21} of Bio3+SM should not be validated due to a technical error in the experiment. Bio3 had the highest GP_{21} (832 Nml/g-VS), while the GP_{21} of sludge mixture was nearly 1,8 times lower (467 Nml/g-VS). It is clearly visible that BMP_{15} values followed the same pattern as GP_{21} values. Bio3 had the highest methane potential (497 Nml-CH_4/g-VS), while sludge mixture showed the lowest methane potential (275 Nml-CH_4/g-VS). These values are consistent with formerly published data (Raposo et al 2011, la Cour Jansen et al 2004). Co-digestion of the substrates resulted in an increased productivity compared to the mono-fermentation of the sludge mixture. This finding has also been reported by previous studies (Raposo et al 2011, Sosnowski et al 2008, la Cour Jansen et al 2004, Kim et al 2003).

The BMP_{15} values formed 56–60% of the GP_{21}, which is consistent with the general composition of biogas (Al Seadi et al 2008). For Bio3, methane potential determined in AMPTS II was lower than measured CH_4 concentration in liquid displacement system; for co-digestion, methane potential in AMPTS II was slightly higher than measured CH_4 concentration in liquid displacement system. The differences might have resulted from different mixing of the reactors: liquid displacement system was mixed manually once a day,

while AMPTS II was mixed automatically every four minutes. Mixing facilitates the contact between microbes and substrates (Angelidaki et al 2009) and therefore impacts the production of methane as well as biogas.

In conclusion, the biogas production was characterised by a high production rate in the first days of the experiment in both test systems. Detected biogas potential of the substrates bore similar pattern to the detected biological methane potential of the substrates. Highest productivity was recorded for Bio3, while sludge mixture showed the lowest productivity. Co-digestion of the substrates resulted in an increased productivity compared to the mono-fermentation of the sludge mixture.

3.3. Digestion residues

Digestion residues from liquid displacement system were characterised in terms of total solids content (TS), volatile solids content (VS), chemical oxygen demand (COD), pH-value and electric conductivity (EC). The TS content ranged from 3,10% (Bio3) to 3,18% (Bio2+SM). The organic fraction comprised 57,3% to 59,4% of the TS, resulting in VS values ranging from 1,78% (SM) to 1,88% (Bio2+SM). According to the VS value of the given substrate and the corresponding values in digestion residues, VS degradation rate was calculated (Table 7). The highest substrate degradation rate was found in Bio3, indicating its best characteristics for microbial digestion. SM showed the lowest degradation rate, which demonstrates its limited qualities for microbial digestion.

Table 7. Volatile solids (VS) degradation rate of the substrates in liquid displacement system.

	Average (%)	**Standard deviation (%)**
MCC	81,0	3,15
Bio1+SM	66,1	0,10
Bio2+SM	57,8	0,85
Bio3+SM	69,1	1,89
SM	57,4	2,27
Bio3	85,1	1,30

Chemical oxygen demand of the digestion residues remained in a limited range for all experimental lines with an average value 28 160 mg-O_2/l (standard deviation 459 mg-O_2/l). The same trend was noticed for the pH-value as well as electric conductivity. An average pH

of the experimental lines was 8,14 (standard deviation 0,10) and average EC at 23,2°C was 14,6 µS/cm (standard deviation 0,58 µS/cm). The average pH-value increased by 0,47 units during the experiment, while the average EC decreased. The rise in pH can be explained by the accumulation of alkaline substances such as ammonia, the decrease in electric conductivity is a result of diminishing ion concentration during the digestion.

3.4. General conclusions

According to the results of substrate and digestate characterisation and fermentation tests with liquid displacement system as well as automatic methane potential test system, food waste proves to be a highly valuable substrate for biogas production. The analysis detected no inhibitory effects, indicating on the appropriate composition of the substrates for anaerobic digestion. Biogas production was characterised by a high production rate in the beginning of the experiment, which shows the ability of the microbial community to start digestion without a prior adaption period.

The substrates used in the experiment were provided by the wastewater treatment plant in Zirl and the state hospital of Innsbruck. The food waste collection system in the hospital is based on a vacuum technology, which has not been taken into use in large scale yet (Bernstad and la Cour Jansen 2012). This study provided evidence of the suitability of the novel system as the results showed a high energetic value of the food waste collected with this technology. Future studies could further analyse the optimal technical solutions as well as implementation of the system to contribute to the spread of the technology.

It is recommended to continue food waste co-digestion with sewage sludge in the WWTP Zirl. Anaerobic digestion accounts for the stabilisation of sewage sludge from wastewater treatment and provides an alternative source of energy. The use of food waste as a co-substrate proved to increase energy yields. The optimal ratio between sewage sludge and food waste in WWTP Zirl should be addressed by future studies in order to fully utilise sewage sludge and maximise biogas production.

Summary

Due to the global growth in energy demand and limited fossil resources, alternative energy sources have gained importance. Biomass represents a sustainable source of renewable energy, characterised by its abundance. During anaerobic digestion of the biomass, organic matter is degraded by the microbial community, producing an energy-rich mixture of gases, the biogas, which is commonly utilised for combined heat and power production. The digestion residues can be applied for soil fertilisation. In the following study food waste co-digestion with sewage sludge was analysed. The study aimed at the characterisation of sewage sludge and food waste as well as the digestion residues, the detection of biogas potential and the detection of biological methane potential of the substrates in lab-scale reactor systems.

The substrates were provided by the wastewater treatment plant Zirl (sewage sludge from the wastewater treatment) and the state hospital of Innsbruck (food waste collected by an automatic vacuum technology). The samples were characterised in terms of total solids content (TS), volatile solids content (VS), chemical oxygen demand (COD), total nitrogen content (N), total phosphorus content (P), electric conductivity (EC), and the pH-value. The results revealed two distinguishable groups: the sludges were characterised by lower TS, VS and COD values, while food waste was distinguished by acidic pH-value. Three food waste samples were found to be similar to one another in terms of all measured parameters.

A fermentation experiment was conducted with two distinct test systems: liquid displacement system was used for the detection of biogas potential GP_{21} and automatic methane potential test system AMPTS II was used for the detection of biological methane potential BMP_{15}. GP_{21} showed similar pattern to the detected BMP_{15} of the substrates. Highest productivity was recorded for food waste sample Bio3 (GP_{21}=832 Nml/g-VS, BMP_{15}=497 Nml-CH_4/g-VS), while sludge mixture from primary and excess sludge showed the lowest productivity (GP_{21}=467 Nml/g-VS, BMP_{15}=275 Nml-CH_4/g-VS). Co-digestion of the substrates resulted in increased productivity compared to the mono-fermentation of the sludge mixture.

According to the results of the study food waste is a highly valuable substrate for anaerobic digestion and co-digestion of sewage sludge with food waste is recommended for the WWTP Zirl. Further studies are suggested to assess the optimal ratio of the co-substrates.

Biogaasi tootmine toidujäätmete kooskääritamisel reoveemudaga

Kärt Kanger

Kokkuvõte

Kasvav globaalne energianõudlus ja piiratud fossiilsete kütuste varud tingivad vajaduse pöörata tähelepanu alternatiivsetele energiaallikatele. Biomass on taastuvenergiaallikas, mida iseloomustab lai levik. Biomassi anaeroobsel lagundamisel tekivad mikroorganismide elutegevuse tulemusel energiarikas gaaside segu, biogaas, mida kasutatakse enamasti energia ja sooja koostootmiseks, ja käärimisjääk, mis on sobiv põllumajandusväetis. Käesolev bakalaureusetöö uuris biogaasi tootmist toidujäätmete kooskääritamisel reoveemudaga. Töö eesmärgid olid iseloomustada kasutatud tooraineid ja käärimisjääki ning määrata toorainete biogaasi potentsiaal ja bioloogiline metaani potentsiaal laboratoorsetes reaktorsüsteemides.

Toorained pärinesid Zirli reoveepuhastusjaamast (reoveepuhastuse primaar- ja liigmuda) ning Innsbrucki haiglast (vaakumsüsteem toidujäätmete kogumiseks). Toorainete iseloomustamiseks määrati nende kuivainesisaldus, orgaanilise kuivaine sisaldus, keemiline hapnikutarve, üldlämmastiku ja üldfosfori kontsentratsioon, pH ning elektriline juhtivus. Tulemused näitasid toorainete jagunemist kahte eraldiseisvasse gruppi: reoveemudasid iseloomustasid madalamad kuivaine ja orgaanilise kuivaine sisaldused ning samuti madalam keemiline hapnikutarve, toidujäätmeid iseloomustas happeline pH. Leiti, et kolm analüüsitud toidujäätmete proovi olid üksteisega sarnased kõigi määratud parameetrite puhul.

Kääritamiskatses kasutati kahte testsüsteemi: eudiomeetriga reaktorsüsteemi biogaasi potentsiaali GP_{21} määramiseks ja automaatset metaanipotentsiaali testsüsteemi AMPTS II bioloogilise metaanipotentsiaali BMP_{15} määramiseks. Kasutatud testsüsteemide tulemused olid omavahel sarnased. Kõrgeima tootlikkusega olid toidujäätmed Bio3 (GP_{21}=832 Nml/g-VS, BMP_{15}=497 Nml-CH_4/g-VS), madalaim tootlikkus tuvastati primaar- ja liigmuda segul (GP_{21}=467 Nml/g-VS, BMP_{15}=275 Nml-CH_4/g-VS). Reoveemuda ja toidujäätmete kooskääritamisel saavutati kõrgem tootlikkus kui reoveemuda mono-kääritamisel.

Vastavalt käesoleva töö tulemustele on toidujäätmed kõrge väärtusega tooraine anaeroobseks lagundamiseks ja Zirli reoveepuhastusjaamas on soovituslik reoveemuda kääritada koos toidujäätmetega. Täiendavad uuringud on vajalikud toorainete optimaalse suhte määramiseks.

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Appendix 1. Food waste collection in the state hospital of Innsbruck

The food waste in the state hospital of Innsbruck originates from several sources, which are listed below.

1. Bread station: bread with minimal amount of water vacuumed to the primary tank.
2. Mixed food waste station: soups, salad sauces etc gathered manually from plates to the collecting system and vacuumed to the primary tank.
3. Vegetable station: vegetable waste vacuumed to the primary tank.
4. Production station: production residues from the kitchen and mixed food waste from the staff canteen vacuumed to the primary tank.
5. Grease trap from wastewater: separated grease vacuumed to the primary tank.

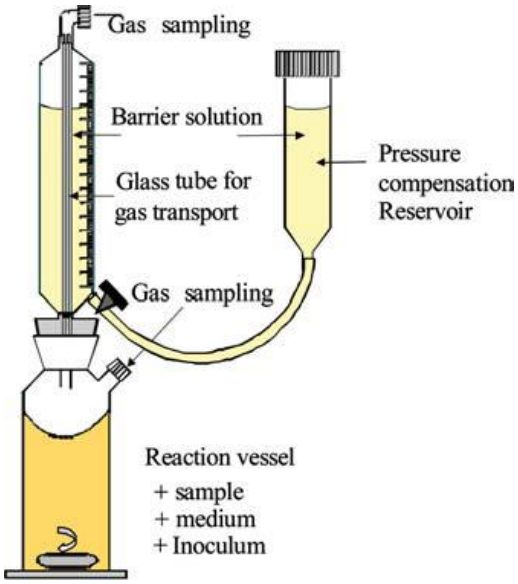
All waste is collected to a closed primary tank, where it is subject to homogenisation and size reduction. Homogenised food waste (A1) is directed to a closed storage tank and eventually transported to the wastewater treatment plant by a liquid waste collecting car as described in chapter 2.1.2.

A1. Homogenised food waste sample from the state hospital of Innsbruck.

Appendix 2. Liquid displacement system with eudimeter unit

The volume of produced biogas was measured by a liquid displacement system. The system was based on the eudiometer unit, which is described in the international standard ISO/DIS 14853 (ISO/DIS 14853 1999).

The unit (A2) consists of reactor vessel with a septum for the extraction of samples, a sealed gas collection tube and a reservoir tank (Guwy 2004). Produced biogas passes from the reactor vessel into the gas collection tube, displacing barrier solution into the reservoir tank (Guwy 2004). The volume of the produced biogas can be recorded from the scale on the collection tube when the liquid levels in the collection tube and reservoir are brought to the same niveau.



A2. Reaction system with eudiometer unit. The main parts of the eudiometer unit include the reaction vessel, the gas collection tube with barrier solution and a reservoir tank (Guwy 2004, modified from ISO/DIS14853 1999).

Appendix 3. Automatic methane potential test system AMPTS II

The methane potential of the substrates was detected using an automatic methane potential test system AMPTS II (Bioprocess Control, Lund, Sweden).

The AMPTS II instrument (A3) consists of a water bath with maximal 15 reactor bottles, followed by carbon dioxide trap bottles and a tipping mechanism for the detection of produced methane. Each reactor is mixed by a slow rotating mixing rod. The produced gas passes from the reactor to the carbon dioxide trap bottles, where CO_2 is absorbed by sodium hydroxide solution. The remaining methane is then directed to the detection system, which measures the number of pulses generated by a pre-defined volume of gas flowing though the device. AMPTS II uses a real-time data recording system which also allows the analysis of the results (Browne and Murphy 2013).

A3. AMPTS II instrument. The reactors in a white water bath are situated in the back right of the image; in the front right CO_2 trap bottles can be seen; the white device in the middle is methane detection unit, which sends online data to the computer.

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